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Riparian Wetland Response to Livestock Exclusion

In the Lower Columbia River Basin

by

Sarah Ann Holmen

A thesis submitted in partial fulfillment of the
requirements for the degree of

Master of Science
In
Environmental Science and Management

Thesis Committee:
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Portland State University
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ABSTRACT

The purpose of this study was to examine the characteristics of riparian plant communities along a succession gradient of livestock exclusion in the Lower Columbia River Basin (LCRB). Livestock exclusion is an example of a passive restoration practice throughout the region. However, few studies have focused on the effects of livestock or livestock exclusion on riparian wetland ecosystems in this area. Two passive restoration sites, 3 and 13 years since livestock exclusion, and a control site with a continued livestock grazing presence were examined. It was hypothesized that native plant species richness would be lower in the excluded wetlands than in the grazed wetland due to the competitive exclusion from an increase in non-native plant dominance in the absence of grazing. Data were collected along six (45-60m) randomly distributed transects which were aligned perpendicular to the wetland shoreline of each site, providing a total of 18 transects with an accumulative length of approximately 990 meters. Vegetation cover data were collected for 10 cm intervals along these transects using the line intercept method during low water periods in August and September of 2009. The Kruskal–Wallis one-way nonparametric analysis of variance by ranks and the Mann–Whitney U test were used to detect significant ($p < 0.05$, $p < 0.0167$ after Bonferroni adjustment) differences in native and non-native plant species richness, diversity indices and relative cover among sites.

A total of 58 plant species were identified among all three study sites: 27 native, 27 non-native and 4 species of unknown origin. The grazed wetland had significantly

($p < 0.0167$) greater average total species richness (23.3), native (10.2) and non-native (12) species richness than both the excluded wetlands. Average species richness did not differ significantly between the 3 year and 13 year excluded wetlands for both native (6.6, 2.8, $p = 0.088$) and non-native (5, 2.7, $p = 0.064$) species or total species richness (12, 5.5, $p = 0.063$). However, native species abundance was significantly ($p < 0.0167$) lower on the 13 year excluded wetland (4.2%) than both the 3 year excluded (51.5%) and grazing (23.2%) wetlands. The invasive grass *Phalaris arundinacea* L., commonly known as reed canarygrass, was found to be the dominant vegetation cover in all three wetlands with average relative cover ranging from 95.2% at the 13 year exclusion site to 52.8% at the grazing site and 43.0% cover at the 3 year exclusion site. These results suggest that livestock exclusion alone may be an ineffective strategy for restoring riparian plant communities in the LCRB where invasive species like *Phalaris arundinacea* L. are abundant. Other more practical management strategies could include short-term livestock exclusion and re-introduced targeted grazing to reduce livestock impacts and control *Phalaris arundinacea* L. dominance.

DEDICATION

I dedicate this thesis to my mom and dad.

Without their support and guidance I would not have been able to get where I am today.

Love you guys.

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TABLE OF CONTENTS

ABSTRACT.....	i
ACKNOWLEDGMENTS	iv
LIST OF TABLES	ix
LIST OF FIGURES	ix
INTRODUCTION	1
BACKGROUND	3
Lower Columbia River Basin	3
Invasive Species.....	4
Invasive Plants and Disturbance	5
Riparian Zone and Riparian Wetlands.....	8
Riparian Wetland Restoration.....	9
Dams and Restoration in the LCRB.....	10
Invasive Plants in the LCRB: Reed Canarygrass.....	11
Livestock Grazing	15
Livestock Grazing and Restoration in the LCRB	20
OBJECTIVES AND HYPOTHESES	28
This Study: Research Objectives & Hypotheses	28
METHODS	30

Study Approach	30
Site Description.....	31
Sites.....	32
Vegetation Survey.....	33
Transect Point Selection	33
Transect Point Location	35
Transect Vegetation Surveying Procedure.....	36
Diversity.....	37
Plant Community Similarity	39
Riparian Wetland Topography.....	40
Approach.....	40
Digital Elevation Model.....	41
Vegetation Transect Data Layer	42
Sample Plot Points	42
Extraction of Elevation Data.....	43
Soil Survey.....	43
Approach.....	43
Soil Bulk Density	44
Soil Organic Matter.....	45

Soil Texture Analysis.....	45
Statistical Methods.....	46
Differences in Vegetation Composition among Sites	46
Reed canarygrass Abundance and Plant Community Diversity	48
Differences in Soil Characteristics among Sites	49
RESULTS	50
Site Vegetation Survey	50
Species Richness.....	53
Shannon Diversity Index.....	56
Relative Cover	58
Plant Community Similarity	66
Reed canarygrass Abundance and Plant Community Diversity	68
Elevation Data.....	72
Soil Survey.....	73
Soil Texture.....	73
Organic Matter	75
Soil Bulk Density.....	76
DISCUSSION	78
Grazing and Wetland Plant Community Characteristics	78

Management Implications.....	84
References.....	86
Appendix A: Complete Plant Species List.....	108
Appendix B: Calculating Study Grazing Intensity	112

LIST OF TABLES

Table 1: Livesock Grazing Literature Reviews..	16
Table 2: Summary of Wetland Grazing Studies that Measured Species Richness Under Different Grazing Regimes/Intensities.	24
Table 3: Summary of Total Site Vegetation Characteristics.	50
Table 4: Summary of Site Average Vegetation Characteristics	51
Table 5: Plant Species Common among All Study Sites and Average % Relative Cover of Each Species Found on Each Site.....	55
Table 6: Comparison of Average % Relative Cover of Non-native Plant Species at Each Site by % Abundance Category.	62
Table 7: Comparison of Average % Relative Cover of Native Plant Species at Each Site by % Abundance Category.	63
Table 8: Bray Cutris Index of Similarity Within and Among Sites.....	67
Table 9: Spearman's Rho Correlation Analysis.....	69
Table 10: Average Slope and Elevation Range of Each Study Site.	73
Table 11: Summary of Average Soil Parameters for Each Site.	74

LIST OF FIGURES

Figure 1: A Conceptual Model Showing Factors Involved in Determining Livestock Grazing Impacts on Riparian Wetlands in the Lower Columbia River Basin (LCRB).	19
Figure 2: Conceptual Figure of Expected Plant Community Diversity and Invasive Species Dominance in Response to Different Disturbance Regimes	26
Figure 3: Map of Study Sites.	32
Figure 4: Map of Transect Placement and Estimated High Water Line in 2009.	34
Figure 5: Number of New Plant Species Found Along Each Transect.....	35
Figure 6: Comparison of Total Species Richness among Study Sites	52
Figure 7: Comparison of Average Species Richness among Study Sites.	53
Figure 8: Plant Species Presence at All Study Sites.	56
Figure 9: Comparison of Each Study Site's Average True Diversity.....	57
Figure 10: Comparison of Each Study Site's Average Shannon Diversity Index H'	58
Figure 11: Comparison of Each Study Site's Average % Relative Vegetation Cover.	60
Figure 12: Comparison of the Average % Relative Cover of Reed canarygrass (<i>Phalaris arundinacea</i>) and Other Non-native Species Among Study Sites.	61
Figure 13: Current Grazing Site % Average Cover of All Plant Species $\geq 1\%$	64
Figure 14: Short-term Exclusion Site % Average Cover of all Plant Species $\geq 1\%$	65
Figure 15: Long-term Exclusion Site % Average Cover of all Plant Species $\geq 0.01\%$	66
Figure 16: Bray-Curtis Index of Plant Community Similarity Ordination of Site Transects by non-metric Multidimensional Scaling.....	68

Figure 17: Total Shannon Diversity Index (H') vs. Relative % Cover Reed canarygrass (<i>Phalaris arundinacea</i>).....	70
Figure 18: Total Native Shannon Diversity Index (H') vs. Relative % Cover Reed canarygrass (<i>Phalaris arundinacea</i>).....	71
Figure 19: Relative % Native Cover vs. % Cover Reed canarygrass (<i>Phalaris</i> <i>arundinacea</i>).....	72
Figure 20: Average Texture Composition of Soil at All Study Sites	74
Figure 21: Boxplot Showing the Spread and Average of the % Soil Organic Matter (OM) by Site	76
Figure 22: Average Soil Bulk Density (g/cm^3) of the Soil at Each Study Site.....	77
Figure 23: Conceptual Figure of Expected Plant Community Diversity and Invasive Species Dominance in Response to Different Disturbance Regimes (such as grazing) Including Study Sites.....	82

INTRODUCTION

In the Pacific Northwest one of the largest networks of riparian wetlands is located in the Columbia River Watershed. Recent declines in anadromous Pacific salmonids (*Oncorhynchus* spp.) in this region have been attributed in part to degradation of freshwater riparian habitats (PACFISH 1994, Kershner et al. 2004, Fullerton et al. 2006). Riparian wetland restoration is an important component of the endangered salmon recovery efforts in the Lower Columbia River Basin (LCRB) (PACFISH 1994, Goodwin et al. 1997, LCFRB 2004, Lev et al. 2004). Some of the greatest documented impacts to the LCRB riparian wetlands are livestock grazing, non-native plant invasion and flood control (Christy and Putera 1992, Leonard and Karl 1995, Lev et al. 2004). The ubiquity of invasive plants and altered riverine hydrology from upstream dams have made understanding land use impacts and conducting restoration projects in the region a complex challenge (Goodwin et al. 1997, Lev et al. 2004).

Livestock grazing has been a widespread use of the land in the LCRB floodplains since the early 1800's (Christy and Putera 1992). Livestock impacts to riparian wetlands have been extensively studied throughout the arid west. In these western riparian areas, livestock grazing has been shown to degrade riparian habitats by altering plant community composition, local geomorphology and water quality (Kauffman and Krueger 1984, Elmore and Kauffman 1994, Mosley et al. 1997, Larsen et al. 1998, Belsky et al. 1999, Krueger and Sanderson 2002, Sarr 2002). The extents of these grazing impacts are determined by a combination of local environmental conditions and grazing management practices (e.g., Kauffman and Krueger 1984, Menke and Bradford 1992, Milchunas and

Lauenroth 1993, Vavra et al. 1994, Green and Kauffman 1995, Krueger and Sanderson 2002). Livestock exclusion from riparian wetlands is a common passive restoration technique used to mitigate grazing impacts and restore native plant community composition (Reichard 1989, Leonard and Karl 1995, Kauffman et al. 1997). The success of livestock exclusion as a restoration technique in the more temperate and developed environment of the LCRB has not been well documented.

In this study, riparian plant communities of two passively restored riparian wetlands three and thirteen years post livestock exclusion were examined and compared to an actively grazed riparian wetland. This study is a step in understanding the effectiveness of livestock exclusion as a passive restoration technique in the LCRB. Results from this study may inform land managers and restoration practitioners regarding the restoration and management of grazed riparian wetlands in the LCRB.

BACKGROUND

Lower Columbia River Basin

The lower 146 river miles of the Columbia River from the Bonneville Dam to the Pacific Ocean compose the LCRB. The hydrology of the LCRB is tidally influenced from the Pacific Ocean to the Bonneville Dam and water levels in this region are regulated by the Bonneville Dam and over 200 other dams located in the upper Columbia basin watershed. It has been estimated that over half of the LCRB's historic riverine and floodplain wetlands have been lost or significantly degraded due to these dams and other local hydrologic regulations such as diking, draining, filling, and dredging (Christy and Putera 1992, USFWS 2004). Historic and current land uses in the basin such as agriculture, timber harvest, grazing and urban development have also caused degradation and loss of the LCRB riparian wetland ecosystems (Christy and Putera 1992, Christy 2004). These land uses and associated hydrologic regulations have aided the invasion of non-native, exotic plant species throughout the basin causing further riparian wetland habitat impairment (Zedler and Rea 1998, Hood and Naiman 2000, Mulhouse and Galatowitsch 2003, Zedler and Kercher 2004, Richardson et al. 2007). Invasive non-native plant species can dominate these habitats to the exclusion of important native species, significantly altering the habitat structure and function (Lesica 1997, Galatowitsch et al. 1999, Hood and Naiman 2000, Mulhouse and Galatowitsch 2003, Zedler and Kercher 2004, Schooler et al. 2006, Richardson et al. 2007, Osland 2009, Spyreas et al. 2010). In 1993 Christy and Putera reported that the invasion of Reed canarygrass (RCG), *Phalaris arundinacea*, along the lower Columbia River had degraded

or destroyed almost as much wetland acreage as wetland diking and clearing in the region.

The LCRB's stream, riparian and wetland environments are essential habitat for many species. Under the Endangered Species Act, 32 species of plants, fish and wildlife found in the LCRB are listed as either threatened or endangered, including 13 salmonid species (LCREP 2010). Improving and providing habitats for these species have been the main focus of basin wide restoration efforts (LCREP 2010). Restoring these habitats as well as protecting the existing riparian habitats is fundamental to maintaining the biological diversity, integrity and health of the LCRB ecosystem.

Invasive Species

Invasive species can be defined as non-native species which have successfully spread beyond the point of their introduction, altering native flora and fauna community compositions and function (Hobbs and Huenneke 1992, Vitousek et al. 1997, Zedler and Rea 1998, Richardson et al. 2000, Zedler 2009). They pose a significant threat to ecosystem biodiversity by overrunning habitats and locally pushing native species to extinction, creating less diverse and poorly functioning ecosystems (Vitousek et al. 1997, Zedler and Rea 1998, Mooney and Hobbs 2000, Herben et al. 2004, Fierke and Kauffman 2006, Richardson et al. 2007). Invasive species have become so abundant that it is almost impossible to conduct ecological field research without including them as a component of the study. The ubiquitous presence of invasive species has created many challenges for restoration ecologists and environmental managers. The economic and environmental

costs of non-native species invasions are very high. On a global scale invasive species are considered a major threat to species diversity coming second only to human land use and habitat loss (Vitousek et al. 1997, Chaplin et al. 2000). The damage and control costs of invasive species are estimated to be over 100 billion dollars a year in the United States alone (Pimentel et al. 2005). The introduction and spread of invasive species threatens the long-term viability of all native fauna and flora.

Invasive Plants and Disturbance

The invasibility of an ecosystem system by a non-native plant species can be determined by a number of factors including soil nutrient levels, hydrology, plant species composition and disturbance regime (Mack 1981, Hobbs and Huenneke 1992, Mack and D'Antonio 1998, Galatowitsch et al. 1999, Lavergne and Molofsky 2004, Zedler and Kercher 2004, Osland 2009, Zedler 2009). Disturbance is the primary factor involved in most successful invasions (Mack 1981, Hobbs and Huenneke 1992, Lavergne and Molofsky 2004, Zedler and Kercher 2004, Locke et al. 2008). An environmental disturbance is commonly defined as “any relatively discrete event in time that disrupts ecosystem, community or population structure and changes resources, substrate availability, (*and*/) or the physical environment” and can be caused by natural or anthropogenic factors (White and Pickett 1985, page 7). An anthropogenic induced change in the intensity, frequency, duration or predictability of an ecosystem’s natural disturbance regime can also be considered a type of system disturbance (White and Pickett 1985). These changes to a natural disturbance regime have also been associated

with non-native species invasions (Mack 1981, Hobbs and Huenneke 1992, Mack and D'Antonio 1998, Lavergne and Molofsky 2004, Zedler and Kercher 2004, Locke et al. 2008, Zedler 2009). Examples of common environmental disturbances (both natural and anthropogenic) associated with the spread and establishment of invasive species include grazing, fire, flooding, drought, erosion, pollution, and anthropogenic; hydrologic regulation, land use, and development (e.g., Mack 1981, D'Antonio and Vitousek 1992, Hobbs and Huenneke 1992, Mack and D'Antonio 1998, Galatowitsch et al. 1999, Lavergne and Molofsky 2004, Zedler and Kercher 2004, Locke et al. 2008, Osland 2009, Zedler 2009).

Once an ecosystem is invaded, the invasive species can alter the ecosystem causing its own disturbance, further enhancing the establishment of itself and other invasive species (D'Antonio and Vitousek 1992, Hobbs and Huenneke 1992, Mack and D'Antonio 1998, Zedler 2009). The invasion of *Bromus tectorum* L. (*Bromus*) in the arid west is a well established example of this invasion disturbance feedback mechanism. *Bromus* invades areas that suffer from heavy overgrazing and cultivation (Mack 1981, Knick and Rotenberry 1997). Once established, *Bromus* alters the natural fire regime of the area increasing fire frequency and intensity (Mack 1981, Knick and Rotenberry 1997). Native plant communities are not adapted to the altered fire regime and this decreases the plant communities' ability to compete with *Bromus*. Through altering the native fire regime *Bromus* creates a disturbance feedback mechanism which enhances its ability to establish and spread (Mack 1981, D'Antonio and Vitousek 1992, Knick and Rotenberry 1997, Mack and D'Antonio 1998). This feedback mechanism has also been

documented with other invasive species altering the local soil nutrient dynamics, hydrology, microenvironment and resource availability degrading ecosystem function and increasing ecosystem invasibility (D'Antonio and Vitousek 1992, Mack and D'Antonio 1998, Davis et al. 2000, Zedler 2009). RCG invasion in tussock sedge meadows has also been found to create a disturbance feedback mechanism. RCG can increase sediment deposition in invaded meadows decreasing the microtopographic relief of the meadow which degrades the native sedge meadow plant species ability to compete with the RCG (Werner and Zedler 2002, Zedler and Kercher 2004, Zedler 2009). This disturbance feedback loop (mechanism) created by some invasive plants, like *Bromus* and RCG, can dramatically increase the restoration and management efforts needed to reverse the resultant habitat degradation.

Disturbance is also an important component of an ecosystem's function. In the absence of invasive species, disturbance can facilitate species diversity and healthy ecosystem function (White and Pickett 1985, Hobbs and Huenneke 1992). The intermediate disturbance hypothesis states that the highest ecological diversity is supported by intermediate frequencies or intensities of disturbance (Grime 1973, Connell 1978, Fox and Connell 1979, Hobbs and Huenneke 1992). Unfortunately, the role of disturbance in maintaining ecological diversity is complicated by its ability to facilitate the establishment of invasive species. The presence of invasive species makes the role of disturbance in an ecosystem a challenging ecological component to evaluate and manage.

Riparian Zone and Riparian Wetlands

The riparian zone can be defined as the transitional area between terrestrial upland and aquatic riverine systems (EPA 2002). Typically, the land directly adjacent to creeks, streams and rivers (and in some cases ponds or lakes) is considered the riparian zone (Mosley et al. 1997). Riparian wetlands are located within the riparian zone and are characterized by a high water table, shallow inundation during the growing season and emergent hydrophytic plant species (EPA 2002). Riparian wetlands are some of the most biologically productive and diverse ecosystems on Earth and are a fundamental component of both the riverine and terrestrial ecosystems (Brinson et al. 1981, Gregory et al. 1991, Mitsch and Gosselink 2007). They contain valuable water resources, diverse and productive plant communities, and provide habitat for both resident and migrating aquatic and terrestrial biota (Brinson et al. 1981, Mitsch and Gosselink 2007). They provide many essential riverine ecological services such as water filtration, nutrient cycling, detritus input, erosion control and hydrologic regulation (Brinson et al. 1981, Gregory et al. 1991, Mitsch and Gosselink 2007). Riparian wetlands are also heavily used by humans and commonly suffer degradation from hydrologic regulation, farming, livestock grazing, development, pollution and invasive species (Brinson et al. 1981, Vitousek et al. 1997, Wissmar and Beschta 1998, Zedler and Rea 1998, Hood and Naiman 2000, Mulhouse and Galatowitsch 2003, Zedler and Kercher 2004, Patten 2006, Mitsch and Gosselink 2007, Richardson et al. 2007). These anthropogenic exploitations of riparian wetlands have degraded the ecological integrity of riverine systems throughout the world (Vitousek et al. 1997, Chapin et al. 2000, Zedler 2000, Patten 2006, Mitsch and

Gosselink 2007, Osland 2009). The destruction of these important habitats and ecological services has been linked to drastic declines in fishery and wildlife populations (Brinson et al. 1981, Goodwin et al. 1997, Kauffman et al. 1997, Vitousek et al. 1997, Chapin et al. 2000, Zedler 2000, Mitsch and Gosselink 2007). Recognition of the importance of these habitats has resulted in large scale efforts to restore and protect riparian wetlands (Brinson et al. 1981, Goodwin et al. 1997, Kauffman et al. 1997, Zedler 2000, Mitsch and Gosselink 2007, Richardson et al. 2007). In the Pacific Northwest, restoration efforts have been especially focused on riparian wetlands that have historically provided habitat for endangered Pacific salmonid species (LCREP 2010).

Riparian Wetland Restoration

Restoring the diversity, function and productivity of riparian wetlands is fundamental to the long-term recovery and maintenance of degraded riverine systems (Kauffman et al. 1997, PACFISH 1994, Goodwin et al. 1997, Wissmar and Beschta 1998, LCFRB 2004, Lev et al. 2004, Patten 2006, Richardson et al. 2007, LCREP 2010). It has been established that restoration of natural hydrologic conditions are fundamental to the successful recovery of a wetland's complete structure and function (Goodwin et al. 1997, Zedler 2000, Mulhouse and Galatowitsch 2003, Patten 2006, Mitsch and Gosselink 2007, Zedler 2009). The dynamic hydrologic characteristics and disturbance regimes of riparian systems make them some of the most vulnerable ecosystems to plant invasion (Hobbs and Huenneke 1992, Hood and Naiman 2000, Richardson et al. 2007, Zedler and Kercher 2004). This growing presence and persistence of invasive species has made

integrating invasive species management and monitoring plans into a comprehensive restoration approach necessary for the success of any restoration project (Goodwin et al. 1997, Vitousek et al. 1997, Zedler 2000, Wilson et al. 2004, Patten 2006, Richardson et al. 2007, Osland 2009, Zedler 2009).

Re-establishment of natural hydrology, native species seeding/planting, and invasive species removal and management are all important components of a successful restoration plan (Goodwin et al. 1997, Wissmar and Beschta 1998, Zedler 2000, Wilson et al. 2004, Patten 2006, Richardson et al. 2007, Zedler 2009). Passive restoration is also a common riparian wetland restoration technique (Reichard 1989, Leonard and Karl 1995, Kauffman et al. 1997). Passive restoration entails halting damaging land use activities in or near the riparian wetlands and letting the area recover naturally, without further restoration or management action (Kauffman et al. 1997). The use of different restoration techniques can vary depending on the pre-restoration condition of the riparian wetland, as well as the goals of and availability of funding for the restoration project. Success of the passive restoration technique in the LCRB has not been widely documented (Lev et al. 2004, LCREP 2010).

Dams and Restoration in the LCRB

The LCRB riparian ecosystems are subjected to large scale anthropogenic hydrologic regulation from the Bonneville Dam and other dams located in the Upper Columbia Basin. Dam and levee induced hydrologic regulation of the LCRB have caused significant changes to the natural flood regime of the basin (Wissmar and Beschta 1998,

Lev et al. 2004). Historically, the LCRB was inundated with annual spring floods for a longer period of time and with a higher level of water than it is today, with large scale flooding occurring approximately every 5 years (Lev et al. 2004, NRCC 2004).

Hydrologic regulation of the Columbia River has resulted in lower summer flows, higher winter flows and no large scale flooding events (Lev et al. 2004, NRCC 2004). Changes to natural hydrologic regimes and land use disturbances have both been shown to degrade wetland habitat and increase the abundance of invasive species making restoration a very complex task in this region (Zedler and Rea 1998, Hood and Naiman 2000, Mulhouse and Galatowitsch 2003, Herben et al. 2004, Zedler and Kercher 2004, Richardson et al. 2007). Local restoration projects focus on restoring historic hydrology by levee removal, installing water control structures, restoring plant communities with native seeding and planting, and passive livestock exclusion (Lev et al. 2004, Jenkins et al. 2008, LCREP 2010). In areas where levee removal and installation of water control structures is not feasible, restoring natural riparian wetland function can be difficult.

Invasive Plants in the LCRB: Reed Canarygrass

The prevalence of invasive plant species in the LCRB riparian wetlands has been documented since the 1970's (Christy and Putera 1992). The historic and current land uses as well as the altered hydrology of the LCRB have likely been the sources and have enhanced the spread of these plants throughout the basin (Barnes 1999, Christy 2004). Reed Canarygrass (RCG), *Phalaris arundinacea* L., is one of the most ubiquitous invasive plants found in the LCRB and can greatly hinder riparian wetland restoration

efforts (Joe Maser, *personal communication*, Christy 2004, Lev et al. 2004, Perkins and Wilson 2005, Fierke and Kauffman 2006). RCG is an aggressive cool-season early emergent. It is a long-lived perennial graminoid documented as an invasive wetland plant throughout North America (Galatowitsch et al. 1999, Maurer et al. 2003, Lavergne and Molofsky 2004). Other problem invasive riparian plant species in the LCRB include: Purple Loosestrife (*Lythrum salicaria*), Yellow Flag Iris (*Iris pseudacorus*), Japanese knotweeds (*Polygonum sachalinense*/*P. cuspidatum*) and Indigo Bush (*Amorpha fruticosa*). These invasive species are not as widely distributed throughout the basin as RCG and were not found in this study's riparian wetland sites.

RCG has native origins in North America; however, the invasive RCG is a hybrid of the native and Eurasian ecotypes, so it is believed that its aggressive characteristics are Eurasian in origin (Galatowitsch et al. 1999). Eurasian ecotypes of RCG have been introduced for forage crops, erosion control and for use in treatment wetlands throughout North America (Galatowitsch et al. 1999, Maurer et al. 2003). This facultative wetland grass has great morphological plasticity giving it a high tolerance for dynamic hydrologic conditions; RCG can be found in wetlands, riparian zones and upland areas (Paveglio and Kilbride 2000, Lavergne and Molofsky 2004, Herr-Turoff and Zedler 2006, Kercher et al. 2007). RCG grows in dense stands producing high amounts of biomass, with individual plants reaching heights of 1 to 2 meters (Barnes 1999, Stannard and Crowder 2001). RCG spreads aggressively through underground rhizomes, tillers and high (fertile) seed production (Stannard and Crowder 2001, Lavergne and Molofsky 2004). RCG growth begins in spring and continues through summer; it is possible for RCG growth to

continue into the autumn, depending on water availability (Barnes 1999, Stannard and Crowder 2001). Significant to its competitive success, RCG growth begins in the early spring, typically before native herbaceous species, with significant vertical growth and germination occurring mid-spring (Stannard and Crowder 2001).

RCG is widely considered a threat to wetlands and riparian ecosystems throughout the Pacific Northwest and Midwest (Apfelbaum and Sams 1987, Hutchinson 1992, Lesica 1997, Uthus 1999, Christy 2004, Perkins and Wilson 2005, Fierke and Kauffman 2006, Jenkins et al. 2008, Ringold et al. 2008, WRMWG 2009, Zedler 2009). The success of RCG invasion in the LCRB can be attributed to its early emergence (often before native species), rapid aggressive growth, and physiological tolerance to dry and wet water regimes (Barnes 1999, Maurer et al. 2003, Lavergne and Molofsky 2004, Herr-Turoff and Zedler 2006). Other competitive characteristics of RCG include a tall, dense growth form and production of copious amounts of persistent litter. In wetland and riparian habitats RCG can establish monospecific stands (Maurer et al. 2003, Zedler and Kercher 2004, Perkins and Wilson 2005, Fierke and Kauffman 2006, Zedler 2009). Stands of RCG vegetation are dense and produce thick mats of slowly decomposing litter reducing light availability and altering soil microhabitat (temperature and nutrient availability) that suppress competing plants and seedlings (Green and Kauffman 1995, Lesica 1997, WRMWG 2009, Zedler 2009). Monotypic stands of RCG can dominate wetland habitats for decades and are especially detrimental to late emergent native species (Emers 1990, Lesica 1997, Barnes 1999, Fierke and Kauffman 2006, Zedler 2009). In a Montana oxbow marsh study, Lesica 1997 found RCG cover increased up to

75% during a 9 year period. These monotypic stands of RCG displaced native species including the endangered marsh plant *Howellia aquatilis* (Lesica 1997). In a study of 58 inland wetlands in Ontario, Houlahan and Findlay 2004 found RCG abundance to be negatively correlated with native species richness and concluded that RCG was capable of competitively excluding native species. Werner and Zedler 2002 found stands of RCG increased sedimentation, decreasing microtopographic variation and species richness in Wisconsin sedge meadows.

RCG invasions are capable of reducing native species richness, altering ecosystem function and degrading habitat quality for native fauna (Werner and Zedler 2002, Maurer et al. 2003, Zedler and Kercher 2004, Perkins and Wilson 2005, Fierke and Kauffman 2006, Schooler et al. 2006, Schooler et al. 2008, Zedler 2009, Spyreas et al. 2010). To successfully restore riparian wetlands in the LCRB, reed canarygrass and other invasive plants need to be suppressed to recover native species biodiversity and natural ecosystem functions (Zedler 2000, Christy 2004, Jenkins 2005, Fierke and Kauffman 2006, Zedler 2009). Restoration techniques used to control the spread of RCG include tilling, shading, herbicide application, pathogenic biocontrol, removal of topsoil (scrapping), flooding, mowing, and grazing (Kilbride and Paveglio 1999, Stannard and Crowder 2001, Hovick and Reinartz 2007, Jenkins et al. 2008, Zedler 2009, Healy and Zedler 2010). Management and eradication of RCG in wetlands has shown to be very expensive and challenging, with few successful eradications documented (Kilbride and Paveglio 1999, Jenkins 2005, Zedler 2009, Healy and Zedler 2010). RCG's ubiquity in the LCRB is a serious threat to riparian restoration efforts. Continued

evaluation of these control techniques are needed to determine which management practices are most successful at suppressing RCG in the LCRB.

Livestock Grazing

Livestock grazing became common in the Columbia River Basin following the settlement of the west in the mid 1800's (Christy and Putera 1992, Harrison 2010).

Today livestock grazing, primarily cattle, continues to be a common land use throughout the basin (Christy and Putera 1992, Leonard and Karl 1995, Harrison 2010). By the late 1800's the ecological impacts of uncontrolled and/or poorly managed grazing on riparian and floodplain habitats became noticeable throughout the region (Harrison 2010).

Impacts of livestock grazing have now become widely studied throughout the world (Table 1).

Livestock Grazing Literature Reviews			
Review Focus	Review Region	Year	Citation
Impacts in Riparian Areas	Western North America	2002	Sarr 2002
Impacts in Riparian Areas	Western North America	1999	Belsky et al. 1999
Impacts in Riparian Areas	Western North America	1998	Larsen et al. 1998
Impacts in Riparian Areas	Western North America	1997	Mosley et al. 1997
Impacts in Riparian Areas	Western North America	1994	Elmore and Kauffman 1994
Impacts in Riparian Areas	Western North America	1984	Kauffman and Krueger 1984
Impacts in Wetlands	World Wide	2004	Reeves and Champion 2004
Impacts in Wetlands	World Wide	1999	van Oene et al. 1999
Wetland Restoration	European Fens	2006	Middleton et al. 2006
Riparian Restoration	North America	2005	Medina et al. 2005
General Impacts	Arid Ecosystems of North America	2000	Jones 2000
General Impacts	Southwestern United States	2006	Milchunas 2006
General Impacts	Western North America	2002	Krueger et al. 2002
General Impacts	Western North America	1994	Fleischner 1994
General Impacts	Western North America	1994	Vavra et al. 1994
General Impacts	World Wide	1995	Trimble and Mendel 1995
General Impacts	World Wide	1993	Milchunas and Lauenroth 1993
General Impacts	World Wide	1992	Menke and Bradford 1992
General Impacts	World Wide	2007	Osmond et al. 2007
General Impacts	World Wide	2001	Hay and Kicklighter 2001
General Impacts	World Wide	2001	Kauffman and Pyke 2001
Impacts in Grasslands	World Wide	1998	Milchunas et al. 1988
Impacts on Soil Properties	World Wide	2001	Greenwood and Mckenzie 2001
Grazing for Weed Control	World Wide	2006	Launchbaugh 2006
Grazing for Weed Control	World Wide	1996	Popay and Field 1996
Weed Introduction	Western North America	2007	Vavra et al. 2007
Weed Introduction	Western North America	2000	DiTomaso JM 2000
Weed Introduction	Western North America	1999	Olsen 1999

Table 1: Livestock Grazing Literature Reviews. Literature reviews conducted on the impacts of livestock grazing are sorted by review focus, region and date.

These studies have shown that livestock grazing can cause ecosystem disturbance through excessive vegetation removal, physical damage to plants and soil (and topography) through trampling, and an alteration in riverine nutrient dynamics through defecation in and/or near streams and wetlands (e.g., Menke and Bradford 1992, Milchunas and Lauenroth 1993, Fleischner 1994, Belsky et al. 1999, DiTomaso 2000, Jones 2000, Hay and Kicklighter 2001, Kauffman and Pyke 2001, Clary and Kinney

2002, Krueger and Sanderson 2002, Kauffman et al. 2004, Reeves and Champion 2004, Garibaldi et al. 2007, Osmond et al. 2007, Sharrow 2007, Bartuszevige and Endress 2008). Through these disturbances, livestock grazing can degrade riparian habitats and alter riparian plant community composition in the following ways:, by decreasing density and biomass of individual plant species, by decreasing canopy cover, by increasing exotic species richness, by decreasing native species richness, by shifting ecological plant succession, by altering riparian geomorphology, by degrading water quality and by altering nutrient cycling (e.g., Menke and Bradford 1992, Milchunas and Lauenroth 1993, Fleischner 1994, Trimble and Mendel 1995, Proulx and Mazumber 1998, Belsky et al. 1999, DiTomaso 2000, Jones 2000, Hay and Kicklighter 2001, Kauffman and Pyke 2001, Clary and Kinney 2002, Krueger and Sanderson 2002, Kauffman et al. 2004, Reeves and Champion 2004, Garibaldi et al. 2007, Osmond et al. 2007, Sharrow 2007, Bartuszevige and Endress 2008). Livestock trampling and herbage removal alters riparian geomorphology and water quality by causing soil erosion, soil compaction and increasing soil bulk density, which collectively may decrease water infiltration and increase runoff and sedimentation into the stream (e.g., Menke and Bradford 1992, Milchunas and Lauenroth 1993, Fleischner 1994, Trimble and Mendel 1995, Belsky et al. 1999, DiTomaso 2000, Jones 2000, Hay and Kicklighter 2001, Kauffman and Pyke 2001, Clary and Kinney 2002, Krueger and Sanderson 2002, Kauffman et al. 2004, Reeves and Champion 2004, Garibaldi et al. 2007, Osmond et al. 2007, Sharrow 2007, Bartuszevige and Endress 2008). These impacts are not consistently present in all grazed riparian wetlands, and past studies have documented extensive variability in the effects of

livestock grazing on riparian habitats (e.g., Kauffman and Krueger 1984, Menke and Bradford 1992, Milchunas and Lauenroth 1993, Vavra et al. 1994, Larsen et al. 1998, Belsky et al. 1999, Jones 2000, Clary and Kinney 2002, Pyke and Marty 2005, Osmond et al. 2007). The extent and types of impacts appear to be dependent on livestock type, grazing season, grazing intensity, local topography, vegetation, climate and hydrology (e.g., Kauffman and Krueger 1984, Menke and Bradford 1992, Milchunas and Lauenroth 1993, Vavra et al. 1994, Green and Kauffman 1995, Mosley et al. 1997, Proulx and Mazumber 1998, Belsky et al. 1999, Stohlgern et al. 1999, van Oene et al. 1999, Clary and Kinney 2002, Lucas et al. 2004, Reeves and Champion 2004, Austin et al. 2007, Osmond et al. 2007, Bartuszevige and Endress 2008). Although past studies provide a wealth of information regarding livestock grazing in rangelands and riparian wetlands, it is difficult to extrapolate their findings to the complex environment of the LCRB (Belsky et al. 1999). As summarized in Figure 1, the varying characteristics of local environment, degree of disturbance and grazing all interact to help determine general impacts of livestock grazing on riparian wetland plant communities in the LCRB.

Figure 1: A Conceptual Model Showing Factors Involved in Determining Livestock Grazing Impacts on Riparian Wetlands in the Lower Columbia River Basin (LCRB). Local environmental factors and specific characteristics of local grazing practices influence how and which wetland characteristics will be altered in the grazed system. The resulting wetland environmental characteristics influence the ecological health and habitat quality of that wetland (Kauffman and Krueger 1984, Proulx and Mazumber 1998, Wissmar and Beschta 1998, Christy and Putera 1992, Menke and Bradford 1992, Milchunas and Lauenroth 1993, Vavra et al. 1994, Green and Kauffman 1995, Belsky et al. 1999, van Oene et al. 1999, Reeves and Champion 2004, Clary and Kinney 2002, Austin et al. 2007, Osmond et al. 2007, Bartuszevige and Endress 2008, Zedler 2009).

Excluding livestock from riparian wetlands is a passive restoration technique used all throughout the LCRB. Livestock exclusion has been accepted as an effective riparian restoration method throughout the West (Elmore and Kauffman 1994). However the success of livestock exclusion has not been well documented in the LCRB where invasive species and hydrologic regulation create a complex restoration environment.

Understanding how livestock grazing and exclusion are affecting the plant communities in the LCRB is important to their future restoration and management.

Livestock Grazing and Restoration in the LCRB

In the LCRB, riparian wetlands are commonly invaded with RCG. Historically RCG was used as grazing forage and commonly planted in rangeland wet meadows throughout the region, which may be why it is so ubiquitous (Galatowitsch et al. 1999, Stanndard and Crowder 2001, Maurer et al. 2003). Livestock successfully feed on RCG throughout the growing season and prefer young RCG stands and re-growth (Decker et al. 1969). As RCG ages the alkaloid content increases, decreasing the palatability of the forage toward the end of the growing season, which is true of most grass species (Decker et al. 1969, Darambazar 2003). However, livestock preference of RCG over native species has not been well documented but observational evidence indicates that the livestock will eat what is palatable, abundant and easily accessible (Rita Beaston, *personal communication*, Heady 1964). In areas with RCG present, it will be grazed along with other palatable plants in the vicinity, likely in proportion to its abundance. Selectivity and patchiness of grazing are affected by herbivore morphology and behavior (Hay and Kicklighter 2001). Cattle are less selective than other grazers because their mouthpart morphology makes it difficult to graze individual plants or plant parts (Hay and Kicklighter 2001). Cattle have been shown to selectively forage on grasses (native and non-native) over forbs and shrubs (Holechek 1984, Kie and Boroki 1996, Reeves and Champion 2004, Beck and Peek 2005, Osmond et al. 2007). Grazed vegetation

community characteristics will vary depending on grazer type, grazing intensity and other environmental characteristics (Holechek et al. 2003, Reeves and Champion 2004, Launchbaugh 2006, Figure 1).

Grazing intensity is commonly defined by the number of animal units released to graze an area and the amount of forage (i.e. carrying capacity) that area is expected to provide (Holechek et al. 2003). An animal unit (AU) is a standardized measure of animals used in rangeland management with a 1,000 lb beef cow equating to 1 AU. One AU is expected to require (ingest) 26 lbs of forage a day. Animal Unit Months (AUM) are used to define a rangeland's carrying capacity. One AUM is defined as the amount of forage required to support 1 AU for 1 month, approximately 780 lbs of forage (dry weight) for a 1,000 lb beef cow. A rangeland's AUM capacity is approximated based on the amount of forage available, the number of AU's to be grazed and the duration of grazing desired. The grazing site in this study was estimated in 2005 to provide 1308 AUM's which equates to enough forage for 218 AU's (i.e., 218 cows) over a 6 month period, which would be considered a moderate grazing regime (Mehlhoff 2005). However in 2009, the year of the study, this rangeland was heavily grazed by 250 cattle for 6 months removing approximately 1500 AUM's of forage (9.84 AUM/ha/yr).

Heavy grazing and overgrazing can occur when more forage is removed than determined available for a moderate grazing regime. Overgrazing can push a vegetation community to support weedy unpalatable native and exotic plants, and severe overgrazing can lead to sparse vegetation coverage due to soil degradation (Holechek et al. 2003). While overgrazing can severely degrade vegetation community characteristics,

low to moderate grazing intensities have been found to positively influence vegetation community biodiversity (e.g., Milchunas et al. 1988, Proulx and Mazumber 1998, Paine and Ribic 2002, Krzic et al. 2004, Reeves and Champion 2004, Marty 2005, Launchbaugh 2006).

In a review of 15 wetland grazing studies that measured change in species richness under different grazing intensities, including the present study, moderate to high (2.9-9.84 AUM/ha/yr) grazing intensities were found to increase in species richness and low (<1.44 AUM/ha/yr) grazing intensities were found to result in no difference or a decrease in species richness when compared to ungrazed exclosures (Table 2, Appendix B). During a 10 year grazing (moderate to high intensity, 7.7-9.2 AUM/ha/yr) exclusion study in Eastern Oregon, Green and Kauffman (1995) found that species richness and evenness were consistently higher in grazed vs. excluded wet and dry meadow study areas. In a 28 year grazing exclusion study, conducted in Australia, researchers found that the abundance of an invasive forage grass (similar to RCG), *Cenchrus Ciliaris*, increased in cattle excluded study plots from <5% to >80% cover over the study period and was accompanied by a stark decrease in plant diversity (Clarke et al. 2005). In general, it has been well established that grazing can result in increased species richness when grazers reduce (or control) a dominant species, creating opportunities for new species to establish and compete (Bakker 1985, Noy-Meir et al. 1989, Buxton et al. 2001, Jutila 2001, Reeves and Champion 2004, Clarke et al 2005, Jutila 1997). However, caution should be used when generalizing impacts of grazing intensity across different habitat types. Vegetation community response to grazing intensity is dependent on a myriad of other

environmental and grazing characteristics (as were summarized in Figure 1) which can vary between different wetland types and management techniques.

Summary Table of Wetland Species Richness Under Different Grazing Intensities

Wetland Type	Location	Grazing Season (Yearly)	Study Length (Years)	AUM/ha/yr	Grazing Intensity	Species Richness	Citation
Montane riparian areas	New Mexico	Varied (2-8 days)	2	0.22-1.44		*	Lucas et al. 2004
Rain-fed wetlands	Nebraska	Varied (1.6-0.67 months)	2	0.3-5.4		*	Hillhouse et. al 2010
Freshwater wetlands	Indiana	Year round	2	0.81 [^]		-	Kellogg and Bridgham 2004
Vernal pools	California	Varied (2-7 months)	3	0.83-2.9		+	Marty 2005
Riverine floodplain	New Zealand	Year round	4	1.56-2.4		+	Buxton et al. 2001
Montane riparian areas	British Columbia	June to July	10	3		+	Krzic et al. 2004
Delta meadow	Finland	Year round	2	3.6-20.4		+	Jutila 1997, 1999
Seashore meadow	Finland	Year round	2	3.6-20.4		-	Jutila 1997, 1999
Riparian vegetation	Kansas	April - Nov	2	4.4		+	Hoover et al. 2001
Riparian pasture	Scotland	August - Sept	9	4.5-5		+	Humphrey and Patterson 2000
Mixed deep marsh	Idaho	Fall (Sept-Oct)	2	4.6-5		+	Austin et al. 2007
Mire (heathland)	United Kingdom	Varied (Summer, May-Oct)	1	6-414 [~]		+	Bullock and Pakeman 1996
Lowland floodplain	Argentina	Year round	13	6		+	Chaneton and Facelli 1991
Saltmarsh	Denmark	May-Sept/Oct	9	7.15-9.35		+	Bakker 1985
Moist & dry meadow	Oregon	Year round	10	7.7-9.2		+	Green and Kauffman 1995
Riparian area	Wisconsin		1	High		+	Paine and Ribic 2002
Riparian wetland	Oregon	May-Oct	1	9.84		+	This study

Table 2: Summary of Wetland Grazing Studies that Measured Species Richness Under Different Grazing Regimes/Intensities. '*' indicates no change in species richness with grazing, '-' indicates a decrease in species richness with grazing and a '+' indicates an increase in species richness with grazing. When possible grazing intensity has been converted to AUM/ha/yr of forage removed a year for comparison, this conversion is based on reported stocking rate and duration of grazing. The table is sorted by low to high minimum AUM/ha/yr (grazing intensity). All studies used cattle for grazing except: ^ deer, ~ combination of sheep, horses and cattle. A detailed summary of reported grazing regimes and AUM/ha/yr calculations can be found in Appendix B.

Exclusion of livestock releases RCG from herbivory, which is a significant factor limiting its dominance in riparian wetland environments (enemy release hypothesis, Huston 1979, Hobbs and Huenneke 1992, Martin and Chambers 2001, Paine and Ribic 2002, Carpenter and Cappuccino 2005). Livestock grazing in RCG invaded wetlands has been shown to reduce RCG biomass, decreasing its ability to outcompete other wetland plant species (Tesauro 2001, Paine and Ribic 2002, Tesauro and Ehrenfeld 2007, Hillhouse et. al 2010). In a controlled greenhouse study, Fraser and Keddy (2005) found that clipping (simulated grazing) reduced RCG's ability to compete with other less aggressive plants and resulted in an increase in the biomass of these competing plants. In a study examining forage grass response to high grazing pressure, Brummer and Moore 2000 found that RCG abundance was reduced by 90% after 2 years of continuous cattle grazing. In Wisconsin, Paine and Ribic (2002) found that grazed riparian buffers had a lower RCG abundance and higher species diversity than ungrazed buffers. These studies support the possibility that livestock grazing in RCG invaded riparian wetlands could release other plant species from resource competition and increase plant community biodiversity (Figure 2). The conceptual model, Figure 2, based on the intermediate disturbance hypothesis and competitive exclusion principle, shows the possible response of a riparian wetland plant community (species richness and invasive plant dominance) to different intensities, frequencies and duration of time post disturbance. This thesis uniquely contributes the hypothesized response of invasive plant dominance (dashed line in Figure 2) to this diversity vs. disturbance relationship. Without intermediate levels (and/or frequencies) of disturbance (such as grazing), plant community diversity will

decrease and if present, invasive species (such as RCG) will become dominant (Figure 2). In areas like the LCRB, where natural disturbance regimes have been altered and invasive species like RCG are prevalent, grazing disturbance and herbivory could possibly be used to help control invasive species dominance and maintain wetland diversity (Figure 2). Developing a better understanding of how RCG and other plants respond to livestock grazing and livestock exclusion restoration efforts in the LCRB is important for the improvement of riparian wetland restoration efforts throughout the region.

Figure 2: Conceptual Figure of Expected Plant Community Diversity and Invasive Species Dominance in Response to Different Disturbance Regimes (such as grazing). This concept is based on the intermediate disturbance hypothesis and the competitive exclusion principle (Grime 1973, Connell 1978, Milchunas et al. 1988, Kercher et al. 2007, Hughes 2010). This thesis uniquely contributes the hypothesized response of invasive plant dominance (dashed line) to the diversity vs. disturbance relationship.

In this study, vegetation cover (native, non-native and invasive species) and species diversity are compared among three LCRB riparian wetlands: one with ongoing

grazing and two that have been excluded from grazing, one for three years and the other for thirteen years. The relationship between grazing and grazing exclusion and the corresponding riparian wetland plant community characteristics are also compared. This study is a step in understanding the effectiveness of livestock exclusion as a passive restoration technique in the LCRB. Results from this study may inform land managers and restoration practitioners regarding the restoration and management of grazed riparian wetlands in the LCRB.

OBJECTIVES AND HYPOTHESES

This Study: Research Objectives & Hypotheses

My study objectives were to examine riparian wetland plant community characteristics along a succession gradient of livestock exclusion, to evaluate the relative roles of grazing (cessation of grazing) in determining plant species richness and dominance, and to develop a general understanding of the effects of grazing and grazing exclusion on plant diversity in riparian wetlands of the LCRB.

My hypotheses were:

1. The native plant species richness will be lower in excluded riparian wetlands than in the grazed wetland.
2. The grazed riparian wetland will have higher native and non-native species richness than the excluded riparian wetlands.

I hypothesized that native plant species richness would be lower in the excluded wetlands than in the grazed wetland due to the competitive exclusion from an increase in non-native plant dominance in the absence of grazing (Grime 1973, Harper 1977, Hobbs and Huenneke 1992, Vavra et al. 1994, Green and Kauffman 1995, Figure 2). I hypothesized that the grazed riparian wetland would have higher native and non-native species richness because of reduced competition due to herbivory (biomass removal). According to the intermediate disturbance hypothesis, grazing will increase species richness and possibly facilitate non-native plant establishment (Grime 1973, Connell 1978, Fox and Connell 1979, Mack 1981, Milchunas et al. 1988, D'Antonio and Vitousek 1992, Hobbs and Huenneke 1992, Fleischner 1994, Vavra et al. 1994, Green and

Kauffman 1995, Bartuszevige and Endress 2008, Figure 2). Through these observations I was also able to document RCG dominance in the riparian wetlands. The riparian wetland habitats of the LCRB are essential to the success and survival of many terrestrial and aquatic species, and understanding how these critical habitats are affected by livestock grazing and exclusion is fundamental to the protection, management and restoration of these wetlands.

METHODS

Study Approach

The overall approach of this study was to compare the plant communities in similar riparian wetlands that have different grazing histories. Working with the Scappoose Bay Watershed Council on their restoration projects in the Lower Columbia River Basin (LCRB), access to private and state grazing lands in the Scappoose Bay Watershed was provided. The watershed council had been excluding livestock from riparian wetland restoration projects on the private grazing land for three years as of 2009. This property is adjacent to Oregon State Park and Recreation (OSPR) grazing lands. The OSPR lands are very similar in riparian wetland composition and have not been excluded from grazing. A complex of riparian wetlands similar to those on the private and OSPR lands (located approximately 10 miles south of these sites) is owned by Metro, Portland's Regional Government, and has been excluded from grazing for 13 years in 2009.

The overall approach to capturing riparian wetland plant community characteristics in these riparian wetlands was to establish 6 transects positioned perpendicularly to the shoreline (or high water line) of each riparian wetland. Vegetation cover was measured along these transects during low water in August and September of 2009. Soil characteristics such as bulk density, texture and percent organic matter, as well as site topography, were also evaluated to determine other possible factors influencing the wetland vegetation communities.

Site Description

The Lower Columbia River Basin (LCRB) is a 146 mile portion of the Columbia River that extends from the Pacific Ocean to the Bonneville Dam. This study was conducted on three sites located in the LCRB along the Multnomah Channel, Oregon USA (Figure 3). The Multnomah Channel is a freshwater tidally influence branch of the Willamette River and is located a few miles upstream of the Willamette's convergence with the Columbia River. The study sites have riparian wetland hydrology, similar topography, and are located along a 10 mile section of the Multnomah Channel near Scappoose, Oregon. The climate of this area can be described as modified west coast marine, with relatively wet, mild winters and clear dry summers (Johnson 1987). Mean annual precipitation for the study sites is 950 mm with the majority falling during the winter months as rain.

Figure 3: Map of Study Sites. Study sites located along an approximately 10 mile stretch of the Multnomah Channel in the LCRB.

Sites

Three riparian wetland sites were used in this study representing currently grazed riparian wetlands, recently excluded riparian wetlands and long-term excluded riparian wetlands. Each site is under different ownership but all have been historically grazed by cattle with similar historic grazing regimes. The three sites will be referred to as current grazing (CG), short-term exclusion (STE) and long-term exclusion (LTE) riparian wetlands (Figure 3). Each riparian wetland area sampled is approximately 10 ha in size. The current (as of 2009) grazing practices on the CG site are heavy continuous grazing with cattle present from April 15th to September 15th (Mehlhoff 2005). The grazed study area is part of a larger range riparian wetland complex composing approximately 152.45 hectares of range estimated to provide 1308 AUM's of forage (Mehlhoff 2005). In 2009

this area was grazed by approximately 250 cattle, 9.84 AUM/ha/yr (Rory, Cattle Operator August 2009, *personal communication*) which would require 1500 AUM's of forage for the grazing period. The site was therefore grazed by approximately 300 AUM's (234,00 lbs of forage) over the prescribed grazing intensity of 1308 AUM's during the 2009 season. The STE site has had cattle exclusion fencing in place for 3 years and the LTE has had no grazing for 13 years. The excluded riparian wetlands are being passively restored with no active vegetation or hydrologic management. Differences in riparian wetland vegetation among sites are assumed to be due to cattle grazing or cessation of grazing. Native ungulate grazing is minimal in these areas and not considered a significant factor in this study (Rita Beaston, *personal communication*). These assumptions are reasonable considering the similar historic land use and environmental characteristics among sites (Sarr 2002, Kauffman et al. 2004). Wetland site elevation, soil bulk density, soil texture and percent soil organic matter were measured to characterize the riparian wetlands and identify other possible underlying influences to riparian wetland vegetation composition.

Vegetation Survey

Transect Point Selection

Using an aerial photograph of the riparian wetlands at each site, taken in June 2005, and a 20m by 20m grid overlay, random transect starting point locations were determined for the vegetation surveys (Jenkins 2005, Figure 4). The overlay grid was numbered along the perimeter of each wetland and the Microsoft Excel Random Number Generator function was used to select each transect's starting point location. When a

transect point was determined inaccessible it was rejected and the next randomized point was used. Each site had 6 transects ranging from 45 to 60 meters in length. Sampling efforts (number of transects) at each site were determined according to the number of new species found along each transect surveyed. The number of new plant species found at each location markedly decreased after the first few transect surveys at each site (Figure 5). After surveying the 6th transect at each site it was determined that the plant communities were adequately represented by 6 transects and further surveying was not necessary for the purpose of this study. A study of 130 lake and wetland sites in Iowa determined that 6 vegetation transects resulted in a 90% probability of capturing 100% of the plant species present in a 10 ha macrophyte study area (Quist et al. 2007).

Figure 4: Map of Transect Placement and Estimated High Water Line in 2009.

Figure 5: Number of New Plant Species Found Along Each Transect. The number of new species found markedly decreased with each additional transect surveyed, with very few new species being found after the 3rd transect surveyed. No new species were found at the Long-term Exclusion Site by the 3rd and 5th transects and no new species were found at the Short-term exclusion site by the 5th transect.

Transect Point Location

The latitude and longitude of each transect starting point was identified using Google Earth and uploaded to a Garmin (eTrex HC series) GPS unit using GPS Utility Software Version 5.02. The Garmin GPS unit was then used to identify the location of each transect starting point in the field. All transect starting points were located to within 3.35 meters (11 feet). When locating the GPS position, for each point the number of satellites communicating with the GPS receiver ranged from 5 to 8, always with a Positional Dilution of Precision (PDOP) < 4. The PDOP is a measure of how accurate the

location identification is, based on the number and position of satellites used, with a PDOP of less than 4 considered excellent (Wing 2005).

Transect Vegetation Surveying Procedure

Vegetation surveys were conducted at the end of the growing season from August 5, 2009 through September 15, 2009. Sampling at this time allowed measurements of ecosystem conditions after cattle had utilized the area for the majority of the grazing season. This time frame for surveying was also the driest part of the year, allowing for the best access to the riparian wetland areas.

Transects were extended perpendicular to the riparian wetland shoreline from the estimated mean high water level into the riparian plant community. Transects ran into the riparian wetland plant community for 60 meters or to the end of the riparian vegetation.

In fenced riparian wetlands, transects began at the fence line (shore line) and then extended to the edge of the riparian plant community. The length of transects varied with topography and riparian plant community width and ranged from 45 to 60 meters.

Vegetation was surveyed along each established transect according to the line intercept method (Brower et al. 1997, Jenkins 2005). Plant percent cover (species of all heights, up to 2 meters) was recorded continuously within every 1 dm measure along each transect, and when possible plants were identified to species (Sharp 2002, Jenkins 2005).

Taxonomic guides to regional flora were consulted (Hitchcock and Cronquist 1973, Pojar and MacKinnon 1994, Guard 1995) to help with species identification and to determine native/non-native status of each plant species. Native, non-native and invasive status determination of each plant species was also identified using the online NRCS PLANTS

database (<http://plants.usda.gov>). Vegetation surveys were periodically cross-checked for quality assurance by a wetland botanist.

Diversity

Native and non-native plant species richness, diversity indices and relative cover were calculated for individual transects and overall for each site. Plant community diversity was determined by calculating the Shannon's Diversity index (H') (Shannon 1948, Shannon and Weaver 1949, Pielou 1975) and species richness along each transect. The Shannon's Diversity index (H') combines components of species richness and evenness into one measure of diversity (Odum 1971). Species evenness is defined as a measure of how equitably species abundances are distributed within a community (Hill 1973). The Shannon's index (H') was used because it provides an intermediate weighing of rare species compared to the Simpson's index that emphasizes species dominance (Peet 1974, Magurran 1988). The Shannon's Diversity index is increased by having either additional unique species, or greater species evenness. The relative abundance (p_i) for each species was calculated as the proportion of that species individual % cover to the total % cover of all species along the transect. These values were then used to calculate the diversity index (H') of each transect:

$$H' = - \sum_{i=1}^S \left(\frac{n_i}{N} \right) \ln \left(\frac{n_i}{N} \right) \text{ can be simplified to } H' = - \sum_{i=1}^S (p_i) \ln(p_i), n_i = \text{the \% cover}$$

(abundance) of individual species i ,

N = the total % cover of all individual species,

S = the total number of species (species richness) found on the transect, and

p_i = the relative abundance of each species in terms of relative % cover of each species (n_i/N).

The H' of each transect was then exponentially transformed by $\exp(H')$ to acquire the effective number of species of each transect for statistical comparison (Hill 1973, Peet 1974, Jost 2006). The effective number of species (also known as true diversity) acquired by exponentially transforming the diversity index represents the maximum number of equally common species that could be found under the given diversity index value (Jost 2006). When comparing diversity indices among several sites, it is easier to conceptualize differences between site diversity when the indices are transformed into the number of equally common species that is represented by the index values. This value is referred to as the true diversity throughout this thesis.

Species richness is defined as the total number of species in a given area and weighs all species equally (Ludwig and Renolds 1988, Chaneton and Facelli 1991). Percent relative cover (p_i) was calculated for all plant species along each transect to determine species dominance. Percent relative cover (p_i) was calculated by dividing the total percent cover of an individual plant species on a transect (n_i) by the total percent cover of all plant species found on that transect (N). The plant species with the highest relative percent cover along a transect was considered dominant.

Plant Community Similarity

Plant community similarity was calculated for individual transect and overall site comparisons using the Bray-Curtis Index of Similarity (BCI), also known as the index of dissimilarity or the Sørensen quantitative index (Magurran 1988). The plant community similarity BIC was calculated by comparing the differences in individual species abundance (and presences/absence) found along each transect both within and among sites. A BCI matrix of similarity was calculated by using this equation to compare all site transect vegetation composition using a similarity index

$$C_N = \frac{2jN}{(N_a + N_b)},$$

N_a = the total abundance of species in transect a,

N_b = the total abundance of species in transect b, and

$2jN$ = the sum of the lowest species abundance between a and b.

Here, C_N is the similarity index between transect a and transect b between 0 and 1, with 0 indicating the transects have no species in common and 1 indicating that the transects are identical in species composition. The C_N value can be multiplied by 100 to get percent similarity. Non-metric multidimensional scaling (NMDS) was used to rank similarities from the BIC data matrix and create an ordination plot of these similarity relationships (Magurran 2004). All BIC and NMDS calculations were computed using R 2.8.1.

Riparian Wetland Topography

Approach

The topography and hydrology of riparian wetlands are closely linked. The elevation or topography of a riparian wetland defines its hydrologic regime, including water depth during flood events, period of inundation, and soil moisture in drier sites and during drawdown events (Welch et al. 2006). In a study on wetland plant communities, elevation was also found to be an important factor in determining the dominant species in a given sample plot (Lenssen et al. 1999). The flooding gradient created by topographical characteristics of a riparian wetland results in distinct vegetation zones populated by particular groups based on species tolerances to flooding (Lenssen et al. 1999, Welch et al. 2006). Hydrology is a more precise way of defining a riparian wetland, but can be difficult and costly to estimate (Page et al. 2003). Due to the costs associated with monitoring riparian wetland hydrology, topography data were considered adequate for the purposes of this study. To address the possible influence of riparian wetland topography on differing plant community characteristics among sites the elevation characteristics of each site were compared. Ground surveying to collect elevation data is often very difficult or impossible because of the difficulty to access and work in many wetland sites (Page et al. 2003). LiDAR (Light Detection And Ranging) data are considered the best way to obtain high resolution elevation data for riparian wetlands because the low-lying terrain of mudflats and marshes do not lend themselves to easy or accurate ground surveying (Lefsky et al. 2002, Morris et al. 2005).

Digital Elevation Model

The digital elevation model (DEM) of the riparian wetland topography used in this analysis was created from LiDAR data obtained from the Scappoose Watershed Council and the Portland Metro Government. The LiDAR data from the Scappoose Watershed Council covered the STE and CG sites. These data were collected on September 24, 2004 between 10:00 am and 12:00 pm at 1100 meters above ground level (AGL). The data were acquired during a period of low water for the best possible wetland elevation data acquisition. All data were collected in a UTM Zone 10 NAD 83, NAVD88 Geoid03 projection with a 0.25 meter resolution and a 4 cm accuracy. The LiDAR data for the LTE site was acquired from the Portland Metro Government. These data were collected in 2002 with a 1 meter resolution. For the best comparison among sites the LiDAR data should be of similar quality and resolution. Unfortunately, acquiring new higher resolution LiDAR data for the LTE site was not possible for this study.

To process the LiDAR data, the original ASCII file listing the coordinates and elevation data were first converted to a text file and imported into a Microsoft Access database. The Access file was converted into an ESRI 3D LiDAR point shapefile in ArcGIS ArcMap with the *Add XY Data* tool. The LiDAR points were then clipped to the scale of the study area. The clipped LiDAR was converted to a Triangulated Irregular Network (TIN) surface using ArcGIS 3D Analyst. From there, the TIN surface was converted to a raster (DEM) surface with a 0.25 meter or 1 meter cell size.

Vegetation Transect Data Layer

A shapefile of the vegetation transects was created by importing the GPS point data as an excel table into ArcMap and using the *Add XY Data* tool to first create a point layer of the beginning and endpoints of the transects. This layer then was re-projected from the WGS 1984 coordinate system of the GPS points into NAD 1983 State Plane Oregon North. The 45 to 60 meter transect lines were created with the Editor Sketch tool to draw the lines and snap them to the endpoints. The transect lines were then converted into routes by using the *Create Routes* tool in the Linear Referencing tools of ArcToolbox to establish a measurement attribute with which to properly distribute the sample data points every 1 dm along the transects.

Sample Plot Points

The sample data points were associated with the transect lines through dynamic segmentation. Dynamic segmentation refers to the process of determining the location of events (vegetation data points) along a route (the transect line) (Chang 2009). An Excel table with all of the 1 dm data points was imported into ArcMap as an event table (Make Route Event Layer tool). This route event representing the vegetation sample points along each of the transects was used to determine the approximate elevation at each sample point, every 1dm along the transect line.

Extraction of Elevation Data

The ArcGIS 9.3 Extract Values to Points tool (Spatial Analyst toolbox, Extraction) was used to determine the elevation at each sample point along the transects. This determination required the use of the LiDAR derived DEM and the vegetation (route event) data layer. After this process was executed, the elevation at each sample point along the transect was added to the vegetation data for site comparison and analysis. Transect slope was calculated from the elevation data by subtracting the beginning elevation with the transect end elevation and dividing this by the length of the transect in between these points. In cases where elevation data were not available for the beginning or ending point, the next available data point was used and the distance between these points was used to divide the difference in elevation in calculating the slope. Elevation range of each transect was determined by locating the highest and lowest elevation points along the transect.

Soil Survey

Approach

In this study, surface soil texture, bulk density, and organic matter (OM) percentage were measured to investigate soil properties at each riparian wetland site. These physical soil properties were evaluated because they can affect nutrient cycling, infiltration, below ground biological activity, and may ultimately influence plant growth and distribution (Albrecht 1971, Dwire et al. 2000, Sharp 2002). Therefore differences in soil characteristics found among sites could be assessed when evaluating the differing

riparian wetlands plant community compositions (Kozlowski 1999, Sharp 2002, Castellano and Valone 2007).

Surface soil samples (0–10 cm depth) for soil bulk density were collected along each of the study transects (1 soil sample per transect or n=6 soil samples per site). Soil from the bulk density sample was also used for the soil texture and the OM analysis. The Microsoft Excel Random Number Generator function was used to select the soil sample locations along each transect. If the location identified could not be penetrated by the bulk density sampler, the next random location along the transect was used. Soil samples were collected prior to vegetation surveying to avoid disturbing the soil prior to collection, while care was taken to insure plant distribution was not altered. Soil samples were analyzed according to the techniques described in Kalra and Maynard's Methods manual for forest soil and plant analysis (Kalra and Maynard 1991). Field and lab replicates, performed during bulk density sampling and soil texture and percent organic matter analysis, showed these methods to be accurate and reproducible.

Soil Bulk Density

Soil bulk density was measured via extraction of a core of known volume (261.09 cm³) (Kalra and Maynard 1991). All live vegetation and litter at each soil sample was removed to reveal the mineral soil surface prior to collecting the soil core. Soil cores were dried at 105 °C in a drying oven for 48 hours to achieve a consistent dry weight. Soil cores were then weighed on an analytical balance and bulk density was calculated as soil dry weight (g) divided by total core volume (cm³).

Soil Organic Matter

Soil organic matter (OM) was calculated by loss-on-ignition method (Kalra and Maynard 1991). This method uses high heat (375°C) to oxidize the organic matter in the soil sample; the organic matter content is then calculated through soil sample weight loss. Dried bulk density samples were ground and mixed thoroughly. Soil from these samples was then used for both the organic matter and soil texture analysis. Porcelain crucibles were prepared by heating to 375°C in a muffle furnace for 1 hour, then cooling to 150°C and placing the crucibles in a desiccator for 30 minutes. The weight of each crucible was then recorded. Soil samples (5 g) were then placed in each crucible and heated to 375°C in a muffle furnace for approximately 16 hours. The soil samples were then cooled to 150°C, placed in a desiccator for 30 minutes, and re-weighed to the nearest milligram. The percent of weight loss of each soil sample was then calculated to determine the percent organic matter of each sample.

Soil Texture Analysis

Soil texture analysis was performed on the dried and ground bulk density samples. Gravel (>2 mm) was separated by dry sieving each sample prior to hydrometer analysis. Bouyoucos hydrometer analysis, without pretreatment, was used to determine the silt (0.002-0.05 mm), clay (< .002 mm) and sand (0.5-2 mm) fractions (Kalra and Maynard 1991). The bouyoucos hydrometer method uses the sedimentation principle based on Stokes' law, relating the size of the soil particle to the rate at which it settles out

of the water column. As soil particles of different sizes settle out of the water column at different rates, the hydrometer measures a change in buoyancy of the water column. Each oven dried soil sample (50 g) was transferred into a dispersion cup with 400 ml of distilled water and 50 ml of calgon solution and then mixed for 15 minutes. Each sample was then transferred into a sedimentation cylinder; distilled water was then added to the 1 liter mark. An additional sedimentation cylinder was prepared as a blank with 50 ml of Calgon solution and 950 ml of distilled water. Soil samples were then covered and left to stand overnight to equilibrate to room temperature. After equilibrating, the soil samples were mixed well with a plunger and a hydrometer was inserted. Hydrometer and temperature readings were then taken 40 seconds and 2 hours after mixing. These readings were used to determine the percent of clay, silt and sand in the soil samples.

Statistical Methods

Differences in Vegetation Composition among Sites

Cover data, percent cover of all taxa every dm, for each transect were entered into an Excel spreadsheet. These data were then used to calculate the total (absolute) percent cover for each taxon recorded along each transect. These cover data were then summed to calculate the total cover recorded (for all taxa) along each transect. The relative cover of each taxon was then calculated by dividing the total percent cover of each taxon along the transect by the total cover of all taxa recorded along that transect. Relative cover of each species was used to sort the taxa by highest to lowest abundance and determine dominance for each transect, with higher abundance indicating greater dominance

(Lundholm and Larson 2004). Species richness was determined by summing the total number of species found on each transect and site species richness was determined by summing all unique species found on each transect within the site (only counting each species once). The species richness data, individual taxon data and total transect cover data were then used to calculate the Shannon Diversity index and the Bray-Curtis index of similarity of each transect and overall for each site. Relative cover and species richness of native species, non-native species and all species, as well as the indices data, were used to compare vegetation communities among all sites.

Prior to statistical analysis, the distributions of the independent summarized transect data were examined. Nonparametric statistical analysis was conducted because the data violated the parametric statistical assumptions of normality before and after data transformations (with various transformations attempted). The Kruskal–Wallis one-way nonparametric analysis of variance by ranks was used to detect significant ($p < 0.05$) differences in relative cover of taxa, species richness and diversity indices among the three study sites (Kruskal and Wallis 1952). When significant differences were detected, the nonparametric Mann–Whitney U test (also known as the Wilcoxon matched pairs signed ranks test) was used to determine significant differences among sites (Wilcoxon 1945, Mann and Whitney 1947). These analysis methods test whether the independent samples (observations) from each site come from the same distributions. The null hypothesis for both of these tests was that the differences among site vegetation data were equal to zero. Because three Mann-Whitney U tests would be performed for each significant Kruskal Wallis test, a Bonferroni correction for multiple comparisons was

used to set a significance level for this test (Abdi 2007). The Bonferroni correction was applied by dividing the standard significance level of $p\text{-value} < 0.05$ by three (for three comparisons) which gives a new significance level of $p\text{-value} < 0.0167$ (Abdi 2007). This new significance level was used for evaluating all Man-Whitney U test comparisons among sites and allowed control for a change in family-wise error (Abdi 2007). The non-parametric multivariate analysis of similarity (ANOSIM) was used to identify significant differences in vegetation community composition among the sites based on the calculated BCI of similarity (Clarke 1993). The ANOSIM test was first conducted on all of the site data to identify if there was a significant difference in community composition among the three sites with a significance level of $p\text{-value} < 0.05$. Then three ANOSIM pairwise tests were conducted to identify which sites were significantly different from one another. The Bonferroni correction for three comparisons was used to set the significance level for these comparisons ($p\text{-value} < 0.0167$). All vegetation statistical analysis was conducted with R 2.8.1.

Reed canarygrass Abundance and Plant Community Diversity

After summarizing the vegetation data from each site, the Spearman's Rho non-parametric correlation coefficient was used to identify significant ($p\text{-value} < 0.05$) relationships between species richness, diversity and reed canarygrass abundance for each site (transects=6) and for all the sites (3 sites, transects=18). This analysis was conducted to identify if there were any larger trends between plant community

composition and reed canarygrass abundance throughout the study region. All correlation analysis was conducted with R 2.8.1.

Differences in Soil Characteristics among Sites

Lab results from the soil analysis were also recorded in an Excel spreadsheet. Soil sample bulk density, texture composition and percent organic matter were compared among sites. Nonparametric statistical analysis was conducted because the data violated the parametric statistical assumptions of normality before and after data transformations. A Kruskal–Wallis one-way nonparametric analysis of variance was used to detect significant ($p < 0.05$) differences in soil characteristics among the three study sites (Kruskal and Wallis 1952). When significant differences were detected, the nonparametric Mann–Whitney U test was used to determine significant differences ($p < 0.0167$) among sites (Mann and Whitney 1947). All soil statistical analysis was conducted with R 2.8.1.

RESULTS

Site Vegetation Survey

A total of 58 plant species were identified among all three study sites: 27 native, 27 non-native and 4 species of unknown origin (Figure 6). A comprehensive list of the Latin name, common name and native or non-native origin of every plant species identified can be found in Appendix A. Among the sites, more native and non-native species were found on the CG site (Figure 7, Table 3 & 4). The greatest non-native cover was found on the LTE site and the greatest native cover was found on the STE site (Table 4). A comparative summary of each site's vegetation community characteristics is given in Tables 3 and 4.

Site Total	CG	STE	LTE	Total
Species Richness	44	25	20	58
Native Species Richness	19	11	11	27
Non-Native Species Richness	22	12	9	27
Unidentifiable Species	3	2	0	4
Site Shannon Diversity Index	1.7	1.5	0.26	
exp(Shannon Diversity Index)	5.2	4.5	1.3	

Table 3: Summary of Total Site Vegetation Characteristics. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion).

Vegetation Characteristics	Sites		
Average Species Richness	CG	STE	LTE
Native Species	10.2(±1.3)*	6.3(±1.4)	2.8(±3.3)
Non-Native Species	12.0(±3.6)*	5.0(±1.7)	2.7(±2.7)
All Species	23.3(±3.9)*	12.0(±3.4)	5.5(±5.7)
Average Relative Cover (%)			
Native Species	23.2(±8.1)^	51.6(±9.5)^	4.2(±4.9)^
Non-Native Species	65.7(±2.6)^	47.9(±10.6)^	95.9(±4.9)^
% <i>Phalaris arundinacea</i> (Non-Native)	52.7(±10.2)	43.0(±9.9)	95.2(±10.2)~
Average Species Diversity (H') (Shannon Diversity Index)			
All Species	1.5(±0.4)	1.4(±0.1)	0.21(±0.3)+
Native Species	0.56(±0.1)^	0.83(±0.1)^	0.13(±0.2)^
Non-Native Species	0.75(±0.2)	0.53(±0.1)	0.08(±0.1)+
Average True Species Diversity exp(Shannon Diversity Index)			
All Species	4.8(±1.8)	4.0(±0.6)	1.3(±0.4)+
Native Species	1.8(±0.2)^	2.3(±0.2)^	1.2(±0.2)^
Non-Native Species	2.2(±0.5)	1.7(±0.2)	1.1(±0.1)+

Table 4: Summary of Site Average (±1 Standard Deviation) Vegetation Characteristics. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). *CG site characteristic is significantly greater than both the LTE and STE sites, ^ all sites are significantly different from each other, ~LTE site is significantly greater than both the CG and STE sites, and + LTE site is significantly lower than both the CG and STE sites (p<0.0167 is considered significant for multiple Mann-Whitney U Tests).

* CG significantly different $p < 0.0167$ (Total, Native and Non-native Species Richness) from STE and LTE

Figure 6: Comparison of Total Species Richness among Study Sites. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). The CG site had significantly ($p < 0.0167$) greater total, native, and non-native species richness than both the STE and LTE sites.

* CG significantly different $p < 0.0167$ (Native and Non-native Species Richness) from STE and LTE

Figure 7: Comparison of Average (± 1 Standard Deviation) Species Richness among Study Sites. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). The CG site had significantly ($p < 0.0167$) greater native and non-native species richness than both the STE and LTE sites.

Species Richness

A total of 44 species were identified on the current grazing (CG) site: 19 native, 22 non-native and 3 unknown species (Figure 6, Tables 3 & 4). A total of 25 species were identified at the short-term exclusion (STE) site: 11 native, 12 non-native, and 2 unknown species. The lowest number of total species ($n=20$) were identified at the long-term exclusion (LTE) site: 11 native and 9 non-native. The CG site had significantly ($p < 0.0167$) greater total species richness (average of 23.3 species per transect), native (average of 10.2 species per transect) and non-native species richness (average of 12 species per transect) than both the STE and the LTE sites (Table 4, Figure 7). Average

species richness did not differ significantly between the STE and LTE sites for native (STE 6.3; LTE 2.8, p-value= 0.076), non-native (STE 5; LTE 2.7, p-value= 0.063) or total species combined (STE 12; LTE 5.5, p-value= 0.063) (Table 4, Figure 7).

Of the 58 plant species identified, only 6 species (3 native, 3 non-native) were common on all three sites and 30 species (17 native, 13 non-native) were found to be unique to one site (Table 5, Figure 8). In total, 17 species (9 native, 8 non-native) were found only on the CG site, 5 species (3 native, 2 non-native) were found only on the STE site and 8 species (5 native, 3 non-native) were found only on the LTE site (Figure 8). The CG and STE sites had 12 species (5 native, 7 non-native) exclusively in common and the CG and LTE had 6 species (3 native, 3 non-native) exclusively in common between them (Tables 5, Figure 8). The STE and LTE site had no species exclusively in common between them (Figure 8).

Plant Species Common Among All Study Sites (Average Relative % Cover)					
Latin Name	Common Name	Status	CG	STE	LTE
<i>Phalaris arundinacea</i>	Reed Canarygrass	NN	52.72	43.03	95.16
<i>Polygonum persicaria</i>	Water pepper	NN	6.38	1.48	0.0003
<i>Ranunculus repens</i>	Creeping buttercup	NN	0.012	0.006	0.005
<i>Ludwigia palustris</i>	Water purslane	NA	15.21	0.376	0.455
<i>Eleocharis palustris</i>	Creeping spikerush	NA	1.05	13.89	0.02
<i>Fraxinus latifolia</i>	Oregon ash	NA	0.055	0.893	0.003

**Plant Species Only Common between the Current Grazing (CG) and
Short-term Exclusion (STE) Sites (Average Relative % Cover)**

Latin Name	Common Name	Status	CG	STE	LTE
<i>Sagittaria latifolia</i>	Broad leaf wapato	NA	1.03	31.24	NC
<i>Carex obnupta</i>	Slough Sedge	NA	0.25	1.65	NC
<i>Equisetum arvense</i>	Horsetail	NA	1.50E-03	0.22	NC
<i>Prunella vulgaris</i>	Self heal	NA	0.05	0.11	NC
family Hypnaceae	Moss	NA	1.6	0.49	NC
<i>Lysimachia nummularia</i>	Moneywort	NN	2.42	2.93	NC
<i>Rubus armeniacus</i>	Himalayan blackberry	NN	0.51	0.12	NC
<i>Plantago major</i> L.	Common plantain	NN	0.12	0.08	NC
<i>Trifolium</i> sp.	Clover	NN	0.79	0.07	NC
<i>Setaria viridis</i>	Green foxtail	NN	5.00E-04	0.05	NC
<i>Myriophyllum spicatum</i> L.	Eurasian watermilfoil	NN	0.17	0.02	NC
<i>Hypochaeris radicata</i>	Hairy cat's ear	NN	0.17	3.33E-04	NC
<i>Festuca</i> sp.	Fescue	U	11.1	0.53	NC

**Plant Species Only Common between the Current Grazing (CG) and
Long-term Exclusion (LTE) Sites (Average Relative % Cover)**

Latin Name	Common Name	Status	CG	STE	LTE
<i>Salix lucida</i>	Pacific willow	NA	0.37	NC	0.5
<i>Eleocharis ovata</i>	Ovate spikerush	NA	0.3	NC	0.13
<i>Epilobium torreyi</i>	Willow herb	NA	5.00E-04	NC	7.50E-03
<i>Sparganium emersum</i>	Bur-reed	NN	0.86	NC	7.50E-03
<i>Callitriche stagnalis</i> Scop.	Hairy crabgrass	NN	0.46	NC	0.12
<i>Alisma plantago-aquatica</i> L.	European water plantain	NN	0.11	NC	0.06

NA = Native, NN = Non-Native, U= Unknown

Table 5: Plant Species Common among All Study Sites and Average % Relative Cover of Each Species Found on Each Site. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion).

Native Plant Species	Site Presence			Non-Native Plant Species	Site Presence		
	CG	STE	LTE		CG	STE	LTE
<i>Eleocharis palustris</i>	X	X	X	<i>Phalaris arundinacea</i>	X	X	X
<i>Fraxinus latifolia</i>	X	X	X	<i>Polygonum persicaria</i>	X	X	X
<i>Ludwigia palustris</i>	X	X	X	<i>Ranunculus repens</i>	X	X	X
<i>Sagittaria latifolia</i>	X	X		<i>Lysimachia nummularia</i>	X	X	
<i>Carex obnupta</i>	X	X		<i>Rubus armeniacus</i>	X	X	
family Hypnaceae	X	X		<i>Plantago major</i> L.	X	X	
<i>Equisetum arvense</i>	X	X		<i>Trifolium</i> sp.	X	X	
<i>Prunella vulgaris</i>	X	X		<i>Setaria viridis</i>	X	X	
<i>Salix lucida</i>	X		X	<i>Myriophyllum spicatum</i> L.	X	X	
<i>Eleocharis ovata</i>	X		X	<i>Hypochaeris radicata</i>	X	X	
<i>Epilobium torreyi</i>	X		X	<i>Sparganium emersum</i>	X		X
<i>Veronica</i> sp.	X			<i>Callitriche stagnalis</i> Scop.	X		X
<i>Bidens cernua</i> L.	X			<i>Alisma plantago-aquatica</i> L.	X		X
<i>Myosotis laxa</i>	X			<i>Anagallis arvensis</i>	X		
<i>Ranunculus sceleratus</i> L.	X			<i>Gnaphalium uliginosum</i> L.	X		
<i>Polygonum aviculare</i>	X			<i>Carduus nutans</i> L. ssp.	X		
<i>Marchantia polymorpha</i>	X			<i>Convolvulus arvensis</i>	X		
<i>Eleocharis acicularis</i>	X			<i>Bellis perennis</i> L.	X		
<i>Galium aparine</i> L.	X			<i>Digitaria sanguinalis</i> (L.) Scop	X		
<i>Urtica</i> sp.	X			<i>Geranium molle</i> L.	X		
<i>Nuphar polysepala</i>		X		<i>Taraxacum officinale</i>	X		
<i>Leersia oryzoides</i> (L.) Sw.		X		<i>Mentha pulegium</i>		X	
<i>Polygonum amphibium</i>		X		<i>Rumex crispus</i>		X	
<i>Scirpus Tabernaemontani</i>			X	<i>Echinochloa crus-galli</i>			X
<i>Juncus tenuis</i>			X	<i>Lotus corniculatus</i>			X
<i>Alnus rubra</i>			X	<i>Cirsium vulgare</i> (Savi) Ten.			X
<i>Juncus articulatus</i>			X				
<i>Crataegus douglasii</i> Lindl.			X				
Unknown Plant Species	Site Presence						
	CG	STE	LTE				
<i>Festuca</i> sp.	X	X					
Scrophulariaceae sp.	X						
<i>Carex</i> sp.	X						
Unknown Grass		X					

Figure 8: Plant Species Presence at All Study Sites. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion).

Shannon Diversity Index

The total diversity was greater in both the CG and STE sites than in the LTE site (Tables 3 & 4, Figures 9 & 10). The CG site had the highest total diversity with an average $\exp(H')$ of 4.83 (Table 4, Figures 9 & 10). The STE site was slightly less diverse than the CG site with an average total $\exp(H')$ of 3.79 (Table 4, Figures 9 & 10). The

LTE site had significantly ($p < 0.0167$) lower average total diversity than both of the other sites with an $\exp(H')$ of 1.28 (Table 4, Figures 9 & 10). Average non-native diversity was lowest on the LTE site and highest on the CG site (Table 4, Figures 9 & 10). Average native diversity was significantly different among all of the sites, with the highest native diversity found on the STE and the lowest found on the LTE site (Table 4, Figures 9 & 10).

* Significantly ($p < 0.0167$) different from the other sites.

Figure 9: Comparison of Each Study Site's Average True Diversity. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). The LTE site had significantly ($p < 0.0167$) lower total and non-native diversity than both the STE and CG sites and native diversity was significantly different among all of the sites.

Figure 10: Comparison of Each Study Site's Average Shannon Diversity Index H'. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). The LTE site had significantly (p<0.0167) lower total and non-native diversity than both the STE and CG sites and native diversity was significantly different among all of the sites.

Relative Cover

Average native and non-native species cover were significantly (p<0.0167) different among all three sites. The LTE site had the highest non-native cover of 95.85% of all the sites (Table 4, Figure 11). The CG site had significantly less non-native cover (65.7%) than the LTE site and significantly higher non-native species cover than the STE (47.95%) site (Table 4, Figure 11). Conversely, native cover was highest on the STE site with 51.52% and lowest at the LTE site with 4.15% native cover. The CG site had 23.18% native cover which was significantly higher than the LTE and lower than the STE site (Table 4, Figure 11). On the CG site 11.1% of the vegetation cover was too

heavily grazed to be positively identified past the genus *Festuca* and therefore was not included in the native or non-native cover site comparisons or analysis (Figure 11).

A comparison of the native and non-native plant species cover abundance among the study sites can be seen in Tables 6 & 7. Reed canarygrass (*Phalaris arundinacea*, RCG) was the dominant plant cover at all study sites. RCG average relative cover ranged from 95.16% at the LTE site to 52.75% at the CG site and 43.03% cover at the STE site (Tables 4 & 6, Figure 12). The LTE site was found to have significantly greater RCG cover than both the STE and CG sites (Table 4, Figure 12). No significant difference in RCG cover was found between the STE (43.03%) and CG (52.75%) sites (Table 4, Figure 12). The second most abundant taxa at each site was native in origin; *Sagittaria latifolia* (broadleaf wapato, native) was the second most abundant plant species at the STE site with 31.24% cover; *Ludwigia palustris* (water purslane, native) was the second most abundant species at the CG site with 15.21% cover; and *Scirpus Tabernaemontani* (softstem bulrush, native) was the second most abundant species at the LTE site with 2.93% cover (Table 7, Figures 13-15). Other important plant species ($\geq 5\%$ average relative cover) found on the study sites include *Festuca sp.* (Fescue, unknown origin) and *Polygonum persicaria* (lady's thumb water pepper, non-native) on the CG site and *Eleocharis palustris* (creeping spikerush, native) on the STE site (Tables 6 & 7, Figures 13& 14).

Figure 11: Comparison of Each Study Site's Average % Relative Vegetation Cover. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). Native and non-native cover was found to be significantly ($p < 0.0167$) different among all sites.

Figure 12: Comparison of the Average (± 1 Standard Deviation) % Relative Cover of Reed canarygrass (*Phalaris arundinacea*) and Other Non-native Species Among Study Sites. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). The average relative % reed canarygrass cover on the LTE site was found to be significantly ($p < 0.0167$) higher than the reed canarygrass cover on both the CG and STE sites.

Non-Native Plant Species Cover									
Cover Category	Current Grazing Site			Short Term Exclusion Site			Long Term Exclusion Site		
	Plant Species	Status	% Cover	Plant Species	Status	% Cover	Plant Species	Status	% Cover
>3%	<i>Phalaris arundinacea</i>	NN	52.72	<i>Phalaris arundinacea</i>	NN	43.03	<i>Phalaris arundinacea</i>	NN	95.16
	<i>Festuca sp.</i>	U	11.10						
	<i>Polygonum persicaria</i>	NN	6.38						
>2%	<i>Lysimachia nummularia</i>	NN	2.42	<i>Lysimachia nummularia</i>	NN	2.93			
>1%				<i>Polygonum persicaria</i>	NN	1.48			
>0.10%	<i>Sparganium emersum</i>	NN	0.86	<i>Festuca sp.</i>	U	0.53	<i>Echinochloa crus-galli</i>	NN	0.40
	<i>Trifolium sp.</i>	NN	0.79	<i>Rubus armeniacus</i>	NN	0.12	<i>Callitriche stagnalis Scop.</i>	NN	0.12
	<i>Anagallis arvensis</i>	NN	0.65	<i>Mentha pulegium</i>	NN	0.11			
	<i>Rubus armeniacus</i>	NN	0.51						
	<i>Callitriche stagnalis Scop.</i>	NN	0.46						
	<i>Myriophyllum spicatum L.</i>	NN	0.17						
	<i>Hypochaeris radicata</i>	NN	0.17						
	<i>Plantago major L.</i>	NN	0.12						
	<i>Taraxacum officinale</i>	U	0.12						
	<i>Gnaphalium uliginosum L.</i>	NN	0.11						
	<i>Alisma plantago-aquatica L.</i>	NN	0.11						
<0.10%	<i>Carduus nutans L. ssp.</i>	NN	0.04	<i>Plantago major L.</i>	NN	0.08	<i>Lotus corniculatus</i>	NN	0.09
	<i>Convolvulus arvensis</i>	NN	0.03	<i>Trifolium sp.</i>	NN	0.07	<i>Alisma plantago-aquatica L.</i>	NN	0.06
	<i>Bellis perennis L.</i>	NN	0.02	<i>Setaria viridis</i>	NN	0.05	<i>Sparganium emersum</i>	NN	0.01
	<i>Scrophulariaceae sp.</i>	U	0.01	<i>Myriophyllum spicatum L.</i>	NN	0.02	<i>Ranunculus repens</i>	NN	0.01
	<i>Ranunculus repens</i>	NN	0.01	<i>Rumex crispus</i>	NN	0.02	<i>Cirsium vulgare (Savi) Ten.</i>	NN	1.00E-03
	<i>Carex sp.</i>	U	5.17E-03	<i>Ranunculus repens</i>	NN	5.67E-03	<i>Polygonum persicaria</i>	NN	3.33E-04
	<i>Digitaria sanguinalis (L.) Scop</i>	NN	4.33E-03	<i>Hypochaeris radicata</i>	NN	3.33E-04			
	<i>Geranium molle L.</i>	NN	1.00E-03	<i>Unknown Grass</i>	U	3.33E-04			
	<i>Setaria viridis</i>	NN	5.00E-04						
NA = Native, NN = Non-Native, U= Unknown									

Table 6: Comparison of Average % Relative Cover of Non-native Plant Species at Each Site by % Abundance Category

Native Plant Species Cover									
Cover Category	Current Grazing Site			Short Term Exclusion Site			Long Term Exclusion Site		
	Plant Species	Status	% Cover	Plant Species	Status	% Cover	Plant Species	Status	% Cover
>3%	<i>Ludwigia palustris</i>	NA	15.21	<i>Sagittaria latifolia</i>	NA	31.24			
	<i>Festuca sp.</i>	U	11.10	<i>Eleocharis palustris</i>	NA	13.89			
>2%				<i>Nuphar polysepala</i>	NA	2.55	<i>Scirpus Tabernaemontani</i>	NA	2.93
>1%	<i>Veronica sp.</i>	NA	1.94	<i>Carex obnupta</i>	NA	1.65			
	family <i>Hypnaceae</i>	NA	1.60						
	<i>Eleocharis palustris</i>	NA	1.05						
	<i>Sagittaria latifolia</i>	NA	1.03						
	<i>Bidens cernua L.</i>	NA	1.02						
>0.10%	<i>Salix lucida</i>	NA	0.37	<i>Fraxinus latifolia</i>	NA	0.89	<i>Salix lucida</i>	NA	0.50
	<i>Eleocharis ovata</i>	NA	0.30	<i>Festuca sp.</i>	U	0.53	<i>Ludwigia palustris</i>	NA	0.46
	<i>Carex obnupta</i>	NA	0.25	family <i>Hypnaceae</i>	NA	0.49	<i>Eleocharis ovata</i>	NA	0.13
	<i>Myosotis laxa</i>	NA	0.16	<i>Ludwigia palustris</i>	NA	0.38			
	<i>Taraxacum officinale</i>	U	0.12	<i>Equisetum arvense</i>	NA	0.22			
	<i>Ranunculus sceleratus L.</i>	NA	0.12	<i>Prunella vulgaris</i>	NA	0.11			
				<i>Leersia oryzoides (L.) Sw.</i>	NA	0.10			
<0.10%	<i>Fraxinus latifolia</i>	NA	0.05	<i>Polygonum amphibium</i>	NA	0.03	<i>Juncus tenuis</i>	NA	0.07
	<i>Prunella vulgaris</i>	NA	0.05	<i>Unknown Grass</i>	U	3.33E-04	<i>Alnus rubra</i>	NA	0.03
	<i>Polugonum aviculare</i>	NA	0.01				<i>Eleocharis palustris</i>	NA	0.02
	<i>Scrophulariaceae sp.</i>	U	0.01				<i>Juncus articulatus</i>	NA	0.02
	<i>Marchantia polymorpha</i>	NA	0.01				<i>Epilobium torreyi</i>	NA	0.01
	<i>Eleocharis acicularis</i>	NA	6.50E-03				<i>Crataegus douglasii Lindl.</i>	NA	3.17E-03
	<i>Carex sp.</i>	U	5.17E-03				<i>Fraxinus latifolia</i>	NA	2.50E-03
	<i>Galium aparine L.</i>	NA	3.33E-03						
	<i>Equisetum arvense</i>	NA	1.50E-03						
	<i>Epilobium torreyi</i>	NA	5.00E-04						
	<i>Urtica sp.</i>	NA	5.00E-04						

NA = Native, NN = Non-Native, U= Unknown

Table 7: Comparison of Average % Relative Cover of Native Plant Species at Each Site by % Abundance Category

Figure 13: Current Grazing Site % Average Cover of All Plant Species ≥1%. Plant species codes: Ph ar = *Phalaris arundinacea* (Reed canarygrass), Lu pa = *Ludwigia palustris* (Water purslane), Fe sp.= *Festuca sp.* (Fescue), Po pe = *Polygonum persicaria* (Lady's thumb water pepper), Ly nu= *Lysimachia nummularia* (Moneywort), Ve sp.= *Veronica sp.*(Speedwell), Moss= family *Hypnaceae*, El pa= *Eleocharis palustris* (Creeping spikerush), Sa la= *Sagittaria latifolia* (Broad leaf wapato), Bi ce= *Bidens cernua L.*(Nodding beggarstick).

Figure 14: Short-term Exclusion Site % Average Cover of All Plant Species ≥1% . Plant species codes: Ph ar = *Phalaris arundinacea* (Reed canarygrass), Sa la= *Sagittaria latifolia* (Broad leaf wapato), El pa= *Eleocharis palustris* (Creeping spikerush), Ly nu= *Lysimachia nummularia* (Moneywort), Nu po= *Nuphar polysepala* (Yellow pond lily), Ca ob = *Carex obnupta* (Slough sedge), Po pe = *Polygonum persicaria* (Lady's thumb water pepper).

Figure 15: Long-term Exclusion Site % Average Cover of All Plant Species $\geq 0.01\%$. Plant species codes: Ph ar = *Phalaris arundinacea* (Reed canarygrass), Sc ta= *Scirpus Tabernaemontani* (Softstem bulrush), Sa lu= *Salix lucida* (Pacific willow), Lu pa = *Ludwigia palustris* (Water purslane), Ec cr= *Echinochloa crus-galli* (Barnyard grass), El ov= *Eleocharis ovate* (Ovate spikerush), Ca st= *Callitriche stagnalis Scop.* (Water starwort).

Plant Community Similarity

The Bray-Curtis Index of Similarity (BCI) of plant community composition among transects within each site was high ranging from 92% similarity within the LTE site and 70% similarity within the CG site (Table 8). Similarity among sites ranged from 43% between the STE site and the LTE site to 53% between the CG site and the LTE site (Table 8). All of the sites were found to be significantly different from each other (p -value < 0.167) using the ANOSIM test (Table 8). The NMDS ordination plot, Figure 16, of the BCI data clearly shows that within each site the plant communities are similar, but among the sites they are distinctly different.

Bray Curtis Index of Similarity Analysis					
Average % Similarity Within and Among Sites					
Sites	Long-term Exclusion		Short-term Exclusion		Current Grazing
	% Similarity	p-value	% Similarity	p-value	% Similarity
Long-term Exclusion	92 (± 3.3)				
Short-term Exclusion	43 (± 0.2)	0.0108*	79 (± 8.3)		
Current Grazing	53 (± 1.2)	0.0022*	47 (± 8.2)	0.0065*	70 (± 13.4)

Table 8: Bray Cutris Index of Similarity Within and Among Sites. * All sites plant community composition (BIC) were found to be significantly different from each other (p-value <0.0167).

Figure 16: Bray-Curtis Index of Plant Community Similarity Ordination of Site Transects by non-metric Multidimensional Scaling. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). Each site had 6 transects. All sites were found to be significantly different from each other (p -value < 0.0167).

Reed Canarygrass Abundance and Plant Community Diversity

Across all of the transects ($n=18$) significant negative correlations ($p < 0.05$) were found between RCG abundance and native and non-native species richness, diversity (H'), and cover (Table 9). The strongest (negative) correlation was found between RCG

abundance and total species diversity (Spearman's rho = -0.86, p-value<0.001), native species diversity (Spearman's rho = -0.83, p-value<0.001) and relative native cover (Spearman's rho = -0.81, p-value<0.001). On all transects with high levels of RCG abundance tended to have lower total diversity and native species abundance. Trends in native and non-native species richness, diversity, cover and RCG cover among the sites can be seen in Table 9 and Figures 17-19. No difference between the true diversity exponentially (exp) transformed Shannon Diversity Index and the untransformed Shannon Diversity Index was found in the correlation analysis.

Spearman's Rho Correlation Analysis % Reed Canarygrass (RCG) vs.		
Among all transects (n=18)	Rho	p-value
Species Richness (-RCG)	-0.55	<0.02
Native Species Richness	-0.47	<0.05
Non-Native Species Richness (-RCG)	-0.54	<0.03
Total H'	-0.86	<0.001
Native H'	-0.83	<0.001
Non-Native H'	-0.78	<0.001
% Native Cover	-0.81	<0.001
% Non-Native Cover (-RCG)	-0.68	<0.002

Table 9: Spearman's Rho Correlation Analysis. Conducted between RCG cover and native and non-native: species richness, Shannon Diversity Index (H'), relative cover for all vegetation transects across all sites (transects=18).

Figure 17: Total Shannon Diversity Index (H') vs. Relative % Cover Reed canarygrass (*Phalaris arundinacea*). Significant Spearman's rank correlation $\rho = -0.86$, $p\text{-value} < 0.05$. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion).

Figure 18: Total Native Shannon Diversity Index (H') vs. Relative % Cover Reed canarygrass (*Phalaris arundinacea*). Significant Spearman's rank correlation $\rho = -0.83$, $p\text{-value} < 0.05$. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion).

Figure 19: Relative % Native Cover vs. % Cover Reed canarygrass (*Phalaris arundinacea*). Significant Spearman's rank correlation rho= -0.83, p-value<0.05. Study sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion).

Elevation Data

After processing the LiDAR data it became apparent that differences in the LiDAR elevation resolution among the sites prohibited detailed comparisons to be conducted. However the LiDAR data were used to determine the average slope and elevation range of the transects at each site (Table 10). Transects in CG riparian wetland had an average slope of 3.41% and ranged from 2.89 to 4.73 meters in elevation. Transects in the STE riparian wetland had an average slope of 4.16% and ranged from 2.58 to 4.45 meters in elevation. The LTE riparian wetland transects had an average slope of 0.76% and ranged from 4.82 to 5.92 meters in elevation. The difference between

elevation range and average slope of each site was small and considered comparable across sites for this study.

Slope (%) and Elevation Range (m)			
Site	Slope	Min	Max
LTE	0.76(±0.6)%	4.82(±0.1)	5.92(±0.4)
STE	4.16(±1.1)%	2.58(±0.1)	4.45(±0.5)
CG	3.41(±0.2)%	2.89(±0.2)	4.73(±0.2)

Table 10: Average (±1 Standard Deviation) Slope and Elevation Range of Each Study Site. Study sites: CG (current grazing), STE(short-term exclusion), LTE (long-term exclusion).

Soil Survey

Soil Texture

Soil was consistent among all of the sites falling into the clay loam texture class (Table 11, Figure 20). Percent of sand ranged from 18% to 22% with an average of 20% on the CG site, from 13% to 24% with an average of 19% on the STE site, and from 12% to 30% with an average of 23% on the LTE site. Percent silt ranged from 49% to 54% with an average of 51% on the CG site, from 42% to 50% with an average of 45% on the STE site, and from 38% to 60% with an average of 46% on the LTE site. Percent clay ranged from 24% to 32% with an average of 28% on the CG site, from 30% to 48% with an average of 36% on the STE site, and from 28% to 30% with an average of 31% on the LTE site. Soil texture did not vary significantly among sites.

Soil Parameters	Sites		
Site Average	CG	STE	LTE
Silt & Clay (%)	79.6(±1.4)	80.9(±4.1)	77.2(±7.0)
Clay (%)	28.4(±2.6)	35.6(±7.7)	31.2(±3.5)
Silt (%)	51.2(±2.1)	44.4(±4.2)	46.0(±6.5)
Sand (%)	20.4(±1.4)	19.1(±4.1)	22.8(±7.0)
Organic Matter (%)	9.71(±1.4)~	7.36(±2.4)	5.79(±1.6)
Bulk Density (g/cm³)	0.71(±0.18)	0.63(±0.08)	0.81(±0.08)

Table 11: Summary of Average (± 1 Standard Deviation) Soil Parameters for Each Site. Each site had 6 soil samples. ~ Indicates that the CG site was significantly (p-value <0.0167) higher than the STE site. Study Sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion).

Figure 20: Average (± 1 Standard Deviation) Texture Composition of Soil at All Study Sites. Study Sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). All sites fell within the clay loam texture class.

Organic Matter

Soil organic matter content was relatively high, averaging over 5%, at all of the sites (Kalra and Maynard 1991, Table 11, Figure 21). The CG site had the higher OM content over the other sites, ranging from 12.4% to 8.76% with an average of 9.71%. The STE site had the lowest OM content ranging from 4.08% to 7.84% with an average of 5.79% and the LTE site soil OM content ranged from 5.04% to 10.8% with an average of 7.36%. The CG site had significantly (p-value <0.0167) higher soil OM levels over the STE site; however, there was no significant difference in soil OM content between either the CG site or the STE site and the LTE site (Table 11, Figure 21).

*CG site was significantly (p-value<0.0167) higher than the STE site.

Figure 21: Boxplot Showing the Spread and Average of the % Soil Organic Matter (OM) by Site (each site = 6 soil samples). Study Sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). The CG site had significantly (p-value<0.0167) higher soil OM levels than the STE site.

Soil Bulk Density

The surface (0-10cm) soil bulk density at all sites averaged below 1 g/cm³, falling into the very low bulk density range (Hazelton and Murphy 2007, Table 11, Figure 22).

Soil bulk density ranged from 0.49 to 1.03 g/cm³ with an average of 0.71 on the CG site, from 0.52 to 0.71 g/cm³ with an average of 0.63 g/cm³ on the STE site, and from 0.54 to

1.09 g/cm³ with an average of 0.81 g/cm³ on the LTE site. All site bulk density measurements fell well below the critical clay loam bulk density of 1.60 g/cm³, at which point plant root penetration is severely restricted (Hazelton and Murphy 2007, Figure 22). These results were lower than expected given the current and historic land uses. However, surface soils with high organic matter content (>5%) can have correspondingly low (<1 g/cm³) bulk density levels (Kalra and Maynard 1991, Soane 1999).

*1.60 g/cm³ = Critical bulk density, root penetration is severely restricted in clay loam soils (Hazelton and Murphy 2007).

Figure 22: Average ($\pm$ 1 Standard Deviation) Soil Bulk Density (g/cm³) of the Soil at Each Study Site (each site=6 transects). Study Sites: CG (current grazing), STE (short-term exclusion), LTE (long-term exclusion). All site bulk density measurements fell well below the critical clay loam bulk density of 1.60 g/cm³, at which point plant root penetration is severely restricted (Hazelton and Murphy 2007).

DISCUSSION

Grazing and Wetland Plant Community Characteristics

The results of this study supported the first hypothesis, showing that the native plant species richness was lower in the excluded wetlands than in the grazed wetland. Soil and elevation characteristics were found to be similar among the study sites, removing the possibility that variations in topography or soil were controlling factors on plant community composition differences among study sites. The grazing site had significantly higher numbers of native species than either the 3 year excluded wetland or the 13 year excluded wetland. Non-native plant dominance, however, was higher on the grazed and the 13 year excluded wetlands than on the 3 year excluded wetland. Relative abundance of native species was significantly higher on the 3 year excluded wetland than in both the grazed and 13 year excluded wetlands. These results suggest that competitive exclusion of native species due to an increase in non-native plant dominance in the absence of grazing was likely a factor on the 13 year excluded site but not on the 3 year excluded site.

In another livestock exclusion restoration project (located adjacent to the short-term exclusion site), the Scappoose Bay Watershed Council (SBWC) found that after three years of cattle exclusion, important native wetland species such as Wapato and Creeping spikerush increased from <3% to 64% cover and <1% to 10% cover respectively (SBWC unpublished report 2010). During this time, overall RCG abundance in the wetland also increased from 9% to 25% cover, with some areas of the wetland supporting monotypic stands of RCG in 2010 (SBWC unpublished report 2010). In

another 3 year grazing exclusion study, conducted in a RCG-invaded freshwater wetland, Kellogg and Bridgham 2004 found that excluded plots had significantly higher native species richness than paired grazed plots. However, based on observed changes in RCG abundance during the study period, they hypothesized that long-term exclusion of grazing in these areas may result in future RCG exclusion of these native species (Kellogg and Bridgham 2004). The prior exclusion grazing regime of both of these studies was similar to the CG and (pre-exclusion) STE sites in the present study. The findings of this study are corroborated by these studies cited above and suggest that short-term exclusion from heavy continuous grazing may benefit the wetland plant community by allowing less grazing tolerant native species to recover, while invasive RCG abundance is low. However, RCG abundance may eventually increase due to a lack of grazing, as seen on the 13 year excluded site, and thus crowd out other species. In these situations RCG abundance will need to be controlled by the re-introduction of grazing and/or other management methods to prevent degradation of the riparian wetland plant community.

It was also hypothesized that the grazed riparian wetland would have higher native and non-native species richness, because grazing can reduce plant competition through herbivory and increase species richness through disturbance (the intermediate disturbance hypothesis; Grime 1973, Connell 1978, Fox and Connell 1979). The findings in this study supported this hypothesis, with the grazed wetland having significantly greater total native and non-native species richness as well as higher total diversity (Shannon Diversity Index) than the excluded wetlands. These findings are similar to other riparian wetland grazing exclusion studies where grazed sites have been found to

have higher diversity than paired ungrazed exclosures (e.g., Bakker 1985, Chaneton and Facelli 1991, Green and Kauffman 1995, Jutila 1997, Humphrey and Patterson 2000, Paine and Ribic 2002, Krzic et al. 2004, Reeves and Champion 2004, Marty 2005, Table 2). Grazing influence on species richness, in riparian wetland habitats, is highly dependent on the intensity and duration of the grazing disturbance, as was discussed earlier in the thesis (Table 2). For example, Hillhouse et al. (2010) conducted a 2 year grazing study to identify if cattle grazing could be used to control RCG dominance in Nebraskan rain-fed wetlands. Under low grazing intensities ranging from 0.3-5.4 AUM/ha of forage removed a year, they found no significant difference between RCG abundance or species richness in grazed and ungrazed plots. It is possible that this intensity of grazing was not high enough to adequately reduce the RCG abundance and provide opportunity for less competitive plants to grow. The CG site's grazing intensity (9.84 AUM/ha/year) was more than twice that of the Hillhouse et al. (2010) study, which could account for the difference in species abundance and diversity findings between our studies. In the comparison of 15 different wetland grazing exclosure studies discussed earlier, moderate-high grazing intensities resulted in an increase in species richness and low grazing intensity resulted in no difference or reduced species richness when compared to ungrazed exclosures (Table 2).

A main objective of this study was to identify if RCG dominance was an issue on any of the study wetlands. RCG was found to be the dominant non-native plant species on all of the wetland sites, composing at least 43% of the total plant cover and making up over 80% of the non-native cover. The overall dominance of RCG was comparable

between the grazed site and the 3 year excluded site. The 13 year excluded site had significantly more RCG cover, approaching a monospecific stand of RCG. On all sites RCG abundance was negatively correlated with plant community diversity. RCG invasion was an issue on all sites, with long-term exclusion from grazing showing by far the highest RCG dominance.

The RCG dominance found in these Lower Columbia River Basin (LCRB) wetlands is not an isolated issue, as RCG invasion of wetland areas is a serious problem throughout the region. In a review of the native freshwater wetland plant associations of Northwestern Oregon, Christy (2004) stated that most Columbia River bottom historic prairie and seasonal willow/ash swamps were invaded with RCG. Restoration practitioners and researchers have reported RCG cover ranging from 40 to 100% in riparian wetland sites throughout the region (Kilbride and Paveglio 1999, Perkins and Wilson 2005, Fierke and Kauffman 2006, Schooler et al. 2006, Jenkins et al. 2008, Miller et al. 2008, Ringold et al. 2008, SBWC unpublished data 2010). Several Oregon researchers (e.g. Perkins and Wilson 2005, Fierke and Kauffman 2006, Schooler et al. 2006, Jenkins et al. 2008), also reported RCG abundance having a strong negative correlation with plant community diversity and native species richness. Locally, the Port of Portland also reported that RCG became increasingly dominant on the wetland edges of Jewett Lake after cattle were excluded in 1993 (FES 2002). In the LCRB, where RCG presence is widespread, long-term livestock exclusion may result in an increase in RCG abundance and reduced plant community diversity. The low diversity and high RCG abundance found in the long-term excluded riparian wetland can be explained by the lack

of disturbance (from cattle or other factors) to help maintain diversity and/or to keep RCG from becoming dominant (the intermediate disturbance hypothesis; Grime 1973, Connell 1978, Fox and Connell 1979).

Orienting the results of this study within the context of the intermediate disturbance hypothesis provides an intriguing picture of how grazing might play a role in riparian plant community dynamics (Figure 23). Placement of the study sites on this graph was based on overall native species diversity and invasive species dominance (RCG) of each site. The long-term exclusion site fell within the low diversity (native) and high invasive plant (RCG) dominance on the left side of the disturbance axis, indicating the low intensity, frequency and time since disturbance (grazing) characteristic to this site. The current grazing site and short-term exclusion sites are located towards the middle of the diversity (native), invasive plant (RCG) dominance range characteristic of their disturbance regimes.

Figure 23: Conceptual Figure of Expected Plant Community Diversity and Invasive Species Dominance in Response to Different Disturbance Regimes (such as grazing) Including Study Sites. Study site position based on grazing disturbance (or lack of), average native species diversity and invasive plant dominance (RCG). Concept based on the intermediate disturbance hypothesis and the competitive exclusion principle (Grime 1973, Connell 1978, Milchunas et al. 1988, Kercher et al. 2007, Hughes 2010). This thesis uniquely contributes the hypothesized response of invasive plant dominance (dashed line) to the diversity vs. disturbance relationship.

Historically, these wetlands would have experienced disturbance in the form of periodic large and small scale flooding events and native ungulate grazing (Christy and Putera 1992, Christy 2004, Lev et al. 2004, NRCC 2004). The current local and regional hydrologic regulation minimizes flooding disturbance and habitat loss-fragmentation has diminished natural ungulate use of these areas (Christy and Putera 1992, Wissmar and Beschta 1998, Christy 2004, Lev et al. 2004). These changes to the natural disturbance regimes, coupled with the introduction of RCG, have made restoration of these areas a complex undertaking. Given the high cost and low success rate of RCG eradication and

control efforts, selective grazing may be an economically feasible restoration tool for maintaining ecological diversity in these RCG invaded riparian wetland habitats.

Management Implications

Based on the differences in plant community composition observed between the grazed and excluded wetlands in this study, the use of long-term livestock exclusion as a passive restoration technique may not be an effective way to restore wetlands in this region. Public pressure to enhance and restore riparian wetlands has encouraged restoration practitioners to eliminate all livestock grazing in these habitats (Reichard 1989, Leonard and Karl 1995, Kauffman et al. 1997, Hoover et al. 2001). However the benefits of livestock exclusion (e.g., reduced soil erosion, improved water quality, restored riparian plant communities) are highly variable and depend on a myriad of variables including local environmental conditions and grazing regime (Figure 1, Table 2). Passive restoration through livestock exclusion has been shown to be successful in some parts of the West, but the benefits of this practice may not extend to the LCRB. When restoring a riparian wetland through livestock exclusion, site conditions and existing plant communities need to be considered. In areas like the LCRB, where invasive plants like RCG are abundant, long-term livestock exclusion may result in further degradation of the wetland through invasive plant dominance.

Short-term exclusion of grazing, however, could be an effective strategy for recovering native species abundance in heavily grazed sites. In areas where livestock are excluded, RCG abundance (and other invasive species) would need to be monitored and

controlled to maintain native species abundance and diversity. Continued research is needed to identify optimal livestock grazing and exclusion regimes that will minimize negative grazing impacts, while maximizing species richness of both native plants and animals in these riparian wetland habitats. Consideration of optimal grazing regimes should include both the intensity (e.g. AUM levels) of grazing as well as the duration of both grazing and exclusion periods. After a short exclusion period (with duration based on native plant recovery), grazing could be re-introduced to control RCG abundance and help maintain site diversity. It is possible that grazing could also be used to help control RCG dominance and recover diversity on previously ungrazed or long-term excluded sites.

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Appendix A: Complete Plant Species List

Plant Species Information						Site Occurrence		
Latin	Common	Status	USDA Weed List*	Duration	Wetland Indicator*	Current Grazing	Short-term Exclusion	Long-term Exclusion
<i>Alisma plantago-aquatica L.</i>	Water plantain (European)	Non-native		Perennial	OBL	X		X
<i>Alnus rubra</i>	Red alder	Native		Perennial	FAC			X
<i>Anagallis arvensis</i>	Pimpernel	Non-native	X	Annual/ Biennial	FAC	X		
<i>Bellis perennis L.</i>	English daisy	Non-native	X	Perennial		X		
<i>Bidens cernua L.</i>	Nodding beggarstick	Native	X	Annual	FACW+	X		
<i>Callitriche stagnalis Scop.</i>	Water starwort	Non-native	X	Perennial	OBL	X		X
<i>Carduus nutans L. ssp.</i>	Thistle species	Non-native	X	Biennial/ Perennial		X		
<i>Carex obnupta</i>	Slough sedge	Native		Perennial	OBL	X	X	
<i>Carex sp.</i>	Sedge species	Unknown				X		
<i>Cirsium vulgare (Savi) Ten.</i>	Bullthistle	Non-native	X	Biennial	FACU			X
<i>Convolvulus arvensis</i>	Field bindweed	Non-native	X	Perennial		X		
<i>Crataegus douglasii L. indl.</i>	Black hawthorn	Native		Perennial	FAC			X
<i>Digitaria sanguinalis (L.) Scop</i>	Hairy crabgrass	Non-native	X	Annual	FACU	X		
<i>Echinochloa crus-galli</i>	Barnyard grass	Non-native	X	Annual	FACW			X
<i>Eleocharis acicularis</i>	Needle spikerush	Native		Annual	OBL	X		

Plant Species Information (Continued)						Site Occurrence		
Latin	Common	Status	USDA Weed List*	Duration	Wetland Indicator*	Current Grazing	Short- term Exclusion	Long-term Exclusion
<i>Eleocharis palustris</i>	Creeping spikerush	Native		Perennial	OBL	X	X	X
<i>Eleocharis ovata</i>	Ovate spikerush	Native		Annual	OBL	X		X
<i>Epilobium torreyi</i>	Watson's willowherb	Native		Annual	FACW	X		X
<i>Equisetum arvense</i>	Field horsetail	Native	X	Perennial	FAC	X	X	
<i>family Hypnaceae</i>	Moss	Native				X	X	
<i>Festuca sp.</i>	Fescue	Unknown				X	X	
<i>Fraxinus latifolia</i>	Oregon ash tree	Native		Perennial	FACW	X	X	X
<i>Galium aparine L.</i>	Bed straw	Native	X	Annual	FACU	X		
<i>Geranium molle L.</i>	Dovefoot geranium	Non-native		All		X		
<i>Gnaphalium uliginosum L.</i>	Cudweed	Non-native		Annual	FAC+	X		
<i>Hypochaeris radicata</i>	Hairy cats ear	Non-native	X	Perennial	FAC*	X	X	
<i>Juncus articulatus</i>	Jointed rush	Native		Perennial	OBL			X
<i>Juncus tenuis</i>	Slender rush	Native	X	Perennial	FACW-			X
<i>Leersia oryzoides (L.) Sw.</i>	Rice cutgrass	Native		Perennial	OBL		X	
<i>Lotus corniculatus</i>	Trefoil	Non-native	X	Perennial	FAC			X
<i>Ludwigia palustris</i>	Water purslane	Native		Perennial	OBL	X	X	X
<i>Lysimachia nummularia</i>	Moneywort	Non-native	X	Perennial	FACW	X	X	
<i>Marchantia polymorpha</i>	Lung liverwort	Native	X			X		

Plant Species Information (Continued)						Site Occurrence		
Latin	Common	Status	USDA Weed List*	Duration	Wetland Indicator*	Current Grazing	Short-term Exclusion	Long-term Exclusion
<i>Mentha pulegium</i>	Pennyroyal	Non-native	X	Perennial	OBL		X	
<i>Myosotis laxa</i>	Forget me not	Native		All	OBL	X		
<i>Myriophyllum spicatum L.</i>	Eurasian watermilfoil	Non-native	X	Perennial	OBL	X	X	
<i>Nuphar polysepala</i>	Yellow pond lily	Native		Perennial	OBL		X	
<i>Phalaris arundinacea</i>	Reed canarygrass	Non-native	X	Perennial	FACW	X	X	X
<i>Plantago major L.</i>	Common plantain	Non-native	X	Perennial	FACU+	X	X	
<i>Polygonum amphibium</i>	Water smartweed	Native	X	Perennial	OBL		X	
<i>Polygonum aviculare</i>	Knotweed	Native	X	Annual/ Perennial	FACW-	X		
<i>Polygonum persicaria</i>	Water pepper (lady's thumb)	Non-native	X	Annual/ Perennial	FACW	X	X	X
<i>Prunella vulgaris</i>	Selfheal	Native	X	Perennial	FACU+	X	X	
<i>Ranunculus sceleratus L.</i>	Celery leaf buttercup	Native		Annual/ Perennial	OBL	X		
<i>Rubus armeniacus</i>	Himalayan blackberry	Non-native	X	Perennial	FACU	X	X	
<i>Rumex crispus</i>	Curly dock	Non-native	X	Perennial	FAC+		X	
<i>Sagittaria latifolia</i>	Broad leaf wapato	Native		Perennial	OBL	X	X	
<i>Salix lucida</i>	Pacific willow	Native		Perennial	FACW+	X		X
<i>Scirpus Tabernaemontani</i>	Bulrush	Native		Perennial	OBL			X
<i>Scrophulariaceae sp.</i>	Figwort species	Unknown				X		

Plant Species Information (Continued)						Site Occurrence		
Latin	Common	Status	USDA Weed List*	Duration	Wetland Indicator*	Current Grazing	Short-term Exclusion	Long-term Exclusion
<i>Setaria viridis</i>	Green foxtail	Non-native	X	Annual		X	X	
<i>Sparganium emersum</i>	Bur-reed	Non-native		Perennial	OBL	X		X
<i>Taraxacum officinale</i>	Dandelion	Unknown	X	Perennial	FACU	X		
<i>Trifolium sp.</i>	Clover sp.	Non-native	X	Perennial	FAC*	X	X	
<i>Unknown grass</i>	Unknown grass	Unknown					X	
<i>Urtica sp.</i>	Nettle species	Native	X	Perennial	FAC+	X		
<i>Veronica sp.</i>	Veronica species	Native		Perennial	OBL	X		

* Wetland Indicator and Weed List status information acquired from the USDA PLANTS Database (www.plants.usda.gov).

Appendix B: Calculating Study Grazing Intensity

In the grazing studies reviewed for comparison with this study, grazing intensity was not consistently reported using the same rangeland management terminology. Ideally a study's grazing regime is reported by providing the type and number of Animal Units (AUs, 1 AU = 1,000 lb beef cow) per hectare (ha) grazed for a given period of time (days-months), during a certain season (Summer-Fall). Not all this information was reported by the studies reviewed. To enable easier grazing intensity comparisons among sites, a grazing intensity metric (AUM/ha/yr) was estimated using the information provided by each study. Using the grazing intensity data available for each study the average Animal Unit Months (AUM, which is equivalent to the amount of forage consumed by 1 AU in 1 month and equivalent to 780 lbs), per hectare per year was calculated for each study. This metric is an estimate of the amount of forage (AUM) removed from 1 hectare of rangeland over a 1 year period of time and was calculated by multiplying the AUM/ha (or AU/ha) reported by the duration of grazing (number of months grazing occurred per year) for each study. When duration was reported being between two months such as Sept/Oct, the number of months used in the calculation was estimated as being half a month, so for May-Sept/Oct the number of grazing months was estimated as being 5.5 months. If number of days was reported instead of months, the number of days was divided by 30 (assuming a standard 30 days in a month) and the fraction of month calculated was used in the metric calculations. Season of grazing was not included in the metric calculations. Detailed information regarding the reported

grazing intensity of each study can be found in the following table (Appendix B Table) and in an abbreviated form in the text (Table 2).

Detailed Summary Table of Wetland Species Richness Under Different Grazing Intensities

Wetland Type	Location	Reported Grazing Regime	AU/ha	Duration (Months/yr)	Years	Total Forage Removed (lb/ha/yr)	AUM/ha/yr	Grazing Intensity	Species Richness	Citation	
Montane riparian areas	New Mexico	3-5 AUs for 2-8 days		0.07-0.27	2	168-1120	0.22-1.44		Low	*	Lucas et al. 2004
Rain-fed wetlands	Nebraska	20AU/7ha-40AU/8ha	2.9-5	1.6-0.67	2	218-4203	0.3-5.4		*	Hillhouse et. al 2010	
Freshwater wetlands	Indiana	38 deer per km2 (100ha), 1 deer = .18AU	0.07	12	2	636	0.81^		-	Kellogg and Bridgham 2004	
Vernal pools	California	1 AU/2.4 ha	0.42	2-7	3	650-2275	0.83-2.9		+	Marty 2005	
Riverine floodplain	New Zeland	.13-.2 AU/ha	.13-.2	12	4	1217-1872	1.56-2.4		+	Buxton et al. 2001	
Montane riparian areas	British Columbia	15 AU/5 ha	3	2	10	2340	3		+	Krzic et al. 2004	
Delta meadow	Finland	.3-1.7 AU/ha	.3-1.7	12	2	2808-15912	3.6-20.4		+	Jutila 1997, 1999	
Seashore meadow	Finland	.3-1.7 AU/ha	.3-1.7	12	2	2808-15912	3.6-20.4		-	Jutila 1997, 1999	
Riparian vegetation	Kansas	1.6 ha/AU	0.63	7	2	3416	4.4		+	Hoover et al. 2001	
Riparian pasture	Scotland	2.25-2.5 AU/ ha	2.25-2.5	6	9	3510-3900	4.5-5		+	Humphrey and Patterson 2000	
Mixed deep marsh	Idaho	2.3-2.5 AUM/ha	2.3-2.5	2	2	3588-3900	4.6-5		+	Austin et al. 2007	
Mire (heathland)	United Kingdom	2-69 AU/ha	2-69	3-6	1	4680-322920	6-414~		+	Bullock and Pakeman 1996	
Lowland floodplain	Argentina	0.5 AU/ha	0.5	12	13	4680	6		+	Chaneton and Facelli 1991	
Saltmarsh	Denmark	1.3-1.7 AU/ha	1.3-1.7	5.5	9	5577-7293	7.15-9.35		+	Bakker 1985	
Moist & dry meadow	Oregon	1.3-1.8 ha/AUM	0.77-0.55	12	10	5196-7200	7.7-9.2		+	Green and Kauffman 1995	
Riparian area	Wisconsin	High			1		High		+	Paine and Ribic 2002	
Riparian wetland	Oregon	250AU/152.5ha	1.64	6	1	7674	9.84		High	+	This study

Appendix B Table 1: Detailed summary of wetland grazing studies that measured species richness under different grazing regimes/intensities. '*' indicates no change in species richness with grazing, '-' indicates a decrease in species richness with grazing and a '+' indicates an increase in species richness with grazing. When possible grazing intensity has been converted to AUM/ha/yr of forage removed a year for comparison; this conversion is based on reported stocking rate and duration of grazing. The table is sorted by low to high minimum AUM/ha/yr (grazing intensity). All studies used cattle for grazing except: ^ deer, ~ combination of sheep, horses and cattle. Total forage removed a year was calculated based on 1 AU removing 780lb of forage a month.