EVALUATION OF RIPARIAN VEGETATION DATA

AND ASSOCIATED SAMPLING TECHNIQUES

by

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ABSTRACT

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Monitoring the vegetation of riparian areas has become important to land managers who need to evaluate and minimize anthropogenic impacts. Riparian vegetation is an important monitoring variable because of its role in maintaining habitat for fish and other animals and plants. This dissertation evaluated data from hundreds of riparian sites across the interior northwestern US, and evaluated tools for summarizing and monitoring the vegetation at those sites. The sites were compared, based on the cover of meta-community types in the riparian zone. Meta-community types were developed based on the similarity of community types from eight riparian vegetation classifications, which had been used to describe the site vegetation. The relationships of vegetation, management, environment, and stream characteristics were evaluated. The strongest relationship between vegetation and management was identified at the site level, where differences were detected between livestock exclosures and adjacent grazed

areas. Differences related to management at larger scales were relatively small or undetectable. A wetland rating, which summarizes the abundance of obligate wetland plants, was developed to facilitate comparison among sites. This wetland rating was one of the more repeatable variables, compared to some other variables in use. The low observer agreement for many of the variables would inhibit change detection, which is a primary objective of monitoring. These findings highlight the need for improved repeatability with existing methods, or the development of alternative methods of data collection and summarization, in order to more accurately characterize and monitor changes in riparian vegetation and associated ecological functions.

(183 pages)

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CHAPTER 1

INTRODUCTION

In recent years concern has grown about the condition of riparian areas, after an extended period of over-use and alteration of riparian areas by human-related activities (National Research Council 2002). Many people, it seems, were unaware or unconcerned with the anthropogenic impacts to riparian areas. This could have been due to their resiliency, which allowed riparian areas to recover from the initial disturbances from livestock grazing and tree harvesting. In addition, riparian areas were a relatively small part of the landscape, so the relative importance of riparian vegetation, for fish and wildlife habitat among other benefits, may have been underestimated.

There are also difficulties associated with studying riparian areas, which may explain why they were not studied as much as upland forests, shrublands, and grasslands. The difficulties of studying riparian areas relate to their dynamic nature, as part of the stream/riparian environment, their irregular and linear shape, and their relatively small proportion of the landscape. The dynamic nature of stream flow, and the associated migration of the channel, makes it difficult to establish permanent plots in riparian areas. Random plots are also difficult to use because of the variable dimensions of the riparian zone and the dramatically variable conditions within that zone. Riparian areas have steep environmental gradients, particularly for moisture availability. In addition, there are continuous, and often dramatic, environmental changes along the stream corridor (in the upstream and downstream direction). For these reasons many researchers probably avoided the study of riparian vegetation and the functions it performs.

With the increased recognition of the value of riparian areas, many land managers and livestock operators changed the management activities in riparian areas (Platts 1991). Riparian buffers were established by states and the national government to minimize the impacts to riparian areas from logging and roads (National Research Council 2002). Season-long grazing was often replaced with other management systems such as rest-rotation, which provided a period of recovery for the vegetation. There have been many examples of dramatic increases in the abundance of riparian vegetation in response to improvements in livestock management. Repeat photography has documented some of these dramatic changes in riparian vegetation and stream characteristics as a result of improved livestock management (Clifton 1989, Bradford et al. 2003, Kay 2003, Frisina and Keigley 2004).

In order to characterize the changes (i.e. improvements) in riparian vegetation, or the differences among sites, methods were needed that could provide quantitative assessments of riparian condition. In the past, researchers in academia and government have used their own methods that they have developed for riparian assessment and monitoring, not a particular protocol. These methods involved collection of species abundance data and population dynamics (such as age-class distribution for woody plants). These data have been used to monitor species abundance over time, or to evaluate differences among sites.

Scientists with the land management agencies (vegetation ecologists, fisheries biologists, hydrologists) sought to develop simpler methods that could be applied by a variety of people, such as land managers and livestock operators. In the 1990's riparian monitoring methods were developed and refined by Forest Service and BLM personnel

(Cagney 1993). Subsequently a riparian monitoring protocol was published by Winward (2000), at which time it was already in use by land managers across the Intermountain West and beyond. The effectiveness monitoring program (EMP) that resulted from the PACFISH/INFISH biological opinion (PIBO) adopted Winward's methods in 1998, when the publication was still in draft form.

An important objective of Winward's methods was to assess the condition of a riparian area and its ability to withstand natural or human disturbance. The focus was on assessing the ability of the vegetation to maintain stable banks in the future (greenline stability), and interpreting what the vegetation indicated about the amount of past disturbance (greenline successional status). These are very important concepts to consider in riparian monitoring.

Winward's methods have been taught to many land managers and livestock operators across the western US. As a result, there is a much greater knowledge about riparian vegetation and its value, especially for supporting stream ecosystems. Alma Winward has been a very effective educator about riparian ecology.

Many studies of riparian areas have had problems with their design that limited the conclusions that could be drawn from the data (Rinne and LaFayette 1991, Platts 1991, Sarr 2002). Often the studies were of relatively small areas, or of relatively few sites or just one site. In addition, the sites studied were usually not randomly selected. The sites were often selected for a particular reason, such as because they were severely impacted, and managers wanted to monitor the changes after a change in management.

In an attempt to better identify relationships between management and stream/riparian condition, the PIBO-EMP developed a study design that randomly

selected a large number of sites to evaluate the condition of stream/riparian ecosystems in the upper Columbia River basin and to monitor them on a 5-year cycle (Kershner et al. 2004).

This dissertation evaluates riparian vegetation data from the PIBO-EMP's initial years (1999 to 2002) when Winward's (2000) monitoring methods were used. These methods use community types as the cover categories to describe the vegetation of riparian areas. Eight riparian vegetation classifications were used to describe the vegetation at 325 sites in the upper Columbia River basin. In order to compare the sites, the community types needed to be consolidated across the different classifications. Chapter 2 describes how the community types from the eight classifications were evaluated and grouped into more general meta-community types. The cover data from the published riparian vegetation classifications were compiled and used to define meta-community types so that data for the entire Columbia River basin could be evaluated together. This proved facilitated the comparison of all PIBO-EMP sites.

A primary objective of PIBO-EMP was to evaluate the impacts of management on riparian systems. Chapter 3 compares the vegetation data from sites that are managed (have livestock grazing, roads, or mining activity) to reference sites, which have had less disturbance from human activities, at least in the past 30 years. This is part of the PIBO-EMP charge to evaluate the effectiveness of management activities, by the Forest Service and BLM, in maintaining riparian conditions. In addition, evaluations are presented about the relationships between vegetation and other variables, such as stream channel characteristics. This is especially important for the issue of fish habitat, which is affected

by activities that alter streambank vegetation and subsequently streambank stability, channel form, and stream conditions.

There is much variability in climate and vegetation, so these types of comparisons require a large sample size, which PIBO-EMP has achieved. Notwithstanding this large sample size of 325 sites, the dramatic environmental variability within the study area makes it difficult to compare vegetation, especially community type cover data, across such a large area. Environmental differences can overwhelm differences due to management (Cousins and Lindborg 2004).

Indices that quantify the status of certain characteristics are a useful tool to evaluate conditions and change over time. Winward (2000) developed synthetic indices, or ratings, for greenline stability and greenline successional status for vegetation community types that are common along streams. These ratings seek to assess the condition of a riparian area as reflected by the vegetation. However, the ratings of Winward (2000) were only developed for Region 4 of the Forest Service, which is only Utah and the southern half of Idaho. That does not include a major part of the PIBO-EMP study area, i.e. Oregon, Washington, northern Idaho, and Montana. Therefore a data summery technique for community types was needed so that all sites could be compared in terms of some important ecological variable.

In response to this need, a wetland rating was developed to assess the abundance of wetland species near the stream. The hypothesis was that functional riparian areas along the low-gradient (< 3%) stream reaches sampled by PIBO-EMP would have a high abundance of obligate wetland species (sensu Reed 1996) because of natural processes, such as overbank flow, infiltration of flood flows and overland flow from the hillslopes,

and a high water table associated with valley bottoms. A low abundance of wetland species near the stream would indicate that there was not as strong a linkage between the stream and the riparian area as presented by Toledo and Kauffman (2001). Such a disconnect could be a result of anthropogenic disturbances that had caused the water table to drop because of eroded streambanks and/or channel incision. Therefore it was believed that the wetland rating could be used to assess the degree to which natural processes were occurring and maintaining wetland species, and whether management activities were impeding those processes.

Chapter 4 presents a method for calculating a wetland rating, and how it was used by PIBO-EMP to compare sites with and without livestock grazing. This involved calculating a wetland rating for community types in the vegetation classifications used by PIBO-EMP to describe site vegetation. The wetland rating provided a way to summarize all of the sites, from across the large study area, in the same way. This was especially important because of the lack of summary techniques for community type data. The quantitative nature of the wetland rating also permitted a broad array of statistical analyses that could be used to compare different sites with similar environments or the same site over time.

Other ratings are needed in order to evaluate the condition of riparian sites and trend over time. PIBO-EMP is seeking to develop such ratings in association with other Forest Service, BLM, and academic ecologists. The environmental and vegetative diversity of riparian areas make it difficult to develop condition ratings for the plant species that occur in riparian areas.

In order to accurately monitor change in riparian areas it is necessary to evaluate observer variability. This allows for a determination of the amount of change that must occur before it can be confidently detected. Chapter 5 presents data from PIBO-EMP quality assurance testing on the methods of Winward (2000) that use community types to describe the vegetation at riparian sites.

The quality assurance testing provided an opportunity to evaluate the usefulness of Winward's techniques for monitoring that inevitably involves different observers. There are many factors that contribute to observer variability, particularly with the use of broad categories of community types. In order to identify some of the sources of variability, data were collected at two scales. At a small scale, comparisons were made for data from multiple observers that were collected at 1-m intervals. This isolated the differences among observers. At the larger scale, data were summarized for riparian sites (110 m of stream) in order to test both the observer agreement as well as the ability of the protocol to capture the same information with different observers.

Quality assurance testing can stimulate the generation of ideas to improve the repeatability of sampling methods. As a result of the quality assurance tests, ideas arose for improving Winward's (2000) methods, in order to improve agreement among observers. That led to tests of modifications to Winward's (2000) methods, particularly the collection of species rather than community type data. In addition to improving precision, it was thought that the modifications would improve the accuracy of the data by capturing more information about the site vegetation. The modifications that resulted from this process are presented in Appendix A.

In summary, this dissertation presents analysis of riparian vegetation data from a broad-scale monitoring project. It uses a variety of techniques – cluster analysis, similarity measures, ordination, random effects models, and synthetic vegetation indices--to facilitate analysis of riparian vegetation data and to improve our ability to detect change over time. The results presented here will improve our understanding of the important functions of riparian vegetation, and will improve our ability to detect actual changes in riparian areas, especially in relation to management activities.

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CHAPTER 2

GROUPING COMMUNITY TYPES FROM RIPARIAN VEGETATION CLASSIFICATIONS TO FACILITATE COMPARISON OF SITES DESCRIBED WITH COMMUNITY TYPES[1]

Abstract

A grouping of community types, from multiple riparian vegetation classifications, was done to facilitate comparison of sites where community type cover data had been collected. Hierarchical agglomerative clustering was performed on a Bray-Curtis dissimilarity matrix with species data from eight vegetation classifications. The species data were importance values, calculated from the product of average cover and constancy, for all species in each community type. The pruning of the clusters was done to create meta-community types that represented similar vegetation across classifications. The result was the grouping of 725 community types into 167 meta-community types. The within-group similarity of meta-community types was evaluated using multi-response permutation procedures. The mean within-group similarity for all meta-community types was 0.43, and a few meta-community types had within-group similarities as high as 0.69. The average chance-corrected within-group agreement for all meta-community types was 0.36. Each meta-community type contained an average of 4.3 community types. The average number of classifications represented in a meta-community type was only 2.5, indicating that relatively few similar community types

[1] Coauthored by Marc Coles-Ritchie, David W. Roberts, Jeffrey L. Kershner.

were detected across classifications. A few community types were ubiquitous, i.e. were observed in at least 7 of the 8 classifications, which were the *Carex nebrascensis*, *Carex rostrata*, *Deschampsia cespitosa*, *Carex aquatilis*, *Eleocharis palustris*, and *Poa pratensis* community types. The grouping of community types into meta-community types improved the ability to compare sites from across the upper Columbia River basin where community types had been used to describe the site vegetation cover. In particular, ordinations of sites became more informative because the artificial distinctions of the classifications were greatly reduced.

Introduction

Vegetation classifications describe assemblages of plant species that occur repeatedly on the landscape. Community types defined by these classifications have been used as tools to describe these common vegetation types. The community concept was developed and promoted by Clements (1936) and became a major paradigm in vegetation ecology. An alternative view was the continuum concept, originally developed by Gleason (1939) and later elucidated by McIntosh (1967) and Whittaker (1967) among others. The continuum concept was based on the idea that species vary along environmental continua, rather than being distributed according to discrete community boundaries. The continuum concept is now the dominant paradigm in vegetation ecology (Hoagland and Collins 1997), but community types continue to be described and used to communicate about vegetation patterns (see a listing of riparian vegetation classifications in Table 2.1).

Table 2.1. Riparian vegetation classifications of the interior northwestern United States that were used for data collection.

Riparian Vegetation Classification	Authors
Initial Riparian and Wetland Vegetation Classification and Characterization of the Columbia Basin in Washington	Crawford (2001)
Mid-Montane Wetland Plant Associations of the Malheur, Umatilla and Wallowa-Whitman National Forests	Crowe and Clausnitzer (1997)
Classification and Management of Montana's Riparian and Wetland Sites	Hansen et al. (1995)
Riparian Zone Associations: Deschutes, Ochoco, Fremont, and Winema National Forests	Kovalchik (1987)
Classification and Management of Eastern Washington's Riparian and Wetland Sites	Kovalchik (2001)
Riparian Community Type Classification for Humboldt and Toiyabe National Forests, Nevada and Eastern California	Manning and Padgett (1995)
Riparian Community Type Classification of Utah and Southeastern Idaho	Padgett et al. (1989)
Riparian Community Type Classification of Eastern Idaho – Western Wyoming	Youngblood et al. (1985)

Community types have been promoted by Winward (2000) as a tool for monitoring changes in the vegetation of riparian areas in relation to livestock grazing or other anthropogenic activities. An advantage of recording community types, rather than species data, is that it only requires the knowledge of the dominant species. It has been

thought that monitoring community types was easier than monitoring species because there are fewer entities to monitor (Alpert and Kagan 1998). However, determining community types and their boundaries is not necessarily easy or repeatable with different observers (Coles-Ritchie et al. 2004).

A challenge with using community types from multiple classifications to describe vegetation is that classifications define and name communities differently. That makes it difficult to compare sites in different geographical areas, where different classifications were used to describe the vegetation. An example of differences in dividing community types among classifications is with the different number of community types that have *Alnus incana* as the dominant overstory, i.e. the first species in the community type name (see Appendix B for list of *Alnus incana* community types). Three classifications (Youngblood et al. 1985; Hansen et al. 1995; Crawford 2001) describe only one community type with *Alnus incana* in the name. Three classifications describe three or four *Alnus incana* community types (Kovalchik 1987; Padgett et al. 1989; Manning and Padgett 1995). Two classifications describe 15 or more *Alnus incana* community types (Crowe and Clausnitzer 1997; Kovalchik 2001). Some of these differences are likely a function of the variable abundance of *Alnus incana* throughout the interior northwestern US. However some of the most dramatic differences (1 vs. 15 *Alnus incana* community types) are for classifications for adjacent geographical areas (Montana vs. eastern Washington). Some classifiers were apparently splitters and others were lumpers for communities dominated by *Alnus incana*.

The community type names were not sufficient to determine equivalent community types across classifications. For example, Padgett et al. (1989) described

conifer community types with generic names such as conifer/mesic forb, whereas Hansen et al. (1995) used more specific names such as *Abies lasiocarpa/Galium trifolium*, and distinguished many more types. Only in a few cases was it possible to match community types from different classifications, such as with the *Carex aquatilis* or *Poa pratensis* community types. All the classifications included references to similar community types in other classifications, but that was not done in a consistent manner for all the classifications.

This paper addresses the need for a tool to compare sites where different riparian classifications were used to describe the vegetation. This need existed because community types had been used to describe the vegetation at riparian sites (as per Winward 2000) across the upper Columbia River basin (Figure 2.1) as part the PACFISH/INFISH Biological Opinion (PIBO) effectiveness monitoring project (EMP) or PIBO-EMP (Kershner et al. 2004). Community types were evaluated, in terms of their similarity of species composition, and grouped across the eight classifications (Table 2.1) to facilitate the comparison of sites, regardless of the classification used.

Methods

Study area

The PIBO-EMP study area includes perennial streams and riparian areas on federal (Forest Service and BLM) land within the upper Columbia River basin, which includes major portions of Washington, Oregon, Idaho, and Montana, and a small part of Nevada (Kershner et al. 2004) (Figure 2.1). The sites sampled were response reaches – which are low gradient reaches of wadeable streams (usually 2nd or 3rd order), within unconstrained valley bottoms, where impacts from excessive disturbance upstream could

be manifested. The stream reaches were sampled during the 2000, 2001, or 2002 field seasons (June to September).

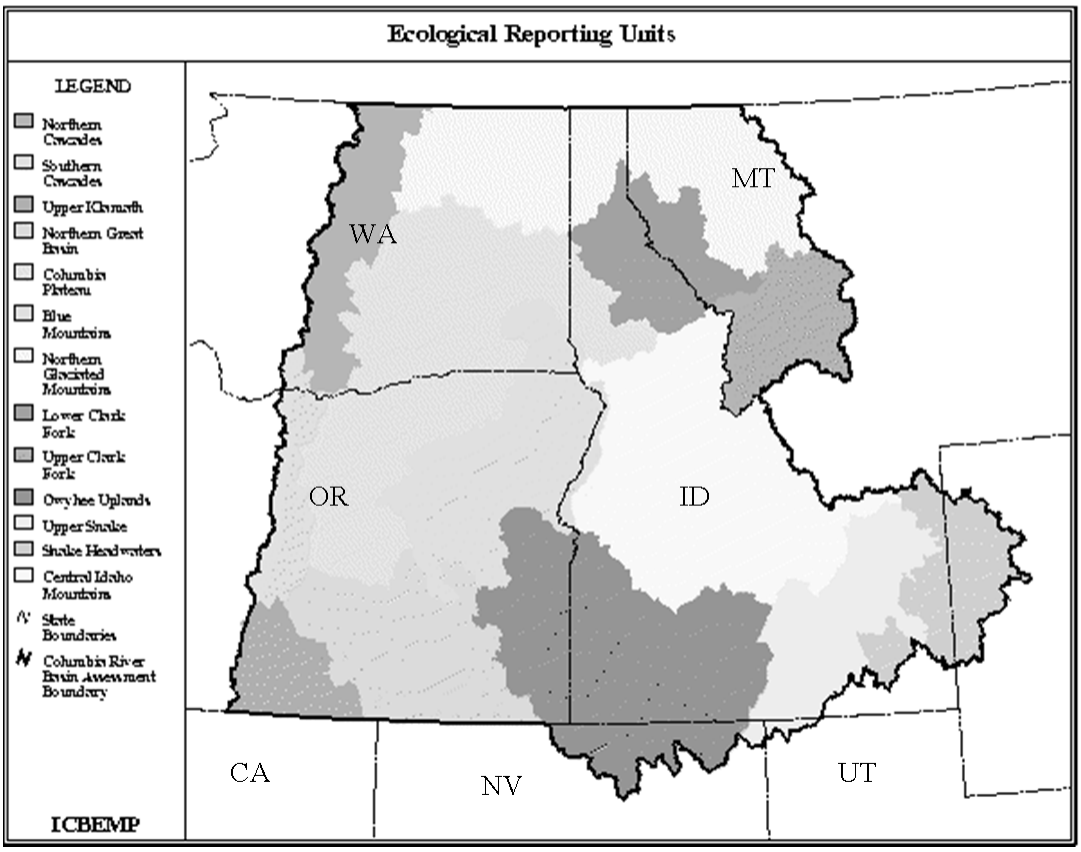

Figure 2. 1. Map with the upper Columbia River basin (within the US) outlined, which is the study area of the PACFISH/INFISH effectiveness monitoring project. States and ecological reporting units are also identified.

Data collection

The PIBO-EMP used the riparian vegetation data collection methods of Winward (2000), which were being used by numerous entities that managed or monitored riparian areas on public lands. The vegetation data were collected over a distance of 110 m of streambank, on both sides of the stream, as per Winward (2000). Data were collected in two areas relative to the stream: the greenline, which is immediately adjacent to the

stream, and vegetation cross-sections, which are transects across the riparian area (perpendicular to the direction of the valley bottom). The data collectors recorded the community type for each step as they walked along the greenline and the cross-sections. Eight riparian vegetation classifications (Table 2.1) were used to determine the community type based on the dominant species or general species composition.

Community type data for cluster analysis

Data on the species composition of community types were needed in order to group the community types from all of the classifications. The average species cover and constancy data for all the community types in the 8 classifications were obtained from the authors of the classifications or from the appendices of their publications. Average cover represented the average for each species for sites where that species was recorded among the sites classified as that community type. Constancy represented the percentage of sites, among those used to define that community type, where the species was recorded. Average cover and constancy were multiplied together to obtain an importance value for all species in the community types.

Many species names listed in the classifications had to be updated in order to perform dissimilarity calculations among the community types of the eight classifications. The PLANTS database (USDA 2002) and the "Flora of the Pacific Northwest" (Hitchcock and Cronquist 1998) were used to determine the currently accepted scientific names. Some plants in the classifications, about 4%, were listed only to genus (i.e. *Trifolium* spp.), so they were not included in the analysis to avoid spurious or inadequate similarity determinations among genera and species. The resulting matrix contained 725 community types and 1,359 species.

Cluster analysis

Cluster analysis is a form of classification that summarizes data by grouping into clusters of similar objects (Gordon 1999). Cluster analysis uses distance measures to create groups with high within-group similarity, and ideally low between-group similarity. In this study, a hierarchical agglomerative clustering technique (Dash et al. 2003) was performed using PC-ORD for Windows (McCune and Mefford 1999). Agglomerative indicates that clusters were formed by joining the closest pair of objects (community types or a cluster of community types) in each iteration. Hierarchical indicates that clusters were identified within clusters. In addition, the method was polythetic, in that multiple species were used to decide on the fusion of clusters, which is preferable to monothetic methods that use only one species to make clustering decisions and thereby ignore much of the information in the data (Greig-Smith 1980).

In order to perform the cluster analysis, a Bray-Curtis dissimilarity index was used to determine the dissimilarity of all the community types based on their species importance values. The linkage method was flexible beta, with a beta of –0.25, which is recommended by McCune and Grace (2002) because it is a space conserving method, which avoids space distortion.

It was important to maintain distinctions between vegetation dominated by different physiognomic groups. Therefore, community types were sorted by the physiognomic group of the species listed in the community type name (which indicates the dominant overstory), with the following 5 groups: conifer, deciduous tree, non-willow shrub, willow, and herbaceous. Cluster analyses were then performed separately for each of the physiognomic groups.

Dendrograms generated by cluster analysis were cut, or pruned, into clusters of similar community types. Pruning was done based on the dominant species, ecological knowledge from observing those community types in the field, and information about the common species of each community type found in the classification.

The objective of clustering for this program was to create clusters that represented similar plant communities from across the study area, which would enable comparison of sites within the entire study area. Therefore it was desirable to have community types from multiple classifications represented in each cluster, and as few community types from the same classification as possible. In cases where a cluster had community types from only one or two classifications, the most similar cluster was evaluated to see if combining the clusters would add community types from different classifications. If so, and the grouping made ecological sense (based on the community type names), then the clusters were combined. If combining clusters merely added community types from the same classification(s), then the clusters were not combined.

The results of the pruning were meta-community types that represented groups of similar community types. In a few cases only one classification was represented in a meta-community type because expanding it would have merely added community types from the same classification. In these cases the individual community types, rather than a group of community types from the same classification, were used in the comparison of sites. Combining community types from the same classification would not have met the objective of improving the ability to compare sites where different classifications had been used, and it would have eliminated distinctions that technicians made between those vegetation types in the field.

Evaluation of clusters

Post-hoc evaluation of the clusters included consideration of both chaining and within-group agreement. Percent chaining reflects the average number of nodes from the tip of a branch to the trunk of the dendrogram. A lower percent chaining value is preferred because that indicates that there are relatively distinct clusters (McCune and Grace 2002).

Within-group agreement, i.e. the level of homogeneity within each meta-community type, was evaluated with multi-response permutation procedures (MRPP) – using PC-ORD for Windows (McCune and Mefford 1999). MRPP uses the distance measures between all entities in a priori groups to determine the level of similarity within the groups, and it does not require that data be from a normal population (Zimmerman et al. 1985). MRPP can also be used to test the hypothesis of no difference between two or more groups of entities (McCune and Grace 2002).

Within-group dissimilarities were calculated with the Bray-Curtis dissimilarity index, in PC-ORD. A p-value (P) indicates the proportion of possible within-group distance values that are less than the observed value (Zimmerman et al. 1985). The agreement statistic (A) represented the "chance-corrected within-group agreement" (McCune and Grace 2002).

The A statistic was calculated as follows:

$$A = 1 - \frac{\text{observed } \delta}{\text{expected } \delta} \qquad (1)$$

where δ is the weighted mean within-group dissimilarity.

The A statistic would be 1 if all entities (community types in this case) in a cluster were identical (i.e. had the exact same species composition). The A statistic would be 0 if the items in the group had the same heterogeneity as expected by chance. A negative value would indicate more heterogeneity than expected by chance. Therefore a larger (more positive) A statistic, indicates greater within-group similarity.

Ordinations

Ordination is a method of arranging sites based on their similarity of vegetation and/or their environmental conditions (Kent and Coker 1992). In this study ordinations were done with the vegetation data for 325 sites, for both the greenline and cross-section data sets. Ordinations of site vegetation data were done two ways: 1) with the community type cover data; and 2) with meta-community types substituted for the community types.

In order to perform the ordinations, a Bray-Curtis dissimilarity index compared community types among all of the classifications, with PC-ORD for Windows (McCune and Mefford 1999). The Bray-Curtis dissimilarity index was recommended by Minchin (1987) and McCune and Grace (2002). The dissimilarity values were used to generate the ordinations with non-metric multidimensional scaling (NMDS). NMDS is considered a relatively accurate low-dimensional representation of a high dimensional space (Clymo 1980; Minchin 1987).

Preliminary ordinations were calculated to determine the appropriate number of dimensions, the best starting configuration, and to perform a Monte Carlo test with randomized runs. The Monte Carlo test was done to evaluate the probability that a similar final stress could be achieved by chance. “Stress” represents the degree to which

the dissimilarity in the ordination (with the number of dimensions selected) differs from the dissimilarity in the original dissimilarity matrix (McCune and Mefford 1999). In PC-ORD (McCune and Mefford 1999) raw stress is first calculated as:

$$S^* = \sum_{i=1}^{n-1} \sum_{j=i+1}^{n} (d_{ij} - \hat{d}_{ij})^2 \qquad (2)$$

Stress is then normalized with the following equation:

$$S = 100 \sqrt{S^* / \sum_{i=1}^{n-1} \sum_{j=i+1}^{n} d_{ij}^2} \qquad (3)$$

For all preliminary runs, a randomized starting configuration was used, the maximum number of iterations was 400, the instability criterion was 0.00001, the starting number of dimensions was 6, the number of runs with real data was 40, and the number of runs with randomized data was 50 (Table 2.2). The calculation for instability is the standard deviation in stress over the preceding x iterations, where x is set by the user (McCune and Mefford 1999).

For each final ordination, the number of dimensions (axes) was selected according to the following criteria: an additional dimension was added if it reduced the stress by 5 or more, and if that stress was lower than 95% of the randomized runs ($P <= 0.05$ for the Monte Carlo test) as recommended by McCune and Mefford (1999).

The final run was done with the best starting configuration for the number of dimensions selected and no step-down in dimensionality (Table 2.2). For final runs the

following information was reported: the number of dimensions that were statistically different than 95% of the randomized runs (Monte Carlo test results), number of iterations, final instability, final stress, and the cumulative R^2 based on the correlation coefficient between the ordination distances and distances in the original n-dimensional space.

Table 2. 2. The settings used to run the NMDS ordinations using the Bray-Curtis dissimilarity index. The preliminary run included runs with randomized data for the Monte Carlo test to determine the appropriate number of dimensions (axes).

	Starting configuration	Maximum iterations	Instability criterion	Dimensions	Runs (real data)	Runs (randomized data)
Preliminary Run	random number	400	0.00001	6	40	50
Final Run	preliminary run with lowest final stress (for dimensions selected)	200	0.0001	1-6	1	none

Ordinations of the sites were presented in graphs and as joint plots that use vectors to represent the relationship (strength, or vector length, and direction) of environmental or site variables to the vegetation ordination. Because there were over 40 variables (environmental, management, and stream characteristics) only the vectors for variables with the strongest correlation to the axes were portrayed in the joint plots, with

an $R^2 > 0.15$ used as the cutoff. The R^2 values for variables in the joint plots were calculated as the length of the hypotenuse of a right triangle for which the other two sides were the R^2 values related to each of the axes of the ordination (McCune and Grace 2002).

The combination of axes to present (of the three possibilities) was selected so that the joint plot included many explanatory variables with vectors pointing in different directions. This was done to elucidate the relationships of different vegetation with different environmental and management variables. All the ordinations were rotated and reflected in order to align the dominant explanatory variables, precipitation and elevation, with the vertical and horizontal axes, respectively. Precipitation increases moving up the vertical axis and elevation generally increases from left to right for all the ordinations. This was done to facilitate detection and reporting of patterns.

Results

Combining community types from different classifications

The cluster analysis generated dendrograms that depicted clusters of similar community types from the eight riparian vegetation classifications. The dendrogram for each of the physiognomic groupings of community types was pruned, as indicated by slashes on the dendrograms (Figure 2.2). The pruning reduced the original 725 community types to 167 meta-community types. Appendix B lists some of the meta-community types and the community types contained within them.

The pruning resulted in relatively distinct clusters, or meta-community types, as indicated by the percent chaining that was below 2% in all of the dendrograms (Table

2.3). This is within the acceptable range and well below what would be considered "extreme chaining" (over 25%) by McCune and Grace (2002).

Post hoc evaluation of the meta-community types with MRPP indicated a mean within-group distance of 0.57, or a mean similarity of 43%. The A statistics were over 0.33 for all 5 of physiognomic groups (Table 2.4), which is considered "fairly high" by McCune and Grace (2002). Each meta-community type represented between 1-13 community types, with an average of 4.3 community types. There were an average of 2.5 classifications, out of eight possible, represented in each meta-community type, indicating that for most community types, there were few similar community types in other classifications. In five cases the meta-community type had only one community type, but four of those were community types that did not appear in our data set, probably because they were uncommon vegetation types.

A few community types were widespread and relatively distinct. The *Carex rostrata* (now referred to as *Carex utriculata*) and the *Deschampsia cespitosa* meta-community types had all eight classifications represented. The *Carex rostrata* meta-community type also included two to three community types from some of the classifications, while *Deschampsia cespitosa* meta-community type included only one classification that had two community types.

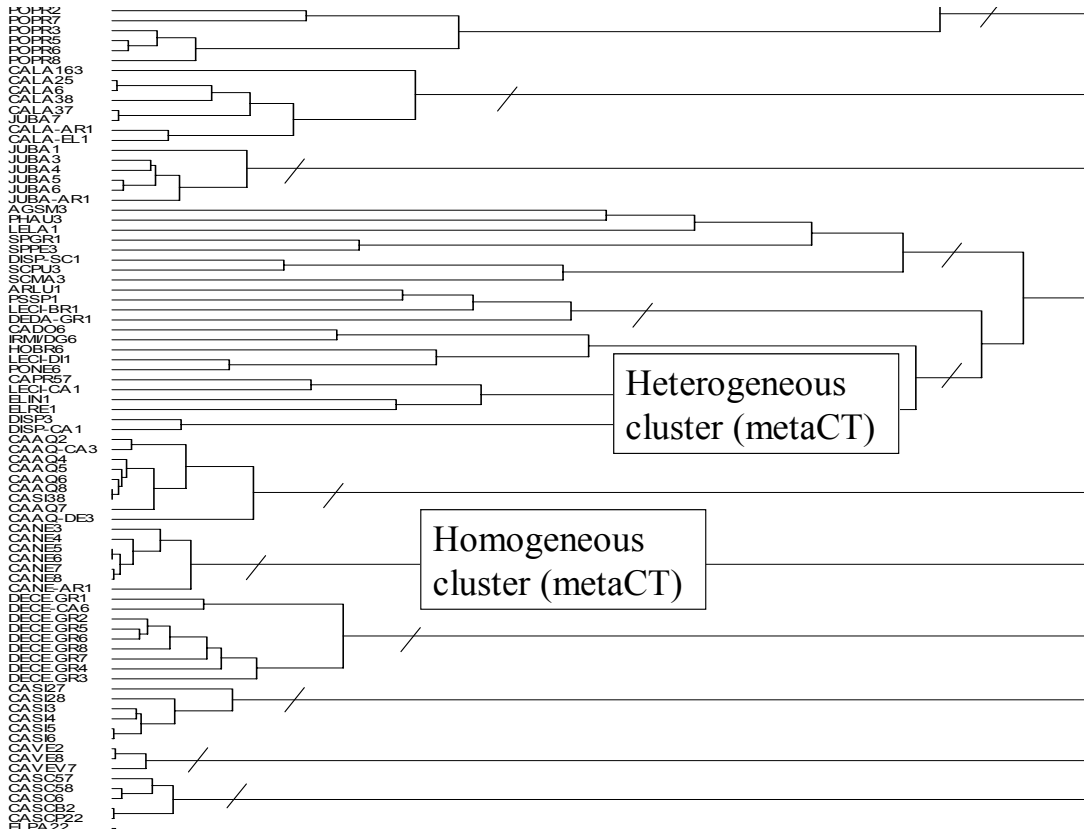

Figure 2. 2. A portion of the dendrogram, generated by cluster analysis, of herbaceous community types from riparian vegetation classifications in the upper Columbia River basin. The pruning of clusters is indicated by diagonal lines.

The *Carex nebrascensis* meta-community type (indicated in Figure 2.2) had seven community types from seven different classifications and was relatively distinct from any other community types. Three other meta-community types had seven classifications represented (*Carex aquatilis* meta-community type [see Appendix B], *Eleocharis palustris* meta-community type, and *Poa pratensis* meta-community type), but there were also multiple community types from the same classification. Meta-community types with

Table 2.3. The number of community types, from eight classifications, used in cluster analysis, within physiognomic groups, and the resulting number of "meta-community types" after pruning of clusters.

Physiognomic group	Community types	Meta-community types after pruning	Percent chaining
Conifer	150	30	1.2%
Deciduous tree	98	23	1.2%
Shrub	138	36	1.1%
Willow	131	34	1.9%
Herbaceous	208	44	1.7%
Total	725	167	

Table 2.4. Results of the multi-response permutation procedure (MRPP) that evaluated the meta-community types generated from cluster analysis. Mean within-group distance is on a scale of 0 (identical) to 1 (completely dissimilar).

Physiognomic group for which cluster analysis was performed	Observed mean within-group distance	Chance-corrected within-group agreement (A statistic)	P
Conifer	0.57	0.33	< 0.01
Deciduous tree	0.58	0.33	< 0.01
Shrub	0.54	0.40	< 0.01
Willow	0.54	0.40	< 0.01
Herbaceous	0.62	0.35	< 0.01
Grand mean	0.57	0.36	

six classifications included four herbaceous meta-community types (*Carex lanuginosa* meta-community type, *Eleocharis pauciflora* meta-community type, *Dodecatheon jeffreyi* meta-community type, and *Carex simulata* meta-community type), one shrub meta-community type (*Potentilla fruticosa*/mesic graminoid meta-community type), and one willow meta-community type (*Salix*/*Carex rostrata* meta-community type).

Ordinations

The greenline and cross-section ordinations, with community types and meta-community types, had six dimensions that were statistically better than 95% of the runs of randomized data, but only three dimensions that each lowered the minimum final stress by five (Table 2.5). Therefore three dimensions were used for the final run with real data. The final instability values for the greenline ordinations (with community type and meta-community type data) were 0.003 and 0.00009 and 0.0005 for cross-section data with community type and meta-community type data (respectively), which is near or below the level recommended by McCune and Grace (2002). The final stress values were 21.5 and 22.8 for the greenline ordinations with community type and meta-community type data (respectively) and 23.9 for the cross-section data (for both community type and meta-community type data), which is considered "poor" according to Kruskal's rules of thumb listed in McCune and Grace (2002). The high stress values could decrease confidence in the results, but guidelines on stress are "over-simplistic" according to Clarke (1993) in McCune and Grace (2002), in part because of the general pattern of stress increasing with sample size.

Table 2.5. The final solutions of the NMDS ordinations for the data sets that used community types and meta-community types to describe riparian sites. The dimensions selected were both statistically significant and decreased the stress by at least 5.

Data set	Dimensions selected	Iterations	Final Instability	Final Stress	R^2
Greenline community type	3	200	0.00294	21.5	43.1
Greenline meta-community type	3	200	0.00267	22.8	46.7
Cross-section community type	3	183	0.00009	23.9	33.6
Cross-section meta-community type	3	200	0.00038	24.2	34.4

The reason that there was some overlap among states in the ordinations is because at some sites multiple classifications were used. If the classification that was geographically appropriate did not adequately describe the vegetation at a site then technicians consulted classifications from adjacent areas. In addition, some areas had no riparian vegetation classification; therefore multiple classifications from adjacent areas were used. For example, in Northern Idaho, where no riparian vegetation classification was available, the classifications for Montana (Hansen et al. 1995) and Washington (Kovalchik 2001) were used (Kershner et al. 2004).

The joint plot vectors showed that the strongest variables in relation to the vegetation ordination were the three Regions of the Forest Service (Figures 2.3 and 2.4).

The other variables of interest – environmental, management, and stream characteristics – were overwhelmed by the geographical variables.

Ordinations with meta-community type site data

The grouping of community types with cluster analysis reduced the greenline data set to 142 meta-community types from the original 436 community types. The cross-section data set was reduced to 140 meta-community types from the original 382 community types.

The ordinations with meta-community types resulted in a mixing of sites from different states (Figures 2.5 and 2.6), compared to the geographical clustering of sites when community type data were used (Figures 2.3 and 2.4). The average R^2 value for a geographical variable, relative to the axis that it was most correlated with, dropped from 0.22 for the ordinations with community type data, to 0.10 with the meta-community type data. In addition, the joint plot vectors do not show the geographical variables as the dominant site variables in relation to the vegetation ordination, as did the joint plots with community type data.

There was still some relationship between vegetation (meta-community types) and geography, as would be expected. The MRPP calculation indicated that the mean within-group distance for the five states was statistically lower ($P < 0.01$) than would be expected by chance for both the greenline and cross-section data (Table 2.6). Some difference in vegetation among states was still evident, as would be expected, but the homogeneity of vegetation within states was not great, as indicated by the relatively low A statistic values of 0.041for the greenline and 0.045 for the cross-sections.

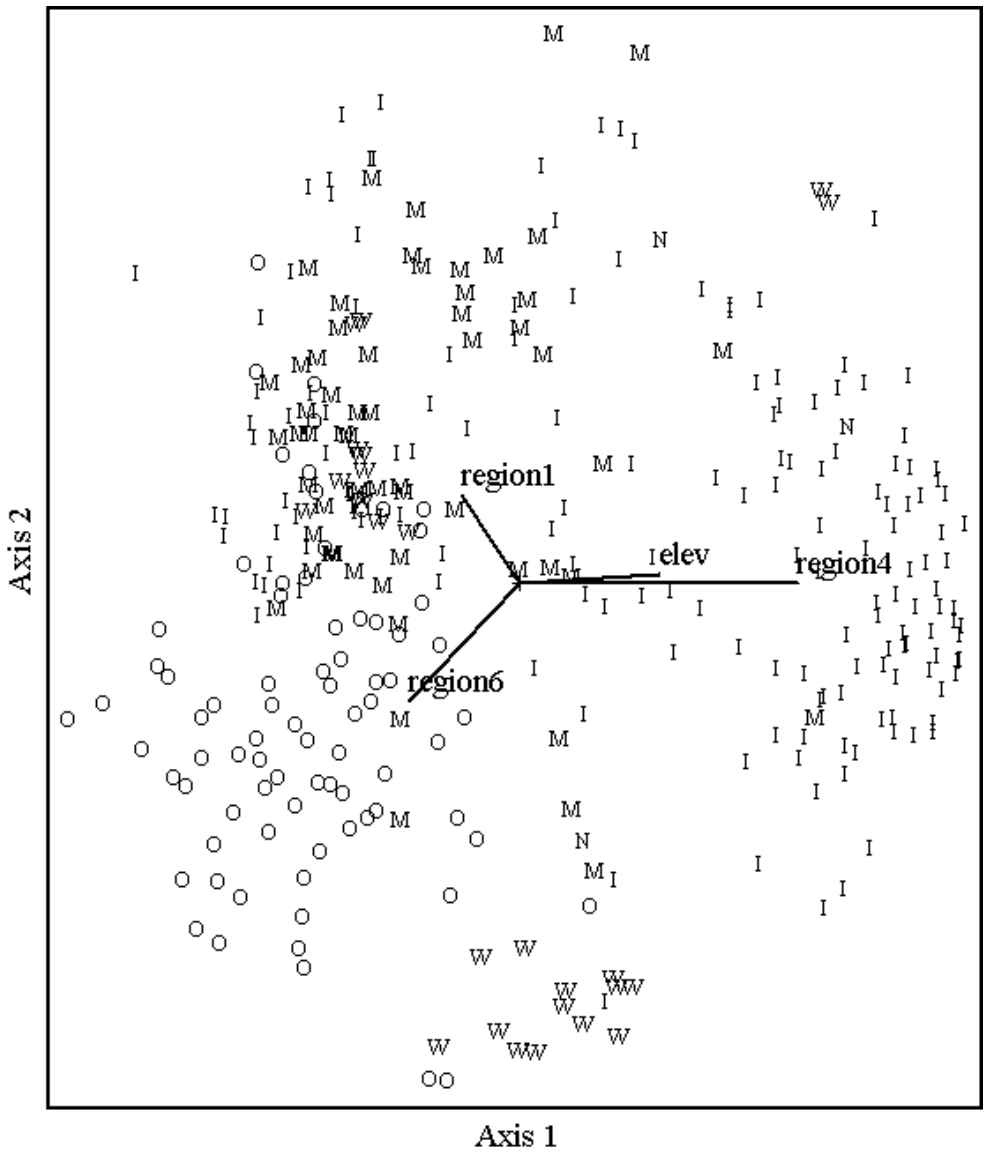

Figure 2.3. An NMDS ordination of site greenline vegetation data, based on community type cover. Sites are represented by the first letter of the state where they are located (Idaho, Montana, Nevada, Oregon, and Washington).

The MRPP analysis with the other geographical variables--National Forest and Region (Forest Service)—also had $P < 0.01$ for both greenline and cross-section data (Table 2.6). The A statistics for Region, 0.056 and 0.050 for greenline and cross-section data, respectively, were slightly higher than for State. Of the three geographical

variables, National Forest had the highest A statistics of 0.095 and 0.084 for greenline and cross-section data respectively. The National Forest category is the smallest geographical unit, so it makes sense that it would have the highest degree of homogeneity.

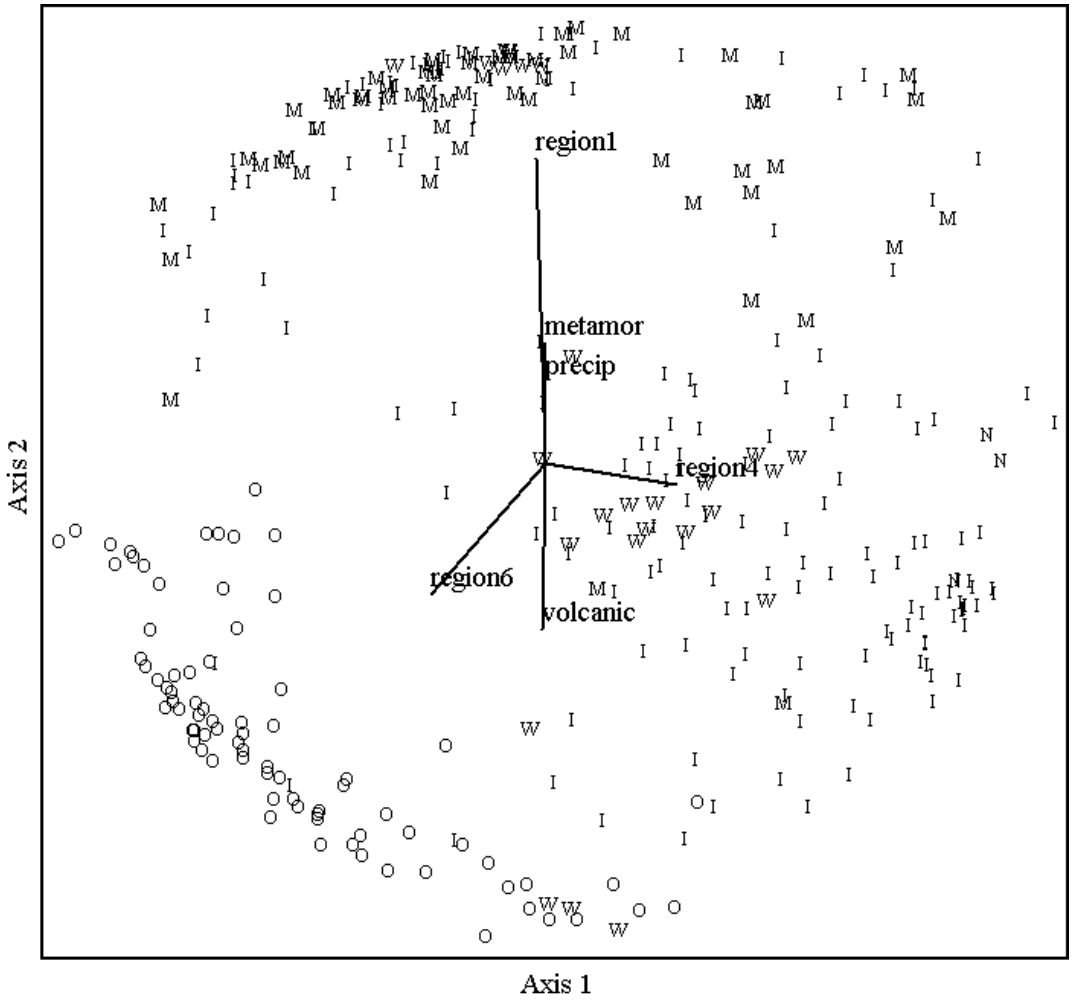

Figure 2. 4. An NMDS ordination of site cross-section vegetation data, based on community type cover. Sites are represented by the first letter of the state where they are located (Idaho, Montana, Nevada, Oregon, and Washington).

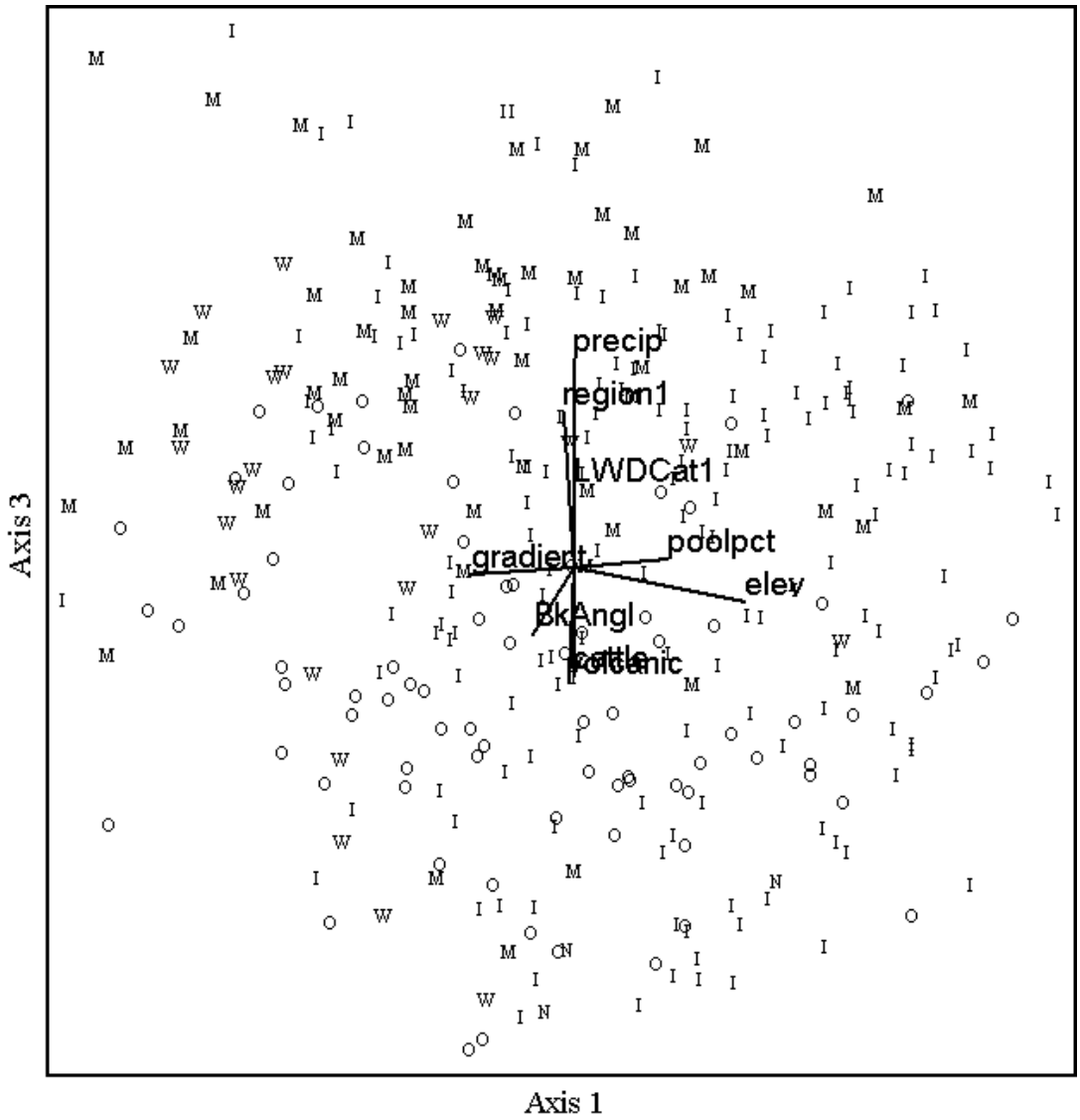

Figure 2. 5. An NMDS ordination of site greenline vegetation data, based on meta-community type cover. Sites are represented by the first letter of the state where they are located (Idaho, Montana, Nevada, Oregon, and Washington).

Discussion

While the concept of discrete communities has largely been replaced by the continuum concept in vegetation ecology (Hoagland and Collins 1997), community types continue to be described and used in communication about vegetation. In fact, the term

"community" continues to be used by most applied ecologists (Austin and Smith 1989). The community concept seems to have an inherent appeal, regardless of the difficulty of defining communities on the ground. An example of this is the promotion of community types for monitoring riparian vegetation (Winward 2000).

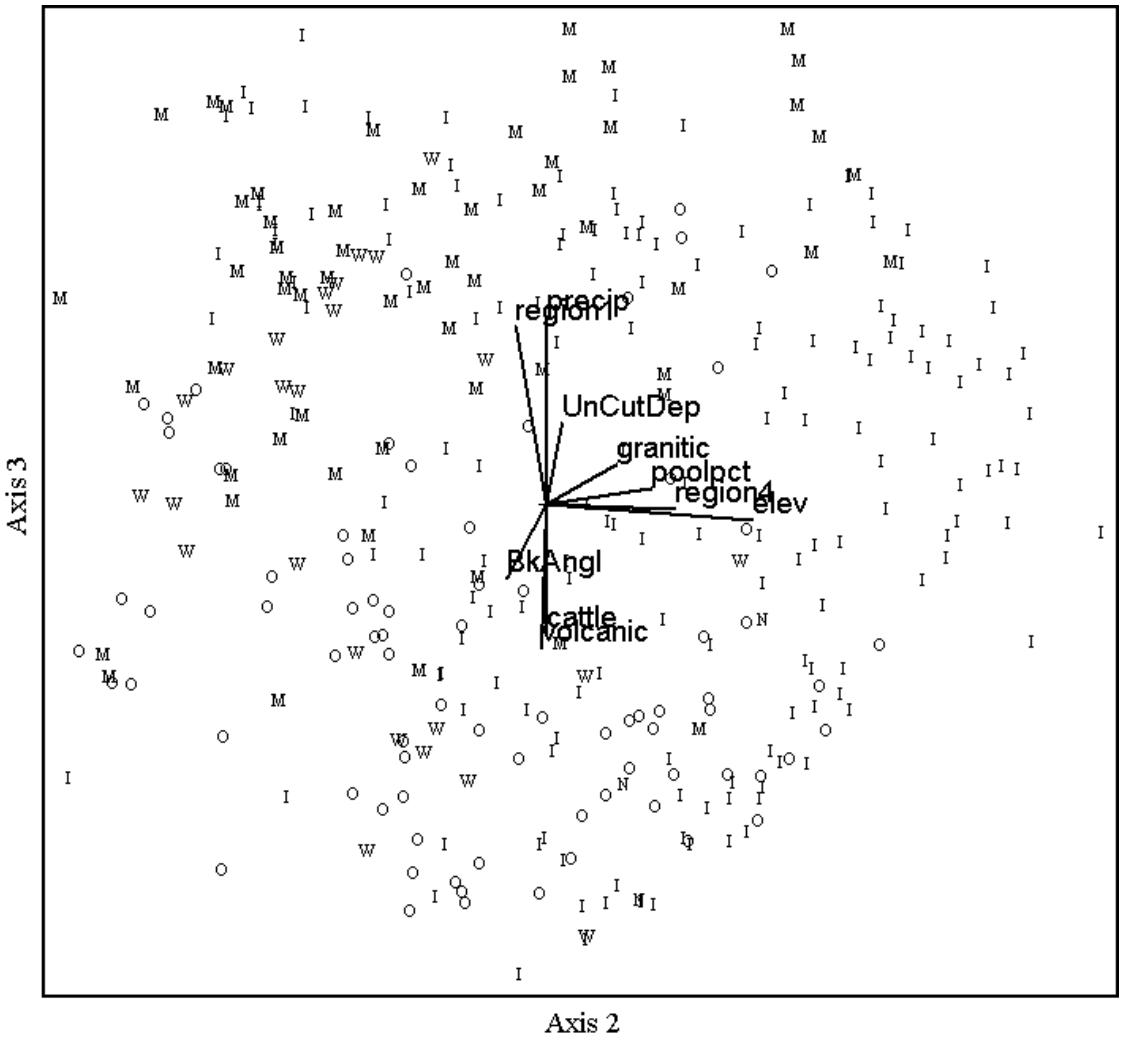

Figure 2.6. An NMDS ordination of site cross-section vegetation data, based on meta-community type cover. Sites are represented by the first letter of the state where they are located (Idaho, Montana, Nevada, Oregon, and Washington).

PIBO-EMP followed Winward's protocol to describe vegetation in riparian areas, but because multiple community type classifications were used, it was necessary to group community types across classifications. This was done with cluster analysis, which converted community types to meta-community types. The mean within-meta-community type similarity was 43%, and the chance-corrected within-group agreement was 0.36, which are satisfactory values, but they indicate that there is variability within meta-community types. Therefore the conversion to meta-community types meant that there was a loss of some distinctions among community types that were made at sites. Most community types were similar to only a few community types from other classifications. This suggests that either (1) there are few distinct community types across large geographical areas, or (2) differences in how the classifiers collected and grouped their data interfered with the recognition of common community types between classifications. The first point could be a function of geographical differences in vegetation, which would support the continuum concept of species distribution. All the

Table 2. 6. Evaluation of within-group homogeneity of vegetation, based on meta-community types, for groups of sites according to categorical geographic variables.

		Greenline		Cross-sections	
Variable	Groups	Agreement statistic	P	Agreement statistic	P
National Forest	22	0.095	< 0.01	0.084	< 0.01
Region	5	0.056	< 0.01	0.050	< 0.01
State	5	0.045	< 0.01	0.041	< 0.01

classifications listed in Table 2.1 refer to “similar” community types in other classifications, but that is based on the community type names and professional judgment rather than calculations of similarity in terms of species abundance.

There are a number of reasons why the differences in how the classifiers collected and grouped their data would have affected the ability to recognize comparable community types across classifications. It could be that the small sample sizes, i.e. the low number of plots used to define each community type, prevented the accurate characterization of community types. The median number of plots used to describe a community type was 7, which is a respectable number of plots, but half the community types were characterized with fewer plots. For two classifications (Crowe and Clausnitzer 1997; Crawford 2001) the median plots per community type were only 3 and 4 respectively. For many community types in those classifications only a couple of plots were used to characterize that community type, which would not always be enough to accurately represent the species composition. That would inhibit the detection of similar community types across classifications.

Another factor that could have prevented the detection of similar community types across classifications is the variability in how community types were defined. Different types of clustering and ordination were used by the different classifiers. There is subjectivity that is unavoidable in classification, which can lead to very different community types among the classifications. This makes it difficult to match community types across classifications.

Another difference in classification methods was whether, or how, classifiers “fine-tuned” their classifications. In some cases classifiers adjusted community types

based on the presence of a single species, which is a monothetic strategy. Whereas other classification methods, such as hierarchical agglomerative clustering, were based on a polythetic approach, which considers multiple species when deciding on community types.

It is noteworthy that a few community types were ubiquitous such as the *Carex nebrascensis*, *Carex rostrata*, *Deschampsia cespitosa*, *Carex aquatilis*, *Eleocharis palustris*, and *Poa pratensis* community types. These community types, which almost always had the same name, had similar species composition in at least seven of the eight classifications, and few differently-named community types with equally similar vegetation. This indicates that these riparian communities recur repeatedly throughout the upper Columbia River basin. These are herbaceous community types that are dominated by rhizomatous species (in all but one case), and all of them can be mat-forming and therefore can dominate a small area. The growth form of those species (clonal and dense) makes it understandable how they could form definable communities, or patches at least. The fact that no community types dominated by woody species were ubiquitous suggests that it is harder to define discrete communities with woody species in the overstory. For example, it is difficult to define criteria that define how much willow cover is needed to distinguish a sedge community type from a willow/sedge community.

Using meta-community types decreased the geographical clustering of sites that was observed in the ordinations with community type data. With meta-community types there were still some correlations between ordination axes and geographical variables, as would be expected. The geographical variable that represented the smallest area, National Forest, had the greatest within-group similarity. That is consistent with the

"distance-decay" concept, which suggests that objects that are near are more similar than objects that are further apart (Palmer and White 1994). But with meta-community types the other variables of interest were not completely overwhelmed.

The conversion of community types to meta-community types will facilitate the comparison of sites where different classifications had been used to describe the riparian vegetation. That in turn, will increase our ability to detect patterns related to vegetation and other variables such as management, environment, and channel characteristics, however, using clustering to combine community types is not a perfect solution because information is inevitably lost when entities are grouped. The variability within meta-community types indicates that some information about vegetation is being lost in the post-hoc conversion of community types to meta-community types. Therefore, some distinctions between community types on the ground (which are imprecise in themselves) were lost when community types were grouped into meta-community types after the data were collected. Avoiding the need to cluster community types may be a good alternative. This could be done by collecting cover data that are compatible across a large landscape, such as species data. An increased skill level would be needed for species identification, but the added cost in time or expertise would likely be offset by the benefits, among others, of having data that can be directly compared among all sites.

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CHAPTER 3

THE RELATIONSHIP OF VEGETATION, ENVIRONMENT, AND MANAGEMENT ACTIVITIES AT RIPARIAN SITES IN THE INTERIOR NORTHWESTERN U.S.[2]

Abstract. Riparian vegetation at sites from across the upper Columbia River basin was evaluated in relation to environment, management, and stream variables. Similarity indices and ordinations were used to compare riparian sites based on the relative representation of meta-community types –groupings of community types from multiple vegetation classifications. Grouping community types facilitated the comparison of sites from across the entire study area, but vegetation information was lost as a result of the grouping. The most important environmental variables in relation to the vegetation ordination of the riparian sites were precipitation and elevation. Of particular interest were the differences between sites in managed vs. reference watersheds. The within-group similarities of sites in managed and reference watersheds were only slightly greater than would be expected by chance for this large data set. Sites in watersheds with cattle grazing and low precipitation generally occurred in proximity to each other in one area of the ordination. After accounting for precipitation, the cattle-grazing variable was still important in explaining the distribution of sites, and hence the vegetation at those sites.

Vegetation, environment, management, and stream characteristics were used to describe four types of riparian sites: 1) conifer-alder forests; 2) wet sedge-willow sites; 3) dry-grazed sites; and 4) steeper-shrub sites. Hypotheses are presented about how the characteristics of these types of sites affect how they respond to management, and livestock in particular. The conifer-alder forests and the wet sedge-willow sites appear to

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[2] Coauthored by Marc Coles-Ritchie, David W. Roberts, Jeffrey L. Kershner.

be less impacted by management than the dry-grazed sites. The combination of low precipitation and cattle grazing seems to make sites more vulnerable, since these sites were dominated by vegetation indicative of disturbance (the *Poa pratensis* community type) and unstable streambanks (flattened streambanks and little bank overhang). Managers can use this information to promote appropriate management of cattle in areas with low precipitation.

INTRODUCTION

The condition of riparian areas in the western US has become increasingly important for government agencies such as the US Forest Service and Bureau of Land Management (USDA/USDI 1995, USDI 2000). In order to understand the condition of these areas it is important to understand the relationships between riparian vegetation, environment, stream characteristics and management activities. The Biological Opinions for salmon, steelhead, and bull trout identified the need for monitoring of streams and riparian areas within the areas covered by the Pacific Anadromous Fish Strategy (PACFISH) and the Inland Fish Strategy (INFISH) (USDA/USDI 1995). The PACFISH/INFISH Biological Opinions (PIBO) resulted in the creation of an effectiveness monitoring project (PIBO-EMP) that was charged with determining whether management was effective in maintaining or improving aquatic and riparian conditions on federal lands within the upper Columbia River basin (Fig. 2.1) (Kershner et al. 2004).

Riparian vegetation provides many functions that have both ecological and social importance. Riparian species have characteristics that help increase bank stability,

dissipate stream energy, trap sediment, shade the stream, promote water infiltration, and mitigate floods (Tabacchi et al. 1998).

The effect of riparian vegetation in stabilizing streambanks and capturing sediment influences channel form (Abernathy and Rutherford 2000, Gran and Paola 2001). Streambanks with less vegetation have wider channels (Ikeda and Izumi 1990, Millar and Quick 1993, Rowntree and Dollar 1999). In floodplain streams, vegetation can be particularly important in reducing bank erosion (Abernathy and Rutherford 1998). The effect of riparian vegetation in maintaining channel form also relates to the degree of habitat complexity (pools and riffles) which is critical to maintain spawning habitat for endangered salmonids (see reviews by: Belsky et al. 1999, Fitch and Adams 1998).

The feedbacks between vegetation, streambank stability, and channel form are consistent with the "dynamical systems perspective," which indicates that there are feedbacks between vegetation and the environment (Roberts 1987). Riparian vegetation affects and is affected by hydrologic processes. Therefore riparian vegetation can serve as an integrative variable, which informs about the condition of the system and the presence of certain functions.

This paper evaluates the relationship of riparian vegetation to other variables including environmental, geographical, management, and stream channel variables. The objectives are: 1) to evaluate the relationship between riparian vegetation and environmental conditions, that will indicate which environmental variables need to be accounted for in comparisons of sites with different management activities; 2) to determine the relationship between vegetation and management activities, which will indicate whether management activities have altered the vegetation at managed sites; and

3) to evaluate the relationship between vegetation and stream variables, in order to identify possible linkages between management activities, changes in vegetation, and changes in stream channel characteristics.

METHODS

Study sites

The PIBO-EMP study area includes federal land (Forest Service or BLM) within the upper Columbia River basin (Fig. 2.1). PIBO-EMP samples riparian areas associated with perennial streams that have a gradient less than 3%, unconstrained valley bottoms, and bankfull widths between 1 and 15 m (see Kershner et al. 2004 for a description of the site selection process). These reaches are considered response reaches because they can manifest the impacts of activities in the upstream watershed.

Data collection

Vegetation data

Riparian vegetation data were collected based on the methods of Winward (2000) as presented in Kershner et al. (2004) for PIBO-EMP. Five to eight technicians per year were trained to apply the protocol in the collection of the vegetation data. The riparian areas analyzed here were sampled during the 2000, 2001, or 2002 field seasons (June to September).

The technicians received two weeks of training on the methods and identification of riparian plants. Each technician collected the vegetation data independently at a

stream site, while data in the stream channel were collected by other PIBO-EMP technicians. Data collection at a site was usually accomplished in one day.

The methods of Winward (2000) rely on the identification of community types as the observed data, rather than species. Eight riparian vegetation classifications (see Table 2.1) were used in this study. At each stream site the most geographically appropriate classification was used to determine the community types present (see Kershner et al. 2004 for a listing of classifications to use by geographical area).

As per Winward (2000) the riparian vegetation data were collected over a distance of 110 m of streambank, on both sides of the stream. The vegetation and stream channel data collection began at the same point, which was considered the downstream end of the reach (hereafter referred to as site). The stream data were collected over a variable distance, based on channel width, which ranged from 80 to 280 m.

Data were collected in two areas relative to the stream: the greenline and vegetation cross-sections. The greenline was considered to be the first line (rooted point) of perennial vegetation adjacent to the stream that was at least 1 foot wide, with at least 25% cover of vegetation. The data collector paced along this greenline and at each step recorded the community type for the area between steps and 1 foot in width. The community type was determined based on the dominant species or general species composition, which was sometimes a difficult and subjective process (Coles-Ritchie et al. 2004).

Five cross-sections were sampled along the 110 m of stream at evenly spaced intervals. For each cross-section the observer paced a transect perpendicular to the direction of the valley bottom (not necessarily perpendicular to the stream) and recorded

the community type for the area between each step, similar to the greenline. The cross-sections extended in both directions from the greenline to the edge of the riparian area or 27.5 m whichever was encountered first. The edge of the riparian area was determined to be the point where the vegetation was no longer a riparian community type listed in the classifications, or did not have greater than 25% cover of riparian species.

Non-vegetation data

Non-vegetation data were collected by the PIBO-EMP about each of the sites where vegetation data were collected. For presentation purposes, the data are broken into groups of environmental, management, and stream channel data. The variables, and the associated abbreviations used in the figures, are explained in Table 3.1 (including all of the classes for categorical variables). A description of how the data for these stream variables were collected is in the PIBO-EMP "Sampling Protocol for Integrator Reaches: Channel Parameters" in Kershner et al. (2004).

The management classes that PIBO-EMP assigned to each watershed, based on information from Forest Service and BLM field units, were managed or reference, grazing management (cattle, sheep or not-currently-grazed), percent federal ownership, road density, riparian roads, and road crossings. Reference watersheds were defined using the following criteria for the watershed upstream of the sampling site: 1) road density < 0.5 km of road / km^2; 2) road density in riparian area < 0.25 km of road / km^2; 3) no grazing within in the last 30 years; and 4) no dredge mining and minimal hardrock mining associated with riparian areas.

Table 3.1. List of variables included in analyses with vegetation data.

Variables (terms used in text)	Abbreviations used in figures	Brief explanation
% federal	owner	Percent of subwatershed with federal ownership
Bank angle	BkAngl	High bank angle means flat banks
Bank stability	Stab2Pct	Percent of streambank that is stable
Bankfull width	bfxs	Bankfull width at cross-sections
Conductivity	cond	Stream water conductivity
Coniferous	conif	Percent of the reach covered by coniferous meta-community types
Cross-section wetland rating	ripwet	Measure of the abundance of hydric species in the riparian area.
D50	D50.Rif	Median particle size in riffle streambed
D84	D84.Rif	Riffle streambed particle size for which 84% of particles are smaller
Ecoregion	Ecoreg	11=Blue Mountains/Oregon, 15=Northern Rockies, 16=Idaho Batholith, 17=Middle Rockies, 41=Canadian Rockies, 80=Northern Basin and Range
Effective ground cover	pergrcov	Percent of riparian area covered by rock, litter, or vegetation
Elevation	elev	
Entrenchment ratio	entrench	Valley width at 2x bankfull depth / bankfull width
Forest		Categories: 22 National Forest units
Geology	geol	Categories: granitic, metamorphic (geol1), sedimentary (geol2), or volcanic (geol3)
Gradient	gradient	Stream gradient
Grazing management	grazemgmt	Categories: cattle grazed, not-currently-grazed , sheep grazed, or reference
Greenline wetland rating	glwet	Measure of the abundance of hydric species adjacent to the stream.
Large woody debris	LWDCat1	Pieces of large woody debris in the channel
Management	mgmt	Categories: managed or reference
Not-currently-grazed	Graze.no	Site in a grazing allotment that is not-currently-grazed
Pool percent	poolpct	Percent of stream that is pools
Pool depth	respodep	Residual pool depth
Pool tail fine sediment	PTFinePct	Percent of pool tails that is fine sediment
Precipitation	precip	
Region	Region	Forest Service region or BLM unit: Region1= Montana & northern Idaho; Region4=southern

		Idaho, Utah, Nevada, & western Wyoming; Region6=Washington & Oregon; NW BLM=Washington and Oregon BLM land; Idaho BLM= Idaho BLM land
Riparian roads	riprddens	Riparian road density within the subwatershed
Riparian width	ripwidth	Width of riparian area
Road crossings	rdcross	Stream road crossings within the subwatershed
Road density	roaddens	Road density within the subwatershed
Sinuosity	sinuos	Sinuosity of stream
State	state	Idaho, Montana, Nevada, Oregon, or Washington
Stream density	strmdens	Stream density within the subwatershed
Undercut depth	UnCutDep	Depth of undercut, i.e. bank overhang
Undercut percent	UnCutPct	Percent of reach with undercut banks
Vegetative cover	perlivec	Percent of riparian area covered by live vegetation
Watershed area	wsareahs	
Width to depth ratio	wd	

Data management and summarization

After the data on the cover of community types at a site were collected, the relative abundance of community types was calculated. This was done separately for the greenline and cross-sections and the result for each site was a list of about 10 community types and the percentage of the site that they covered.

The community types were classified to meta-community types, which were developed by cluster analysis to group community types, based on their average cover values. The cluster analysis resulted in the grouping of 725 community types from the 8 riparian vegetation classifications into 167 meta-community types.

In the data collection process along the stream, if there were more than two steps where the vegetation did not match any of the community types listed in the geographically appropriate classifications, technicians collected species cover data for a

"new community" or patch of vegetation (this was done in 2001 and 2002, but not 2000). These new communities were grouped with a meta-community type based on the similarity values or the dominant species. In a few cases there were insufficient data to group the new community with any meta-community type so it was deleted from the data set. For sites where data were deleted, the data were relativized again, so that the cover summed to 100%, just as it did for all other sites. The new communities were a very small portion of the area represented in the greenline (4%) and cross-section (2%) data sets.

Data Analysis

Ordination and within-group similarity

The greenline and cross-section vegetation data were used to evaluate the similarity of the 325 sites to each other, and the relationships among the vegetation, environment, management, and stream variables. A Bray-Curtis dissimilarity index was used to calculate the dissimilarity in vegetation among sites. This was used to generate non-metric multidimensional scaling (NMDS) ordinations with PC-ORD for Windows (McCune and Mefford 1999).

The dissimilarity index was also used to evaluate within-group similarities using multi-response permutation procedures (MRPP). MRPP was used to determine whether the vegetation at sites in managed watersheds was different than at sites in reference watersheds – where there was minimal or no current management activity. A similar comparison was done by Ponzetti and McCune (2001), who used MRPP to evaluate the relationships of vegetation with environmental variables and livestock activity.

A measure of the similarity of site vegetation within groups was represented by the agreement (A) statistic, which is the "chance-corrected within-group agreement" (McCune and Mefford 1999). The A statistic would be 1 if all items in the group were identical, 0 if the items in the group had the same heterogeneity that would be expected by chance, or negative if there were more heterogeneity than expected by chance (McCune and Grace 2002).

Generalized additive models

Generalized additive models (GAMs) were performed using S-Plus statistical program (Insightful Corp 2001) to evaluate the strength of the relationship between the vegetation ordinations and the environmental, management, and stream variables. GAMs are a useful tool to explore the influences of variables, which need not be normally distributed, on an ordination (Yee and Mitchell 1991). The GAMs were calculated with each continuous variable as the response, and the vegetation ordination (the axes of the ordination) as the predictors. A spline smoother was used to fit a smooth relationship between the transformed response and the predictors. The D^2 goodness of fit measure (Chambers and Hastie 1992, Yee and Mitchell 1991) was used to determine the amount of deviance explained by the model, which was calculated as:

D^2 = (1 – [residual deviance/null deviance])

The D^2 is similar to the R^2 goodness-of-fit statistic used in linear regression.

Generalized linear models

Generalized linear models (GLMs), calculated with the S-Plus software program (Insightful Corp 2001), were used to separate the influence of correlated variables

relative to the vegetation ordinations. GLMs are useful because they can be used to do logistic, Poisson, and other types of non-Gaussian regression (Krause and Olson 2000). GLMs are useful for data that violate the basic assumptions of linear regression, which is generally the case with vegetation abundance data.

GLMs were also used to calculate an "effect size" for some dominant variables relative to an axis of the ordination. The cattle-grazing, precipitation, and geology variables were all correlated with axis 3 of the ordination. Therefore a GLM was generated to evaluate the relative importance of the cattle-grazing and precipitation variables on the distribution of sites within the ordination. The cattle-grazing variable was the dependent variable and axis 3 of the ordination, precipitation and geology were the independent variables (see Figs 2.5 and 2.6). From that GLM, the regression coefficients for axis 3, precipitation, and geology were multiplied by the first and third quantiles of that variable and then the difference between those two values was calculated. The result was the effect size for each variable, which allows an evaluation of the relative importance of axis 3, precipitation, and geology in terms of explaining the distribution of cattle-grazed watersheds. This provides an indication of the importance of cattle-grazing on vegetation, after accounting for precipitation and geology.

RESULTS

Vegetation cover data were collected at 325 riparian/stream sites in the upper Columbia River basin. The greenline data included 436 community types (as well as 54 unclassified communities, and an "upland" category). At each site there was an average of 10 community types recorded along the greenline, with a range from 1 to 28. The 436 community types were grouped into 142 meta-community types, based on the similarity of species recorded in the different vegetation classifications, to facilitate comparison among sites. After grouping the community types into meta-community types, there was still an average of 10 meta-community types per site for the greenline, while the range was from 2 to 22.

The cross-section data set included 382 community types (as well as 76 unclassified patches of vegetation, and an "upland" category). The average number of community types along the cross-sections was 7 per site, with a range from 1 to 19. The 382 community types were grouped into 140 meta-community types. The grouping of community types into meta-community types reduced the average number of meta-community types per site to 6 for the cross-sections, with a range from 1 to 16.

The grouping of community types into meta-community types resulted in a decrease of about 1 vegetation type per site. While that is a small number, it represented 9% of the community types recorded along the greenline and 16% of the community types recorded along the riparian cross-sections at a site. Therefore grouping the community types resulted in some loss in the distinctions made between community types at the sites.

Evaluations with dissimilarity index

The meta-community type data were used to perform ordinations of the greenline and cross-section data. For both data sets, the NMDS ordinations had six dimensions that were statistically better than 95% of the runs of randomized data, but only three dimensions that each lowered the minimum final stress by 5 (see Table 2.5). Therefore thee dimensions were used for the final run with real data. The final instability values for the greenline and cross-section ordinations were 0.003 and 0.0005 (respectively), which is near or below what is recommended by McCune and Grace (2002). The final stress values were 22.8 and 23.9, which is considered "poor" according to Kruskal's rules of thumb listed in McCune and Grace (2002).

With meta-community types used in the place of community types the artificial separation of sites by vegetation classification was eliminated, but there were still some relationships between vegetation and geography. For the greenline and cross-section data, the mean within-group distance, using MRPP, was statistically significant ($P < 0.01$) for all the geographical data (state, region, and forest), indicating that there is significant spatial autocorrelation coded in the geographic variables (Table 3.2). The geographical patterns are evident in Figs. 2.5 and 2.6. Of the categorical data for geography, National Forest had the highest "chance-corrected within-group agreement" or A statistic (McCune and Grace 2002) for the greenline and cross-section data (0.095 and 0.084 respectively), followed by region (0.056 and 0.050), and state (0.041 and 0.045).

Table 3.2. The within-group vegetation homogeneity for groups of sites according to categorical variables, calculated with multi-response permutation procedures (MRPP).

		Greenline		Cross-sections	
Variable	Groups	Agreement statistic (A)	P	Agreement statistic (A)	P
Geology	4	0.032	0.00	0.029	0.00
Grazing	4	0.020	0.00	0.018	0.00
Management	2	0.006	0.00	0.005	0.00
National Forest	22	0.095	0.00	0.084	0.00
Region (Forest Service)	5	0.056	0.00	0.050	0.00
State	5	0.045	0.00	0.041	0.00

Vegetation variables

The meta-community types with the strongest explanatory relationships relative to both the greenline and cross-section ordinations were water sedge (*Carex aquatilis*) (D^2 = 0.501 and 0.303, respectively for greenline and cross-sections) (Fig. 3.1), western red cedar (*Thuja plicata*)/fern (D^2 = 0.435 and 0.421) (Fig. 3.2), thinleaf alder (*Alnus incana*)/mesic graminoid (D^2 = 0.402 and 0.392) (Fig. 3.3), and Kentucky bluegrass (*Poa pratensis*) (D^2 = 0.393 and 0.346) (Fig. 3.4).

Environmental variables

The environmental variables with the strongest explanatory relationship relative to the greenline vegetation ordination were: precipitation (D^2 = 0.453), elevation (D^2 = 0.385), NW BLM (Oregon and Washington BLM land) (D^2 = 0.385), Idaho BLM (D^2 = 0.344), Region 1 (the Forest Service region that includes northern Idaho and Montana) (D^2 = 0.329), and volcanic geology (D^2 = 0.252) (see Fig. 2.5 and Table 3.3).

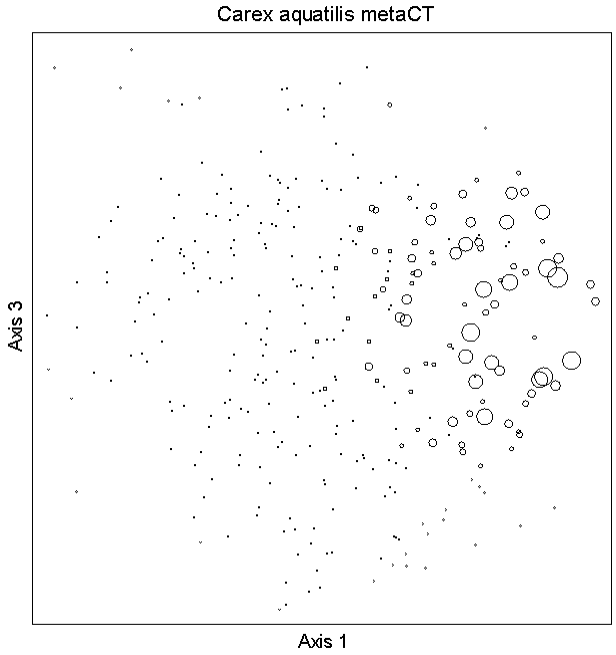

Figure 3.1. NMDS ordination of greenline vegetation data with the size of the symbols corresponding to the abundance of the water sedge meta-community type.

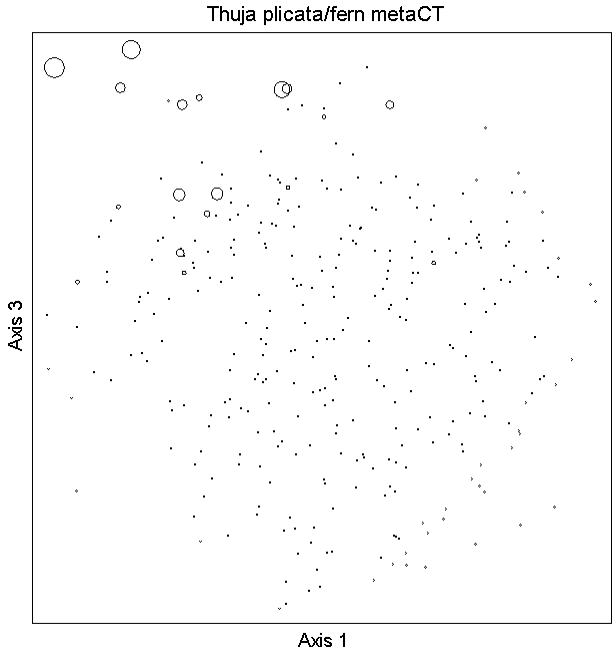

Figure 3. 2. NMDS ordination of greenline vegetation data with the size of the symbols corresponding to the abundance of the western red cedar /fern meta-community type.

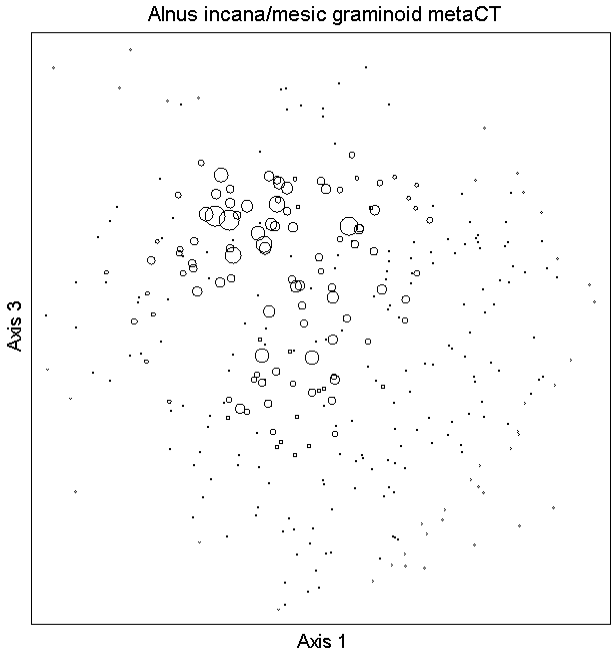

Figure 3. 3. NMDS ordination of greenline vegetation data with the size of the symbols corresponding to the abundance of the thinleaf alder /mesic graminoid meta-community type.

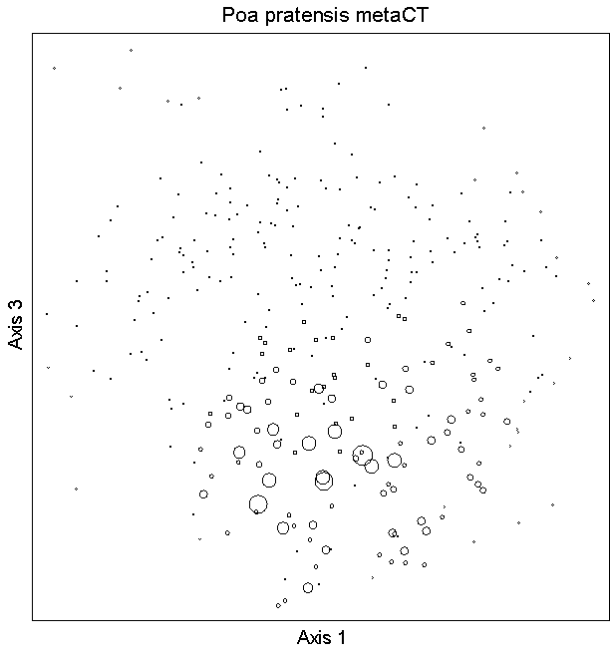

Figure 3. 4. NMDS ordination of greenline vegetation data with the size of the symbols corresponding to the abundance of the Kentucky bluegrass meta-community type.

Table 3.3. The relationship of environmental and stream variables to the greenline vegetation ordination based on the D^2 goodness-of-fit measure (variables listed in descending order of D^2).

Variable	D^2
Precipitation	0.453
Elevation	0.385
NW BLM [a]	0.385
Idaho BLM [a]	0.344
Region1 [a]	0.329
Bank angle	0.282
Volcanic [a]	0.252
Gradient	0.252
Pool percent	0.241
Undercut depth	0.236
Large woody debris	0.215
Conductivity	0.193
Region6 [a]	0.186
Region4 [a]	0.185
granitic [a]	0.184
D50	0.177
Metamorphic [a]	0.169
D84	0.163
Bankfull width	0.163
Pool depth	0.158
Bank stability	0.150
Riparian width	0.142
Sedimentary [a]	0.126
Sinuosity	0.119
Pool tail fine sediment	0.105
Riffle fine sediment	0.102
Stream density	0.080
Entrenchment ratio	0.069
Watershed area	0.068
Width to depth ratio	0.052

[a] binomial variable

For the cross-section ordinations, the environmental variables with the strongest explanatory power were: elevation ($D^2 = 0.455$), Region 1 ($D^2 = 0.388$), precipitation ($D^2 = 0.384$), NW BLM ($D^2 = 0.314$), volcanic geology ($D^2 = 0.295$), Region 4 (the Forest Service region that includes the southern half of Idaho, Utah, Nevada, and western Wyoming) ($D^2 = 0.284$), Idaho BLM ($D^2 = 0.280$), Region 6 (the Forest Service region that includes Oregon and Washington) ($D^2 = 0.245$), and granitic geology ($D^2 = 0.216$) (see Fig. 2.6 and Table 3.4).

The categorical geology variable was also used to perform MRPP to test for similarity of vegetation within geologic categories. The mean within-group distance was statistically significant ($P < 0.01$) for both the greenline and cross-section data. This indicates that sites with the same geology had greater similarity of vegetation than would be expected by chance, but only slightly greater as indicated by the small A statistics of 0.032 for the greenline and 0.029 for the cross-sections (Table 3.2).

Management variables

Ordinations of sites were evaluated based on whether sites were in watersheds with management activities (roads, livestock grazing, or mining) vs. watersheds with minimal management activities. For the management category, 250 sites were managed, and 75 were reference (Table 3.5). The management category was further divided according to whether the watershed had grazing by: cattle (n=165), sheep (n=27), or was not-currently-grazed (n=58).

Table 3.4. The relationship of environmental and stream variables to the cross-section vegetation ordination based on the D^2 goodness-of-fit measure (variables listed in descending order of D^2).

Variable	D^2
Elevation	0.455
Region1 [a]	0.388
Precipitation	0.384
NW BLM [a]	0.314
volcanic [a]	0.295
Region4 [a]	0.284
Idaho BLM [a]	0.280
Bank angle	0.271
Pool percent	0.263
Region6 [a]	0.245
Granitic [a]	0.216
Undercut depth	0.215
Conductivity	0.194
Gradient	0.194
Metamorphic [a]	0.192
Pool depth	0.192
Bankfull width	0.184
Large woody debris	0.167
D84	0.146
D50	0.143
Sinuosity	0.130
Riparian width	0.122
Stream density	0.107
Watershed area	0.096
Bank stability	0.088
Pool tail fine sediment	0.087
Riffle fine sediment	0.075
Sedimentary [a]	0.067
Width to depth ratio	0.058
Entrenchment ratio	0.033

[a] binomial variable

Table 3.5. The geographical distribution of reference sites for the data set of 325 sites.

Management	Idaho	Montana	Nevada	Oregon	Washington	Total
Managed	123	37	3	65	22	250
Reference	36	22	0	14	3	75

The management variables with the strongest explanatory relationship to both the greenline and cross-section vegetation were: not-currently-grazed ($D^2 = 0.200$ and $D^2 = 0.210$ respectively) and cattle-grazed ($D^2 = 0.198$ and $D^2 = 0.236$ respectively) (Tables 3.6 and 3.7; see Figs. 2.5 and 2.6). The sites in watersheds categorized as not-currently-grazed were most common in the upper part of the ordinations, as indicated by the strong positive correlation ($r = 0.331$) with axis 3 (Fig. 3.5). The sites in watersheds that were cattle-grazed were most common in the lower part of the ordinations, as indicated by the strong negative correlation with axis 3 ($r = -0.464$) (Fig. 3.5).

The D^2 for the reference and managed categories was only 0.124 for the greenline ordination and 0.194 for the cross-section ordination, indicating a weak relationship between vegetation and the management category. The sites in reference watersheds tended to be in the upper part of the ordinations (Figs. 3.5 and 3.6), but sites in both reference and managed watersheds were scattered throughout the ordination diagrams.

The MRPP test indicated that there was some relationship between management and vegetation in both the greenline and cross-section ordinations. The groups of managed and reference sites had greater within-group similarity than would be expected by chance ($P < 0.01$), but the A statistics (0.006 and 0.005 for greenline and cross-sections respectively) were very small indicating only slight homogeneity (Table 3.2). The mean within-group similarity for reference sites was only 12% and for managed sites only 10%.

Table 3.6. The relationship of management variables to the greenline vegetation ordination based on the D^2 goodness-of-fit measure (variables listed in descending order of D^2).

Variable	D^2
Not-currently-grazed [a]	0.200
Cattle-grazed [a]	0.198
Management [a]	0.124
Road crossings	0.086
Sheep-grazed [a]	0.078
Riparian roads	0.073
Road density	0.073
% federal	0.043

[a] binomial variable

Table 3.7. The relationship of management variables to the cross-section vegetation ordination based on the D^2 goodness-of-fit measure (variables listed in descending order of D^2). The D^2 value was calculated as 1- (residual deviance/null deviance) from a generalized additive model (GAM).

Variable	D^2
Cattle-grazed [a]	0.236
Not-currently-grazed [a]	0.210
Management [a]	0.194
Riparian roads	0.157
Road crossings	0.128
Road density	0.125
Sheep-grazed [a]	0.106
% federal	0.087

[a] binomial variable

For the grazing categories, the MRPP for both the greenline and cross-sections also indicated a small, but significant difference in vegetation among groups ($P < 0.01$, A statistics = 0.02). The A statistics for the grazing categories were slightly higher than for the more general management category.

All three variables related to roads for the greenline ordination had D^2 values < 0.09 and between 0.12 and 0.16 relative to the cross-section ordination.

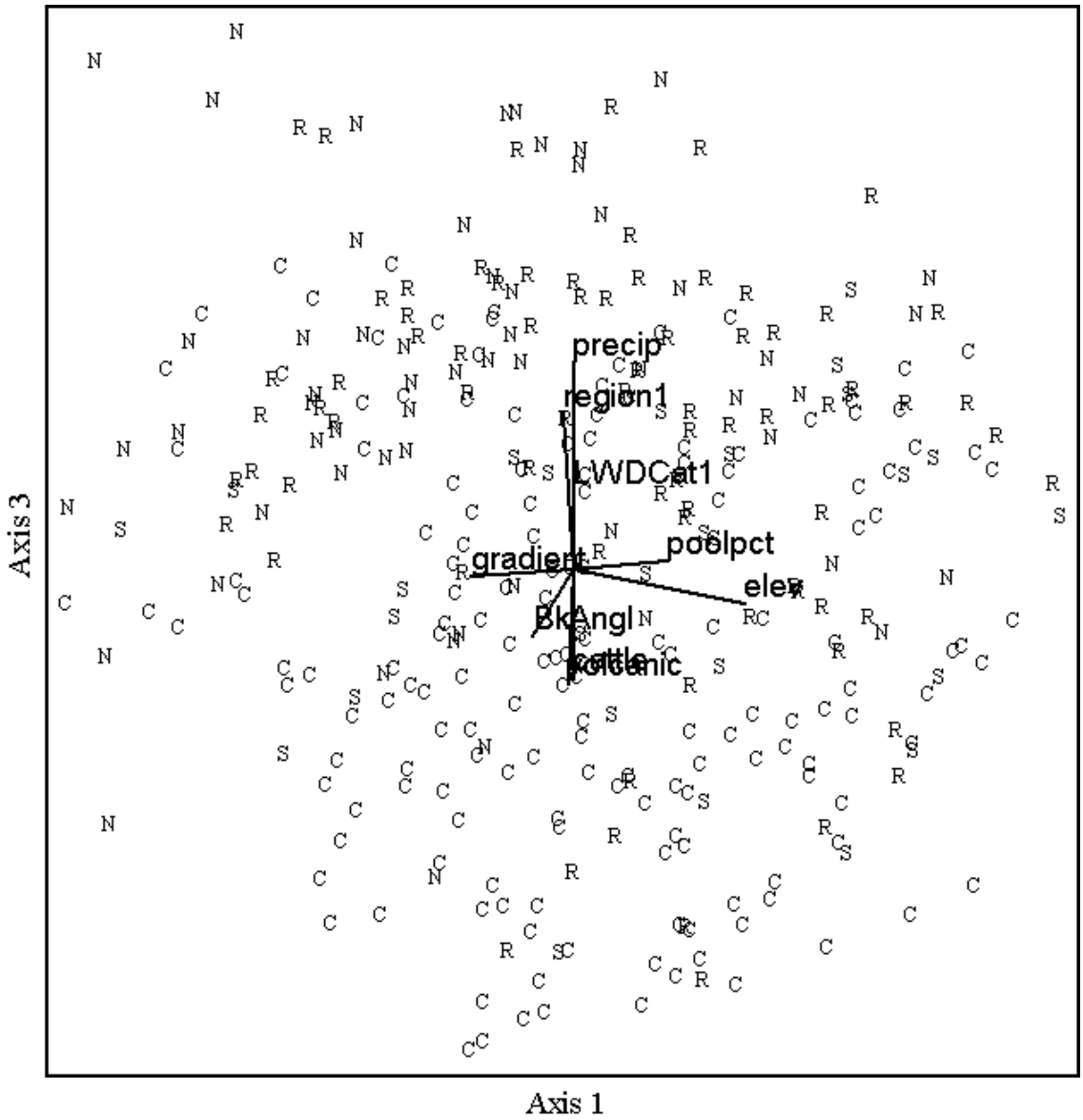

Figure 3. 5. NMDS ordination of greenline vegetation data with the type of grazing management represented by different symbols and colors.

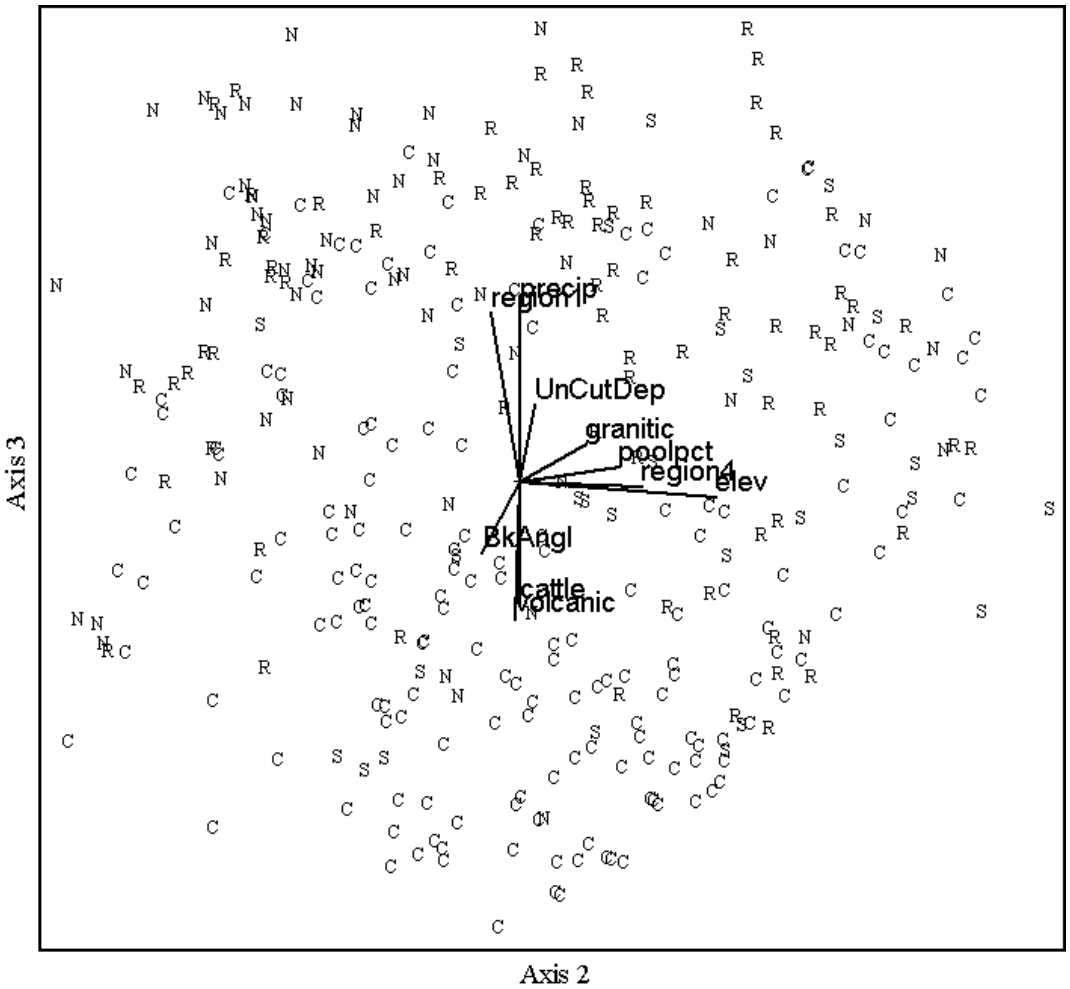

Figure 3. 6. NMDS ordination of cross-section vegetation data with the type of grazing management represented by different symbols and colors.

Cattle grazing and precipitation were negatively correlated in both of the ordinations, therefore it was not clear which of the two variables, or what combination of the two, most influenced the vegetation. In both ordinations precipitation was aligned with the vertical axis, referred to as axis 3, and as a result the cattle-grazing variable was also generally aligned with axis 3, although with the opposite pattern. Therefore it was important to determine the influence of precipitation vs. cattle grazing in the distribution of sites along axis 3. A generalized linear model (GLM) was used to determine the

regression coefficients of precipitation, geology and axis 3 (the highest correlated ordination axis) in relation to the distribution (in the ordination) of sites in cattle-grazed watersheds (a binomial probability). Precipitation had the largest influence, but axis 3 was 38% as strong as precipitation in explaining the distribution of sites from cattle-grazed watersheds in the greenline ordination and 69% as strong in the cross-section ordination. Therefore, after accounting for precipitation axis 3 was statistically significant ($P < 0.05$) in terms of explaining the distribution of cattle sites. This suggests that sites from cattle-grazed watersheds have similar vegetation (are more abundant in the lower part of the ordinations) partly because of lower precipitation but also because of the grazing management.

Vegetation and stream channel data

For the greenline, the stream variables with the strongest explanatory relationship to the ordination were bank angle ($D^2 = 0.282$), gradient ($D^2 = 0.252$), pool percent ($D^2 = 0.241$), undercut depth ($D^2 = 0.236$), large woody debris ($D^2 = 0.215$) (Table 3.3). Most of these variables are visible as vectors in Fig. 2.5.

For the cross-sections, the stream variables with the strongest explanatory relationship to the ordination (Fig. 2.6) were bank angle ($D^2 = 0.271$), pool percent ($D^2 = 0.263$), and undercut depth ($D^2 = 0.215$) (Table 3.4).

Summary of ordinations

In the greenline and cross-section ordinations there are two dominant environmental variables: precipitation and elevation. In both ordinations the joint plot vectors for these variables form 100 degree angles. The lack of positive correlation

between precipitation and elevation is likely due to the fact that the northern part of the study area (Washington, Montana, and northern Idaho) has high precipitation but has many of the lower elevation sites in the study area. A third variable, with a different direction of influence in the ordination was stream gradient, although it was not as important as precipitation and elevation.

The variables precipitation, elevation, and stream gradient radiate out in different directions, like "spokes" on a wheel. Because these variables influence riparian vegetation, and because they appear to have a different relationship with the vegetation of this data set, they were used to characterize these data. In the ordinations the variable for precipitation was aligned with the vertical axis and the variable for elevation was generally aligned with the horizontal axis. The vegetation of these ordinations were broken into four types of sites that correspond to the four directions of the axes. The patterns for the greenline and cross-section NMDS ordinations were very similar so the summaries below apply to both the greenline and cross-section data.

Conifer-alder forests

The upper part of the ordination is generally "conifer-alder forests." They are mostly Montana and northern Idaho sites (Region 1 of the Forest Service). They have relatively high precipitation, are middle to low elevation, and have granitic or metamorphic geology. The vegetation at these sites, especially those upper-most in the ordination, is dominated by conifers, i.e. meta-community types with western red cedar, Engelmann spruce (*Picea engelmannii*), grand fir (*Abies grandis*), and subalpine fir (*Abies lasiocarpa*). Just below the conifer zone in the ordination, the vegetation is dominated by shrubs, i.e. meta-community types with thinleaf alder, Sitka alder (*Alnus*

viridis), redosier dogwood (*Cornus sericea*), and rose spirea (*Spiraea douglasii*) with mesic forb, graminoid, and fern understories. A large proportion of the reference watershed sites, and the sites in not-currently-grazed watersheds, are in this "conifer-alder forests" section of the ordination, but there are also many sites in managed watersheds as well. Relative to the other sites in the ordination, the streams associated with these sites can be characterized by relatively low bank angles (steeper banks), overhanging banks, deep pools, more large woody debris, wide channels, and low conductivity.

In the upper-most part of the ordination the sites are either in reference watersheds or are those that are not-currently-grazed. This could be due in part to the forested habitat that makes these areas less favorable for livestock grazing. Below the upper-most part of the ordination there are sites from managed as well as reference watersheds, suggesting that something allows these sites from managed watersheds to maintain vegetation (at least meta-community types) similar to some of the sites in reference watersheds.

Wet sedge and willow sites

The right part of the greenline ordination is dominated by "wet sedge-willow sites." These sites are mainly in Idaho, at higher elevations, with granitic or volcanic geology. The vegetation is dominated by hydric graminoids (water sedge meta-community type primarily), or meta-community types with willows (*Salix* spp.) and hydric graminoid understories. Some of the sites in reference watersheds are in this part of the ordination, but they are mostly sites from managed watersheds (cattle or sheep

grazed). The streams are characterized by relatively low gradients, a high percentage of pools, small streambed substrate, and high sinuosity.

Dry-grazed sites

The lower part of the ordination could be called the "dry-grazed sites." These are principally sites in Idaho (Region 4 and BLM) and Oregon (Region 6 and BLM), with volcanic geology, low precipitation, and somewhat higher elevation. Most of these sites are in managed watersheds, and specifically with cattle grazing. The dominant vegetation at these sites is the Kentucky bluegrass meta-community type. These sites also include communities that grow in the driest conditions, such as meta-community types with Woods' rose (*Rosa woodsii*), ponderosa pine (*Pinus ponderosa*), big sagebrush (*Artemisia tridentata*), silver sagebrush (*Artemisia cana*), and the general upland category. These streams are characterized by little bank overhang, high bank angles (flattened banks), high conductivity, shallow pool depths, and little if any large woody debris.

The lower right part of the ordination is a sub-type that is transitional between the dry-managed and the wet sedge-willow sites. The vegetation at these sites is dominated by sedge, rush, and willow communities -- Nebraska sedge (*Carex nebrascensis*), Baltic rush (*Juncus balticus*), woolly sedge (*Carex pellita*), Northwest Territory sedge (*Carex utriculata*), and Booth willow (*Salix boothii*)/mesic graminoid) meta-community types that have more obligate wetland species (sensu Reed 1996) than the sites in the lower part of ordination, which are dominated by the Kentucky bluegrass meta-community type, but less obligate wetland species than in the right part of the ordination. These meta-community types represent vegetation that could replace the Kentucky bluegrass meta-

community type, considered a "grazing disclimax" by Hansen et al. (1995), along a successional pathway that moves toward the wet sedge-willow vegetation to the right in the ordination.

Steeper-shrub sites

The left part of the ordination could be called the "steeper-shrub sites" (recognizing that stream gradients were less than 3%). These sites are mainly in Oregon and Washington (Region 6), as well as some Idaho and Montana sites, which are lower elevation and have volcanic geology. The vegetation is primarily shrub-dominated communities (meta-community types with thinleaf alder, redosier dogwood, and currant (*Ribes* spp.) with hydric or mesic graminoid understories, and some cottonwood and upland communities as well. These are predominantly sites in managed watersheds, including all types of grazing management — cattle or sheep grazing and no current grazing — and a few sites from reference watersheds. The streams are characterized by relatively steep gradient, a low percentage of pools, and large substrate. Those characteristics are somewhat similar to the dry-grazed sites, which could be due to the natural disturbance of the steeper gradient or the result of management activities, or some combination of the two.

DISCUSSION

The ordinations and MRPP tests suggest that management activities, and livestock grazing in particular, have had an influence on the vegetation at riparian sites in the upper Columbia River basin.

Vegetation within the groups of managed and reference sites – as well as within the grazing groups (cattle-grazed, sheep-grazed, or not-currently-grazed) – had slightly greater similarity than would be expected by chance. The within-group similarity is probably not large enough to say that the vegetation at grazed sites is converging or becoming more similar. Statistical significance with MRPP can be an artifact of a very large sample size (McCune and Grace 2002).

The grazing categories had higher within-group similarity than the more general managed and reference categories. This suggests that grazing by livestock, sheep, or a lack of current grazing lead to different impacts. On the other hand, it could be that there are different types or levels of grazing in different geographical areas that have different susceptibilities to impact from grazing.

Of the two activities that can be broken out of the management category—roads and livestock (data on mining and logging were not available) —livestock has a larger effect on riparian vegetation. The variables that represent roads are not strongly correlated with vegetation in the ordinations.

In the ordinations precipitation was the strongest variable in relation to the vegetation, with sites in the lower part of the ordinations being associated with low precipitation. However, cattle grazing still had a significant relationship to vegetation in terms of the vertical axis of the ordination. It seems that there is an interaction between

low precipitation and cattle grazing that results in certain vegetation along the stream, such as Kentucky bluegrass communities, which is not the expected streambank vegetation.

For sites in the upper part of the ordination (conifer-alder forests) the abundance of rain might prevent or offset some of the impacts of management on vegetation. Cattle, which are one of the major management influences, spend a disproportionate amount of time in riparian areas, where there is water, forage, cooler temperatures, and relatively flat terrain, compared to the uplands (Bryant 1982, Kauffman and Krueger 1984, Fleischner 1994). In areas of higher precipitation, where more forage is available in the uplands, cattle may spend less time in riparian areas and therefore have fewer impacts on riparian areas. This is consistent with the finding of Milchunas and Lauenroth (1993) that environmental variables, including precipitation, were more important than grazing management in terms of explaining vegetation and ecosystem variables.

The sites in the right part of the ordination (wet sedge and willow sites) are in both reference and managed watersheds and they have late successional riparian communities (as described in the vegetation classifications listed in Table 2.1). This suggests that there is something about these sites that makes them less vulnerable to the impacts of management. It may be that high elevation sites in managed watersheds receive relatively lower amounts of use and disturbance than sites in lower elevation managed watersheds due to difficulty of access because of steep terrain and/or snow.

Sites in the lower portion of the ordination (dry-grazed sites) seem to have been more vulnerable to negative impacts from cattle grazing, perhaps for the opposite reasons that the high precipitation sites were able to withstand management activities (as

discussed above). Riparian areas provide a disproportionately high amount of forage consumed by livestock compared to uplands (Kauffman and Krueger 1984). In a study in semi-arid northeast Oregon, the riparian zone -- though only 2% of the landscape -- provided 81% of livestock forage due to steep slopes in the uplands (Roath and Krueger 1982). Therefore in drier environments, where upland forage is limited, riparian areas likely experience greater impacts from livestock grazing.

Of the few (5) reference sites in the dry-grazed part of the ordinations, two were in the Bob Marshall wilderness, an area that historically experienced stand-replacing fires (Arno et al. 2000). The vegetation and channel conditions at those sites may exhibit the legacy of large fire and flood events that caused the channels to widen and the water table to drop resulting in more upland species as at the dry-grazed sites (personal observation).

In addition, the GLM showed that after accounting for the influence of precipitation, cattle grazing still had a strong and statistically significant correlation to sites in the lower part of the ordination, which were dominated by Kentucky bluegrass communities. Similarly, other studies have documented a decrease in Kentucky bluegrass communities with the exclusion of livestock from riparian areas (Leege et al. 1981, Schultz and Leininger 1990, Green and Kauffman 1995).

The cross-section data showed a stronger relationship than the greenline data between cattle and the lower part of the ordination, which is associated with precipitation, suggesting that vegetation some distance from the stream (cross-sections) changed more due to cattle grazing than the vegetation along the stream (greenline). Disturbance from cattle grazing can lead to bank erosion and a drop in the water table (Gebhardt et al.

1989), which can cause the riparian area to be drier, whereas the greenline is generally close to water because of the proximity of the stream or groundwater.

One of the challenges of evaluating management impacts is the lack of reference sites, i.e. sites with minimal disturbance from human activities, in some areas. In this study, as many reference sites as could be found were sampled, but that still left the proportion of reference sites at only 23% of the total sites sampled (Table 3.5). For the four states (excluding Nevada, which only had three sites total) reference sites ranged from 12-37% of the sites for each state. Therefore reference sites were not as well represented in the ordinations as managed sites. In addition, reference sites are not randomly distributed across the landscape. These factors could inhibit the detection of some relationships between management and vegetation.

Vegetation and streambank relationships

A decrease in the abundance, or certain types, of vegetation can lead to a decrease in bank stability and channel widening. The dense and deep root structure of many obligate wetland species (sensu Reed 1996) makes them effective bank stabilizers (Toledo and Kauffman 2001). Species that grow in drier conditions, such as Kentucky bluegrass, have lower very-fine root-length density, less biomass, and a shallower distribution of roots in the soil profile than riparian communities dominated by species that grow in wetter environments, such as Nebraska sedge and Baltic rush (Dunaway et al. 1994, Kleinfelder et al. 1992, Manning et al. 1989). That explains, at least in part, why the dry-grazed sites had flattened banks and little bank overhang, which are indicative of lower bank stability. A comparable pattern was observed in a meadow

dominated by silver sagebrush and annual grasses, which had much greater streambank erosion and lateral migration than a comparable meadow dominated by sedges and rushes (Micheli and Kirchner 2002a, Micheli and Kirchner 2002b).

Livestock activity can lead to channel widening due to destabilization of streambanks by shearing and trampling, and by reducing vegetation which lowers the resistance to erosive stream flows (Trimble and Mendel 1995) and overland flows (Flenniken et al. 2001). Kentucky bluegrass is considered an increaser species with livestock grazing (Hansen et al. 1995). At sites in managed watersheds, livestock grazing probably led to an increase in Kentucky bluegrass, and other facultative or upland species, and an associated decrease in bank stability. Myers and Swanson (1995) documented greater bank instability and wider channels (larger width to depth ratio) at streams with livestock grazing compared to comparable streams rested from livestock grazing.

Problems with community types as cover categories

The use of community types as cover categories was problematic because it limited the options for describing the vegetation at a site. Community types did not represent all of the vegetation that was encountered, especially for transition zones and disturbed areas. In fact, some classifications (Crowe and Clausnitzer 1997, Hansen et al. 1995, Kovalchik 2001) were developed to describe late successional vegetation and not necessarily sites disturbed by human activities. In some cases the closest fit between the vegetation described by the classification and what technicians observed was not a good match. An alternative was developed for patches of vegetation that were poorly

described by a classification, which was to record species data and call it a "new" community, but having a mix of community type data and species data was problematic for analysis.

Community type data are particularly difficult to analyze for a large area, as in this study, because multiple classifications have to be used. Each classification had defined somewhat different communities, therefore it was necessary to group community types across classifications. This was done by converting community types to meta-community types. In other words, community types from different classifications were grouped together into more general meta-community types, but this resulted in the loss of a fair amount of distinction in communities that had been made on the ground.

The use of community types as cover categories also prevents the recognition of similarities in community types when sites are evaluated with a similarity index. This is because the community type is a name that represents an assemblage of species, but it does not list those species. Therefore if two sites are compared that have similar community types, such as water sedge community type and Booth willow/ water sedge community type, a dissimilarity index considers those community types as completely different, even though they have species in common. Fuzzy set ordination could be used to overcome this, but the only data available for all of the classifications are the average cover values for each community type, which do not necessarily reflect the vegetation at the site.

If species data were collected, then the similarity index, and hence the ordination, would better recognize the similarity among sites, because it would recognize all species those sites have in common. That might facilitate a better understanding of the

relationships between vegetation, management, environment and stream variables. A modification of Winward's methods is now being employed by PIBO-EMP that incorporates species data collection, in an effort to obtain more accurate and informative data.

The community type data did indicate strong relationships between environment and certain vegetation types, particularly for precipitation and elevation. This is consistent with the published community type classifications (see Table 2.1) which document some different communities in different geographical areas.

Relationships between vegetation and stream channel variables were detected that were consistent with other research. The relationships between management and vegetation were generally not strong, although there was a relationship between sites in cattle-grazed watersheds and Kentucky bluegrass communities. These were sites with relatively low precipitation, which would effect vegetation. After accounting for precipitation there was still a strong relationship between the distribution of sites in the ordination and whether a site was in a watershed with cattle-grazing. This suggests that the combination of livestock grazing and low precipitation has resulted in abundant cover of the Kentucky bluegrass community, an indicator of poor riparian condition. In contrast, wet sedge and willow communities had low stream gradients and more pools, which is consistent with their status as indicators of good riparian condition.

Improving our understanding of the relationships between vegetation, environmental variables, stream channel variables, and management will help the PIBO-EMP in its effort to answer the question of how management activities impact the functions of streams and riparian areas within the upper Columbia River basin. The

patterns identified here can also help land managers determine which locations are more susceptible to impacts from management activities, such as livestock grazing. This will be useful in efforts to maintain the important ecological and social functions that riparian and stream ecosystems provide.

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CHAPTER 4

A WETLAND RATING SYSTEM FOR EVALUATING RIPARIAN VEGETATION[3]

ABSTRACT: A system of rating riparian sites was developed to assess ecological integrity through the abundance of hydric species along streams. In order to calculate wetland ratings for sites, wetland ratings were first calculated for community types, from riparian vegetation classifications, used to describe the sites. The wetland rating was compared for 14 livestock exclosures and adjacent grazed areas. It was hypothesized that exclosures would have higher wetland ratings than adjacent grazed sites where disturbance may have altered the hydrologic processes that affect vegetation. The wetland ratings for exclosures were more than 12 percent higher than grazed-site wetland ratings; differences were statistically significant based on paired t-tests. Channel widths and width-to-depth ratios were significantly ($p < 0.10$) smaller in exclosures, and streambank stability was significantly higher in exclosures. The increase in hydric species with livestock exclusion was probably associated with the increase in bank stability that confined the stream in a narrower/deeper channel and thereby raised the water level. This likely corresponded to an increase in moisture availability in the riparian area, and hence a higher wetland rating. The wetland rating is recommended as a tool for monitoring the same sites over time, or sites with comparable environmental conditions.

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INTRODUCTION

Riparian areas in the western US have become increasingly important for government agencies such as the US Forest Service and Bureau of Land Management (National Research Council, 2002). Streams and riparian areas perform many functions that have both ecological and social importance. The vegetation within riparian areas helps to: stabilize streambanks, dissipate energy of floods, support perennial flows, trap sediment, and moderate stream temperature (Gregory *et al*., 1991; Elmore and Kauffman, 1994; Gurnell, 1997; Naiman and Decamps, 1997; Tabacchi *et al.*, 1998; Tabacchi *et al.,* 2000). Many of these functions are important for maintaining wildlife habitat, especially for endangered salmonids (see reviews by Kauffman and Krueger, 1984; Platts*,* 1991; Fitch and Adams, 1998; Naiman *et al.,* 2000).

There is a steep moisture gradient from the uplands to the stream edge, particularly in semi-arid regions of western North America (Chapin *et al.,* 2002; Dwire *et al.,* 2004). Compared to adjacent uplands, riparian areas have high moisture availability due to the high water table and the interaction of the channel and the floodplain through infiltration, overbank flow, and bank storage (Gebhardt *et al.*, 1989; Hughes, 1997). Therefore, riparian areas support unique vegetation, in particular an abundance of obligate and facultative wetland species that are rarely if ever found in the surrounding uplands. The wetland rating employed in this research assesses the abundance of wetland species, with the expectation that functioning riparian areas associated with low-gradient streams (< 3 percent in this study) will have an abundance of wetland species.

A low gradient perennial stream without wetland species indicates poor stream/floodplain connectivity. A severe lack of connectivity is referred to as

"entrapment" by Sarr (2002), which is the condition where the stream is trapped within the channel and is not able to reach the former floodplain. This has negative consequences because "floods that would normally escape the baseflow channel and dissipate sediment and energy over the floodplain are trapped in the gully and highly scouring flows occur" (Sarr 2002). That can have negative consequences for the stream (increased bank instability, channel widening, bank flattening, and shallower water depths) and riparian ecosystems (a drier environment and conversion to more upland species), which has negative implications for fish and wildlife habitat.

Monitoring of riparian areas is essential to assess the impacts of human-caused activities -- such as livestock grazing -- on ecological functioning and ecosystem services. Concern for fish populations within the Columbia River basin were expressed in the Pacific Anadromous Fish Strategy (PACFISH) and Inland Fish Strategy (INFISH) biological opinion (PACFISH, U.S. Forest Service and U.S. Bureau of Land Management, 1994; U.S. Department of Interior, Fish and Wildlife Service, 1998). The PACFISH/INFISH biological opinion (PIBO) led to the establishment of an effectiveness monitoring program (EMP) charged with determining whether management practices in the upper Columbia River basin were maintaining or restoring the structure and function of riparian and aquatic habitats (Kershner *et al.,* 2004a).

PIBO-EMP used the methods of Winward (2000) to collect data on riparian vegetation. These methods rely on the use of community types to describe the cover of vegetation of riparian sites, and community type ratings to evaluate sites. Winward (2000) assigned a rating to each community type indicating bank stability and successional status of streambank vegetation. These were developed to facilitate the

ratings of the riparian area at a stream reach (hereafter "site") based on the community types present, and to monitor change over time. However, these ratings were developed only for Region 4 of the Forest Service, which covers just the southern Idaho portion of the PIBO-EMP study area – which includes eastern Washington, eastern Oregon, Idaho, northeastern Nevada, and northwestern Montana. In addition, these ratings applied only to vegetation along the streambank and not to data collected away from the stream, along riparian cross-sections. This led to two problems: 1) there was no summary technique to evaluate all the vegetation data in a uniform way; and 2) for part of the data was no summary technique at all.

A Wetland Rating

In response to the need for evaluation tools, a wetland rating for riparian community types and sites was developed. This wetland rating is based on the wetland indicator status (WIS) of plant species (Reed, 1996), which has been used by others to classify riparian vegetation, and to evaluate changes over time or differences among sites (Stromberg, 2001; Toledo and Kauffman, 2001; Chapin *et al.,* 2002).

The wetland rating summarizes the relative abundance of species, or community types, in relation to the continuum of the wetland indicator status, from obligate wetland species at one end, to upland species at the other end of the continuum (Table 4.1). This wetland rating is a "synthetic vegetation index" because its uses values assigned to each wetland indicator status to generate quantitative values for a site. These values facilitate comparisons of sites, or the same site over time.

Table 4.1. The Wetland Indicator Status of Reed (1996) and the Synthetic Wetland Indicator Value Assigned to Each Status.

Wetland Indicator Status	Estimated probability a species will occur in a wetland	Wetland indicator value
Obligate	Almost always (99%)	100
Obligate -		92
Facultative Wet +		83
Facultative Wet	Usually (67% - 99%)	75
Facultative Wet -		67
Facultative +		58
Facultative	Sometimes (34% - 66%)	50
Facultative -		42
Facultative Upland+		33
Facultative Upland	Not usually (1% - 33%)	25
Facultative Upland -		17
Upland +		8
Upland	Almost never (1%)	1

Livestock and Riparian Ecology

Livestock can have a variety of impacts on riparian areas (see reviews by Fleischner, 1994; Trimble and Mendel, 1995; Belsky *et al.,* 1999). Livestock grazing in riparian areas can lead to bank instability (Myers and Swanson, 1995) because of the loss of above-ground and below-ground biomass, and direct damage to banks by hooves. Livestock also compact the soil which decreases infiltration (Bohn and Buckhouse, 1985; Wheeler *et al.,* 2002). Livestock grazing can also lead to changes in species composition (Leege *et al.,* 1981; Dobkin *et al.,* 1998), and particularly declines in deciduous woody vegetation (Schulz and Leininger, 1990; Green and Kauffman 1995; Case and Kauffman, 1997; Brookshire *et al.,* 2002). These impacts from livestock grazing can lead to bank instability, channel widening or incision, and an overall decrease in moisture availability in the riparian area, and hence a lower wetland rating (Beschta and Platts, 1986; National Research Council, 2002).

Vegetation plays a major role in stabilizing streambanks and affecting channel form because of the ability of riparian species to stabilize streambanks, dissipate stream energy, and trap sediment (Hughes, 1997; Naiman and Decamps, 1997). There is also a positive feedback between riparian species and hydrologic processes which support those species (Manci, 1989). Therefore the abundance of obligate wetland species, quantified with the wetland rating, can serve as an indicator of riparian condition, in terms of hydrologic connectivity between the stream and floodplain (sensu Toledo and Kauffman, 2001), and to some degree bank stability.

In this study the wetland rating is used to assess the abundance of wetland species (using a scale from upland to obligate wetland species) in livestock exclosures and

adjacent grazed areas. The hypothesis was that sites in exclosures would have higher wetland ratings than adjacent areas outside the exclosures that are still grazed sites because of disturbance from livestock grazing. "Disruption of normal geomorphic or hydrologic function, or the vegetation on which it depends, usually results in impairment to overall riparian resource values" (Manci, 1989). Differences in the wetland rating would suggest that livestock activities have altered the vegetation or the local environmental conditions (primarily moisture availability) that determine which species occupy a site.

METHODS

Study Sites

The PIBO-EMP study area includes stream/riparian areas on federal land (Forest Service and BLM) within the upper Columbia River basin (Kershner *et al.,* 2004a). In this paper a subset of the PIBO-EMP sites with livestock exclosures are evaluated. These are areas where federal agencies had previously set up fences around the stream, and part, or all, of the riparian area. The exclosures were designed to prevent livestock from accessing that part of the stream and riparian area, although the efficacy of the exclosure in that regard is not known. Ten of the exclosure sites examined in this study were on Forest Service and BLM land in Idaho (Boise, Custer, Gem, Idaho, and Valley counties), and four were on Forest Service land in eastern Oregon (Grant, Harney, Union, and Wallowa counties). The average age of the exclosures was approximately 11 years, based on estimates from agency personnel.

The exclosure sites were all on perennial stream reaches in unconstrained valley bottoms, with stream gradients from 0.1 to 2.2 percent, and bankfull widths between 1 and 6 m.

Data Collection

The riparian vegetation component of the PIBO-EMP used the monitoring methods of Winward (2000) to evaluate the condition of riparian vegetation along the immediate streambank and in the larger riparian area. The methods of Winward (2000) require community types to describe the vegetation at a riparian site, along a 110-m length of stream. At each stream site the geographically appropriate vegetation classification was used to determine the community types present.

The data collector recorded the community types as he/she walked along the streambank vegetation, referred to as the greenline. The greenline is the first line of perennial vegetation adjacent to the stream. Community types were also recorded along five vegetation cross-sections spaced evenly along the 110 m of stream length. The cross-sections extended in both directions from the greenline to the edge of the riparian area or 27.5 m, whichever was encountered first. The data collectors recorded the extent of each community type, using steps to quantify the area. The total number of steps for each community type along the greenline and the cross-sections were summed separately for the site, and then converted to percentages.

Each year there were five to eight observers who collected the vegetation data. They received approximately 8 days of training at the start of the field season. The training included instruction in identification of the dominant riparian species, how to use

community type classifications, and the sampling methods. For 11 of the 14 sites, the same observer collected the data for both the exclosure and the adjacent grazed site.

Stream channel data were also collected at these sites by pairs of observers who had also received about 8 days of training. Stream variables that affect the hydrology of the riparian area are presented here, including channel dimensions and bank characteristics. The sites were sampled during the 1999, 2000, or 2001 field seasons (June to September).

Calculating the Wetland Rating

The wetland indicator status (WIS) of the US Fish and Wildlife "1996 National List of Vascular Plant Species that Occur in Wetlands" (Reed, 1996) is commonly used to evaluate and classify wetland vegetation. The WIS ranges from obligate wetland species, which are found in wetlands 99 percent of the time, to upland species, which are found in wetlands only 1 percent of the time, with various distinctions in between (Table 4.1). The WIS is based on the observations of scientists and they "represent the probability of a species occurring in wetlands versus non-wetlands in the region" (Reed, 1996).

Wetland vegetation is often evaluated in relation to the WIS to delineate wetlands, using a weighted average to generate a score from 1 to 5 (for examples see Michener, 1983; Wakeley and Lichvar, 1997; Brown, 1999). Recently riparian vegetation has also been evaluated with the WIS -- by calculating the percentage of species in WIS classes (Toledo and Kauffman, 2001; Chapin *et al.,* 2002) or with a weighted average score from 1 to 5 (Stromberg, 2001).

For this study weighted average scores were also calculated with the 13 WIS classes currently available. Values were assigned to each class (Table 4.1) by evenly spacing the values between 1 (for the upland class) and 100 (obligate wetland class), similar to Michener (1983). The values closely correspond to the class midpoint of the probability of occurrence described by Reed (1996), at least for the 5 classes for which frequency estimations were provided, as recommended by Michener (1983) and Brown (1999).

In order to calculate a wetland rating for a site, a wetland rating was needed for the community types recorded at the site. To calculate a wetland rating for a community type, data were compiled about the species composition of the community types in the eight classifications used by PIBO-EMP.

The wetland rating for a community type was calculated based on the average cover and constancy of each species in the community type, as reported in the classifications. The average cover was based on the plots in which a species occurred (for the plots used to define that community type). The constancy was the percentage of plots with that species present (for the plots used to define that community type). An importance value was generated for each species within a community type by multiplying the average cover of the species by the constancy of that species. The relative importance was calculated by dividing the importance value by the sum of importance values for that community type. A community type wetland rating (CTWR) was calculated as

$$\boldsymbol{CTWR} = \sum_{i=1}^{i=1} (RImp_i * WIV_i) \qquad (1)$$

where $RImp_i$ is the relative importance of species i in the community type, WIV_i (wetland indicator value) is a value from 1 to 100 for species i based on the wetland indicator status (Table 4.1), and n is the number of species in the community type. The maximum community type wetland rating was 100, which would indicate all obligate wetland species in the community type, and the lowest possible score was 1, which would indicate all upland species in the community type. An example of a community type wetland rating calculation for a community type is shown in Table 4.2.

For sites, the relative cover of each community type was calculated by dividing the area covered by each community type by the total area sampled at the site. A site (greenline or cross-section) wetland rating was calculated as

$$\textbf{Site Wetland Rating} = \sum_{j=1}^{n} (RCov_j * CTWR_j) \qquad (2)$$

where $RCov_j$ is the relative cover of community type j at the site, $CTWR_j$ is the community type wetland rating from Equation (1), and n is the number of community types at the site. An example of the calculation of a site wetland rating is shown in Table 4.3.

A site wetland rating can also be calculated with species data. To do that the equation above is used, except that *"j"* refers to species instead of community types and the WIV value for each species is taken directly from Table 4.1 instead of using the CTWR.

Table 4.2. An example of a community type wetland rating calculation for the Carex rostrata (now Carex utriculata) community type of Padgett et al. (1989). See text for explanation of headings and abbreviations.

Species	Constancy	Average Cover	Importance	RImp	WIS	WIV	Species score
Carex utriculata	100	89	89.0	86.7	OBL	100	86.7
Carex nebrascensis	22	11	2.4	2.4	OBL	100	2.4
Carex aquatilis	22	6	1.3	1.3	OBL	100	1.3
Deschampsia cespitosa	17	4	0.7	0.7	FACW	75	0.5
Equisetum arvense	17	4	0.7	0.7	FAC	50	0.3
Agrostis stolonifera	11	25	2.8	2.7	FACW	75	2.0
Poa pratensis	11	17	1.9	1.8	FAC	50	0.9
Glyceria spp.	11	13	1.4	1.4	OBL	100	1.4
Calamagrostis canadensis	11	8	0.9	0.9	FACW+	83	0.7
Juncus balticus	11	2	0.2	0.2	FACW+	83	0.2
Salix boothii	6	10	0.6	0.6	FACW+	83	0.5
Caltha leptosepala	6	6	0.4	0.4	OBL	100	0.4
Mentha arvensis	6	5	0.3	0.3	FACW-	67	0.2
Dasiphora floribunda	6	3	0.2	0.2	FAC-	42	0.1
					Community type wetland rating (CTWR):		97.4

Table 4.3. An example of a wetland rating calculation for a riparian site (Kennally Creek, Idaho), based on the cover of community types described by Youngblood et al. (1985), Padgett et al. (1989), Manning and Padgett (1995), and Crowe and Clauznitzer (1997).

Community Type	Steps (Cov)	% of steps (RCov)	Community type wetland rating (CTWR)	Community type wetland contribution (RCov * CTWR)
CAAQ_Padgett	37	16.1	96.5	15.5
ALIN/MG_Padgett	34	14.8	71.4	10.6
ALIN/MF_Padgett	29	12.6	62.4	7.9
LUPO-SETR_Manning	25	10.9	69.8	7.6
CACA_Padgett	18	7.8	85.2	6.7
SAEX/MG_Padgett	16	7.0	81.2	5.6
CONIF/CACA_Padgett	15	6.5	62.0	4.0
SETR_Crowe	15	6.5	69.4	4.5
SAEX/MF_Padgett	8	3.5	67.5	2.3
SABO/CAAQ_Padgett	5	2.2	87.2	1.9
CONIF/MF_Manning	4	1.7	47.9	0.8
SABO/MG_Padgett	4	1.7	79.0	1.4
PICO/CAAQ_Crowe	3	1.3	76.7	1.0
SABO/MF_Padgett	3	1.3	68.5	0.9
SABO/CACA_Padgett	3	1.3	72.6	0.9
RHAL_Youngblood	3	1.3	43.2	0.6

SAEX/BRN_Padgett	2	0.9	91.2	0.8
CONIF/ACRU_Padgett	2	0.9	42.6	0.4
PICO/ALIN/MF_Crowe	2	0.9	62.8	0.5
RHAL2/MF_Crowe	1	0.4	32.8	0.1
CONIF/ACCO_Padgett	1	0.4	49.7	0.2
			Site wetland rating:	74.3

Data Analysis

The exclosure data set of 14 sites was used to compare the wetland ratings for each pair of exclosure and grazed sites, for both the greenline and cross-section wetland ratings. A one-sided paired t-test, with a 95 percent confidence level, was used to test if the wetland ratings were statistically higher at exclosures compared to grazed sites, using S-PLUS software program (Insightful Corp., 2001). Some stream channel variables that have documented relationships with livestock exclusion, and expected relationships to the wetland rating, were also compared for the pairs of exclosure and grazed sites.

RESULTS

Community Type Wetland Ratings

Wetland ratings were calculated for 725 community types from eight riparian vegetation classifications for the upper Columbia River basin. The average community type wetland rating was 64.3, and the range was from 5 to 100, although less than 1 percent of community types had a rating below 24 (Figure 4.1).

Wetland Ratings

Site wetland ratings, based on community type cover data and community type wetland ratings, were compared for the 14 pairs of exclosure and adjacent grazed sites. The greenline wetland rating had a mean pair-wise difference of 7.3 units between the exclosure and grazed sites, with the average exclosure rating being 12 percent higher than for grazed sites (Figure 4.2). A one-sided paired t-test confirmed that the greenline exclosure sites had a statistically higher wetland rating (P = 0.003) than the grazed sites (Table 4.4). For the 11 of the 14 sites where the exclosure greenline wetland rating was higher than the grazed-site wetland rating, the difference was between 1.8 percent and 31.0 percent. The three sites that had lower greenline wetland ratings in the exclosures were between 0.6 percent and 4.5 percent lower.

The cross-section wetland ratings were also higher in the exclosure sites, by an average of 7.6 units, or 14 percent, compared to the grazed sites (Figure 4.2). The one-sided paired t-test also indicated that the cross-section exclosure sites had a statistically higher wetland rating (P = 0.006) than the grazed sites (Table 4.4). For the 10 of the 14 sites where the exclosures had a higher cross-section wetland rating than the grazed sites, the difference was between 0.5 percent and 32.7 percent. The four sites that had lower exclosure cross-section wetland ratings were between 0.1 percent and 8.5 percent lower.

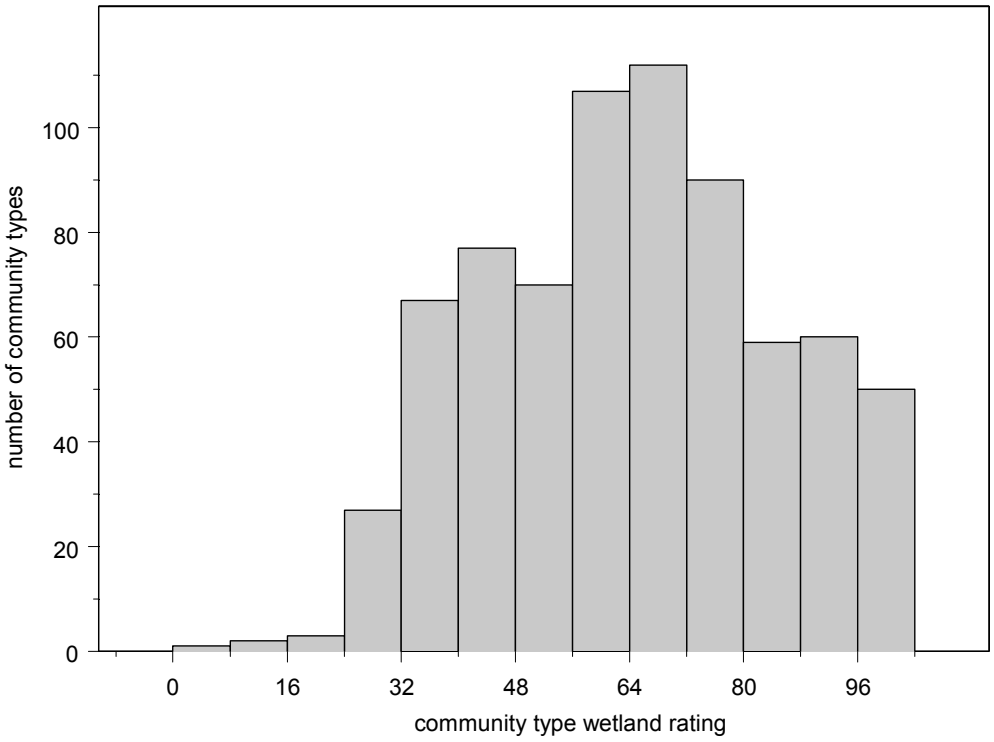
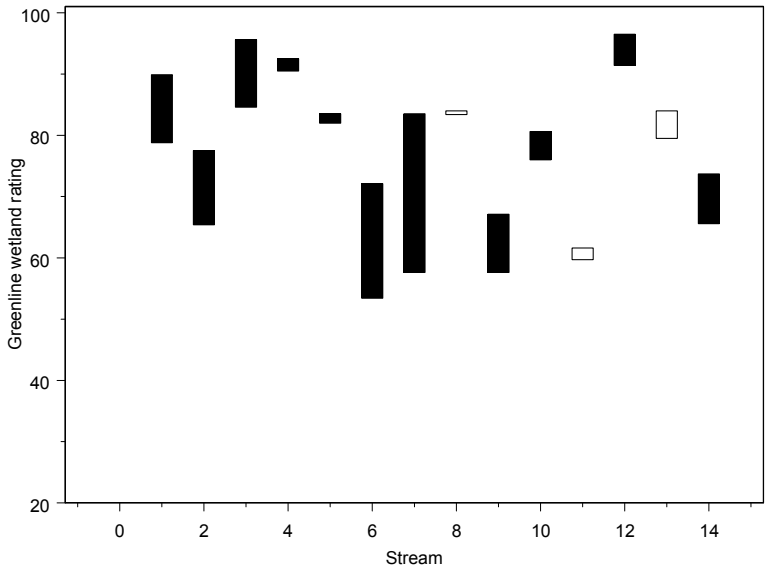
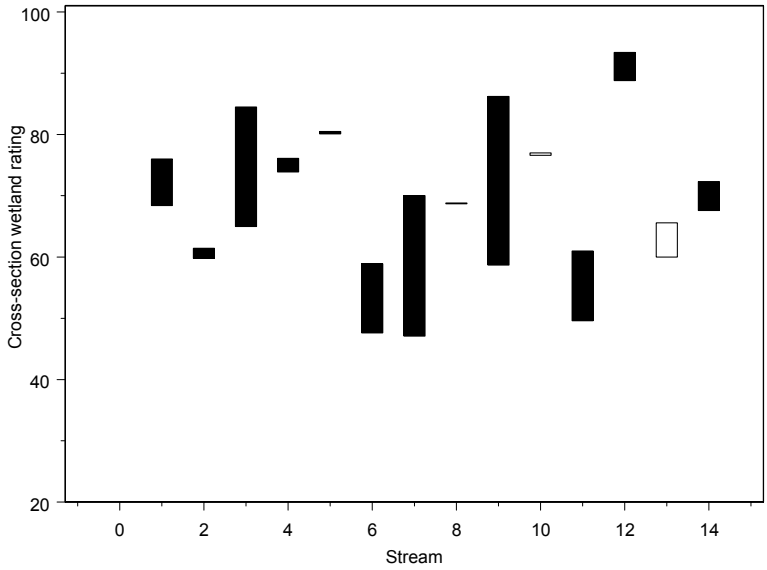


Figure 4.1. Community Type Wetland Ratings for 725 Community Types of the Interior Northwestern United States.

The 11 sites with the same observer were also compared to determine the effect of observer variability. The same patterns were found for both the greenline and cross-sections, with the exclosures having statistically higher wetland ratings (P = 0.02) in a pair-wise comparison (Table 4.4). The t-values and the mean differences were somewhat smaller, suggesting that observer variability slightly affected the differences at the 3 sites where different observers collected the exclosure vs. grazed site data; however, the differences were always significant and in the same direction, i.e. higher wetland ratings at exclosures.



(A)



(B)

Figure 4.2. Wetland rating differences for 14 pairs of exclosure and adjacent grazed riparian sites in Idaho and Oregon. Boxes represent the magnitude of the difference in the wetland rating based on the greenline (A) and cross-section (B) vegetation. Filled boxes were pairs where the exclosure had a higher wetland rating (top of box) than the grazed site (bottom of box). Open boxes represent pairs where the grazed site had a higher wetland rating than the exclosure.

Table 4.4. Results of one-sided, paired t-test for the wetland ratings and stream channel variables, comparing exclosure and grazed (outside exclosure) sites (n=14 pairs). The confidence level was 95 percent.

Variable	Direction of one-sided t-test for exclosures	Mean difference (exclosure – grazed)	t-value	p-value
Greenline wetland rating	greater	7.33	3.3	0.003*
Cross-section wetland rating	Greater	7.56	2.9	0.007*
Stream gradient (%)	lower	-0.03	-0.3	0.388
Bankfull width (m)	lower	-0.33	-1.3	0.100*
Width-to-depth ratio	lower	-3.86	-3.0	0.005*
Entrenchment ratio	greater	0.12	1.5	0.078*
Bank stability	greater	15.79	3.7	0.002*
Bank angle (%)	lower	-3.93	-0.9	0.192
Undercut as % of reach	greater	2.00	0.6	0.293

* significant at $p \leq 0.10$

Stream Variables

One-sided paired t-tests indicated that streambank stability and entrenchment ratio were statistically higher ($p < 0.10$) in exclosures compared to adjacent grazed areas (Table 4.4). Bankfull width and width-to-depth ratios were statistically lower ($p < 0.10$) in exclosures compared to the adjacent grazed areas. Other variables that were higher in exclosures, but not statistically significant (at $p < 0.10$) were: undercut percent, and undercut depth. Variables that were lower in exclosures, although not statistically significant (at $p < 0.10$), were: stream gradient and bank angle.

DISCUSSION

In semi-arid regions, livestock spend a disproportionate amount of time in riparian areas, where there is water, forage, cooler temperatures, and relatively flat terrain (Bryant, 1982; Kauffman and Krueger, 1984; Fleischner, 1994). This can lead to negative impacts for stream and riparian ecosystems, such as channel widening due to bank destabilization and reduced infiltration due to soil compaction. These impacts can decrease moisture available to plants, which along with pressure from herbivory, can lead to changes in species composition. The wetland rating is a tool to measure changes in species composition that are indicative of disturbance, particularly the important hydrologic processes that support riparian vegetation.

As hypothesized, the sites where livestock had been excluded had higher wetland ratings. The higher wetland rating indicated that livestock exclusion had resulted in different types of vegetation, i.e. a greater abundance of species that are commonly found in wetlands. This is consistent with other studies that have documented changes in vegetation as a result of livestock exclusion and a general increase in the abundance of obligate wetland species. Schultz and Leininger (1990) observed a dramatic increase in obligate wetland willows (*Salix* spp.) in a 30-year-old exclosure, and a dramatic decline in the facultative species Kentucky bluegrass (*Poa pratensis*). Green and Kauffman (1995) observed increases in obligate and facultative wet woody species on gravel bars in 15-year-old exclosures compared to grazed areas. Dobkin *et al.* (1998) recorded greater cover of obligate wetland sedges (primarily *Carex nebrascensis*) in a 34-year-old exclosure compared to the grazed area outside the exclosure.

Shifts from facultative to obligate wetland species are indicative of increased moisture availability. Such a shift would occur if the water table became higher and/or overbank flows were more frequent, which could result from channel narrowing as described by Gebhardt *et al.* (1989). The exclosure channels were slightly narrower than the channels of adjacent grazed areas. Magilligan and McDowell (1997) state that channel narrowing is one of the most consistent adjustments after livestock exclusion, which has also been documented by Hubert *et al.* (1985), Platts (1981), Myers and Swanson (1995), and Clary (1999).

The width-to-depth ratios were lower in the exclosures, indicating that the exclosure channels were generally deeper and/or narrower than the adjacent grazed areas, which is significant considering the close proximity of the sites (within a hundred meters of each other). The bank stability was significantly higher in the exclosures, which could indicate that the banks are better able to confine the stream during high flows, which promotes more overbank flow. While not statistically significant, there were more undercut banks and steeper banks (higher bank angle) in exclosures as well. These patterns are consistent with the higher entrenchment ratios found in the exclosures, indicating that over-bank flows have access to a wider floodplain area, which facilitates more dissipation of stream energy and a larger area of infiltration for flood waters.

A greater abundance of streambank vegetation has been associated with narrower and deeper channels (Ikeda and Izumi, 1990; Millar and Quick, 1993; Rowntree and Dollar, 1999). Obligate wetland species are particularly good bank stabilizers because they have more long, deep, and/or dense roots than facultative species (Manning *et al.,* 1989; Kleinfelder *et al.,* 1992; Dunaway *et al.,* 1994). This is consistent with the findings

of Micheli and Kirchner (2002) who observed that banks with obligate wetland sedges (*Carex* spp.) and rushes (*Juncus* spp.) had lower susceptibility to erosion than banks with dry meadow species (sagebrush (*Artemisia cana*) and annual grasses). Similarly, Toledo and Kauffman (2001) detected more obligate wetland species, and twice the root biomass, along narrower, unincised reaches compared to wider, incised reaches on the same stream. The cause of that channel incision, and the shift to species that grow in drier environments, likely included grazing, roads, or mining (Toledo and Kauffman, 2001). Converting their data to the wetland rating (as described above) showed that the unincised reaches had a wetland rating 25 units, or 30 percent, higher.

There are advantages and disadvantages to studying riparian vegetation in livestock exclosures. A significant advantage of exclosure studies is that environmental site variability (precipitation, geology, flow regime, etc.) is practically eliminated. Therefore the influence of livestock grazing can be isolated, while holding other variables generally constant.

Problems with studying exclosures include their small size, haphazard placement, and variable or unknown site histories in terms of livestock activity, which have led to conflicting results in some cases (Sarr, 2002). This is especially problematic when a study includes exclosures from only one stream, as was the case with all but one (Leege *et al.,* 1981) of nine exclosure studies evaluated here (Kauffman *et al.,* 1983; Schulz and Leininger, 1990; Allen and Marlow, 1994; Green and Kauffman, 1995; Case and Kauffman, 1997; Dobkin *et al.,* 1998; Clary, 1999; Brookshire *et al.,* 2002). Broad-scale studies are a way to overcome some of the weaknesses of the individual exclosures, and

to detect patterns that emerge from a group of sites (Sarr, 2002). This study sought to do that by evaluating sites from multiple watersheds in two states.

A limitation of exclosure studies is that the pairs of exclosure and grazed sites are nearby, so there are inevitably linkages between them. Rinne and LaFayette (1991) suggest that grazing be conducted only in downstream reaches, presumably to minimize the impacts from grazing on the exclosure site. In this study, a different approach was taken, in that the grazed sites were actually upstream of the exclosure sites. This was done so that the grazed site would not benefit from having an exclosed area just upstream that would buffer some of the impacts from livestock grazing. As a result, some impacts from livestock grazing were likely transferred to the exclosure site as well. Therefore the differences detected in this study between exclosure and grazed sites likely underestimate the recovery associated with livestock exclusion.

CONCLUSION

The wetland rating provided a uniform way to evaluate greenline and cross-section riparian data for riparian sites from a large geographical area. It was particularly useful as a summary technique for community type cover data, for which there were no known summary techniques applicable across the large PIBO-EMP study area. Wetland ratings can also be calculated for sites where species data are collected, as is currently being done by PIBO-EMP.

The wetland rating was used to compare exclosure and the adjacent grazed sites, demonstrating one application of this technique for evaluating riparian vegetation. The wetland rating was consistently higher in the exclosures for both the greenline and cross-

section data. This indicates that vegetation, and the wetland rating in particular, was responsive to livestock exclusion. The comparable environmental conditions between the exclosure and grazed sites isolated the influence of livestock, which likely enhances our ability to detect differences in the wetland ratings. If studies compare sites from different geographical areas, then site variability (precipitation, geology, flow regime, etc.) needs to be accounted for, as done by Kershner *et al.* (2004b) for stream variables.

The wetland rating can be broadly applied because a wetland indicator status has been assigned to most species that occur in wetlands across the United States (Reed, 1996). It should be noted that the wetland indicator status is a "best approximation of wetland affinity" and therefore not a perfect indication of site conditions, but the assignments had to be unanimous among a group of ecologists who know the region of interest (Reed, 1996).

The sites evaluated here were in relatively arid regions, where there are distinct differences in vegetation between riparian areas and uplands. In wetter areas, where the vegetation is more similar between riparian and upland settings, the responsiveness and usefulness of the wetland rating would need to be evaluated.

Monitoring riparian vegetation is difficult because change is a natural part of stream/riparian systems (Hughes, 1997; Tabacchi *et al*., 1998). Therefore, it can be unclear whether a change in riparian vegetation is important or not in terms of the functioning of the system. The wetland rating provides a technique for evaluating how disturbance affects processes, particularly hydrology, which determine the vegetation.

The wetland rating can be used to compare sites with similar environmental conditions, as presented here, or to monitor the same sites over time. It is a technique

that can be applied consistently across a large geographical area, as long as environmental differences are considered. Therefore, the wetland rating provides a useful tool for summarizing riparian vegetation data and for evaluating site condition as reflected by the types of species that are most abundant at a site.

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CHAPTER 5

EVALUATION OF THE REPEATABILITY OF METHODS THAT USE COMMUNITY TYPES FOR MONITORING RIPARIAN VEGETATION[4]

Abstract. The repeatability was evaluated for riparian vegetation monitoring methods that use vegetation community types as cover categories. When multiple observers collected data along a permanent transect, agreement for community type determination was about 39% at the scale of 1-m increments. At the scale of the site, the mean similarity among observers was 0.53 for the greenline, and 0.40 for the riparian cross-sections. This level of observer variability would make it difficult to use community type cover as a monitoring variable. The community type data were also converted to ratings of greenline bank stability, greenline successional status, and a wetland rating. The coefficients of variation (CVs) for these ratings were relatively good; the stability, successional, greenline wetland, and cross-section wetland ratings all had a CV under 14. The percent of total variability attributable to observers ranged from 16% for the greenline wetland rating, to 44% for the cross-section wetland rating. The sample size estimates (which includes equal sample sizes from two populations) necessary to detect a 10% change in the ratings, with an 80% confidence level, ranged from 56 sites (greenline stability rating) to 224 sites (greenline successional rating). Reducing the community types to ratings seemed to provide better precision than the community type cover data, because differences among observers that were not important from the perspective of the rating were eliminated. However, in order to overcome observer

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variability, numerous sites would need to be sampled to detect differences between two populations. Small differences among populations would be difficult to detect with a high degree of confidence, unless hundreds of sites were sampled.

INTRODUCTION

Many government agencies, and other entities, need to do monitoring to determine the level of anthropogenic influence on ecosystems. In order for actual differences to be detected across space and through time, methods must be objective and repeatable by different observers who sample the same site. Monitoring of riparian vegetation has become increasingly important because of the many important ecological and social benefits and functions of riparian ecosystems. However, the study of riparian vegetation is a relatively new field, compared to other branches of vegetation ecology, and there are few methods specifically designed to monitor these dynamic transitional zones between streams and uplands.

The methods of Winward (2000) have been promoted as a tool for monitoring riparian vegetation. These methods have been used by the effectiveness monitoring program (EMP) for streams and riparian areas of the upper Columbia River basin, which grew out of the PACFISH/INFISH Biological Opinion (PIBO). The objective of the PIBO-EMP is to determine whether stream habitat and riparian conditions on federal lands of the upper Columbia River basin are changing over time (Kershner et al. 2004). In order to be able to accurately detect differences, it is necessary to determine the level of observer variability associated with these methods.

This study evaluates the repeatability of Winward's riparian monitoring methods that rely on the use of community types as cover categories. I evaluate observer variability associated with Winward's greenline and cross-section data collection methods at the small scale (1-m increments) and at the larger scale (a stream site 110 m long). I also discuss possible modifications to the riparian vegetation monitoring methods of Winward (2000) in order to improve repeatability, and to capture more information about the vegetation at each site.

Observer variability is often overlooked in studies of vegetation, or it is assumed to be zero (Elzinga et al. 1998, Gotfryd and Hansell 1985). This assumption is dangerous because differences due to observers can arise from a number of sources, including: methods that allow observer subjectivity to affect decisions, methods that do not permit consistent measurement, recording errors, and incorrect species identification (Elzinga et al. 1998). Differences due to observers can be minimized with proper protocol development and training of observers. However, some observer difference is inherent in all sampling methods, so quality assurance testing must be conducted to determine how much observer variability exists. Understanding the degree of difference due to observer allows researchers to know the level of change detection that is possible.

Early researchers recognized observers as a source of variability in vegetation sampling data (Greig-Smith 1957, Hope-Simpson 1940). Imprecision due to observer difference has been documented for: frequency of species observance (Hope-Simpson 1940, Kirby et al. 1986, Leps and Hadincova 1992, Nilsson and Nilsson 1985); species cover estimates (Gotfryd and Hansell 1985, Kennedy and Addison 1987, Leps and Hadincova 1992, Sykes et al. 1983); and mapping of cover types (Cherrill and Mcclean

1999). Studies have also found that the same observer varied when repeated sampling was done at a site (Kennedy and Addison 1987, Smith 1944, Sykes et al. 1983). As would be expected, within-observer variability (the same observer doing repeat sampling) has been found to be lower than between-observer variability (Kirby et al. 1986, Smith 1944, Sykes et al. 1983).

Studies have also tested the repeatability of riparian vegetation sampling methods, although there are fewer studies than for upland vegetation, based on a review of articles listed in the bibliography by Elzinga and Evenden (1997). Some studies of repeatability in riparian vegetation methods were part of stream habitat studies that measured general categories of vegetation. One such study found "fairly high" variation among observers (95% confidence interval 1-6 times the field measurement precision) for a bank vegetation metric that categorized vegetation into broad categories (woodland, shrub, meadow, residential) (Wang et al. 1996). Another study found coefficients of variation (CVs) over 33% for estimation of general riparian vegetation categories in Oregon, such as amount of canopy, mid-layer, and ground-layer cover (Kaufmann et al. 1999). A method of characterizing vegetation was somewhat precise for the categories of herbaceous plants and shrubs (CV of 8%) but was less precise for a tree category (CV of 25%) (Barker et al. 2002). The use of such broad categories may have created difficulty distinguishing the categories, especially in transition zones, of which there are many in riparian areas.

Few studies have incorporated the implications of observer variability into their study design (Gotfryd and Hansell 1985, and Sykes et al. 1983). Understanding the level of observer difference is essential if researchers are to make appropriate assessments of

change over time or across space. Without data on observer-based variability, incorrect conclusions can be made about changes in vegetation and the role of management. Spurious results can include detection of changes that did not occur but were an artifact of observer difference (type I errors) and the failure to detect changes that did occur (type II errors). To minimize these errors, quality assurance testing must be done to identify and develop highly repeatable methods that generate monitoring data that will effectively detect change.

METHODS

Study Sites

Data were collected throughout the PIBO-EMP study area, which is federal (BLM and Forest Service) land within the upper Columbia River basin (Fig. 5.1). The sites sampled by PIBO-EMP are randomly selected each year from a subset of the 25 groups of 6th code watersheds, with at least 50% federal ownership (Forest Service or BLM).

The stream reaches analyzed here are a subset of the sites selected according to the process noted above, except for a few sites, which were chosen to represent different stream and habitat types. All reaches were within unconstrained valley bottoms with gradients less than 2% (Table 5.1). All were perennial streams with wadeable channels and bankfull widths between 2 and 9 m.

The study sites were in central Idaho on lands managed by the Nez Perce and Payette National Forests (Fig. 5.1). Data were collected at individual sites along six different streams: Boulder and Little Goose Creeks within the Little Salmon River drainage; Lost Creek within the Weiser River drainage; and Big, Jack, and Meadow

Creeks within the South Fork Clearwater River drainage (Table 5.1). These stream sites are numbered 1 through 6, respectively, in all figures.

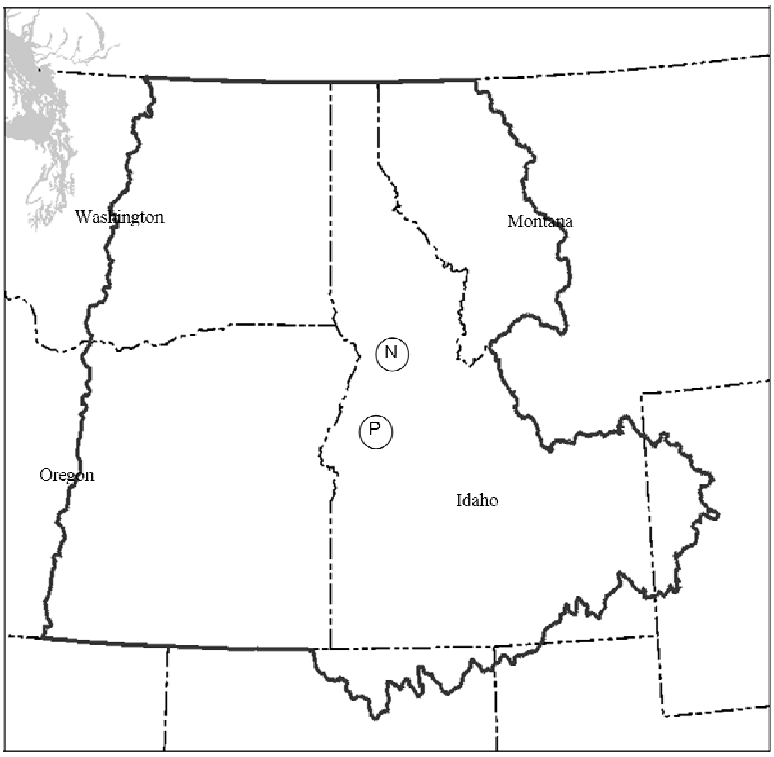

Figure 5.1. Map of the PIBO-EMP study area. The locations of the quality assurance test sites in Idaho are labeled "N" for the three sites on the Nez Perce National Forest and "P" for the three sites on the Payette National Forest.

Table 5.1. Information on stream sites used in the measurement and repeat tests of riparian vegetation sampling methods.

Site Code	Stream	National Forest	Geology	Elevation (ft.)	Gradient (%)	Bankfull width (m)
1	Boulder	Payette	Volcanic	4750	0.34	7.7
2	Little Goose	Payette	Volcanic	5000	1.57	3.5
3	Lost	Payette	Volcanic	4855	0.59	5.4
4	Big	Nez Perce	Granitic	6360	0.33	4.4
5	Jack	Nez Perce	Granitic	5280	0.73	2.7
6	Meadow	Nez Perce	Granitic	3200	0.37	8.3

The study sites were in central Idaho on lands managed by the Nez Perce and Payette National Forests (Fig. 5.1). Data were collected at individual sites along six

different streams: Boulder and Little Goose Creeks within the Little Salmon River drainage; Lost Creek within the Weiser River drainage; and Big, Jack, and Meadow Creeks within the South Fork Clearwater River drainage (Table 5.1). These stream sites are numbered 1 through 6, respectively, in all figures.

An additional 44 randomly selected sites within the two forests were used to estimate site variability. The quality assurance sites and the randomly selected sites represent a range of stream sizes, gradients, elevations, geology, and valley types that are sampled by the PIBO-EMP within these two national forests.

Data Collection

The riparian vegetation component of the PIBO-EMP adopted the riparian vegetation monitoring methods described by Winward (2000), which use community types to evaluate the vegetation at a site. These methods were chosen because: 1) they generate quantitative site values; and 2) they are currently used within land management agencies. Monitoring with community types allows rapid assessment of vegetation, and a certain degree of subjectivity.

In 2000 and 2001 PIBO-EMP undertook a quality assurance program to define the variability associated with the evaluation of stream and riparian habitat. The evaluation of stream channel parameters is presented in Archer et al. (2004) and Roper et al. (2002). This riparian component of that quality assurance program is presented here as well as in Coles-Ritchie et al. (2004).

The vegetation sampling methods of Winward (2000) were designed to evaluate the condition of riparian vegetation along the streambank (greenline vegetation

composition) and in the riparian area (vegetation cross-section composition). Winward's (2000) method to quantify woody species regeneration is evaluated in Coles-Ritchie et al. (2004).

An explanation of how the PIBO-EMP applied Winward's methods is in the PIBO-EMP vegetation sampling protocol (Kershner et al. 2004). That protocol was used to train the technicians (those who collected the vegetation data) in the sampling methods.

As per Winward (2000) the riparian vegetation data were collected over a distance of 110 m of streambank, on both sides of the stream. The vegetation and stream channel data collection began at the same point, which was considered the downstream end of the stream reach (hereafter referred to as site).

Eight riparian vegetation classifications (see Table 2.1) were used by PIBO-EMP for data collection. At each stream site the geographically appropriate classification was used to determine the community types present (see Kershner et al. 2004) for a listing of classifications to use by geographical area). When the primary classification for that area did not adequately describe the vegetation at the site, classifications from adjacent geographical areas could also be used. For sites in areas without a classification, multiple classifications from nearby areas were used in order to have access to community types that might be encountered.

The greenline data were collected along the first line (rooted point) of perennial vegetation adjacent to the stream that was at least 1 foot wide, with at least 25% cover of vegetation. The data collector walked along this greenline for 110 m and recorded the number of steps in each community type on each bank. The classifications (see Table

2.1) were used to determine the community type based on the dominant species or general species composition.

Five vegetation cross-sections were sampled along the 110 m of stream where the greenline was sampled. The cross-sections were evenly spaced along the stream reach, at meters 0, 27.5, 55, 82.5, and 110. For each site, the number of steps in each community type was recorded as the data collector walked a line perpendicular to the direction of the valley bottom (not necessarily perpendicular to the stream). The cross-sections extended in both directions from the greenline to the edge of the riparian area or 27.5 m, whichever was encountered first. The edge of the riparian area was determined to be the point where the vegetation was no longer a riparian community type listed in the classifications, or did not have greater than 25% cover of riparian species.

The cross-section width was also calculated. The number of steps for each cross-section, from one edge of the riparian area to the other, were averaged and then multiplied by the length of the observers step to determine the average riparian width in meters. Observers terminated the cross-section 27.5 m on each side of the stream at sites with wider floodplains, for a maximum width of 55 m. These data were used to identify a potential source of observer variability, not as a monitoring tool.

Vegetation data collectors (observers) received a total of 12 days of training at the start of the field season, which is extensive for a seasonal field crew. The training included eight days dedicated to learning the dominant riparian species, how to use community type classifications, and the sampling methods. Subsequent days were spent working in pairs to practice the methods at several training sites. The observers received a few additional days of training in their assigned geographical areas (Idaho, Oregon, and

Montana) on species and community type identification, and they sampled two additional training sites.

Synthetic vegetation indices

Two ratings of Winward (2000) were used to summarize the vegetation data. The greenline stability rating is based on community type stability classes, ranging from 1 (low stability) to 10 (high stability). This rating was designed to assess the vegetation's ability to withstand the force of moving water. The percent of each community type at a site was multiplied by the community type stability class and the resulting values were summed to obtain the greenline stability rating for the site. There are no defined units for this rating, so the term "units" is used.

The greenline successional rating is based on the categorization of community types as "early" or "late" successional by Winward (2000). These classes were used to calculate the percent of steps that were late successional community types at a site.

A limitation of these ratings is that they only apply to three of the eight community type classifications used within the PIBO-EMP study area. In addition, there is no rating for cross-section community type data. In order to have a uniform and quantitative rating system for our entire study area, and for both the greenline and cross-section methods, a "wetland rating" was developed for the community type data collected at each site.

Wetland ratings (between 0 and 100) were computed using data on the average species cover and constancy (percentage of plots with the species) from riparian vegetation classifications and the species wetland indicator status (Reed 1996). The

wetland rating was calculated for 725 vegetation types described in eight classifications for the upper Columbia River Basin.

In this study wetland ratings were calculated to summarize community type data as well as species data. Wetland ratings (based on species cover data and/or community type data) were compared at the quadrat level and at the site level. The site level community type wetland rating was calculated in the following manner. The sum all of steps for the greenline or cross-section for a site was calculated and converted to a percentage of the site. The percentage of each community type was multiplied by the wetland rating for that community type and then summed to obtain a site wetland rating. A wetland rating was also calculated with the community types for only those steps where a species quadrat was located.

Another wetland rating was calculated with the species data from the quadrats. This was done by calculating the average cover of each species at quadrats where that species occurred at a site. That was multiplied by the frequency of each species, which was the percentage of quadrats with that species. The product of average cover and frequency generated an importance value for each species at a site. The importance value was multiplied by a value corresponding to the wetland indicator status and then summed to obtain a site wetland rating based on species data.

Evaluating observer variability

Three separate components of variability associated with the evaluation of riparian vegetation, using community types, were assessed: 1) variability in application of a method at 1-m scale (measurement data); 2) variability in data summarized from an entire site (repeat data); and 3) variability among sites (site variability).

Riparian vegetation data were collected using the classifications by Padgett et al. (1989) on the Payette and Nez Perce National Forests and for many of the other 44 sites. The classification by Hansen et al. (1995) was used for 12 of the sites located on the Nez Perce National Forest.

The "measurement" component of the study was conducted with six to seven technicians collecting data at each of the six stream sites in August of 2001. This measurement study was designed to detect the causes of variability in each method at a small scale. For greenline and vegetation cross-section data collection, observers recorded the community type for each meter along a 10-m transect (right side of Fig. 5.2). This allowed an estimation of observer variability in assigning community types to describe vegetation.

The "repeat" component of the study included 6 sites that were sampled by six observers in both years, except for two sites in 2000 that were sampled by seven observers. Data collection occurred in late June of 2000, and August of 2001 at the three sites on the Nez Perce National Forest and in early August both years at the three sites on the Payette National Forest. Observers were randomly assigned to a site each day. Each observer established the boundaries of the site, based on a fixed starting point, and

followed the normal data collection procedures (left side of Fig. 5.2). These repeat sites were located adjacent to the area where measurement data were collected. The

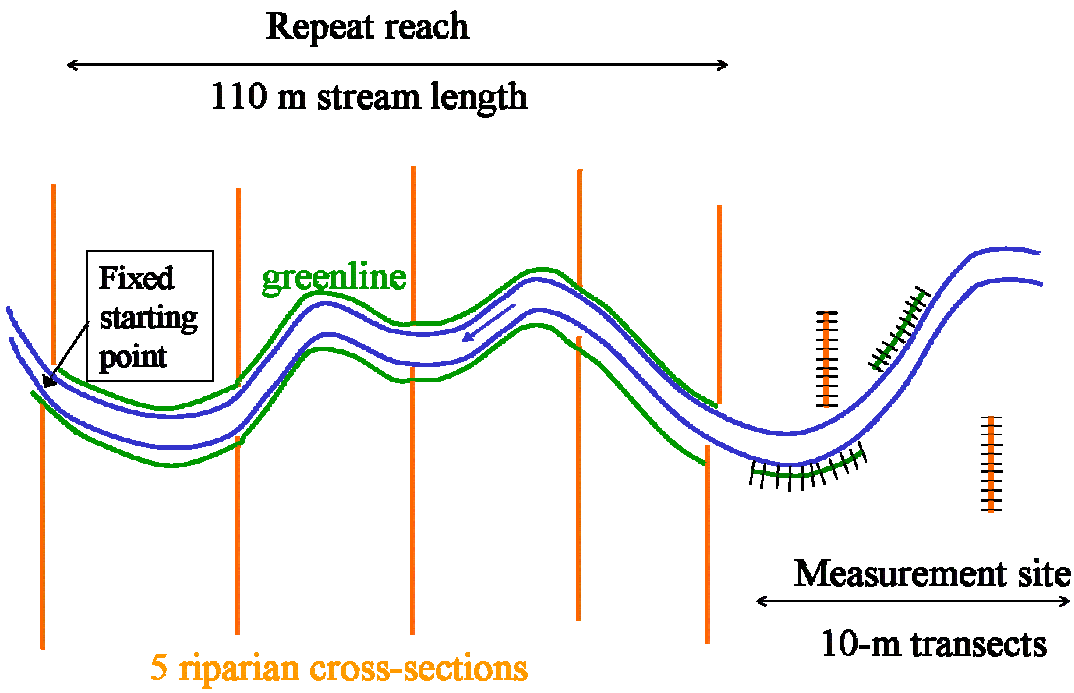

Figure 5. 2. Schematic of a repeat reach (left side) and a measurement site (right side). The repeat reach did not have fixed sampling points, whereas the measurement site transects were fixed, so that all technicians described the same 1-m segments.

repeat study include the small scale variability quantified with measurement data, and would indicate how that translated to variability at the site scale. In addition, the site level variability would include differences in interpretation and application of the methods, such as determination of where the greenline was located and differences in sample area due to using steps as the distance measure.

"Site variability" was estimated by calculating the variance among all 50 sites, six of which were the repeat sites. The additional 44 sites were used to better

characterize the variability of sites on the two forests. Of the 50 sites, the six repeat sites were sampled 12-13 times by 12 different observers over two years, six sites were sampled by two different observers in different years, and the remaining 38 were sampled once.

Measurement data analysis

An agreement matrix was used to calculate the between-observer agreement for all observer pairs at the 1-m units along the greenline and cross-sections. The percentage of 1-m units where pairs of observers agreed was the average between-observer agreement.

A fuzzy agreement matrix was also calculated by using the similarity of the two community types rather than "agreement." At 1-m units where a pair of observers recorded the same community type, the similarity was 100%. At 1-m units where observers recorded different community types the agreement was not zero, but rather the similarity (from 0 to 100%) of the two community types that each recorded. Those similarities were calculated with a Bray-Curtis dissimilarity index for all pairs of community types recorded. For example, the *Agrostis stolonifera* community type and the *Poa pratensis* community type (from Hansen et al. 1995) are 27% similar. When two observers used those two community types to describe the same 1-m area, the fuzzy agreement was 27% rather than 0. An average fuzzy agreement was calculated for all pairs of observers, based on the similarity of the community types they recorded, for all the 1-m units.

Greenline community type data were also converted to numeric values using the stability, successional, and wetland ratings, while the cross-section data were converted

to a wetland rating (the only rating available for this method). The ratings were substituted for the community type and a mean, standard deviation (SD), and CV were calculated for each 1-m unit. A grand mean, mean SD, and mean CV were calculated for all the 1-m units to represent the precision of each data reduction technique. The number of times that all technicians recorded the same value at a 1-m unit was also calculated. Because the greenline successional rating has only two classes it was only summarized with an average between-observer agreement.

Repeat data analysis

Community type data were compared among all observers and sites to determine observer agreement. The percent cover of community types recorded by an observer at each site was used to generate a dissimilarity matrix among sites, which was used to calculate within-group similarities and to generate ordinations.

Data from six to seven observers at the same site, for the same year, were compared with multi-response permutation procedures (MRPP) in PC-ORD (McCune and Mefford 1999). MRPP is a tool "to detect concentration within a priori groups" (Zimmerman et al. 1985). MRPP calculated the mean within-group similarity to indicate the similarity of community type data recorded by different observers at the same site.

Within-group dissimilarities were calculated with the Bray-Curtis dissimilarity index, in PC-ORD. A p-value (P) indicated the probability of obtaining the observed weighted mean within-group dissimilarity, relative to the distribution of possible values (McCune and Grace 2002). With these repeat data, it was expected that the p-values would be significant, as they were in each case. This is not very informative, because the

data are from the same sites and therefore should be more similar than expected by chance. Therefore p-values were not reported.

What is more informative is the level of similarity of data among technicians. This is represented by the agreement (A) statistic, which is the "chance-corrected within-group agreement" (McCune and Mefford 1999). The A statistic would be 1 if all items in the group were identical, 0 if the items in the group had the same heterogeneity that would be expected by chance, or negative if there were more heterogeneity than expected by chance (McCune and Grace 2002). As used in this analysis, the A statistic provides an indication of observer agreement, and hence the repeatability of the method.

The similarity of community type data, for different observers at the same site, was represented in ordination diagrams. Ordination is a method of arranging sites based on their similarity of vegetation and/or their environmental conditions (Kent and Coker 1992). The ordination method used was non-metric multidimensional scaling (NMDS), which Minchin (1987) and Clymo (1980) found to be the most robust of the ordination techniques that they evaluated. McCune and Grace (2002) concluded that NMDS was "the most generally effective ordination method for ecological community data and should be the method of choice, unless a specific analytical goal demands another method."

The NMDS ordinations were done with a Bray-Curtis dissimilarity index, which is what Minchin (1987) used in his study that found NMDS to be robust. In addition, McCune and Mefford (1999) recommend the Bray-Curtis dissimilarity index.

Preliminary ordinations were calculated to determine the appropriate number of dimensions, the best starting configuration, and to perform a Monte Carlo test with

randomized runs. The Monte Carlo test was done to evaluate the probability that a similar final stress could be achieved by chance. "Stress" represents the degree to which the dissimilarity in the ordination (with the number of dimensions selected) differs from the dissimilarity in the original dissimilarity matrix (McCune and Mefford 1999). PC-ORD (McCune and Mefford 1999) first calculated raw stress as:

$$S^* = \sum_{i=1}^{n-1} \sum_{j=2}^{n} (d_{ij} - \hat{d}_{ij})^2$$

Stress was normalized with the following equation:

$$S = 100 \sqrt{S^* / \sum_{i=1}^{n-1} \sum_{j=2}^{n} d_{ij}^2}$$

For all preliminary runs, a randomized starting configuration was used, the maximum number of iterations was 400, the instability criterion was 0.00001, the starting number of dimensions was 6, the number of runs with real data was 40, and the number of runs with randomized data was 50 (see Table 2.2). "Instability is calculated as the standard deviation in stress over the preceding x iterations, where x is set by the user" (McCune and Mefford 1999).

For each final ordination, the number of dimensions (axes) was selected according to the following criteria: an additional dimension was added if it reduced the stress by 5 or more, and if that stress was lower than 95% of the randomized runs ($P <= 0.05$ for the Monte Carlo test) as recommended by McCune and Mefford (1999).

The final run was done with the best starting configuration for the number of dimensions selected and no step-down in dimensionality (see Table 2.2). For final runs the following information was reported: the number of dimensions that were statistically different than 95% of the randomized runs (Monte Carlo test results), number of

iterations, final instability, final stress, and the cumulative R^2 based on the correlation coefficient between the ordination distances and distances in the original n-dimensional space.

The community type data for each site were also converted to ratings and a random effects model (Littell et al. 1996; SAS 2000 (PROC MIXED)) estimated the mean and variance associated with each rating. The variance was calculated for: site, year, site with year interaction, observer, and residual. All five variance components were estimated from the combined 2000 and 2001 data, but only the variance estimates for site, observer, and residual were used to evaluate observer variability.

Site variability described the variation among sites not attributable to observer or interannual variability (year effect). Interannual variability was partitioned out and not included in the analysis of observer variability.

Observer variability was defined as the sum of the observer and residual terms, which includes all error associated with observer and all unexplained error (residual). The unexplained error was included as part of the observer variability because that variability will be part of the data that are used to monitor sites, or groups of sites.

The objective was to assess the precision with which riparian vegetation attributes could be measured. Observer precision was evaluated by calculating the standard deviation (SD) and coefficient of variation (CV) for each variable. In addition, the intraclass correlation coefficient was calculated as the magnitude of the observer variability (observer and residual) relative to the overall variability (site, observer and residual).

Calculation of minimum sample sizes

The minimum sample sizes necessary to detect a specified difference between the means of two populations were calculated. For this study, observer and site variability (based on estimated variance components described above) were both included, in estimating sample sizes. Sample sizes were evaluated for means that ranged from 5 to 50%. This range was chosen because differences of these magnitudes likely included changes in attributes that would result in a biological response. A type I error rate of α = 0.1 was used, and a type II error rate of β= 0.1.

Estimates of sample size were calculated following the iterative procedure outlined by Zar (1996, page 133, equation 8.22):

$$n \geq \frac{2S_p^2}{d^2}\left(t_{\alpha(2),v} + t_{\beta(1),v}\right)^2$$

where:

$S_p^2 = \dfrac{(n_1 - 1)S_1^2 + (n_2 - 1)S_2^2}{(n_1 - 1) + (n_2 - 1)}$ is the pooled estimate of variance

$v = (n_1 - 1) + (n_2 - 1)$ is the degrees of freedom for S_p^2

$t_{\alpha(2),v}$ = 2 - tailed t - value on v df for a type I error rate of α

(also used for 1 - α, two - sided confidence intervals)

$t_{\beta(1),v}$ = 1 - tailed (upper) t - value on v df where β is the acceptable type - II error rate

d = minimum difference to be detected.

For these calculations, variance estimates from the repeat study were used as estimates of S_p^2. Total variance was calculated as the sum of site and observer variability (Clark et al. 1996, Montgomery 1984, Ramsey et al. 1992). This equation calculates the

number of samples needed from each population and assumes equal sample sizes. If the number of samples from one population is constrained (for example, few ungrazed sites), it would be necessary to adjust the sample size of the unconstrained population. When *n* exceeded 30, values for infinite sample size were substituted because differences in results were minimal.

One final consideration is that when taking a sizeable sample (more than 5%) without replacement from a finite population, each observation "carries" more information than when sampling with replacement or from an infinite population. This extra information results in a slight decrease in the variance, accomplished by multiplying the usual variance by the finite population correction factor, $(1-n/N)$ where N is the number of elements of the population and n is the sample size. The value n/N is known as the sampling fraction. Corrections for finite populations were not included in our sample size estimates, so our estimates are conservative.

RESULTS

Greenline measurement data

The greenline measurement data had a mean community type agreement for all observers of 38% for the 119 1-meter units of vegetation along the greenline (Table 5.2).

Table 5.2. Measurement site summary statistics describing observer agreement for different riparian vegetation sampling methods and data summary techniques. Values represent the average agreement for 119 1-m units (n) from six measurement sites, for six to seven observers. The * symbol indicates not applicable.

Method	Data summary technique for 1-m units	Mean (all 1-m units)	Mean SD	Mean CV	Mean agreement (%)	% of 1-m units with unanimous agreement
Greenline	Community type agreement	*	*	*	38	5
	Community type fuzzy agreement	*	*	*	48	*
	Stability rating	8.2	1.0	14.6	*	39
	Successional rating	*	*	*	83	62
	Wetland rating	81.1	8.1	10.7	*	*
Cross-section	Community type agreement	*	*	*	39	10
	Community type fuzzy agreement	*	*	*	50	*
	Wetland rating	81.7	10.1	13.0	*	*

The maximum agreement of community types at a measurement site was 49% and the minimum agreement was 29%. All seven observers recorded identical community types at 5% of the 1-meter units on the greenline. The fuzzy agreement calculation increased the average agreement to 48%, because of the similarity in species composition between community types.

The measurement community type data were also converted to the corresponding value for the three ratings and then compared among observers for every 1-m unit (Table 5.2). The greenline stability rating had a mean CV of 14.6% and complete agreement at 39.5% of the 1-m units. The greenline wetland rating had a CV of 10.7%. The greenline successional rating had 83% average agreement among observers, although with only two possible categories random agreement would be 50%. There was complete agreement among observers at 62.2% of the 1-m units.

Greenline repeat data

For repeat sites, observers recorded an average of 10 community types per site on the greenline. The maximum number of community types recorded by an observer at one site was 18 and the minimum was 3.

The final solutions of the 2000 and 2001 greenline NMDS ordinations are presented in Table 5.3. The ordination graphs (Figs. 5.3 and 5.4) depict the greenline data for the 6-7 observers in 2000, and the 6 observers in 2001 at each of the six sites. The average within-site similarity among observers for the greenline community type data was 65% in 2000 and 51% in 2001 (Table 5.4). The chance-corrected agreement statistic (A statistic) was 0.4 for the 2000 data and 0.3 for the 2001 data.

The community type data were converted to ratings and were then evaluated by site. The greenline stability rating had a CV of 7.4% and an SD of 0.6 units (Table 5.5, Fig. 5.5). The greenline successional rating had a CV of 13.4% and an SD of 10.3% (Table 5.5, Fig. 5.6). The greenline wetland rating had a CV of 5.9% and an SD of 4.4 units (Table 5.5, Fig. 5.7).

The percent of the total variability attributed to observers was 35.7% for the stability rating, 27.8% the successional rating, and 15.8% for the greenline wetland rating (Table 5.5). The sample sizes necessary to detect change were calculated to describe the implications of site and observer variability for monitoring efforts. To detect a change of 10% (power was set to 0.9 for these calculations) between two populations, considering both observer and site variability, the following number of sites would be needed for each

Table 5.3. The final solutions of the NMDS ordinations of riparian sites, based on cover of community types. Dimensions selected were those that were both statistically

significant and that each decreased stress by at least 5. The R^2 value is based on the correlation coefficient between the ordination distances and distances in the original n-dimensional space.

Data set	Dimensions selected	Iterations	Final Instability	Final Stress	R^2
Greenline (2000)	2	44	0.00007	11.3	.893
Greenline (2001)	3	200	0.03312	13.1	.882
Cross-section (2000)	3	200	0.00268	12.8	
Cross-section (2001)	4	200	0.00483	10.2	.593

Table 5.4. Repeat site observer variability statistics generated from MRPP. Analyses were done on community type cover data for six to seven observers who each collected data at six riparian sites. The same six sites were sampled in 2000 and 2001.

Data	Within-site similarity (%)	Agreement statistic
Greenline (2000)	64.9	0.436
Greenline (2001)	50.5	0.285
Cross-section (2000)	49.0	0.314
Cross-section (2001)	39.6	0.221

Table 5.5. Repeat site observer variability statistics for quantitative summaries, or indices generated from community type cover data. The mean values are based on 50 sites to account for site variability. All other values are based on the six repeat sites. The stability and wetland ratings do not have units.

Method	Data summary technique	Mean	SD	CV	% of total variability attributable to observer
Greenline	stability rating	7.6	0.6	7.4	35.7
	successional rating (percent)	76.3	10.3	13.4	27.8
	wetland rating	75.6	4.4	5.9	15.8
Cross-section	wetland rating	70.7	7.6	10.8	44.0
	width (meters)	40.0	7.3	18.3	44.7

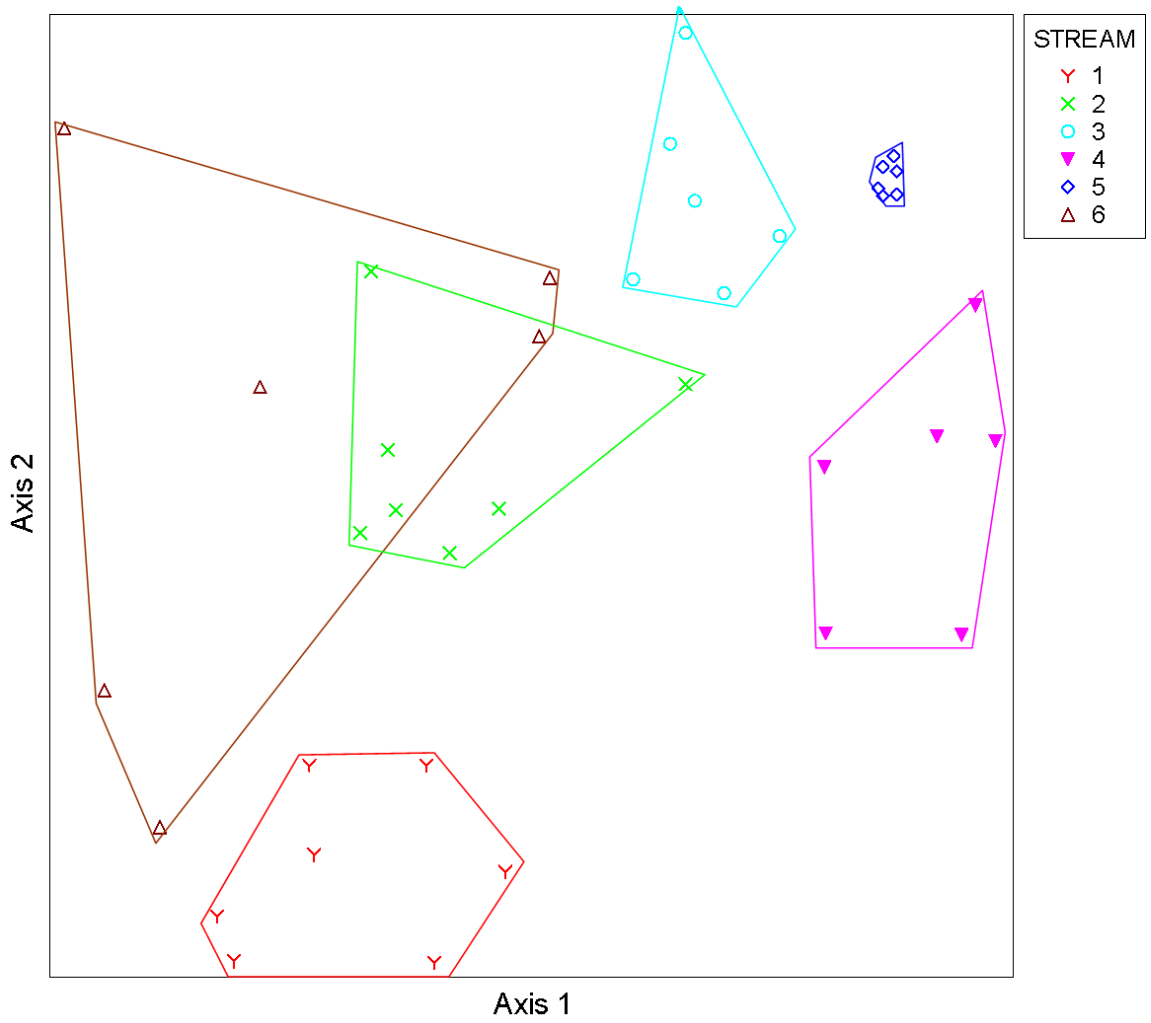

Figure 5. 3. Non-metric multidimensional scaling (NMDS) ordination of greenline community type data for the six to seven observers in 2000 at the six repeat sites. A convex hull surrounds all the observers at a given site.

technique (double the value in the 10% column of Table 5.6): 56 for stability, 224 for successional, and 78 for greenline wetland.

Cross-section measurement data

The measurement results of the cross-section method indicated a mean community type agreement for all observers of 39% for a given 1 m of vegetation (Table

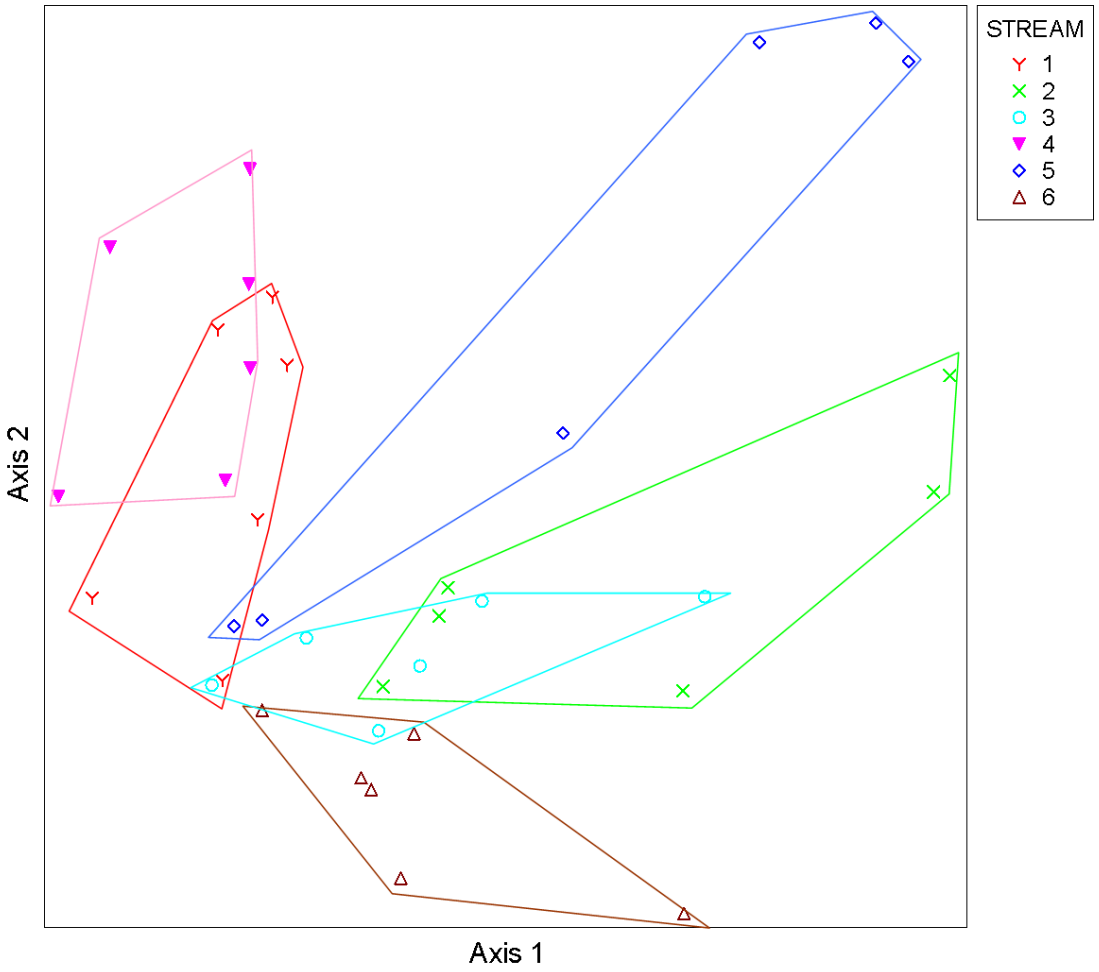

Figure 5. 4. Non-metric multidimensional scaling (NMDS) ordination of greenline community type data for the six observers in 2001 at the six repeat sites. A convex hull surrounds all the observers at a given site.

5.2). The maximum between-observer agreement at a site was 55% and the minimum was 29%. All seven observers recorded the same community type at 10% of the 1-m units at measurement sites. The fuzzy agreement calculation increased the community type agreement to 50%.

Cross-section repeat data

Observers recorded an average of 7.4 community types per repeat site for the cross-sections. The maximum number of community types recorded by an observer at one site was 12 and the minimum was four.

The final solutions of the 2000 and 2001 cross-section NMDS ordinations are presented in Table 5.3. The ordination graphs (Figs. 5.8 and 5.9) depict the cross-section data for the six to seven observers in 2000, and the 6 observers in 2001 at each of the six sites.

The average within-site similarity for the 6-7 observers for the cross-section community type data was 49% in 2000 and 40% in 2001 (Table 5.4). The A statistic was 0.3 for the 2000 data and 0.2 for the 2001 data.

Table 5.6. The minimum sample sizes needed to detect difference/change with riparian vegetation sampling methods when both observer variability and site variability are considered. Sample size estimates assume equal size samples, so values listed below indicate half the total sample needed. The value listed in each column is the sample size needed to detect the stated change with a Type I error of 0.1 and a Type II error of 0.1.

Method	Data summary technique	Sample sizes needed to detect a change of:					
		5%	10%	20%	30%	40%	50%
Greenline	stability rating	107	28	8	4	3	3
	successional rating	446	112	28	14	8	6
	wetland rating	150	39	11	6	4	3
Cross-section	wetland rating	183	46	13	6	4	3

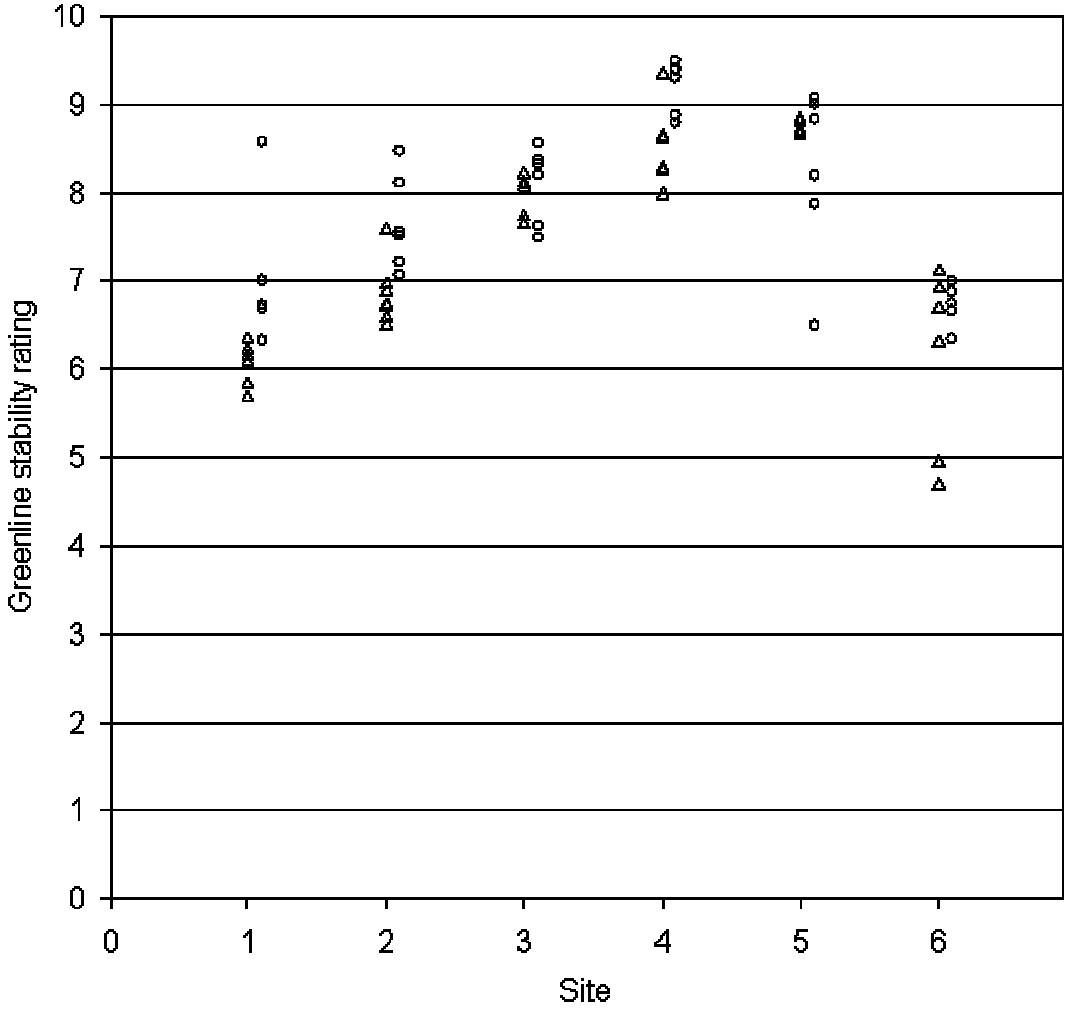

Figure 5. 5. Greenline stability ratings for all observers at the six repeat sites. Observers in 2000 are represented by triangles and observers in 2001 are represented by circles, which are offset slightly to the right.

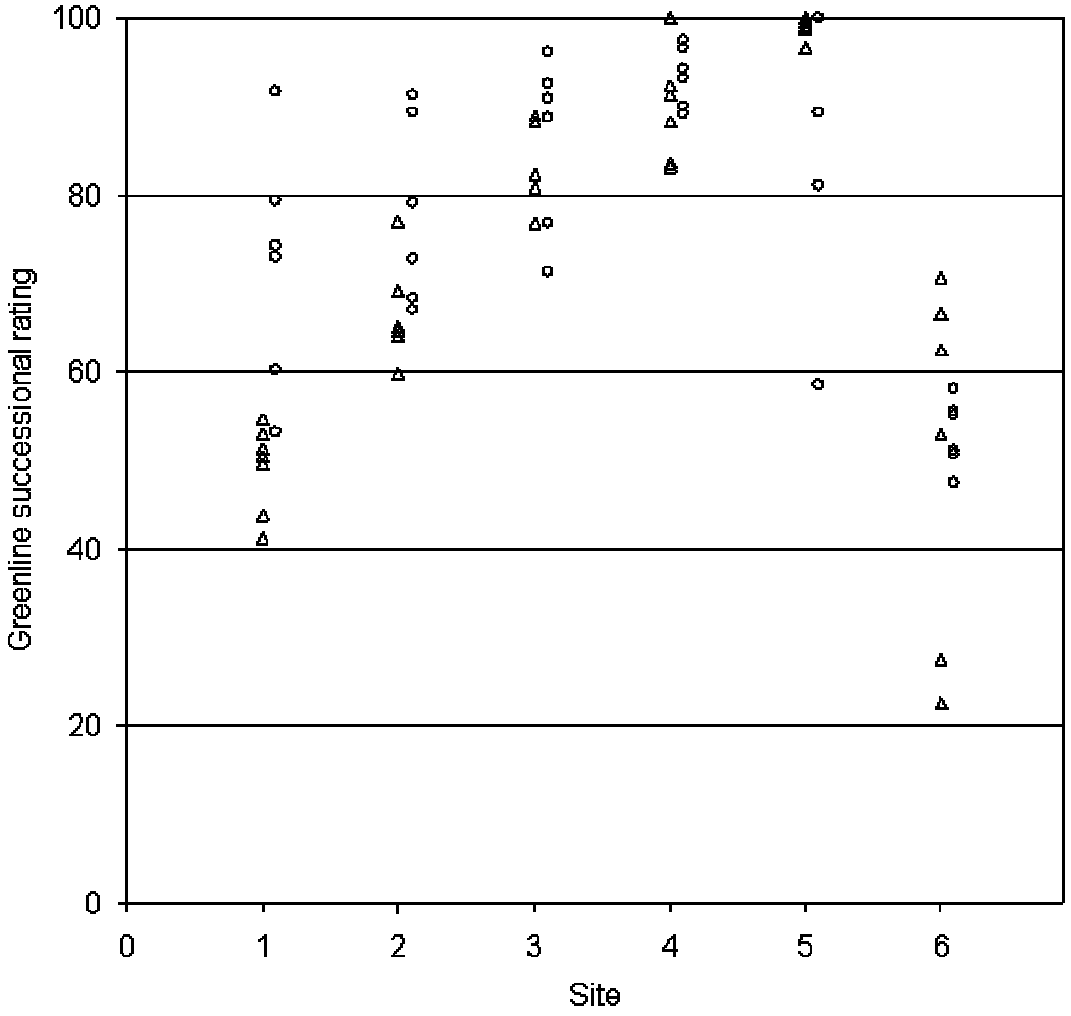

Figure 5. 6. Greenline successional ratings for all observers at the six repeat sites. Observers in 2000 are represented by triangles and observers in 2001 are represented by circles, which are offset slightly to the right.

The cross-section wetland rating had a CV of 10.8% and an SD of 7.6 units (Table 5.5, Fig. 5.10). The cross-section width had a CV of 18.3% and an SD of 7.3 m (Table 5.5, Fig. 5.11). The percent of the total variability attributed to observers was 44.0% for the vegetation cross-section wetland rating (Table 5.5). The total number of sites needed to detect a 10% change (power was set to 0.9 for these calculations) would be 92 sites for the cross-section wetland rating (Table 5.6).

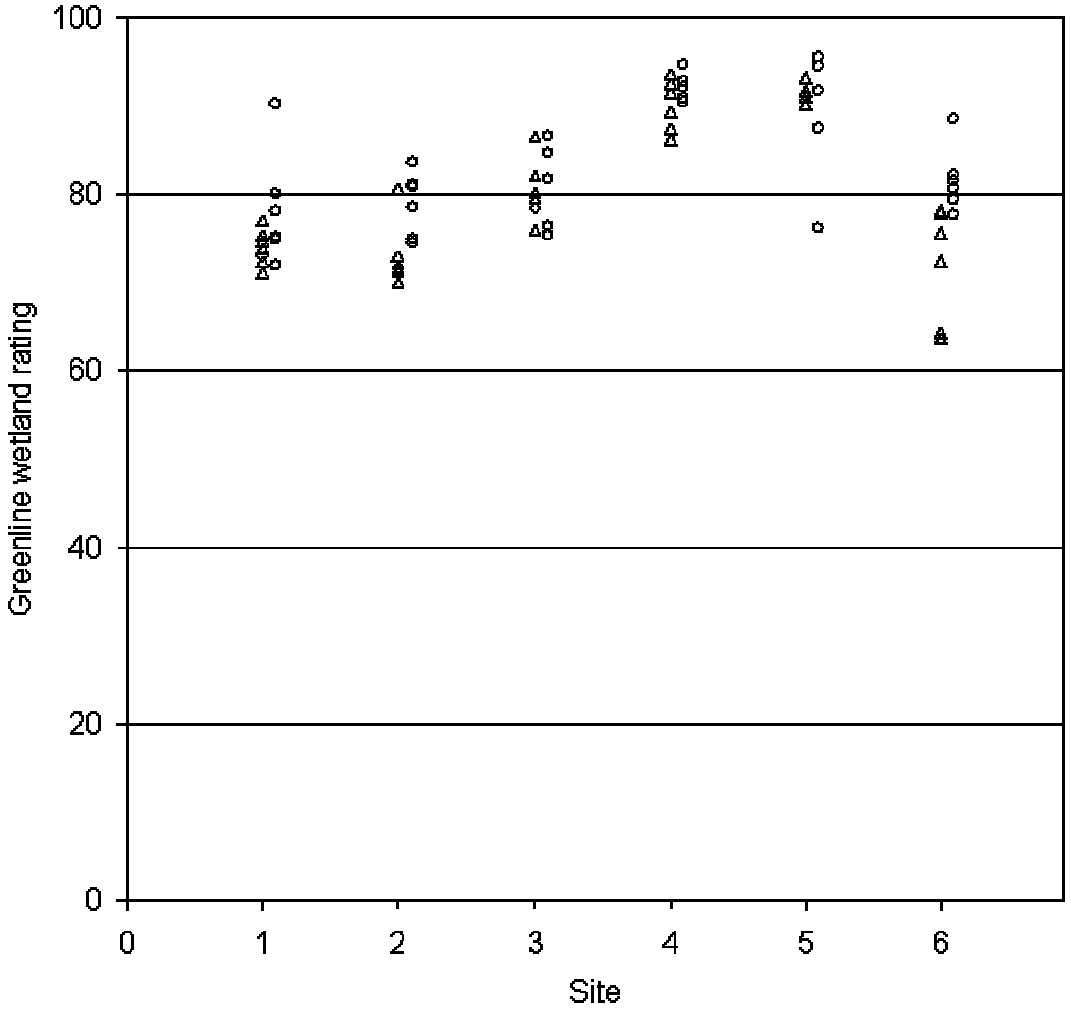

Figure 5. 7. Greenline wetland ratings for all observers at the six repeat sites. Observers in 2000 are represented by triangles and observers in 2001 are represented by circles, which are offset slightly to the right.

DISCUSSION

Observer agreement for community type data was relatively low as indicated by the lack of clustering of sites in the ordinations (Figs. 5.3, 5.4, 5.8, and 5.9). There was also overlap among many of the sites in the ordinations, as shown by the overlap in convex hulls, each of which is the convex set that includes all points from a site. There

was better clustering, and little overlap among sites, in the 2000 greenline ordination (Fig. 5.3), indicating a moderate level of agreement among observers. In general, there were relatively large distances among some observers at the same site, and overlap among many of the sites. This would make it difficult to use community types to detect differences in vegetation among sites with different observers.

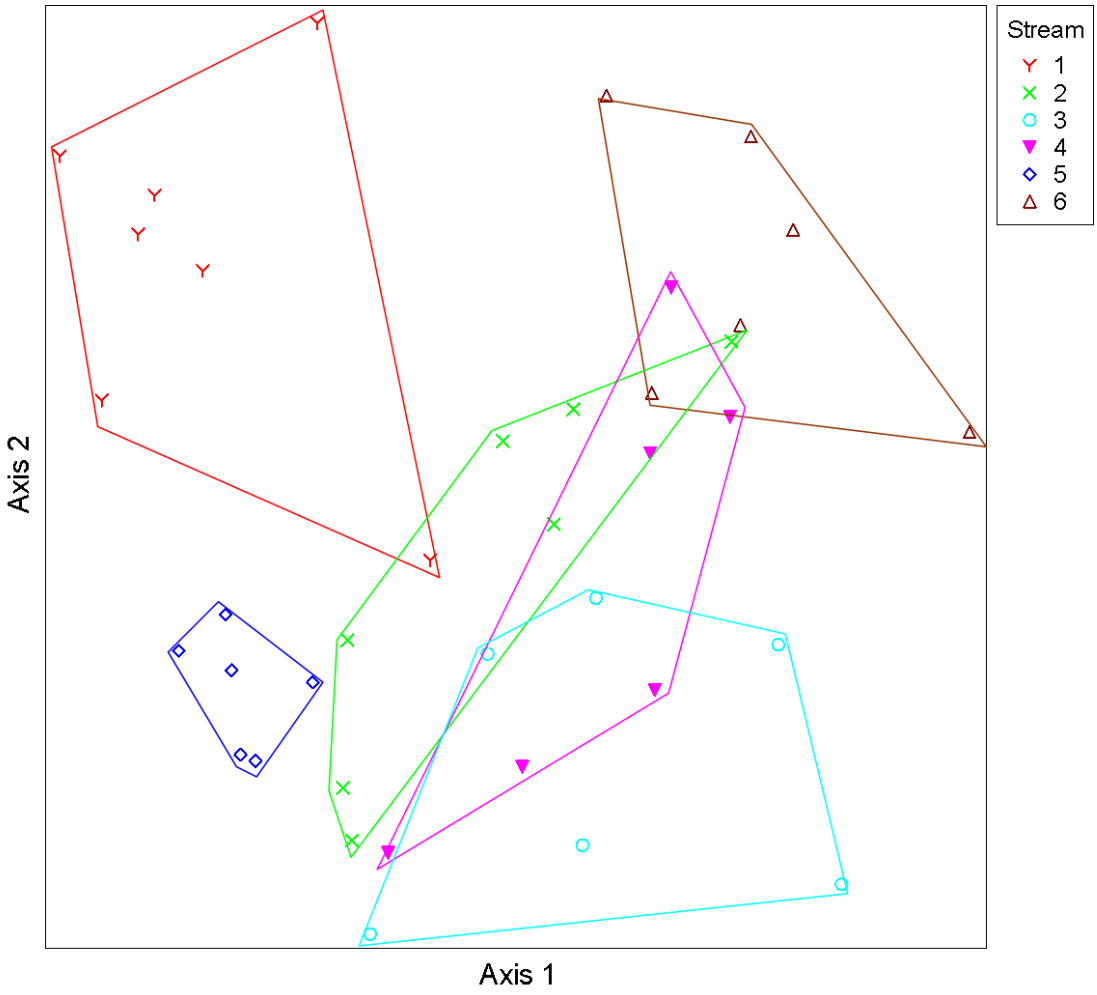

Figure 5. 8. Non-metric multidimensional scaling (NMDS) ordination of the cross-section community type data for the six to seven observers in 2000 at the six repeat sites. A convex hull surrounds all the observers at a given site.

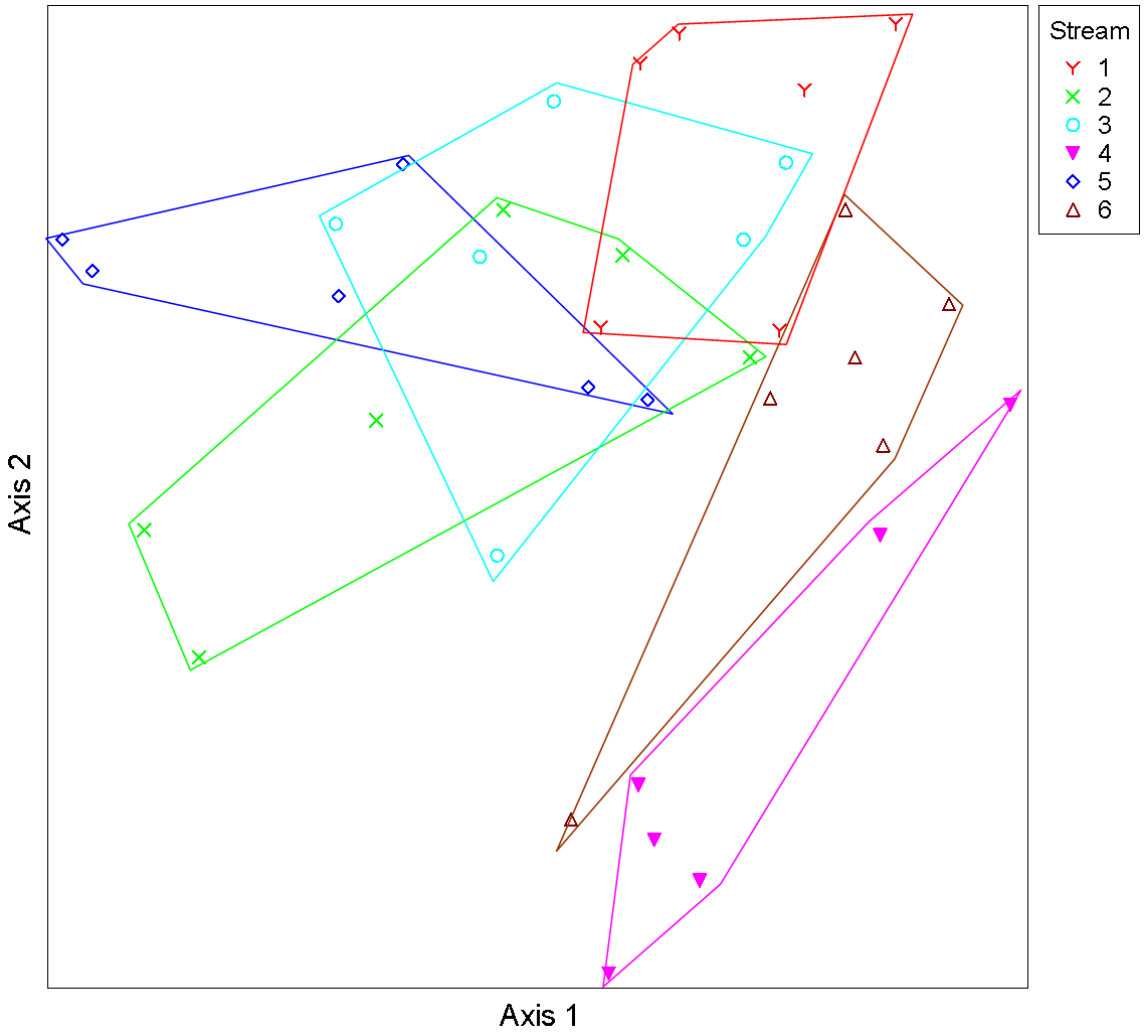

Figure 5. 9. Non-metric multidimensional scaling (NMDS) ordination of the cross-section community type data for the six observers in 2001 at the six repeat sites. A convex hull surrounds all the observers at a given site.

The A statistics, between 0.2 and 0.4, would be relatively good for community ecology, given that a value greater than 0.3 is considered "fairly high" (McCune and Grace 2002). But in this case, where observers were collecting data at the same site, a much higher A statistic would be possible, and expected. The A statistics were well below one, and only somewhat better than for geographical variables that grouped sites (see Fig. 3.2). The relatively low within-site similarity, and the overlap in sites in the

ordination, mean that using community type data with multiple observers would make it difficult to detect differences in vegetation among sites or over time.

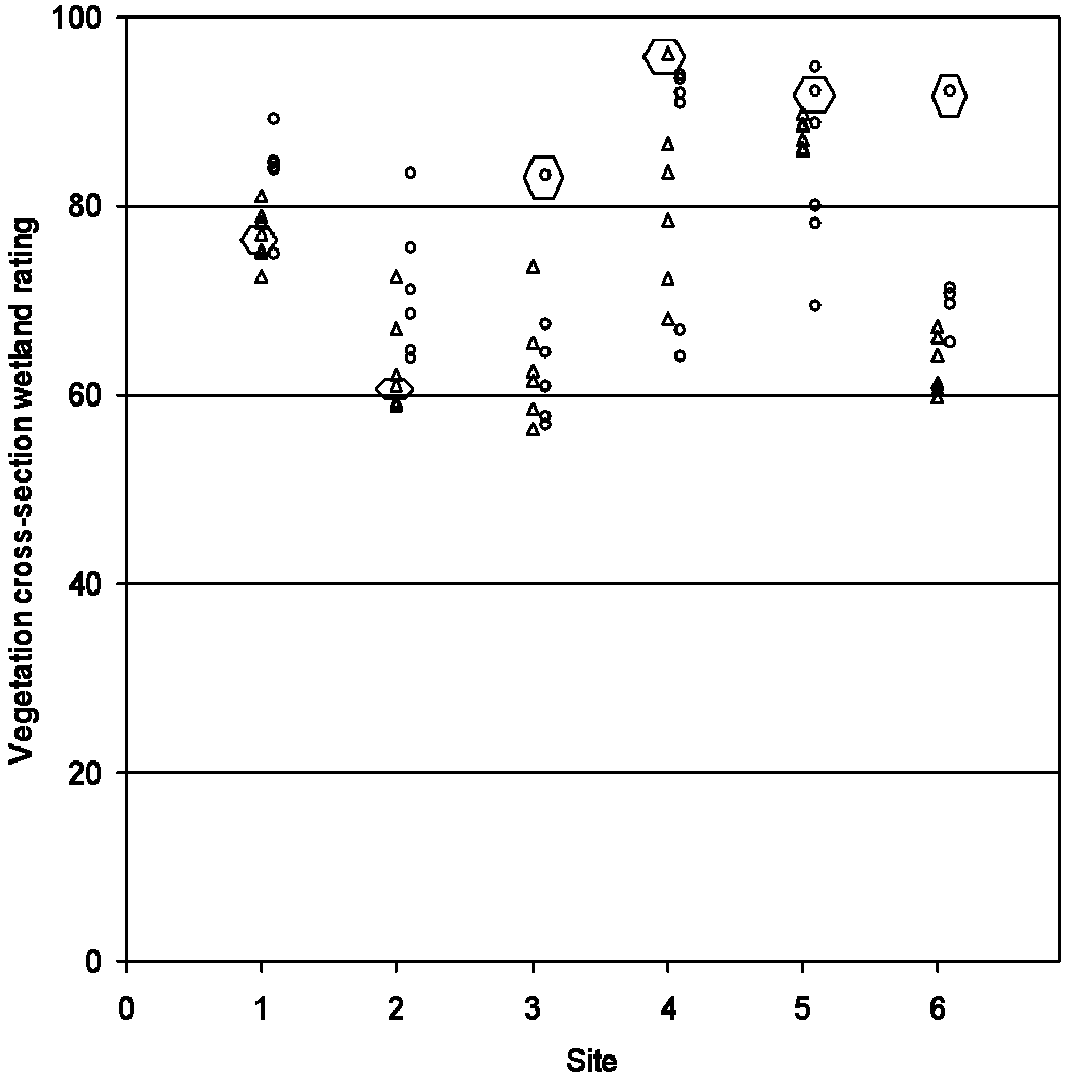

Figure 5. 10. Vegetation cross-section wetland ratings for all observers (triangles for 2000 and circles for 2001) at the six repeat sites. The observers circled (with a hexagon) are the same observers circled in Fig. 5.11.

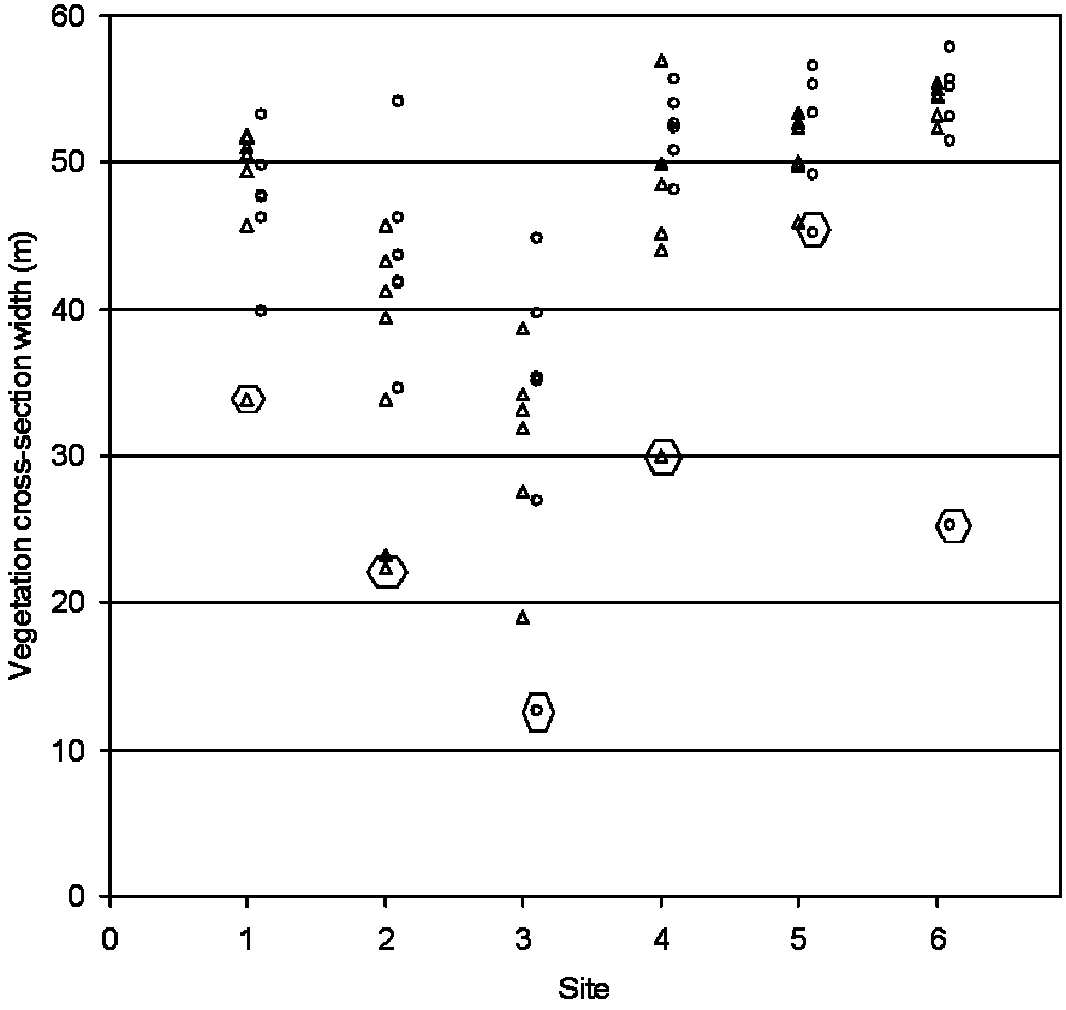

Figure 5. 11. Average width of cross-sections for all observers (triangles for 2000 and circles for 2001) at the six repeat sites. The observers circled (with a hexagon) are the same observers circled in Fig. 5.10.

The ideal would be to have very low observer variability, which would be represented by tight clustering of observers at the same site in the ordination (such as for stream 5 in Fig. 5.3). That would result in an increased ability to detect changes in vegetation at a site, or differences between sites.

The final instability values for the NMDS ordinations were higher than what McCune and Grace (2002) recommended when they said to "strive for instability < 10^{-4}." The final stress values were "fair" to "poor" according to Kruskal's rules of thumb listed in McCune and Grace (2002). The high stress values could decrease confidence in the results, but guidelines on stress are considered "over-simplistic" by Clarke (1993, cited in McCune and Grace 2002), in part because "stress tends to increase with increasing sample size."

The three rating techniques improved observer agreement, although some information is inevitably lost in the process. The percent of 1-m units where all six or seven observers agreed increased from 5-10% for community types to 39% for the greenline stability and 62% for greenline successional ratings. The low CV's (less than eight) for the greenline stability and greenline wetland ratings indicate that observer precision was relatively good. The cross-section wetland rating and the greenline successional rating were slightly less precise, with CVs over 10.

However, precision is not the only variable of interest. It is also important to consider how much of the total variability is attributable to the observer. Other studies have suggested that it becomes difficult to assess differences between sites or changes through time when the total variability attributed to observers is greater than 20% (Clark et al. 1996, Ramsey et al. 1992) or greater than 33% (Kaufmann et al. 1999). All summary techniques but the greenline wetland rating had observer variability that exceeded the 20% criteria for, and the greenline stability and cross-section wetland rating exceeded the 33% criteria.

These results exemplify the difficulty of evaluating the usefulness of each method for monitoring. For example, the stability rating was relatively precise (CV of 7.4) but had high variability attributed to observers (35.7%). The sample size of 56 sites (28 for two different groups) to be able to detect a 10% change would be feasible only for a well-funded study.

The greenline wetland rating was precise (CV of 5.9) and had a relatively low variability attributed to observers (15.8%), but it still had a sample size of 78 sites (39 for two groups) to detect a 10% change. Even though the greenline wetland rating had a lower CV and percent variability attributed to observer than the stability rating, it had a higher sample size requirement because site variability (among all 50 sites) was higher than for the stability rating (Tables 5.5 and 5.6).

The cross-section wetland rating was relatively precise (CV of 10.8), yet almost half of the total variability was due to observers. The variability among observers at a site was about as large, or larger than the variability between the mean values for the six sites (Fig. 5.10). The combination of observer and site variability resulted in a sample size of 92 sites (46 for two different groups) needed to detect a 10% change.

All of the summary techniques had sample size estimates over 56 sites to detect a change of 10%, and over 200 sites to detect a change of 5%. Those sample size requirements would be feasible for well-funded studies, but would limit the ability to detect change for smaller studies.

One source of variability for the cross-sections was differences in defining the edge of the riparian area. The location of the riparian edge was determined independently by each observer based on vegetation and landform, both of which are

subjective judgments. Comparison of the vegetation cross-section widths and the wetland ratings for the same observers suggests that precision was sometimes affected by differences in defining the edge of the riparian area. Differences in the wetland rating would be expected because there is generally a decrease in obligate wetland plants moving away from the stream. To highlight this influence, the lowest value for the cross-section width for each site is circled (with a hexagon, Fig. 5.11), and the value for the same observer is also circled (with a hexagon) on the cross-section wetland rating (Fig. 5.10).

For four of the six sites the lowest riparian width was associated with the highest or second highest wetland rating for that site. This suggests that defining a shorter riparian width can sometimes cause the wetland rating to be higher than it would be if a larger area were sampled. These data underscore the importance of using methods that result in a consistent sampling area. Winward (2000) suggested using permanent markers, which would alleviate the problem of variable width. In some situations permanent markers are not feasible, in which case a more repeatable method of identifying the riparian edge, or techniques to always consider a comparable area, would be needed.

The misidentification of species, or differences in estimating the species with the greatest cover, also contributed to variability in community type determination. Converting the vegetation data to ratings often decreased observer variability due to different identification of the same species, or different judgment of the dominant species, because ratings were sometimes similar for the different species. For example, at one repeat site observers recorded three different sedge dominated community types

(*Carex rostrata*, *Carex nebrascensis*, and *Carex aquatilis* community types) along the greenline. This resulted in disagreement in community type data, however it had little affect on the greenline ratings since all three community types had the same successional (late) and stability (9) ratings, and very similar wetland ratings (96 to 97, for community types of Padgett et al. 1989). The dominant species in these community types are probably ecological equivalents, in that they grow in similar habitats and perform similar ecosystem functions – such as bank stabilization, sediment trapping, and soil development. In this and other cases, taxonomic differences, which were sometimes not consistently recorded in the field, were not always important to understand the functioning of the riparian ecosystem.

While the previous example showed that converting community types into more general categories increased observer agreement, in other situations converting data to ratings did not improve agreement. This was evident at a site where some observers recorded more of the *Calamagrostis canadensis* community type (found in moist conditions) and others recorded more of the *Poa pratensis* community type (found in a drier environment). The differences in these community types are reflected in the values of the stability ratings (7 versus 3), greenline successional ratings (late versus early), and the wetland ratings (85 versus 48). The fact that observers recorded these different grass community types for the same area led to high variability in the ratings, as sell as for community type data.

CONCLUSION

The objective of riparian monitoring efforts is generally to detect change in habitat characteristics that are caused by anthropogenic disturbances. Our ability to detect these changes is affected by observer variability, which is greatly affected by the methods used. Minimizing observer variability requires that methods generate consistent data when applied by multiple observers at the same site. Understanding the magnitude of observer variability, and the sources of that variability, allows us to improve methods and to implement useful monitoring.

This study examined observer variability, and its sources, for vegetation sampling methods that utilize community types as the cover categories. Multiple observers collected data at specific locations (measurement data) to identify sources of observer differences. Multiple observers collected data for entire sites (repeat data) to estimate and partition observer and site variability. Different classifications were used for the same site to determine the influence of classifications as a source of spurious variability. community type and species data were collected to evaluate differences between the two methods at the step and site level.

Repeatability of community type composition is poor

When multiple observers recorded community types at the same 1-m increments (measurement data), the average pair-wise agreement was only 39%. There was unanimous agreement (for six or seven observers) at only 5% and 10% of 1-m units for the greenline and cross-section methods respectively. Greater agreement was detected

with a fuzzy agreement matrix (about 49%), which shows the overlap in community types that is not captured by using community type cover categories alone.

Some of the small-scale variability among observers was reduced when data were summarized over the site level. The repeat data showed within-site agreement for community type data of 53% for the greenline and 40% for cross-sections. But the ordinations and group analyses show a high degree of variability among observers at the same site, and overlap among sites (Figs. 5.3, 5.4, 5.8, and 5.9). The MRPP evaluations of both the greenline and cross-section data resulted in A statistics below 0.3 which would be considered fair for community ecology (McCune and Grace 2002) but poor for data that represent the same site.

In general, the greenline and vegetation cross-section methods show that community type data, at both the small scale (1-m units) and large scale (site level aggregations), had low similarity among observers. That would impede the ability to detect change in vegetation unless the change was wholesale.

Community type data would be more repeatable on a study of a single site, by a single observer, because observer variability would be greatly reduced and site variability would not be an issue. However, for most monitoring efforts it is inevitable that different observers will collect data at different times. Therefore, for studies involving multiple sites and multiple observers over time, community type cover data would only be useful for detecting very dramatic changes in vegetation.

Synthetic vegetation indices improve repeatability

Community type data were also converted to ratings, which seek to represent attributes important to managers. These data summary techniques improved observer agreement because they minimized the influence of differences in species or community types that were irrelevant for a given rating.

With the measurement data, all observers could be compared at the same 1-m units. At only 5% of those 1-m units was there 100% agreement in the community type recorded among the six to seven observers. When the community types were converted to ratings, the unanimous agreement increased to 39% for the greenline stability rating, and to 62% for the greenline successional rating.

Some of the small-scale variability among observers was reduced when data were summarized over the site level. Repeat data for sites showed that most of the ratings were relatively precise, with CVs between 5 and 14. The greenline stability, greenline wetland, and cross-section wetland ratings, would be precise enough to detect large changes (greater than 20%) with feasible sample sizes (less than 13 sites in each of two populations). Detecting a smaller change (10%) with these three ratings would require larger sample sizes (between 28-46 sites in each of two populations).

The greenline successional rating had much higher observer variability, suggesting that change detection would be very difficult with that rating technique. One reason the greenline wetland and stability ratings may have been more precise than the greenline successional rating was that they were based on a numeric rating scale, with more distinctions. Conversely, the successional rating had only two categories (early and late) and therefore observer differences were always very influential on the rating.

While the ratings showed greater precision among observers than the raw community type data, ratings would still only permit detection of relatively large changes in riparian vegetation. If a large change occurred quickly then the wetland and stability ratings could prove useful. If change occurred gradually, over a long time period, these ratings would be less effective at detecting change. It may take so long to detect a negative change, that resource damage would have occurred before change could be confidently detected. Positive change could also take a long time to be confidently detected, failing to provide prompt feedback about the effectiveness of management.

In addition, information is inevitably lost with such data reduction techniques, which can be problematic for accurate change detection.

Why repeatability is low when using community types

Community types are imprecise

Vegetation classifications have relatively few categories compared to species data, and therefore it has been presumed that they are easier to use. Alpert and Kagan (1998) noted that "easy detection" is one of the advantages of using community types for management. Winward (2000) stated that one of the objectives of the greenline and cross-section methods was "to be efficient in both time and cost." While detection may appear easy, our results indicate that community type detection is not always easy or repeatable.

Using community types as the cover category is difficult because plant communities generally do not have distinct boundaries. This makes it difficult for observers to consistently categorize vegetation with community types. There has been

considerable debate in vegetation ecology about the appropriateness of community types because they are not "real entities" (Alpert and Kagan 1998). Some community types may seem obvious on the landscape, but as our results indicate, communities cannot always be consistently determined.

Scale is also a complicating factor when using community types as cover categories. There is no defined spatial area that community types cover. The classifications evaluated here considered areas from $4m^2$ to $50m^2$ when creating the community types. But there is no clear definition of what spatial area (i.e. how large or small an area) can be called a community type, and how to distinguish a change in a community type from variability within a community type. There are keys in the classifications that are designed to help determine the community type in the field, but they are intended at the scale of many square meters, not at the scale of a step, which is what the methods evaluated here require.

In a large community, steps are a useful way to estimate the size of the community type at a site. However, often there is variability at every step, which may or may not be considered a change in the community type. In this study, more often than not observers recorded different community types at each step. This exemplifies the confusion of trying to determine the community type at the scale of a step. When data were summarized for an entire site, some of the variability was reduced, but there was still much variability among observers in the community type cover recorded.

Classifications are variable or not available

The vegetation composition of similarly named community types in different classifications was relatively different. This means that using classifications

interchangeably at the same site over time will add a source of variability for monitoring. Variability in vegetation composition is inevitable due to geographical variability in vegetation, the limitations of sampling, and differences in the way each classification was done.

The classification of community types can be performed in various ways, with community types being lumped or split, depending on the prerogative of the classifier. In addition, the site selection criteria may differ. For example, some classifications seek to describe only natural, preferably late successional, vegetation (Crowe and Clausnitzer 1997, Hansen et al. 1995), whereas other classifications include areas disturbed by human activities (Manning and Padgett 1995, Padgett et al. 1989, Youngblood et al. 1985). So the objectives of the classification will influence the community types that are generated. The keys in the classifications, which help to determine the community type in the field, are also done differently.

A problem encountered by PIBO-EMP was that a riparian vegetation classification has not been developed for all areas. For example, there was no classification for central Idaho, which was the location of many of the sites analyzed here. The community types described in the classifications by Hansen et al. (1995) and Padgett et al. (1989) covered much, but not all, of the vegetation encountered at these sites. Plant assemblages that were not documented in the classifications caused uncertainty and therefore variability among observers. Any geographically large study that relies on community types will encounter the problem that classifications have not been done for all geographical areas.

Differences in identification and cover estimation

Differences in community types recorded at 1-m units are partially due to different identification of species, or different judgment of the abundance of species. The accuracy of species identification was not evaluated, but evidence indicated that at some 1-m units observers recorded different community types of the same life form (i.e. *Carex aquatilis* community type and *Carex nebrascensis* community type), when there was likely only one of those species/community types abundant for that 1-m area.

Variability in estimating cover, and which species is most abundant in a plot, are other reasons for variability in vegetation data. This affects community type determination because the keys in the classifications are based on the dominant species in the area considered. Studies that have assessed the precision in estimating canopy cover found results ranging from low variability in a controlled experiment (Hatton et al. 1986) to moderate to high variability using field sampling methods (Barker et al. 2002, Pollard and Smith 1999, Smith 1944). The results from these studies, and ours, suggest that estimations of canopy cover contribute to observer variability when using community types as the cover category.

The results from this study underscore the need for repeatable methods of monitoring riparian vegetation. Modifications to the Winward (2000) sampling methods, which could potentially improve repeatability, are summarized in Appendix A, and are described in detail in the Coles-Ritchie et al. (2003).

Numerous researchers and land managers have collected data on riparian vegetation, but few protocols exist for systematic monitoring of riparian areas. A riparian monitoring protocol is in development by Weixelman et al. (2001), but quality assurance

testing has not yet been completed. The protocol of Winward (2000), evaluated here, was one of the first for monitoring riparian vegetation and it has been important in focusing attention on the importance of riparian vegetation. These methods have motivated many people to look more closely at riparian areas, the ecosystem functions they provide, and how these important functions can be altered by land management activities.

The observer variability, associated with methods that use community types to describe riparian vegetation, limits their usefulness for monitoring with multiple observers--which is generally unavoidable. When community type cover data are reduced to site-level ratings, such as the wetland rating and the greenline stability rating, large changes may be detected if an adequate number of sites are sampled. Small but potentially important changes in riparian vegetation would be more difficult to detect when multiple observers use these methods, unless hundreds of sites were sampled.

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CHAPTER 6

CONCLUSIONS

Riparian vegetation monitoring is a relatively new area of ecology and natural resource management compared to other fields such as forest and rangeland monitoring. This is likely due to the challenges of monitoring riparian systems because they are dynamic (due to fluctuations in stream flow) and have irregular shapes (generally narrow and linear). As recognition of the importance of riparian areas has increased, there has been an effort to better understand and monitor riparian vegetation. The publication of Winward (2000) and his training efforts have been instrumental in improving the general understanding of riparian ecology and the importance of riparian areas for plants, wildlife, and social benefits.

As a relatively new field of study, the methods of riparian vegetation data collection and summarization are still being evaluated and refined. Therefore it has been important to evaluate the usefulness of the different methods of data collection and data summary for detecting impacts from management activities. As part of that it has been important to determine the precision of the data in terms of change detection capability.

For the data to be useful for monitoring management impacts, the data need to be comparable for all sites in the study area and the data need to be sensitive to management activities. PIBO-EMP initially used the methods of Winward (2000) which recommend the use of community types to describe vegetation cover at riparian sites. Community type classifications have been developed for distinct geographical areas, which in this study only covered portions of the study area. Different classifications have different community types that were derived in different ways. This made it difficult to compare

different sites based on their cover of community types which were from numerous classifications.

In order to evaluate the PIBO-EMP data community types from the different classifications were combined so that sites from across the large study area (the upper Columbia River basin) could be compared. Cluster analysis proved to be a useful tool for combining community types from multiple classifications. The combining of community types from the different classifications enabled the evaluation of relationships between vegetation types, management, environment and stream channel characteristics. However, some degree of information about sites was inevitably lost when data for individual sites were lumped. An alternative that does not require grouping, such as using species rather than community types as the cover categories, would have advantages. As a result of the work presented here, PIBO-EMP has modified its riparian vegetation data collection methods to record species rather than community types at riparian sites (Appendix A). This has improved the ability of PIBO-EMP to make comparisons of all sites in the study area.

Tests of the repeatability of Winward's (2000) methods were done to determine if different observers generated similar vegetation data or ratings derived from the data. The repeatability of those methods, which use community types as the cover categories, was fair but not great. In some cases there were dramatic differences among observers who collected data at the same point (1-meter increments) or site (110 m stream reaches). The differences were most pronounced when comparing the actual community type cover data.

When the community types were converted to ratings, there was moderate agreement among observers. This suggests that large changes in the synthetic ratings for greenline bank stability or greenline wetland rating could be detected with these methods, but smaller changes would be more difficult to detect. Observer variability would reduce the ability to detect change in riparian vegetation.

As a result of these tests, and the difficulties of comparing sites where community types from different classifications were used (noted above), modifications to Winward's methods have been developed and are now being implemented and tested by PIBO-EMP (see Appendix A). An important objective of the changes has been to improve observer agreement, which would increase change detection capability. Initial testing of these modified methods has shown improvements in observer agreement. That is in addition to the increase in detailed information about the site vegetation, such as information about the presence of noxious weeds, tree species, etc.

The data from PIBO-EMP have provided an opportunity to evaluate riparian vegetation data and its relationship to stream variables. Ordinations facilitated the identification of four general vegetation/site types and their relationships to management and stream characteristics: 1) conifer-alder forests; 2) wet sedge-willow sites; 3) dry-grazed sites; and 4) steeper-shrub sites. The dry-grazed sites seem to be the most impacted by management as indicated by the relative abundance of atypical vegetation for riparian areas, such as the *Poa pratensis* meta-community type. The combination of low precipitation and cattle grazing probably makes those sites more vulnerable to impacts from livestock grazing and associated changes in vegetation and stream characteristics. Even when accounting for the low precipitation in cattle-grazed

watersheds, there was still a strong relationship between cattle grazing and the *Poa pratensis* meta-community type. This grass community is generally considered to be an increaser with livestock grazing.

Across the basin as a whole, there was a weak relationship between vegetation and the general management category. It may be that management of riparian areas has improved such that managed sites are not unusually different than reference sites. That is probably the case in some areas. On the other hand, differences were detected between the wetland rating for vegetation (meta-community types) at exclosure sites compared to adjacent non-exclosed sites. Statistically higher wetland ratings (for the greenline and cross-section) were detected at exclosure sites versus the outside exclosure sites. This suggests that livestock grazing does have a significant impact on riparian vegetation, at least at sites where managers have chosen to place livestock exclosures.

The data presented here suggest that different sites respond differently to management activities and the associated disturbances. Some sites are impacted less by management, or are better able to withstand certain disturbances. Other sites seem to be more vulnerable, such as the low precipitation sites that were grazed by cattle. Impacts from management may be difficult to detect in comparisons across a large area because of the enormous environmental variability among the sites.

The PIBO-EMP data presented here has enabled an evaluation of the relationships between vegetation, management, environment, and stream variables. While community types may not be the most sensitive and repeatable cover categories, they did facilitate the identification of numerous relationships among these variables.

The wetland rating, as a synthetic index for monitoring, proved to be sensitive to impacts from management, such as livestock grazing at the site level. The wetland rating was significantly higher in the exclosures, indicating that this rating is responsive to changes in management (removal of livestock in this case), at least for riparian areas along low-gradient streams in Idaho and Oregon. The wetland rating was less informative in comparisons of different sites, probably because of the environmental differences which create different potentials for the abundance of hydric vegetation. The wetland rating can be a useful tool for land managers who need to assess the recovery of riparian systems after changes in livestock management.

Additional techniques are needed help managers and researchers evaluate riparian vegetation in terms of ecosystem functioning. To be useful these variables need to be ecologically important and measurable in a repeatable manner. The large PIBO-EMP dataset provides opportunities for testing and identification of such monitoring variables, which will be an important component of future analyses with their riparian vegetation data.

As a follow-up to the work presented here, testing of the modified riparian sampling methods is being done by PIBO-EMP. This includes tests of the repeatability in terms of the data or indices generated by the methods. PIBO-EMP data are also being used to develop new data summary techniques that will be useful for land managers who evaluate anthropogenic impacts. Empirical data will be used to develop metrics of species based on their indicator status of reference vs. managed watersheds. This will allow evaluations of site vegetation relative to vegetation in reference watersheds. The relationships between vegetation and streambank measurements will also be used to

develop ratings of species in terms of their ability to stabilize streambanks. This is a variable that is of great interest to land managers, but there is no data-based measure of species' abilities to stabilize streambanks.

Ideally those new techniques for evaluating the condition of riparian sites will be applicable across large geographical areas. It may be that different monitoring variables, or different potentials for a given variable, are identified for different environmental conditions.

The testing, evaluation, and refinement of riparian monitoring variables is essential in order to identify ecologically important variables that are responsive to anthropogenic activities, and that can be measured in a repeatable manner. Such efforts improve our ability to detect meaningful change in riparian areas, which helps land managers and researchers to evaluate and adapt management activities to maintain the functions of these important riparian ecosystems.

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APPENDICES

APPENDIX A. Modifications to Riparian Protocol

MODIFICATIONS TO WINWARD'S RIPARIAN VEGETATION MONITORING PROTOCOL

Based on data presented in this dissertation, the PIBO-EMP vegetation sampling methods, which were based on Winward (2000), were modified for the 2003 field season. The modified methods are outlined in the "Sampling Protocol for Integrator Reaches, Vegetation Parameters" (Coles-Ritchie et al. 2003). The modifications are briefly outlined here.

Aspects retained from the protocol of Winward (2000):

1. Greenline definition and sampling location;
2. Riparian cross-sections (5) as location to sample; and
3. Sampling area includes 110 m of stream (but is somewhat larger).

Modifications to protocol of Winward (2000):

1. Species cover data are collected in quadrats (0.2 m x 0.5 m) at intervals along the greenline and cross-sections, rather than collecting community type cover data at every step. Data are collected for all species that can be identified in the field, and for all species over 5% cover based on specimens if necessary.
2. Greenline area sampled
    a. Species data are collected in quadrats but not at every step as with community type data. There are 40 quadrats on the greenline (20 on each bank), which correspond to stream channel transects that are 8 to 16 m apart. This association of vegetation quadrats and

stream transects will provide an opportunity to identify relationships between streambank vegetation and bank/channel characteristics.

b. The distance of the greenline sampling area was increased from 110 m to between 160-280 m to match stream sampling area.

3. Cross-section sampling area

a. A fixed distance of 10 m on each side of the stream is sampled, regardless of whether it is riparian or not. Therefore the subjective judgment of where the riparian area ends is no longer left to the technicians.

b. Quadrats are placed every 3 three meters along the 5 cross-sections (on each side of the stream) for a total of 30 cross-section quadrats.

APPENDIX B. Meta-Community Types

META-COMMUNITY TYPES

The following table is a partial list of the meta-community types (14 of the 167) created by grouping community types. Meta-community type names were created by the author, based largely on community type names. Community types include a classification code, after the underscore, which is based on the first author, and a geographic code (WA or OR) in the case where the same author (i.e. Kov) did two classifications. The physiognomic category is based on the dominant species of the community type. For explanation of the species codes used in the community type (and sometimes used in meta-community type) names, see the classifications listed in Table 2.1 (see References for complete citation). Codes for meta-community types include: ab = alluvial bar; hg=hydric graminoid; mf=mesic forb; and mg= mesic graminoid.

Meta-community types	Community Type	Physiognomic group
ALIN/AB	ALIN/AB_KOV-WA	shrub
ALIN/AB	ALIN/CALEL2_Crowe	shrub
ALIN/AB	ALIN-SPDO_Kov-OR	shrub
ALIN/AB	ALIN-SPDO_KOV-WA	shrub
ALIN/CACA	ALIN/CACA_Crowe	shrub

ALIN/CACA	ALIN/CACA_KOV-WA	shrub
ALIN/CALU	ALIN/CALU_Crowe	shrub
ALIN/CASP	ALIN/CASCP2_KOV-WA	shrub
ALIN/CAUT	ALIN/CAUT_Crowe	shrub
ALIN/CAUT	ALIN/CAUT_KOV-WA	shrub
ALIN/COSE	ALIN/CALA3_Crowe	shrub
ALIN/COSE	ALIN/COSE_CRAWFORD	shrub
ALIN/COSE	ALIN/COSE_Padgett	shrub
ALIN/COSE	ALIN/RIHU_YOUNGBLOOD	shrub
ALIN/HG	ALIN/ATFI_Crowe	shrub
ALIN/HG	ALIN/ATFI_KOV-WA	shrub
ALIN/HG	ALIN/CAAM_Crowe	shrub
ALIN/HG	ALIN/EQAR_Crowe	shrub
ALIN/HG	ALIN/EQUIS_KOV-WA	shrub
ALIN/HG	ALIN/GLEL_Crowe	shrub
ALIN/HG	ALIN/GLEL_KOV-WA	shrub
ALIN/HG	ALIN/GYDR_KOV-WA	shrub
ALIN/HG	ALIN/LYAM_KOV-WA	shrub
ALIN/HG	ALIN-COST/MF_Crowe	shrub
ALIN/HG	ALIN-COST-SYAL_KOV-WA	shrub
ALIN/HG	ALIN-RIBES/MF_Crowe	shrub
ALIN/MF	ALIN/BENCH_Manning	shrub

ALIN/MF	ALIN/CADE_Crowe	shrub
ALIN/MF	ALIN/GYDR_Crowe	shrub
ALIN/MF	ALIN/MF_KOV-WA	shrub
ALIN/MF	ALIN/MF_Manning	shrub
ALIN/MF	ALIN/SPRINGS_Kov-OR	shrub
ALIN/MF	ALIN_Kov-OR	shrub
ALIN/MG	ALIN/EQAR_Padgett	shrub
ALIN/MG	ALIN/HELA_Crowe	shrub
ALIN/MG	ALIN/MF_Padgett	shrub
ALIN/MG	ALIN/MG_Manning	shrub
ALIN/MG	ALIN/MG_Padgett	shrub
ALIN/MG	ALIN/PHAR_KOV-WA	shrub
ALIN/MG	ALIN/POPR_Crowe	shrub
ALIN/MG	ALIN_Hansen	shrub
ALIN/SCMI	ALIN/CAAQ_Crowe	shrub
ALIN/SCMI	ALIN/SCMI_Crowe	shrub
ALIN/SCMI	ALIN/SCMI_KOV-WA	shrub
ALIN/SY	ALIN-SYAL_Crowe	shrub
ALIN/SY	ALIN-SYAL_Kov-OR	shrub
ALIN/SY	ALIN-SYAL_KOV-WA	shrub
CAAQ	CAAQ_Crowe	herb
CAAQ	CAAQ_Kov-OR	herb

CAAQ	CAAQ_KOV-WA	herb
CAAQ	CAAQ_Manning	herb
CAAQ	CAAQ_Padgett	herb
CAAQ	CAAQ_YOUNGBLOOD	herb
CAAQ	CAAQ-CAAQ_Hansen	herb
CAAQ	CAAQ-DECE_Hansen	herb
CAAQ	CASI3_Kov-OR	herb
PICEA/CACA	ABLA/CACA-LICA_Hansen	conif
PICEA/CACA	CONIF/DECE_Padgett	conif
PICEA/CACA	CONIF/ELGL_Padgett	conif
PICEA/CACA	PICEA/CACA_Hansen	conif
PICEA/CACA	PICEA/GATR_YOUNGBLOOD	conif
SAGE/MG	SAGE/DECE_Padgett	willow
SAGE/MG	SAGE/MF_Padgett	willow
SAGE/MG	SAGE/MG_Manning	willow
SAGE/MG	SAGE/MG_Padgett	willow
SAGE/MG	SAGE/POPR_YOUNGBLOOD	willow

APPENDIX C. Permission Letters

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Education

Candidate for PhD in vegetation ecology, Utah State University, Logan, UT.

M.S. in environmental studies, Bard College, Annandale-on-Hudson, NY, 1996
Thesis on the harvest and marketing of tagua seeds in Ecuador.

B.A. in comparative literature, University of Massachusetts, Amherst, Minor in resource economics, 1991.

Work experience

Riparian vegetation ecologist, USDA Forest Service, Effectiveness Monitoring Program for Streams and Riparian Areas within the Upper Columbia River Basin, Logan, UT. 2004 to present.

Graduate research assistant, USDA Forest Service, Fish and Aquatic Ecology Unit, Logan, UT. Collected, managed, and summarized riparian vegetation data. Trained technicians in riparian sampling methods and plant identification. 1999-2004

Environmental policy analyst, Border Ecology Project, Bisbee, AZ. Research and policy development to address environmental contamination in the US-Mexico border. 1995-1998.

Researcher, SUBIR Project of CARE International, Quito, Ecuador. Evaluated the sustainability of harvesting tropical forest fibers for production of baskets and related products. 1994.

Researcher, Fundacion IDEA, Quito, Ecuador. Performed research on the tagua industry, from the harvest of tagua seeds from tropical forests to the export of tagua buttons. 1993.

Teaching experience

Riparian Ecology and Management (for undergraduate and graduate students), Utah State University, College of Natural Resources, Spring 2002 and Spring 2005.

English literature, Cochise College, Douglas, AZ 1997.

Publications

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