Evaluation of Bureau of Land Management Protocols for Monitoring Stream Condition

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(ABSTRACT)

The goal of the Aquatic Indicators of Land Condition (AILC) project is to develop analytical tools that integrate land condition information with stream condition for improved watershed management within the United States Bureau of Land Management (BLM). Based on the goal of the AILC, two objectives for this study were: to determine the effect of four GIS-derived distance measurements on potential relationships between common BLM landscape stressors (mining and grazing) and changes in benthic macroinvertebrate community structure; and to assess the effectiveness of individual questions on a commonly-used Bureau-wide qualitative stream assessment protocol, the proper functioning condition (PFC) assessment.

The four GIS distance measurements assessed for biotic relevance included: straight-line distance, slope distance, flow length, and travel time. No significant relationships were found between the measured distance to stressor and macroinvertebrate community structure. However, the hydrological relevance of flow length and travel time are logically superior to straight-line and slope distance and should be researched further.

Several individual questions in the PFC assessment had statistically significant relationships with the final reach ratings and with field-measured characteristics. Two of the checklist questions were significantly related to the number of cow droppings. This may indicate a useful and efficient measure of stream degradation due to grazing. The handling and use of the PFC assessment within the BLM needs further documentation and examination for scientific viability, and the addition of quantitative measurements to the PFC in determining restoration potential would be desirable.

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Chapter 1: Introduction

The Bureau of Land Management (BLM), an agency of the U.S. Department of the Interior, manages large areas of land for multiple uses. The BLM manages 264 million acres in 34 states (BLM 2003), particularly in the western U.S. and Alaska for uses such as livestock grazing, mineral and oil development, and recreation.

The Clean Water Act (CWA) of 1972 mandates that the United States and its agencies "restore and maintain the chemical, physical, and biological integrity of the Nation's waters" (Angermeier and Karr 1994, Chaney et al. 1993, Karr and Kerans 1991). In the process of fulfilling this requirement, federal land management agencies such as the BLM are currently assessing and monitoring the condition of biotic communities and physical processes in waterways. The composition of biotic communities in waterways reflect the physical conditions of those waterways and their watersheds.

In response to the need for water resource monitoring, the BLM's Aquatic Indicators of Landscape Condition (AILC) project aims to develop analytical tools to integrate information on indicators of land use and condition with stream biotic information. This information will be critical to aid in evaluation of the effects of BLM land use activities on stream biotic communities at a regional scope. The analytical tools would consist of a repeatable, objective, and scientifically sound monitoring strategies for BLM jurisdiction streams. This monitoring protocol can, in turn, aid science-based land management planning for the BLM at local field office, district, regional, and national scales. The evaluation of the use of biotic metrics as quantitative measures of stream health is presently a central activity of the AILC.

1.1 Monitoring the Biotic Integrity of Water Resources

The term "biotic integrity" as used in the CWA consists of more than biological diversity- it encompasses entire biotic communities and the physical processes that sustain their abundance, presence, and health (Angermeier and Karr 1994, Karr and Kerans 1991). It follows that the concept of biotic integrity views streams and communities as part of a larger system including the terrestrial landscape that influences them. Biotic integrity is "a quantitative expression of a number of known relationships between human disturbance and the characteristics of the resident biota"(Karr and Kerans 1991). Therefore, measuring biotic integrity requires the establishment of reference conditions and ongoing monitoring of stream and watershed conditions (Angermeier and Karr 1994, Chessman 1999, Harrelson et al. 1994, Lundquist and Beatty 1999, Platts et al. 1987). Reference conditions are physical and biotic conditions of stream reaches that have had little to no human influence. The physical and biotic conditions of other, more human-influenced, streams can be compared with the conditions of reference streams of a similar location to assess the effects of human activity.

Monitoring is a system of observation and planning designed to track changes in water resource conditions over time, particularly those changes influenced by management (Platts et al. 1987). Variables monitored and measured in stream systems may include physicochemical parameters (Meador and Goldstein 2003, Townsend et al. 1997), stream and floodplain morphology (Leopold 1994, Muhar and Jungwirth 1998), riparian soils, riparian vegetation (Townsend et al. 1997, Wallace et al. 1997), catchment land use and land cover (Bryce et al. 1999, Lammert and Allan 1999, Richards et al. 1997, Weigel et al. 2000), historical information on stream conditions and land use (Harding et al. 1998), and biotic community composition (Delong and Brusven 1998, Plafkin et al. 1989, Platts et al. 1987, Roth et al. 1996). These measurements of stream resources can be taken using direct field observations or remote sensing. Remote sensing utilizes aerial photos and digital spatial data to make measurements.

Water resource physical and biotic conditions can vary both temporally and spatially (Poff and Ward 1990). Effective monitoring should therefore encompass a range of spatial and temporal scales (Karr and Kerans 1991). Landscape influences on aquatic systems may occur at multiple spatial scales and through multiple processes (Allan and Johnson 1997). Water resource data can be organized at the watershed/ catchment level, and the riparian/reach level. Watershed-level data incorporate all information about an entire drainage area for a particular stream outlet point. Riparian-level data include information about a particular stream reach and its surrounding zone of influence. Stream reach-level data can also consist of the particular data collected at a point within the reach, including detailed information about biotic communities, stream physicochemistry, and stream morphology. Successful monitoring of stream conditions requires integration of both levels of water resource data.

1.2 Development of BLM stream monitoring

In order to track the condition of streams under the BLM's jurisdiction, a monitoring protocol must be in place (Platts et al. 1987). The BLM has acquired stream physical condition data for areas of its jurisdiction using the proper functioning condition assessment protocol.

The term proper functioning condition (PFC) refers to the functionality of a stream reach's physical processes. Physical processes determine the hydrology, morphology, and riparian vegetation of a particular stream segment or reach. PFC assessments are based on the assumption that if physical processes are correctly functioning, a riparian area will be consistently resilient after flooding (Prichard et al. 1998). Resiliency refers to a condition of dynamic equilibrium, where the processes of aggradation (deposition) and degradation (erosion) are offset by the channel's physical properties, such as riparian rooting, channel slope, and sediment size (DeBano and Schmidt 1989). According to the PFC assessment manual (Prichard et al. 1998), if an area were not functioning properly, an imbalance of aggradation and degradation and subsequent major channel alterations during flood events would be expected.

The qualitative (judgment-based rather than measurement-based) PFC process is widely used by the BLM. PFC is simple to implement, requires little to no quantitative (measured) data or sample collection, and is designed for use across diverse BLM lands (Prichard et al. 1998). PFC is a ranking procedure designed to help land managers rank restoration potential of stream reaches. The procedure is to be completed by a team of experts, who visit the site in question and fill out a standard lotic checklist to determine the PFC ranking of the area (Appendix A). To complete a PFC assessment, after answering the field sheet questions, observers place stream reaches in one of three condition categories based on a qualitative assessment that focuses on the riparian area.

Questions about hydrologic characteristics, riparian vegetation, and erosional characteristics of the site are addressed on the standard lotic checklist as yes/no answers (Prichard et al. 1998). To determine the final ranking, the team is to consider their answers to the check sheet questions and then determine an overall reach rating based on their expert judgment. A rating of PFC or Proper Functioning Condition indicates that the area is resilient to flooding. A rating of Functioning- At Risk (FAR) means that the area has some important physical processes in place, but there is a high likelihood that a flood would severely damage the area. Damage may consist of streambank damage, drastic channel relocation, and accelerated erosion and deposition rates. A direction of change (upward, downward, or trend not apparent) is associated with the FAR rating, referring to the reach's movement toward or away from PFC. A rating of Not Functioning (NF) indicates that none of the physical processes assessed are functioning for the reach and flood damage is usually already evident or damage is imminent with the next flood event (Prichard et al. 1998).

For the remainder of this document, the entire PFC protocol or process will be referred to as the "PFC assessment". The questions on a checklist used during the PFC assessment will be referred to as "checklist questions" and the checklist itself will be called the "standard lotic checklist" or "checklist", the final rating of a reach (PFC, FAR, or NF) will be referred to as the "reach rating", and a final rating of PFC for a specific reach will be referred to as a "reach rating of PFC" to avoid confusion between the overall name of the assessment and the ranking of a particular reach.

The PFC protocol is intended for use as an initial assessment of an area in order to rank its restoration potential, not for use as a long- term monitoring tool or a watershed analysis tool (Prichard et al. 1998, Pyke et al. 2002). Restoration is defined as the "reestablishment of the structure and function of an ecosystem" (Williams et al. 1997). It is unknown to what extent the PFC is used to rank restoration potential with follow-up including restoration activities. Additionally, the current state of knowledge about the PFC process and its scientific effectiveness is limited. Researchers are aware that the PFC relies solely on professional opinion that is categorized as scientific procedure (Stringham 2004).

Due to the widespread acceptance of the PFC process and its substitution for quantitative science, although not the original intent of the authors (Prichard et al. 1998) it would be prudent to scientifically assess its viability. The procedure's widespread use and acceptance in the BLM justifies its further scientific study.

In order to meet the AILC project's objective of examining relationships between stream conditions and land use, the land uses within BLM lands must be identified and studied in addition to the PFC and instream biotic conditions. The BLM manages land for many different uses, including recreation, livestock grazing, mining, and oil and gas development. Management activities related to these land uses may include construction of dams or water diversions, fencing, extermination of invasive and noxious weed species, construction of roads, construction of oil and gas well heads, and construction of mines and mine infrastructure. This study will focus on the impacts of grazing and mining on macroinvertebrate communities.

To examine possible landscape factors affecting biotic integrity, a measure of biotic integrity must be used. Biotic integrity can be quantified through the use of several types of indices and metrics, which are discussed further in the literature review. These indices can be calculated from collection of benthic macroinvertebrates or fish and quantifying the community composition. The quantification method is determined by the index used. Benthic macroinvertebrate communities are commonly-used indicators of watershed health. Collection of macroinvertebrates is simple to implement, requires little specialized equipment, and specimens can be kept in a laboratory for further analysis (Plafkin et al. 1989, Weber 1973). Macroinvertebrates are also plentiful in most streams and the wide range of macroinvertebrate species can represent a gradient of pollution tolerances (Rosenberg and Resh 1993).

The BLM manages 264 million acres of surface land, and is also responsible for overseeing the mineral rights to 700 million acres of land in the U.S (BLM 2003). Mining and mineral claims, along with oil and gas development, represented 92% of the revenue generated by the BLM in 2003 (BLM 2003). Mining impacts on streams vary according to the type of mining or mineral extraction, but can include direct pollution of heavy metals, increased turbidity, and decreased pH (Grigorovich and Angermeier 2004).

The BLM also manages huge areas of range (livestock grazing areas). For example, 18,186 grazing permits and leases were issued in 2002 (BLM 2003). Streams are frequently the only source of drinking water for livestock in rangelands, which can lead to overuse and degradation of riparian areas (Platts 1991). The impact of rangeland grazing on riparian areas is of increasing concern to many stakeholders (Clary and

Leininger 2000). The management of rangelands may affect stream quality both directly and indirectly through alteration of water chemistry, soil properties, and vegetation cover of riparian areas. Stream biotic and physical indicators assessed in this study may reflect grazing and mining practices and help inform future management decisions.

1.3 BLM data resources

The BLM has access to many resources to aid in reaching the AILC project's goals of developing a sound stream monitoring system and providing objective information to assist in making sound land management strategies at multiple levels. The PFC protocol, macroinvertebrate data, land use information, spatial data, and Geographic Information Systems (GIS) are all easily accessible to the BLM and will be integral in reaching the goals of the AILC project.

The PFC process represents an excellent starting point for developing a quantitative, objective, and repeatable monitoring protocol within BLM because of its wide acceptance and implementation within the agency. Therefore, the strength of correlation between PFC and landscape factors and instream conditions is an important topic of study, and relevant to the goals of the AILC.

In addition, the BLM has access to entomologists who can identify and compile macroinvertebrate data at the National Aquatic Monitoring Center (NAMC) in Logan, Utah. The NAMC also trains field collection crews to collect macroinvertebrates with a standardized sampling protocol.

The BLM also maintains several databases of management activities. The Rangeland Assessment System (RAS) is one of these databases, used to catalog grazing activity on all BLM lands (BLM 2003). Data on grazing allotment boundaries, vegetation cover, land ownership, climate, mining development, and dam locations are also available within the BLM or readily available through other federal agencies. Grazing and mining location information are the most readily available landscape stressor data at this time.

Furthermore, watershed and riparian area characteristics, including elevation, stream order, road density, watershed area, and land cover, can all be derived remotely from spatial datasets in a GIS. Using spatial datasets to derive these variables, rather than field measurements, will reduce the amount of resources needed to generate information for the AILC project. The correlation of PFC sampling points and macroinvertebrate assemblages with these spatial variables will involve large amounts of spatial data at varying spatial and temporal scales. GIS provides a system of data management, analysis, and display of spatial data that will be of critical importance to the goals of the AILC. This study will provide selected GIS analysis and data management for the AILC project.

1.4 Study objectives

Based on the goals of the AILC project and the resources available to the BLM, the proposed research will include two related studies: *a stream distance study* and a *PFC checklist study*.

Stream distance study

Potential relationships between landscape stressors and macroinvertebrate community structure are of central importance to the goals of the AILC. If macroinvertebrate community structure represents a monitoring structure that fulfills the goals of the AILC, then the relationship between macroinvertebrate communities and landscape activities will help guide management decisions. However, the location of a stressor relative to a macroinvertebrate sampling site may affect the observed relationship between the stressor and the biota at the sampling site.

The distance between stressor and sample point can be calculated in a variety of different ways within a GIS. Which distance measure will result in the strongest relationship between stressor and biotic sampling point? Does the intensity of the stressor, i.e. the grazing pressure at a site, affect any relationship, quantified by distance, between stressor and sample?

These questions were addressed using a GIS analysis of four distance measurements for two stressors in Wyoming and Utah. The distance measures tested included: straight-line (crow's flight) distance, slope distance, flow length, and travel time.

PFC checklist study

The study of the PFC assessment, its applicability to a wide range of western BLM managed lands, and its overall viability will also be important to the goals of the AILC in implementing an effective stream monitoring system. Study of the PFC assessment will address two questions, outlined below. The analysis of the PFC process will help to strengthen responses and the scientific relevance of the qualitative method. If no relationships between checklist questions and final reach ratings, or between checklist questions and field crew observations are found, then the use of the PFC process in the BLM should be examined further.

Question A

Within the PFC assessment, do some questions contribute more than others to the reach rating? Are some checklist questions redundant in determining the reach rating? The answer to these questions may illustrate which parts of the PFC procedure, which does not use any formal ranking or weighting measures, can be overlooked or simply not considered by observers when assigning a final PFC ranking based on collective responses to the 17 individual PFC questions.

Question B

Do standard lotic checklist responses correspond to measured field conditions? To assess this question, measurements and observations recorded by a summer data collection field crew (Appendix B) at sites with reach ratings previously assigned were assumed to represent actual field conditions. Not all checklist questions have logical corresponding field data, so a subset of the checklist questions was studied. Individual PFC statement responses with stronger spatial correlations to instream characteristics identify areas of the standard lotic checklist that effectively characterize measured field conditions, while weak correlations of checklist responses and instream characteristics may identify questions less effective in characterizing measured field conditions or a need for further study. The response to each checklist question within the subset was compared with the measurement (continuous) or observation (categorical) of the summer field data.

Chapter 2: Literature Review

Relevant literature in the study of riparian areas, macroinvertebrate sampling, land use effects on riparian areas, and GIS analysis will be reviewed.

2.1 Riparian areas and their monitoring

The PFC assessment relies heavily on observable, moment-in-time riparian properties to assess the "functioning condition" of a stream reach. The exact area that determines a riparian zone, or area of influence around a stream, is subject to interpretation. Chaney and others (1993) state that riparian areas are places next to bodies of water where the vegetation is influenced by or dependent on the water body. Riparian areas, particularly in the semiarid west, represent areas of rich resources for humans, livestock, wildlife, and aquatic populations.

Riparian areas are often subject to continuous fluctuations in water levels, sediment loads, and biotic communities and are therefore dynamic and complex systems (Naiman et al. 2000). Snapshot-type monitoring of riparian areas is not sufficient to encompass the wide range of fluctuations riparian areas undergo. Restoration activities should take into account the highly dynamic nature of riparian areas (Ebersole and Liss 1997).

The term riparian area is ambiguous, although many researchers use a 100-m straight-line buffer on either side of a stream as the standard area of riparian influence (Pess et al. 2002, Richards et al. 1996). The problem with the use of straight-line buffers around a stream is that they are determined by humans and not by the landscape. The areas surrounding streams could differ greatly in topography or soil type, which could affect the distribution of riparian vegetation, but the use of straight-line buffers does not take these factors into account. However, the use of hydrologic travel time, or the time it takes for precipitation to enter into a stream via the overland flow network, can be used to delineate a riparian area of influence as well (Heatwole and Burcher 2003). Each point in a watershed has a specific time to channel output associated with it for a given amount of rainfall. These times could be generalized into zones (e.g. 30 minutes or less, 90 minutes or less) and an appropriate zone chosen to represent the riparian area. An appropriate travel time zone would closely follow the riparian and upland vegetation ecotone; however, this research has not yet been done.

The determination of travel time may allow for better partitioning of the effects of specific land uses on stream quality (Heatwole and Burcher 2003) at the landscape scale than the use of riparian buffers. One disadvantage of this method compared to the use of straight-line buffers is the time-consuming nature of calculations required for one riparian area. However, GIS offers an excellent platform for calculation of these areas, and script programs offer the possibility of full or partial automation of these processes.

In addition to the consideration of the area that makes up a riparian zone, the method of communicating the physical and biotic properties of the riparian area is also of importance. Indices are often used to categorize riparian monitoring data, and can be calculated from measurements or qualitative observations. In North Carolina, scientists conducted a pilot study in two sets of paired watersheds comparing the predictive power of several qualitative indices of watershed health against an index of biotic integrity (IBI) for fish (McQuaid and Norfleet 1999). Qualitative indices like the PFC are based on several weight-of-evidence (McMahon et al. 2001) rather than measured observations. The study found that all qualitative indices had a low correlation with IBI and suggested that qualitative measures of watershed health have little utility and need further examination. Although the PFC process is qualitative, the manual claims that the questions are based on quantitative science (Prichard et al. 1998), and should therefore have the potential for correlations with instream biotic condition.

The PFC manual describes for each of the 17 checklist questions supporting science for the development of the question and any quantitative methods of evaluating the question (Prichard et al. 1998). The reasoning behind the wording of most questions is based on interpretation and assumption of conclusions of other studies. However, the scientific relevance of a large portion of the questions is not clearly explained (questions 1, 2, 4, 7, 12, 14, 15, 16, 17, Appendix A) and assumes reader knowledge about literature on the subject and provides no explicit scientific basis for the questions (Prichard et al. 1998). Several questions do have a clearly outlined scientific basis (questions 3, 5, 6, 8, 9, 10, 11, 13, Appendix A) with measurable characteristics, including Manning's channel roughness, Rosgen's (1996) stream channel classification, empirical studies of channel response to streamflow changes, Myers (1989) wetland plant classification, and Platts and others' (1987) method for determining streambank stability (Prichard et al. 1998).

However, the vague scientific background of many of the checklist questions further warrants the scientific study of the PFC process.

2.2 The use of macroinvertebrates in stream monitoring

The relationship between PFC ratings and instream biotic conditions, as well as riparian and watershed variables, is unknown. Studies of these correlations are integral to the aforementioned goals of the AILC. Instream biotic conditions will be assessed for this study using benthic macroinvertebrates. There are several advantages of using macroinvertebrates over other biota, such as fish, including ease of collection, relatively stationary nature, and the need for little specialized collection equipment (Plafkin et al. 1989, Weber 1973). There are also direct relationships between macroinvertebrate abundance and fish production (Waters 1995). However, processing of macroinvertebrate samples does require specialized knowledge and considerable time. Furthermore, macroinvertebrate abundance fluctuates seasonally (Rosenberg and Resh 1993, Weber 1973). The plethora of available indices involving macroinvertebrates also suggests that finding an appropriate measure of biotic integrity is difficult and variable (Rosenberg and Resh 1993).

Macroinvertebrate community compositions can reflect changes over time. Because of their stationary nature, many researchers believe that macroinvertebrates tend to reflect changes in local or riparian conditions more than watershed-wide conditions (Lammert and Allan 1999, Plafkin et al. 1989, Rosenberg and Resh 1993), although some authors have disagreed (Weigel et al. 2000). Changes in macroinvertebrate communities may reflect changes in substrate type, depth of stream, and velocity of streams (Weber 1973), which may be directly or indirectly caused by natural variation or anthropogenic influences, on local or watershed-wide scales. The partitioning of these wide ranges of influences on macroinvertebrate communities will be discussed further.

There are several types of indices that can be generated using macroinvertebrates. Diversity and biotic indices (Johnson et al. 1993) offer two distinct ways to compare macroinvertebrate community structure with environmental factors. Useful indices would show large differences between reference and disturbed sites at the onset of disturbance, and diminishing difference as the disturbed site recovers over time (Stone and Wallace 1998).

Diversity indices are based on the total number of individuals and total number of taxa present (Norris and Georges 1993). Species abundance and species richness are taken into account in the formulation of diversity indices. Diversity is assumed to be low in environmentally stressed areas (Norris and Georges 1993). Common diversity indices include Simpson’s index (Simpson 1949) and species per 1,000 individual organisms.

Biotic indices, based on known pollution tolerances, assign a ranking for each type of organism observed or captured within a given area. The index is composed of these individual species rankings, and sometimes will allow for physical and seasonal fluctuations as well. Biotic indices are developed using assumptions about pollution type and geography (Johnson et al. 1993).

The Hilsenhoff (1987) biotic index is one widely used biotic index. This index ranks species by their organic pollution tolerance from 1 – 10 and also takes stream current, temperature, and seasonal fluctuations into account (Hilsenhoff 1987). Because this index is suited mainly for organic pollution impacts, it is suited for response to grazing impacts (Grigorovich and Angermeier 2004). Another common biotic index is the count of Ephemeroptera, Trichoptera, and Plecoptera (EPT) taxa. EPT taxa are usually indicative of high-quality sites (Weigel et al. 2000), so the relative abundances of these taxa are used to indicate the quality of a site. This index is simple to calculate, and is suited for responding to impacts from development and direct, inorganic pollution such as impacts from mining (Grigorovich and Angermeier 2004). The ratio of EPT to Chironomidae taxa is thought to also include response to organic pollution impacts, such as those from grazing (Grigorovich and Angermeier 2004). Many other biotic indices exist as well (Rosenberg and Resh 1993) but these represent a small fraction of simple, commonly-used, and available metrics for this study.

The relationship between macroinvertebrate community structure and riparian cover has been widely studied, and is important when assessing land use influences on stream condition. Riparian vegetation also represents a large portion of the variables assessed in the PFC process (Prichard et al. 1998) and studying vegetation- related variables may help illustrate any links between reach rating, land use, and instream biotic conditions.

The presence of riparian plants provides stability for streambanks and stream shading. Percent vegetation cover has been related to land use (Townsend et al. 1997), particularly grazing, development, and forestry.

Riparian vegetation provides organic matter in the form of leaves and woody debris, which are important as food sources for specific groups of macroinvertebrates. Riparian vegetation represents a significant portion of organic inputs to stream systems (Kauffman and Kreuger 1984). Vegetation also provides shading and filtering properties for streams (McEldowney et al. 2002). It follows that a decrease in riparian vegetation would result in a loss of suitable habitat (increased temperature, increased turbidity) and loss of food source for macroinvertebrates, which would be evident through changes in the biotic community, for example, increased numbers of shredders. Scrimgeour and Kendall (2003) found that the total biomass of invertebrates in grazed streams was significantly affected by grazing practice.

Correlation between functional feeding groups and riparian vegetation is apparent in some cases; however, this division of macroinvertebrate communities is not always effective or necessary (Cummins 1974, Norris and Georges 1993). Macroinvertebrate functional feeding groups can be indicators of land use and its effect on riparian vegetation (Townsend et al. 1997). Shredders represent one functional feeding group with direct connection to the amount and type of riparian vegetation. Shredders ingest the largest particles of organic matter, mainly leaves, taking in about 40% of the matter for internal processes and excreting the remaining 60%. Other feeding groups, such as the collectors, feed on these particles (Cummins 1974). Reed and others (1994) found that shredder biomass was higher in forested stream sites than in non-forested stream sites. Similarly, Stout and others (1993) found that the response of shredders to disturbance paralleled the response of the surrounding vegetation to disturbance.

Classifying the sources of variation in macroinvertebrate communities is important to the goals of the AILC. Results of correlations between land use and physical characteristics with macroinvertebrate biotic indices can be interpreted only by understanding which community characteristics are a result of natural variation and which are a result of land management practices and anthropogenic influences. Classification of natural variance will be especially important due to the large regional scope of the AILC project. There is a large body of literature on the partitioning of variance in stream biota.

Poff and Ward (1990) identify several scales, or levels at which variation can be classified. Spatial (regional to local), temporal (seasonal to geological), and ecological (physiology and behavior to species migration) differences are all considerations in the partitioning of variation in biotic communities (Poff and Ward 1990). Ecological scale for this study will focus on macroinvertebrate assemblages, rather than individuals or species migrations.

Classification methods for partitioning natural variance in benthic macroinvertebrates are typically categorized using a large spatial scale (encompassing large regions) or a small spatial scale (focusing on local variation). Large-scale classifications use ecoregions, basins, or geology to account for natural variation in biotic communities. There are several systems of ecoregionalization currently used, including Major Land Resource Areas, the National Hierarchy of Ecological Units, and Level III Ecosystems (McMahon et al. 2001). Ecoregions are large stratifications based on some combination of any of the following factors: geology, land use, land cover, climate, vegetation, and physiography (Omernik 1987). Hughes and others (1993) found that ecoregions could not account for certain fauna and could not successfully predict fish abundance. Ecoregions also have limited success in regionalizing biotic communities and can be useful as a rough stratification framework for sampling design (Hawkins et al. 2000, Omernik and Bailey 1997) but not as a sole source for stratifying variation. Basins are often difficult to use for classification, particularly in some geological areas (Omernik and Bailey 1997).

Angermeier and others (2000) compared the effects of ecoregions and basins for on variation in fish community composition. The authors found that a combination of ecoregions and basins was more desirable than the use of just one of the two systems. Van Sickle and Hughes (2000) studied the classification strengths of ecoregions and large basins among other techniques. The authors concluded that large-scale geographic partitioning by ecoregions or basin could account for only small amounts of natural variation.

Some authors propose using geology alone as a large-scale stratification (Allan and Johnson 1997). Harrellson and others (1994) also recommend classifying reference sites based on underlying geomorphology. However, this is not likely to work as a stand-alone large-scale classification for all geographic areas, as a study by Delong and Brusven (1998) did not find that geologic patterns successfully partitioned macroinvertebrate variation. This may be because the relative influence of geology on stream biota can vary by geographic area.

Small-scale stratification considers smaller, more site-specific factors in partitioning natural variation. Local characteristics are thought to be more effective predictors of natural variation in stream macroinvertebrates (Hawkins et al. 2000). Chessman (1999) proposes a method of classification that predicts species abundance directly from a stream's departure from local reference conditions: latitude, longitude, temperature, elevation, and stream size. Stream substratum, stream flow rate, and temperature have also been suggested as local stratification methods (Poff and Ward 1990).

Hawkins and Vinson (2000) state that due to the continuous nature of environmental variation in streams, a classification system using small-scale environmental gradients will be most effective in partitioning variation. The river continuum concept (Vannote et al. 1980) also supports this theory. The river continuum concept states that as a stream's physical properties change from headwaters (the uppermost part of a channel with flow significant enough to sustain macroinvertebrates) to outlet, stream biotic properties should also change. This should result in a change in the relative abundance of functional feeding groups. Shredders should dominate upper reaches of the stream, where organic matter inputs are high, and the biotic community composition should gradually shift towards collectors where the stream widens and organic matter inputs are less important (Delong and Brusven 1998, Vannote et al. 1980). In areas where the community abundances do not change longitudinally, this may indicate a departure from reference conditions (Delong and Brusven 1998). Quantifying these continua may prove more challenging than using large-scale stratification.

In addition to spatial variability, both benthic macroinvertebrates and stream systems are subject to temporal variability. Over time, stream processes can change naturally and benthic communities would change as well. The temporal regime of stream systems is often ignored (Muhar and Jungwirth 1998) but is important to address. Land use along a stream corridor also changes temporally (Allan and Johnson 1997). Karr and Kerans (1991) state that monitoring of stream areas should occur at several temporal as well as spatial scales. The effects of land use on macroinvertebrate communities and their structure encompasses a large body of literature. The effect of livestock grazing and mining as land uses of interest will be explored in this study.

2.3 Impacts of Mining and Livestock Grazing

Mining and livestock grazing on BLM lands are both economically important and extensive current and historical land uses. Mining claims represent a large proportion of BLM revenues (BLM 2003).

Impacts of mining and mineral extraction depend on the type and size of the mines. Mining activities in the study area of interest included phosphate mining, hard rock mining, and some extraction of construction materials (BLM 2003). However, there are some generalized impacts of mining. Stream pollution by mine tailings (waste) can increase heavy metal concentration in streams (Beasley and Kneale 2002, Marqués et al. 2003). Stream turbidity can also increase with increased erosion caused by surface mining. Coal mining can lower stream pH dramatically in streams with low buffering capacities (Jarvis and Younger 1997). In one study of several mining-related stream characteristics as well as agriculture and physiography related characteristics, the biota responded most strongly to the mining factors (García - Criado et al. 1999). Concentrations of toxic metals, pH, and electrical conductivity are effective measures of mining impacts on stream systems (Marqués et al. 2003).

Chaney and others (1993) state that grazing has more wide-reaching landscape effects in the west than any other land use. Dividing grazing areas into allotments became a common management practice in the 1960's and is the most common organization of range management used in the west today, particularly by the BLM (Platts 1991). Each allotment may represent a different leasing ranch or use of a different grazing system. Each allotment is further divided into any number of pastures, the number of which is determined by the rancher's needs, the number of livestock, and the grazing system in use.

The amount of forage necessary to sustain one mature cow and her suckling calf for thirty days is referred to as an animal unit month (AUM), where the cow and calf are considered one animal unit. Other definitions of animal units also exist for other foraging livestock. AUM's are used to quantify the amount of forage resources needed, used, or leased within an allotment or pasture.

Grazing systems refer to the rotation patterns of livestock within an area. There are many types of grazing systems in use; continuous, rest-rotation, deferred, and high-intensity short-duration systems are the most common systems in use on BLM lands in the intermountain west. Holochek (1983) gives a through description of these patterns, and other, lesser-used patterns. The following is a summary of Holochek's descriptions. Continuous grazing is the name given to any type of grazing using one pasture for consecutive years. Continuous systems can cause some areas of pasture to be severely overused, particularly riparian areas. In deferred rotation grazing, the rancher waits until maturity has been reached on the most important feed grasses to rotate livestock into a pasture. This system has proved advantageous in mountainous areas and areas where plant availability and desirability are different. However, this system can also lead to rapid degradation of riparian areas. Rest-rotation typically uses 4 pastures, where one pasture is rested one full year every 4 years. Productivity of cattle may not be as high as for other systems, however this system favors aesthetics and wildlife. High intensity short duration grazing typically uses a 'wagon wheel' layout of pastures, where cattle are moved between pastures at a high rate (4-6 weeks). Theoretically, high intensity short duration would decrease infiltration of water due to the hoof impacts, and cause even use of the entire allotment. In mountainous rangelands, this system can cause irreversible damage to plants and soils. However, lowlands and areas with uniform plant palatability and longer growing seasons may not be harmed by this system.

Both cattle and sheep graze BLM lands. Cattle grazing will be the focus of this study, as they are more commonly grazed and their effects on stream habitats are heavily researched. Additionally, there was no data or other evidence to suggest that sheep grazing occurs on any of the study area lands. Cattle prefer riparian areas over upland grazing areas, due to the increased water, thermal cover, high quality forage, and gentle topography available in riparian areas (Clary and Leininger 2000, Kauffman and Kreuger 1984). Historically, overuse of rangeland riparian areas has been a problem (Chaney et al. 1993, Platts 1991), leaving the remaining functioning riparian areas with an increased value to humans and wildlife (Kauffman and Kreuger 1984). This increased value of rangeland riparian areas points to a need for informed, science-based management to protect and restore remaining resources.

Grazing has many effects on water quality, both from upland and direct riparian effects. Upland grazing areas, although not as heavily used as riparian areas, nevertheless affect water quality. Upland pressures include trampling, an increase in soil bulk density under intense grazing, the formation of trails which can lead to gully erosion, and removal of vegetation which can lead to a decrease in infiltration (Trimble and Mendel 1995). This study took into account the effects of these upland (entire-watershed) impacts through the flow-length and travel time distances.

There are many direct effects of cattle overuse on riparian communities, including changes in stream morphology, water quality, wildlife habitat, and riparian vegetation (Kauffman and Kreuger 1984). Grazing can cause compaction of soil in the riparian area, which leads to decreased infiltration (Chaney et al. 1993). Erosion was found to increase three to six times in grazed areas versus ungrazed areas in one study (Trimble 1994). Cattle can also cause direct pollution through urine and manure, cause streambank shearing by hooves, and influence the widening and shallowing of streams (Chaney et al. 1993, Clary and Leininger 2000, Waters 1995). Grazing can also lead to a loss of fish habitat including smothering of spawning gravels and removal of riparian cover (Chaney et al. 1993, Kauffman and Kreuger 1984, Waters 1995).

Overgrazing of riparian grasses can lead to the replacement of deep- rooted riparian grasses with shallow-rooted species. This leads to increased bank width, decreased bank stability, and increased turbidity (Clary and Leininger 2000, Manning et al. 1989, Platts 1991). In a study of grazing effects on sediment transport, stem density was found to be one of the most important factors influencing sediment transport to streams, with higher grass stem density reducing sediment transport. Grazing by cattle was found to reduce stem density by 40% (McEldowney et al. 2002).

When examining potential relationships between land use and riparian condition, it is important to acknowledge the dynamic nature of stream systems and the wide range of natural and anthropogenic influences on these systems (Platts 1991). The natural morphology and soil types of riparian areas and streams may render some areas more sensitive to grazing pressure than others (Chaney et al. 1993). In modeling applications, including the use of GIS, the diversity of rangeland types must be addressed. Rangelands should be grouped according to their responses to precipitation and relationships among physical characteristics for the development of valid models (Pierson et al. 2002).

2.4 Use of GIS and Spatial Scale

The conditions and processes of watersheds, riparian areas, and stream reaches undoubtedly affect stream biota (Angermeier and Bailey 1992). A GIS has the ability to store, process, display, and analyze spatial data quickly and efficiently, particularly at large spatial scales. GIS can combine general scientific knowledge in visual form (maps) with specific information in the form of a database (Longley et al. 2001). These capabilities make GIS technology an effective tool in aquatic ecosystem management (Angermeier and Bailey 1992).

Furthermore, GIS are equipped to handle multi-dimensional problems (Longley et al. 2001). The multi-faceted nature of the interaction between land use and stream biota is therefore well suited to GIS analysis. In addition, GIS can handle spatial distributions of biotic characteristics, and can effectively model environmental processes (Longley et al. 2001), which will be vital to this study.

Watershed-scale variables such as catchment area, slope, stream gradient, road density, vegetation cover, and similar relevant variables can be found by using GIS algorithms and nationally available data. These types of variables have been used in a variety of studies of impacts of land use on aquatic communities (Pess et al. 2002, Richards et al. 1996, Roth et al. 1996, Sharma and Hillborn 2001). The ability of GIS to determine variables remotely may save resources by eliminating the need for some field measurements and calculations.

Distance from one geographic area to another is a common calculation that can be made in GIS. Houlahan and Findlay (2004) found that different chemical and sediment stressors created by deforestation significantly affected wetlands at different distances, and the distances were fairly large, up to 4km. Additionally, there is scientific evidence that not all methods of calculating distance are equally relevant (Yuan 2004b).

In the case of this study, the distance from a point stream sample to a point or polygon landscape stressor is of interest. There are several ways that distance can be calculated in GIS, each with an increasing level of required inputs and time.

The simplest way to calculate distance within a GIS is by using a spatial join. A spatial join adds the attributes of one layer to another based on a spatial property, in this case, distance. The distance from a feature in one layer to the closest feature in another layer is recorded, along with the attributes of the closest feature (ESRI 2002).

Another method of calculating distance is by finding the change in elevation from one feature to another. The net elevation change and the straight-line distance are used as inputs to the Pythagorean theorem to find the distance between two points incorporating the net elevation change. This is referred to as slope distance.

A third method of calculating distance to a hydrologic feature, such as a stream sample point, can be calculated using the ArcGIS function flow length, which uses an elevation dataset to calculate a flow path, rather than straight-line distance, based on terrain. The flow length distance is the distance from the stressor to the nearest ephemeral drain and the length of all ephemeral, intermittent, and perennial streams to the outlet point. This function, when set to calculate downstream distance, will find the distance in map units from each cell in the watershed of the sample point to the sample point (ESRI 2002).

GIS can also be used to determine hydrologic travel time based on time of concentration (Heatwole and Burcher 2003). Extensive hydrology equations can be calculated within GIS for display and analysis. This method requires the input of both elevation and land cover data and the calculation of velocity for the watershed. The flow length function is part of these calculations, although they are much more complex and time-consuming than simply calculating flow length.

The specific limitations of the data inputs will be discussed in the review of data and methods. With increasing modeling capability and increased inputs comes increased potential error in the final distance measurement.The calculation of spatial attributes and characteristics of watersheds must fit into the larger framework of the actual landscape being studied, and the processes and extent of interest within that landscape. Therefore, the determination of spatial scale is an important aspect to consider when designing a study using GIS. Caution should be used when approaching problems from a large-scale (regional extent) view. Regional data can be easily managed and processed in a GIS, but study results may be extremely temporally variable (Wiley et al. 1997). Roth and others (1996) did find that watershed (large) scale land use data analyzed using a GIS were effective predictors of biotic indices and conditions of fish communities. A combination approach of large-scale data and small-scale (local extent), temporally repeated (same measurement made multiple years) data would also be appropriate (Wiley et al. 1997) and would reduce variability due to temporal under-sampling. However, Lammert and Allan (1999) found that a single-extent approach was often sufficient to identify relationships between land use and biota, although they agreed with Wiley and others (1997) that the nested approach would be ideal. Allan and Johnson (1997) also found that changing spatial scale could result in different results, particularly with partitioning variation. The authors also recommend using multiple spatial scales when approaching a landscape level problem.

Chapter 3: Data and Methods

Because this study will focus heavily on GIS analysis, several types of spatial and attribute data will be used. Base physical data, macroinvertebrate data, and stressor data will all be important in this study (Table 1). In addition to a review of data types, sources, and accuracy, GIS and statistical methods for both studies will be explained.

3.1 Base data

Base GIS data for this project consisted of elevation, hydrography, land cover, and land features. These data are used to derive other landform and watershed characteristics, such as slope, stream order, flow accumulation, travel time, elevation, percent vegetative cover, surface roughness, and other characteristics.

Elevation grids known as digital elevation models (DEMs) are available with nationwide coverage through an online database. USGS topographic quad sheets partition DEM coverage. The accuracy of DEM grids is categorized based on grid development methods. Level 3 grids, the most accurate, are developed directly from hydrography and hypsography topographic data. Level 3 DEMs are permitted a root mean square error (RMSE; similar to standard deviation), of one-third of the topographic contour interval (USGS 2002). Level 2 grids are the middle accuracy level, and are produced by smoothing or filtering of older grids. Some hydrographic and hypsographic data are used to increase accuracy of these grids. The maxium permitted RMSE for Level 2 grids is one-half the topographic contour interval (USGS 2002). DEM data for the study area selected are all level 3 DEM data (USGS 2002).

High-resolution DEM grids are usually 10 meters or 30 meters [USGS makes some DEM grids with much lower resolution, like 1 km or 100 m]. Although the higher (10-m) resolution would be more desirable, the accuracy level of data in the area should prove more important than resolution level in deriving accurate measurements. In one study, 10-m Level 3 DEMs were found to be superior, particularly to the 30-m Level 1 DEMs, in several categories of analysis. This was due more to the drainage enforcement (enforced consistency between elevation-derived flow paths and hydrology) than the increased resolution in the opinion of the authors (Clarke and Burnett 2003). Ten-meter

Table 1. Data types and sources used in the study.

Data name	Type	Source	Spatial/ Attribute only	Scale/ Resolution	Derived products
			Base Physical Data		
Digital Elevation Model (DEM)	Elevation	seamless.usgs.gov	spatial	10m to 30m	flow paths, travel time, slope, elevation, watershed area, stream gradient
National Hydrography Dataset (NHD)	Hydrography	nhd.usgs.gov	spatial	1:100,000	reach codes, stream location
Gap Analysis Program (GAP)	Land Cover	gap.uidaho.edu	spatial	1:100,000	Land cover, stewardship, roughness coefficients
Digital Raster Graphic (DRG)	Land Features	USGS	Spatial	1:24,000	Validation of DEM-generated flow networks
			Macroinvertebrate Data		
National Aquatic Macroinvertebrate Center (NAMC) database	Macroinvertebrate	NAMC	attribute only		biotic indices, patterns of spatial distribution
Field crew-visited Summer 2003 sites	Macroinvertebrate, stream characteristics	NAMC	attribute only		stream basin properties, percent cover, macroinvertebrate indices
Proper Functioning Condition (PFC) assessments	PFC ratings	BLM field offices	attribute only		PFC question breakdowns
			Stressor Data		
Allotment boundaries	Grazing areas	BLM	spatial		Linking of allotment management atrributes and grazing pressure
Rangeland Administrative System (RAS) Database	Grazing attributes	BLM	attribute only		allotment management attributes
Mine locations	Mine, quarry, and oil and gas head areas and attributes	BLM	Spatial		Mining and oil and gas management attributes, intensity of land use

data were used for this study due to the available coverage in the western United States. All DEM grids in the study area are 1/3 arc second (10-m) coverage (USGS 2002).

Digital elevation models can be used to derive watershed areas, flow networks, watershed travel time, and other hydrologic and terrain variables in a GIS (Jenson and Domingue 1988, Morris and Heerdegen 1988). It is important to note that any digital data source will have inherent error in accordance with its source data and the technique used to generate the data product. DEM elevations may not match field collected elevation data, but when applied at a larger spatial scale, DEM derivatives can useful. There are procedures and algorithms to reduce some of the error in DEM-derived flow networks (Turcotte et al. 2001). A seamless version of DEM data is available through the National Elevation Dataset (NED), which will be useful in study areas spanning more than one topographic quadrangle map.

Hydrography data, including stream reach data and Hydrologic Unit Code (HUC) boundaries are available through the National Hydrography Dataset (NHD), which is also available online with nationwide coverage. Stream reach data for the NHD originates from digital line graphs and EPA reach files (USGS 2000). The NHD contains streams, bodies of water, coastlines, and wetlands in some areas. The NHD contains geocoded reach data reach numbers as well as common stream names. Each stream reach has a unique 14-digit permanent, non-transferable numeric identifier code, with the first 8 digits being the HUC code that the stream falls in, and the last 6 digits being the reach code (USGS 2000). Producers of the NHD intend that these reach codes will be used to identify and link information to stream reaches in a uniform manner. In addition to the reach code, each reach also has a 10-digit common identifier code. The NHD also has flow relationships and stream levels (the reverse of stream order, with the largest rivers having the lowest level number) encoded (USGS 2000).

Land cover for areas in the stream distance study was determined through another nationally available source, the gap analysis program (GAP). GAP data are available online, free of charge, through the USGS. The GAP aims to provide vegetation and stewardship mapping, in addition to species mapping, on a state-by-state basis (Jennings and Scott 1997). Maps are produced at a 1:100,000 scale and are updated at regular intervals, the length of which depends on the state.

Land features for areas in the stream distance study were needed to assess the validity of DEM-generated flow networks. Digital raster graphics (DRG) are rasterized USGS topographic maps, which show land features and topography at a 1:24000 scale. DRG were used to determine flow networks mapped on USGS topographic quads for selected study areas.

3.2 Macroinvertebrate Data

Macroinvertebrate point sampling data and biotic indices for specific sites were available from the National Aquatic Monitoring Center (NAMC) in Logan, Utah. This dataset provided the sampling points used in the stream distance study. A large (15,000 record) database of stream sample points covers many areas in the western U.S. The dataset includes the sampling type (qualitative, quantitative), habitat type (coarse particulate organic matter, riffle, multiple), total individuals counted, total taxa counted, number of unique families, the number of unique genera, the number of unique Ephemeroptera, Trichoptera, and Plecoptera genera. It can also include the calculated Hilsenhoff (1987) biotic index, calculated Shannon's diversity index, latitude and longitude of the sample point, the water body type (stream, lake) and the organization or individual who took the sample (Vinson 2003), it can also include many other calculated metrics.

A field crew, referred to as the "summer field crew", also sampled 79 sites with previously assigned reach ratings in summer of 2003 and 84 sites in the summer of 2004. Macroinvertebrate data were collected for both qualitative and quantitative assessments and the NAMC processed these samples. Additionally, measurements of stream width, percent overhead vegetative stream cover, stream substrate, flow rate, and temperature were taken at each site, along with observations of land use at each stream site (Appendix B). The NAMC catalogs the macroinvertebrate data into a database. The site measurements that correspond with PFC statements were used in this study, which will be discussed further.

PFC assessment sheets were acquired from BLM field offices for the points sampled in summer 2003 and 2004. These ratings will provide the necessary individual PFC responses as well as the overall ratings for each of these points. These PFC sheets were entered into a Microsoft Access database and georeferenced to the summer data points, along with the physical site data from the sampling field sheets.

3.3 Stressor Data

Specific allotment data were available through a large online BLM-managed database, the Rangeland Administration System (RAS). RAS maintains information on permitted rangelands for BLM in several states, including all of the states within the Interior Columbia Basin (ICB). RAS contains information sorted by allotment on permittees, grazing management, and grazing intensities. RAS data can be easily associated with spatially linked grazing allotments using GIS. Digitized allotment boundaries are available for some areas within individual BLM field offices.

Individual BLM field offices catalog mines, quarries, and oil and gas wells. There is normally attribute data associated with these locations. Each field office records varying degrees of detail about these stressors, however, so only location of the mines were used for consistency within the study.

3.4 Study area

An appropriate study area for the stream distance study needed to fulfill several characteristics. The area had to contain numerous sampling sites; contain areas of low land use intensity; encompass a large geographic area in keeping with the desired outcomes of the AILC; contain a large amount of BLM managed land; have grazing and mining development as predominant land uses; and have spatial data readily available. The selection of a study area was approached in three narrowing sets of criteria: available land use data, differences in topographic relief and climate, and sufficient numbers of existing macroinvertebrate sample plots and BLM managed land.

Areas within the AILC major study areas (northwest Wyoming, north central Idaho, south central Utah, and northeast Oregon) that had readily available land use data (grazing and mining) included only Utah and Wyoming. This strictly limited the further selection of a study area. Eight-digit Hydrologic Unit Codes (HUC-8s) from the NHD were used as outlines for selection areas where sufficient sample points and BLM managed lands were present. Two HUC-8 areas from each state were selected. The study areas between the two states differ in climate and are characterized by aquatic ecoregions (Omernik 1987) (Table 2). They also differ in topographic relief, with the Wyoming study areas having lower mean slopes and Utah having higher mean slopes (Table 2).

Table 2. Characteristics of study areas chosen for the stream distance study.

Study Area	HUC-8	Number of Macroinvertebrate Sample points	Aquatic ecoregions	Mean slope of entire HUC-8
Utah 1	14070005	150	Wasatch and Uinta Mountains Colorado Plateaus	23.6%
Utah 2	14070007	18	Colorado Plateaus	20.9%
Wyoming 1	14040101	41	Wyoming Basin Middle Rockies	17.1%
Wyoming 2	14040102	48	Wyoming Basin Middle Rockies	18.8%

*slope determined using a 10-meter Digital Elevation Model using the ArcGIS slope function

The Utah 2 study area does contain a small amount of Arizona land, however, none of the macroinvertebrate sample points fall within this area, so the Arizona portion of Utah 2 was not included in any further calculations (Figure 1). The two Wyoming study areas do differ considerably in area, however, the areas of interest in this study are the watersheds above individual sample points, so the area of the HUC-8s were not a factor (Figure 2).

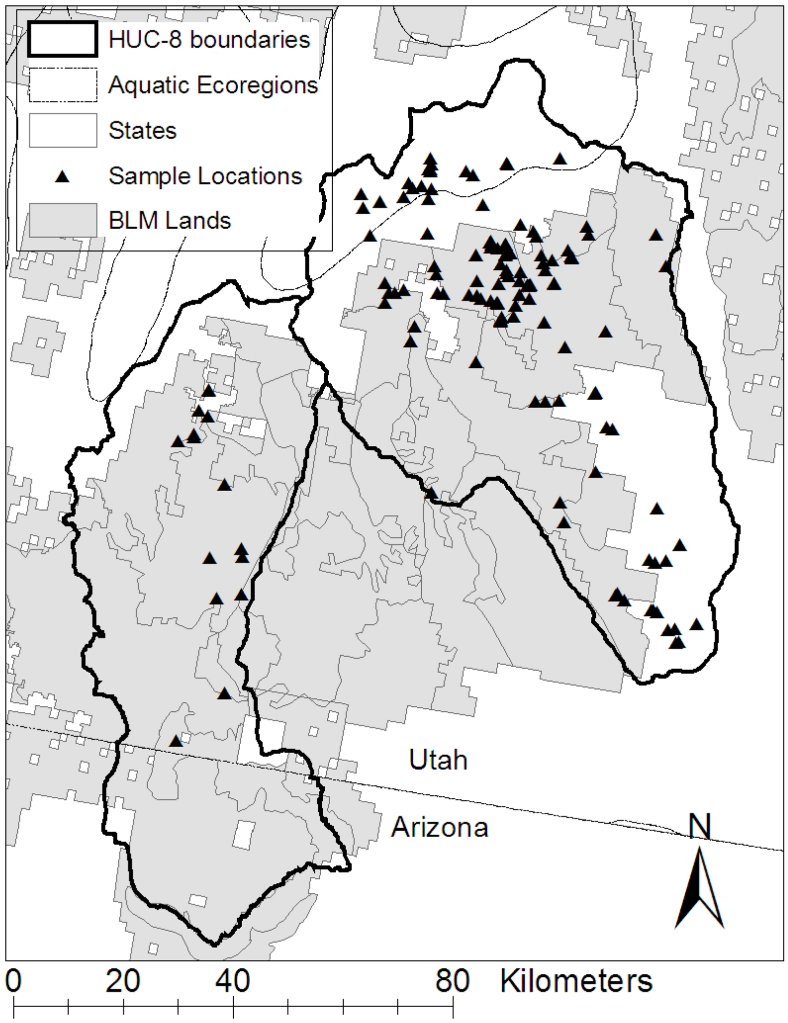

Figure 1. Utah study areas. Note the area within Arizona, however, there are no sample points within Arizona.

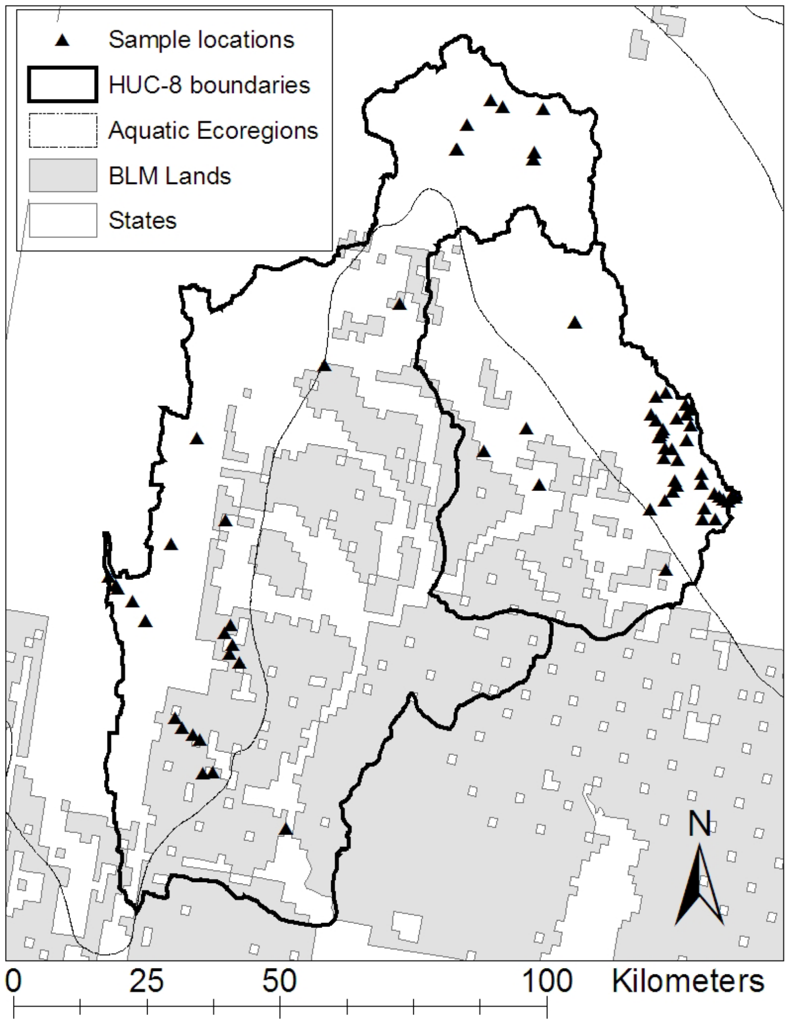

Figure 2. Wyoming study area for stream distance study.

A different study area was used for the PFC checklist study. For objective A, all available reaches from participating field offices with complete standard lotic checklists were used, a total of 150 reaches. Seven BLM field offices contributed complete standard lotic checklists to the study. The location of these sites and the participating BLM districts is shown in Figure 3. Field offices are within districts (Table 3). It should be noted that the Idaho Falls field office contributed information to the project, but uses a different lotic checklist, so their checklists were not included in the analysis. For objective B, only those reaches with both a complete PFC assessment and a summer field crew physical site data collection could be included. This totaled 119 reaches.

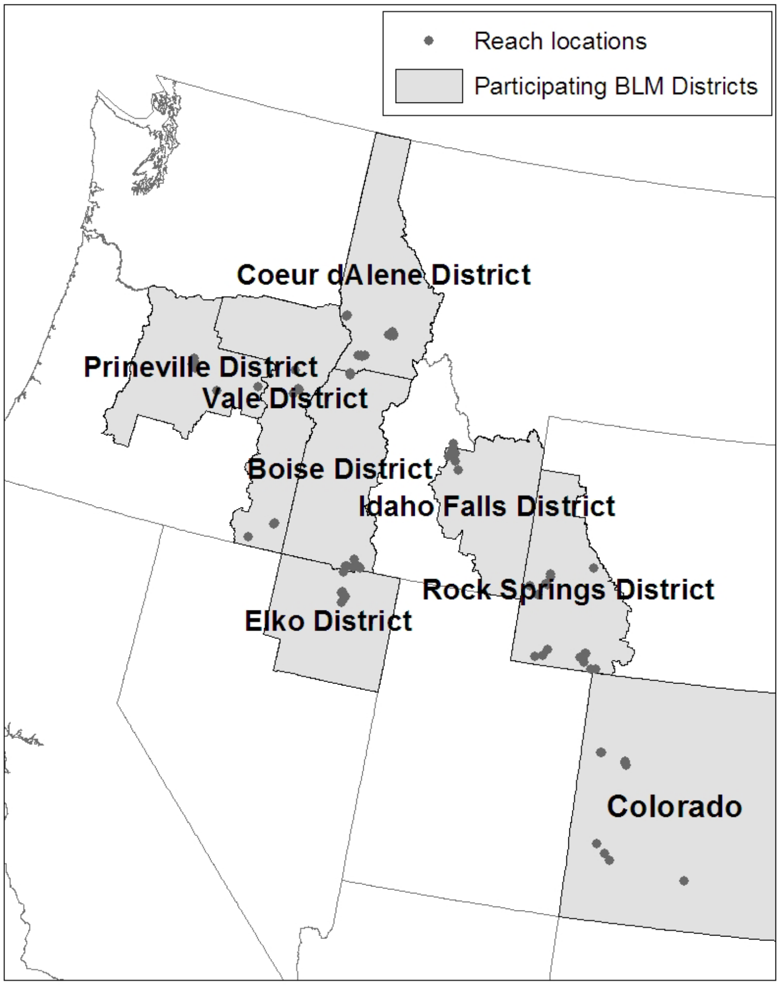

Figure 3. Locations of participating BLM districts and reach locations for the checklist question study.

Table 3. District offices and participating field offices within each district.

District Office Name	Participating Field Office Name(s)
Prineville	Prineville
Vale	Baker
Boise	Jarbidge
Elko	Elko
Coeur D'Alene	Cottonwood
Idaho Falls	Idaho Falls
Rock Springs	Rock Springs, Kemmerer, Pinedale
Colorado	Colorado (southwest)

3.5 Methods for stream distance study

Four distances between stressors and macroinvertebrate samples were calculated within GIS for the stream distance study: straight-line, slope distance, flow path, and hydrologic travel time. Yuan (2004b) found, using a GIS, that flow length distance may be more relevant than straight-line distance. Similar results should be expected, that the two methods using straight-line distance (straight-line and slope distance) would be less relevant to actual landscape-sample relationships than flow length or travel time. However, different types and magnitudes of errors are possible with each measurement technique. Each measurement method (Table 4) uses an increasing number of inputs as the complexity of the distance model increases (including elevation, hydrology, and land cover). Each input (elevation, land cover, assumptions for hydrologic parameters) assumes a certain amount of error due to spatial uncertainty and the inherent limitations of computer-generated models.

Table 4. Comparison of distance calculation methods used in the stream distance study. With increased model attributes and number of inputs comes increased potential error and decreased spatial certainty.

	Straight-line	Slope distance	Flow length	Travel Time
Number of input layers	2	3	5	10
Total number of raster layers in the GIS	0	1	3	25
Processing time for one sample (approximate)	< 1 minute	3 minutes	10 minutes	20 minutes
Includes Elevation?	N	Y	Y	Y
Includes Hydrology?	N	N	Y	Y
Includes land cover?	N	N	N	Y
Units	Meters	Meters	Meters	Minutes

Data used for this portion of the study included macroinvertebrate sample points, grazing polygons, and mining polygons. The macroinvertebrate sample points were samples from the NAMC database taken within the study areas. At locations with multiple macroinvertebrate samples taken over the same amount of time, 115 of the 228 sites, only the most recent sample was included in the study. The grazing polygons were obtained from the field offices of the respective study areas at the allotment level. The mining polygons were also obtained from each field office and included areas of mining leases, mining claims, and gravel pits. All three types of mining polygons characterize areas with active mining activities. Wyoming contained only gravel pits, while Utah contained mining leases and claims, some of which were gravel extraction pits and some of which represented other mining activities.

Pre-processing for the distance calculations included the calculation of grazing and mining polygon centroids collectively referred to as stressor points, and the overlay of sample points and stressor points and centroids with elevation. The centroid of all stressor polygons was found using the centroid field calculation available in ArcGIS (ESRI 2002). All calculated distances were to the centroid of polygon stressors. The overlay of elevation from a 10-m DEM with both the stressor and sample points was necessary for the calculation of slope distance. This can be accomplished using the GRIDSPOT script for ArcMap (Rathert 2004) which appends a field onto the attribute table of the point shapefile with the grid value directly beneath it.

Straight-line and slope distance

Straight-line distance or “crow’s flight” distance does not take elevation or hydrology into account. Straight-line distance was calculated using a spatial join, and the closest stressor centroid to each sample point was recorded, as well as the distance to the stressor (Figure 4). This function is pre-programmed in ArcGIS (ESRI 2002).

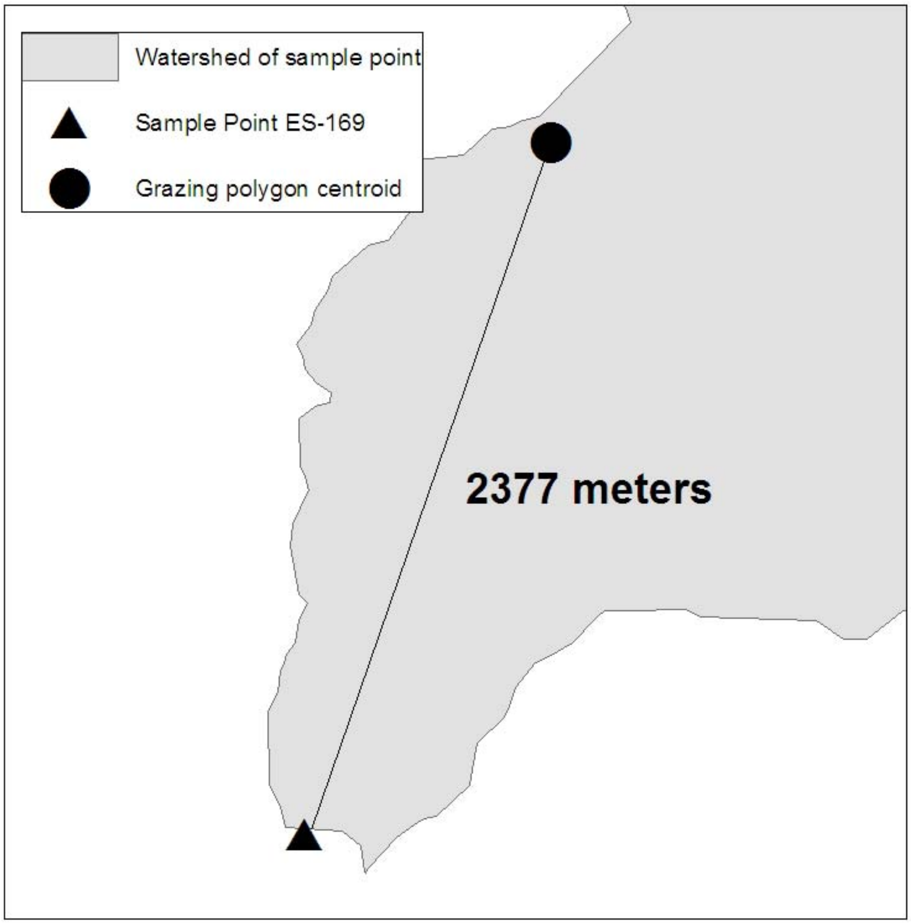

Figure 4. Straight-line distance for sample point ES-169 in Utah (triangle). The delineated watershed is the watershed for sample point ES-169. The black line shows the path used to measure distance between the sample point and the stressor (circle), 2377 meters.

Slope distance measures the distance between two points using the net elevation change. Slope distance between samples and stressors was found using the joined table created by calculating straight-line distance. This table contained the elevation of both the sample point and the stressor location, and the sample elevation was subtracted from the stressor elevation. This amount was used in the Pythagorean theorem to represent change in rise, and the straight-line distance previously calculated was used as change in run between the two points. The length of the hypotenuse of the resulting triangle was recorded as the slope distance.

Flow length

Flow length takes elevation and topography information into account. This is accomplished within a GIS by first calculating the flow direction of each cell within the watersheds of interest using a DEM. Flow direction is calculated for one nine-cell block, or neighborhood, at a time. A pre-programmed algorithm computes all calculations for flow length. The flow direction is defined as the steepest downhill slope from the center cell (ESRI 2002). From the flow direction, the length of the path water would take to an outlet can be determined. In this case, outlets were the macroinvertebrate sample points. Flow length distance was calculated using the flow length command available in Raster Calculator in ArcMap (ESRI 2002) (Figure 5). The downstream distance was calculated from each stressor to the closest outlet point, or sample. GRIDSPOT was used to record the distance to outlet at each stressor location.

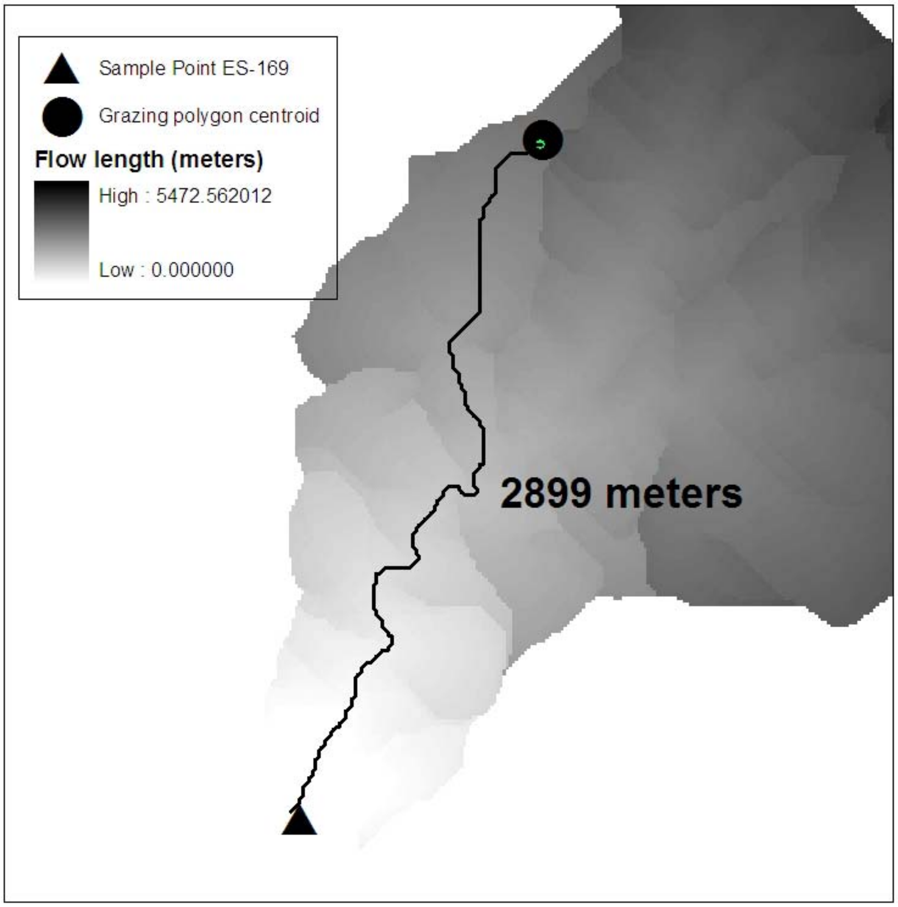

Figure 5. Flow distance for sample point ES-169 in Utah (triangle). Flow distance in meters was calculated using Raster Calculator, which resulted in a grid. The path of the flow distance from the stressor (circle) to the sample is shown by the black line, 2899 m.

Travel time

Travel time, the final stream distance measurement, uses information about elevation and topography as well as land cover to determine the time, rather than distance, separating two locations in a watershed. Travel time calculations are complex and include partitioning the area of interest by flow types and them performing multiple calculations. Many of these calculations have parameter values that were not known, so sensitivity analyses were included in the study and are reported with these methods.

The travel time concept is based on time of concentration, which is defined as the amount of time for water to travel from the most hydrologically remote point in the watershed to the watershed outlet (Bedient and Huber 2002). The time of concentration measurement can be adapted to find the time it takes for water to travel from any given point within a watershed to the watershed outlet by using a GIS to complete the calculations. As in the flow length calculation, macroinvertebrate sample points are watershed outlets (Figure 6).

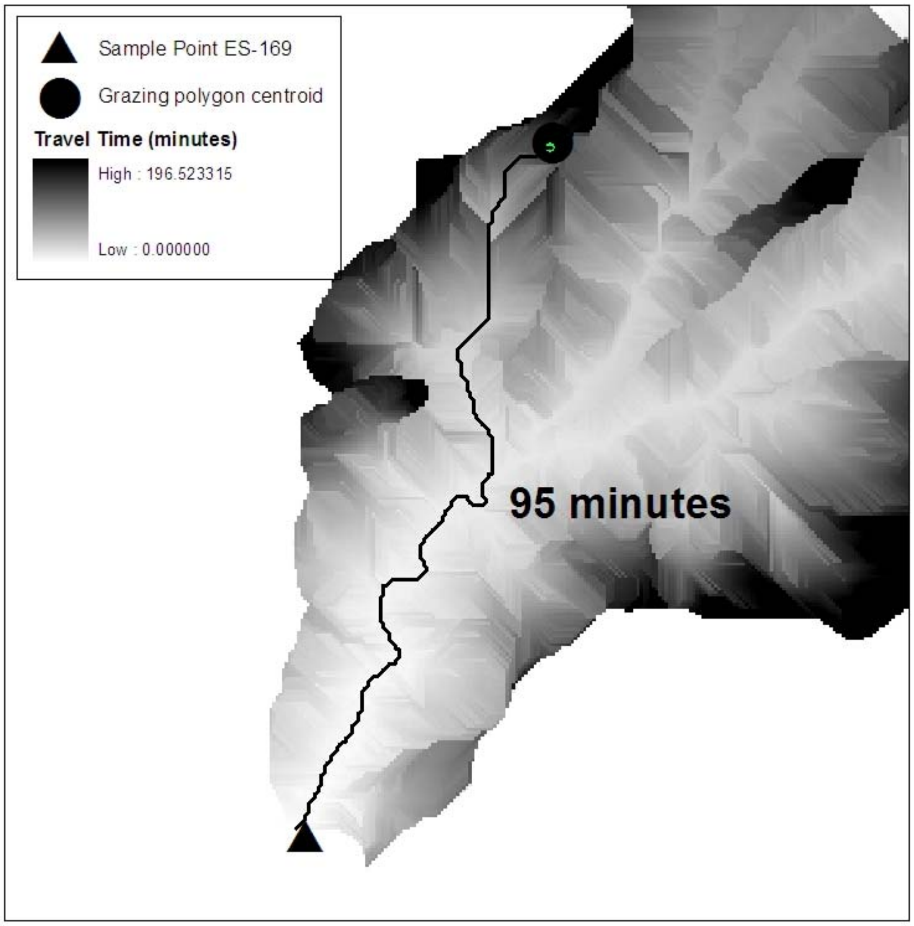

Figure 6. Travel time for sample point ES-169. Travel time follows the same path as flow distance (black line) but also incorporates information about ground cover. The travel time from the stressor (circle) to the sample point (triangle) was 95 minutes.

Partitioning of flow types for travel time

In order to calculate travel time, the velocity of water as it moves through the watershed must first be calculated. The velocity is calculated in three separate segments (SCS 1973): channel flow velocity, overland or sheet flow velocity, and rill or shallow flow velocity. A GIS can be used to determine where channels occur, and where the separation between overland and shallow flow occurs. Each 10 by 10-meter cell in a watershed represents only one type of flow. The resulting grid from the GIS operations to separate the three types of flow will be referred to as a partition grid. The partition grid will contain an integer in each cell designating which of the three types of flow occurred in that grid cell. Channel locations can be determined using a flow accumulation grid produced using a pre-programmed function in GIS (ESRI 2002). Flow accumulation is calculated using flow direction (see above) and based on the direction of flow and the position of the cell in the watershed, measures the number of cells "flowing" into that cell. This is calculated for every cell in the watershed, resulting in a flow accumulation grid. This can be used to estimate channel location, were cells with high flow accumulations are assumed to be channels. To separate channels from non-channels, a conditional statement it used based on the minimum flow accumulation a cell must have to be considered a channel. This minimum flow accumulation will be referred to as the flow accumulation cutoff, or FA cutoff. The resulting flow network will be referred to as a digital elevation model-generated flow network, or a DEM-FN. A sensitivity analysis was performed on each HUC8 in the study area to determine the most desirable FA cutoff. One subwatershed in each HUC8 was chosen. The subwatersheds were each about 25,000 ha in area, and were chosen to represent the upper to middle reaches of the entire study areas.

A USGS Digital Raster Graphic (DRG) was used as a reference to determine the most desirable FA cutoff for the DEM-FN. The most desirable FA cutoff would be that which most closely aligns with the DRG stream channel. A DRG is a scanned and rasterized USGS topographic quad that shows all intermittent stream channels marked by the USGS and was assumed to represent ground truth for the purposes of this study. For each subwatershed, the stream color was separated out of the DRG and ArcScan was used to generate a continuous network. Then, DEM-FN were generated at the FA thresholds of 200, 400, 800, 1000,1400, 1600, and 2000 (10 m cells). Each DEM-FN was overlaid on the DRG-FN to visually determine which DEM-FN had the best match to the intermittent stream networks from the DRG.

For Utah subwatershed 1, the FA cutoff of 800 cells appeared to give the closest match (Figure 7). In Utah subwatershed 2, the FA cutoff of 1600 cells appeared to best match the DRG-FN, however, results are not as clearly interpretable as for subwatershed 1. In Wyoming subwatershed 1, an FA cutoff of 2000 cells was chosen as the best representation of the DRG flow network, and in Wyoming subwatershed 2, an FA cutoff of 1600 cells was chosen. In Utah 1 and Wyoming 1, choosing the DEM-FN was relatively straightforward. The flow network lengths were relatively homogeneous throughout the watershed and closely matched the DRG-FN in almost all areas. However, in Utah 2 and Wyoming 2 there were many areas that were erroneous, due to lack of relief in the DEM and due to water bodies other than streams in the DRG. These errors also made a qualitative approach to choosing a FA cutoff difficult. When faced with such a watershed, and an appropriate FA cutoff is not obvious, the analyst should choose a reasonable FA cutoff (between 200 and 2000 cells) that appears to best represent the stream network as a whole. It is the author's opinion that each HUC8 should be tested for the most appropriate FA cutoff, as the FA cutoff will change according to climatic and topographical differences, even among adjacent HUC8s.

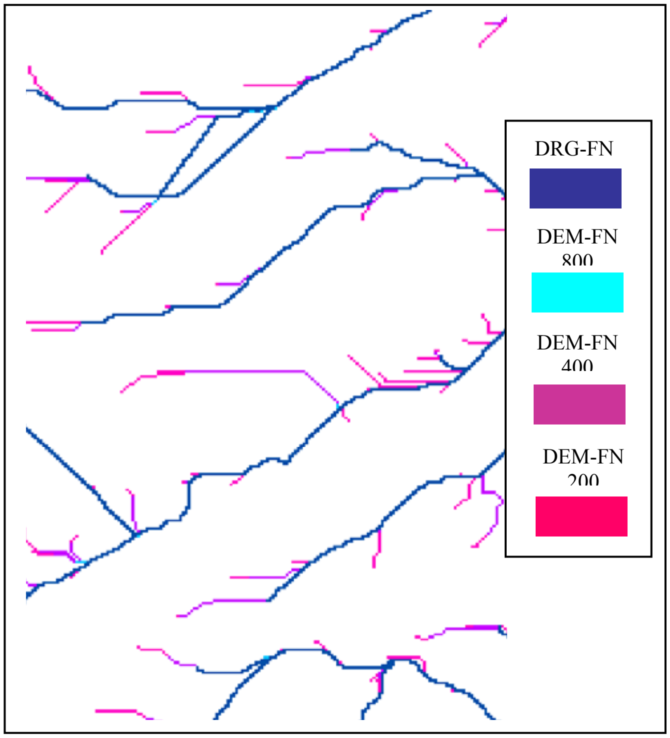

Figure 7. Selection of a DEM-generated Flow Netwotk (DEF-FN) for the Utah 1 subwatershed. Each DEM-FN was compared with the Digital Raster Graphic flow network (DRG-FN), the same as those delineated on USGS topographic maps. Note how the DEM-FN 800 is almost completely hidden by the DRG-FN, however, the other generated FN are not.

To determine the separation between overland and shallow flows, a distance from the ridgetop where overland (sheet) flow stops and shallow flow begins must be chosen, referred to as the overland flow cutoff (OFC). The shallow flow area can be thought of as similar to a riparian zone (Figure 8) and is the area where water travels in small channels (rills) into the stream channel. The overland area is where water travels in sheets. A sensitivity analysis was done to determine the effect that changing the OFC would have on the magnitude of potential error in finding the travel time. The U.S. Army Corps of Engineers HEC-HMS manual suggests that most sheet flow turns into shallow flow after 100 meters (Feldman 2000).

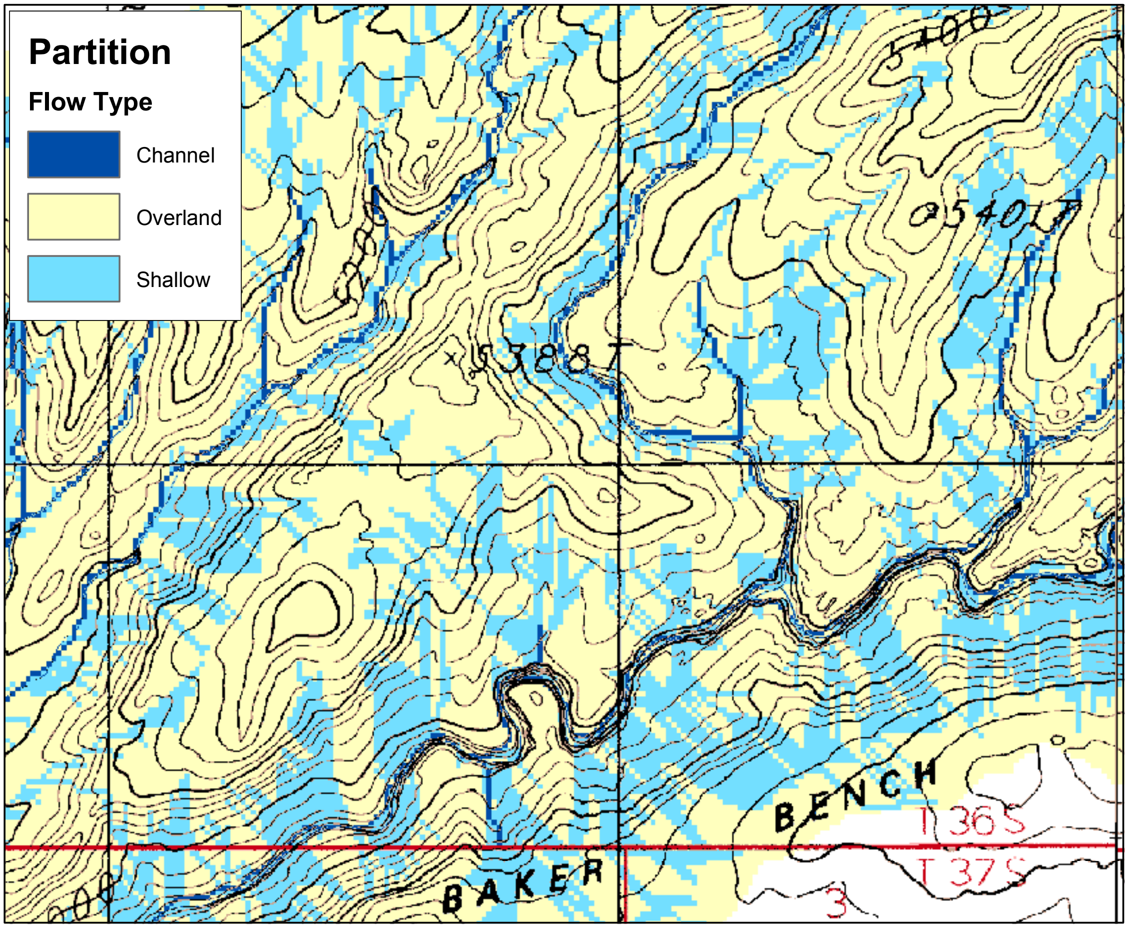

Figure 8. Types of flow partitioned for the travel time calculations. Channel flow (dark blue) includes channels similar in length to those delineated on a UGSG topographic map. Overland flow (light yellow) includes the area from the top of the ridge (shown by USGS topo quad in black) 100 meters down, following the flow path distance from the ridge top, which results in an irregular border. Shallow flow (light blue) includes the areas in between the overland and channel flows and can indicate ephemeral drains and areas of rill flow.

In order to quantify the differences in travel time caused by a variation in overland flow cutoff distances, the effects of changing the OFC by 50 meters, 50% of the currently assumed OFC (100 meters) were analyzed. A 25,000 ha subwatershed of the Utah 1 HUC8 was used. The watershed was partitioned once for an OFC of 50 meters and once for an OFC of 150 meters, all other inputs remaining the same. Then, each partition dataset was used to find travel time (further travel time calculations described hereafter). The travel time results were compiled and the mean travel time for the watershed was found. The mean travel time for the 50-meter OFC was 132 minutes and the mean travel time for the 150-meter OFC was 133 minutes. Since less than one minute (0.7 %) mean difference between 50 meter and 150 meter OFC's were found in the final travel time, this study assumes that an overland flow distance of 100 meters will be sufficient and appropriate as suggested in the HEC-HMS manual (Feldman 2000). If ground truth data or researcher knowledge in this area improves, then other distances should be tested for their final effect on travel times.

Computing channel velocity for travel time

Once the partitioning of flow types was complete, velocity was computed for each separate flow type: channel, shallow, and overland. To determine channel velocity, Manning's equation was used:

$$V_{channel} = (CR^{2/3}S^{1/2})/n;$$

where R= Hydraulic radius of the stream,

S= slope of stream channel, and

n= Manning roughness coefficient (Feldman 2000, Haan et al. 1994).

Hydraulic radius was determined by the equation

$R = A/Wp$,

where A is the cross-sectional area of the stream channel and Wp is the wetted perimeter of the cross-sectional area. This calculation presents some problems when incorporated into a GIS. First, R changes slightly at each cross-section along a stream network, so the possible values of R within a study area are nearly infinite. Additionally, velocity and channel dimensions are dependent on R (as calculated here) but R is also dependent on velocity. Therefore, some generalizations must be made about the average R based on assumptions about flow in a study area. Additionally, the resolution of the DEMs used in this study is too coarse to be used in determining stream widths and depths. However, equations are available for most regions that relate channel properties, such as width and depth, and sometimes R directly to watershed properties, such as drainage area. Drainage area can be calculated in a GIS and then a regional flow equation can be used to determine average channel width and depth, or R if available, from the drainage area. If an equation for R is not available, then the width and depth can be used to find the area and wetted perimeter of the stream after a representative stream cross-sectional shape is chosen from those listed in Rosgen (1994) and a simplified R value can be calculated.

The regional flow equation used for Wyoming was from Lowham (1976). Lowham used regions to separate areas of Wyoming that were largely different in hydrologic regime. The region used for this study was region 1, or the Wyoming mountainous areas. Lowham calculated hydraulic radius (R) at each gauging station used for data collection in region 1. Lowham assumed from his observations Rosgen stream type B, a moderately entrenched triangular shape with a high width: depth ratio, and both streambanks generally at equal angles to the stream bottom. The equation

$$\log_{10}R = 0.1170357 + 0.4583036 \log_{10}DA;$$

where DA = acres of drainage area, was used and had an r^2 value of 0.78 (Lowham 1976). Individual equations for width and depth were also available, but had lower r^2 values than the equation for R and would introduce more potential error into the final calculation.

The regional flow equations used for Utah were from Fields (1975). The equations

$$W = 3.27 * DA^{0.51} \ (r^2 = 0.83) \text{ and}$$

$$D = 0.79 * DA^{0.24} \ (r^2 = 0.58)$$

where W = stream width in feet and

D = stream depth in feet were used (Fields 1975).

A Rosgen (1994) stream type of B was used and the equation used to determine R was:

$$R = (1/2W^2 + D^2)/2$$

substituting the W and D equations above. An equation for R directly from drainage area would have been ideal, however, no such information was found in the literature.

A sensitivity analysis was performed to determine the effect that the choice of stream shape for calculating an approximate hydraulic radius would have on travel time. Ideally, flow rate could be held constant with R calculated for differing channel shapes. However, flow rate (velocity) is the ultimate product in this calculation, so channel cross-sectional area will be used as a constant. The change in R given a constant cross-sectional channel area (4 m^2) across five common channel shapes, corresponding to Rosgen stream types, was calculated (Table 5). Travel time was calculated twice, once using the maximum R, 0.68, and once using the minimum R, 0.60, with all other variables held constant. The average travel time for R=0.68 was 871 minutes and the average travel time for R=0.60 was 943 minutes, a difference of 7% between the mean high R and low R calculated travel times. A visual comparison of the two grids also reveals a relatively small difference in the number of cells in each category (Figure 9). Because the R of the standard right triangle (0.60) was not highly different from the R of the trapezoid (0.68), this study will continue to assume a right triangle channel shape, Rosgen's stream type B, for simplicity of calculation. Care should be taken when choosing a channel shape to represent the watershed although shape appears to have little impact on determining the watershed travel time. When the channel shape is known for an area, the known shape, rather than the assumed shape should be used.

Table 5. Calculation of Hydraulic radius, R, given a constant cross-sectional area for five common channel shapes. R = Area/Wetted Perimeter.

Shape	Rosgen (1994) stream type	Width (m)	Depth (m)	Area (m^2)	Wetted Perimeter	R (Hydraulic Radius)
equilateral triangle	A	3.04	3.39	4	6.08	0.66
irregular triangle	C	3	2.66	4	6.16	0.65
right triangle	B	3	2.66	4	6.67	0.60
trapezoid	G	3	1.33	4	5.84	0.68
rectangle	F	3	1.33	4	6.66	0.60

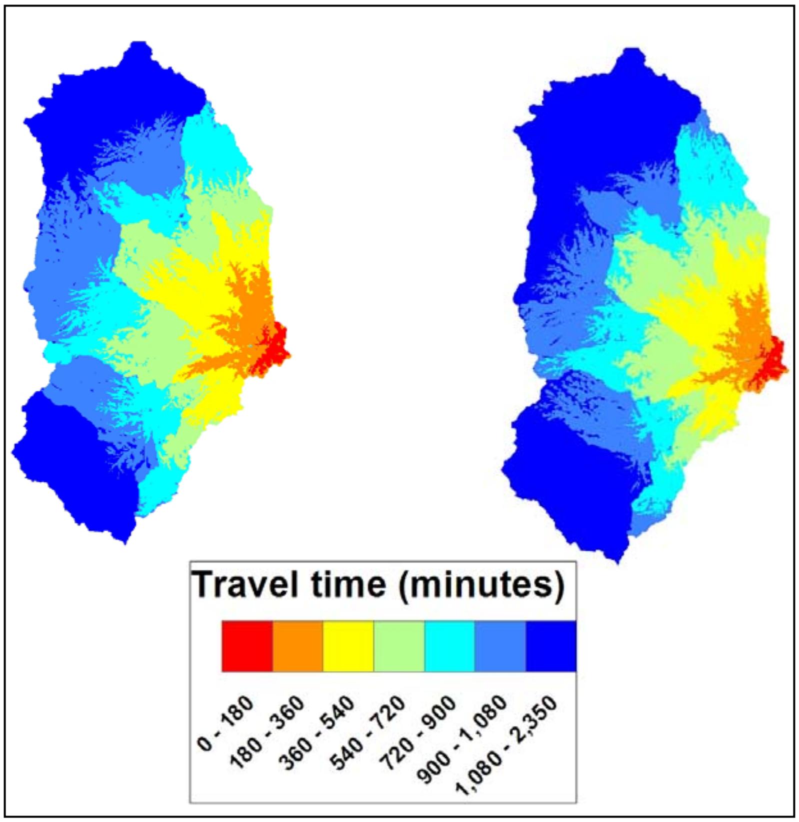

Figure 9. Visual comparison of travel time zones using high R (hydraulic radius) - value of 0.68 (grid on the right) and low R-value of 0.60 (grid on the left). Grids were separated by 180-minute intervals of time to outlet in a subwatershed in Utah 1.

The slope of the stream channel was found using the focal range function in the following equation: focalrange([DEM]) / focalrange(flowlength([flow direction grid])). The focal range operator finds the range of occurring values in a 9-cell window around each cell on the grid (ESRI 2002). The above equation divides the range in elevation (rise) by the range in flow length (run) to determine channel slope.

Manning's roughness coefficient was estimated using known channel physical properties and the same properties were assumed to occur in all streams throughout the entire watershed. The coefficients and stream types used (Ward and Trimble 2004) in the analysis are shown in table 6. Determining Manning's n for all channels in a watershed without field-collected data requires more assumptions. Neither DEMs nor land cover datasets can provide information about stream roughness. User knowledge of the area and general assumptions must be made for this segment of the calculation. A sensitivity study was also performed on channel n to determine the effect of high and low n values (table 3.6) on the magnitude of error inherent in travel time calculations. A subwatershed of 25,000 ha in the Utah 1 study area was used.

Table 6. Stream types and Manning's n values used for the study(Ward and Trimble 2004).

Stream type description	Associated n value
Clean, straight, no rifts/deep pools	0.03
Clean, winding, some pools	0.045
Clean, winding, some pools, more stones	0.05
Sluggish, weedy, deep pools	0.07
Mountain stream, cobbles and few boulders	0.04
Mountain stream, cobbles and large boulders	0.05

The mean travel time for the lowest n value in the subwatershed, 0.03, was 132.2 minutes. The mean travel time for the highest n value, 0.07, was 297.8 minutes. The increase between these two average times is 165.6 minutes, or 55% of the largest travel time.

When all other inputs were held constant, a very large difference in the resulting travel times were observed between the maximum and minimum n used in this study.

This illustrates that the travel time calculation is very sensitive to the input of channel n, determined by the channel type. For this study, because dominant channel type is not known, an n value of 0.05 "Clean, winding, some pools, more stones" and "Mountain stream, cobbles and large boulders"(Ward and Trimble 2004) will be assumed. This n value was chosen because it avoids the extremes of the n values available and it is viable for two distinctly different stream types. Researcher knowledge of the stream type would have been preferable, but was beyond the sampling scope of this study.

Computing shallow and overland velocities for travel time

The velocity of cells in the watershed partitioned as shallow flow was calculated as: $V_{shallow}$ =16.1345(sqrt S) for unpaved areas and =20.3282(sqrt S) for paved areas, where: S= percent land slope (Feldman 2000). Paved and unpaved areas were determined using GAP data (Table 2). Areas classified as urban, suburban industrial, or commercial were considered "paved" areas and all other classifications "unpaved". Percent land slope was determined using the pre-programmed slope function in ArcGIS, which uses the Z-values (elevation) of an eight-cell neighborhood surrounding each cell to determine the slope of that cell. Cells directly adjacent to the center cell are weighted more heavily than the cells diagonal to the center cell (ESRI 2002).

The velocity of cells partitioned as overland (sheet) flow was determined by: $V_{overland}$= $(Lm/((0.007\ (nL)^{0.8})/(P_2^{0.5}*S^{0.4})))/60$ where: Lm = flow length to outlet in meters; n= Manning's roughness coefficient for land; L= flow length to outlet in feet; P_2=2-year, 24-hour rainfall; S= percent land slope (Feldman 2000, Haan et al. 1994, Heatwole and Burcher 2003).

Manning's roughness coefficient for the land surface was found using GAP data to determine land cover, and Manning's n charts for each land cover type provided in a hydrology text (Ward and Trimble 2004). Land cover types in the Manning's n chart did not exactly match the GAP categories for either state, so some assumptions were made to assign n values to each land cover type present in the study areas (Table 7 and Table 8). There were also differences in the detail of coverage between the two states, where Wyoming had many detailed cover types and Utah had fewer, less detailed cover types.

Table 7. Wyoming GAP categories, cover types, and Manning's n values assigned (Ward and Trimble 2004)

Wyoming GAP Descriptions	Manning's n Cover Type	Manning's n
Human settlements, Mixed grass prairie, short grass prairie, Unvegetated playa, Basin bare, Mining operations, alpine areas	Development, short grass, bare	0.03
Irrigated crops, Foothills grassland, Grass-dominated wetland and riparian	Row crops, high grass	0.035
Dry-land crops	Field crops	0.04
Burned conifer, Clearcut conifer	Young forest (clearcut, burned)	0.06
Mesic and Xeric upland shrub types, Bitterbrush shrub, Mountain big sagebrush, Wyoming big sagebrush, Black sagebrush steppe, Basin big sagebrush, Desert shrub, Salt fans and flats, Greasewood fans and flats, Vegetated dunes, Shrub riparian	Shrubland	0.08
Spruce-fir, Douglas fir, Lodgepole, Whitebark pine, limber pine, Ponderosa pine, Juniper woodland type, Bur Oak woodland	Forest	0.1
Forest dominated riparian	Riparian forest	0.12
Aspen forest	Dense straight willows	0.15

Table 8. Utah GAP categories, cover types, and assigned Manning's n values.(Ward and Trimble 2004)

Utah GAP cover types	Manning's n cover type	Manning's n value
Barren, Urban, Salt Desert Scrub	Short grass, Cultivated with no crop, bare	0.03
Grassland, Alpine, Dry Meadow, Wet Meadow, Agriculture, Desert Grassland, Blackbrush, Creosote-Bursage, Greasewood, Pickleweed Barrens	High grass, mature row crops	0.035
Sagebrush/perennial grass	Medium-to-dense brush, in winter	0.07
Juniper, Pinyon, Pinyon-Juniper, Mountain Mahogany, Sagebrush	Brush, slightly more sense than Sagebrush/perennial grass cover type	0.08
Ponderosa Pine, Lodgepole, Spruce-Fir/Mountain Shrub, Mountain Fir/Mountain Shrub	Medium density stand of timber	0.09
Spruce-Fir, Mountain Fir, Oak, Maple, Lodgepole/Aspen	Heavy stand of timber	0.1
Mountain Riparian, Lowland Riparian, Wetland	Heavy stand of timber, with flood stage reaching branches	0.12
Aspen, Aspen/Conifer	Dense straight willows	0.15

The 2-year, 24-hour rainfall, the empirically determined average amount of rain to fall in a 24-hour period at a 2-year recurrence interval, was found using data provided from the National Weather Service (NWS) precipitation data frequency server (NWS 2005). The centroid of each HUC8 was used to determine the amount of rainfall. The 2-year, 24-hour rainfall amount was determined from the precipitation data frequency server as 1.23 inches for both Utah study areas, and 1.4 inches for both Wyoming study areas. Land slope was the same as calculated for the shallow flow velocity.

Calculating travel time from velocities

After velocities for each partition type were calculated (channel, shallow, and overland) the travel time was found. Travel time for the entire watershed was calculated by combining all the velocities (channel, shallow, and overland) into one grid. The flow length function was used with a weight of the inverse of the combined velocities. This resulted in a travel time in minutes for each individual grid cell across the entire watershed. This travel time was used to determine how far, in minutes, a stressor is from a sample point.

Automating the travel time calculations

Calculations for travel time are more complex than the other distance measurements used in this study. The computation of travel time, even in a GIS, is a tedious and time-consuming task. The automation of travel time calculation, using ArcGIS scripts, was completed as part of this study.

Two separate programs were created using ArcObjects in ArcGIS 8.3(ESRI 2002) in order to automate the calculation of travel time. The first program, the Partition Calculator, determines which cells in the watershed are channel flow, overland flow, or shallow flow, and assigns an integer to represent each flow type. The result of the partition calculator, as well as several other grids, can be input into the second program, the Watershed Velocity Calculator. The watershed velocity calculator will need to be revised for study areas other than Wyoming and Utah, but the basic framework of the program is complete. A watershed velocity calculator for Idaho was also completed. The main purpose of these programs was to reduce the time necessary to derive travel time, because the calculations are numerous and tedious, even within a GIS. Both programs may also provide useful tools for others wishing to study travel time.

User inputs for the partition calculator (Figure 10) include a DEM-FN where channels are 0 and all other cells are 1 (0,1 flow network); a flow direction grid; a name for the new grid; and the distance separating shallow flow and overland flow. The output for the partition calculator is an integer grid where channels = 1, shallow flow = 2, and overland flow = 3.

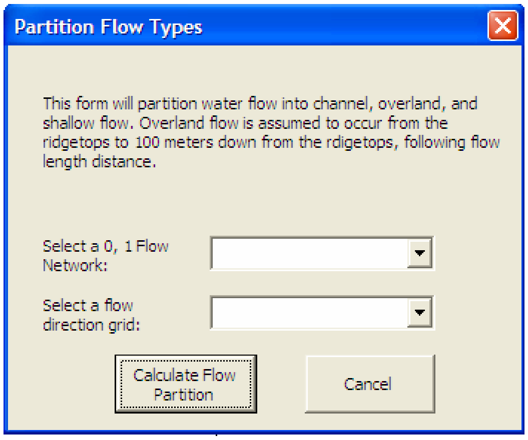

Figure 10. Partition calculator created using ArcObjects (ESRI 2002).

User inputs for the watershed velocity calculator (Figure 11) include flow network grids, flow accumulation grid, flow direction grid, land cover grid, and elevation grid, as well as the dominant channel type and the 2-year 24-hour rainfall for the area. Outputs of the watershed velocity calculator include the overall watershed velocity grid, and the separate velocities of the channels, overland, and shallow flow. The user must then use this velocity grid as a weight to calculate final travel time.

The partition calculator and watershed velocity calculator were used to find the travel times within the study areas in Wyoming and Utah.

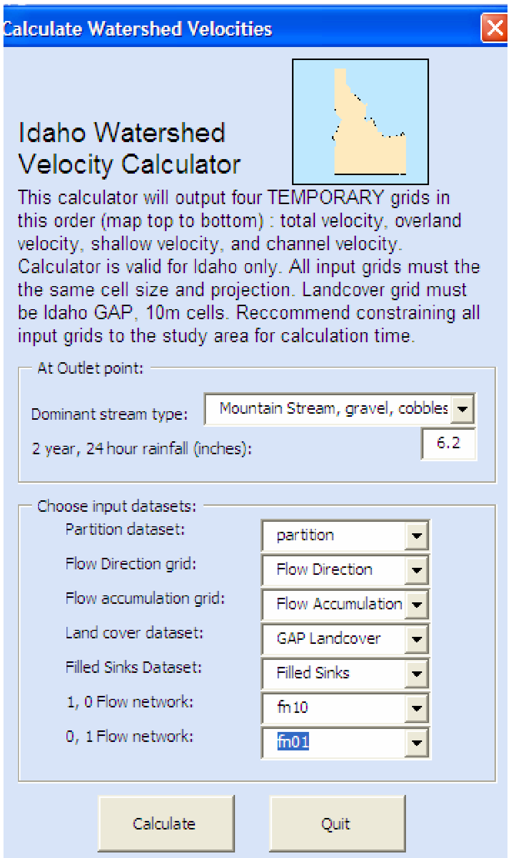

Figure 11. Watershed velocity calculator programmed using ArcObjects (ESRI 2002).

Characterization of stressor and sample conditions

For the stream distance study, to characterize macroinvertebrate communities, the ratio of the number of individuals in Ephemeroptera, Plecoptera, and Trichoptera (EPT) taxa to the number of individuals in the Chironomidae taxa will be used. EPT taxa are sensitive to pollution, and decrease in number when water quality decreases (Yuan 2004a). Chironomidae taxa are generally tolerant of pollution, and should not decrease in number as water quality decreases (McCormick et al. 2004). The ratio of EPT to Chironomidae individuals should therefore increase with less impacted sites, and decrease with more impacted sites. For sites with no EPT individuals present, the EPT/Chironomidae ratio was calculated as zero. For sites with no Chironomidae individuals present, the ratio was calculated as the number of EPT individuals present. This index was calculated from the macroinvertebrate data supplied by the NAMC. It is also appropriately responsive to both of the land uses of interest (Grigorovich and Angermeier 2004).

In addition, for stream distance study question A, to assess the intensity of range management practices, stocking of grazing allotments of the sample years were used. Stocking was estimated by the acres per permitted AUM on the allotment (from the RAS database). This measures the grazing acreage in the allotment available for each AUM. Low acres per AUM indicate high levels of stocking, while high acres per AUM indicate low levels of stocking. More detailed measures of grazing and mining pressure may be illustrative, but were beyond the resource constraints of this pilot study. As the AILC progresses, measures of stressors will become more detailed and descriptive.

The spatial and temporal accuracy of the data to be used for this study was assessed before completing analysis. A pilot study was conducted, using macroinvertebrate data from the large (15,000 record) database, grazing allotment data, and stream location data for five watersheds, and these data were analyzed in a GIS. Using the National Hydrography Dataset (NHD) stream locations as a reference, distances from macroinvertebrate stream locations (MSLs) to streams were calculated and assumed to represent spatial errors in MSLs. Simulations of spatial errors in MSLs were then performed for each map, following the distribution of distance-to-stream errors. The number of MSLs that changed grazing allotment was noted for each iteration. Changes in grazing allotment due to low spatial accuracy would be significant in any further analyses of the grazing dataset. This study found that 12% of the points used would change allotment number when random spatial errors were added or subtracted from point coordinates. This illustrates a need for data accuracy screening and testing.

Temporal variation of the data for this study was also minimized as much as possible. At this stage in the AILC, temporally consecutive sampled studies were not available. However, as the AILC progresses, future studies should be able to further reduce temporal variability through consecutive samples. The use of temporally similar data was emphasized during this analysis with most data collected between 1999 and 2004; however, the sample data had a range of nine years (Figure 12).

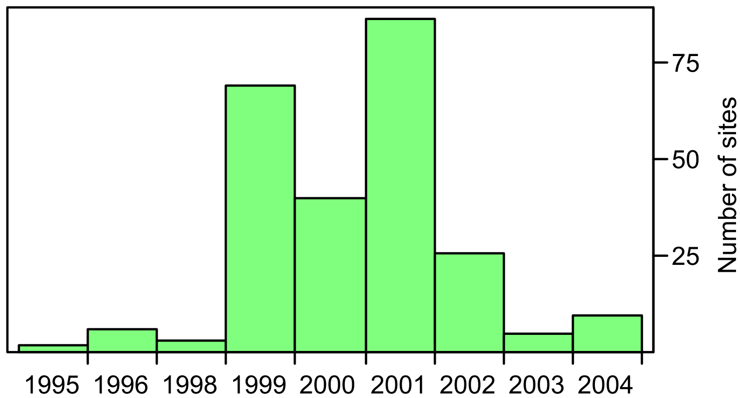

Figure 12. Distribution of macroinvertebrate samples among years for the 247 sites used in the stream distance study.

The calculations for the stream distance study will result in one data set. The dataset includes, for each study area (Utah and Wyoming), stream sample point to stressor location distance calculated in four ways, an identification number of the closest stressor measured each of four ways, a measure of macroinvertebrate community structure by the EPT/ Chironomidae ratio, and acres per AUM as an indicator of grazing pressure for the closest grazing allotment using both the flow distance and travel time methods. A t-test was done to determine the difference in mean EPT/Chironomidae ratio of samples with no stressors in their watersheds and samples with stressors in their watersheds. The effects of each distance measure, number of allotments or mines, and the intensity measure for grazing, and combined effects of distance and intensity on macroinvertebrate metrics were assessed using multiple linear regression.

3.6 Methods for PFC checklist study

A database with spatial reference (latitude, longitude) was designed and created in Microsoft Access for the PFC checklist study. This database allows analysis of the relationship between individual checklist question responses and reach ratings as well as spatial reference of each reach rating with instream conditions. All statistical analyses were performed with the program JMP version 4.0 (SAS Institute 2001) and Canoco version 4.5 (Ter Braak and Smilauer 2004).

PFC Question A

All reaches with available, fully completed standard lotic checklists, 150 reaches in all, were used for this study. Multiple logistic regression (Sall et al. 2001) with an alpha level of 0.05 was used to find the relationship between the reach rating (dependent variable) and each checklist question (independent variables). Ordinal logistic regression was used in place of binary logistic regression (often referred to as simply logistic regression) because the response variable has more than two categories and the explanatory variables have any number of categories. In this case, the response variable is PFC rating with three possible categories, and the explanatory variables are the 17 checklist questions, each with three possible categories as well. Apparent trends for the FAR category (downward, upward, not apparent) were ignored in this analysis because of the inconsistency between observers in assigning apparent trends. The log odds between the three separate categories are simulated at the same time in multiple ordinal logistic regression. For example, log odds of PFC vs. FAR, FAR vs. NF, and NF vs. PFC are all estimated together.

Principal components analysis was also used to illustrate any response patterns. PCA analysis produces a biplot with multiple axes, where axes represent eigenvalues, which are composites of several variables, arrows represent checklist questions, and points on the plot represent sample points. In a PCA biplot, the length of the arrows indicates the strength of the question's relationship to the PCA axes. The direction of the question's arrow in relationship to the other arrows indicates the degree of association with other questions; where arrows with a large angle of separation are less likely to be associated, and those with a small angle of separation are more likely to be associated.

Axes were scaled in the PCA analysis, for both this question and PFC question B (below). Scaling of the PCA axes causes all eigenvalues to sum to one and varies from non-scaled analyses in this regard. A non-scaled eigenvalue, which is a descriptor of one Principal Component (axis), would typically have a value greater than one. A scaled eigenvalue will have a value between zero and one, and the sum of all eigenvalues for all principal components will sum to one. Similarly, since eigenvalues are composites of several variables, individual variable (in this case checklist questions) loadings will also be scaled proportionately.

PFC Question B

A subset of checklist questions with additional applicable site data collected in the field by the summer field crew was selected (Table 9) totaling 119 sites. Potential relationships between PFC question responses and site measurements and observation were tested using contingency tables for ordinal data, or observations, and logistic regression for continuous data, or measurements (Sall et al. 2001), using the Chi-Squared (X^2) statistic and an alpha level of 0.05 for both tests.

Principal components analysis (PCA) was also used as an exploratory technique for this objective to determine if variation among sites, as described by field measures and observations or by factors in PFC ratings, could be adequately explained by a few variables. Axes were also scaled for this analysis, as in PFC question A.

Table 9. PFC questions with applicable measured site characteristics.

PFC Field Sheet Question	Measurable characteristics	Sources for characteristics	Observed or measured characteristics
4. "Riparian-wetland area is widening or has achieved potential extent"	Potential extent by topographic break, channel substrate, w/d ratio	(Rosgen 1994)	substrate and width/depth, main management objective
5. "Upland watershed is not contributing to riparian- wetland degradation"	Erosion apparent from surrounding slopes into the stream reach	(DeBano and Schmidt 1989)	Qualitative ranking of 1 or 2 = "no" in erosional deposition, main management objective
9. "Streambank vegetation is comprised of those plants or plant communities that have root masses capable of withstanding high-streamflow events"	Amount of streambank covered by deep, binding root masses, except in bedrock or boulder channels.	(Manning et al. 1989)	Visual estimation of % streambank with binding roots, score of 3 or 4, main management objective
10. "Riparian-wetland plants exhibit high vigor"	Hard to quantify, amount grazed by livestock reflects the ability of plants to establish communities and produce vigorous roots.	(Prichard et al. 1998)	Visual estimation of percent cover, score of 2, 3, or 4. Visual estimation of percent consumed by livestock, score of 3 or 4. Main management objective.
11. "Adequate riparian-wetland vegetative cover is present to protect banks and dissipate energy during high flows"	An ideal measurement is a greenline stability rating (Winward 2000), however, percent vegetative cover of the stream is also a measure of this variable. Percent cover necessary is	(Platts et al. 1987, Rosgen 1994)	Visual estimation of percent cover, score of 2, 3, or 4. Visual estimation of percent consumed by livestock, score of 3 or 4. Main

	determined by the channel and bank type.		management objective.
13. “Floodplain and channel characteristics are adequate to dissipate energy”	Measurement of sinuousity, w/d ratio, stream gradient	(Cowan 1956, Rosgen 1994)	W/d ratio and stream gradient.

Chapter 4: Results

4.1 Objective 1

There were 247 total sites used in the stream distance study, including sites from both the Utah and the Wyoming study areas. For each site, straight-line distance, slope distance, flow distance, and travel time were recorded. In instances where there were no stressors in the sample point's watershed, a flow distance and travel time of 1,000,000 were recorded, representing infinite distance. Of the 247 original sites, 70 had either mining or grazing stressors in their watersheds and 177 sites had no mining or grazing stressors in their watersheds. Of all sites, 140 were sampled quantitatively, meaning that a standard size sampling net was used to collect insects that only fell within the randomly placed sampling net.   and only these sites were used in any further analyses. Of the quantitatively sampled sites, 38 had mining or grazing stressors in their watersheds.

Scatter plots comparing the various distance measures within grazing stressors (Figure 13) illustrate that straight-line distance and slope distance are very similar (r^2 = 0.999), flow length and travel time and flow length and straight-line are slightly similar (r^2 = 0.29 and r^2 = 0.32, respectively) and travel time and straight-line are dissimilar (r^2 = 0.06). Scatter patterns and r^2 values were similar for mining stressors despite larger mean differences.

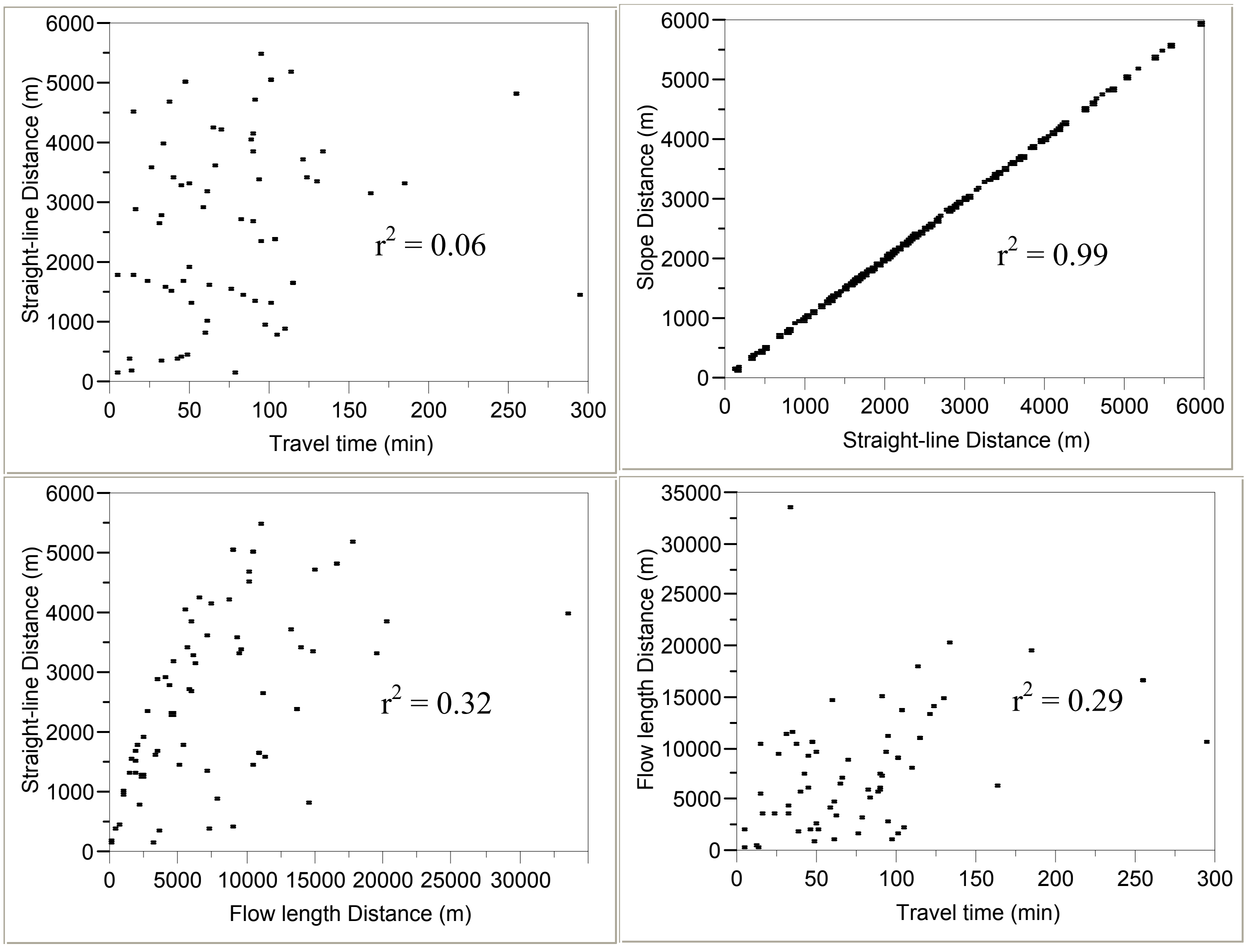

Figure 13. Scatter plots comparing the various distances calculated from grazing stressor centroids to sample points. Two outliers with travel times greater than 1000 minutes were excluded.

Two metrics were used for stream macroinvertebrates in this study: the EPT/Chironomidae ratio and the EPT abundance. The ranges of values for these metrics differ (Table 10). The EPT/Chironomidae ratio is the EPT abundance divided by the Chironomidae abundance, so EPT/Chironomidae ratios are smaller than EPT abundance scores, as would be expected.

Table 10. Differences in mean and range of values for the macroinvertebrate metrics used in the stream distance study.

Metric	Mean (both study areas)	Maxium	Minimum
EPT	822	20957	0
EPT/Chir	5.15	247	0

A t-test was used to test for a difference in mean EPT/Chironomidae ratio and mean EPT abundance between the sites with stressor centroids and the sites without stressor centroids. A difference in means was expected between the two groups. There was no statistically significant evidence ($p > 0.10$) that the mean EPT/Chironomidae ratio or mean EPT abundance differed between sites with and sites without mining or grazing stressor centroids in their watersheds. Furthermore, one-way ANOVAS were run to test for differences in mean EPT/Chironomidae ratio in sampling method (Surber net, kick net, plankton net, Hess net) and habitat (riffle, pool, pelagic, littoral, backwater, tinaja, multiple habitats). There was no statistically significant evidence that the mean EPT/Chironomidae ratio or EPT abundance differed between sampling techniques ($p > 0.10$) or habitats sampled ($p > 0.10$). A one-way ANOVA was also used to determine the difference in mean EPT/Chironomidae ratio and EPT abundance between aquatic ecoregions (Omernik 1987) and the same test was run for differences in means between states. There was no statistically significant evidence that mean EPT/Chironomidae ratio difference between ecoregions or states ($p > 0.10$). However, there was statistically significant evidence that mean EPT abundance differed between ecoregions and states ($p < 0.0001$, both tests). Significant differences in mean EPT abundance between ecoregions was tested using Tukey's HSD. Significant (< 0.05) differences occurred in all pairs of ecoregions except the Middle Rockies and Wasatch and Uinta Mountains, and between the Colorado Plateaus and the Wasatch and Uinta Mountains (Table 11).

Table 11. Comparison of mean EPT (Ephemoptera, Plectoptera, Trichoptera) abundances between Omernik's (1989) aquatic ecoregions using Tukey's HSD (Significant probabilities are in bold).

Abs(Dif)-LSD	Wyoming Basin	Middle Rockies	Wasatch and Uinta Mountains	Colorado Plateaus
Wyoming Basin	-4374.8	**735.9**	**1200.3**	**2888.7**
Middle Rockies	**735.9**	-1374.9	-1320.9	**877.7**
Wasatch and Uinta Mountains	**1200.3**	-1320.9	-2838.7	-955.2
Colorado Plateaus	**2888.7**	**877.7**	-955.2	-1013.7

Standard least squares multiple linear regression was used to determine the contribution to EPT abundance of each distance measure and distance*intensity. The EPT metric was partitioned by both ecoregion and by state. Only grazing stressors were tested. Results were statistically insignificant ($P > 0.10$) for all models tested.

Sampling sites that occurred within grazing allotment polygons and those that did not occur within grazing polygons were identified (Figure 14). This analysis eliminated the need to use only grazing polygon centroids. In Utah, all but 4 sites were within grazing polygons. Each sample within an allotment was joined to the allotment it fell within. Using linear regression, the relationship between EPT abundance and allotment acres and Acres per AUM was tested. Two tests were run: separation of EPT abundances by aquatic ecoregion and without separation of EPT abundances. There was no statistically significant evidence that EPT abundances were related to allotment size in acres or grazing pressure in acres per AUM ($p > 0.10$) in either test.

In the Wyoming study area, 15 sites were within allotments and 75 sites were not. There was slightly statistically significant evidence that the mean EPT value differed between sites within and sites outside of grazing allotments ($p = 0.05$). All 15 sites within allotments in Wyoming had an EPT of 0. Using linear regression, the relationship between EPT abundance and allotment acres and Acres per AUM was tested. Two tests were run: separation of EPT abundances by aquatic ecoregion and without separation of EPT abundances. There was no statistically significant evidence that EPT abundances were related to allotment size in acres or grazing pressure in acres per AUM ($p > 0.10$) in either test.

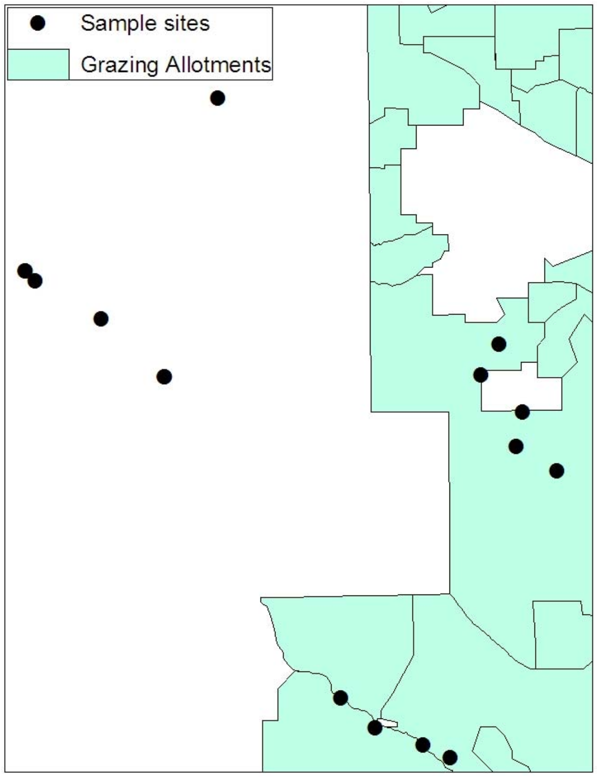

Figure 14. Sample sites that occurred within grazing polygons were identified using a GIS.

4.2 Objective 2A

There were 7 BLM field offices that contributed completed standard lotic checklists to the study, totaling 150 reaches. Of the 150 reaches used in the study, most field offices contributed between 6 and 13 percent of the reaches, and the Wyoming office contributed over 40 percent of the reaches (Figure 15). It is important to note that the Wyoming office is actually made up of three different field offices: Pinedale, Kemmerer, and Rock Springs. Considering this breakdown, the percentage of reaches contributed by each field office is about 6 to 15 percent.

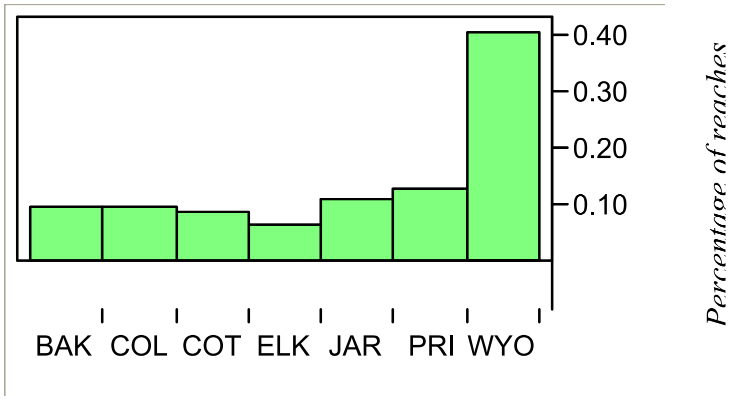

Figure 15. Distribution of reaches among contributing BLM Field Offices. BAK = Baker, Oregon; COL = Colorado; COT = Cottonwood, Idaho; ELK = Elko, Nevada; JAR = Jarbidge, Idaho; PRI = Prineville, Oregon; and WYO = Wyoming, which includes three separate field offices.

Of the 150 sites, 53 were rated PFC, 83 were rated FAR, and 14 were rated NF (Figure 16). A breakdown of percentage of reaches in each reach rating category by field office using the chi-squared statistic revealed that there is statistically significant ($p = 0.002$) evidence that the reach rating is dependent on the field office. An examination of the resulting mosaic plot (Figure 17) reveals that not all field offices had the same percentages of sites in the same reach rating categories. However, with sample sizes averaging 17 reaches per field office, this data is not sufficient for drawing conclusions and the chi-squared value was suspect. Suspect chi-squared values indicate that there were expected cell counts less than 5 in over 1/5 of all the cells. The Pearson chi-squared statistic (used in all chi-squared tests in this study) works better in suspect chi-squared value situations than other chi-square tests (Sall et al. 2001). However, results should be scrutinized and other tests used whenever available.

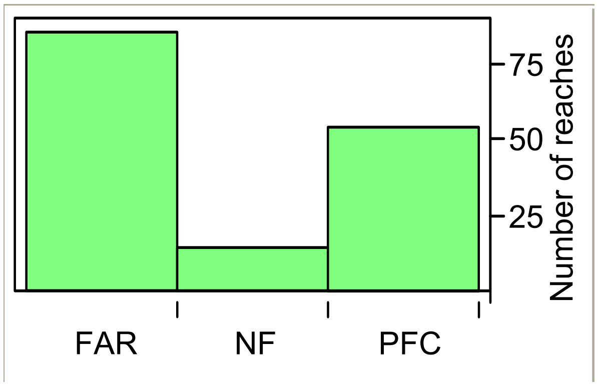

Figure 16. Number of study sites used in objective 2A with reach ratings of PFC, FAR, and NF.

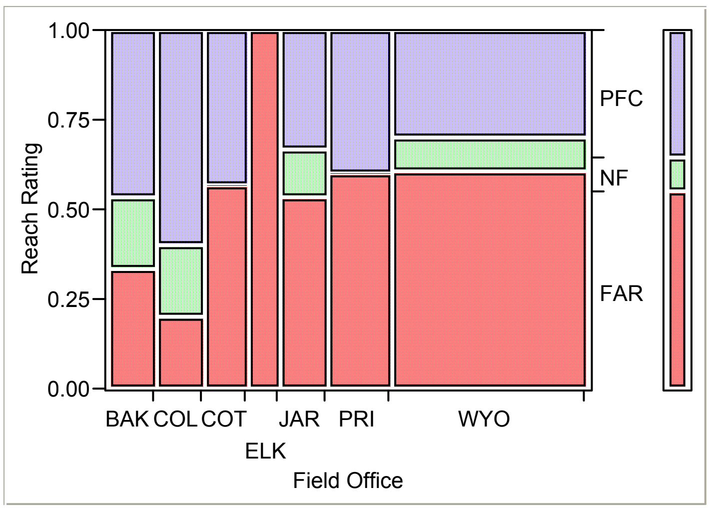

Figure 17. Mosaic plot of the proportion of reaches in each reach rating category by field office. Mosaic plots are calculated by dividing the X axis (Field Office) by the proportions of samples in each category, and then dividing the Y axis (Reach Rating) by the estimated probability responses of each office, or category. The final mosaic plot represents the frequency of each combination of reach rating (Y) and office (X) in relationship to all other possible combinations by the proportional area of the cell. BAK = Baker, Oregon; COL = Colorado; COT = Cottonwood, Idaho; ELK = Elko, Nevada; JAR = Jarbidge, Idaho; PRI = Prineville, Oregon; and WYO = Wyoming, which includes three separate field offices. Note that the Elko field office had only FAR reaches, but also had the smallest sample size (n = 10).

The standard lotic checklist does not incorporate any weighting or ranking or questions to determine the reach rating. The reach rating was significantly affected by question responses (multiple ordinal logistic regression, model $X^2 = 139.068$, $N = 150$, $P < 0.0001$). When individual questions were tested using the Wald Statistic, questions 2, 3, 5, 11, 12, and 16 were significant at the 0.05 alpha level (Table 12). For questions or terms significant in the ordinal logistic regression model where responses included “yes”, “no”, and “N/A” answers, the Chi-Squared test was performed for the responses of “yes” and “no” and again for responses of “yes” and “N/A”. Question 16 had only responses of “yes” or “no” so only the yes/no relationship was tested. Parameter estimates of the significant questions show that only the yes or no responses to the checklist questions, not the N/A responses, were significant in relationship to the reach rating ($\alpha = 0.05$) (Table 13)

Table 12. Wald statistic test results on individual questions in the ordinal logistic regression model.Significant probabilities ($\alpha = 0.05$) are given in bold. Where degrees of freedom (DF) = 1, responses only consisted of “yes” and “no”; where DF = 2, responses included “yes”, “no” and “N/A”.

Question	DF	Wald ChiSquare	Prob>ChiSq
Q1	2	0.0577	0.9715
Q2	2	11.028	**0.0040**
Q3	2	9.611	**0.0082**
Q4	1	1.530	0.2161
Q5	2	6.460	**0.0396**
Q6	1	0.707	0.4006
Q7	1	3.054	0.0806
Q8	1	0.199	0.6558
Q9	1	0.103	0.7487
Q10	1	0.037	0.8470
Q11	2	19.154	**0.0001**
Q12	2	8.457	**0.0146**
Q13	2	3.324	0.1897
Q14	2	2.765	0.2509
Q15	1	0.550	0.4581
Q16	1	9.142	**0.0025**
Q17	1	0.964	0.3263

Table 13. Parameter estimates and Chi-Squared test results for individual questions for both the yes, no responses and the yes, N/A responses for each significant question from Table 12. Significant probabilities (α = 0.05) are given in bold.

Term	Estimate	ChiSquare	Prob>ChiSq
Q2[y:n]	-1.9418105	5.70	**0.0170**
Q2[y:N/A]	-0.1287748	0.02	0.8784
Q3[y:n]	-2.0163043	9.36	**0.0022**
Q3[y:N/A]	1.05094603	0.19	0.6613
Q5[y:n]	-2.08418	6.45	**0.0111**
Q5[y:N/A]	15.876534	0.01	0.9189
Q11[y:n]	-3.9531076	19.15	**<.0001**
Q11[y:N/A]	-7.9972028	0.00	0.9581
Q12[y:n]	-3.1569883	8.45	**0.0036**
Q12[y:N/A]	0.96931098	2.00	0.1571
Q16[y:n]	2.73319869	9.14	**0.0025**

The principal components analysis (PCA) was centered by standard lotic checklist questions and values of “N/A” as responses to checklist questions were included in the analysis. The first two axes accounted for 47.4% of the variance in sample scores (Table 14).

Table 14. PCA results for PFC checklist study objective A. The first two axes account for 47.4% of the variance in samples, with diminishing percent variance explained for higher-level axes.

Axis	1	2	3
Eigenvalues	0.306	0.168	0.122
Cumulative % variance of checklist question data	30.6	47.4	59.5
Individual % variance of checklist question data	30.6	16.8	12.1

Table 15. Question loadings (eigenvalues) in the PCA analysis. Question 2 had the highest loading of axis 2, question 13 has the largest loading of axis one, and question 12 has the smallest and only positive loading of axis one.

Axis	AX1	AX2	AX3	WEIGHT
Eigenvalues	0.31	0.17	0.12	
Q1	-0.26	-0.04	0.11	1
Q2	-0.46	**0.88**	0.01	1
Q3	-0.64	0.05	0.29	1
Q4	-0.63	-0.30	0.03	1
Q5	-0.17	-0.16	-0.14	1
Q6	-0.63	-0.31	-0.20	1
Q7	-0.57	-0.31	-0.28	1
Q8	-0.60	-0.25	-0.32	1
Q9	-0.71	-0.30	-0.08	1
Q10	-0.67	-0.23	0.16	1
Q11	-0.74	-0.23	0.17	1
Q12	**0.02**	0.11	-0.96	1
Q13	**-0.78**	-0.06	0.17	1
Q14	-0.47	-0.05	-0.23	1
Q15	-0.63	-0.28	-0.10	1
Q16	-0.56	0.03	0.02	1
Q17	-0.65	-0.06	0.00	1

In the question loadings table of the PCA analysis (Table 15), question 2 has the highest loading of axis two (0.88), question 13 has the largest loading of axis one (-0.78), and question 12 has the smallest positive loading of axis one (0.02). All questions were weighted equally in the analysis (Table 15).

From table 18, principal component one (AX1) was influenced by many questions, so little variation in PFC score can be explained by a few questions. However, the most important questions for AX1 were questions 9, 10, 11, and 13. Principal component two (AX2) was mostly influenced by question 2. Thus, when sites are plotted in space defined by the first two principal components, the plot depicts a gradient in responses to questions 9, 10, 11, and 13 versus a gradient in response to question 2.

The Euclidean distance biplot of the first two PCA axes illustrates the position of questions and sites in relationship to the first two PC axes (Figure 18). Three distinct large linear clusters of points appeared, and PFC and the NF reach rating clusters were at opposite ends of the diagram along the larger linear clusters. Several clusters of reach rating PFC samples were apparent, as well as one cluster of reach rating NF samples. The linear clusters represent the responses to question 2 for each site, sites with an answer of "N/A" are in the topmost cluster, a response of "yes" in the middle cluster, and a response of "no" in the bottom cluster.

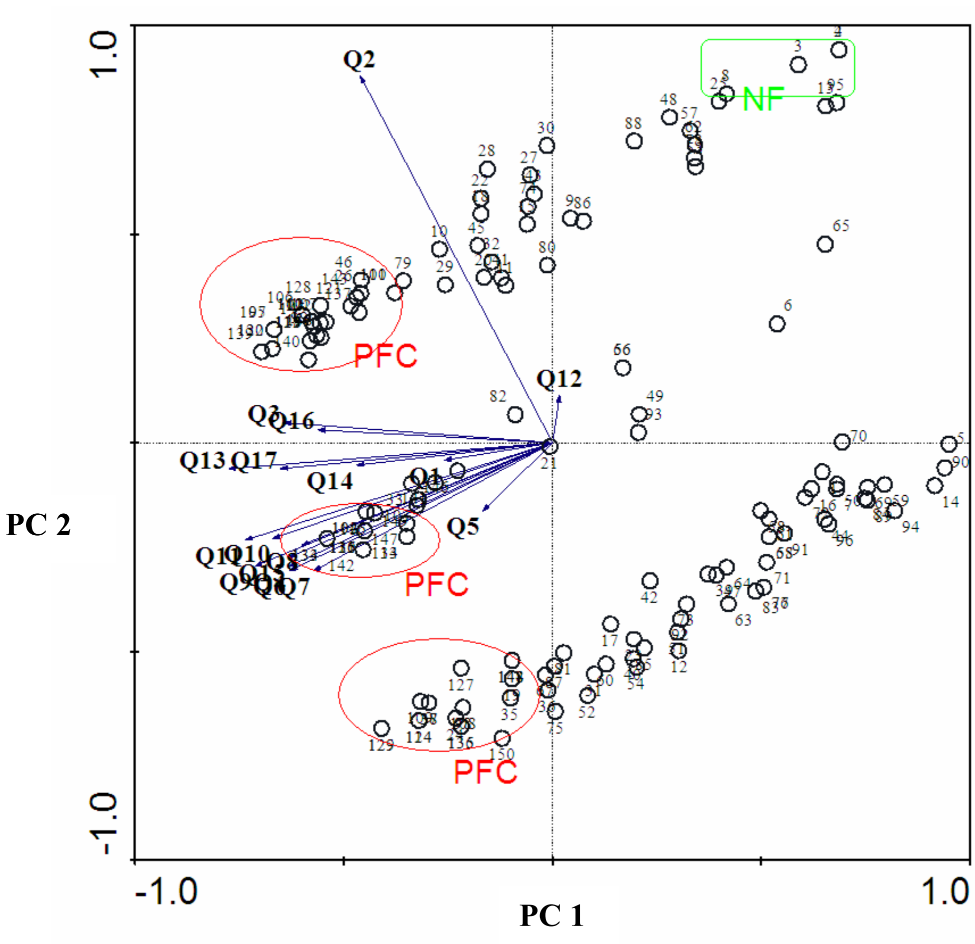

Figure 18. Euclidean distance biplot of the first two principal components, PC 1 (x-axis) and PC 2 (y-axis). Clusters of points are outlined and labeled by their reach rating. All unlabeled points had a reach rating of FAR, the most common reach rating. Each numbered point represents one site and each arrow represents one checklist question.

4.3 Objective 2B

Sites with incomplete corresponding field crew responses were eliminated from the analysis, with 119 sites having complete observations. There was statistically significant evidence from the questions tested that only questions 10 and 11 of the six questions tested were related to field observed characteristics (Table 16) at the 0.10

alpha-level. For Question 10, there was statistically significant evidence that the response to the yes/no question, “Riparian-wetland plants exhibit high vigor” was related to the sum of livestock droppings counted on two transects adjacent to the stream reach ($P = 0.02$).

For question 11, there was statistically significant evidence that the response to the yes/no question “Adequate riparian-wetland vegetative cover is present to protect banks and dissipate energy during high flows” was related to the field crew ranking of percent of streambank covered by deep, binding root masses ($P = 0.0003$), however, Chi-Squared values were suspect. There was also statistically significant evidence that the response to question 11 was related to the field crew ranking of percent covered by vegetation ($P = 0.06$), also with suspect Chi-Squared values. Additionally, there was statistically significant evidence that the response to question 11 was related to the sum of livestock droppings counted on two transects adjacent to the stream reach ($P = 0.03$).

All other questions paired with applicable field crew measurements or observations for analysis were not statistically significant at the 0.10 alpha- level.

Table 16. Comparison and results of PFC responses with field crew measured or observed characteristics. Significant probabilities are given in bold.

Standard Lotic Checklist question	Field-crew collected data	Test used	Chi-Squared (X^2) value	Parameter Estimate if applicable	P-value
4. "Riparian-wetland area is widening or has achieved potential extent"	Width/depth ratio as measured by field crew, averaged across 10 transects	Logistic Regression	0.12	0.776	> 0.10
5. "Upland watershed is not contributing to riparian-wetland degradation"	Field crew ranking (1-4) of "Erosional deposition from surrounding slopes"	Chi-Squared contingency analysis	1.21		> 0.10
9. "Streambank vegetation is comprised of those plants or plant communities that have root masses capable of withstanding high-streamflow events"	Field crew ranking (1-4) of "percent streambank covered by deep, binding root masses"	Chi-Squared contingency analysis	10.48		> 0.10
	Field crew ranking (1-4) of "Consumption of trees & shrubs by livestock"	Chi-Squared contingency analysis	10.57		> 0.10
	Count of number of droppings on two transects (sum)	Logistic Regression	1.76	0.011	> 0.10
10. "Riparian-wetland plants exhibit high vigor"	Field crew ranking (1-4) of "Consumption of trees & shrubs by livestock"	Chi-Squared contingency analysis	12.62		> 0.10
	Count of number of droppings on two transects (sum)	**Logistic Regression**	**5.66**	**0.020**	**0.02**
11. "Adequate riparian-wetland vegetative cover is present to protect banks and dissipate energy during high flows"	**Field crew ranking (1-4) of estimation of percent vegetative cover**	**Chi-Squared contingency analysis**	**16.46** * X^2 values suspect		**0.06**
	Field crew ranking (1-4) of "percent streambank covered by deep, binding root masses"	**Chi-Squared contingency analysis**	**24.27** * X^2 values suspect		**0.0003**
	Count of number of droppings on two transects (sum)	**Logistic Regression**	**4.56**	**0.018**	**0.03**
13. "Floodplain and channel characteristics are adequate to dissipate energy"	Width/depth ratio as measured by field crew, averaged across 10 transects	Logistic Regression	0.72	-1.185	> 0.10
	Stream slope as measured by field crew	Logistic Regression	0.03	-0.015	> 0.10

The PCA biplot (Figure 19) of the field crew observations and measurements illustrates that the number of cow droppings counted on two transects contributed the most to variation among sites. There were no clear clusters of points by reach rating on this graph. The cow droppings total had the only large PCA loading (Table 17). This measurement represents the first axis of the biplot, which explained 98.4% of the variance in the data. Stream slope had the strongest influence on axis two, which explained less than one percent of the variance in the data.

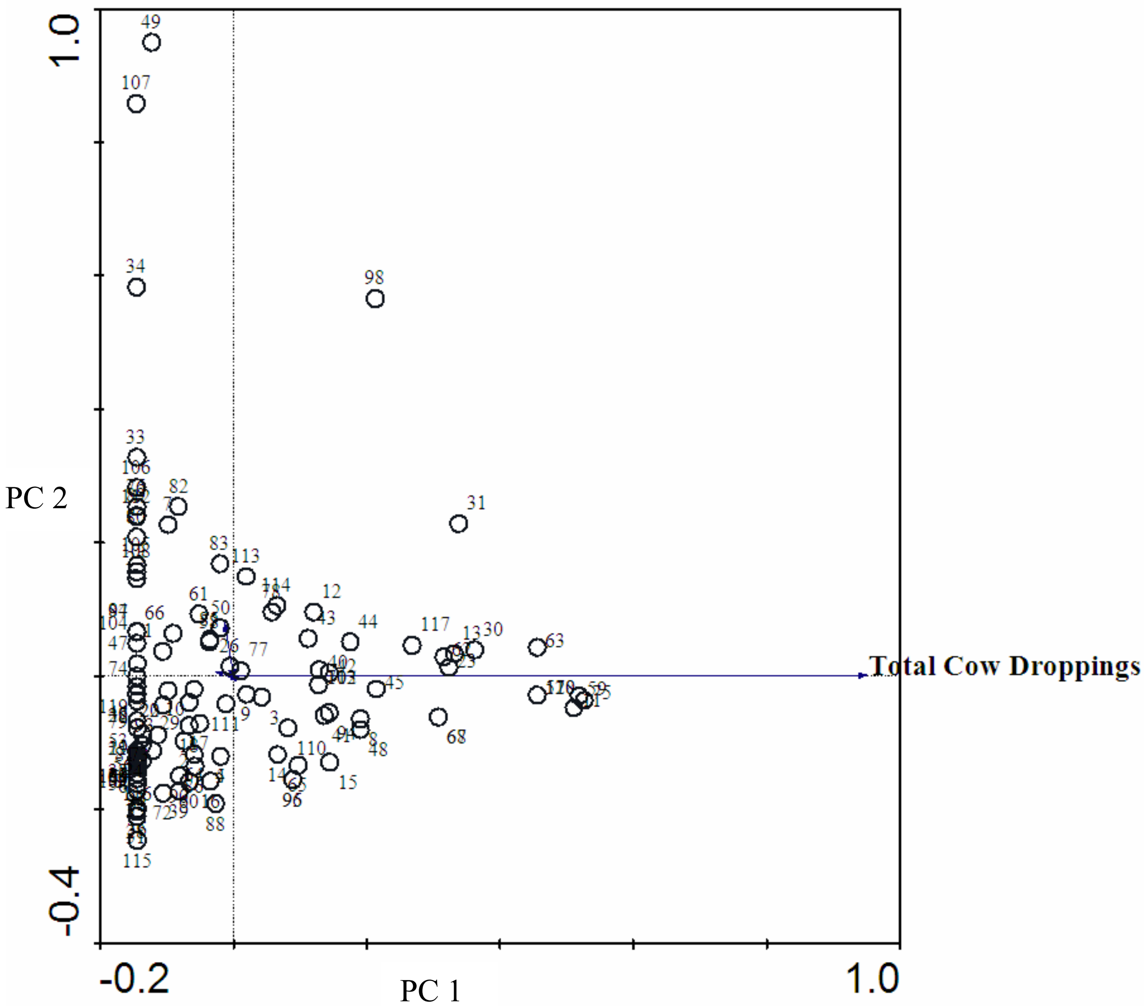

Figure 19. Biplot of the principal components (PC 1 and PC 2) for the analysis of field crew observations and measurements. There were no clear clusters of points by reach rating. Each numbered point represents one site.

Table 17. Principal Components loadings of the PCA analysis of field-crew measured characteristics.

Axis	AX1	AX2	AX3	WEIGHT
Eigenvalue	0.984	0.0072	0.0038	
Vegetative Cover	-0.0408	0.012	-0.0989	1
Erosion	-0.0204	0.026	-0.0552	1
Consumption	-0.0811	0.0199	-0.0596	1
Incisement	0.0061	0.0205	-0.01	1
Root Mass	-0.0126	0.0097	-0.1157	1
Cow total	**2.8039**	0.0042	-0.0038	1
Stream Slope	-0.0383	**0.2365**	0.0218	1
Width/Depth ratio	-0.0009	-0.0021	0.0005	1

Chapter 5: Discussion

5.1 Objective 1

There was no statistically significant evidence that the mean EPT abundances or EPT/Chironomidae ratios of sites with stressor centroids in their watersheds and sites without stressor centroids in their watersheds differed ($p > 0.10$).

A large body of literature supports the idea that landscape stressors in the contributing area to a sample point will affect the riparian condition, and ultimately, biota composition at that site (Beasley and Kneale 2002, García - Criado et al. 1999, Kauffman and Kreuger 1984, Maret et al. 2001, Marqués et al. 2003, Townsend et al. 1997). However, the relationships between biotic community structure and landscape stressors are extremely complex (Hawkins and Vinson 2000, Muhar and Jungwirth 1998). There also exists a wide range of biotic metrics for the researcher to test, with each metric having strengths and weaknesses in the amounts and types of stress measured by the biotic signals (Grigorovich and Angermeier 2004). Additionally, some sites are inherently more sensitive to impacts than others, particularly livestock grazing impacts (Chaney et al. 1993).

The lack of separation between sites with stressor centroids and sites without stressor centroids may indicate that using stressor centroids for polygon stressors is not relevant. It may potentially indicate that BLM management techniques of land use and riparian areas are effective in mitigating the effects of grazing and mining on stream communities. However, no statistical inference can be made in direct relationship to the BLM's management practices, because not all study sites fell on BLM lands. It is important to note that all stressor data were cataloged by the BLM, which may mean that some stressors on non-BLM land were omitted from the study.

There was slightly statistically significant evidence that in the Wyoming study area, the mean EPT value differed between sites within and sites outside of grazing allotments ($p = 0.05$) without the use of centroids. It is interesting to note that all sites within allotments had an EPT of 0. However, using linear regression, there was no statistically significant relationship between EPT and allotment acres or acres/AUM ($p > 0.10$). This may demonstrate that acres per AUM is unsuitable as an indicator of grazing pressure.

There was insufficient statistical power to detect a difference, and an increased sample size was needed, particularly of sites within allotments in the Wyoming study area.

There was also statistically significant evidence that the mean EPT abundances differed both between aquatic ecoregions and between states ($p < 0.001$, both tests). This may further the case that the presence or absence of stressor centroids within sample watersheds is not relevant in partitioning biotic variance.

The biological significance of the four distance measurements in characterizing stressor impacts could not be quantified using multiple linear regression. While travel time and flow length distances appear logically superior to straight-line and slope distances, this study was not able to assess the biological significance of these measures. The mean differences in distances measured using slope distance and flow length (5197 meters for grazing and 14951 meters for mining mean difference) indicate some degree of difference in the distance measurements. Grazing allotments occurred throughout the watersheds, and mining leases occurred in clusters, leading to a difference in mean distances between the two stressors. Additionally, when comparing distance measures to one another using linear regression, the greatest difference occurred between straight-line and travel time. There was almost no difference between straight-line and slope distance, and nearly equal differences between straight-line and flow length, and flow length and travel time. The lack of correlation (r^2=0.06) between straight-line distance, which is the simplest measure tested and also the least hydrologically relevant; and travel time, which is the most detailed distance measure tested and the most hydrologically relevant of those studied, may indicate that these distances and their biological relevance warrant further study.

The results of this study illustrate the need for a more controlled study approach. A controlled paired watershed study of the potential relationship of distance (including travel time) with biotic signals would also enable the researcher to compare travel time with other distances without the hindrance of trying to directly compare two distinct units of measurement (time and distance). Additionally, research should take Omernik's (1989) aquatic ecoregions for partitioning regional variances in EPT abundance.

Further study is necessary, with a more focused and controlled study approach. This study, in keeping with the goals of the AILC, attempted to use a large study area and also used pre-existing data and locations. Sample locations and times of collection were already present when this study began. The study was constrained to already existing data. When measuring landscape stressors as diverse and dynamic as cattle grazing, which can change from month to month, a smaller temporal scale, such as monthtly measurements of both grazing pressure and macroinvertebrate communities, may prove helpful. Cattle grazing impacts can also be extremely localized (Grigorovich and Angermeier 2004) so a focused spatial approach may also be helpful. Focused temporal testing of sites in a limited spatial area may yield useful data.

5.2 Objective 2A

Checklist questions 2, 3, 5, 11, 12, and 16 significantly affected the reach rating in the multiple logistic regression analysis (MLR).

From the PCA biplot (Figure 18), questions 3 and 16 are similar, indicated by the small angle of separation between the two arrows on the biplot. In the PFC assessment, questions 3 and 16 are not in the same groups of questions on the standard lotic checklist. Question 3 is in the erosion/deposition group and question 16 is in the hydrology group. Question 3, “Sinuosity, width/depth ratio, and gradient are in balance with the landscape setting” is designed to assess the dissipation of energy through the reach, assuming that when the reach is not in balance with its setting, rapid erosion will occur due to less dissipation of energy (Prichard et al. 1998). Question 16, “System is vertically stable” is designed to assess if the streambed is lowering rapidly as a result of erosion (Prichard et al. 1998). Both of these questions assess the stability and dissipation of energy through a reach. Question 3 addresses a broad view of reach stability and question 16 addresses an individual effect of reach stability. Question 3 also has a longer arrow than question 16 in the PCA diagram (Figure 18) indicating a stronger relationship with the data. It may be that question 3 is sufficient to assess reach stability for the reach rating.

Question 5, “Upland watershed is not contributing to riparian- wetland degradation,” had a p-value of 0.04 in the MLR. From a mosaic plot of the responses to question 5, it appears that there is little separation between FAR and NF with yes/no answers as would be expected (Figure 20), as the height of the cells in the mosaic plot for “no” and “yes” answers are similar. Additionally, in the PCA biplot (Figure 18), question

5 has a shorter arrow than most other questions, indicating a weaker relationship with the response data. The arrow is also separated from the other questions, indicating that question 5 is not highly correlated with the other questions. These additional analyses indicate that question 5, although significant in the MLR analysis, may be weaker in its correlation with reach rating than other questions significant in the MLR.

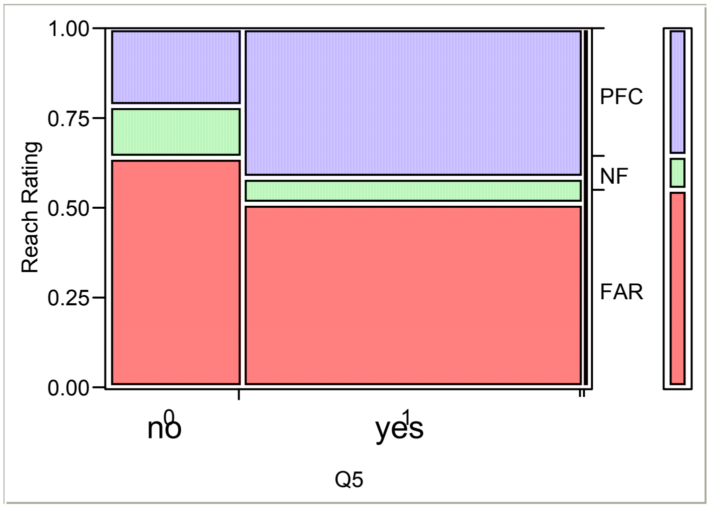

Figure 20. Mosaic plot of the dependence of PFC rating on the response to question 5 ($p > 0.10$). The mosaic plot represents the frequency of each combination of reach rating (Y) and answer to question 5 (X) in relationship to all other possible combinations by the proportional area of the cell.

Question 5 is the only standard lotic checklist question that is worded negatively "Upland watershed is not contributing to riparian- wetland degradation," which may affect the response of checklist users. The PFC assessment manual clearly points out this difference in the wording of question 5 (Prichard et al. 1998) , however, this may still contribute to the weak relationship between response to question 5 and reach ratings. It is also important to note that question 5 was not intended to assess the condition of uplands, but only to assess the contribution of uplands to any riparian degradation (Prichard et al. 1998). This may also be a concept of variable definition between PFC assessors. It may be difficult for assessors to rate only the riparian processes without incorporating upland condition into their responses, particularly when both the riparian and upland areas are in view of the assessor. Re-wording the question similarly to "The riparian-wetland area is positively influenced by upland conditions" would still allow for a "yes" answer in functioning reaches without a negatively worded question.

There were two questions significant in the MLR that had “N/A” response rates of over 20 percent. Question 2, “Active/stable beaver dams” and question 12, “Plant communities in the riparian area are an adequate source of coarse and/or large woody debris” both had high levels of “N/A” responses. However, both of these questions were statistically significant at the 0.05 alpha level in the MLR. Additionally, in the PCA biplot (Figure 18), questions 2 and 12 were both separated from the majority of the checklist questions as indicated by the angles between the arrows on the diagram. This indicates low correlation with other checklist questions, which may suggest that these questions assess unique characteristics of the reaches. In the PCA diagram, question 2 has the longest arrow of any checklist question, indicating a very strong relationship to the response data, and question 12 has the shortest arrow of any checklist question, indicating the weakest relationship to the response data. It should also be noted that question 2 is the only checklist question with more “no” and “N/A” than “yes” responses. All other questions had more “yes” than “no” responses. This difference may affect the importance of this question in the final reach rating dramatically and may contribute to its overwhelming dominance in the PCA biplot (Figure 18).

From the high levels of N/A response rates, it appears that these questions may not be universally applicable to diverse western rangelands. When applicable, they may be helpful in assessing reach rating, as illustrated by the MLR and PCA analysis results, but over 20 percent of the time, they are not applicable or present. Question 2, “Where beaver dams are present, they are active and stable” is only intended to be answered with a “yes” or “no” where beaver dams are present, and should be answered “N/A” where they are not present (Prichard et al. 1998). The high level of “N/A” responses to the question indicates that many sites do not have beaver dams. However, the question does appear to have a strong, unique relationship to the response data and appears to overwhelm the presence of all other questions when beaver dams are present (Figure 18), affirmed by its dominant loading of the second PCA axis, while its universal applicability to all sites appears weak. The PFC assessment manual acknowledges that question 12, “Plant communities in the riparian area are an adequate source of coarse and/or large woody debris,” will have high levels of “N/A” responses in the west. Coarse woody debris is only important to the functioning of those stream reaches in forested areas, and the majority of BLM western rangelands are not forested. It appears from the PCA analysis that question 12 has a unique but weak relationship with the response data.

Question 11 was significant in the MLR but was not correlated with any other significant questions and did not have high levels of N/A responses. Question 11, "adequate riparian-wetland cover is present to protect banks and dissipate energy during high flows, " is part of the vegetation group of questions. Of all the checklist questions pertaining to vegetation (Appendix A), question 11 appears to be the most general and directly related to the overall goal of the PFC assessment. For a reach to receive a rating of PFC, it must have the ability to withstand and recover from high flow events. The overall goal of the PFC assessment is to assess resilience and dissipation of flows. Question 11 directly addresses this issue for the vegetation segment of the assessment, and this may be why question 11 is not correlated with any other significant questions, significantly affects the reach ratings, and has one of the strongest relationships with the data of any checklist question as shown in the PCA biplot (Figure 18). Question 11 appears to be a vital facet of the PFC assessment.

Questions 1, 4, 6, 7, 8, 9, 10, 13, 14, 15, and 17 did not significantly affect reach rating at the 0.05 alpha level in the MLR analysis. It also appears from the PCA biplot (Figure 18) that all these questions are highly correlated with each other or with other significant questions. This may indicate a level of redundancy in the reach rating resulting from the question responses, or it may indicate that assessors are not taking these questions into account when assigning the final reach ratings. It was not possible with the data available for this study to determine if these results were due to redundancy or assessor use of the checklist in determining reach rating. The inclusion of a ranking, addition, or weighting system based on the current standard lotic checklist questions may address this issue by allowing assessors to take all checklist questions into account when assigning the reach ratings. A weighting system is already in use by the Idaho Falls BLM field office for a protocol that uses slightly different assessment questions, but the same principle could be applied to the PFC assessment.

5.3 Objective 2B

Standard lotic checklist questions 10 and 11 were found be statistically related to applicable field measured and observed characteristics.

Question 10, “Riparian plants exhibit high vigor,” was significantly related at the 0.10 alpha-level to the sum of livestock droppings counted along two transects, as was question 11, “Adequate riparian-wetland vegetative cover is present to protect banks and dissipate energy during high flows.” The parameter estimates for both questions were small and positive numbers (Q10 = 0.020, Q 11 = 0.018), indicating that as the number of cow droppings counted increases, the probability of question 10 or question 11 receiving an answer of “no” also increases, but at a small rate of change. Livestock grazing is known to reduce riparian vegetative cover through direct removal and trampling (Kauffman and Kreuger 1984). It is logical that indicators of livestock presence, damage, and consumption of vegetation would statistically correlate with riparian plant vigor and amount of cover. This correlation may indicate that both checklist questions 10 and11 are effective as binary responses within the PFC assessment to assess riparian damage due to livestock presence and livestock removal. It may also indicate these questions are truly applicable to diverse western rangelands.

For question 11, there was statistically significant evidence that the response to “Adequate riparian-wetland vegetative cover is present to protect banks and dissipate energy during high flows” was related to the field crew ranking of percent of streambank covered by deep, binding root masses. There was also statistically significant evidence that question 11 was related to the field crew estimation of percent vegetative cover in the riparian area (ranked 1-4). This indicates an agreement between PFC assessors and the independent field crew on the observation of vegetative cover. This may indicate that the streambank coverage for this checklist question is related to both the amount and type of vegetation within the reach, although the PFC assessment manual explains that this question is designed to assess only the amount of vegetation (Prichard et al. 1998). This relationship may also indicate that a binary response is sufficient to represent the amount of vegetation present, when compared to a four-category response used by the field crew. Additionally, the relationship between these two sets of observations may indicate that adequate riparian cover is a repeatable, easily observable, and universal stream reach characteristic.

It is important to note that the two contingency table analyses on question 11 had suspect chi-squared values. This may indicate that the results are unstable. However, there is statistically significant evidence that both question 10 and 11 are related to the count of cow droppings, tested by logistic regression, and chi-squared values were not suspect for these analyses.

Of the PFC questions with applicable field crew measured or observed stream reach characteristics, four of the six tested questions had no statistically significant relationship with field crew observations. There are several potential reasons for lack of correlation with these responses, including design of the study, design of field crew measurements, lack of regionalization, temporal differences between PFC and field crew observations, small sample size, and potential problems inherent in the PFC process.

The study design may not have accounted for all possible combinations of field crew measurements and PFC responses. Potential relationships were only explored when logical relationships between the PFC question of field measurement could be justified using the current knowledge base (Table 9) and description of techniques in the PFC assessment field manual (Prichard et al. 1998). Additionally, field crew measured and observed characteristics were not specifically designed for this study. The field crew measurements and observations were designed previous to this study as part of a more detailed macroinvertebrate study, and represented the only known data source sufficient to meet needs for this objective. If field crew observations had been specifically tailored to measure each aspect of the PFC assessment at the same sites, all 17 questions could have been studied in greater detail.

Some of the lack of statistical relationship between the two protocols may have been due to a need for regionalization of certain characteristics. For example, a stream gradient of 2% may have been considered “in balance” in response to question 3 in a mountainous area, but could be considered “out of balance” in an area with more gentle topography. Regionalization by ecoregion, topographic relief, or other large-scale characteristics may have resulted in more statistically significant relationships. However, in this study, regionalization would have resulted in sample sizes far too small for statistical analysis of these questions with the currently available data. A larger sample size would be necessary for an analysis of this type.

Additionally, the temporal differences between the year of PFC observation and field crew measurement may have been a factor. The average time in years between PFC and field crew observation was 5 years with a range of eleven years difference between the two site visits (Figure 21). A difference in years of negative one indicates that the PFC assessment was done after the field crew observations for that reach. There was little variation in the means of years difference among the checklist questions assessed.

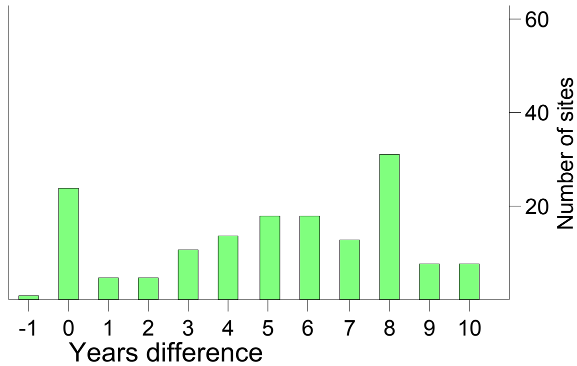

Figure 21. Distribution in years of difference between the field crew visit and PFC assessment for all sites used in objective 2B.

For sites with large differences between PFC and field crew observation, a more recent PFC would be helpful in strengthening the analysis.

There may also be some problems inherent in the PFC process. A qualitative ranking such as the PFC is relatively quick and easy to accomplish. However, PFC alone can not possibly provide all the information a land manager needs about the reach, and was not intended to do so (Prichard et al. 1998). Additionally, qualitative rankings may not always be applicable to field conditions. McQuaid and Norfleet (McQuaid and Norfleet 1999) found that in a paired watershed study testing several quantitative methods with a previously established, qualitative fish index of biotic integrity (IBI) that there was little statistical correlation between such methods with the IBI. Qualitative methods such as PFC have usefulness as screening procedures, but should not serve as the sole basis for determining stream health and subsequent management decisions.

Additionally, only questions 10 and 11 (out of questions 9, 10, 11, 13, and 2) varied substantially among sites (based on PCA in objective A) and showed statistically significant relations with empirical data. This indicates that the empirical basis for using most PFC questions to assess stream condition is weak. The ecological basis of PFC assessments needs to be studied in more detail. Without further scientific justification, reach ratings based on PFC assessments should be used very cautiously to inform management decisions.

There is currently no national structure within the BLM to catalog reach ratings and the subsequent restoration activities that are taking place on those reaches. Cataloguing and follow-up of reaches is typically the responsibility of the field office. There is typically no standardized information available within the BLM on how the results of reach ratings are being used in the field. According to the purpose of the PFC process, the reach ratings should be used to rank restorative potential. From that ranking, it is unknown what is done with the stream except for information kept within each field office. Furthermore, an area receiving a reach rating of PFC could still have questions with "no" answers, which indicate needs for restoration activity and improvement (Prichard et al. 1998). However, at first glance, the overall reach rating of PFC may lead to the conclusion that the reach is functioning properly in every aspect.

Chapter 6: Conclusions

6.1 Objective 1

A controlled experimental approach would best suit the further exploration of the biological significance of stream distance from stressor to sample measured in four different ways in a GIS. Literature strongly suggests a link between landscape stressors and biological community structure; however, this study was constrained by pre-existing data and wide spatial and temporal variation. An appropriate study would cover multiple spatial scales, both regional and local, and would incorporate temporal repetition. A well-designed study of the effects of distance, stressor type, and stressor intensity could contribute immensely to current knowledge about land management/stream health relationships. Such a study could be performed in various aquatic ecoregions that fall within the BLM's jurisdiction for improved regional applicability.

Additionally, the lack of separation between sites with stressor centroids and sites without stressor centroids may suggest that BLM management of mining and grazing activities is successful in relationship to stream health. However, a controlled study would also be necessary to validate this relationship.

The logical reasons for using flow path distance or travel time still remain. Straight-line distance and slope distance may assign stressors to the wrong watersheds if not carefully calculated, and are not hydrologically relevant. The use of travel time and/or flow length distance in assigning distances between stream and stressor within a GIS should be pursued.

6.2 Objective 2A

Standard lotic checklist questions 2, 3, 5, 11, 12, and 16 influence the overall PFC outcome, even without any formal ranking, averaging, or weighting and addition of the individual question responses by field observers. Question 11 appears to be very important to the reach rating. Consideration of the wording of question 5 may lead to a stronger relationship between this question and the checklist responses. The assessment of questions 2 and 12, which had over 20% of responses in the N/A category, for their broad-range applicability may also help to strengthen the PFC assessment. The abnormality of response rates to question 2, where there were more “no” and “N/A” than “yes” responses; rather than a higher number of “yes” than “no” answers as was present in all other questions, may indicate the need for clearer wording of the question. It may also indicate that question 2 is extremely influential in the final reach rating when beaver dams are present and not active or stable. Assessment of multiple questions that are correlated, such as questions 3 and 16, could result in a simpler and more robust procedure. Clarification of some checklist questions and the implementation of a weighting system may also result in more uniform influence of all checklist questions on reach ratings.

6.3 Objective 2B

Questions 10 and 11 on the PFC field checklist appear to be related to field measured characteristics. These questions may represent easily repeatable, sufficiently descriptive, and widely applicable parts of the PFC protocol. Both of these questions are related to vegetation of the stream reach. Additionally, question 11 had a strong, unique relationship to the data in objective 2A, further supporting the applicability and strength of this question in the PFC assessment.

The strong statistical relationship between questions 10 and 11 with the number of cow droppings counted on two transects along the stream reach may indicate that the number of cow droppings is a useful observation to record. This quantitative technique is repeatable, requires no specialized training, can be measured while on site for other reasons, and can provide insight for land managers of the grazing pressure and vegetative characteristics of a stream reach.

The lack of statistical correlation between other PFC questions and field measured or observed characteristics indicate that repeatable, applicable measurements of hydrologic and erosive properties of stream reaches may need further research. This research should consist of a regionalized analysis, a larger dataset, and more specific field measurement techniques if necessary. Until such research is completed, reach ratings based on PFC assessments should be viewed skeptically when used to inform management decisions. If no further relationships between PFC and quantitative or categorical data are found, the wording and/or broad-range applicability of these questions should be investigated.

Additionally, changes in the handling and use of the PFC protocol may assist in reaching the goals of the AILC. Field offices should be made aware that the PFC protocol alone may not provide a clear ranking of restoration potential and should be used as a screening technique only. Several field offices already practice this policy. A national database structure containing reach names, locations, and standard lotic checklist responses, restoration activities, and subsequent reach ratings would greatly assist in tracking the improvement of stream health at a national level. This database would also provide validity of the PFC protocol as applicable to diverse areas. Furthermore, field offices should be aware that reach ratings of PFC mean restorative activities may still be necessary.

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Appendix A: PFC Assessment checksheet

Standard Lotic Checksheet

Name of riparian-wetland area:___________________________

Date:_________ Hydro Unit/Segment ID: ____________ Quad: ___________________

Twnshp: ___________ Rng: _____ Sect: ______ Photos: _________________________

ID Team Observers: ____

Yes	No	N/A	HYDROLOGIC
			1) Floodplain inundated in "relatively frequent" events (1-2 years) rationale:
			2) Active/Stable Beaver Dams rationale:
			3) Sinuousity, width/depth ratio, and gradient are in balance with the landscape setting (i.e. landform, geology, and bioclimatic region) rationale:
			4) Riparian Zone is widening or has achieved potential extent rationale:
			5) Upland watershed not contributing to riparian degradation . rationale:
			VEGETATIVE
			6) Diverse age-class distribution (recruitment for maintenance/recovery) rationale:
			7) Diverse composition of vegetation (for maintenance/recovery) rationale:
			8) Species present indicate maintenance of riparian soil moisture characteristics rationale:
			9) Streambank vegetation is comprised of those plants or plant communities that have root masses capable of withstanding high stream flows. rationale:
			10) Riparian plants exhibit high vigor rationale:
			11) Adequate vegetative cover present to protect banks and dissipate energy during high flows rationale:
			12) Plant communities in the riparian area are an adequate source of coarse and/or large woody debris rationale:
			SOILS-EROSION DEPOSITION
			13) Flood plain and channel characteristics (i.e. rocks,

			overflow channels, coarse and/or large woody debris) is adequate to dissipate energy rationale:
			14) Point bars are revegetating rationale:
			15) Lateral stream movement is associated with natural sinuosity rationale:
			16) System is vertically stable rationale:
			17) Stream is in balance with the water and sediment being supplied by the watershed (i.e. no excessive erosion or deposition) rationale:

Summary Determination

Function Rating
Proper Functioning Condition ___
Functional At-risk ___
Nonfunctional ___
Unknown ___

Trend for functional at-risk:
Upward ___
Downward ___
Not Apparent ___

Rationale for rating:

Are factors contributing to unacceptable conditions outside BLM's control or management?

Appendix B: Stream sampling data sheet used by AILC

BLM BugLab Stream Assessment Data Sheet

Date: Crew:	Latitude:
Site Name:	Longitude:
Site ID:	Elevation(ft) Map= GPS=
District/Forest/Park:	Start time:
County:	State:
Reference site? Yes or No	Temp, Air: Water:

Site Evaluation	Score	
Vegetative Cover		4 = >95% 3 = 85-94% 2 = 75-84% 1 < 75%
Erosional deposition from surrounding slopes		4 = None 3 = Some in specific, limited locales 2 = Obvious signs 1 = Mass wasting
Consumption of trees & shrubs by livestock		4 = 0-5% 3 = 5-25% 2 = 25-50% 1 = >50%
Stream Incisement		4 = no incisement 3 = Old incisement 2 = Deep incisement, new floodplain developing 1 = Deep incisement, active down cutting
% Bank with lateral cutting		4 = < 5% 3 = 5-15% 2 = 15=35% 1 = > 35%
% streambank with deep, binding root masses		4 = > 85% 3 = 65-85% 2 = 35-64% 1 = <35%

Management Activities	Rank	Describe
Logging		Notes:
Agriculture		Notes:
Recreation		Notes:
Mining		Notes:
Roads		Notes:
Stream Diversion		Notes:
Urbanization		Notes:
Livestock Grazing		Notes:

Livestock use index: Number of fecal droppings	Left Transect:				Right transect:			
	Cow-old	Cow-new	Sheep-old	Sheep-new	Cow- old	Cow-new	Sheep-old	Sheep-new

Site Measurements					
Conductivity (S/cm)	P Alkalinity (ppm CaCO3)	Total Alkalinity (ppm CaCO3)	Stream Travel Time (s/50m) Lead/trail	Stream Slope (%)	Periphyton sample Volume (mL)
			/		
Channel Classification: Braided Regime Pool-Riffle Plane-Bed Step-pool Cascade Bedrock Colluvial					
Dominant Erosional Habitat Type: Rapid Riffle Run Steprun					
Dominant Depositional Habitat Type: Lateral Scour Plunge Dammed					
Photographs, Exposure #'s, Upstream : Downstream: Overview:					

	Mean Depth (cm): Mean width (m):									
Transect	1	2	3	4	5	6	7	8	9	10
Width (m)										
Depth (cm) at 0.25 width										
Depth (cm) at 0.5 width										
Depth (cm) at 0.75 width										

Mean =	Densiometer Measurements (number of points shaded of 96)				General Site Comments
Direction	Unit 1	Unit 2	Unit 3	Unit 4	
Upstream					
Left bank					
Right bank					
downstream					

		Stream Bed Particle Size Counts							
		Unit 1		Unit 2		Unit 3		Unit 4	
Particle size class (mm)	Totals	Tallies	Count	Tallies	Count	Tallies	Count	Tallies	Count
Bedrock									
180									
128									
90									
64									
45									
32									
22.6									
16									
11									
8									
<8									

Total # of particles counted: Median: