Chapter 5

Riparian Restoration through Grazing Management: Considerations for Monitoring Project Effectiveness

Alvin L. Medina, John N. Rinne
U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station
2500 South Pine Knoll Drive, Flagstaff, Arizona 86001, USA
almedina@fs.fed.us, jrinne@fs.fed.us

Philip Roni
Northwest Fisheries Science Center, National Marine Fisheries Service
2725 Montlake Boulevard East, Seattle, Washington 98112, USA
phil.roni@noaa.gov

Introduction

Many riparian areas throughout the United States were altered by the European settlement and the westward migration. Riparian areas currently comprise 1–5% of the landscape in the conterminous United States (Swift 1984; Knopf 1988), depending on the region. Their use is disproportionate to their relative extent and resource value. Swift (1984) estimated about 67 million riparian acres existed before European settlement. These areas were obligate settlement locations because of the presence of water. By the early twentieth century, the demand for surface water initiated a period of water resource development for irrigation, hydropower, and flood control. These early impacts were followed, midcentury, by mining (extraction) of subsurface waters to provide for rapidly developing metropolitan areas. Rivers and streams were dammed or diverted, and wetlands were drained, resulting in a drastic reduction in riparian habitats.

The settlement of rural areas included livestock grazing in riparian areas. The demand for beef increased as the population grew and as urban centers expanded in the late 1800s and early 1900s. Wild ungulates (e.g., bison *Bison bison*, elk *Cervus elaphus*, deer *Odocoileus* spp.) were a limited and unreliable food source for a developing nation; hence, livestock were extensively substituted. In the late 1800s, livestock in the West numbered in the millions and, coupled with drought, depleted vegetative resources across the landscape, including water-rich wetlands (Hendrickson and Minckley 1984) and riparian areas (Young 1998). Although the U.S. Forest Service recognized the need to control livestock numbers on public lands, not until the 1934 Taylor Grazing Act was livestock grazing management affected on public lands.

The grazing of public lands by livestock has been a highly contentious issue since the enactment of the National Environmental Policy Act of 1969. In the 1970s, a series of acts, including the Endangered Species Act and Public Rangelands Improvement Acts, further affected livestock management and reduced herd numbers across the West (Rinne and Medina 1996). Livestock numbers on western rangelands have been reduced by about half since the early part of the twentieth century (Medina and Rinne 1999), and considerable reductions were witnessed recently in response to drought and litigation over critical habitat issues. However, the relative decline in livestock numbers on rangelands has been replaced by an expansion of elk populations throughout western U.S. rangelands and Canada. Successful elk reintroductions in many western states have resulted in large herds with similar grazing effects of cattle. Elk ranching has even replaced cattle ranching in parts of the Rocky Mountain regions of North America. This change may satisfy political needs but does little for restoration of riparian habitats. Ungulate grazing issues will remain contentious for some time to come.

Although impacts to riparian and aquatic systems resulting from overgrazing are not limited to western North America, much research has focused on this region, primarily because of its large tracts of public and private rangelands used for grazing livestock (Belsky et al. 1999). Yet grazing in watersheds and riparian areas is an important issue throughout the world. Impacts of grazing on aquatic systems have been studied and reported in the eastern and midwestern United States (Wohl and Carline 1996; Weigel et al. 2000; Meals and Hopkins 2003), Europe (Diaz et al. 1996; Humphrey and Patterson 2000; O'Grady 2002), and Australia (Robertson and Rowling 2000; Jansen and Robertson 2001). Impacts to aquatic systems undoubtedly occur in other areas where humans use and manage livestock or ungulate populations. Impacts of grazing of riparian areas and uplands have been well documented (see Platts 1991 and Belsky et al. 1999 for thorough reviews). Grazing can alter natural riparian and channel processes (e.g., upland and streambank erosion, channel sedimentation and widening), increase stream temperatures, decrease water quality, change the water table and hydrologic regime, and, ultimately, affect aquatic biota (Elmore and Beschta 1987; Armour et al. 1991; Platts 1991; Belsky et al. 1999). Impacts on fishes and other biota are apparent but are more difficult to measure and are less well documented than physical changes (Platts 1991; Rinne 1999a).

Resource managers faced with the challenge of monitoring and mitigating grazing effects have devised many assessment models, such as the General Aquatic Wildlife Survey (GAWS) and the COWFISH habitat model used by the U.S. Forest Service (Lloyd 1985; USFS 1998), or the Rapid Bioassessment Habitat Evaluation Protocols or RBP (Plafkin et al. 1989) used by the Environmental Protection Agency (EPA). The RBPs and the qualitative habitat assessment approaches were developed as inexpensive screening tools for determining whether a stream is supporting a designated aquatic life use (Plafkin et al. 1989; Barbour and Stribling 1991, 1994; Rankin 1991, 1995). Barbour and Stribling (1991, 1994) modified the habitat assessment approach originally developed for the RBPs to include additional assessment parameters for high-gradient streams and a more appropriate parameter set for low-gradient streams. Habitat Evaluation Procedures (HEP) can be used to document the quality and the quantity of available habitat (USFWS 1980). Habitat Evaluation Procedures provide information for general instream and riparian habitat comparisons of the relative value of different areas at the same or future points in time. By combining the two types of comparisons, the impact of proposed or anticipated land- and water-use changes on instream and riparian habitat is quantified.

However, to date, there are no comprehensive models or methods for the assessment of ungulate grazing effects on fish habitats that have been validated across various ecosystems. For example, Contor and Platts (1991) evaluated COWFISH (Lloyd 1985) and determined that its capabilities were limited to qualitative estimations of stream or riparian health and, more importantly, that the model is based on vital assumptions of fish–terrestrial habitat linkages (Platts 1990). Deficiencies in habitat assessment methods have lead resource managers to develop and deploy many methodologies, most often with little validation but with honorable intentions, to meet monitoring requirements. The fundamental problem lies in establishing the direct linkages between fish and other aquatic biota health factors and the complex interactions of ungulates and many environmental variables (e.g., vegetation, hydrology, geomorphology, geology, climate). Many studies report negative interactions between fish and ungulates (Kauffman and Krueger 1984; Platts 1991; Fleischner 1994; Belsky et al. 1999; Rinne 1999a), but none have established direct linkages of cause and effect. This is not to deny that livestock impair various riparian functions when habitats are overused, but we emphasize that grazing is a function that can occur even in the absence of livestock, such as by various herbivores (e.g., elk, deer). While the assessment of direct effects often is difficult, it should be performed with objectivity and validated methods. For example, Medina and Steed (2002) found that elk have impacts similar to cattle on vegetation and stream channels. The impacts of grazing vary within and among ecoregions, depending on a suite of factors (e.g., geology, channel type, vegetation, ungulate species, elevation, hydrology). Some riparian areas can sustain little to no ungulate grazing, while others (e.g., Nebraska sedge *Carex nebraskensis* sites) can sustain very high use. However, a divergence from ecosystem science occurs when "ungulate" grazing is selectively applied to livestock, especially where habitats are managed for fisheries with critical habitat designations for threatened or endangered populations (Rinne 2003a). These fisheries and their associated riparian habitat may require some form of protection from grazing of all ungulates (e.g., elk, deer, cattle), as well as recreation, or other land uses for vegetation and channel recovery on selected reaches.

Numerous strategies to restore or to improve riparian areas and aquatic habitats impacted by grazing have been developed and implemented throughout North America and other parts of the world. Platts (1991) listed several innovative management strategies for addressing and corroborating livestock grazing impacts in riparian areas. While numerous and sometimes complex, all grazing strategies include control of livestock numbers, distribution, duration, timing of grazing, control of forage use, or some combination of these factors (Platts 1991). Strategies such as rest–rotation and seasonal use often require active management to periodically move livestock, reduce their numbers, or prevent them from grazing riparian areas. Fencing and complete grazing removal are the most common and potentially successful "grazing strategies" used to provide short- and long-term exclusion of ungulates and allow for riparian recovery. To ensure success, all grazing strategies should be approached in a context similar to a U.S. Forest Service multiple-use paradigm and in a multidisciplinary (e.g., fisheries, hydrology, botany, geology) frame of reference. We try to adhere to these frames of reference as we discuss and suggest monitoring needs and approaches.

Clearly, the success of various grazing strategies depends on thorough monitoring and evaluation to determine the effectiveness of different grazing reduction and removal strategies. In this chapter, we provide the reader with practical information on issues of monitoring riparian areas, with emphasis on fisheries. First, we provide an overview of considerations in designing a monitoring and evaluation program (e.g., questions and hypotheses, study design, and duration) and selecting useful monitoring parameters. Next, we present three grazing case studies wherein we describe the purpose of the study, problems and issues, methods, and what was measured and what was learned (results). Lastly, we synthesize general principles from the case studies that should apply for any monitoring program when addressing grazing and fencing in riparian areas. While several types of monitoring exist (status, compliance, effectiveness, validation, etc.), our discussion focuses on effectiveness monitoring and validation monitoring: determining whether the fencing or grazing strategy had the desired physical and biological effects.

Design Considerations for Monitoring Grazing Strategies

The development of a monitoring program to evaluate a grazing or a fencing strategy to restore or to improve stream and riparian conditions—similar to other types of habitat restoration and improvement—should follow several logical steps. These include determining project objectives and hypotheses, appropriate experimental design (e.g., scale, duration, replication), selecting appropriate parameters and sampling protocols and strategies, and implementing the monitoring program. Chapter 2 discusses these steps, as well as statistical considerations, in detail. Rather than reiterate those here, we briefly describe unique study design and parameter selection considerations for grazing and fencing projects.

Defining Project Objectives and Monitoring Hypotheses

One of the initial steps in developing any monitoring and evaluation program is to clearly articulate the specific objectives of the project (i.e., to allow for the recovery of herbaceous and woody riparian vegetation through the removal of grazing) and specific hypotheses. Without overarching goals and specific objectives, it will be difficult to determine key questions and phrase these as testable hypotheses the monitoring program will answer. Examples of key questions might be:

- What is the effect of removal of grazing on riparian vegetation species and growth over time?
- What is the effect of grazing removal on physical habitat (e.g., bank stability, fine sediment, channel type)?
- What is the effect of grazing removal on aquatic biota (e.g., fishes, macroinvertebrates)?

Articulating the project objectives and testable hypotheses sets the stage for subsequent steps in designing a monitoring program.

Study Design and Spatial and Temporal Considerations

Selecting an appropriate study design for monitoring a grazing strategy depends upon a number of factors, including the scale (reach or watershed) of the project or projects, duration, and replication. Many evaluations of grazing strategies have attempted to use a before–after (BA) or before–after control–impact (BACI) study design. Indeed, because grazing and fencing are treatments that can be relatively easily manipulated (i.e., fencing or changing livestock numbers), monitoring and evaluation of grazing and fencing projects lend themselves to BA and BACI designs. However, most studies that have used this design included little pretreatment monitoring, whereas conducting a thorough BA study requires considerable preproject planning and commitment of resources (Platts 1991; Rinne 1999a). Thus posttreatment designs with paired treatment and reference (control) reaches or watersheds with sampling immediately after treatment (implementation of new grazing strategy) are particularly common (e.g., Myers 1989; Wohl and Carline 1996; Clary 1999). However, finding sites with similar grazing strategies and physical features to serve as replicates and controls can be very difficult. Harrelson et al. (1994) and Downes et al. (2002) provide good criteria in selecting control reaches or streams. These typically include finding stream reaches with similar geology, channel type, substrate, vegetation, and more. Establishment of short- or long-term control over a study reach by using fencing or by excluding ungulates requires not only knowledge of technical aspects of a monitoring plan but also specific knowledge of other wildlife needs (Kie et al. 1994; Acorn 1997; Huedepohl 2000), livestock (Worley and Heusner 2000), fencing designs, costs and materials (BLM 1985; Craven and Hygnstrom 1996; De Calesta and Witmer 1996; Mayer 1999; Bekaert 2002), and environmental conditions (Hygnstrom et al. 1996). Retrospective or posttreatment studies can provide useful comparisons of physical and biological conditions in treated and untreated areas but typically require extensive replication to detect changes in fish abundance due to grazing or removal of grazing.

The spatial scale of the project also will help determine the type of study design. If a grazing project occurs along a short stream reach and changes are expected to be localized, a reach-scale approach may be needed. A grazing or fencing project that occurs along multiple reaches or in upland areas may require monitoring the entire watershed or subwatershed. In the absence of extensive preproject data, a reference or control reach or watershed is needed to account for variability not associated with the treatment (grazing or reduced grazing). As with evaluations of other types of restoration projects, one of the pitfalls of studies evaluating the effects of grazing strategies on instream conditions is ignoring watershed-scale effect (Kondolf 1993; Rinne 1999a). For example, both Myers and Swanson (1995) and Kondolf (1993) demonstrated that while riparian vegetation conditions improved along fenced treatment areas, channel width and sediment levels did not change due to upstream and upslope roads and grazing, which contributed sediment and altered hydrology in treatment reaches.

Measurements usually are made at sites, within reaches, along a stream course, or positioned within a watershed or catchment basin. A very basic consideration is the mobility of a response variable across these scales within the riparian corridor. Plants, streambanks, and channels are stationary at a site within a reach of a riparian-stream corridor. By contrast, fishes, invertebrates, substrates, and chemical factors are very dynamic and move through these spatial scales. That is, response variables transcend controls and treatments frequently positioned linearly within riparian corridors (Rinne 1999a, 2000). Based on these functions and processes, monitoring protocols, analyses, and interpretations must be markedly different for these two response groups. Fences preclude grazing impacts on the stationary response variables (e.g., vegetation and streambanks) within an exclosure (i.e., a site or a reach of stream). Outside exclosures, the effects on static (stationary) variables are directly linked and relatively easily measured and interpreted. In stark contrast, the mobile response variables (e.g. sediment, biota) are indirectly and independently linked to potential grazing impacts. Accordingly, their measurement, analyses, and interpretations are more complex and often are subject to misinterpretation. All study design and interpretations of results of study must be made within the context of the stream continuum (Vannote et al. 1980). To do otherwise may fail to obtain viable, defendable monitoring information that could benefit land managers.

Placing controls and treatments linearly upon a stream can lead to pseudoreplication (Hurlbert 1984). Scientific controls are difficult to achieve in the natural world because of habitat complexity and our inadequate knowledge of interactions (Likens 1984). Rinne (1999a) reported that a third of the grazing studies he examined in the literature had no controls and most of the remaining two-thirds were positioned linearly upon the same stream that contained treated (grazed) reaches. Using "control" reaches interspersed with treatments on the same stream to satisfy the lack of pretreatment data for the entire stream is problematic: streams are continuums, and treated areas can influence conditions in interposed control areas. Some of the studies reviewed by Rinne (1999a) were inconclusive because they did not consider channel type. Considering both the channel type and the positioning of treatments and controls is crucial when designing riparian monitoring.

Replication of study areas in time is another common problem with evaluation of fencing riparian areas for restoration (Rinne 1988, 1999a). Only half of studies examined in the literature reviewed by Rinne (1999a) were replicated in time. Temporal, pretreatment information in the target area to be restored is desirable, but usually lacking (Rinne 1999a, 2000). When possible, a minimum of 2–3 years of information and preferably 5–7 years is desirable before rehabilitative treatment (Rinne 1999a). The duration of monitoring realistically should extend to twice the minimum period of time (i.e., 4–6 years) or longer (e.g., 10 years) in an attempt to address natural variability. Studies on juvenile salmonids and other fishes suggest that more than 10 years (>5 before and 5 after) are needed to detect significant changes in fish abundance after habitat changes unless the magnitude of change in fish abundance is large (>threefold) or the treatments and controls are extensively replicated (Bisson et al. 1997; Roni et al. 2003). Decades of before or after monitoring may be needed to detect significant responses of anadromous salmonids (Bisson et al. 1997), likely because other factors, such as ocean conditions and migration, are affecting their survival and are increasing interannual variability. Estimating the duration of monitoring needed to detect statistical significance can be done if estimates of the variability of the parameter of interest are available (see Chapter 2 for examples and statistical texts such as Zar 1999 for details). Finally, the frequency of monitoring a response variable should be determined based on the ecology of biological entities and seasonal and climatic considerations for physical and chemical variables.

Selecting Monitoring Parameters

The potential parameters to monitor the response of riparian restoration through grazing management fall into three general categories: physical, chemical, and biological (Table 1). Livestock grazing has been suggested to affect changes in all three areas (Kauffman and Krueger 1984; Platts 1991; Rinne 1999a). However, too frequently, monetary and human resources will limit the feasibility of measuring all variables within the three general response areas. Selection of appropriate parameters to monitor in these three categories will depend upon the objectives and hypotheses of the grazing strategy and the monitoring program. For example, if the project objective is to restore riparian vegetation by fencing to exclude cattle and the hypothesis is that riparian vegetation will recover in x number of years after fencing, then one of the most obvious categories of parameters to monitor will be riparian vegetation. The vegetation component of riparian resources is linked most directly to the herbivory response variable. Accordingly, specific monitoring parameters for herbaceous and woody vegetation should become a priority. On the other hand, if the stream is the water supply for domestic use in a downstream municipality and the key questions or hypotheses relate to water quality, then the chemical responses, including concentrations of pollutants (e.g., *Escherichia coli*) and nutrients that potentially affect water quality, should be monitored. Finally, if threatened or endangered fishes occupy the stream and if the hypotheses regarding the effects of grazing management on fish are part of the monitoring program, then fish and fish habitat parameters should be part of the primary study focus of monitoring.

Unfortunately, selecting a single response variable normally results in more questions asked than answered at the completion of the monitoring activity. Notwithstanding that, one must begin with the key questions and hypotheses and must select response variables that can test these hypotheses. These variables should be the highest priority for monitoring. It also is important to consider whether variables will respond directly or indirectly to grazing management. For example, vegetation growth typically responds directly to grazing management, but stream shade and stream physical characteristics and biota respond indirectly (i.e., they depend on vegetation growth). Thus, indirect response variables need to be selected and linked (Rinne 1999a) to the primary response variable for greater definition and reliability of conclusions of monitoring. As we shall see, detecting changes in indirect response variables, which often are the objective of riparian restoration, can be difficult. Below, we discuss the major physical and biological parameters that often are monitored to determine the response of riparian and aquatic systems to fencing and other grazing strategies and provide some examples of their use (see also Table 1). We briefly discuss chemical parameters to monitor in the "other parameters" subsection. They are discussed in more detail in Chapter 9.

Physical Parameters

There are several key categories of parameters of physical instream and riparian variables that may respond to changes in grazing, including channel morphology, channel stability, sediment (fine and coarse), bank stability, stream flow, and water temperature.

Table 1.
Major categories and potential variables (parameters) to monitor to determine effectiveness of riparian grazing or fencing project. Which parameters are appropriate will depend on objectives of project and questions and hypotheses determined before initiating project and monitoring. Not all parameters are appropriate for every fencing or grazing project.

General categories	Common parameters monitored	References with protocols
Physical parameters		
Channel morphology	Channel type, longitudinal and cross sections,channel maps, aerial photographs, channel migration	Rosgen 1996; Montgomery and Buffington 1997
Bank stability	Rooting depth, vegetation cover, soil types, slope angles, soil moisture, bank erosion	Pfankuch 1975, Platts et al. 1987, Myers 1989, Bauer and Burton 1993, Schuett-Hames et al. 1994, FISRWG 1998; Newton et al. 1998; Casagli et al. 1999, Rinaldi and Casagli 1999, Simon et al. 1999; Burton and Cowley 2002; Simon and Collison 2002
Channel stability	Channel substrates	Rosgen 1996, 2001b; Montgomery and Buffington 1997
Hydrology	Flow, velocity	Dunne and Leopold 1978; Gordon et al. 1992; Harrelson et al. 1994
Substrates	Embeddedness, fine sediment levels, particle size, subsurface analysis	Gangmark and Bakkala 1958; Schuett-Hameset al. 1994; Bevenger and King 1995; Waters 1995; Bunte and Abt 2001; Sylte and Fischenich 2002; see also Chapter 3
Temperature	Daily continuous, maximum and minimum	Newton et al. 1998
Chemical parameters		
Nutrients	Cations (Ca, Fe, Mg, Mn), Nutrients (NO_2, e.g., NO_3, PO_4), pollutants (e.g. sulfates)	NRCS 1996, 1999; Newton et al. 1998; Clesceri et al. 1999; see also Chapter 9
Water quality	Dissolved oxygen, pH, alkalinity, conductivity, hardness, salinity	Newton et al. 1998; Clersceri et al. 1999; NRCS 1999
Biological parameters		
Vegetation	Cover, frequency, density, composition, production, utilization	Elzinga et al. 1998; BLM 1999a, 1999b; Clary and Leininger 2000
Fish	Abundance, survival; species, length, age composition	Platts et al. 1983; Murphy and Willis 1996; Barbour et al. 1999; Moulton et al. 2002; see also Chapter 8
Macroinvertebrates	Abundance, diversity functional feeding groups,various metrics and indices of integrity	Platts et al. 1983; Rosenburg and Resh 1993; Barbour et al. 1999; Karr and Chu 1999; Moulton et al. 2002
Other biota	Primary productivity (periphyton, algae, etc.), aquatic plants	Barbour et al. 1999; Moulton et al. 2002; see also Chapters 6 and 9

Channel morphology and complexity

Channel type, geometry, width, sinuosity, movement, and channel units (i.e., pools and riffles) frequently are monitored to demonstrate changes due to various riparian and grazing strategies. For example, channels may narrow after removal of grazing, growth of vegetation, and stabilization of banks; several studies have demonstrated this by measuring cross sections or stream widths (Platts 1991; Clary 1999; Myers and Swanson 1995).

However, other factors, such as sediment supply and streamflow, have a greater potential to affect channel morphology and complexity (see sediment subsection below and the case studies for further discussion). Habitat units or reaches of a stream with a similar nature (i.e., gradient, sinuosity, substrate) should be used cautiously when evaluating the effectiveness of grazing management strategies. We concur with Poole et al. (1997) that instream habitat unit classifications (i.e., pools, riffles, glides) should not be used to quantify and monitor aquatic habitat and channel morphology response to grazing management or riparian restoration, owing to subjectivity, lack of validation, and statistical limitations. As indicted previously, considering or stratifying reaches by channel type is a critical component of selecting study reaches. Details on methods for monitoring channel morphology are provided in Chapters 3 and 8.

Sediment

Grazing generally is thought to increase levels of fine sediments and to decrease the size of bed materials in stream channels (Platts 1991). The ecology of aquatic biota such as fishes are intimately linked to substrates they occupy for breeding, feeding, and cover (Rinne and Stefferud 1996; Rinne and Deason 2000; Rinne 2001a, 2001b). Sediment most often is considered synonymous with "fines" or materials of less than a millimeter or two in size. Parent geology, location within a watershed, and hydrology are major factors influencing fine sediment and substrate size within a stream reach. A number of methods exist for measuring substrate size. The zigzag transect method of Bevenger and King (1995) provides an efficient, reliable method of assessing change in the surface nature of bedload composition, because it responds to impacts on a watershed or within the riparian area proper. More detailed methodologies are available for examining surface and subsurface sediments or substrates (Everest et al. 1987; Chapman 1988). Typically, in these types of studies, the focus is on the levels of fine sediment in redds and the levels potential effects on salmonid egg development and survival. For example, Rinne (1988) used a standpipe (Gangmark and Bakkala 1958) driven into the substrate to determine substrate permeability in one montane stream subjected to grazing removal. However, aside from salmonids, most cypriniforms, and many other species, are nest builders or surficial substrate spawners or broadcast spawners. The pebble count methodology of Bevenger and King (1995) or a similar method can assess changes in substrates that would, in turn, affect these spawning behaviors. Given that a stream is a continuum, sediment impacts from a grazed area above an ungrazed area could affect the substrate composition bedload through the ungrazed area. Additional details on sampling fine and coarse sediment is provided in Chapter 3.

Major off-site disturbances may cause excessive sedimentation changes within the channel, such as to increase substrate embeddedness. Various techniques for measuring substrate embeddedness have been developed (Platts et al. 1987; Bain and Stevenson 1999). Sylte and Fischenich (2002) provide an in-depth review of seven methods, definitions, and various rating systems, and of the significance of embeddedness to fisheries. Despite many methods, there are fundamental problems with them, including wide disparity in methodologies, fundamental defects, and poor guidance criteria (Sylte and Fischenich 2002). The use of existing embeddedness measures is not recommended for monitoring grazing impacts on streams, because the results are unreliable.

As indicated previously, subsurface analysis methods sometimes are used to determine the composition of spawning substrates, especially where fine sediments are of concern. The McNeil sampler (McNeil and Ahnell 1964) is an instrument commonly used in salmonid monitoring studies in Washington State (Schuett-Hames et al. 1994). The "shovel method" also has been used with some success (Grost et al. 1991). A comprehensive review of particle-size analysis and sampling is provided by Bunte and Abt (2001) and also is discussed in Chapter 3 and will not be covered here.

Channel and bank stability

Bed and channel stability generally is a prerequisite to streambank stability (FISRWG 1998). The complex hydrological interactions of streamflow, erosion, and sediment can act upon the channel to cause instability (Rosgen 2001c), though channel movement and migration are natural stream processes and occur at higher rates in some channels types and geologies than others. Channel instability can lead to major shifts in lateral migration of the channel but is a natural phenomenon that can occur gradually, as in the erosion of meander bends of low-gradient meadow streams. While channel avulsions occur during bank-full events, their scouring effects can be magnified by flood flows, excessive sedimentation, debris jams, or mass wasting (Keller and Swanson 1979). Stream types with wide valley floors and low entrenchment ratios typical of Rosgen (1996) types E and C would be most susceptible.

Streambank integrity and stability are principal concerns of riparian monitoring for grazing impacts, because the streambank is the zone where ungulates can exert their greatest physical impact. Since the early 1970s, several methods were devised to estimate the relative stability of streambanks (Pfankuch 1975; Platts et al. 1987; Myers 1989; Bauer and Burton 1993; Schuett-Hames et al. 1994; FISRWG 1998; Burton and Cowley 2002). The basis for most methods is founded on the premise that streambank integrity is a direct factor affecting fish populations through sequential alteration of streambanks, channel stability, and riparian habitat. The term "integrity" connotes the use of a measure that integrates, at a minimum, specific types of herbaceous and woody vegetation, the soil type, the hydrologic regime, and geomorphic conditions. However, because of the complex interactions between physical and biological factors, to date, there is no one single approach that accurately provides a measure of streambank stability. Qualitative approaches rely on visual estimates of cover of aboveground biomass, root density and rooting depth, and relative bank erosion rates to estimate stability. Quantitative approaches apply physical laws (e.g., force and resistance) to estimate bank instability (FISRWG 1998). Hybrid models may incorporate both qualitative and quantitative aspects of biological and physical factors. Most streambank stability assessment methods apply to low-gradient meadow habitats, where ungulates tend to congregate, because higher-gradient habitats generally are stabilized by rock–boulder substrates. The type of vegetation that occupies the streambank is linked to elements for measuring bank stability (Rosgen 1996; Clary 1999). Many stream types are morphologically dependent on herbaceous vegetation (e.g., sedges; Medina 1995), deeply rooted woody species (e.g., willows), or large and coarse woody debris (Rosgen 1996; Montgomery and Buffington 1997).

Newton et al. (1998) provide a typical example of a qualitative streambank stability rating protocol (Table 2). Four categories of condition are delineated based on bank height, percent of exposed bank actively eroding, roots, and stability of overhanging woody vegetation. Platts et al. (1983) suggested rating streambank erosion and vegetative stability separately. Examples of modifications to previous models or hybrid models are Johnson et al. (1998), Rosgen (1996), and others. For example, Rosgen (1996) developed a hybrid model that derives an erodibility hazard index for streambank assessment that incorporates bank heights, angles, materials, presence of layers, rooting depth and density, and percent bank protection. The index, coupled with calculated near-bank stresses, is used to develop a quantitative prediction model of streambank erosion rates

Table 2.
Descriptions of qualitative bank stability criteria and ratings (modified from Newton et al. 1998). Numeric values at bottom row indicate rating on a scale of 1–10 that fit the narrative descriptions. Intermediate numeric scores would be given to bank stability that falls between the qualitative verbal descriptions.

Banks are stable	Moderately stable	Moderately unstable	Unstable
Banks are low (at elevation of active floodplain); 33% or more of eroding surface area of banks in outside bends is protected by roots that extend to the base-flow elevation.	Banks are low (at elevation of active floodplain); less than 33% of eroding surface area of banks in outside bends is protected by roots that extend to the base-flow elevation.	Banks may be low but typically are high (flooding occurs 1 year out of 5 or less frequently); outside bends are actively eroding (overhanging vegetation at top of bank, some mature trees falling into steam annually, some slope failures apparent).	Banks may be low but typically are high; some straight reaches and inside edges of bends are actively eroding, as well as outside bends (overhanging vegetation at top of bare bank, numerous mature trees falling into stream annually, numerous slope failures apparent).
10	7	3	1

(Rosgen 2001a). Platts et al. (1983, 1987) presented hybrid approaches in which various attributes of the streambank (e.g., soil alteration, vegetative stability, streambank undercut, and channel-bank angle) were rated and measured to obtain estimates of bank stability. Intensive stream stability assessment methods such as those developed by Rosgen (2001b, 2001c), or other similar intensive approaches, are very useful for monitoring riparian management but require hydrological expertise. Examples of quantitative estimations of bank stability include those of Casagli et al. (1999), Rinaldi and Casagli (1999), Simon et al. (1999), and Simon and Collison (2002). These models examine the mechanics of soil and water interactions that result in the failure of streambanks. Simon and Collison (2002) have developed an interactive software model that examines bank geometry material, root strength, pore pressure, and other factors to assist in the quantitative monitoring of streambank stability. Refinements to traditional streambank monitoring protocols to increase consistency and reliability are described by Burton and Cowley (2002) and have shown promising results for monitoring streambank alteration. Their protocol strives for objectivity and consistency, with warnings that appropriate training to understand and identify streambank alteration processes is essential.

Stream flow

The hydrologic regime of a stream has a direct impact on riparian areas, stream channels, and the biota that inhabit them. Differences in flow regimes may have marked effects on fish assemblages and their sustainability (Rinne 2002). For example, in the Southwest, many of the native, mostly threatened and endangered, fishes depend upon elevated flow or flood events (Minckley and Meffe 1987; Rinne and Stefferud 1997; Rinne 2002). The antithesis of elevated flow is base or sustained surface flow (Neary and Rinne 1997). The removal of water from streams and aquifers can effectively and completely override any grazing reduction or other riparian restoration efforts (Neary and Rinne 2001; Rinne 2002).

Monitoring flow is essential to evaluation of grazing activities and of a stream restoration. At a fine scale, characteristics of the water column, such as width, depth, the ratio of the two, and velocity, can be monitored (Platts et al. 1983, 1987). Because of characteristic habitat use or selection by aquatic biota (Rinne and Stefferud 1996), changes in these flow attributes may result from riparian restoration and may explain presence, absence, or abundance of respective species. Changes in ratios of pool to riffle habitat can markedly affect reduction in numbers or complete disappearance of some species. Changes in water column characteristics and dynamics ultimately may change fish assemblages (Rinne et al. 1998; Rinne 1999b).

Common measures for quantifying flow include staff gauges, stream gauges, and flowmeters, described in Chapter 6. In larger rivers, U.S. Geological Survey stream gauging stations are available and provide invaluable resources to define base and peak flows and variability of the system. Unfortunately, only a small portion of streams have stream gauging stations.

Temperature

Water temperature of streams varies seasonally and diurnally. In rivers and streams in the Southwest, water temperatures can vary 10°C in a 24-h period (Rinne et al. 2002). Measurement of water temperature is relatively straightforward, given the availability of relatively inexpensive data loggers that can be placed throughout the stream network and programmed to record temperatures at desired time intervals (daily, hourly, etc.) and for several months at a time. Stream water temperatures are affected by climate, ambient air temperatures, subsurface flow, springs, topography, and exposure. The variability of stream temperature and the multiple factors that can affect stream temperature make this parameter problematic for monitoring changes in grazing. In the context of grazing management and riparian restoration, stream exposure to solar radiation is most often of concern. Water temperature, indeed, can be affected by streamside vegetation or channel morphology that affects the surface area of the water. However, because of the complicating factors listed above, it is difficult to delimit with precision the relative effects of either stream shading by vegetation or relative stream surface-area exposure as related to stream channel morphology. Some researchers argue for streamside vegetation (Betscha 1997) as a controlling factor of water temperature. Others (Larson and Larson 1996) suggest that temperatures in streams are influenced by ambient temperatures and surface exposure as affected by channel morphology. Clearly, water temperature is affected by a suite of factors, including water sources (i.e., groundwater, snowmelt, surface runoff) stream size, channel morphology, geology (alluvial versus bedrock), vegetation type, and riparian canopy. Whether this parameter responds to reduction or removal of grazing in a particular stream or stream reach may not be as straightforward as many researchers contend.

The thermal limits of aquatic biota also should be considered when measuring water temperature. Cold versus cool or warmwater species will respond differently to changes in temperature and may help explain changes in the fish community associated with removal or change in grazing management. Rinne et al. (2002) demonstrated that several species of cypriniforms in one Southwest desert river sustain markedly elevated heart rates with increasing water temperatures. Two points that monitoring programs need to embrace are that (1) most often, specific information on thermal tolerances of fishes is not available; and (2) because of seasonal, diel, and diurnal variations of water temperatures in rivers and streams, a wide range of temperature tolerances by species should be expected. These ranges may be broader than those achievable by streamside vegetation restoration and its effect on water temperatures. Further, the common sectioning or partitioning of stream reaches by fencing and varying the grazing strategies characteristic of many studies (Rinne 1999a) results in temperature sinks and sources in the stream continuum, one potentially canceling the influence of the other.

In summary, water temperature can be an important parameter to monitor, but the results need to be interpreted with caution because a number of factors outside the study area may influence temperature. This emphasizes the need for temperature monitoring beyond a reach scale to tease out effects of grazing and fencing activities from other factors that may have a substantial influence on stream temperatures. Linking changes in temperature resulting from grazing management to changes in fish populations is, indeed, more problematic. The continual mixing of water, the array of factors influencing temperatures within a reach or habitat, the frequent lack of thermal data for fishes, and their probable wide range of tolerances of different fish species can confound linking changes in temperature to changes in vegetation and, ultimately, in grazing.

Biological Parameters

Many biological parameters are used to monitor direct and indirect changes in terrestrial and aquatic habitats in response to ungulate grazing, including herbaceous and woody vegetation, fishes, macroinvertebrates, periphyton, and other benthos components.

Vegetation

Streambank vegetation is the most common parameter used in monitoring of ungulate grazing. It is expected that presence, reduction or absence of ungulate foraging, or trampling of streamside habitats will invoke a change in the vegetation community, and such changes can be quantified by monitoring various attributes of the vegetation. Quantitative, semiquantitative, and qualitative approaches can be used, depending on project constraints. Several vegetation attributes should be measured to provide a better estimation of grazing effects on streambanks, but six principal attributes are cover, frequency, density, composition, structure, and production or utilization.

Many quantitative methods are available for measuring changes in vegetation attributes, including frequency, dry weight rank, Daubenmire, line intercept, step-point, point-intercept, cover board, density, double-weight, harvest, comparative yield, visual obstruction (robel pole), and other methods. The applicability and description of quantitative methods that measure these attributes are described in detail in BLM (1999a, 1999b). Additional information on general vegetation monitoring can be found in Kent and Coker (1994) and Elzinga et al. (1998). A quantitative method used to specifically monitor changes in streamside habitats, commonly known as the greenline method, is described in Winward (2000). This method measures the first perennial vegetation that forms a lineal grouping of community types on or near the waters' edge (i.e., the "greenline") typically slightly below bank-full stage, an important area for determining effects of grazing activities on streambank stability. Winward (2000) also describes methods for measuring vegetation cross-section composition and woody species generation. Similarly, the U.S. Forest Service is developing a quantitative methodology for detecting ungulate effects on streambank vegetation (Medina, unpublished). The protocol differs from the greenline method in that (1) aquatic and terrestrial plant- and ground-cover attributes are measured at the land–water interface, (2) vegetation transects follow the contour of the water interface, and (3) it yields repeatable quantitative estimates of plant and ground cover. The method also yields information on species frequency and composition. Many aquatic plants function as colonizers of the streambank interface zone and, as such, are important indicators of trend and condition (Medina 1995). The method is flexible to accommodate other parameters of interest (e.g., trampling, geomorphology) that can be used in combination to estimate ungulate impacts, as well as to provide measures of aquatic plant attributes. Quantitative methods are recommended where endangered species or litigative issues are present.

Similarly, semiquantitative and qualitative methods commonly are used in habitat assessments and generally are built into habitat assessment models (e.g., GAWS, COWFISH). For example, Platts et al. (1987) described a qualitative method to assess stream cover and habitat conditions by using a ranking approach. These qualitative approaches often are favored because they require considerably less field time and no technical botanical expertise. They may be adequate as diagnostic tools to identify potential problems and trends but typically are not good indicators for long-term monitoring.

The measurement of any single vegetation attribute (e.g., vegetative cover) is insufficient to estimate grazing effects, despite the fact that vegetative cover has been linked to salmonid abundance in many studies (Platts 1991). The type of plant cover, whether herbaceous, woody, or aquatic, yields important information about the habitat type and its functional state. For example, herbaceous cover provided by dense sedge stands is functionally different from woody plant cover for the same range site. In the latter case, the woody component may be indicative of a disturbed habitat condition versus a stable high-successional stage. However, the woody condition may provide better cover to ameliorate water temperatures. Frequency and density attributes are used to describe relative abundance of plant species or groups. Species composition provides a description of the individual plants that comprise the site. All these measures collectively provide evidence of ungulate effects. The interpretation of these attributes into causal effects requires technical knowledge of individual plant and community functions. For example, plant cover may remain constant, but plant composition may change from a sedge type to mesic species (e.g., Kentucky bluegrass *Poa pratensis*), a condition indicative of a general loss in soil moisture through the soil profile, which may be in response to a hydrological change exclusive of ungulates, or a combination thereof. The combination of vegetation attributes also can aid in interpretations of successional dynamics, functionality, trend, and disturbance regime, or to quantify and describe riparian habitat types. Additional information about the habitat type may be derived from vegetation attributes that measure plant community structure (i.e., diameter, height). Platts et al. (1983) and others have suggested that fish productivity can be determined from the type and diversity of habitat types of a given stream.

Various methods are available for measuring the production and utilization of biomass (e.g., browse removal, stubble height, residue measuring, herbaceous removal, landscape appearance) on riparian areas (BLM 1999a) and are used in conjunction with other vegetation attributes to assess grazing impacts. The assessment of production or utilization can require extensive time and effort, especially if causal factors are sought. There are several considerations (e.g., seasonal and annual effects, differences due to methods and observers) that should be weighed prior to selecting a specific method (Krueger 1998). The stubble height method has been advocated for use in riparian areas by Clary and Leininger (2000), who recommend a streamside stubble height of 10 cm as near optimal in many situations. They further suggest that the recommended height be adjusted to meet site conditions, such as increasing the height to 15–20 cm where willow browsing is of concern. The criterion is suggested primarily for small streams or sensitive streambanks, not for dry meadows or other similar sites. Users are cautioned not to emphasize the method as the management goal; rather, the resource manager should have a clear picture of the ecological structure and function of the area before settling on a specific height.

In summary, the use of vegetation attributes to assess grazing impacts on riparian areas requires measurement of several attributes and careful interpretation of the results. Vegetation changes may result from a variety of intrinsic (e.g., community dynamics) and extrinsic factors (e.g., ungulates, floods, drought, disease). As such, it is important to at least control for habitat or vegetation type, channel type, seasonal influences on vegetative production, ungulate class, and methodology.

Fishes

Like many restoration and habitat improvement techniques, the objective of grazing and fencing projects often are to increase fish abundance and survival. Monitoring of fish often includes examining fish size, age, abundance, and survival at various life stages, as well as species composition and diversity. Monitoring riparian-stream restoration projects for fish response must be based on the ecology and basic biology of the species. Most of the information on effects of grazing and for restoration activities, in general, is based on studies of salmonid fishes (Rinne 1988; Medina and Rinne 1999; Roni et al. 2002). In regions other than the Pacific Northwest and northern Rockies, salmon and trout are not the dominant species taxonomically, politically, or economically. For example, in the Southwest, there are about 40 native species of fishes, only 3 of which are salmonids. Although some general principles may apply to fishes as a group, the biology and ecology of most of the cypriniform species in the Southwest, and we suggest elsewhere in North America, is highly specialized and differs significantly from that of salmonids. To adopt a "one size fits all" approach may be expedient for managers, but it has a high probability of being detrimental to sustainability of many native cypriniform species. In a context of restoration of riparian-stream areas, the antithesis, namely destruction, extirpation, and possible extinction of a native species of fish or their assemblages, may be the end result because alien fish species are favored by vegetation changes (see case studies below). Many of the cypriniform species in the West (Minckley and Deacon 1991; Rinne and Minckley 1991; Rinne 1999a) and elsewhere throughout the United States (Williams et al. 1989) are threatened, endangered, or of special concern. Others are candidate and sensitive species (Rinne 2003a). Therefore, it is necessary to design monitoring protocols relative to the biology and ecology of the species of interest.

An equally important consideration to the salmonid versus nonsalmonid issue is that of native versus nonnative (alien) fish species. Introduction of nonnative species of fishes is cosmopolitan, with more than 500 nonindigenous species documented throughout inland waters of the United States (Fuller et al. 1999). For example, more than 100 species have been introduced into the waters of Arizona since 1890 (Rinne 1995; in press), and almost half have become established, self-sustaining populations. Through mechanisms of competition, displacement, and direct predation by nonnatives, native fish species have declined markedly. In restoration projects, the impact of nonnative fish species on natives may be more direct and negative than the positive influences of riparian restoration. Further, changes affected by restoration (e.g., instream vegetation increase and streambank changes) may favor nonnative fishes over natives (Rinne and Neary 1997).

Finally, one may approach fish monitoring in restoration projects from the "guild" or fish assemblage context. For example, in the Southwest, there are native "cryptic species," such as roundtail chub *Gila robusta*, Colorado pikeminnow *Ptychocheilus lucius*, Apache trout *Oncorhynchus apache*, Gila trout *O. gilae*, and Rio Grande cutthroat trout *O. clarki virginalis*, which are predators and feed in open waters but spend much of their life in deep pools or under banks and woody debris. There are "pelagic water" species, such as spikedace *Meda fulgida*, Little Colorado spinedace *Lepidomeda vitatta*, longfin dace *Agosia chrysogaster*, pupfishes *Cyprinodon* spp., and topminnow *Poeciliopsis* spp., that are normally in the open water column. Then there are "demersal species" such as loach minnow *Rhinichthys cobitis*, speckled dace *R. osculus*, and several sucker species *Catostomus* spp. that spend almost their entire existence on the bottom, within and upon stream substrates. A shortcoming of the guild approach is that, as noted above, each fish species has specific habitat requirements (Rinne 1992; Rinne and Stefferud 1996). Monitoring must address these specific, diverse habitat requirements. Further, in areas with naturally low diversity of fishes, such as the Pacific Northwest or Alaska or higher elevation streams in the West, examining diversity of fishes or fish guilds is difficult when only a handful or few species may be present.

The techniques or gear for monitoring fishes in response to riparian and grazing projects are similar to those for other habitat restoration techniques, including electrofishing, visual observation (snorkeling), seining, trapping, and more. The techniques and gear vary in their effectiveness in different stream habitats and stream types. For example, snorkeling has been used widely in the larger, less turbid montane streams of the Pacific Northwest and northern Rockies inhabited by salmonid fishes (Hankins and Reeves 1988). If snorkeling is selected as a sampling approach, snorkel estimates should be calibrated with a more accurate technique, such as electrofishing, to estimate the precision of snorkel estimates and to provide defendable monitoring of restoration efforts (Thompson 2003). Many streams of lower elevations in the West and Southwest have very meager flows (<0.01 m3/s) and depths (<10 cm), which render them less conducive to snorkel sampling. Methods for enumerating fish and their limitations are discussed in more detail in Chapter 8.

Aquatic macroinvertebrates

Macroinvertebrates are highly sensitive indicators of habitat change and may be useful for examining the effects of changes in grazing on stream health and biota (Merritt and Cummins 1996; Karr and Chu 1999). In addition to individual species or genus information, species richness, indexes of biotic integrity (i.e., Karr and Chu 1999), or presence and density of Ephemeroptera, Plecoptera, and Trichoptera (EPT index) may be useful indicators of changes in habitat associated with changes in grazing management. Similar to other biota, one must determine how often and where to sample macroinvertebrates. Increase in fine sediments often is offered as a negative impact of livestock grazing. Assessing or sampling of the fine (<2 mm) component of these same stream substrates is an important response factor to relate to any change in density and diversity of invertebrates. Sampling of more diverse substrates (i.e., pebble, gravel, and cobble) may be desirable to detect a change in diversity of macroinvertebrate communities. Sampling should be timed to get the maximum diversity of invertebrates; this generally is in the spring or fall, depending on the region. Temporally, sampling quarterly should be adequate to define seasonal variability and yet detect changes resulting from restoration. Equipment and protocols for sampling aquatic macroinvertebrates are readily available (Platts et al. 1987; Merritt and Cummins 1996). Invertebrate analyses are time and money consuming, and managers should consider contracting to entities that specialize in this field. Additional information on macroinvertebrate sampling can be found in Chapters 6, 8, and 9, as well as in texts on identification and sampling of macroinvertebrates, such as Rosenburg and Resh (1993) and Merritt and Cummins (1996).

Other Parameters

Other biotic and abiotic parameters also may be of interest in examining biological responses to grazing and fencing activities, depending upon project objectives. These may include algal (primary productivity) and other aquatic plants, sediment, nutrients, pollutants, potentially harmful bacteria, and vertebrates (e.g., amphibians). Algal and bacteria production may respond quickly to changes in light, nutrients, and sediment, and may be useful in assessing in-channel effects of the reduction in grazing. Methods for sampling algae and bacteria are fairly straightforward; two common measures include ash-free dry mass and chlorophyll a (see Steinman and Lamberti 1996). Additional information on methods for sampling algae and primary productivity are provided in Chapter 9.

Pollutants are important when the stream restoration has the object of improving water for domestic consumption and recreation, as well as fish and fisheries. With ever-increasing human use of riparian-stream areas, there is always a degree of organic or bacterial contamination or pollution in rivers and streams throughout the United States. Pollution, in general, comes from sources such as mining, industrial activities, irrigational return waters, municipal waste water systems, and sand and gravel operations within riparian areas.

Efforts in monitoring and estimating nutrient levels, contaminants, and major cations and anions in the waters of streams and rivers should be based on careful considerations. State departments of environmental quality and the EPA must be considered as sources of information on general water-quality parameters. Not all pollution or other parameters will be useful for a project. As with our general thesis in this chapter and book, always first determine the project objectives and hypotheses, and then determine if specific parameters can test the hypotheses or help link a change in grazing management to physical or biological factors of interest.

Case Studies in the Southwest

Below, we present three case studies of monitoring riparian-stream restoration by grazing removal in two montane and one lower-elevation aquatic ecosystem in the southwestern United States. The case studies emphasize pitfalls and confounding factors that may, potentially, strongly affect valid, defendable monitoring of riparian-stream areas that are restored through alteration of grazing strategy. Temporal-spatial problems, fish species considerations, fisheries management in restored streams, and natural impacts, such as flood and drought, can quickly and effectively alter a well-designed study and greatly reduce its value for land managers in their attempts to restore riparian-stream areas. We submit that objectivity is foremost in defining the scope of the resource problem and developing a monitoring plan that meets legal, statistical, and biological requirements. We recognize legal and regulatory mandates upon resource managers to monitor management actions. However, as we illustrate in these case studies, monitoring to meet legal mandates is different from monitoring to discover the underlying relationships, thresholds, and linkages between fish and their habitats. The former assumes that the relationships and linkages are well established and, hopefully, uses a valid methodology; the latter takes into account the spectrum of ecological facts, recognizes that cause and effect linkages may be weakly defined or lacking, and proceeds to develop an understanding of relationships, with validation of methods and results.

Case Study One: Long-Term Monitoring of the Rio de Las Vacas Exclosures

Project overview

The Rio de Las Vacas, located on the Santa Fe National Forest, New Mexico, was fenced to improve riparian habitat during the early to late 1970s. Two stream reaches, approximately 1 km long and 50 m wide, were fenced in 1972 and in 1975 to exclude livestock. These exclosures were separated by private lands from a downstream grazed study area. The objective of the research and monitoring was to determine if livestock exclusion benefited the riparian ecosystem (habitat and fishes). The stream supported three fish species native to upper elevation tributaries to the Rio Grande: Rio Grande sucker *Catostomus plebeius*, Rio Grande chub *Gila pandora*, and Rio Grande cutthroat trout (Rinne 1985, 1988; Calamusso and Rinne 1995). In addition, two nonnative trouts, brown trout *Salmo trutta* and rainbow trout *Oncorhynchus mykiss*, were present in the stream.

Monitoring approach

Six sample units (50-m sections) were studied in both grazed and ungrazed reaches of a stream (Rinne 1985, 1988). To estimate fish populations, 50-m, blocked-netted sections of the stream were sampled, from 1982 to 1985, with electrofishing gear. Initially, the grazed sample units were separated from the ungrazed units by about 4 km of stream on private lands that originally were not available for sampling. Ultimately, permission was received to sample these private holdings in 1985. In addition, the water quality, substrate permeability, and streambank stability and vegetation were measured.

Results

Fish population densities were highly variable within and among reaches and years in 50-m study reaches (21–181 and 33–545 fish/50 m reach in grazed and ungrazed reaches, respectively) and not significantly different between grazed and ungrazed reaches of stream throughout the 4 years of study. After 10 years of streambank protection from grazing, streambank stability was 100% in exclosures, but 64% were unstable in grazed areas, and both percentages of streambank (8% versus 1%) and overhanging vegetation (17.2% versus 0%) were greater in grazed compared to ungrazed reaches of stream. Substrate permeability of streambed generally was lower in ungrazed areas, and intergravel flow was slightly higher in ungrazed areas, but the differences were not significant (Rinne 1988). There was no difference in the basic nutrients NO_3, PO_4, and SO_4 in waters among the grazed and ungrazed reaches of stream. Conclusions from this case study of an upper-elevation montane stream removed from grazing for about a decade were that three major design problems were inherent: (1) absence of pretreatment data on fishes, vegetation, bank stability, or water quality; (2) spatial–temporal design of the study confounded results; and (3) fisheries management actions (e.g., stocking, fishing regulations) induced additional variation into the study.

The lack of pretreatment data, as seen in the Rio de Las Vacas study, has been a common flaw in monitoring physical and biological responses to grazing and other habitat protection and restoration activities. Rinne (1999a) reported that only 1 in 10 studies of grazing effects on riparian habitats and fishes across the West included adequate prestudy information. Often, the necessity and importance of such data are ignored. As was the case with the Rio de Las Vacas study, researchers most often adapt their study design to fit the situation that presents itself, with all its inherent shortcomings.

In the Rio de Las Vacas, grazed and ungrazed study reaches were separated by several kilometers because of the interpositioning of private land that was not originally available for study. Since completion of the study, it has been confirmed that stream-channel type (Rosgen 1996) had changed in that distance. The lower, grazed study reaches were within a C-type meadow reach, with less streamside vegetation and a different substrate composition. Based on intragravel flow rates, substrates characterized by a greater fine component also were present in the lower gradient, C-type channel. In the upstream, grazed, higher gradient, B-type channel, less fines and greater intragravel flow were present. These changes in riparian habitat cannot only affect fish abundance (Rinne and Stefferud 1996; Rinne and Neary 1997; Rinne 2001b) but also fish species distribution linearly in a channel. Higher densities in the ungrazed, lower gradient, C-type meadow reaches of the Rio de Las Vacas could be attributed, in part, to the gradient (Rinne 1988). Without information on channel and habitat preferences of various species, one could obviously, alternatively, and erroneously explain the increase of this species to degradation of habitat caused by livestock grazing.

Another major consideration in conducting studies on the effects of livestock exclosure concerns fish species, their interactions, and their management. Studies need to consider salmonids, as well as nonsalmonid species; their interactions and their habitat requirements and ecology are very different. Further, most studies never consider the widespread, common practice of "put and take" fisheries for salmonid species in the West (Platts and Nelson 1988). In one given year in the Rio de Las Vacas study, more than 9,000 catchable rainbow trout and 800 catchable brown trout were stocked. These introductions increased the density estimates of brown trout the following year. Thus, fisheries management actions, such as stocking or access for fishing, need to be carefully controlled or they make detecting a fish response to grazing activities even more difficult.

Case Study Two: The West Fork Grazing Allotment Riparian Grazing Study

Project overview

Ungulate grazing studies were initiated in 1993 on the West Fork Grazing Allotment of the Apache Sitgreaves National Forest (Medina and Steed 2002). The problem defined by the U.S. Fish and Wildlife Service and the Arizona Game and Fish Department was overgrazing of riparian habitats by cattle, with negative effects on Apache trout habitat and populations. The Forest Service contended that various factors, including cattle grazing, were causing changes in riparian and stream channels (Figure 1; top photograph). Ranchers argued that elk were responsible for limited forage resources. Environmental activists cited studies from other regions, noting cattle as the culprit. The species at issue was the threatened Apache trout, which reportedly occupied three streams on the allotment: Wildcat, Boggy, and Centerfire. The project objectives were to improve habitat for trout and to improve riparian conditions. The principal objectives included determining the relative effects of ungulate grazing on riparian habitats and Apache trout, assessing the utility of GAWS as a monitoring protocol, and developing grazing prescriptions for livestock. Resource managers agreed to resolve the highly contentious issue by soliciting the U.S. Forest Service Rocky Mountain Research Station to monitor riparian conditions, cattle grazing, and Apache trout for 6 years, and they agreed to incorporate newfound knowledge into allotment management plans.

Monitoring approach

The streams were variously fenced to impose three grazing treatments: no grazing (elk exclosures), elk use only (standard 5-strand barb wire), and both elk and cattle grazing. Monitoring parameters included streambank vegetation, water quality, channel geomorphology, ungulate trampling (crossings), assessment of Habitat Condition Indices used in GAWS (USFS 1998), production and utilization of herbaceous vegetation, and fish. Sampling of riparian vegetation, streambanks, channel morphology and fish habitat, and fish abundance occurred within permanent 40-m stream reaches dispersed among the streams and treatments. Sampling stations were established within Boggy and Centerfire Creeks on alder and nonalder sites, whereas, in Wildcat Creek, transects were established based on flow conditions (perennial and intermittent streams), since there were no alder stands. Study progress was reviewed annually, with frequent field sessions to discuss current findings

Results

Changes in riparian conditions varied relative to the hydrogeomorphic condition of the excluded stream reaches. Herbaceous vegetation (i.e., biomass, production) responded favorably across all treatments and controls, including the common use areas (Figures 1 and 2). Pretreatment baseline monitoring occurred in 1993–1994, with grazing reduction treatments occurring in 1995. Sedges generally increased in vigor and abundance in the grazing exclosures (Figure 1; middle photograph) and slightly in the common use areas. General increases in herbaceous biomass production across the study period were noted but were offset by an increase in relative utilization (spring >85%; mean annual >45%) by elk. Trampling of streambanks was most common in stream crossings, though relatively minor changes (<6 cm^2/channel profile/year) in channel geomorphology were noted across all streams. In 4 of the 6 years of the study, various sections of the three streams

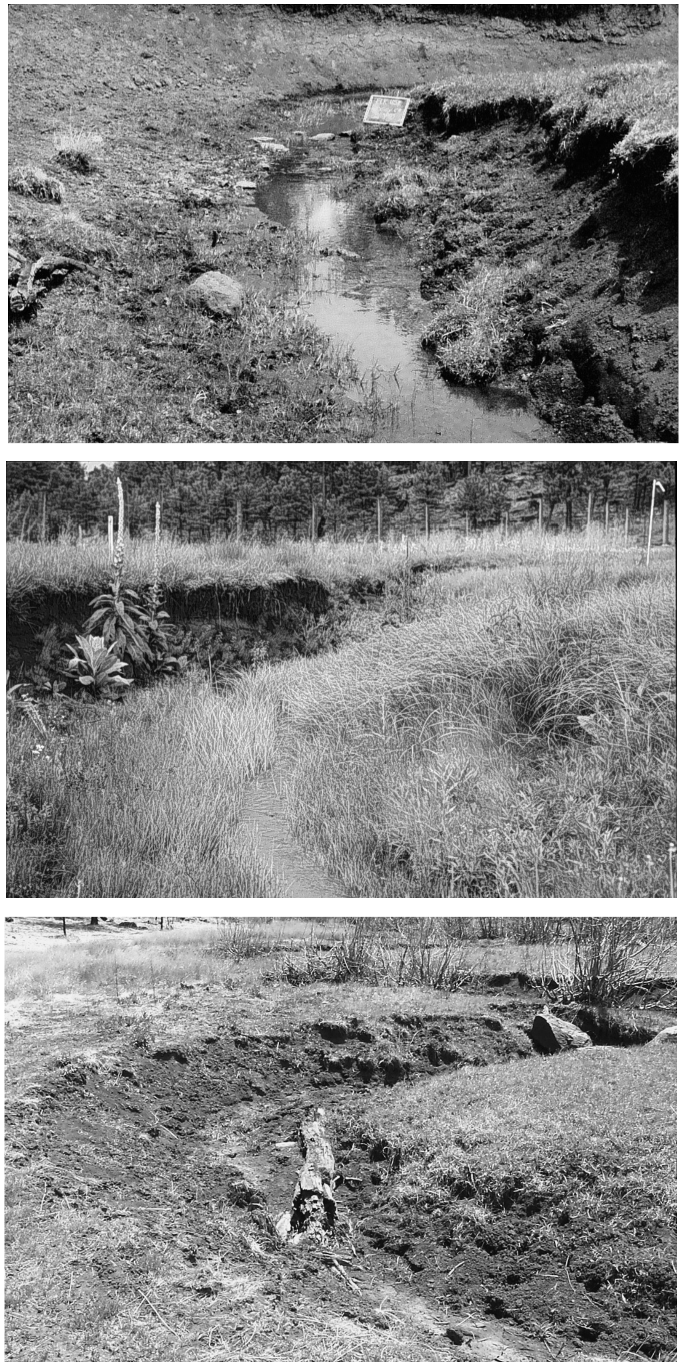

Figure 1.
Photographs of Boggy Creek, showing effects of cattle use before initiation of study and removal cattle in May 1991 (top photograph); 5 years after cattle exclusion in August 1996 (middle photograph); and same reach after completion of study in May 2000, showing heavy elk damage (bottom photograph).

dried in the lower meadow reaches. Rodents (i.e., voles, shrews, and mice) were largely responsible for sedimentation effects and the collapse of overhanging streambanks (Figure 3). Freezing and thawing also promoted bank instability, and the fine soil material became available for transport during the next flow event. Cattle were thought to be the principal agent, but these studies illustrate that many other interactions also are likely factors affecting riparian condition. Despite habitat condition indices (HCI) meeting a 70% criterion set by managers, Apache trout did not respond positively. The natural die-off of alder stands left the channel

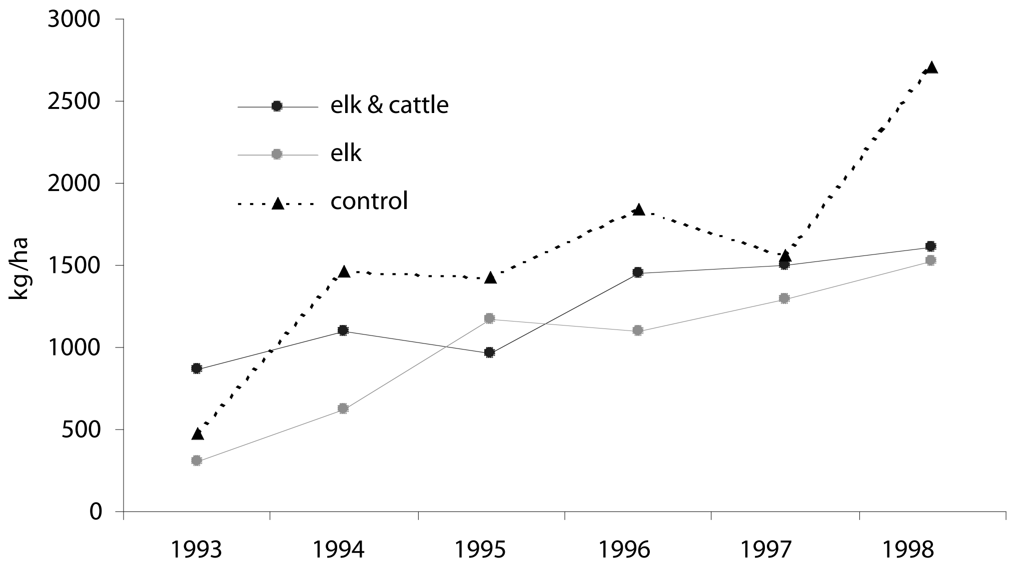

Figure 2.
Increase in mean standing biomass (dry weight) of vegetation at grazing treatments and controls on Boggy Creek between 1993 and 1998. Similar results were found on sites at Centerfire and Wildcat Creek study sites.

exposed to higher solar irradiation and water temperature fluctuations (Figure 4; top photograph). The presence of woody plants is a major constituent of the HCI methodology, resulting in higher index values compared with herbaceous streambanks. Additionally, woody debris from the alders caused debris jams to form, which eroded the streambanks or caused braiding (Figure 4; bottom photograph). Continued observations between 1998 and 2003 indicated similar trends, despite continued improvement in vegetative conditions. However, despite the fact that cattle have not grazed the allotment since 1998, streambank conditions on many reaches are similar to pre-1993 conditions, owing to heavy grazing by elk after cattle removal (Figure 1; bottom photograph).

After 4 years of treatment, fishes did not respond to grazing treatments. Trout density per kilometer successively decreased in two of the streams (Table 3), and various external factors, such as stream intermittency and drying (climatic), limited fish sampling, underlying geology, soils, and hydrology, appear to limit our ability to detect changes in fish production in the study streams. Additional assessments of channel habitats and contrasts with reference streams suggest that these streams are marginal at best for sustaining an Apache trout fish-

Figure 3.
Rodent burrowing (right bank) at an grazing exclusion site, causing increased bank instability.

Figure 4.
Multiple alder debris jams causing braiding of channel (top photograph) and debris jam redirecting streamflow and causing bank erosion on Centerfire Creek, April 1998.

ery (Medina and Steed 2002). Ongoing research suggests that parent geology and resulting substrate composition of streams are far more important factors defining Apache trout habitats and their populations. The West Fork streams are positioned upon basaltic soils, which naturally have high amounts of fines and do not provide optimal salmonid spawning habitat (16–64-mm gravels) compared to glaciated, alluvial soils in the White Mountains that typify salmonid reference streams. Finally, data suggest that the GAWS and HCI are inappropriate methods for monitoring and evaluating grazing effects in the Southwest. Stream reaches in common use by both cattle and elk improved (e.g., streambank stability, vegetation density, and composition) outside the exclosures. We conclude that optimal, high-quality fish habitat is primarily defined by the inherent hydrogeomorphological structure of the stream. The extent of grazing influences seems minor comparatively but, nonetheless, important on sensitive riparian habitats. In short, various other factors, both intrinsic (e.g., nonnative fishes, limited substrate availability) and extrinsic (e.g., drought, floods, vegetation–animal interactions, wildlife), may exert a greater influence on fishery habitats and populations than a grazing strategy and must be a primary consideration when monitoring fish-grazing relationships.

In this study, the combined interactions of animals, vegetation, climate, and natural attributes among study areas partially masked detection of ungulate grazing effects on streambanks. Complex interactions are a common confounding factor in grazing studies but often are ignored. The relative utilization of key riparian species was easily observed throughout most of the year, except during the latter phase of the growing season, when herbaceous species (e.g., sedges, rushes) grew faster than they were consumed. Biomass production and

Table 3.
Mean number of trout per kilometer in three West Fork allotment streams, autumn, 1993–1996. Common use treatment included grazing by elk and cattle, cattle exclusion = excluded cattle but not elk (elk use only), and the elk exclusion = excluded all ungulates. NA = not applicable (not collected).

Treatment	Stream		
Year/type	**Centerfire**	**Boggy**	**Wildcat**
1993			
Common use	240	330	12.5
Cattle exclusion	7	4	0
Elk exclusion	0	0	NA
1994			
Common use	160	180	30
Cattle exclusion	40	4	30
Elk exclusion	20	0	NA
1995			
Common use	38	60	25
Cattle exclusion	129	5	0
Elk exclusion	33	0	NA
1996			
Common use	NA	19	NA
Cattle exclusion	6	4	NA
Elk exclusion	0	0	NA

utilization steadily increased over the study period, probably in response to the initial release from cattle grazing for 3 years and the cattle grazing treatments, which limited cattle use to near the end of the growing season. Species differences were observed, such as with beaked sedge *Carex rostrata*, which produced minimal biomass in response to spring use by elk. Conversely, Nebraska sedge was grazed extensively yearlong by elk and exhibited overcompensation. These responses to grazing at the species level may account for the presence of a grazing effect or may account for a different response at the community level (species–species interactions). Secondly, climate was a major factor controlling the hydrological (i.e., flow, water quality) and biological (i.e., vegetation, fish) variables, which exhibited a chaotic response to drought and floods. Grazing effects are most evident and lasting during drought and are minimized during wet periods. Unusually high precipitation during study years may explain why vegetation growth increased in both grazed and ungrazed sites. The conclusions differed with each additional year of study. Hence, monitoring should encompass a period that adequately spans a period of time to account for climatic influences and carryover effects.

Case Study Three: The Verde River Threatened and Sensitive Fish and Riparian Studies

Project overview

The upper Verde River study area lies below the Mogollon Rim of central Arizona, at an elevation of 1,000 m, and encompasses about 50 km of river principally within the pinyon–juniper woodland type. The contentious question of cattle grazing effects on warmwater fishes was the impetus for this study, similar to the West Fork case, except that litigation over livestock grazing of riparian areas on a regional basis invoked a greater need for information of potential grazing effects on fish. Many biologists indicated that riverine conditions were impaired because of livestock grazing on the river and the watershed. A principal theory was that "the river was sediment enriched." In response to endangered fishes and critical habitat designations, including the riparian corridor of the study area, and to improve the riparian area, all livestock on public lands within the watershed were removed in 1998. Strategic fencing and rough terrain were used to exclude livestock, which had grazed the river bottom with varying strategies for over a century. Trespassing animals were monitored for and removed upon discovery. Exclusion of livestock grazing continues to the present, but elk have moved in and replaced cattle. The objectives of the monitoring study were multifold, including inventory of the fish populations, assessment of fish and riparian habitats, inventory and classification of vegetation and channel types, assessment of sediment influences, and water quality.

Monitoring approach

Riparian studies were initiated in 1996 and included monitoring at different scales. A complete inventory of upper Verde River channel conditions ensued to classify the channel types (Rosgen 1996; Neary et al. 2001). Permanent riparian vegetation transects (n = 48) were established across various habitat types and were superimposed on channel sample sites. Streambanks were intensively sampled by using a modified green line approach developed specifically for monitoring ungulate–streambank influences (Medina and Steed 2002). Vegetation and channel measurements were taken each year to provide an estimate of type and rate of change. Water-quality studies were initiated in 2000 to assess potential sedimentation and turbidity problems. Automated water samplers were used to collect water samples for sediment analyses. Pebble count transects (Bevenger and King 1995) were established in all major tributaries to detect changes in substrate composition and to identify potential sources of fine sediments. In 2001, macroinvertebrate sampling was initiated to provide an additional measure of aquatic habitat quality. Samples were taken across seasons and habitat types. To address questions of channel stability, annual aerial flights of the riverine corridor were used to detect changes in channel position across the entire 50-km reach. In 2000, a long-term river change detection study was initiated to quantify changes in channel position over the past 70+ years for which aerial imagery was available.

Fisheries studies were initiated in 1994 after elevated levels of flooding in the winter of 1992–1993 (Stefferud and Rinne 1995). Additional elevated flow events occurred in spring 1995; however, the upper 60 km of river has sustained low flow drought conditions since that time. Seven permanent sites 300–500 m long were established in the upper Verde and were sampled, with seines and electrofishing gear, annually for fishes. The relative proportions of native versus nonnative fish species were used as an indicator of fish responses to flooding. Six native species were common in spring 1994 (Stefferud and Rinne 1995; Rinne and Stefferud 1997; Table 4). In addition, several nonnative fishes were present but not abundant.

Results

The series of diagnostic studies did not identify any physical, biological, or chemical parameter to suggest an impairment of water quality or riparian functions. All parameters (i.e., dissolved oxygen, temperature, conductivity, pH, total suspended sediments) measured for water quality were within established standards for warmwater fisheries (Medina 2001). Vegetation cover, density, and composition increased across grazed and ungrazed sites (Medina 2001; Figure 5). Increased plant cover and species diversity were attributed to post-

Table 4.
Changes in fish assemblages in the upper Verde River, 1994–2002.

Species	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003
Longfin dace	1319	12	282	21	13	2	1	2	1	1
Spikedace	428	72	141	0	0	0	0	0	0	0
Speckled dace	172	25	68	1	12	2	7	0	0	0
Desert sucker *Catostomus clarki*	2644	328	471	231	126	167	137	376	148	128
Sonora sucker *C. insignis*	1810	322	654	240	128	118	197	163	90	75
Roundtail chub	776	341	259	50	64	25	20	43	20	4
Smallmouth bass	14	10	32	35	66	104	48	163	211	193
Red shiner	1473	97	275	2238	1047	545	1594	1608	276	632
Green sunfish	4	29	6	8	21	49	95	192	53	139

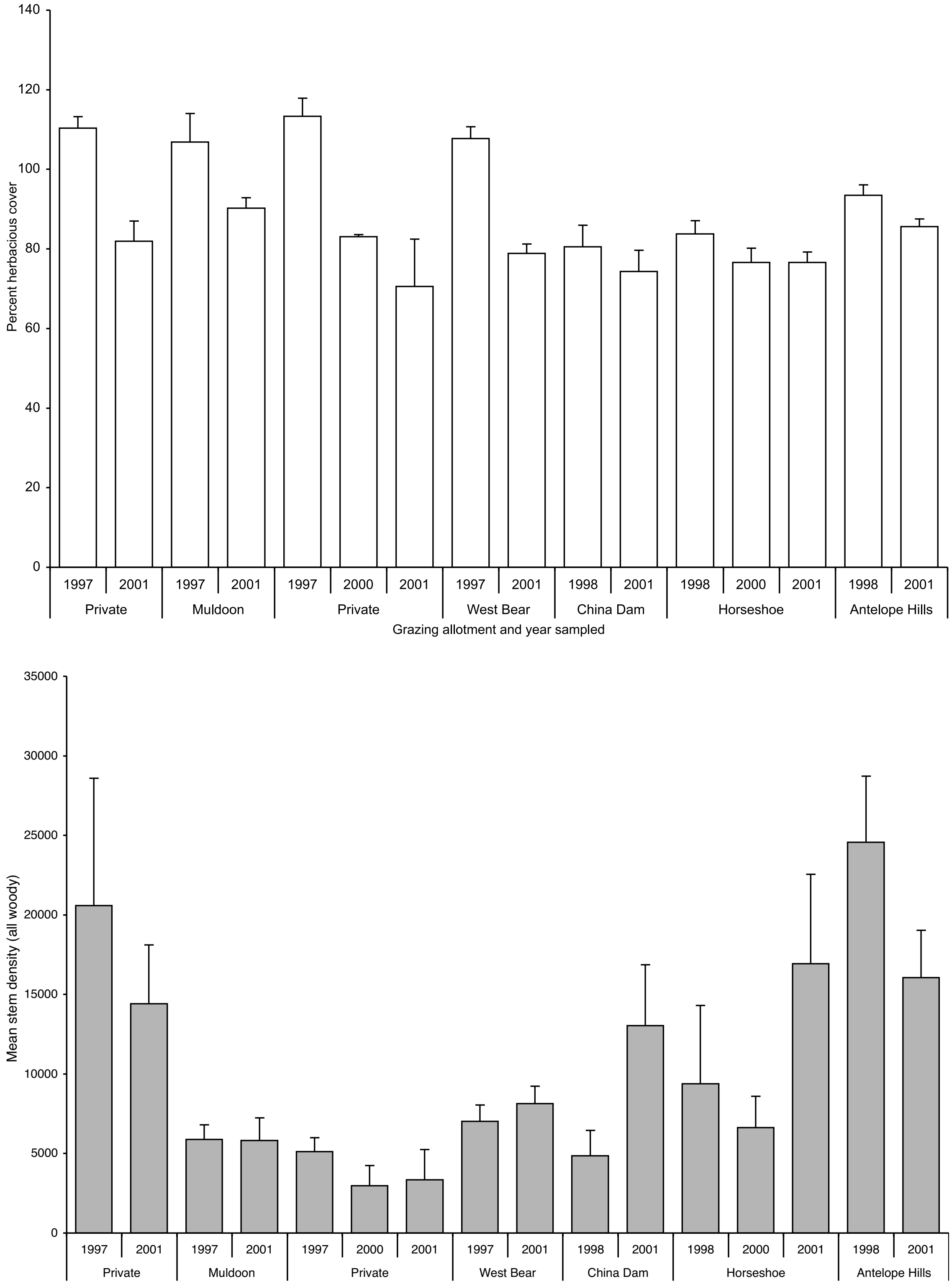

Figure 5.

Comparisons of mean percent cover of total herbaceous vegetation and mean stem density or woody vegetation at six allotments on the Verde River (51 transects) repeatedly sampled between 1997 and 2001. Grazed lands included only private lands, and ungrazed lands included all other U.S. Forest Service allotments (Muldoon, West Bear, China Dam, Horseshoe, and Antelope Hills). Sites are in order from upstream to downstream (left to right). Error bars represent standard error.

flood disturbance dynamics that masked effects of livestock removal. Invasive and exotic species also increased in both grazed and ungrazed reaches. The distribution of channel types, substrates, and associated geomorphological attributes were consistent with reference reaches for alluvial desert rivers (Medina et al. 1997; Neary et al. 2001). A preliminary analysis of aerial imagery indicates that the main channel has remained relatively static since 1947. However, since 1979, the channel has incised 1–5 m and has become dominated by woody vegetation (Figure 6). Channel degradation was attributed to a sediment imbalance caused by dams upstream, a common phenomenon observed elsewhere (Collier et al. 2000). Encroachment of woody vegetation was attributed to a 1993 historic flood and ongoing channel degradation associated with sediment impoundment by the dams. Reaches where sedges and bulrushes prevailed on grazed sites remained relatively stable in channel form and vegetation communities.

Immediately after the floods (1994), native fishes dominated the fish community (Table 4). Native species comprised over 80% of the total fishes sampled each spring from 1994 to 1996. In 1997, the relative composition of native fishes dropped dramatically to 19% and has remained below 30% since that time. The three small-sized, short-lived species—longfin dace, speckled dace, and spikedace—have declined to zero or near zero. The last species is listed as threatened under the U.S. Endangered Species Act. The longfin dace was very abundant in 1994, averaging almost 200 individuals per sample section. In the past 4 years of sampling, only six of this threatened species have been collected during sampling at all seven established sample sites. Although the three longer-lived, larger-sized species have survived, even these species may become extirpated. Presently, primarily adults of each species have been captured and are represented in samples. These remaining native species have been reduced 20- to 30-fold in total numbers from spring samples of 1994. By contrast, introduced red shiner *Cyprinella lutrensis*, green sunfish *Lepomis cyanellus*, and smallmouth bass *Micropterus dolomieu* have steadily increased. The increase in nonnative species abundance has been attributed to a general increase in nearstream and instream aquatic herbaceous vegetation (Medina and Rinne 1999). Bass and sunfish are cover inhabitants in aquatic systems (Pflieger 1975) and have responded very favorably to the changes in habitat conditions. By contrast, the native species are either absent from samples or are rapidly being extirpated from the Verde River system.

This study suggests that improved conditions in vegetative and channel conditions have not benefited native fishes (Rinne 1999a, 2003a, 2003b). On the other hand, nonnatives species have benefited from the changes. The results of a recently initiated predator removal study (i.e., nonnative species from reaches of a stream; Rinne 2001a) suggest that direct predation by nonnative species may be the primary cause for the dramatic reduction of native fishes. As with both the Rio Las Vacas and West Fork studies, fish species interactions and

Figure 6.
Verde River Bear Siding site before (left) and in 2002, 4 years after removal of grazing (right slide), demonstrating recovery of vegetation. It was not possible to take right photograph from exactly the same location because vegetation recovery and channel incision created a deep pool where earlier photograph had been taken.

fishery management are major contributors to the outcome of monitoring studies of grazing exclusion. Habitat change with grazing removal obviously has an influence on fishes (Rinne and Neary 1997); however, it is an unintended response. That is, the increase in habitat complexity had unintended positive benefits for nonnative fish species that prey upon native fishes. The absence of recent (post-1993) flood events also has benefited the nonnative species and harmed the native species (Rinne and Stefferud 1997).

Synthesis of Case Studies: Confounding Factors in Monitoring Effects Grazing on Fishes

These three case studies point out some inherent problems that must be addressed in the design and monitoring of effectiveness of grazing and riparian restoration projects on fishes. Several factors apparent in these case studies can confound interpretation of data, including (1) species interactions, (2) management practices, (3) spatial–temporal factors (replication and spacing of treatments and controls), (4) geology and geomorphology, and (5) climate and hydrology. We suggest these same factors are applicable to all studies evaluating grazing and riparian restoration. Some additions, explanations, and interpretations of these factors are appropriate and offered here.

Species interactions

First, native and exotic species and their interactions have to be taken into consideration for study design, monitoring, and interpretation of the effects of grazing management (Rinne 2002), for example, in the Verde River, the positive influence of grazing removal on nonnative compared to the negative influence on native fish species. One has to go a step further and examine the interactive influence of predation by the nonnative species, such as red shiner, smallmouth bass, and green sunfish. Based on our data (Rinne 1995, 2001a, 2003a, 2003b; Rinne and Alexander 1995) and that of others (Minckley 1983; Minckley and Deacon 1991), predation is one of the primary negative impacts of nonnative species on the native species. The change in cover habitat in the upper Verde has strongly and positively influenced the nonnative predators, which, in turn, have negatively impacted all native species, perhaps to the point of extirpation of smaller-sized, short-lived species (i.e., longfin dace, speckled dace, and spikedace, Rinne 1999b). Similarly, on the Rio de Las Vacas, regular stocking of rainbow trout and brown trout has negatively impacted native Rio Grande cutthroat and two native cyprinids through hybridization and predation, respectively. Changes in native and nonnative vegetation also influenced the vegetation and bank stability, both of which may influence instream factors such as fishes. These examples emphasize the need to examine species interactions for both vegetation and aquatic biota when conducting studies on riparian restoration.

Management practices

A second consideration involves interactions of both historic and contemporary management activities. Grazing management in the form of livestock removal from the Verde River corridor was assumed beneficial for the riparian habitat and native fish species. Fisheries management in the form of stocking nonnative species (Rinne et al. 1998; Rinne, in press) was initiated 50 years ago, and grazing, another half century earlier. So, while grazing and fish stocking occurred together for at least a half century, native fish species persisted. One possible explanation for their persistence is the repression of vegetative growth in this period, which benefited the native species. In the same line of reasoning, native species in this region appear to be adapted to natural disturbances such as floods (Rinne 2003a, 2003b) and also can withstand a certain level of anthropogenic disturbances (e.g., grazing). More recently, catch limits have been removed for nonnative sport species in the Verde River by the Arizona Game and Fish Department. However, the U.S. Forest Service, as the manager of the river habitat, has closed road access to formerly easily accessed areas of the river, making removal of nonnative predators by fishermen unlikely. Similarly, the removal of cattle grazing in the case studies often was followed by an increase in grazing of native ungulates (elk). Here again, we see one management activity counteract another. All three case studies demonstrate the importance of examining the influence and the interactions of current and past management practices on multiyear studies.

Spatial and temporal factors

Spatially fencing is problematic, because, by design, it is linear and it fragments the riparian habitat. Both the Rio de Las Vacas and West Fork Case studies had this inherent design flaw. The only alternative to such an intrastream approach is to design interstream comparison of data. However, variation (as suggested above for fishes) is present in all physical and biological factors of respective streams, rendering such comparisons suspect or invalid. Change in stream type (see habitat influences below) was present on both the Rio de Las Vacas and the West Fork and private lands in the former affected the initial study design. These results indicate that one should control for additional factors, including stream type, geology, and, especially, habitat and channel type, when selecting study sites. To reemphasize, perhaps, the most important spatial consideration to remember is that the stream is a continuum (Vannote et al. 1980). Sectioning by fences to exclude grazing and to improve riparian-stream habitat, although it segregates the riparian area, will induce recovery within the treated areas but probably will not affect upstream and downstream processes that operate on large scales.

Temporal influences often are intricately linked to these spatial influences. Large interannual variation (>50%) in wild fish populations is the norm rather than the exception. This was noted over several years on all three case study areas and has been documented elsewhere (Platts and McHenry 1988; Platts and Nelson 1988). For example, after year one on the Rio de Las Vacas, it appeared that the conventional knowledge of that time (i.e., grazing removal enhanced trout populations) was corroborated. However, additional years of data suggest no effect of grazing removal on fishes. As with instream restoration and other long-term monitoring, it is necessary to adequately quantify interannual variation in fish populations before drawing conclusions on the effectiveness of riparian restoration projects on fish or habitat.

Geology and geomorphology

Understanding the underlying geology and changes in geomorphology and habitat resulting from grazing are critical to valid interpretation of riparian restoration. For example, the longitudinal change in channel type along the stream continuum is a basic tenet of stream morphology: streams change from one channel type to another in montane areas as they flow through higher gradient reaches and meadow reaches. Both the Rio de Las Vacas and the West Fork study designs were influenced by stream type. Although the Verde is primarily composed of C-type channels, Rinne and Neary (1997) demonstrated that with a change in channel type, a change in fish assemblage occurred. The respective biology of native fish species in the Verde and their habitat preferences are very specific (Rinne and Stefferud 1996). Accordingly, even slight changes in habitat, whether natural or by artificial, anthropogenic activities, such as grazing (or its removal), can dramatically affect fish distribution and abundance. Substrate composition is largely dictated by parent geology and is very strongly and directly influenced by stream gradient, which dictates water velocity. Velocity and substrate are strong influences of native fish presence and abundance (Heede and Rinne 1990; Rinne 1992, 2001b, 2001c; Rinne and Deason 2000). This relationship between channel type, velocity, and substrate, and fish distribution has been well documented in other areas (e.g., Montgomery et al. 1999; Weigel and Sorensen 2001). Thus, geology, geomorphology, and channel type are important factors to control for when designing a monitoring program.

Climate and hydrology

Finally, the results of these studies must be interpreted in the context of natural hydrologic patterns and climate (Platts 1991; Rinne 2003a, 2003b). Hydrology has been a major controlling factor on both the West Fork and the Verde case studies. In the former, drought in most of the years of the study affected fish populations more than did grazing treatment. In the Southwest (Rinne, in press) and throughout many areas of the West (Platts et al. 1985), cycles of flood and drought are the norm. The Verde study, for example, was initiated during a period of high flow (1993–1995); however, most monitoring (1996– 2002) was under drought conditions. Floods in the Verde (Rinne and Stefferud 1997) and elsewhere in the Southwest (Minckley and Meffe 1987) have been demonstrated to have a marked positive influence on native fishes. By contrast, drought or low and more stabilized flows, which occur below dams in the West, often favor nonnative fishes. Indeed, a portion of the changes in fish community structure in the Verde can be attributed to changes in hydrology. Similarly, changes in vegetation growth (or lack thereof) between grazed and ungrazed sites appear to be related to changes in climate (rainfall). Differences between vegetation attributes in treatment and controls are most pronounced during drought years but are absent during wet years. Thus, it is important to consider changes in hydrology and climate when interpreting the results of grazing studies. This emphasizes the need for long-term monitoring to completely understand the effects of changes in grazing management on riparian recovery.

Summary

As with most restoration actions, determining the objectives and hypotheses of both the grazing and fencing project and the monitoring program are critical steps. The monitoring design is equally important, and many grazing studies have been limited by not considering the location of control and treatments or by not adequately replicating them in space or time. Many parameters typically monitored for grazing project, such as temperature, bank stability, and fishes, while important, can be confounded by complex interactions of many factors, including species of fishes and other biota, fishery management practices (e.g., stocking, fishing pressure and regulations, and exotics), spatial and temporal scale and replication, channel and habitat type, hydrology and climate, and others. Vegetation and bank stability typically respond directly to changes in grazing intensity, but they also may be confounded by factors such as hydrology, climate, and grazing by wild ungulates. Interpreting the results of riparian restoration should be done cautiously, particularly for instream variables (e.g., fish, sediment, temperature), which may respond secondarily to improvement in vegetation after grazing (a direct response). That is not to say that monitoring the response of instream parameters is not valuable; but, as we have demonstrated in the case studies, many other factors (e.g., fish species and management, habitat influence, hydrology) can confound interpretation of results in the absence of adequate spatial, temporal replication, and other monitoring design considerations. One must not jump to quick conclusions without considering at least the influencing and controlling factors discussed in this chapter. Only by addressing the issue, resource, or questions to be answered and by carefully arriving at conclusions and coming up with, and continually refining, new models, will riparian-stream areas be properly understood, managed, and restored.

References

Acorn, R. C. 1997. Using electric fences to protect stored hay from elk and deer. Alberta Agriculture, Food, and Rural Development Bulletin, Agdex 684–17. Alberta Queen's Printer, Edmonton, Alberta.

Armour, C. L., D. A. Duff, and W. Elmore. 1991. The effects of livestock grazing on riparian and stream ecosystems. Fisheries 16(1):7–11.

Bain, M. B., and N. J. Stevenson. 1999. Aquatic habitat assessment: common methods. American Fisheries Society, Bethesda, Maryland.

Barbour, M. T., and J. B. Stribling. 1991. Use of habitat assessment in evaluating the biological integrity of stream communities. Pages 25–39 *in* Environmental Protection Agency, editor. Biological criteria: research and regulation, 1991. U.S. Environmental Protection Agency, Office of Water, EPA-440/5–91-005, Washington, D.C.

Barbour, M. T., and J. B. Stribling. 1994. A technique for assessing stream habitat structure. Pages 156–178 *in* National Association of Conservation Districts, editor. Proceedings of the conference on riparian ecosystems of the humid U.S.: management, functions, and values. National Association of Conservation Districts, Washington, D.C.

Barbour, M., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates, and fish. 2nd edition. U.S. Environmental Protection Agency, Office of Water, EPA-841-B-99–002, Washington, D.C.

Bauer, S. B., and T. Burton. 1993. Monitoring protocols to evaluate water quality effects of grazing management on western rangeland streams. U.S. Environmental Protection Agency, Washington, D.C.

Bekaert. 2002. Tightlock™ game fence: high tensile fencing. Bekaert Corporation, Akron, Ohio.

Belsky, A. J., A. Matzke, and S. Uselman. 1999. Survey of livestock influences on stream and riparian ecosystems in the western United States. Journal of Soil and Water Conservation 54(1):419–431.

Betscha, R. L. 1997. Riparian shade and stream temperatures: an alternative view. Rangelands 19(2):25–28.

Bevenger, G. S., and R. M. King. 1995. A pebble count procedure for assessing watershed cumulative effects. U.S. Forest Service, GTR-RM-RP-319, Fort Collins, Colorado.

Bisson, P. A., G. H. Reeves, R. E. Bilby, and R. J. Naiman. 1997. Watershed management and Pacific salmon: desired future conditions. Pages 447–474 *in* D. J. Stouder, P. A. Bisson, and R. J. Naiman. Pacific salmon and their ecosystems. Chapman and Hall, New York.

BLM (Bureau of Land Management). 1985. Fencing. U.S. Bureau of Land Management Handbook H-1741–1, Denver.

BLM (Bureau of Land Management). 1999a. Utilization studies and residual measurements: interagency technical reference. Bureau of Land Management Technical Reference 1734–3, Denver.

BLM (Bureau of Land Management). 1999b. Sampling vegetation attributes: interagency technical reference. Bureau of Land Management Technical Reference 1734–4, Denver.

Bunte, K., and S. R. Abt. 2001. Sampling surface and subsurface particle-size distributions in wadeble gravel-and cobble-bed streams for analyses in sediment transport, hydraulics, and streambed monitoring. U.S. Forest Service, Rocky Mountain Research Station, General Technical Report RMRS-GTR-74, Fort Collins, Colorado.

Burton, T., and E. R. Cowley. 2002. Monitoring streambank stability. Bureau of Land Management, Idaho State Office Report, Boise, Idaho.

Calamusso, B., and J. N. Rinne. 1995. Distribution of Rio Grande cutthroat trout *Oncorhynchus clarki virginalis* and its co-occurrence with Rio Grande sucker *Catostomus plebeius*, and Rio Grande chub, Gila pandora, on the Carson and Santa Fe National Forests. Pages

157–167 *in* D. W. Shaw, and D. M. Finch, editors. Proceedings of the symposium of desired future conditions for southwestern riparian ecosystems: bringing interests and concerns together. U.S. Forest Service General Technical Report RM-272, Fort Collins, Colorado.

Casagli, N., M. Rinaldi, A. Gargini, and A. Curini. 1999. Pore water pressure and streambank stability: results from a monitoring site on the Sieve River, Italy. Earth Surface Processes and Landforms 24:1095–1114.

Chapman, D. W. 1988. Critical review of variables used to define effects of fines in redds of large salmonids. Transactions of the American Fisheries Society 117:1–21.

Clary, W. P. 1999. Stream channel and vegetation responses to late spring cattle grazing. Journal Range Management 52(3):218–227.

Clary, W. P., and W. C. Leininger. 2000. Stubble height as a tool for management of riparian areas. Journal of Range Management 53:562–573.

Clesceri, L. S., A. E. Greenburg, and A. D. Eaton, editors. 1999. Standard methods for the examination of water and wastewater. 20th edition. American Public Health Association, American Water Works Association, and Water Environment Federation, Washington, D.C.

Collier, M., R. H. Webb, and J. C. Schnidt. 2000. Dams and rivers: a primer on the downstream effects of dams. U.S. Geological Survey Circular 1126, Reston, Virginia.

Contor, C. R. and W. S. Platts. 1991. Assessment of COWFISH for predicting trout populations in grazed watersheds of the intermountain west. U.S. Forest Service, Intermountain Research Station General Technical Report INT-278, Ogden, Utah.

Craven, S. R., and S. E. Hygnstrom. 1996. Deer: damage prevention and control methods. Pages D25–D40 *in* S. E. Hygnstrom, R. M. Trimm, and G. E. Larson, editors. Prevention and control of wildlife damage. Diane Publishing Company, Collingdale, Pennsylvania.

De Calesta, D. S., and G. W. Witmer. 1996. Elk: damage prevention and control methods. Pages D-41-D50 *in* S. E. Hygnstrom, R. M. Trimm, and G. E. Larson, editors. Prevention and control of wildlife damage. Diane Publishing Company, Collingdale, Pennsylvania.

Diaz, M. E., Gonzalez, R. Munoz-Pulido, and M. A. Naveso. 1996. Habitat selection patterns of common cranes *Grus grus* wintering in holm oak *Quercus ilex dehesas* of central Spain: Effects of human management. Biological Conservation 75(2):119–123.

Downes, B. J., L. A. Barmuta, P. G. Fairweather, D. P. Faith, M. J. Keough, P. S. Lake, B. D. Mapstone, and G. P. Quinn. 2002. Monitoring ecological impacts: concepts and practice in flowing waters. Cambridge University Press, Cambridge, UK.

Dunne, T., and L. Leopold. 1978. Water in environmental planning. Freeman, New York.

Elmore, W., and R. L. Beschta. 1987. Riparian areas: Perceptions in management. Rangelands 9:260–265.

Elzinga, C. L., D. W. Salzer, and J. W. Willoughby. 1998. Measuring and monitoring plant populations. U.S. Bureau of Land Management Technical Reference 1730–1, Denver.

Everest, F. H., R. L. Betscha, J. C. Scrivener, K. V. Koski, J. R. Sedell, and C. J. Cederholm. 1987. Fire sediment and salmonid fish production—a paradox. Pages 98–142 *in* E. Salo and T. Cundy, editors. Streamside management and forestry and fishery interactions. University of Washington, College of Forest Resources, Contribution 57, Seattle.

FISRWG (Federal Interagency Stream Restoration Working Group). 1998. Stream corridor restoration: principles, processes, and practices. Federal Interagency Stream Restoration Working Group, GPO Item No. 0120-A; SuDocs No. A 57.6/2:EN 3/PT. 653, Washington, D.C.

Fleischner, T. L. 1994. Ecological costs of livestock grazing in western North America. Conservation Biology 8:629–644.

Fuller, P. L., L. G. Nico, and J. D. Williams, editors. 1999. Nonindigenous fishes introduced into inland waters of the United States. American Fisheries Society, Bethesda, Maryland.

Gangmark, A. A., and R. G. Bakkala. 1958. Plastic standpipe for sampling streambed environment of salmon spawning. U.S. Fish and Wildlife Service, Special Scientific Report on Fisheries 1, Washington, D.C.

Gordon, N. D., T. A. McMahon, and B. L. Finlayson. 1992. Stream hydrology: an introduction for ecologists. Wiley, Chichester, England.

Grost, R. T., W. A. Hubert, and T. A. Wesche. 1991. Field comparison of three devices used to sample substrate in small streams. North American Journal of Fisheries Management 11:347–351.

Hankin, D. G., and G. H. Reeves. 1998. Estimating total fish abundance and total habitat area in small streams based on visual estimation methods. Canadian Journal of Fisheries and Aquatic Sciences 45:834–844.

Harrelson, C. C., Rawlins, C. L., and J. P. Potyondy. 1994. Stream channel reference sites: an illustrated guide to field technique. U.S. Forest Service, Rocky Mountain Forest and Range Experiment Station General Technical Report RM-245, Fort Collins, Colorado.

Heede, B. H., and J. N. Rinne. 1990. Hydrodynamics and fluvial morphologic processes: implications for fisheries management and research. North American Journal of Fisheries Management 10:249–268.

Hendrickson, D. L., and W. L. Minckley. 1984. Cienegas—vanishing climax communities of the American Southwest. Desert Plants 6:131–175.

Huedepohl, C. 2000. The deer and elk procedures manual. Alberta Agriculture, Food, and Rural Development. Alberta Queen's Printer, Edmonton, Alberta.

Humphrey, J. W., and G. S. Patterson. 2000. Effects of late summer cattle grazing on the diversity of riparian pasture vegetation in an upland conifer forest. Journal of Applied Ecology 37:986–996.

Hurlbert, S. H. 1984. Pseudoreplication and the design of ecological field experiments. Ecological Monographs 54:187–211.

Hygnstrom, S. E., R. M. Trimm, and G. E. Larson. 1996. Prevention and control of wildlife damage. Diane Publishing Company, Collingdale, Pennsylvania.

Jansen, A., and A. I. Robertson. 2001. Relationships between livestock management and the ecological condition of riparian habitats along an Australian river floodplain. Journal of Applied Ecology 38:63–75.

Johnson, C. F., P. Jones, and S. Spencer. 1998. A guide to classifying selected fish habitat parameters in lotic systems of west central Alberta. Alberta Conservation Association, Foothills Model Forest, Hinton, Alberta.

Karr, J. R., and E. W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington, D.C.

Kauffman, J. B., and W. C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications—a review. Journal of Range Management 37(5):430–438.

Keller, E. A., and Swanson, F. J. 1979. Effects of large organic debris on channel form and fluvial processes. Earth Surface Processes 4:361–380.

Kent, M., and P. Coker. 1994. Vegetation description and analysis: a practical approach. Wiley, Chichester, England.

Kie, J. G., V. C. Bleich, A. L. Medina, J. D. Yoakum, and J. W. Thomas. 1994. Managing rangelands for wildlife. Pages 663–688 *in* T. A. Bookout, editor. Research and management techniques for wildlife and habitats. The Wildlife Society, Bethesda, Maryland.

Knopf, F. L. 1988. Riparian wildlife habitats: more, worth less, and under invasion. Pages 20–22 *in* K. M. Mutz, D. J. Cooper, M. L. Scott, and L. K. Miller, editors. Restoration, creation and management of wetland and riparian ecosystems in the American West. PIC Technologies, Denver.

Kondolf, G. M. 1993. Lag in stream channel adjustment to livestock exclosure, White Mountains, California. Restoration Ecology 1:226–230.

Krueger, W. C. 1998. Integrating utilization measurements into monitoring programs. Pages 71–72 *in* Oregon State University, Agriculture Experiment Station, editor. Stubble heights and utilization measurements: uses and misuses. Oregon State University, Station Bulletin 682, Corvallis.

Larson, L. L., and S. L. Larson. 1996. Riparian shade and stream temperature: a perspective. Rangelands 18(4):149–152.

Likens, G. E. 1984. Beyond the shoreline: a watershed-ecosystem approach. Internationale Vereinigung fur theoretische und angewandte Limnologie Verhandlungen 22:1–22.

Lloyd, J. 1985. COWFISH: habitat capability model. U.S. Forest Service, Northern Regional Office, Missoula, Montana.

Mayer, R. 1999. Estimated costs for livestock fencing. Iowa State University, Extension Circular FM 1855, Ames, Iowa.

McNeil, W. F., and W. H. Ahnell. 1964. Success of pink salmon spawning relative to size of spawning bed materials. U.S. Fish and Wildlife Service Special Scientific Report-Fisheries No. 469, Washington, D.C.

Meals, D. W., and R. B. Hopkins. 2003. Phosphorus reductions following riparian restoration in two agricultural watersheds in Vermont, USA. Water Science and Technology 45(9):51–60.

Medina A. L. 1995. Native aquatic plants and ecological condition of southwestern wetlands and riparian areas. Pages 329–335 *in* D. W. Shaw, D. M. Finch, editors. Proceedings of the symposium of desired future conditions for southwestern riparian ecosystems: bringing interests and concerns together. U.S. Forest Service General Technical Report RM-272, Fort Collins, Colorado.

Medina, A. L., and J. N. Rinne. 1999. Ungulate/fishery interactions in southwestern riparian ecosystems: pretensions and realities. Proceedings of the North American Wildlife and Natural Resources Conference 62:307–322.

Medina, A. L. 2001. A preliminary view of water quality conditions of the upper Verde River. Pages 25–33 *in* C. Decarlo, C. Schlinger, and A. Springer, editors. Verde Watershed Symposium: state of the watershed in 2001. Cliff Castle Lodge and Conference Center, Camp Verde, AZ. May 17–19, 2001. Northern Arizona University, Flagstaff.

Medina, A. L., and J. E. Steed. 2002. West Fork Allotment riparian monitoring study: 1993–1999. U.S. Forest Service, Rocky Mountain Research Station Final Project Report, Volume I, Flagstaff, Arizona.

Medina, A. L., M. B. Baker, Jr., and J. R. Turner. 1997. Channel types and geomorphology of the upper Verde River. Journal of the American Water Resources Association 33(4):465–473.

Merritt, R. W., and K. W. Cummins. 1996. An introduction to the aquatic insects of North America. 3rd edition. Kendall and Hunt Publishing Company, Dubuque, Iowa.

Minckley, W. L. 1983. Status of razorback sucker, *Xyrauchen texanus*, in the lower Colorado River basin The Southwestern Naturalist 28:165–187.

Minckley, W. L., and G. K. Meffe. 1987. Differential selection by flooding in stream fish communities of the arid American Southwest. Pages 93–104 *in* W. J. Matthews and D. C. Heins, editors. Community and evolutionary ecology of North American stream fishes. University of Oklahoma Press, Norman.

Minckley, W. L., and J. E. Deacon. 1991. Battle against extinction: native fish management in the American West. University of Arizona Press, Tucson.

Montgomery, D. R., and J. M. Buffington. 1997. Channel-reach morphology in mountain drainage basins. Geological Society America Bulletin 109(5):596–611.

Montgomery, D. R., E. M. Beamer, G. R. Pess, and T. P. Quinn. 1999. Channel type and salmonid spawning distribution and abundance. Canadian Journal of Fisheries and Aquatic Sciences 56:377–387.

Moulton, S. R. II, J. G. Kennen, R. M. Goldstein, and J. A. Hambrook. 2002. Revised protocols for sampling algal, invertebrate and fish communities as part of the National Water-Quality Assessment Program. U.S. Department of Interior, U.S. Geological Service, Open File Report 02–150, Reston, Virginia.

Murphy, B. R., and D. W. Willis. 1996. Fisheries techniques, 2nd edition. American Fisheries Society, Bethesda, Maryland.

Myers, L. H. 1989. Grazing and riparian management in southwestern Montana. Pages 117–120 *in* R. E. Gresswell, B. A. Barton, and J. Kershner, editors. Practical approaches to riparian resource management: an education workshop. U.S. Bureau of Land Management, Billings, Montana.

Myers, T. J., and S. Swanson. 1995. Impact of deferred rotation grazing on stream characteristics in central Nevada: a case study. North American Journal of Fisheries Management 15:428–439.

NRCS (Natural Resources Conservation Service) 1996. National handbook of water quality monitoring: Part 600. U.S. Department of Agriculture, Natural Resources Conservation Service, National Water and Climate Center, Portland, Oregon.

NRCS (Natural Resources Conservation Service). 1999. A procedure to estimate the response of aquatic systems to changes in phosphorus and nitrogen inputs. U.S. Department of Agriculture, Natural Resources Conservation Service, Washington, D.C.

Neary, D. G., A. L. Medina, and M. B. Baker, Jr. 2001. Geomorphic conditions of the riparian zone, Upper Verde River. Pages 45–52 *in* C. Decarlo, C. Schlinger, and A. Springer, editors. Proceedings, Verde Watershed Symposium: state of the watershed in 2001. Cliff Castle Lodge and Conference Center, Camp Verde, AZ. May 17–19, 2001. Northern Arizona University, Flagstaff.

Neary, D. G., and J. N. Rinne. 1997. Baseflow trends in the upper Verde River relative to Fish Habitat Requirements. Hydrology and Water Resources in Arizona and the Southwest 27:57–63.

Neary, D. G., and J. N. Rinne. 2001. Baseflow trends in the Verde River revisited. Hydrology and Water Resources of the Southwest 31:37–44.

Newton, B., C. Pringle, and R. Bjorkland. 1998. Stream visual assessment protocol. U.S. Natural Resources Conservation Service, National Water and Climate Center, Technical Note 991, Washington, D.C.

O'Grady, M. 2002. Proceedings of the 13th international salmonid habitat enhancement workshop. Central Fisheries Board, Dublin.

Pflieger, W. L. 1975. The Fishes of Missouri. Missouri Department of Conservation, Columbia.

Pfankuch, D. J. 1975. Stream reach inventory and channel stability evaluation. U.S. Forest Service, Northern Region, Missoula, Montana.

Plafkin, J. L., M. T. Barbour, K. D. Porter, S. K. Gross and R. M. Hughes. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. U.S. Environmental Protection Agency, EPA/444/4–89-001, Washington, D.C.

Platts, W. S. 1990. Managing fisheries and wildlife on rangelands grazed by livestock: a guidance and reference document for biologists. Nevada Department of Wildlife, Reno.

Platts, W. S. 1991. Livestock grazing. Pages 389–423 *in* W. R. Meehan, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. American Fisheries Society, Special Publication 19, Bethesda, Maryland.

Platts, W. S., C. Armor, B. D. Booth, M. Bryant, J. L. Bufford, P. Cuplin, S. Jensen, G. W. Leinkaemper, G. W. Minshall, S. B. Monsen, R. L. Nelson, J. R. Sedell, and J. S. Tuhy. 1987. Methods for evaluating riparian habitats with applications to management. U.S. Forest Service, Intermountain Forest and Range Experiment Station, General Technical Report INT-221, Ogden, Utah.

Platts, W. S., K. A. Gebhardt, and W. L. Jackson. 1985. The effects of large storm events on basin-range riparian stream habitats. Pages 30–34 *in* R. R. Johnson, C. D. Ziebel, D. R. Patton, P. F. Ffolliott, and R. H. Hamre, editors. Proceedings of the first North American riparian conference. Riparian ecosystems and their management: reconciling and conflicting uses. U.S. Forest Service General Technical Report RM-120, Fort Collins, Colorado.

Platts, W. S., and M. L. McHenry. 1988. Density and biomass of trout and char in western streams. U. S. Forest Service, Intermountain Research Station, GTR-INT-241, Ogden, Utah.

Platts, W. S., W. F. Megahan, and G. W. Minshall. 1983. Methods for evaluating stream riparian, and aquatic conditions. U.S. Forest Service, Intermountain Forest and Range Experiment Station General Technical Report INT-138, Ogden, Utah.

Platts, W. S., and R. L. Nelson. 1988. Fluctuations in trout populations and their implications for land-use evaluation. North American Journal of Fisheries Management 8:333–345.

Poole, G. C., C. A. Frissell, and S. C. Ralph. 1997. In-stream habitat unit classification: inadequacies for monitoring and some consequences for management. Journal of the American Water Resources Association 33(4):879–896.

Rankin, E. T. 1991. The use of the qualitative habitat evaluation index for use attainability studies in streams and rivers in Ohio. Page 25–38 in G. Gibson, editor. Biological criteria: research and regulation. U.S. Environmental Protection Agency, Office of Water, EPA 440/5–91-005, Washington, D.C.

Rankin, E. T. 1995. Habitat indices in water resource quality assessments. Pages 181–208 *in* W. S. Davis and T. P Simon, editors. Biological assessment and criteria: Tools for water resource planning and decision making. Lewis Publishers, Boca Raton, Florida.

Rinaldi, M., and Casagli, N. 1999. Stability of streambanks formed in partially saturated soils and effects of negative pore water pressures: the Sieve River (Italy). Geomorphology 26(4):253–277.

Rinne J. N. 1985. Livestock grazing effects on southwestern streams: A complex research problem. Pages 295–299 *in* R. R. Johnson, C. D. Ziebel, D. R. Patton, P. F. Ffolliot, and R. H. Hamre, editors. Proceedings of the first North American riparian conference. Riparian ecosystems and their management: reconciling and conflicting uses. U.S. Forest Service General Technical Report RM-120, Fort Collins, Colorado.

Rinne, J. N. 1988. Grazing effects on stream habitat and fishes: research design considerations. North American Journal of Fisheries Management 8:240–247.

Rinne, J. N. 1992. Physical habitat utilization of fishes in a Sonoran Desert stream, Arizona, southwestern United States. Ecology of Freshwater Fish 1992:35–41.

Rinne, J. N. 1995. Interactions of predation and hydrology on native southwestern fishes: Little Colorado spinedace in Nutrioso Creek, Arizona. Hydrology and Water Resources in Arizona and the Southwest. 22/ 25:33–38.

Rinne, J. N. 1999a. Fish and grazing relationships: the facts and some pleas. Fisheries 24(8):12–21.

Rinne, J. N. 1999b. The status of spikedace, *Meda fulgida*, in the Verde River, 1999. Implications for research and management Hydrology and Water Resources in Arizona and the Southwest 29:57–64.

Rinne J. N. 2000. Fish and grazing relationships in southwestern United States. Pages 329–371 *in* R. Jamison, and C. Raish, editors. Ecological and socioeconomic aspects of livestock management in the Southwest. Elsevier, Amsterdam.

Rinne, J. N. 2001a. Nonnative, predatory fish removal and native fish response: Verde River, Arizona, 1999–2000. Hydrology and Water Resources of the Southwest 31:29–36.

Rinne, J. N. 2001b. Effects of substrate composition on Apache trout *Oncorhynchus apache*. Journal of Freshwater Ecology 16(3):355–365.

Rinne, J. N. 2001c. Relationship of fine sediment and two native southwestern fish species Hydrology and Water Resources of the Southwest 31:67–70.

Rinne, J. N. 2002. Hydrology, geomorphology and management: implications for sustainability of native southwestern fishes. Hydrology and Water Resources of the Southwest 32:45–50.

Rinne, J. N. 2003a. Native and introduced fishes: their status, threat, and conservation. Pages 193 to 213 *in* P. F. Ffolliott, M. B. Baker, L. F. DeBano, and D. G. Neary, editors. Ecology, hydrology and management of riparian areas in the southwestern United States. CRC Press, Boca Raton, Florida.

Rinne, J. N. 2003b. Fish habitats: conservation and management implications. Pages 277 to 297 *in* P. F. Ffolliott, M. B. Baker, L. F. DeBano, and D. G. Neary, editors. Ecology, hydrology and management of riparian areas in the southwestern United States. CRC Press, Boca Raton, Florida.

Rinne, J. N. In press. Riparian restoration: grazing management considerations for monitoring project effectiveness. In J. Rinne, R. Hughes, and R. Calamusso, editors. Changes in large river fish assemblages in North America: implications for management and sustainability of native species. American Fisheries Society, Bethesda, Maryland.

Rinne, J. N., and M. A. Alexander. 1995. Nonnative salmonid predation on two threatened native species: preliminary results of field and laboratory studies. Proceedings of the Desert Fishes Council 26:114–116.

Rinne, J. N. and W. L. Minckley. 1991. Native fishes in arid lands: A dwindling natural resource of the desert Southwest. U.S. Forest Service Rocky Mountain Forest and Range Experiment Station, General Technical Report 206, Fort Collins, Colorado.

Rinne, J. N., and A. L. Medina. 1996. Implication of multiple use management strategies on southwestern (USA) native fishes. Pages 110–123 *in* R. M. Meyer, editor. Fisheries resource utilization and policy, proceedings of the World Fisheries Congress, Theme 2. Oxford and IBH Publishing Company, New Deli.

Rinne, J. N., and D. G. Neary. 1997. Stream channel and fish relationships: preliminary observations, Verde River, Arizona. Pages 475–482 *in* J. J. Warwick, editor. Proceedings of the American Water Resources Agency Symposium, water resources education, training, and practice: opportunities for the next century. American Water Resources Agency, Las Vegas, Nevada, June 29–July 3. American Water Resources Association, Herndon, Virginia.

Rinne, J. N., and J. A. Stefferud. 1996. Relationships of native fishes and aquatic macrohabitats in the Verde River, Arizona. Hydrology and Water Resources in Arizona and the Southwest 26:13–22.

Rinne, J. N., and J. A. Stefferud. 1997. Factors contributing to collapse yet maintenance of a native fish community in the desert Southwest (USA). Pages 157–162 *in* D. A. Hancock, D. C. Smith, A. Grant, and J. P. Beaumer, editors. Developing and sustaining world fisheries resources: the state of science and management. Second World Fisheries Congress, Brisbane, Australia.

Rinne, J. N., and B. P. Deason. 2000. Habitat availability and utilization by two native, threatened fish species in two southwestern rivers. Hydrology and Water Resources in the Southwest 30:43–52.

Rinne, J. N., J. A. Stefferud, A. Clark, and P. Sponholtz. 1998. Fish community structure in the Verde River, Arizona, 1975–1997. Hydrology and Water Resources in Arizona and the Southwest 28:75–80.

Rinne, J. N., B. Holland, and G. Sundnes. 2002. Comparative study of heart rates in fishes from cold and temperate sea water and warm (hot) desert rivers. Pages 13–26 *in* K. Gamperl, T. Farrel, and D. McKinlay, editors. Proceedings of the Symposium on Cardiovascular physiology of fish. 5th International Congress on Biology of Fishes. Vancouver, BC.

Robertson, A. I., and R. W. Rowling. 2000. Effects of livestock on riparian zone vegetation in an Australian dryland river. Regulated Rivers: Research and Management 16:527–541.

Roni, P., T. J. Beechie, R. E. Bilby, F. E. Leonetti, M. M. Pollock, and G. R. Pess. 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest watersheds. North American Journal of Fisheries Management 22:1–20.

Roni, P., M. Liermann, and A. Steel. 2003. Monitoring and evaluating responses of salmonids and other fishes to in-stream restoration. Pages 318–339 *in* D. R. Montgomery, S. Bolton, D. B. Booth, and L. Wall, editors. Restoration of Puget Sound rivers. University of Washington Press, Seattle.

Rosenburg, D. M., and V. H. Resh. 1993. Freshwater biomonitoring and benthic macroinvertebrates. Chapman Hall, New York.

Rosgen, D. L. 1996. Applied river morphology. Wildland Hydrology, Pagosa Springs, Colorado.

Rosgen, D. L. 2001a. A practical method of computing streambank erosion rate. Pages 9–17 *in* Interagency Advisory Committee on Water Quality, editors. Proceedings of the 7th Federal Interagency Sedimentation Conference, March 25–29, Reno, Nevada. Volume II. Interagency Advisory Committee on Water Quality, Reston, Virginia.

Rosgen, D. L. 2001b. A stream channel stability assessment methodology. Pages 18–26 *in* Interagency Advisory Committee on Water Quality, editors. Proceedings of the 7th Federal Interagency Sedimentation Conference, March 25–29, Reno, Nevada. Volume II. Interagency Advisory Committee on Water Quality, Reston, Virginia.

Rosgen, D. L. 2001c. A hierarchical river stability/watershed based sediment assessment technology. Pages 97–106 *in* Interagency Advisory Committee on Water Quality, editors. Proceedings of the 7th Federal Interagency Sedimentation Conference, March 25–29, Reno, Nevada. Volume II. Interagency Advisory Committee on Water Quality, Reston, Virginia.

Schuett-Hames, D., A. Pleus, L. Bullchild, and S. Hall. 1994. Timber-Fish-Wildlife ambient monitoring manual. Northwest Indian Fisheries Commission, TFW-AM9-94-001, Olympia, Washington.

Simon, A., A. Curini, S. E. Darby, and E. J. Langendoen. 1999. Streambank mechanics and the role of bank and near-bank processes in incised channels. Pages 123–152 *in* S. E. Darby, and A. Simon, editors. Incised river channels: processes, forms, engineering and management. Wiley, Chichester, England.

Simon, A., and A. J. C. Collison. 2002. Quantifying the mechanical and hydrologic effects of riparian vegetation on streambank stability. Earth Surface Processes and Landforms 27(5):527–546.

Stefferud, J. A., and J. N. Rinne. 1995. Preliminary observations on the sustainability of fishes in a desert river: the roles of streamflow and introduced fishes. Hydrology and Water Resources in Arizona and the Southwest 22/25:26–32.

Steinman, A. D., and G. A. Lamberti. 1996. Biomass and pigments of benthic algae. Pages 295–314 *in* F. R. Hauer and G. A. Lamberti, editors. Methods in stream ecology. Academic Press, San Diego, California.

Swift, B. L. 1984. Status of riparian ecosystems in the United States. Water Resources Bulletin 20(2):223–228.

Sylte, T. L., and J. C. Fischenich. 2002. Techniques for measuring substrate embeddedness. U. S. Army Engineer Research and Development Center, EMRRP Technical Notes Collection, ERDC TN-EMRRP-SR-36, Vicksburg, Maryland.

Thompson, W. L. 2003. Hankin and Reeves' approach to estimating fish abundance in small streams: limitations and alternatives. Transaction of the American Fisheries Society 132:69–75.

USFS (U. S. Forest Service). 1998. General aquatic wildlife surveys-manual. U.S. Forest Service Region 3, Albuquerque, New Mexico.

USFWS (U. S. Fish and Wildlife Service). 1980. Habitat evaluation procedures (HEP). U.S. Fish and Wildlife Service, Division of Ecological Services, Manual ESM 02, Washington, D.C.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130–137.

Waters, T. F. 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society, Monograph 7, Bethesda, Maryland.

Weigel, B. M., J. Lyons, L. K. Paine, S. I. Dodson, and D. J. Undersander. 2000. Using stream macroinvertebrates to compare riparian land use practices on cattle farms in southwestern Wisconsin. Journal of Freshwater Ecology 15:93–106.

Weigel, D. E., and P. W. Sorensen. 2001. The influence of habitat characteristics on the longitudinal distribution of brook, brown, and rainbow trout in a small Midwestern stream. Journal of Freshwater Ecology 16:599–614.

Williams, J. E., J. E. Johnson, D. A., Hendrickson, S. Contreras-Balderas, J. D. Williams, M. Navarro-Mendoza, D. E., McAllister, and J. E. Deacon. 1989. Fishes of North America endangered, threatened, and of special concern: 1989. Fisheries 14(6):2–21.

Winward, A. H. 2000. Monitoring the vegetation resources in riparian areas. U.S. Forest Service, Rocky Mountain Research Station General Technical Report RMRS-GTR-47, Ogden, Utah.

Wohl, N. E., and R. F. Carline. 1996. Relations among riparian grazing, sediment loads, macroinvertebrates, and fishes in three central Pennsylvania streams. Canadian Journal of Fisheries and Aquatic Sciences 53:260–266.

Worley, J. W., and G. Heusner. 2000. Fences for horses. University of Georgia, Cooperative Extension Service Bulletin 1192, Augusta.

Young, D. W. 1998. The history of cattle grazing in Arizona. Hydrology and Water Resources in Arizona and the Southwest 28:13–17.

Zar, J. H. 1999. Biostatistical analysis. Prentice Hall, Upper Saddle River, New Jersey.