Recovery of Channel Morphology and Benthic Macroinvertebrate Assemblages after Livestock Exclusion

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Abstract

Measurements in paired stream reaches with and without livestock access in southwestern Virginia suggest that livestock exclusion practices installed on short, isolated stream reaches result in improved geomorphic and riparian vegetation condition, but do not significantly improve the benthic macroinvertebrate assemblage. Detailed longitudinal and cross-sectional surveys, pebble counts, and rapid geomorphic assessments were conducted on contiguous, paired stream reaches (5 pairs) with and without active livestock access across a range of time since livestock exclusion was implemented. In addition, bank characteristics were quantified by measuring groundcover biomass, shrub crown volume, tree density and diameter, soil bulk density, and particle-size analysis. Benthic macroinvertebrates were collected with a D-frame dip net and quantified using the Virginia Stream Condition Index (SCI), and other benthic macroinvertebrate metrics. We determined that: 1) there were significant differences in stream morphology, streambank soils, and riparian vegetation between paired grazed and livestock exclusion reaches; however, the benthic macroinvertebrate assemblages were not significantly different; and, 2) the Reach Condition Index (RCI), a qualitative geomorphic assessment methodology, increased with time since livestock exclusion. All other parameters did not show a clear temporal response.

These observations suggest that a more targeted and holistic approach that addresses watershed-wide impacts must be implemented to restore aquatic ecosystems.

Dedication

To my late grandmother - her hard work and perseverance in India allowed my parents, siblings, and me to lead prosperous lives in America.

To my friends Julia Pryde, Brian Bluhm, and Matt Gwaltney - their contributions to the field of water resources engineering will never be forgotten.

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Chapter 1. Introduction

1.1 Background and Motivation

Many streams in the United States are experiencing severe bank retreat and habitat destruction due to livestock access (VDCR, 2006). Unrestricted stream access results in frequent defecation in and near the water, a decrease in riparian vegetation diversity, and weakened streambanks that are highly erodible and physically degraded (Chaney et al., 1990). The loss of riparian vegetation often results in increased water temperatures and increased fine bed sediment; hence, aquatic habitat diversity is greatly reduced (Chaney et al., 1990). Platts (1982) concluded that direct livestock access to streams had the largest impact on degraded stream and riparian environments when compared to other non-point source pollution sources. A review by Belsky et al. (1999) indicated that approximately 85% of riparian livestock studies concluded livestock access negatively impacts aquatic habitat and stream morphology.

The USDA-NRCS has made a notable effort to keep livestock out of streams. One of the Agency's programs focused on limiting livestock access to streams is the Conservation Reserve Enhancement Program (CREP). CREP is an expansion of the federal Conservation Reserve Program (CRP), which was established in 1985 and has been implemented on more than 39 million acres nationwide. CREP is Virginia's most funded conservation program, with $91 million in federal and state funding allocated for the program yearly (VDCR, 2006). The goal of CREP is to reduce the incidence of runoff, sediment, and nutrients from agricultural production into streams through livestock exclusion and the establishment of pollution-filtering vegetation. This is accomplished through the installation of streambank fencing, and the establishment of riparian buffers, filter strips, wetland areas, trees, and grasses to improve and protect water quality (USDA-NRCS, 2005). Other CREP-like fencing and buffering projects have been implemented by individual farmers. The intent of these conservation efforts are frequently similar to those associated with CREP.

1.2 Research Objectives

Many Best Management Practices (BMPs) are not monitored to evaluate their effectiveness following implementation, including livestock exclusion practices (Blann et al., 2002). The goal of this research was to determine the success of livestock exclusion practices in improving channel morphology, bank characteristics, and benthic macroinvertebrate assemblages over time. To accomplish this, we focused on two main objectives:

1. Assess the effectiveness of existing livestock exclusion projects in southwest Virginia; and,
2. Evaluate the time distribution for improved channel morphology and benthic macroinvertebrate assemblages once exclusion projects have been implemented.

To test these research objectives, the following hypotheses were developed:

<u>Hypothesis 1:</u> There is a significant difference in the geomorphology, bank characteristics, and benthic macroinvertebrate assemblages between the grazed and fenced reaches.

<u>Hypothesis 2:</u> There is a gradient of improvement associated with the length of time livestock exclusion practices have been in place.

Chapter 2. Literature Review

2.1 Riparian Buffers

Riparian buffer implementation is encouraged to protect the physical characteristics of stream ecosystems and to filter potential sediment, and nutrient inputs (Peterjohn and Correll, 1984; Cooper et al., 1987; Welsch, 1991; Jordan et al., 1993; USDA-NRCS, 1997; Figure 2.1). Wooded riparian buffers are established to improve physical habitat for macroinvertebrates by creating overhead shade and instream woody debris (Blann et al., 2002). They also influence geomorphologic processes, bank stability, and water retention and infiltration (Barton et al., 1985; Rinne, 1988; Gregory et al., 1991; Gordon et al., 1992; Montgomery, 1997).

The Chesapeake Executive Council adopted a plan in October 1996 to plant 3,235 km (2,010 miles) of riparian forest buffers along streambanks and shorelines in the Chesapeake Bay basin jurisdictions of Maryland, Virginia, Pennsylvania, and the District of Columbia by the year 2010 (Blankenship, 1996; USEPA, 1997). This goal was established to prevent polluted runoff from entering Chesapeake Bay tributaries (Chesapeake, 2000). Section 303(d) of the Clean Water Act requires that a Total Maximum Daily Load (TMDL) Priority List is submitted to the EPA for each state; waters that do not meet state water quality standards are reported on the 303(d) Impaired Waters in Virginia list (VDEQ, 2006). Programs such as the United States Department of Agriculture-Natural Resources Conservation Service (USDA-NRCS) Conservation Reserve Program (CRP) and Elizabeth River Project in Eastern Virginia were used to accomplish the Chesapeake Executive Council's riparian buffer goal, eight years before the deadline August 2002 (USEPA, 1997; Brown, 2003).

2.1.1 Livestock Grazing Impacts on Riparian Buffers

Many riparian areas have been degraded by intensive livestock grazing; however, livestock removal results in the improvement of these ecosystems (Rickard and Cushing, 1982; Stuber, 1985). Often restoration programs, such as Environmental Quality Incentives Program (EQIP) and the Conservation Reserve Enhancement Program (CREP), are implemented to remove livestock from the stream (VDCR, 2006). The goal of these programs is to reduce the incidence of runoff, sediment, and nutrients from agricultural production into streams through the installation of streambank fencing and riparian buffers (Blann et al., 2002; USDA-NRCS, 2005; VDCR, 2006).

Riparian zones degraded by livestock grazing have little vegetation to protect and stabilize the banks and poor habitat diversity (Chaney at al., 1990). With limited herbaceous and groundcover vegetation to protect the streambanks, bank erosion, and mass failure are more likely to occur (Blann et al., 2002). In contrast, restored riparian areas with successional buffers have diverse riparian vegetation, relatively less bank erosion, stable banks, wider channels, and improved aquatic habitat diversity (Zimmerman et al., 1967; Murgatroyd and Ternan, 1983; Beschta and Platts, 1986; Chaney et al., 1990; Peterson, 1993; Sweeney, 1993; Trimble, 1993, 1997, 1999; Sovell et al., 2000; Nerbonne and Vondracek, 2001).

The impact of livestock grazing on vegetation growth can be separated into soil compaction and vegetation removal (Severson and Boldt 1978). Trampling and compaction of streambank soils can reduce macropore size and infiltration therefore inhibiting new vegetative growth (Orr, 1960; Rauzi and Hanson, 1966). Grazing in riparian areas can destabilize and break down streambanks through the loss of vegetation and physical degradation of the banks (Clary, 1999); consequently, soil erosion and channel width increase (Laubel et al., 2003; Williamson et al., 1992).

Soil compaction produced by livestock trampling can exert pressures on the upper 15 cm of soil equivalent to those of heavy tractors (reviewed by Lull, 1959). In a paired grazed and ungrazed field study in Utah, Laycock and Conrad (1967) took bulk density samples on the rangeland and determined there was no significant difference between study sites. In a study at the Sonora Agricultural Research Station in Texas, bulk density was similar in continuously grazed, heavily stocked, rotationally grazed pastures, and fenced study sites (McGinty et al., 1979). In contrast, Wood and Blackburn (1984) conducted a study in Texas; they compared pastures with livestock exclusion to pastures with rotational, light, moderate, or continuous grazing. They determined that soil bulk density was significantly higher in the lightly and continuously grazed pastures than the livestock exclusion areas. Alderfer and Robinson (1949), Bryant et al. (1972), Orr (1960), and Rauzi and Hanson (1966) also determined soil compaction increased with grazing intensity.

Livestock grazing and trampling in riparian areas can reduce vegetation on streambanks, making them more susceptible to erosion (Kauffman and Krueger 1984; Trimble and Mendel 1995; Fitch and Adams 1998; Soto-Grajales 2002; Strand and Merritt 1999; Williamson et al., 1992). In a study conducted by Williamson et al. (1992) in Southland New Zealand the effects of grazing and channelization on stream morphology, vegetation, and aquatic habitat were studied. The treatments were grazed versus ungrazed and channelised versus unchannelised reaches in third and forth order streams. Williamson et al. (1992) found that cattle trampling the banks resulted in the removal of vegetation on the banks and subsequent erosion and widening. Sovell et al. (2000) conducted a study in Minnesota on 17 sites on first to forth order streams. The study design compared first order streams in two paired watersheds and a longitudinal design was utilized on the third to forth order streams. Different types of riparian vegetation management were compared; the treatments were fenced with a wooded buffer, fenced with a grass buffer, and unfenced continuously and rotationally grazed sites. The percent fines in the streambed were significantly higher at sites with wood buffers than grass and rotationally grazed areas, canopy cover was similar at sites with wood and grass buffers, a higher percentage of streambank soil was exposed at the continuously grazed site than the rotationally grazed site, and there were no significant differences in the benthic macroinvertebrate metrics between sites. In a study on Sheep Creek, a headwater stream in the Roosevelt National Forest in northcentral Colorado, Schulz and Leininger (1990) compared the riparian vegetation density and cover in grazed and livestock exclusion reaches in July-August 1985 and 1986. They found the grazed riparian areas had approximately five times as much bare ground than areas with long-term livestock exclusion (29 years).

Paine and Ribic (2002) conducted a study on spring-fed stream reaches in the unglaciated area of Southwestern Wisconsin. They compared four different management types including woody buffer strip, grassy buffer strip, and unfenced continuously and rotationally grazed stream reaches. The woody buffer strip had the greatest amount of native vegetation remaining at the end of the study, while the reach with continuous grazing had the least amount of vegetative growth. Flenniken et al. (2001) conducted a plot study on the hydrologic response of a montane riparian community in northern

Colorado following cattle use. Four treatments were created: a grazed plus trampled plot; a trampled plot; a mowed plot; and, a control plot with natural vegetation and no treatment. This study showed there was a 50% decrease in stem density in the grazed and trampled plot as compared to the control. The reduction in stem density increased the amount and rate of runoff from the plot. It also decreased the sinuosity of the micro channels created in the plot due to overland flow.

2.2 Stream Morphology

2.2.1 Livestock Grazing Impacts on Stream Morphology

Channel geometry is influenced by bank erosion processes, which may be grouped into three categories: weakening processes, direct fluid entrainment, and mass failure (Lawler, 1992). The main factors influencing these three erosion processes are stream hydraulics, bank soil and vegetation composition, climate, subsurface soil and hydrologic conditions, biological and human-induced factors such as livestock grazing (Knighton, 1998). Large groups of wild and domesticated animals including herds of elephants, hippopotami, bison, moose, elk, caribou, mountain goats, bighorn sheep, and cattle increase bank erosion through intensive trampling, which results in vegetative and soil degradation, reduced infiltration capacity and accelerated runoff and channel entrenchment (Butler, 2002). Trimble and Mendel (1995) suggested livestock access to the stream results in the reduction of erosion resistance and increased overall channel retreat. Numerous studies have found that livestock access creates a combination of reduced channel boundary resistance and increased stream power so that bank erosion and mass failure occur (Chaney et al., 1990; Platts, 1991; BLM, 1994; Marston, 1994).

Specific changes in channel physical dimensions indicate improved channel morphology, including mean depth at bankfull stage elevation, width to depth ratio, entrenchment ratio, and channel slope (FISRWG, 1998). Figure 2.2 depicts bankfull width, depth, and elevation, which form the basis of some channel physical parameters.

Nine fifty meter stream reaches consisting of pasture, forest, and transitional (pasture/forest) landuse on three first and second order streams were studied by Scarsbrook and Halliday (1999) in Waikato, New Zealand. They determined that the channel width was significantly higher in pasture reaches than the forested reaches. Nagle and Clifton (2003) compared second order grazed and fenced stream reaches on Wickiup Creek in eastern Oregon in 1986 and 1998. Both times the reaches were surveyed, significant differences were found in the shape of the channel between the reaches with and without cattle access to the stream. The authors determined the fenced reach had a smaller width to depth ratio and a lower mean depth at bankfull stage. The width to depth ratio is the ratio of the mean bankfull width over the mean bankfull depth.

McDowell and Magilligan (1997) found similar results for a study conducted on eight streams in Oregon. Grazed sites were compared to sites where cattle were excluded for periods ranging from six to thirty-one years. Most enclosed sites had a lower bankfull width and width to depth ratio than the grazed sites. In a study on a third order stream in the Sawtooth National Forest in central Idaho, Clary (1999) established grazing treatments ranging from no grazing, to light (20-25% impact), and medium (35-50% impact) grazing. He determined that as cattle grazing intensity increased, the stream channels widened, the width to depth ratio increased, and the banks became less stable.

A low entrenchment ratio indicates a wider active floodplain that helps to dissipate energy during flood flows, while a high entrenchment ratio indicates a narrow or inaccessible floodplain typically observed in incised channels. Deeply incised channels are typically subjected to higher shear forces during flood flows (Agouridis et al., 2005). Dobkin et al. (1998) conducted a study in eastern Oregon on three paired grazed and livestock exclusion 1.5 ha plots. They observed that following livestock exclusion from the riparian meadow, the entrenchment ratio increased, indicated the stream had improved access to a larger floodplain.

2.2.2 Geomorphic Assessments

Ecological and geomorphic assessments are used to compare disturbed stream reaches to reaches in their natural undisturbed state, prior to anthropogenic alterations (Steedman, 1994; Hughes, 1995; Jackson and Davis, 1995; Davies and Jackson, 2006). Following European settlement in the late 1800s, logging, channelization, and poor agricultural practices led to changes in flow regimes, sediment loads, and subsequent erosion (Waters 1977). Poor stream maintenance practices such as widening and straightening increased channel erosion by exposing underlying alluvium and increasing water velocities (Laubel et al., 2003; Williamson et al., 1992). The Unified Stream Methodology (USM) was underdeveloped by U.S. Army Corps of Engineers, Norfolk District and the Virginia Department of Environmental Quality (VDEQ) to assess human impacts on the ecological and geomorphic condition of stream reaches (USACE and VDEQ, 2007). The USM assigns a condition index score to a stream reach based on the historical impacts on the stream and the Best Management Practices (BMPs) used to mitigate the impacts (USACE and VDEQ, 2007).

Other similar rapid geomorphic assessments have been used to relate changes in channel geometry, bank instability, and aquatic habitat to human influences (Stoddard et al., 2006). Scarsbrook and Halliday (1999) used the Pfanfuch index to quantify channel stability; the Pfanfuch index scores fifteen variables in three regions of the stream channel, the transitional zone between the forest and cattle reaches had a significantly higher degree of channel instability. In a similar paired grazed and fenced study in New Zealand Parkyn et al. (2003) compared nine fenced and buffered reaches to grazed reaches. They found that the grazed reaches had significantly greater instability than the fenced reaches, according the Pfanfuch index.

Jansen and Robertson (2001) studied the effect of cattle management practices on riparian ecological condition in south-eastern Australia. An ecological condition index based on an index developed by Ladsen et al. (1999) was used to assess the habitat quality in an agricultural drainage basin. The index was used to evaluate habitat continuity and extent, vegetation cover and structural complexity, bank and soil structure and stability, standing and fallen debris, dominance of native vs. exotic vegetation, and indicator species. The authors determined that this rapid appraisal method was a valuable tool to assess habitat conditions in research studies with many sites and large spatial scales.

2.3 Instream Habitat and Water Chemistry

2.3.1 Instream Habitat

Many different methods and indexes have been developed to evaluate the population and diversity of benthic macroinvertebrates in streams (Resh et al., 1995). The Rapid Bioassessment Protocol (RBP) is a standard assessment method developed by the

EPA to rapidly assess habitat quality. It can be use to characterize the existence and severity of impairment to the water resource and to evaluate the effectiveness of control actions and restoration activities (Barbour et al., 1999). Braccia and Voshell (2006) used the RBP habitat assessment to determine reach-scale habitat quality. In five first-order streams in the Little River drainage basin in Floyd County Virginia, USA based on the Rapid Habitat Assessment (RHA) methodology, the study reaches received an overall habitat score between zero (poor habitat condition) and two-hundred (optimal habitat condition) based on streambed characteristics, channel morphology, bank structure, and the riparian zone (Barbour et al., 1999). The control site had not been subject to cattle grazing for 15 years. Site two had a rotational grazing practice, while sites three through five had continuous cattle access to the stream. The reference reach and light rotationally grazed RHA scores were much higher (159 and 142) than the intermediate, heavily, and very heavily grazed reaches (113-116) indicating cattle access to streams degrades aquatic habitat (Braccia and Voshell, 2006).

2.3.2 Water Chemistry

Agricultural activities can influence water chemistry measurements; Ott et al. (1991) and Galeone et al. (2006) determined nitrate concentrations increase with increase agricultural land use. In stream phosphorus concentrations were significantly lower after cattle fencing (Galeone et al., 2006). Braccia and Voshell (2006) found higher electrical conductivity and maximum diurnal water temperature increased with cattle grazing intensity. Similarly, Harding et al. (2006) found conductivity and temperature were consistently higher in agricultural reaches as compared to forested reaches. Scarsbook and Halliday (1999) determined that there was an increase in pH and a decrease in electrical conductivity in the grazed reach relative to the transitional forest reach. Generally unrestricted cattle access to the stream tends to result in increased nutrient concentrations, electrical conductivities, and pH values (Ott et al., 1991; Scarsbrook and Halliday, 1999; Braccia and Voshell, 2006; Galeone et al., 2006; Harding et al., 2006).

PH values that are outside of an organism's tolerance range can create toxic conditions (USEPA, 2006). In addition, high water temperatures can reduce the aquatic diversity in a stream reach (Thyssen and Erlandsen, 1987; Richardson et al., 1994; Wilcock et al., 1995; USEPA, 2006). Streams with high temperatures and elevated nutrient concentrations can develop uncontrolled algae growth. Elevated total suspended solids, nitrate, and phosphate concentrations can lead to excess growth of undesirable algal species, which can block access to microhabitat features and reduce the growth of other filamentous algae needed for an organism's survival (reviewed by Belsky et al., 1999; USEPA, 2006).

2.4 Benthic Macroinvertebrate Assemblages

2.4.1 Benthic Macroinvertebrates Assessment Methods

Biological communities reflect the overall ecological integrity of a stream, while integrating the effects of different stressors and thus providing a measure of their overall impact (Barbour et al., 1999). Benthic macroinvertebrate assemblages are specifically used because they are good indicators of localized conditions (Rosenberg and Resh, 1993). They are particularly well-suited for assessing site-specific impacts because many benthic macroinvertebrates are sedentary (Barbour et al., 1999; Rosenberg and Resh, 1993).

The Hilsenhoff Biotic Index (HBI) can be used to compare organisms identified to the family or genus level, with each organism assigned an environmental tolerance value of zero to ten. Intolerant species rate a value of zero, while the most pollution tolerant species rate a value of ten (Bode, 1991). The Ephemeroptera, Plecoptera, Tricoperta (EPT) metric is used to measure the number of taxa in the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Tricoperta (caddiesflies); as the stream becomes more disturbed the value of the metric decreases (Barbour et al., 1999). The Virginia Stream Condition Index (VSCI) integrates eight metrics including the HBI and the EPT; it was developed specifically for streams in Virginia based on biomonitoring data from 1994 to 1998 (USEPA and VDEQ, 2003). This standard index provides a predetermined scale of biological condition so that the relative biological condition between study reaches can be compared (Resh et al., 1995; USEPA and VDEQ, 2003).

2.4.2 Livestock Grazing Impacts on Benthic Macroinvertebrates

A wide range of studies have been conducted to determine the response of benthic macroinvertebrate assemblages to livestock exclusion, some of which are summarized in Table 2.1. These studies quantified their data using a large range of metrics to determine the impact of livestock on the biological integrity of the stream.

Braccia and Voshell (2006) determined the percent of Coleoptera and crawler taxa strongly decreased with increased cattle grazing. They also quantified benthic macroinvertebrate response to various environmental stressors including organic matter, periphyton, flow, depth, and inorganic sediments: the authors determined the percentage of fine sediment in the bed had the largest impact on macroinvertebrate assemblage structure. Wohl and Carline (1995) found similar results in a riparian grazing study conducted on three streams in central Pennsylvania. Pairs of fenced and grazed reaches were compared on the three streams; it was determined that the grazed reaches consistently had highly erodible streambanks and an increase percent of bed fines. The macroinvertebrate density decreased with an increase in percent fines and a decrease in bed substrate size (Rabeni and Minshall, 1977; Minshall, 1984; Wohl and Carline, 1995).

Kyriakeas and Watzin (2006) conducted a study in Vermont involving nine sites within five watersheds. The land use at each site consisted of a cattle pasture where animals had unrestricted access to the stream, cornfields with either no buffer or a small buffer and where concentrated surface runoff entered the stream, and reference areas that were mostly forested and without adjacent agricultural activity. Benthic macroinvertebrates were collected at each of the sites and compared using multiple metrics. The percent crawler taxa (Elmidae) and nutrient-tolerant taxa (Hydropsyidae) were significantly higher at the grazed sites than at the cornfield or reference sites. Mean HBI values differed among all three site types with cattle sites showing highest values (5.3), corn sites intermediate values (4.8), and reference sites lowest values (3.8); however, these differences were not significant. Similarly, in the Sovell et al. (2000) implemented the HBI in their study. Sampling took place in spring 1995, fall 1996, and spring 1996; although the HBI was consistently higher in the continuously grazed reaches, a significant difference between the continuous and rotationally grazed sites was only detected in spring 1995.

Stone et al. (2005) conducted a study in an agricultural region of Illinois on benthic macroinvertebrate assemblages over a range of riparian forest cover. The treatments consisted of low (6%), medium (22%), and high (31%) percent forest cover. Study results showed that insect density and biomass was inversely related to forest cover. The HBI showed no significant differences among the reaches, since the lower cover reach value (3.9) was only slightly higher than the medium cover reach value (3.6). A multiple regression analysis on the HBI revealed that in-stream habitat was the most influential variable. The EPT values were extremely low and variable; this index failed to reveal any meaningful statistical comparisons. Harding et al. (2006) conducted a similar study on four streams in South Island, New Zealand. Eight study sites were located on forested stream reaches, while another four study sites were located on agriculturally impacted streams. Significantly more EPT taxa were collected in the forested than the agricultural fragments. In addition, the agricultural reaches experienced greater sedimentation than the forest fragments.

Nerbonne and Vondracek (2001) found significant correlations between channel shape and benthic macroinvertebrates communities in twenty-seven paired stream reaches in southeastern Minnesota. As the streambanks were eroded and width to depth ratio increased, the substrate percent fines and the density of tolerant taxa increased. Nerbonne and Vondracek (2001) concluded that reduced local sedimentation in the stream will result in improved habitat conditions. Their study also suggested grass buffers will maintain streambank stability and reduce the percent fines in stream substrates (Lyons et al. 2000).

Figure 2.1.
Three zone riparian buffer (USDA-NRCS, 1997)

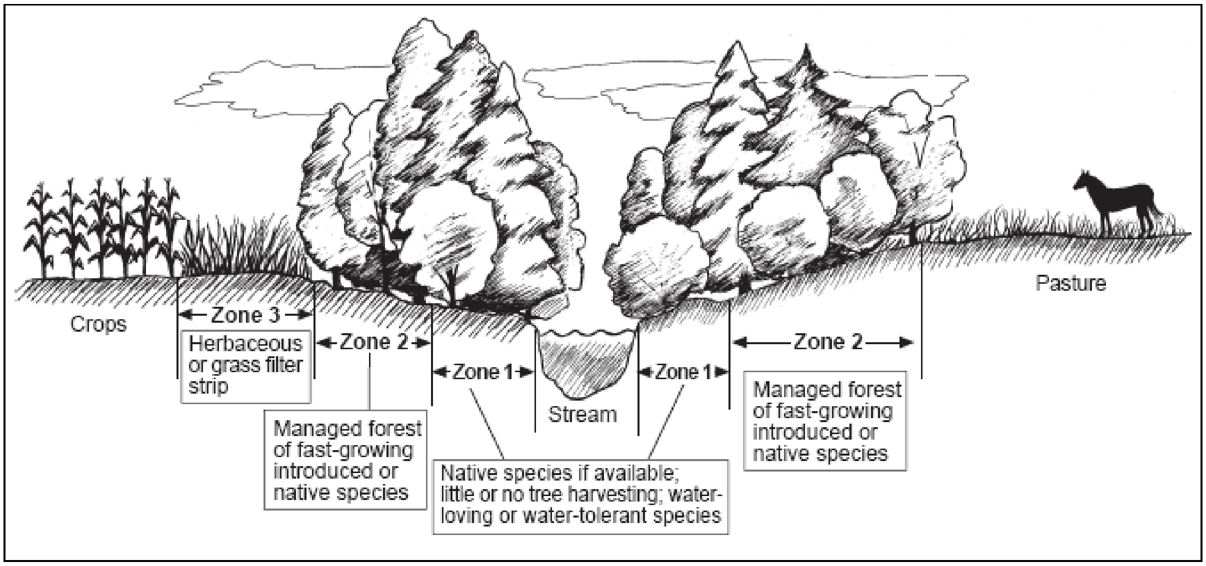


Figure 2.2.
Channel cross-section (FISRWG, 1998).

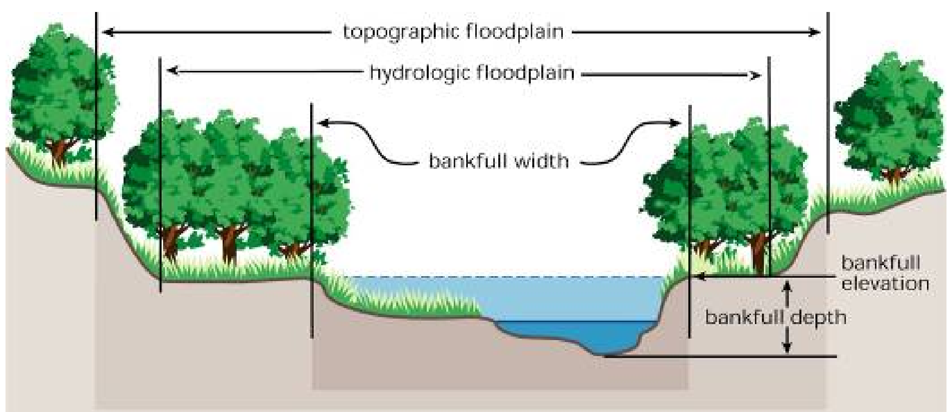


Table 2.1. Effect of riparian grazing on benthic macroinvertebrate assemblages

Author(s)	Study Location	Major Result
Braccia and Voshell (2006)	Virginia	percentage of fine sediment in the bed had the largest impact on macroinvertebrate assemblage structure
Wohl and Carline (1995)	Pennsylvania	macroinvertebrate density decreased with an increase in percent fines and a decrease in bed substrate size
Kyriakeas and Watzin (2006)	Vermont	percent crawler taxa and nutrient-tolerant taxa higher at the grazed sites than the cornfield or reference sites
Sovell et al. (2000)	Minnesota	HBI higher in the continuously than rotationally grazed reaches
Stone et al. (2005)	Illinois	in-stream habitat most influential variable on benthic assemblages
Harding et al (2006)	South Island, New Zealand	more EPT taxa were collected in the forested than the agricultural fragments
Nerbonne and Vondracek (2001)	Minnesota	as percent fines increased the tolerant taxa density increased

Chapter 3. Methods

3.1 Study Sites

The main goal of this study was to focus on reach-level impacts of livestock on the aquatic and geomorphologic integrity of streams. Five contiguous pairs of livestock exclusion and livestock reaches in southwest Virginia were evaluated (Figure 3.1). Since the pairs have the same contributing watershed any differences found would be due to local conditions (e.g. livestock or no livestock). Five study sites in southwest Virginia consisting of paired livestock exclusion and grazed stream reaches were identified for use in this study. The grazed reaches had unrestricted livestock access and the livestock exclusion reaches were implemented as part of the CREP program or have CREP-like practices implemented on the reach. The paired study reaches were located on the North Fork of the Roanoke River, Tom's Creek, Johns Creek, and Sinking Creek.

Comparison of the grazed reaches with livestock access and the adjacent livestock exclusion reaches allowed us to evaluate the effectiveness of livestock exclusion in improving benthic macroinvertebrate habitat and channel morphology. Four of the livestock exclusion reaches range in time since implementation from one to fourteen years, while one site is a grazed reach paired with a forested reach (Table 3.1).

Historical aerial photography of the study sites was obtained from the Virginia Tech Newman library and the Christiansburg and Bonsack United States Department of Agriculture - Natural Resources Conservation Service (USDA-NRCS) offices (Appendix A). In addition, digital orthophotography was reviewed for each of the paired research sites and the study sites were adjusted as to avoid obvious physical irregularities such as significant upstream urban impacts (VBMP, 2004).

Watershed characteristics for each reach including ecoregion, Total Maximum Daily Load (TMDL) impairments, land use, soils, drainage area, and watershed slope were obtained from analysis of digital spatial data in ArcGIS 9.0 (ESRI, 2007). All of the study sites are located in the Ridge and Valley Ecoregion. This forested mountainous region is composed of a large variety of geologic materials, including limestone, dolomite, shale, siltstone, sandstone, chert, mudstone, and marble (USEPA, 2007).

Total Maximum Daily Load (TMDL) impairments for each watershed were determined using the Virginia Department of Environmental Quality Geological Environmental Mapping System (GEMS). The North Fork of the Roanoke River and Johns Creek are listed as impaired in Virginia's 2002 Section 303(d) List of Impaired (Category 5) Waters due to water quality violations of the state fecal coliform standard (VADEQ, 2004). The source of the impairment in the North Fork of the Roanoke River and Johns Creek is believed to be nonpoint source pollution from urban, agricultural, and wildlife activities (VGEMS, 2007).

Data layers for land use, soils, drainage area, and watershed slope were obtained from the National Hydrology Database (NHD, 2007), the National Land Cover Database (NLCD, 2007), the National Watershed Boundary Dataset (NWBD, 2007), and the United States Geological Service (USGS, 1982; Table 3.2; Appendix A). A global positioning system (GPS) unit was used to locate the inlet and outlet of each study reach; this data was then georeferenced to USGS digital topographic maps.

3.2 Field Methods

3.2.1 Stream Morphology

To quantify the physical and biological condition of each reach, surveying and standard protocol methods for assessing stream health were used. The Virginia Department of Environment Quality Reach Condition Index (VDEQ RCI) was used to evaluate stream geomorphic condition, physical habitat condition, adjustment processes, and reach sensitivity (USEPA and VDEQ, 2006). This was done through the field analysis of channel condition, evaluation of in-stream habitat, and riparian land use/land cover. We utilized the VDEQ RCI results to quantify the physical condition of all ten study reaches, to compare the livestock exclusion and grazed reach conditions, and to evaluate the time required for streams to respond to livestock exclusion and riparian buffer improvement activities (Appendix B).

Longitudinal and cross-sectional profiles were developed for a minimum reach length of 20 times bankfull width at each of the ten study reaches. Detailed surveying using a Leica© TC 307 total station was used to quantify stream morphology parameters. The inlet and outlet of each reach was located with a global positioning system (GPS) with differential correction. Longitudinal profiles were surveyed using a total station to determine stream gradient, water surface slopes, and the location of bed features such as riffles and pools. Channel cross-sectional profiles were surveyed for two riffles and two pools along each reach and used to quantify channel dimensions and floodplain features, including bankfull width and hydraulic bankfull depth. Photos of the left and right streambanks and upstream and downstream views of each cross-section were taken (Appendix H).

The longitudinal survey data were analyzed to determine thalweg length, straight line distance, sinuosity, channel water surface and bed slope, and the number and length of the pools and riffles in each reach. The Geo-hydraulic Diversity Index (GDI) was used to compute the longitudinal thalweg reach depth variability (Skinner et al., 1998; Equation 3.1).

$$R_D = \frac{D_{98} - D_{02}}{D_{50}} \tag{3.1}$$

where R_D = reach depth variability

$D_{02,}$ $D_{50,}$ D_{98} = the 2nd, 50th, and 98th percentiles on the distribution curve for measured thalweg depths

The cross-sectional data were evaluated to determine water surface and bankfull width, average depth, width-depth ratio, cross-sectional area, and wetted perimeter (Appendix C). Average depth in each cross-section was determined by dividing the cross-section area by the width at bankfull stage (Mecklenburg, 2004; Equation 3.2). The width to depth ratio was calculated in each cross-section as the bankfull top width divided by the bankfull depth (Mecklenburg, 2004; Equation 3.3). The mean depth and average width to depth ratio were then calculated over the four surveyed cross-sections in each reach.

$$D_{bkf} = \frac{A_{bkf}}{W_{bkf}} \tag{3.2}$$

$$W_{bkf}/D_{bkf} = \frac{W_{bkf}}{D_{bkf}} \tag{3.3}$$

where D_{bkf} = Mean depth at bankfull stage (m)

A_{bkf} = Area at bankfull stage (m^2)

W_{bkf} = Width at bankfull stage (m)

W_{bkf}/D_{bkf} = Width to depth ratio

The reach-averaged grain-size distribution was determined using a modification of the Wolman (1954) pebble count. Reach-averaged median particle size was determined through the evaluation of each type of bedform (pools and riffles) (Riley et al., 2003). At each reach, one hundred pebbles were measured in two pools and two riffles for a total of 400 pebbles using a US Forest Service gravelometer (Appendix C). The reach d_{50} and riffle d_{50} were calculated for each study reach. The reach d_{50} is the median particle size for the 400 pebbles that were measured in each reach. The riffle d_{50} is the median bed particle size of two-hundred pebbles in two riffles. The geometric standard deviation of the particle size distribution was calculated for each reach using equation 3.4 (Chang, 1988).

$$\sigma_g = \sqrt{\frac{d_{84.1}}{d_{15.9}}} \tag{3.4}$$

where σ_g = geometric standard deviation of the particle size distribution

$d_{15.9}$ = 15.9% of the bed substrate distribution is finer (mm)

$d_{84.1}$= 84.1% of the bed substrate distribution is finer (mm)

Data from the longitudinal and cross-sectional surveys and pebble counts were entered into the Reference Reach Spreadsheet Version 4.0L (Mecklenburg, 1999), a Microsoft Excel® worksheet used to organize survey data from a single stream reach (Appendix G).

3.2.2 Streambank Soils and Riparian Vegetation

Soil bulk density was measured at each reach by taking an undisturbed soil core at each horizon using a 5 by 5 cm aluminum cylinder with a slide hammer (Wynn and Mostaghimi, 2006). Soil cores were taken at two random locations on the side of each streambank for a total of four locations (Appendix D). These samples were weighed and dried at 105°C within eight hours of sampling, following procedures outlined in the Soil Survey Laboratory Methods Manual (USDA-NRCS, 2004).

A particle-size analysis was conducted on each soil horizon following the methods described by the USDA Soil Survey Manual (USDA-NRCS, 2004). A composite sample of each soil horizon was collected from the streambank. The sample was dried and mixed to produce a homogenous sample. Approximately 3,400 cm^3 of the material was passed through a 2-mm sieve (No. 10). The soil samples were separated into sand, silt, and clay portions using sieve meshes ranging in size from 2 mm (No. 10) to 0.044 mm (No. 325). These subsamples were weighed to determine the overall composition of the soil at each study reach. Also, the particle-size distribution of coarse material was determined by sieving approximately 3,400 cm^3 through sieves ranging from 25 mm to 2 mm meshes (No. 10). Soils were also identified at each study site using the Field Book for Describing and Sampling Soils (USDA-NRCS, 2002) and compared to the soil-type listed in the Soil Survey Manual in Montgomery County, Virginia (Creggar et al., 1985). Since a soil survey has not been completed for the three study sites located in Craig County, an NRCS soil scientist identified the soil types present in the study reaches (Freyman, 2007; Appendix D).

The riparian buffer vegetation was assessed through measurements of trees, shrubs, and groundcover. Vegetation type and density were evaluated using three nested 1, 25, and 100 m^2 quadrats at two random locations on each streambank (Hession et al., 2000). Riparian buffer width was measured two ways: 1) by finding the distance from the edge of the streambank to the end of vegetation; and, 2) by measuring the distance from the edge of the streambank to the fenceline. These measurements were taken in the two random locations on each bank using a measuring tape (Appendix D).

All groundcover below 1m in height was cut to ground level in $1m^2$ areas; groundcover greater than 1m in height within the $1m^2$ area left uncut (Bonham, 1989). These samples were dried in an oven at 60°C and then weighed to determine dry biomass (kg/ha). Shrub crown volume was measured within the 25 m^2 quadrats. A geometric shape was determined for each shrub and appropriate measurements were made to calculate volume (Bryant and Kothmann, 1979; Bonham 1989). Basal stem area was determined by finding the geometric mean of the largest and smallest diameter of all the trees within the 100 m^2 quadrat (Davidson et al., 1991). Measurements were taken at breast height (1.4 m) using a tree caliper (Bonham, 1989). Trees were distinguished from shrubs based on the general size and shape of the plant (Wynn et al., 2004). Both trees and shrubs were identified to the family level using the taxonomic nomenclature in the USDA-NRCS PLANTS Database (USDA-NRCS, 2001; Appendix D).

3.2.3 Instream Habitat and Water Chemistry

The Rapid Bioassessment Protocol (RBP) was developed as a tool to evaluate reach-scale stream habitat quality using a Rapid Habitat Assessment (RHA) an evaluation of the macroinvertebrate community (Barbour et al., 1999). We used the RBP to rapidly assess the reach-scale habitat quality, to compare the benthic macroinvertebrate assemblages in the livestock exclusion and grazed reaches, and to evaluate the time required for the benthic macroinvertebrate assemblages to recover. The RHA was used to evaluate the habitat quality of each reach (Barbour et al., 1999). The RHA assigns a score ranging from 0 (poor habitat condition) to 200 (optimal habitat condition) to a stream reach based on streambed characteristics, channel morphology, bank structure, and riparian zone (Appendix E).

The stream water chemistry conditions were evaluated by measuring of conductivity, pH, temperature, and nutrient concentrations on each benthic macroinvertebrate sampling day. Conductivity, pH, and temperature were measured with a TDSTestr 3 meter (MP Biomedical, Solon, OH), a Hach meter (Hach Company, Loveland, CO), and a Control Company thermometer (Control Company, Friendswood, Texas), respectively. A water grab-sample was collected at each site and analyzed for nitrate and orthophosphate concentrations (Appendix F). The margin of error for the laboratory analysis of the nitrate and orthophosphate concentrations is 0.02 and 0.01 mg/L respectively.

3.2.4 Benthic Macroinvertebrates

Benthic macroinvertebrate samples were taken between the middle of June and the end of August with a D-frame dip net; each reach was sampled within a day of its pair, the short sampling period eliminated any temporal variation in sampling between the paired reaches. Samples were taken at three riffles within each study reach at the left and right side of each riffle for a total of six sampling areas. The substrate was disturbed in area of 0.91m x 0.30 m (3 ft x 1 ft) upstream of the net by rubbing the surface of the large rocks and kicking vigorously for 30-60 seconds. The contents of the net were empty onto a table and the benthic macroinvertebrates were picked in the field after two samples were taken in each riffle. All benthic macroinvertebrates for each site were composited and stored in 95% ethanol. In the lab, the samples were soaked with water and then sieved through a 500 µm mesh. The samples were then subsampled using the methods described in Caton (1991) to produce 200 ± 10% or 20% of the organisms for each reach. The samples for each study reach were identified to the family level following Voshell (2002) and Merritt & Cummins (1996).

The Virginia Stream Condition Index (SCI) was calculated to evaluate the benthic macroinvertebrate assemblages at each reach (USEPA and VDEQ, 2003). The SCI is composed of eight metrics including taxa richness; ephemeroptera, plecoptera, and trichoptera (EPT) index; percent ephemeroptera; percent plecoptera and trichoptera minus hydropsychidae (% P+T-Hydropsych.); percent scrapers; percent chironomidae;

percent dominant taxon; and the Modified Family Biotic Index (MFBI). The SCI assigns a score ranging from 1 (severe stress) to 100 (excellent) to each reach based on the benthic assemblages identified. Tetra Tech developed the Aquatic Life Use (ALU) tiers ranging from severe stress (<42) to excellent (>73) based on data collected from 350 streams in Virginia (Table 3.2). The scores determined for each reach were compared to the ALU tiers. In addition, other metrics including percent crawlers and clingers, percent sprawlers and burrowers, percent elmidae and psephenidae, and percent hydropsychidae were used to compare the benthic macroinvertebrates collected at the paired reaches (Appendix F).

3.3 Data Analysis

Percent difference in paired reach-scale parameters was calculated for each paired data set including geomorphic characteristics, substrate size, vegetation and soil parameters, water chemistry, benthic macroinvertebrates metrics, and rapid geomorphic and habitat assessments (equation 3.5). A one-sample Wilcoxon Signed Rank test was run in SAS-JMP® to determine if the percent difference between paired reaches differed significantly from zero (SAS-JMP, 2006).

$$\%diff = \frac{Exclusion - grazed}{grazed} \tag{3.5}$$

One to one (1:1) plots were used to visually assess the relationship of the grazed versus livestock exclusion reaches over a range of time. The grazed reach parameter was plotted on the x-axis and the livestock exclusion parameter was plotted on the y-axis.

Also, a diagonal equal-value line was plotted to determine if the data were greater in the livestock exclusion than the grazed reaches. If the data lay above the equal value line, the data were greater in the livestock exclusion reach; however, if the data lay below the equal value line the data tended to be larger in the grazed reaches. If the data followed an upward trend parallel to the equal value line, then the data set followed an upward trending time distribution.

Figure 3.1
Location of study sites in Virginia, USA.

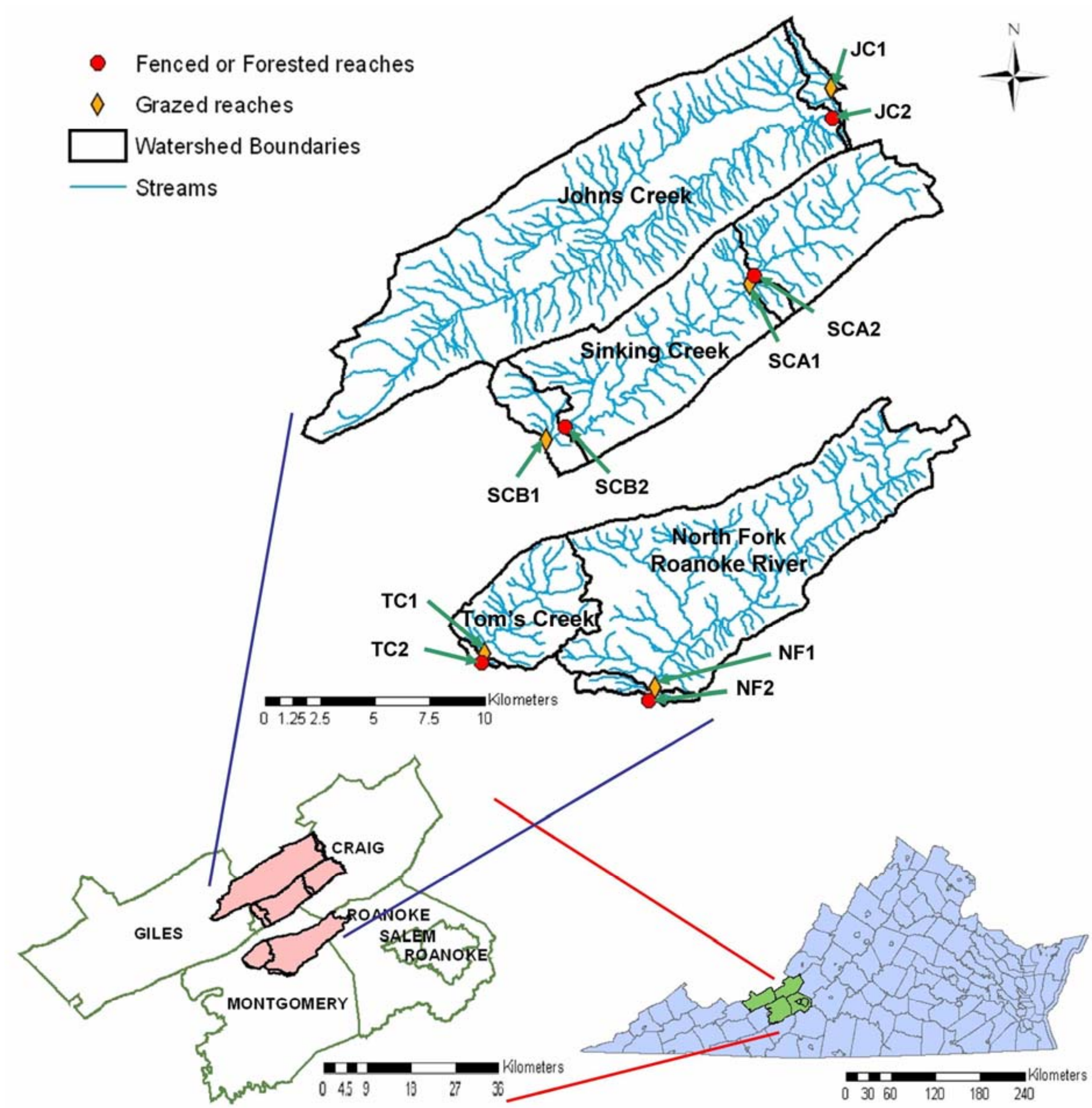


Table 3.1. Watershed land use.

Site*	Grazed vs. Exclusion	Forest (%)	Urban areas/ roads (%)	Natural Grass (%)	Open water/ Wetlands (%)	Residential/ Developed (%)	Pasture/ Hay (%)	Croplands (%)	Other (%)
TC	Grazed	43.1	18.5	1.9	0.5	18.3	6.4	10.2	1.0
	Exclusion	43.8	18.3	2.1	0.5	17.9	6.3	10.1	1.0
	% diff.	1.6	-1.3	10.8	-2.6	-2.0	-2.1	-0.9	-2.6
SCA	Grazed	56.2	5.7	0.6	0.4	0.9	9.4	26.9	0.0
	Exclusion	56.7	5.8	0.6	0.4	0.8	9.0	26.7	0.0
	% diff.	0.9	2.0	5.8	5.8	-9.9	-4.1	-0.7	0.0
SCB	Grazed	58.3	8.1	1.6	0.2	1.2	8.6	21.8	0.1
	Exclusion	57.2	8.0	1.7	0.2	1.2	8.7	22.8	0.1
	% diff.	-1.8	-1.9	5.5	10.0	0.8	1.1	4.4	-3.6
NF	Grazed	60.9	6.3	1.6	0.2	4.9	7.9	18.0	0.1
	Exclusion	60.5	6.5	1.6	0.2	5.2	7.9	18.0	0.1
	% diff.	-0.7	3.5	-2.4	0.3	6.1	0.1	-0.4	-3.2
JC	Grazed	78.0	6.2	11.1	3.2	1.0	0.0	0.0	0.4
	Forested	78.0	6.4	10.9	3.3	1.1	0.0	0.0	0.4
	% diff.	0.0	1.8	-1.6	1.5	2.7	2.7	0.0	-6.1

* TC – Tom's Creek, SCA – Sinking Creek A, SCB – Sinking Creek B, NF – North Fork of the Roanoke River, JC – Johns Creek

Table 3.2. Virginia Stream Conditions Index (SCI) aquatic life use (ALU) tiers (VDEQ, 2006).

SCI Score	ALU tiers
<42	Severe Stress
42-55	Moderate Stress
55-63	Fair (Gray Zone)
63-73	Good
>73	Excellent

Chapter 4. Results

4.1 Comparison Between Grazed and Livestock Exclusion Reaches

4.1.1 Stream Morphology

In our study the hydraulic depth increased and width to depth ratio decreased following livestock exclusion ($p = 0.031$, $p = 0.031$; Table 4.2; Figure 4.1). There were no significant differences in width, cross-sectional area, or amount of LWD between grazed and livestock exclusion reaches (Table 4.2; Figure 4.1). There were also no significant differences in the sinuosity, the overall water surface slope, or the percent of riffles and pools between the paired reaches (Tables 4.1 and 4.3). The average thalweg depth was significantly lower in the livestock exclusion reaches ($p = 0.031$; Table 4.3), while there were no significant differences in the GDI (Skinner et al., 1998; $p = 0.59$; Table 4.3).

The RCI ranged from 1.3 and 5.8; the livestock exclusion reaches had a significantly higher score than the grazed reaches ($p = 0.031$; Table 4.1; Figure 4.2). All reaches were typed as gravel-bed streams with pool-riffle morphology using Rosgen's method for stream classification (Rosgen, 1998; Table 4.1)

There were no significant differences in the median (d_{50}) bed substrate size, percent fines, or the riffle or overall geometric mean particle size between the grazed and livestock exclusion reaches (Chang, 1988; Table 4.4). Although not significant, the riffle d_{50} was usually smaller in the grazed than the livestock exclusion reaches ($p = 0.063$; Table 4.4).

4.1.2 Streambank Soils and Riparian Vegetation

There were no significant differences in bulk density or percent silt and clay in the streambank soils of the paired reaches (Table 4.5 and Figure 4.3). The amount of groundcover biomass was significantly greater in the livestock exclusion than the grazed reaches ($p = 0.031$; Table 4.6; Figure 4.4). No significant differences were found in shrub-crown volume, basal stem area, or tree density between the grazed and livestock exclusion reaches (Table 4.6).

4.1.3 Instream Habitat and Water Chemistry

There was a significant difference in the RHA score between the grazed and livestock exclusion reaches: the grazed reaches' RHA scores ranged from 74-118, while the livestock exclusion scores ranged from 103-175 ($p = 0.031$; Table 4.7; Figure 4.5). No significant differences were found in the nitrate and orthophosphate concentrations between the grazed and livestock exclusion reaches (Table 4.7; Figure 4.6).

4.1.4 Benthic Macroinvertebrates

No significant differences were found in the benthic macroinvertebrate metrics (Taxa Richness, EPT index, and SCI) between paired reaches (Table 4.8 and Figure 4.7).

Table 4.1. Reach characteristics including drainage area, reach length, sinuosity, water surface slope, Rosgen stream type (Rosgen, 1998), and Reach Condition Index (RCI; VDEQ, 2006).

Site*	Grazed vs. Exclusion	Drainage area (km^2)	Reach length (m)	Sinuosity (m/m)	Water Surface Slope (%)	Rosgen stream type	RCI
TC	Grazed	25.7	237.0	1.6	0.15	C4	2.1
	Exclusion	26.4	159.8	1.4	0.27	E4	3.4
	% diff.	3	-33	-10	88	---	62
SCA	Grazed	40.5	201.3	1.1	0.15	C4	2.9
	Exclusion	38.2	117.0	1.1	0.20	C4	4.2
	% diff.	-6	-42	2	34	---	45
SCB	Grazed	107.2	142.9	1.1	0.32	F4	2.9
	Exclusion	97.4	198.7	1.2	0.24	C4	4.2
	% diff.	-9	39	7	-24	---	45
NF	Grazed	105.8	174.2	1.6	0.34	C4	1.3
	Exclusion	109.3	267.2	1.1	0.33	C4	4.2
	% diff.	3	53	-26	-2	---	223
JC	Grazed	171.0	421.0	2.1	0.24	C4	2.1
	Exclusion	166.5	184.3	1.1	0.18	C4	5.8
	% diff.	-3	-56	-47	-26	---	176
p-value†	---	0.63	0.81	0.31	0.81	---	**0.031**
Tested Hypothesis	---	% diff ≠ 0	% diff ≠ 0	% diff ≠ 0	% diff ≠ 0	---	% diff > 0

* TC – Tom's Creek, SCA – Sinking Creek A, SCB – Sinking Creek B, NF – North Fork of the Roanoke River, JC – Johns Creek

† Probability that the percent difference between paired reaches differ significantly from zero (Wilcoxon Signed Rank test); values less than 0.05 were considered significant.

Table 4.2. Cross-section dimensions and large woody debris count (LWD).

Site*	Grazed vs. Exclusion	Cross-section area (m^2)	Width (m)	Mean depth (m) ‡	Width-depth ratio	LWD
TC	Grazed	6.4	9.0	0.7	13.1	11
	Exclusion	4.2	5.4	0.8	7.0	8
	% diff.	-34	-40	13	-46	-27
SCA	Grazed	3.9	9.1	0.4	21.8	5
	Exclusion	2.7	5.6	0.5	12.2	7
	% diff.	-32	-38	6	-44	40
SCB	Grazed	9.4	15.2	0.6	25.0	7
	Exclusion	8.3	13.6	0.6	23.5	0
	% diff.	-13	-11	2	-6	-100
NF	Grazed	8.1	12.2	0.7	19.7	12
	Exclusion	10.3	13.4	0.8	19.4	4
	% diff.	27	10	15	-1	-67
JC	Grazed	10.1	12.5	0.9	16.9	26
	Forested	18.0	17.1	1.1	16.5	8
	% diff.	77	37	17	-2	-69
p-value†	---	0.5	0.22	**0.031**	**0.031**	0.094
Tested Hypothesis	---	% diff < 0	% diff < 0	% diff > 0	% diff < 0	% diff < 0

* TC – Tom's Creek, SCA – Sinking Creek A, SCB – Sinking Creek B, NF – North Fork of the Roanoke River, JC – Johns Creek

† Probability that the percent difference between paired reaches differ significantly from zero (Wilcoxon Signed Rank test); values less than 0.05 were considered significant.

‡ Hydraulic depth over four cross-sections was calculated by dividing the cross-sectional area by the top width at bankfull stage.

Table 4.3. Percent riffles and pools, average thalweg depth, and reach depth variability index in each study reach.

Site*	Grazed vs. Exclusion	Percent riffles (%)	Percent pools (%)	Average thalweg depth (m)	Geo-hydraulic Diversity index (GDI)
TC	Grazed	22	79	0.21	1.4
	Exclusion	66	34	0.20	1.3
	% diff.	209	-57	-4	-5
SCA	Grazed	29	71	0.29	1.1
	Exclusion	55	45	0.24	0.9
	% diff.	90	-37	-16	-17
SCB	Grazed	45	55	0.34	1.2
	Exclusion	53	47	0.26	1.8
	% diff.	18	-15	-23	43
NF	Grazed	47	53	0.43	1.3
	Exclusion	45	55	0.34	1.3
	% diff.	-3	3	-20	4
JC	Grazed	24	76	0.61	1.3
	Forested	55	45	0.41	1.1
	% diff.	128	-41	-32	-17
p-value†		0.063	0.063	**0.031**	0.59
Tested Hypothesis	---	% diff > 0	% diff < 0	% diff < 0	% diff > 0

* TC – Tom's Creek, SCA – Sinking Creek A, SCB – Sinking Creek B, NF – North Fork of the Roanoke River, JC – Johns Creek

† Probability that the percent difference between paired reaches differ significantly from zero (Wilcoxon Signed Rank test); values less than 0.05 were considered significant.

Table 4.4. Substrate characteristics including d_{50} (median bed substrate particle size), percent fines, and geometric standard deviation (σ_g).

Site*	Grazed vs. Exclusion	d_{50}[‡]	Riffle d_{50}[§]	% fines	Geometric standard deviation (σ_g)[¶]	Riffle geometric standard deviation (σ_g)[#]
TC	Grazed	7	11	24	3.3	1.9
	Exclusion	18	37	9	4.2	2.6
	% diff.	147	236	-65	29	36
SCA	Grazed	42	38	1	2.5	2.8
	Exclusion	17	24	1	2.9	3.1
	% diff.	-60	-37	-23	18	14
SCB	Grazed	19	17	2	3.2	3.2
	Exclusion	49	49	1	2.9	3.2
	% diff.	158	188	-40	-9	-3
NF	Grazed	26	27	10	3.3	2.7
	Exclusion	41	39	2	2.6	2.1
	% diff.	58	44	-77	-22	-23
JC	Grazed	45	36	2	2.1	2.3
	Forested	40	53	5	2.5	1.9
	% diff.	-11	47	113	20	-19
p-value[†]		0.22	0.063	0.31	0.31	0.5
Tested Hypothesis	---	% diff > 0	% diff > 0	% diff < 0	% diff > 0	% diff > 0

* TC – Tom's Creek, SCA – Sinking Creek A, SCB – Sinking Creek B, NF – North Fork of the Roanoke River, JC – Johns Creek

[†] Probability that the percent difference between paired reaches differ significantly from zero (Wilcoxon Signed Rank test); values less than 0.05 were considered significant.

[‡] d_{50} - the median substrate size of four-hundred pebbles in two riffles and two pools.

[§] Riffle d_{50} - the median substrate size of the two-hundred pebbles counted in two riffles.

[¶] Geometric standard deviation of four-hundred pebbles in two riffles and two pools.

[#] Riffle geometric standard deviation of two-hundred pebbles counted in two riffles.

Table 4.5. Streambank soil bulk density median, minimum, and maximum, and percent silt and clay from particle size analysis.

Site*	Grazed vs. Exclusion	Median (g/cm^3)	Min (g/cm^3)	Max (g/cm^3)	Silt and Clay (%)
TC	Grazed	1.4	1.0	1.5	48
	Exclusion	1.5	0.9	1.4	36
	% diff.	4	-10	-9	-26
SCA	Grazed	1.3	1.0	1.7	42
	Exclusion	1.4	0.7	1.3	44
	% diff.	2	-32	-25	4
SCB	Grazed	1.2	1.2	1.4	26
	Exclusion	1.1	1.0	1.5	26
	% diff.	-11	-21	5	-2
NF	Grazed	1.6	0.9	1.4	46
	Exclusion	1.0	1.0	1.5	50
	% diff.	-38	9	8	9
JC	Grazed	1.1	1.3	1.6	34
	Forested	1.1	1.2	1.6	25
	% diff.	1	-13	4	-25
p-value†	---	0.41	0.063	0.41	0.31
Tested Hypothesis	---	% diff < 0	% diff < 0	% diff < 0	% diff < 0

* TC – Tom's Creek, SCA – Sinking Creek A, SCB – Sinking Creek B, NF – North Fork of the Roanoke River, JC – Johns Creek

† Probability that the percent difference between paired reaches differ significantly from zero (Wilcoxon Signed Rank test); values less than 0.05 were considered significant.

Table 4.6. Buffer type and aboveground vegetation measurements.

Site*	Grazed vs. Exclusion	Buffer type ‡	Average buffer width (m)	Ground-cover dry biomass (kg/ha)	Shrub crown volume (m^3/ha)	Basal stem area (m^2/ha)	Tree density (stems/ha)
TC	Grazed	H	---	1240	0	24	175
	Exclusion	H	34.9	2688	0	20	50
	% diff.	---	---	117	0	-17	-71
SCA	Grazed	H	---	1350	1713	0	0
	Exclusion	F	24.0	1765	780	5	50
	% diff.	---	---	31	-54	0	0
SCB	Grazed	H	---	2345	962	2	275
	Exclusion	H/F	11.9	3628	497	3	75
	% diff.	---	---	55	-48	50	-73
NF	Grazed	H	---	760	133	21	975
	Exclusion	F	7.2	2675	1063	17	350
	% diff.	---	---	252	699	-19	-64
JC	Grazed	H/F	---	453	174	13	150
	Exclusion	F	52.3	1318	280	29	725
	% diff.	---	---	191	61	123	383
p-value†	---	---	---	**0.032**	0.31	0.31	0.59
Tested Hypothesis	---	---	---	% diff > 0	% diff > 0	% diff > 0	% diff > 0

* TC - Tom's Creek, SCA -Sinking Creek A, SCB - Sinking Creek B, NF - North Fork of the Roanoke River, JC - Johns Creek

† Probability that the percent difference between paired reaches differ significantly from zero (Wilcoxon Signed Rank test); values less than 0.05 were considered significant.

‡ H- Herbaceous, F- Forested

Table 4.7. Nitrate and orthophosphate concentrations, Rapid Habitat Assessment score, and percent embeddedness for each reach.

Site*	Grazed vs. Exclusion	Nitrate (mg/L)	Orthophosphate (mg/L)	Rapid habitat score	Percent embeddedness‡
TC	Grazed	0.081	0.122	74	25
	Exclusion	0.062	0.020	103	35
	% diff.	-23	-84	39	40
SCA	Grazed	0.060	0.017	87	55
	Exclusion	0.055	0.023	129	75
	% diff.	-8	35	48	36
SCB	Grazed	0.050	0.038	118	75
	Exclusion	0.041	0.024	141	70
	% diff.	-18	-37	20	-7
NF	Grazed	0.058	0.099	95	30
	Exclusion	0.062	0.039	131	75
	% diff.	7	-61	38	150
JC	Grazed	0.044	0.023	98	55
	Forested	0.040	0.014	175	100
	% diff.	-9	-39	79	82
p-value†	---	0.063	0.063	**0.031**	0.063
Tested Hypothesis	---	% diff < 0	% diff < 0	% diff > 0	% diff > 0

* TC – Tom's Creek, SCA – Sinking Creek A, SCB – Sinking Creek B, NF – North Fork of the Roanoke River, JC – Johns Creek

† Probability that the percent difference between paired reaches differ significantly from zero (Wilcoxon Signed Rank test); values less than 0.05 were considered significant.

‡ From Rapid Habitat Assessment for high gradient streams, a score from 0 (low)-20 (high) was chosen for embeddedness and adjusted to a percent value.

Table 4.8. Taxa richness, EPT index, and Stream Condition Index (SCI).

Site*	Grazed vs. Exclusion	Taxa Richness	EPT index	SCI
TC	Grazed	14	7	60.6
	Exclusion	17	7	60.1
	% diff	21	0	-1
SCA	Grazed	20	11	73.2
	Exclusion	23	13	69.5
	% diff	15	18	-5
SCB	Grazed	21	11	74.5
	Exclusion	17	9	72.8
	% diff	-19	-18	-2
NF	Grazed	14	7	60.0
	Exclusion	17	7	59.3
	% diff	21	0	-1
JC	Grazed	18	9	70.5
	Forested	18	8	73.7
	% diff	0	-11	5
p-value†	---	0.19	0.50	0.78
Tested Hypothesis	---	% diff > 0	% diff > 0	% diff > 0

* TC – Tom's Creek, SCA – Sinking Creek A, SCB – Sinking Creek B, NF – North Fork of the Roanoke River, JC – Johns Creek

† Probability that the percent difference between paired reaches differ significantly from zero (Wilcoxon Signed Rank test); values less than 0.05 were considered significant.

Figure 4.1
Bankfull width (W_{bkf}), depth (D_{bkf}), and width to depth ratio (W/D) based on cross-section surveys of two riffles and two pools in each reach. Error bars indicate +1 standard deviation with n=4.

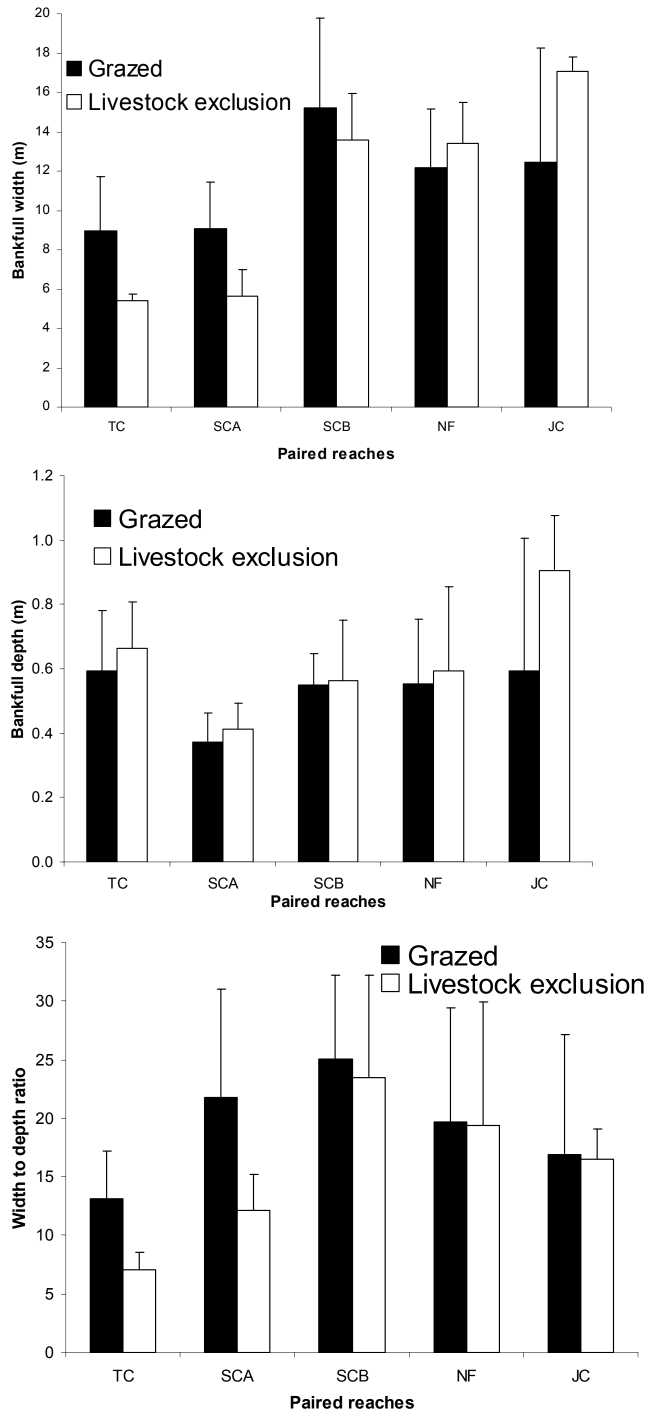


Figure 4.2
Reach Condition Index (RCI) score in each study reach. Scores range from 0 (poor) to 7 (excellent) and indicate geomorphic reach condition.

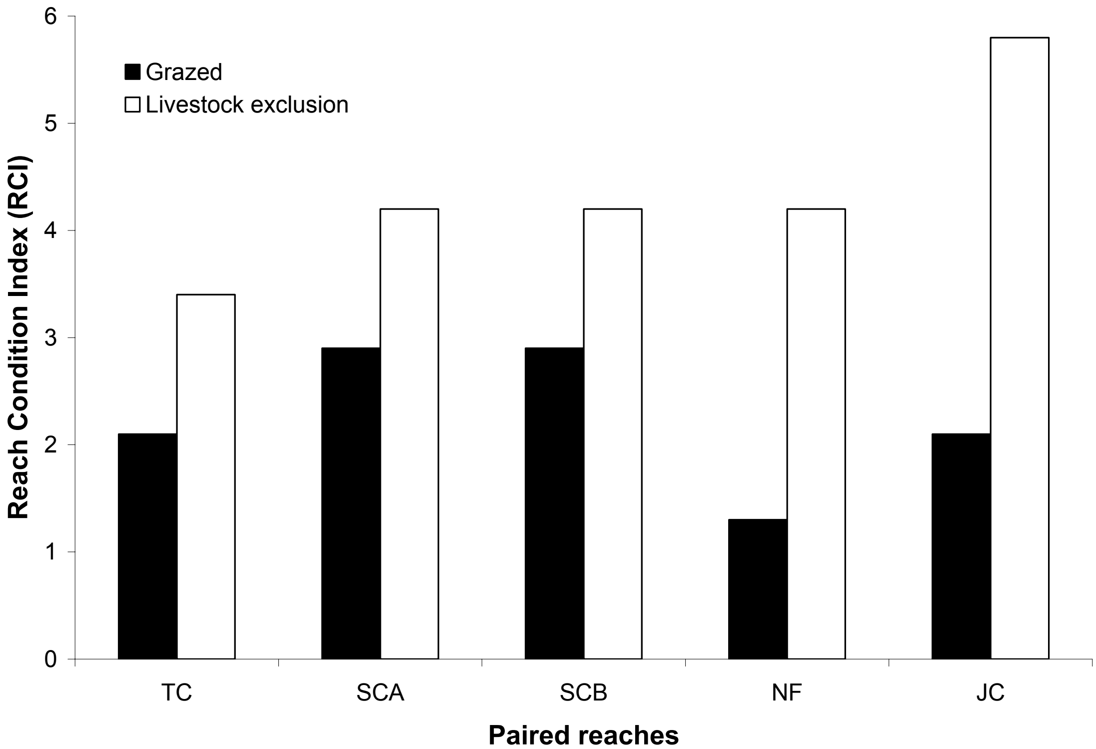


Figure 4.3
Median bulk density for each paired reach. Error bars indicate +1 standard deviation with n ≥ 4.

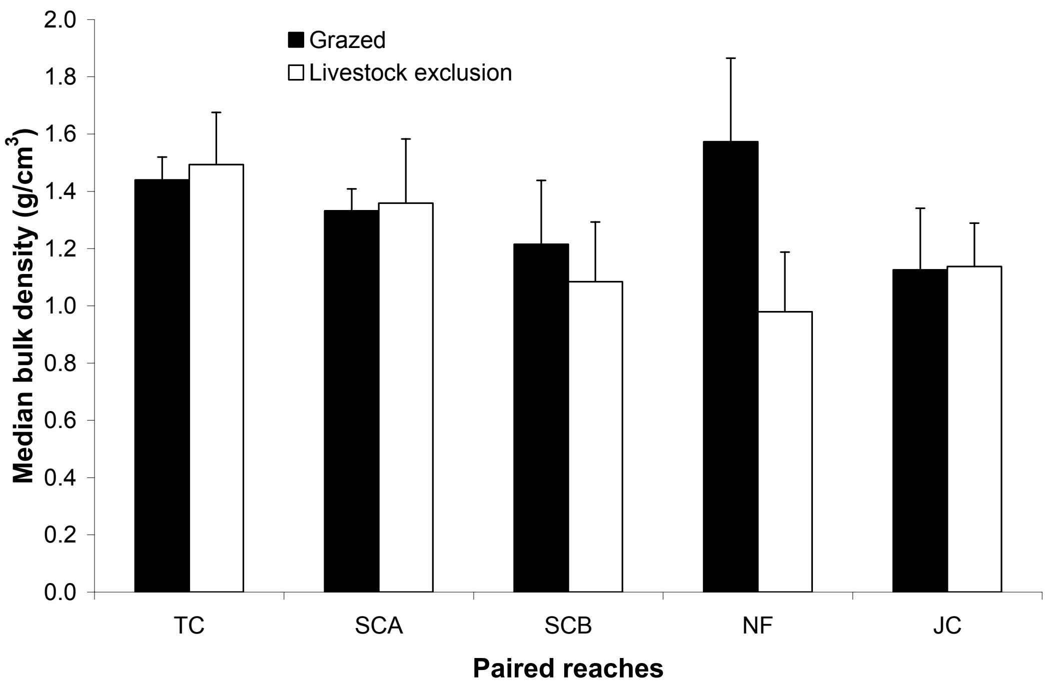


Figure 4.4
Groundcover vegetation biomass for each paired reach.

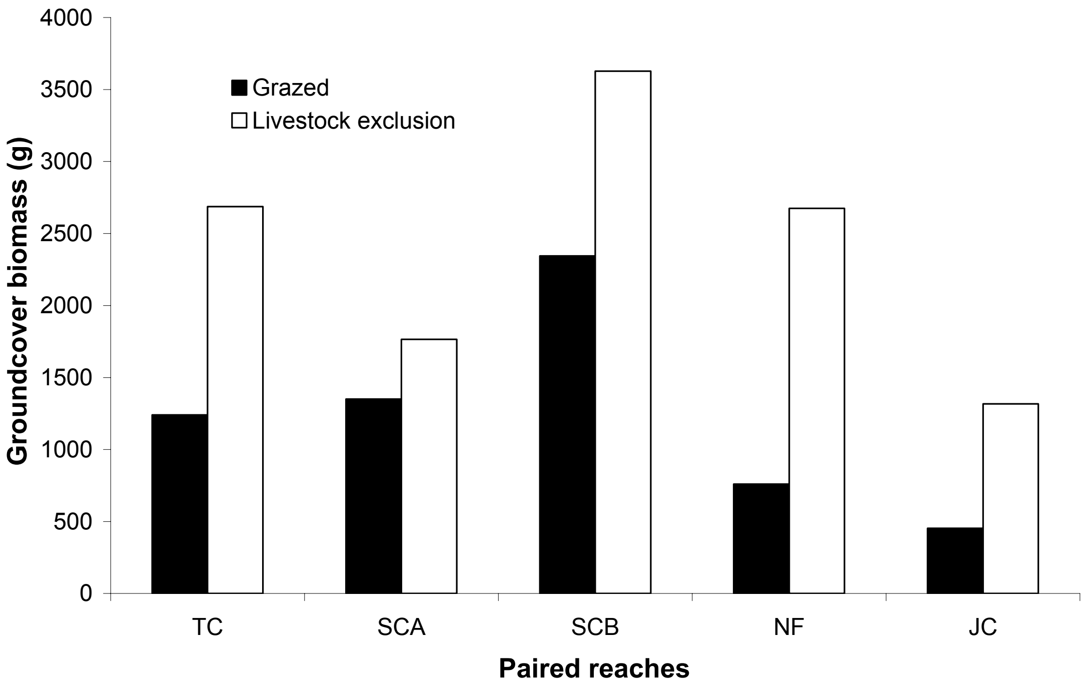


Figure 4.5
Rapid Habitat Assessment (RHA) score for each paired reach. Scores range from 0 (poor) to 200 (excellent) indicating reach-scale habitat quality.

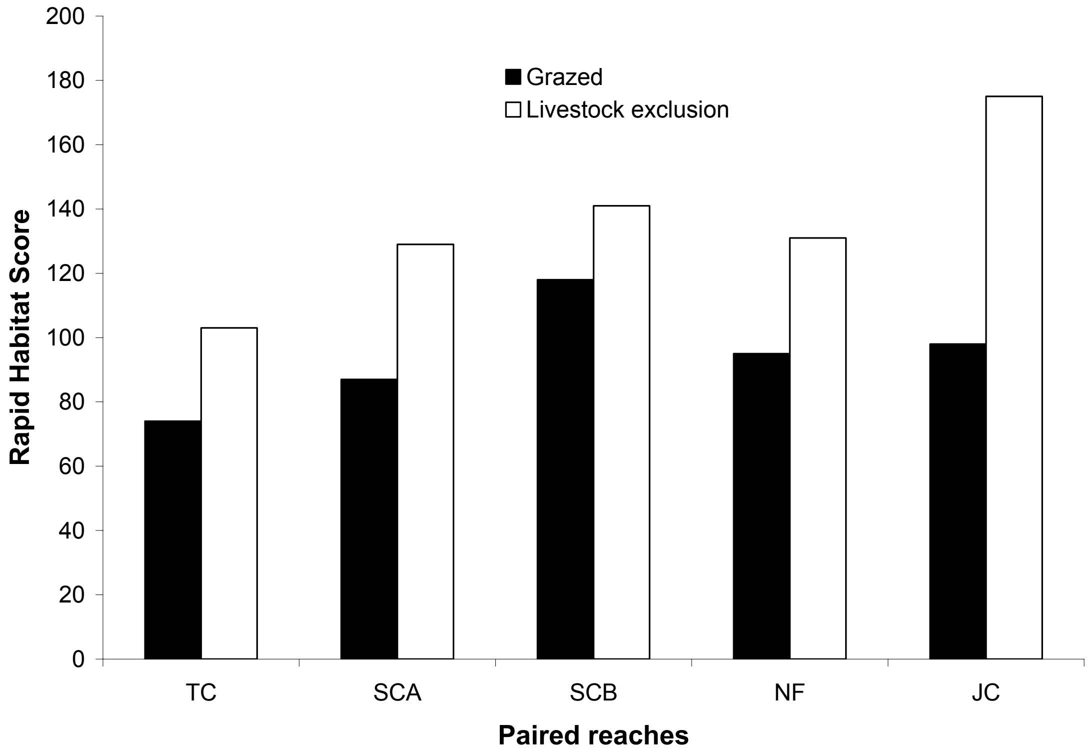


Figure 4.6

Nitrate and orthophosphate concentrations for each paired reach. Error bars indicate laboratory test precision: 0.02 and 0.01 mg/L for nitrate and orthophosphate concentrations, respectively.

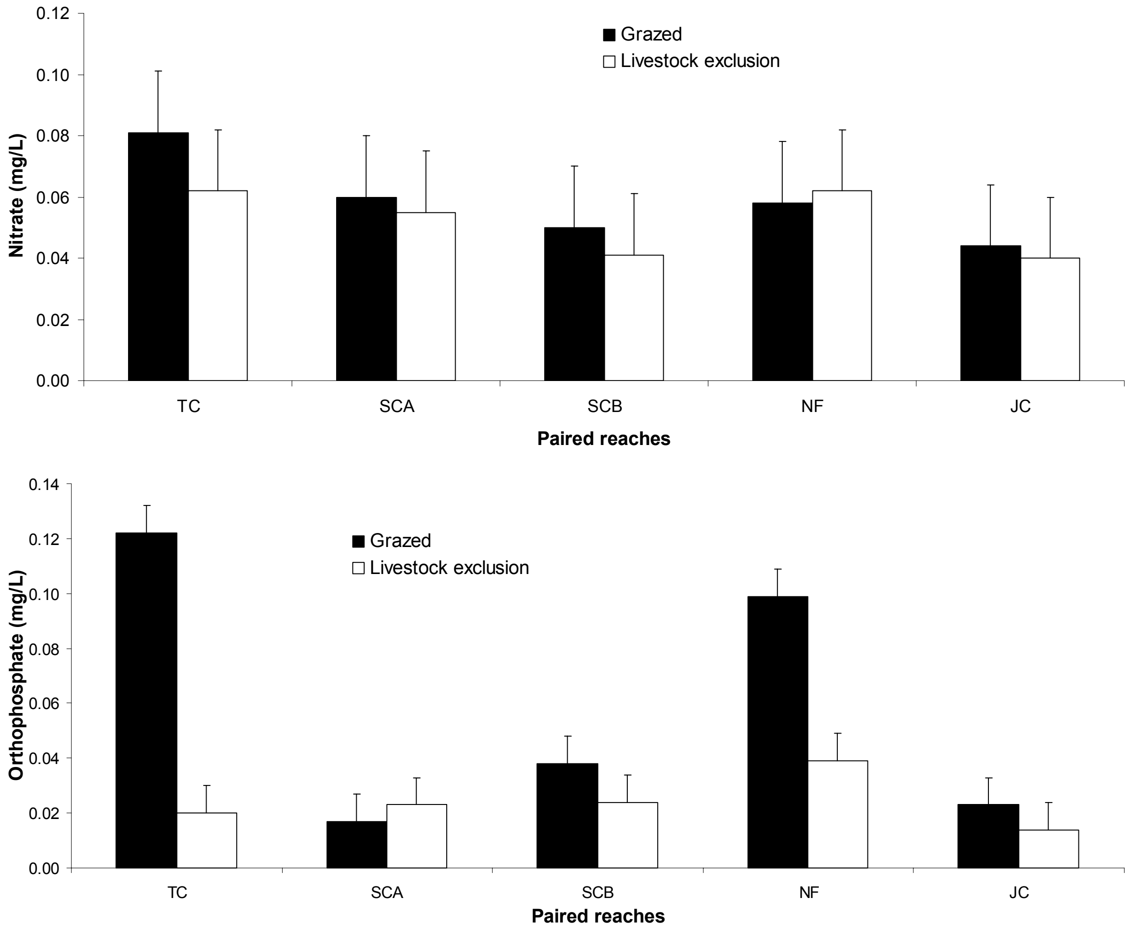
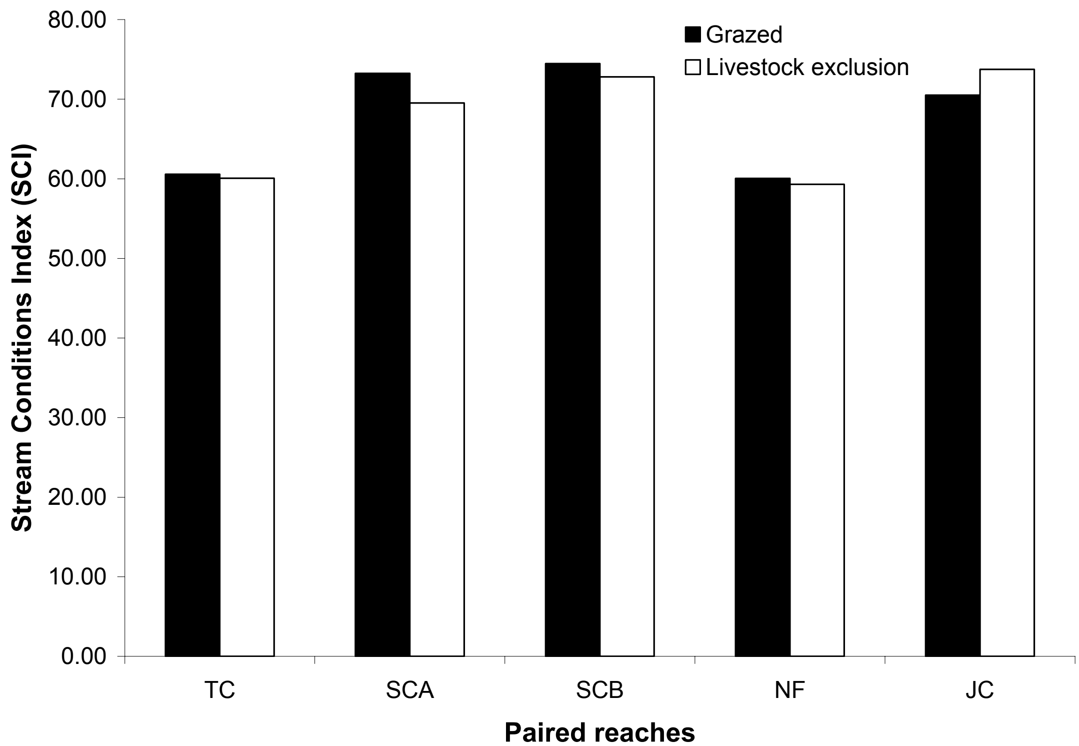


Figure 4.7
Stream Conditions Index (SCI) in each paired reach. SCI values range from 0 (severe stress) to 100 (excellent) indicating aquatic habitat condition.



Chapter 5. Discussion

5.1 Comparison Between Grazed and Livestock Exclusion Reaches

5.1.1 Stream Morphology

In our study reaches the bankfull depth and width to depth ratio were significantly different in the grazed and livestock exclusion reaches; however, there were no significant differences in bankfull width. The bankfull depth, the cross-section area divided by the width at bankfull stage, was significantly higher in the grazed reach. Consequently, the width to depth ratio was lower in the livestock exclusion reaches.

Parkyn et al. (2003) similarly determined that stream width did not change with increased livestock access to the stream. Wohl and Carline (1996) found that the width and depth of the stream did not alter with increased livestock access. In contrast, Clary (1999) and Scarsbrook and Halliday (1999) determined that wider reaches resulted from increased livestock access to the stream. Numerous studies conducted on the geomorphic response to livestock exclusion have determined that the width to depth ratio decreases after livestock exclusion, the non-dimensional width to depth ratio allows for focus on local grazing impacts (McDowell and Magilligan, 1997; Clary, 1999; Nagle and Clifton, 2003).

The there were no significant differences in the amount of LWD between grazed and livestock exclusion reaches. Wohl and Carline (1996) and Harding et al. (2006) also found no significant differences in LWD between forested and agricultural reaches. Wohl and Carline (1996) attributed the lack of LWD in the grazed reaches to the young age of trees in the riparian zone. Harding et al. (2006) hypothesized that storm events had moved LWD from the forested reach downstream to the agricultural reaches. The age of trees in most of our study riparian zones was less than 14 years, which most likely contributed to our finding that the amount of LWD was not significantly greater in the livestock exclusion reaches. Since all of our reaches have upstream stream segments with a wide range of land uses, storm events could have moved LWD debris downstream causing the variability in the amount of LWD between study reaches.

The Reach Condition Index (RCI) was significantly higher in the livestock exclusion reaches than the grazed reaches. Similar qualitative geomorphic assessments were conducted by Parkyn et al. (2003), Scrimgeour and Kendall (2003), and Harding et al. (2006), and all of them found an increase in a bank stability index following livestock exclusion. Galeone et al. (2006) also conduct qualitative visual geomorphic assessments and determined that the available substrate cover, pool to riffle ratio, and bank stability showed improvements after streambank fencing. Qualitative geomorphic assessments are helpful in determining if positive visual improvements are occurring in the stream reach, especially if livestock have recently been removed from the stream.

There were no significant differences in the percent fines, embeddedness, or d_{50} between grazed and livestock exclusion reaches. However, the riffle d_{50} substrate size was usually larger in the livestock exclusion reaches ($p=0.063$). Clary (1999) showed no significant differences existed in percent fines or embeddedness between grazed and ungrazed reaches. In contrast, Braccia and Voshell (2006) and Nerbonne and Vondracek (2001) determined the substrate percent fines and the degree of embeddedness increase with increased cattle access to the stream. Wohl and Carline (1996) determined the amount of fine substrates increased with increased cattle access to the stream. The lack of difference in d_{50}, percent fines, and embeddedness in our study was probably due to multiple upstream sediment sources rather than localized livestock impacts.

5.1.2 Streambank Soils and Riparian Vegetation

Many studies have found that soil bulk density, a measure of soil compaction, increases with grazing intensity (Alderfer and Robinson, 1949; Orr, 1960; Rauzi and Hanson, 1966; Bryant et al., 1972; and Wood and Blackburn, 1984); however, there were no significant differences in bulk density between the paired reaches in our study. The lack of difference in soil bulk density might be due to the low grazing intensity in our study reaches (Appendix A).

The amount of groundcover vegetation was significantly greater in the livestock exclusion reaches than the grazed reaches. Schulz and Leininger (1990), Williamson et al. (1992), and Flenniken et al. (2001) found similar responses of groundcover vegetation to livestock removal. In fact, Schulz and Leininger (1990) found the grazed riparian areas had approximately five times as much bare ground than areas that had livestock exclusion for twenty-nine years.

5.1.3 Instream Habitat and Water Chemistry

The VDEQ screened several sets of Virginia habitat quality data to determine that the Ridge and Valley Ecoregion usually has a range of RHA scores between 120 and 140. A habitat score below 120 indicates that the reach has poor habitat condition, while a score above 140 indicates that the reach has good habitat conditions (VDEQ, 2006). Scores from our study reaches scores' were generally on the lower end of the VDEQ

Ridge and Valley Ecoregion scores in the grazed reaches (74-118) and on the higher end in the livestock exclusion reaches (103-175). Stone et al. (2005) and Braccia and Voshell (2006) used the RHA to distinguish the habitat conditions between study reaches in Virginia with varying degrees of cattle density. Braccia and Voshell (2006) assigned a RHA score of 156 to there reference reach, which is lower than the score for the forested reach in our study (175). Since the RHA is a qualitative assessment, the scores assigned to stream reaches are subjective. However, the RHA is helpful in comparing the reach-scale physical condition of the stream; Stone et al. (2005) concluded that in general physical habitat quality was the limiting factor improving aquatic habitat.

Although not significant, the nitrate and orthophosphate concentrations tended to were generally higher in the grazed reaches. Galeone et al. (2006) found similar results to our study; they determined that nitrate and phosphorus concentrations increase with increased agricultural land use. The overall summer orthophosphate concentrations for streams in the Ridge and Valley Ecoregion range from 0.00050 to 0.12 mg/L (USEPA, 2000). The orthophosphate concentrations for the study reaches (0.017-0.12 mg/L) were within this range and were higher than the mean concentration (0.016 mg/L) for streams in the Ridge and Valley Ecoregion (USEPA, 2000). The nitrate concentrations for our study reaches (0.040-0.081 mg/L) fell in the lower range of values for the nitrite/nitrate concentrations in Ridge and Valley Ecoregion streams (0.003-8.950 mg/L; USEPA, 2000).

5.1.4 Benthic Macroinvertebrates

There were no significant differences in the benthic macroinvertebrate assemblages between our paired reaches. Nerbonne and Vondracek (2001), Wohl and Carline (1995), and Parkyn et al. (2003) similarly determined no significant differences in the benthic macroinvertebrate metrics between grazed and livestock exclusion reaches. Stone et al. (2005) found no significant differences in the EPT scores between reaches with range of agricultural land use. However, Scarsbrook and Halliday (1999) and Galeone et al. (2006) found higher EPT scores with decreased livestock access to the stream. In addition, Galeone et al. (2006) found that taxa richness increased with increased agricultural land use.

Nerbonne and Vondracek (2001) attributed the lack of benthic macroinvertebrate response in their study in southeastern Minnesota to the influence of watershed characteristics rather than localized livestock grazing impacts. Watershed topography and land use dictate the conditions to which benthic macroinvertebrates respond, including temperature, discharge, flood frequency and magnitude, and delivery of sediment and nutrients (Troelstrup and Perry, 1989; Nerbonne and Vondracek, 2001). Parkyn et al. (2003) attributed the lack of benthic macroinvertebrate response to livestock exclusion to variations in shading between reaches. The age, length, and vegetation type in buffers affects the amount of shading available to the stream reach and alters stream temperature to which certain benthic macroinvertebrate assemblages are senstitive (Quinn et al., 1992; Rutherford et al., 1999; Parkyn at el, 2003).

In our study reaches, four of the livestock exclusion reaches were most likely not mature enough (2-14 years) to significantly increase shading, and therefore influence stream temperature. Also, there was a large range of watershed and riparian buffer characteristics that could have contributed to the variability in the physical habitat conditions. Variability in streambank vegetation and watershed size is common in livestock exclusion studies; therefore, aquatic invertebrate populations are frequently attributed to differences in watershed land use and geologic characteristics (Belsky et al., 1999).

5.2 Temporal Distribution

To evaluate the time distribution for improved channel morphology and aquatic habitat once exclusion projects are implemented, we plotted several parameters on 1:1 line plots (Figures 5.1-5.8). If the points are above the 1:1 line the parameter is larger in the livestock exclusion reaches; however, if the points are below the 1:1 line the parameter is larger in the grazed reaches. Each data point was labeled with the amount of time the livestock exclusion practices have been in place. To observe a time distribution for improved geomorphic and aquatic stream conditions, the data points would follow a trend with 2 years as being the closest to the 1:1 line, and 50 years as the furthest away. Depending on the measured parameter, the data would lie towards either the grazed or livestock exclusion axis. Only the RCI plot displays a relationship to time (Figure 5.3); however, the RCI is a qualitative measure estimated using visual observation. Therefore, we might expect RCI to have the most rapid response to livestock exclusion practices.

The bankfull width, width to depth ratio, RCI, d_{50}, soil bulk density, and SCI were not significantly correlated to duration of livestock exclusion (Figures 5.1, 5.2, 5.4, 5.5, 5.6, and 5.7). McDowell and Magilligan (1997) similarly determined that the degree of channel aquatic and geomorphic recovery was not directly related to the amount of time livestock were excluded. They attributed this finding to possible differences in initial conditions, local controls, and hydrologic conditions during the period of livestock exclusion. Similarly, our paired study sites are in different watersheds with different watershed characteristics, upstream disturbances, and initial conditions.

Closer inspection of the bankfull widths (Figures 4.1 and 5.1) highlights the complexity involved in evaluating changes over time since exclusion. Numerous studies have reported that livestock access results in wider stream channels (Scarsbrook and Halliday, 1999; Trimble and Mendel, 1995; Clary, 1999). Trimble and Mendel (1995) presented a schematic representing the recovery of a stream channel after livestock exclusion that suggested that, once the cattle were removed, the grasses along the streambank would be allowed to grow, resulting in increased sediment trapping and channel narrowing. Additionally, there are numerous studies that suggest that small streams with riparian forests are wider than those with non-forested, or grassy, vegetation (Davies-Colley, 1997; Trimble, 1997; Hession et al., 2003). While our data set is small (n=5), the sites where cattle were recently excluded (2, 2.5, and 4 years) fall below the 1:1 line in Figure 5.1 (livestock exclusion narrower than grazed) and the sites with older exclusion (14 and >50 years) plot above the line. In addition, the oldest site plots furthest above the 1:1 line, suggesting that it has widened the most. Such a progression from narrow, grassy channel to wider, forested channel has been predicted by several authors (Davies-Colley, 1997; Trimble, 2004). Trimble (2004) suggested the following sequence: a) starting with a narrow stream with grassy riparian vegetation; b) widening occurs as trees begin to shade out grass on banks and LWD begins to “clog” the channel (10-20 years); and c) years later, a wider stream with mature forest and dense shade results. While our data set is small, we do have some evidence that the streams may narrow after livestock exclusion, and eventually widen as the riparian forests mature. Most importantly, the situation with bankfull width helps to highlight the complexities involved in studying the impacts of livestock exclusion on stream channel recovery.

To more thoroughly evaluate the changes over time, we would need a much larger data set to fill in the gaps in our temporal distribution and to provide multiple sites with similar exclusion durations. However, our findings suggest the first parameters in which we would expect to see changes are riparian vegetation (groundcover) the visual qualitative physical characteristics (RCI). The riparian and aquatic wildlife might take longer to recover, or even require improved upstream and watershed conditions.

Figure 5.1
Bankfull width in two riffle and two pool cross-sections in the grazed and livestock exclusion reaches. The number in the parenthesis indicates the duration of time that livestock exclusion practices have been in place on the livestock exclusion reaches.

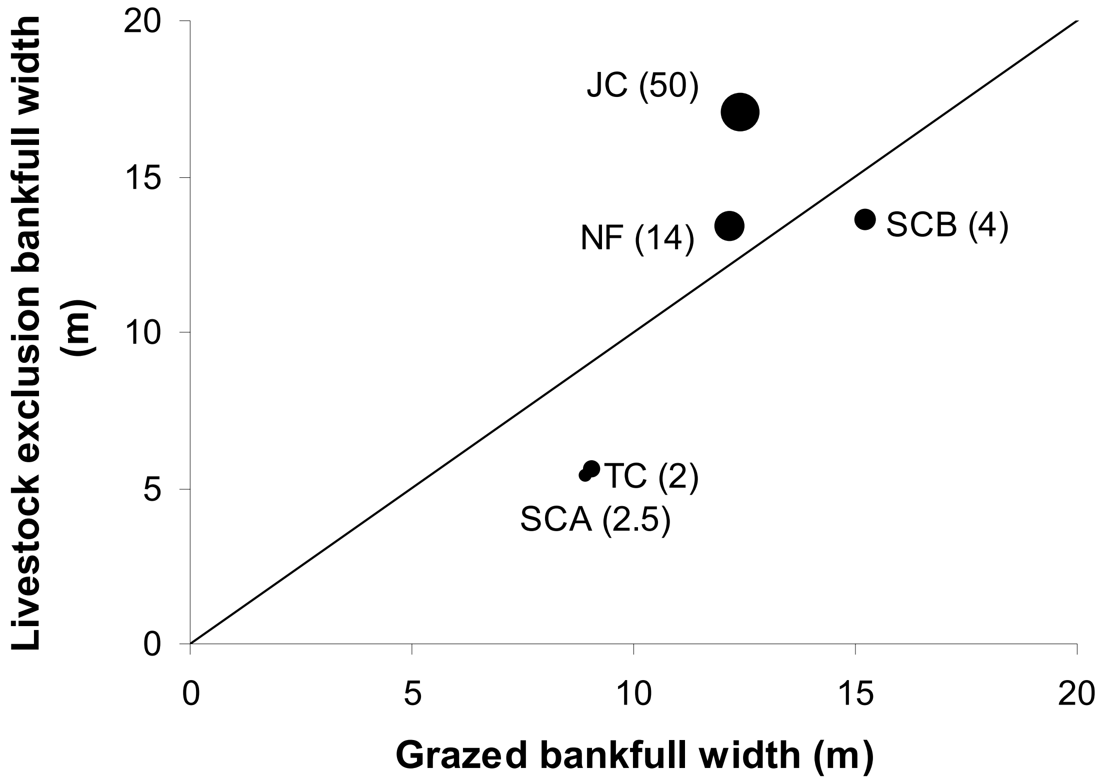


Figure 5.2
Average bankfull width to depth ratio in two riffle and two pool cross-sections in the grazed and livestock exclusion reaches. The number in the parenthesis indicates the duration of time that livestock exclusion practices have been in place on the livestock exclusion reaches.

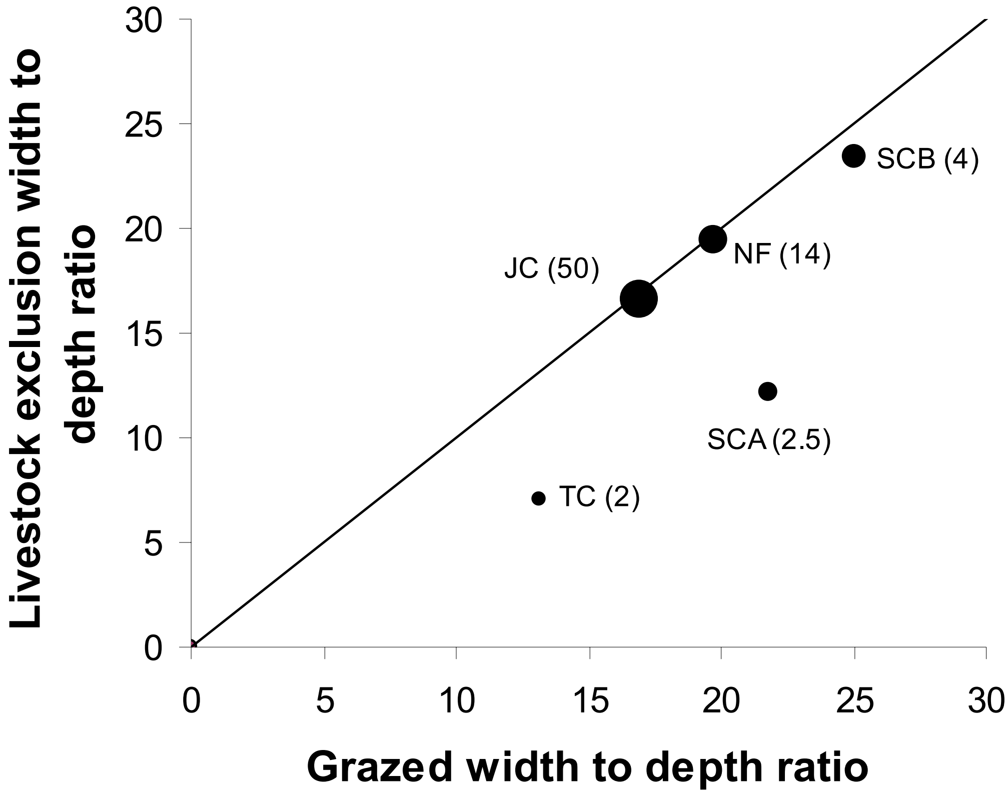


Figure 5.3
Reach Condition Index (RCI) for the grazed and livestock exclusion reaches. The number in the parenthesis indicates the duration of time that livestock exclusion practices have been in place on the livestock exclusion reaches.

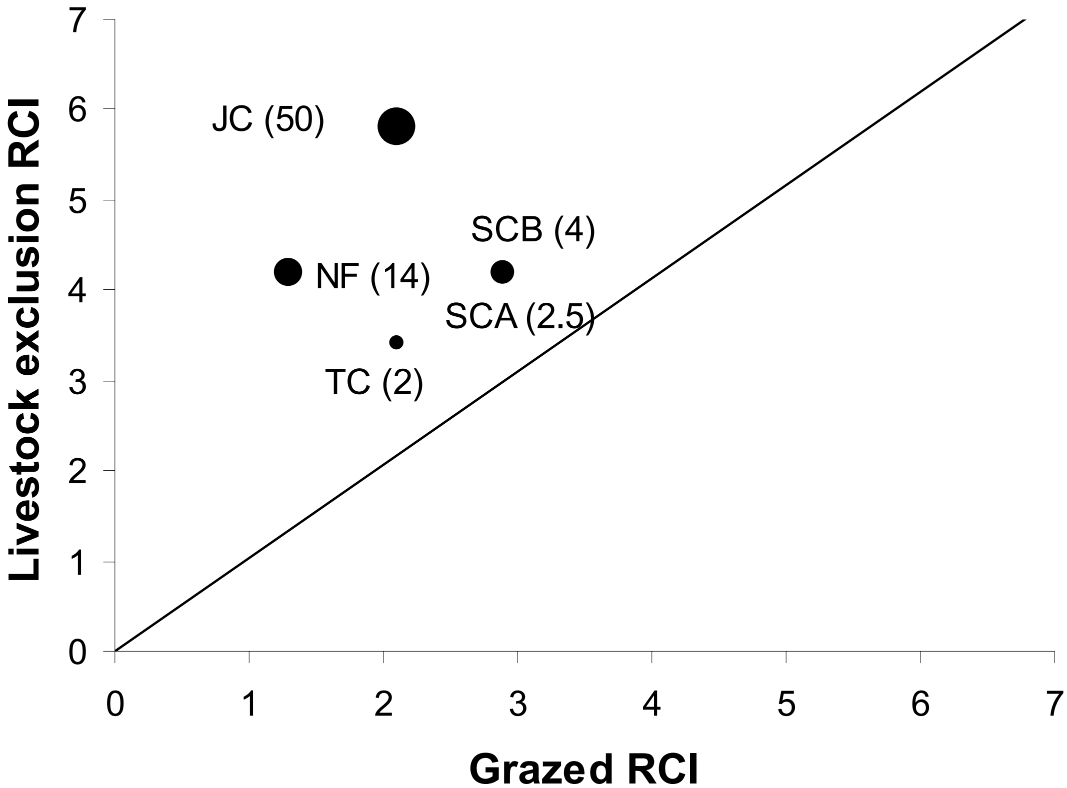


Figure 5.4
Riffle median particle size (d_{50}) in the grazed and livestock exclusion reaches. The riffle median particle size (d_{50}) of 200 pebbles measured in two riffles in each study reach. The number in the parenthesis indicates the duration of time that livestock exclusion practices have been in place on the livestock exclusion reaches.

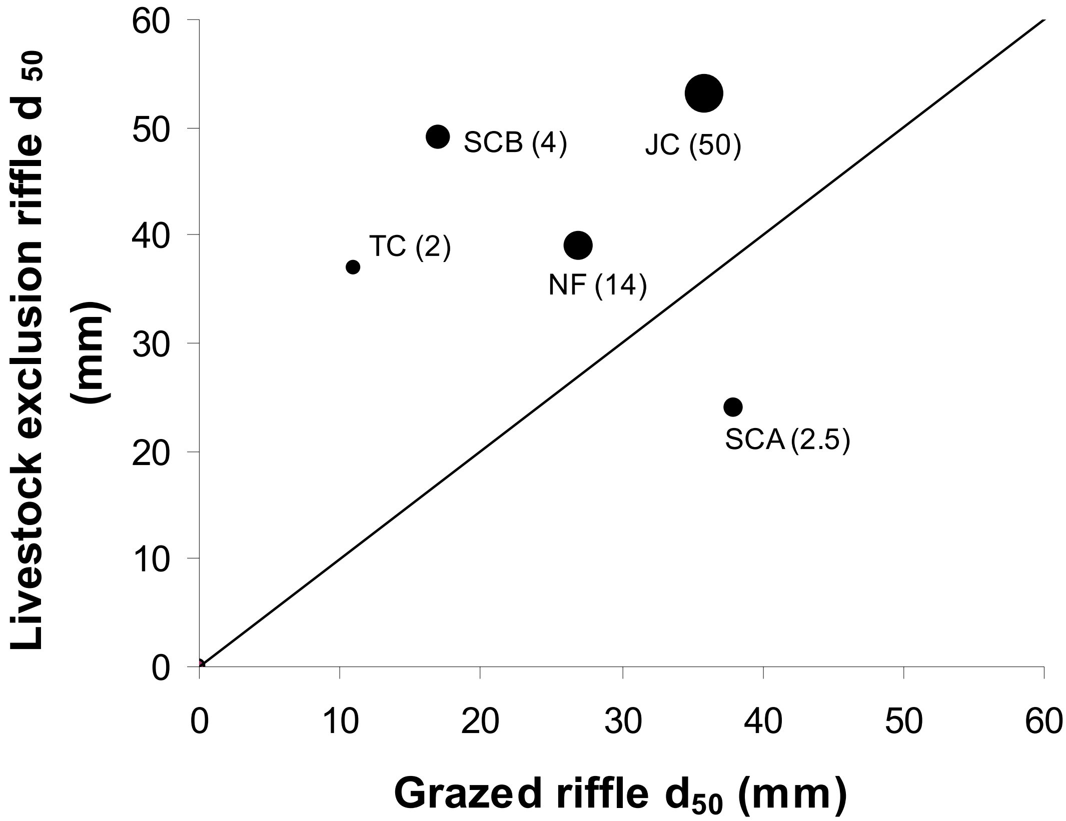


Figure 5.5
Bulk density (g/cm^3) in the grazed and livestock exclusion reaches. The number in the parenthesis indicates the duration of time that livestock exclusion practices have been in place on the livestock exclusion reaches.

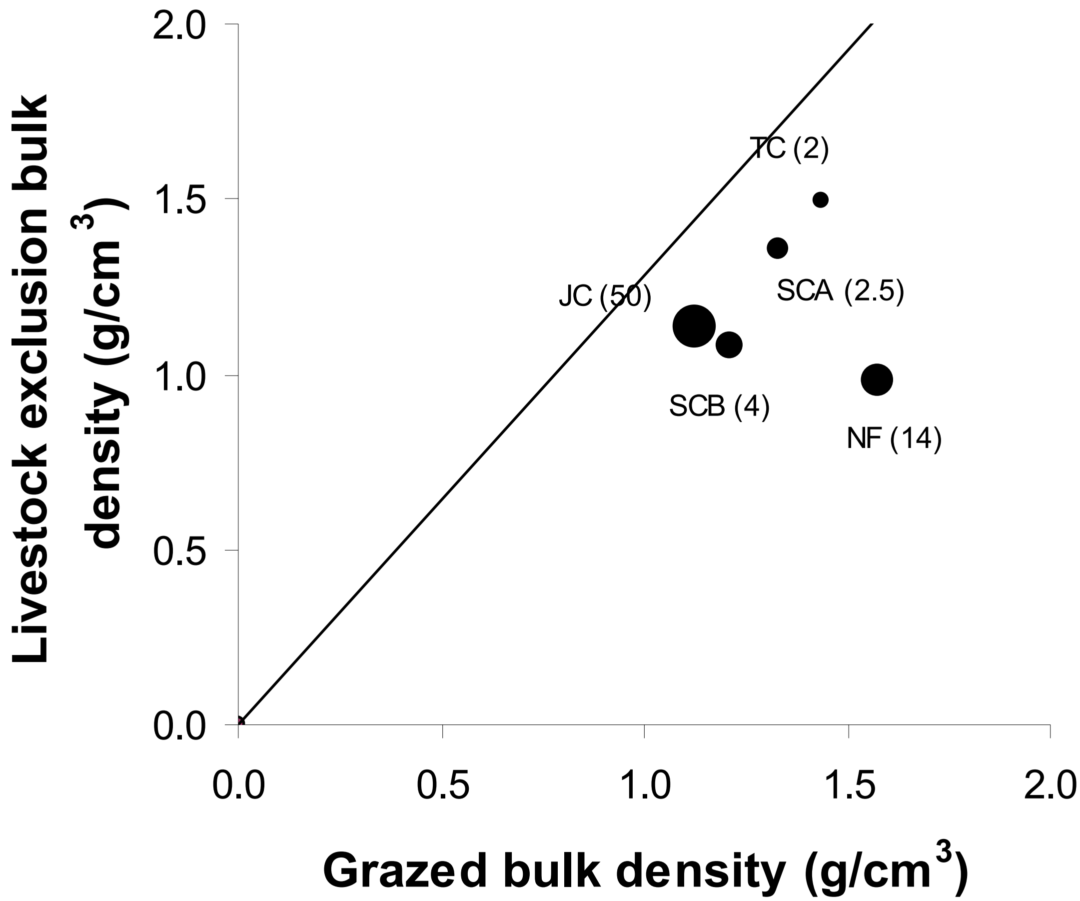


Figure 5.6
Rapid Habitat Assessment (RHA) for the grazed and livestock exclusion reaches. The number in the parenthesis indicates the duration of time that livestock exclusion practices have been in place on the livestock exclusion reaches.

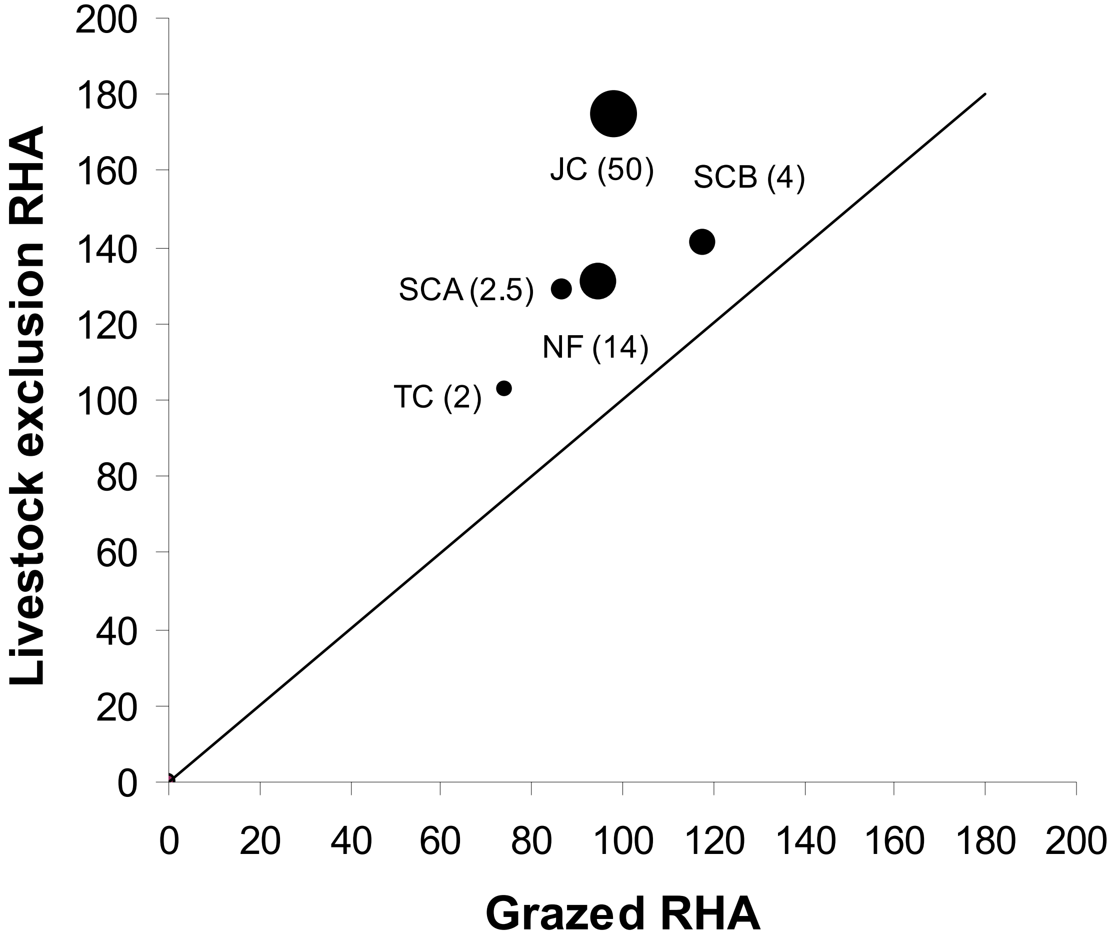
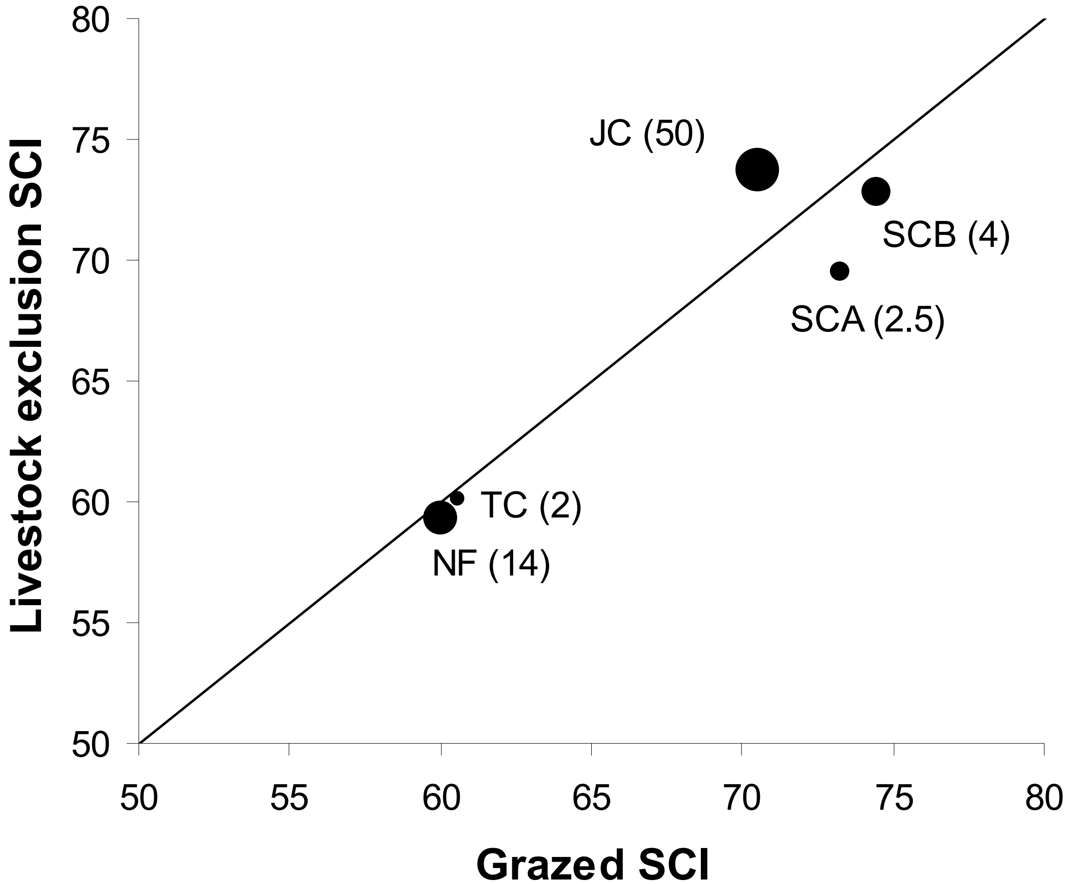


Figure 5.7
Stream Conditions Index (SCI) for the grazed and livestock exclusion reaches. The number in the parenthesis indicates the duration of time that livestock exclusion practices have been in place on the livestock exclusion reaches.



Chapter 6. Conclusions

We studied paired stream reaches with and without cattle access at five sites in southwest Virginia to: a) assess the effectiveness of existing livestock exclusion projects in improving stream morphology, streambank and riparian characteristics, and benthic macroinvertebrate assemblages; and b) determine the range of improvement expected over time since livestock exclusion implementation. The following conclusions can be drawn in response to our original hypotheses:

> Hypothesis 1: There is a significant difference in the geomorphology, bank characteristics, and benthic macroinvertebrate assemblages between the grazed and fenced reaches.
>
> *There were significant differences in stream morphology, streambank soils, and riparian vegetation between paired grazed and livestock exclusion reaches; however, the benthic macroinvertebrate assemblages were not significantly different.*
>
> Hypothesis 2: There is a gradient of improvement associated with the length of time livestock exclusion practices have been in place.
>
> *The Reach Condition Index (RCI), a qualitative geomorphic assessment methodology, increased with time since livestock exclusion. All other parameters did not show a clear temporal response.*

The lack of benthic macroinvertebrate assemblage response was likely due to the overall land use, soils, and geologic features of the watershed rather than the localized impacts of livestock access to the stream reach. These results suggest that short sections of livestock exclusion may not result in improved biological integrity. Rather, a watershed-wide initiative to reduce negative impacts (including livestock access) from streams must be taken for improvements to occur. About $91 million is allotted annually to the CREP program to fence livestock out of streams; however, the fencing and buffering of small isolated stream reaches may not be the best restoration strategy without addressing additional stressors at the watershed scale. The implementation of CREP is currently on a volunteer basis; this research suggests federal regulation to require and enforce livestock exclusion would have to be established to achieve widespread recovery of aquatic ecosystems.

A typical CREP buffer is approximately 30 meters wide, which can reduce sediment, nitrogen, and phosphorus inputs to the stream by at least 77% (VDCR, 2006). The anticipated reduction in sediment and nutrient loads can help Virginia meet water quality improvement goals, such as reducing nutrient loads to the Chesapeake Bay drainage basin by 40% (VDCR, 2006). In addition, the results from this study suggest that wider, grazed streams will begin to narrow upon livestock exclusion, but may widen over time as the woody vegetation matures.

Chapter 7. Future Research Needs

This research was conducted as a pilot study to investigate the effectiveness of livestock exclusion BMPs in improving stream morphology and benthic macroinvertebrate assemblages. Future research needs include:

1. Similar research should be conducted on more paired reaches across a wide range of ecoregions, watersheds, and local conditions.
2. To evaluate the improvements or changes over time the temporal range of the study reaches needs to be expanded. Since livestock exclusion programs such as CREP have only been in place since 1985, finding a large range in time since livestock exclusion is challenging (VDCR, 2006). Initiatives that farmers have taken to keep livestock out of streams prior to 1985 would have to be found by speaking to individual landowners. Alternatively, the recovery of a single reach after livestock exclusion could be studied over time.
3. There were many factors that contributed to the variability in the data including length and type of riparian buffer, upstream pollution impacts, watershed characteristics, stream order, drainage area, and order of the paired grazed and livestock exclusion reaches along the stream. To reduce this variability, headwater stream reaches with livestock exclusion practices upstream of the paired grazed reach could be used.
4. Future studies should include replicate sampling of benthic macroinvertebrates for increased statistical power when examining the benthic macroinvertebrate assemblages' responses to livestock exclusion.

Definition of key variables

A = cross-section area (m^2)

W_{fpa} = width of the flood prone area (m)

W_{bkf} = bankfull width (m)

D_{bkf} = bankfull (hydraulic) depth (m)

W_{bkf}/D_{bkf} = width to depth ratio

R_D = reach depth variability

D_{02} = the 2nd percentile on the distribution curve for measured thalweg depths

D_{50} = the 50th percentile on the distribution curve for measured thalweg depths

D_{98} = the 98th percentile on the distribution curve for measured thalweg depths

$d_{15.9}$ = 15.9% of the bed substrate distribution is finer (mm)

d_{50} = median particle size of four-hundred pebbles in two riffles and two pools (mm)

Riffle d_{50} = the median bed particle size of two-hundred pebbles in two riffles.

$d_{84.1}$= 84.1% of the bed substrate distribution is finer (mm)

σ_g = geometric standard deviation of the particle size distribution

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