Responses of Stream Channels, Riparian Habitat,

and Aquatic Invertebrate Community Structure to

Varied Livestock Grazing Exposure and Management in the

West Walker River Watershed (Mono County, California)

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Introduction

Studies of the effects of livestock grazing on stream and riparian habitats have seldom examined linkages between the physical environment of the stream and resident invertebrate aquatic life, or how these respond to differing exposures to grazing and land management practices. Changes in land use management related to livestock grazing after 1998 in the drainages of the West Walker River of the eastern Sierra Nevada provided a setting for conducting integrated monitoring on streams of varied size and grazing regime. Data were collected to evaluate differences between streams at the start of wetlands restoration projects and after three years under these new practices.

Biological structure and integrity of stream environments can be ascertained from a quantitative description of the inhabitant organisms. Aquatic insects and other invertebrates are central to the function of stream ecosystems, consuming organic matter (wood and leaf debris) and algae, and providing food to higher trophic levels (fish and riparian birds). These native organisms also tolerate pollution and disturbance to varying degrees and integrate such impacts over their entire lives. Thus, their community composition can be a powerful indicator of short- and long-term water quality and habitat conditions. Collections of the zoobenthos (bottom-dwelling fauna) may be used to evaluate the relative abundance of different taxa, feeding guilds, pollution indicators, and diversity, in order to develop a quantitative basis for measuring ecological attributes of the stream. Monitoring relative to reference sites (having little or no impact but similar physical setting), and/or over time, then permits impact problems or recovery to be quantified (Rosenberg and Resh 1993, Davis and Simon 1995, Karr and Chu 1999).

Use of quantitative data on the structure of biological communities in evaluating stream habitat quality is known as bioassessment (USEPA 1999). In this study, bioassessment surveys of baseline conditions (i.e. 1999) and conditions three years later at managed grazing sites (i.e. "treatment" sites) were compared with those at sites where grazing was minimal and unchanged (i.e. "control" sites) to examine the effects of management practices implemented at the managed grazing sites. Minimal grazing sites were treated as a control group to characterize whether trends found at managed grazing sites could be attributed to management action (little or no change was expected for the minimal grazing sites). Several physical habitat parameters (i.e. riparian cover, bank stability, substrate particle-size distribution) were measured concurrently to establish and link between the effects of grazing on physical habitat and the quality of the biological community.

There is a distinctive syndrome of degraded stream habitat conditions that characterize overgrazing by livestock (*see* Herbst and Knapp 1999). This often includes unstable and eroded banks, sedimentation, burial or embedding of rock substrates, loss of riparian vegetation cover and associated organic matter inputs, increased width-to-depth ratio, reduced current in shallow water, nutrient enrichment, and increased algae growth (Belsky *et al* 1999). Potential ecological impacts of overgrazing on benthic invertebrate communities include suspended sediment releases, loss of habitat through choked and silted substrate conditions, algae-microbial mats covering substrates, reduced dissolved oxygen, higher temperatures, and reduced leaf and wood litter as food and habitat structure.

Field Monitoring Study Design, Sampling Strategy, and Methods

Approach

Bioassessment monitoring was designed to accomplish the following objectives:

1. Describe and quantify the baseline condition of biological health and habitat at several sites within and near the project allotments on small to large sized streams
2. Compare baseline conditions with those found three years later
3. Examine the relationship between grazing regime and management practices with biological health and habitat in small to large streams

The invertebrate communities of control sites were used here to reflect the potential range of ecological conditions found in stream habitats matched to the treatment sites, but with minimal or reduced impacts related to land use. Sampling was conducted to frame the natural background spatial and temporal variability of streams nearby and within the upper West Walker River watershed. This was accomplished by sampling a varied size range of reference streams on multiple occasions in 1999 at the onset of changes in grazing practices and in 2002 after three years of altered land use. Quantitative description of biological communities at sites over a range of grazing management practices permitted development of a dose-response linkage between grazing stress and biological signals.

The goal of this project was to determine the extent of biological response in streams to changes in grazing management practices. This information may be used by managers to determine if recently implemented management practices have been effective and to what degree, whether and where improvements may be needed, and the nature and extent of future monitoring efforts.

Site Selection

A variety of physical habitat features of streams can affect benthic invertebrate communities (Resh and Rosenberg 1984). In addition to natural erosion and sedimentation, the size, gradient and elevation of stream reaches may contribute to shaping communities, as may water quality impacts associated with land use. Site selection for bioassessment was thus guided by the need to account and control for varied environmental background influences.

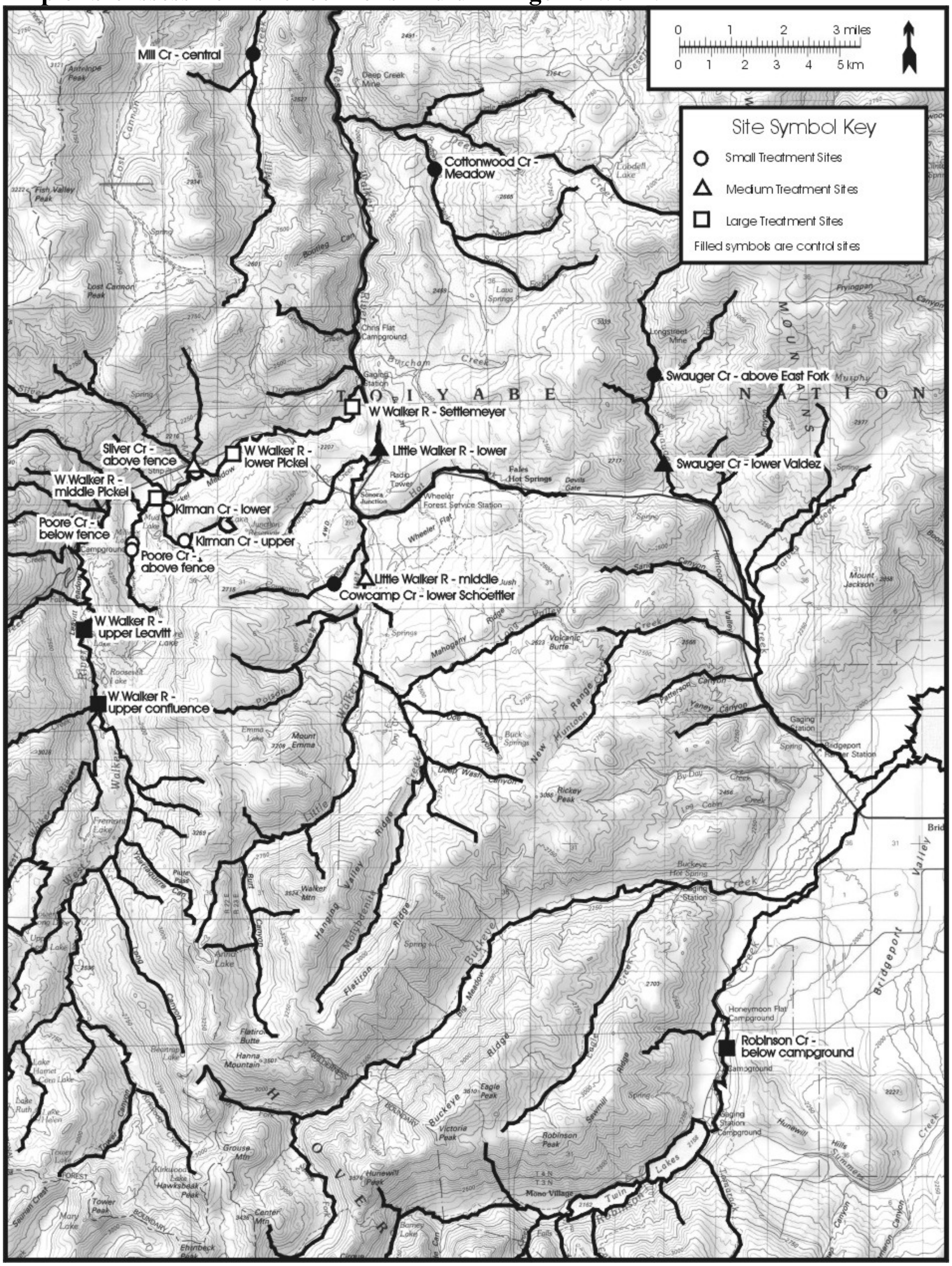
A total of 18 bioassessment sites were selected, nine managed grazing, or treatment sites, and nine minimal grazing, or control sites (refer to list below and map). Eight of the treatment sites were within the Junction Cattle and Horse Allotment on the West Walker River and tributaries from Pickel Meadow to Sonora Junction. The ninth treatment site was within the Little Walker Cattle and Horse Allotment just east of the West Walker River sites. The nine control sites were dispersed more widely, including three sites in the East Walker River drainage. Control sites were selected to reflect the natural spatial and temporal variability expected for similar stream types. These sites were selected to provide contrasting land use conditions, and yet be proximal enough to minimize spatial heterogeneity in environmental conditions.

Each of the sites were classified by mean channel width and discharge as small (width < 2.5 m; Q < 10 cfs), medium (width < 10 m; Q < 75 cfs), and large (width > 10 m; Q > 75 cfs). For both treatment and control site classes, there were four small sites, two medium sites, and three large sites each (Table 1). In 1999, all sites were surveyed between 2 and 20 August. In 2002, they were each sampled about a month earlier, between 8 and 22 July, to best match flow conditions, which were lower in the latter year.

List of treatment and control bioassessment sites by size class, and date surveyed

Treatment Sites		**Control Sites**	
Large			
W Walker River – middle Pickel	8/10/99 & 7/11/02	W Walker River – upper confluence	8/20/99 & 7/17/02
W Walker River – lower Pickel	8/12/99 & 7/11/02	W Walker River – upper Leavitt	8/20/99 & 7/17/02
W Walker River – Settlemeyer	8/18/99 & 7/8/02	Robinson Cr – below campground	8/17/99 & 7/12/02
Medium			
Silver Cr – above fence	8/10/99 & 7/19/02	Swauger Cr – lower Valdez	8/19/99 & 7/16/02
Little Walker River – middle	8/2/99 & 7/10/02	Little Walker River – lower	8/3/99 & 7/8/02
Small			
Poore Cr – above fence	8/9/99 & 7/18/02	Cowcamp Cr – lower Schoettler	8/2/99 & 7/10/02
Poore Cr – below fence	8/9/99 & 7/18/02	Cottonwood Cr – meadow	8/13/99 & 7/12/02
Kirman Cr – lower	8/19/99 & 7/9/02	Mill Cr – central	8/19/99 & 7/19/02
Kirman Cr – upper	8/18/99 & 7/22/02	Swauger Cr – above East Fork	8/17/99 & 7/16/02

Map of bioassessment site locations and drainage network.



Two of the bioassessment sites were subject to drastic alteration between 1999 and 2002. The Mill Cr – central site and portions of its watershed burned prior to sampling in 2002. Thus, measurements from 2002 provided insight into the effects of fire on stream physical habitat and biota, but were not comparable to the other data, and were excluded from the analysis of the effects of the grazing management practices. The original Kirman Cr – upper site was dry in 2002, so an alternate site (i.e. alternate upper) with perennial flow, located approximately 500 m downstream from the original site, was sampled. Conditions at this site were assumed to be comparable to the original site, and thus comparisons between these sites are included in the analysis.

Sampling Methods

The data gathered consisted of physical habitat surveys and biological sampling of benthic macroinvertebrates, algae and organic matter. Each site was defined as a 150-meter length study reach, located by GPS-UTM coordinates and elevation (near lower end of each site). The longitudinal distribution and length of riffle and pool habitats were first delineated and then used to determine random locations for sampling of benthic macroinvertebrates from riffle habitat. Slope over the reach was measured with a survey transit and stadia rod, and sinuosity was estimated from straight-line distance over the 150 m channel, or using maps and measuring 500-1000 meters of stream length centered on the study reach. Physical habitat was measured over the length of each reach using 15 transects spaced at 10 meter intervals. Water depth, substrate type, and current velocity were measured at five equidistant points on each transect, along with stream width, bank structure (cover/substrate type and stability rating), riparian canopy cover, and bank angle. Bank structure between water level and bankfull channel level was rated as open, vegetated, or armored (rock or log), and as stable or eroded (evidence of collapse or scour scars). Bank angles were scored as shallow, moderate, or undercut (<30°, 30-90°, and >90°, respectively), and riparian cover was measured at each stream edge and at mid-stream facing up- and downstream using a densiometer (i.e. number of concave mirror grid points reflecting vegetation). The type and amount of riparian vegetation along the reach was also estimated by qualitative visual evaluation. The embeddedness of cobble-size substrate was estimated as the volume of the rock buried by silt or fine sand for 25 cobbles (encountered during transect surveys and supplemented with randomly selected cobbles as needed). Discharge was calculated from each transect as the sum of one-fifth the width times depth and current velocity at each of the five transect points, and averaged. Basic water chemistry and related measures performed in 1999 and 2002 included temperature, pH, alkalinity, hardness, dissolved oxygen (DO), conductivity, turbidity, and sulfate. Nitrate, total Kjeldahl nitrogen (TKN), total phosphorus, and silica were measured in 2002 only. Documentation of site conditions also included photographs taken from mid-stream channel looking upstream at 0, 50, and 100 meters, and downstream at 150 meters.

Biological sampling consisted of 5 replicate benthic samples taken in riffle zones with a 30-cm wide, 250 µm mesh, D-frame kick-net. Each replicate was comprised of a composite of three 30 x 30 cm sample areas taken across the riffle transect or over riffle areas of varied depth, substrate, and current. This composite of microhabitats provides a more representative sampling and reduces the variability among replicate samples. Samples were processed in the field by washing and removing large organic and rock debris in sample buckets followed by repeated elutriation of the sample to remove invertebrates from remnant sand and gravel debris. Remaining debris was inspected in a shallow white pan to remove any higher density organisms, such as cased caddisflies (e.g., Glossosomatidae), snails, or other molluscs. Elutriated and inspected sample fractions were then preserved in ethanol, and a small volume of rose bengal stain added to aid in lab processing.

Invertebrate field samples were subsampled in the laboratory using a rotating drum splitter, sorted from subsamples under a magnifying visor and microscope, and identified to the lowest practical taxonomic level possible (usually genus; species when possible based on the availability of taxonomic keys) except for oligochaetes and ostracods, which were identified to order only. A minimum count of 250 organisms was removed from each replicate for identification (in practice averaging about 300-500). Data analysis yielded information on taxonomic composition by density and relative abundance. Metrics of community structure were calculated to express biological health in terms of diversity, composite community tolerance, number of sensitive taxa (mayfly-stonefly-caddisfly), dominance, and other measures of composition. All stages of sample processing and identification were checked using quality control procedures to assure uniformity, standardization and validation (QAPP; Herbst 2001).

The benthic food resources of stream invertebrates were also quantified by collecting three replicate samples of both organic matter and periphyton. Particulate organic matter was sampled from undisturbed, stream bottom riffles as above for invertebrates using a 250 µm mesh, D-frame net. Samples were poured through a 1-mm screen, with the retained wood and leaf particle debris then weighed as a wet biomass measure of coarse particulate organic matter (CPOM). The fine fraction passing through the screen (particle range 250 to 1,000 µm) was collected in a 100 µm mesh aquarium net, placed in a sample vial, and preserved in formalin. At the laboratory, samples were dried and ashed in a muffle furnace to quantify ash-free dry mass of fine particulate organic matter (FPOM).

Algal periphyton was quantified by scrubbing attached algae off the surface of a cobble-size rock selected randomly from undisturbed, mid-stream riffle habitat. Algae was removed using a wire brush, and the rinsate homogenized using a large syringe, and scissors if needed for filamentous algae. The homogenized rinsate was subsampled for (a) chlorophyll *a* by filtration through a 1 µm pore-size glass fiber filter, recording the volume filtered, and (b) archival of algae for cell counts and taxonomic identifications (preserved in formalin and Lugol's stain). The area of the rock was estimated from measures of length, width, height, and circumference. The filters, preserved by freezing, were analyzed for chlorophyll *a* in the laboratory using cold ethanol extraction for 24 hours followed by fluorometry. Concentrations were determined using a standard curve developed using a spectrophotometric extinction coefficient found in the literature.

Data Analysis (exposure and response variables)

Several biological response variables were calculated using the invertebrate taxa data. The variables selected are based on measures that have been commonly applied in bioassessment analyses and have an expected (and documented) response to environmental stress. Selected metrics of invertebrate community structure and function were used to evaluate responses to differences in the grazing regime among streams of different size, and these included measures of diversity, tolerance, and food resource utilization.

Stream habitats with minimal land use related disturbance, heterogeneity in stream bed substrates and food resources, stable banks, mixed riparian cover, and unaltered flow regime typically contain a diverse array of sensitive taxa inhabiting varied microhabitats, using different food resources, and having varied life cycles. Environmental stressors compromise the quality and variety in stream habitats, resulting in the loss of structural and functional diversity, and of organisms intolerant of stress (diversity is lost, composition changes).

List of examples of invertebrate community structure metrics and expected responses to stress associated with livestock over-grazing habitat degradation

Biological Metric	Metric Definition	Expected Response to Stress
Taxa Diversity (mean of samples)	Total number or richness of taxa found in a sample (reflecting resource variety)	Decrease
EPT Diversity Index (ephemeroptera, plecoptera, and trichoptera)	Number of taxa belonging to mayfly, stonefly, and caddisfly orders, usually regarded as intolerant of pollution	Decrease
%EPT	Percent of the organisms present belonging to one of the EPT orders	Decrease
Biotic Index	Composite measure of community tolerance to pollution (based on tolerance values and relative abundance)	Increase
No. of Sensitive Taxa (0-2)	Number of taxa with tolerance values of 0, 1, or 2 (scale of 10; least to most tolerant)	Decrease
No. of Tolerant Taxa (7-10)	Number of taxa with tolerance values of 7, 8, 9, or 10 (scale of 10)	Increase
%Dominance	Percent of organisms comprising the most abundant taxon (resource imbalance)	Increase

In this study, our objective was to elucidate the biological response to varied grazing regimes and practices. The primary contrast was between the minimal-grazed sites (controls), and grazed sites where new land use practices were initiated (treatments), examined within streams grouped by size class.

Several measures taken during physical habitat surveys can serve as indicators of grazing stress, and thus could be used in evaluating a dose and response. These included the percent fines and sand that along with embeddedness of cobble provided an indication of sedimentation from erosion. Bank erosion, riparian vegetation cover also provided indications of grazing impacts. Direct estimates of grazing exposure were also calculated from records of grazing intensity maintained by the US Forest Service (i.e. AUMs), normalized to the length of stream channel occurring within each allotment.

Potential for Mercury Contamination Impacts

Elevated concentrations of mercury in tissues of migratory common loons that stage at Walker Lake, Nevada, and of the lake's tui chub, prompted a study of potential source areas of mercury in the Walker Lake basin and impacts to aquatic invertebrates and stream sediment (Wiemeyer 2002). Invertebrate tissue samples and sediment samples were collected from a number of sites in the basin, primarily downstream of suspected source areas associated with historic mining. Three of the sites sampled were in the vicinity of bioassessment sites in this study. Invertebrate samples were collected in the West Walker River near the Highway 108 crossing about 3.25 km upstream of the Settlemeyer site (stonefly larvae), in the Little Walker River about one km below the lower site (stonefly larvae), and in Robinson Cr near the outlet of Twin Lakes, about 3.5 km upstream of the below campground site (stonefly larvae and crayfish). At these three sites, no evidence of upstream mining activity was found.

Concentrations of mercury in invertebrate tissues from the West Walker River site (n=1) and the Robinson Creek site (n=5) were all below 0.1 μg/g (dry weight), and were considered indicative of background, or reference, conditions. Mercury concentrations in tissue samples at the Little Walker River site (n=2) were slightly higher, but below 0.15 μg/g, and considered unimpaired in comparison to sites associated with upstream mining. In addition, mercury concentrations in sediment at all three sites were less than 0.05 μg/g (dry weight), below a reported sediment effects threshold for mercury for freshwater invertebrates of 0.2 μg/g. Though this is a limited data set from which to draw conclusions, it indicates that mercury contamination was not likely to be have been a factor in this study, given that none of the bioassessment sites are known to be associated with upstream mining, and therefore is not considered further.

Results and Discussion

Physical Habitat Measurements

This section summarizes trends in physical habitat measurement data for the bioassessment sites. To investigate the response of treatment sites to these actions, seven physical habitat parameters expected to have the potential to exhibit an effect (i.e. recovery response) over the three years of the study were examined:

Parameter	Recovery Response
Percent stable banks	Increase
Percent vegetative bank cover	Increase
Percent riparian cover	Increase
Percent undercut banks	Increase
Percent fines + sand	Decrease
D_{50} particle size	Increase
Percent cobble embeddedness	Decrease

Trends for these parameters at each site are summarized quantitatively in Table 1, qualitatively in Table 2, and discussed below. Results for all physical and chemical measurements are compared graphically in Figures 1 through 28.

Comparison of Treatment and Control Sites at the Start of the Study

At the start of the study, each size class of the control sites had three to ten times more riparian cover (Figure 1), eleven to 24 percent more stable banks (Figure 2), and seven to eleven percent more undercut banks (Figure 3), than corresponding size classes of treatment sites. Differences in percent vegetated banks (Figure 4), percent fines + sand (Figure 5), D_{50} particle size (Figure 6), and percent cobble embeddedness (Figure 7) did not exhibit a consistent pattern between treatment and control sites. Comparison within size classes, however, reveals some contrasts in these measurement values. Percent fines + sand was about three times higher at medium control sites than medium treatment sites; conversely it was about two times higher at small treatment sites than at small control sites. D_{50} particle size was about three times larger at small control sites than small treatment sites.

Morphologic and hydrologic characteristics (i.e. channel width [Figure 8], depth [Figure 9], velocity [Figure 10], and discharge [Figure 11]) were similar between treatment and control sites within each size class, validating the comparability of treatment and control sites with regard to these factors. Mean mass of fine particulate organic matter (FPOM) was 33 percent higher for small treatment sites (Figure 12), while mean mass of coarse particulate organic matter (CPOM) was about three times higher for small control sites (Figure 13); medium and large sites did not exhibit such disparity in organic matter. Similarly, mean concentrations of periphyton (i.e. attached algae; reported as mass of chlorophyll *a* per surface area of cobble substrate) were similar for medium and large sites, but were about four times higher for small treatment sites (Figure 14). Such higher productivity at small treatment sites was also exhibited by higher occurrences of algae and aquatic macrophytes on channel substrates, although the pattern was not as consistent site-by-site (Figure 15). Herbaceous cover scores for the riparian area adjacent to the creeks were similar between managed and control sites, with the exception that woody cover was substantially higher for the control sites (Figure 16). This was also reflected in the greater occurrence of tree bank cover (i.e. Vt) at control sites (Figure 17).

For the chemical analytes measured in 1999, pH was similar between managed and control sites (range 7.4 to 8.9; Figure 18). Large and small treatment sites generally had higher values for alkalinity (36 to 118 vs. 24 to 61 mg HCO_3 l^{-1}; Figure 19), hardness (24 to 111 vs. 14 to 68 mg l^{-1}; Figure 20), and conductivity (53 to 276 vs. 24 to 156 µS; Figure 22), while the opposite was true for medium sites. Turbidity was higher at small control sites (Figure 23). An anomalously high conductivity value was measured at a medium treatment site (i.e. Silver Cr – above fence). Sulfate values were higher at small treatment sites (4 to 17 vs. <2 to 6 mg l^{-1}; Figure 26), but lower at medium treatment sites (<2 to 2.6 vs. 2.4 to 7.3 mg l^{-1}).

Comparison of Trends in Treatment and Control Sites Between 1999 and 2002

Trends at treatment and control sites between 1999 and 2002 are described in detail below. In general, most small treatment sites showed a recovery response for most of the seven recovery parameters, while most small control sites did not, indicating that management actions at small treatment sites were effective in inducing improved channel and riparian conditions over the three year period (Tables 1 and 2). Most large and medium treatment sites showed a recovery response for most parameters as well, but so did large and medium control sites, such that recovery of medium and large treatment sites cannot necessarily be attributed to management actions for this three year period.

The Mill Cr site and portions of its watershed burned two to three weeks prior to the 2002 sampling event, thus it was excluded from the comparison of treatment and control sites over the three year period. Measurements at this site did provide insight into the effect of fire on physical habitat. Compared to other small control sites, the Mill Cr site showed substantial declines in percent riparian cover, percent stable banks, percent cobble embeddedness, and herbaceous and woody cover, all findings that would be expected to accompany the loss of vegetation and increased rates of erosion associated with a fire. The Mill Cr site also showed a substantial increase in CPOM and substrate modifiers (i.e. algae, macrophytic vegetation, wood, and moss), and had the highest concentration of total phosphorus and total Kjeldahl nitrogen (i.e. organic forms of nitrogen) compared to all other sites in 2002 (the only year in which these analytes were measured), but one of the lowest concentrations of nitrate (i.e. inorganic forms of nitrogen). This is indicative of a substantial release of nutrients and increase in stream productivity associated with the fire. Finally, following the fire the maximum mean daily water temperature was 18 ^{o}C; it had not exceeded 15 ^{o}C the previous two years. Increased stream temperatures were likely attributable to the loss of riparian cover due to the fire.

Riparian Cover

Riparian cover would be expected to increase as management actions reduce its removal from grazing. Riparian cover shades the stream, lowering water temperatures, serves as a source of large woody debris and habitat heterogeneity in the channel, and also provides habitat for terrestrial organisms. Riparian cover was estimated using a densiometer to measure the density of reflected canopy over the stream. Herbaceous and woody cover were estimated by scoring the abundance of cover based on a visual inspection of the stream reach.

Mean riparian cover increased at all small and medium treatment sites, except Poore Ck – below fence, between 1999 and 2002, while there was not a substantial change at the large treatment sites (Figure 1). Riparian cover also increased at small and medium control sites, but on average, riparian cover increased two to three times more at small and medium treatment sites.

Results were more ambiguous for herbaceous and woody cover scores. At treatment sites, herbaceous cover scores increased by one at one site, decreased by one at one site, and did not change at all other sites (Figure 16). At control sites, herbaceous cover scores declined by one and two at two sites, increased by one at one site, and remained the same at all others (note that the Mill Cr site is excluded). Woody cover scores increased at one large, medium, and small treatment site each, and remained the same at all others. At control sites, woody cover scores declined at all large sites, at one medium control site, and none of the small sites.

Bank Stability and Cover

Implementation of effective grazing management would be expected to increase bank stability by limiting trampling and disturbance at the stream margins. Vegetation on banks would be expected to increase as well. With improved vegetative bank cover stabilizing soils, natural erosion processes would be expected to increase the amount of undercut banks.

By 2002, bank stability improved at six of the nine treatment sites, while it either declined or did not change at each of the control sites (Figure 2). Quantitatively, the mean percent of stable banks for each size class increased 3 to 7 percent at treatment sites. Trends in the amount of undercut banks were not as consistent (Figure 3). All small and large treatment sites showed increased undercut banks, or did not change, a 3 to 4 percent increase relative to small and large control sites. But, medium treatment sites showed a decline in undercut banks relative to medium control sites. Percent vegetated bank cover increased by 12 and 15 percent, respectively, at the small and medium treatment sites relative to small and medium control sites (Figure 4). However, it declined by 9 percent at the large treatment sites relative to large control sites.

Channel Substrate

Erosion upstream and within a stream channel reach, as is often associated with grazing, will tend to shift a stream's substrate particle size distribution toward smaller particles, such that the percent fines + sand increases and the median particle size (i.e. D_{50}) decreases. In addition, larger quantities of fines and sand particles will tend to fill interstitial spaces around larger substrates, increasing cobble embeddedness. But, there is often considerable temporal and spatial variability in the substrate properties of a stream, due to factors such as the occurrence of flushing flows and dynamics associated with sediment source areas. And source areas may occur throughout a watershed, so associating stream channel substrate size distribution with adjacent land use can be problematic.

Percent fines + sand declined or did not change at all treatment sites and all large and medium control sites, while it increased slightly at the three small control sites (Figure 5). Thus, only small treatment sites exhibited a recovery trend relative to control sites over the three year period, although it occurred at only two of the small treatment sites (i.e. Poore Cr – below fence and Kirman Cr – upper).

D_{50} particle size increased at all large sites and all but one medium site (Figure 6). This increase was greater for control sites, thus such a recovery response cannot necessarily be attributed to management action for these sites. For small sites, three of four treatment sites had an increased D_{50} particle size, while it declined at all small control sites, indicating that management action probably improved substrate size distribution at small sites over the three year period.

Despite these indications of improved substrate conditions at small treatment sites, percent cobble embeddedness increased at all four small treatment sites, and at small control sites as well (Figure 7). It did decline at all large and medium treatment sites, as well as most large and medium control sites.

Morphologic and Hydrologic Characteristics
There were no substantial differences in mean channel width, depth, or velocity for any site between 1999 and 2002 (Figures 8 – 10). Results were similar for discharge, except in Silver Cr – above fence, where discharge declined by about two-thirds, and in all small control sites, where discharge declined by 50 to 80 percent (Figure 11). Some of the decline in discharge found at the latter sites may be attributable to the difficulty in estimating such low discharge values (i.e. $< 0.1\ m^3\ s^{-1}$).

Food Resources
Several measurements were made to characterize the quantity and quality of food available to the stream invertebrate community, including estimates of FPOM, CPOM, periphyton biomass, and the occurrence of substrate modifiers on the stream bottom (e.g. algae, aquatic macrophytes, leaf, wood, roots, detritus, and moss). The type of food resources available can also be a function of physical habitat disturbance, and profoundly influence the invertebrate community. For example, a dense riparian canopy will shade the stream channel and inhibit primary productivity, making FPOM and CPOM the principal food resource for invertebrates, rather than algae.

FPOM mass increased at all but one control site (Swauger Cr – lower Valdez) and at medium treatment sites (Figure 12). Conversely, FPOM declined at all small treatment sites. At nearly all sites, the standard deviations (i.e. error bars) overlapped, indicating that these differences are not statistically significant at probabilities typically assumed (i.e. $p = 0.95$). Mean differences in FPOM mass between managed and control sites for large, medium, and small size classes were 0.1, 0.2, and 1.4 g in 2002, indicating the level of similarity between the treatment and control groups. Trends in the mass of CPOM were similarly obscure within and among site and size classes, with error bars between year overlapping for most sites (Figure 13). Periphyton biomass increased at all but one site (Swauger Cr – lower Valdez; Figure 14). Small treatment sites continued to have substantially greater periphyton biomass, although a maximum value measured at Cowcamp Cr – lower Schoettler somewhat tempers this conclusion. Trends in substrate modifiers within and among site and size classes were largely inconsistent (Figure 15). One exception was a substantial increase in the occurrence of aquatic macrophytes at all small treatment sites, and a decline in the occurrence of algae at two of these sites.

Bank Cover
Decreased vegetation removal, trampling, and erosion resulting from improved grazing practices would be expected to increase the occurrence of vegetative cover and reduce the amount of un-vegetated, or open, banks. Open bank cover decreased at all small

treatment sites and large and medium control sites (Figure 17). Trends in other site and size classes were not consistent. The largest increases in vegetative bank cover occurred at Silver Cr – above fence (30 percent) and Poore Cr – above fence (33 percent), medium and small treatment sites, respectively.

Water Chemistry
Similar to stream channel substrate properties, stream water chemistry can be influenced by factors throughout the watershed, as well as local land use practices. Small treatment and control sites generally had higher concentrations of many of the chemical analytes (i.e. alkalinity, hardness, conductivity, total phosphorus, silica, sulfate, nitrate, and TKN) compared to large and medium sites (Figures 18-28). This may be attributable to the lower discharge at small sites, and thus reduced dilution effects.

At all but one site (Poore Cr – above fence), pH declined between 1999 and 2002 (Figure 18). Alkalinity increased at all small treatment and small control sites, but trends were inconsistent for large and medium sites (Figure 19). Hardness declined at all large and medium managed and control sites, and increased at most small managed and control sites (Figure 20). Conductivity increased at all but a few of the sites (Figure 22). Turbidity increased at all but one treatment site, and at most control sites (Figure 23). Similar to alkalinity, sulfate concentrations increased at all small treatment and small control sites, but trends were smaller and inconsistent for large and medium sites (Figure 26).

Stream Temperature
A few trends were apparent in mean water temperature measurements between 1999 and 2002 (Figure 29), available for five treatment sites and eight control sites (only a partial record was available for two of the treatment sites). Except for the Robinson Cr site and the Mill Cr site post-fire, maximum mean daily temperatures never or rarely exceeded 15 ^{o}C at each of the control sites. At four of the five treatment sites, sustained mean daily temperatures above 15 ^{o}C were measured during the summers. At the Poore Cr site, mean daily temperatures increased each summer of the study, exceeding 27 ^{o}C in 2002. Cottonwood Cr was the only control site that showed a trend of increasing temperature over the period of record.

Stream Invertebrate Community
With improvement in habitat quality conditions expected with reduced grazing intensity at the managed treatment sites, there should be associated enhancements in the capacity of the aquatic environment to support more species (total diversity and EPT diversity), reduced enrichment (lower eutrophication evidenced in decreased abundance of algae and invertebrate biomass, especially of periphyton grazers), increased numbers of sensitive indicator taxa while tolerant forms are reduced, an increased proportion of mayfly/ stonefly/ caddisfly (EPT) taxa comprising the community, greater diversity within groups that must partition shared resources, reduced dominance of the community by few taxa (more even distribution), and an increase in the proportion of organisms using decaying riparian vegetation as a food resource (shredders and collector-gatherers). The data are reported here as summed values from the combination of all replicates taken at each site.

No differences in total and EPT diversity between control and treatment groups were observed at the outset of the study for either medium or large size streams, but consistently lower diversity was evident for the small size class of grazed streams (Figure 30a) compared to the small stream controls. This pattern was still evident in 2002, without any indication of enhanced diversity on grazed treatment streams (Figure 30b). Invertebrate density, indicative of productivity, was relatively low across all large and medium size streams (around 10,000 invertebrates per square meter), elevated to the range of 10-20,000 m^{-2} for minimal grazed small control streams, and then usually increased further to over 20-30,000 m^{-2} in small grazing treatment streams in both 1999 and 2002 (Figure 31 a and b). Contribution of mayflies, stoneflies and caddisflies to the total numbers in the invertebrate community was variable but usually high (>50%) for large and medium sized streams at both the beginning and end of the study, with no consistent differences evident between treatment and controls (Figure 32 a and b). Small streams that were minimally grazed usually had more than 40% EPT, but the managed grazing streams mostly held less than 30% EPT to no more than 40%. Large and medium size streams held equivalent numbers of sensitive indicators (25 to 35 taxa with tolerance values of 0-2) and tolerant indicators (5 to 10 taxa with tolerance values of 7-10) in both years and in managed and minimal grazed groups (Figure 33 a and b). Small streams with minimal grazing were also in about this same range of indicators, but managed grazing treatments held only about 10 to 15 sensitive taxa and in the range of about 5 to 15 tolerant forms, and showed no improvement over time. Diversity within groupings of taxonomically related organisms with similar resource requirements did not show a consistent pattern between grazing treatment and control groups among large or medium size streams (Figure 34 a and b). Small managed grazing streams again showed a substantial loss in diversity relative to their minimal grazing counterparts, and though diversity in these groups increased in 2002, this also occurred in the minimal grazing controls. Functional feeding groups were dominated by collector-gatherers across all streams and both years of study except in small managed grazing treatments where algae grazers were equally or more abundant (Figure 35 a and b). This observation is also consistent with the abundance of algae observed as cover and chlorophyll a in small treatment streams.

Mill Creek contrasts from 1999 to 2002 indicated more grazers and shedders in the post-wildfire stream community, with lower %EPT (73% to 48%), more tolerant taxa (from 6 to 15), and fewer sensitive taxa (30 to 26). Other measures showed little change.

Livestock Grazing Histories

Records maintained by the US Forest Services on actual use (time period, number and type of stock) were reliable for most sites back to 1980. Grazing intensity was calculated as AUMs (animal unit months) using the standard assumption that one cow = 1.0 AUM, one sheep = 0.2 AUM, and one cow-calf pair = 1.32 AUM. The AUMs calculated from stocking numbers reported and duration of stay were then divided by the total length of channel within an allotment to distribute use according to intensity of riparian/channel area exposed to yield AUMs per km of channel for each stream site. This was done for the period before 1998, and after 1998, when the new management practices became established. These practices ranged from complete removal to rest rotation and seasonal grazing (see project background report, Curry et al. 2004). Though stocking rates decreased after 1998, use was still high on 3 of 4 small treatment streams, and prior grazing use on Cow Camp Creek control (ungrazed after 1995) was intense (Figure 36).

Discussion

To evaluate stream habitat responses to grazing it is essential to have appropriate indicator variables and a gradient of grazing exposure impacts that allow description of the range of response to environmental stress. Within contrasts of grazing it is also important to recognize that stream watersheds possess differing susceptibility and resilience of riparian and upland vegetation communities to grazing impacts (Laycock 1991; Friedel 1991). Bank, channel, and aquatic habitat components may also respond over different time frames to disturbance or restoration and so require long-term monitoring to detect integrated changes (Herbst and Knapp 1999). Riparian vegetation is known to recover rapidly when grazing exposure is limited or eliminated, but channel structure and in-stream recovery of ecological health may require longer and be dependent on not only local conditions, but on upstream watershed land use, sediment sources, and the timing of sediment-flushing flows. Thus, grazing impacts and potential for recovery are likely to be conditional on the intrinsic resistance conferred by such features as bank armor, substrate size, gradient, flows, and other geomorphic and ecological attributes of each stream.

Other studies of similar streams have found such variability in recovery response to changed grazing management. At Cottonwood Cr in the White Mountains, channel dimensions within a 24-year-old exclosure showed no significant difference with adjacent grazed channel downstream, despite lush growth of stream bank vegetation within the exclosure (Kondolf 1993). This lack of recovery of pre-grazing channel morphology after so many years was attributed to increased runoff and sediment load from continued grazing upstream of the exclosure, as well as a relatively low-magnitude hydrologic flow regime with infrequent high flows. In another example, 15 years of complete rest from grazing on Mahogany Creek and implementation of rotation of rest grazing on Summer Camp Creek in northwest Nevada lead to improved bank stability and tree cover on both creeks, but pool quantity and quality decreased (Myers and Swanson 1996). This latter finding was attributed to the removal of large woody debris from the stream channels prior to the study period by fisheries managers. This material had not been replaced after nearly two decades, and thus recovery of natural pool-riffle sequences lagged behind other physical habitat improvements.

The condition of range habitat under exposure to overgrazing may not change in a linear fashion, but rather remain in a state of relative ecological stability until grazing impacts cause an alteration that pushes habitat condition across a threshold to another state (Friedel 1991; Laycock 1991). Such thresholds may be in soil stability, flood events in susceptible channels, and vegetation community types. The transitions from one stable state to another often appear to be short-lived and difficult to reverse, suggesting that some range improvements may not occur immediately and only through intensive management and habitat manipulations. Such degradation and recovery response dynamics have been documented in a variety of ecosystems (Scheffer *et al* 2001).

The physical and biological responses measured in this study are likely to vary significantly within and between years. A similar study of nine sites subjected to a range of grazing intensity demonstrated that habitat conditions varied at sites substantially in Spring (i.e. before grazing commenced) compared to Fall, following the grazing season (Herbst and Knapp 1999). The physical habitat parameter that most affected invertebrate community structure also shifted from embeddedness and fine sediment deposition in Spring, to algae cover of channel substrates in Fall. The most sensitive biological indicator metric shifted as well, from richness measures in Spring to tolerance measures in Fall. The four year study also found substantial within-site variation between years, independent of grazing. The primary factor cited was climatic alternation of a sequence of heavy and mild winters and associated late-spring flood discharges.

The results of the present study provide evidence that grazing impacts depend in part on stream size. The biota of medium to large size streams (as defined here) may be more resilient to the long-term effects of livestock grazing, and though large-scale features such as entrenchment of river channels, incision, and bank erosion may occur with overgrazing, the in-stream habitat may remain stable enough to resist impacts to benthic invertebrate inhabitants. The ecology of small streams by contrast appears to be consistently impaired by greater exposure to livestock grazing, and do not respond to reduced grazing pressure at least over the short-term three year period examined here. Small streams have greater connectivity with the landscape and so may be more vulnerable to adjacent land disturbance, and have more exposure to AUMs per channel length than larger and wider streams.

Small grazed streams were in general less diverse and more productive than controls that were exposed to minimal grazing, suggesting that adapted organisms were thriving under disturbed conditions (e.g. grazers, tolerant organisms). Enrichment and algal growth in these streams did not compensate for habitat degradation, and losses occurred in total diversity and among sensitive taxa, and the functional organization of food webs were altered (more algae grazers and reduced diversity within groups of taxa with overlapping resource utilization).

These same patterns were consistent in 1999 and 2002, with no indication of any improvement or changes that permitted the managed grazing sites to become more biologically similar to the minimal grazing sites. As with geomorphic processes over short time periods, there appeared to be no enhanced response in the structure and function of the benthic invertebrate community to reduction or elimination of livestock grazing after 3 years under this modified land use management. The duration of this lag in response may be more closely related to in-stream recovery of geomorphic processes and channel structure than to the riparian regeneration that occurs more rapidly on stream banks in the absence or reduction of grazing. Long-term monitoring of future conditions in perhaps another ten years may provide a better indication of the potential of these small streams for ecological recovery.

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Table 1: Differences in select physical habitat parameters indicative of a recovery response between 1999 and 2002.

			Percent stable banks	Percent vegetative bank cover	Percent riparian cover	Percent undercut banks	Percent fines + sand	D_{50} Particle Size	Percent cobble embeddedness
Expected recovery response			Increase	Increase	Increase	Increase	Decrease	Increase	Decrease
Managed Grazing Sites	**Large**	W Walker R - middle Pickel	+ 3	- 3	- 1	+ 3	- 7	+ 32	- 7
		W Walker R - lower Pickel	+ 13	+ 13	0	+ 10	- 11	+ 14	- 22
		W Walker R - Settlemeyer	- 3	- 13	- 1	+ 7	- 13	+ 11	- 8
	Medium	Silver Cr - above fence	+ 23	+ 30	+ 16	- 17	0	+ 24	- 2
		Little Walker R - middle	- 10	- 7	+ 9	+ 3	- 4	- 14	- 11
	Small	Poore Cr - above fence	+ 23	+ 33	+ 28	+ 27	0	+ 23	+ 4
		Poore Cr - below fence	- 30	+ 7	- 9	0	- 4	+ 3	+ 28
		Kirman Cr - lower	+ 7	+ 20	+ 38	+ 7	0	- 18	- 5
		Kirman Cr - alternate upper	+ 13	+ 10	+ 44	+ 13	- 9	+ 100	+ 2
	Number of sites exhibiting recovery response (n = 9)		6	6	5	7	6	7	6
	Mean response		+ 4	+ 10	+ 14	+ 6	- 5	+ 19	- 2
Minimally Grazed Sites	**Large**	Robinson Cr - below campground	- 3	+ 13	+ 6	+ 7	- 5	+ 25	- 13
		W Walker R - upper confluence	- 3	0	- 3	- 7	- 11	+ 54	- 2
		W Walker R - upper Leavitt	0	+ 10	+ 2	+ 7	- 9	+ 57	+ 4
	Medium	Swauger Cr - lower Valdez	0	- 3	+ 3	0	- 8	+ 126	- 21
		Little Walker R - lower	- 3	- 3	+ 4	+ 40	- 8	+ 22	- 4
	Small	Cow Camp Cr - lower Schoettler	0	+ 7	+ 25	+ 13	+ 8	- 16	+ 4
		Cottonwood Cr - meadow	0	0	+ 21	- 3	+ 4	- 35	+ 28
		Mill Cr - central	- 33	- 3	- 38	- 3	- 5	- 19	- 5
		Swauger Cr - above East Fork	0	+ 10	+ 1	+ 13	+ 1	- 24	+ 2
	Number of sites exhibiting recovery response (n = 8) *		0	4	7	5	5	5	4
	Mean response *		- 1	+ 4	+ 7	+ 9	- 4	+ 26	0

* Mill Cr site excluded due to fire prior to 2002 sampling event.
Shading indicates site exhibited recovery response for given parameter.

Table 2: Summary of physical habitat recovery trends between 1999 and 2002.

Treatment Sites	Control Sites
• At least five of the nine sites showed a recovery response for each of the seven recovery parameters. • The mean response for each parameter was in the direction of recovery. • None of the sites showed a recovery response for all of the parameters. • None of the parameters improved at all of the sites.	• At least five of the eight[*] sites showed a recovery response for four of the seven recovery parameters. • The mean response for five of the parameters was in the direction of recovery. • None of the sites showed a recovery response for all of the parameters. • None of the parameters improved at all of the sites. • None of the sites showed an improvement in percent stable banks, with five showing no change.
Large sites (n=3): • All three large sites showed a recovery response with regard to undercut banks, fines + sand, D_{50} particle size, and cobble embeddedness. • Two sites showed in increase in stable banks. • One site showed an increase in vegetative bank cover. • No sites showed an increase in riparian cover.	**Large sites (n=3):** • All sites showed a recovery response for fines + sand and D_{50} particle size. • Two sites showed a recovery response for vegetative bank cover, riparian cover, undercut banks, and cobble embeddedness. • No sites showed an increase for stable banks.
Medium sites (n=2): • Two sites showed an increase in riparian cover and a decrease in cobble embeddedness. • Other parameters only improved at one or the other site over the period (i.e. not both).	**Medium sites (n=2):** • Both sites showed a recovery response for riparian cover, fines + sand, D_{50} particle size, and cobble embeddedness. • One site showed an increase in undercut banks. • No sites showed an increase in vegetative bank cover.
Small sites (n=4): • All sites showed an increase in vegetative bank cover. • Three sites showed a recovery response for bank stability, riparian cover, undercut banks, and D_{50} particle size. • Two sites showed a decrease in fines + sand • One site showed a decrease in embeddedness	**Small sites (n=3[*]):** • All sites showed a recovery response for riparian cover. • Two sites showed a recovery response for vegetative bank cover and undercut banks. • One site showed a recovery response for fines + sand and cobble embeddedness. • No sites showed an increase in D_{50} particle size

* The Mill Creek site burned prior to sampling in 2002 and is not included in this analysis.

Figure 1a: Percent Riparian Cover
Large Sites - 1999 and 2002

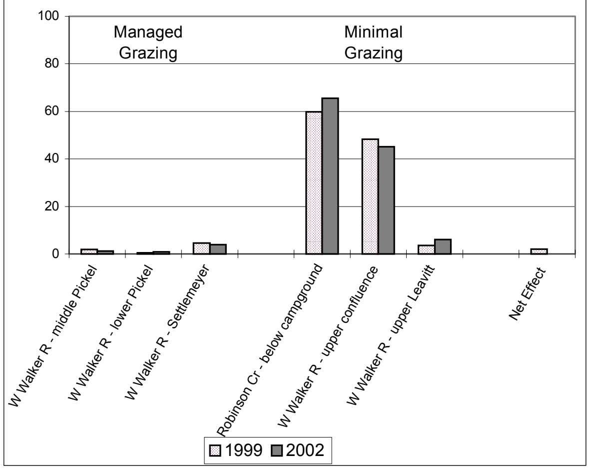


Figure 2a: Percent Stable Banks
Large Sites - 1999 and 2002

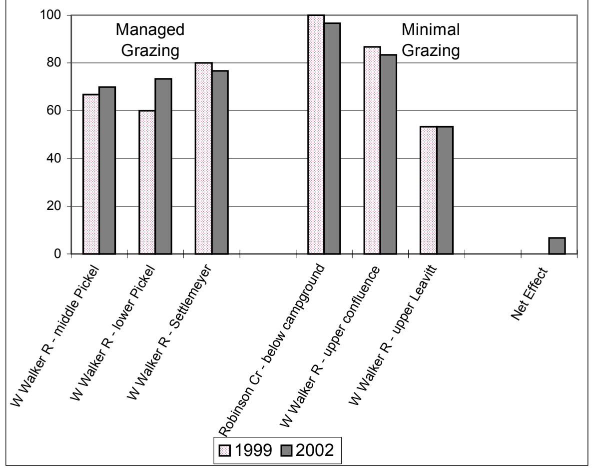


Figure 1b: Percent Riparian Cover
Medium Sites - 1999 and 2002

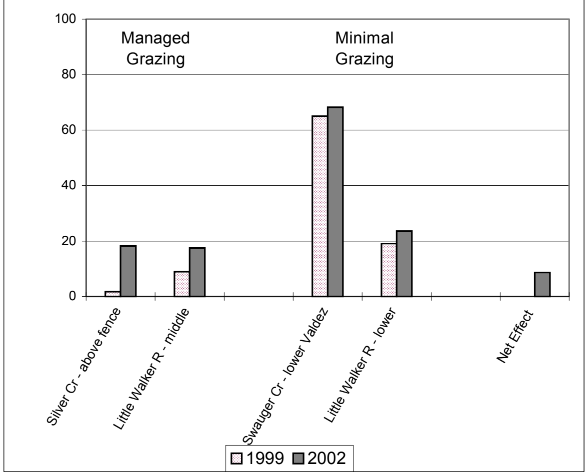


Figure 2b: Percent Stable Banks
Medium Sites - 1999 and 2002

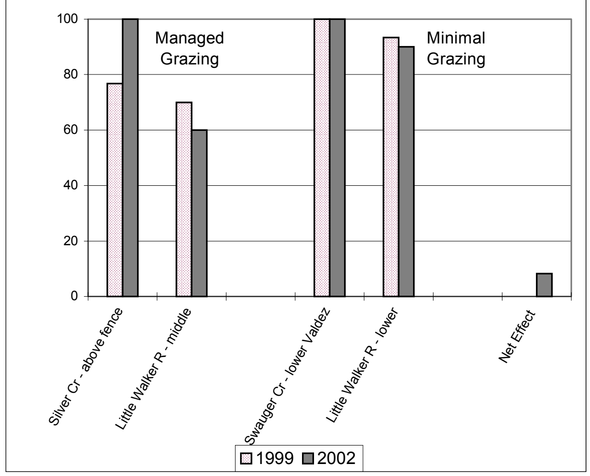


Figure 1c: Percent Riparian Cover
Small Sites - 1999 and 2002

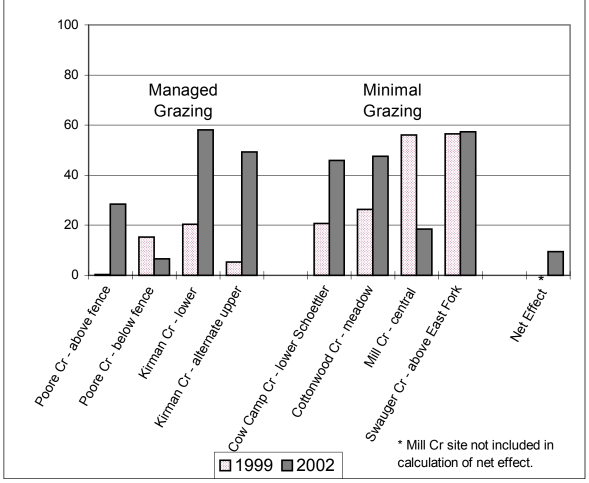


Figure 2c: Percent Stable Banks
Small Sites - 1999 and 2002

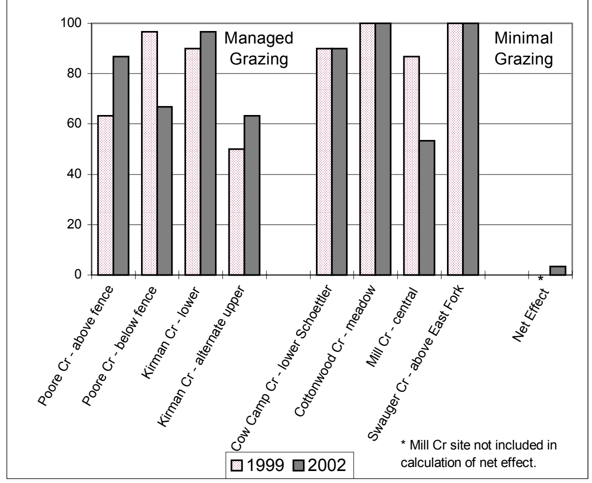


Figure 3a: Percent Undercut Banks
Large Sites - 1999 and 2002

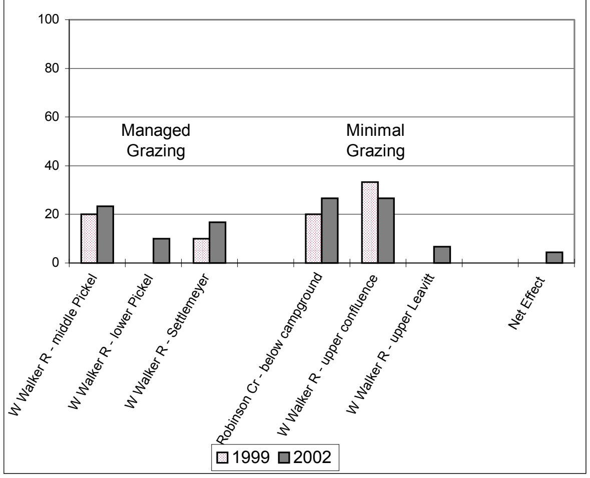


Figure 4a: Percent Vegetated Bank Cover
Large Sites - 1999 and 2002

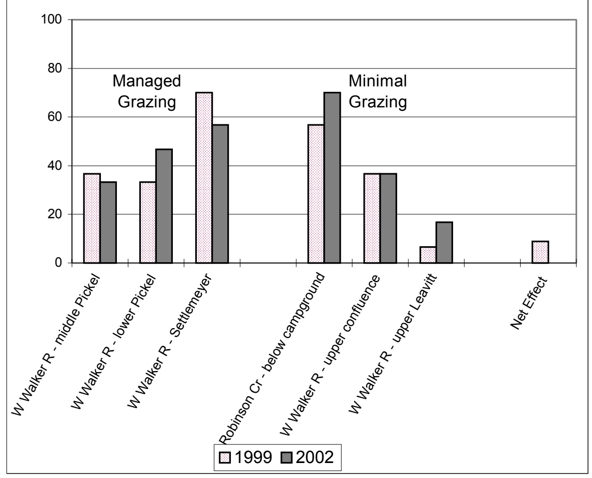


Figure 3b: Percent Undercut Banks
Medium Sites - 1999 and 2002

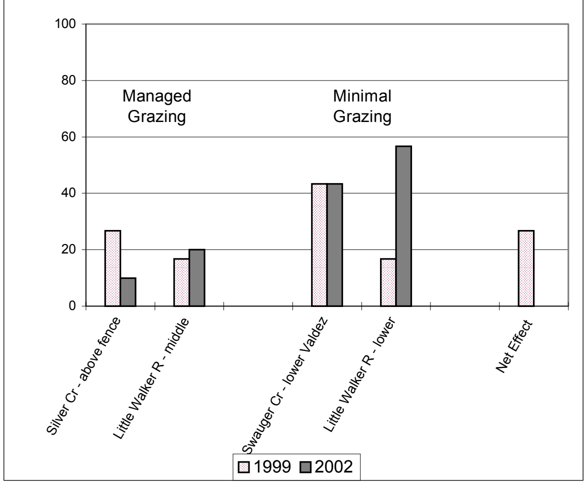


Figure 4b: Percent Vegetated Bank Cover
Medium Sites - 1999 and 2002

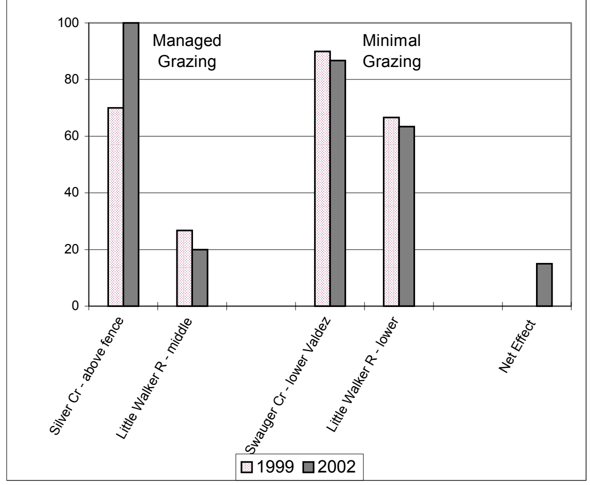


Figure 3c: Percent Undercut Banks
Small Sites - 1999 and 2002

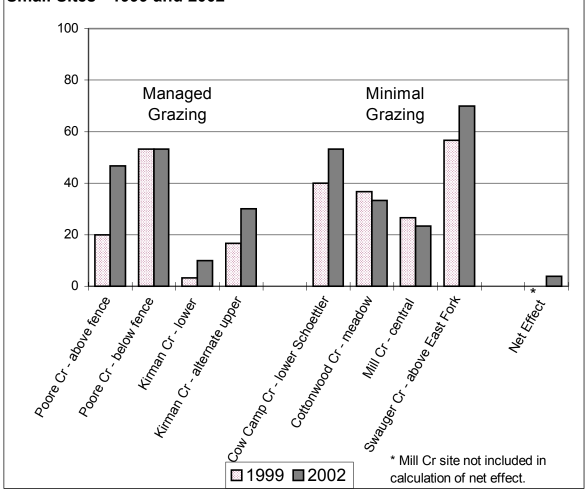


Figure 4c: Percent Vegetated Banks
Small Sites - 1999 and 2002

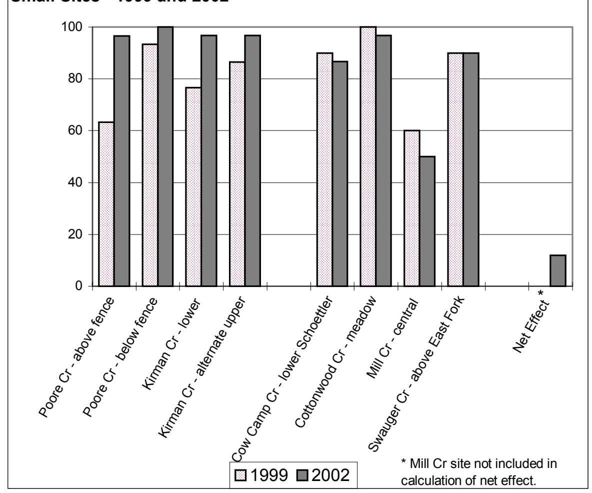


Figure 5a: Percent Fines + Sand
Large Sites - 1999 and 2002

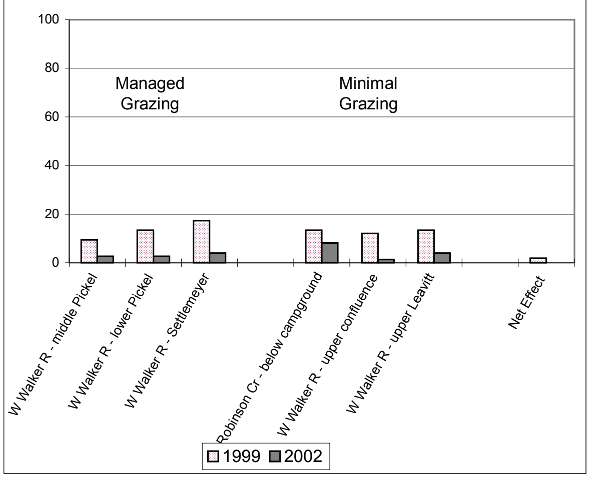


Figure 6a: D_{50} Particle Size (mm)
Large Sites - 1999 and 2002

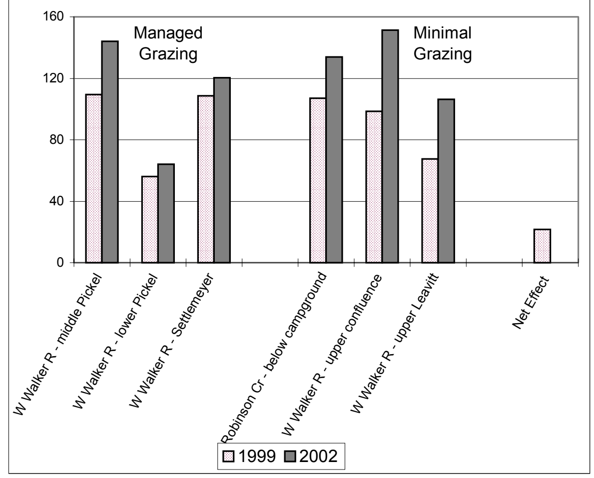


Figure 5b: Percent Fines + Sand
Medium Sites - 1999 and 2002

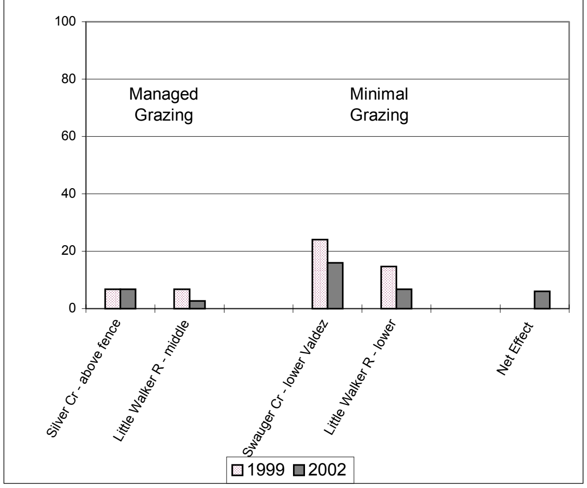


Figure 6b: D_{50} Particle Size (mm)
Medium Sites - 1999 and 2002

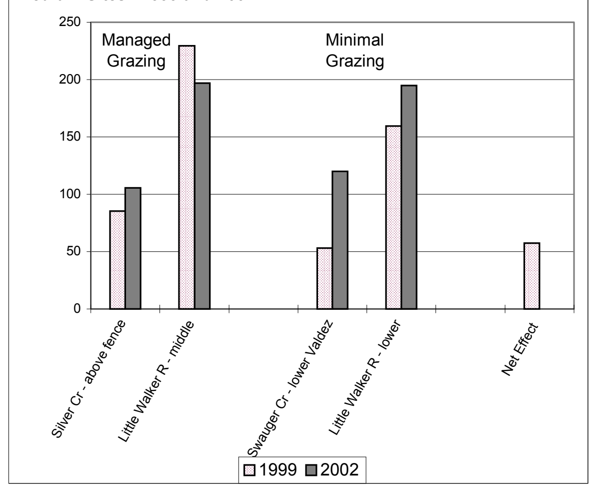
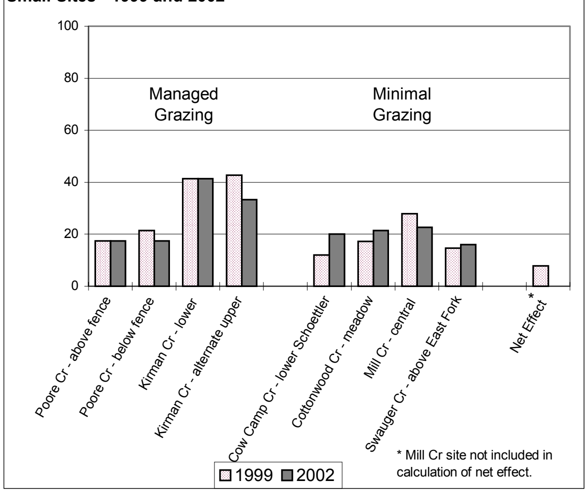
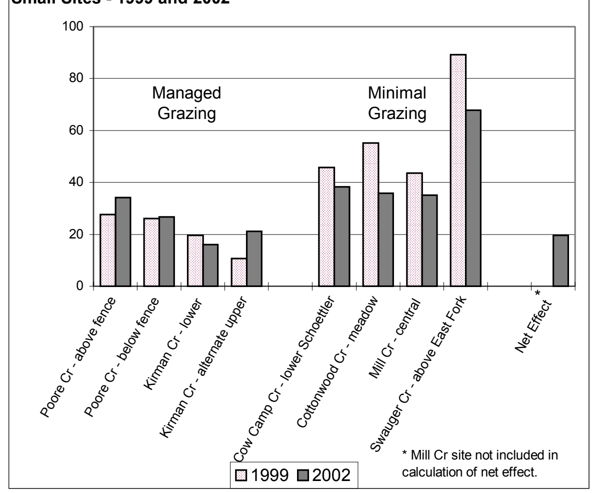


Figure 5c: Percent Fines + Sand
Small Sites - 1999 and 2002



* Mill Cr site not included in calculation of net effect.

Figure 6c: D_{50} Particle Size (mm)
Small Sites - 1999 and 2002



* Mill Cr site not included in calculation of net effect.

Figure 7a: Percent Cobble Embeddedness
Large Sites - 1999 and 2002

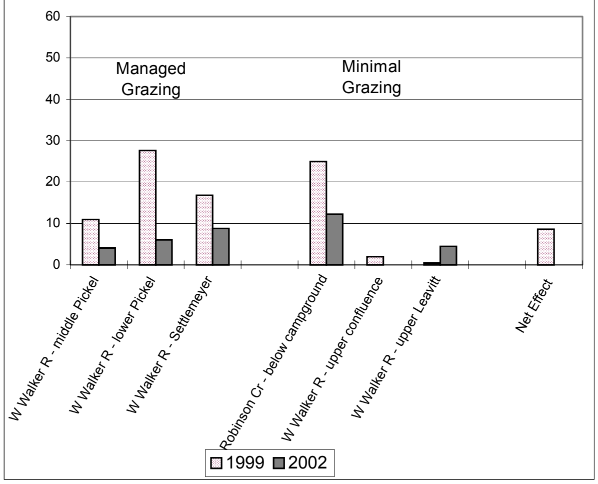


Figure 7b: Percent Cobble Embeddedness
Medium Sites - 1999 and 2002

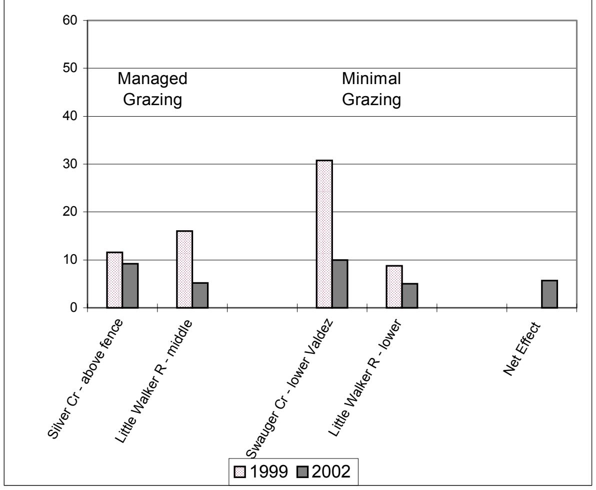
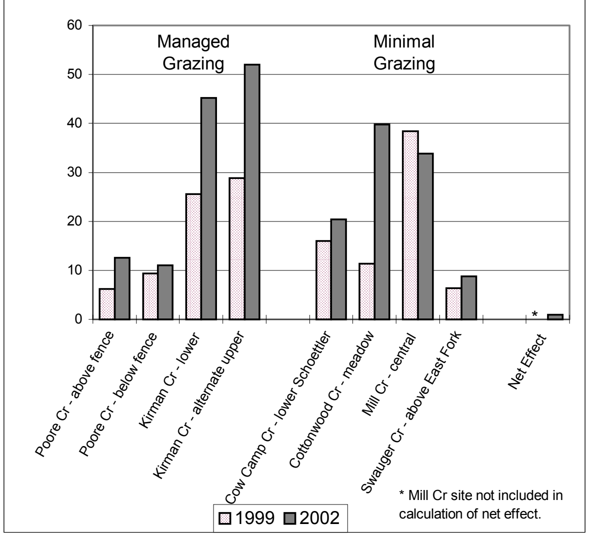


Figure 7c: Percent Embeddedness
Small Sites - 1999 and 2002



* Mill Cr site not included in calculation of net effect.

Figure 8a: Mean Channel Width (cm) - 1999 and 2002
Managed Grazing Sites

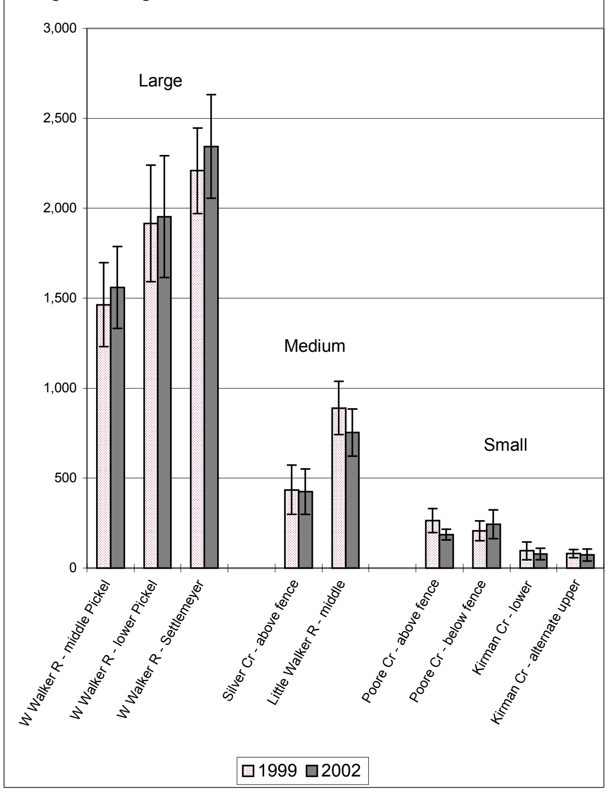


Figure 8b: Mean Channel Width (cm) - 1999 and 2002
Minimal Grazing Sites

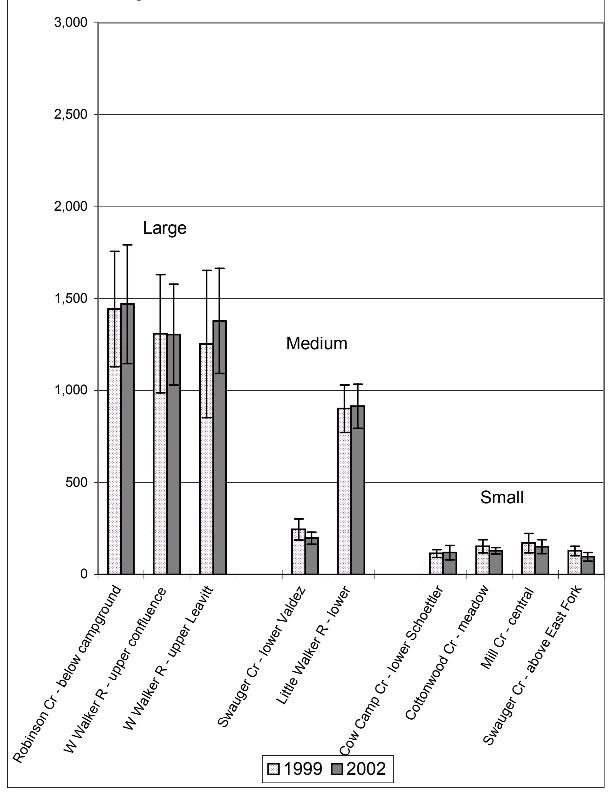


Figure 9a: Mean Channel depth (cm) - 1999 and 2002
Managed Grazing Sites

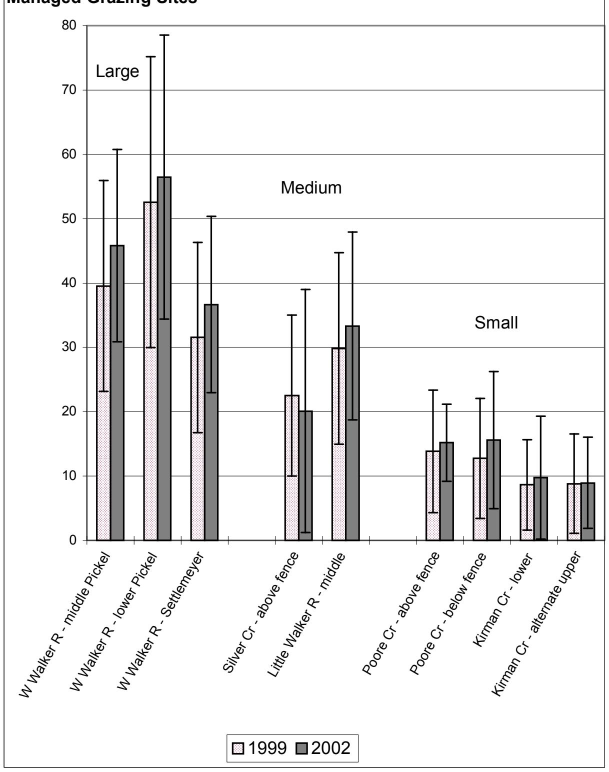
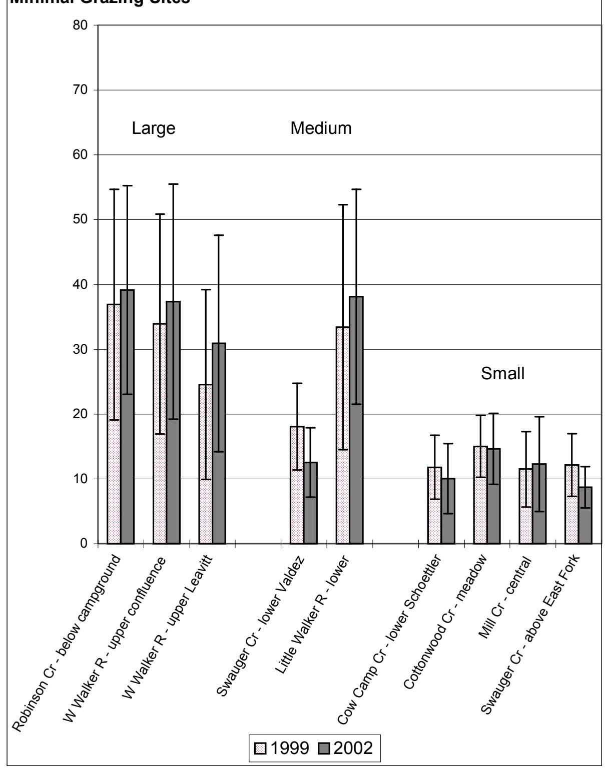


Figure 9b: Mean Channel depth (cm) - 1999 and 2002
Minimal Grazing Sites



Error bars represent one standard deviation.

Figure 10a: Mean Velocity (cm s^{-1}) - 1999 and 2002
Managed Grazing Sites

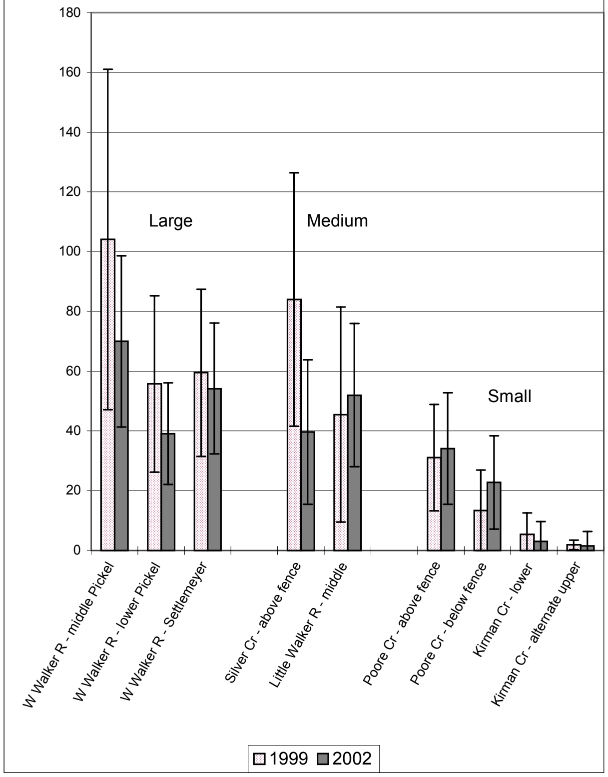


Figure 10b: Mean Velocity (cm s^{-1}) - 1999 and 2002
Minimal Grazing Sites

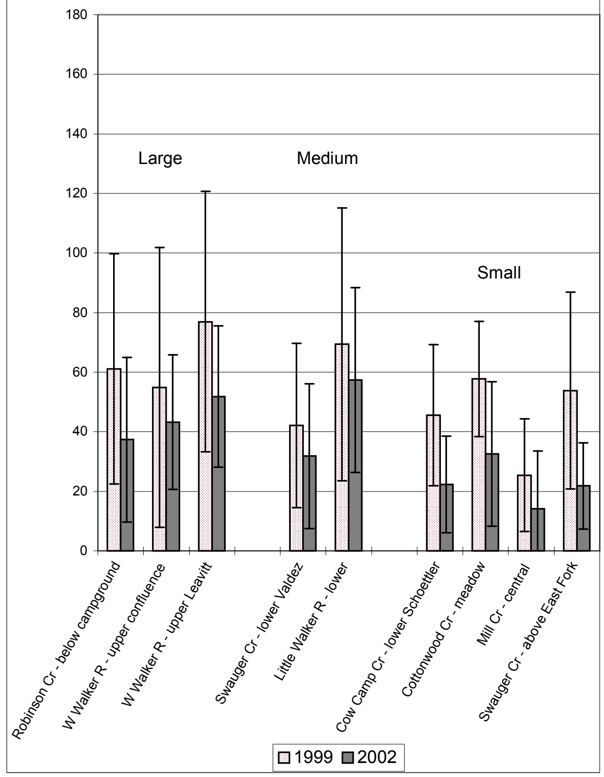


Figure 11a: Mean Discharge (m^3 s^{-1}) - 1999 and 2002
Managed Grazing Sites

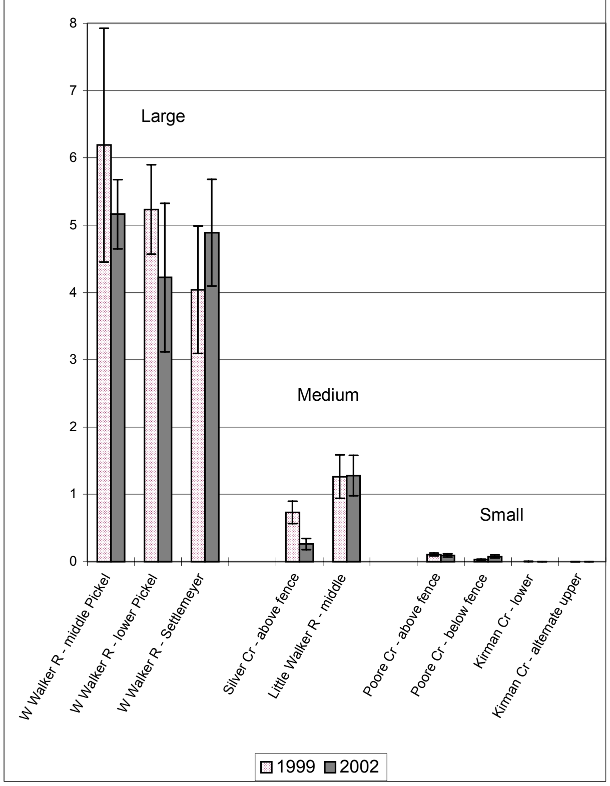
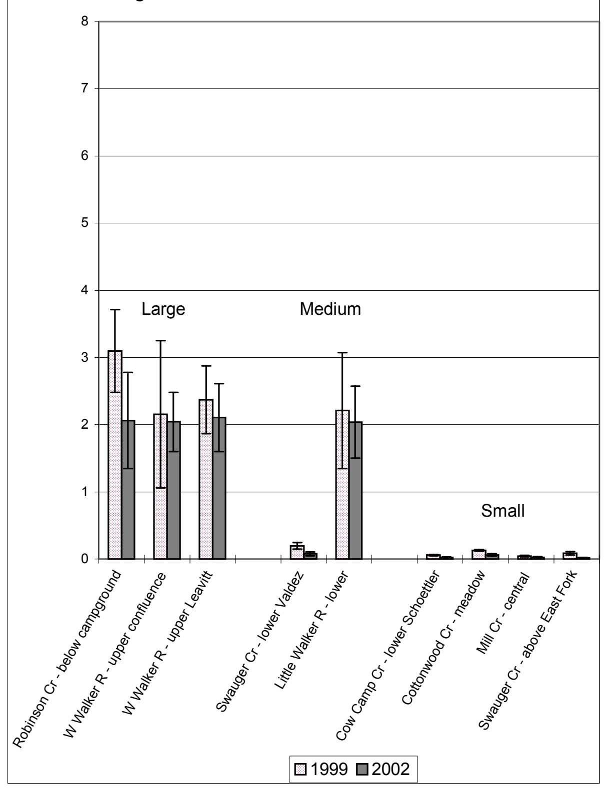


Figure 11b: Mean Discharge (m^3 s^{-1}) - 1999 and 2002
Minimal Grazing Sites



Error bars represent one standard deviation.

Figure 12a: Mean FPOM (g) - 1999 and 2002
Managed Grazing Sites

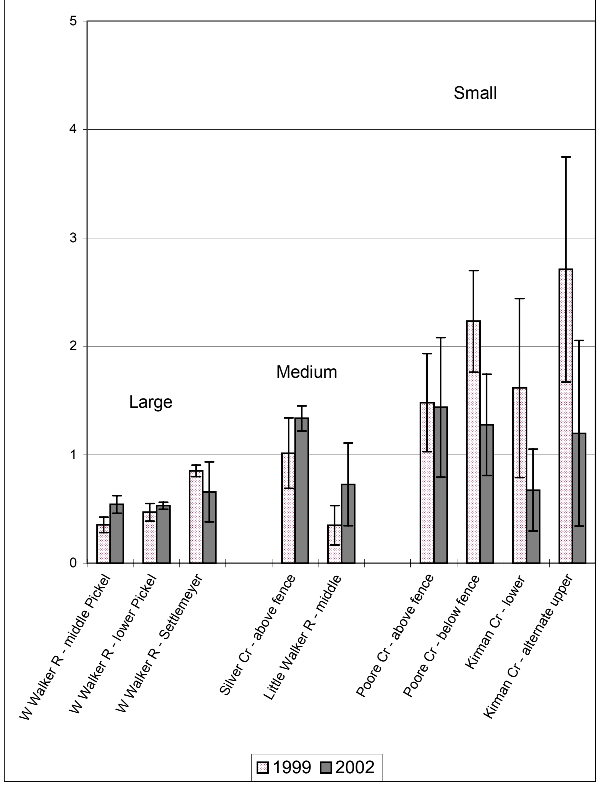


Figure 12b: Mean FPOM (g) - 1999 and 2002
Minimal Grazing Sites

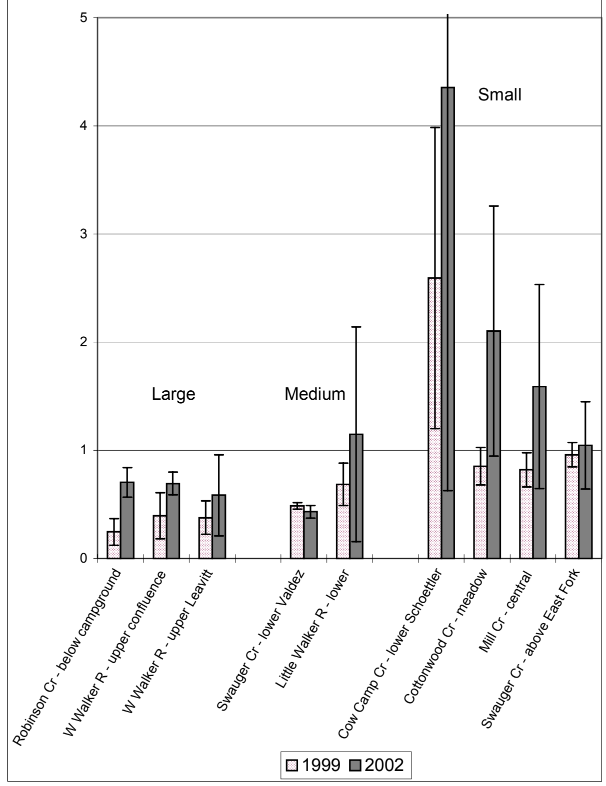


Figure 13a: Mean CPOM (g) - 1999 and 2002
Managed Grazing Sites

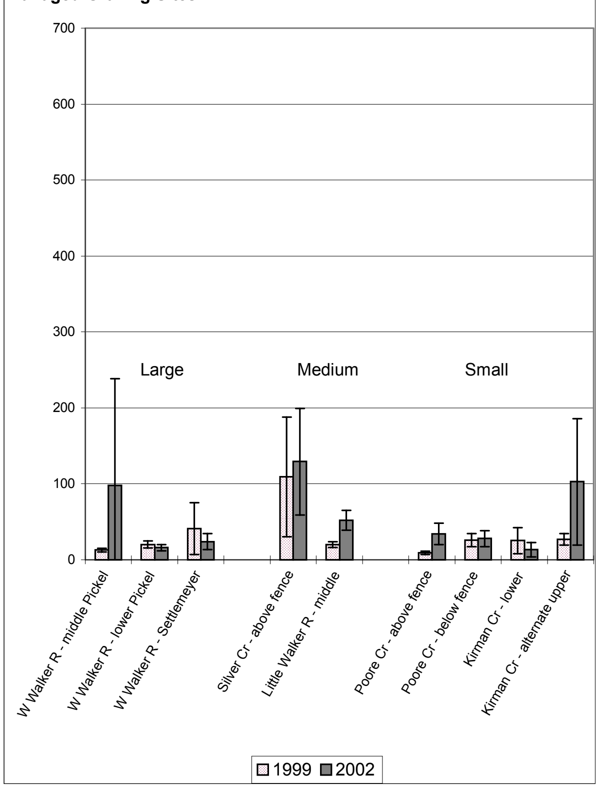
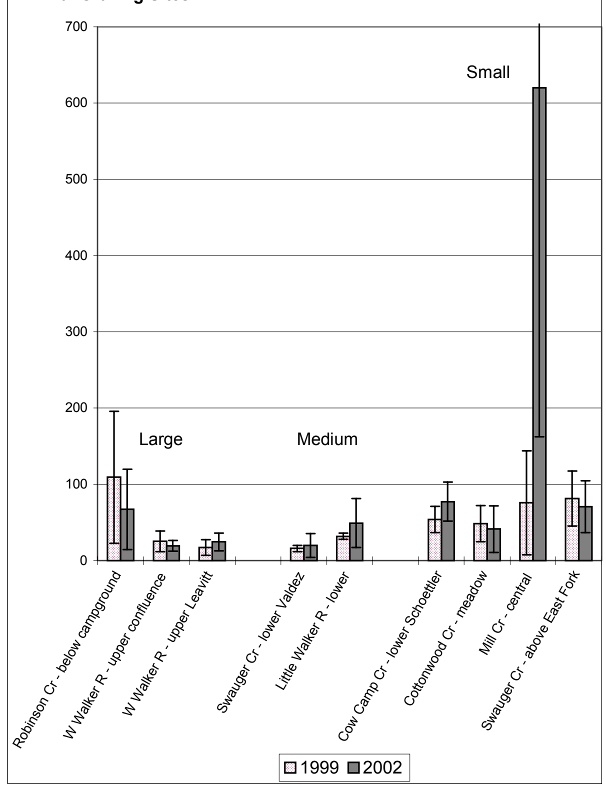


Figure 13b: Mean CPOM (g) - 1999 and 2002
Minimal Grazing Sites



Error bars represent one standard deviation for three replicate samples at each site.

Figure 14a: Mean Chlorophyll *a* (ug cm^{-2}) - 1999 and 2002
Managed Grazing Sites

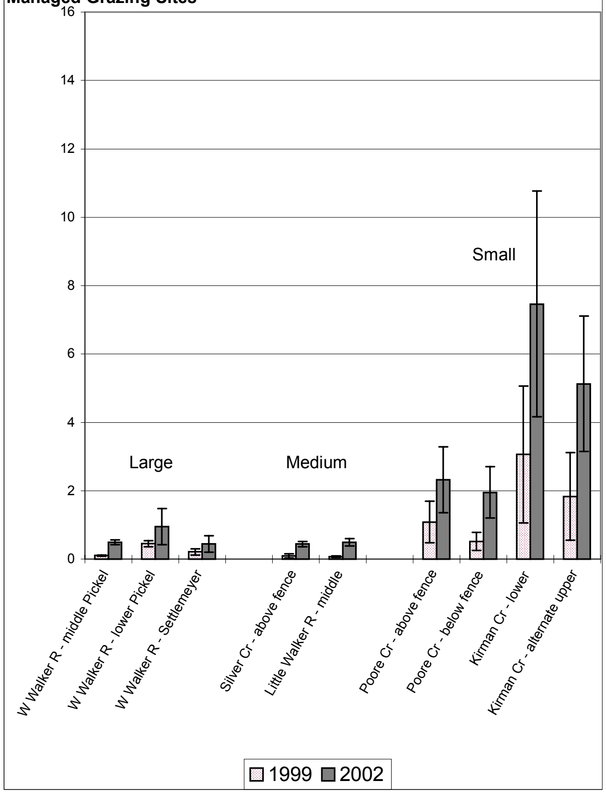
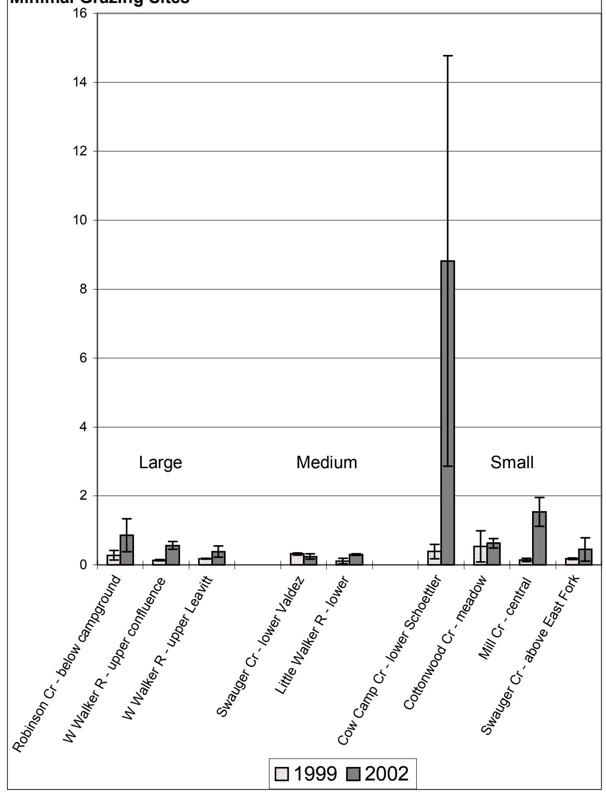


Figure 14b: Mean Chlorophyll *a* (ug cm^{-2})- 1999 and 2002
Minimal Grazing Sites



Error bars represent one standard deviation for three replicate samples at each site.

Figure 15a: Percent Substrate Modifiers - 1999 and 2002
Managed Grazing Sites

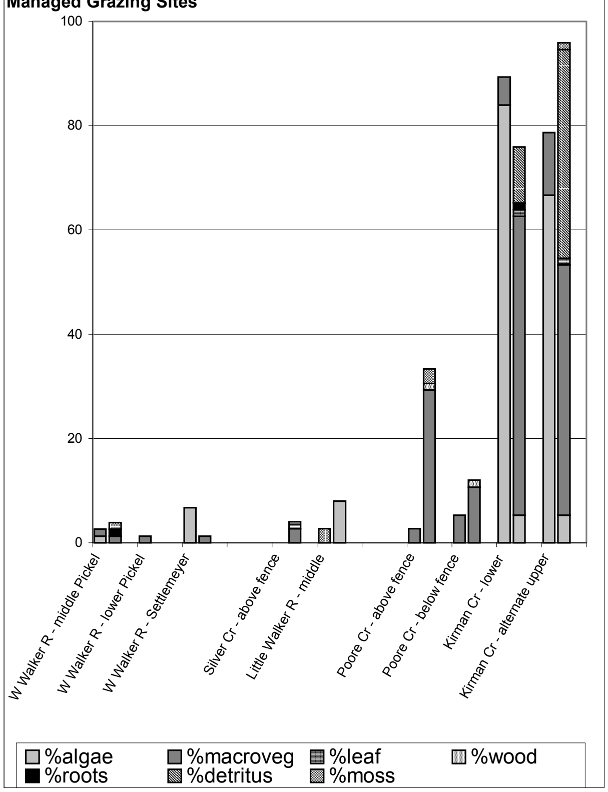


Figure 15b: Percent Substrate Modifiers - 1999 and 2002
Minimal Grazing Sites

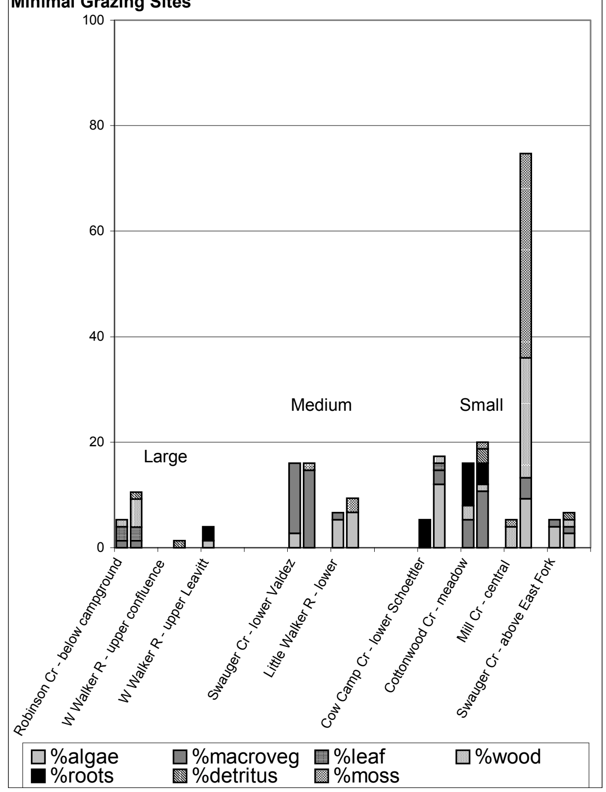


Figure 16a: Herbaceous and Woody Cover-1999 and 2002
Managed Grazing Sites

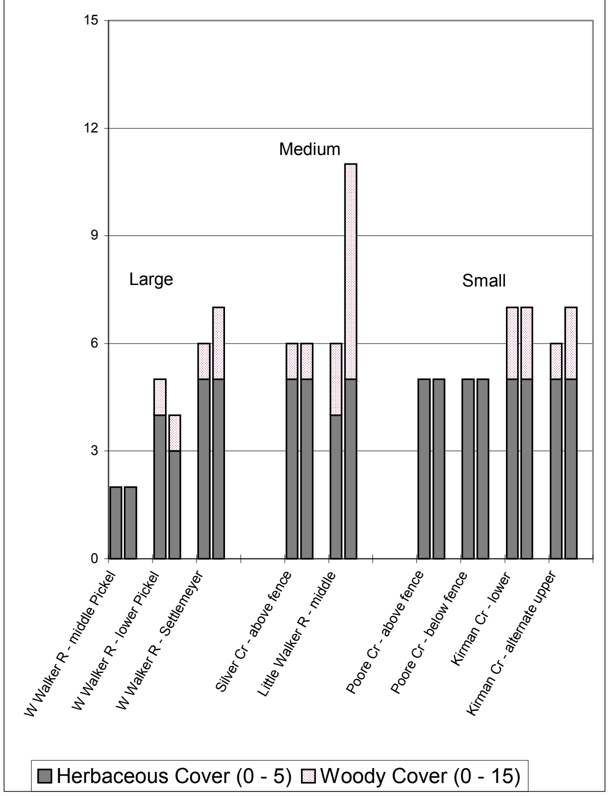


Figure 16b: Herbaceous and Woody Cover-1999 and 2002
Minimal Grazing Sites

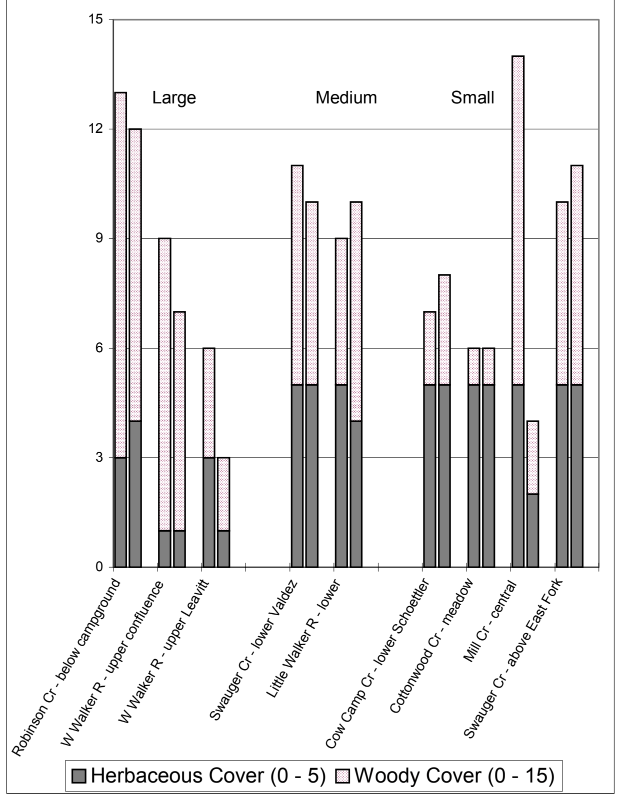


Figure 17a: Percent Bank Cover - 1999 and 2002
Managed Grazing Sites

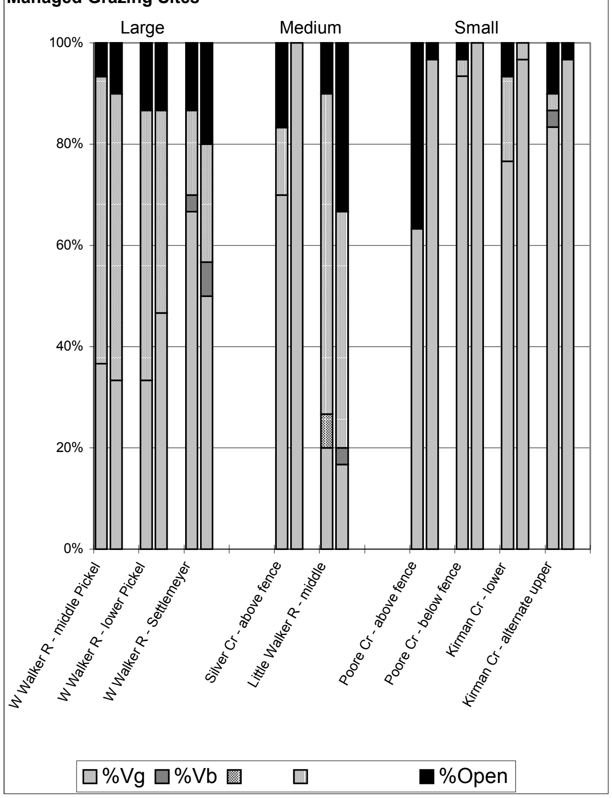


Figure 17b: Percent Bank Cover - 1999 and 2002
Minimal Grazing Sites

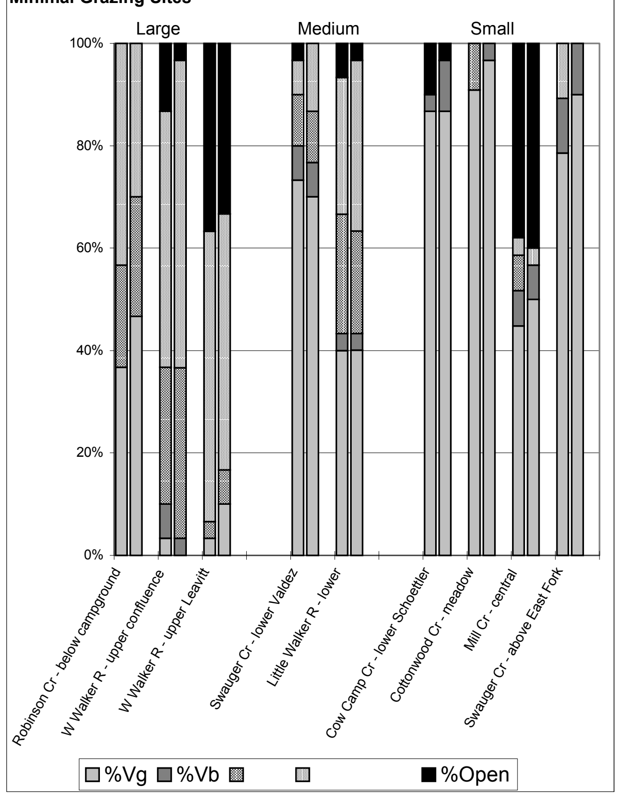


Figure 18a: pH - 1999 and 2002
Managed Grazing Sites

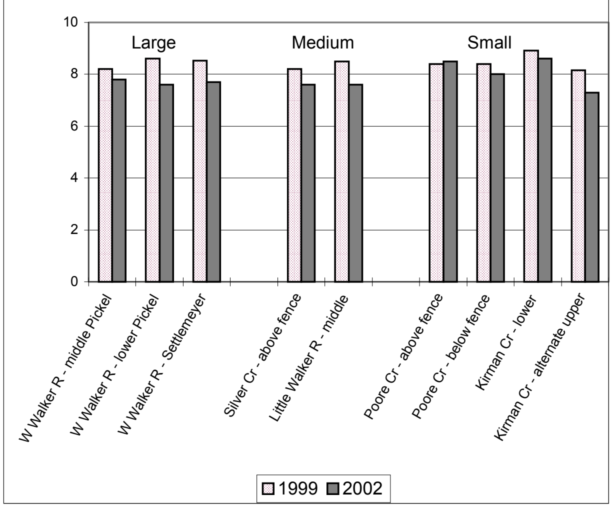


Figure 18b: pH - 1999 and 2002
Minimal Grazing Sites

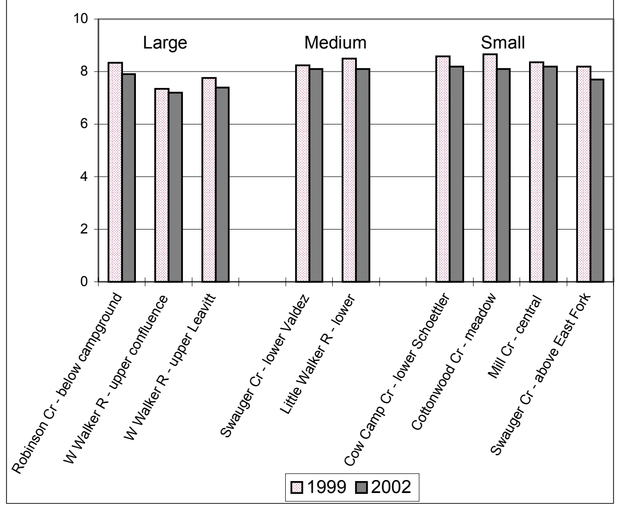


Figure 19a: Alkalinity (mg l^{-1}) - 1999 and 2002
Managed Grazing Sites

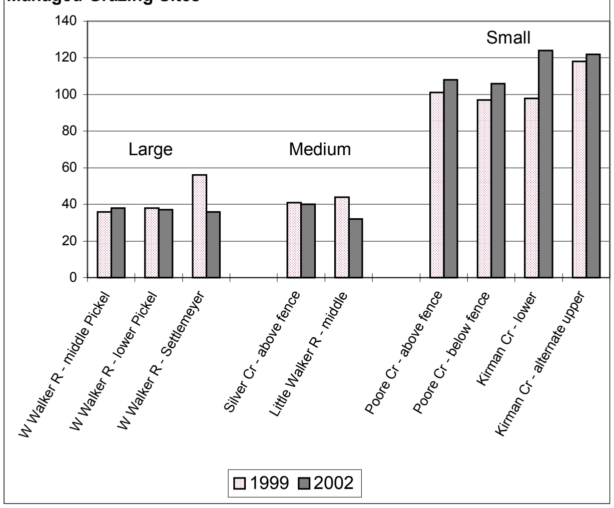


Figure 19b: Alkalinity (mg l^{-1}) - 1999 and 2002
Minimal Grazing Sites

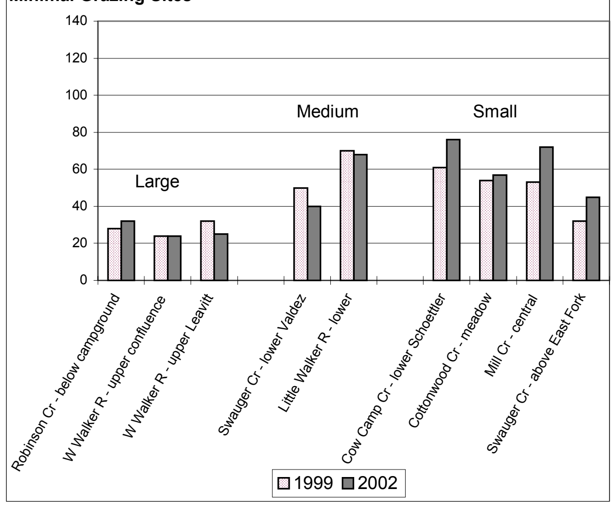


Figure 20a: Hardness - 1999 and 2002
Managed Grazing Sites

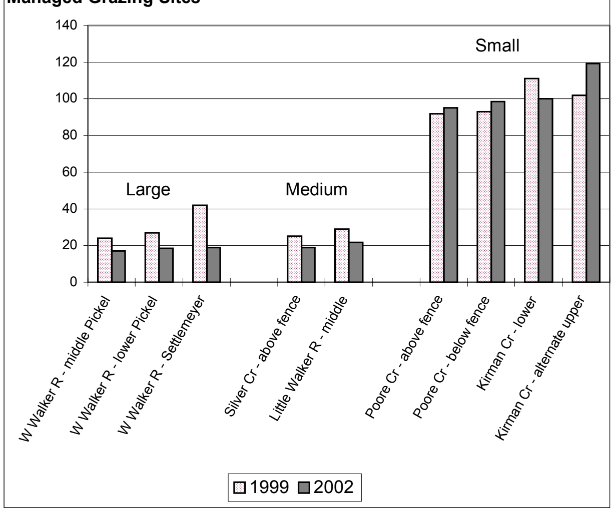


Figure 20b: Hardness - 1999 and 2002
Minimal Grazing Sites

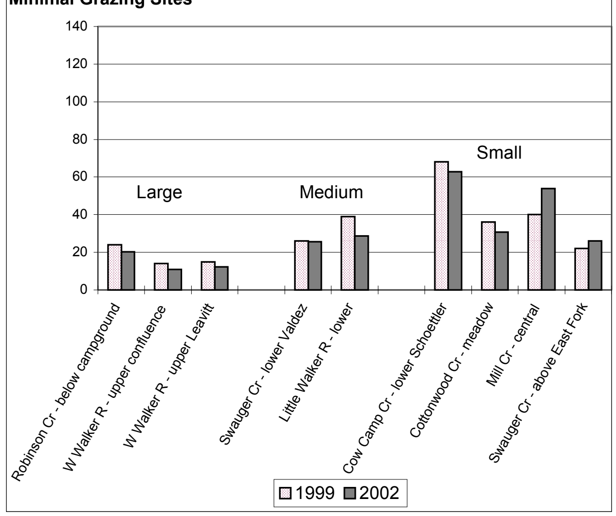


Figure 21a: DO Concentration (mg l^{-1}) - 1999 and 2002
Managed Grazing Sites

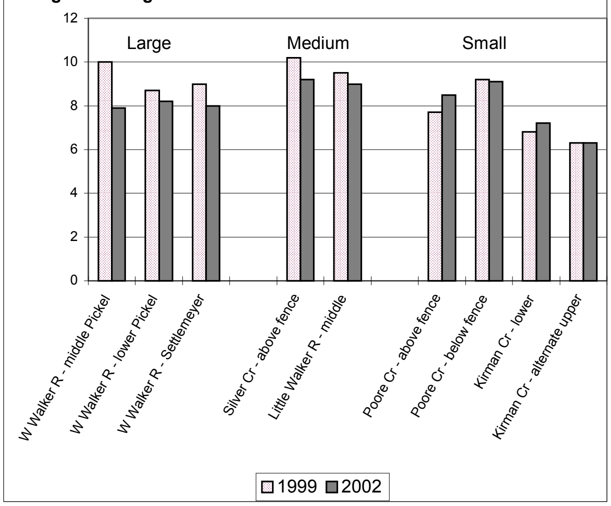


Figure 21b: DO Concentration (mg l^{-1}) - 1999 and 2002
Minimal Grazing Sites

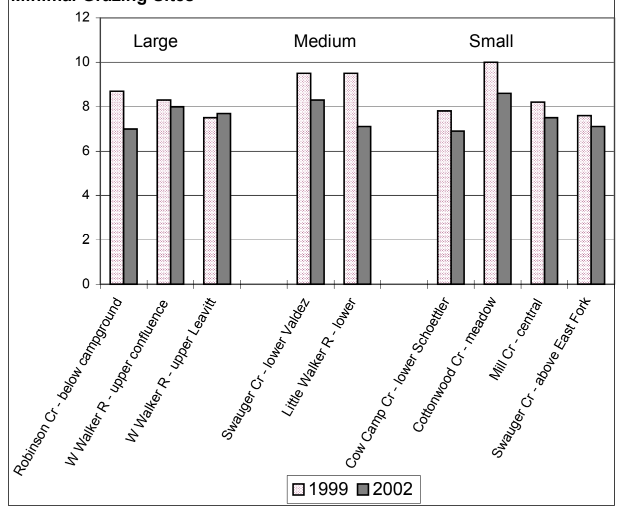


Figure 22a: Conductivity (uS) - 1999 and 2002
Managed Grazing Sites

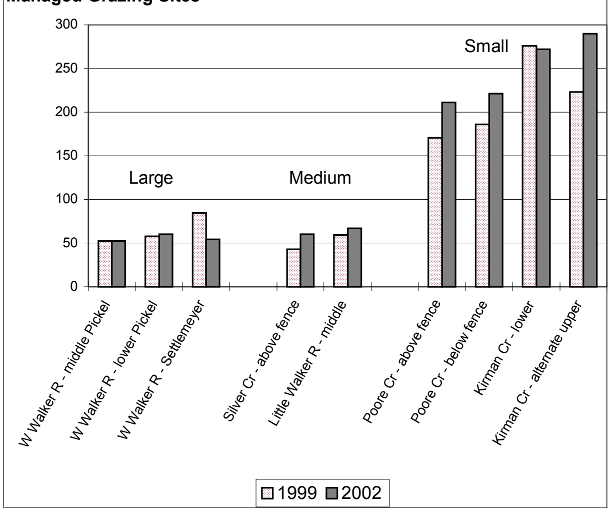


Figure 22b: Conductivity (uS) - 1999 and 2002
Minimal Grazing Sites

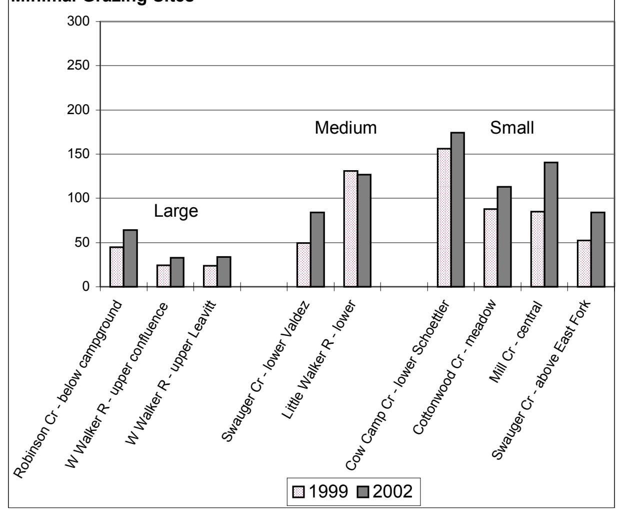


Figure 23a: Turbidity (ntu) - 1999 and 2002
Managed Grazing Sites

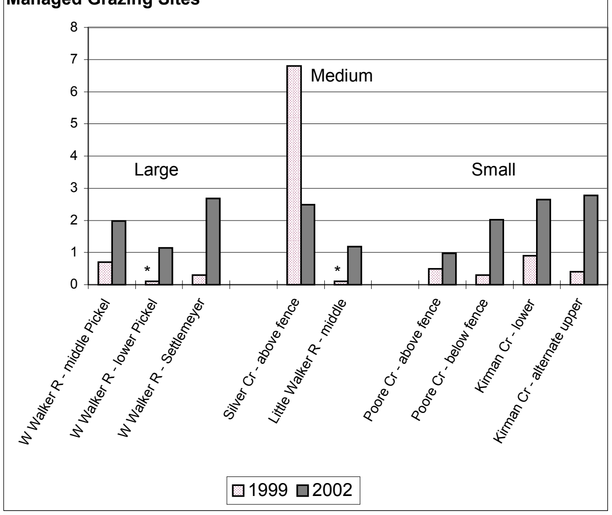
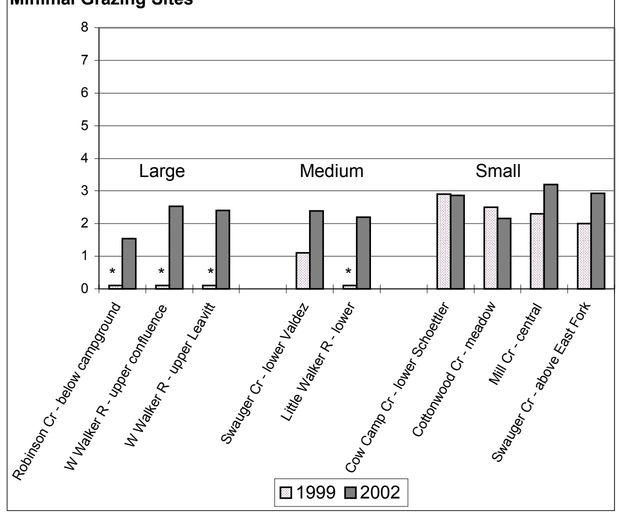


Figure 23b: Turbidity (ntu) - 1999 and 2002
Minimal Grazing Sites



* Measurement below laboratory reporting limit (0.2 ntu); one-half this value is plotted.

Figure 24a: Total Phosphorus (mg l^{-1}) - 2002
Managed Grazing Sites

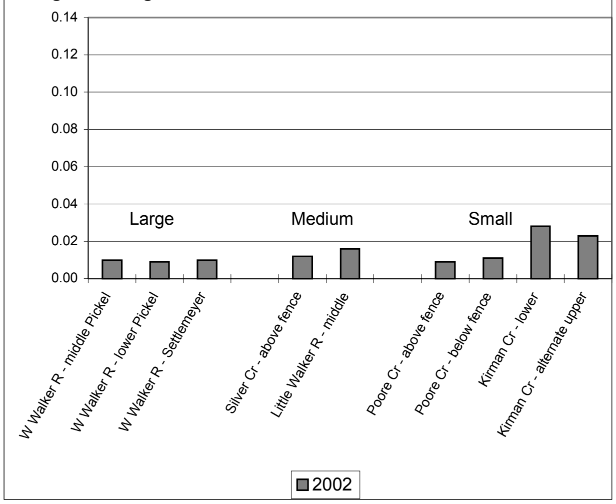


Figure 24b: Total Phosporus (mg l^{-1}) - 2002
Minimal Grazing Sites

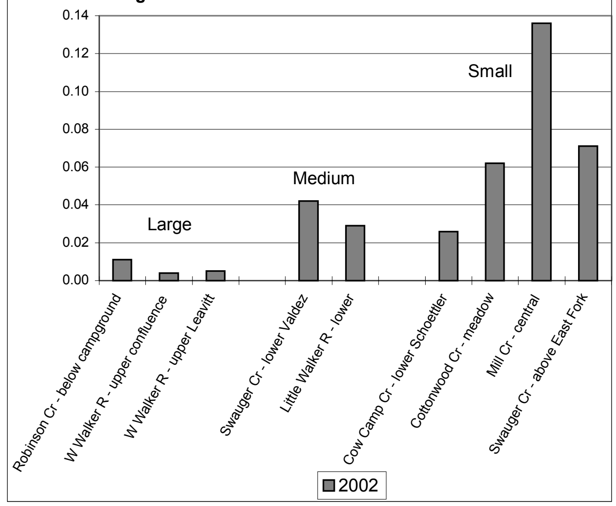


Figure 25a: Silica (mg l^{-1}) - 2002
Managed Grazing Sites

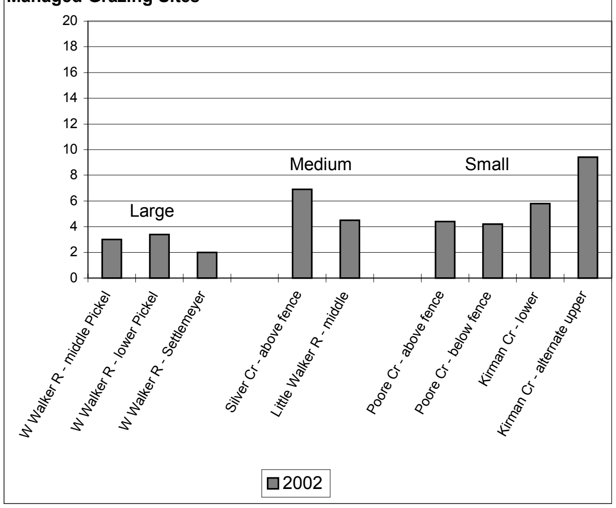


Figure 25b: Silica (mg l^{-1}) - 2002
Minimal Grazing Sites

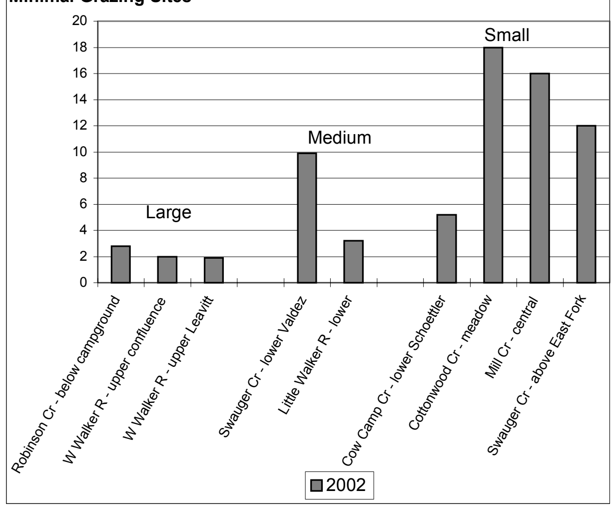


Figure 26a: Sulfate (mg l^{-1}) - 1999 and 2002
Managed Grazing Sites

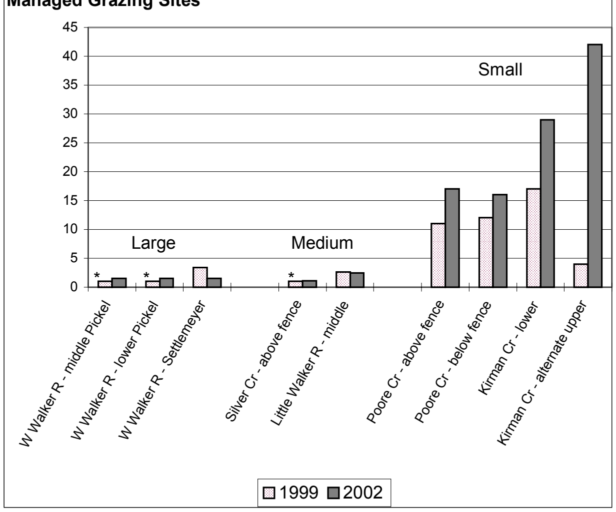
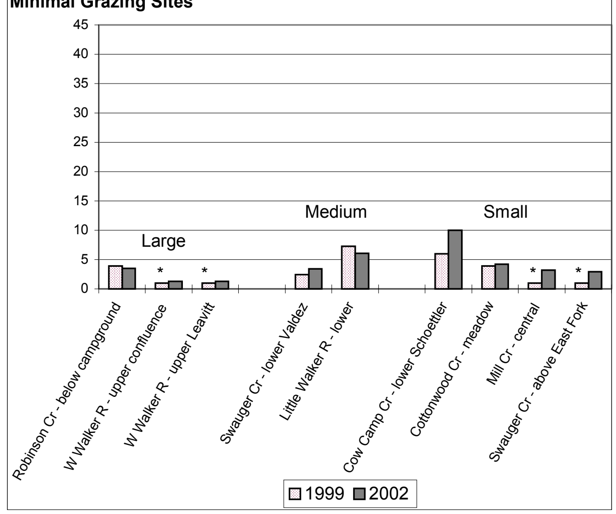


Figure 26b: Sulfate (mg l^{-1}) - 1999 and 2002
Minimal Grazing Sites



* Measurement below laboratory reporting limit (2 mg l^{-1}); one-half this value is plotted.

Figure 27a: Nitrate (mg l^{-1}) - 2002
Managed Grazing Sites

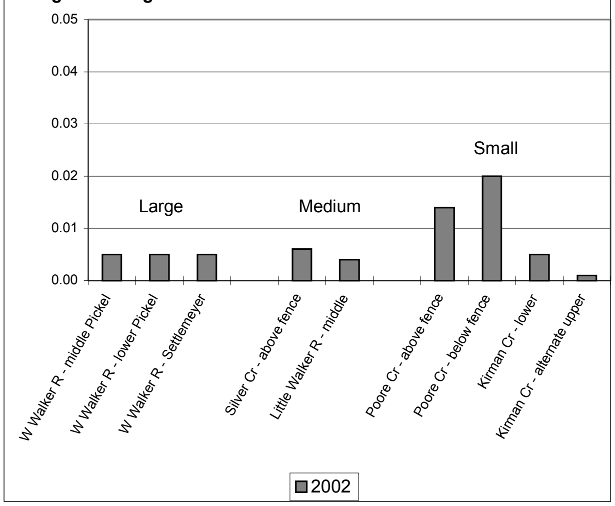


Figure 27b: Nitrate (mg l^{-1}) - 2002
Minimal Grazing Sites

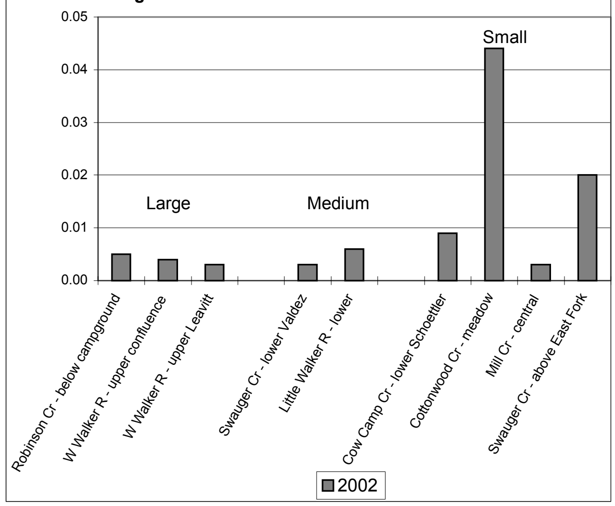


Figure 28a: Total Kjeldahl Nitrogen (mg l^{-1}) - 2002
Managed Grazing Sites

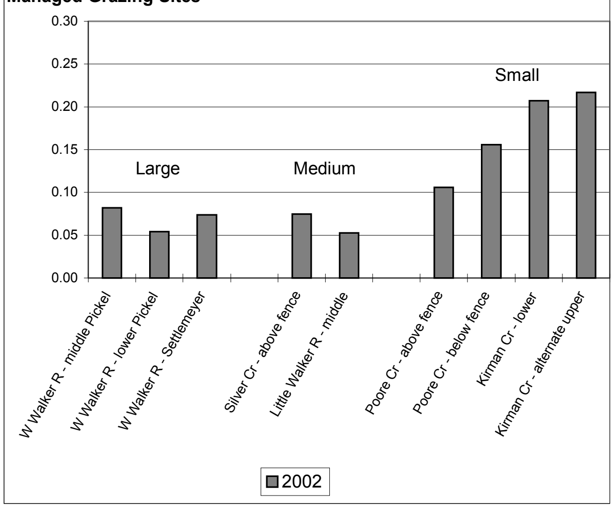


Figure 28b: Total Kjeldahl Nitrogen (mg l^{-1}) - 2002
Minimal Grazing Sites

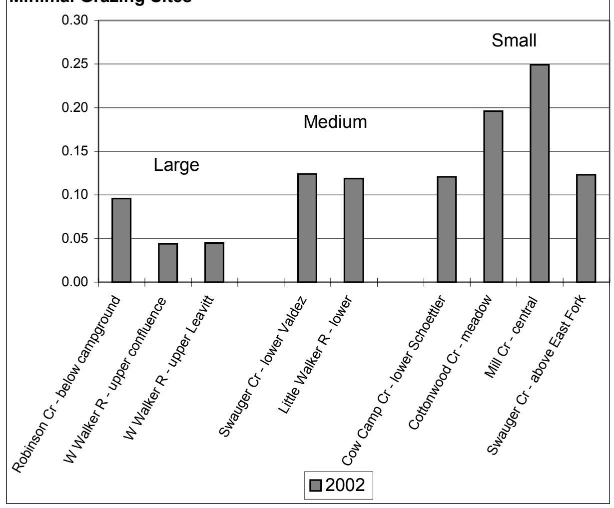
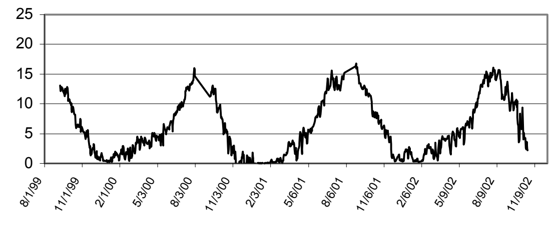


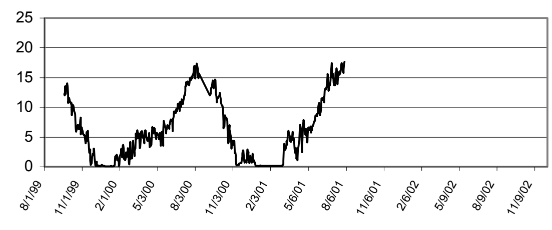
Figure 29: Mean daily water temperature (°C) - 1999 to 2002

Large Managed Grazing Sites

a: W Walker R - middle Pickel

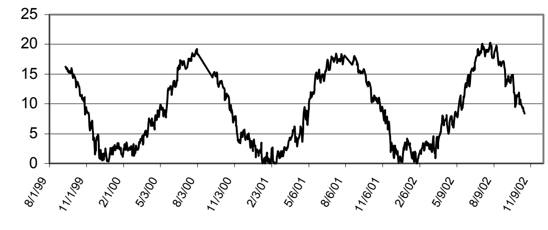


b: W Walker R - Settlemeyer

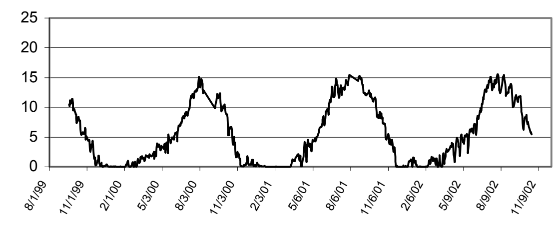


Large Minimal Grazing Sites

f: Robinson Cr - below campground

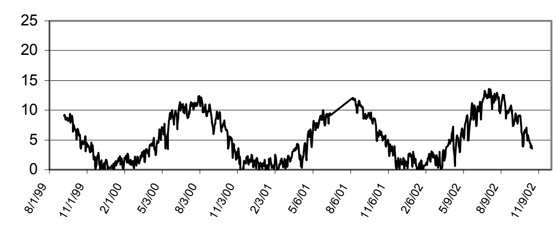


g: W Walker R - upper Leavitt



Medium Minimal Grazing Sites

h: Swauger Cr - lower Valdez



i: Little Walker R - lower

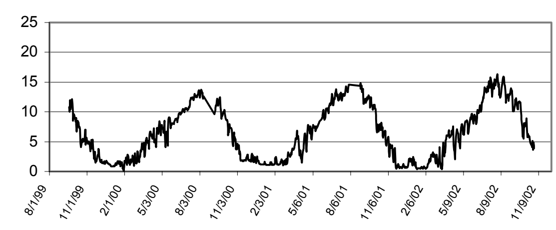
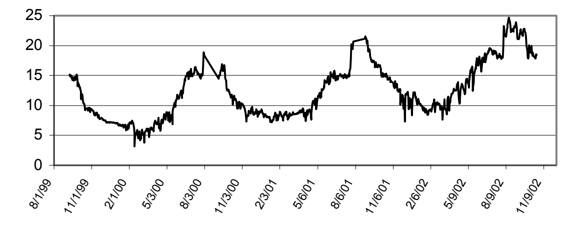


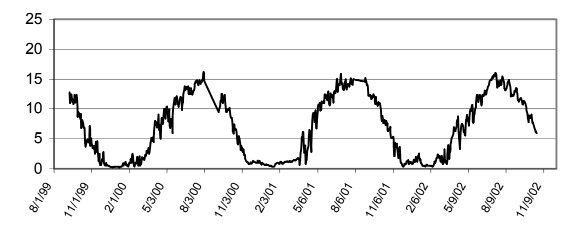
Figure 29 (continued): Mean daily water temperature (^oC) - 1999 to 2002

Small Managed Grazing Sites

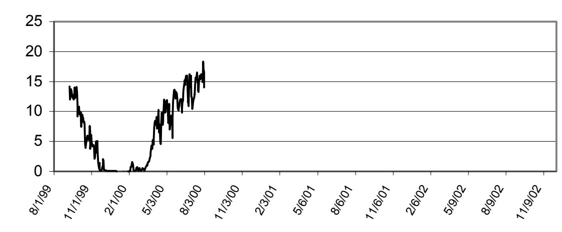
c: Poore Cr - above fence



d: Kirman Cr - lower

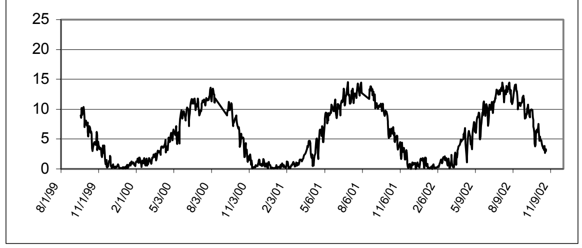


e: Kirman Cr - alternate upper

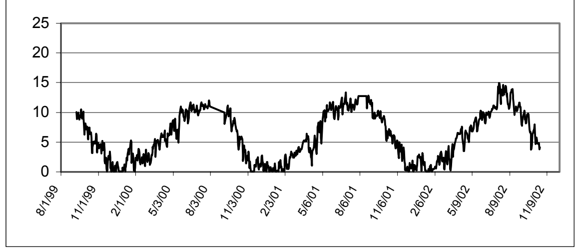


Small Minimal Grazing Sites

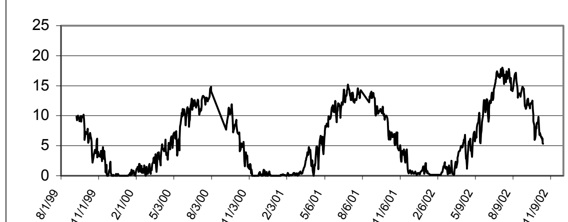
j: Cow Camp Cr - lower Schoettler



k: Cottonwood Cr - meadow

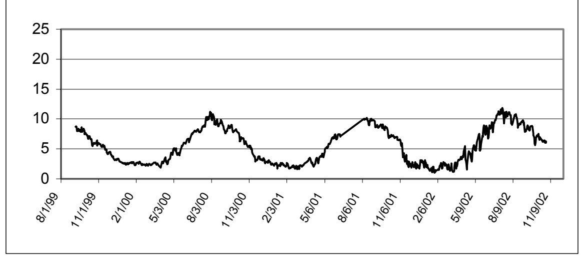


l: Mill Cr - central *



* This site burned in early July 2002.

m: Swauger Cr - above East Fork



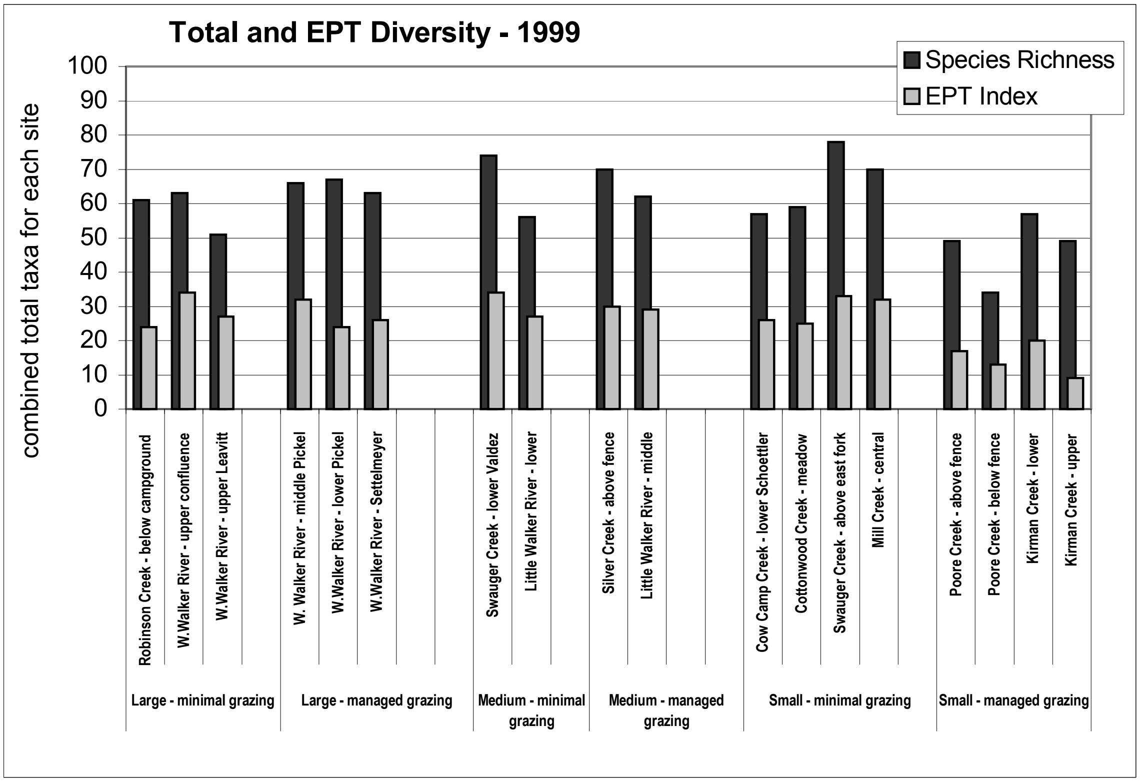


Figure 30a. Composite total diversity and EPT diversity in 1999 (taxa from all 5 replicate samples taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

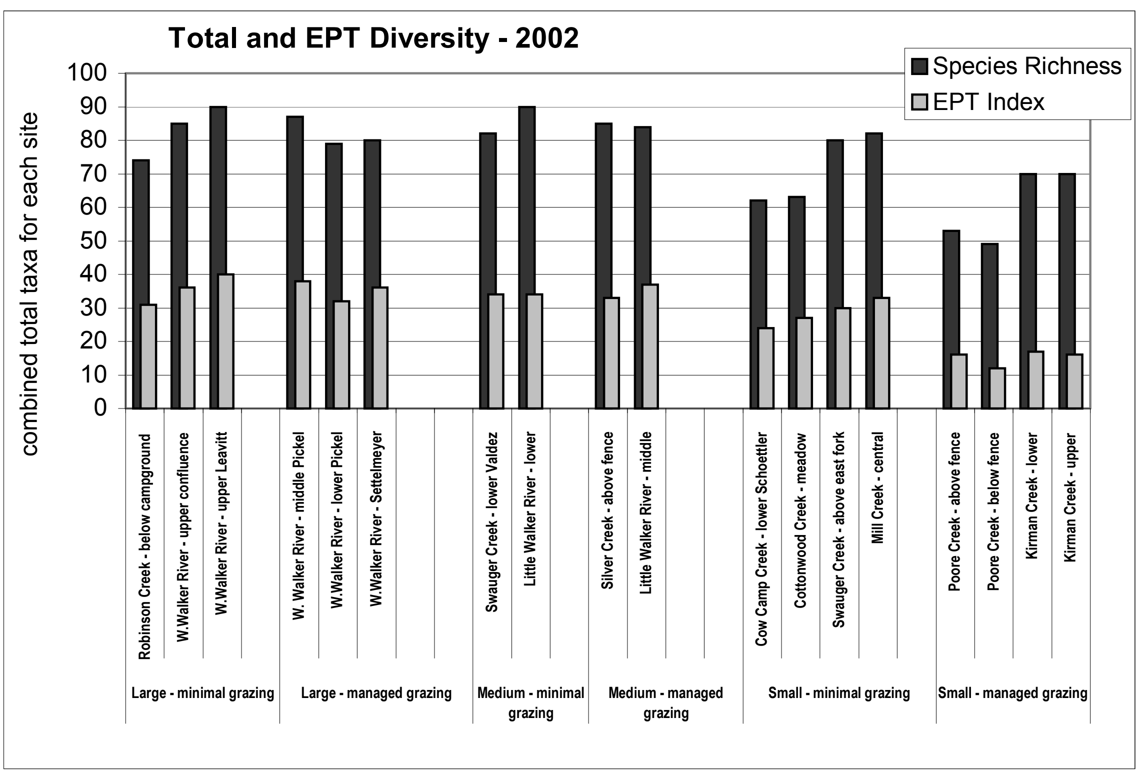


Figure 30b. Composite total diversity and EPT diversity in 2002 (taxa from all 5 replicate samples taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

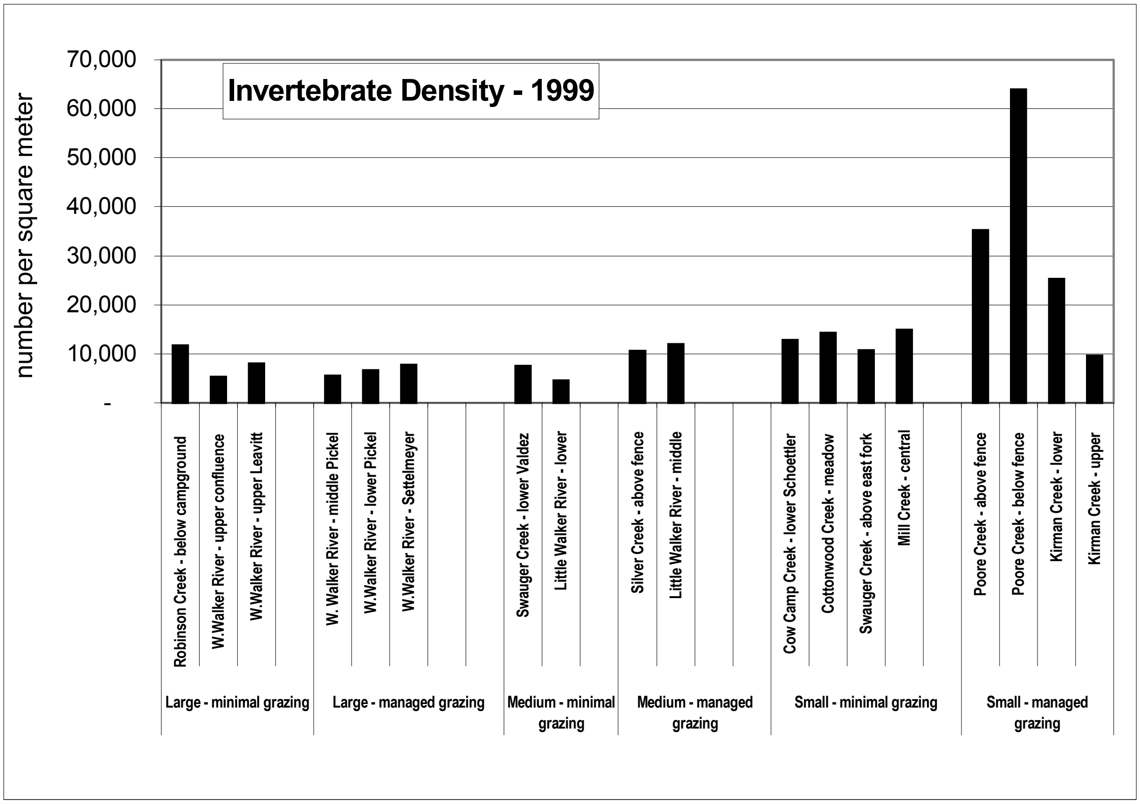


Figure 31a. Average invertebrate density in 1999 (coefficient of variation averages 29%) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

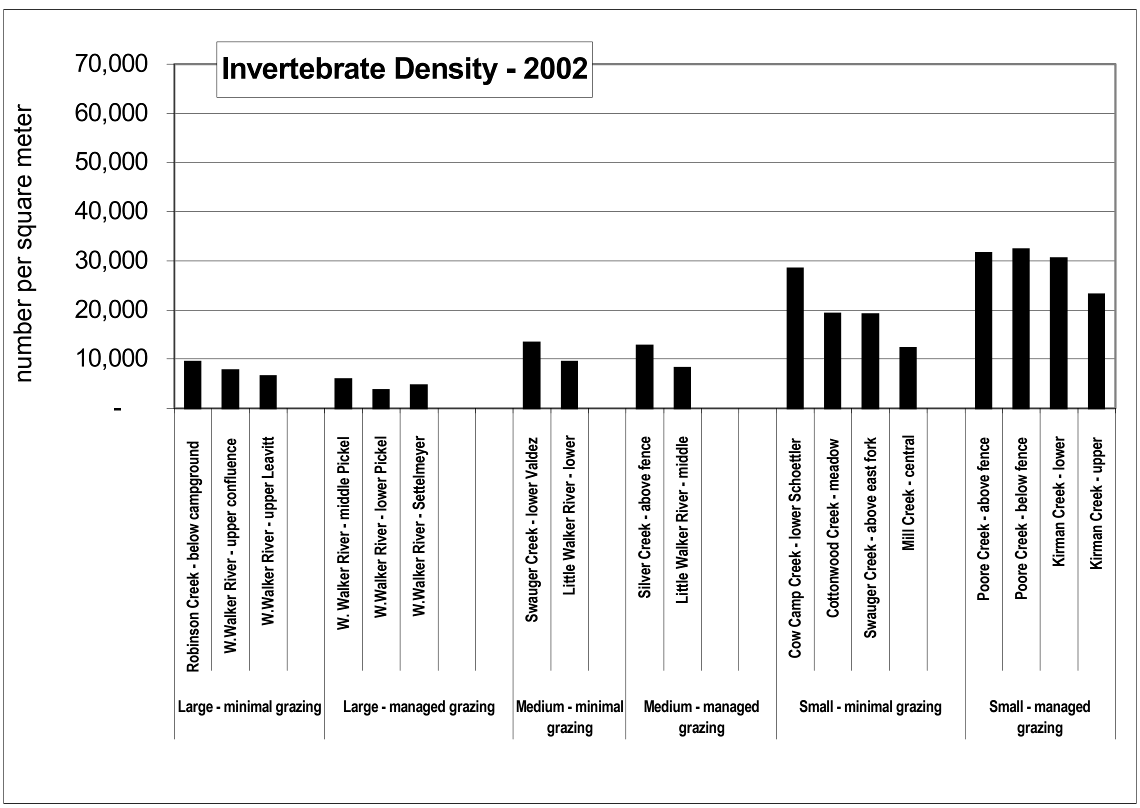


Figure 31b. Average invertebrate density in 2002 (coefficient of variation averages 24%) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

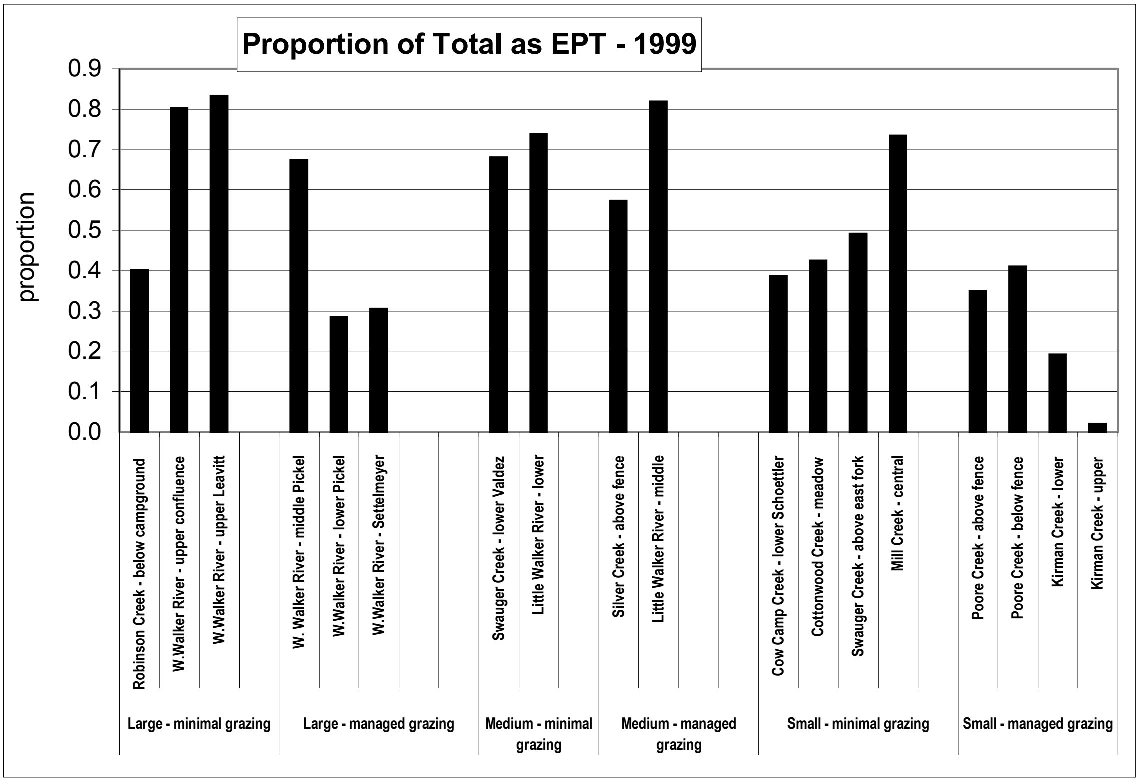


Figure 32a. Composite percent EPT of total counts in 1999 (total EPT from all 5 replicate samples taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

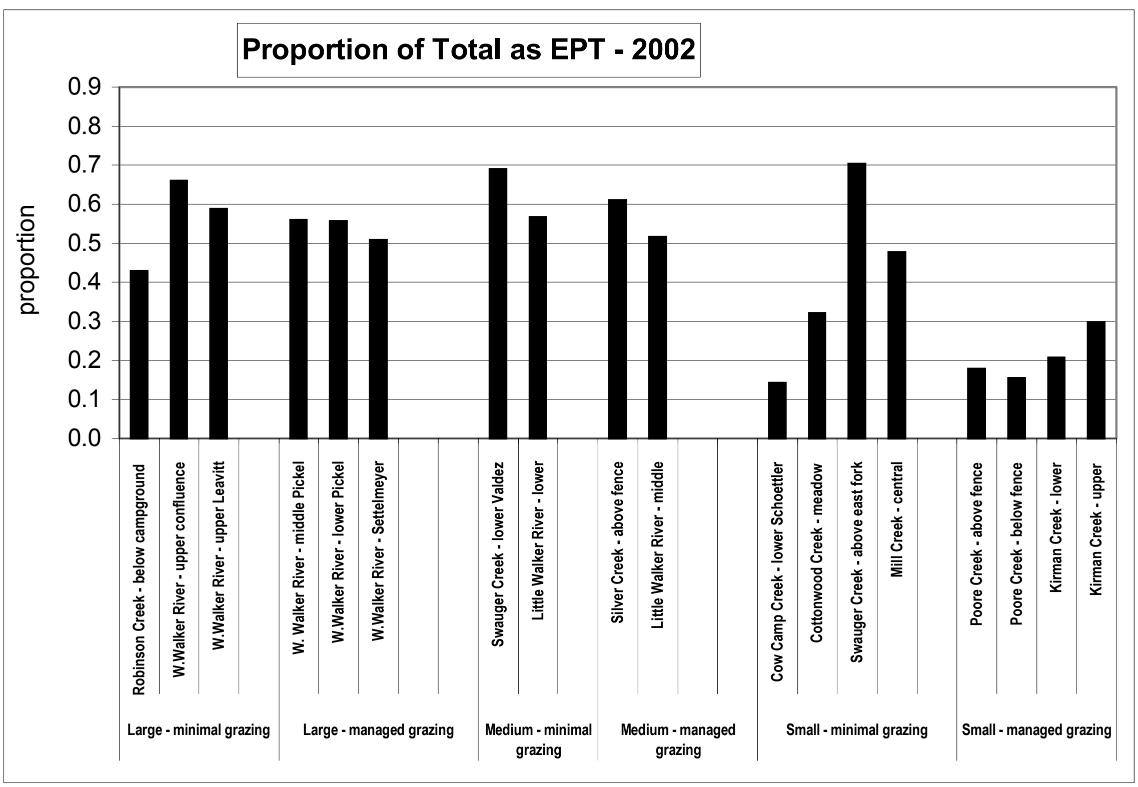


Figure 32b. Composite percent EPT of total counts in 2002 (total EPT from all 5 replicate samples taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

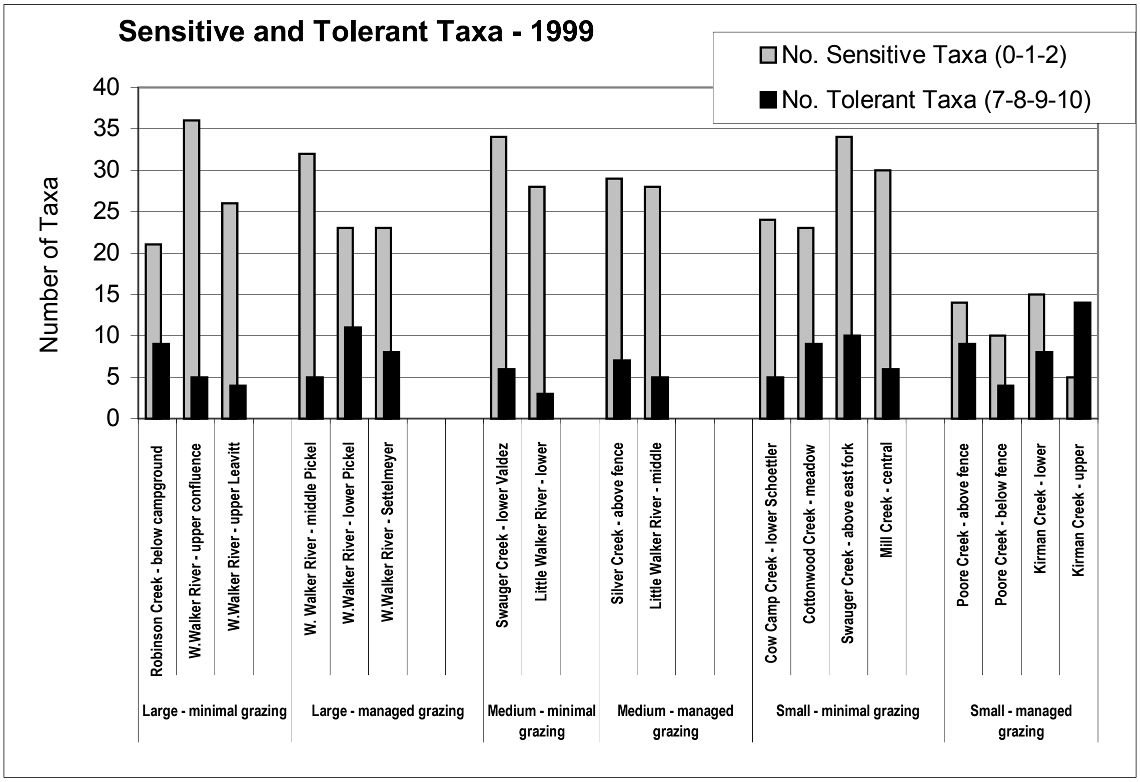


Figure 33a. Composite numbers of sensitive and tolerant taxa in 1999 (from all 5 replicate samples taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

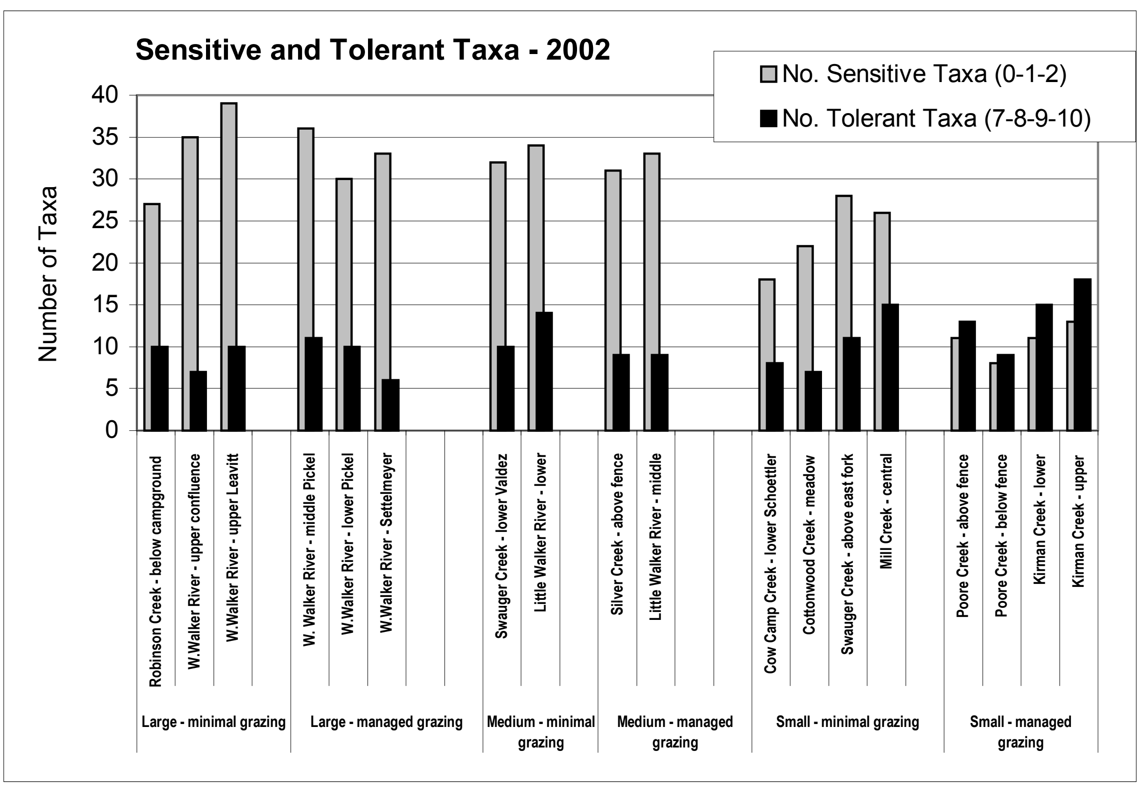


Figure 33b. Composite numbers of sensitive and tolerant taxa in 2002 (from all 5 replicate samples taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

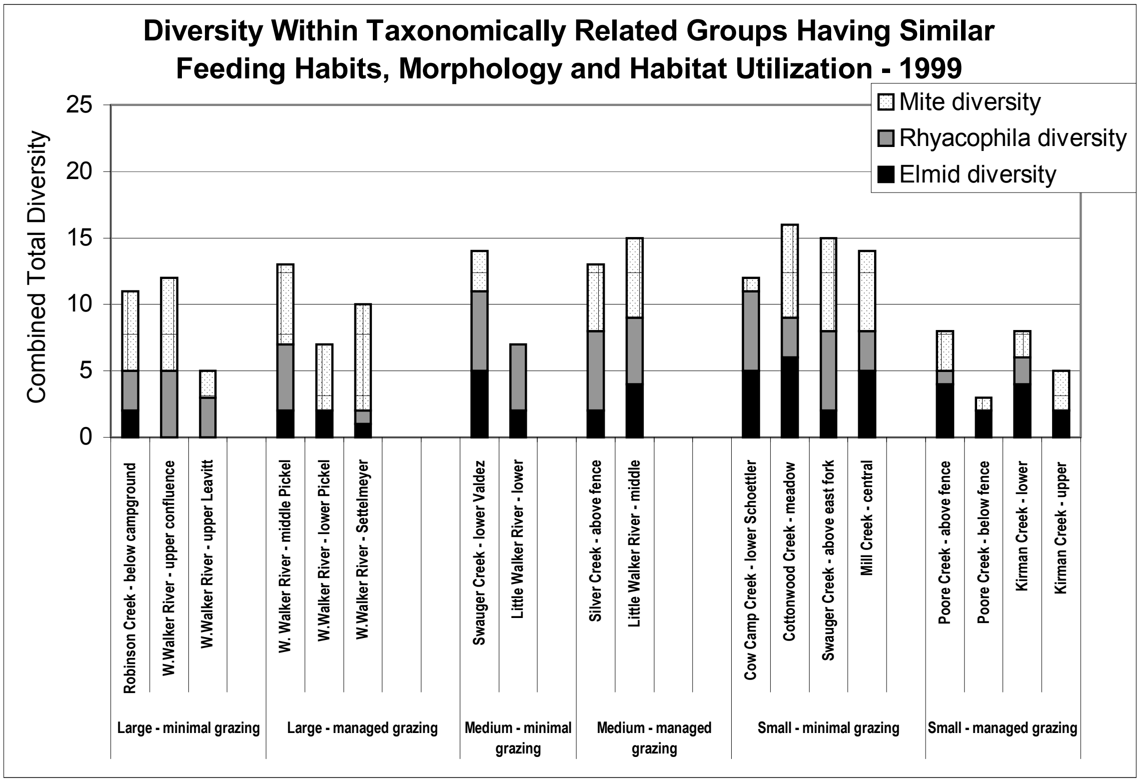


Figure 34a. Composite diversity of mites, Rhyacophila (free-living predatory caddis), and elmids (riffle beetles) in 1999 (taxa from all 5 replicate samples taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

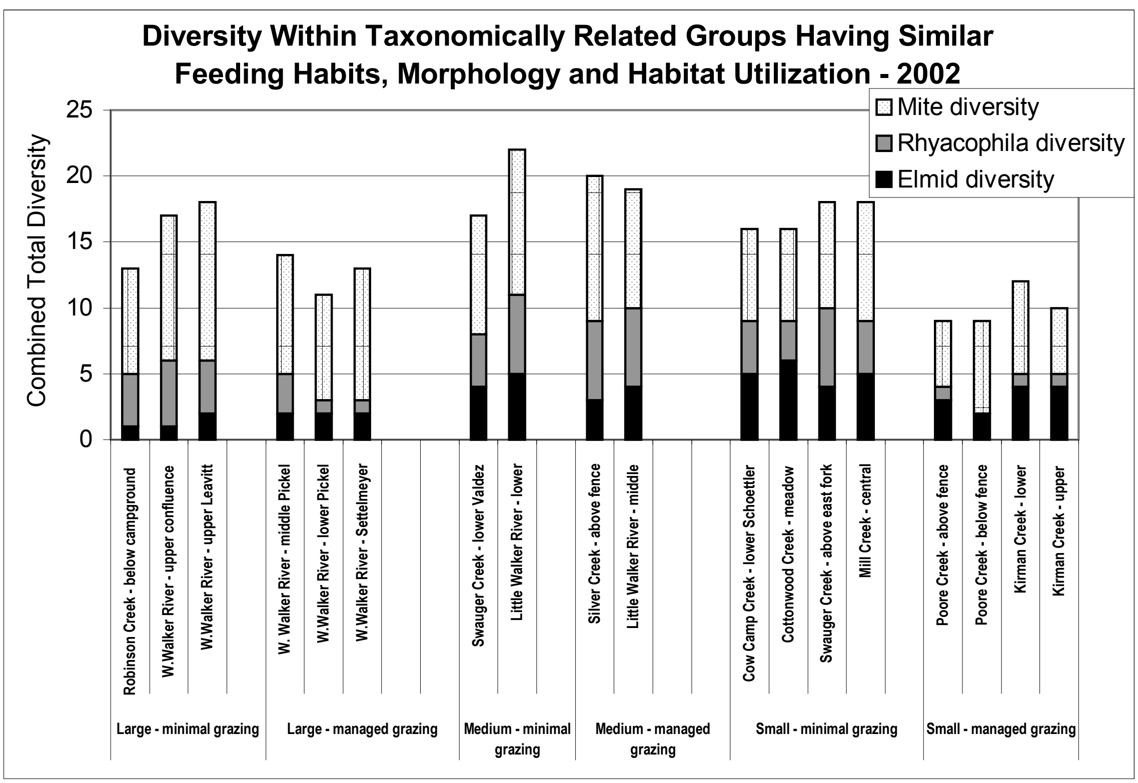


Figure 34b. Composite diversity of mites, Rhyacophila (free-living predatory caddis), and elmids (riffle beetles) in 2002 (taxa from all 5 replicate samples taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

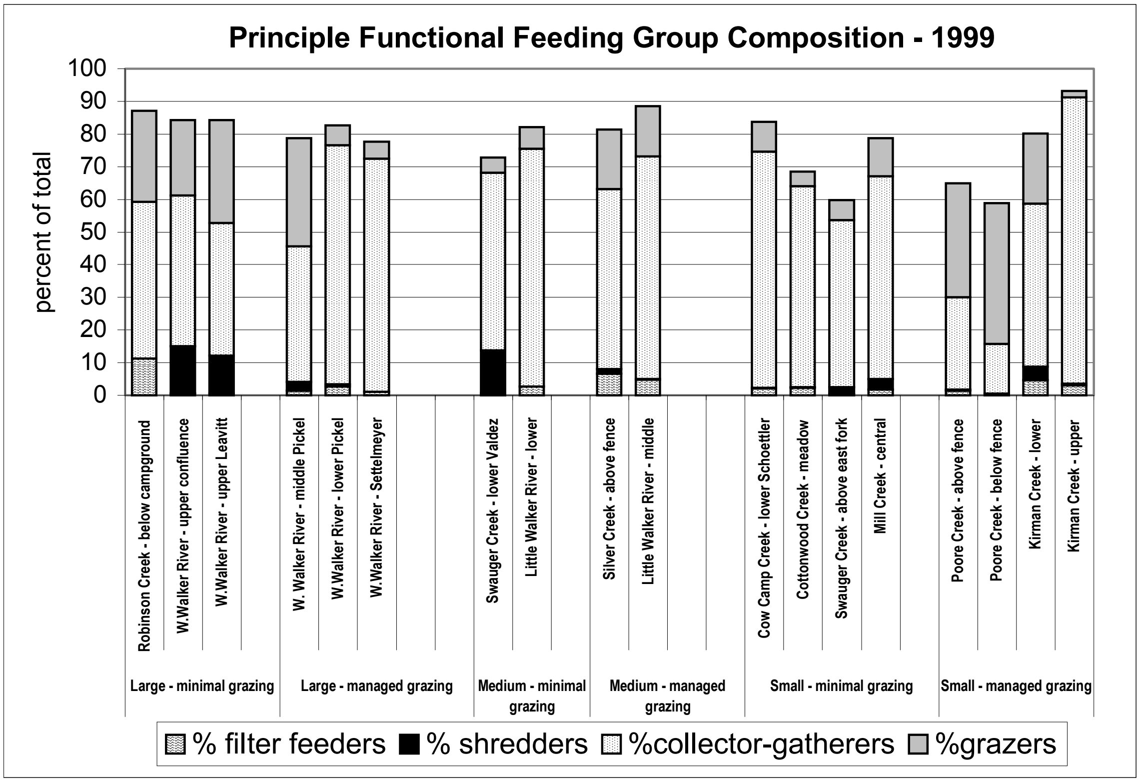


Figure 35a. Composite percentages of the main feeding habit types in 1999 (from all 5 replicates taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

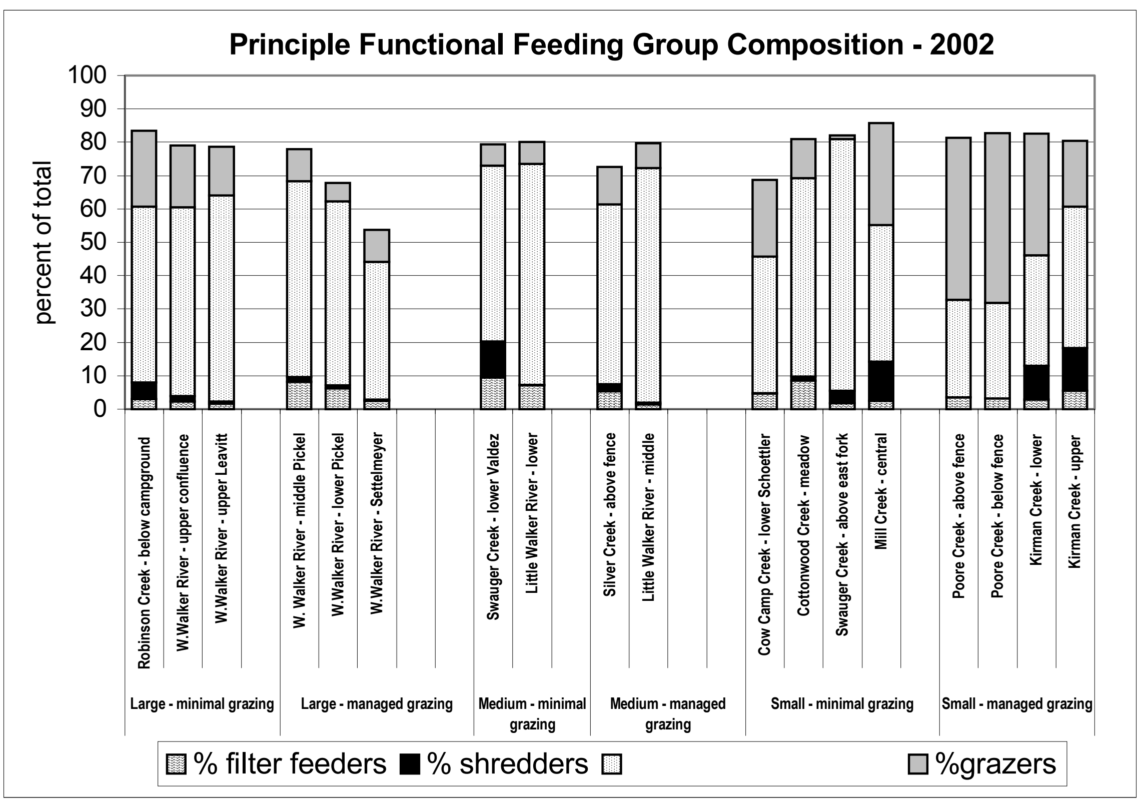


Figure 35b. Composite percentages of the main feeding habit types in 2002 (from all 5 replicates taken at each site combined) of streams in the West Walker watershed. Grouped first by size class (large to small), and as minimal grazed controls and managed grazing treatments within each group.

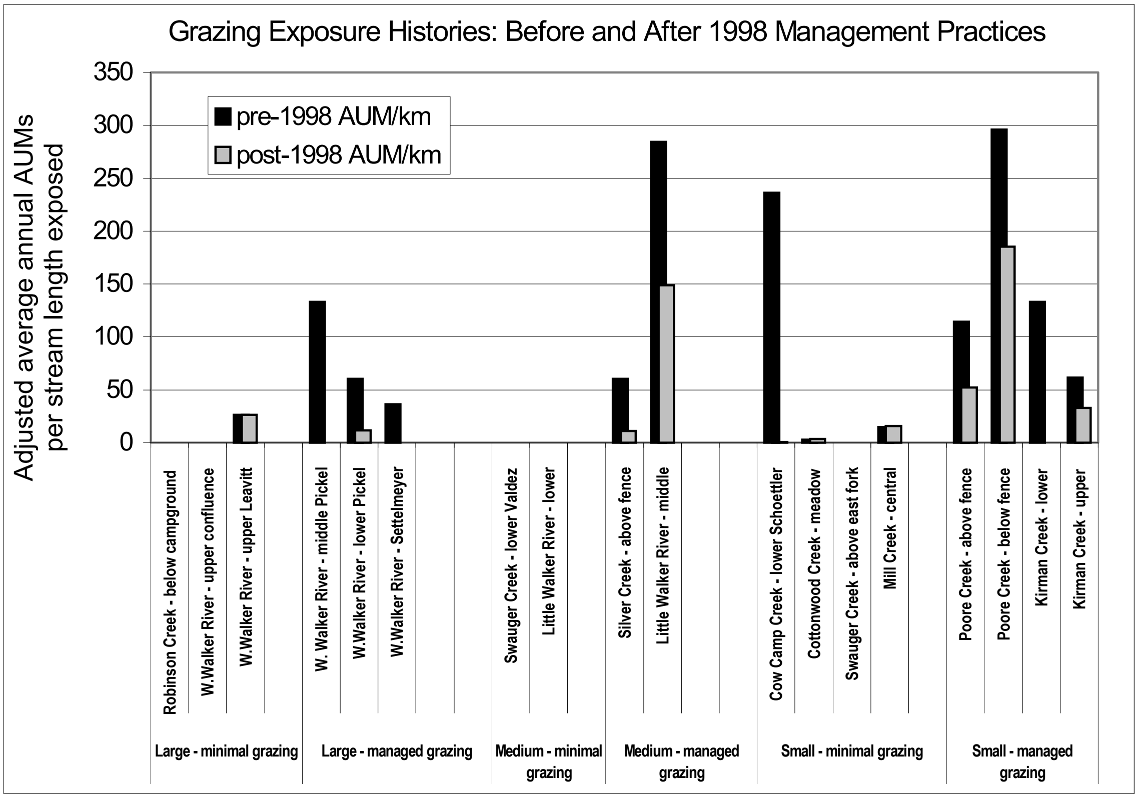


Figure 36. Livestock grazing histories on streams of the West Walker watershed before (since 1980) and after changes in grazing management practices in 1998. The AUMs per channel length were calculated as annual averages distributed over the length of channel within each allotment where a given stream site was located.