Biological impairment in three Kansas reservoirs and associated lotic ecosystems due to sediment and nutrients.

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EXECUTIVE SUMMARY

A series of impoundment and stream water quality measures were determined for a reference impoundment (Banner Creek Reservoir) and two non-reference impoundments (Centralia and Atchison County Lakes). In addition to core sample chemistry from these impoundments, water quality, habitat and biological measures from major tributaries to these impoundments were also collected to assess overall watershed impacts from erosion and sediment additions to these aquatic ecosystems. Due to limited number of study watersheds, the robustness of this study is limited by low sample size, both temporally and spatially, rendering some conclusions more speculative than analytical. In general there were few significant water quality differences between reference and non-reference impoundments, except for turbidity and nitrogen. However phosphorus, total suspended solids (TSS) and volatile suspended solids (VSS) values were higher in non-reference impoundments, as were nitrogen and turbidity. Banner Creek (i.e. reference stream) had statistically lower concentrations of nutrients than either non-reference steam, but no stream differences were found for stream turbidity, TSS, VSS or inorganic suspended solids (ISS). V* and A*ave values, which are measures of unconsolidated stream bed materials in a stream reach, were higher in Banner Creek due to loose, un-compacted sands while the non-reference stream bottoms were mostly silt or silt and sand mixtures. Stream nutrients were highly related to impoundment nutrient values suggesting that normal flows were a major contributor to impoundment concentrations. No meaningful relationships between stream turbidity, TSS, VSS and impoundment measures were found, and core chemistry related to few other ecosystem parameters. Banner Creek Reservoir cores had a higher % silt than other impoundments, and overall % clay strongly correlated to TP in the cores.

Phytoplankton and zooplankton communities showed few differences between these two treatment groups (reference vs. non-reference). The reference impoundment had higher phytoplankton richness and somewhat higher diversity values, but zooplankton richness in this same reference impoundment was lower, as were most measures of zooplankton diversity. Stream habitat and macroinvertebrate community metric values were not significantly different for the most part unless the data from one of the non-reference site-dates was removed from the ANOVA analysis. Macroinvertebrate values were highly variable within and between dates and sites, thus preventing a clear separation between reference and non-reference stream conditions. In general, reference stream macroinvertebrate communities were more diverse and had more taxa than non-reference stream sites. V* and A*ave measures were not good predictors of macroinvertebrate metrics and were marginally associated with stream or impoundment water quality.

Overall, reference and non-reference watershed groups did not exhibit significant differences in suspended or bed sediment for the aquatic ecosystem sites that were studied, but nutrient differences for those sites were significant. It appears that high nutrient concentrations generally associated with low macroinvertebrate metric scores, suggesting a causal relationship between the two. There were no clear differences between baseline (normal flows) sediment measures for the reference and non-reference impoundments and streams. For this same study, USGS reported “annual sediment yields were 360, 400, and 970 tons per square mile per year at Atchison County, Banner, and Centralia Lake watersheds respectively”. Despite marked differences in land use the reference stream and watershed (i.e. Banner Creek) had similar baseline values for turbidity, TSS and other indicators of instream sediment concentrations while the estimate sediment yield for this watershed was higher than Atchison but lower than Centralia. Collectively, this information suggests that factors other than land use are contributing to sediment yields and concentrations even within watersheds that are predominately in

permanent ground cover (e.g. pasture, hay meadows range land). Differential nutrient loading may instead be associated with differences in sediments derived from within the stream channel, rather than directly from contemporary overland flow.

BACKGROUND

Sedimentation in reservoirs has become an increasing concern in Kansas, leading to collaboration of various agencies and research units working to address issues such as sedimentation assessment methods, management practices to control sedimentation, and economic issues of reservoir rehabilitation (KSU 2008). These issues led to creation in 2008 of a Sediment Baseline Assessment Work Plan whose goal is to identify baseline conditions of Kansas streams and watersheds. The various academic and state groups examined seven watershed characteristics for assessment: geomorphology, hydrology, and geology/soils which comprise the physical setting and process portion of the baseline assessment methodology; riparian condition and land use which encompass the management opportunities in the watersheds; and biology and chemistry which will be used to assess the current condition and then measure movement toward the desired outcome in the streams and lakes of the watersheds. For more details of the plan see the Sediment Baseline Assessment Work Plan on-line at the workgroup's website http://www.kwo.org/reservoirs/Sediment_Baseline_Group.htm.

The workgroup compared a "reference" reservoir, Banner Creek Lake that appeared to have a low sedimentation rate with two reservoirs in the same general physiographic setting that appeared to have much higher sedimentation rates, Atchison County Lake and Centralia Lake (Fig. 1). The Work Plan states that "the ultimate goal would be to use policy and management (where applicable) to change the characteristics of the higher sedimentation rate reservoir to emulate those of the lower sedimentation rate reservoir."

In summer and fall of 2010, the Central Plains Center for BioAssessment (CPCB) sampled Banner Creek Lake, Centralia Lake, and Atchison County Lake (named Clear Creek Lake on some maps) and their tributaries to assess biological impairment due to sedimentation. Three stream sites on Banner Creek were sampled, while two stream sites were sampled on Centralia Lake's tributary Black Vermillion River, and one stream site was sampled on Atchison Co. Lake's tributary Clear Creek (Table 1). The Work Plan provides details about each watershed.

Table 1. Reservoirs and tributary study sites with stream site codes and locations. Coordinate datum is NAD83 and transects at which coordinates were taken are indicated. See Appendix 1 for specific site maps.

Impoundment	Stream	Code	Location	Latitude	Longitude	Transect	Description
Banner	Banner Creek	B1	upper site	39.44754	-95.81076	1	Downstream of USGS 392652095484100 (BA1). Follow foot path on east side of road M.
Banner	Banner Creek	B2	middle site	39.44747	-95.81005	1	
Banner	Banner Creek	B3	lower site	39.44709	-95.80898	1	
Centralia	Black Vermillion	C1	upper site	39.69001	-96.12675	6	Downstream of USGS 394126096073500
Centralia	Black	C2	lower site	39.69060	-96.12693	1	

Impoundment	Stream	Code	Location	Latitude	Longitude	Transect	Description
	Vermillion						(CE1).
Atchison	Clear Creek	A1	only site	39.63734	-95.43303	5	Upstream of USGS 393817095260100 (CL1), between 326th and Decatur Rds.

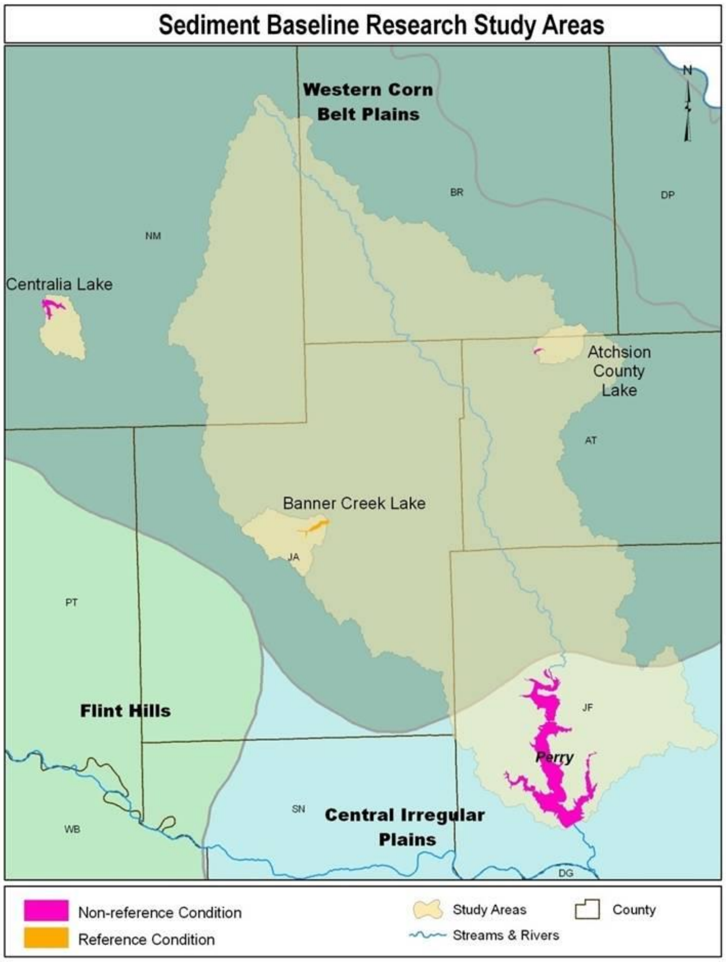

Figure 1. Northeastern Kansas and the three study reservoirs: Centralia Lake in Nemaha Co., Atchison County Lake, and Banner Creek Lake in Jackson Co. From the Sediment Baseline Assessment Work Plan (see http://www.kwo.org/reservoirs/SedimentGroup/Rpt_Sediment_Baseline_Assessment_Work_Plan_022009_cbg.pdf).

METHODS

Sampling dates

CPCB's first stream sampling event occurred between 1 – 14 July 2010 with lakes sampled July 13 and 14. The second stream sampling period was 6 – 14 October 2010, with lakes sampled October 6 and 7.

Tributary sampling

Reach layout

We established three sampling sites along Banner Creek, the tributary to Banner Creek Lake; two along Black Vermillion River, a tributary to Centralia Lake; and one on Clear Creek, the primary tributary to Atchison County Lake. The number of sites in each stream system was limited by the scarcity of permanent flowing water that allowed for macroinvertebrate colonization and water quality sampling at normal flows. While small watershed size typically limits watershed heterogeneity and can reduce sampling efforts, stream systems draining these watersheds often experience intermittent flows and support limited faunal assemblage. Therefore, site selections were limited to stream segments that were least likely to be stressed by low or no flow conditions. At each site, a center transect was marked with flagging tape and latitude and longitude was recorded. A reach length of 20 times the average of five wetted widths was delineated around the center transect, and 10 – 12 transects were laid out and numbered sequentially from downstream to upstream (Fig. 2). The establishment of transects along each stream study reach was, in part, to facilitate the sediment depth sampling of the modified V* method used in this study (see Sediment sample section below).

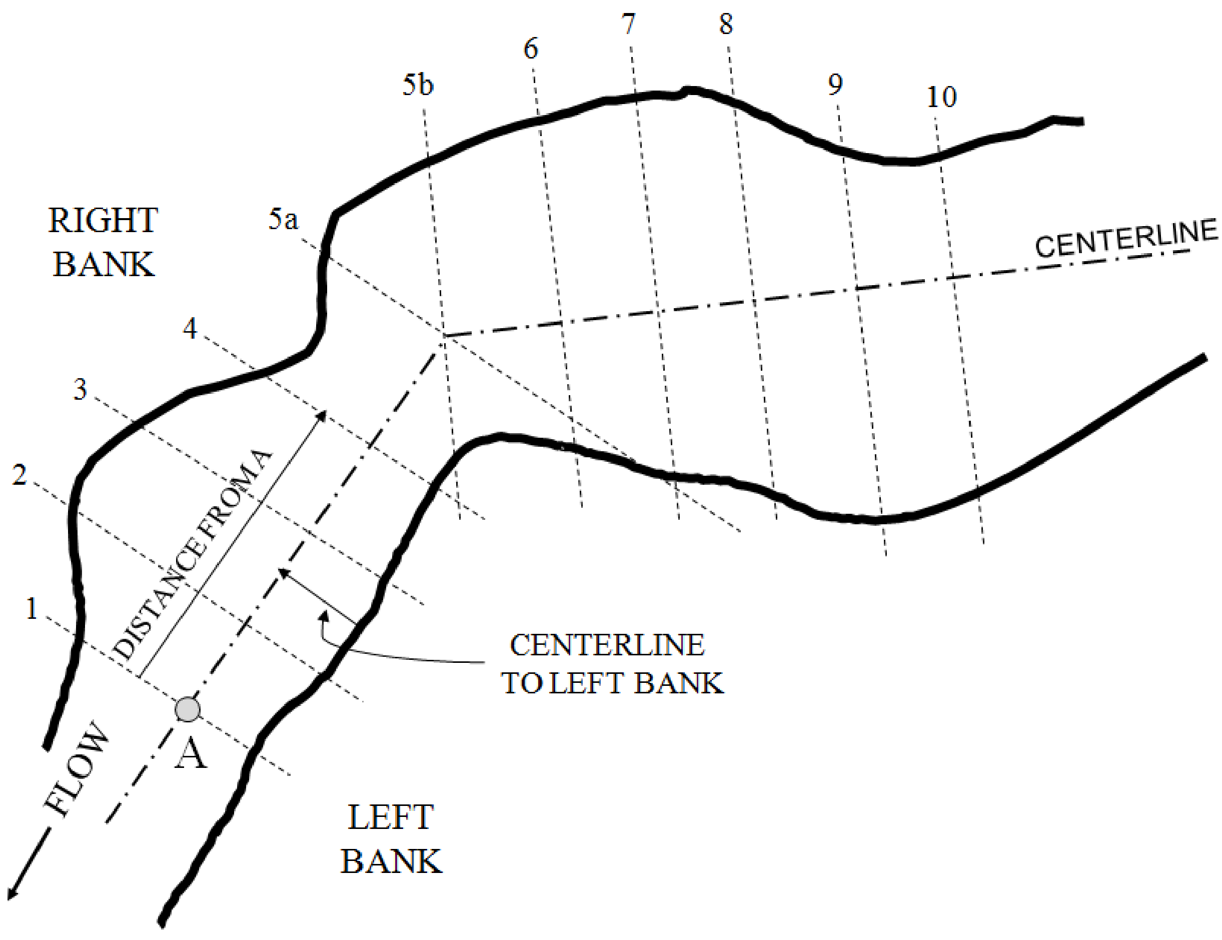

Figure 2. Placement of transects in each stream reach which is 20 times the average wetted stream width. Sediment depths were measured at 10 – 20 locations along each transect using a stainless steel probe.

Water quality

At the downstream transect (transect 1), before the crew entered the water, a 1-liter surface sample from mid-channel was collected in a labeled amber glass bottle that was preserved on ice and returned to the lab for processing suspended chlorophyll *a*, filtered and unfiltered TN and TP (Ebina *et al.* 1983), TSS, and VSS (APHA *et al.* 2005). Nutrient analyses on unfiltered water samples represented total phosphorus (TP) and total nitrogen (TN). Filtered (0.45 μm, 47 mm diameter glass fiber filter) water samples analyzed for phosphorus and nitrogen represented total dissolved phosphorus (TDP) and dissolved nitrogen (TDN). *In situ* measurements (DO, pH, conductivity, salinity, air and water temperature, and turbidity) were measured with a Horiba U-10 water quality checker at the same location. The Horiba U-10 was two-point calibrated prior to each sampling event. The Horiba and chemistry measurements were taken within two weeks of the habitat, macroinvertebrate, and sediment assessment. The lag period between these measurement efforts were characterized by no runoff events and all were taken at normal flow levels.

Habitat

To assess habitat we used the Habitat Development Index (HDI, Huggins and Moffet 1988) and the Ohio EPA's Qualitative Habitat Evaluation Index (QHEI) (Ohio EPA 2006). The same person evaluated habitat at all sites and all events. Velocity was measured at one transect with a Swoffer flow meter following protocol established by the United States Geological Survey (Rantz *et al.* 1982) and using a form developed for the USEPA National Stream Surveys (USEPA 2007). Digital photos were taken at each site (available upon request).

Sediment deposition

We examined the extent of sedimentation using a modification of the V* methodology of the U.S. Forest Service (Lisle and Hilton 1992, Hilton and Lisle 1993). By definition V* ("v star") is the ratio of the volume of fine sediment in a pool relative to the total volume of fine sediment and water in the same pool. V* is most appropriately used in permanent pools of stream reaches with riffle-run-pool morphology, hard substrates, and mild gradients, such as Rosgen B2, B3, or C channel types (see Rosgen 1996). However, preliminary work in the sand-bottom streams examined in this study suggested that the majority of pools to be measured were scour pools, where little to no sediment was deposited due to prevailing hydraulic conditions. Instead, the majority of fine sediment deposition appeared to be in stream runs, where flow velocity decreased and larger particles tended to settle to the bottom. Given the proven utility of the V* approach in previous studies, we believe that the V* concept may provide valuable insights into sediment deposition in sand-bottom streams. However, based on our initial findings, we determined that an adaptation of the Lisle and Hilton V* methods would be necessary to describe sediment deposition for these systems.

As a first step, rather than measuring sediment only in pools, we measured the depths of fine sediment and water along each transect of the study reach, including runs, riffles, and bars. Cross-sectional areas and reach volumes were calculated from these measurements (Fig. 3 - 5). To account for stream sinuosity, at each bend of the centerline, two transects were placed and the inside angle between them was recorded (see 5a/5b in Fig. 2).

Field Measurements

Each site was visited twice, and each site had between 10 and 20 transects. At each transect, a survey rod was placed perpendicularly to the center line of the stream. Starting at the waterline on the left bank, sediment and water depths were measured using a graduated stainless steel probe at 10 to 20 intervals along the survey rod, ending at the waterline on the right bank. For each measurement location, the following data were recorded: distance from the left bank, water depth to the bottom surface, sediment depth, and dominant substrate (sand, silt, clay, cobble, gravel, bedrock, or other). Sediment sizes for dominant substrate were classified using USEPA EMAP methodology (USEPA 2007). In fall sampling events, detritus (e.g., leaf litter, sticks, etc.) covered some portions of the stream bed, and the detritus layer depth was also measured (if greater than > 0.5 cm thick). For purposes of calculation, the detritus layer depth was subtracted from the sediment depth measurement.

Calculation of Metrics

Cross-sectional area was estimated as a series of trapezoids, similar to the velocity-area method commonly used in flow calculation (USEPA 2007). Knowing the distance of each measurement from the left bank, the depth of the sediment, and the depth of the water, we were able to estimate the total cross-sectional area of the stream (sediment plus water) and the cross-sectional area of the sediment for each transect. Then, using the spacing between transects, we were able to estimate the sediment volume and total volume of the reach. Both V* (the ratio of the sediment VOLUME to the total VOLUME for each cross-section) and A* (the ratio of the sediment AREA to the total AREA for each cross-section) were subsequently calculated.

Based on these data, five metrics of sediment deposition were calculated to estimate sediment parameters: A*, V*, A*ave, mean A*ave, and mean V*. V* represents the sediment volume across all site transects for one site visit, A* represents the sediment cross-section of one transect for one site visit, and A*ave represents the average of the sediment cross-section for all transects at a site for one site visit. Calculations designated as mean represent the mean for a given metric across multiple site visits.

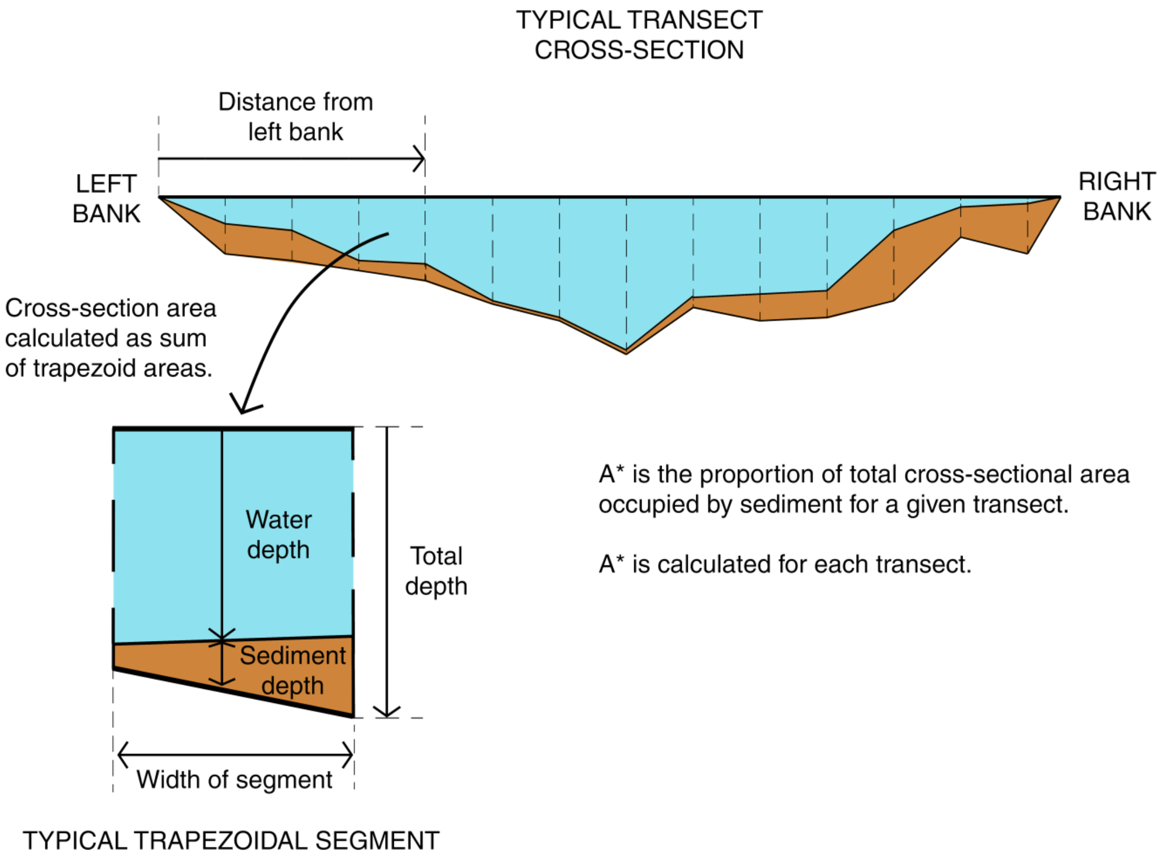

Figure 3. Typical transect cross-section for estimation of stream and sediment volume. Measurements of water depth, sediment depth, and distance from the left bank were used to calculate the area of roughly 10 to 20 trapezoidal segments across the channel. Estimations of total cross-sectional areas were made by summation of the area of these trapezoidal segments.

For a given transect, the cross-sectional area of the sediment and the cross-sectional area of the whole stream are calculated as a sum of trapezoidal areas (Fig. 3):

$$\mathrm{A}_{sediment} = \sum_{i}^{\substack{number\ of \\ measurements}} \frac{1}{2}\left(\mathrm{d}_{sediment_i} + \mathrm{d}_{sediment_{i+1}}\right)(\mathrm{L}_{i+1} - \mathrm{L}_i)$$

and

$$\mathrm{A}_{total} = \sum_{i}^{\substack{number\ of \\ measurements}} \frac{1}{2}\left(\mathrm{d}_{total_i} + \mathrm{d}_{total_{i+1}}\right)(\mathrm{L}_{i+1} - \mathrm{L}_i)$$

where $\mathrm{d}_{\mathrm{sediment}}$ is the depth of sediment, d_{total} is the total depth, L is the distance from the left bank, and the subscripts *i* and *i+1* indicate consecutive measurements along the transect.

A*, the proportion of the cross-sectional area of a transect occupied by sediment, is then calculated for each transect as:

$$\mathrm{A}_j^* = \frac{\mathrm{A}_{sediment}}{\mathrm{A}_{total}}$$

where *j* indicates the transect number for a given site and sampling event.

To estimate the sediment and total volumes of a site, a sum of smaller volumes was calculated. The volume between each transect was calculated by multiplying the area of the downstream cross-section by the spacing between it and the next transect upstream (Fig. 4), then these volumes were added to get the total volume estimate for the study reach:

$$\mathrm{V}_{sediment} = \sum_{j}^{\substack{number\ of \\ transects - 1}} \left(\mathrm{A}_{sediment_j}\right)\left(\mathrm{S}_j\right)$$

and

$$\mathrm{V}_{total} = \sum_{j}^{\substack{number\ of \\ transects - 1}} \left(\mathrm{A}_{total_j}\right)\left(\mathrm{S}_j\right)$$

where *j* indicates the transect number for a given site and sampling event, A*j* is the cross-sectional area of transect *j* as calculated above, and S*j* is the distance along the centerline upstream from transect *j* to transect *j+1*. V*, the proportion of site total volume occupied by sediment is then calculated as:

$$\mathrm{V}^* = \frac{\mathrm{V}_{sediment}}{\mathrm{V}_{total}}$$

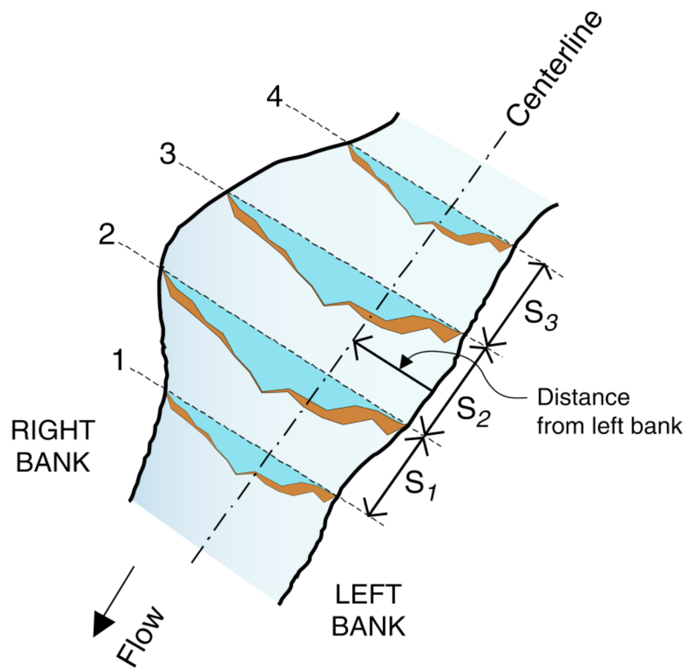

Figure 4. Plan view of transect layout and spacing for use in stream and sediment volume calculations. Distances between transects (*e.g.*, S1, S2, S3) are measured along the established centerline of the stream. Volumes are calculated by multiplying the cross-sectional area of each transect by the spacing to the next transect.

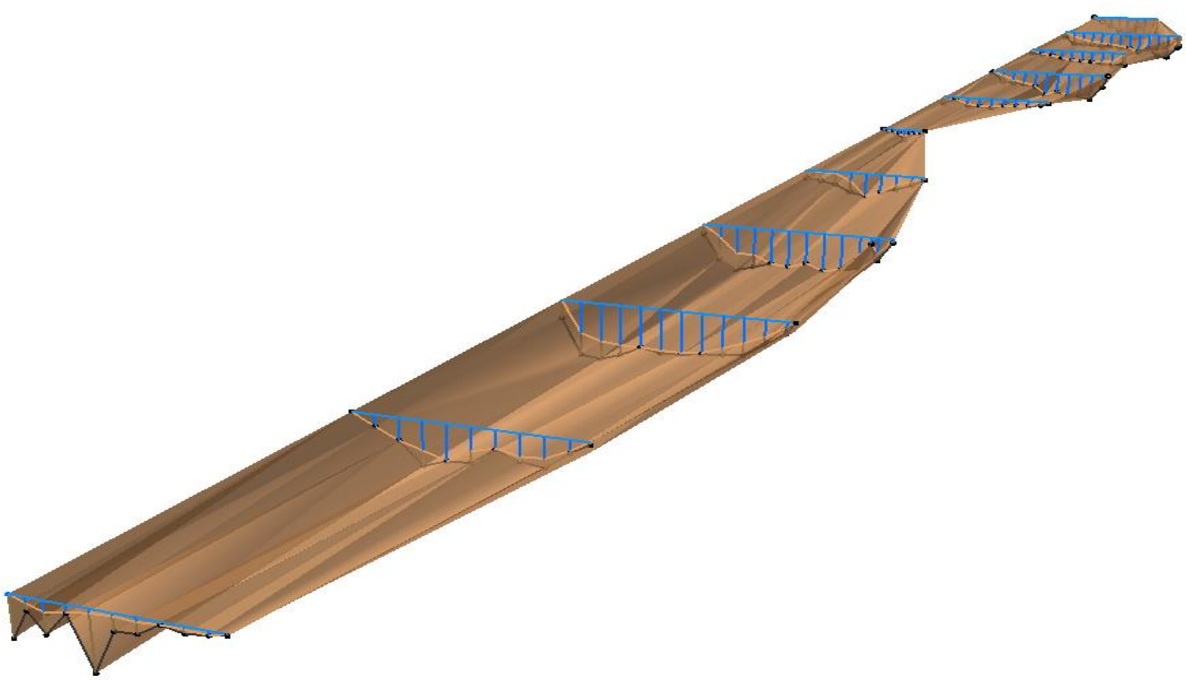

Figure 5. Visualization of sediment thickness, water depth, and transect layout as measured at Clear Creek in July 2010. This illustration reflects spacing, depth, and width, but not direction. Not to scale.

Additionally, we calculated three summary metrics to represent the ranges of condition that occur for a given site across space and time: A*ave (the average of A* for all transects of a given site for a given sampling event), mean A*ave (the mean of A*ave for all sampling events for a given site), and mean V* (the mean of V* for all sampling events for a given site):

$$\mathrm{A}^*{}_{ave} = \frac{1}{n}\sum_{j=1}^{n} \mathrm{A}^*{}_j = \frac{1}{n}(\mathrm{A}^*{}_1 + \mathrm{A}^*{}_2 + \cdots + \mathrm{A}^*{}_n)$$

$$\text{mean } \mathrm{A}^*{}_{ave} = \frac{1}{k}\sum \mathrm{A}^*{}_{ave} = \frac{\mathrm{A}^*{}_{ave_1} + \mathrm{A}^*{}_{ave_2}}{2}$$

$$\text{mean } \mathrm{V}^* = \frac{1}{k}\sum (\mathrm{V}^*{}_1 + \mathrm{V}^*{}_2 + \cdots + \mathrm{V}^*{}_k) = \frac{\mathrm{V}^*{}_1 + \mathrm{V}^*{}_2}{2}$$

where j indicates the transect number, n is the number of transects, and k indicates the number of visits to the site.

Overall, five metrics of sediment deposition were calculated to estimate sediment parameters: V*, A*, A*ave, mean A*ave, and mean V*. V* represents the sediment volume across all site transects for one site visit, A* represents the sediment cross-section of one transect for one site visit, and A*ave represents the average of the sediment cross-section for all transects at a site for one site visit. Calculations designated as "mean" represent the arithmetic mean of a given metric across multiple site visits.

Macroinvertebrates

HDI protocols were used to collect macroinvertebrate samples. Within the stream reach, an aquatic kick net (500-µm mesh) was used to collect macroinvertebrates from a variety of habitats for a total of three minutes. Habitats within each macrohabitat (i.e. pool, riffle, run, or glide) in each site were subsampled in proportion to occurrence in the site. On bottom substrates, approximately 0.09 m^2 (1ft^2) of substrate was disturbed to a depth of 1-2 cm. A sweep of similar area was used in vegetated habitats, root wads, and areas associated with woody debris. The subsamples from each site were combined into a single sample jar and preserved with 10% buffered formalin and rose bengal solution.

The samples were returned to the CPCB lab for sorting and identification using the CPCB Standard Operating Procedures (SOP). These and other SOPs are available to download from the CPCB webpage at http://www.cpcb.ku.edu/datalibrary/assets/library/protocols/BenthicLabSOP.pdf. Samples were sorted to remove at least 300 organisms (300 + 10%) from the sample, using a modified Caton gridded tray. Sorted organisms were placed into 80% alcohol for storage and identification to the lowest practical taxonomic level. Macroinvertebrates were identified to the proper taxonomic level described in the SOP for each taxa group. Chironomid and oligochaete specimens were slide-mounted prior to taxonomic identification. References for each taxon are listed in the SOP. Voucher specimens of difficult to identify taxa as well as rare taxa are retained for a minimum of three years after project end dates.

Macroinvertebrate metrics were calculated in Ecomeas 1.6 (http://cpcb.ku.edu/media/cpcb/datalibrary/assets/databases/ecomeas01_6.mdb), a software program developed at CPCB that calculates most commonly used diversity indices and other ecological measures of community structure. Nondistinct taxa were disregarded in the taxa richness calculation so as not to elevate the richness estimates, but were included in the calculation of all other metrics. Metrics calculated and examined included total abundance, taxa richness, richness/abundance, a number of diversity indices, and Fager's Number of Moves (an estimation of alpha diversity). These same metrics were used in examining plankton community differences.

Impoundment sampling

Water samples

Throughout the riverine, transitional, and main basin of each lake ten sampling sites were evenly distributed to capture variance in lake conditions. Latitude and longitude of each site was recorded so that re-sampling of these original sites could be easily accomplished. During each sampling event, *in situ* water chemistry (DO, pH, conductivity, salinity, air and water temperature, and turbidity) was measured with a Horiba U-10 water quality checker at each of the ten sites. In addition, Secchi depth measurements were obtained from the shaded side of the boat.

From the ten sampling sites, a main basin site found to be one of the deepest points in each lake was designated for depth profiles of *in situ* chemistry, sediment core sample, a vertical plankton tow and a liter, surface (i.e. 0.25 m depth) grab sample for laboratory analysis obtained with a Van Dorn sampler. . *In situ* measurements a this site were taken at approximately 1 m depth increments to determine if the lake was stratified. If stratified, a bottom water sample was collected with a Van Dorn sampler for laboratory analysis. Additionally, at the larger Banner and Centralia Lakes with larger more defined riverine areas a surface grab sample was also collected from one riverine site in each lake to assess possible spatial difference within lab chemistry. Water samples were transferred to labeled 1-liter amber glass jars, stored on ice, and returned to the CPCB lab for processing of suspended chlorophyll *a*, TP, TDP, TN and TDN (Ebina *et al.* 1983), total suspended solids (TSS), and volatile suspended solids (VSS) (APHA et al. 2005). Inorganic suspended solids measurements (ISS) were calculated as TSS minus VSS. During each sampling event, a duplicate field sample was taken either at a lake or a stream site, as well as a sample in a nutrient-spiked jar. For details regarding accuracy and precision requirements, see EPA Award X7 97703210 QAPP (http://www.cpcb.ku.edu/research/assets/2009MODIS/QAPP_modis_r1_2009Jul25.pdf).

Sediment core samples

A single sediment core was taken at each primary water chemistry site, kept upright on ice, and delivered to the KU Department of Geography where subsamples (0 – 10, 10 – 20, and 20 – 30 cm depth if possible) were analyzed for particle size, bulk density, TP and TN. Sediment subsamples were sent to Kansas State University for analysis of TP and TN. A total of 10 sediment cores were collected and analyzed during this project.

Table 2. Samples collected at each lake during each sampling event in July (Jul) and October (Oct) 2010. One-liter water samples were returned to the CPCB lab for analyses of TN, TP, TDN, TDP, chlorophyll *a*, TSS, and VSS.

Impoundment	*In situ* water chemistry		Secchi depth		Primary water samples (1-liter)				Zooplankton tow		Phyto-plankton (1-liter)		Sediment cores	
					Surface		Bottom							
	Jul	***Oct***	***Jul***	***Oct***	***Jul***	***Oct***	***Jul***	***Oct***	***Jul***	***Oct***	***Jul***	***Oct***	***Jul***	***Oct***

Banner	10	10	10	10	2	2	1	1	1	1	1	1	2	2
Centralia	10	10	10	7	2	2	1	0	1	1	1	1	2	2
Atchison	10	8	10	10	1	1	0	0	1	1	1	1	1	1

Zooplankton

A single vertical plankton net tow was conducted at each main basin primary site to collect quantitative samples for zooplankton identification and enumeration. Zooplankton were collected with 80-µm mesh plankton net having a mouth diameter of 20 cm; the sample was transferred to a 500-ml plastic bottle and preserved with 70% ethanol (70 ml of 100% ethanol for each 30 ml of sample volume) then placed in the cooler for transport to the lab for processing. Each vertical tow started approximately 10 cm above the substrate surface and extended to the surface. The tow distance was recorded and the filtered volume of water was calculated for each tow and used to determine the taxon count of organisms per liter.

Zooplankton samples were sub-sampled using a Hensen-Stempel 1 ml pipette. These subsamples were transferred to a 65mm diameter Syracuse glass dish and specimens identified and enumerated at 20-40x magnification, against a black microscope stage. When necessary, multiple subsamples were enumerated until at least 250 individuals, including cladocerans, copepods, and rotifers were counted and the total volume enumerated was then calculated. Cladocerans were identified to species when possible. Copepods were identified to sub-order. Rotifers were identified to phylum. As previously stated, at least 250 individuals were identified from each study sample. All data were recorded on standard datasheets. Once counts were completed, correction factors were calculated for each sample and densities (i.e. numbers per liter) were determined for each of the major groups listed above, based on the original volume of reservoir water filtered in the tow and the total subsample volume used in reaching the ≥250 individual specimen counts (pers.com., A. Dzialowski 2010).
Zooplankton metrics were calculated using Ecomeas 1.6.

Phytoplankton

At each main basin primary site, a near-surface (≈ 0.25 m) phytoplankton sample was obtained using a 1.5 L Van Dorn bottle submerged vertically so that the top of the Van Dorn bottle was about 10 cm below the water surface. A 250 or 500 ml sample was preserved with 1 to 3 ml of Lugol's solution. Different water chemistry and densities of algal material require different concentrations of preservative; hence a general guideline was that there be sufficient Lugol's to turn the sample the color of weak tea.

To facilitate phytoplankton enumeration, the preserved field samples were shaken vigorously and 100 ml aliquots were removed and allowed to settle in 100 ml glass beakers. Beakers with samples were covered with Parafilm© and left to settle for two weeks. After two weeks, 80 ml of liquid was pipetted off each sample with a 5 ml pipette, with care taken not to disturb bottom materials, and discarded. The remaining 20 ml was put into a 100 ml bottle for long-term storage and 5 ml of water was added to it. Sub-samples were shaken vigorously for a 25 seconds and then 1 ml, 3 ml, or 5 ml of algal concentrate was settled overnight in 10 cm long fiberglass settling chambers, each with a 12.5 mm diameter opening. For each sample, 50 fields were counted under 400x magnification on a calibrated Wild Heerbrugg inverted microscope with ocular eyepiece attachment.

Algae were typically identified to genus. Within some genera, distinct species difference were noted and separate species were assigned a species number (e.g. *Scenedesmus* sp. 1, sp. 2, sp. 3).

RESULTS and DISCUSSION

Our assessment of the potential impact of erosion and sedimentation within the stream and reservoirs of this study was based on the *a priori* assumption that the Banner Creek watershed represented a reference condition in regards to upland soil loss and sedimentation of aquatic ecosystems. The recognition of Banner watershed as a reference watershed and both Atchison and Centralia reservoir watersheds as sediment-impaired watersheds in general comes from past information and data collected by various agencies and organizations over the past several decades. In fact, all studies conducted as part of this "Baseline Sediment Studies" effort were, in part, designed around these past determinations and the current *a priori* assumption that good land management and limited cultivation lends itself to reduced sediment loading to aquatic ecosystems. Our study attempted to identify the relationship between sediment losses, stream loadings (reference vs. non-reference watersheds), and changes in the aquatic biological quality of streams and impoundments located within the same drainage areas.

In presenting our results we first compared the water quality and sediment quality/quantity in both the streams and impoundments that comprise both watershed groups (reference vs. non-reference treatments). Treatment group comparisons were the only way we could achieve a large enough sample size ($\geq$ 3 samples) to perform standard parametric statistical comparisons. Our assumption was that a number of key water quality indicators such as turbidity, TSS, VSS, ISS (TSS - VSS), and nutrients would be lower in Banner Creek watershed samples, reflecting better overall water quality. In addition we also expected that biological community metrics showing a more diverse community composed of a large number of sensitive species would be found in the Banner Creek ecosystems. Both one- and two-way GLM ANOVAs were performed on most water quality metrics calculated for impoundments and streams (Hintze 2004). However, only the stream macroinvertebrate metrics could be statistically analyzed since just two phytoplankton and two zooplankton samples were collected during the study.

Stream and impoundment water chemistry

One-way ANOVAs (i.e. season <u>or</u> reference/nonreference) for stream and impoundment water chemistry showed few significant differences except for season and nutrients. Seasonal differences were limited to water temperature and pH for streams and impoundments, while stream salinity and impoundment dissolved oxygen values also varied seasonally. These differences were expected considering the temporal span between sampling events and the close relationship of these parameters with air temperature, hydrology, and normal biological phenology. Nutrients variables (TP, TDP, TN and TDN) were typically lower in the reference (Banner Creek) ecosystems. Two-way ANOVAs that considered both time and treatment differences together showed similar results to those of the one-way ANOVAs. Except for the significant interaction terms between time and treatment for water temperature, pH, and salinity in stream samples, no other interactions were found to be significant. These findings allowed us to combine the seasonal data for those variables of most interest (e.g. nutrients, turbidity) and calculate one-way ANOVAs using all measurements for these variables. These results were similar to both the original one-way and two-way ANOVAs (Table 3).

Table 3. Results of Analysis of Variance (ANOVA) tests for treatment effects (i.e. reference vs. non-reference) for various stream and impoundment chemistry parameters. Significantly ANOVA models are noted in bold print. The last column “Difference in mean values” shows actual differences in non-reference mean values when compared to reference mean values for significant models where + indicates and increase and – a decrease in mean values

Waterbody	Parameter	n	p	F-ratio	Difference in mean values
Stream	Turbidity	12	0.44	0.64	
Stream	TSS	12	0.30	1.22	
Stream	VSS	12	0.43	0.68	
Stream	ISS	12	0.29	1.22	
Stream	**TDP**	**12**	**0.00**	**12.91**	**+ 87.2 µg/L**
Stream	**TP**	**12**	**0.00**	**15.36**	**+ 163.1 µg/L**
Stream	**TDN**	**12**	**0.00**	**14.86**	**+ 1823.8 µg/L**
Stream	**TN**	**12**	**0.00**	**20.93**	**+ 2063.0 µg/L**
Lake	**Turbidity**	**99**	**0.01**	**6.79**	**+ 38.9 NTU**
Lake	TSS	13	0.20	1.90	
Lake	VSS	13	0.07	3.97	
Lake	ISS	13	0.20	1.87	
Lake	TDP	13	0.07	4.01	
Lake	TP	13	0.08	3.81	
Lake	**TDN**	**13**	**0.01**	**10.12**	**+ 997.2 µg/L**
Lake	**TN**	**13**	**0.00**	**22.67**	**+ 1164.9 µg/ L**

Overall, water samples from the Banner Creek stream sites had lower nutrient concentrations than stream sites in the two non-reference watersheds for both total and dissolved forms (Fig. 6). Examination of these box plots suggest that most all of the nitrogen in these streams is in a dissolved form (TDN), probably as nitrate nitrogen. However, based on differences between the median and geometric mean values for TP and TDP, it appears that over one-half (about 54%) of the phosphorus in these streams is in a particulate form. While there were no statistical differences between reference and non-reference, box plots of turbidity, TSS, VSS, and ISS suggest that while turbidity was somewhat lower in the reference stream, all forms of suspended solids were higher (Fig. 7). It would seem that most TP in these streams is attached to suspended material (e.g. sediment, fine particulate organics), and that the higher non-reference TP values are due to the amount of TP attached to suspended material and not the amount of sediment itself.

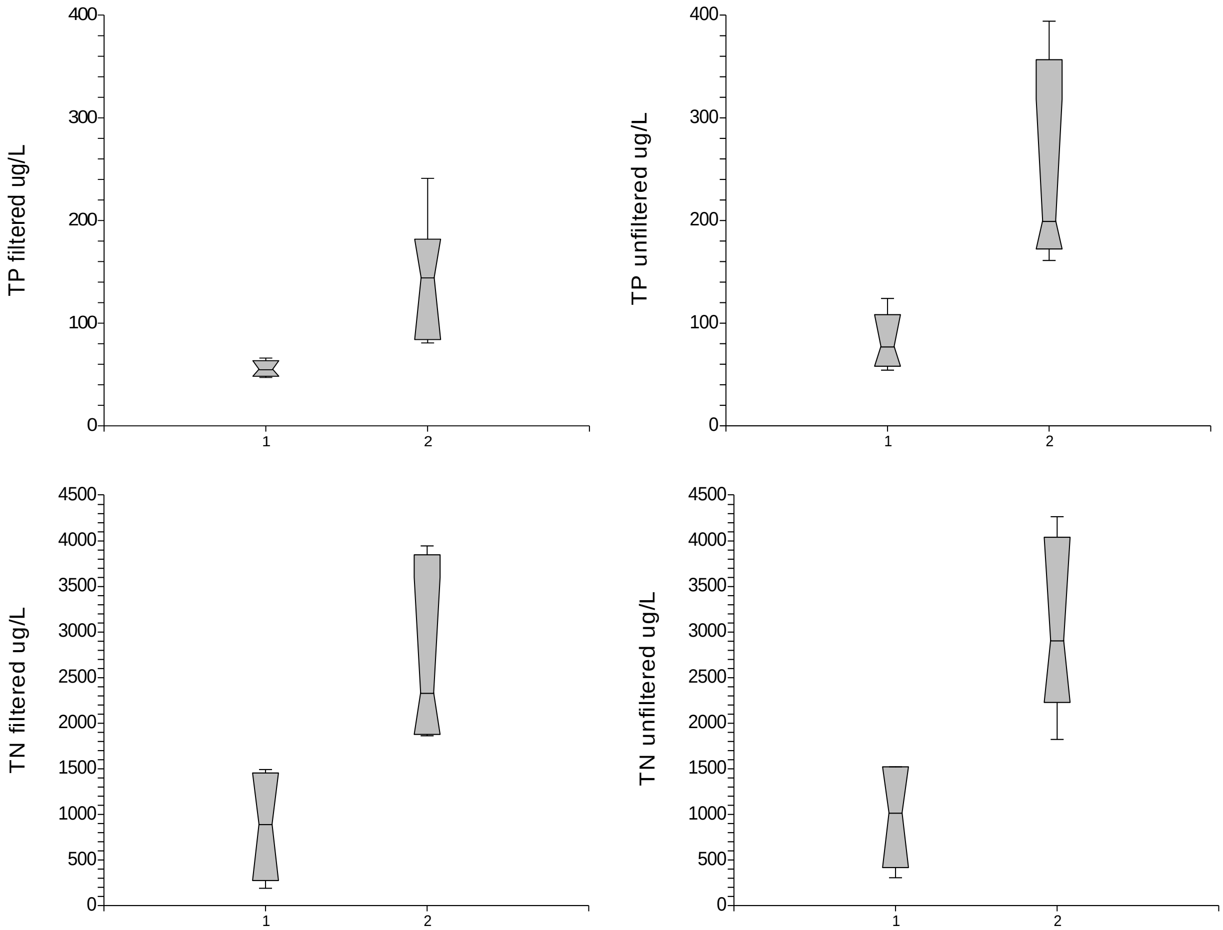

Figure 6. Box plots of total phosphorus and nitrogen concentrations in both filtered and unfiltered water samples from reference (1) and (2) stream sites.

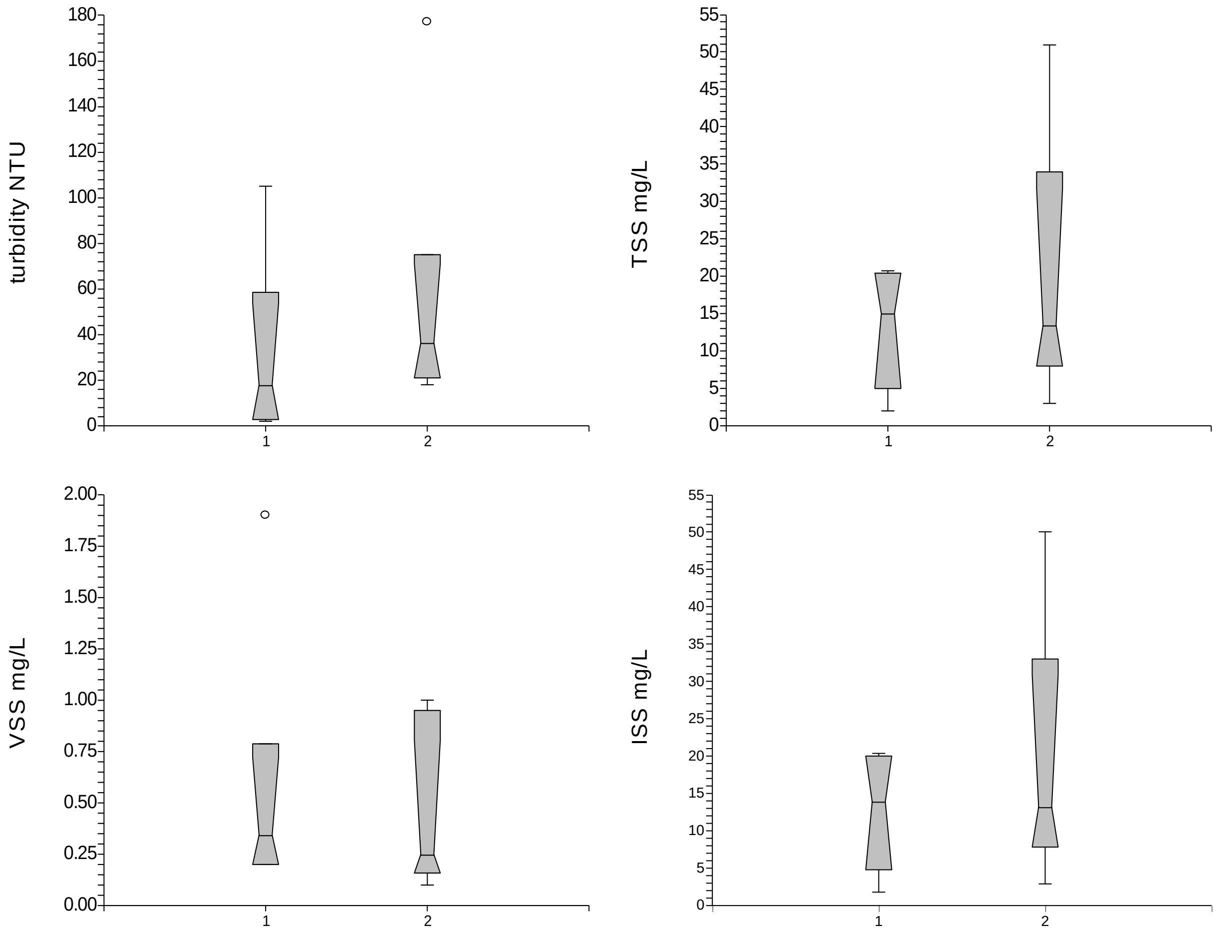

Figure 7. Box plots of turbidity TSS, VSS, and ISS concentrations in water samples from reference (1) and non-reference (2) stream sites. A single outlier TSS value (352 mg/L) was removed from the non-reference group because of suspected bottom disturbance by the Horiba Water Checker® sonde during *in situ* sampling.

Two-way ANOVA results for lakes indicated that there were significant interactions between sampling period and treatment for water temperature, conductivity, turbidity, pH, TN and TDN. Again while we could expect all measured parameters to show seasonal differences, the significant interaction term associated with the above parameters suggests the occurrence of a time/treatment effect that could influence the direct interpretation of both factors (time and treatment effects). However, the ANOVA outcomes for impoundments tend to be supported by box plots for these and other parameters (Fig. 8 and 6) and by previously noted differences in stream values. Only turbidity and TN were found to be significantly different in impoundment groups (Table 3), which is similar to the finding for the streams, where turbidity was not significant but was generally lower in the reference stream (Fig. 7). Box plot results for TN and TDN show a distinct separation in reference and non-reference values. A similar pattern was observed for TP and TDP, but with a larger overlapping data cloud (Fig. 8). Impoundment turbidity was very different between reference and non-reference groups. While the median values for TSS, VSS, and ISS were also noticeably different between groups, individual measurements were highly variable causing the upper and lower quartiles to broadly overlap (Fig. 9).

It should be noted that nearly all the TSS measured in these impoundments was ISS and probably represented eroded and resuspended soils and other inorganic materials.

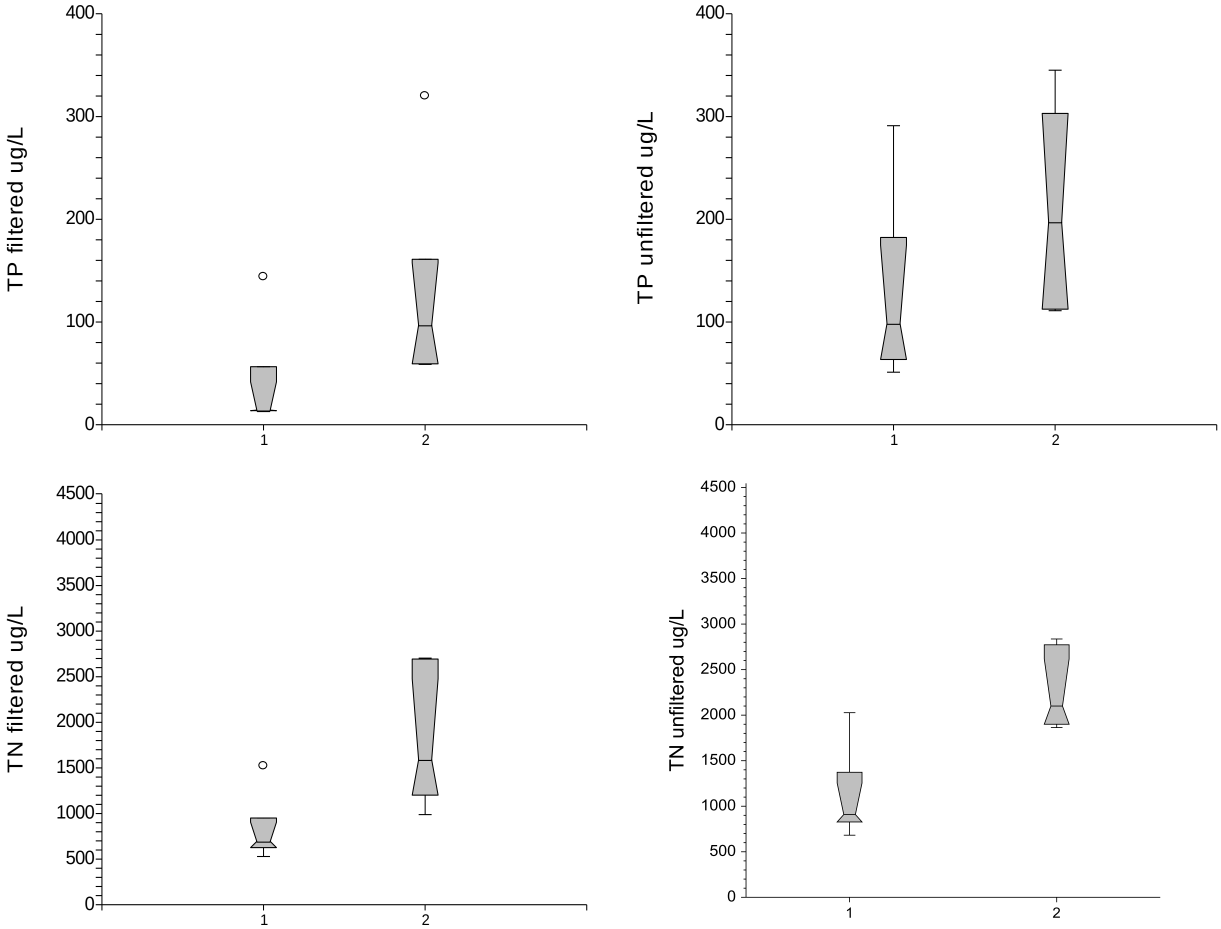

Figure 8. Box plots of total phosphorus (TP) and total nitrogen (TN) concentrations in both filtered and unfiltered water samples from reference (1) and non-reference (2) impoundment sites.

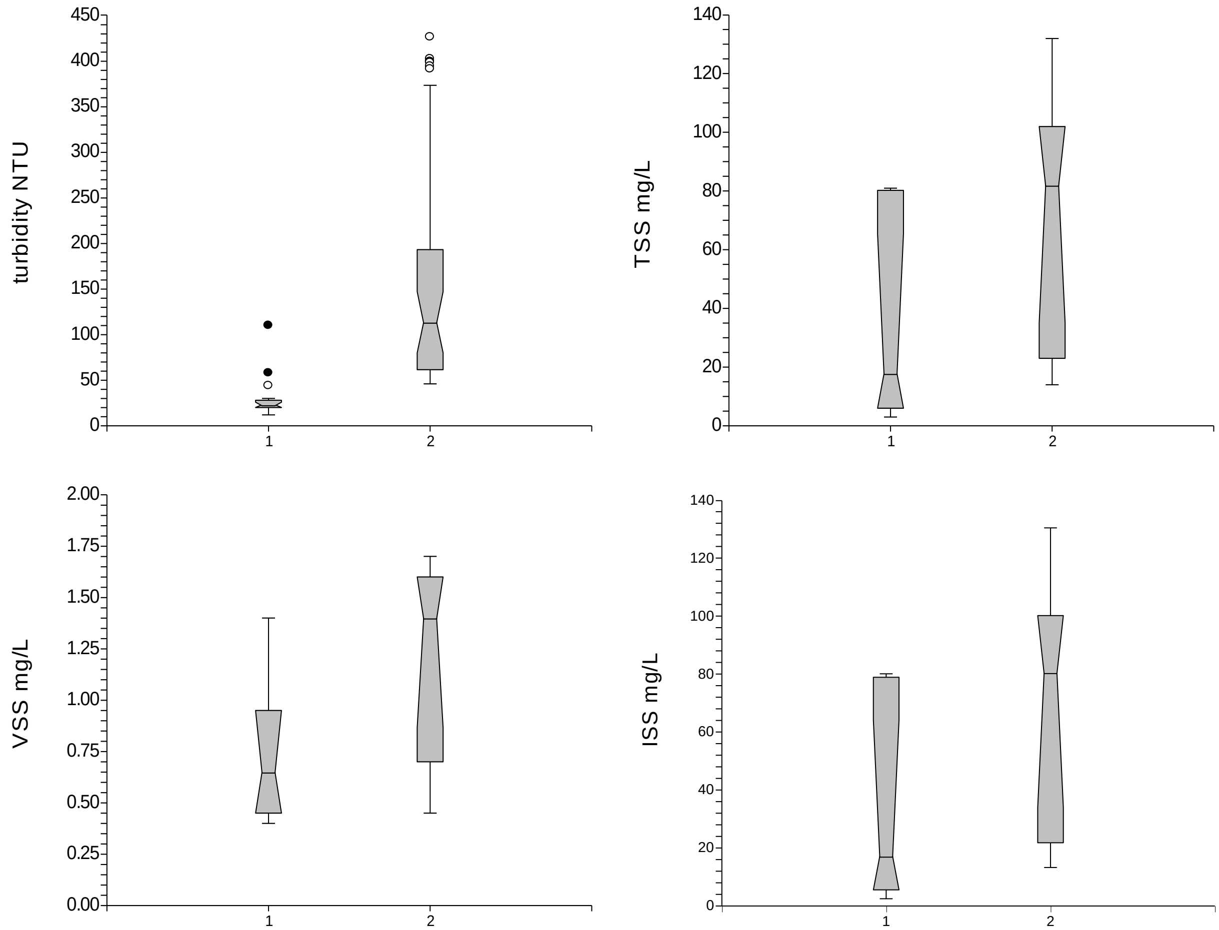

Figure 9. Box plots of turbidity, total suspended solids (TSS), volatile suspended solids (VSS), and inorganic suspended solids (ISS) concentrations in water samples from reference (1) and non-reference (2) impoundment sites.

In general, reference stream nutrient concentrations at normal flows were a good predictor of impoundment nutrient concentrations (Table 4). All significant models in Table 4 were positively related to the independent variable that comprised the simple regression models. These findings suggest that nearly all of the nutrient load is being delivered to these impoundments. However, stream turbidity, TSS, VSS, and ISS were not related to impoundment measures of these same parameters suggesting that impoundment characteristics and dynamics (e.g. mean depth, mixing) may be as important of determinants as incoming stream concentrations in regards to these parameter concentrations.

Table 4. Robust regression information for significant models (alpha = 0.05) except for the model where impoundment TP is the dependent and stream TP is the independent which had a p value of 0.06. However this model was thought to be biologically significant and the p value just failed the alpha value cutoff so it was included for discussion.

Dependent variable	Independent variable	N	Model p value	Intercept p value	Relationship	R^2
Impoundment TP	Stream TP	12	0.06	0.11	+	0.31
Impoundment TN	Stream TN	12	0.00	0.00	+	0.64
Impoundment TSS	Stream TSS	12	0.41	0.00	+	0.07
Impoundment TP	Impoundment VSS	11	0.00	0.97	+	0.81
Impoundment TDP	Impoundment VSS	12	0.00	0.19	+	0.69
Impoundment TP	Impoundment TSS	11	0.00	0.00	+	0.70
Impoundment TDP	Impoundment turbidity	13	0.00	0.00	+	0.55
Impoundment TDN	Impoundment VSS	9	0.00	0.02	+	0.87
Impoundment TN	Impoundment VSS	12	0.00	0.19	+	0.75
Impoundment TDN	Impoundment TSS	12	0.00	0.00	+	.79
Impoundment TN	Impoundment TSS	13	0.00	0.01	+	0.58

A number of significant robust regression models were found in which both phosphorus and nitrogen variance could be explained by TSS or VSS (Table 4). The best models for impoundment phosphorus were generated when impoundment VSS or TSS was used as the independent with as much as 81% of the variance in TP being explained by VSS alone. Similarly a portion of the variance in impoundment TN could be explained with impoundment VSS and TSS. The model with the highest R^2 (0.87) was the model where the dependent was TDN and VSS was the independent variable. In addition VSS and TSS were both good predictors of impoundment TN. No significant TN model could be found when impoundment turbidity was used as the independent variable.

Impoundment sediment

Lake sediment core samples were taken at one or two sites per lake and analyzed in 10cm segments. Only % silt in all the core segments differed based on reference condition (p=0.03). The remainder of the parameters did not differ between reservoir groups (reference vs. non-reference) amongst either all the depths of the cores or when restricted to just the first 10 cm (Table 5).

Table 5. Results of Analysis of Variance (ANOVA) on core samples from impoundments draining reference/non-reference watershed ecosystems as the treatment groups. Significant differences ($p<0.05$) between treatment groups were found for those parameters in bold. A filter was also applied to restrict analysis to the first 10 cm of the sediment cores. The last column "Difference in mean values" shows actual differences in non-reference mean values when compared to reference mean values for significant models where + indicates and increase and – a decrease in mean values

Parameter	n	p	F-ratio	Filter	Difference in mean values
Bulk Density	26	0.82	0.05	none	
% clay	26	0.20	1.78	none	
% silt	**26**	**0.03**	**5.13**	**none**	**–12.2 %**
% sand	26	0.28	1.21	none	
TN	26	0.13	2.40	none	
TP	26	0.09	3.19	none	
Bulk Density	10	0.86	0.03	top 10 cm	
% clay	10	0.47	0.57	top 10 cm	
% silt	10	0.21	1.86	top 10 cm	
% sand	10	0.28	1.32	top 10 cm	
TN	10	0.43	0.69	top 10 cm	
TP	10	0.27	1.39	top 10 cm	

Interestingly, Banner Reservoir sediment had a significantly higher percentage of silt than the non-reference impoundments thus the negative 12.2 % in mean values for silt between reference and non-reference groups (Table 5), although there was considerable spread in the silt values within treatments (Fig. 10). This significant difference in silt did not occur when considering only the upper 10 cm of the core length. Banner Reservoir is a much younger impoundment than the non-reference impoundments, which might have affected the overall contribution of silt to the cores that were taken.

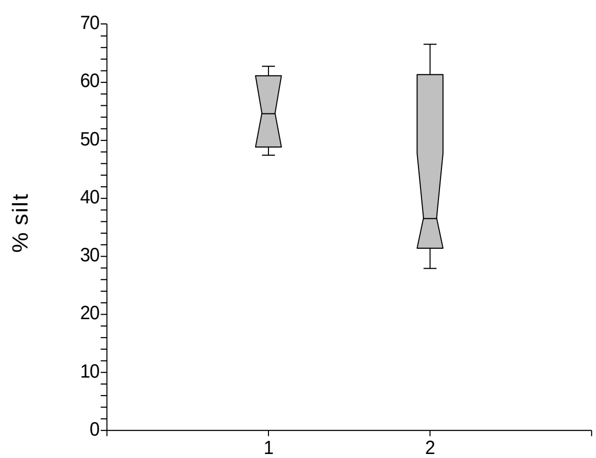

Figure 10. Box plot of % silt in core samples from reference (1) and non-reference (2) impoundments.

We also tested (one-way ANOVAs where seasonal data was combined) for differences between impoundment core locations (i.e. main basin vs. riverine segment). ANOVA tests on total core values indicated that there were significant differences between basin and riverine mean values for all parameters listed in Table 4. It appears that TP, TN, and % clay values are higher in the basin while mean values for bulk density, % silt, and % sand were higher in the riverine segment.

Analyses of relationships within sediment core properties revealed several interesting relationships between physical properties of the sediment and nutrients. Bulk density was positively related to the percent of sand and silt but negatively related to clay (Table 6). The best predictor of sediment TP and TN was % clay, then bulk density. While % clay and bulk density are highly related to each other, the % clay explained more of the variance of both TP and TN. Phosphorus is often observed bound to clay particles in runoff and stream flows (e.g. Ulen 2003, Schroeder *et al.* 2004). Danish researchers (de Jonge *et al.* 2004) found that particulate inorganic phosphorus (PIP) positively correlated with clay content while dissolved inorganic phosphorus (DIP) was negatively related to clay, suggesting that the sediment TP in our impoundments might be mostly PIP.

Table 6. Robust regression information for significant sediment core models (alpha ≤ 0.05) of bulk density and nutrients.

Dependent variable	Independent variable	n	Model p value	Intercept p value	Relationship	R^2
Bulk density	% Sand	26	0.00	0.00	+	0.76
Bulk density	% Silt	25	0.00	0.00	+	0.82
Bulk density	% Clay	25	0.00	0.00	–	0.83
TP	Bulk density	25	0.00	0.00	–	0.90
TP	% clay	26	0.00	0.01	+	0.94
TN	Bulk density	26	0.00	0.00	–	0.71
TN	% clay	25	0.00	0.02	+	0.88

No meaningful relationships (i.e. robust regression) or correlations (i.e. Pearson's correlation coefficient) were found between impoundment water chemistry and core chemistry. Stream chemistry also was not related to core chemistry except for some weak correlations ($r \leq 0.61$) between stream TDP and % clay (positive) and silt (negative). Though clay and silt have both been related to particulate phosphorus, it is difficult to determine whether those relationships extend to dissolved phosphorus in these systems given the limited number of samples available.

Stream sediment volumes

Originally we calculated only the V* values for each stream segment studied, but later we also calculated two other related variables. This was done to investigate the inter-relationships between these variables and their potential relationship(s) other stream factors. These stream bed variables were V* (the calculated ratio of reach sediment volume to total reach volume), A* (the calculated ratio of cross-section sediment area to cross-section total area), and the final A*ave value (average of ratios for all cross-sections in a particular stream segment). We expected these variable values to be lower in the reference watershed stream (Banner Creek) assuming that this stream would receive less sediment input and retain less sediment in the wetted channel. As expected, V* and A*ave were highly correlated ($r = 0.83$, $p = 0.00$), but both were used to explore possible relationships with other stream or lake variables. Interestingly, Banner Creek had the two highest V* and A*ave values which was unexpected but was the result of the inclusion of measures of loose, unconsolidated sand throughout most of the open channel flow areas (Table 7). Consistently high V* values in Banner Creek contributed to a significant one-way ANOVA for treatment effects (reference vs. non-reference) when using V* as the response variable (p = 0.02) but not when A*ave was used (p = 0.09). This may or may not indicate that these two stream bed factors are measuring different bed phenomena.

Table 7. Mean values for V* and A* variables from each sampling events, with events ranked from low to high by V*.

Stream system and site	Reference or Non-reference	Sampling event	V* rank	V*	A* average
Clear Creek Site 1	Non-ref	July	2	0.24	0.28
Clear Creek Site 1	Non-ref	October	1	0.17	0.23
Banner Creek Site 1	Ref	July	10	0.40	0.41
Banner Creek Site 1	Ref	**October**	**11**	**0.48**	**0.66**
Banner Creek Site 2	Ref	July	7	0.32	0.33
Banner Creek Site 2	**Ref**	**October**	**12**	**0.64**	**0.60**
Banner Creek Site 3	Ref	July	9	0.37	0.36
Banner Creek Site 3	Ref	October	5	0.29	0.30
Black Vermillion Site 1	Non-ref	July	8	0.33	0.34
Black Vermillion Site 1	Non-ref	October	3	0.25	0.25
Black Vermillion Site 2	Non-ref	July	4	0.26	0.37
Black Vermillion Site 2	Non-ref	October	6	0.32	0.42

Interpretation of Table 7 and ANOVA results suggest that the reference stream had more unconsolidated material occurring in its wetted channel than did the non-reference stream channels. It should be noted that the material stored in the non-reference stream segments was almost all soft silt while the reference bed materials were mostly sand. If we consider that the historic stream condition in this region was a primarily sandy-bottom substrate, then Banner Creek might still be thought of as a reference stream. However, we did not find difference in variables that, in part, represented suspended sediment and other matter (TSS, VSS, ISS, turbidity), suggesting that these streams do not differ in respect to suspended sediment (see Table 3). Additionally, we expect TSS, VSS, and turbidity to increase with the volume of unconsolidated sediment on the streambed as measured by V* and A* ave. This was not the case as both V* and A*ave were not significantly correlated (Pearson and Spearman correlations, alpha = 0.05) with any of the suspended sediment measures including turbidity.

In summary, V* and A*ave show little relationship to either traditional measures of suspended sediment or reference condition, if Banner Creek is in fact a reference stream with regards to geomorphology and substrate condition.

These stream bed variables did seem to be marginally related to impoundment chemistry (Table 7), but these relationships may not be causal and only two were significant ($p \leq 0.05$). Both mean V* and A*ave were significantly and negatively correlated with TP and turbidity in study impoundments.

Table 8. Pearson correlation coefficients between mean V* and A*ave values and mean impoundment nutrient and turbidity values.

Impoundment parameters	Mean V*		Mean A*ave	
	r	p	r	p
TDP (µg/L)	-0.56	0.25	-0.55	0.26
TP (µg/L)	**-0.90**	**0.01**	**-0.86**	**0.03**
TDN (µg/L)	-0.70	0.12	-0.66	0.16
TN (µg/L)	-0.77	0.08	-0.70	0.13
Turbidity (NTU)	**-0.80**	**0.05**	**-0.74**	**0.09**

We also examined the relationships between mean V* and A*ave and stream nutrients. Using both Pearson (parametric) and Spearman (nonparametric) correlations, we found that both correlation methods indicated that V* was significantly correlated to both stream TP and TN values. Spearman correlations using ranked data found that TDP, TP and TN were significantly correlated with mean V* values (Fig. 11). As with Pearson correlations, significant Spearman correlations were negative in nature with r values that varied from -0.57 (V* and TN) to -0.73 (V* and TP). None of these correlations may represent causal relations, but may only be predictive associations since it is difficult to understand why phosphorus and nitrogen values would rise when V* values decrease or conversely why stream nutrients would decrease with increases in V* values. It might be that TP and TN values go down in streams because of settling of particulates that increase the V* estimates since the TP and TN values were most strongly correlated with V*.

Lastly, the relationship between V* and impoundment TP values might be related to the fact that V* shows the same relation with stream TP and stream values have already been determined to be good predictors of impoundment nutrients, although marginally for TP (see Table 4).

Phytoplankton (impoundments)

A number of phytoplankton metrics were calculated from the impoundment samples taken during the course of this study (Table 9). Because only two samples were available for each impoundment, only a visual comparison of the data was attempted. In general, Banner Reservoir had consistently higher total abundance, taxa richness, and diversity values. The highest Shannon and Brillouin's diversity index values were noted in Atchison in July while the highest total abundance was in Centralia Reservoir. Typically, summer community metric values were higher than those for October samples. Based on taxa richness and consistently higher diversity values, it is tempting to say that Banner Reservoir has a more diverse phytoplankton community compared to the two other impoundments.

Table 9. Community metric values for phytoplankton samples taken during the two periods for reference and non-reference impoundments. All phytoplankton grab samples were taken at the surface (0.25m) at the deepest station within the main basin.

Metric	Atchison		Banner		Centralia	
	13 Jul 2010	7 Oct 2010	14 Jul 2010	7 Oct 2010	13 Jul 2010	6 Oct 2010
Total Abundance	66,039	9,811	1,279,716	439,620	3,171,697	20,440
Taxa Richness	14	9	35	24	14	12
Gleason's Index	2.90	2.25	5.73	4.25	2.15	2.78
Margalef's Index	1.17	0.87	2.42	1.77	0.87	1.11
Menhinick's Index	0.05	0.09	0.03	0.04	0.01	0.08
McIntosh's Index	0.61	0.45	0.47	0.53	0.12	0.57
Simpson's Index	0.15	0.31	0.28	0.22	0.77	0.19
Simpson's Compliment	0.85	0.69	0.72	0.78	0.23	0.81
Simpson's Reciprocal	6.60	3.23	3.55	4.48	1.30	5.22
Shannon's Index (H')	0.95	0.66	0.87	0.87	0.25	0.82
Standard Deviation	5,184	1,547	110,472	39,069	733,716	2,028
Brillouin's Index	0.95	0.66	0.87	0.87	0.25	0.82

When we examined the ratio of cyanobacteria cells to total algal cell counts we found that Banner Creek Reservoir samples had over 75% (i.e.78 – 79 %) of all algal cells that were cyanobacteria taxa. Both non-reference impoundments experienced at least one high cyanobacteria event (i.e. July). The summer sample for Atchison was 33% cyanobacteria while the summer sample for Centralia was 99% cyanobacteria. It appears that high cyanobacteria abundances can be common in all impoundments.

Lastly, a one-way ANOVA test, where data from all dates was used, indicated no significant difference in the mean concentrations of chlorophyll *a* between reference and non-reference impoundments. However, it should be noted that the mean chlorophyll *a* value for Banner Creek Reservoir was 25 µg/L compared to the non-reference mean of 18 µg/L. Robust regression produced three significant models when nutrient variables were used as the independent variable, but neither turbidity, TSS, VSS, or ISS values were found to explain any chlorophyll *a* variance. The best chlorophyll model was with TN as the independent variable. TN explained about 73% of the chlorophyll variability and was negatively related to chlorophyll concentrations. This relationship is difficult to explain biologically and may only represent a correlative agreement between measured variables. The other chlorophyll *a* model of interest was when TP was used as the independent variable and had a R^2 value of 0.39. This model indicated that TP had a negative relationship with chlorophyll similar to the TN model.

Interestingly, stream chlorophyll *a* showed relationships with both stream nutrients and TSS, VSS, and ISS (Table 10). All of the suspended solids models had R^2 values greater than 0.99 and positive relationships with chlorophyll *a*. Dissolved nutrient model R^2 values were not as strong, with total dissolved (i.e. filtered) phosphorus values explaining over 80% of the recorded variation in chlorophyll *a* concentrations and TDN explaining 60%. Typically TP is noted to be the limiting nutrient in aquatic ecosystems in our region which is probably the reason TP produced a more explanative model than TN (filtered).

While TSS, VSS, and ISS are all highly correlated with each other (R^2 >0.99) nearly all of the suspended solids are ISS (e.g. soils, minerals). However, the most likely meaningful biological model is between VSS as an independent measure of organic matter (e.g. algal biomass) and chlorophyll *a* concentration as the dependent variable.

Table 10. Robust regression information between chlorophyll *a*, total nitrogen (TN) and phosphorus (TP), and measures of suspended solids TSS, VSS, and ISS.

Dependent variable	Independent variable	N	Model p value	Intercept p value	Relationship	R^2
chlorophyll *a*	TSS	12	0.0000	0.2149	+	0.9923
chlorophyll *a*	VSS	10	0.0000	0.0267	+	0.9911
chlorophyll *a*	ISS	12	0.0000	0.2483	+	0.9925
chlorophyll *a*	TN (filtered)	10	0.0084	0.2946	+	0.6017
chlorophyll *a*	TP (filtered)	10	0.0003	0.1775	+	0.8197
TSS	VSS	10	0.0000	0.0010	+	0.9988
TSS	ISS	10	0.0000	0.0008	+	1.0000
VSS	TSS	10	0.0000	0.0008	+	0.9988
VSS	ISS	10	0.0000	0.0008	+	0.9987
ISS	TSS	10	0.0000	0.0008	+	1.0000
ISS	VSS	10	0.0000	0.0010	+	0.9987

Zooplankton (impoundments)

Zooplankton community diversity and richness were higher in both non-reference impoundments, although some of these differences were relatively small (Table 11). Atchison "lake" had the highest taxa richness (8-9) while Banner had the lowest (4-6), with Centralia falling in between these values. The more commonly used diversity indices (e.g. Gleason's, Margalef's, Shannon's and Brillouin's) suggested that zooplankton diversity with both non-reference impoundments were higher than those for Banner Creek Reservoir except for the two information-based indices (i.e. Shannon's and Brillouin's), which were slightly higher in July than those in Centralia for this same time period.

Table 11. Community metric values for zooplankton samples taken during the two periods for both reference and non-reference impoundments. All zooplankton tows were taken at the deepest station within the main basin.

Metric	Atchison		Banner		Centralia	
	13Jul 2010	7 Oct 2010	14 Jul 2010	7 Oct 2010	13 Jul 2010	6 Oct 2010
Total Abundance	n/a	n/a	n/a	n/a	n/a	n/a
Taxa Richness	8	9	6	4	7	6
Gleason's Index	3.31	3.70	2.48	1.65	2.90	2.48
Margalef's Index	1.26	1.43	0.90	0.54	1.08	0.90
Menhinick's Index	0.50	0.55	0.37	0.25	0.43	0.37
McIntosh's Index	0.55	0.61	0.45	0.33	0.41	0.37

Simpson's Index	0.24	0.18	0.33	0.47	0.38	0.42
Simpson's Compliment	0.76	0.82	0.67	0.53	0.62	0.58
Simpson's Reciprocal	4.24	5.66	2.99	2.13	2.62	2.36
Shannon's Index (H')	0.73	0.80	0.54	0.40	0.47	0.47
Standard Deviation	33	25	48	71	52	59
Brillouin's Index	0.70	0.77	0.52	0.39	0.46	0.46

n/a = not applicable since only a subsample of about 250 individuals were identified to calculate metrics.

Macroinvertebrates (streams)

Macroinvertebrates were collected from all stream sites while QHEI and HDI were concurrently evaluated. The two highest HDI scores were on Banner Creek, but otherwise no significant HDI relations between reference versus non-reference stream samples were observed (p=0.35). However Banner Creek had significantly higher QHEI scores than the non-reference stream sites (p=0.001). QHEI is an index that is scaled to evaluate fish habitats, thus large stream reaches included large habitat parameters such as percent fish cover, stream depth, canopy cover, while HDI was developed to evaluate macroinvertebrate habitats actually collected as part of the macroinvertebrate sampling process. Thus the HDI is focused on small scale features sampled for macroinvertebrates such as leaf packs, root wads, macrophytes, and algal mats (Huggins and Moffet 1988).

Assuming similar water quality condition exists in all study reaches (which is not true), we would expect most macroinvertebrates indices to mirror habitat indices (e.g. high habitat richness → high taxa richness). As with HDI, one-way ANOVA (seasonal data collections were combined) results revealed that indices did not differ between reference and non-reference sites (alpha = 0.05, Table 12). Black Vermillion site 2 in October had the highest values for taxa richness and also for Gleason’s, Margalef’s and Menhinick’s diversity indices. When this site is filtered from analyses, ANOVA tests indicated that there was a difference in these scores based on reference condition (see filter = yes, Table 12). However, we have no statistical or biological reason for removing this site and date from the analysis other than to show that biologically this non-reference site was more like reference sites.

Table 12. ANOVA tests results for treatment effects (reference condition and non-reference) for stream habitat and macroinvertebrate indices, with count of samples (n), p value, F-ratio, and degrees of freedom (DF). The last column “Difference in mean values” shows actual differences in non-reference mean values when compared to reference mean values for significant models where + indicates and increase and – a decrease in mean values.

Parameter	n	p	F-ratio	filter	Difference in means values
HDI	12	0.35	0.95	none	
QHEI	**12**	**0.00**	**19.93**	**none**	**+14.0**
Gleason's	12	0.20	1.91	none	
Margalef's	12	0.20	1.90	none	
Menhinick's	12	0.18	2.10	none	
Richness:Abundance	12	0.16	2.34	none	
Taxa richness	12	0.22	1.69	none	
HDI	11	0.24	1.58	yes*	

QHEI	**11**	**0.00**	**15.49**	**yes***	**+13.3**
Gleason's	**11**	**0.01**	**12.63**	**yes***	**+3.3**
Margalef's	**11**	**0.01**	**12.63**	**yes***	**+1.4**
Menhinick's	**11**	**0.01**	**12.43**	**yes***	**+0.12**
Richness:Abundance	**11**	**0.01**	**11.38**	**yes***	**+3.0**
Taxa richness	**11**	**0.01**	**11.46**	**yes***	**+8**

* Excluded Black Vermillion Site 2 for October sample period.

Examination of box plots (Fig. 11) for selected macroinvertebrate metrics indicates that there was considerable variability within treatment groups (i.e. reference, non-reference) which may have been due to the combining of seasonal samples as well as within treatment group macroinvertebrate habitat variability. The relationship between HDI scores and most diversity and richness measures is often quite strong and positive for indices that have positive scales (see Fig. 12).

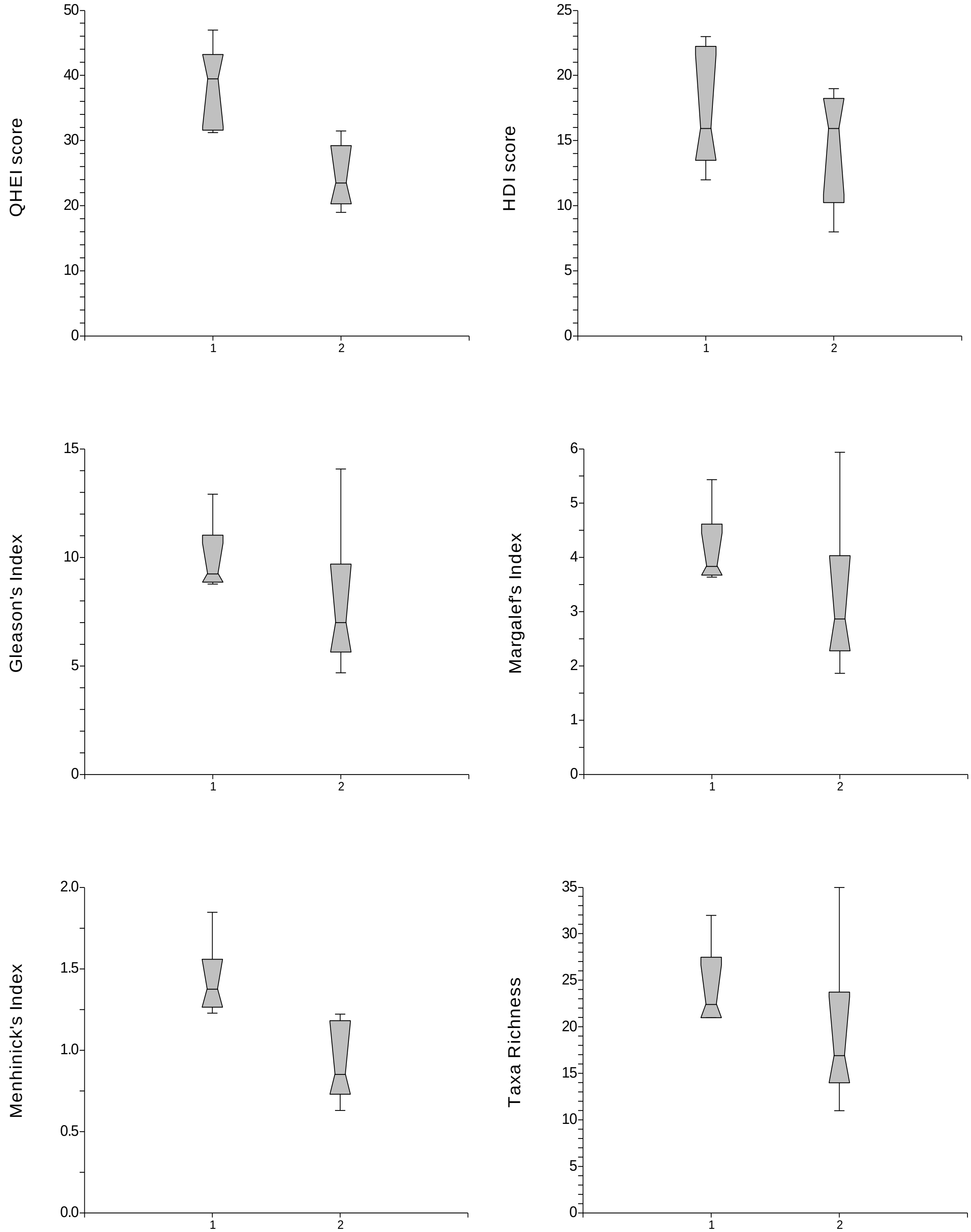

Figure 11. Box plots of habitat and selected macroinvertebrate metrics for stream sites grouped by reference (1) and non-reference (2) watersheds.

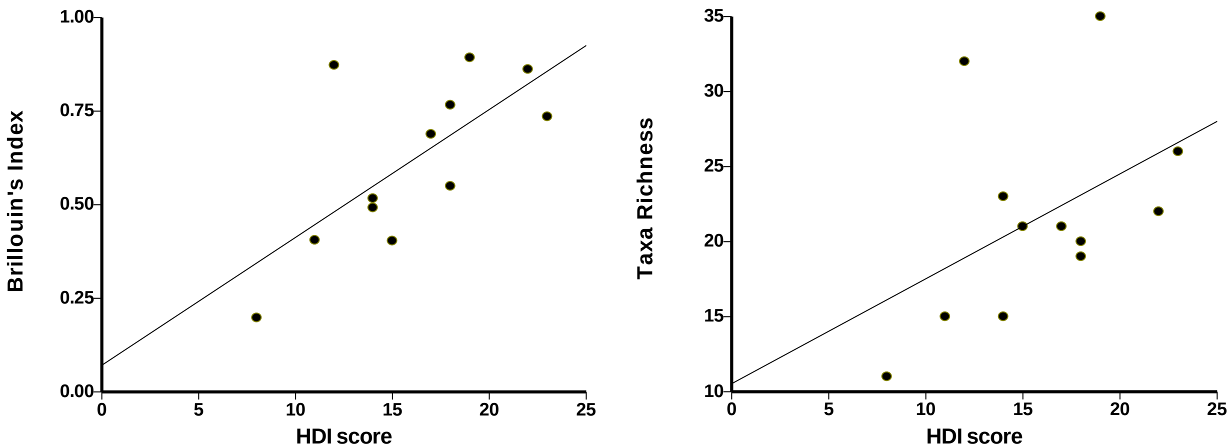

Figure 12. Scatter plots with linear trend lines for HDI habitat scores and Brillouin's diversity index scores or taxa richness for all stream sites and dates.

It has already been noted that significant differences for most macroinvertebrate metric values occurred between reference and non-reference stream groups <u>if</u> one of the non-reference site values was removed from the analysis (Table 12). The means for these metrics indicated that the reference stream often had higher taxa diversity and more taxa than did the non-reference streams. These results also suggested that for the most part there were little or no overall habitat differences between these treatment groups. Robust regression analyses indicated that these macroinvertebrate metric differences were not related to V*, A*ave, TSS, VSS, or turbidity, but were significantly related to stream nutrients (Table 13). TDP was found to be a significant independent variable in only two of the three metrics listed in Table 13. Both the Taxa Richness and Gleason's Index models included TDP but similar models using TP were produced, however, these models fall short of being significant ($p \approx 0.06$). Most of these regression models were models that identified both TN and TDN as a significant independent variable that explains 50 – 60% of the variance in richness, Gleason's, and Shannon's diversity index values. All simple regression models generated in examining the relationships between macroinvertebrates and nutrients indicated that increases in either nutrient resulted in decreases in diversity and richness.

Several multiple regression models that included both phosphorus and nitrogen as independent variables were found to be significant; however in all cases one of the independent variables was noted not to be a significant variable causing us to reject the model outcomes. Even if these multiple regression models were considered biologically significant, they explained little additional variance in the macroinvertebrate metric values (≤ 8% increase). It appears that while macroinvertebrates richness and diversity is adversely impacted by stream nutrient levels found in this study, there are no clear relationships between these organisms and any measure of sediment either suspended or incorporated on the stream bed.

Table 13. Robust regression information for significant models (alpha = 0.05) where macroinvertebrate metrics are the independent variable.

Dependent variable	Independent variable	n	Model p value	Intercept p value	Relationship	R^2
Taxa Richness	Stream TDP	12	0.01	0.00	–	0.48
Taxa Richness	Stream TN	12	0.00	0.00	–	0.65
Taxa Richness	Stream TDN	12	0.00	0.00	–	0.62
Gleason's Index	Stream TDP	12	0.00	0.00	–	0.56
Gleason's Index	Stream TN	12	0.00	0.00	–	0.65
Gleason's Index	Stream TDN	12	0.00	0.00	–	0.65
Shannon's Index	Stream TN	12	0.02	0.00	–	0.45
Shannon's Index	Stream TDN	12	0.01	0.00	–	0.49

However, it must be remembered that these finding like all other findings in this study should be viewed with care and linked back to other published works in this field of study due to the limited numbers of spatial and temporal samples available for use in this study. In addition, from an ecological perspective the study design adopted for this study has its limitations and did not fully allow researchers to use more definitive analytical approaches. It may well be that all study ecosystems are impacted to a degree where distinguishing subtle differences was not possible with our limited sample size and study design.

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Appendix 1. Photos of Atchison, Banner, and Centralia Lakes showing approximate CPCB stream sampling sites.

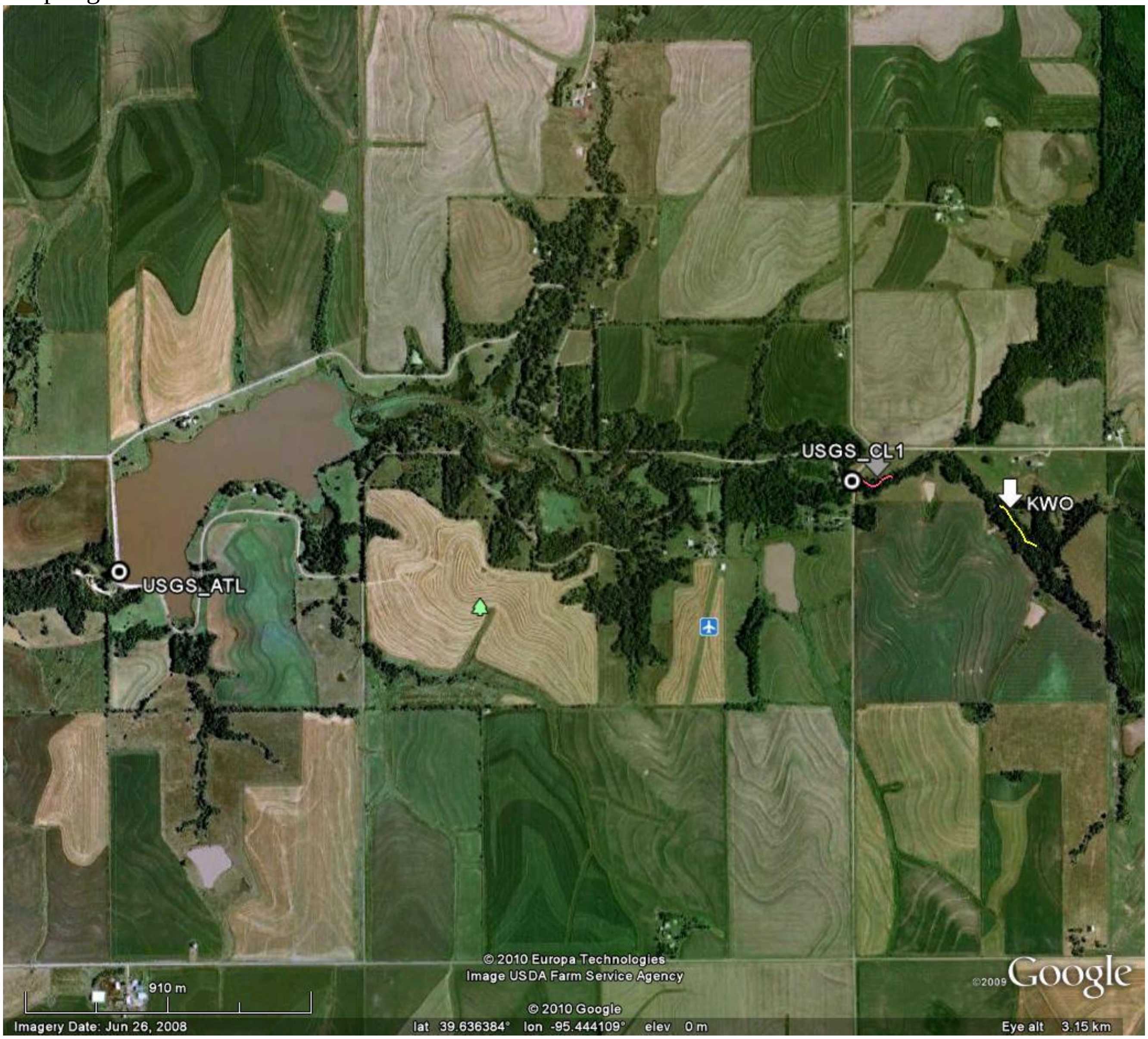

Figure 1. Atchison County Lake showing USGS gaging stations ATL and CL1 and CPCB's (pink) and KWO's (yellow) survey reaches on Clear Creek.

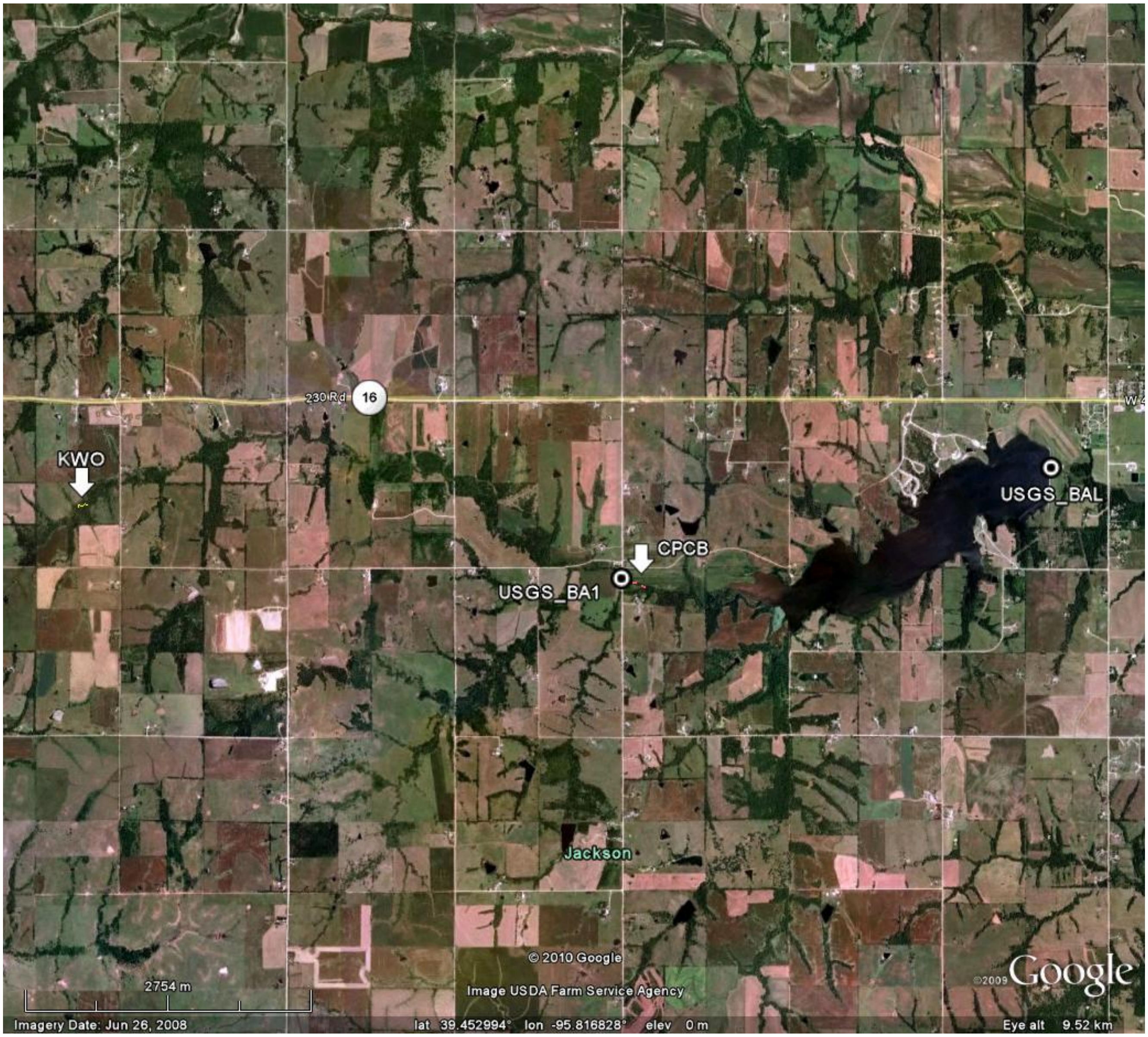

Figure 2. Banner Creek Lake showing USGS gaging stations BA1 and BAL and CPCB's (pink) and KWO's (yellow) survey reaches on Banner Creek.

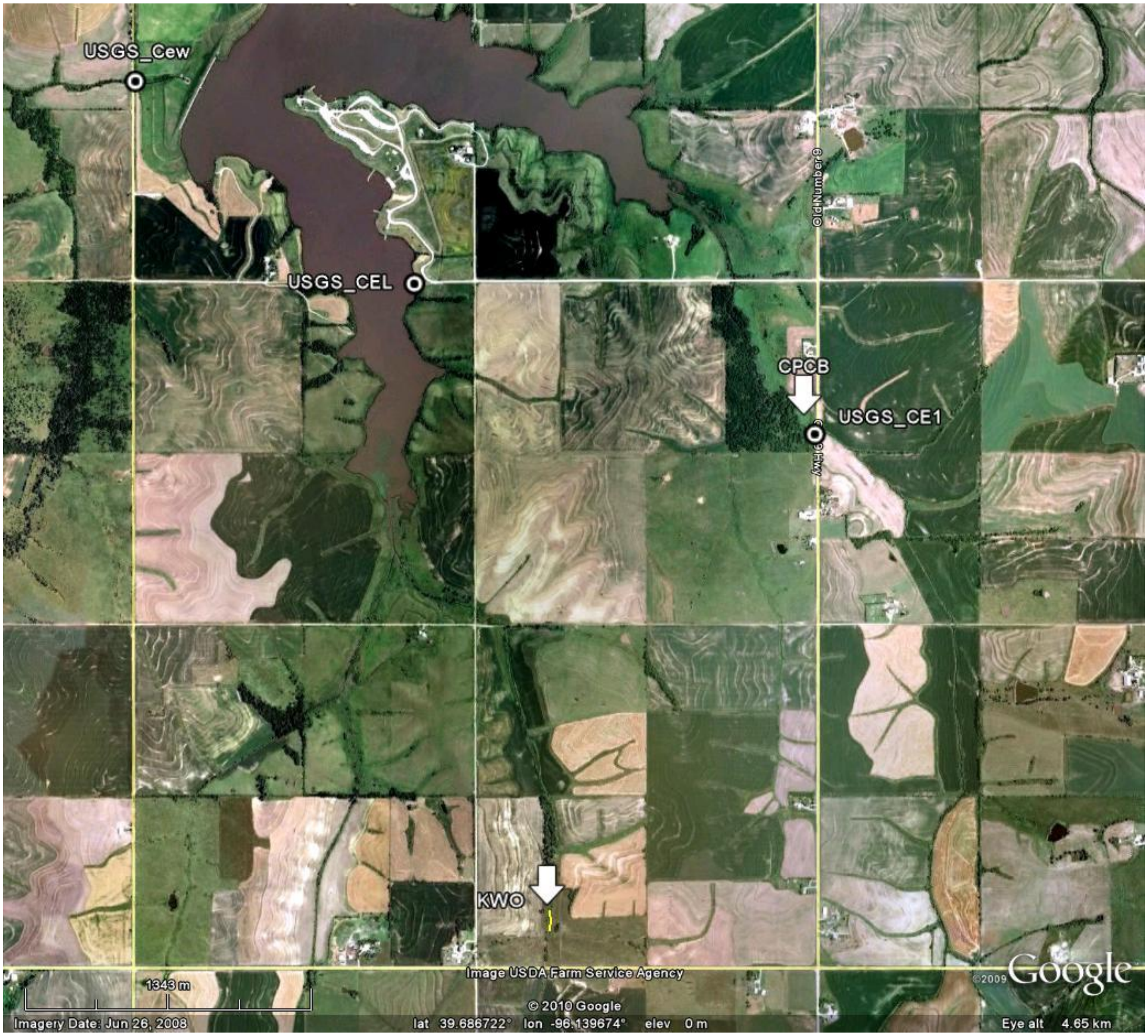

Figure 3. Centralia Lake showing USGS gaging stations CE1, CEL, and Cew, and CPCB's (pink) and KWO's (yellow) survey reaches on the Black Vermillion River.

Appendix 2. Sediment V* form modified by the Central Plains Center for BioAssessment.

Project________________ **Date**________________ **Stream** ____________________
site_________

Crew __________________________ Transect 1 latitude _____________ longitude _____________
Dec.degrees, circle: NAD83 or WGS84

A. Reach length (m) _______
B. Intended distance between transects (m) _______
C. # of transects _______ (Tally at end)

Site comments and sketch (indicate flow, center line, angles from center line, etc.)

Transect measurements

Transect # 1	Distance from transect #1 = 0 m							Center line (string) to left bank (cm)						
Measurements (cm)	1	2	3	4	5	6	7	8	9	10	11	12	13	14
Distance from left bank														
1. Water depth														
2. Fines depth														
3. Detritus layer depth*														
4. Dominant substrate														
comments														

Transect #	Distance from previous transect (m) =								Center line (string) to left bank (cm) =					
Measurements (cm)	1	2	3	4	5	6	7	8	9	10	11	12	13	14
Distance from left bank														
1. Water depth														
2. Fines depth														
3. Detritus layer depth*														
4. Dominant substrate														
comments														

* If a fines deposit has an organic or detritus layer on it (leaves, sticks, etc), estimate the depth of the detritus layer.

Appendix 3. Nutrient, Total Suspended Solids (TSS), Volatile Suspended Solids (VSS), and other values by site and sampling event in 2010.

Table A. Impoundment values for nutrients, TSS, and VSS for the main sampling sites.

Impoundment	Sampling Event	Site	Total Dissolved Phosphorus (µgP/L)	Total Phosphorus (µgP/L)	Total Dissolved Nitrogen (µgN/L)	Total Nitrogen (µgN/L)	Total Suspended Solids (TSS) (mg/L)	Volatile Suspended Solids (VSS) (mg/L)	Inorganic Suspended Solids (ISS) (mg/L)	Turbidity (NTU)
Atchison	Jul	9	320.0	345.0	1590.0	1900.0	23.5	0.4	23.1	127.0
Atchison	Oct	9	100.0	405.0	2040.0	2270.0	82.0	1.4	80.6	373.0
Banner Creek	Jul	1	27.3	104.0	759.0	910.0	27.0	0.8	26.2	58.0
Banner Creek	Jul	9	14.0	93.3	658.0	902.0	8.7	0.5	8.2	14.0
Banner Creek	Oct	1	14.3	67.6	663.0	882.0	7.0	0.4	6.6	44.0
Banner Creek	Oct	9	14.2	51.1	528.0	678.0	3.0	0.5	2.5	23.0
Centralia	Jul	1	161.0	213.0	1200.0	2100.0	91.0	1.6	89.4	157.0
Centralia	Jul	6	59.3	111.0	987.0	1870.0	23.0	1.2	21.9	46.0
Centralia	Oct	1	58.7	182.0	2700.0	2780.0	132.0	1.4	130.6	268.0
Centralia	Oct	6	71.7	113.0	2690.0	2840.0	14.0	0.7	13.3	52.0

Table B. Impoundment core values for measured parameters.

Impoundment	Sampling Event	Site	Depth Range (cm)	Bulk Density (g/cm^3)	Clay %	Silt %	Sand %	Total Nitrogen (ppm)	Total Phosphorus (ppm)
Atchison	July	9	0-10.0	0.4	65.2	32.7	2.2	1999.0	1065.6
Atchison	July	9	10.0-20.0	0.4	65.7	32.2	2.1	2181.8	905.6
Atchison	July	9	20.0-31.0	0.4	70.6	27.9	1.5	1816.0	853.0
Atchison	October	9	0-10.0	0.3	69.7	29.7	0.6	2401.7	931.0
Atchison	October	9	10.0-20.0	0.4	69.3	30.1	0.6	2437.4	961.1
Atchison	October	9	20.0-30.0	0.4	70.0	28.8	1.1	2285.2	821.8
Banner Creek	July	1	0-10.0	0.7	36.3	59.8	3.9	1144.1	554.5

Banner Creek	July	1	10.0-21.0	1.0	35.7	62.7	1.6	984.7	502.5
Banner Creek	July	9	0-10.0	0.3	46.9	48.6	4.5	1772.4	687.8
Banner Creek	July	9	10.0-19.0	0.5	47.5	47.4	5.0	1223.3	664.0
Banner Creek	October	1	0-10.0	0.8	33.0	61.4	5.6	1252.2	481.8
Banner Creek	October	1	10.0-20.5	1.1	31.3	60.1	8.6	1105.6	467.1
Banner Creek	October	9	0-10.0	0.5	46.2	49.6	4.1	2032.3	629.7
Banner Creek	October	9	10.0-16.0	0.5	47.1	49.4	3.5	1787.7	627.4
Centralia	July	1	0-10.0	1.0	23.1	61.5	15.4	975.9	395.5
Centralia	July	1	10.0-20.0	1.4	26.8	58.1	15.0	1054.3	476.6
Centralia	July	1	20.0-21.5	1.8	25.5	61.2	13.3	949.6	413.3
Centralia	July	6	0-10.0	0.2	57.0	31.8	11.2	2213.2	1005.8
Centralia	July	6	10.0-20.0	0.3	58.4	36.4	5.1	1787.2	941.1
Centralia	July	6	20.0-30.0	0.4	60.5	36.9	2.7	1829.1	942.3
Centralia	October	1	0-10.0	1.1	20.7	66.5	12.8	918.4	341.7
Centralia	October	1	10.0-20.0	1.0	23.5	63.2	13.4	1037.8	346.7
Centralia	October	1	20.0-23.0	1.1	20.8	61.9	17.3	949.9	383.8
Centralia	October	6	0-10.0	0.2	59.6	34.2	6.2	3106.7	1003.0
Centralia	October	6	10-20.0	0.3	61.3	37.0	1.8	2643.0	884.4
Centralia	October	6	20.0-24.5	0.4	57.8	37.7	4.5	2057.9	864.9

Table C. Stream values for nutrients, TSS, VSS and turbidity.

Watershed	**Sampling Event**	**Site**	**Total Dissolved Phosphorus (µgP/L)**	**Total Phosphorus (µgP/L)**	**Total Dissolved Nitrogen (µgN/L)**	**Total Nitrogen (µgN/L)**	**Total Suspended Solids (TSS) (mg/L)**	**Volatile Suspended Solids (VSS) (mg/L)**	**Inorganic Suspended Solids (ISS) (mg/L)**	**Turbidity (NTU)**
Atchison	Jul	9	241	344	1990	2360	51.00	0.90	50.10	177
Atchison	Oct	9	162	208	2680	2950	3.00	0.10	2.90	37
Banner Creek	Jul	1	61	124	1430	1500	20.33	0.42	19.92	33
Banner Creek	Jul	9	50.1	59.3	365	544	2.00	0.20	1.80	3

Banner Creek	Oct	1	47.1	103	1490	1520	20.20	0.34	19.86	43
Banner Creek	Oct	9	62.6	62	303	453	6.00	0.20	5.80	2
Centralia	Jul	1	66	93.5	1440	1520	20.75	0.35	20.40	105
Centralia	Jul	6	48.7	54.2	189	305	10.00	1.90	8.10	3
Centralia	Oct	1	137	161	3940	4260	13.50	0.25	13.25	22
Centralia	Oct	6	80.7	394	1880	2870	352.00	4.80	347.20	41