The effectiveness of riparian buffer zones to protect Manitoba streams from agricultural impacts

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Abstract

The study area includes the rural municipalities of Dauphin, Gilbert Plains, and Grandview. This is an area that is intensively used for livestock and crop production. Streams running through such areas have the potential to be physically degraded as well as impacted by agricultural pollutants such as fertilizers, pesticides and manure. These impacts may affect the amount and quality of habitat for benthic invertebrates and fish within stream ecosystems. It was the aim of this study to quantify the effectiveness of riparian buffer areas to mitigate the impacts of agriculture on streams. Measures of mitigation included stream channel physical habitat variables, water quality variables in the form of chemical and fecal coliform analyses, and macroinvertebrate community composition. It was hypothesized that there would be a significant difference between discrete reaches of the same stream with and without a riparian buffer zone. Canopy closure was found to be more dense and bank overhang was more developed at mitigated sites. The invertebrate community at mitigated sites had higher proportions of Diptera, and impacted sites had higher proportions of Mollusca and Crustacea. These were the only invertebrate taxa for which a significant difference between sites was detected. There was no significant difference in taxa richness or overall abundance between mitigated and impacted sites. The effects of agriculture in the study area were widespread encompassing large portions of entire watersheds. It is likely that the predominance of nonsignificant results in this study is a reflection of the limited and fragmented nature of riparian areas in the study area. This study sheds light on the prevalence of agricultural impacts in western Manitoba and how fragmented and narrow buffer strips are not adequate forms of mitigation.

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1.0 Introduction

Since the colonization of Canada, agricultural practices have affected the ecological integrity of the mid-western prairies such that only one percent of the original tall grass prairie remains (Scott 1995). The native vegetation of the prairies has been replaced with various forms of agriculture and the towns that are sustained by it (Samson and Knopf 1994). Prior to colonization, the Native American populations of the prairies had subsistence lifestyles largely oriented towards the hunting of bison, with agricultural activities limited to growing a few indigenous plants (Fowler 2003). These anthropogenic influences on the land had a far lesser effect on the prairie ecosystem than did those that were to follow.

The end of the nineteenth century brought an era of settlement to the Canadian prairies. By 1913, the population of early prairie colonizers occupied an area the size of Germany (Mackintosh 1974). Agricultural land use was concentrated to the south-west of Lake Winnipeg and Lake Manitoba due to the sandy and less fertile soil, and the shorter growing season typical of north-eastern Manitoba (England 1936). Along with extensive colonization came new pressures on the land in the form of agricultural and livestock rearing techniques that were highly concentrated as compared to the preceding indigenous practices. Farming became increasingly widespread and industrialized through the decades due to new technologies and the increasing demand for vegetative yield and animal products. This has lead to increased pressures on the environment by the diverse machinery and chemicals used in agriculture, as well as the intensification of animal husbandry.

The area between Dauphin and Ethelbert is a zone of delineation between the northern coniferous forest and the agricultural lands of north-western Manitoba and is an area heavily impacted by agriculture. The total area of farmland in Division No. 17, which includes Dauphin, Grandview and Gilbert Plains, is 859,720 hectares. In the 2001Canadian Agricultural Census (Statistics Canada 2001), the top five crops within Division No. 17 were spring wheat, alfalfa, canola, barley, and other types of hay and fodder crops. Land use in this area includes 65 hectares under irrigation, as well as substantial land area subjected to various chemical applications including: fertilizer on 254,798 hectares; herbicide on 246,767 hectares; insecticide on 22,500 hectares; and fungicide on 26, 264 hectares (Statistics Canada 2001). Livestock counts in the Division No. 17 included 175,794 cows and calves, and 14,955 pigs.

Grazing has always been part of the prairie ecosystem: before European settlement, wild ungulates grazed within the carrying capacity of the environment. This was because herds of bison were able to migrate freely across the land thus distributing their impacts over a wider area (Abernethy and Rutherfurd 1998). Livestock compact the soil, eat riparian vegetation and change growing conditions in riparian areas (Abernethy and Rutherfurd 1998) and this is magnified with the intensification of modern livestock practices. Modern grazing strategies are more highly concentrated, and impacts relating to non-point source pollutants and physical forms of aquatic habitat degradation are highly concentrated (Sovell *et al*. 2000). Agouridis *et al*. (2005) reported that water largely controls the grazing patterns of cattle by concentrating animals on riparian edges. Streams affected by livestock are characterized by elevated phosphorus and nitrogen, higher fecal coliform counts, and lower dissolved oxygen (Sovell *et al*. 2000).

Localized impacts to stream communities in the form of agricultural pollutants may affect a larger area by flowing through tributaries and entering lakes. All livestock wastes and agricultural chemicals have the potential to enter the stream environment, especially under conditions of reduced riparian protection (Pozo *et al*. 1997; Hickey and Doran 2004). In Manitoba, streams running through agricultural lands are tributaries of the many productive lakes in the province. These lakes provide important resources such as drinking water and recreational and commercial fisheries. The ecological integrity of streams is essential to the surrounding area. It is therefore important to consider impacts to the aquatic environment at an ecosystem level. Agriculture has been found to be detrimental to the communities of organisms residing in stream habitats through a variety of chemical, physical and biological mechanisms (Berkman *et al*. 1986; Hickey and Doran 2004). Research to improve land management practices to effectively minimize agricultural pollutants and physical degradation of aquatic environments is needed. Vegetative buffer strips are a widely promoted as an effective technique for the protection of streams against the negative impacts from agriculture (Quinn *et al*. 1997; Hickey and Doran 2004).

Malanson (1993) defines a riparian zone as an ecotone that links terrestrial and aquatic environments, characterized by a high water table and lateral water flow. Riparian vegetation provides shelter and food, reduces bank erosion, filters both surface flow and groundwater before it enters the stream, and provides important instream habitat. The effectiveness of a riparian buffer zone at mitigating agricultural impacts depends on the type of vegetation and the width of the vegetative buffer strip (Barling and Moore 1994). The entire structure of the vegetation within a riparian area, from the stabilizing roots to the canopy of the vegetation, is important to the quality of habitat that a stream provides for its various inhabitants.

The community of benthic macroinvertebrates within streams is important to community dynamics of the adjacent riparian zone. The emergence of adult invertebrates originating from earlier aquatic stages constitutes 25-100% of the organic carbon utilized by riparian predator species (Baxter *et al.* 2005). This demonstrates only one of the many, important riparian links between the terrestrial and aquatic environments. It is therefore important to consider riparian zones and their adjoining watersheds as systems with reciprocal influences.

The removal of riparian vegetation may cause physical forms of degradation such as increased sedimentation and changes to stream bank and stream channel morphology (Knapp and Matthews 1996). There may also be an alteration of flowing waters and depletion of aquifers (Samson and Knopf 1994), by lowering the water table and reducing the amount of water available to streams (Karr *et al*. 1985). The erosion of stream banks also affects the productivity of the adjacent waterway (Platts 1991). The decrease in primary production caused by increased sedimentation resulting from destabilization of the riparian zone has cascading effects through depletion of food resources of the zooplankton, mollusks, insects and fish (Bain *et al.* 1999; Henley *et al.* 2000).

Riparian zones that are not affected by agriculture in western Manitoba are generally characterized by native grasses with a well developed rhizosphere. Riparian zones in this condition are more capable of decreasing nutrients such as nitrogen and phosphorus by sequestering these nutrients into the structure of plants, thus reducing their concentrations in the riparian soil (Montgomery and MacDonald 2002). Other mitigative effects provided by the riparian zone are the filtration of surface runoff filtration and the riparian vegetation acting as a physical barrier restricting livestock access to stream perimeters (Quinn *et al.* 2001). When a riparian zone no longer has the deep, binding root masses of riparian vegetation, there may be increased inputs into the aquatic environment (Abernethy and Rutherfurd 1998).

Organic pollution entering streams may have a number of detrimental effects on aquatic communities. Inputs to the stream environment change drastically when a riparian zone is removed. The combination of base flow and over-bank flows during precipitation events leads to inputs from the riparian zone when a sufficient riparian buffer is absent (Newson and Newson 2000). Pollutants existing in the riparian area, including agricultural pesticides and fertilizers, may enter the aquatic environment through such patterns of flow. Fine suspended sediments that enter aquatic environments through runoff have high affinities for adsorption of nutrients and toxic material and are often considered to be a main cause of deterioration in freshwater habitats (Rosenberg and Resh 1993).

The stabilization of the banks achieved by riparian vegetation is also vital to the water quality of streams by preventing erosion and sedimentation (Abernethy and Rutherfurd 1998; Moret *et al.* 2006). Tree roots contribute to the shear strength of banks and minimize the occurrence of the gravitationally-induced mass movement of the banks (Abernethy and Rutherfurd 1998), which is known as mass wasting. Grasses, a natural feature of in the riparian zones of the prairies, are especially important for building bank stability which significantly reduces erosion, which is the gradual wearing away of the bank material (Platts 1991). Erosion and mass wasting destroys important in-stream habitat by reducing bank overhangs and infilling interstitial spaces in the substrate (Hauer and Lamberti 1996; Henley *et al*. 2000). Areas of the stream that are beneath bank overhang are essential to stream organisms such as invertebrates and fish by providing microhabitats that are more protected (Holtrop and Fischer 2002). Thus, the presence of riparian vegetation is essential to the physical stability and biological integrity of stream ecosystems.

The presence of coarse organic material in the stream channel is important to the heterogeneity of the stream ecosystem (Gordon *et al.* 2003). Organic matter in the form of leaf litter and other coarse detritus may enter a stream by water, wind, or direct deposition. Large woody debris (LWD) enters streams as the riparian vegetation is affected by processes such as geological shifts, windthrow, fire, insect attack, disease, and competition (Harmon *et al*. 1986). Large woody debris is important for creating microhabitats, controlling hydraulic connectivity, contributing to the heterogeneity of flow (pools, riffles, runs), filtering other large debris, and accelerating stabilization of the mid-channel (Tabacchi *et al*. 2000). If such inputs are significantly reduced, the stream becomes degraded and becomes channelized (Platts 1991). Channelization of the stream results in a reduction in hydraulic heterogeneity and increases the amount of fine sediment found at the streambed (Holtrop and Fischer 2002).

Riparian vegetation is a direct source of allocthonous organic matter inputs, and its loss may significantly affect stream metabolism. Allocthonous inputs are those originating from outside the stream boundaries, and they may be a significant source of carbon for stream metabolism in lower order streams (Delong and Brusven 1998; Newson and Newson 2000). The inputs of such organic matter are greater in autumn due to the shedding of leaves from the trees close in proximity to streams. Alterations in allochthonous inputs change the amount and character of food present for benthic macroinvertebrates (Hauer and Lamberti 1996). Shifts in allocthonous resources result in shifts in the functional feeding categories present in the benthic macroinvertebrate community (Baxter *et al.* 2005). Allochtonous inputs also provide organic material for heterotrophic organisms such as denitrifying bacteria (Tabacchi *et al*. 2000), which mitigate agricultural inputs by transforming nitrogen into nitrous oxide (Dandie *et al*. 2007) thus reducing and releasing nitrogen inputs from the aquatic environment.

The degree of canopy closure provided by riparian vegetation affects the amount of solar radiation reaching the surface of a stream (MacDonald *et al.* 2003) and is important in regulating water temperature. Water temperature and the amount of light that reaches the surface of the water are among the most important habitat variables in a stream. Temperature naturally fluctuates within a reach, especially when patches of riparian vegetation are present (Hauer and Lamberti 1996; MacDonald *et al*. 2003). This contributes to the creation of microhabitats throughout the reach, which increases the overall biodiversity of the stream community (Steinman *et al.* 2003). The presence of even a narrow riparian zone can significantly reduce water temperature; however, buffer strips that are a minimum of 30-100 meters are most effective (Hickey and Doran 2004). When a riparian buffer zone is removed, both average temperature and diurnal fluctuations in temperature are generally increased (Hauer and Lamberti 1996; MacDonald *et al*. 2003). The increased incidence of solar radiation arriving at the stream surface also significantly affects the amount of primary production within the stream reach (Platts and Nelson 1989; Fletcher *et al.* 2000), in turn affecting the type and amount of food that is available for the primary consumers in the stream and thereby the functional feeding categories of benthic invertebrates present (Baxter *et al*. 2005).

Community composition of benthic macroinvertebrates is an effective biological indicator of stream conditions. Benthic invertebrates are effective indicators because they are widespread, there are many different species, they have relatively short lifespans, they are sedentary, and there is a great deal of information known about their environmental preferences and tolerance (Rosenberg and Resh 1993; Hauer and Lamberti 1996; Kleine and Trivinho-Strixino 2005). A wide variety of measures of benthic macroinvertebrate community composition have been used including: abundance, taxa richness, diversity, biotic metrics and multivariate analyses of species composition (Rosenberg and Resh 1993; Lydy *et al*. 2000).

Total abundance is the simplest measure of the invertebrate community and is expected to decrease in severely affected streams (Hawkins 1982). Richness, which is the number of taxa in a community, is a common measure of macroinvertebrate community structure, and is expected to decrease along with the physical and chemical degradation of water quality (Merritt and Cummins 1996). Measures of community diversity, such as Simpson’s index, combine measures of richness with information regarding relative abundance of each taxon (Magurran 1988; Lydy *et al*. 2000) and this is expected to decline with increasing agricultural pollutants (Giulio *et al.* 2001) Invertebrate biotic metrics are based on the abilities of specific taxonomic groups to tolerate environmental stressors. For example, biotic metrics have been used to detect stressors such as organic pollution (Hilsenhoff 1998; Lenat 1993), changes in pH (Davy-Bowker *et al.* 2005) and metals (Hoiland and Rabe 1992; Hickey and Clements 1998).

Biotic metrics use the presence and/or abundance of sensitive or tolerant taxa to indicate environmental conditions (Rosenberg and Resh 1993; Hauer and Lamberti 1996; Kleine and Trivinho-Strixino 2005). The proportion of invertebrates belonging to the EPTs is a metric that is commonly used as an indicator of pollutants because these taxa are considered to be relatively intolerant (Fritz *et al.* 1999), and are therefore expected to be less abundant in physically and chemically degraded environments. The intolerance of these groups is due to their respiratory strategy, which requires the passing of relatively highly oxygenated water over their gills (Merritt and Cummins 1996). Oxygen levels in many instances would be reduced in streams affected by organic pollutants due to respiration of microorganism (Correll 1998).

Benthic macroinvertebrates belonging to the order Diptera have varying abilities to withstand levels of organic pollution. Larval Diptera belonging some Families, such as the Tipulidae, are known to be intolterant of organic pollutant; families such as the Chironomidae, Ceratopongidae, Empididae, Ephydridae, Muscidae, Simuliidae, Tabanidae are known to be moderately tolerant; and families such as the Psychodidae and Syrphidae and some species of Chironomidae are know to be extremely tolerant to organic pollutants (Hilsenhoff 1988). The relative abundance of the Family Chironomidae belonging to the Order Diptera is a commonly used indicator of aquatic pollution and sedimentation because of this group's ability to survive in low oxygen environments (Rosenberg and Resh 1993) and their affinity for the additional soft substrate provided by fine materials (Berkman *et al.* 1986). Some of the species within the Family Chironomidae have hemoglobin in their bodies, which allows then to survive in anaerobic conditions (Merritt and Cummins 1996). Quinn *et al*. (1997) reported that macroinvertebrate total density was three times larger in agriculturally-impacted streams due to increased densities of tolerant taxa such as Chironomidae.

Other benthic macroinvertebrates, such as those belonging to the Subclass Oligochaeta, are used as indicator organisms for the presence of organic pollutants due to their abilities to survive in environments with lower dissolved oxygen (Rosenberg and Resh 1993). The Subphylum Crustacea and Phylum Mollusca are also commonly used as an invertebrate metric and have been identified as being intolerant to acidification and organic pollution, for the most part with some families exhibiting higher levels of tolerance (Rosenberg and Resh 1993; Hauer and Lamberti 1996). Within the Phylum Mollusca, the Families Physidae and Sphaeridae have higher tolerances to organic pollution, while those belonging to the Lymnaeidae are known to have a moderate tolerance level (Hauer and Lamberti 1996).

When a riparian zone is removed from a stream there is a negative impact on the community of organisms due to a dramatic change in habitat (Hauer and Lamberti 1996; Baxter *et al.* 2005). Impacts that may occur due to agricultural activities include increases in sedimentation, organic matter, nutrients and contaminants, and reductions in dissolved oxygen. A study looking at effects of cattle grazing on instream environments and benthic macroinvertebrate community composition found that sedimentation and the proportion of organic matter in the substrate had a significant impact on benthic macroinvertebrate communities (Braccia and Voshell Jr. 2006). The macroinvertebrates remaining in a stream following replacement of riparian zones with agriculture are those that are less sensitive to these stressors (Braccia and Voshell Jr. 2006; Delong *et al.* 1998). For example, Quinn *et al.* (1997) reported that agriculturally-impacted streams had densities of larval EPTs that were three times less than those observed in forested streams. As a result of the loss of sensitive species, impacted streams have a more homogenous taxonomic assemblage compared with communities in streams that are more protected from agricultural inputs (Delong and Brusven 1998; Kleine and Trivinho-Strixino 2005).

The detrimental effects of agriculture on soil nutrients and terrestrial habitat quality are known to be widespread in grassland ecosystems (Hopkins and Holtz 2006). However, our knowledge of how riparian vegetation may potentially mitigate agricultural impacts is somewhat more limited. The objective of this study was to quantify the effectiveness of riparian vegetation to mitigate the impacts of agriculture on streams and to determine whether riparian buffer strips should be promoted at tools for mitigating agricultural impacts. The measures of effect that were assessed included stream channel physical habitat such as bank overhang and substrate composition, water quality in the form of chemical and fecal coliform analyses and macroinvertebrate community composition. It was hypothesized that there would be a significant difference between discrete reaches of the same stream with and without riparian buffer zones. For ease of description these sites were referred to as mitigated and impacted, for those sites having riparian or no riparian zones, respectively.

2.0 Methods

2.1 Study Area and site selection

The study area includes the Red and part of the Assiniboine River valleys within the rural municipalities of Dauphin, Grandview, and Gilbert Plains. Many of the waterways in the study area are affected by culverts and have been channelized. All of the sites were selected outside of the Riding Mountain National Park and Duck Mountains Provincial Park boundaries because a preliminary assessment determined that sites of the same stream order in and out of park boundaries were very different in altitude and channel gradient, which would have been significant confounding factors. Sampling sites were selected based on their proximity to each other, distance from obvious sources of confounding impact (i.e. roads and culverts), and habitat characteristics. All streams were third order. The majority of the sites that were visited during a preliminary survey had substrates of sand, gravel and cobble and therefore streams that had entirely muddy bottoms were not included to ensure that the same sampling device could be used at all sites. The study utilized a paired comparison design to determine whether the presence of a riparian buffer zone was effective in mitigating agricultural impacts. A paired design essentially tests to see if the difference between impacted and mitigated sites on the same stream is consistent between streams. The use of a paired design is similar to blocking in that it removes confounding variability due to between-stream differences. Each stream had a mitigated and an impacted site, for a total of twelve sites. These were selected based on the presence or absence of an established riparian buffer zone. All streams were found on the Valley River (Appendix A) and Wilson River (Appendix B) drainage system.

Potential sites were first identified by examining topographic maps and GIS data on the Department of Fisheries and Oceans Canada GIS website (http://intraca.dfo-mpo.gc.ca/hfomgeomatics/PrairiesArea/index_e.htm). This allowed for the viewing of patterns of channelization, the locations of roadways and the dominant land use on the area. Potential sites were then visited in order to select sites along the same streams that were characterized by either having a riparian buffer zone present or absent. A minimum of fifty meters was used as the minimum distance from roads, culverts, and other sites of the same stream to avoid potential influences associated with these factors. This distance was chosen arbitrarily and was decided upon for practical reasons within the study area. Mitigated sites were relatively rare and could not be consistently located upstream of impacted sites. This placement also removed any potential confounding influence of the natural changes in biota that are expected as a result of downstream progression (Vannote *et al.* 1980)All sites were sampled within a period of approximately twelve days so as to ensure that evaporation, precipitation, temperature, and other dramatic changes in weather, as well as emergence phenology of aquatic invertebrates didn't confound results. The sampling period ran between June 24th and of July 1st. The assessment of the mitigation of riparian vegetation was achieved by comparing a number of ecological indicators. These included physical, chemical, and biological variables which were assessed quantitatively and qualitatively.

2.2 Procedures for physical and water quality measurements

All field measurements were collected using standardized methods and by the same individuals across the different sites in order to maximize consistency. A 50 meter reach was initially selected based on its riparian characteristic. The 50 meter reach was then divided into five ten-meter sections in order to spread sampling along the entire reach without the influence of biases from focusing on particular areas (Appendix C). The physical characteristics of each section were measured using a variety of metrics. For measures that had highly variable values, for example stream velocity, multiple recordings were taken for each section and the mean value was used in analysis.

Temperature measurements were taken over the period of a week for all sites. Water temperature was recorded by Hobo Water Temp Pro v2 data loggers that had been placed at every site on the day prior to the first day of the twelve day sampling period. The Hobo data loggers were programmed to record water temperature every half hour. Average, minimum, maximum, and daily range temperatures were obtained from these data and were compared between sites. At the West Wilson River impacted site, the temperature meter was exposed by dropping water levels after four days and the data from these days was therefore omitted from the data analysis.

A YSI 556 MP5 handheld multi-parameter instrument was used to measure dissolved oxygen (mg/L), oxidation-reduction potential (mv), specific conductivity (ms/mc), and pH. This was the first set of measurements taken at all of the sites in order to ensure that instream activities associated with sampling did not affect the results.

Water turbidity was measured in every 10m section with a turbidity tube constructed from a 4.5 cm diameter polycarbonate cylinder that was marked with a ruler in 10 cm intervals for a total of 120cm. This was based upon a design which is commonly used as a cost-effective turbidity measuring device in the field (Dahlgren *et al*. 2004). The bottom was sealed with a polyethylene cap that had a miniature Secchi disk fastened to the inside (Appendix D). The water column was filled with water from the sampling site until the secchi disk could no longer be seen, and the value was recorded. Values for each section were converted to a median for the whole reach for comparison between sites. All readings were taken by the same individual to ensure that readings would have a degree of consistency. Turbidity measures taken with this method were top-censored because the length of the tube did not permit measures exceeding 120 cm.

At each sampling site a number of physical measurements were taken. These measurements were taken in each ten meter section and averaged for the entire reach. Bank angle and overhang measurements were taken by using the clinometer and stadia-rod method of Platts *et al.* (1983). These measurements were highly variable and therefore three measures were averaged for each section to obtain a more representative value. Wetted and bankfull widths were taken by stretching a surveyor's tape across the width of the stream at water level and bank height level, respectively. Stream channel slope is the gradient of the entire reach and was measured using a stadia-rod and leveling scope, which gave a measure in meters of the rise or fall of the stream bed over the 50 m distance. This value was then used to determine slope with the use of basic trigonometry (Gordon *et al.* 2003). The percentage of the stream channel that was riffle, run, or pool areas were also visually estimated and later converted to averages.

Stream velocity was measured in every section; five times across the width of the stream (Appendix A). Velocity was measured using an E-waterscience BackPackFlowMeter (SN: BP-004) digital turbine flow meter, which recorded the stream average velocity in meters per second. Wetted depth was measured with a stadia-rod at each point in the stream in which water velocity was also measured. Stream velocity measurements were taken at a depth of approximately 40% of the height from the stream bed to the water-air interface in order to gain accurate velocity readings (Gordon *et al.* 2003). Measurements were taken by holding the instrument upstream of the researcher and avoiding riffles caused by water passing over the researcher's legs. Stream discharge was calculated using the following equation: Q=VA or Q = sum of $Q_n = w_1D_1V_1 + w_2D_2V + ... + w_nD_nV_n$ (Gordon *et al.* 2003). Where Q represents discharge, and W, D, and V represent channel width, wetted depth and stream current velocity, respectively. Bankfull depth was only measured once at the center of the channel at each section and was used as an average for the whole reach. The wetted and bankfull widths were measured with a surveying tape measure.

Measurements of the width of the buffer zone were taken using a surveyor's tape from the edge of the bank to the edge of the buffer strip, where the bank sharply cut down to the water, at all five sections along the reach and for both bank sides. Measurements were taken using a one-hundred meter surveying tape measure. This was the maximum distance recorded and those that were greater were recorded as being greater than 100 meters. This was done for practical reasons within the study area, since sites were often too densely vegetated to take accurate measures with a tape over 100 meters. This data was therefore top-censored at 100 meters. The types of riparian vegetation (*i.e.*, trees, bush, grass, willow) and dominant land uses (*i.e.*, pasture, crops, forest) were also recorded.

A densiometer is a small box with a convex mirror that has a 96-point grid and used to measure the canopy cover in each of the five, ten meter sections. The percent canopy cover was calculated by counting those points on the mirror which were not covered. The number of points not covered is then deducted from ninety-six, divided by ninety-six and multiplied by one-hundred to give the exact value for canopy cover as a percentage. This measurement was performed by standing at the center of the stream and taking four consecutive readings: on the left bank and right bank, facing upstream, and then downstream. Mean canopy cover was calculated for the sites by taking an average of all of the recordings.

A qualitative, four-point ranking system was used to rank stream attributes and was rated on a scale of one to four (Appendix E). Qualitative values assessed in this manner were: substrate embeddedness, rock brightness, aquatic vegetation, presence of large woody debris, presence of fish and presence on manure on banks. The types of aquatic vegetation (Appendix F) were also noted as being either present or absent for each ten meter section of the reach. Each ten meter section of the reach had a corresponding left bank, right bank, upstream and downstream photograph taken. A site diagram was also drawn for each site to record the important features of the reach.

2.3 Water, sediment and benthic macroinvertebrate sampling

Water samples were collected for chemical analysis and for fecal coliform counts. Water samples for chemical analysis were collected from all sites on the last day of sampling in acid-washed, Nalgene™ low density polyethylene 500 ml bottles. Bottles were first rinsed with site water three times, then submerged and filled, and capped while under water. These samples were immediately placed on ice and transported to the Freshwater Institute Chemistry Laboratory, where chemical analyses were conducted. Analytes included: total dissolved phosphorus (TDP), total dissolved nitrogen (TDN), soluble reactive phosphorus (SRP), suspended nitrogen (SUSPN), suspended phosphorus (SUSPP) and carbon (SUSPC), soluble reactive silica (SRSI), total suspended solids (TSS), suspended iron (SUSFE) and concentrations of Na, K, Mg, Ca, Fe, and Mn (µg/L) as well as pH and alkalinity.

Samples for fecal coliform analysis were taken at each site over a single day, one week following the main sampling period. Sealed bottles were obtained from ALS Laboratories Inc., which was the private laboratory responsible for the analyses. Samples were again collected by submerging the bottle and placing the cap on the bottle while under water. These samples were packed on ice for the return to Winnipeg and stored in a refrigerator until they could be delivered to the lab the next day.

Stream-bed particles were sampled at every 10 m section of the 50 m reach. A bottomless pail with a 50 cm diameter was used to sample the stream substrate for particle analysis as is outlined in Atkinson *et al.* (1993). The sediment was dug to approximately 10 cm standard depth using a small spade. The sediment was then transferred into a double polyethylene bag for transportation to the lab.

Five macroinvertebrate samples were taken at each site, one in each ten-meter section. A 0.1 m^2 U-net (Scrigmour *et al.* 1992) was used with a 250 micrometer mesh size, and was placed hap hazardously within the sections. It was important to have the wire of the U-net flat on the substrate so that none of the disturbed benthic invertebrates would escape under the net, therefore areas with large protruding rocks could not be sampled. The substrate within the area of the U-net was disturbed to a standardized depth of approximately 10 cm. The net was then removed from the substrate and anything on the net was carefully washed into the cod end at the end of the net with clean water. The contents of the cod end were then placed in 50 ml jars with internal and external labels, and were filled with a 10% formalin solution. Sample containers were then sealed with electrical tape and stored in a sample box until they were processed, approximately two weeks later.

2.4 Processing of samples

Particle size and substrate organic composition was analyzed at the University of Winnipeg, Geography Department Soils and Geology Lab. The three samples that were analyzed for sediment composition corresponded to the same reaches in which benthic macroinvertebrate samples were processed. Sediment samples were first dried at 28.5°C for two weeks, and then were placed in an oven at 400°C overnight to remove the remaining water and assure that the sample was completely dry. Samples were then broken up using a mortar and pestle and run through a Soil Test sample splitter/sub-sampler. For particles that were larger than cobble, a four paper technique was used. This involves the use of four eight-by-eleven inch standard printer papers that were laid on a table with their edges touching to form a sixteen-by-twenty-two inch rectangle. The large particles were then set in the middle of where the four papers touched. One paper was randomly pulled away to obtain a 25% sub-sample or two papers were randomly pulled away to obtain a 50% sub-sample. This technique allows for the sub-sample to be proportional to the sub-sample of the finer sediment and is one that is commonly used in the University of Winnipeg Geography Department (pers. comm.). The total sub-sample weight was then recorded using a digital scale to four decimal places. The finer portions of the sub-samples were then sieved for 15 minutes using a No. 200 U.S.A. standard test sieve and a sieve shaker to separate the sand from the silt-clay fraction.

The same three out of five samples were used for the analyses of the organic content by loss on ignition. From the full samples obtained in the field, a sub-sample of approximately 65 grams was taken for the analysis of the percent organics by scooping the material into a pre-weighed crucible. The weight difference with the material was then recorded. Samples were handled using tongs as not to add any weight from oils or dirt on fingers. The sub-samples, which had been previously dried, were weighed with a digital scale to four decimal points. They were then placed in an oven set to between 450-600°C for 24 hours. Samples were then re-wetted and dried at approximately 350°C for an additional 24 hours. The samples were weighed directly after drying so that atmospheric water would not be reabsorbed by hydrates in the sediment and increase the weight of the sub-sample.

Three of the five benthic macroinvertebrate samples were randomly chosen for the each of the twelve sites due to processing time constraints. Macroinvertebrate samples were first transferred from the formalin solution to a 70% ethanol solution for storage until they could be processed. Samples were placed into gridded Petri dishes with approximately 70% ethanol solution and were examined with a stereomicroscope under approximately 12x magnification using incident lighting. All macroinvertebrates were removed from the samples using forceps or a pipette (in the case of the more delicate organisms). Organisms were counted and transferred into labeled patent-lip vials for storage. Organisms were identified to Family except for organisms belonging to the Phylum Nematoda and some of the Annelida, as well as organisms belong to the Class Ostracoda, Subclass Hyrudinea and Oligochaeta, and Order Lepidoptera. These groups were not identified to Family due to time constraints.

Quality assurance procedures were used so that a sorting efficiency of 90 % would be maintained. As the samples were sorted, they were recorded in a book in the order that they were processed. The first 10 of 36 samples were reprocessed by an experienced sorter for any missed macroinvertebrates present in the samples. For the remaining 26 samples, an additional 10% of the processed samples were randomly chosen, and sorted by an experienced sorter. These numbers were then recorded and compared to the number of inverts found by the original sorter. If this procedure did not produce a value of 90% efficiency, which was the required standard, the previous ten samples were resorted.

2.5 Statistical analyses

All quantitative data including physical, chemical and sediment measurements were converted into means or medians for each site. Descriptive statistics were calculated including the averages, ranges, standard deviations, and standard errors. A paired test was used in this study to compare the means because the mitigated and impacted sites were not independent from one another (Zar 1996). Analyses were performed using Minitab version 12 software. In cases where the data was top-censored, a non-parametric comparison of medians (Wilcoxon paired rank sum test) was performed.

Lydy *et al.* (2000) reported that when diversity, similarity and biotic metrics were compared for their ability to detect changes in water quality, proportional biotic metrics were the most accurate. Biotic metrics were therefore used in this study. Numbers obtained from the samples were first converted to density then converted to their proportional values by dividing the number of individuals within a metric by the total number of individuals. The proportion of Ephemeroptera, Plecoptera, and Trichoptera (EPTs); Diptera; Chironomidae; Crustacea and Mollusca; and Oligochaeta were the main biometrics tested in this study because they are known to be good indicators of anthropogenic impacts to aquatic environments (Rosenberg and Resh 1993; Hauer and Lamberti 1996; Metzling *et al.* 2003). The Crustacea and Mollusca were grouped and tested as a proportional metric as is common procedure in macroinvertebrate bioassessments (Barbour *et al*. 1999; Griffith *et al*. 2001). Taxa richness was compared between mitigated and impacted sites. Simpson's diversity index was calculated, which takes into account both the taxa richness and the relative abundance of taxonomic groups. Metrics were then analyzed using a one-tailed or two-tailed paired t-test depending on the expected differences as reported in current literature. Bonferroni adjustments were made for the alphas used in the paired t-tests so as to reduce the possibility of incorrectly accepting the alternate hypotheses. The corrections were applied to the benthic macroinvertebrate metrics, the water chemistry analysis from the water sample, the water chemistry analysis from the YSI, the sediment and substrate organics analysis and the channel morphology analysis. Power analyses were also performed with Minitab 15 via the 1-mean t-test using the observed differences of the means, the standard deviations of the differences and the Bonferroni corrected p-values.

3.0 Results

3.1 Water quality

There were no significant differences in water temperature metrics between mitigated and impacted sites. The West Wilson mitigated site dried up sufficiently that the Hobo^{TM} at this site was exposed to air on the fifth day and therefore data from this period was included in the statistical analysis. Mean daily water temperature ranged from 14.53 ^{o}C to 34.92 ^{o}C in mitigated sites and 13.45 ^{o}C to 26.16 ^{o}C in impacted sites (Figure 1). Sites collectively had mean daily temperature ranges of 10.17 ^{o}C and 8.61 ^{o}C for mitigated and impacted sites, respectively, and these were not found to be statistically different (paired t= 0.15, $t_{\alpha 0.0125\ DF = 5}$, P>0.05). Mitigated and impacted sites also had mean temperatures of 20.3 ^{o}C and 19.8 ^{o}C, respectively, which were not significantly different (paired t= 0.80 $t_{\alpha 0.0125,\ DF = 5}$, P>0.05).

There were no significant differences between treatments for measurements obtained with the YSI^{TM} metre. Mean specific conductivity was 0.961 mS/cm and 1.413 mS/cm (milliSiemens per centimetre) for mitigated and impacted sites, respectively. The highest conductivity was found at Sulphurspring Creek at both mitigated and impacted sites, and the lowest was at the East Wilson impacted site (Table 1). Mean values for specific conductivity were not found to be significantly different between sites (paired t= -0.70, $t_{\alpha 0.0125\ DF = 5}$, P>0.05). Dissolved oxygen (DO) ranged between 1.397 mg/L and 8.77 mg/L and 0.38 mg/L to 8.105 mg/L, at mitigated and impacted sites, respectively. The impacted and mitigated sites for Sulphurspring Creek had the lowest DO value for all sites, which were 1.40 mg/L and 0.38 mg/L, respectively. The mean DO was 6.42 mg/L for mitigated sites and 5.71 mg/L for impacted sites and this was not significantly different between sites (paired t= -0.74, $t_{\alpha 0.0125\ DF = 5}$, P>0.05).

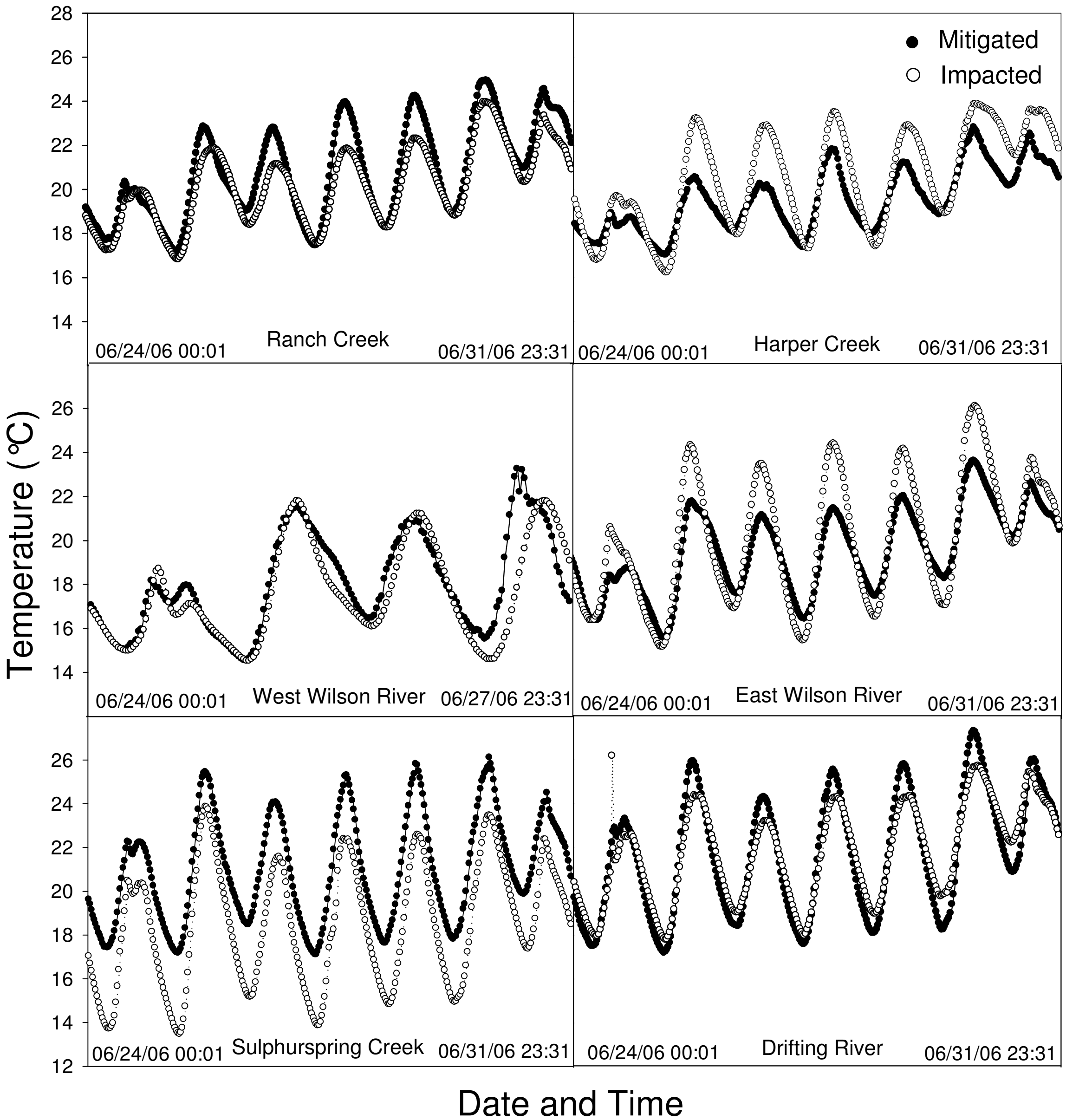

Figure 1. Water temperature as a function of date and time, recorded every 30 minutes with Hobo Water Temp Pro v2. temperature meters.

Oxidation reduction potential (ORP) at the impacted sites showed a much wider range of -291.33 mv to 190.06 mv than did the mitigated sites' range of 76.76 mv to 190.77 mv. This larger range is the product of the lowest recorded value, which was at the Sulphurspring Creek impacted site. Oxidation reduction potential was not significantly different between mitigated and impacted sites (paired t= 0.56, $t_{\alpha 0.0125\ DF = 5}$, P>0.05). The mean pH for mitigated and impacted sites were 7.91 and 8.07, respectively and these were not significantly different (paired t= -1.32, $t_{\alpha 0.0125\ DF = 5}$, P>0.05).

Water turbidity readings were, in several cases, restricted by the height of the water column in the sampler, thus medians were compared. Turbidity measures were very similar between mitigated and impacted sites, with ranges of 58.4 cm to >120 cm and 24.1 to >120cm, respectively. A paired Wilcoxon rank sum test was used to test median values for mitigated and impacted sites. Median values were 73.9 cm and 67.8 cm for mitigated and impacted sites respectively. These results were not significantly different (n = 6, W+ = 11, W- = 10, $P \leq 1$).

Detailed water chemistry results are in Appendix E. Each water chemistry variable was tested individually for differences between mitigated and impacted sites (Table 2). There was no significant difference between treatments for any of the water chemistry variables. There were several values that were notably high when compared to the collective means at mitigated and impacted sites. The impacted site on Harper Creek had a value of 50 µg/L dissolved phosphorus, as compared to the mean value of 22 µg/L at impacted sites. The impacted site on the East Wilson River site also had water quality variables that had high values when compared to overall means at impacted sites.

Table 1. Water quality descriptive statistics for sites in the study area measured with an YSI 556 MP5 handheld multi-parameter instrument.

		Mitigated				Impacted			
Site	**Variable**	**Mean**	**n**	**SD**	**SE**	**Mean**	**n**	**SD**	**SE**
Ranch Creek	**pH**	8.18	104	0.0075	7.35 E-04	8.01	118	0.025	0.0023
	SpCond (mS/cm)	0.720	104	1.682 E-04	1.649 E-05	0.698	118	0.0539	4.958 E-03
	ORP (mv)	137	104	9	1	178	118	17	2
	DO (mg/L)	8.77	104	0.39	0.038	7.09	118	0.21	0.020
Harper Creek	**pH**	7.89	119	0.041	0.0038	8.00	104	0.012	0.0012
	SpCond (mS/cm)	0.801	119	2.700 E-03	2.475 E-04	0.777	104	1.451 E-03	1.423 E-04
	ORP (mv)	129	119	39	4	186	104	13	1
	DO (mg/L)	5.38	119	1.42	0.13	8.11	104	1.17	0.11
East Wilson River	**pH**	8.16	128	0.024	2.15 E-03	8.09	113	0.062	5.83 E-03
	SpCond (mS/cm)	0.675	128	0.060	5.27 E-03	0.661	113	0.089	8.340 E-03
	ORP (mv)	178	128	12	1	184	113	11	1
	DO (mg/L)	7.46	128	0.96	0.085	7.95	113	0.56	0.052
West Wilson River	**pH**	8.13	125	0.018	1.62 E-03	8.11	141	0.011	9.02 E-04
	SpCond (mS/cm)	5.332	125	2.220 E-03	1.985 E-04	0.781	141	1.317 E-03	1.109 E-04
	ORP (mv)	191	125	11	1	190	141	12	1
	DO	7.08	125	1.13	0.10	7.53	141	0.94	0.080
Sulphur-spring Creek	**pH**	7.314	112	0.158	0.0149	7.997	99	0.0195	1.69 E-03
	SpCond (mS/cm)	1.879	121	0.012	1.046 E-03	5.354	99	0.0146	1.471 E-03
	ORP (mv)	165	121	26	2	-291	99	35	4
	DO (mg/L)	1.40	121	0.61	0.055	0.38	99	0.81	0.081
Drifting River	**pH**	7.992	115	0.024	2.19 E-03	8.25	113	0.030	2.77 E-03
	SpCond (mS/cm)	0.873	115	2.850 E-03	2.658 E-04	0.830	113	5.407 E-03	5.087 E-04
	ORP (mv)	77	115	5	0.5	35	113	6	0.6
	DO (mg/L)	6.57	115	0.23	0.021	6.73	113	0.35	0.033

There was an extremely high ammonia (NH_4) value, 2988 µg/L, obtained at the East Wilson impacted site, compared to the mean for impacted sites of 510 µg/L. There were also high values obtained at this site for total dissolved nitrogen (TDN) at 1430 µg/L, and suspended particulate carbon at 130 µg/L, as compared to the impacted treatment means of 751 µg/L and 37 µg/L, respectively.

Table 2. Paired t-test results for the water chemistry parameters sampled on the last day of the sample period.

Chemical Parameter	Paired-t	tα	DF	P-Value
Dissolved N	-0.98	0.0025	5	>0.05
Dissolved P	0.78	0.0025	5	>0.05
NO_3	1.42	0.0025	5	>0.05
NO_2	-0.85	0.0025	5	>0.05
NH_4	-0.99	0.0025	5	>0.05
SRP	0.98	0.0025	5	>0.05
SUSPN	-0.77	0.0025	5	>0.05
SUSPC	-0.84	0.0025	5	>0.05
SUSPP	-0.70	0.0025	5	>0.05
SRSI	-1.72	0.0025	5	>0.05
TSS	-0.17	0.0025	5	>0.05
SUSFE	0.64	0.0025	5	>0.05
Na	-0.99	0.0025	5	>0.05
K	-0.75	0.0025	5	>0.05
Mg	1.26	0.0025	5	>0.05
Ca	0.88	0.0025	5	>0.05
Fe	-1.00	0.0025	5	>0.05
Mn	1.02	0.0025	5	>0.05
Alkalinity	-1.10	0.0025	5	>0.05

Fecal coliform colonies were counted to a maximum count of 200 colony forming units (CFU). None of the mitigated streams had values above this number and the medians value for these sites was 70.5 CFU. The impacted sites had two sites with coliform counts above 200 CFU, which were Sulphurspring Creek and the West Wilson River. When the median values were compared using a Wilcoxon paired test, the sites were not found to be statistically different ($n = 6$, $W+ = 7$, $W- = 14$, $P \leq 0.5625$).

3.2 Stream morphology

Descriptive statistics were calculated for all stream morphological features, and means were then compared with paired t-tests (Table 2). The proportion of the stream's channel that was riffles (paired t= -0.47, $t_{\alpha 0.01\ DF} = 5$, P>0.05), runs (paired t= -0.18, $t_{\alpha 0.01\ DF} = 5$, P>0.05), or pools (paired t= 0.40, $t_{\alpha 0.01\ DF} = 5$, P>0.05) did not differ statistically. The mitigated site for Sulphurspring Creek did, however, stand out in terms of the percentage of pools, with a mean pool proportion of 84% as compared to the mean for all mitigated sites of 43.3%.

Bank slope and bank overhang measurements were averaged for each section of the reach. These averages were then converted to means for each site so that they could be used in the paired t-test. Bank slope averaged 32.53^o at mitigated sites and 38.56^o at impacted sites, with respective ranges of 24.1^o to 44.77^o and 32.4^o to 45.3^o. These differences were not found to be statistically significant (paired t= -1.71, $t_{\alpha 0.025\ DF = 5}$, P>0.05). Bank overhang means ranged from 0.06 cm to 9.72 cm and 1.63 cm to 8.33 cm at mitigated and impacted sites, respectively. These means were not statistically different (paired t= 2.13, $t_{\alpha 0.01\ DF = 5}$, P>0.05), with mitigated sites having higher a bank overhang mean of 6.85 cm and the impacted bank overhang a mean of 3.85 cm.

There was no difference in channel slope between the mitigated and impacted sites (paired t=1.27 $t_{\alpha 0.01\ DF = 5}$, P>0.05). The ranges were of the sites were almost identical with mitigated sites ranging from 0.03 m to 0.07 m and impacted sites ranging from 0.03 to 0.08 m over a 50 m reach. Bankfull depths of mitigated and impacted sites ranged from 1.07 m to 2.29 m and 0.71 m to 3.21 m, respectively. Wetted widths ranged from 3.96 m to 5.67 m and 2.27 m to 6.46 m at mitigated and impacted sites, and had averages of 3.58 m and 3.78 m, respectively, which were not found to be statistically different (paired t=

-0.19, $t_{\alpha 0.01\ DF=5}$, P>0.05). Bankfull widths ranged from 4.02 m to 8.65 m and 3.55 m to 8.95 m for mitigated and impacted sites, and had averages of 6.54 m and 6.19 m, respectively and were not found to be significantly different (paired t= 0.27, $t_{\alpha 0.01\ DF=5}$, P>0.05). Channel discharge ranged from 0 m^3/s to 6.07 m^3/sec for mitigated sites and 0 m^3/sec to 2.55 m^3/s for impacted sites, respectively. The mean channel discharge was 1.52 m^3/sec and 0.82 m^3/sec for mitigated and impacted sites, respectively. The means were not found to be statistically significantly different (paired t= 1.20, $t_{\alpha 0.05\ DF=5}$, P>0.05).

Table 3. Descriptive statistics for stream morphology features including bank slope (degrees), bank overhang (cm), channel slope (m), bankfull width (m), wetted width (m) and stream discharge (m^3/sec).

	Mitigated			**Impacted**		
Morphology	**Mean**	**SD**	**SE**	**Mean**	**SD**	**SE**
% Riffle	22.17	9.75	3.98	27.17	18.20	7.43
% Run	34.50	18.96	7.74	36.00	10.43	4.26
% Pool	43.3	25.2	10.3	36.8	20.6	8.4
Bank slope (Degrees)	32.53	8.01	3.27	38.56	4.89	1.99
Bank overhang (cm)	6.85	5.26	2.15	3.85	2.79	1.14
Channel slope (m)	0.0542	0.0196	0.008	0.048	0.022	0.009
Bankfull depth (m)	1.61	0.44	0.18	1.71	0.96	0.39
Bankfull width (m)	6.54	1.67	0.68	6.18	2.57	1.05
Wetted width (m)	3.58	1.26	0.51	3.73	1.48	0.60
Stream discharge (m^3/sec)	1.52	2.27	0.93	0.82	0.94	0.38

Table 4. Paired t-test results for stream morphological features including bank slope, bank overhang, channel slope, bankfull width, wetted width and stream discharge.

Morphology	**SD**	**SE**	**α**	**Paired-t**	**P value**
% Riffle	25.9	10.6	0.01	-0.47	>0.05
% Run	19.87	10.6	0.01	8.11	>0.05
% Pool	40.1	16.4	0.01	0.40	>0.05
Bank slope	8.64	3.53	0.01	-1.71	>0.05
Bank overhang	3.45	1.41	0.01	2.13	0.087
Channel slope	0.0112	0.0046	0.01	1.27	>0.05
Bankfull depth	0.864	0.353	0.01	-0.29	>0.05
Bankfull width	3.20	1.31	0.01	0.27	>0.05
Wetted width	1.96	0.80	0.01	-0.19	>0.05
Stream discharge	1.43	0.58	0.05	1.20	>0.05

3.3 Riparian Vegetation Measurements

The median widths of the vegetated buffer strip, which were top censored at 100 m, were compared with the Wilcoxon paired Rank Sum test. When statistically tested, these values did not produce significant differences (n = 6, W+ = 15, W- = 0, $P \leq 0.0625$). However, all mitigated sites had some sections with buffer zones that were over 100 meters, and only two of the sites had sections with buffer zones narrower than 100 meters (Appendix D). In contrast, the impacted sites had only a single stream, Sulphurspring Creek, which had some sections with a riparian buffer exceeding 100 meters in width.

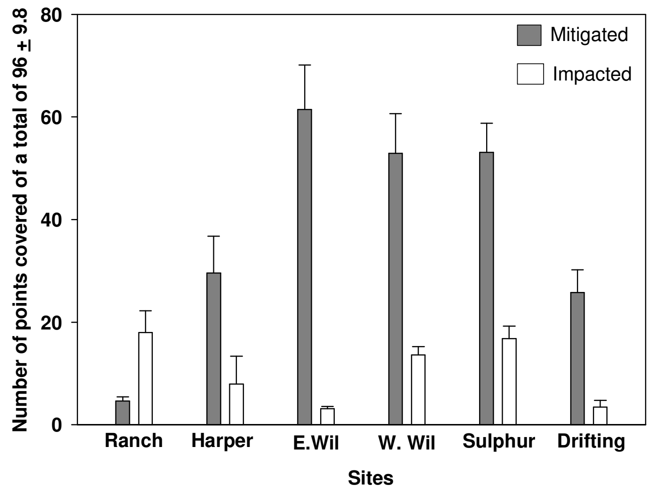

Figure 2. Mean canopy cover measured by the average number of points covered on a 96 point densiometer (paired t= 2.79, $t\alpha_{0.05\ DF = 5}$, P=0.019).

Canopy cover between treatments ranged from 4.64% at Ranch Creek and 61.435% at the East Wilson River for mitigated sites, and 3.11% at the East Wilson River and 17.96% at Ranch Creek for impacted sites (Figure 2). Mean canopy closure was 37.91% for mitigated sites and 10.48% for impacted sites, which were significantly different (paired t= 2.79, $t_{\alpha 0.05\ DF = 5}$, P=0.019).

3.4 Sediment Samples

The proportions of cobble, sand, and fines were statistically analyzed for differences between mitigated and impacted sites. There were no significant differences between the sites and this can be easily seen when the values are plotted on a ternary graph (Figure 3). One site varied considerably in terms of the percentage of organic material found in the substrate. This was the impacted site at Ranch Creek, which had a mean percent organic value of 34.98% as compared to the mean value for impacted sites of 7.19%. The mitigated and impacted sites collectively showed no significant difference for the mean proportion of organics material (paired t= -1.07, $t_{\alpha 0.0125\ DF=5}$, P>0.05).

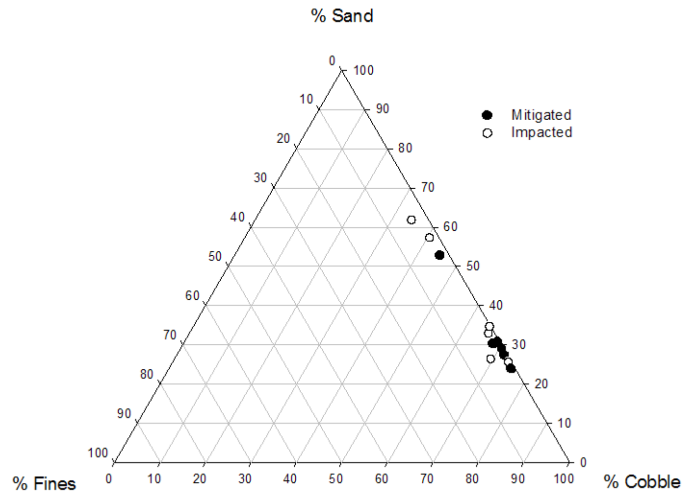

Figure 3. Percentage substrate composition represented by percent cobble (SD= 13.42, SE=5.48, paired t= 1.38, $t\alpha_{0.0125\ DF=5}$, P>0.05), sand (SD= 12.15, SE=4.96, paired t= -1.57, $t\alpha_{0.0125\ DF=5}$, P>0.05), and fines (SD= 1.53, SE=0.63, paired t= -1.86, $t\alpha_{0.0125\ DF=5}$, P>0.05) for mitigated and impacted sites of all six streams.

3.5 Qualitative assessments

Ranked qualitative assessments were compared by using medians due to the fact that not all classes of the ranking scale represented even differences between the classes. However, median values were calculated for comparisons between sites (Table 5). Median values for scores describing the extent of vegetative bank protection and the presence of large woody debris (LWD) were different between sites, with LWD being far more prevalent at mitigated sites. The median ranks for vegetative bank protection were 1 and 3, and 1 and 3 for large woody debris for mitigated and impacted sites, respectively.

Table 5. Median values for stream qualitative features for mitigated and impacted sites in the study area.

Qualitative Variable	Median rank-Mitigated	Median rank-Impacted
Mass wasting	2	3
Vegetative bank protection	1	3
Embeddedness	2	2
Large Woody Debris	1	3
Presence of manure	0	0

3.6 Benthic macroinvertebrate metrics

Benthic community diversity was described with Simpson's diversity index. Index values ranged at mitigated sites between 0.2068 and 0.4165 at the East Wilson River and the Drifting River for mitigated sites, respectively. At impacted sites it ranged from 0.1121 at Ranch Creek and 0.7694 at Sulphurspring Creek (Appendix I).There was no statistically significant difference between mitigated and impacted sites (paired t= -0.19, $t\alpha_{0.001\ DF=5}$, P>0.05). Taxa richness was also tested compared between sites, and there was no observable difference (paired t= -0.19, $t_{\alpha 0.001\ DF=5}$, P>0.05).

The proportion of the organisms collected belonging to the EPTs was not significantly different between sites (paired t= 0.16, $t_{\alpha 0.001\ DF=5}$, P>0.05). However, the impacted site Sulphurspring Creek had no EPTs in the impacted site, but had 45.2% of the benthic macroinvertebrates belonging to these orders in the mitigated site (Figure 4). The mean proportion of the community belonging to the Diptera at mitigated sites was statistically different from that at the impacted sites (paired t= 3.07, $t_{\alpha 0.001\ DF=5}$, P=0.028), with mitigated sites having the larger proportion. However, the mean proportion of Chironomidae within the order Diptera did not produce significant differences between mitigated and impacted sites (paired t= -1.09, $t_{\alpha 0.001\ DF=5}$, P>0.05). Invertebrates belonging to the Phylum Mollusca and the Subphylum Crustacea, when grouped and represented as mean proportions, did show nearly significant differences between sites (paired t= -2.05, $t_{\alpha 0.001\ DF = 5}$, P=0.054). These groups had a higher prevalence at impacted sites. The mean proportion of Oligochaeta was the last group of benthic invertebrates that was tested between treatments and was found not to differ significantly (paired t= 0.57, $t_{\alpha 0.001\ DF = 5}$, P>0.05). Total invertebrate abundance in the sites were also tested and were found to not be significantly different between sites (paired t= -1.07, $t_{\alpha 0.001\ DF = 5}$, P>0.05).

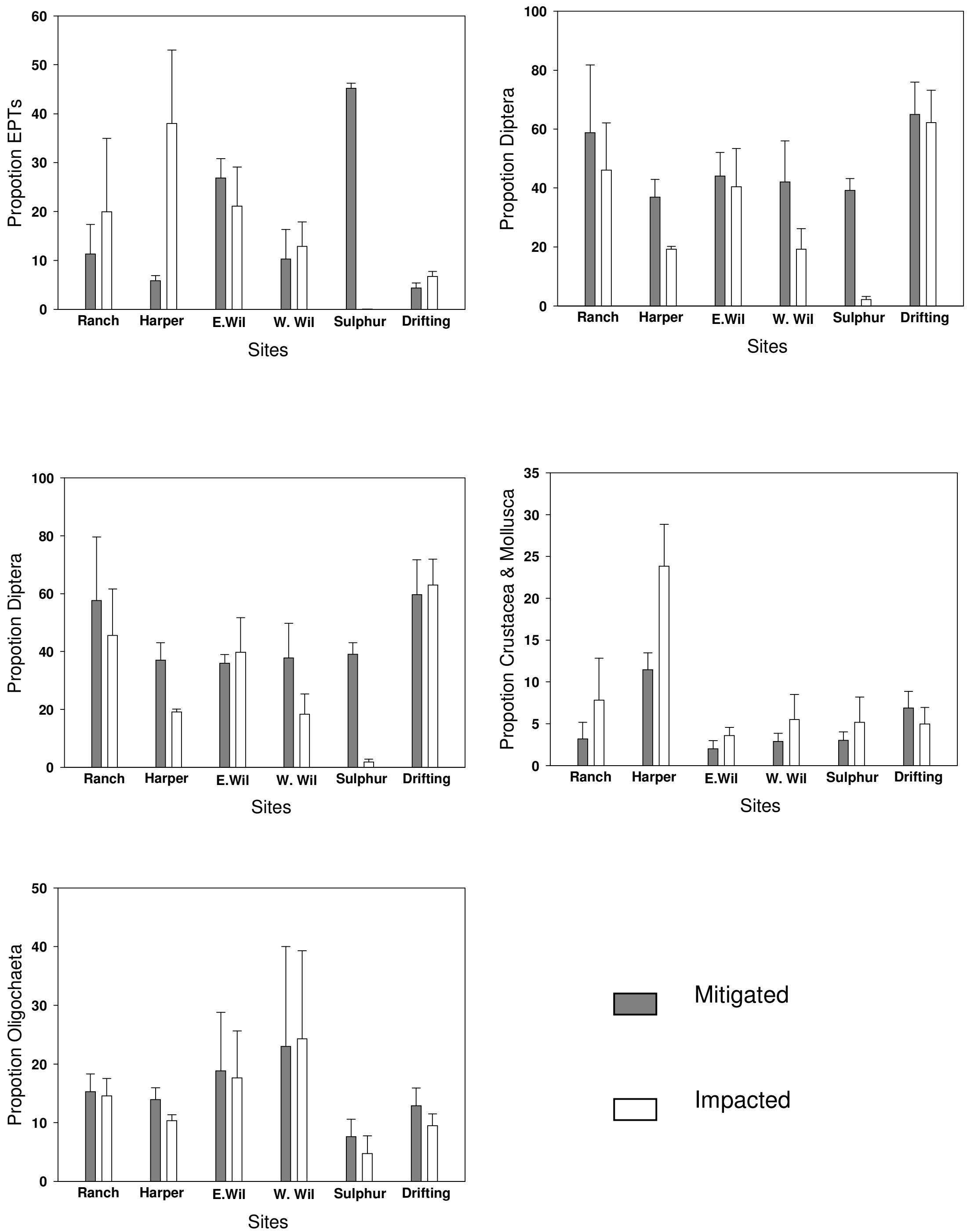

Figure 4. Proportions of macroinvertebrates belonging to major taxonomic groupings. Samples were collected with a 0.1 m^2 U-net, between June 20^{th} and July 1^{st}.

4.0 Discussion

Certain variables, such as channel slope and discharge were not expected to differ between mitigated and impacted sites due to the selection of all sites at the third order of stream classification. This was a trend that has been observed in previous studies (Gordon *et al.* 1993; Rott *et al*. 1998). With geomorphologic variables being constant across treatments, it was possible to make comparisons between streams based on their riparian characteristics (Rott *et al.* 1998; Newson and Newson 2000).

Canopy closure is a riparian characteristic that was expected to affect the solar radiation arriving at the surface of the water and thereby the primary production of the reach (Hauer and Lamberti 1996; Fletcher et al. 2000). However, the types of aquatic vegetation in this study were only looked at in a qualitative fashion therefore making it difficult to make assertions of a higher prevalence of aquatic vegetation and algae at mitigated or impacted sites. Canopy cover was found by Sovell *et al.* (2000) to not be affected by whether the riparian zone was dominated by either woody vegetation or grasses, therefore these variables are less important to the overall effects of incoming solar radiation. The canopy closure of the mitigated sites was more substantially developed than at impacted sites as would be expected (Platts and Nelson 1989; Fletcher *et al*. 2000; England and Rosemond 2004).

With the significant difference in canopy closure between sites, it would be expected that mitigated sites would have had a narrower range and lower mean temperature than impacted sites (Platts and Nelson 1989; MacDonald *et al*. 2003; Kishi, *et al.* 2004). However, the temperature means and ranges for mitigated and impacted sites did not follow this expectation. The data from the mitigated sites produced higher mean temperatures and wider temperature ranges except at the East Wilson River and Harper

Creek. This is opposite to what would be typically expected for sites with a large amount of canopy closure. This could have been due to the placement of the temperature meters in areas that eventually changed in depth or dried up over the study period, as was the case with the West Wilson River. Future studies should aim to seek out maximum depths for the placement of water temperature meters, especially when dealing with shallow low-order streams that are prone to drying. It is also possible that sites were more affected by upstream temperature conditions more than the shading effects of riparian vegetation. It has been observed that in streams that have dense canopy cover, the heat from the air and flows from groundwater are more important than the direct effects of solar radiation (Hauer and Stanford 1982). Therefore, mitigated sites may have been too close to areas that were highly affected by solar radiation as well as being affected by ambient air temperature. Site selection in future studies should take upstream conditions farther along the stream into consideration.

The amount of LWD in a stream is an important stream habitat characteristic that is expected to affect the overall hydrology by hindering flow velocity of the stream (Abernethy and Rutherfurd 1998). Higher mean values for the proportion of channel that was pools at certain mitigated sites, namely Sulphurspring Creek, also reflects what would be expected in reaches with larger proportions of debris dams (Tabacchi *et al.* 2000; Montgomery and MacDonald 2002). Sites that were mitigated by the presence of riparian vegetation also had a much higher level of vegetative bank protection, with mitigated sites having a median value of 1, compared with a median value of 3 at impacted sites. This kind of bank protection is achieved by the roots of riparian vegetation of a well developed riparian zone.

The presence of riparian vegetation greatly shapes the geological and hydraulic processes at the land-water interface and therefore has an important effect on bank formation (Hauer and Lamberti 1996; Tabacchi *et al*. 2000; Gordon *et al*. 2003). Bank slope was expected to be steeper at mitigated sites (Gordon *et al.* 2003) but did not differ significantly from mitigated to impacted streams. This may have been due to operational errors in the measurements or because this difference simply was not present at the sites. A power analysis for bank slope determined that there was an 8.7 % chance of falsely accepting the null hypothesis at $\alpha=0.05$. This indicates there was actually no significant difference in the bank slope between sites and we may accept the null hypothesis here with a high degree of confidence.

Even though the t-test results were not significant we may assert that impacted sites had smaller values for bank overhang than did mitigated sites as was expected due to the potential for erosion at impacted sites (Abernethy and Rutherfurd 1998; Moret *et al*. 2006). The Bonferroni correction is one that is extremely conservative (Zar 1996) and a p-value of 0.087 for bank overhang should still be considered as demonstrating a difference between sites. The well developed bank overhang at mitigated sites is attributable to the presence of the deep binding root masses of a well developed riparian buffer strip (Abernethy and Rutherford 1998; Smith 1976; Tabacchi *et al.* 2000). Higher mass wasting median values at impacted sites were somewhat reflective of the bank morphology differences between sites that are affected by agricultural land practices (Abernethy and Rutherford 1998; Moret *et al*. 2006). The widths of the buffer strips encountered at mitigated sites were almost statistically different from impacted sites. It was noticed at many of the mitigated sites that riparian buffer zones often extended for well over 200 meters (pers. observation) and future studies should include a maximum riparian measurement as a necessary component of the data set.

Water quality variables, such as nutrient concentrations (*i.e.,* elements essential for plant growth such as nitrogen and phosphorus) and fecal coliform content, as well as the pH, conductivity, and potassium are commonly used in the assessment of water quality in agricultural regions (Airaksinen *et al*. 2007). Lower dissolved oxygen is often observed in streams impacted by agriculture because of the increased aerobic decomposition of organic inputs (Salles *et al.* 2005). However, there were no significant differences in these variables when the means were tested with a paired design. This was likely due to the continuous nature chemical parameters throughout the watershed and the discontinuous riparian vegetation of the study area. Power analysis revealed that the sample size would have had to been be very large to detect a difference in dissolved nitrogen between the sites and that the differences in dissolved phosphorus were very small.

The overall water chemistry between sites was not significantly different, however, there were sites that had chemical variables that were considerably higher than the means the site types. Sulphurspring Creek had several variables that were the maximum values encountered over all sites. The Sulphurspring impacted site had a conductivity value of 5.354 mS/cm, which was the maximum conductivity value for both mitigated and impacted sites. Oxidation reduction potential is ability of stream to contribute or accept electrons, thus measuring the level of removal of contaminants from the water (Yu and Yu 2000). Sulphurspring Creek also had a minimum value for ORP. These values were likely impacted by the surrounding agricultural practices, as may be deduced from the maximum fecal coliform count of 200 CFU. However, certain characteristics are also likely attributable to the fact that the creek is the recipient of an active geothermal sulphur spring, and is therefore predisposed to increased acidity and extremely high conductivity (Karakaya *et al*. 2007). ORP should also be affected since there is an inverse relationship between pH and ORP (Yu and Yu 2000); however the opposite relationship was observed at this site, therefore other factors would explain this occurance. The presence of lower levels dissolved oxygen (DO) predisposes water to a decreased ORP value since (Lie and Welander 1994). It is therefore expected that the impacted site at Sulphurspring Creek with the lowest recording for DO also had the lowest recording for ORP. In the case of Sulphurspring Creek, the comparison of levels of fecal coliform levels in the mitigated and impacted sites is also a good indicator of agricultural inputs entering the aquatic environment.

Fecal coliforms from livestock feces enter the aquatic environment *via* base flow or storm overflow (Schoonover and Lockaby 2006). Inputs from runoff may be especially pronounced in circumstances where riparian vegetation is removed (Maillard and Santos 2007). Impacted sites at Sulphurspring Creek and the West Wilson River exhibited the maximum values for fecal coliform values at 200 CFU, which is the maximum number of bacteria colonies that may be found. Fecal coliforms in aquatic systems are indicators of the potential presence of dangerous pathogens to animals including humans. Fecal coliforms also indicate the presence of fecal matter which contributes to the biological oxygen demand of the system (Servias *et al.* 2007). The dissolved oxygen value of 7.533 mg/L for the West Wilson impacted site was very close to the mean DO value of 6.12 mg/L. However, the Sulphurspring impacted site had the lowest of all of the values recorded for DO at 0.375 mg/L, reflecting what is expected of sites with higher levels of coliforms (Servias *et al*. 2007).

Other water quality variables that were high when compared with mean values were those for dissolved phosphorous for the Harper Creek impacted site and ammonia, total dissolved nitrogen and suspended phosphorus (SUSPP) for the East Wilson River impacted site. High levels of ammonia and total dissolved nitrogen in agricultural streams have been associated with grazing along the stream margins (Walker *et al.* 2002), while elevated levels of phosphorous have been found to attributable to erosion, runoff and sediment loading (Barbro and Kalisky 2005). It is not surprising then that these maximum values were encountered at impacted sites. It would be interesting in future studies to test the chemical variables of individual streams against baseline levels for the study area.

There was no difference found in the substrate particle fractions between sites. Power analysis of the proportion of fine sediments revealed that this difference was indeed insignificant and that the null hypothesis may be correctly accepted. Sedimentation has been found in previous studies to be higher in riparian areas dominated by trees as opposed to grasses (Sovell *et al.* 2000). Future studies should also aim to measure the exact width of vegetation, vegetation types present at riparian buffer zones and their effects on runoff, sedimentation and non-point source pollutants (Tabacchi *et al.* 2000; Montgomery and MacDonald 2002).

The comparison of Simpson's diversity index values suggested that taxa richness was not different between mitigated and impacted sites. This was not expected, as other studies have shown that agricultural impacts normally have an overall detrimental effect on taxonomic richness (Merritt and Cummins 1996; Delong and Brusven 1998; Sandin 2003). Power analysis found that there was a 95% chance of falsely accepting the null hypothesis at α=0.05. This indicates that the number of replicates was likely not sufficient to detect a significant difference. Further identification to genus would also likely prove to be useful for extracting the more subtle community assemblages between sites, as genus is often the highest taxonomic level used for the determination of diversity (Rosenberg and Resh 1993; Merritt and Cummins 1996).

The difference in community assemblage between mitigated and impacted sites was found only among the Diptera, and the Crustacea and Mollusca. The Diptera were more common at mitigated sites, and the Crustacea and Mollusca were more common at impacted sites. The higher prevalence of the Mollusca and Crustacea is likely due to higher tolerance levels of some ostrocod species towards inputs from agricultural activities (Roca *et al.* 2000). These organisms are therefore able to be more ubiquitous in environments with non-point source pollutants. However, with the Diptera it is more difficult to reason for the difference and further identifications, especially among the Chironomidae may be useful in future studies. A significant prevalence of sensitive species such as the EPTs was expected in mitigated sites as compared to impacted sites. EPTs are commonly used in impact assessments because of their known sensitivities to organic pollution and sedimentation (Rosenberg and Resh 1993). There were, however, only two cases in which EPTs were more prevalent at mitigated sites. These were the East Wilson River and Sulphurspring Creek mitigated sites. However, collectively, this didn't account for enough of a difference for it to be considered statistically significant.

Power analyses were conducted for the taxonomic groups that were found not to be significantly different between sites. It was found that the metric proportion of Chironomidae had a 5.8 % power to detect a difference, and the total invertebrate abundance metric had a 0.3 % power to detect a difference in community composition. These metrics therefore may have shown differences had the sample size been larger or had all of the 5 samples of a reach been processed. The similarities in community assemblages of the benthic macroinvertbrates may also have been a function of the low taxonomic resolution of the study, but may also be reflective of homogeneity in the study area. It was noted in a paired comparison impact study by Liljaniemi (2002), that finding adequate reference sites (*i.e.*, mitigated sites) was a major challenge, resulting in benthic invertebrate assemblages differing very little between treatments. However, it was found in a study by Harding *et al*. (1999) that even though the macroinvertebrate community didn't differ significantly along a reach from the sources of impact, there was a significant influence of the physical condition and agricultural activities of the area on community composition.

The No. 17 agricultural district of Manitoba is an area that is intensively altered by agriculture, and it was observed that riparian vegetation in this area was highly fragmented. Although there were areas with well-developed riparian buffer strips, these were often preceded by long stream lengths that had no riparian vegetation at all. Lotic systems are characterized by unidirectional flow and, thus, inputs to the stream ecosystem are transported for a distance downstream. The concept of nutrient spiraling in streams (Pringle *et al*. 1988; Woodward and Hildrew 2002) recognizes the inherent longitudinal spatial scale along which element cycling occurs in these ecosystems. As a result, water quality and benthic community composition at any point in a stream reflect conditions not only at that location but also upstream influences. Lotic systems also demonstrate what has been called "self purification" (Cazelles *et al.* 1991; Elosegui *et al*. 2006), in that recovery from disturbance also has a spatial component; conditions and communities return to increasingly "normal" conditions with increasing distance from the point of impact. The effectiveness of mitigation by riparian buffer strips is likely to occur and to be detectable only if the stream length within this mitigated zone is sufficient to allow the in-stream processes of nutrient absorption and cycling, biological and chemical decomposition of materials and recolonization by “sensitive” biota to occur. In this study there were few observed differences between the mitigated and impacted stream segments. This is although riparian zones have been clearly demonstrated to play an important role in the reduction of sedimentation and nutrient and contaminant transport into streams as well as the structural and metabolic functioning of the stream ecosystem (Hauer and Lamberti 1996; Abernethy and Rutherfurd 1998; Henley *et al*. 2000). It is likely that the predominance of nonsignificant results in this study is a reflection of the limited and fragmented nature of riparian areas in the study area. Future studies should attempt to determine the width and length of stream riparian buffer zones that are required to produce measurable, effective mitigation in order to further the adoption of land use practices that favour the integrity of our aquatic habitats.

5.0 Conclusions

It was found that sites were similar in terms of their geomorphology and it was therefore possible to use a paired comparison to investigate the effects of the presence or absence of riparian buffer zones. Bank overhang was the only morphological variable that was notably different between sites, with mitigated sites having a larger amount of overhang. There was a higher median value for vegetative bank stabilization at mitigated sites, which coincides with the difference in bank overhang measurements. Large woody debris also had a higher median value for mitigated sites as would be expected for sites with more vegetation.

Other habitat variables such as the proportion of sediment at the substrate and water quality variables did not differ between sites. Canopy closure, however, did differ significantly between sites but this did not affect the mean or range of temperatures at the sites. Benthic macroinvertebrate assemblages differed in the proportion of Diptera, and Crustacea and Mollusca, but only these groups. There was also no significant difference in species richness found between sites. Further taxonomic identifications may be useful in future studies to pull out subtleties in the data. Futures studies should also factor in measurements of the entire riparian buffer zone, and should record the type of terrestrial and aquatic vegetation in a quantitative fashion. The riparian vegetation of the study area was sparse and adequately mitigated sites were largely unavailable. It should be the aim of future studies to determine the width and length of a riparian buffer zone required to produce measurable effective mitigation.

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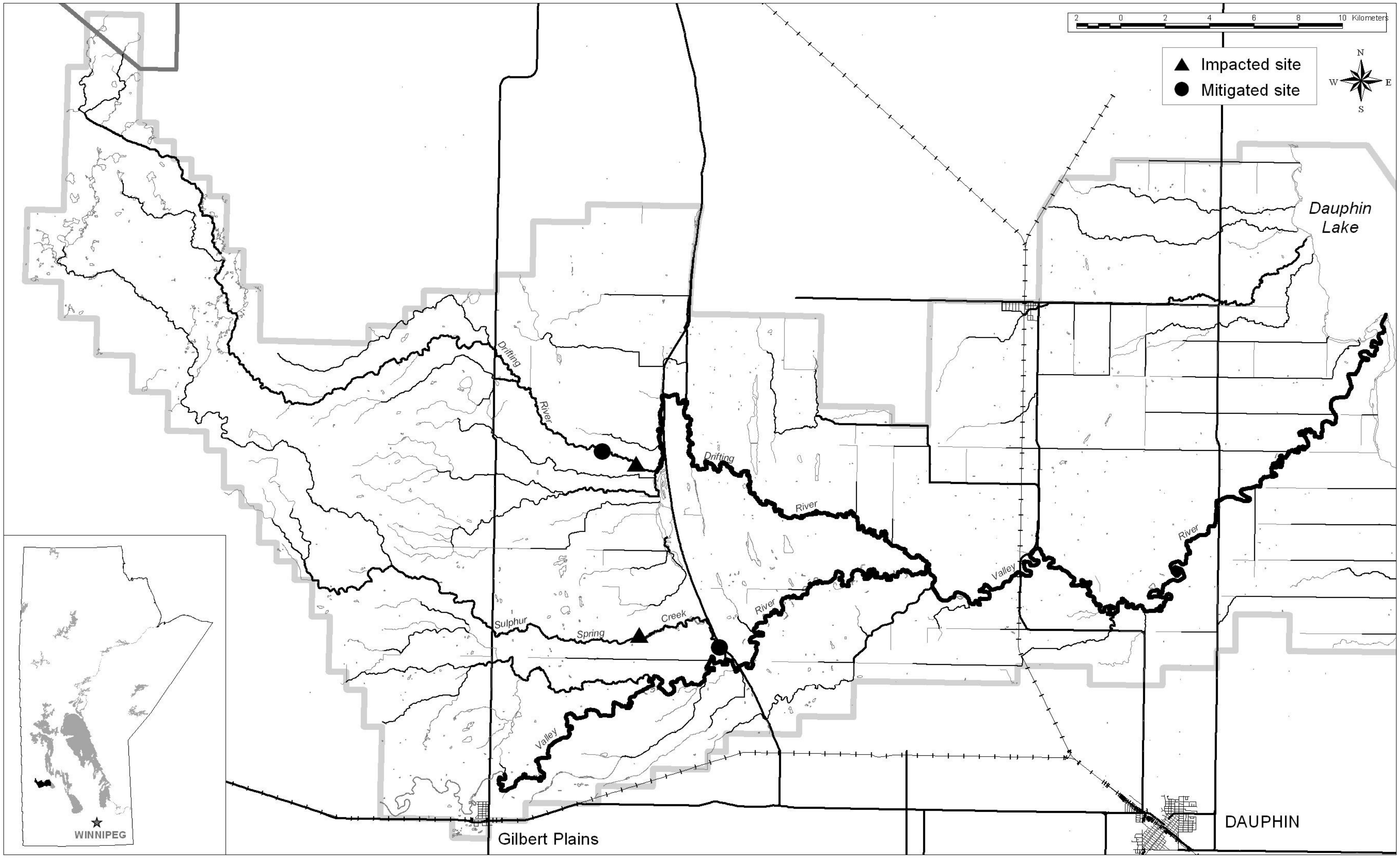
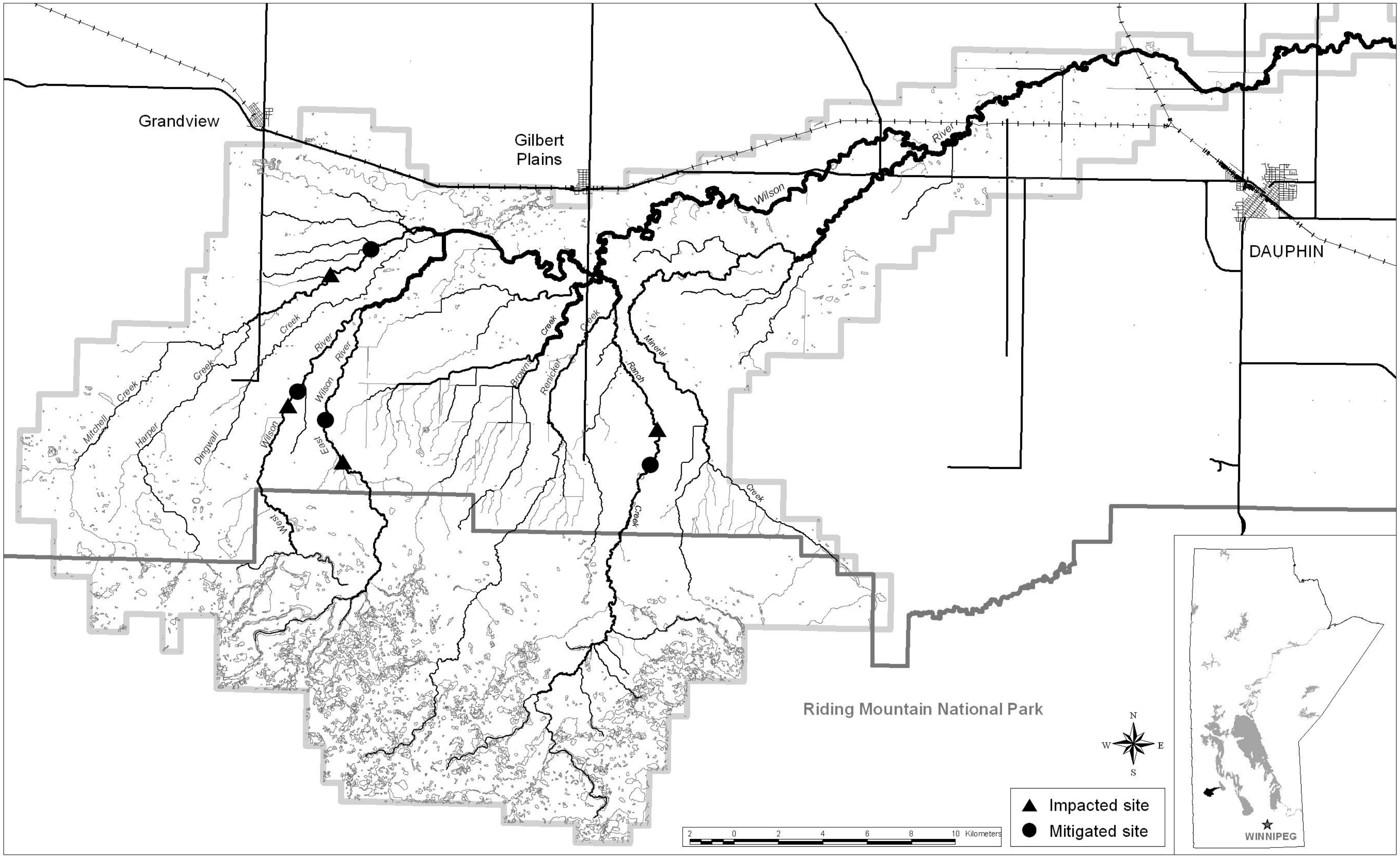
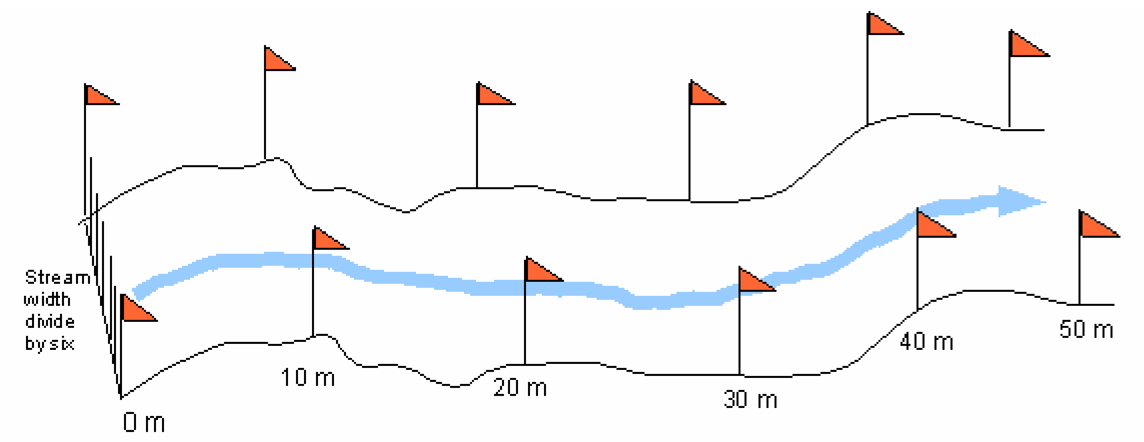


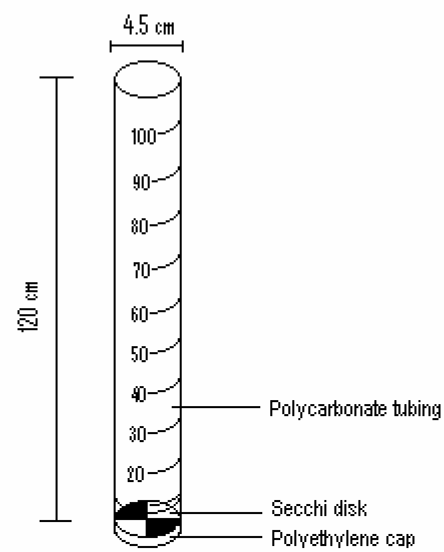
Appendix A. Valley River drainage



Appendix B. Wilson River drainage



Appendix C. Reach sampling design for the selected sites. A 50 meter reach was selected then divided into five, ten-meter sections. This was marked with flagging tape. The width of the reach was divided by six so that there would be five points at which to take depth and velocity measurements that were equally spaced.



Appendix D. Design of a turbidity tube, a water turbidity measuring device. Material consists of 120 cm graduated polycarbonate tubing, a polyethylene cape and a secchi disk fixed to the inside of the cap.

Appendix E. Qualitative ranking system used for stream attributes.

	Qualitative value			
Stream Quality	**1**	**2**	**3**	**4**
Mass Wasting Hazard	No evidence of past potential for future mass wasting	Infrequent and/or very small, mostly healed over. Low future potential	Moderate frequency and size. Erosion is some spots	Frequently or large, causing sedimentation and imminent danger of erosion
Vegetative Bank Protection	>90% plant diversity; vigour and variety suggest a deep, dense soil building root mass.	70-90% plant density; fewer plant species or lower vigour suggests a less dense or deep root mass.	50-70% density; lower vigour & still fewer species form a somewhat shallow and discontinuous root mass.	<50% density; even fewer species & less vigour indicate poor, discontinuous & shallow root mass.
Substrate embeddedness	Gravel, cobble and boulder particles and 0-25% surrounded by fine sediment.	Gravel, cobble and boulder particles are 25-50% surrounded by fine sediment.	Gravel, cobble and boulder particles are 50-75% surrounded by fine sediment.	Gravel, cobble, and boulder particles are >75% surrounded by fine sediment.
Rock brightness	Surfaces dull, darkened, or stained (growth or film of organic stain); <5% of total bottom bright	5-35% affected. Scour at constrictions and where gradients are steep. Some deposition in pools	Mixture, 35-65% bright materials	Predominantly bright, >65% exposed or scoured surfaces (i.e. in motion recently)
Aquatic vegetation	Periphyton and macrophytes abundant. Growth perennial, in swift water too	Peripyton and macrophytes common in slower portions of reach, but thins out or absent in swift flows	Periphyton and macrophytes present, but spotty, mostly in slow or still water areas, almost totally absent from swifter potions	Periphyton and macrophytes scarce or absent
Large woody debris	Large quantities of large woody debris, “natural stream”	Some large woody debris-intermediate	Large woody debris present but not a dominant feature	No large woody debris present at site
Presence of fish	Many fish present	Some fish present - intermediate	Some fish present- sparse	No fish present
Presence of manure	Large quantity of new manure on banks	Small quantity of new manure on banks	Large quantity of old manure present on banks	Small quantity of old manure present on bank

Appendix F. Types of aquatic vegetation assessed as being either present or absent within each 10m section of the 50 m reach.

Type of aquatic vegetation	**Description**
Periphyton	Micro-algae which is firmly attached to the substrate
Rooted floating	Vegetation which is rooted in the substrate and is also floating at the surface of the water
Rooted submergent	Vegetation that is rooted in the substrate but is not floating at the surface of the water
Free floating vegetation	Vegetation not attached to the substrate
Floating algae	Algae not attached to the substrate

Appendix G. Water chemistry raw data obtained from the Freshwater Institute Chemistry Laboratory. Samples were taken on the last day of the 12-day sampling period and packed on ice until they could be delivered to the lab.

Water Chemistry Variable	Ranch Creek		Harper Creek		East Wilson River		West Wilson River		Sulpurspring Creek		Drifting River	
	Mit	Imp	Mit	Imp	Mit	Imp	Mit	Imp	Mit	Imp	Mit	Imp
NO3 (µg/L)	38	25	15	3	10	5	6	6	1	1	5	10
NO2 (µg/L)	0.1	1	1	0.1	0.1	0.1	0.1	0.1	1	6	0.1	0.1
NH4 (µg/L)	13	13	17	9	26	9	17	15	43	2988	26	26
SUSPN (µg/L)	71	59	171	100	79	140	381	185	252	1083	204	280
TDN (µg/L)	583	581	835	746	564	541	437	445	136	1430	670	765
SRP (µg/L)	7	7	24	34	9	7	15	11	21	*	0.2	0.2
SUSPP (µg/L)	12	12	31	17	8	19	42	19	51	130	18	27
TDP (µg/L)	17	16	44	50	18	16	23	19	34	20	10	12
SUSPC (µg/L)	780	690	1500	910	550	1300	3110	1550	2350	8450	1290	2200
SRSI (mg/L)	9.500	9.900	6.250	7.360	8.520	8.570	8.890	9.000	4.510	5.870	6.230	5.930
TSS (mg/L)	7	6	13	3	3	14	47	11	20	57	8	17
SUSFE (µg/L)	251	263	475	214	179	415	1348	353	674	654	465	757
Na (mg/L)	37.40	36.10	23.20	20.10	25.00	23.40	27.80	28.00	181.0	1130.0	27.50	26.20
K (mg/L)	4.23	4.26	5.19	3.96	3.54	3.21	4.35	4.38	5.65	12.80	3.66	3.66
Mg (mg/L)	26.30	25.50	40.00	38.30	32.40	30.80	33.80	33.20	42.50	23.50	40.80	41.50
Ca (mg/L)	69.60	70.00	93.00	95.80	72.80	72.00	84.40	84.40	106.0	38.60	92.60	97.00
Fe (mg/L)	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	0.02	0.04	0.02	0.02
Mn (mg/L)	0.02	0.09	0.16	0.13	0.02	0.02	0.02	0.02	0.69	0.06	0.09	0.03
pH	8.54	8.50	8.45	8.41	8.51	8.53	8.48	8.48	8.38	8.38	8.49	8.46
ALK (ueq/L)	5330	5440	6900	7330	5800	5760	6260	6210	9240	14020	6940	6910

* Sample lost

Appendix H. Riparian vegetation measurements, types of vegetation, and main land used practices at mitigated and impacted sites in the study area. Riparian vegetation was not found to be significantly different between sites when tested with a Wilcoxon non-parametric test (n = 6, W+ = 15, W- = 0, $P \leq 0.0625$).

	Riparian Measurement		**Types of Riparian Vegetation**		**Main Land Use Practices**	
Stream	**Mitigated**	**Impacted**	**Mitigated**	**Impacted**	**Mitigated**	**Impacted**
Ranch Creek	100+	48.9	grass, trees, bushes	willow, grass, shrubs	field, pasture, agriculture	filed, pasture
Harper Creek	100+	13	grass, trees, bushes, willow	thistle, grass	filed, pasture	field, pasture
East Wilson River	100+	45.75	trees, willow, grass	grass, willow	agriculture	agriculture
West Wilson River	some 100+	29.4	grass, trees, bushes	dense willow	agriculture	field, pasture, agriculture
Sulphurspring Creek	some 100+	some 100+	oak, phrag, rush, grass	grass	forest	field, pasture, agriculture
Drifting River	100+	4.37	trees, shrubs, fern. horsetail, grass	willow, grass, sedge	forest	field, pasture

Appendix I. Simpson's diversity values for mitigated and impacted sites for six streams in the study area. No significant difference was observed for diversity between mitigated and impacted sites $t_{\alpha 0.05\ DF} = 5$, T= -0.19, P>0.05.

Sites	D-Mitigated	D-Impacted
Ranch Creek	0.3568	0.1221
Harper Creek	0.2297	0.2260
East Wilson River	0.2068	0.2103
West Wilson River	0.2237	0.2031
Sulphurspring Creek	0.3917	0.7694
Drifting River	0.4165	0.3846

Appendix J. Glossary of terms

BOD- Biological oxygen demand

CFU- Colony forming units

DO- Dissolved oxygen

EPTs- Ephemeroptera, Plecoptera, Trichoptera

LWD- Large woody debris

ORP- Oxidation-reduction potential

SRP- Soluble reactive phosphorous

SUSFE- Suspended iron

SUSPC- Suspended carbon

SUSPN- Suspended nitrogen

SUSPP- Suspended phosphorous

SRSI- Soluble reactive silica

TDN- Total dissolved nitrogen

TDP- Total dissolved phosphorous

TSS- Total suspended solids