Volume 3:

AESA Water Quality Monitoring Project

Assessment of Environmental Sustainability in Alberta's Agricultural Watersheds Project

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EXECUTIVE SUMMARY

All land-use activities impact watershed function to some degree. Research has shown that agricultural practices can lead to the deterioration of surface water quality by contributing contaminants such as nutrients (i.e., phosphorus and nitrogen), pesticides, sediments, and fecal bacteria to surface water bodies. The consequences of agricultural contamination to aquatic ecosystem and human health may be significant. Water quality issues in streams and receiving waterbodies include eutrophication, cyanobacterial blooms, and sedimentation of aquatic habitat, to name a few. Water quality concerns affecting human and livestock health are related to the presence of nitrate, pathogens, or pesticide contaminants in drinking water. While all surface water in Alberta must be treated prior to human consumption, livestock consume untreated or raw water. Furthermore, treatment of drinking water does not necessarily remove pesticides. The presence of pesticides or pathogens in irrigation water also poses a threat to food safety.

The impact of agricultural activities on water quality depends on the amount and distribution of land under cultivation, farming practices employed, soil type, topography, weather, and climate patterns. Agricultural activities that have potential to contribute contaminants to surface water include, but are not limited to, manure or fertilizer application, intensive livestock operations (e.g., feedlots, dairies), non-intensive livestock operations (e.g., pasture, cow-calf, watering sites), some tillage methods, pesticide application, and irrigation.

The Provincial Stream Survey, conducted under the Canada-Alberta Environmentally Sustainable Agricultural (CAESA) Agreement (1992 to 1997), included provincial-scale surface water quality monitoring in agricultural watersheds in Alberta. The monitoring goal of the CAESA Provincial Stream Survey was to determine the potential effects of agriculture on surface water quality. Fifteen streams in agricultural basins with different levels of agricultural intensity were monitored for two years (1995 and 1996). Higher peak, median, and flow weighted mean concentrations of total and dissolved nutrients and more frequent pesticide detections were observed in streams draining high agricultural intensity watersheds.

In 1998, the Province of Alberta took the lead in facilitating the promotion of environmentally sustainable agriculture practices by initiating the Alberta Environmentally Sustainable Agriculture (AESA) Program. The AESA Program followed the cessation of the Canada-Alberta Environmentally Sustainable Agriculture (CAESA) Agreement. The AESA program included soil and water quality monitoring programs as well as an extension program.

The AESA Stream Survey, initiated in 1999, was designed to assess temporal and spatial patterns in water quality in watersheds with agricultural activity. Twenty-three watersheds were selected to encompass the range of agricultural intensities throughout the province including low-agricultural intensity watersheds, watersheds already subject to high intensity farming, watersheds with the potential for intensified agriculture, and those draining irrigation return flows. Specifically, the two objectives for AESA Stream Survey were

i) to learn more about how stream water quality is impacted by low, moderate, and high intensity (dryland and irrigated) agriculture in Alberta; and
ii) to track changes in water quality as the industry grows and agricultural management practices change.

The purpose of this report is to describe water quality in representative agricultural streams in Alberta and any changes that occurred during the period of monitoring. This report compiles and evaluates water quality and quantity data collected from 1999 to 2006, as well as those data available from the earlier CAESA Provincial Stream Survey. This dataset provides reliable reference information for current and future studies aimed at quantifying and mitigating agricultural impacts on water quality.

The following section summarizes key conclusions from chapters 2 through 5 of the technical report. These findings contributed to Volume 1 of the Assessment of Environmental Sustainability in Alberta's Agricultural Watersheds project (Palliser Environmental Services Ltd. and ARD 2008).

Chapter 2: Hydrology, land cover, and agricultural intensity metrics.

Overland runoff is one of the dominant pathways for agricultural contaminants to reach waterways. As such, water quality is highly dependent on the timing and magnitude of stream flow.

The province-wide and multi-year AESA Stream Survey captured inherent variability in the timing and magnitude of stream flow among study streams and years that permitted generalization of the flow patterns in nine of the major ecoregions of the province's agricultural zone.

- Watersheds under high intensity (dryland) agriculture were predominantly located in the Aspen Parkland Ecoregion where streams often flowed only in the spring in response to overland runoff from snowmelt. In years with smaller snow packs, hydrological connectivity with land was low, as there was little response to summer rain events.
- Watersheds under low or moderate intensity agriculture were situated in the Boreal Transition and Western Alberta Upland Ecoregions where streams generally flowed continuously during the open water season in response to snowmelt and summer precipitation events.

The relative proportion of land cover categories was often similar among watersheds of the same agricultural intensity.

- The relative proportion of cropland, forage, grassland, trees, and shrubs varied among watersheds. Watersheds with high agricultural intensity (dryland or irrigated) generally had higher proportions of cropland compared to watersheds with low or moderate agricultural intensity.

Agricultural intensity percentiles differed among census years (1996, 2001, and 2006) in some watersheds, corresponding to changes in manure production and chemical and fertilizer sales percentiles.

- Evaluation of the agricultural intensity metric from 1999 to 2006 revealed a decrease in agricultural intensity categories in seven watersheds, generally as a result of decreases in manure production percentiles.
- Agricultural intensity percentiles increased in one watershed (Prairie Blood Coulee) between 1996 and 2006 as a result of increases in fertilizer and chemical sales percentiles.

- Differences between 1996 and 2006 percentiles in some watersheds were small and just spanned the cutoff between two agricultural intensity categories (e.g., Strawberry), while some watersheds remained within the same category even though a greater decrease (e.g., Rose Creek) or increase (e.g., Hines Creek) in the overall agricultural intensity percentile was observed.

Chapter 3: Nutrients.

Excessive nutrients, such as phosphorus (P) and nitrogen (N), may increase waterbody primary production resulting in eutrophication. Eutrophication has a negative effect on aquatic ecosystem health by decreasing oxygen levels and degrading water quality.

The results of the study suggest that small to moderately sized agricultural watersheds (3200 to 95000 ha effective drainage area) throughout the province have elevated levels of P and N, and concentrations increase as agricultural intensity increases.

- Compliance with provincial and national surface water quality guidelines decreased with increasing agricultural intensity.
  - Watersheds under high intensity agriculture had lower compliance with guidelines for the protection of aquatic life than low and moderate agricultural intensity watersheds. In high intensity watersheds, on average, only 9% of total nitrogen (TN) samples and 7% of total phosphorus (TP) samples met guideline values.
  - Protection of aquatic life guidelines set for nitrite N (NO_2-N), nitrate N (NO_3-N), and ammonia N (NH_3-N) were rarely exceeded.
  - Nitrite N and NO_2^--N+NO_3^--N livestock watering guidelines were never exceeded.
- Median annual total and dissolved N and P flow-weighted mean concentrations (FWMCs) and the proportion of dissolved to total N (TN) and P (TP) varied among streams and were influenced by agricultural intensity.
  - Total N and P FWMCs increased with agricultural intensity. Mean TP FWMCs were 0.15 and 0.53 mg L^{-1} for low and high intensity watersheds, respectively. Mean TN FWMCs were 1.09 and 3.12 mgL^{-1} for low and high agricultural intensity streams, respectively.
  - Total particulate P was not influenced by agricultural intensity.
  - A stepwise increase was observed for the ratio of dissolved inorganic N to total N (DIN/TN) with agricultural intensity (low<moderate<high<irrigated). The ratio of total dissolved P to total P (TDP/TP) was highest in the high agricultural intensity watersheds but did not show a stepwise, statistically significant trend among low and moderate or moderate and irrigated watersheds (low≤moderate=irrigated<high).
  - A higher proportion of TP was comprised of the dissolved fraction (mean TDP/TP=0.5) than was observed for TN (mean DIN/TN=0.2) for all watersheds.

Correlations between the dissolved nutrient fractions and the overall agricultural intensity metric supported use of the metric as an indicator of agricultural influence on nutrient concentrations in agricultural streams.

- Overall, dissolved nutrient FWMC fractions and the ratios of TDP/TP and DIN/TN were positively correlated with the agricultural intensity metrics (chemical and fertilizer expenses and manure production percentiles).
- The ratio of TDP/TP was positively correlated with fertilizer and chemical expenses, while DIN/TN was strongly, positively correlated with all three agricultural intensity metrics (including manure production percentiles).
- Strong correlations were not observed for the median annual P and N exports with the agriculture intensity metric.

Export coefficients were influenced by climatic and geographic characteristics including interannual and seasonal variation in stream flow, which differed among ecoregion areas.

- Total nutrient exports were highest in the Boreal ecoregion area and influenced by factors such as runoff depth, landscape, and climate, while nutrient exports were lowest in the Grassland ecoregion area. Higher dissolved nutrient proportions were observed in the Parkland Ecoregion as a result of higher agricultural intensity in the watersheds.
- Export coefficients for most forms of P and N were higher in the early spring in the Boreal and Parkland ecoregion areas, particularly in April in the Boreal streams. Streams in the Grassland ecoregion area did not show a seasonal trend in nutrient export coefficients, while higher TN export coefficients were observed in June in the Continental Divide Ecoregion.
- Similar to the CAESA findings, median annual TDP and NH_3-N export coefficients and the ratio of DIN/TN showed an increasing stepwise trend with increasing agricultural intensity, while the ratio of TDP/TP was higher in high agricultural intensity watersheds but similar among low and moderate agricultural intensity streams.
- Total P and N export coefficients in the AESA watersheds ranged from 0.012 to 0.214 $kg \cdot ha^{-1} yr^{-1}$ and 0.142 to 1.412 $kg\ ha^{-1}\ yr^{-1}$, respectively and were similar to values reported in other studies in Alberta and within the range of exports measured in Canada, the United States, and Europe.

Although there were some changes in agricultural intensity, temporal patterns in P and N FWMCs and nutrient loads were generally not observed during the period of monitoring (1995 to 2006).

- Many watersheds that exhibited inter-annual patterns in nutrient FWMCs were strongly influenced by flow.
- Median annual loading values were influenced by annual flow volumes, with high annual loading typically observed in years with high flow volumes.
    - Deviations from the flow-loading pattern were attributed to sampling regime (e.g., sampling that missed peak flows), climatic variability (e.g., floods or droughts), and a possible change in land management or land use.
- Based on visual inspection of the data, watersheds where a statistical trend analysis should be considered include Battersea Drain, Prairie Blood Coulee, Blindman River, Kleskun Drain, Meadow Creek, Tomahawk Creek, Buffalo Creek, Renwick Creek, and

Wabash Creek. A statistical trend analysis would verify whether an increasing or decreasing temporal trend in nutrient concentrations was present.

Chapter 4: Bacteria.

Bacteria are naturally found in the intestinal tract of mammals; however, coliforms, including *E. coli*, are often found in surface waters and may indicate a risk to human health. Agricultural activities that have the potential to contribute bacteria to surface waters include manure spreading, allowing direct cattle access to streams, and improper storage and handling of manure.

Peaks in fecal bacteria occasionally occur in agricultural streams and may indicate a risk to human or animal health.

- *E. coli* and fecal coliforms were commonly found in agricultural streams in Alberta. Sixty-nine percent of all samples collected (1999 to 2006) had detectable levels of *E. coli,* and 79% had detectable levels of fecal coliforms.
- Fecal bacteria counts in Alberta's agricultural streams were extremely variable ranging three orders of magnitude, from below method detection limits (<10 CFU·100 mL^{-1}) to 60 000 CFU·100 mL^{-1}.
- Annual counts of ambient fecal bacteria for individual streams were typically < 100 CFU·100 mL^{-1}, falling below water quality guidelines for irrigation and recreational use. However, occasional extreme peaks in fecal bacteria (> 1000 CFU·100 mL^{-1}) occurred, most often in summer months in association with peaks in discharge or suspended sediment.

The agricultural intensity metric was not a good predictor of streams with the highest risk of fecal contamination.

- Unlike nutrient concentrations, fecal bacteria counts did not show an increasing pattern with increasing agricultural intensity.
    - Annual geometric mean *E. coli* and fecal coliform counts were significantly higher in moderate intensity and irrigated streams than in low and high agricultural intensity watersheds.
    - Export coefficients were also highest for the moderate agricultural intensity streams.

The highest risk of fecal contamination occurred in watersheds in the Fescue, Moist, and Moist Mixed Grasslands Ecoregions in Southern Alberta.

- Annual geometric mean fecal coliform and *E. coli* counts were highest in the Grassland ecoregion area, followed by the Irrigated Grassland and Boreal ecoregions areas. Values were the lowest in the Parkland and Continental Divide Ecoregions, areas which are dominated by high and low intensity agriculture, respectively.
- *E. coli* and fecal coliform exports were significantly higher in the Grassland and Boreal ecoregions areas. These regions of the province have higher runoff potential and higher ambient bacteria counts. Export coefficients were not calculated for the Irrigated Grassland, but load calculations suggest that fecal loading was high from watersheds in this region.

Streams with high risk for fecal contamination and high bacteria exports do not necessarily coincide with streams with high in-stream nutrient concentrations or high nutrient exports.

- There was an inverse correlation between ambient fecal coliforms and nutrient parameters, specifically TN and TDP, indicating that streams with high nutrient concentrations do not necessarily have water quality issues related to fecal bacteria contamination.
- Bacteria exports from the Grassland ecoregion area were higher than would have been expected from nutrient exports. For example, nutrient exports for Trout and Meadow Creeks were among the lowest of the 23 AESA watersheds, while fecal bacteria exports were among the highest.

Chapter 5: Pesticides.

During the last few decades, pesticides have been essential for increasing agricultural productivity; however, pesticide exposure has been linked to human and aquatic health issues. Currently, there is widespread pesticide use throughout the agricultural industry in Alberta.

Low level concentrations of a variety of pesticides were commonly found in surface waters of agricultural watersheds. Each of the 23 AESA watersheds had at least one pesticide compound detected in stream water during the monitoring period.

- One or more of the 68 pesticide compounds monitored were detected in 64% of samples from 1999 to 2006.
- Thirty-seven of the 68 compounds monitored were detected. Of the total 68 compounds analyzed, detections included
    - 29 of 40 herbicides plus breakdown products and isomers,
    - 4 of 20 insecticides plus breakdown products and isomers,
    - 4 of 8 fungicides.
- Herbicides were detected more frequently than insecticides or fungicides, a finding that corresponds with the pesticide sales information.
    - The top two detected herbicides, 2,4-D and MCPA, were detected in 46% and 31% of samples, respectively. Another six herbicide active ingredients were detected in ≥10% of analyzed samples (clopyralid, triclopyr, dicamba, picloram, imazethabenz-methyl, and MCPP). The remaining 21 herbicide compounds were detected in <10% of samples.
    - The top detected insecticide was gammabenzehexachloride (lindane), found in 0.6% of samples.
    - The top detected fungicide was iprodione, found in 3.3% of samples.

Pesticide detection frequency, total pesticide concentration, and the total number of compounds detected increased significantly as agricultural intensity increased from low to high.

- In low intensity watersheds, 24% of samples had detectable levels of at least one pesticide. Total detection frequency increased to 80% in high intensity dryland watersheds and 91% in high intensity irrigated watersheds.
- Higher pesticide detections in high intensity watersheds were mirrored by higher total concentrations.

- The likelihood of pesticide mixtures occurring was also greater in high intensity agricultural watersheds, regardless of whether dryland or irrigated agriculture was practiced.
    - Low and moderate agricultural intensity watersheds generally had only one pesticide in a sample, while high intensity and irrigated watersheds typically had a mixture of pesticide compounds (two or more) per sample.
    - The median number of compounds detected per sample was two.
- Irrigated watersheds had a higher toxicity risk than high intensity dryland watersheds indicating total pesticide concentration alone is not a good measure of threat to aquatic life; it is important to know which compounds are present and in what concentrations.

Correlations between total pesticide detection frequency and the overall agricultural intensity metric supported the use of the metric as a predictor of the degree of pesticide contamination in small agricultural watersheds.

- There was a strong correlation between the intensity of agriculture in a watershed (as % cropland and fertilizer and chemical expense percentiles) and total pesticide detection frequency.
- Total pesticide concentrations appeared to be influenced by the type of water management used (irrigated versus dryland) as well as by the intensity of chemical use.

At a broad level, temporal patterns in pesticide detection frequency and total concentration were not observed during the period of monitoring (1999 to 2006); however, temporal patterns were observed for certain active ingredients.

- Though a potential downward pattern in total detections frequencies may be emerging for the 40 compounds routinely monitored from 1999 to 2006, this pattern is dampened when all 68 compounds were included in the analysis. Active ingredients in pesticides are changing with time as new products come on the market and specific pest outbreaks occur.
- Temporal patterns in certain individual active ingredients were observed with time either at provincial, regional, or site specific scales.
    - Imazamethabenz-methyl concentrations showed a qualitative increase from 2001 to 2006 at a provincial scale.
    - Picloram detection frequencies showed significant declines in high intensity dryland watersheds from 1999 to 2005 but slightly increased again in 2006.
    - Simazine detection frequencies increased in New West Coulee from 2002 to 2006.
- Of the top two detected herbicides, 2,4-D and MCPA, only MCPA demonstrated a substantial increase in pesticide sales from 1998 to 2003. A corresponding increase in detection frequency was not observed from 1999 to 2006.

The types of pesticides detected and the timing of peak concentrations varied geographically and were related to the type of agriculture practiced (irrigated versus dryland).

- Eight pesticide compounds were detected solely in watersheds under irrigated agriculture, reflecting the greater diversity of specialty crop types grown (e.g., simazine application to corn crops).

- Nine pesticide compounds were detected only in watersheds under dryland agriculture. For example, the fungicide iprodione, which is applied to canola and bean crops, was only detected in high intensity dryland streams.
- High agricultural intensity and irrigated watersheds showed peaks in total pesticide concentrations in spring (March) and summer (June or July).
    - In irrigated streams, total pesticide concentrations were highest in June following pesticide application, while total pesticide concentrations were highest in March during snowmelt runoff in watersheds with high intensity agriculture.

Guidelines for the protection of aquatic life (PAL) and irrigation application were exceeded in some samples.

- Watersheds under irrigated agriculture exceeded guidelines more frequently than high, moderate, and low agricultural intensity watersheds. Low agricultural intensity watersheds exceeded guidelines the least.
- Irrigation guidelines for MCPA and dicamba were exceeded most frequently (11.2 and 11.4% of samples, respectively), indicating potential for damage to sensitive plant species if stream water was used for irrigation purposes.
- Guidelines for PAL were exceeded for 2,4-D, MCPA, chlorpyrifos, lindane, and triallate but only in a small proportion of samples (0.2 to 0.5%).

The AESA Stream Survey confirmed the impact of agricultural activities on surface water quality in Alberta, echoing findings of the CAESA study (Anderson et al. 1998a, b). In general, higher agricultural intensity watersheds had the highest concentrations of nutrients and pesticides. Overall, it was evident that water quality in the AESA watersheds (as determined by nutrients, pesticides, and bacteria) was influenced by agricultural intensity factors (e.g., variability in land use and management) and ecoregional characteristics (e.g., climate and topography). Seasonal and inter-annual variability was observed in most watersheds for the majority of parameters.

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All partners provided technical input at AESA Water Quality Committee meetings and individually made the following contributions:

- ARD coordinated the program and provided funding, technical expertise in data analysis, report preparation, and extension material preparation.
- AENV provided technical expertise in data collection and management and provided funding for pesticide analyses.
- AHW provided bacterial analyses.

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LIST OF ABBREVIATIONS

AAWQI	Alberta Agriculture Water Quality Index
ac	Acres
AENV	Alberta Environment
AESA	Alberta Environmentally Sustainable Agriculture
AI	Agricultural Intensity
ANOVA	Analysis of Variance
ARD	Alberta Agriculture and Rural Development
CAESA	Canada-Alberta Environmentally Sustainable Agriculture
CCME	Canadian Council of Ministers of the Environment
CEQG	Canadian Environmental Quality Guidelines
CFU	Colony Forming Units
E. coli	*Escherichia coli*
FWMC	Flow Weighted Mean Concentration
GIS	Geographic Information Systems
ha	Hectares
km	Kilometers
LIND	Lethbridge Northern Irrigation District
MDL	Minimum Detection Limit
N	Nitrogen
NH_3-N	Ammonia Nitrogen
NH_4+	Ammonium
NFR	Non-Filterable Residue
NO_2-N	Nitrite Nitrogen
NO_3-N	Nitrate Nitrogen
NO_2^--N+NO_3^--N	Nitrite Nitrogen + Nitrate Nitrogen
ORG-N	Organic Nitrogen
P	Phosphorus
PAL	Protection of Aquatic Life
Q	Stream Discharge
QA	Quality Assurance
QC	Quality Control
SMRID	St. Mary Irrigation District
spp.	Species
TDP	Total Dissolved Phosphorus
TKN	Total Kjeldahl Nitrogen
TN	Total Nitrogen
TP	Total Phosphorus
TPP	Total Particulate Phosphorus
WDS	Water Data Server

Chapter 1: Study Objectives and Rationale

All land-use activities impact watershed function to some degree. Research has shown that agricultural practices can lead to the deterioration of surface water quality by contributing contaminants such as nutrients (i.e., phosphorus (P) and nitrogen (N)), pesticides, sediments, and bacteria to surface water bodies (Daniel et al. 1998; Baxter-Potter and Gilliland 1988; Sharpley and Syers 1979; Wauchope 1978). The consequences of nutrient enrichment, bacteria and sediment loading, and pesticide contamination to aquatic ecosystem and human health may be significant. Resulting water quality issues in streams and receiving waterbodies include eutrophication, cyanobacterial blooms, and sedimentation of aquatic habitat, to name a few. Water quality concerns affecting human and livestock health relate to the presence of nitrate, pathogens, or pesticide contaminants in drinking water. While all surface water in Alberta must be treated prior to human consumption, livestock consume untreated or raw water. Furthermore, treatment of drinking water does not necessarily remove pesticides. The unintended application of pesticides to crops through irrigation water can dramatically affect crop production. The presence of pesticides or pathogens in irrigation water also poses a threat to food safety (Van de Venter 2000).

Ecoregions are areas of similar climate, geology, landscape, soils, and natural vegetation. With this natural variability across the province, ecoregions are an important scale for examining water quality. The natural physical and climatic characteristics within an ecoregion have a tendency to lead to similar land-use patterns. Correlations between land use (e.g., bare ground, vegetative cover, wetlands, and urban spaces) and water quality have been found in many countries, including Canada (Galbraith and Burns 2007; Houlahan and Findlay 2004; Herlihy et al. 1998). Specifically pertaining to agricultural land uses, Mattikalli and Richards (1996) found an increasing trend of P and N and with the intensification of agriculture and fertilizer usage. Other studies have found P and N concentrations in intermittent prairie watersheds to increase where there was an increased density of cropping (Dodds and Oakes 2006; Little et al. 2003). The impact of agricultural activities on water quality will depend on not only the proportion of land under cultivation, but also on the farming practices employed (i.e. timing, amounts, and placement), soil type and topography, weather and climate patterns. Agricultural activities that have potential to contribute contaminants to surface water include but are not limited to: applying manure or fertilizer to agricultural fields, intensive livestock operations (i.e. feedlots, dairies, wintering sites), non-intensive livestock operations (i.e. pasture, cow-calf, watering sites), tilling fields to release sediment, applying pesticides, and irrigating (releases trace elements and salts).

The environmental concerns arising from agricultural activities, including nutrient and pesticide loading and bacterial contamination, are also a reality in Alberta. The Provincial Stream Survey, which was conducted under the Canada-Alberta Environmentally Sustainable Agricultural (CAESA) Agreement (1995 to 1996), was specific to agricultural watersheds in Alberta and found higher peak, median, and flow weighted mean concentrations of total and dissolved nutrients and more frequent pesticide detections in streams draining high agricultural intensity watersheds (Anderson et al. 1998a, b, c).

The agriculture industry has continued to grow in Alberta. In 2006, Alberta had 21.1 million ha of agricultural land, representing 31.2% of the national total (Statistics Canada 2006). There were 49,500 farms with an average size of 427 ha of agricultural land. Out of these farms, the two major sectors were beef cattle (41.5%) and wheat, grain and oil seed (25.2%). With the majority of agriculture in Alberta relating to the beef cattle sector, it is not surprising that Alberta has the largest cattle and calf inventories compared to other provinces. In 2006, Alberta had 6.3 million head followed by Saskatchewan at 3.5 million head. Other livestock inventories including pigs, sheep, and lambs were less dominant. Overall, these large numbers of livestock across Alberta have the potential to contribute significant quantities of nutrients and bacteria into surface water through hydrologically linked pathways, or by direct access of livestock to waterways.

Given the large area of the province designated as crop land, Alberta had the highest pesticide sales of all Canadian provinces in 2003, the majority of them being herbicides (Byrtus 2007). Within the Alberta agricultural sector, the top active ingredients sold in 2003 include glyphosate, MCPA, and 2,4-D (Byrtus 2007). Pesticide sales in the Oldman River basin accounted for 21% of the provincial sales followed closely by the Red Deer River basin (18%), the North Saskatchewan River basin (15%), and the Battle River basin (15%). These values are not surprising given all four basins have the highest area of cultivated land in Alberta (Byrtus 2007).

Nutrients, pesticides, and bacterial contamination are also a concern in Alberta's irrigated areas. A small portion (approximately 4%) of Alberta's agricultural land is irrigated; however, it provides approximately 16 % of the total agricultural production (Hecker 2002). The 13 irrigation districts provide water to 550,000 ha of farmland with an 8000-km conveyance system. All 13 irrigation districts lie in the southern portion of the province and are used to irrigate cereal, forage, and specialty crops including potato, sugar beet, and sunflower. Agricultural production is always the most intense on irrigated land. Inputs on irrigated land are far greater than those for dryland agriculture. Thus, increased fertilizer and pesticide use combined with water application (and drainage) predispose irrigated lands to environmental risks.

The increased awareness of these environmental concerns across Alberta is evident in the increased use of environmentally friendly practices such as no-till seeding and conservation tillage. In 2006, 75% of land prepared for seeding used environmentally improved practices, which is much higher than in 1991 when only 27% of the land base used environmentally improved practices (Statistics Canada 2006).

In 1998, the Province of Alberta took the lead in facilitating the implementation of environmentally sustainable agriculture practices by initiating the Alberta Environmentally Sustainable Agriculture (AESA) Program (AAFRD 1999). The AESA Program followed the cessation Canada-Alberta Environmentally Sustainable Agreement (CAESA), a five-year (1992 to 1997) cost-shared program that aimed to decrease the environmental impacts of agriculture on the environment. The CAESA and AESA Programs included provincial-scale surface water quality monitoring components. In the CAESA Provincial Stream Survey, the monitoring goal was to define the effects of agriculture on surface water quality. Small streams in agricultural basins with different levels of agricultural intensity streams were monitored for two years (1995 and 1996). The AESA Stream Survey, initiated in 1999, was designed to be longer term and assess temporal and spatial patterns in water quality in watersheds with agricultural activity, as recommended in the CAESA findings. Furthermore, watersheds were selected to encompass the range of agricultural intensities across the province including low-agricultural intensity reference watersheds, watersheds already subject to high intensity farming, watersheds with the potential for intensified agriculture, and those under irrigated agriculture. Specifically, the two objectives for AESA Stream Survey were as follows:

iii) to learn more about how stream water quality is impacted by low, moderate, and high intensity (dryland and irrigated) agriculture in Alberta; and

iv) to track changes in water quality as the industry grows and agricultural management practices change.

The purpose of this report is to address the two AESA Stream Survey objectives using water quality data collected from 1999 to 2006, as well as data available from the earlier CAESA Provincial Stream Survey. Due to the large volume data, this report is divided into five chapters:

- Chapter 1: Study Rationale and Objectives
- Chapter 2: Study Design, Hydrology, Land Cover, and the Agricultural Intensity Metric
    - Provides background information on the study design and discusses watershed characteristics such as hydrology (including precipitation), land cover, and the census-based agricultural intensity classifications from 1996, 2001, and 2006.
- Chapter 3: Nutrients
    - Discusses nutrient findings including compliance with surface water quality guidelines and interpretation of flow weighted mean concentrations, mass transport, and export coefficients. Seasonal and temporal patterns as well as the influence of agricultural intensity and ecoregional characteristics are examined.
- Chapter 4: Bacteria
    - Discusses ambient *E. coli* and fecal coliform concentrations, geometric means, export coefficients, and loads. Seasonal patterns are examined as well as the influence of agricultural intensity and watershed location within the province (ecoregion area).
- Chapter 5: Pesticides
    - Evaluates pesticide concentrations and detection frequencies with respect to pesticide guidelines, the Alberta Pesticide Toxicity Index, and agricultural intensity. Seasonality and interannual variability are examined. The relationship between pesticide occurrences and land cover, chemical and fertilizer expenses, and pesticide sales data is also discussed.

Figures and tables not illustrated within the chapters can be found in the Appendices at the end of the report.

Chapter 2: Study Design, Hydrology, Land Cover, and the Agricultural Intensity Metric

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INTRODUCTION

Other studies have shown that patterns in water quality may be affected by activity on land (Daniel et al. 1998; Baxter-Potter and Gilliland 1988; Sharpley and Syers 1979; Wauchope 1978) as well as the magnitude and timing of runoff and stream flow (Tate et al. 1999; Anderson et al. 1998b). The AESA Stream Survey study design is discussed in Chapter 2 to provide a background for subsequent chapters. Precipitation, hydrology, sampling regime, land cover, and agricultural intensity are also examined in order to facilitate interpretation of nutrient, bacteria, and pesticide data. These influential factors are frequently discussed throughout the report (Chapters 3, 4, and 5).

Objectives

The objectives of Chapter 2 are as follows:

i. Understand precipitation and hydrology patterns (1995 to 2006) in the 23 AESA watersheds and the similarities and/or differences among Ecoregions;
ii. Examine the flow biased sampling regime and its implications in subsequent data interpretation;
iii. Characterize land cover in the AESA watersheds and understand the variations in land cover among watersheds under differing agricultural intensities; and
iv. Evaluate the agricultural intensity metric and examine changes in agricultural intensity percentiles over the three census years (1996, 2001, and 2006).

MATERIALS AND METHODS

Project Design and Site Descriptions

The AESA Stream Survey examined water quality in 23 small, representative agricultural watersheds across Alberta. The watersheds were selected through a process that took into account agricultural intensity, regional climate, and runoff likelihood (Anderson et al. 1999). Watersheds were intentionally biased toward soil and landscape features that promote runoff as this is a primary transport mechanism for agricultural contaminants between land and water (Anderson et al. 1999). Runoff potential was determined by considering annual rainfall, landform characteristics, soil texture (clay, loam, silt), and depth of the topsoil layer in the watershed. Watersheds were relatively small in size (3200 to 137000 ha) to minimize the influences of non-agricultural activity on water quality.

Agricultural intensity (AI) metrics are used to link a watershed's agricultural activities with potential environmental impacts. Agricultural intensity was evaluated using three metrics: fertilizer expenses, chemical expenses, and manure production (Johnson and Kirtz 1998; Anderson et al. 1999). Data were obtained from the 1996 Canada Census of Agriculture (Statistics Canada 1996). Fertilizer and chemical expenses were used as indicators of crop production (e.g., oilseeds/grains), and manure production was used as an indicator of livestock production (e.g., cow-calf, intensive livestock operations). Chemical expenses, fertilizer expenses, and manure production per unit area were determined for all watersheds in the province, and each watershed was ranked with respect to the rest of the watersheds. The sum of the ranks for the three metrics was ranked again, and the final rank represented the measure of agricultural intensity as defined in the AESA water quality monitoring program (Anderson et al. 1999). The AESA watersheds were selected based on these final ranks.

Agricultural intensity classes based on stream-basin percentile ranking were defined as follows:

Non-agricultural = 0
0 < Low Agricultural Intensity > 40
40 < Moderate Agricultural Intensity > 75
High Agricultural Intensity > 75

Of the 23 watersheds selected, five had low agricultural intensity, six had moderate agricultural intensity, eight had high agricultural intensity, and four had high intensity agriculture in addition to receiving irrigation return flows. These watersheds were selected because they covered the range of agricultural intensities that typify the province (Anderson et al. 1999) and irrigated and dryland agriculture. Most of the high agricultural intensity watersheds have low to moderate runoff potential, whereas watersheds with low and moderate agricultural intensity are in areas of moderate to high runoff (Anderson et al. 1999).

AESA watersheds also capture the range of climatic and ecological characteristics by spanning nine different Ecoregions across the province (Figure 2.1). For the purpose of reporting, the specific Ecoregions were grouped into five general 'ecoregion areas' that are similar to the Natural Regions of Alberta. The five general ecoregion areas include the Boreal (includes four Ecoregions), Aspen Parkland (includes one Ecoregion), Continental Divide (includes one Ecoregion), Grassland (includes one Ecoregion), and Irrigated Grassland (includes two Grassland Ecoregions with the irrigated watersheds) (Table 2.1).

The 23 AESA watersheds are in four of Alberta's seven major river basins: South Saskatchewan River (Bow, Red Deer, and Oldman River sub-basins), North Saskatchewan River (Battle River sub-basin), Peace/Slave River, and Athabasca River (Table 2.1).

Fifteen of the watersheds in the AESA Stream Survey were originally monitored as part of the CAESA study (1995 to 1998), and eight watersheds were added to the monitoring suite in 1999 (Table 2.1). The additional watersheds included Hines Creek, Grande Prairie Creek, and Kleskun Drain in north western Alberta; Wabash Creek northwest of Edmonton; Willow Creek in the South-Western foothills region; and New West Coulee, Drain S-6, and the Battersea Drain in the irrigated, southern portion of the province.

Table 2.1. Agricultural intensity, runoff potential, major river basin (or sub-basin), and ecoregion area categories of the watersheds studied from 1999 to 2006.

AESA stream	HYDAT station code	Gross drainage area (ha)	Effective drainage area (ha)	Runoff potential	Ecoregion Area	Major river basin or sub-basin
Low intensity- dryland						
Hines Creek (HIN)	07FD011	37400	37400	High	Boreal	Peace River and Slave River
Paddle River (PAD)	07BB011	25300	25300	Moderate	Boreal	Athabasca River
Prairie Blood Coulee (PRA)	05AD035	22600	22600	High	Grassland	Oldman River
Rose Creek (ROS)	05DE007	55900	55900	Moderate	Boreal	North Saskatchewan River
Willow Creek (WIL)	05AB040	6530	6530	High	Continental Divide	Oldman River
Moderate intensity- dryland						
Blindman River (BLI)	05CC008	35300	35300	Moderate	Boreal	Red Deer River
Grande Prairie Creek (GRA)	07GE003	14000	14000	High	Boreal	Peace River and Slave River
Kleskun Drain (KLE)	07GE002	3200	3200	High	Boreal	Peace River and Slave River
Meadow Creek (MEA)	05AB029	13000	13000	High	Grassland	Oldman River
Tomahawk Creek (TOM)	05DE009	9530	9530	High	Boreal	North Saskatchewan River
Trout Creek (TRO)	05AB005	44100	44100	High	Grassland	Oldman River
High intensity- dryland						
Buffalo Creek (BUF)	05FE002	71400	14700	Moderate	Parkland	Battle River
Haynes Creek M1 (HM1)[z]	-	2400	2400	Moderate	Parkland	Red Deer River
Haynes Creek M6 (HM6)	05CD006	16600	16600	Moderate	Parkland	Red Deer River
Ray Creek (RAY)	05CE010	4440	4440	Moderate	Parkland	Red Deer River
Renwick Creek (REN)	05CE011	5900	5810	Moderate	Parkland	Red Deer River
Strawberry Creek (STW)	05DF004	59200	58900	High		North Saskatchewan River
Stretton Creek (STT)	05EE005	7400	5630	Low	Parkland	North Saskatchewan River
Threehills Creek (THR)	05CE018	19900	13800	Moderate	Parkland	Red Deer River
Wabash Creek (WAB)	07BC007	34400	34400	Moderate	Parkland	Athabasca River
Streams receiving irrigation return flows (irrigated watersheds)[y]						
Battersea Drain (BAT)	-	8700	-	High	Irrigated	Oldman River
Crowfoot Creek (CRO)	05BM008	137000	-	High	Irrigated	Bow River
Drain S6 (DS6)	-	3200	-	High	Irrigated	South Saskatchewan River
New West Coulee (NEW)	05BN006	31800	-	High	Irrigated	Bow River

[z] Haynes Creek M1 is a sub-watershed of Haynes Creek M6.

[y] The effective drainage area for each irrigated watershed is not show as there was uncertainty in the actual contributing areas for these watersheds.

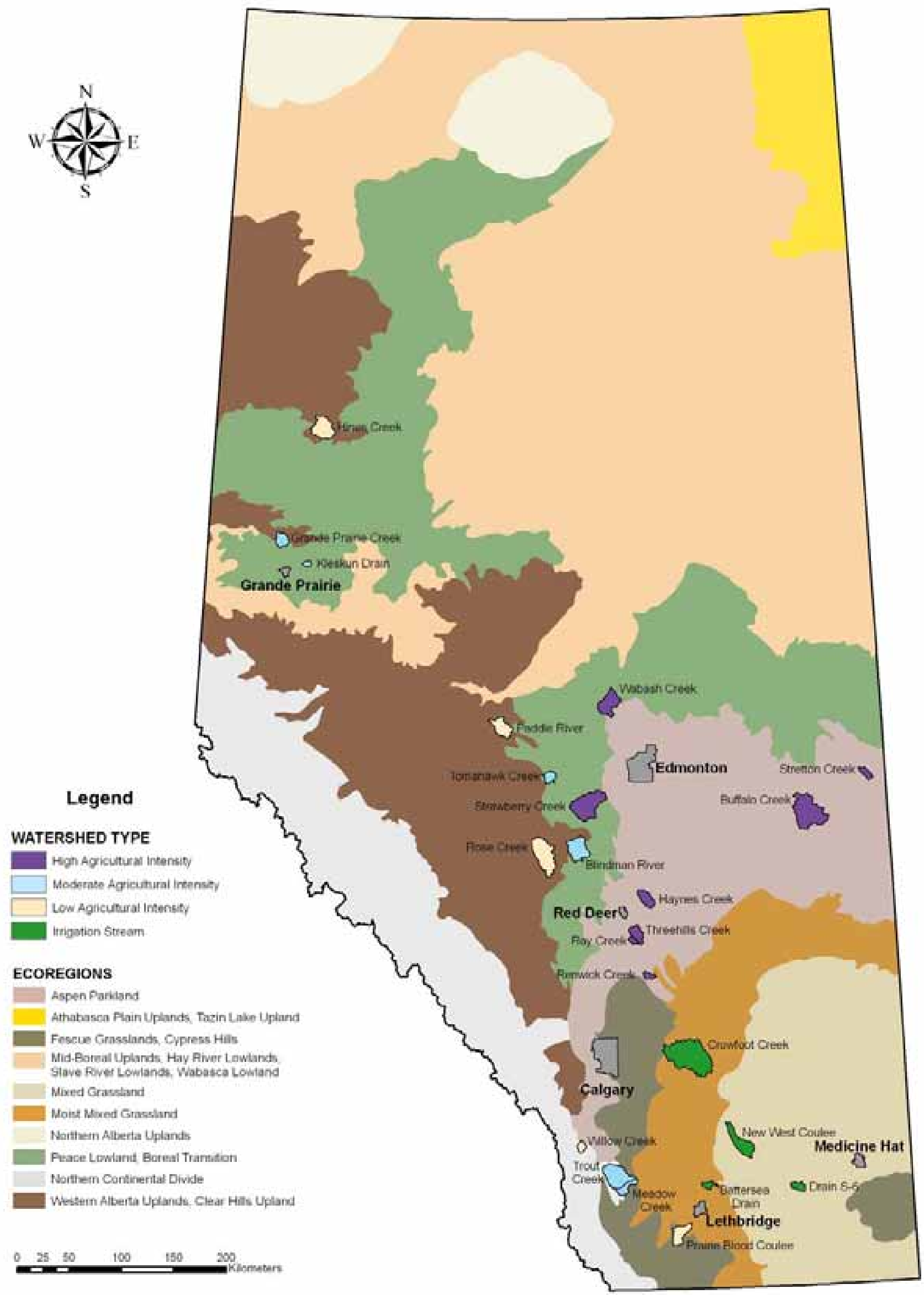

Figure 2.1. Map of Alberta showing the location of the 23 AESA watersheds in relation to Ecoregion and white zone (agricultural zone) boundaries.

Monitoring Period

There are up to 12 years of continuous water quality data for streams sampled under the CAESA and AESA Stream Surveys (1995 to 2006) and eight years of data for the streams sampled under the AESA Stream Survey alone (1999 to 2006) (Table 2.2).

Sampling records for five of the 15 CAESA/AESA watersheds were interrupted in 1998 during the transition between programs. The sampling location in Willow Creek was changed in 1999, and data from this watershed were only analyzed between 1999 and 2006.

In this report, the water quality comparisons made among watershed categories (i.e., agricultural intensity or ecoregion area categories) were completed on the eight-year 1999 to 2006 dataset for the 23 AESA streams. This ensured that observed differences could be attributed to the factors under investigation and that data were collected during years with comparable weather and stream flow conditions.

Water quality patterns in individual streams (i.e., inter-annual or seasonal patterns) were examined using the entire period of record available (up to 15 years).

Stream Sampling Methods

All sampling locations were in close proximity to the watershed outlet and situated in the immediate vicinity of an active Water Survey of Canada flow gauging station or a gauging station operated by AENV or the Irrigation District. Note that Willow Creek was sampled in the headwaters rather than the watershed outlet in order to monitor a low agricultural intensity dryland watershed in the southern part of the province (irrigation occurs in the lower portions of the watershed). The AESA Stream Survey field procedures were based on Alberta Environment's protocols and were followed for all sample collections (Depoe and Fountain 2003).

Grab samples were collected throughout the open water season (March 1 to October 31), beginning with spring melt/freshet and continuing until fall. A flow-proportionate (flow-biased) sampling regime was followed, with more frequent sample collection during periods of high flow:

- Nutrient and bacteria samples were collected twice per week during runoff periods, once per week as runoff subsided, then every two weeks and monthly as stream flow returned to base-flow conditions.
- Pesticide samples were collected once per week during peak runoff, then once every two weeks to once monthly as stream flow decreased.

Table 2.2. Watersheds monitored in the CAESA and AESA Stream Surveys from 1995 to 2006.

Watershed	CAESA				AESA							
	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Battersea Drain near the mouth				*	*	*	*	*	*	*	*	*
Blindman River near Bluffton	*	*	*	*	*	*	*	*	*	*	*	*
Buffalo Creek at Highway No. 41	*	*	*		*	*	*	*	*	*	*	*
Crowfoot Creek near Cluny	*	*	*	*	*	*	*	*	*	*	*	*
Drain S-6 near Bow Island					*	*	*	*	*	*	*	*
Grande Prairie Creek near Sexsmith					*	*	*	*	*	*	*	*
Haynes Creek near Haynes	*	*	*	*	*	*	*	*	*	*	*	*
Hines Creek above Gerry Lake					*	*	*	*	*	*	*	*
Kleskun Hills Main Drain near Grande Prairie					*	*	*	*	*	*	*	*
Meadow Creek near the mouth	*	*	*		*	*	*	*	*	*	*	*
New West Coulee near the mouth					*	*	*	*	*	*	*	*
Paddle River near Anselmo	*	*	*		*	*	*	*	*	*	*	*
Prairie Blood Coulee near Lethbridge	*	*			*	*	*	*	*	*	*	*
Ray Creek near Innisfail	*	*	*	*	*	*	*	*	*	*	*	*
Renwick Creek near Three Hills	*	*	*	*	*	*	*	*	*	*	*	*
Rose Creek near Alder Flats	*	*	*	*	*	*	*	*	*	*	*	*
Strawberry Creek near the mouth	*	*	*	*	*	*	*	*	*	*	*	*
Stretton Creek near Marwayne	*	*	*	*	*	*	*	*	*	*	*	*
Threehills Creek below Ray Creek	*	*	*	*	*	*	*	*	*	*	*	*
Tomahawk Creek near Tomahawk	*	*	*	*	*	*	*	*	*	*	*	*
Trout Creek near Granum	*	*	*		*	*	*	*	*	*	*	*
Wabash Creek near Pibroch					*	*	*	*	*	*	*	*
Willow Creek at Secondary 532					*	*	*	*	*	*	*	*

The total number of samples collected for each stream varied from year to year as a function of stream flow and runoff episodes.

Hydrology and Stream Flow Monitoring

For the majority of AESA watersheds, continuous daily stream discharge ($m^3\ s^{-1}$) was recorded at Water Survey of Canada (WSC) gauging stations (HYDAT station codes in Table 2.1). Alberta Environment maintained many of the gauging stations on the water quality monitoring network, with assistance from irrigation districts (St. Mary River Irrigation District (SMRID), Lethbridge Northern Irrigation District (LNID), and Bow River Irrigation District). Flow records for Drain S6 were maintained by SMRID. Battersea Drain flow records were maintained by LNID. Staff gauge readings and subsequent discharge curves for Battersea Drain were calculated by ARD staff.

Complete flow records were not available for all 23 watersheds:

- Tomahawk Creek was missing flow data for May and June of 1999.
- Willow Creek was missing flow data in April and parts of May 2000.
- New West Coulee was missing flow data from March of every year (1999-2006), while Drain S6 was missing flow data from March and April of every year (1999-2006). Stream flow was not monitored in these two streams before the irrigation canals began diverting flow as flow was minimal.
- Battersea Drain was missing flow data from March to mid or late April of every year (1998 to 2006) and at the very end of October for most years when flow was minimal or had stopped. No data were available for part of May in 2002.

Mean annual unit runoff was calculated each monitoring year by dividing the total volume of water in stream flow from March to October by the effective drainage area and expressed as millimeters (mm) depth of water per year (Anderson et al. 1998b). Monthly unit runoff was calculated as total monthly volume divided by the effective drainage area. Similarly, historical values were calculated for each stream using all available historical flow information. Unit runoff is the standard unit of measure that facilitates comparison of stream flow for basins of different sizes. However, unit runoff measures were not calculated for irrigation drains as canals may enter the watershed and drain water from outside the natural watershed boundary.

Historic hydrometric records ranged in length from 10 years (Willow Creek) to 98 years (Trout Creek). The Willow Creek AESA monitoring site was moved to its current location in 1999, and flow information had only been collected at this site since 1996. Consequently, Willow Creek had a short flow record, whereas other streams had an average of 31 years of flow monitoring data available for calculations of historical means. Historical mean annual runoff depths were calculated using Water Survey of Canada data.

Precipitation

Annual precipitation data (1995 to 2006) were obtained for the township located in the geographic centre of each AESA watershed. Township centers were identified using the *pointed* *polygon* function of ARC GIS software. Due to the nature of precipitation data collection, the majority of township rainfall data were interpolated based on surrounding weather stations. Weather station data were supplied by Alberta Environment, Environment Canada, and Alberta Agriculture and Rural Development and interpolated using Ab-Clim 1.0.

Provincial-scale historical precipitation data (1971 to 2000) were also examined.

Water Sample Analysis

Stream water samples were analyzed for the following parameters:

- nutrients (total and dissolved forms of nitrogen and phosphorus)
- fecal bacteria (fecal coliforms and *Escherichia coli* (*E. coli*))
- pesticides (herbicides, insecticides, and fungicides), and
- pH, temperature, non-filterable residue (NFR), total dissolved solids, and conductivity.

Table 2.3 lists all water quality parameters investigated in the AESA Stream Survey. Nutrients, fecal bacteria, and pesticide data are presented in detail in other chapters in this report. Summary statistics for additional chemical parameters can be found in the Appendix 18. Note that the pesticide analytical suite was expanded throughout the study when additional compounds of concern and their degradation products were identified. Additional details on the pesticide analytical suite can be found in Chapter 5.

The AESA Stream Survey was part of Alberta Environment's Quality Assurance (QA) program, which included quality control samples (QC) and data management in their Water Data System (WDS). This was an essential part of the program to ensure that the data collected were reliable and accurate for future analyses and reporting. The QA/QC sampling program included replicate samples, split samples, and field and lab blanks for all water quality parameters and spiked samples for pesticides. These samples were included to assess accuracy (spikes), precision (splits), and contamination (blanks). Quality control data were removed from the data set prior to the analysis of the various parameters. A discussion of the QA/QC data is not included in this report.

Nutrient analyses were conducted at the Alberta Research Council in Vegreville (1996 to 1998) and Envirotest Laboratories in Edmonton (1999 to 2006), and pesticide analyses were conducted at the Alberta Research Council in Vegreville. The Provincial Laboratories for Public Health in Edmonton and Calgary provided bacterial enumeration.

Table 2.3. Water quality parameters measured in the AESA Stream Survey.

Nutrients		
Total phosphorus (TP)	Total kjeldahl nitrogen (TKN)	Total nitrogen (TN), calculated
Total dissolved phosphorus (TDP)	Nitrite and nitrate nitrogen (NO_2 + NO_3-N)	
Total particulate phosphorus (TPP), calculated	Nitrite (NO_2-N)	
	Ammonia Nitrogen (NH_3-N)[z]	
Other measurements related to inorganic chemistry		
Suspended solids/ non-filterable residue	pH	Temperature
Fecal bacteria		
Escherichia coli	Fecal coliforms	
Pesticides[y]		
2,4-D	Dicamba	MCPA
2,4-DB	Dichlorprop	MCPB
2,4-Dichlorophenol	Diclofop-methyl	MCPP
Aldicarb	Dieldrin	Metalaxyl-m
Aldrin	Dimethoate	Methomyl
Alpha-Benzenehaxachloride (Alpha-BHC)	Disulfoton	Methoxychlor
Alpha-endosulfan	Diuron	Metolachlor
Atrazine	Ethalfluralin	Metribuzin
Desethyl atrazine	Ethion	Napropamide
Desisopropyl atrazine	Ethofumersate	Oxycarboxin
Azinphosmethyl (Guthion)	Fenoxaprop-p-ethyl	Parathion
Bentazon	Fluazifop	Phorate
Bromacil	Fluroxypyr	Picloram
Bromoxynil	Gamma-Benzenehexachloride (Lindane)	Propiconazole
Carbathiin	Glyphosate	Pyridaben
4-Chloro-2-Methylphenol	Aminomethyl Phosphonic Acid	Quinclorac
Chlorothalonil	Glyfosinate	Quizalofop
Chlorpyrifos	Hexaconazole	Simazine
Clopyralid	Imazamethabenz-methyl	Terbufos
Clodinafop-propargyl	Imazamox	Thiamethoxam
Clodinafop Acid Metabolite	Imazethapyr	Triallate
Cyanazine	Iprodione	Triclopyr
Diazinon	Linuron	Trifluralin
	Malathion	Vinclozolin

[z] Analytical method detects both NH_4^+ and NH_3 forms of N
[y]Note: Not all pesticides listed were monitored every year.

Land Cover

Land cover in the AESA watersheds was assessed using geometrically corrected aerial photographs (or orthophotos) taken in 1991. The percent land cover for each major land cover classification (e.g., cropland, forage, grassland, trees and shrubs) was calculated in ArcMap GIS to show spatial variation throughout the province. Individual watershed maps, color-coded by land cover type, are included in Appendix 1.

Agricultural Intensity Metrics: Changes with Time

Agricultural intensity (AI) in the context of the AESA Stream Survey was defined by three Census of Agriculture parameters (fertilizer expenses, chemical expenses, and manure production) that were highly correlated to the presence of agricultural contaminants in surface water (Anderson et al. 1998a, b, c). Intensity metrics were generated from data collected in 1996, 2001, and 2006 (Statistics Canada 1996, 2001, and 2006) to evaluate changes in the agriculture industry with time. The goal was to relate changes in the agricultural intensity of each watershed with time (e.g., increases or decreases in manure production) to patterns in surface water quality.

Changes in AI metrics must be interpreted with caution as slightly different methods were used each Census year to deal with the issue of confidentiality and data suppression. A minimum of 15 farms is required in a polygon (Soil Landscape of Canada (SLC) polygon or PFRA watershed scale) for the Census data to be reported. If there were fewer than 15 farms, polygons were amalgamated with a nearby polygon, preferably of similar size (area) and density to avoid geographic dilution (i.e., a large polygon with few points combined with a small polygon with a high density of points). The polygon groupings differed among years. In 1996, SLC polygons were grouped with the closest polygon of similar size (Anderson et al. 1999). In 2001, AESA watersheds were grouped into polygons by flow path if the original AESA watershed had less than 15 farms. In 2006, AESA watershed boundaries were grouped into polygons by agricultural land practices if there were less than 15 farms in the original polygon. By grouping an AESA watershed with the closest polygon with similar agricultural land practices, some amalgamated polygons may have crossed major basin boundaries. Thorough method comparisons were conducted between 1996 and 2001 but less so from 2001 to 2006. Watersheds with a differing amalgamated area between 2001 and 2006 include Meadow Creek, Renwick Creek, Prairie Blood Coulee, Stretton Creek, and Willow Creek. Also, AI percentiles in Hines Creek, Prairie Blood Coulee, and Kleskun Drain were calculated on amalgamated areas that exceeded the watershed boundary.

It should also be noted that Kleskun Drain was classified as a high AI watershed according to 1996 Census of Agriculture data; however, expert review of the watershed determined that a moderate AI category better described the basin (pers. comm. A-M. Anderson, 2008). The Kleskun Hills, not under agriculture, formed a large portion of the watershed and contributed much of the flow. In contrast, the lower portion of the watershed was intensively farmed.

Correlations between average Census of Agriculture metrics (1996, 2001 and 2006 for agricultural intensity, manure production, and fertilizer and chemical sales percentiles) were run in SYSTAT 10 (SPSS Inc. 2000). Spearman Rank correlations were run on untransformed data for all AESA streams excluding Drain S6. The data available for Drain S6 were based on areas larger than the actual watershed study area and deemed not suitable for the analyses.

RESULTS AND DISCUSSION

Precipitation

The 30-year average total annual precipitation (1971 to 2000) for the province as a whole (Figure 2.2) illustrates that the Boreal ecoregion area and south-western part of the Fescue Grasslands Ecoregion received the highest amounts of precipitation. Precipitation data for the AESA watersheds (March to October, 1995 to 2006 and 1999 to 2006) showed similar regional patterns (Table 2.4). However, it is noteworthy that the watersheds situated in the Peace region (Hines Creek, Grande Prairie Creek, and Kleskun Drain) and Wabash Creek, located northwest of Edmonton, received somewhat lower precipitation than other watersheds in the Boreal ecoregion area. Within the Fescue Grasslands Ecoregion, Prairie Blood Coulee received less precipitation than Trout or Meadow Creeks.

The irrigated watersheds, located in the Mixed Grassland and Moist Mixed Grasslands Ecoregions, had the lowest average precipitation totals for the period of study (about 300 mm), while the Continental Divide had the highest average precipitation total of the Ecoregions (about 490 mm). Of the 23 individual streams under study, Rose Creek (Boreal ecoregion area, low agricultural intensity) had the highest March to October precipitation totals, and Drain S-6 (Mixed Grassland Ecoregion, irrigated agriculture) had the lowest precipitation.

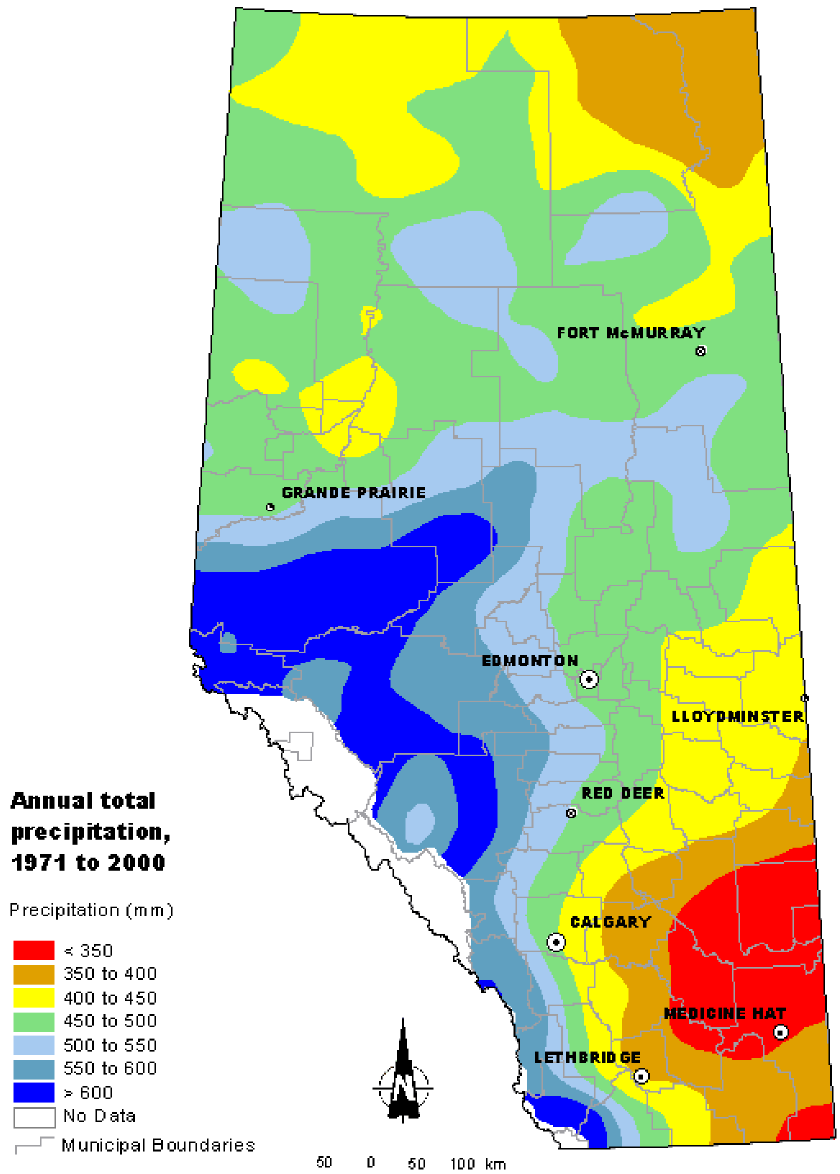

Figure 2.2. The 30-year average annual total precipitation (1971 to 2000) in Alberta based on data from Environment Canada, Alberta Environment, and the National Climate Data Centre.

Table 2.4. Annual precipitation (mm) estimated for the centre of each AESA watershed. Precipitation totals are from March 1 to October 31 annually to correspond with the water quality sampling period.

Stream (Township Centre ID)	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	95-06 Average	99-06 Average
Boreal														
Blindman River (T044R03W5)	440	527	529	494	473	537	397	357	380	395	452	394	448	423
Grande Prairie Creek (T074R06W6)	378	434	472	269	220	439	361	334	319	514	312	297	362	350
Hines Creek T086R02W6)	242	425	463	236	281	543	360	282	306	372	274	283	339	338
Kleskun Drain (T072R04W6)	402	433	479	294	222	402	335	354	319	495	309	301	362	342
Paddle River (T056R11W5)	472	564	521	380	362	552	448	306	373	476	500	380	444	424
Rose Creek (T044R07W5)	520	512	570	575	524	587	430	399	390	542	528	544	510	493
Strawberry Creek (T049R02W5)	377	510	513	424	436	476	396	293	378	464	348	488	425	410
Tomahawk Creek (T051R06W5)	449	511	506	439	450	524	418	311	379	497	359	470	443	426
Wabash Creek (T059R27W4)	380	558	504	291	290	402	376	260	396	469	311	451	391	369
Average	**407**	**497**	**506**	**378**	**362**	**496**	**391**	**322**	**360**	**469**	**377**	**401**	**414**	**397**
Aspen Parkland														
Buffalo Creek (T047R08W4)	249	422	381	214	364	393	222	187	364	342	398	381	331	326
Haynes Creek (M6) (T039R25W4)	405	430	403	422	480	429	298	333	341	375	370	432	382	393
Ray Creek (T035R26W4)	405	392	388	411	505	385	266	340	376	390	416	363	380	386
Renwick Creek (T032R25W4)	364	357	333	328	457	312	237	308	345	345	428	355	348	347
Stretton Creek (T051R02W4)	224	365	345	240	343	408	217	183	411	383	516	382	355	335
Threehills Creek (T036R26W4)	396	402	389	419	509	402	278	338	365	386	404	386	383	389
Average	**340**	**395**	**373**	**339**	**443**	**388**	**253**	**281**	**367**	**370**	**422**	**383**	**363**	**363**
Grassland														
Meadow Creek (T011R28W4)	540	293	353	570	395	236	222	575	316	475	668	351	405	416
Prairie Blood Coulee (T006R24W4)	580	314	346	439	314	241	241	576	251	346	632	300	362	382
Trout Creek (T012R29W4)	581	316	375	587	400	258	232	579	313	471	661	372	411	429
Average	**567**	**308**	**358**	**532**	**370**	**245**	**232**	**577**	**293**	**430**	**654**	**341**	**393**	**409**
Continental Divide														
Willow Creek (T015R03W5)	705	357	421	665	459	342	351	637	418	569	748	377	**488**	**504**
Mixed and Moist Mixed Grassland (Irrigated)														
Battersea Drain (T011R20W4)	372	234	257	405	292	209	142	512	262	305	536	245	313	314
Crowfoot Creek (T024R22W4)	375	325	303	425	394	277	160	337	301	328	426	308	316	330
Drain S-6 (T010R12W4)	378	222	203	334	281	180	118	453	255	248	370	250	269	274
New West Coulee (T015R17W4)	300	188	219	336	328	194	111	451	278	270	465	260	294	283
Average	**356**	**242**	**245**	**375**	**323**	**215**	**133**	**438**	**274**	**288**	**449**	**266**	**298**	**300**

Precipitation was highly variable among the years of study, and the timing of floods and droughts varied spatially. During the CAESA period of monitoring, 1995 and 1998 were wetter years in the Grassland Ecoregions (Mixed, Moist Mixed, and Fescue), while during the AESA period of monitoring, 2002 and 2005 were wet in the Grassland Ecoregions and the Continental Divide Ecoregion. Precipitation totals were notably higher in 1996 and 1997 in the Boreal Ecoregion and again in 2000 and 2004. In the Aspen Parkland Ecoregion, 1999 and 2005 were the wettest years (Figure 2.3).

The driest years of the monitoring period were 2000, 2001, and 2002. In 2000 and 2001, drought conditions were concentrated in the southern part of the province (Figure 2.3). In 2002, the northern part of the province (including the Boreal ecoregion area) had very low amounts of precipitation, while the south experienced a major rain event in June (Figure 2.3).

Precipitation across the agricultural zone of Alberta from March 1 to October 31 of each year (1995 to 2006) is shown as a percentage of the 46-year average (1961 to 2006) in Appendix 2.

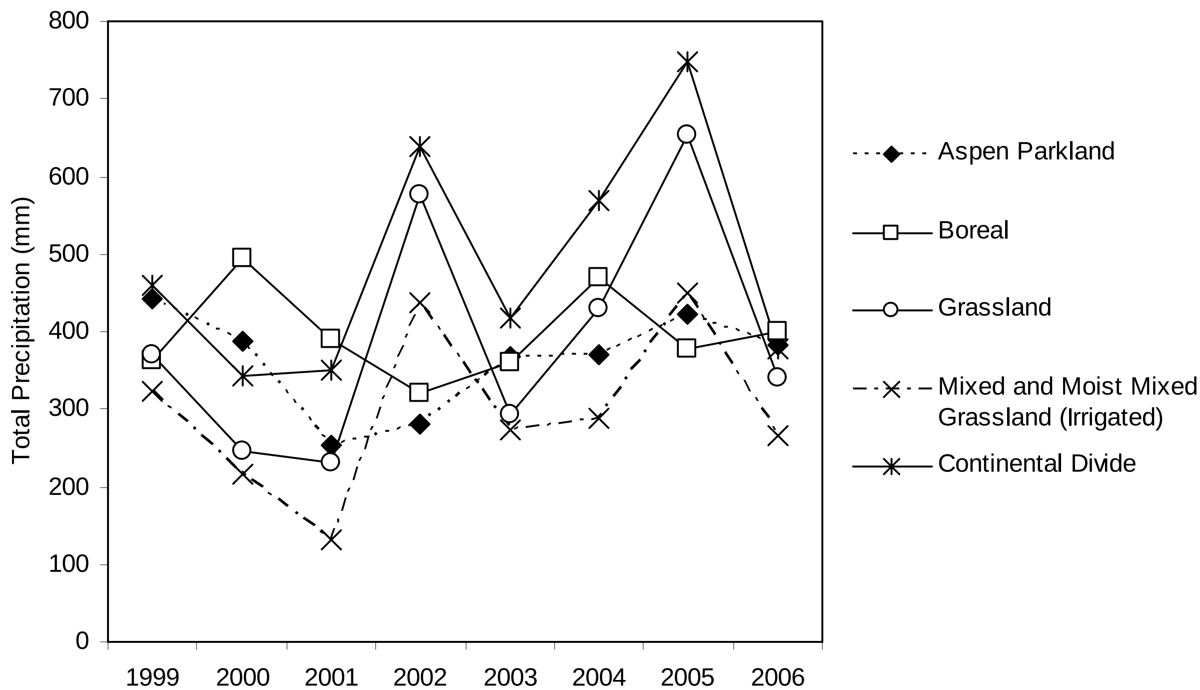

Figure 2.3. Total precipitation by ecoregion area from March 1 through October 31.

Hydrology

Historical mean annual runoff depths for the watersheds reflect the Ecoregions in which they are found (Figures 2.4 and 2.5). The lowest historical runoff depths for the AESA watersheds were found in those watersheds within the Aspen Parkland Ecoregion, which are also typically watersheds of a high agricultural intensity (Figure 2.4). Watersheds within the Boreal Transition zone and Western Alberta Uplands showed higher historical annual runoff, whereas intermediate runoff depths occurred in the Fescue Grasslands in the southwest and the Peace Lowlands Ecoregion in the north. Willow Creek, located in the Northern Continental Divide Ecoregion (i.e., sub-alpine), generated the highest runoff.

Watersheds containing irrigation activity deviate from ecoregion-based trends due to flow regulation. Crowfoot Creek, Battersea Drain, Drain S-6, and New West Coulee fall into the category of irrigation return flow streams. Two of the irrigated AESA watersheds (New West Coulee and Drain S-6) were in the driest part of the province, the Mixed Grassland Ecoregion. Since the streams receive irrigation return flows, their mean annual unit runoff is typically similar to that of watersheds that receive high precipitation.

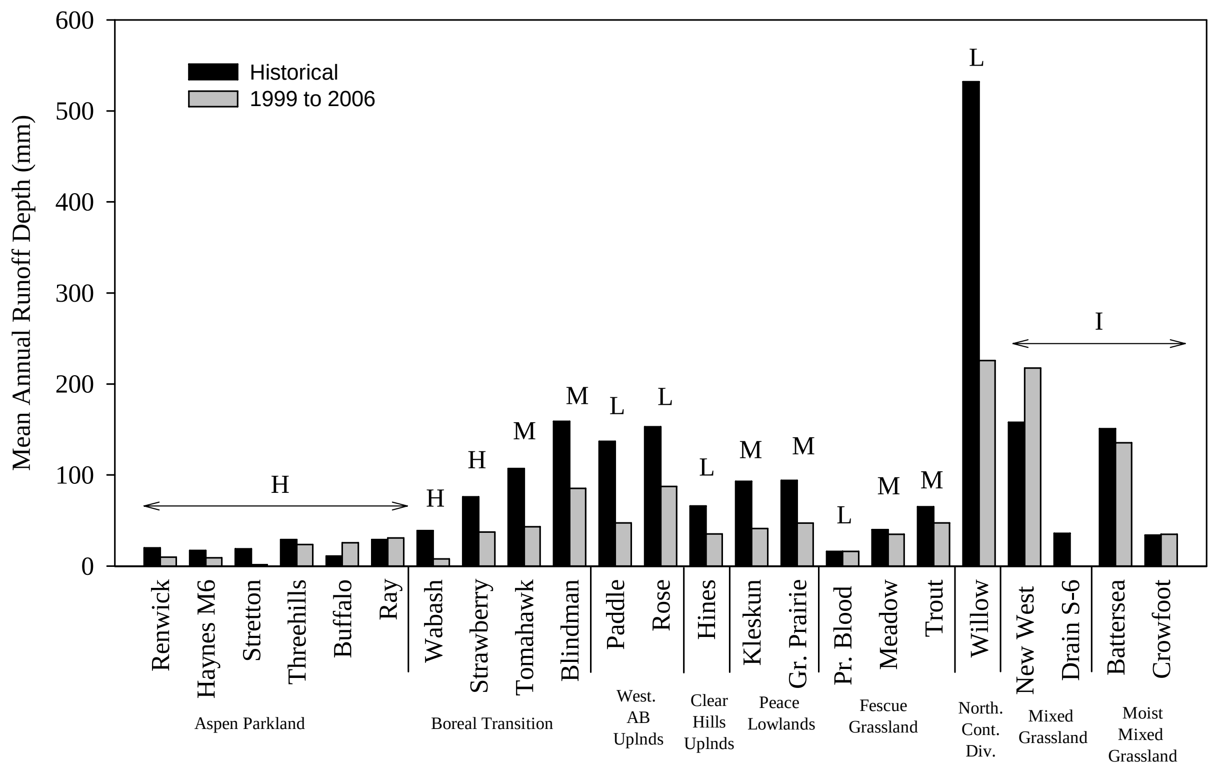

Figure 2.4. Annual mean unit runoff depth for the AESA watersheds. Bars are a comparison of the historical annual mean unit runoff depth (period of record varies from 10 to 98 years depending on the stream) to the AESA monitoring period of record (1999 to 2006). Watersheds are grouped by Ecoregion. Agricultural intensity (1996 categories) is represented by the letter above the bar with “H” representing high, “M” representing moderate, “L” representing low, and “I” representing irrigated watersheds.

The proportion of annual runoff that occurs in early spring (March and April) is notably higher in watersheds in the Aspen Parkland Ecoregion and some of the watersheds in the Boreal ecoregion area (e.g., Wabash Creek and Kleskun Drain) than in other areas (Figure 2.5). This suggests that the majority of runoff is generated from spring snowmelt. Spring snowmelt and summer rain events also influenced stream flow in other watersheds in the Boreal ecoregion area. In contrast, stream flow occurred later in the year (May to October) in the Continental Divide and Grassland (Fescue, Mixed and Moist Mixed) Ecoregions. In the Continental Divide, this is likely due to delayed melt in the mountain headwaters, while the Grasslands typically do not receive much snow and are prone to sublimation during the winter. Generally speaking, the Grassland and Continental Divide Ecoregions receive the largest amount of precipitation as rainfall in June (Environment Canada 2008; Figure 2.16, pg. 2-40).

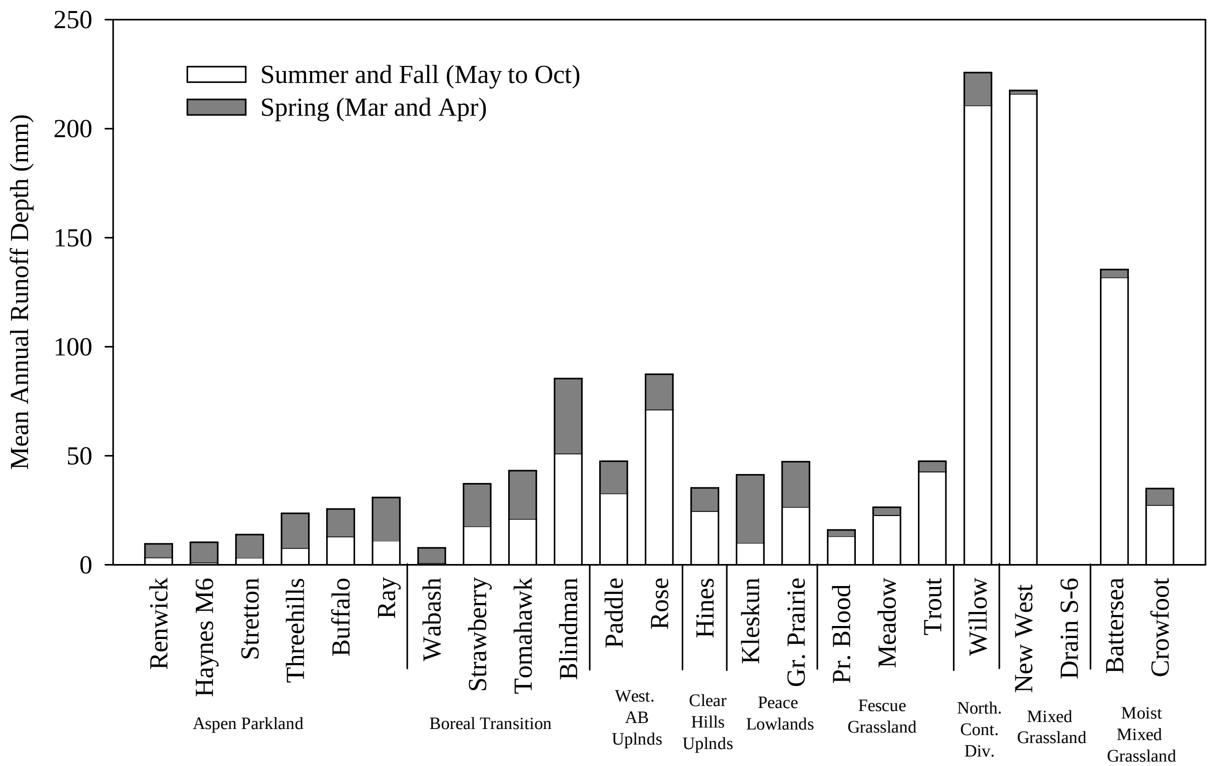

Figure 2.5. Spring (March through April) and Summer and Fall (May through October) mean annual runoff depths (1999 to 2006) for the AESA watersheds. Runoff depths for the irrigated watersheds (New West, Drain S6, Battersea, and Crowfoot) include return water. Bars are stacked.

The magnitude and timing of discharge has a substantial influence on water quality, particularly for pollutants that reach streams via overland pathways. Summary statistics for mean annual discharge (Q_{mean}, Q_{median}, Q_{max}; Tables 2.5, 2.6. 2.7, respectively) were used to characterize stream flow in each watershed during the period of record.

Table 2.5. Mean daily discharge ($m^3 s^{-1}$; Q_{mean}) by year (1995 to 2006) for the 23 AESA. Daily discharge was obtained from Water Survey of Canada for March 1 to October 31. Only years with accompanying water quality data are presented.

	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	95 to 06 Average	99 to 06 Average
Boreal														
Blindman River	0.546	2.979	2.126	1.031	3.026	2.699	0.991	0.559	1.442	0.547	1.565	0.559	1.506	1.423
Grande Prairie Creek					0.119	0.071	0.287	0.464	0.485	0.413	0.573	0.089	0.313	0.313
Hines Creek					0.007	1.001	0.805	0.501	1.067	0.524	1.055	0.024	0.623	0.623
Kleskun Drain					0.078	0.002	0.026	0.120	0.113	0.079	0.079	0.003	0.062	0.062
Paddle River	0.775	1.989	2.148		1.055	0.438	0.930	0.385	0.410	0.324	0.786	0.209	0.859	0.567
Rose Creek	1.931	3.656	3.192	2.455	4.035	3.163	1.324	1.409	1.768	1.563	3.826	1.372	2.475	2.308
Strawberry Creek	0.301	1.807	2.155	0.357	2.089	2.564	0.693	0.467	0.392	0.249	1.503	0.326	1.075	1.035
Tomahawk Creek	0.127	0.510	0.604	0.104	0.428	0.131	0.192	0.109	0.184	0.145	0.351	0.014	0.242	0.194
Wabash Creek					0.004	0.014	0.003	0.108	0.355	0.122	0.364	0.032	0.125	0.125
Average	**0.736**	**2.188**	**2.045**	**0.987**	**1.205**	**1.120**	**0.583**	**0.458**	**0.691**	**0.441**	**1.123**	**0.292**	**0.809**	**0.739**
Aspen Parkland														
Buffalo Creek	0.178	0.249	0.518		0.158	0.160	0.103	0.082	0.132	0.217	0.290	0.281	0.215	0.178
Haynes (M6) Creek	0.015	0.262	0.191	0.007	0.156	0.041	0.003	0.038	0.120	0.001	0.137	0.075	0.087	0.071
Ray Creek	0.018	0.072	0.114	0.017	0.095	0.034	0.008	0.012	0.089	0.057	0.115	0.108	0.062	0.065
Renwick Creek	0.028	0.072	0.172	0.010	0.043	0.004	0.002	0.001	0.051	0.032	0.044	0.034	0.041	0.026
Stretton Creek	0.041	0.018	0.088	0.009	0.025	0.019	0.000	0.000	0.001	0.056	0.081	0.110	0.037	0.037
Threehills Creek	0.031	0.269	0.394	0.025	0.287	0.059	0.018	0.040	0.289	0.059	0.225	0.253	0.162	0.154
Average	**0.052**	**0.157**	**0.246**	**0.014**	**0.127**	**0.053**	**0.022**	**0.029**	**0.114**	**0.070**	**0.149**	**0.143**	**0.101**	**0.088**
Grassland														
Meadow Creek	0.421	0.235	0.219		0.051	0.017	0.019	0.291	0.169	0.046	0.512	0.191	0.197	0.162
Prairie Blood Coulee	0.243	0.126			0.007	0.028	0.022	0.292	0.134	0.028	0.477	0.374	0.173	0.170
Trout Creek	2.415	1.071	1.043		0.300	0.083	0.146	1.437	0.905	0.380	3.495	1.160	1.130	0.988
Average	**1.027**	**0.477**	**0.631**		**0.120**	**0.042**	**0.062**	**0.673**	**0.403**	**0.151**	**1.495**	**0.575**	**0.500**	**0.440**
Continental Divide														
Willow Creek					0.367	0.192	0.380	1.138	0.463	0.597	1.958	0.472	**0.696**	**0.696**
Mixed and Moist Mixed Grassland (Irrigated)														
Battersea Drain				0.014	0.349	0.563	0.434	0.510	0.491	0.520	0.632	0.494	0.445	0.499
Crowfoot Creek	1.047	2.314	3.350	1.491	1.235	1.364	0.902	1.127	1.687	1.875	1.957	2.406	1.730	1.569
Drain S-6					1.888	1.819	1.926	1.159	0.006	0.004	0.008	0.012	0.853	0.853
New West Coulee					1.080	1.397	1.256	0.879	1.250	0.772	0.994	0.752	1.048	1.048
Average	**1.047**	**2.314**	**3.350**	**0.752**	**1.138**	**1.286**	**1.129**	**0.919**	**0.858**	**0.793**	**0.898**	**0.916**	**1.019**	**0.992**

Table 2.6. Median daily discharge ($m^3 s^{-1}$; Q_{median}) by year (1995 to 2006) for the 23 AESA watersheds. Daily discharge was obtained from Water Survey of Canada for March 1 to October 31. Only years with accompanying water quality data are presented.

	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	95 to 06 Average	99 to 06 Average
Boreal														
Blindman River	0.142	0.687	0.598	0.180	0.600	0.526	0.166	0.042	0.067	0.204	0.466	0.204	0.324	0.284
Grande Prairie Creek					0.000	0.002	0.049	0.000	0.000	0.008	0.008	0.000	0.008	0.008
Hines Creek					0.000	0.447	0.272	0.011	0.007	0.414	0.114	0.001	0.158	0.158
Kleskun Drain					0.000	0.000	0.001	0.000	0.000	0.000	0.001	0.000	0.000	0.000
Paddle River	0.420	0.740	0.744		0.208	0.330	0.098	0.030	0.044	0.297	0.311	0.129	0.305	0.181
Rose Creek	0.716	1.870	1.560	0.735	1.250	1.140	0.399	0.200	0.140	0.740	2.340	0.803	0.991	0.877
Strawberry Creek	0.153	0.340	0.384	0.195	0.426	0.240	0.097	0.011	0.008	0.082	0.132	0.076	0.179	0.134
Tomahawk Creek	0.066	0.156	0.136	0.035	0.038	0.031	0.014	0.006	0.011	0.060	0.076	0.005	0.053	0.030
Wabash Creek					0.000	0.007	0.001	0.000	0.007	0.011	0.021	0.000	0.006	0.006
Average	**0.299**	**0.759**	**0.684**	**0.286**	**0.280**	**0.303**	**0.122**	**0.033**	**0.032**	**0.202**	**0.385**	**0.135**	**0.225**	**0.187**
Aspen Parkland														
Buffalo Creek	0.057	0.145	0.177		0.086	0.123	0.085	0.049	0.099	0.155	0.146	0.155	0.116	0.112
Haynes (M6) Creek	0.002	0.000	0.000	0.000	0.000	0.015	0.001	0.000	0.000	0.000	0.001	0.003	0.002	0.003
Ray Creek	0.006	0.003	0.002	0.005	0.011	0.009	0.005	0.000	0.000	0.013	0.025	0.013	0.008	0.010
Renwick Creek	0.003	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.004	0.000	0.001	0.001
Stretton Creek	0.000	0.000	0.000	0.000	0.000	0.001	0.000	0.000	0.000	0.000	0.015	0.000	0.001	0.002
Threehills Creek	0.005	0.004	0.007	0.004	0.020	0.010	0.002	0.000	0.000	0.014	0.059	0.019	0.012	0.016
Average	**0.012**	**0.025**	**0.031**	**0.002**	**0.020**	**0.026**	**0.016**	**0.008**	**0.017**	**0.030**	**0.042**	**0.032**	**0.023**	**0.024**
Grassland														
Meadow Creek	0.182	0.150	0.080		0.040	0.000	0.000	0.107	0.063	0.032	0.209	0.133	0.091	0.073
Prairie Blood Coulee	0.100	0.004	0.002		0.001	0.001	0.002	0.016	0.005	0.012	0.027	0.099	0.024	0.020
Trout Creek	0.977	0.760	0.344		0.237	0.065	0.106	0.468	0.433	0.354	1.590	0.896	0.566	0.519
Average	**0.420**	**0.305**	**0.142**		**0.093**	**0.022**	**0.036**	**0.197**	**0.167**	**0.133**	**0.609**	**0.376**	**0.227**	**0.204**
Continental Divide														
Willow Creek					0.267	0.142	0.125	0.274	0.192	0.418	0.728	0.259	**0.301**	**0.301**
Mixed and Moist Mixed Grassland (Irrigated)														
Battersea Drain				0.015	0.398	0.652	0.477	0.583	0.520	0.646	0.663	0.619	0.508	0.570
Crowfoot Creek	1.080	1.350	1.580	1.480	1.160	1.580	0.990	0.858	1.290	1.570	1.410	1.360	1.309	1.277
Drain S-6					1.605	2.040	2.058	1.141	0.004	0.004	0.003	0.007	0.858	0.858
New West Coulee					0.968	1.490	1.540	0.701	1.250	0.804	1.070	0.777	1.075	1.075
Average	**1.080**	**1.350**	**1.580**	**0.748**	**1.033**	**1.440**	**1.266**	**0.821**	**0.766**	**0.756**	**0.786**	**0.691**	**0.937**	**0.945**

Table 2.7. Maximum daily discharge ($m^3 s^{-1}$; Q_{max}) by year (1995 to 2006) for the 23 AESA watersheds. Daily discharge was obtained from Water Survey of Canada for March 1 to October 31. Only years with accompanying water quality data are presented.

	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	95 to 06 Average	99 to 06 Average
Boreal														
Blindman River	7.4	66.5	30.4	28.3	58.9	107.0	70.9	15.3	31.4	9.5	23.0	10.1	38.2	40.8
Grande Prairie Creek					2.4	4.0	4.0	7.3	9.3	14.1	6.2	3.5	6.4	6.4
Hines Creek					0.1	14.1	7.7	6.7	21.9	2.4	10.3	0.3	7.9	7.9
Kleskun Drain					3.2	0.0	3.2	5.8	5.2	4.0	2.0	0.1	2.9	2.9
Paddle River	14.9	25.3	43.7		19.0	5.9	32.3	6.0	4.4	1.2	11.0	1.2	15.0	10.1
Rose Creek	38.2	19.0	22.5	70.5	100.0	50.9	53.6	22.9	32.4	34.1	37.5	13.3	41.2	43.1
Strawberry Creek	5.7	35.1	33.4	5.0	41.2	95.3	51.6	20.3	9.5	5.7	41.2	15.1	29.9	35.0
Tomahawk Creek	1.2	10.8	12.8	1.3	5.8	1.7	13.4	2.2	5.0	2.6	5.5	0.2	5.2	4.5
Wabash Creek					0.1	0.1	0.1	3.4	14.0	6.6	8.4	0.7	4.2	4.2
Average	**13.5**	**31.3**	**28.6**	**26.3**	**25.6**	**31.0**	**26.3**	**10.0**	**14.8**	**8.9**	**16.1**	**4.9**	**16.8**	**17.2**
Aspen Parkland														
Buffalo Creek	2.5	2.3	5.1		0.8	1.2	0.3	0.8	0.6	1.3	3.1	4.4	2.0	1.5
Haynes (M6) Creek	0.3	18.5	4.1	0.07	4.8	1.4	0.015	1.6	7.1	0.005	3.5	3.9	3.8	2.8
Ray Creek	0.2	3.3	4.3	0.2	2.1	0.5	0.1	0.6	4.0	0.7	1.8	5.9	2.0	2.0
Renwick Creek	1.4	3.5	4.3	0.2	2.1	0.1	0.0	0.1	1.7	1.3	1.5	1.2	1.5	1.0
Stretton Creek	1.6	1.2	4.8	0.5	0.8	0.5	0.0	0.0	0.1	3.5	1.7	2.9	1.5	1.2
Threehills Creek	0.5	14.6	17.2	0.3	9.7	1.1	0.2	1.0	14.3	0.9	4.2	13.1	6.4	5.6
Average	**1.1**	**7.2**	**6.6**	**0.2**	**3.4**	**0.8**	**0.1**	**0.7**	**4.6**	**1.3**	**2.6**	**5.2**	**2.9**	**2.3**
Grassland														
Meadow Creek	6.8	2.8	5.2		0.6	0.1	0.5	8.1	6.9	0.5	7.3	1.6	3.7	3.2
Prairie Blood Coulee	13.7	7.0	10.4		0.1	0.3	0.3	23.0	13.0	0.5	27.9	22.6	10.8	11.0
Trout Creek	30.5	6.3	20.2		1.7	0.4	0.9	24.9	23.0	1.4	62.1	7.8	16.3	15.3
Average	**17.0**	**5.4**	**11.9**		**0.8**	**0.3**	**0.6**	**18.7**	**14.3**	**0.8**	**32.4**	**10.7**	**10.3**	**9.8**
Continental Divide														
Willow Creek					0.3	0.1	0.1	0.3	0.2	0.4	0.7	0.3	**0.3**	**0.3**
Mixed and Moist Mixed Grassland (Irrigated)														
Battersea Drain				0.0	1.2	1.7	1.2	2.3	1.4	1.5	2.7	1.1	1.5	1.6
Crowfoot Creek	4.4	45.7	40.6	11.1	6.6	3.5	2.9	3.6	22.6	12.6	12.5	27.6	16.1	11.5
Drain S-6					5.7	4.2	4.1	3.1	0.0	0.0	0.0	0.1	2.2	2.2
New West Coulee					4.9	4.1	3.8	3.4	4.1	2.5	8.4	2.2	4.2	4.2
Average	**4.4**	**45.7**	**40.6**	**5.6**	**4.6**	**3.4**	**3.0**	**3.1**	**7.0**	**4.2**	**5.9**	**7.8**	**6.0**	**4.9**

In the following sub-sections, each stream is examined for inter-annual variation in the timing and magnitude of flow and to verify that the number of samples collected each year and timing of collection relative to the hydrograph was representative. Sampling frequency is summarized quantitatively by year (Table 2.8) and graphically by month (Figures 2.6 to 2.21). The sub-sections are arranged by Ecoregion for ease of discussion.

Aspen Parkland Ecoregion. The Aspen Parkland Ecoregion includes the Buffalo Creek, Stretton Creek, Haynes Creek, Ray Creek, Renwick Creek, and Threehills Creek watersheds.

Buffalo Creek - Buffalo Creek watershed represents a relatively large area of agricultural land (71,400 ha) in the Eastern edge of the Aspen Parkland Ecoregion. Despite its size, only a small proportion (21%) of the gross drainage area comprises the effective drainage area (14,700 ha). Management recommendations for this basin would apply primarily to the portion of the watershed with hydrologic connectivity.

Buffalo Creek was monitored from 1995 to 1997 and 1999 to 2006 (Table 2.2, pg. 2-6). On average, 14 water quality samples were collected per year. The highest number of samples collected was 24 in 2005, and the lowest was 2 in 1996 (Table 2.8). It is unknown why only two samples were collected from the stream in 1996 although there may have been insufficient manpower to implement the study design. There was not a large amount of inter-annual variation in discharge in this stream, as reflected by the similar number of samples collected in most years (typically 11 to 17 samples).

Historically, flows in Buffalo Creek peak in April and receded thereafter (historical data set: 1972 to 2006). High annual flow volumes were typically attributable to high flows during spring melt in April, and the majority of samples were collected in this month (Figure 2.7). However, there were three years in the monitoring record when flows persisted a little longer and exceeded the historical monthly average in June and July as well (1996, 1997, and 2004) (Figure 2.6). Peak spring flows (i.e., discharge > 3.0 $m^3 s^{-1}$) were observed in 1997, 2005, and 2006. The two years with lowest annual and median discharge were 2001 and 2002 (Table 2.6). The eastern part of Alberta experienced drought conditions during this time period.

When assessing water quality in Buffalo Creek over the period of record, it is important to note the data gaps in 1996 (only two samples) and 1998 (zero samples) (Table 2.8). For the continuous monitoring period from 1999 onwards, it is noteworthy that there were two consecutive years of ‘higher flows’ near the end of monitoring record (2005 and 2006) when spring freshets returned and drought conditions were alleviated.

Stretton Creek - The Stretton Creek watershed drains 7400 ha of land in the northeast corner of the white (agricultural) zone. Water quality monitoring commenced in 1995 and continued until 2000. The stream ceased to flow from 2001 to 2003 as a result of severe drought conditions and sampling was halted. Monitoring resumed from 2004 until 2006 (Table 2.2, pg. 2-6).

On average, eight water quality samples were collected annually in the nine years with flow. The fewest samples (two samples) were collected in 1996 and the most (12 samples) in 1999 (Table 2.8). No water quality testing was done in August, September or October (Figure 2.7).

There is very limited opportunity for collecting samples in this stream due to the nature of flows. Historically (1979 to 2006), Stretton Creek flowed mainly in March (Q_{mean} = 0.09 m^3 s^{-1}) and April (Q_{mean} = 0.21 m^3 s^{-1}).

In 1997 and 2006, mean monthly discharge in April was above the mean historical values. In 2006, flows remained above average in May and June. Like Buffalo Creek, there were three consecutive years of above average flow at the end of the monitoring record. A storm event in August 2005 (August 24th, 94 mm) generated a flow peak in Stretton Creek. This was anomalous and not captured by AESA sampling. In 1997, 2004, and 2006, instantaneous peaks in stream discharge > 3 m^3 s^{-1} were observed with snow melt (Figure 2.6).

Haynes Creek - Haynes Creek watershed is a small tributary (16,600 ha) of the Red Deer River and is located east of the Town of Lacombe.

Haynes Creek is one of the most intensively studied of the AESA watersheds. As part of the CAESA study (1995 to 1996), 18 sites were chosen to assess the impact of specific agriculture practices (e.g., cattle wintering, field cultivation) on water quality (Anderson et al. 1998a). From 1998 to 2001, a follow-up study was conducted to assess the effectiveness of Beneficial Management Practices at improving water quality (Wuite and Chanasyk 2003; Wuite et al. 2007). Water quality samples collected at the mouth of the watershed (M6 – 16,600 ha) and at a sub-basin level (M1 - 2400 ha) are included in the AESA dataset (1995 to 2006); however, the data for the M1 sub-watershed are not reported here.

Sampling frequency from 1995 to 2000 was much more intensive (21 to 35 samples/year) than from 2001 to 2006 (eight to 13 samples/year) as a result of different study goals (Table 2.8). The M6 site was not sampled in 2004 due to low flows.

The majority of flow in Haynes Creek occurred in April (Figure 2.7). The highest spring peaks were observed in 1996 (Q_{max}= 18.5 m^3 s^{-1}) and 2003 (Q_{max}= 7.1 m^3 s^{-1}) (Table 2.7, Figure 2.6). In contrast, there were four years (1995, 1998, 2001, 2004) with no spring peak and very low mean annual flows (Q_{mean}< 0.02 m^3 s^{-1}) (Table 2.5). During the time when the detailed studies were being conducted, intensive sampling continued despite low flow volumes. In 1995 and 1998, 35 and 23 samples were collected, respectively (Table 2.8). Nine and zero samples were collected in 2001 and 2004, respectively. Another example of a discrepancy in sampling frequency was between April 2000 and 2002 when hydrographs were similar but sampling frequencies were not (i.e., 21 vs. nine samples, respectively). The difference in sampling frequency needs to be taken into consideration when interpreting long-term trends.

Ray Creek - Ray Creek drains a small area (4400 ha) of agricultural land in the Red Deer River watershed. Water quality was monitored from 1995 to 2006 (Table 2.2, pg. 2-6). On average, 19 water quality samples were collected per year (Table 2.8). The highest number of samples collected was 32 in 1999, and the lowest was seven in 2002.

Like most streams in the Aspen Parkland Ecoregion, flows in Ray Creek were highest in April with spring snowmelt and receded thereafter (Figure 2.9). Historical (1967 to 2006) mean discharge in April was 0.22 m^3 s^{-1}, declining to <0.05 m^3 s^{-1} from May through October. During the monitoring period, the highest magnitude spring freshets were observed in 1996, 1997, 2003, and 2006 (Figure 2.8). In contrast, freshets were virtually absent in 1995, 1998, and 2000 to 2002.

A summer storm in mid-July 1999 (69 mm) generated sufficient runoff to produce a peak in the hydrograph and sustain flows in the summer, resulting in higher sampling numbers in 1999.

Renwick Creek - Renwick Creek is a small (5800 ha) tributary to the Red Deer River. This system was monitored from 1995 to 2006 (Table 2.2, pg. 2-6). On average, 15 samples were collected per year, with as many as 27 samples collected in 1999 and as few as five collected in 2001 (Table 2.8).

Historically (1967 to 2006), flows were high in March (Q_{mean} = 0.104 m^3 s^{-1}), then peaked in April (Q_{mean} = 0.124 m^3 s^{-1}), and receded thereafter. In 1996 and 1997, spring flows (March and April) exceeded historical values. The peaks did not appear to be related to spring rain but may instead be the result of a large snowpack. Four consecutive years, 2003 through 2006, had above average flow in March and below average flow in April, indicating these years had earlier peak flows in the spring (Figure 2.8). In previous years (2000 to 2002), low flows were observed throughout the entire open water season, from March through October. In contrast, years with above average flow in the summer months include 1995 (June and July), 1998 (July), 2003 (May), and 2005 (May, June, August, September, and October). Noteworthy are July flows in 1999 that were 18-fold higher than average flows due to a 60-mm summer storm event.

Threehills Creek - Threehills Creek drains 13,800 ha and is part of the Red Deer River watershed. Historically, Threehills Creek had peak flows in April followed by receding flows through October (Figure 2.9).

Threehills Creek was monitored from 1995 to 2006 (Table 2.2, pg. 2-6). In many of the years of study, flows were very low. Years with low flow include 1995, 1998, a three year drought period from 2000 through 2002, and 2004 (Table 2.6, Figure 2.8). In contrast, high flows were observed in 1996, 1997, 2003, and 2006.

On average, 18 samples were collected each year (Table 2.8). Like Ray and Renwick Creeks, the highest number of samples (33) was collected in 1999 in response to a summer storm event, and the lowest number of samples (8) was collected during a drought year (2002).

Table 2.8. Annual number of nutrient samples collected (March 1 through October 31) for each AESA watershed. Watersheds are presented by ecoregion area. Bacteria numbers were similar. Pesticide samples were collected less frequently (Appendix 17).

	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	95 to 06 Average	99 to 06 Average
Boreal														
Blindman River	18	19	23	31	34	29	17	15	14	19	32	19	23	22
Grande Prairie Creek					12	20	13	9	11	14	15	10	13	18
Hines Creek					8	17	12	14	12	22	18	12	14	14
Kleskun Drain					8	14	7	10	8	11	11	5	9	12
Paddle River	21	6	26		24	26	15	14	13	12	33	19	19	14
Rose Creek	18	19	23	31	31	29	17	15	15	19	32	24	23	21
Strawberry Creek	17	18	25	18	26	30	11	11	11	17	16	10	18	20
Tomahawk Creek	20	18	26	28	23	26	14	11	15	17	31	17	21	18
Wabash Creek					10	18	5	8	8	13	23	15	13	16
Average	**19**	**16**	**25**	**27**	**20**	**23**	**12**	**12**	**12**	**16**	**23**	**15**	**17**	**17**
Aspen Parkland														
Buffalo Creek	12	2	19		14	11	9	12	16	16	24	17	14	15
Haynes (M6) Creek	35	25	22	23	22	21	9	8	13		12	10	18	14
Ray Creek	18	21	19	23	32	26	12	7	14	15	20	15	19	18
Renwick Creek	15	15	18	18	27	12	5	6	11	13	19	15	15	14
Stretton Creek	10	2	10	8	12	11	0	0	0	6	10	10	7	6
Threehills Creek	15	20	18	23	33	24	12	8	14	16	20	18	18	18
Average	**18**	**14**	**18**	**19**	**23**	**18**	**8**	**7**	**11**	**13**	**18**	**14**	**15**	**14**
Grassland														
Meadow Creek	4	4	21		19	13	11	21	21	22	22	23	16	19
Prairie Blood Coulee	7	3			13	9	9	17	8	19	17	18	12	14
Trout Creek	4	4	21		23	19	17	21	21	23	21	23	18	21
Average	**5**	**4**	**21**		**18**	**14**	**12**	**20**	**17**	**21**	**20**	**21**	**15**	**18**
Continental Divide														
Willow Creek					26	23	21	21	16	20	21	19	**21**	**21**
Mixed and Moist Mixed Grassland (Irrigated)														
Battersea Drain				18	15	18	16	16	20	25	21	21	19	19
Crowfoot Creek	12	14	18	18	15	19	17	16	23	21	22	22	18	19
Drain S-6					15	16	15	14	16	22	15	21	17	17
New West Coulee					14	14	14	16	20	25	20	21	18	18
Average	**12**	**14**	**18**	**18**	**15**	**17**	**16**	**16**	**20**	**23**	**20**	**21**	**18**	**18**

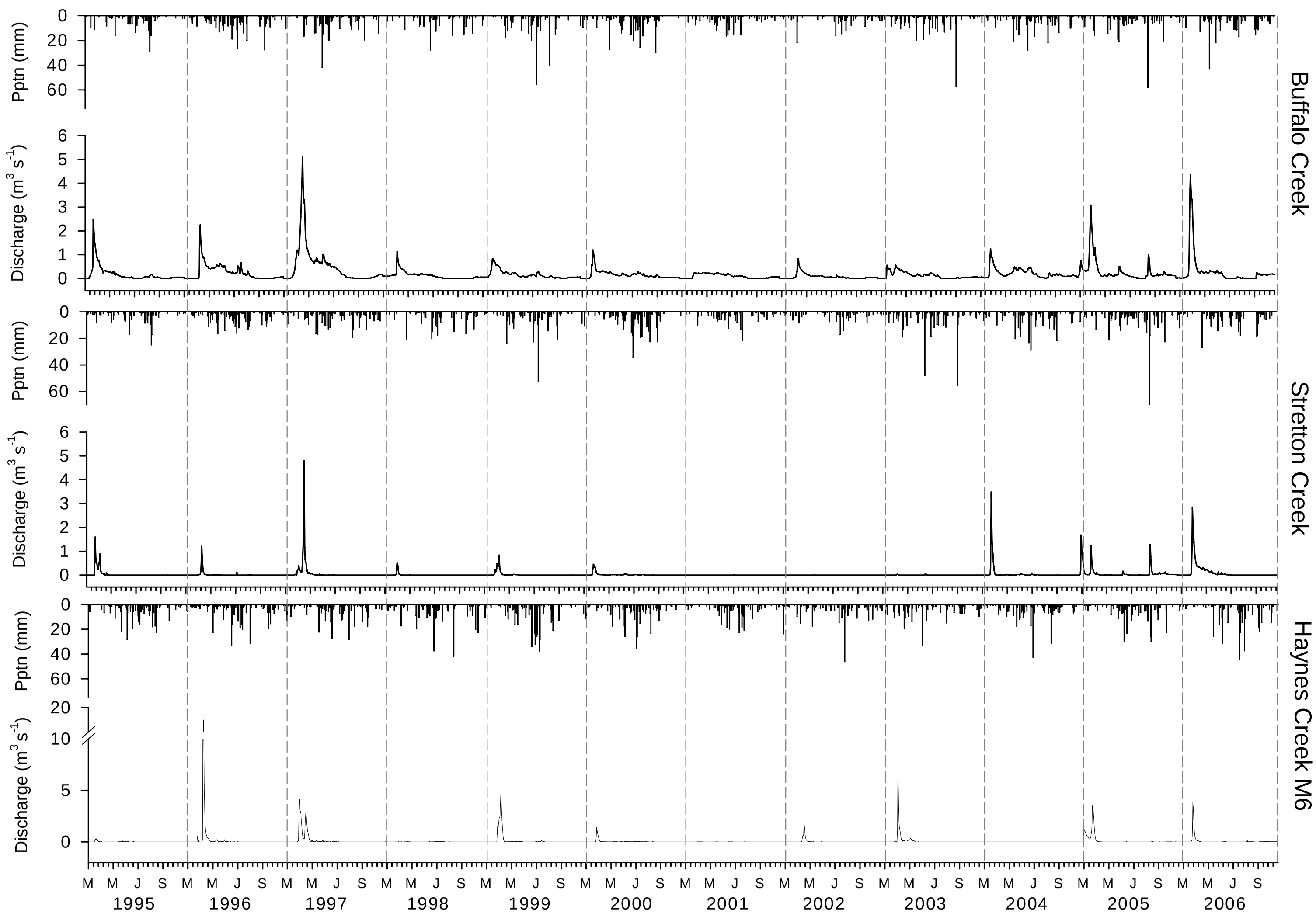

Figure 2.6. Daily discharge (lower y-axis) and precipitation (Pptn) (upper y-axis) for Buffalo Creek, Stretton Creek, and Haynes Creek M6 from 1995 through 2006. Discharge and precipitation data shown were recorded from March 1 to October 31 for each year. Note the different y-axis scales for discharge.

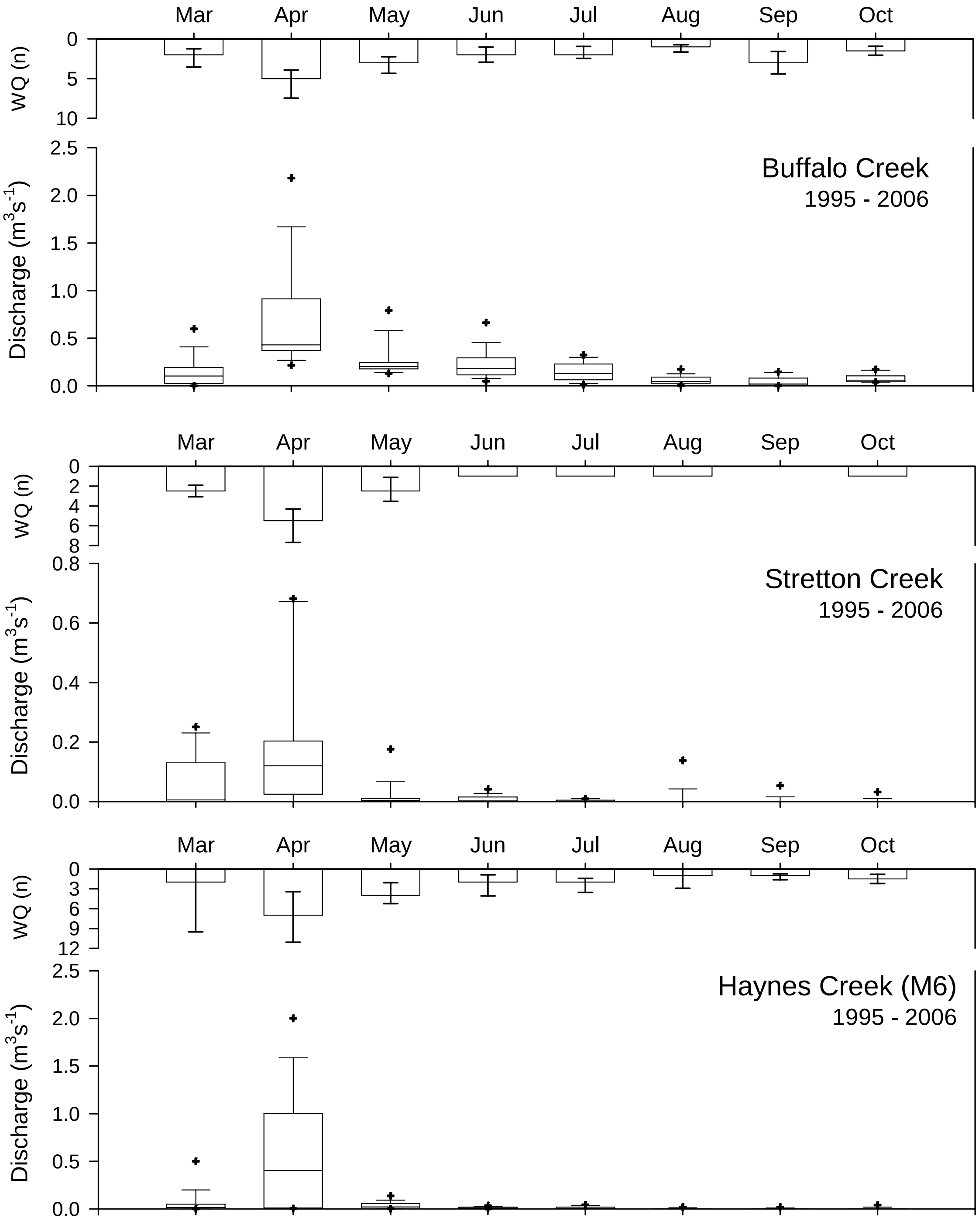

Figure 2.7. Monthly discharge (lower y-axis) and number (n) of water quality samples (WQ) (upper y-axis) for Buffalo Creek (1995 to 2006), Stretton Creek (1995 to 2006), and Haynes Creek (1995 to 2006) from March through October. Discharge box plots stretch from the 25th percentile to the 75th percentile with the horizontal line in the middle of the box representing the median. Vertical lines represent 1.5 times the interquartile range while crosshairs denote the maximum and minimum data point. Note: sample numbers represent the number of nutrient and bacteria samples, not the number of pesticide samples.

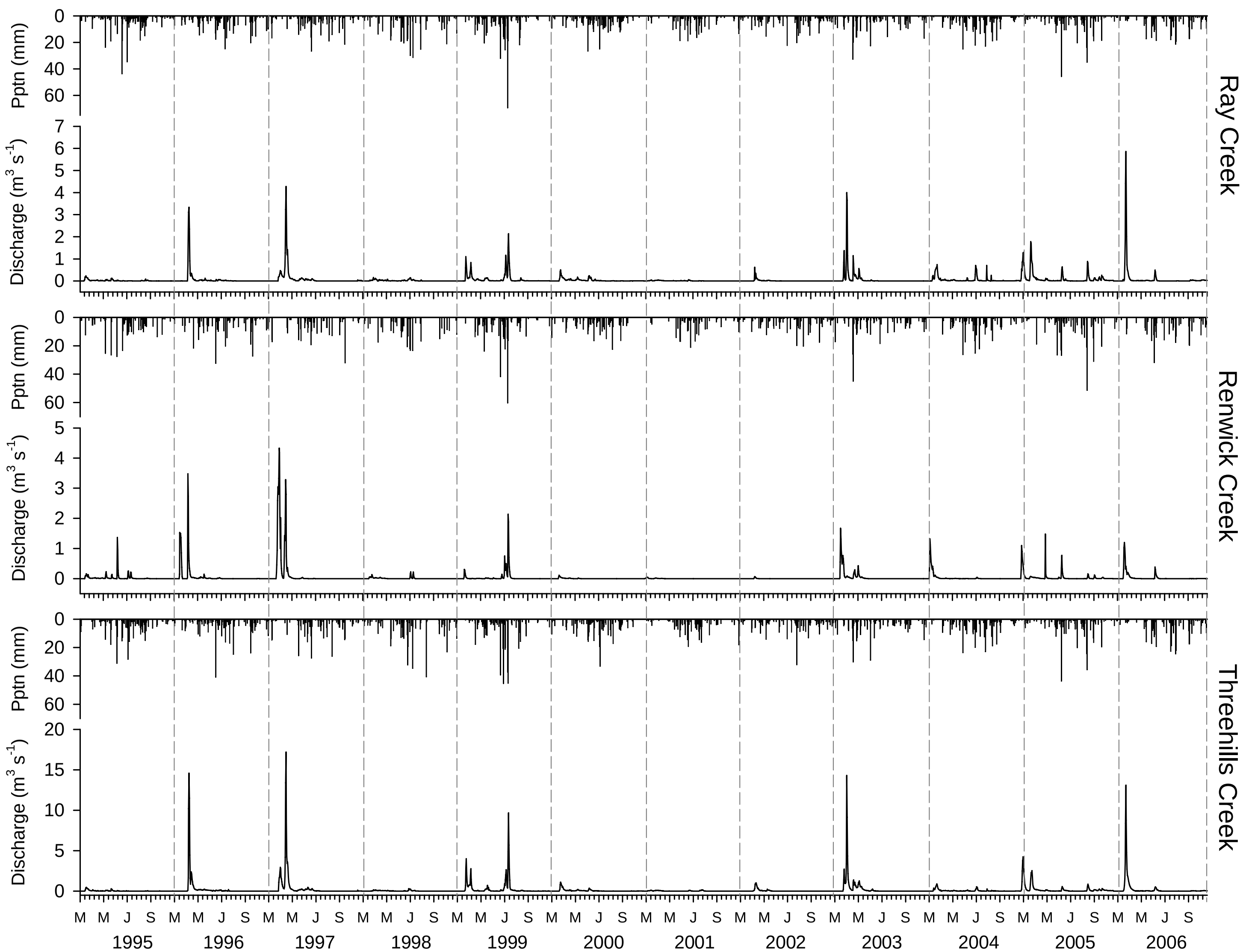

Figure 2.8. Daily discharge (lower y-axis) and precipitation (Pptn) (upper y-axis) for Ray Creek, Renwick Creek, and Threehills Creek from 1995 through 2006. Discharge and precipitation data shown were recorded from March 1 to October 31 for each year. Note the different y-axis scales.

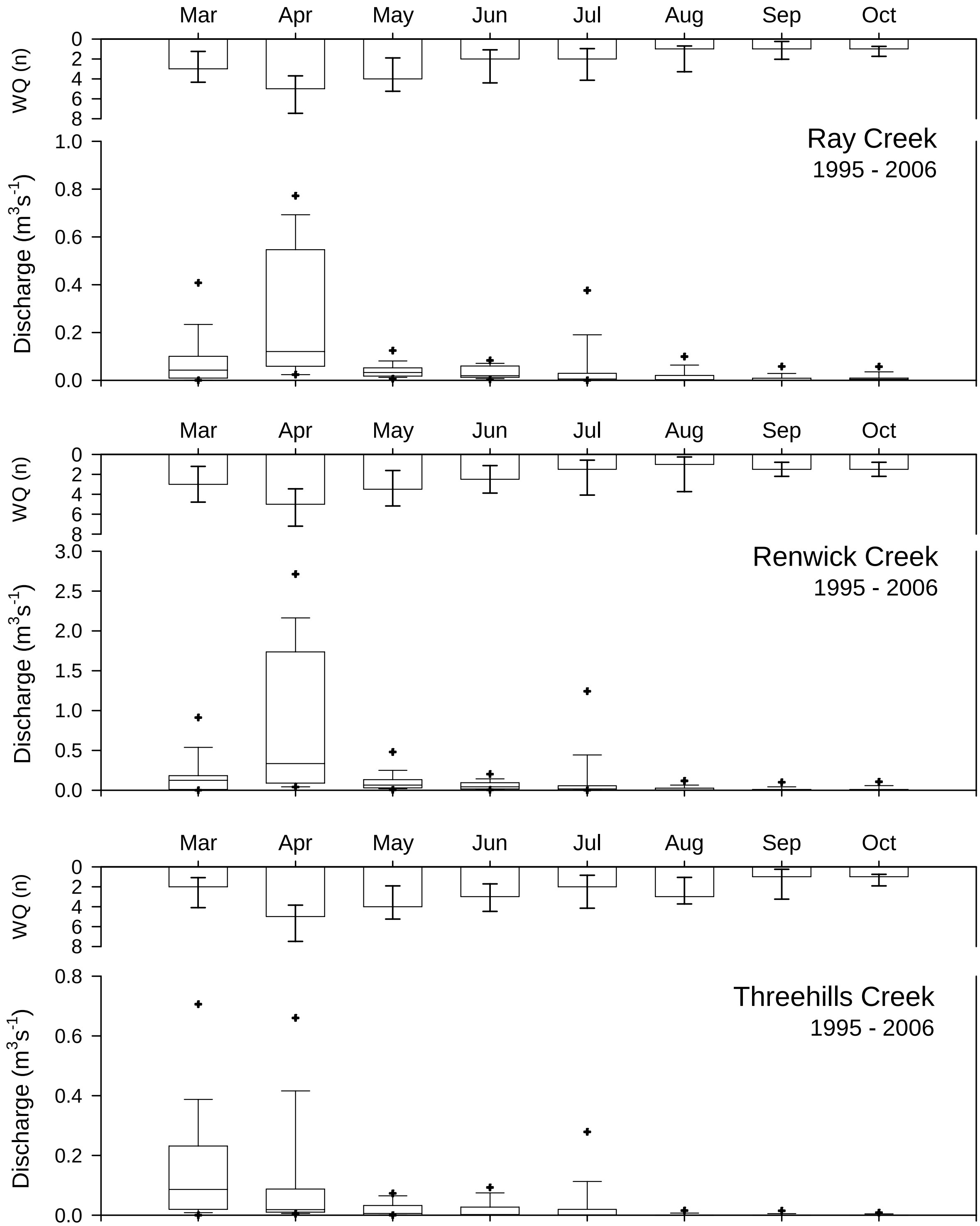

Figure 2.9. Monthly discharge (lower y-axis) and number (n) of water quality samples (WQ) (upper y-axis) for Ray Creek (1995 to 2006), Renwick Creek (1995 to 2006), and Threehills Creek (1995 to 2006) from March through October.

Boreal Forest Ecoregion (Boreal Transition and Western Alberta Upland). The Boreal Forest Ecoregion includes the Blindman River, Rose Creek, Strawberry Creek, Paddle River, Tomahawk Creek, and Wabash Creek watersheds.

Blindman River - The Blindman River watershed is 35,300 ha in size and is a tributary of the Red Deer River. Highest annual flows (Q_{median} > 0.5 m^3 s^{-1}; mean annual flow >2.0 m^3 s^{-1}) occurred in 1996, 1997, 1999, and 2000. The highest peak flow was observed in July 2000 (Q_{max} = 107 m^3 s^{-1}) (Tables 2.6 and 2.7, Figure 2.10).

The Blindman River flows throughout the open water season and was normally sampled continually from March through October (Figure 2.11). On average, 22 samples were collected every year from 1995 to 2006 (Table 2.8). The fewest samples (14 to 15 samples per year) were collected in the two years with lowest median annual flow (Q_{median} < 0.068 m^3 s^{-1} in 2002 and 2003), with >30 samples collected in two years of high flow (1999 and 2000).

Historically, discharge in the Blindman River was highest in April and July (4.35 and 3.06 m^3 s^{-1}, respectively (historical monthly averages, 1965 to 2006)) coinciding with spring melt and summer storm events. During the monitoring period, there were five years when average flows during spring melt in April exceeded historical averages (1996, 1997, 1999, 2003, and 2005) and four years when average flows during summer storms in July exceeded historical averages (1998, 1999, 2000, and 2001). There were also four years where flows were below historical monthly means in all months: 1995, 2002, 2004 and 2006.

Rose Creek - Rose Creek is a tributary of the North Saskatchewan River, and the watershed is 55,900 ha in size. The highest monthly flow volumes generally occurred in May and July, though flows were usually sustained from April through July (Figure 2.11).

Rose Creek was monitored from 1995 to 2006 (Table 2.2, pg. 2-6). On average, 23 samples were collected per year (Table 2.8). The fewest samples (15) were collected in 2002 and 2003 when flows receded completely after the May peak (Q_{median} = 0.2 and 0.14 m^3 s^{-1}, respectively) and did not increase again in July. The most samples (32) were collected in 2005 when median annual flows were approximately 10-fold higher (2.34 m^3 s^{-1}) and the creek flowed from March through October.

In 1999, daily discharge in Rose Creek peaked at 100 m^3 s^{-1}on July 9 following a precipitation event totaling 69 mm on July 7 (Figure 2.10). High magnitude flood peaks (>50 m^3 s^{-1}) in Rose Creek were clustered in the four-year span from 1998 to 2001 and were absent from 2001 onwards. Spring freshets (March to May) were virtually absent in five of the fifteen years of study (1995, 1998, 2001, 2004, and 2006), presumably due to low snowpack or a gradual spring melt.

Strawberry Creek - Similar to Rose Creek, Strawberry Creek is a tributary of the North Saskatchewan River, and the watershed is 59,200 ha in size. Historically, stream flow peaks in April and July, with higher flows occurring during spring melt than during summer storm events.

Strawberry Creek was monitored from 1995 to 2006. On average, 21 samples were collected per year (Table 2.8). The fewest samples (10) were collected in 2006 and the most (30) in 2000 when monthly flows were above historical values in May, June, and July. The highest flood peak during the monitoring period was in July 2000 (Q_{max} = 95 m^3 s^{-1}), and the second highest was 52 m^3 s^{-1} in July 2001 (Table 2.7, Figure 2.10). In several other years, there was very little flow in Strawberry Creek. For example, spring and summer peaks were virtually absent in 1995, 1998, 2003, 2004, and 2006. For the evaluation of water quality with time, it is important to note (similar to Rose Creek) the absence of summer flows in the latter part of the monitoring record.

Paddle River - The Paddle River watershed (25,300 ha) is situated in the Western Alberta Upland (Boreal) Ecoregion and is one of two AESA watersheds that contribute to the Athabasca River system. The hydrograph typically followed the same pattern as Rose Creek, also located in the Western Alberta Upland Ecoregion (Figure 2.12).

Paddle River was monitored from 1995 to 1997 and 1999 to 2006 (Table 2.8). On average, 19 water quality samples were collected per year (Table 2.8). The highest number of samples collected was 33 in 2005, and the lowest was six in 1996. The years with highest flows occurred earlier in the sampling record (1996 and 1997) and were not characterized equally well with respect to water quality (e.g., 6 vs. 26 samples, respectively). Like Buffalo Creek, it was unclear why the system was under-sampled in 1996.

As noted for other streams in the Boreal ecoregion area, annual discharge near or below historical values were concentrated in the later part of the sampling period (2000, 2002, 2004, and 2006). The exception was 2001, when Q_{max} reached 32 m^3 s^{-1} in July 2001 after a 54-mm storm event.

Tomahawk Creek - Tomahawk Creek drains 9500 ha of agricultural land to the North Saskatchewan River watershed. Historically (1984 to 2006), mean monthly discharge was highest in April (Q_{mean}= 0.7 m^3 s^{-1}) and July (Q_{mean}= 0.53 m^3 s^{-1}), a pattern similar to Strawberry Creek.

Tomahawk Creek was monitored from 1995 to 2006 (Table 2.2, pg. 2-6). In the wet years of 1996 and 1997, higher flows were observed in April and June; however, these are the only two years in the monitoring record when spring and summer peaks in flow were observed (Figure 2.12). In 2001, a summer peak was observed after the late July rain event (63 mm) that triggered a hydrologic response in Tomahawk Creek (Q_{max} = 13.4 m^3 s^{-1}) and other AESA streams in the region (Paddle River, Rose Creek, Strawberry Creek, and Blindman River) (Table 2.7).

From 1999 to 2006, the highest mean annual flows were observed in 1999 and 2005, and flows were concentrated in April. On average, 21 water samples were collected per year (Table 2.8).

Wabash Creek - Wabash Creek is a 34,400 ha watershed in the Boreal Transition Ecoregion and is a tributary to the Athabasca River system. Water quality monitoring in Wabash Creek took place from 1999 to 2006 (Table 2.2, pg. 2-6).

Historically (1979 to 2006), discharge from this creek peaked in April and receded thereafter (April Q_{mean} = 1.61 m^3 s^{-1}) (data not shown). This pattern is slightly different from other watersheds in the southern Boreal Ecoregion where spring and summer flow peaks occurred. Wabash Creek watershed received less annual precipitation than other Boreal watersheds (Table 2.4).

During the years 1999 to 2001, flows were extremely low all year (Table 2.5). Stream flow was also low in 2006. It wasn't until 2002 that the first spring peak in four years occurred in Wabash (monthly April Q_{mean}= 0.73 $m^3 s^{-1}$) (Figure 2.12), but mean monthly flows still fell below historical values. In 2004, spring flows were similar to 2002. Only two of the eight years had spring flows that exceeded historical monthly averages (April 2003 and March 2005).

The average number of samples collected per year was 13 (Table 2.8). The fewest (5) and the most (23) water samples were collected in years of lowest (2001) and highest (2005) mean flow years, respectively. Water chemistry in 2000 was slightly overrepresented relative to other years (18 samples), particularly when compared to 2003 (only eight samples). The majority of samples were collected in April (Figure 2.13).

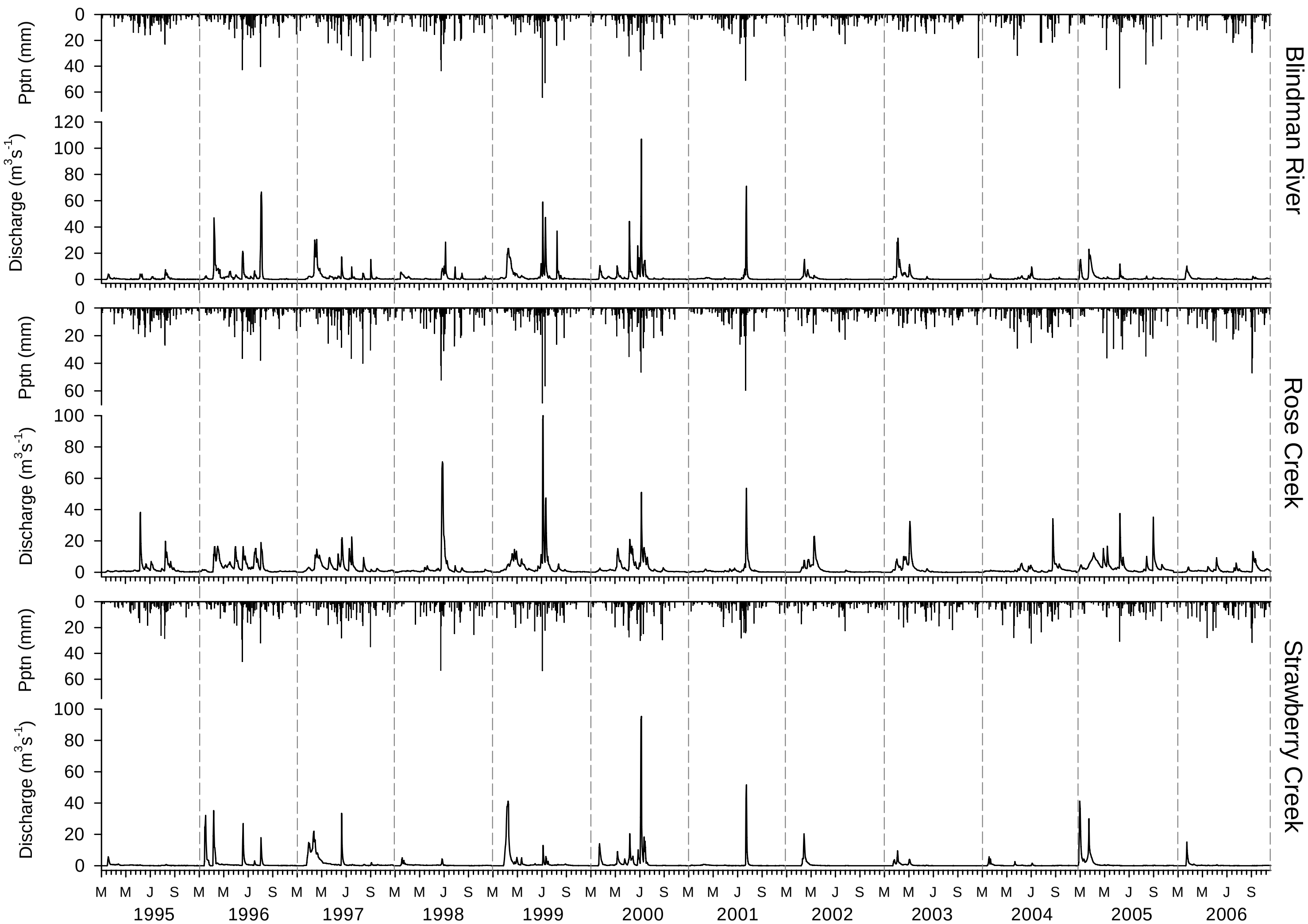

Figure 2.10. Daily discharge (lower y-axis) and precipitation (Pptn) (upper y-axis) for Blindman River, Rose Creek, and Strawberry Creek from 1995 through 2006. Discharge and precipitation data shown were recorded from March 1 to October 31 for each year. Note the different y-axis scales.

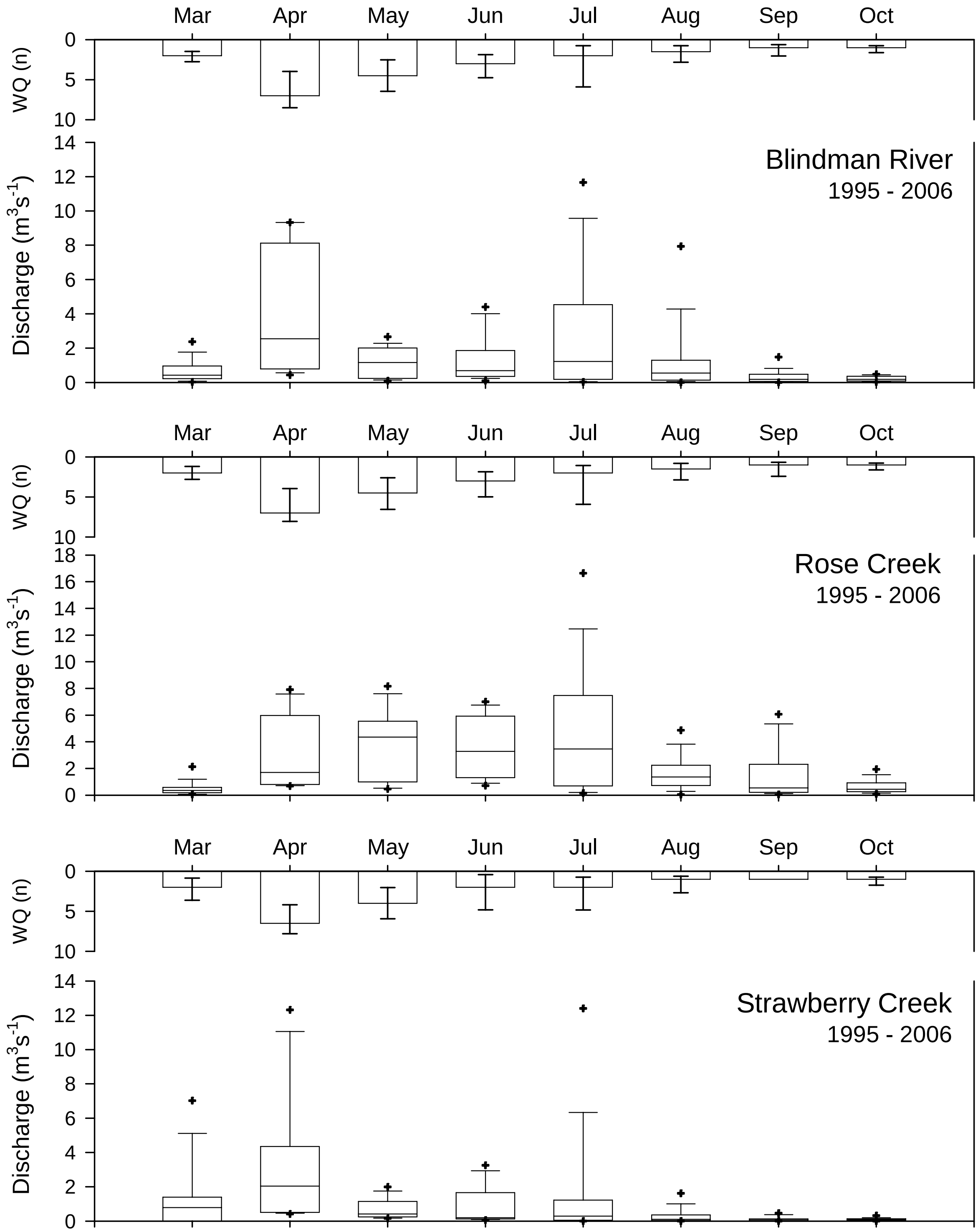

Figure 2.11. Discharge (lower y-axis) and number (n) of water quality samples (WQ) (upper y-axis) for Blindman River (1995 to 2006), Rose Creek (1995 – 2006) and Strawberry Creek (1995 to 2006) from March to October.

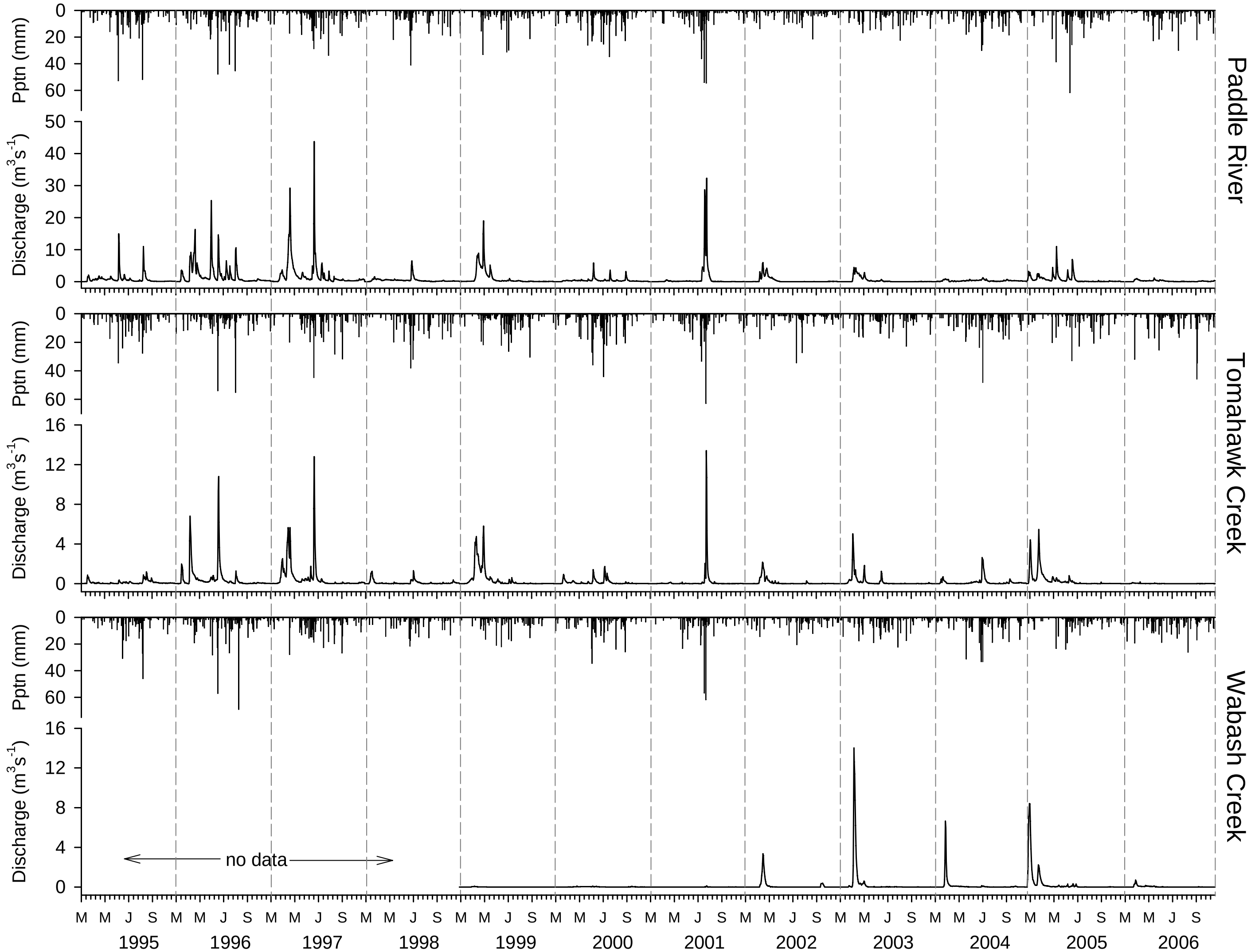

Figure 2.12. Daily discharge (lower y-axis) and precipitation (Pptn) (upper y-axis) for Paddle River, Tomahawk Creek, and Wabash Creek from 1995 to 2006. Discharge and precipitation data shown were recorded from March 1 to October 31 for each year. Note the different y-axis scales.

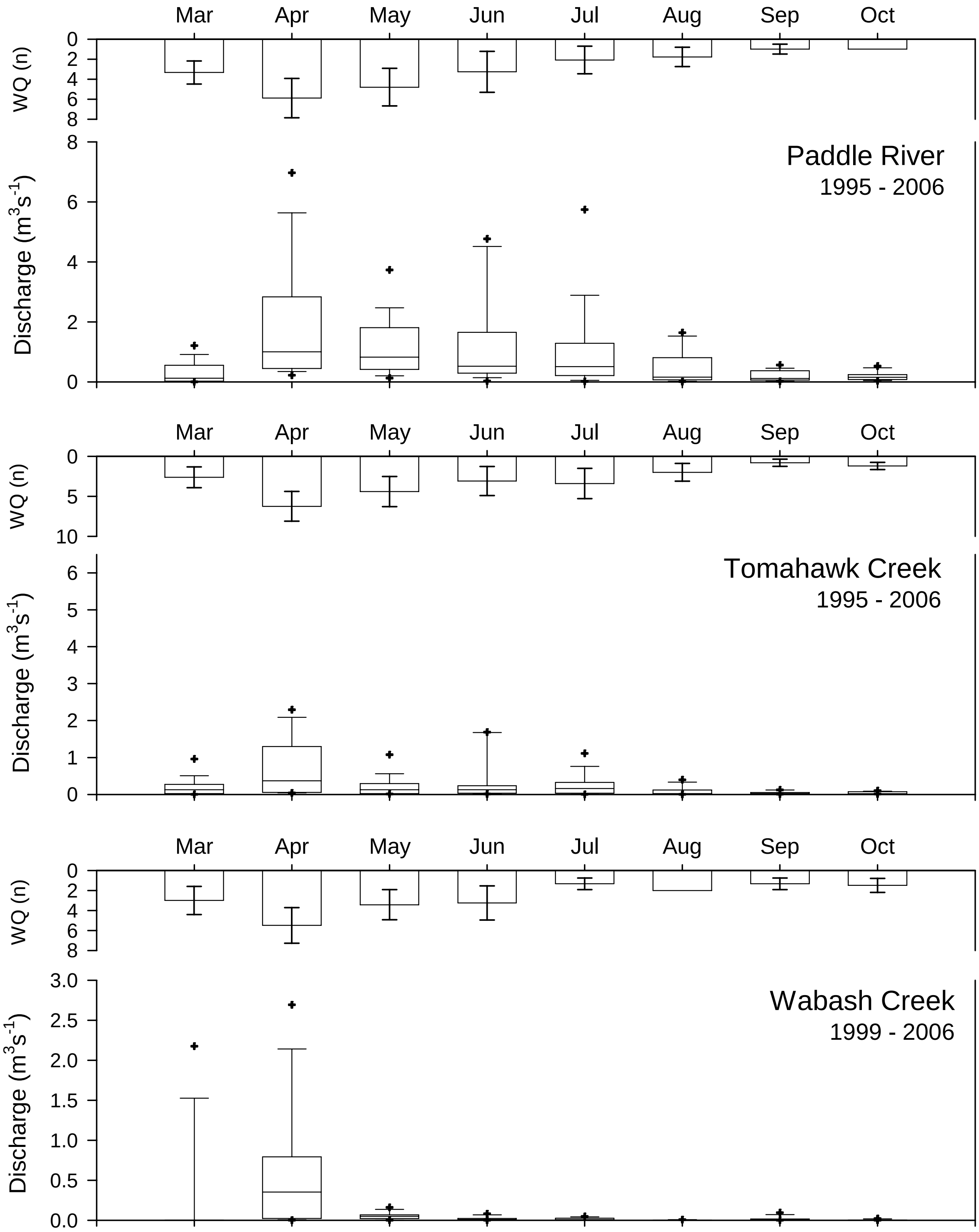

Figure 2.13. Mean monthly discharge (lower y-axis) and number (n) of water quality samples (WQ) (upper y-axis) for Paddle River (1995 to 2006 discharge; 1999 to 2006 sample number), Tomahawk Creek (1995 to 2006), and Wabash Creek (1999 to 2006) from March to October.

Peace Lowland and Clear Hills Upland Ecoregions. The watersheds in these Ecoregions include Kleskun Drain, Grande Prairie Creek, and Hines Creek.

Kleskun Drain **-** Kleskun Drain is a 3200 ha watershed in the northwestern corner of Alberta's agricultural zone. Water quality monitoring in Kleskun Drain took place from 1999 to 2006 (Table 2.2, pg. 2-6).

Historically (1966 to 2006), peak flows in Kleskun Drain occurred in April with spring snowmelt. The historical April Q_{mean} is 0.453 $m^3 s^{-1}$. During the AESA Stream Survey, there were three years (1999, 2002, and 2003) when April Q_{mean} was above the historical average and one year (2005) when the mean monthly flow in March was above the historical average. In 2000 and 2006, there were no peaks in discharge, and further, the lowest flow conditions of the monitoring period were recorded in 2006 (Figure 2.14).

On average, nine water quality samples were taken from Kleskun Drain each year (Table 2.8), and the majority of samples were collected in April (Figure 2.15). The maximum (14) and minimum (5) number of samples collected occurred in the two low flow years, 2000 and 2006, respectively. The inconsistency between discharge and number of samples indicates a problem with the flow-biased sampling regime in this stream. Under-representation in high flow years may have influenced flow-weighted mean concentrations and mass loading results.

Grande Prairie Creek **-** Grande Prairie Creek drains 15,200 ha of agricultural land east of the City of Grande Prairie. Grande Prairie Creek is a tributary to the Peace and Slave River watershed.

Historical records from 1970 to 2006 showed that peak monthly mean flows occurred in April (April Q_{mean}=1.55 $m^3 s^{-1}$). Similar to Kleskun Drain, the peak discharge in April was in response to snowmelt. From 1999 to 2006, the highest median annual flows occurred in 2001 (Q_{median}= 0.049 $m^3 s^{-1}$) when flows were sustained through June despite lower flow volumes in spring (Table 2.6). Highest mean annual flows (Q_{mean}> 0.4 $m^3 s^{-1}$) occurred in 2002, 2003, 2004, and 2005 when flows exceeded historical values in one or more months (Table 2.5).

The timing of peak discharge was variable among years (Figure 2.14). Peaks were recorded in the spring of 2002 (May), 2003 (April and May), and 2005 (April), and in the summer of 2004 (July and September). Low flow years included 1999, 2000, and 2006 (Q_{mean}< 0.15 $m^3 s^{-1}$).

On average, 13 water quality samples were taken each year (Table 2.8). The fewest number of samples (9) was taken in 2002, and the highest number of samples (20) was taken in 2000. Although 2000 had the highest number of samples, the flow was modest in all months except September, indicating a discrepancy in the flow-biased sampling.

Hines Creek **-** Hines Creek is the most northern watershed studied in the AESA Stream Survey. In fact, of the 37,400 ha drainage area, approximately half of the watershed lies outside of Alberta's agricultural zone and is mainly forested. Water quality monitoring in Hines Creek took place from 1999 to 2006 (Table 2.2, pg. 2-6).

Historically, between 1975 and 2006, the highest monthly discharge occurred in May (Q_{mean}= 2.14 $m^3 s^{-1}$). Further, flows were typically sustained from April through July indicating a reliance on snowmelt and precipitation. There were two years (2003 and 2005) with large peaks during spring flow in April (Figure 2.15) and three years where the highest peak flows occurred later in the open water season, including 2000 (July and September), 2001 (June), and 2002 (May). In the remaining years of monitoring, spring flows were below historical values. Specifically, both 1999 and 2006 were years of very low flow (annual Q_{mean}< 0.03 $m^3 s^{-1}$) (Table 2.5).

The average number of water quality samples taken was 14 (Table 2.8). The minimum number of samples (eight) was taken in 1999, and the maximum number of samples (22) was taken in 2004. In 2004, the highest flows of the year were atypical, occurring later in the season from August through October, thus extending the sampling season.

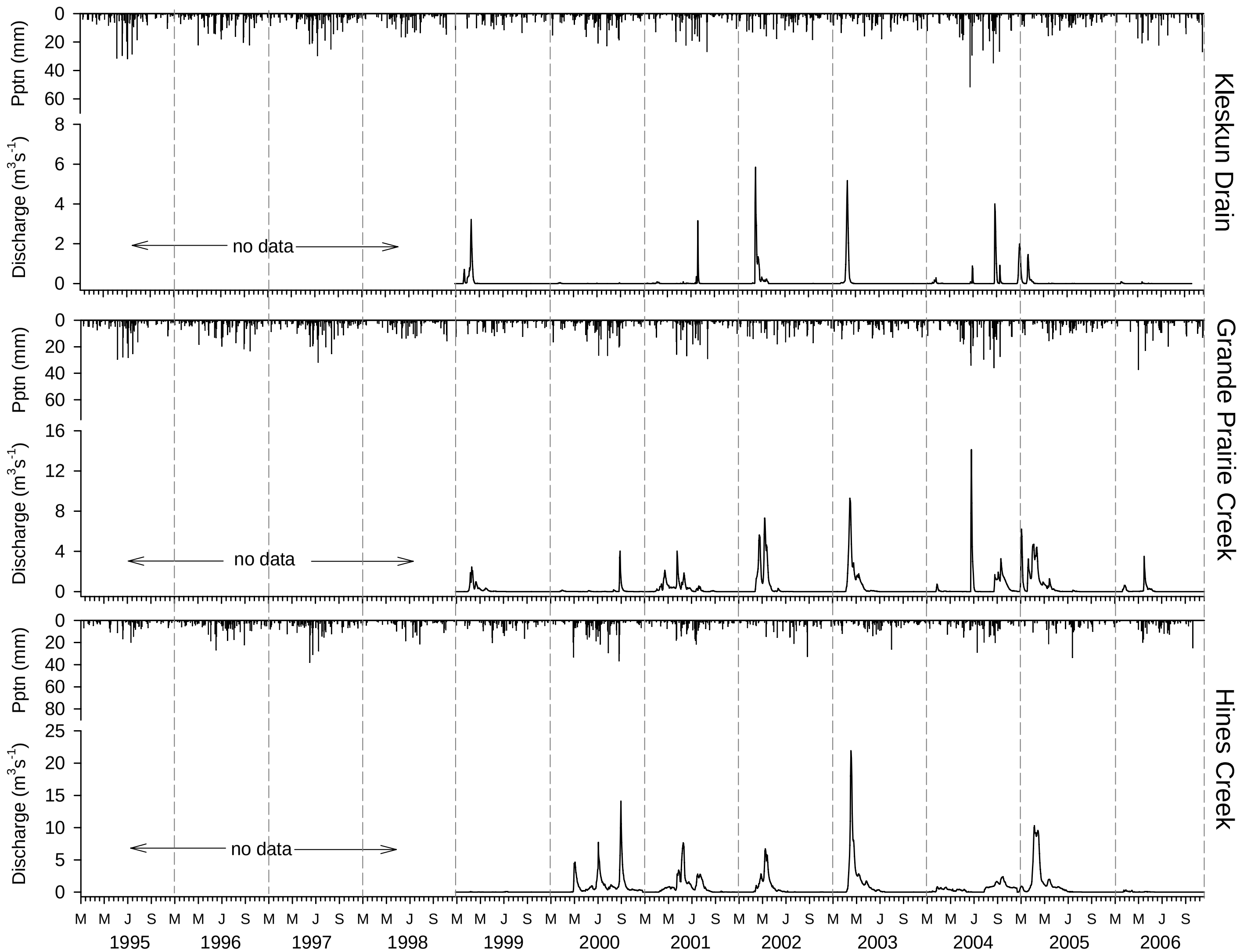

Figure 2.14. Daily discharge (lower y-axis) and precipitation (Pptn) (upper y-axis) for Kleskun Drain, Grande Prairie Creek, and Hines Creek from 1995 through 2006. Discharge and precipitation data shown were recorded from March 1 to October 31 for each year. Note the different y-axis scales.

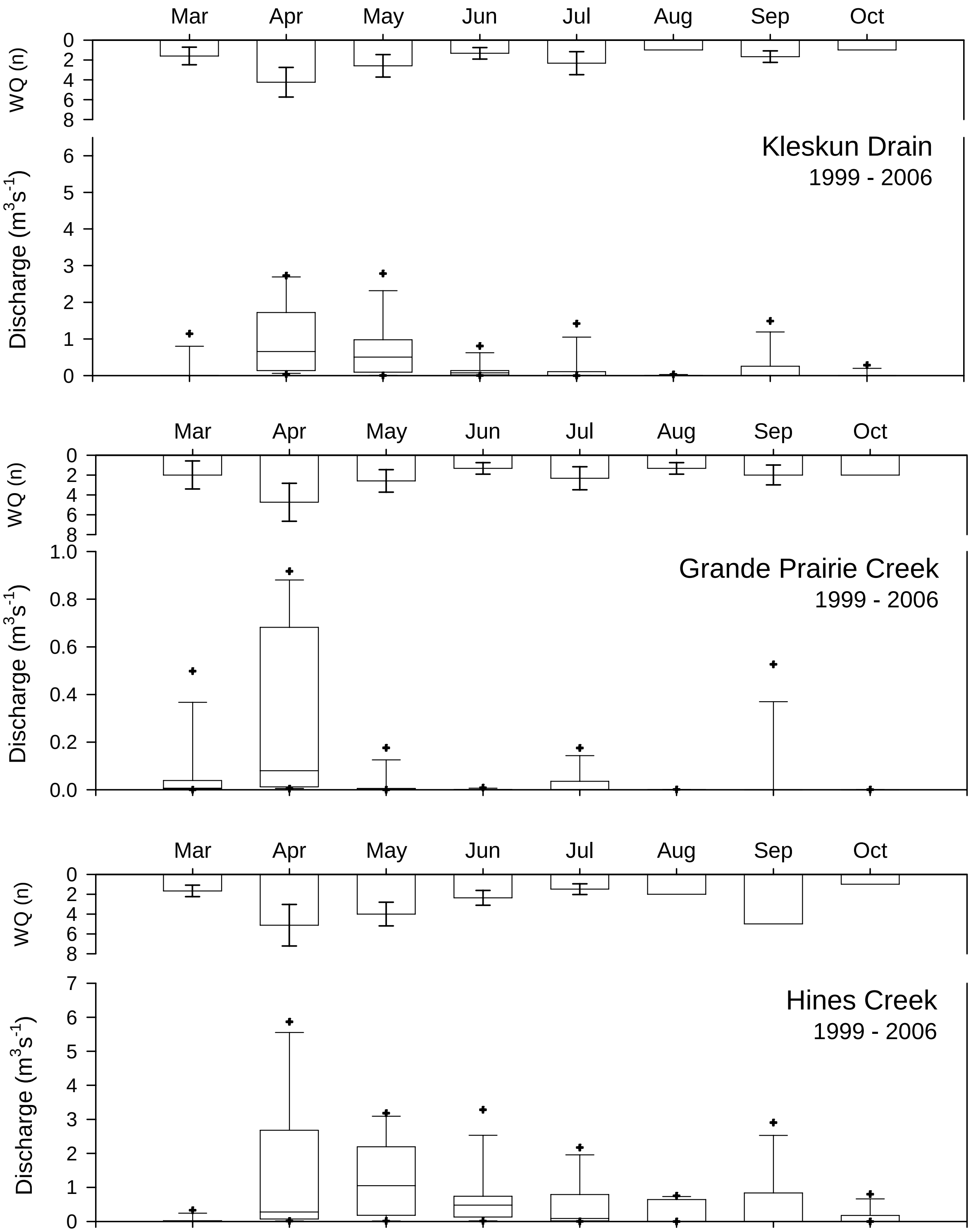

Figure 2.15. Monthly discharge (lower y-axis) and number (n) of water quality samples (WQ) (upper y-axis) for Kleskun Drain (1999 to 2006), Grande Prairie Creek (1999 to 2006), and Hines Creek (1999 to 2006) from March through October.

Fescue Grassland Ecoregion. The Fescue Grassland Ecoregion includes the Meadow Creek, Trout Creek, and Prairie Blood Coulee watersheds.

Meadow Creek - Meadow Creek is a 13,000 ha tributary to the Oldman River watershed. Historically, June was the month with highest flows (June Q_{mean} = 0.38 $m^3 s^{-1}$; 1966 to 2006) in this watershed.

Meadow Creek was monitored from 1995 to 2006 (excluding 1998) (Table 2.2, pg. 2-6). In general, years with high stream flow coincided with high precipitation. For example, in 1995, 1998, 2002, and 2005, annual Q_{mean} ranged from 0.42 to 0.58 $m^3 s^{-1}$ (Table 2.5). March to October precipitation totals ranged from 540 to 668 mm (Table 2.4). In contrast, annual Q_{mean} was approx. 10-fold less (e.g., Q_{mean} <0.06 $m^3 s^{-1}$) in the low flow years (1999, 2000, 2001 and 2004), and the annual precipitation total ranged from 220 to 480 mm. Flow patterns in 2006 followed a near typical hydrograph for this stream (annual Q_{mean}=0.191 $m^3 s^{-1}$) (Figure 2.16).

Certain high flow years (1995 and 2005) sustained slightly higher than average flows later in the summer (July, August, and September), while others (2002) experienced faster recession with a peak in June only (Figure 2.16). In other years (1996, 1997, 2003), discharge peaks were higher in the spring and early summer (March, May and June).

Generally, we would expect similar chemistry in years with similar flow conditions. However, 1995 and 1996 were under-sampled in Meadow and Trout Creeks, with only four samples collected. On average, 16 samples were collected each year from 1995 to 2006, increasing to an average of 19 samples per year from 1999 to 2006 (Table 2.8). For trends with time, data were analyzed from 1999 onwards due to unrepresentative data sets for 1995 and 1996 and lack of data for 1998. It is also noteworthy that there was less flow-bias in the sampling frequency of Meadow, Trout, and Prairie Blood Coulee compared to other streams. Logistics dictated sample frequency early in the spring and later in the summer and fall.

Trout Creek - Trout Creek is a 44,100 ha tributary to the Oldman River watershed.
The Trout and Meadow Creek hydrographs paralleled one another (Figure 2.16), but discharge from Trout Creek was approximately 10-fold higher (Table 2.5). Typically, June was the month with highest mean monthly flow (historical June Q_{mean} = 3.01m^3s^{-1}; 1911 to 2006).

Refer to Meadow Creek, above, for discussion of sampling frequencies.

Prairie Blood Coulee - Prairie Blood Coulee is a 22,600 ha tributary of the Oldman River watershed. Historically, monthly mean discharge was highest in this stream in March and June. However, flow peaks were typically observed in either spring or summer.

Water samples were collected from 1995 to 1996 and again from 1999 to 2006. There were three years with high spring flow (March): 1996, 1997 (not sampled), and 2003. In contrast, there were four years with summer peaks: 1995, 2002, 2005, and 2006 (Figure 2.16).

On average, 12 water samples were collected per year (Table 2.8). In 1996, only three samples were collected. In 2004, 19 samples were collected; the stream was over-sampled relative to discharge in 2004.

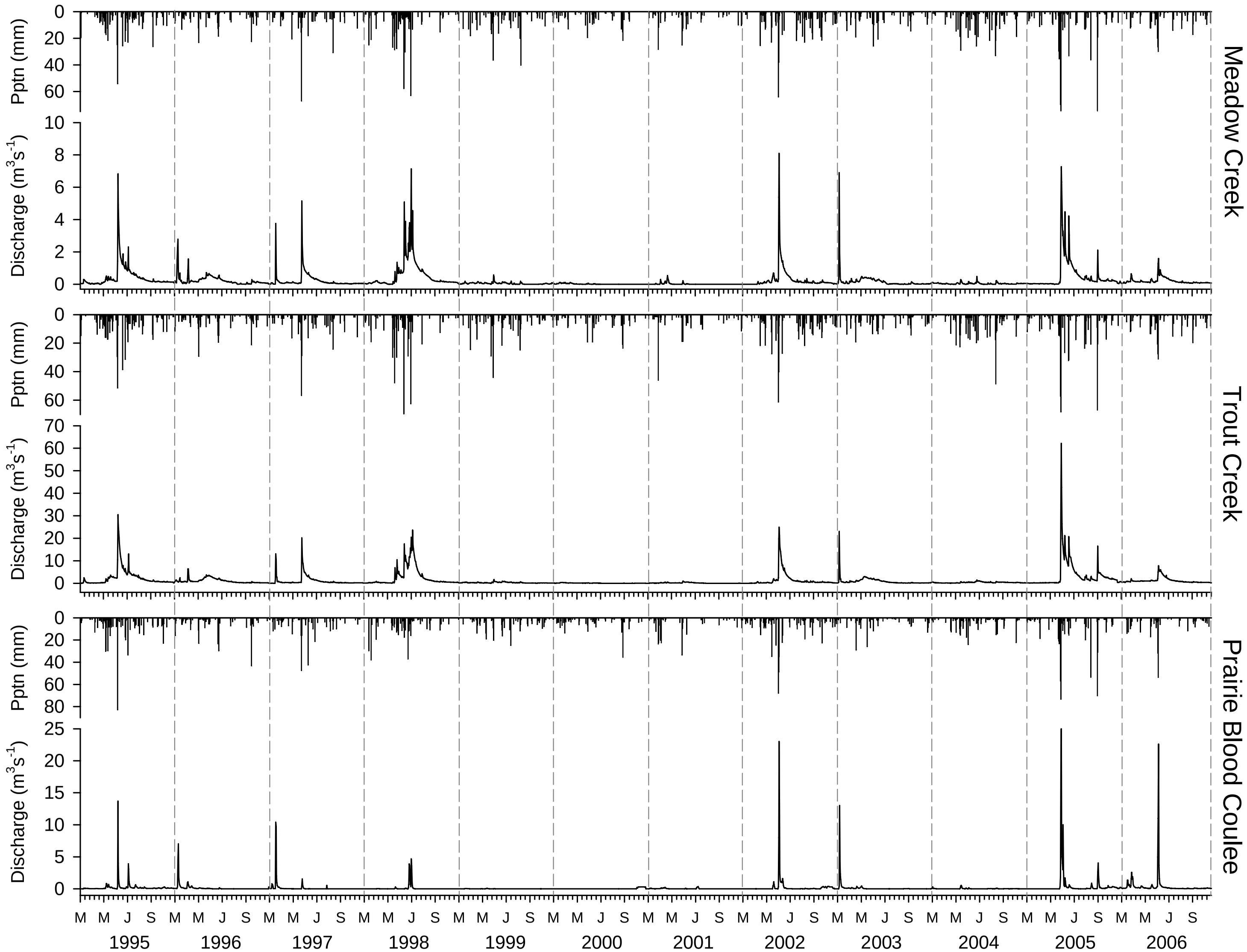

Figure 2.16. Daily discharge (lower y-axis) and precipitation (Pptn) (upper y-axis) for Meadow Creek, Trout Creek, and Prairie Blood Coulee from 1995 through 2006. Discharge and precipitation data shown were recorded from March 1 to October 31 for each year. Note the different y-axis scales.

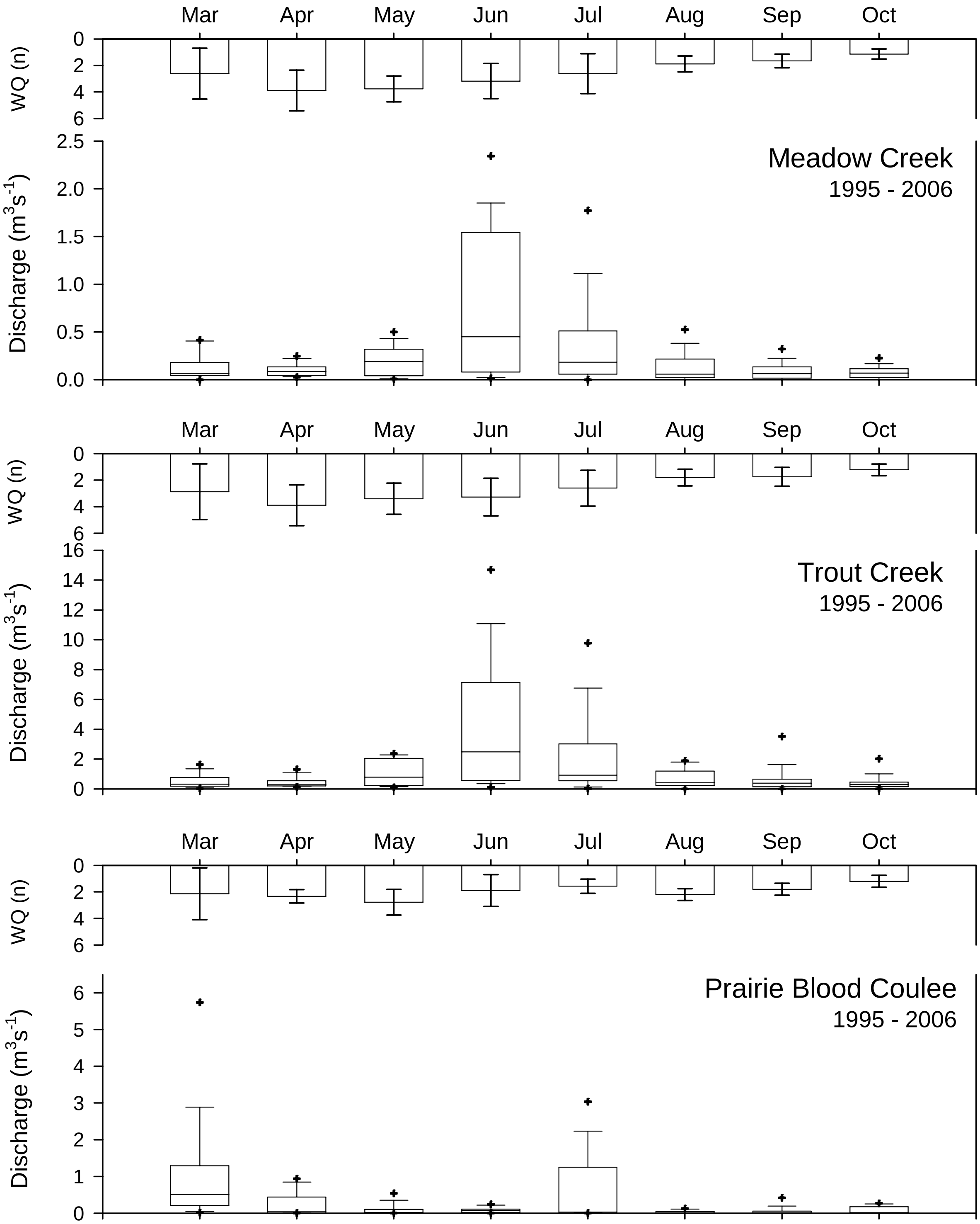

Figure 2.17. Monthly discharge (lower y-axis) and number (n) of water quality samples (WQ) (upper y-axis) for Meadow Creek (1995 to 2006), Trout Creek (1995 to 2006) and Prairie Blood Coulee (1995 to 2006) from March through October.

Northern Continental Divide Ecoregion. Willow Creek is the only AESA watershed that lies in the Northern Continental Divide Ecoregion and is a tributary (6500 ha) to the Oldman River.

Seasonal patterns in stream flow in Willow Creek were similar to watersheds in the Grassland Ecoregion (Trout and Meadow Creeks) (Figures 2.17, 2.21), where the highest monthly mean flows occurred in June (historical June Q_{mean} = 2.89 $m^3 s^{-1}$; 1997 to 2006). During the AESA monitoring period from 1999 to 2006, the years without a peak in flow in June included 1999, 2000, 2001, and 2003 (Figure 2.20). In 2000, however, flow data were missing from March through May. High flow years included 2002 and 2005 where the June Q_{mean} was 2- to 3-fold higher than the historical average (i.e., 6.2 and 9.2 $m^3 s^{-1}$, respectively). Mean annual discharge was high in 2004 and 2005 when flows were sustained through the summer until September (Table 2.5). Note that the historical record for Willow Creek is relatively short (1997 to 2006).

On average, 20 samples were collected per year, with the fewest (16) collected in 2003 and the most (26) in 1999 (Table 2.8).

Moist Mixed Grassland and Mixed Grassland (under Irrigation) Ecoregions. The watersheds in these ecoregions include the Battersea Drain, Crowfoot Creek, Drain S-6, and New West Coulee.

Battersea Drain - Battersea Drain receives irrigation return flows from the Lethbridge Northern Irrigation District (LNID). The topographically-defined watershed covers 7800 ha of intensively farmed agricultural land, but the drain also receives water from outside the drainage area. For this reason, calculations of runoff depth (discharge per watershed area) were not relevant, as they generated inaccurately large values. However, since much of the basin is under irrigation, the amount of runoff generated exceeds that in natural systems in the Moist Mixed Grasslands Ecoregion.

Battersea Drain was sampled from 1998 to 2006 (Table 2.2, pg. 2-6). Flows were typically low prior to the irrigation season (in March and April) and elevated from May to September. The number of samples collected each year was relatively constant as a bi-weekly sampling regime was generally followed (Figure 2.19). The most samples were collected in 2004 (25 samples) and the least in 1999 (15 samples). On average, 19 samples were collected per year (Table 2.8)

In 2002 and 2005, rain events were > 65 mm and generated sufficient runoff to generate discharge peaks > 2 $m^3 s^{-1}$ (Figure 2.18).

Battersea Drain has been well studied by researchers. Other studies in the Battersea Drain include Riemersma et al. (2004), Rodvang et al. (2004), and Rock and Mayer (2004). In addition, several sites in the Battersea Drain were studied from 1999 to 2007 by ARD in partnership with the Oldman Watershed Council (Little 2003; Saffran 2005).

Crowfoot Creek - Crowfoot Creek is a flow-regulated system that serves as a return flow stream for the Western Irrigation District. The watershed (137000 ha) is located east of Calgary and is a tributary of the Bow River. Natural flows in the creek were generated by snowmelt runoff in the spring and rainfall events in summer and fall. Highest monthly mean discharge typically occurred in April, then decreased in May, and then increased to a constant discharge throughout the irrigation season (May and June to September) (Figure 2.18, 2.19).

During the water quality monitoring period (1995 to 2006), the highest mean annual discharge (Q_{mean} > 2.3 m^3 s^{-1}) occurred in 1996, 1997, and 2006 when April discharge was far above historical values due to snowmelt contributions (Table 2.5). Of the 15 years of monitoring, only five years (1996, 1997, 2003, 2004, and 2006) had spring peaks in March or April, while the remaining seven years (1995, 1998 to 2002, and 2005) had much lower spring flows (Figure 2.18). The lowest median annual flows were observed in two drought years: 2001 and 2002 (Table 2.6).

There were peaks in flow that occurred in months other than the spring in response to rainfall events. In June and/or July of 1998, 2005, and 2006, peaks in summer flow exceeded those in the historical record. In 2005, there were also elevated flow conditions toward the end of the season (August and September).

The average number of samples collected was 18 per year but ranged from 12 samples in 1995 to 23 samples in 2003 (Table 2.8).

Like Battersea Drain, Crowfoot Creek is also well studied. Additional studies carried out in the Crowfoot Creek include Ontkean et al. (2003), Cross (2003, 2006), and Patterson et al. (2006).

***Drain S-6* -** Drain S-6 is a small (3300 ha) irrigated watershed in the Mixed Grassland Ecoregion of southern Alberta. Flows in the channel were maintained from May to early October by the addition of return flows from the St. Mary's Irrigation District (SMRID). Historical data were not available for this stream.

Monitoring was conducted from 1999 to 2006. On average, 17 samples were collected per year, with the fewest (14) collected in 2002 and the most (22) in 2004 (Table 2.8). Despite negligible flows (annual Q_{mean}< 0.015 cms) in Drain S-6 from 2003 to 2006 (Table 2.5, Figure 2.21), the sampling frequency was not reduced from the bi-weekly regime. Thus, water quality was somewhat over characterized in the last four years of the monitoring record.

***New West Coulee* -** New West Coulee is a flow-regulated system, which serves as a return flow for the Western Irrigation District. The effective drainage area (10,200 ha) is a small proportion (33%) of the gross drainage area (31,200 ha). This system was monitored from 1999 to 2006. On average, 18 samples were collected per year (Table 2.2, pg. 2-6, and Table 2.8).

The hydrograph showed an increase in flow with a peak in July and flows declining thereafter. Prior to the commencement of irrigation (March and April), there was little flow in this canal, indicating that the channel flow was derived mainly from irrigation water (Figure 2.18). Flows in New West Coulee did not vary substantially among years. Even in drought years (e.g., 2001), flows and sampling regimes remained relatively constant.

One storm event (>100 mm over two days) in September 2005 generated an anomalous flow peak. Water samples were collected on the receding limb of the peak.

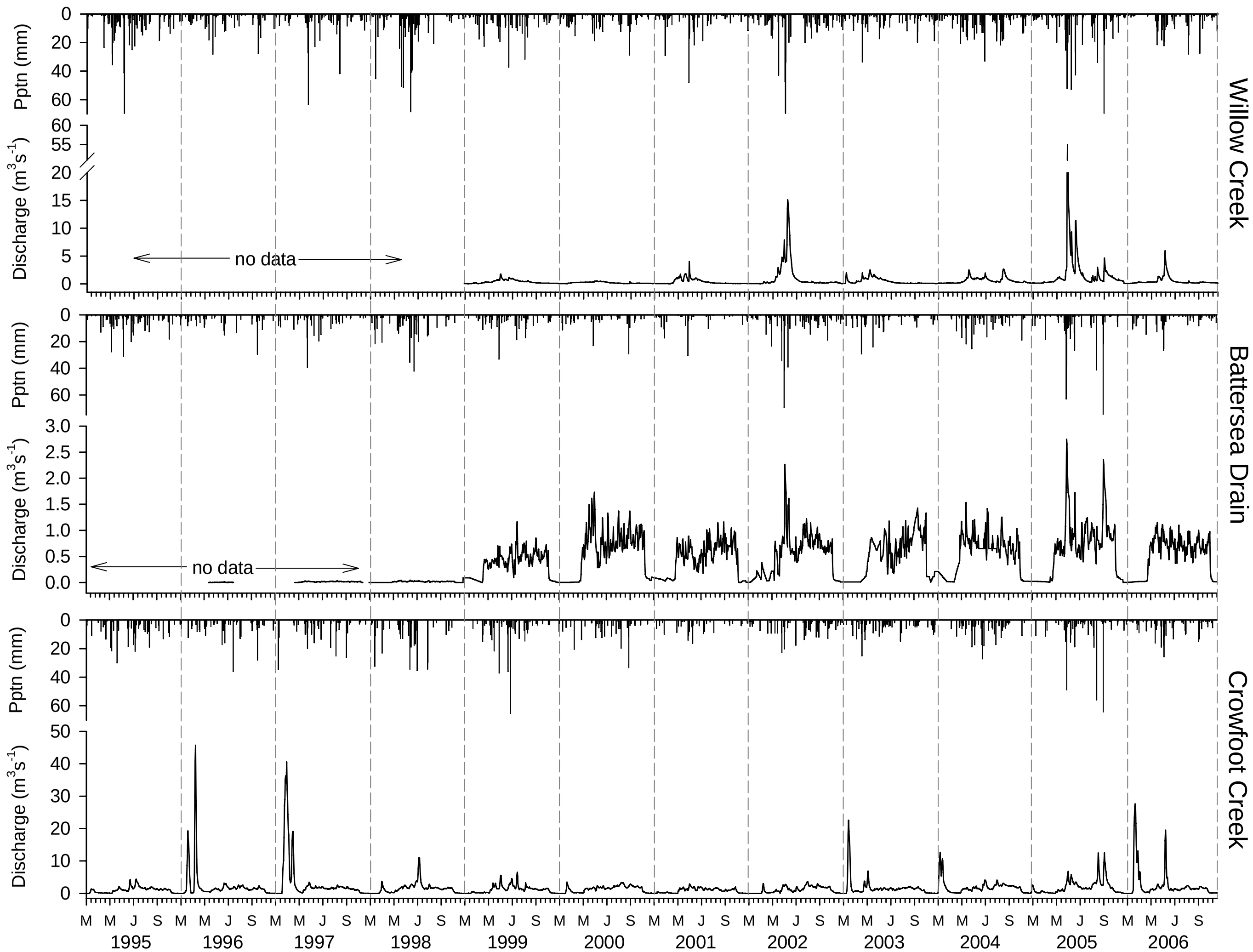

Figure 2.18. Daily discharge (lower y-axis) and precipitation (Pptn) (upper y-axis) for Willow Creek, Battersea Drain, and Crowfoot Creek from 1995 through 2006. Note that there was a poor or absent flow record for Battersea Drain in the spring before the commencement of irrigation. Discharge and precipitation data shown were recorded from March 1 to October 31 for each year. Note the different y-axis scales.

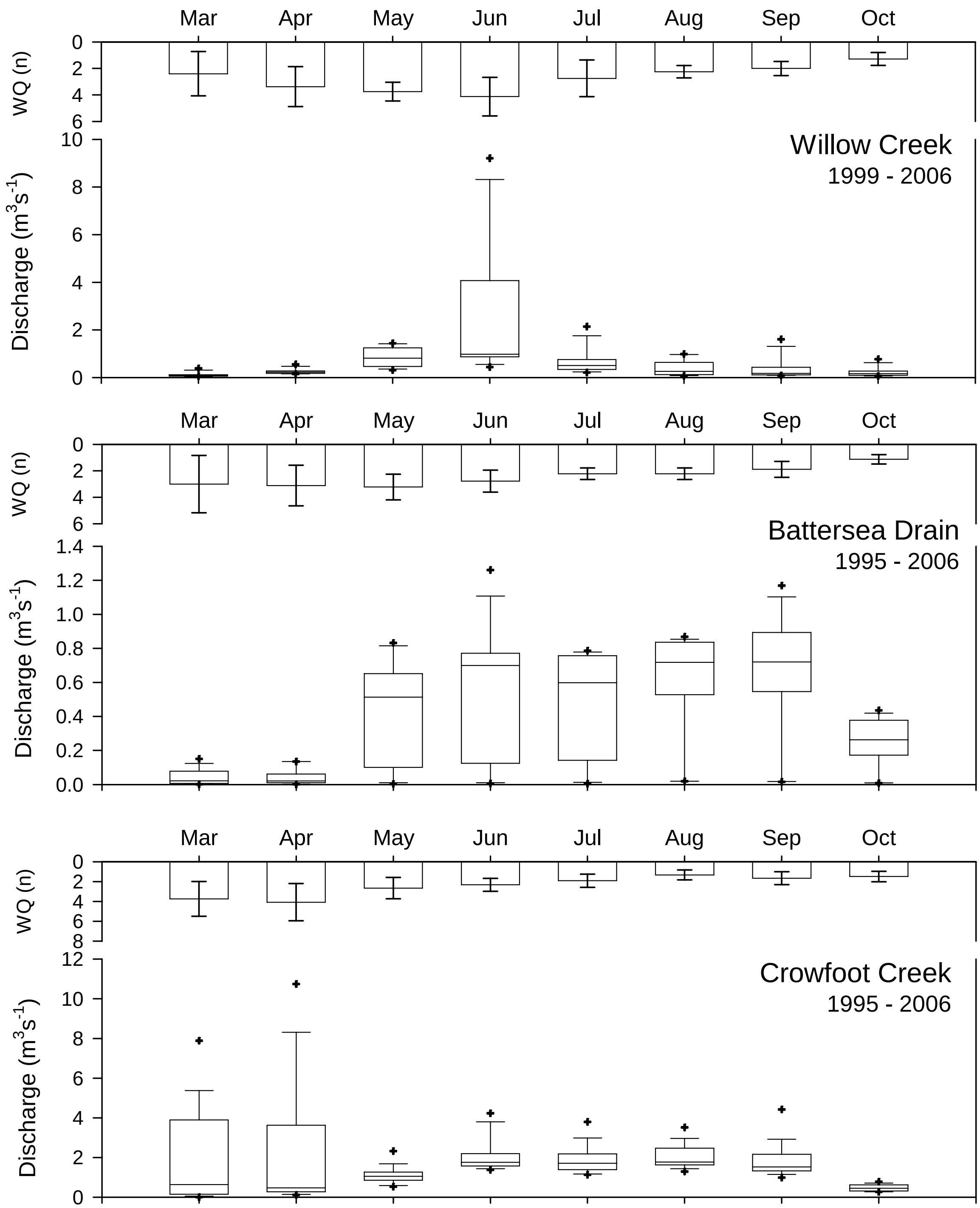

Figure 2.19. Monthly discharge (lower y-axis) and total number (n) of water quality samples (WQ) (upper y-axis) for Willow Creek (1999 to 2006), Battersea Drain (1995 to 2006), and Crowfoot Creek (1995 to 2006) from March through October.

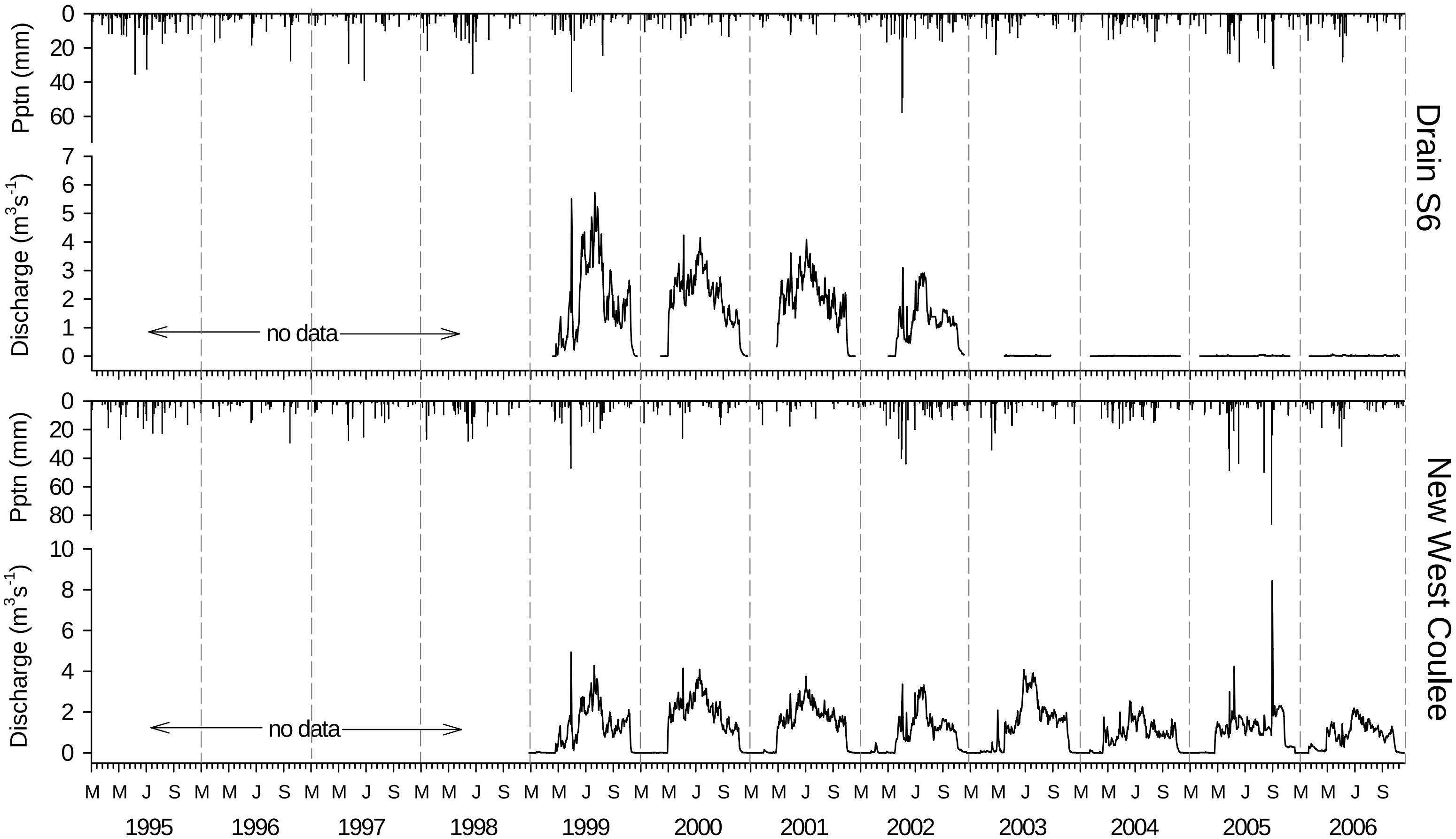

Figure 2.20. Discharge (lower y-axis) and precipitation (Pptn) (upper y-axis) for Drain S6 and New West Coulee from 1995 through 2006. Discharge and precipitation data shown were recorded from March 1 to October 31 for each year. Note the different y-axis scales.

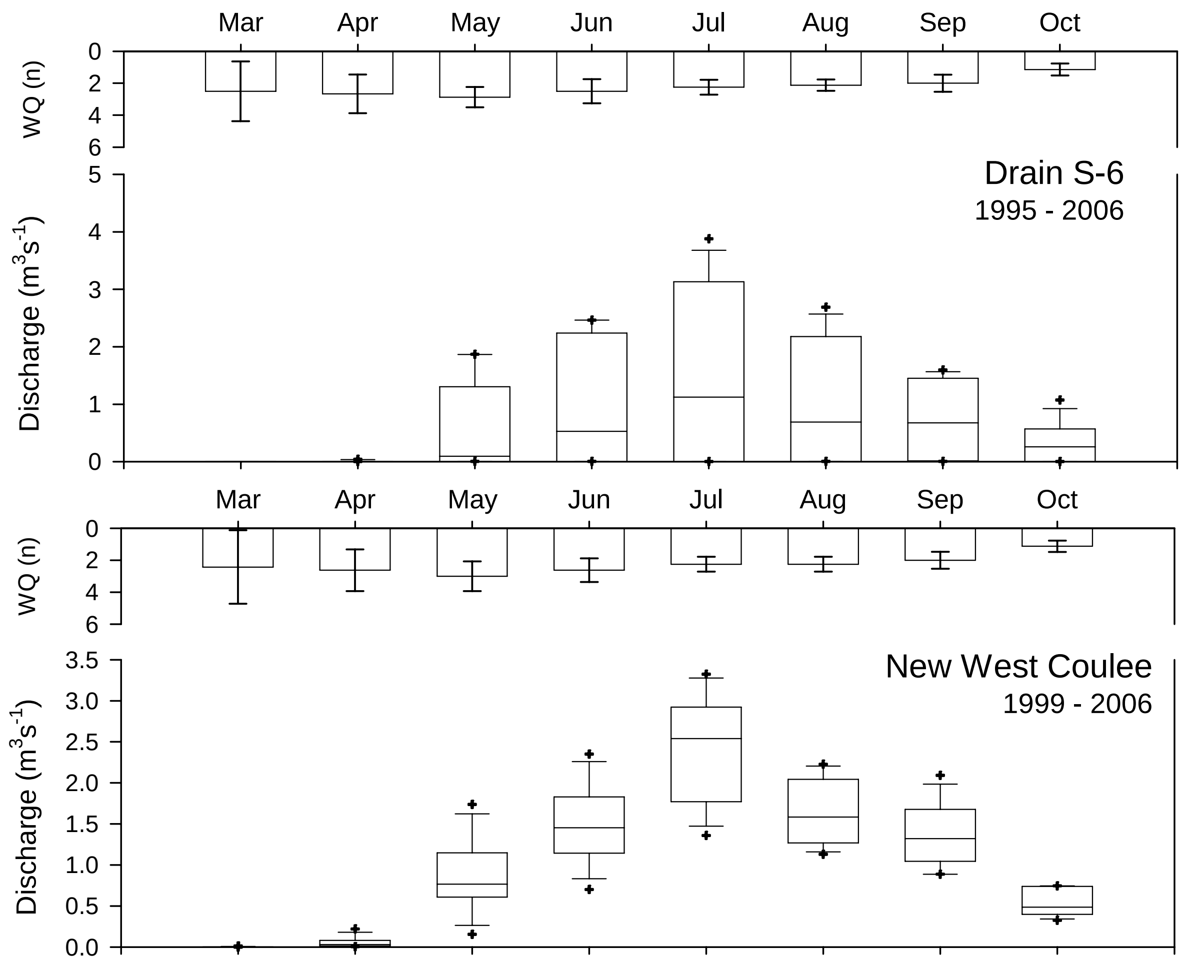

Figure 2.21. Monthly discharge (lower y-axis) and number (n) of water quality samples (WQ) (upper y-axis) for Drain S6 (1995 – 2006) and New West Coulee (1999 – 2006) from March through October.

Land Cover

Land cover in the 23 AESA watersheds varied throughout the province (Table 2.9) although some similarities were observed in watersheds in the same agricultural intensity category. The proportion of cropland, forage, grassland, and trees and shrubs varied among watersheds under different agricultural intensities. Variations in 1991 land cover distributions are illustrated using four representative AESA watersheds: Paddle River, Blindman River, Haynes Creek, and Battersea Drain (Figure 2.22). Land cover maps for the rest of the watersheds can be found in Appendix 1.

Paddle River is located in the Western Alberta Upland Ecoregion where trees and shrubs are the dominant land cover (Table 2.9). The watershed was characterized by low intensity farming (low agricultural intensity ranking in 1996), with beef cattle production as the predominant form of agriculture. Common crops included oat, tame hay, and alfalfa, which were cultivated on the relatively small proportion of crop (3%) and forage (6%) land in the basin. Blindman River, a moderate agricultural intensity watershed in the Boreal Transition Ecoregion, had a lower proportion of trees and shrubs (38%), more forage land (44%), and a similar amount of cropland (4%) compared to Paddle River. Common crops in the watershed included oat, tame hay, barley and alfalfa. Agriculture in the watershed was best described as moderately intensive mixed farming and beef cattle production. Haynes Creek, a high intensity stream in the Aspen Parkland Ecoregion, was characterized by highly intensive mixed farming. Sixty-eight percent of the watershed was under cropland, and dominant crops included barley, canola, alfalfa, and wheat. Livestock were typically beef cattle and hogs. The Battersea Drain watershed is an irrigation return flow stream and was characterized by highly intensive mixed farming dominated by large beef cattle feedlots. Cropland covered 67% of the watershed, and common crops included barley and wheat (Figure 2.22).

Table 2.9. Percent land cover for all major classifications in the AESA watersheds in 1991. Areas that exceeded the boundaries of Alberta's White zone were categorized under the heading "No Data." Bolded values represented the dominant land cover category (>60%).

				Percent Land Cover				
Watershed	Cropland	Forage	Grassland	Trees and Shrubs	Wetland	Water	Anthropogenic	No Data
Aspen Parkland								
Buffalo Creek	**62**	1	31	5				
Haynes Creek	**68**	16	10	5				
Ray Creek	**67**	10	18	5				
Renwick Creek	**81**	6	12					
Stretton Creek	**78**	2	16	3				
Threehills Creek	52	8	32	7		1		
Boreal								
Hines Creek		1		52	5			41
Grande Prairie	21	13	2	57	6			
Kleskun Drain	48	49	2	2				
Paddle River	6	3	23	**67**				
Rose Creek		7	6	**72**				14
Blindman River	4	44	14	38				
Strawberry Creek	25	50	7	17				
Tomahawk Creek	11	24	34	31				
Wabash Creek	40	20	34	5			1	
Continental Divide								
Willow Creek			9	33				58
Grassland								
Prairie Blood Coulee	**81**	0	18					
Meadow Creek	7	4	**74**	16				
Trout Creek		1	**67**	32				
Mixed/ Moist Mixed Grassland (Under Irrigation)								
Battersea Drain	**67**	20	11			1	1	
Crowfoot Creek	**77**	5	17					
Drain S6	**64**	16	18	1		1	1	
New West Coulee	51	21	26	1		2		

Bold: dominant land cover (>60%)

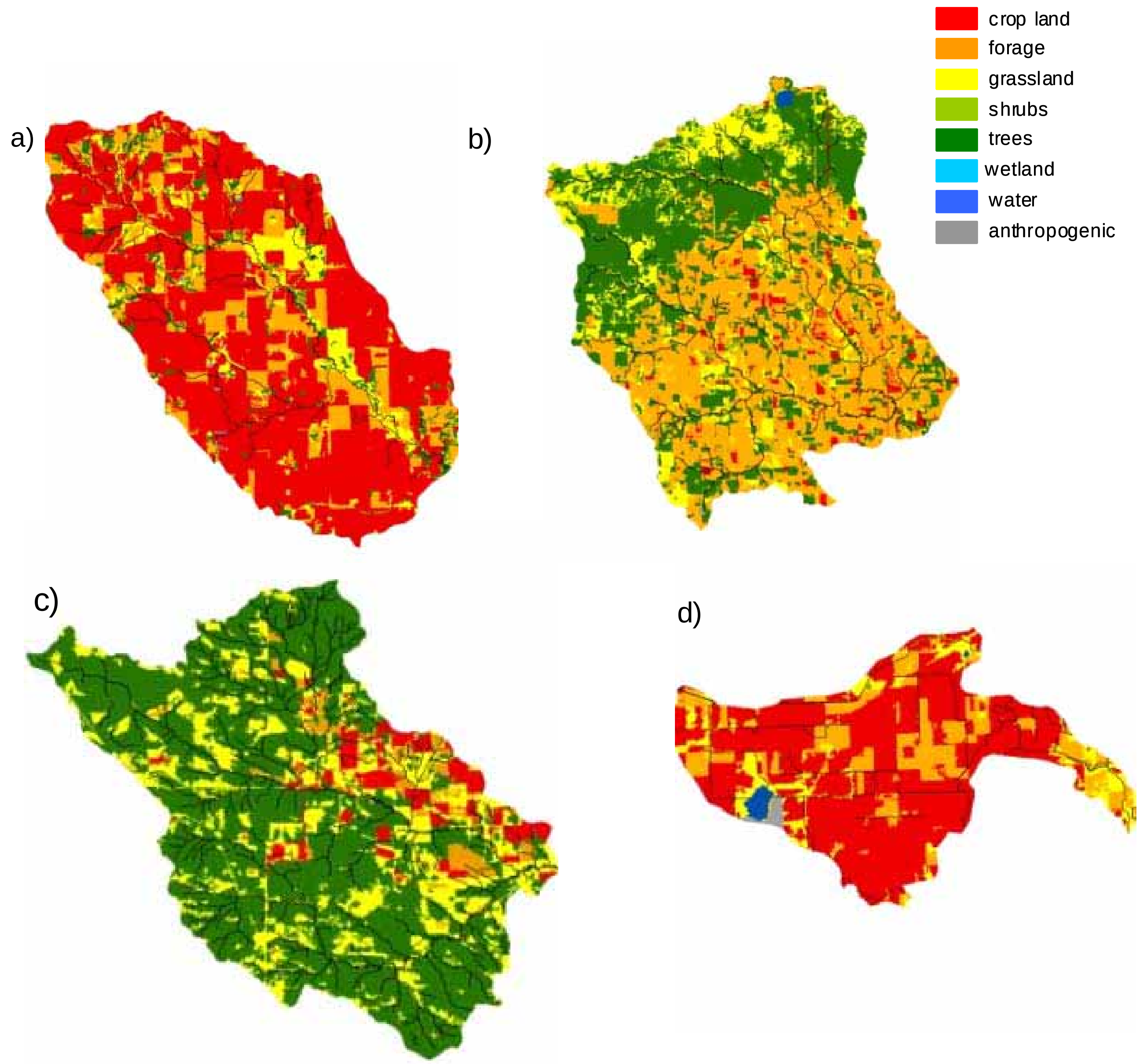

Figure 2.22. Land cover (1991 classification) of four representative AESA watersheds covering the four agricultural intensity categories. High, Moderate, Low agricultural intensity and Irrigation agriculture (1996 classification) are represented by Haynes Creek (a), Blindman River (b), Paddle River (c), and Battersea Drain (d), respectively.

Agricultural Intensity Classification in 1996, 2001 and 2006

The agricultural intensity ranks (low, moderate, and high) for the AESA watersheds, as calculated from the 1996, 2001, and 2006 Census of Agriculture data, were compared to assess whether major changes in agricultural intensity occurred in the AESA watersheds during the timeframe of the project. Individual metrics were also examined to identify if changes were a result of variability in one or more of the three metrics used to calculate overall agricultural intensity.

When the AESA Stream Survey was initiated, and agricultural intensity was based on the 1996 Census of Agriculture data, there were five low agricultural intensity streams, six moderate agricultural intensity streams, and eleven high agricultural intensity watersheds (including four streams receiving irrigation return flows) (Table 2.10).

In 2006, eight streams changed AI categories relative to their 1996 ranking. Blindman River, Buffalo Creek, Kleskun Drain, Meadow Creek, Strawberry Creek, Tomahawk Creek, and Trout Creek decreased in agricultural intensity (four from moderate to low, three from high to moderate), while Prairie Blood Coulee increased from a low to moderate agricultural intensity category. Some watersheds appeared to decrease (e.g., Paddle River) or increase (e.g., Hines Creek) in AI percentile between Census years but still remained within the same category (Figure 2.23, Table 2.11).

It is also important to note the changes in methodologies among years. Specifically, some watersheds were grouped into smaller or larger polygons before percentiles were calculated as a result of the 15-farm minimum reporting requirement (see Materials and Methods: Agricultural Intensity Metrics). As a consequence, calculations were not based on the same area in all three years for the following seven streams: Meadow Creek, Renwick Creek, Prairie Blood Coulee, Stretton Creek, Willow Creek, Hines Creek, and Kleskun Drain. Note that the changes in methodologies only affect those polygons with under 15 farms, typically watersheds with little agricultural activity. Those watersheds with higher agricultural intensity were generally not impacted by the change in aggregation method. Further information is provided on the methodology differences among years in the footnote of Table 2.10.

Agricultural intensity percentiles for each watershed were examined to assess which of the three sub-metrics metrics (fertilizer expenses, chemical expenses, and manure production) influenced the overall ranking (Figures 2.24 to 2.26, Table 2.11). Note that the examinations made were based on the actual agricultural intensity score (between 0.1 and 1) rather than solely on changes in agricultural intensity categories. The rankings are not absolute scores but relative scores (rankings). Changes in relative intensity are to be expected.

Table 2.10. Agricultural intensity rank (low, moderate, high) for 22 AESA streams from 1996, 2001, and 2006 Census of Agriculture data. Ranks were based on three metrics: manure production, fertilizer sales, and chemical sales percentiles. For full stream names refer to Table 2.1 (pg. 2-3).

Stream	1996 Census Rank	2001 Census Rank	2006 Census Rank	Average
HIN^	Low	Low	Low	Low
PAD	Low	Low	Low	Low
PRA*^	Low	**Mod**	**Mod**	**Mod**
ROS	Low	Low	Low	Low
WIL**	Low	Low	Low	Low
BLI	Mod	**Low**	**Low**	Mod
GRA	Mod	**Low**	Mod	Mod
KLE^	Mod	**Low**	**Low**	**Low**
MEA*	Mod	**Low**	High	Mod
TOM	Mod	**Low**	**Low**	**Low**
TRO	Mod	**Low**	**Low**	**Low**
BUF	High	High	**Mod**	High
HM6	High	High	High	High
RAY	High	High	High	High
REN*	High	High	High	High
STT*	High	High	High	High
STW	High	**Mod**	**Mod**	**Mod**
THR	High	High	High	High
WAB	High	**Mod**	**Mod**	High
BAT	High	High	High	High
DS6	High	High	-	High
CRO	High	High	High	High
NEW	High	High	High	High

***Watersheds that have ag intensity calculated on a larger sized polygon in 2006**

MEA REN

PRA STT

****Watersheds that have ag intensity calculated on a smaller sized polygon in 2006**

WIL

^Watersheds that have ag intensity calculated on an area that greatly exceeds the AESA watershed in 2001 and 2006

PRA HIN

KLE

Bolded entires indicate a change from the 1996 AI classification

Table 2.11. Agricultural intensity percentiles (between 0 and 1) for 22 AESA streams from 1996, 2001, and 2006 Census of Agriculture data. Percentiles are also presented for the three metrics: manure production, fertilizer sales, and chemical sales percentiles. For full stream names refer to Table 2.1 (pg. 2-3).

Stream Name	Overall Agricultural Intensity Percentile				Manure Production Percentile			Fertilizer Expenses Percentile			Chemical Expenses Percentile		
	Ave. (1996, 2001, 2006)	1996	2001	2006	1996	2001	2006	1996	2001	2006	1996	2001	2006
Battersea Drain	0.9900	0.9800	0.9901	1.0000	0.9900	0.9877	1.0000	0.9300	0.9727	0.9948	0.8800	0.9772	0.9948
Blindman River[z]	0.4658	0.6200	0.3867	0.3907	0.9000	0.7389	0.6314	0.5400	0.3648	0.3246	0.4400	0.1091	0.2448
Buffalo Creek[z]	0.7806	0.8300	0.8177	0.6941	0.6300	0.6232	0.6211	0.8700	0.8387	0.6545	0.8700	0.8173	0.7292
Crowfoot Creek	0.8278	0.8000	0.8350	0.8483	0.5300	0.5271	0.5773	0.8400	0.8784	0.8770	0.9200	0.9061	0.9193
Grande Prairie Creek	0.4825	0.5700	0.3892	0.4884	0.3700	0.1232	0.0928	0.6900	0.5806	0.7539	0.6800	0.5330	0.6875
Hines Creek[y]	0.2783	0.1200	0.3473	0.3676	0.1400	0.1724	0.1521	0.2100	0.4342	0.4921	0.1500	0.4746	0.5026
Haynes Creek	0.9602	0.9800	0.9828	0.9177	0.9200	0.8670	0.6753	0.9600	0.9901	0.9346	0.9400	0.9873	0.9297
Kleskun Drain[z]	0.3238	0.8000	0.0813	0.0900	0.5700	0.0419	0.0464	0.8700	0.1166	0.1361	0.8600	0.1777	0.1536
Meadow Creek[z]	0.6120	0.5800	0.3768	0.8792	0.8200	0.5665	0.7603	0.5100	0.3400	0.7801	0.4000	0.2741	0.9063
New West Coulee	0.9231	0.8000	0.9975	0.9717	0.6600	0.9828	0.8119	0.7900	0.9826	0.9712	0.8200	0.9949	0.9714
Paddle River[y]	0.2661	0.3800	0.3103	0.1080	0.5200	0.2660	0.1959	0.3900	0.4045	0.1492	0.3100	0.3350	0.0703
Prairie Blood Coulee[z]	0.4699	0.3900	0.4926	0.5270	0.2700	0.1552	0.2835	0.5500	0.6948	0.6806	0.4200	0.6472	0.6302
Ray Creek	0.8835	0.9900	0.8276	0.8329	0.9500	0.4187	0.6443	0.9700	0.9926	0.9686	0.9800	0.8883	0.7344
Renwick Creek	0.8870	0.9600	0.7857	0.9152	0.7600	0.2734	0.6186	1.0000	0.9504	0.9555	0.9900	0.9924	0.9635
Rose Creek[y]	0.1688	0.3800	0.0493	0.0771	0.5800	0.1108	0.1082	0.3700	0.0447	0.0864	0.2500	0.0305	0.0807
Stretton Creek	0.8167	0.9100	0.7611	0.7789	0.7800	0.6626	0.7036	0.8700	0.7345	0.6754	0.9300	0.7640	0.8333
Strawberry Creek[z]	0.7046	0.7600	0.7266	0.6272	0.8200	0.8300	0.7938	0.7100	0.6625	0.5471	0.6500	0.5533	0.4870
Threehills Creek	0.9616	0.9800	0.9458	0.9589	0.8600	0.9704	0.9253	0.9700	0.8933	0.9110	0.9700	0.8325	0.8281
Tomahawk Creek[z]	0.3748	0.5700	0.2512	0.3033	0.8100	0.4581	0.4871	0.5100	0.2630	0.3115	0.4000	0.1523	0.1875
Trout Creek[z]	0.3165	0.4800	0.3177	0.1517	0.7900	0.5936	0.4897	0.4000	0.2333	0.0812	0.2600	0.1929	0.0208
Wabash Creek	0.8047	0.9600	0.7291	0.7249	0.9100	0.8153	0.8582	0.9100	0.6824	0.6466	0.9300	0.5558	0.5781
Willow Creek	0.1338	0.1000	0.1034	0.1979	0.2100	0.2980	0.5670	0.1500	0.0496	0.0628	0.0900	0.0558	0.0781

[z] Change in agricultural intensity category 1996 to 2006.
[y] Changes in agricultural intensity percentiles observed among Census years (1996, 2001, and 2006) without a change in category.

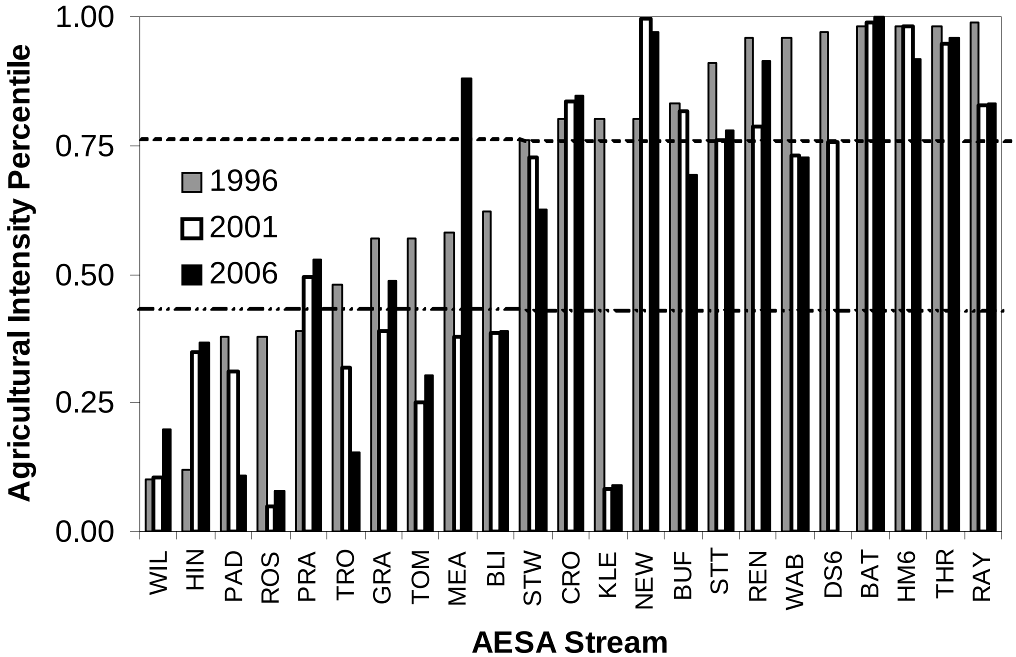

Figure 2.23. Agricultural intensity percentiles for 23 AESA streams from 1996, 2001, and 2006 Census of Agriculture data. In the context of the AESA water quality monitoring program, agricultural intensity was based upon the following percentile categories: low (<0.40), moderate (0.40 to 0.75) and high (>0.75). Refer to Table 2.1 for full stream names (pg. 2-3).

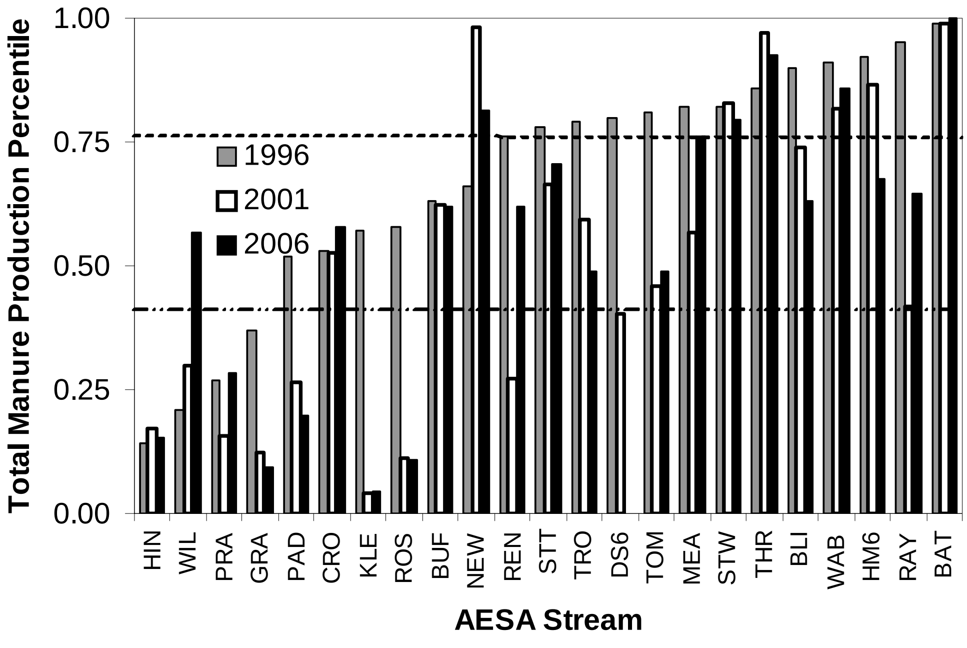

Figure 2.24. Manure production percentiles for 23 AESA streams from 1996, 2001, and 2006 Census of Agriculture data. In the context of the AESA water quality monitoring program, manure production percentiles were categorized as follows: low (<0.40), moderate (0.40 to 0.75) and high (>0.75). For full stream names refer to Table 2.1 (pg. 2-3).

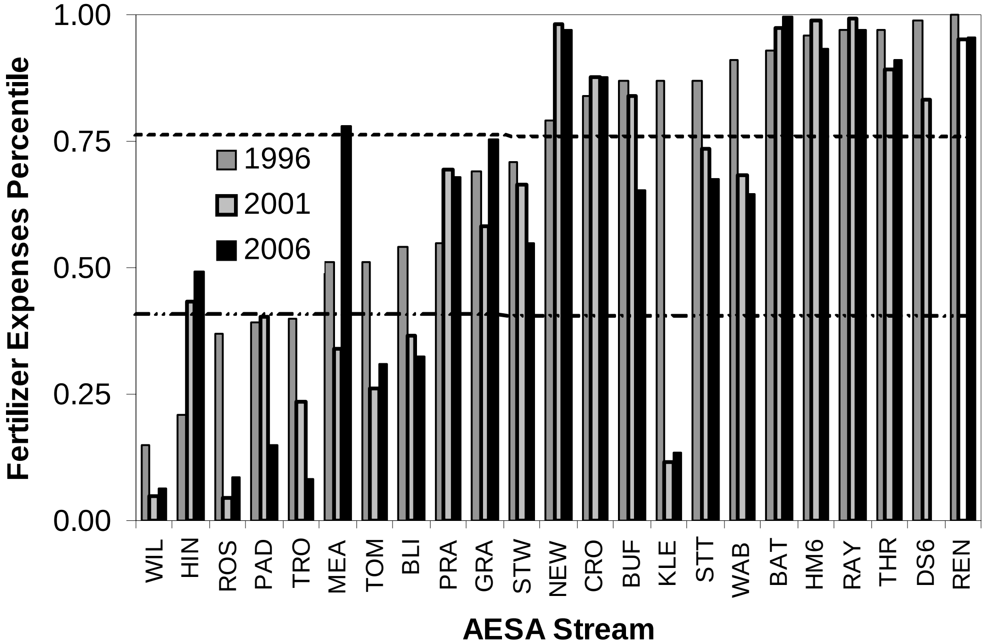

Figure 2.25. Fertilizer expenses percentiles for 23 AESA streams from 1996, 2001, and 2006 Census of Agriculture data. In the context of the AESA water quality monitoring program, fertilizer expenses percentiles were categorized as follows: low (<0.40), moderate (0.40 to 0.75) and high (>0.75). For full stream names refer to Table 2.1 (pg. 2-3).

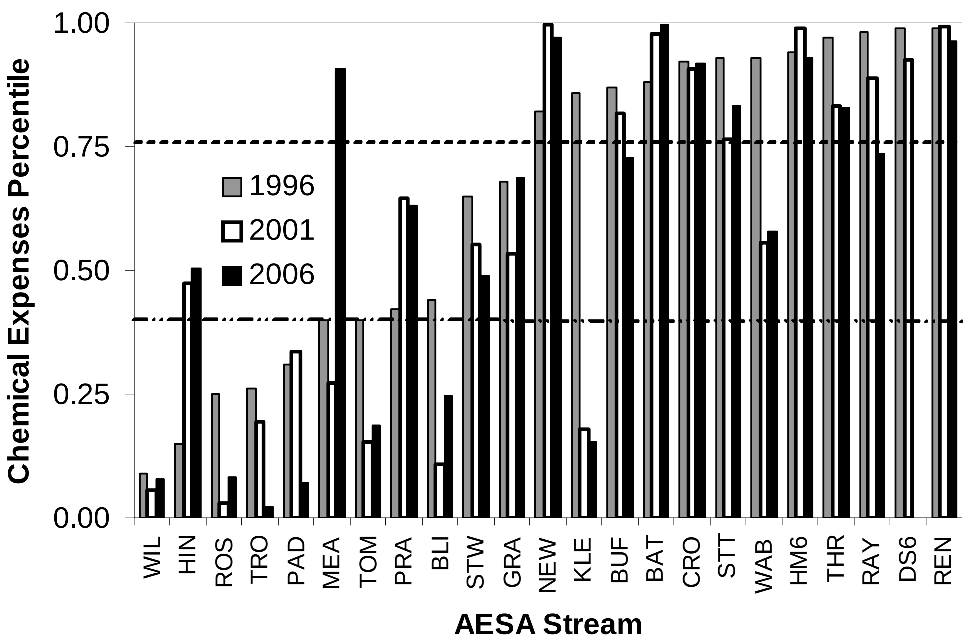

Figure 2.26. Chemical expenses percentiles for 23 AESA streams from 1996, 2001, and 2006 Census of Agriculture data. In the context of the AESA water quality monitoring program, chemical expenses percentiles were categorized as follows: low (<0.40), moderate (0.40 to 0.75) and high (>0.75). For full stream names refer to Table 2.1 (pg. 2-3).

Prairie Blood Coulee (PRA). An increase in agricultural intensity percentile was observed in Prairie Blood Coulee from 1996 (0.3900) to 2006 (0.5270). This increase was driven by increases in chemical (0.4200 to 0.6302) and fertilizer (0.5500 to 0.6806) expenses percentiles (Figures 2.25 and 2.26). Its rank in manure production remained fairly constant. Prairie Blood Coulee was classified as a low agricultural intensity watershed in 1996 but had one of the higher rankings within the low agricultural intensity category and was close to the category cutoff. Although the watershed would have been classified as a moderate agricultural intensity watershed (from low), it was already close to the low and moderate category cutoff (0.4). Furthermore, the size of the amalgamated polygons used to calculate the agricultural intensity of the watershed was larger in 2006 than previous years. Percentiles in 2001 and 2006 were also calculated based on an amalgamated polygon area that greatly exceeded the AESA watershed boundary. It is uncertain if the differences in amalgamated polygon area influenced the observed change in agricultural intensity percentiles.

Blindman River (BLI). Blindman River was classified as a moderate agricultural intensity watershed based on 1996 Census of Agriculture data (Table 2.10 and 2.11); however, it ranked among the high agricultural watersheds for manure production (0.9) and the low agricultural watersheds for chemical expenses (0.44). The manure production percentile for Blindman River decreased between 1996 and 2006 (from 0.9000 to 0.6314) (Figure 2.24, Table 2.11). Moreover, fertilizer and chemical expenses percentiles also decreased (Figures 2.25 and 2.26). Although the percentiles for each metric decreased, the average AI from the 1996, 2001, and 2006 Census data still remained within the range of percentiles for moderate agricultural intensity (0.4658).

Meadow Creek (MEA). Agricultural intensity percentiles in Meadow Creek went from moderate to low to high in 1996, 2001, and 2006, respectively (Table 2.10 and 2.11, Figure 2.23). All AI metrics decreased in 2001 compared to 1996. However, dramatic increases in fertilizer and chemical sales percentiles drove up the agricultural intensity rank in 2006 (Figures 1.26 to 1.28). As in Prairie Blood Coulee, a larger amalgamated polygon area was used in 2006 in the agricultural intensity calculations than in 2001, which may have influenced the increase in fertilizer and chemical sales rankings. The averaged agricultural intensity percentile (1996, 2001, and 2006) was 0.6120; therefore, the Meadow Creek was still examined as a watershed under moderate intensity agriculture.

Kleskun Drain (KLE) and Tomahawk (TOM) and Trout (TRO) Creeks. All three streams ranked as moderate AI watersheds in 1996 and decreased to low AI in 2001 and 2006 (Table 2.11). All three metrics decreased from 1996 to 2001 in each watershed and remained lower in 2006 than was previously reported in 1996 (Figures 2.24 to 2.26, Table 2.11). Chemical and fertilizer percentiles decreased from a moderate to low rank in Tomahawk Creek, while the manure percentile decreased from high to low. Trout Creek showed similar patterns to those observed in Tomahawk Creek. In contrast, manure production percentiles in Kleskun Drain showed a smaller decrease than the fertilizer and chemical sales metrics (Figures 2.24 to 2.26). There was uncertainty in the initial agricultural intensity assessment for Kleskun drain as a larger amalgamated polygon area was used to calculate the metric percentiles in 2001 and 2006. The uncertainty in the watershed's initial agricultural intensity assessment and the larger amalgamated polygon area used to calculate the metric percentiles in 2001 and 2006 make changes in agricultural intensity in this watershed questionable.

Buffalo Creek (BUF). Buffalo Creek remained a high AI watershed in 2001 (0.8177) but decreased to an overall moderate AI percentile in 2006 (0.6941; Figure 2.23, Table 2.11). Its rank in manure production percentile remained fairly constant during the monitoring period (0.63 to 0.6211; Figure 2.24); however, the fertilizer expenses percentile decreased between 1996 and 2006 (from 0.8700 to 0.6545; Figure 2.25, Table 2.11). Chemical expenses percentiles also decreased each Census year (Figure 2.26). Although the overall AI decreased from a high to moderate agricultural intensity category, the differences in ranking were not that great and were close to the cutoff between the two agricultural intensity categories (0.75). Furthermore, the average AI percentile (1996, 2001, and 2006) was still within the high agricultural intensity category (Table 2.11).

Strawberry (STW) and Wabash (WAB) Creeks. Strawberry and Wabash Creeks decreased from high to moderate AI ranks from 1996 to 2001. The 1996 AI percentile in Strawberry Creek was already close to the cutoff between the moderate and high agricultural intensity ranks (1996 AI: 0.7600). Although the manure production metric remained relatively constant above the high agricultural category cutoff (0.7938-0.8300), the fertilizer and chemical sales percentiles decreased from the upper boundary to lower boundary of the moderate AI percentile range (Figures 2.24 to 2.26). Manure production and chemical and fertilizer sales metrics in Wabash Creek showed similar patterns as Strawberry Creek: manure percentiles remained above the high agricultural intensity cutoff and fertilizer and chemical sales percentiles decreased. However, the AI percentile in 1996 was already high (0.96). Even though the AI percentile in Wabash Creek dropped in 2001 and 2006 (0.7291 and 0.7249, respectively), the percentiles were close to the cutoff between the moderate and high agricultural intensity categories (0.75). The average AI percentile (0.8047) for the monitoring period (1996, 2001, and 2006) also remained above the 0.75 cutoff for the high agricultural intensity category.

Although AI ranks changed in many streams among Census years (Table 2.10), only four streams changed in agricultural intensity categories when the 1996 rankings were compared to the average percentiles from the 1996, 2001, and 2006 Census rankings: Kleskun Drain, Prairie Blood Coulee, Strawberry Creek, and Tomahawk Creek. (Table 2.11). All of the watersheds but Kleskun Drain had agricultural intensity percentiles that were close to the agricultural intensity category cutoffs. Furthermore, it was uncertain how changes in methodologies affected the percentiles for watersheds such as Kleskun Drain. Despite the changes found in AI with time, it was deemed the 1996 agricultural intensity classifications for the 23 AESA watersheds would be used for subsequent nutrient, bacteria and pesticide data interpretation and analyses.

Spearman Rank correlations were computed between an average of the AI percentiles from 1996, 2001, and 2006 and average of the three metrics. As expected, all correlations were positive. The AI percentile for all 23 AESA watersheds was very strongly correlated with fertilizer and chemical sales percentiles (0.95 for both), while average fertilizer and chemical sales percentiles were also strongly correlated with each other (0.98). The average manure production percentiles were strongly correlated with the average agricultural intensity percentiles

(0.66) but only weakly correlated with fertilizer or chemical sales percentiles (0.46 and 0.43, respectively).

Interestingly, the correlations imply that livestock and crop production may be somewhat independent. The results indicate that some agricultural watersheds may be better suited to either crop or livestock production. In Alberta, intensive livestock operations tend to be geographically concentrated, as observed in Battersea Drain, which ranks among the top watersheds with a high density of intensive livestock operations (Statistics Canada 2006). Similarly, many of the foothills areas are better suited to cow-calf production than crop production, including the Willow Creek watershed.

SUMMARY AND CONCLUSIONS

Other studies have shown that patterns in water quality may be affected by activity on land as well as the magnitude and timing of runoff and stream flow. To aid in the interpretation of AESA water quality data, specifically nutrient, bacteria, and pesticide data (as discussed in subsequent chapters), it is important to understand land cover and stream hydrology within a given watershed.

The province-wide and multi-year scale of the AESA Stream Survey captured inherent variability in the timing and magnitude of stream flow among study streams and years. This natural variability introduces some challenges in addressing the second overall study objective of determining changes in water quality during the 8- to 15- years of monitoring. There were major differences in stream flow conditions and sampling intensity among years in individual streams, data gaps during years of drought, and anomalous flood events that could have had a large influence on water quality. These challenges are discussed further in the interpretations of nutrient, fecal bacteria, and pesticide data sets in Chapters 3 to 5, respectively.

Objective 1: Understand precipitation and hydrology patterns (1995 to 2006) in the 23 AESA watersheds and the similarities and/or differences among Ecoregions.

- For the period of study (1995 to 2006; 1999 to 2006), the timing and magnitude of stream flow varied naturally among Ecoregions and years. As such, water quality will be characterized at different times of the year in different regions of the province:
    - Aspen Parkland Ecoregion
        - Streams in this Ecoregion flowed primarily in the spring in response to overland runoff from snow melt. In years with smaller snow packs, hydrological connectivity with land was low, as there was little response to summer rain events.
    - Boreal Transition and Western Alberta Upland Ecoregions
        - Streams in these Ecoregions typically flowed continuously during the open water season in response to snowmelt and summer precipitation events.
    - Clear Hills Upland and Peace Lowland Ecoregions
        - The timing of runoff and flow was highly variable among years. Flow peaks occurred in spring or summer, both, or not at all.

  - Grassland and Northern Continental Divide Ecoregions
    - Stream flow in these ecoregions generally peaked in June and receded thereafter.
  - Mixed/Moist Mixed Grassland Ecoregions (Irrigated Agriculture)
    - The hydrology of watercourses receiving irrigation return flows was largely influenced by the timing of crop irrigation. Flows tended to be uniformly high from May through September.
- Precipitation was highly variable among the years of study, and the timing of floods and droughts varied spatially. In the early years of monitoring, 1995 and 1998 were wetter years in the Grassland Ecoregions (Mixed, Moist Mixed and Fescue), while in the latter years of monitoring, 2002 and 2005 were wet in the Grassland Ecoregions and the Continental Divide Ecoregion. Precipitation totals were notably higher in 1996 and 1997 in the Boreal Ecoregion and again in 2000 and 2004. In the Aspen Parkland Ecoregion, 1999 and 2005 were the wettest years.
- Winter precipitation totals were not examined but would likely provide insight on the magnitude of spring runoff.
- There were two noteworthy weather events during the period of study:
  - Drought in 2000, 2001, 2002 that impacted each Ecoregion in one or more years.
  - Heavy rains in southern Alberta in 2005.

Objective 2: Examine the flow biased sampling regime and its implications in subsequent data interpretation.

- The flow biased sampling regime was not without difficulties. There were years when certain streams were over or under characterized with respect to water quality sampling and relative to stream flow.
- Sampling frequency became more representative of changes in stream flow and storm events as staff became more familiar with the watersheds and their respective hydrological responses.

Objective 3: Characterize land cover in the AESA watersheds and understand the variations in land cover among watersheds under differing agricultural intensities.

- The relative proportion of land cover categories, including cropland, forage, grassland, and trees and shrubs, varied among watersheds. In general, watersheds with high agricultural intensity (dryland or irrigated) tended to have higher proportions of crop land compared to watersheds with low or moderate agricultural intensity.

Objective 4: Evaluate the Agricultural Intensity metric and examine changes in Agricultural Intensity percentiles over the three census years (1996, 2001, and 2006).

- Evaluation of the agricultural intensity metric from 1999 to 2006 revealed a decrease in agricultural intensity categories in seven watersheds, generally as a result of decreases in manure production percentiles. Agricultural intensity percentiles increased in one watershed (Prairie Blood Coulee) between 1996 and 2006 as a result of increases in fertilizer and chemical sales percentiles.
  - Some of the perceived changes may be artifacts of changes in polygon areas for which agricultural intensity data were reported in successive census years. These uncertainties mainly apply to watersheds with lower agricultural intensity.

    - Differences between 1996 and 2006 percentiles in some watersheds were small and just spanned the cutoff between two agricultural intensity categories (e.g., Strawberry), while some watersheds remained within the same category even though a greater decrease (e.g., Rose Creek) or increase (e.g., Hines Creek) in the overall agricultural intensity percentile was observed
- The general observations appeared to support the conclusion that agricultural intensity has either remained constant or decreased in these representative AESA watersheds from 1996 to 2006.
- As the changes in the agricultural intensity percentiles were generally minimal and the average agricultural intensity from the three Census years was similar to the 1996 categories for the majority of watersheds, it was determined that all data analyses in subsequent chapters would be conducted with 1996 classifications in order to ensure consistency throughout the report.

Chapter 3: Nutrients

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INTRODUCTION

Nutrients, such as phosphorus (P) and nitrogen (N), are essential to plant growth and development but can have a negative impact on surface water quality if present in excess of plant requirements (Carpenter et al. 1998; Correll 1998). In aquatic ecosystems, excess nutrients, particularly those in dissolved form, have been shown to create toxic algal blooms, deplete oxygen, and decrease biodiversity, all of which are negative impacts on aquatic ecosystems and degrade surface water quality (Schindler 1977; Vollenweider 1976; Carpenter et al. 1998). Major agricultural sources of P and N include commercial fertilizers and animal manure. Many applications of fertilizer and animal manure in the United States exceed the nutrient requirements of the crop to which they are applied (Carpenter et al. 1998). Most soils in Alberta are deficient in soil-test phosphorus (Manunta et al. 2000; Svederus et al. 2006). However, manure application is based upon soil N concentrations that approximate crop requirements, and P is often over applied even though STP levels are low. Thus, P losses from agricultural land have been recognized as a significant contributor to surface water quality degradation in streams and tributaries. Little et al. (2007) found that the concentration of phosphorus in runoff water increased as the amount of phosphorus in the upper soil profile increased, regardless of whether the soil P was from nonmanured or manured soil. The over-application and resulting excess P and N has the potential to accumulate in the soil and subsequently be transported to surface waters through direct runoff of nutrient enriched water or by subsurface or shallow groundwater pathways (Carpenter et al. 1998). The transport of agricultural nutrients and sediment into surface water bodies is a considerable environmental concern in many agricultural watersheds (Moog and Whiting 2002).

The relationship between agricultural watersheds and surface water quality in Alberta has previously been investigated through the Provincial Stream Survey funded under the Canada-Alberta Environmentally Sustainable Agriculture (CAESA) agreement in Alberta (Anderson et al. 1998b). From 1995 to 1996, the Provincial Stream Survey monitored 27 agricultural streams, of different agricultural intensities, using a similar flow-biased sampling regime. Fifteen of these streams were carried over into the AESA water quality program. The Provincial Stream Survey found differences in nutrient levels among agricultural intensity groupings (Anderson et al. 1998b). Specifically, high agricultural intensity streams had a higher degree and frequency of non-compliance with total P (TP) and total N (TN) guidelines and higher peak and median in-stream total nutrient concentrations and flow-weighted mean concentrations (FWMC) compared to moderate and low agricultural intensity streams. Furthermore, the ratio of dissolved nutrients to total nutrients was also higher in high agricultural intensity watersheds. However, the Provincial Stream Survey showed ecoregional characteristics (i.e., flow patterns and runoff zones) often superseded the influence of agricultural intensity on nutrient loading and exports. Suspended solids, mass loads, and exports tended to be highest in streams that had the highest discharge (generally low agricultural intensity watersheds) and lowest in streams that had the lowest discharge (generally high agricultural intensity watersheds). Exceptions to this pattern included loads and exports for nitrite-N + nitrate-N (NO_2^--N+NO_3^--N) and total dissolved P (TDP), which were highest in the high agricultural intensity watersheds. These dissolved forms of nutrients are typical of agricultural runoff; thus, agricultural intensity remained the prominent factor (Anderson et al. 1998b).

Objectives

The AESA water quality program (AESA Stream Survey) emerged from the CAESA study recommendation that a comprehensive, long term provincial water quality monitoring program be developed. From 1999 through 2006, water quality samples were collected and analyzed for nutrients in 23 agricultural watersheds across Alberta to address two objectives:

i) Learn more about how stream water quality is impacted by low, moderate, and high intensity agriculture in Alberta; and
ii) Track changes in water quality as the industry grows and agricultural management practices change.

The purpose of this chapter is to address the two objectives of the AESA Stream Survey using the nutrient data collected. Nutrient data are presented in four ways in order to address the objectives: instream concentrations (compliance to guidelines), flow weighted mean concentrations (FWMCs), mass transport, and export coefficients. Flow weighted mean concentrations are mass normalized for flow and thus allow for comparison of streams with different flow regimes. Loads represent the mass of nutrient (e.g., TP) passing through a stream during a given time period, while exports permit the comparison of nutrient loads among watersheds of different sizes. However, both loads and exports tend to be highly influenced by stream discharge as they are not normalized for flow like FWMCs. Exports and loads are primarily a function of soil type, land use, landscape characteristics, and the amount, timing, and intensity of precipitation.

The specific objectives of this chapter were as follows:

i. Compare instream nutrient concentrations to Canadian Water Quality Guidelines for the Protection of Aquatic Life (PAL) and livestock watering to assess the potential impact of agricultural activity on water quality as well as the impact of different intensities (low, medium, and high) and types (dryland versus irrigation) of agriculture.
ii. Compare nutrient FWMCs and exports among agricultural intensity categories and ecoregion areas to assess whether agricultural and/or ecoregional characteristics influenced FWMCs and exports.
iii. Examine nutrient FWMC and export relationships with agricultural intensity metrics as a means to assess the impact of different intensities (low, moderate, and high) and types of agriculture on water quality (Objective 1 of the AESA stream survey).
iv. Identify changes in water quality with time (Objective 2 of the AESA stream survey).
v. Examine seasonal patterns to assess whether nutrient concentrations and exports vary during different times of the monitoring period (i.e., spring, summer, and fall).

MATERIALS AND METHODS

Nutrient Sample Collection

Samples were collected from 1999 to 2006 from 23 streams (Table 2.8, Chapter 2). In addition, 16 of the AESA streams were sampled from 1995 to 1996 as part of the CAESA program. Several of the CAESA streams were also sampled in 1997 and 1998. Years with no sample collection in a stream indicate that stream flow did not occur (Table 2.8, Chapter 2); however, the lack of sample collection was a result of logistic problems in some cases (see Chapter 2: Results and Discussion, Hydrology - Buffalo and Stretton Creeks). Note that the sampling location in Willow Creek changed in 1999 because the flow gauging station at the original CAESA site had been discontinued; data from this watershed were only analyzed between 1999 and 2006 in this report. Also, any comparisons carried out between watersheds, agricultural intensity classifications, or location by ecoregion were completed on data from 1999-2006 to ensure results could be attributed to the factors under investigation rather than the presence or lack of data for each stream.

The AESA program was part of Alberta Environment's Quality Assurance (QA) program, which included quality control (QC) samples and data management in their Water Data System (WDS). This was an essential part of the program to ensure that the data collected were reliable and accurate for future analyses and reporting. Quality control samples were included as part of the quality assurance check in the sampling program for the purposes of evaluating the quality of the data. The QA/QC sampling program included replicate samples, split samples, and field and lab blanks for all water quality parameters. These samples were included to assess precision (splits) and potential contamination (blanks). Quality control data were removed from the data set prior to the analysis of the various parameters.

Laboratory analyses of water samples. Water samples were analyzed for the following:

- nutrients (total and dissolved forms of nitrogen and phosphorus) (Table 3.1),
- pH, temperature, non-filterable residue (NFR), and conductivity (Table 3.1).

Table 3.1. Water quality parameters measured in AESA streams.

Nutrients	
	Total Phosphorus (TP)
	Total Dissolved Phosphorus (TDP)
	Total Particulate Phosphorus (TPP), calculated as TP-TDP
	Total Kjeldähl Nitrogen (TKN)
	Nitrite Nitrogen plus Nitrate Nitrogen (NO_2^--N+ NO_3^--N)
	Nitrite Nitrogen (NO_2^--N)
	Ammonia Nitrogen (NH_3-N)[1]
	Total Nitrogen (TN), calculated as TKN + NO_2^--N + NO_3^--N
Other measurements related to inorganic chemistry	
	Suspended Solids / Non-filterable residue
	pH
	Temperature
	Conductivity

Inorganic data. Nutrient analyses were conducted at the Alberta Research Council (ARC) in Vegreville from 1995 to 1998 and the ALS Laboratory Group, Environmental Division in Edmonton from 1999 to 2006 (previously known as Envirotest labs). Water samples were submitted to the laboratory as unfiltered, filtered, or unfiltered and preserved samples depending on the parameter of interest (Depoe and Fountain 2003). Samples were analyzed by following standard procedures developed by ALS in accordance with their accreditation by the Canadian Association of Environmental Analytical Laboratories (CAEAL). Specific methods for each parameter are available in Appendix 3. For each stream, concentrations were reported from the laboratory in milligrams per liter (mg L^{-1}). The laboratory reported measurements for total phosphorus (TP), total dissolved phosphorus (TDP), total kjeldahl nitrogen (TKN), nitrite-N (NO_2^--N), nitrite+nitrate-N (NO_2^--N+NO_3^--N), and ammonia-N (NH_3-N). Note that ammonium (NH_4^+) was the form actually measured, but the laboratory determined that the analyses did not exclude the unionized form and therefore reported NH_3-N. Nitrate-N (NO_3^--N) was also intermittently reported.

Data Management and Analyses of Water Quality Data

Nutrient data analysis. All data were validated to ensure fractions of nitrogen and phosphorus did not exceed the totals (TN and TP). If the measured TDP value was higher than the measured TP in a sample, the TDP value was decreased to equal the measured TP value. All values of NO_2^--N and NH_3-N were reported as less than NO_2^--N + NO_3^--N and TKN, respectively.

All values less than the method detection limit (MDL) were set at one-half the MDL, and parameters with multiple detection limits were set at one-half the highest MDL (Westbrook and McEachern 2002; Appendix 3). Total nitrogen (TN) was calculated by adding TKN and NO_2^-+ NO_3^--N values. Organic nitrogen (Org N) was calculated by subtracting NH_3-N from TKN. Total particulate phosphorus (TPP) was calculated by subtracting the TDP value from the TP value. Nitrate-N (NO_3^--N) was calculated by subtracting NO_2^--N from NO_2^--N +NO_3^--N when it was not determined by the lab.

Compliance with surface water quality guidelines. Percent compliance of in-stream concentrations was determined for parameters with existing guidelines: TN, NO_2^-, NH_3-N, NO_2^--N +NO_3^--N, NO_3^-, TKN, TP, and TDP (Table 3.2). The most current provincial guidelines were used – *Surface Water Quality Guidelines for Use in Alberta* (Alberta Environment 1999). Several of Alberta's guidelines are adopted from federal water quality guidelines – *Canadian Environmental Quality Guidelines* (*CEQGs*) (CCME 1992, 1999a, b, 2002). Percent compliance was calculated for each stream for each year from 1999 to 2006.

The Canadian Council of Ministers of the Environment (CCME) guidelines for ammonia are dependent on pH and temperature and are based on pre-defined pH and temperature values (Table 3.3). To calculate compliance, ambient data needed to fit within these set categories defined in Table 3.3. Given this, a complete dataset was needed. If an ambient pH or temperature value was missing, a value was interpolated based on measurements taken before and after the sample was collected. Using the complete data set, ambient pH and temperature data were converted to match the categories listed in Table 2.3. For example, if the ambient pH value was greater than 7.5 and less than or equal to 8, the converted pH value was set at 8. The same steps were followed for the ambient temperature measurement. Therefore, if the ambient temperature measurement was 10°C, the value for this category would be 0.832 (Table 3.3). Once categories and respective values had been determined, compliance was calculated by subtracting the ambient ammonia concentration from the set values in Table 3.3. If the value was negative (i.e., the ambient concentration was less than the set values in Table 3.3) the sample was compliant.

Table 3.2. Existing Canadian Council of Ministers of the Environment (CCME) guidelines for the Protection of Aquatic Life and other agricultural uses such as livestock watering.

Nutrient Parameters	Protection of Aquatic Life[z]	Livestock Water[z]
Total Phosphorus (TP)	0.05 mg L^{-1}	-
Total Nitrogen (TN)	1.0 mg L^{-1}	-
Nitrite and Nitrate (NO_2^--N + NO_3^--N)	-	100 mg L^{-1}
Nitrite (NO_2^--N)	0.06 mg L^{-1}	10 mg L^{-1}
Nitrate (NO_3^--N)	2.9 mg L^{-1}	-
Ammonia (NH_3-N)	see Table 3.3	-

[z] Alberta Surface Water Quality Guidelines (Alberta Environment 1999, CCME 1999a); CWQGs (CCME 1999a, b, 2002, 2003)

To convert NO_2^--N values to NO_2^-, multiply by 3.29; to convert NO_3^--N values to NO_3^-, multiply by 4.43.

Table 3.3. Canadian Council of Ministers of the Environment (CCME) ammonia-N (NH3-N) guidelines for the Protection of Aquatic Life. Source: CCME 2000.

Temp.				pH				
(°C)	**6**	**6.5**	**7**	**7.5**	**8**	**8.5**	**9**	**9.5**
0	184.8	58.4	18.48	5.856	1.864	0.5992	0.2	0.0336
5	122.4	38.64	12.24	3.872	1.232	0.4016	0.1376	0.0272
10	81.6	25.92	8.24	2.608	0.832	0.2744	0.0968	0.0232
15	55.76	17.6	5.584	1.776	0.572	0.1912	0.0712	0.0208
20	38.4	12.16	3.856	1.232	0.3992	0.1368	0.0536	0.0192
25	26.8	8.48	2.696	0.864	0.2832	0.1	0.0424	0.0176
30	18.96	6	1.912	0.6136	0.2048	0.0752	0.0344	0.0168

* Note that values are presented as total NH_3-N versus NH_3.

Load, flow weighted mean concentration (FWMC), and export calculations. Mass loads were determined from instantaneous daily discharge data and periodic sample data using version 4.5 of the program FLUX (U.S. Army Corps of Engineers 1995). Six algorithms are available for mass load calculations; method #3, used by the International Joint Commission, was chosen as it best suited the data set. The entire data record from 1995 (or the earliest record available for each stream) to 2006 for in-stream chemistry and flow were used in FLUX. Data were grouped by sampling year (i.e., March 1 to October 31). The date range option in FLUX permitted load calculations for each year of monitoring. Annual (March 1 to October 31) and monthly loads were computed. As a complete flow record could not be achieved for Drain S6, the data were not used in FLUX, and load and subsequent FWMC calculations were not completed for that watershed. Furthermore, FLUX requires chemistry data for at least three sampling dates per monitoring year in order to calculate a load. Annual loads were not calculated in the following streams as a result of too few annual chemistry data: Buffalo Creek (all N and P in 1996); Blindman River, Paddle River, Prairie Blood Coulee and Meadow, Rose, Ray, Renwick, Strawberry, Tomahawk, and Trout Creeks (TDP and TPP in 1995); and Threehills Creek (TDP and TPP in 1995 and 1998). Load and FWMCs were not reported in Haynes Creek M6 (TDP and TPP in 2002, both N and P in 2004) and Stretton Creek (TDP and TPP in 1995, both N and P in 2000 through 2004) as FLUX was not capable of accurately estimating the values with few chemistry samples and/or very low or zero flow records for years of interest. Data were treated in the same manner in FLUX for monthly load calculations.

Flow weighted mean concentrations (FWMC) were determined from total mass load estimates for a given time period divided by the total flow volume estimates for the same time period. These were part of the output generated in the FLUX program and were calculated on a monthly and annual (March to October) basis. Monthly and annual loads and FWMCs were determined in FLUX. Ratios of TN/TP and DIN/TDP were calculated from median annual FWMCs although the ratios would be the same if instream concentrations were used.

Export coefficients were calculated by dividing the total mass load by the total active drainage area to express mass export of a watershed per unit time for comparison among watersheds (Chapter 2: Table 2.1). The mass export was expressed per year but only accounted for months during the open water season (i.e., annual exports are represented by data from March to October). As discussed in the Site Selection Report (Anderson et al. 1999), the effective drainage basin size of most of the AESA watersheds was similar to the gross drainage basin size (based on topographic features), which confirmed that the watersheds were well-drained and that most of the basin could potentially contribute to stream loading. Exceptions included Buffalo and Crowfoot Creeks where a fairly substantial portion of the basins were determined to be non-contributing. Furthermore, the actual effective drainage area for watersheds that receive irrigation return flows is not well defined. As a result, exports were not calculated for the irrigated watersheds (Battersea Drain, Crowfoot Creek, Drain S6, and New West Coulee). The irrigation return flow streams in this study are used to convey irrigation source water and collect irrigation return flow; this makes the hydrology and loading difficult to estimate as water from outside of the drainage area may be channeled in and runoff originating from the basin may be moved out. Thus, export coefficients were not calculated for the irrigated streams (Battersea Drain, Crowfoot Creek and New West Coulee) as a result of the uncertainty around their respective effective drainage areas (calculations in Drain S6 were not considered as a result of insufficient flow data). Export coefficients for Buffalo Creek were calculated, and the fact that some areas of the watershed may not have contributed to flow was noted accordingly.

Statistical analysis of nutrient data. All statistical analyses were computed in SYSTAT 10 (SPSS Inc. 2000). Summary statistics were computed for in-stream, load, FWMC, and export data for all forms of N and P. Data from 1999 to 2006 were used for statistical analyses. Normality was first assessed using normal probability plots in SYSTAT 10 (SPSS Inc. 2000) before any other statistical analyses were computed.

Statistical significance statistics and correlations were computed for compliance, FWMC, and exports; however, the methods and types of statistical test used differed, as discussed below. All statistical analyses were completed using median annual data. This data was calculated by taking the median of the annual data from 1999 to 2006 for each stream. When statistical analyses were applied to groups by agricultural intensity or ecoregion, the median of the annual medians for the streams within each group was calculated. Medians were used as they best represented the data overall. High stream flow or nutrient concentrations in one or two years in a watershed influenced the mean of the data and were only reported for reference purposes.

FWMCs- Statistical differences were calculated for FWMCs to identify differences among agricultural intensity categories (low, moderate, high, and irrigated) as well as among streams within each agricultural intensity category. Flow weighted mean concentration data were transformed using Log10(x) or Log10(x+1) transformations to normalize the data. The transformed data were analyzed using ANOVA and Tukey's post hoc test. Where data did not follow a normal distribution after transformation, a nonparametric test (Kruskal Wallis One-Way ANOVA) was used on untransformed data followed by a Mann-Whitney test.

Export coefficients- Kruskal Wallis One-Way ANOVA and Mann-Whitney statistics were conducted on untransformed N and P export data as a suitable transformation to normalize the data was not found. Statistical significance was examined for N and P exports among agricultural intensity categories of the AESA dryland streams (low, moderate, and high) and the major Ecoregions in which the dryland AESA streams are located. Note that exports were not calculated for the watersheds receiving irrigation return flows and were not included in the statistical analyses. Exports for Willow Creek were included in comparisons between agricultural intensity categories but not between Ecoregions as this was the only watershed located in the Continental Divide. Statistical differences using the same nonparametric tests were examined among streams within each agricultural intensity category and ecoregion area group.

Correlations- Correlations between median annual (1999 to 2006) N and P FWMCs and exports and average agricultural intensity metrics (1996, 2001, and 2006 agricultural intensity, manure production, and fertilizer and chemical sales percentiles) were completed using the untransformed data and a Spearman's Rank correlation (r_s). Twenty-two AESA watersheds (all but Drain S6) were used in assessing the overall correlations with N and P FWMCs, while the 19 dryland watersheds were used in correlations with N and P exports. Correlations were also completed for N and P FWMCs and exports grouped by ecoregion area.

Trend Analysis- A formal statistical monotonic time series trend analysis was not carried out on the data. Preliminary statistical analyses by Depoe (data not reported) on data from 1999 to 2003 showed that sufficient power was not present in the sample size to detect the trends of interest. It was determined that at least 10 years of data under flow biased sampling would be required to complete a statistical trend analysis, even if done with a nonparametric test such as the Seasonal Mann Kendal. After initial assessment of the data set, it was decided that a statistical trend analysis would not be completed for this report. Even with the addition of CAESA data, many streams did not have enough data points as a result of a short sampling record (sampling in some streams only started in 1999), missing years resulting from low or no stream flow, and missing years resulting from too few chemistry samples for FWMC and load calculations in FLUX. Although the Seasonal Mann Kendal test is robust toward censored data and missing observations, more than 10 years of data is required, especially if serial correlation is present in the data set (Westbrook and McEachern 2002). A visual examination of annual P and N FWMCs was carried out for each watershed to flag potential increasing or decreasing temporal trends in the data set, recognizing that a statistical monotonic trend analyses would be required in the future to verify the observations.

RESULTS AND DISCUSSION

Compliance with Water Quality Guidelines: Protection of Aquatic Life (PAL)

Nutrient parameters were compared to water quality guidelines for the protection of aquatic life and livestock use. Water quality guidelines were used as a measure to assess the potential impact of agricultural activity on water quality as well as the impact of different intensities (low, medium, and high) and types (dryland versus irrigation) of agriculture.

Total Phosphorus (TP) and Total Nitrogen (TN). Compliance of TP and TN was low for all agricultural intensities (Table 3.4). Total P percent compliance ranged from an average of 7 % (± 3 % standard deviation) of 889 samples in high agricultural intensity watersheds to 48 % (± 6 %) of 730 samples taken in low agricultural intensity watersheds. Total N percent compliance ranged from an average of 9 % (± 4 %) in high agricultural intensity watersheds to 63 % (± 11

%) in irrigated watersheds. These averages from 1999 to 2006 not only demonstrate high agricultural intensity areas have the lowest compliance, but also show the wide range of compliance among the years of monitoring for each agricultural intensity (Table 3.4).

For both TP and TN, Hines Creek had the lowest compliance, while Willow Creek had the highest compliance among watersheds under low agricultural intensity (Table 3.5). Note that the AESA sampling site in the Willow Creek watershed is in the headwaters and is not the same as the AENV site at the watershed mouth (see Chapter 2: Results and Discussion - Hydrology). In moderate agricultural intensity watersheds, Tomahawk Creek had the lowest compliance and Trout Creek had the highest compliance (Table 3.5). In high agricultural intensity watersheds, Haynes Creek M6, Stretton Creek (TP), and Renwick Creek (TN) had the lowest compliance, while Strawberry Creek had the highest compliance with guidelines. Within irrigated watersheds, Crowfoot Creek had the lowest compliance, while Drain S6 had the highest compliance.

Comparisons of TP and TN compliance were also made among the Grassland, Parkland, Boreal, and Irrigated Grassland ecoregion areas from 1999 through 2006. For both TP and TN, compliance in the Grassland, Irrigated Grassland, and Boreal ecoregion areas were higher than the Parkland ecoregion, which had the lowest compliance overall. These patterns among ecoregion areas can be related back to agricultural intensities as the Parkland Ecoregion is solely composed of high agricultural intensity watersheds.

Nitrite-N (NO_2^--N). Nitrite compliance with the PAL guidelines was high among all agricultural intensities (Table 3.4). Average compliance from 1999 to 2006 ranged from 89 % (± 3 %) in high agricultural watersheds to 100 % (± 1 %) in low agricultural intensity watersheds. Percent compliance for low and moderate agricultural intensity categories was significantly lower than high and irrigated percent compliance. Unlike TP and TN, there generally was little fluctuation in percent compliance amongst years from 1999 to 2006 (Table 3.4).

Nitrate-N (NO_3^--N). A similar percent compliance was observed for NO_2^- and NO_3^-. Average compliance for all years ranged from 95 % (± 3 %) in irrigated watersheds to an average of 100 % (± 0 %) in low agricultural intensity watersheds. There was annual variation within each agricultural intensity although the fluctuation was small, ranging from 92 % to 100 % (Table 3.4). The lower average compliance in the irrigated streams was due to a low compliance in the Battersea Drain. Rodvang et al. (2004) found high levels of NO_3^- in shallow unconfined aquifers in Battersea Drain, which was attributed to the application of manure and fertilizer on irrigated lands. This difference, however, was not significant.

Ammonia (NH_3-N). On average, close to 100 % compliance with NH_3-N PAL guidelines were observed in most years for all agricultural intensity categories (Table 3.4). High agricultural intensity watersheds and irrigated streams had the lowest compliance at 94 % (± 5 %) and 95 % (± 5 %), respectively. Compliance with the guideline was 100 % (± 1%) and 99% (± 1%) in low and moderate intensity watersheds, respectively.

Other applicable guidelines and objectives. Guidelines for livestock watering exist for NO_2^- and NO_2^- + NO_3^- parameters. The NO_2^- guideline of 10 mg L^{-1} was never exceeded. The largest NO_2^- instream concentration was measured in Blindman River at 4.873 mg L^{-1}. Note that this value was measured in 1996 and was therefore not included in the average PAL compliance calculation (1999 to 2006, Table 3.4). This NO_2^- value in Blindman River would have exceeded the PAL guideline. The NO_2^- + NO_3^- guideline of 100 mg L^{-1} was also never exceeded. For comparison, the largest value recorded for the nitrite + nitrate parameter was 10.2 mg L^{-1}, which was measured in the Battersea Drain on March 18, 2003.

Additional tables with percent compliance for each year (1999 to 2006) by stream are located in Appendix 5.

Table 3.4. Interannual patterns in compliance with Protection of Aquatic Life nutrient guidelines among agricultural intensity categories from 1999 through 2006. All 23 AESA watersheds were included. The total number of samples collected each year (n) is located in parentheses beside the year.

	TP	TN	NO_2-N	NO_3-N	NH_3-N
Guideline	ASWQG (PAL)	ASWQG (PAL)	ASWQG (PAL)	ASWQG (PAL)	ASWQG (PAL)
Guideline Value	0.05 mg L^{-1}	1 mg L^{-1}	0.06 mg L^{-1}	2.9 mg L^{-1}	pH and Temp
	% Compliance				
Low Agricultural Intensity					
1999 (102)	47	60	100	100	100
2000 (104)	58	67	100	100	100
2001 (74)	49	61	100	100	100
2002 (81)	41	53	99	100	98
2003 (65)	45	54	100	100	100
2004 (92)	57	84	99	100	99
2005 (121)	46	57	100	100	100
2006 (92)	43	53	99	99	100
Average	**48**	**61**	**100**	**100**	**100**
Standard Deviation	**6**	**10**	**1**	**0**	**1**
Moderate Agricultural Intensity					
1999 (119)	20	33	100	98	100
2000 (121)	29	29	99	98	97
2001 (79)	27	38	99	100	100
2002 (87)	24	37	98	98	100
2003 (92)	30	39	100	100	100
2004 (106)	27	46	98	100	99
2005 (133)	17	31	100	100	99
2006 (97)	19	30	100	100	99
Average	**24**	**35**	**99**	**99**	**99**
Standard Deviation	**5**	**6**	**1**	**1**	**1**
High Agricultural Intensity					
1999 (176)	5	4	88	100	96
2000 (153)	8	10	89	100	94
2001 (63)	8	8	89	100	83
2002 (60)	8	9	85	100	95
2003 (88)	9	10	90	100	94
2004 (96)	7	15	87	99	98
2005 (144)	11	14	95	100	97
2006 (110)	1	5	87	96	96
Average	**7**	**9**	**89**	**99**	**94**
Standard Deviation	**3**	**4**	**3**	**1**	**5**
Irrigation Streams (n = 585)					
1999 (59)	41	80	97	97	100
2000 (67)	60	69	93	93	98
2001 (62)	46	75	94	95	97
2002 (62)	34	56	94	97	95
2003 (79)	32	45	78	94	97
2004 (93)	31	61	91	98	91
2005 (78)	20	60	95	99	99
2006 (85)	18	57	87	92	85
Average	**35**	**63**	**91**	**95**	**95**
Standard Deviation	**14**	**11**	**6**	**3**	**5**

Table 3.5. Percent compliance for the Protection of Aquatic Life nutrient guidelines in watersheds from 1999 through 2006. The total number of samples collected for each stream from 1999 to 2006 is located in parentheses beside each stream.

		TP	TN	$N0_2$-N	$N0_3$-N	NH_3-N
Guideline		ASWQG (PAL)	ASWQG (PAL)	ASWQG (PAL)	ASWQG (PAL)	ASWQG (PAL)
Guideline Value		0.05 mg L^{-1}	1 mg L^{-1}	0.06 mg L^{-1}	2.9 mg L^{-1}	pH and Temp
	Abbreviation			% Compliance		
Low Agricultural Intensity						
Hines Creek (115)	**HIN**	6	23	100	100	99
Paddle River (156)	**PAD**	14	64	100	100	100
Prairie Blood Coulee (110)	**PRA**	78	51	97	99	99
Rose Creek (182)	**ROS**	51	68	100	100	100
Willow Creek (167)	**WIL**	92	98	100	100	100
Moderate Agricultural Intensity						
Blindman River (179)	**BLI**	6	30	100	100	100
Grande Prairie Creek (104)	**GRA**	16	13	99	96	99
Kleskun Drain (74)	**KLE**	5	6		100	100
Meadow Creek (152)	**MEA**	35	68	100	100	96
Tomahawk Creek (154)	**TOM**	1	2	98	100	100
Trout Creek (168)	**TRO**	82	94	100	100	99
High Agricultural Intensity						
Buffalo Creek (119)	**BUF**	6	30	97	100	85
Haynes Creek M1 (104)	**HM1**	0	2	87	91	93
Haynes Creek M6 (95)	**HM6**	0	0	92	97	89
Ray Creek (141)	**RAY**	4	9	96	100	98
Renwick Creek (108)	**REN**	1	0	90	98	100
Strawberry Creek (132)	**STR**	33	40	98	99	96
Stretton Creek (49)	**STT**	0	2	96	100	100
Threehills Creek (145)	**THR**	0	0	90	100	94
Wabash Creek (100)	**WAB**	2	3	85	100	94
Irrigation Streams						
Battersea Drain (152)	**BAT**	30	50	79	84	93
Crowfoot Creek (155)	**CRO**	5	49	89	99	98
Drain S-6 (134)	**DS6**	69	83	100	100	93
New West Coulee (144)	**NEW**	31	65	94	99	97

Flow Weighted Mean Concentrations (FWMC) of P and N

Phosphorus and N FWMCs, which allow for comparison of streams with different flow regimes, were compared among agricultural intensity categories and ecoregion areas to assess whether agricultural and/or ecoregional characteristics influenced nutrient concentrations. Flow weighted mean concentrations are mass normalized for flow and were thus used to visually identify potential changes in water quality with time. Relationships with agricultural intensity metrics were examined as the metrics can be used to link the type and intensity of agriculture in a watershed with changes in water quality. Lastly, seasonal patterns were examined to assess whether nutrient concentrations varied during different times of the monitoring period (i.e., spring, summer, and fall.

Median annual TP FWMCs from all watersheds in the study (1999 to 2006) ranged from 0.009 to1.300 $mg \cdot L^{-1}$ (1999-2006, median: 0.230 $mg \cdot L^{-1}$) and generally appeared lower than values reported in other agricultural watersheds in Alberta (0.363-1.119 $mg \cdot L^{-1}$) (Mitchell 1985; Sosiak and Trew 1986; Trew et al. 1987; Cooke and Prepas 1998). The median TP FWMCs (0.142, 0.275, and 0.510 $mg \cdot L^{-1}$ for low, moderate and high agricultural intensity streams, respectively) for the AESA streams in this study were similar to the median values reported for the low (0.190 $mg \cdot L^{-1}$), moderate (0.333 $mg \cdot L^{-1}$), and high (0.619 $mg \cdot L^{-1}$) agricultural intensity watersheds in the CAESA stream survey (Anderson et al. 1998b). However, the range of TP FWMCs for the CAESA stream survey (0.075-1.960 $mg \cdot L^{-1}$) was higher than the range for the AESA watersheds. Note that the values referenced for the CAESA stream survey were from two years of monitoring with a slightly different set of watersheds, while the AESA values represent eight years of monitoring. Several studies have found lower median TP FWMCs for forested watersheds in Alberta (0.160 $mg \cdot L^{-1}$ with a range of 0.026-0.330 $mg \cdot L^{-1}$ as reported in Anderson et al. 1998b and 0.123 to 0.198 $mg \cdot L^{-1} yr^{-1}$ as reported in Cooke and Prepas 1998).

Median annual TN FWMCs ranged from 0.120 to 8.600 $mg \cdot L^{-1}$ (1999-2006 median: 1.90 $mg \cdot L^{-1}$) and were generally similar to some of the reported values for Alberta. The median TN FWMCs for the low, moderate, and high agricultural intensity AESA watersheds in this study (1.310, 2.120, and 3.316 $mg \cdot L^{-1}$, respectively) were similar to the median FWMCs reported by Anderson et al. (1998b) for the CAESA stream survey (1.177, 2.128, and 3.773 $mg \cdot L^{-1}$ for the low, moderate, and high categories, respectively) as well as those reported for other agricultural streams in Alberta such as Pine Creek (1.594 $mg \cdot L^{-1}$, Sosiak and Trew 1986), Wabamun Creek (3.03 $mg \cdot L^{-1}$, Mitchell 1985), and Baptiste River (4.325 $mg \cdot L^{-1}$, Trew et al. 1987). However, the range of TN FWMCs for the AESA watersheds was larger than the range reported for the watersheds in the CAESA stream survey (0.853-6.685$mg \cdot L^{-1}$). The median TN FWMC for the AESA streams was higher than the median of Wabamun streams under forest (1.100 $mg \cdot L^{-1}$, Mitchell 1985) but lower than those reported by Trew et al. (1987) for Baptiste (2.300 $mg \cdot L^{-1}$).

Comparisons of P and N FWMCs by agricultural intensity. Nutrient FWMCs were compared among agricultural intensity categories to assess the impact of different agricultural intensities (low, moderate, and high) and type (dryland versus irrigation) on water quality (Objective 1 of the AESA stream survey). Phosphorus and nitrogen FWMCs grouped by agricultural intensity are illustrated in Figures 2.1 through 2.9.

Total and dissolved P FWMCs were influenced by the level of agricultural intensity within a watershed. Total P and TDP FWMCs increased with agricultural intensity in dryland watersheds, with significant differences observed among all (dryland and irrigated) agricultural intensity categories (H=57.754 and 72.950 for TP and TDP, respectively; $p<0.005$) (Figures 3.1 and 3.2, respectively). No statistically significant difference was found among agricultural intensity categories for TPP FWMCs (F=3.068, $p>0.05$) (Figure 3.3). Mean rank TP and TDP FWMCs were significantly higher in the high agricultural intensity category than in the low (U=290 and 108, respectively; $p<0.005$), moderate (U=577 and 496, respectively; $p<0.005$), or irrigated (U=1192 and 1168, respectively; $p<0.005$) watershed categories. Mean rank TP FWMCs in the moderate agricultural intensity category were significantly higher than both low (U=611, $p<0.005$) and irrigated (U=748, $p<0.05$) watershed categories. However, TDP FWMCs were significantly higher in the moderate agricultural intensity watersheds than in the low agricultural intensity watersheds (U=578, $p<0.005$) but similar to irrigated watersheds.

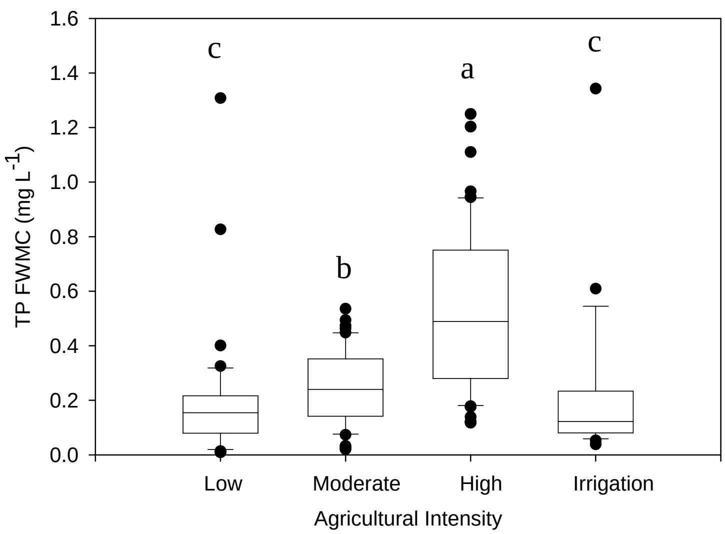


Figure 3.1. Box plots of median annual TP FWMCs (1999 to 2006) in the four agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Significant differences among agricultural intensity categories were observed at the $p<0.005$ significance level with the exception of differences between the moderate and low agricultural intensity watersheds ($p<0.05$). Boxplots stretch from the 25th percentile to the 75th percentile with the horizontal line in the middle of the box representing the median. Vertical lines represent 1.5 times the interquartile range while dots represent minima and maxima data points.

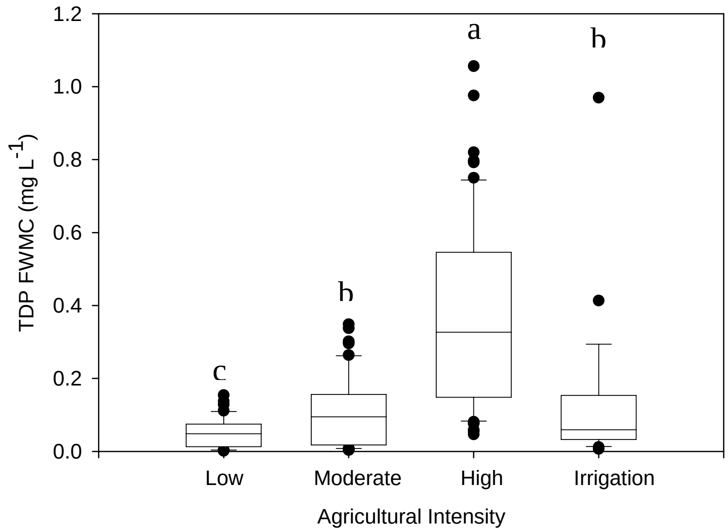


Figure 3.2. Box plots of median annual TDP FWMCs (1999 to 2006) in the four agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Significant differences among agricultural intensity categories were observed at the $p<0.005$ significance level with the exception of differences between the irrigated and low agricultural intensity watersheds ($p<0.05$).

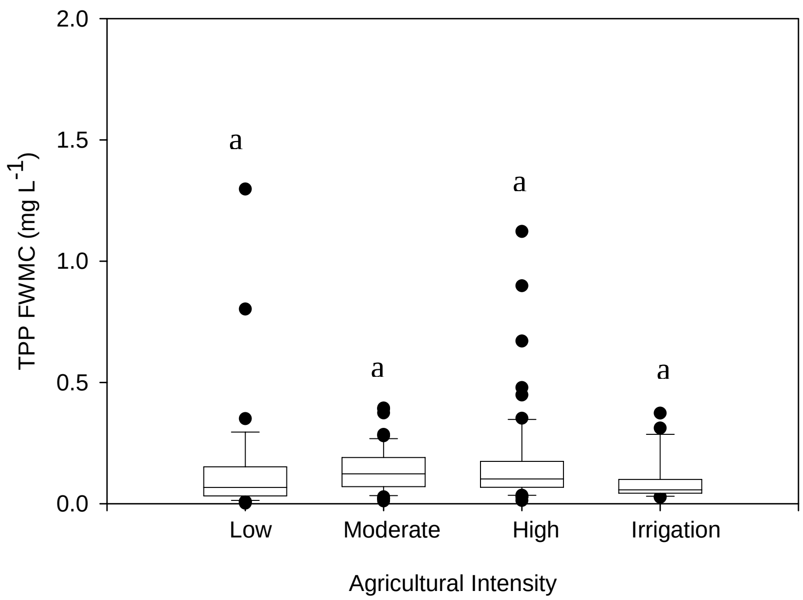


Figure 3.3. Box plots of median annual TPP FWMCs (1999 to 2006) in the four agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey’s post hoc after a Log10 transformation was applied to normalize the data.

As for TP and TDP FWMCs, the ratio of TDP/TP increased with increasing agricultural intensity for the dryland watersheds ($H=47.529$, $p<0.005$) although mean rank TDP/TP in moderate agricultural intensity watersheds was not significantly different ($p>0.05$) from low agricultural intensity streams (Figure 3.4). The mean rank TDP/TP ratio was significantly higher in high agricultural intensity watersheds than in low ($U=382$, $p<0.005$), moderate ($U=417$, $p<0.005$), or irrigated streams ($U=1079$, $p<0.005$). Moreover, irrigation return flow streams also had significantly higher mean rank TDP/TP than low agricultural intensity watersheds ($U=332$, $p<0.05$).

Despite having similar agricultural intensity ratings (i.e., high), there was a significant difference in P concentrations between dryland and irrigated streams. High intensity dryland streams had higher nutrient concentrations. A contributing factor to the lower P concentrations observed in irrigated watersheds may be that the source water for some irrigated streams contains lower nutrient concentrations, which may have a diluting effect (Madawaska Consulting 1997, Greenlee et al 2000, Saffran 2005, Little et al *in prep*).

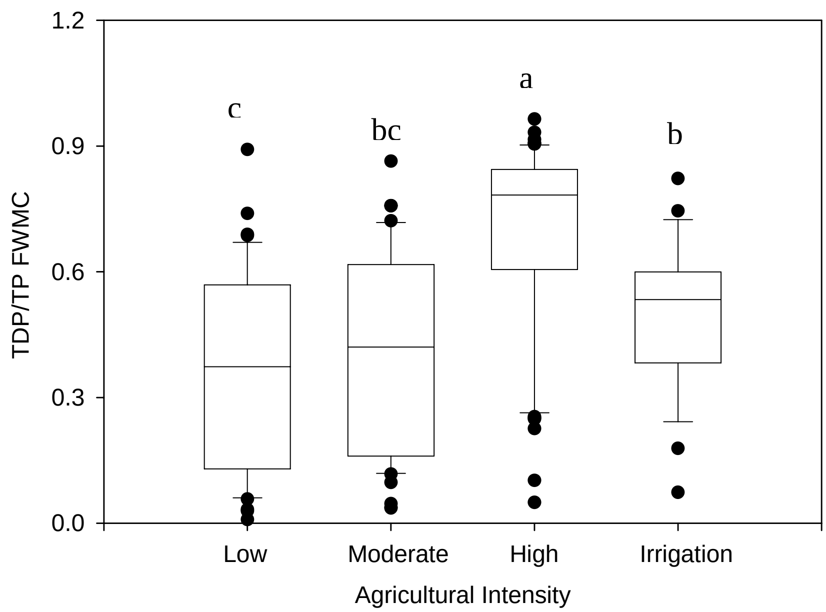


Figure 3.4. Box plots of median annual TDP/TP FWMC ratios (1999 to 2006) in the four agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal Wallis One-Way ANOVA and Mann-Whitney tests on untransformed data. Significant differences among agricultural intensity categories were observed at the $p<0.005$ significance level with the exception of differences between the irrigated and low agricultural intensity watersheds ($p<0.05$).

Flow weighted mean concentrations for all forms of N showed a stepwise increase with agricultural intensity for dryland watersheds, with significant differences observed among all (dryland and irrigated) agricultural intensity categories (TN ($F = 30.4$, $p<0.005$, 3df), Org N ($F = 36.474$, $p<0.005$, 3df), NO_2^--N +NO_3^--N ($F = 28.237$, $p<0.005$, 3df), and NH_3-N ($F = 31.691$, $p<0.005$, 3df)). Streams draining high agricultural intensity watersheds had significantly higher TN, Org N, NO_2^--N+NO_3^--N, and NH_3-N FWMCs than those under low, moderate, or irrigated agriculture ($p<0.005$ for all but NH_3-N in irrigated streams where $p<0.05$) (Figures 3.5-3.8). Moreover, streams with moderate intensity agriculture in the watershed had significantly higher TN and Org N FWMCs than those watersheds under low agricultural activity ($p<0.005$) and irrigation ($p<0.05$ and 0.005, respectively). Mean TN and Org N FWMCs between low and irrigated watersheds were not statistically different ($p=0.854$ and 0.947, respectively). Mean NH_3-N and NO_2^--N+NO_3^--N FWMCs were not significantly different between moderate intensity and irrigated watersheds ($p=0.908$ and 0.867, respectively). Also, NO_2^--N+NO_3^--N FWMCs for irrigated watersheds were not statistically different than those measured in the high agricultural intensity watersheds ($p=0.233$).

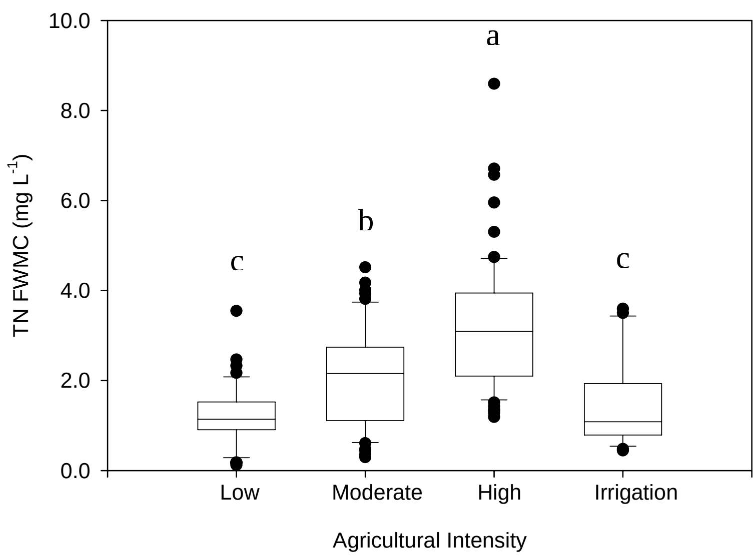


Figure 3.5. Box plots of median annual TN FWMCs (1999 to 2006) in the four agricultural intensity categories. Means of box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey’s post hoc after a Log(x+1) transformation was applied to normalize the data. Statistical differences were observed at the $p<0.005$ significance level with the exception of differences between mean TN FWMCs in the moderate and irrigated watersheds ($p<0.05$).

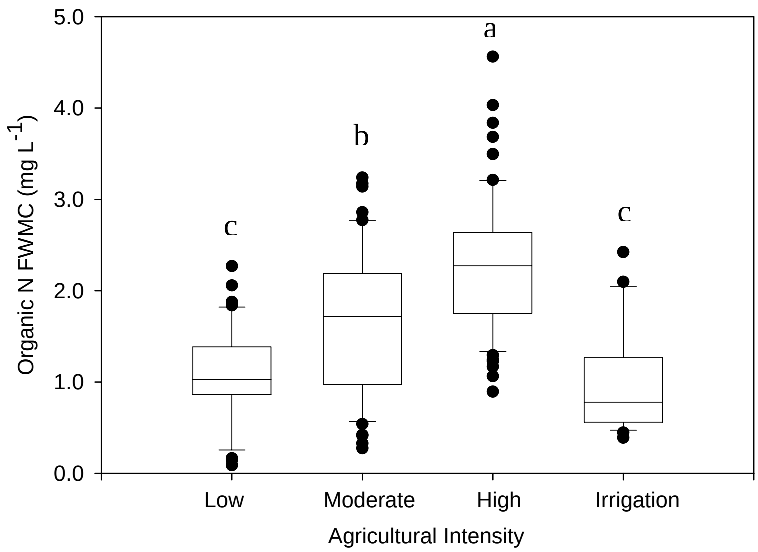


Figure 3.6. Box plots of median annual Org N FWMCs (1999 to 2006) in the four agricultural intensity categories. Means of box plots with the same letter are not significantly different from one another at the 0.005 level as tested with ANOVA and Tukey's post hoc after a square root transformation was applied to normalize the data.

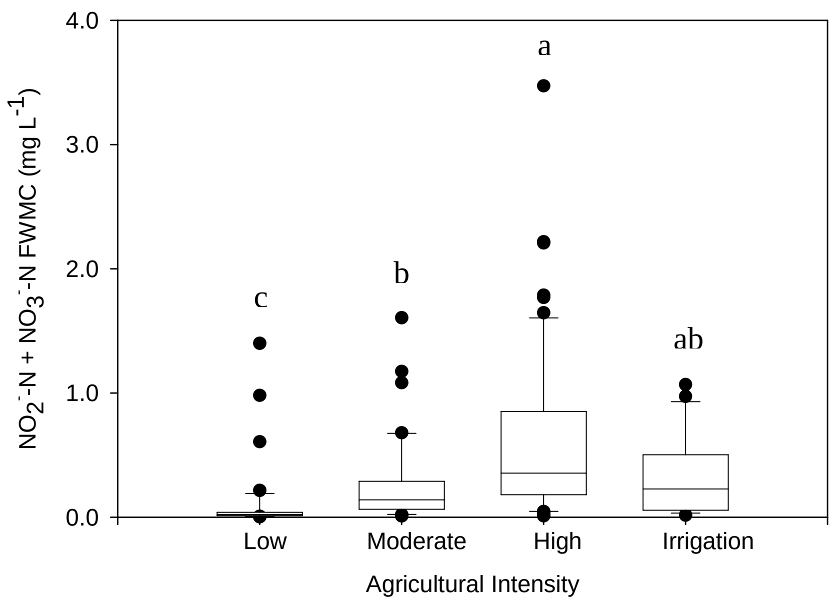


Figure 3.7. Box plots of median annual NO_2^--N+NO_3^--N FWMCs (1999 to 2006) in the four agricultural intensity categories. Means of box plots with the same letter are not significantly different from one another at the 0.005 level as tested with ANOVA and Tukey's post hoc after a Log10 transformation was applied to normalize the data.

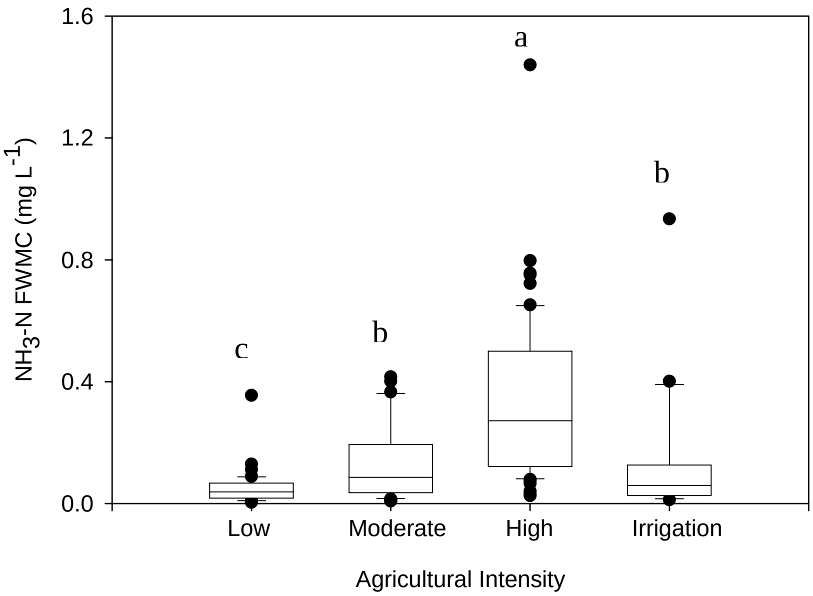


Figure 3.8. Box plots of median annual NH_3-N FWMCs (1999-2006) in the four agricultural intensity categories. Means of box plots with the same letter are not significantly different from one another at the 0.005 level as tested with ANOVA and Tukey's post hoc after a Log10 transformation was applied to normalize the data.

The relationship between DIN/TN and agricultural intensity was different from that of the N parameters and TDP/TP in that a statistically significant stepwise trend with agricultural intensity was found ($H=243$, $p<0.005$, 3df) with the highest DIN/TN ratios measured in the irrigated watersheds (Figure 3.9). All agricultural intensity categories were statistically different from one another at the $p<0.01$ significance level. Note that the absolute concentrations of N in the irrigated watersheds were similar to those observed in low and moderate streams; however, the relative amount of TN as DIN was similar to the amount measured in high agricultural intensity streams. It is uncertain whether the higher ratios of DIN/TN in irrigated watersheds was the result of naturally occurring N in the area (Rodvang et al. 2004) or was influenced by agricultural practices.

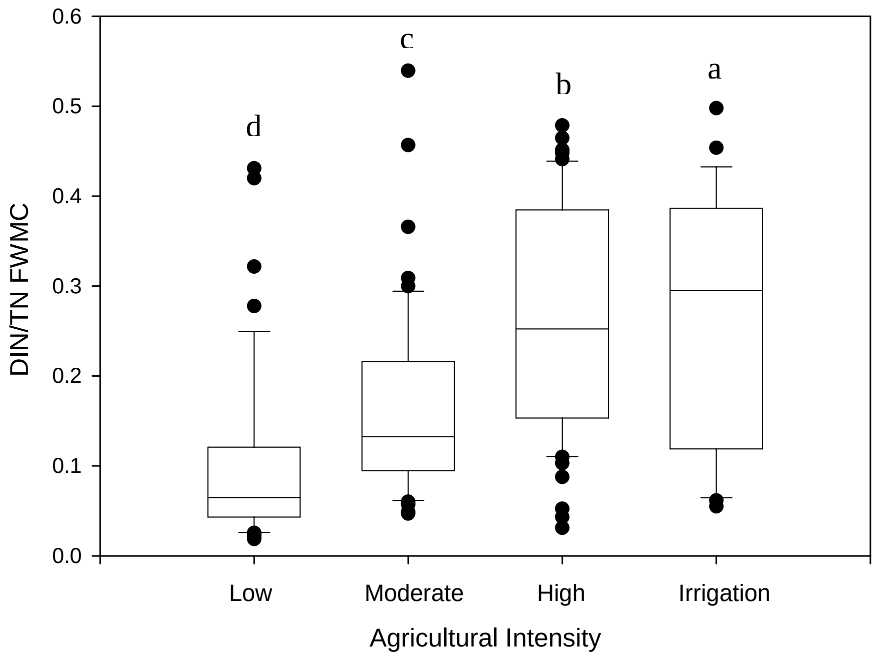


Figure 3.9. Box plots of median annual DIN/TN FWMC ratios (1999-2006) in the four agricultural intensity categories. Box plots with different letters are significantly different from one another at the 0.005 level as tested with Kruskal Wallis One-Way ANOVA and Mann-Whitney tests on untransformed data. Note that the significant differences between irrigated watersheds and moderate and agricultural intensity categories were observed at the $p<0.01$ significance level.

Nutrient FWMCs and ratios were also compared among streams and within each agricultural intensity category to assess whether watersheds under the same intensity level had similar nutrient concentrations or whether there were other factors to consider (Figures 3.10 and 3.11). The lowest and highest P and N FWMCs were found in streams draining watersheds under low and high agricultural intensity, respectively. Both TDP/TP and DIN/TN varied by one order of magnitude among streams with the lowest (Rose and Willow Creeks, low agricultural intensity) and highest ratios (Haynes Creek, high agricultural intensity). A higher ratio of TP was found to be comprised of the dissolved fraction (TDP/TP ranged from 0.08 in Willow Creek to 0.892 in Haynes Creek) than was observed for TN (DIN/TN ranged from 0.046 in Rose Creek to 0.399 in Haynes Creek).

Within the low agricultural intensity watersheds, Rose Creek generally had the highest P and N FWMCs, while Willow Creek had the lowest TP and TN FWMCs, which were lower than all other streams monitored in the study (Table 3.6). Although Rose Creek had some of the highest nutrient FWMCs and Willow Creek had some of the lowest N and P FWMCs, both watersheds had very low ratios of TDP/TP. Furthermore, DIN/TN was lowest in Rose and Hines Creeks. Rose Creek and Paddle River are both located in the Western Alberta Uplands Ecoregion; however, Paddle River had a higher TDP/TP ratio than Rose Creek, indicating a higher ratio of bioavailable P despite similar landscapes and climate. Land cover data shows that there is a higher ratio of grassland in the Paddle River watershed (Appendix 1 or Chapter 2, Land Cover). It was not surprising that the TDP/TP ratio in Willow Creek was low. According to 1991 orthophotos, the Willow Creek watershed was covered by 9% grassland and 33% tree cover (Appendix 1 or Chapter 2, Land Cover). As mentioned previously, the sampling site for Willow Creek was located in the headwaters and was not the same as AENV's long term river network site at the watershed outlet (see Chapter 2: Materials and Methods – Stream Sampling Methods). Over 58% of the watershed was not mapped for land cover as it extends beyond Alberta's White (agricultural) Zone. Thus, it appears there is very little agricultural influence around the mouth of the watershed where sampling occurred. Prairie Blood Coulee had the highest N and P ratios of the low agricultural intensity watersheds; TDP/TP was also higher in Prairie Blood Coulee than in many of the moderate agricultural intensity watersheds. Noteworthy, TDP/TP in Hines Creek was higher than what was expected based on the DIN/TN ratio. It was the second highest median annual ratio of the low agricultural intensity streams (Table 3.6). The higher TDP/TP ratio in Hines Creek may be a reflection of ecoregional differences as flow patterns more closely matched those observed in Grande Prairie Creek, a moderate agricultural intensity watershed also located in the Peace Lowland Ecoreigon.

Nutrient FWMCs and ratios measured in the moderate agricultural intensity watersheds were influenced by the watershed's location in the province: they were lowest in the Grassland ecoregion area and highest in the Boreal ecoregion area (Table 3.6). The lowest TDP/TP and DIN/TN ratios within the moderate agricultural intensity watersheds were found in the two Grassland streams – Trout and Meadow Creeks. Trout Creek also had the lowest nutrient FWMCs. Kleskun Drain, a watershed in the Boreal ecoregion area, had the highest total and dissolved FWMCs, with higher TDP/TP than some of the watersheds under high agricultural intensity (e.g., Buffalo, Strawberry, and Wabash Creeks). Tomahawk Creek, also located in the Boreal, generally had the highest N FWMCs and highest ratio of DIN/TN.

Of the high agricultural intensity watersheds, all but TPP FWMCs were highest in Haynes Creek; these medians were also the highest FWMCs of all watersheds in the program (Table 3.6). Strawberry Creek had the highest median annual TPP FWMC. Buffalo Creek generally had the lowest median annual P and N FWMCs of the high agricultural intensity watersheds, which were lower than those FWMCs observed in some of the moderate and low agricultural intensity watersheds (Table 3.6).

Ratios of dissolved to total N and P in Haynes Creek were the highest of the high agricultural intensity watersheds and of all streams in the study (Table 3.6). Ray Creek had the lowest DIN/TN ratio. Strawberry Creek, like Rose Creek, had a low TDP/TP ratio. This contrasts with all of the other streams under high agricultural intensity with the exception of Wabash Creek, which, like Strawberry, is also located in the Boreal ecoregion area. Furthermore, the median TDP/TP in Strawberry Creek (0.25) was significantly lower than all other high agricultural intensity watersheds ($p<0.008$). Higher TDP ratios did not appear to correspond to lower annual flow volume even though increased dissolved fractions were measured in years with lower/fewer peaks in spring discharge. Climatic characteristics and landscape features likely influenced the higher particulate fractions of P observed in the watershed and the resulting TDP/TP ratios.

Although Strawberry Creek drains high agricultural intensity land, a focus on reduction of the particulate fraction would be most beneficial in this watershed.

Of the three irrigated watersheds, Crowfoot Creek had the highest median annual P FWMCs, while New West Coulee had the lowest median annual TN and TP FWMCs (Table 3.6). As noted earlier, TDP/TP in the irrigated streams was similar to those measured in the moderate intensity dryland watersheds. Ratios were highest in Crowfoot Creek and New West Coulee and slightly lower in Battersea Drain. In contrast, DIN/TN was slightly higher in Battersea Drain and lower in Crowfoot Creek and New West Coulee.

Table 3.6. Medians of the median annual nutrient FWMCs and ratios for 22 AESA watersheds by agricultural intensity category (1999 to 2006).

Agricultural Intensity	AESA Watershed	Total Phosphorus	Total Dissolved Phosphorus	Total Particulate Phosphorus	Total Nitrogen	Nitrite-N + Nitrate-N	Ammonia-N	Organic-N	TDP:TP	DIN:TN
Low	Hines Creek	0.142	0.059	0.063	1.310	*0.011**	0.054	1.236	0.500	0.054
	Paddle River	0.201	**0.093**	0.095	1.347	0.052	**0.068**	1.145	0.456	0.100
	Prairie Blood Coulee	0.087	0.054	*0.026**	1.110	**0.105**	0.028	1.000	0.611	0.133
	Rose Creek	**0.268**	0.028	**0.248**	**1.411**	0.016	0.055	**1.350**	0.116	*0.046**
	Willow Creek	*0.043**	*0.004**	0.039	*0.283**	0.020	*0.010**	*0.256**	*0.080**	0.093
Moderate	Blindman River	0.297	0.142	0.150	1.973	0.130	0.224	1.732	0.590	0.171
	Grande Prairie Creek	0.253	0.092	0.127	2.268	**0.308**	0.063	1.863	0.528	0.171
	Kleskun Drain	**0.362**	**0.237**	0.101	2.741	0.265	0.115	2.287	**0.715**	0.157
	Meadow Creek	0.140	0.018	0.105	1.108	0.071	0.030	0.973	*0.129*	*0.099*
	Tomahawk Creek	0.356	0.120	**0.204**	**2.916**	0.289	**0.230**	**2.387**	0.402	**0.191**
	Trout Creek	*0.057*	*0.008*	*0.050*	*0.538*	*0.042*	*0.014*	*0.479*	*0.135*	*0.101*
High	Buffalo Creek	*0.157*	*0.089*	*0.048*	*1.982*	*0.048*	0.255	*1.630*	0.639	0.184
	Haynes Creek (M6)	**0.880^**	**0.808^**	0.089	**4.321^**	**0.788**	**0.518^**	**3.098^**	**0.892^**	**0.399^**
	Ray Creek	0.245	0.214	*0.042*	*1.995*	0.201	*0.094*	1.760	0.808	0.128
	Renwick Creek	0.787	0.692	0.098	3.453	0.626	0.257	2.543	0.872	0.290
	Stretton Creek	0.433	0.362	0.071	2.969	**0.952^**	0.145	1.986	0.836	0.387
	Strawberry Creek	0.692	0.123	**0.463^**	3.296	0.321	0.313	2.516	*0.250*	*0.284*
	Threehills Creek	0.550	0.439	0.104	3.571	0.580	0.494	2.461	0.807	0.304
	Wabash Creek	0.470	0.223	0.189	3.336	0.646	0.464	2.095	0.501	0.348
Irrigation	Battersea Drain	0.105	*0.024*	*0.049*	1.062	0.272	0.066	0.680	*0.338*	**0.355**
	Crowfoot Creek	**0.234**	**0.154**	**0.094**	**1.814**	**0.446**	**0.127**	**1.265**	**0.551**	0.273
	New West Coulee	*0.098*	0.044	0.052	*0.724*	*0.057*	*0.021*	*0.573*	0.536	*0.135*

Note: The highest and lowest values within an agricultural intensity category are **bolded** and *italicized*, respectively.

^ denotes the highest value of all watersheds from 1999 to 2006.

* denotes the lowest value of all watersheds from 1999 to 2006.

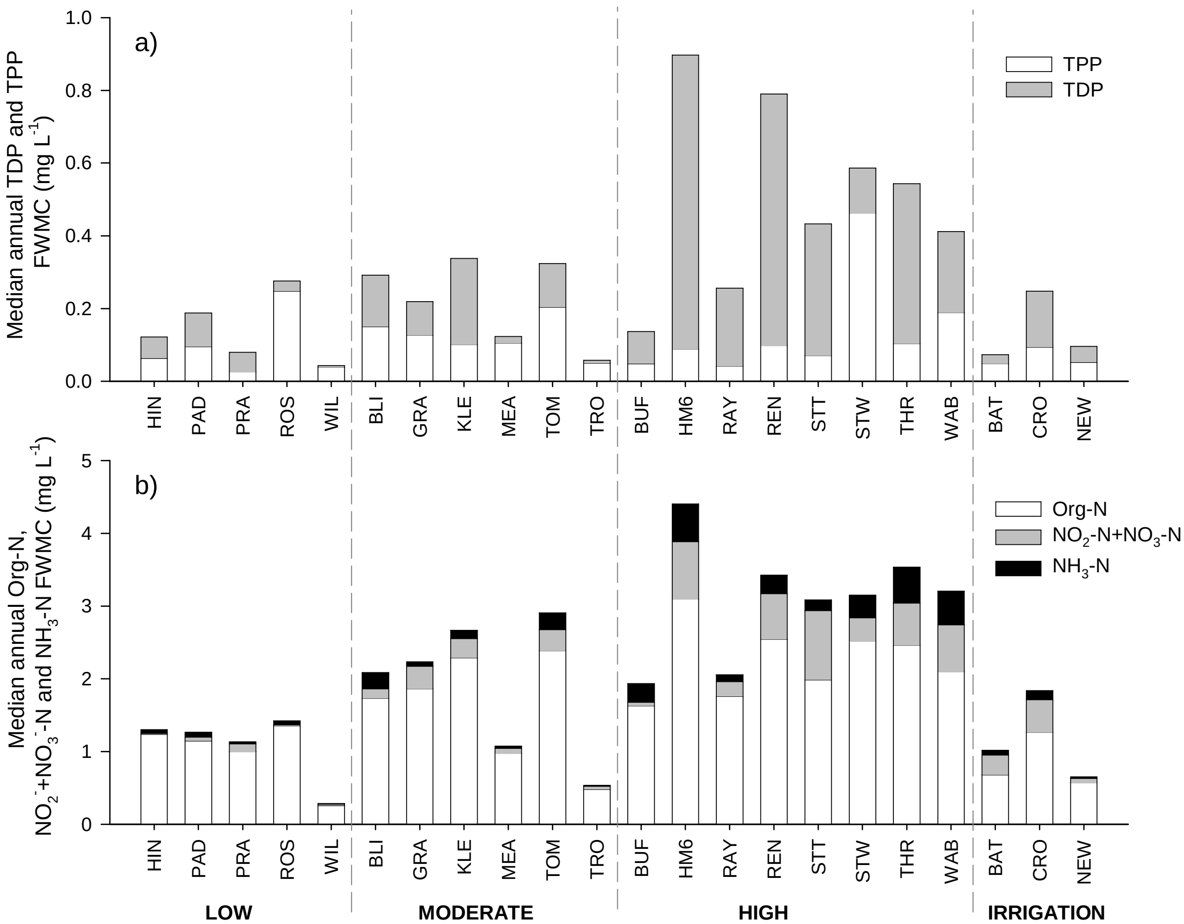


Figure 3.10. Median of the median annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N and NH_3-N (b) FWMCs for the AESA watersheds (1999 to 2006). Refer to Table 3.5 for full stream names.

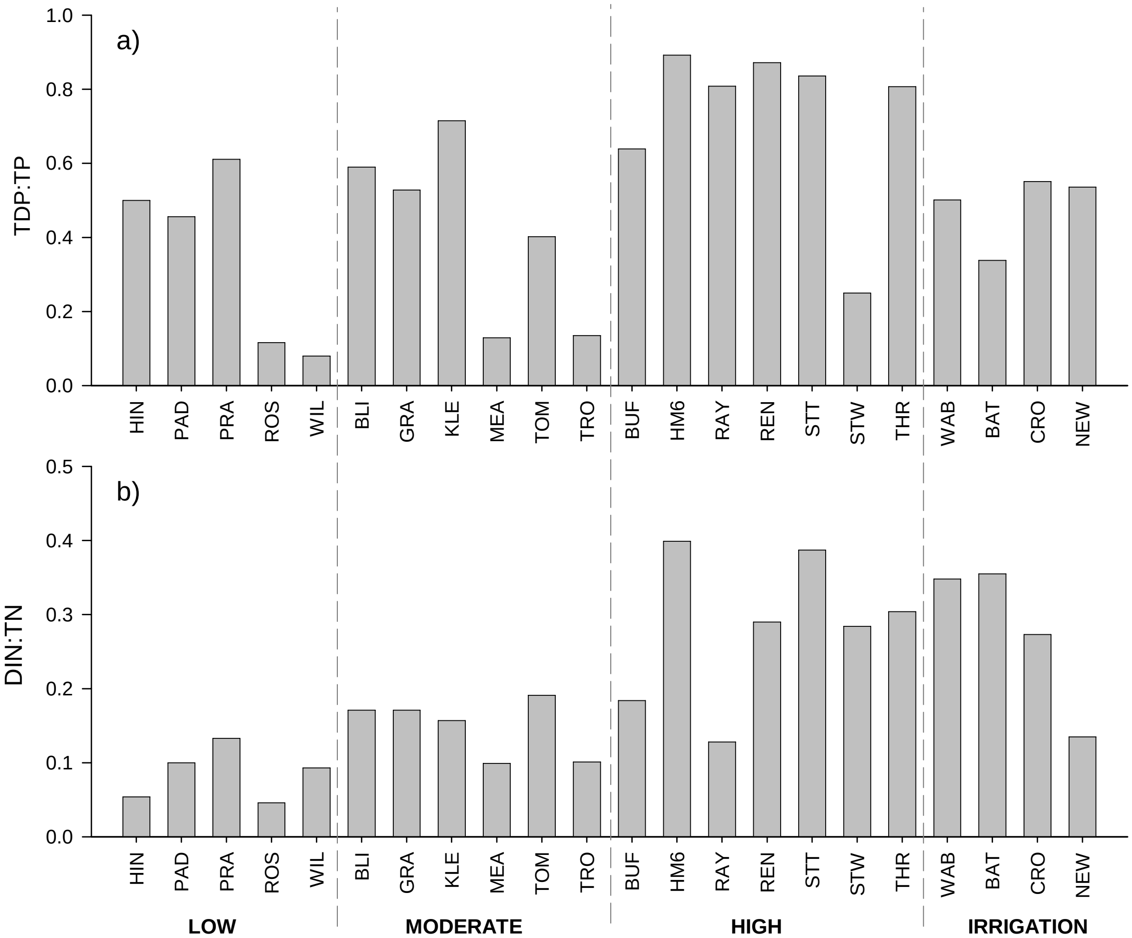


Figure 3.11. Median of the median annual TDP/TP (a) and DIN/TN (b) in the AESA watersheds grouped by the 1996 agricultural intensity rankings. Note that the ratios were obtained from FWMCs. Refer to Table 3.5 for full stream names.

Comparisons of P and N FWMCs by ecoregion area. Although the agricultural intensity in a watershed is not completely independent of where a watershed is located in the Province, the influence of ecoregional characteristics were examined by comparing nutrient FWMCs in watersheds located in similar ecoregion areas. Box plots of P and N FWMCs grouped by ecoregion area are found in Figures 3.12 through 3.20.

Significant differences in TP FWMCs were observed among ecoregion areas (H=54.092, p<0.005) with the highest TP FWMCs measured in the Parkland ecoregion area (p<0.005) (Figure 3.12). Mean rank TP FWMCs were significantly higher in the Boreal than Grassland (U=1496, p<0.005) and Irrigated Grassland (U=1277, p<0.005) ecoregion areas. Mean rank TP FWMCs were not significantly different between the Grassland and Irrigated Grassland ecoregion areas (U=221, p=0.167). Although TP FWMCs measured in the Continental Divide only represented one stream and were not included in the statistical analyses, concentrations appeared lower than those measured in the Grassland and Irrigated Grassland ecoregion areas. Interestingly, watersheds situated in ecoregion areas located in the southern part of the Province (Grassland, Irrigated Grassland, and Cont. Div.) tended to have lower TP FWMCs.

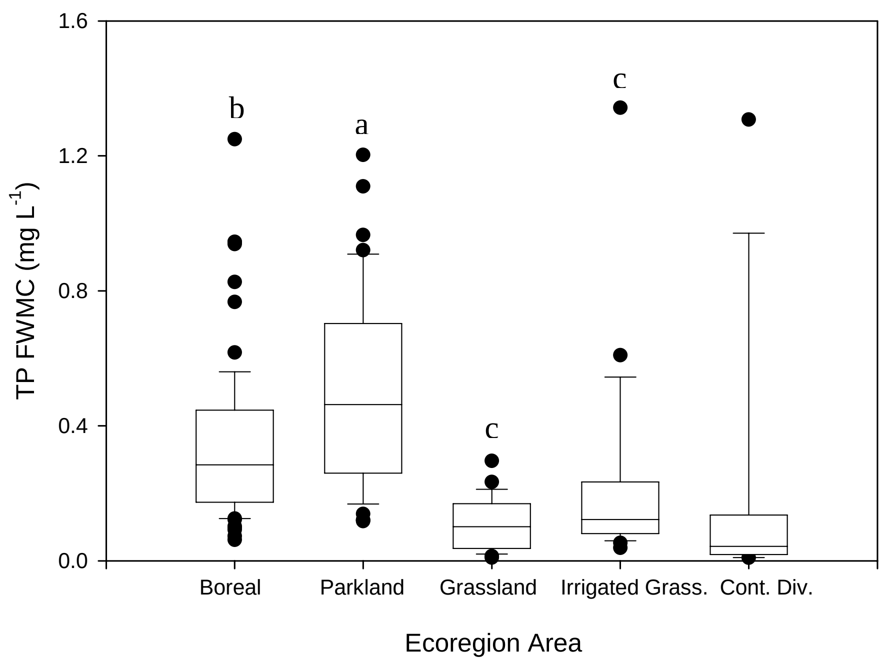


Figure 3.12. Box plots of median annual TP FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Significant differences between agricultural intensity categories were observed at the p<0.005 significance level.

As for TP, mean rank TDP FWMCs were significantly different among all ecoregion areas (H=77.842, p<0.005) with the highest TDP FWMCs measured in the Parkland ecoregion (p<0.005) (Figure 3.13). Furthermore, mean rank TDP FWMCs in the Boreal ecoregion area were also significantly higher than the Irrigated Grassland (U=1102, p<0.05) and Grassland (U=1522, p<0.005) ecoregion areas. However, the lowest TDP FWMCs were measured in the Grassland and Continental Divide ecoregion areas with significantly higher TDP FWMCs measured in the Irrigated Grassland watersheds.

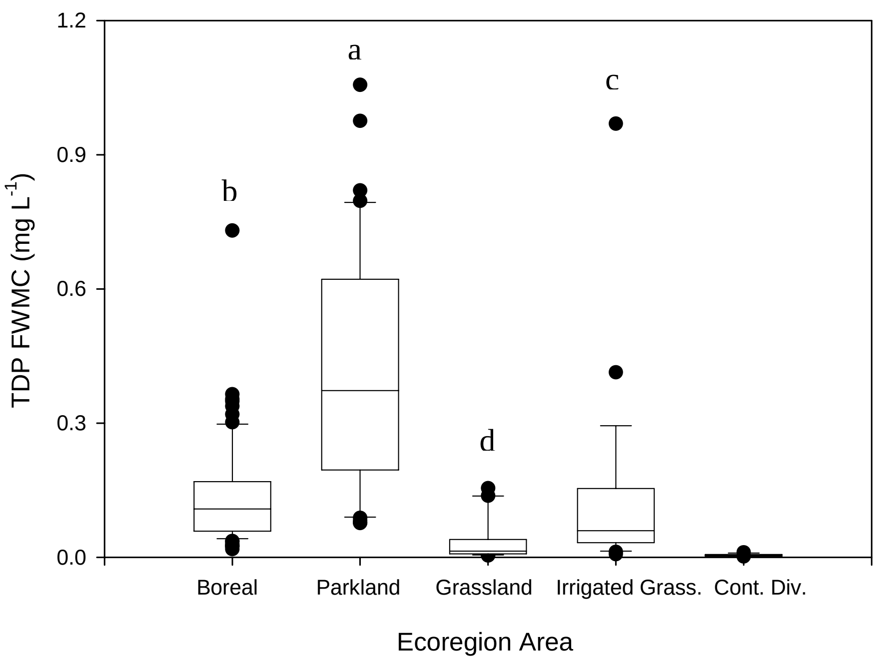


Figure 3.13. Box plots of median annual TDP FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with different letters were significantly different from one another at the 0.005 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Significant differences between the Boreal and Irrigated Grasslands ecoregion areas were observed at the p<0.05 significance level.

In contrast to TP and TDP, the highest TPP FWMCs were observed in the Boreal ecoregion area (F=13.538, p<0.005), while a significant difference in TPP FWMCs was not observed among the other ecoregion areas (p>0.05) (Figure 3.14). Total particulate P FWMCs in the Continental Divide were similar to those observed in the Parkland, Grassland, and Irrigated Grassland ecoregion areas. Higher TPP in the Boreal may be attributable to topography (i.e., steeper slopes), climate, hydrology, and possibly forest soils and sediment substrate in streams.

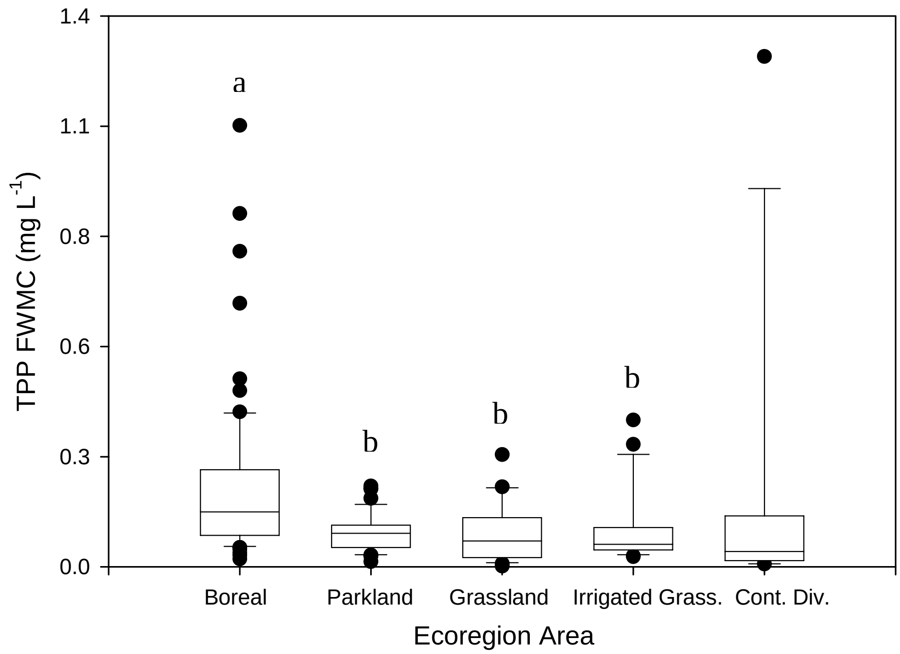


Figure 3.14. Box plots of median annual TPP FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey's post hoc after a Log10 transformation was applied to normalize the data. The mean TPP FWMC for the Boreal ecoregion area was significantly different from all other ecoregions at the $p<0.005$ level.

The ratio of TDP/TP by ecoregion area showed a similar pattern to TP and TDP FWMCs (Figure 3.15). Again, the Parkland ecoregion had the highest ratio of TDP/TP ($p<0.005$); however, a significant difference was not found between mean rank TDP/TP values in the Boreal and Irrigated Grassland ecoregion areas ($U=759$, $p>0.05$). Unlike TP, the ratio of TDP/TP was significantly lower in the Grassland ecoregion area than in the Boreal ($U=1131$, $p<0.005$) and Irrigated Grassland ($U=171$, $P<0.05$) ecoregion areas. Ratios in the Continental Divide were similar to those measured in the Grassland watersheds. The interquartile range appeared to be quite large for the TDP/TP ratio within the ecoregion areas, specifically the Boreal and Grassland ecoregion areas. However, there appeared to be less variability within the Parkland ecoregion area than within the high agricultural intensity category. The Parkland ecoregion contains all of the high agricultural intensity watersheds except Strawberry and Wabash Creeks. These two watersheds, located in the Boreal, may have higher ratios of TPP as a result of increased runoff. Furthermore, watersheds in the Parkland ecoregion area do not typically flow in the summer and may be less likely to respond to summer storm events that could influence increased particulate fractions in the Boreal and Grassland ecoregion areas. Variability within the Boreal and Grassland watersheds, as well as within those under low and moderate agricultural intensity, indicates that both agricultural intensity and factors such as runoff influenced the ratios of dissolved or particulate P measured in a stream.

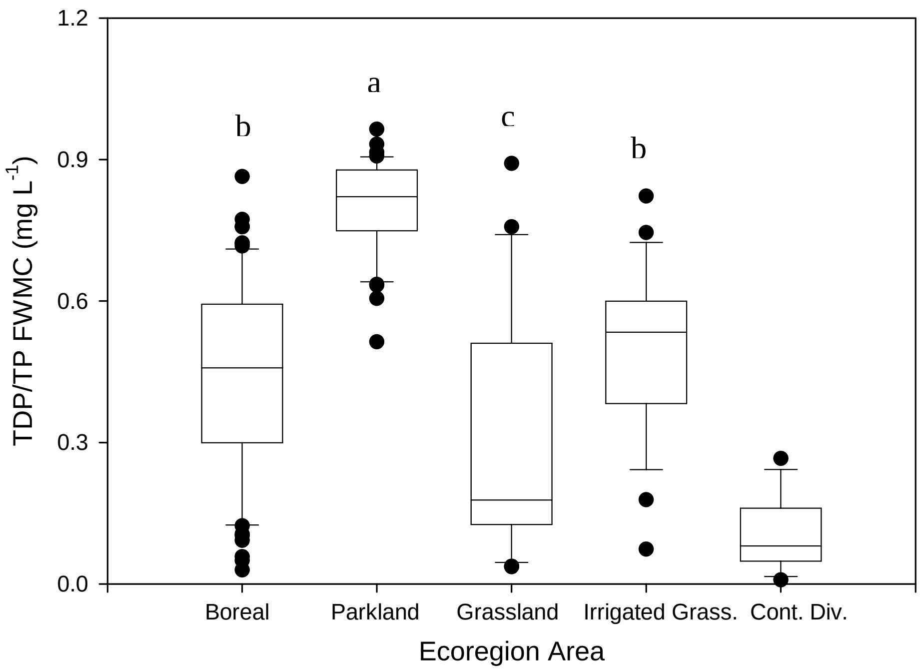


Figure 3.15. Box plots of median annual TDP/TP FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots with different letters were significantly different at the $p<0.005$ level except between the Grassland and Irrigated Grassland ecoregion areas ($p<0.05$).

Significant differences were observed among ecoregion areas for TN ($F=27.103$, $p<0.005$) and Org N FWMCs ($F=37.910$, $p<0.005$), which showed the same pattern among ecoregion areas as TP (Figures 3.16 and 3.17). Total N and Org N FWMCs were significantly higher in the Parkland ecoregion than in the Boreal, Grassland, and Irrigated Grassland ecoregion areas ($p<0.005$ or 0.05). Total and organic N FWMCs were also significantly higher in the Boreal than Grassland and Irrigated Grassland ecoregion areas ($p<0.005$). A significant difference was not found between the Grassland and Irrigated Grassland ecoregion areas for TN and Org N ($p=0.616$ and 0.984, respectively). Although FWMCs measured in the Continental Divide only represented one stream and were not included in the statistical analyses, TN and Org N concentrations appeared to be lower than those measured in the Grassland and Irrigated Grassland ecoregion areas. As for TP, TN appeared to be higher in watersheds located in the more northern ecoregion areas (Boreal and Parkland), which may be a reflection of the soils, hydrology, topography, and land cover/use.

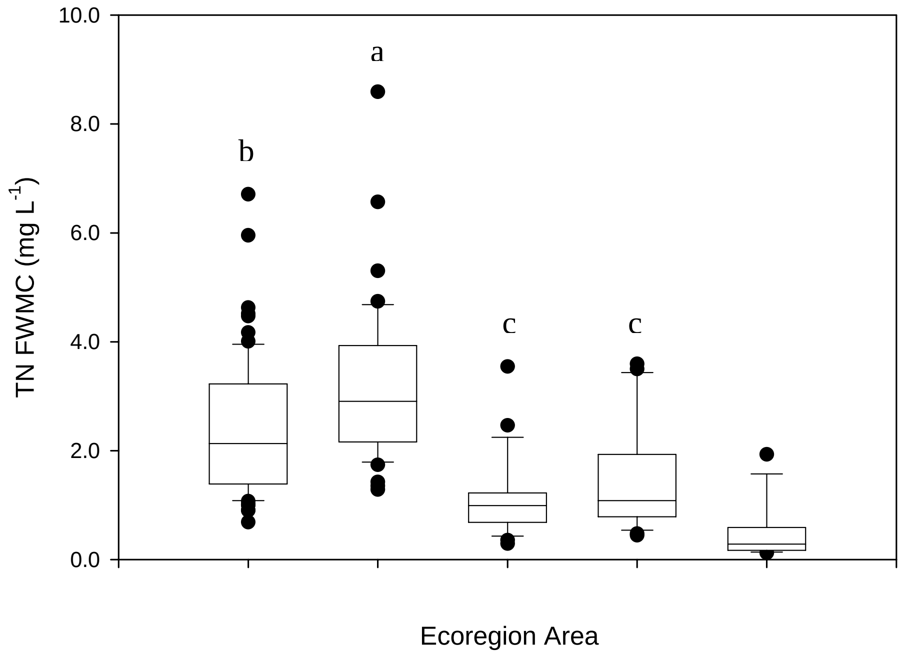


Figure 3.16. Box plots of median annual TN FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey's post hoc after a Log(x+1) transformation was applied to normalize the data. Box plots with different letters were significantly different at the p<0.005 level.

The highest NO_2^-+NO_3^--N FWMCs were measured in the Parkland ecoregion but were only significantly higher than the Boreal and Irrigated Grassland ecoregion areas ($p<0.005$) (Figure 3.18). Furthermore, a significant difference was not found between each of the other ecoregion areas (Boreal, Grassland, or Irrigated Grassland) ($p>0.1$). Nitrite N +Nitrate-N concentrations were lowest in the Continental Divide.

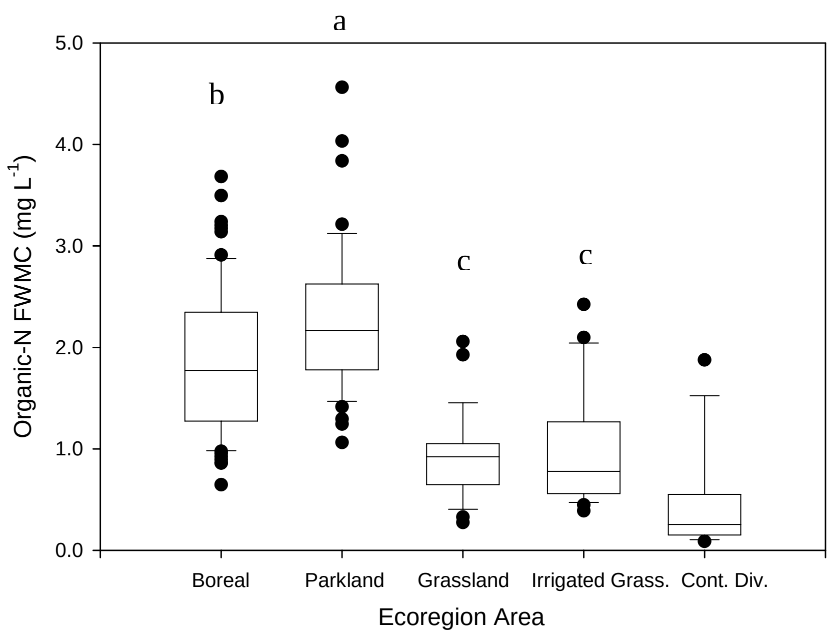


Figure 3.17. Box plots of median annual Org N FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey's post hoc after a square root transformation was applied to normalize the data. Box plots with different letters were significantly different at the $p<0.005$ level.

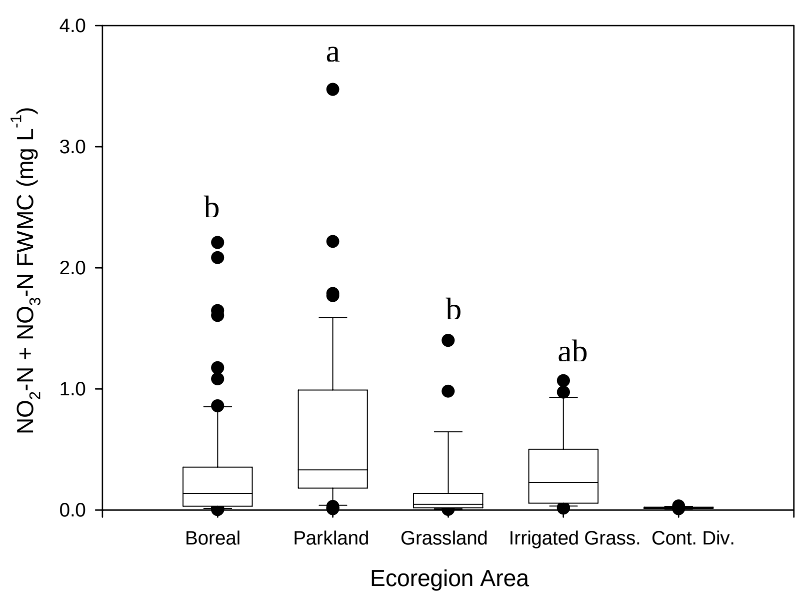


Figure 3.18. Box plots of median annual NO_2^--N+NO_3^--N FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey's post hoc after a Log10 transformation was applied to normalize the data. Box plots with different letters were significantly different at the $p<0.005$ level.

Ammonia-N FWMCs showed a similar pattern by ecoregion area as TDP/TP (Figure 3.19). The Parkland ecoregion had significantly higher NH_3-N FWMCs than the Boreal ($p<0.05$) or Grassland and Irrigated Grassland ecoregion areas ($p<0.005$). As for TDP/TP, a significant difference was not found between NH_3-N values in the Boreal and Irrigated Grassland ecoregion areas ($p=0.546$). Furthermore, NH_3-N was significantly lower in the Grassland than in the Boreal ($p<0.005$) and Irrigated Grassland ($p<0.01$) ecoregion areas. The Continental Divide appeared to have the lowest NH_3-N FWMCs.

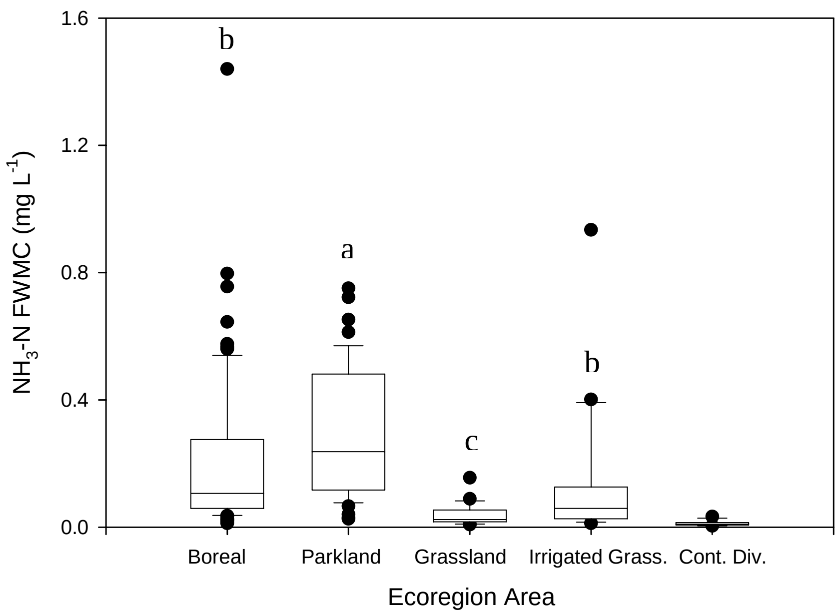


Figure 3.19. Box plots of median annual NH_3-N FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey's post hoc after a Log10 transformation was applied to normalize the data. Box plots with different letters were significantly different at the $p<0.005$ level with the exception of means between the Boreal and Parkland and between the Grassland and Irrigated Grassland ecoregion areas ($p<0.01$).

The mean rank DIN/TN ratio was significantly different among all ecoregion areas ($H=21.759$, $p<0.005$); however, DIN/TN ratios were not statistically different between the Parkland and Irrigated Grassland ecoregion areas ($U=534$, $p=0.714$) (Figure 3.20). Mean rank ratios in the Boreal were significantly higher than in the Grassland ecoregion area ($U=1056$, $p<0.005$). Ratios of DIN/TN in the Continental Divide were similar to DIN/TN in the Grassland ecoregion area. Interestingly, TPP, NO_2^--N+NO_3^--N, and DIN/TN were the only parameters where FWMCs were not significantly higher in the Parkland ecoregion than all other ecoregion areas. Concentrations of TPP, NO_2^--N+NO_3^--N, and DIN/TN in the Irrigated Grassland ecoregion were not significantly different from measurements in the Parkland ecoregion. Again, the ratio of DIN/TN was higher in the irrigated watersheds and close to those values of the high agricultural intensity, dryland watersheds – the opposite pattern observed for the absolute fractions of N. The split between northern and southern ecoregion areas was not observed for DIN/TN.

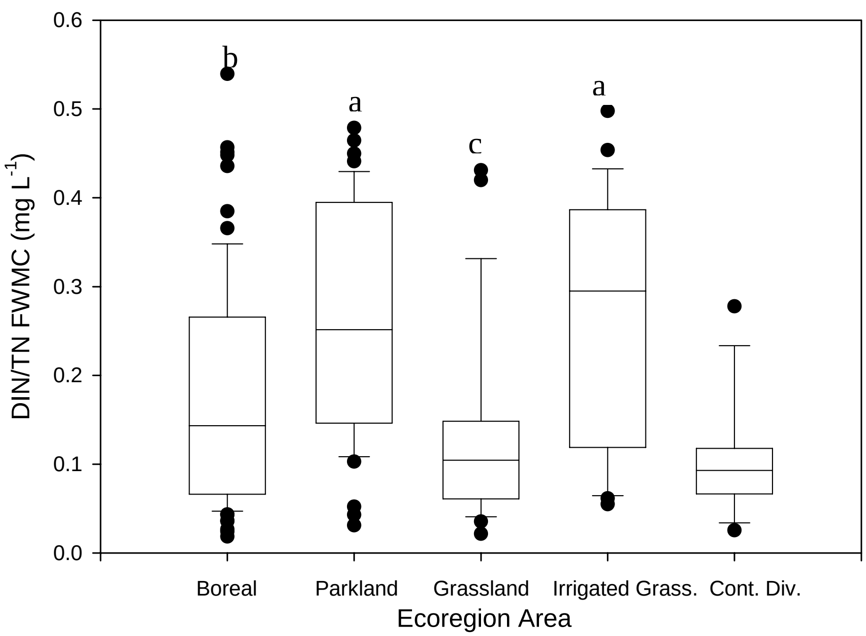


Figure 3.20. Box plots of median annual DIN/TN FWMCs (1999 to 2006) for the five ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots with different letters were significantly different at the $p<0.005$ level.

Molar ratios of TN/TP and DIN/TDP. The molar ratios of TN/TP and DIN/TDP were examined overall for the AESA watersheds, by agricultural intensity, and by ecoregion area to assess potential nutrient limitation. Eutrophication of surface waters has been attributed to increases in dissolved N and P and the ratio of N to P, in addition to other factors such as climate (Isermann 1990). Nitrogen to phosphorus ratios can be used to understand nutrient limitations in a stream. This information can then be used to help target the most effective beneficial management practices to decrease contributions of the limiting nutrient and over-enrichment of the waterbody.

The ratios of TN/TP for the AESA watersheds ranged from 1.475 to 114.338; however, the median ratio was relatively low at 7.421 (Table 3.7). The DIN/TDP ratio also spanned a wide range, but the median was relatively low at 10.636 (Table 3.7).

Table 3.7. Summary statistics for annual TN/TP and DIN/TDP ratios for all AESA watersheds (1999-2006).

Statistic	TN/TP	DIN/TDP
N	171	170
Minimum	1.48	0.410
Maximum	114	1940
Median	7.42	10.6
Mean	9.30	52.5

The TN/TP ratio differed among agricultural intensity categories (Table 3.8); watersheds draining high agricultural land had statistically lower ($p<0.05$) TN/TP ratios than low or moderate agricultural intensity watersheds suggesting a relative abundance of P (Figure 3.21a). The TN/TP ratio for high agricultural intensity watersheds did not differ from irrigated streams. A statistical difference was not found between low, moderate, or irrigated watersheds.

In contrast, irrigated watersheds had DIN/TDP ratios that were significantly lower than both moderate and high agricultural intensity watersheds ($p<0.01$) (Figure 3.21b). The ratios were also higher in high agricultural intensity watersheds than in the low agricultural intensity streams ($p<0.005$), with similar DIN/TDP ratios between moderate agricultural intensity streams and those in high or low agricultural intensity categories. The median DIN/TDP ratios for the low and irrigated watersheds indicated that N was potentially limiting with ratios <10 (Shanz and Juon 1983).

Table 3.8. Median and Average TN/TP and DIN/TDP ratios for agricultural intensity categories (1999-2006).

Agricultural Intensity	Median DIN/TDP	Ave. DIN/TDP	Median TN/TP	Ave. TN/TP
Low	3.91	52.6	8.19	12.7
Moderate	13.6	33.8	8.77	9.40
High	20.1	86.1	7.12	3.15
Irrigated	3.49	8.23	7.36	8.76

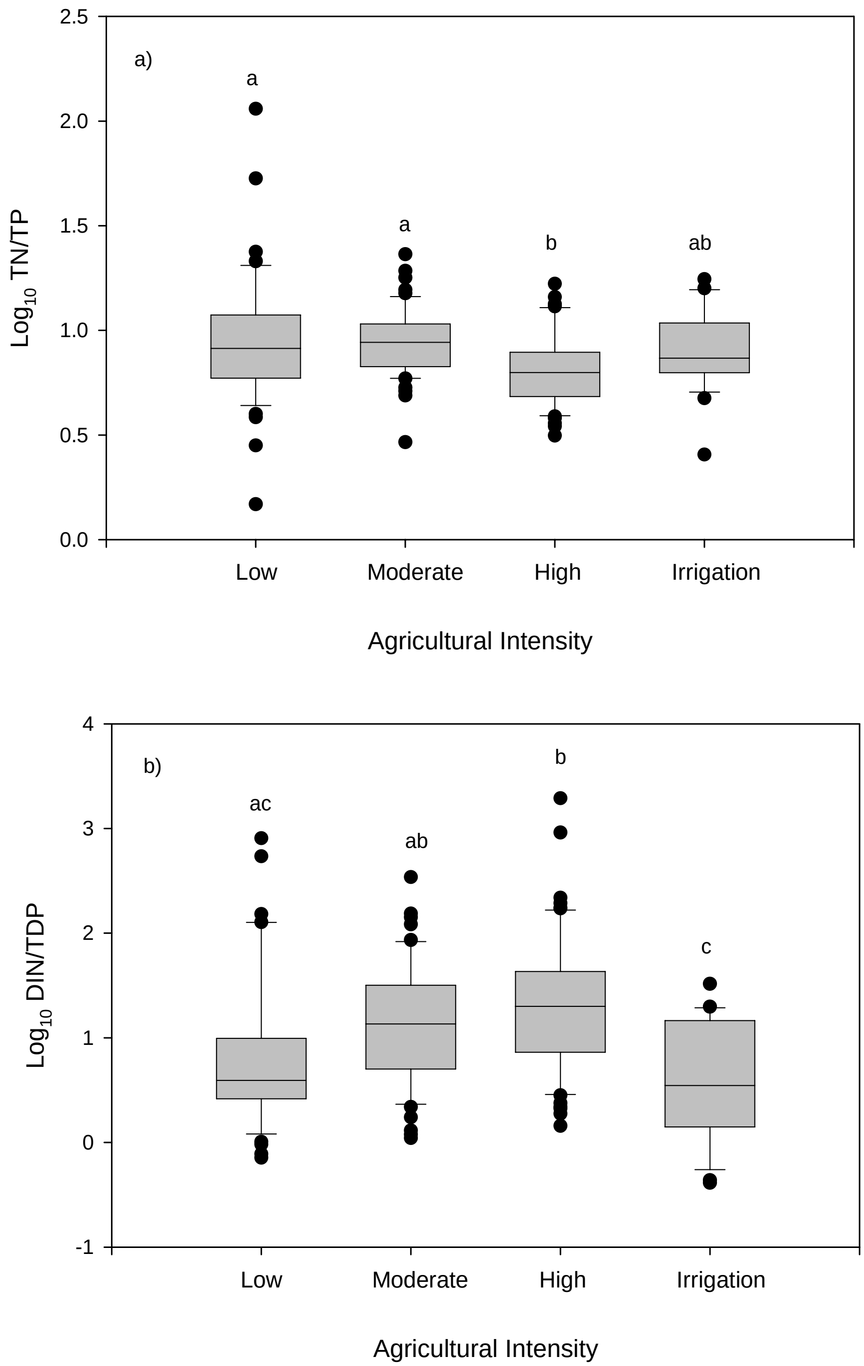


Figure 3.21. Box plots of median annual TN/TP (a) and DIN/TDP (b) ratios (1999-2006) in the four agricultural intensity categories. Medians of box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey's post hoc after a Log10 transformation was applied to normalize the data.

The increase in DIN/TDP from low to moderate to high suggests that the relative abundance of plant available P decreases as agricultural intensity increases. Low ratios (<4) in low intensity and irrigated watersheds suggest that there is a relative overabundance of plant available P in these watersheds and that N would be more likely to be limiting plant productivity and driving eutrophication. These findings are counter intuitive to what is believed about the drivers of eutrophication in freshwater streams (i.e., that P is the key nutrient) (Schindler 1977, Sharpley et al 1987, Isermann 1990) but support other studies which have also found nitrogen to be the limiting nutrient in temperate streams across Canada and the United States (Dodds et al. 2002). The data may indicate that the generalization about P being the key limiting nutrient may not be valid in all Ecoregions or soil types, though some studies have shown that P control is essential in decreasing eutrophication as controlling N alone is not sufficient (Carpenter 2008, Schindler et al 2008).

Nitrogen and phosphorus ratios were also compared by watershed location by ecoregion (Table 3.9). The ratio of TN/TP was significantly higher in the Grassland streams than the other ecoregion areas ($p<0.05$), while no statistical difference was found among the remaining ecoregion areas (Figure 3.22a). The TN/TP ratios by ecoregion area suggest potential N limitation in the agricultural watersheds as they are all below the Redfield ratio of 16 (Redfield 1958); however, ratios in the Grassland ecoregion did exceed 16 and indicate that the N may not always be the limiting nutrient of primary producers in the Grassland watersheds.

Again, the Grassland ecoregion had a significantly higher median DIN/TDP ratio than watersheds in the other ecoregions ($p<0.05$); the ratio observed for the Irrigated Grassland streams was significantly lower than those measured in the Parkland ($p<0.01$) and Boreal ($p<0.05$) ecoregion areas (Figure 3.22b). The Continental Divide had similar DIN/TDP ratios to the Irrigated Grassland watersheds. The median and average DIN/TDP ratios in the Grassland ecoregion again suggest P limitation. With the exception of the Grassland watersheds, streams located in the more southern ecoregions (i.e., Irrigated Grasslands and Continental Divide) appear to have an overabundance of plant available P, with N limiting plant productivity and influencing eutrophication. Geologic sources of N have been found in groundwater in some areas of southern Alberta (Rodvang et al. 2004); therefore, it is possible that other sources, in addition to agriculture, may contribute to eutrophication of N limited surface waters in southern Alberta.

Note that the N/P ratio only indicates potential nitrogen or phosphorus limitations. Previous studies have found instream nutrient concentrations are also important to consider (Dodds et al. 2002; Stelzer and Lamberta 2001). High instream nutrient concentrations can lead to a saturation point of algal growth, negating the suggested limitation of the N/P ratio.

Table 3.9. Median and Average TN/TP and DIN/TDP FWMC ratios by ecoregion area (1999-2006).

Ecoregion	Median DIN:TDP	Ave. DIN:TDP	Median TN:TP	Ave. TN:TP
Boreal	5.73	75.3	7.37	8.15
Parkland	12.4	28.9	6.86	7.43
Grassland	34.8	84.9	10.4	17.2
Irrigated Grassland	3.49	8.23	7.36	8.76
Continental Divide	5.77	6.51	6.11	7.62

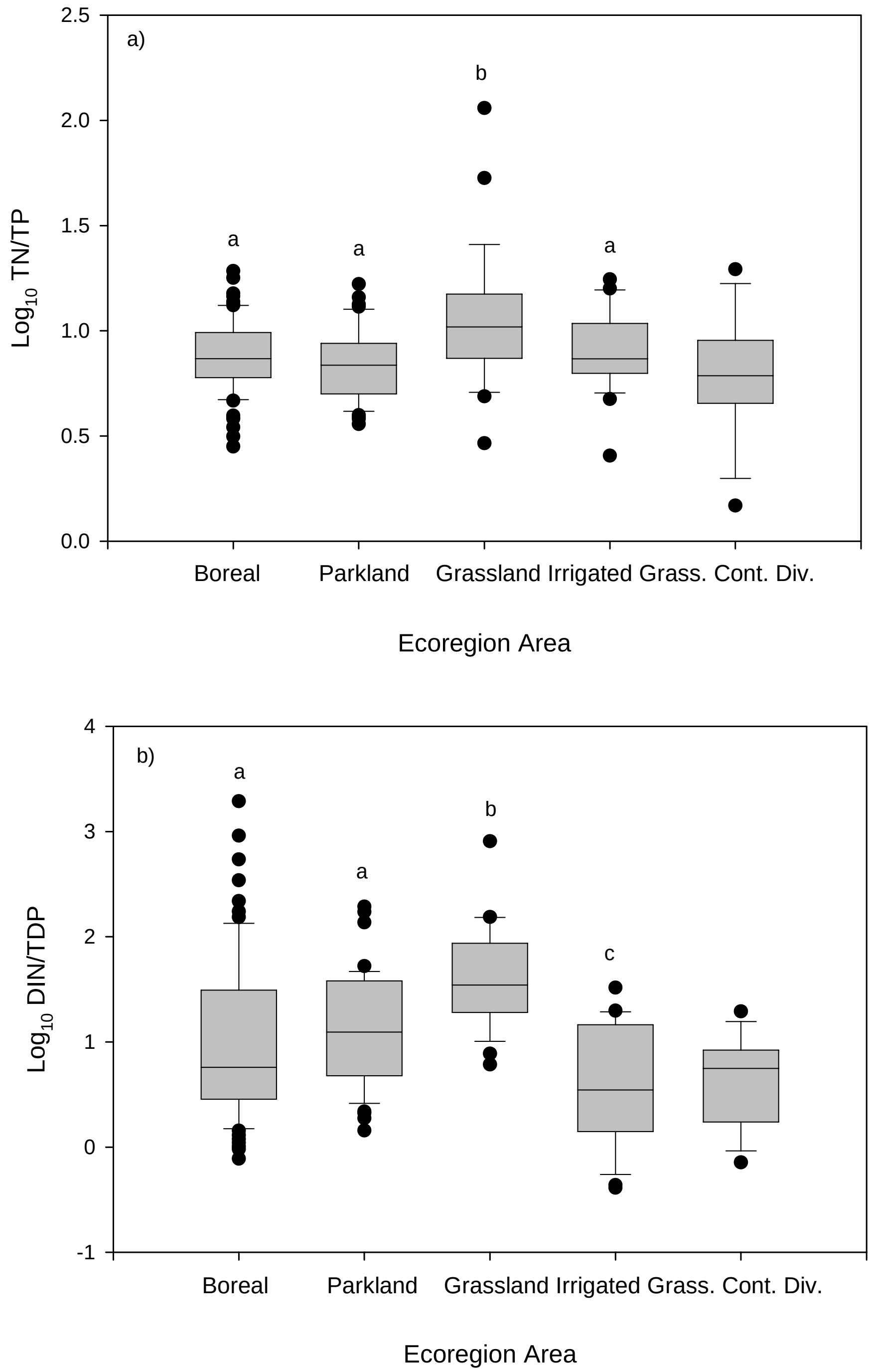


Figure 3.22. Box plots of median annual TN/TP (a) and DIN/TDP (b) FWMC ratios (1999-2006) for the five ecoregion areas. Medians of box plots with the same letter are not significantly different from one another at the 0.05 level as tested with ANOVA and Tukey's post hoc after a Log10 transformation was applied to normalize the data.

Correlations of P and N FWMCs with Agricultural Intensity metrics. Spearman Rank Correlations were run between the medians of median annual P and N FWMCs for the AESA streams and the averaged Census of Agriculture metrics (1996, 2001, and 2006) to examine the relationship between nutrient FWMCs and agriculture. As mentioned previously, relationships with agricultural intensity metrics were examined as the metrics can be used to link the type and intensity of agriculture in a watershed with changes in water quality.

Correlations between the overall agricultural intensity metric and the dissolved nutrient fractions support use of the metric as an indicator for agricultural influence of stream nutrient concentrations. Strong, positive correlations (0.50 to 0.75, $p<0.02$) were observed between the overall agricultural intensity percentile and the dissolved nitrogen fractions (NO2-+NO3--N and NH3-N) and ratios of TDP/TP and DIN/TN (Table 3.10). The overall metric was also weakly correlated with TDP ($p<0.05$). The same strong, positive correlations were observed with both fertilizer and chemical sales as for the overall agricultural intensity metric (Table 3.10). However, the DIN/TN ratio was the only parameter strongly correlated with the average manure production percentile ($p<0.02$), with weaker correlations observed between the manure production percentile and dissolve N fractions (0.46, $p<0.05$). Note that it was the dissolved fractions of N and P rather than the total that correlated most closely with the agricultural intensity metric at a provincial scale.

Table 3.10. Spearman Rank Correlations between median annual P and N FWMCs (1999 - 2006) and average Census of Agriculture metrics (1996, 2001, 2006) for 22 AESA watersheds (n=22).

Parameter	Overall Agricultural Intensity Percentile	Manure Prod. Percentile	Fertilizer Sales Percentile	Chemical Sales Percentile
TP	0.35	0.27	0.32	0.35
TDP	0.48	0.22	**0.52**	**0.53**
TPP	-0.12	0.09	-0.24	-0.20
TDP/TP	**0.58**	0.14	**0.69**	**0.68**
TN	0.42	0.25	0.42	0.42
Org N	0.33	0.17	0.33	0.34
$NO_2^-+NO_3^-$-N	**0.67**	0.46	**0.60**	**0.65**
NH_3-N	**0.50**	0.46	0.46	0.46
DIN/TN	**0.75**	**0.63**	**0.65**	**0.71**

Note: **Bold** values indicate significance at $p<0.02$.

As agricultural practices and intensity may vary for different regions of the province, Spearman Rank Correlations were run between annual P and N FWMCs for the AESA streams and each average Census of Agriculture metric (1996, 2001, and 2006) grouped by ecoregion area (Tables 3.11 to 3.14). In the Boreal ecoregion area, TN, DIN/TN, and the dissolved fractions of N were strongly, positively correlated (0.75 to 0.92, $p<0.05$) with the average agricultural intensity percentile (Table 3.11, Figures 3.23 and 3.24). There appeared to be a strong relationship with P, but the correlations were not significant ($p>0.05$). Total N was the only significantly correlated parameter with the average agricultural intensity percentiles in the Parkland ecoregion (0.89, $p<0.05$, Table 3.12). Note that the agricultural intensity metric was more strongly correlated overall with N than with P in the Boreal and Parkland ecoregion areas. All parameters with the exception of TDP/TP and DIN/TN ratios showed a strong, positive relationship with the average agricultural intensity percentiles in the Grassland ecoregion area, but with such a small sample size, it is unclear whether that relationship is real. The ratios of TDP/TP and DIN/TN showed an inverse relationship (Table 3.13). Nutrient parameters appeared to be inversely correlated with the average agricultural intensity percentile in the Irrigated Grassland ecoregion area (Table 3.14), but as for the Grassland ecoregion area, a greater number of samples would be required to determine whether the relationship was significant.

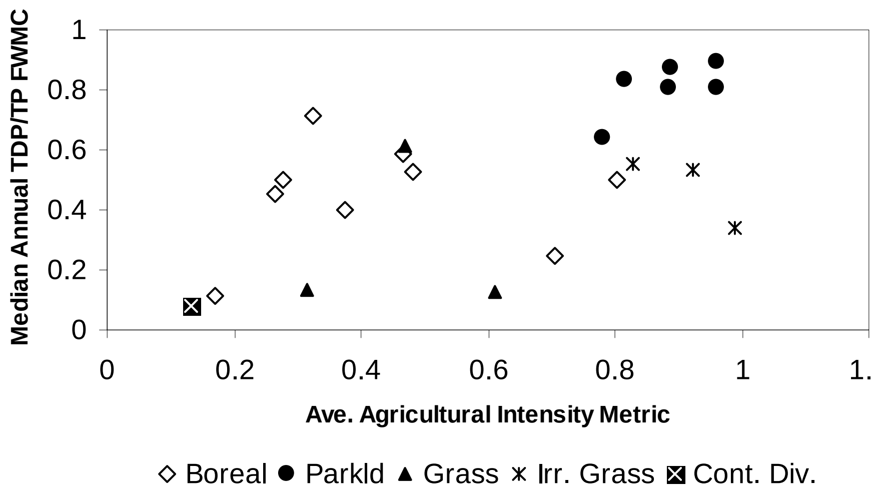


Figure 3.23. Average agricultural intensity metric (1999, 2001, 2006) and median annual TDP/TP FWMC for the AESA watersheds.

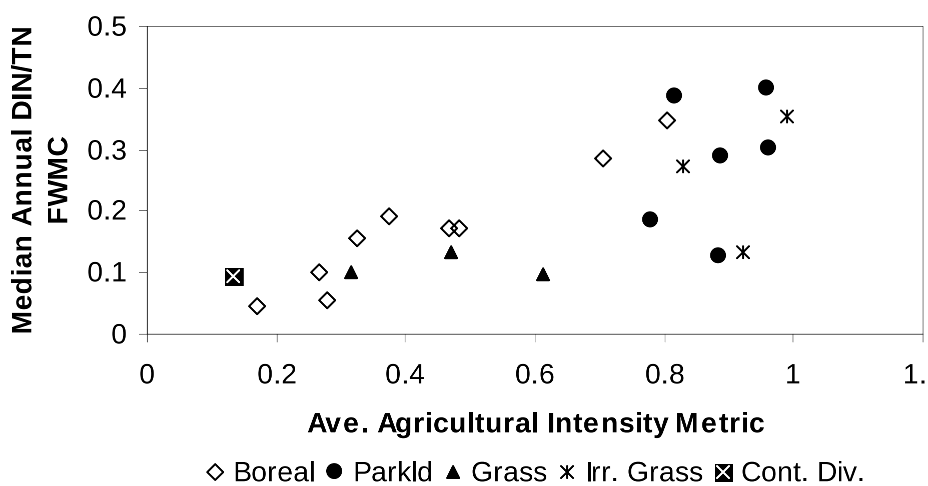


Figure 3.24. Average agricultural intensity metric (1999, 2001, 2006) and median annual DIN/TN FWMC for the AESA watersheds.

The manure production metric appeared to be better indicator of nutrient contamination in the Boreal ecoregion area than in the Parkland ecoregion area where the majority of high agricultural intensity AESA watersheds were located. Total P and N, ammonia-N, and the DIN/TN ratio were strongly, positively correlated with manure production percentiles in the Boreal ecoregion (0.68 to 0.90, $p<0.02$) (Table 3.11). In the Parkland ecoregion area, TN and DIN/TN were the only parameters with a strong relationship with the manure production, though these correlations were not significant ($p>0.05$) (Table 3.12). The fertilizer metric was only significantly correlated with NO_2^-+NO_3^--N and DIN/TN in the Boreal ecoregion area (0.72 and 0.70, respectively, $p<0.05$). Due to the limited sample size and inability to determine the significance of any relationships, correlations between nutrient parameters and agricultural intensity metrics for the Grassland and Irrigated Grassland ecoregion areas are shown in Tables 3.13 and 3.14 but not discussed.

Unlike the relationships observed between nutrients and the manure and fertilizer metrics, strong, positive relationships were noted between the chemical sales percentiles and the majority of P and N fractions in the Parkland ecoregion area. Note that TP, TDP, and Org N were the only parameters with significant correlations (0.89, $p<0.05$) (Table 3.12). The stronger correlation between chemical sales and nutrient concentrations in the Parkland ecoregion area were unexpected considering fertilizer and manure are nutrient sources. It is uncertain whether the metrics for nutrient sources are inadequate or if there is something inherent about the chemical sales percentiles (i.e., land use type) that is a better surrogate of nutrient use.

Table 3.11. Spearman Rank Correlations between median annual P and N FWMCs (1999 to 2006) and average Census of Agriculture metrics (1996, 2001, 2006) for watersheds in the Boreal ecoregion area (n=9).

Parameter	Overall Agricultural Intensity Percentile	Manure prod. Percentile	Fertilizer Sales Percentile	Chemical Sales Percentile
TP	0.65	**0.73**	0.38	0.45
TDP	0.57	0.55	0.42	0.48
TPP	0.35	0.63	0.07	0.00
TDP/TP	0.28	-0.18	0.43	0.47
TN	**0.80**	**0.68**	0.55	0.62
Org N	0.65	0.45	0.38	0.48
NO_2^-+NO_3^--N	**0.90**	0.62	**0.72**	**0.75**
NH_3-N	**0.75**	**0.90**	0.47	0.43
DIN/TN	**0.92**	**0.78**	**0.70**	0.63

Note: **Bold** denotes significance at $p<0.05$.

Table 3.12. Spearman Rank Correlations between median annual P and N FWMCs (1999 to 2006) and average Census of Agriculture metrics (1996, 2001, 2006) for watersheds in the Parkland ecoregion area (n=6).

Parameter	Overall Agricultural Intensity Percentile	Manure Prod. Percentile	Fertilizer Sales Percentile	Chemical Sales Percentile
TP	0.77	0.31	0.26	**0.89**
TDP	0.77	0.31	0.26	**0.89**
TPP	0.77	0.37	-0.09	0.66
TDP/TP	0.37	0.03	0.31	0.71
TN	**0.89**	0.60	0.14	0.77
Org N	0.77	0.31	0.26	**0.89**
NO_2^-+NO_3^--N	0.26	0.31	-0.26	0.37
NH_3-N	0.71	0.43	-0.03	0.60
DIN/TN	0.43	0.60	-0.43	0.31

Note: Bold denotes significance at $p<0.05$.

Table 3.13. Spearman Rank Correlations between median annual P and N FWMCs (1999 - 2006) and average Census of Agriculture metrics (1996, 2001, 2006) for watersheds in the Grassland ecoregion area (n=3).

Parameter	Overall Agricultural Intensity Percentile	Manure Prod. Percentile	Fertilizer Sales Percentile	Chemical Sales Percentile
TP	1.00	0.50	0.50	0.50
TDP	0.50	-0.50	1.00	1.00
TPP	0.50	1.00	-0.50	-0.50
TDP/TP	-0.50	-1.00	0.50	0.50
TN	0.50	-0.50	1.00	1.00
Org N	0.50	-0.50	1.00	1.00
NO_2^- +NO_3^--N	0.50	-0.50	1.00	1.00
NH_3-N	1.00	0.50	0.50	0.50
DIN/TN	-0.50	-1.00	0.50	0.50

Table 3.14. Spearman Rank Correlations between median annual P and N FWMCs (1999 - 2006) and average Census of Agriculture metrics (1996, 2001, 2006) for watersheds in the Irrigated Grassland ecoregion area (n=4).

Parameter	Overall Agricultural Intensity Percentile	Manure Prod. Percentile	Fertilizer Sales Percentile	Chemical Sales Percentile
TP	-0.50	-0.50	-0.50	-0.50
TDP	-1.00	-1.00	-1.00	-1.00
TPP	-1.00	-1.00	-1.00	-1.00
TDP/TP	-1.00	-1.00	-1.00	-1.00
TN	-0.50	-0.50	-0.50	-0.50
Org N	-0.50	-0.50	-0.50	-0.50
$NO_2^- + NO_3^-$-N	-0.50	-0.50	-0.50	-0.50
NH_3-N	-0.50	-0.50	-0.50	-0.50
DIN/TN	0.50	0.50	0.50	0.50

Temporal trends in P and N FWMC. Annual nutrient FWMCs were visually examined in each watershed to assess whether water quality changed with time and are illustrated in Figures 3.25 through 3.50. Temporal monotonic trends were not found in the majority of watersheds. However, nine watersheds appeared to have a potential increasing or decreasing temporal trend and require further statistical investigation to assess whether a temporal pattern was indeed present and a result of agricultural influences or attributed to flow (Battersea Drain, Blindman River, Buffalo Creek, Kleskun Drain, Meadow Creek, Prairie Blood Coulee, Tomahawk Creek, Renwick Creek, and Wabash Creek). It was noted that watersheds that are prone to having large flow events will be more difficult to evaluate with respect to changing agricultural intensity as a result of flow related “event” years in the data record (i.e., decreasing TDP/TP as a result of increased flow). Watersheds have been grouped by 1996 agricultural intensity categories for ease of discussion.

Low Agricultural Intensity Watersheds (Hines, Paddle, Prairie, Rose, Willow)

A temporal trend in P or N FWMCs was not observed in Hines Creek (Figure 3.25, Table 3.15). There was some variability among years for the total and dissolved ratios of P and N.

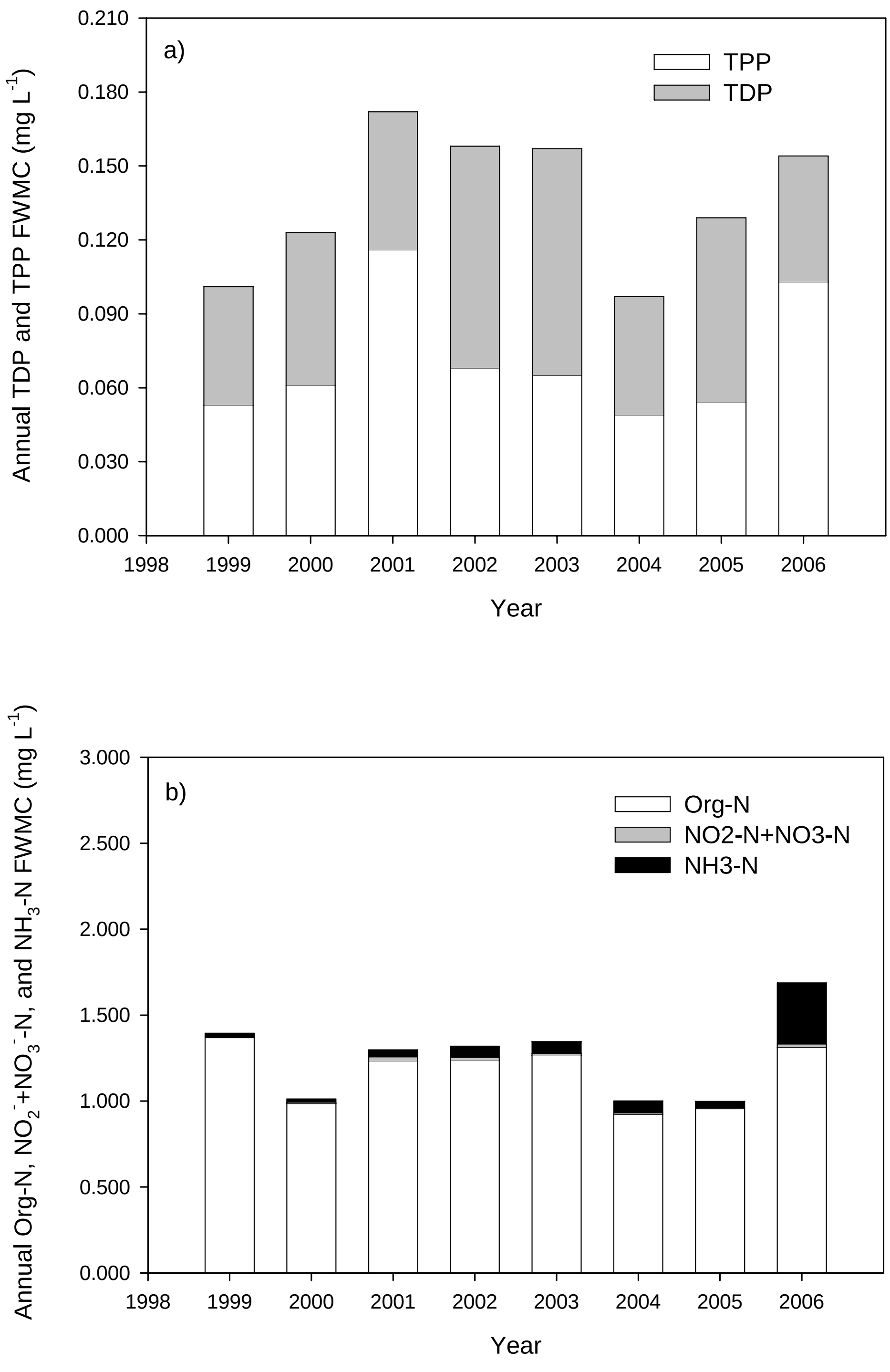
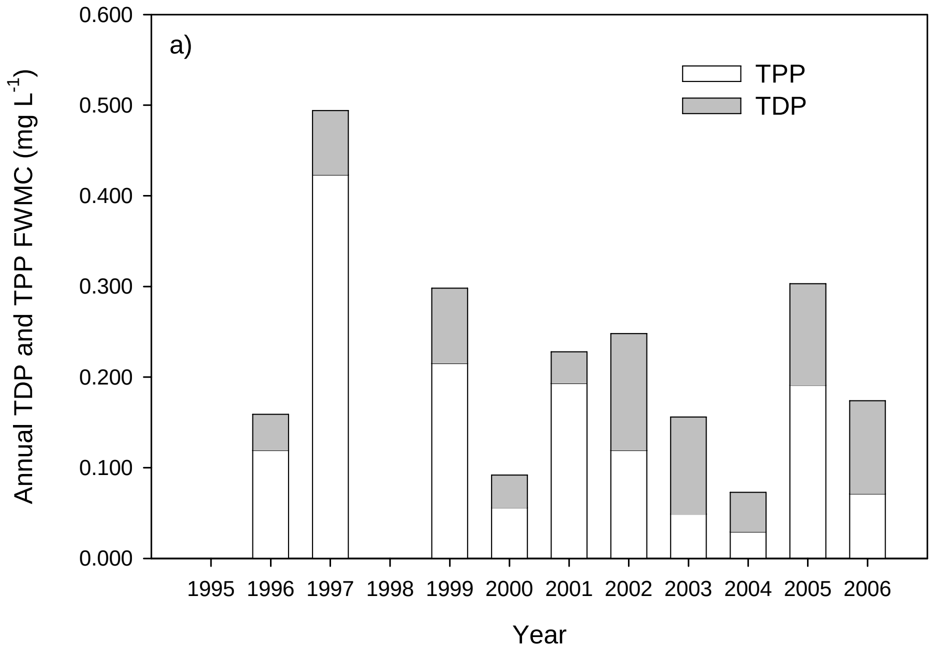


Figure 3.25. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Hines Creek. Bars are stacked (i.e., summed totals for TDP and TPP are equal to TP, and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year).

Table 3.15. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Hines Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg{\cdot}L^{-1}$)	Minimum ($mg{\cdot}L^{-1}$)	Year	Maximum ($mg{\cdot}L^{-1}$)	Year
TP	0.140	0.098	2004	0.173	2001
TDP	0.059	0.048	1999 and 2004	0.092	2002
TPP	0.063	0.049	2004	0.116	2001
TDP/TP	0.416	0.327	2001	0.587	2003
TN	1.310	0.998	2005	1.687	2006
Org N	1.237	0.923	2004	1.369	1999
NH_3-N	0.010	0.018	2000	0.355	2006
NO_2^-+NO_3^--N	0.010	0.003	1999	0.023	2001
DIN/TN	0.054	0.019	1999	0.222	2006
Annual stream volume (hm^3)	14.1	0.2	1999	22.6	2003

Overall, there did not appear to be any trend with time in TP or TN FWMC in Paddle River (Figure 3.26). The majority of TP FWMCs were measured in the particulate fraction with a higher ratio of TDP between 2002 and 2004 (0.52-0.69) (Figure 3.26(a), Table 3.16). The ratio of TPP to TP was generally highest during years with higher stream volume (ex., 45hm^3, 0.86 TPP/TP in 1997; 19.7hm^3, 0.85 TPP/TP in 2001), while the ratio of TDP to TP was generally higher in years of low flow (8.7hm^3, 0.69 TPP/TP in 2003; 6.9 hm^3, 0.60 TPP/TP in 2004). The ratio of TP measured as TDP appeared to increase over time although annual stream volume also appeared to decrease over time, indicating less particulate movement from land to water (Figure 3.27).



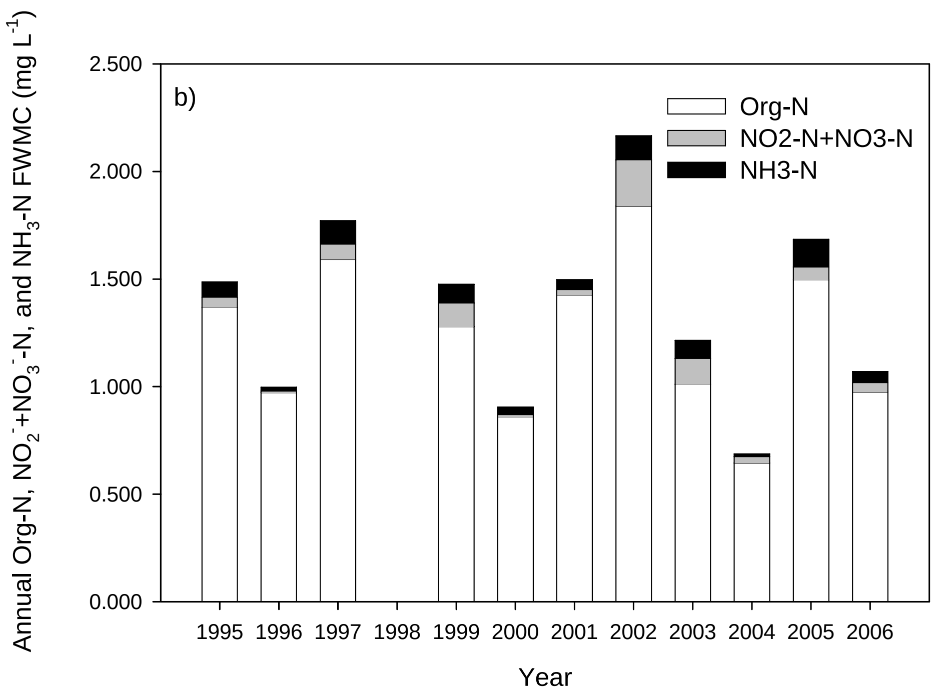


Figure 3.26. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Paddle River. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.16. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Paddle River. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg{\cdot}L^{-1}$)	Minimum ($mg{\cdot}L^{-1}$)	Year	Maximum ($mg{\cdot}L^{-1}$)	Year
TP	0.228	0.073	2004	0.494	1997
TDP	0.077	0.035	1997	0.129	2002
TPP	0.119	0.029	2004	0.423	1997
TDP/TP	0.334	0.144	1997	0.686	2003
TN	1.476	0.687	2004	2.167	2002
Org N	1.274	0.645	2004	1.839	2002
NH_3-N	0.072	0.012	2004	0.129	2005
NO_2^-+NO_3^--N	0.047	0.007	1996	0.216	2002
DIN/TN	0.089	0.025	1996	0.168	2003
Annual stream volume (hm^3)	9.0	4.4	2006	45.4	1997

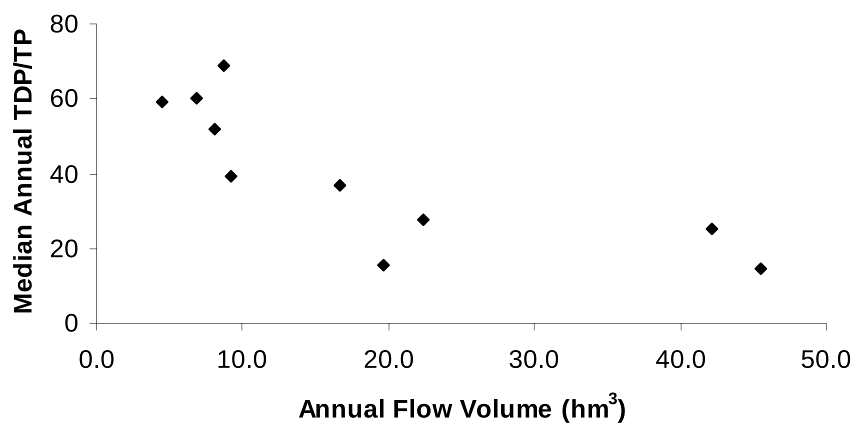


Figure 3.27. Median annual TDP/TP versus annual flow volume in Paddle River.

There appeared to be a potential temporal trend in TP and TN FWMCs with time in Prairie Blood Coulee, with variability among years between 1999 and 2006 but slightly higher values in the latter years of the study (Figure 3.28, Table 3.17). There also seemed to be an increasing ratio of DIN from 2001 through 2006. Although Prairie Blood Coulee was initially ranked as a low agricultural intensity watershed, the amount and type of agricultural activity in the watershed is uncertain as a result of the different calculation methods used for the agricultural intensity metric (Chapter 2: Materials and Methods- Agricultural Intensity Metrics). Note that irrigation pivots were found in 1991 air photos of Prairie Blood Coulee. However, Anderson et al. (1998b) also found higher NO_2^-+NO_3^--N FWMCs in Prairie Blood Coulee but could not attribute the observations to land use. A statistical trend analysis would be required to determine whether a temporal trend in TP and TN FWMC and the ratio of DIN/TN was actually present.

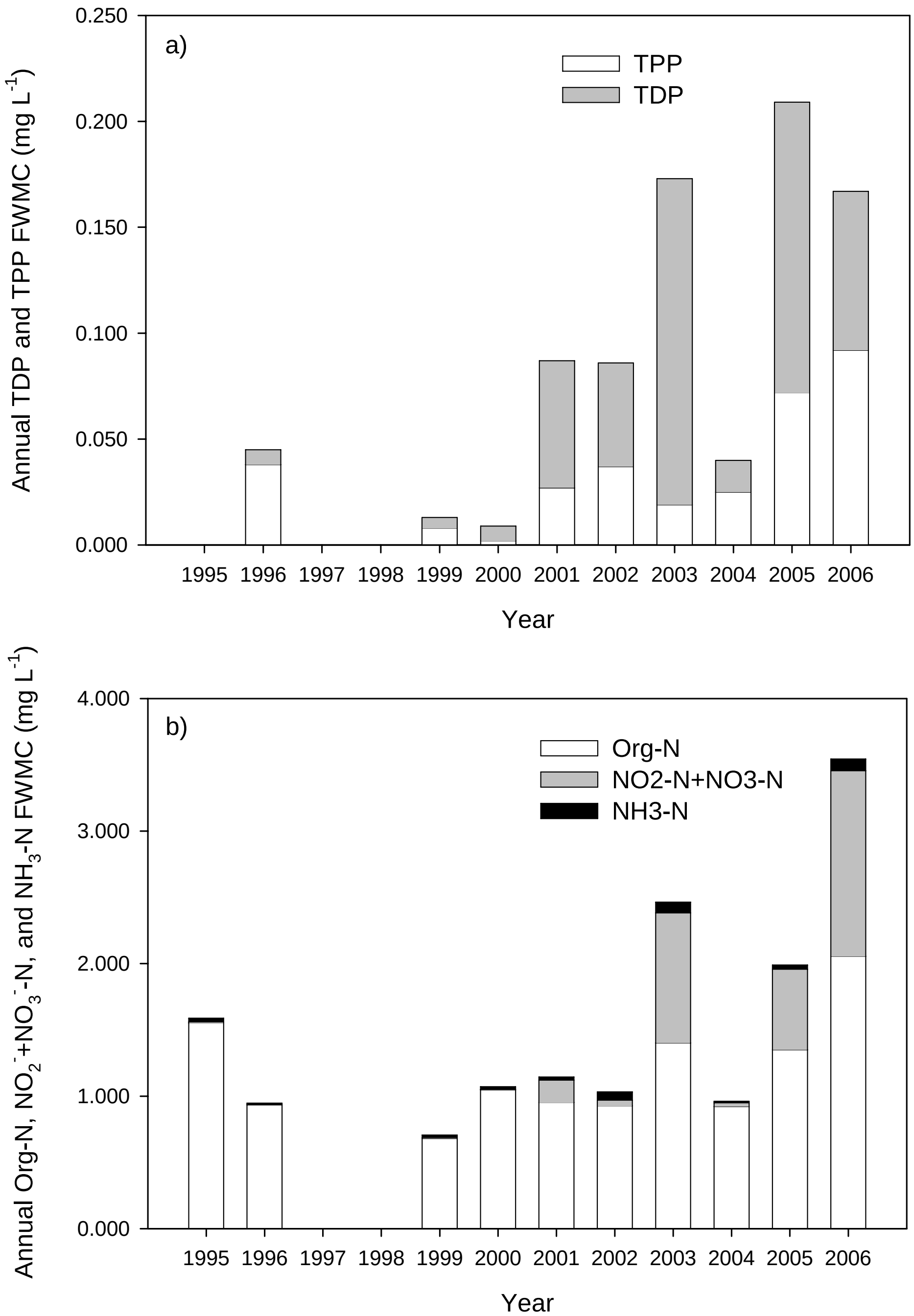


Figure 3.28. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Prairie Blood Coulee. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.17. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Prairie Blood Coulee. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg \cdot L^{-1}$)	Minimum ($mg \cdot L^{-1}$)	Year	Maximum ($mg \cdot L^{-1}$)	Year
TP	0.086	0.009	2000	0.210	2005
TDP	0.049	0.049	2002	0.154	2003
TPP	0.027	0.002	2000	0.092	2006
TDP/TP	0.570	0.158	1996	0.892	2003
TN	1.110	0.705	1999	3.545	2006
Org N	1.000	0.681	1999	2.056	2006
NH_3-N	0.026	0.012	1996 and 2004	0.089	2006
NO_2^-+NO_3^--N	0.035	0.001	1996	1.399	2006
DIN/TN	0.070	0.014	1996	0.431	2003
Annual stream volume (hm^3)	1.7	0.1	1999	10.1	2005

No monotonic temporal trend in TP or TN FWMCs was observed in Rose Creek between 1995 and 2006 (Figure 3.29). As observed in the annual loads, annual TP FWMCs were much higher in 1998 and 1999 (0.955 and 0.826 $mg \cdot L^{-1}$, respectively). Nitrogen FWMCs for all fractions with the exception of NO_2^-+NO_3^--N also peaked in 1998 (Table 3.18). Furthermore, Rose Creek had higher flow than all other streams sampled in 1998 (Chapter 2: Tables 2.5 to 2.7). These high levels appeared to be linked to the effects of high runoff in the basin, specifically high flows resulting from heavy rains experienced during the summer of 1998. Almost all of the TP measured in Rose Creek was in the form of TPP, suggesting sediment movement as the main source of P to the stream.

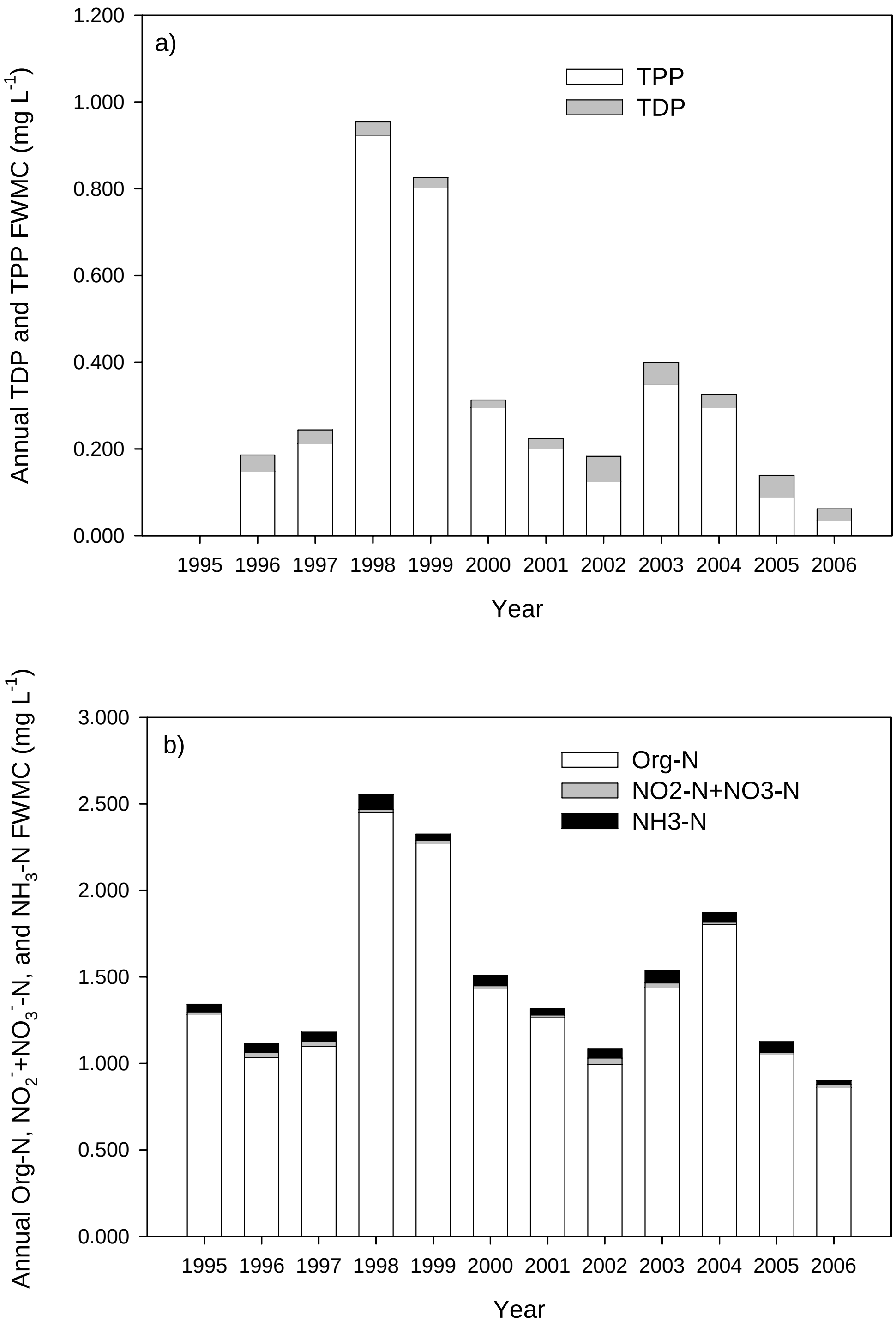


Figure 3.29. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Rose Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.18. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Rose Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg·L^{-1}$)	Minimum ($mg·L^{-1}$)	Year	Maximum ($mg·L^{-1}$)	Year
TP	0.234	0.062	2006	0.955	1998
TDP	0.030	0.018	2000	0.058	2002
TPP	0.212	0.035	2006	0.924	1998
TDP/TP	0.130	0.029	1999	0.435	2006
TN	1.332	0.900	2006	2.551	1998
Org N	1.276	0.862	2006	2.453	1998
NH_3-N	0.054	0.023	2006	0.084	1998
NO_2^-+NO_3^--N	0.016	0.011	2001	0.036	2002
DIN/TN	0.047	0.024	1999	0.082	2002
Annual stream volume (hm^3)	46.4	28.0	2001	85.4	1999

There were no temporal patterns in P or N FWMCs in Willow Creek (Figure 3.30, Table 3.19). Noteworthy, TP and TN FWMC in 2005 were much higher in Willow Creek than in other years of monitoring. Flows in 2005 were the highest of the AESA monitoring period, and the flows measured at Claresholm, Alberta were said to have had a 1 % chance of occurring in any given year (Alberta Environment 2005). Although the data are standardized for flow, the rare occurrence of these anomalously high flows could have resulted in the anomalously high FWMCs in 2005.

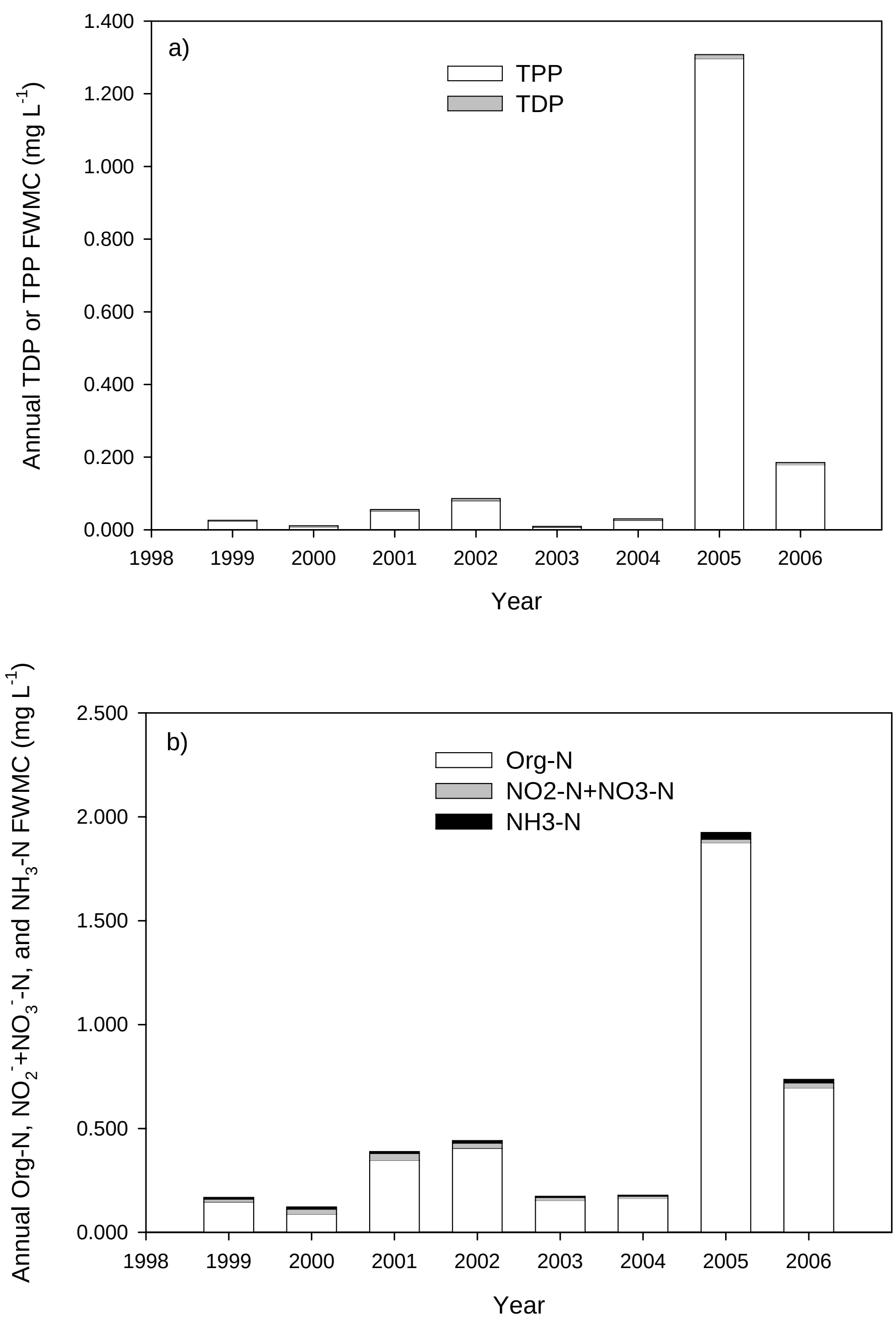


Figure 3.30. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Willow Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.19. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Willow Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg^{.}L^{-1}$)	Minimum ($mg^{.}L^{-1}$)	Year	Maximum ($mg^{.}L^{-1}$)	Year
TP	0.043	0.009	2003	1.308	2005
TDP	0.004	0.002	1999 and 2003	0.011	2005
TPP	0.039	0.007	2003	1.297	2005
TDP/TP	0.088	0.008	2005	0.266	2000
TN	0.283	0.123	2000	1.929	2005
Org N	0.225	0.088	2000	1.876	2005
NH_3-N	0.010	0.004	2004	0.033	2005
NO_2^-+NO_3^--N	0.020	0.010	2004	0.033	2001
DIN/TN	0.091	0.026	2005	0.280	2000
Annual stream volume (hm^3)	9.9	4.1	2000	41.4	2005

Moderate Agricultural Intensity Watersheds (Blindman, Grande Prairie, Kleskun, Meadow, Tomahawk, and Trout)

A temporal pattern in TP and TN FWMCs was not observed in Blindman River (Figure 3.31, Table 3.20). Higher ratios of TPP were observed between 1996 and 2001 (0.57-0.67), while higher ratios of TDP were measured between 2002 and 2006. Total flow volume was higher and peaks in flow occurred more frequently between 1996 and 2001, lending to the higher TPP fractions observed. Flow volume was generally lower between 2002 and 2006 compared to 1996 to 2001 (Chapter 2: Figures 2.10 and 2.11, Tables 2.5 to 2.7). It appeared that flow influenced the higher TP and TPP FWMCs observed in the earlier years of the program, while agricultural practices may have influenced the TP and dissolved P FWMCs measured in 2005 and 2006. The annual TDP/TP ratio appeared to increase over the study period (Figure 3.32); however, a statistical trend analysis would be required to determine whether the apparent increasing trend in TDP/TP was statistically significant or a result of trends in flow. Unlike TDP/TP, a trend was not observed for the ratios of TN as DIN.

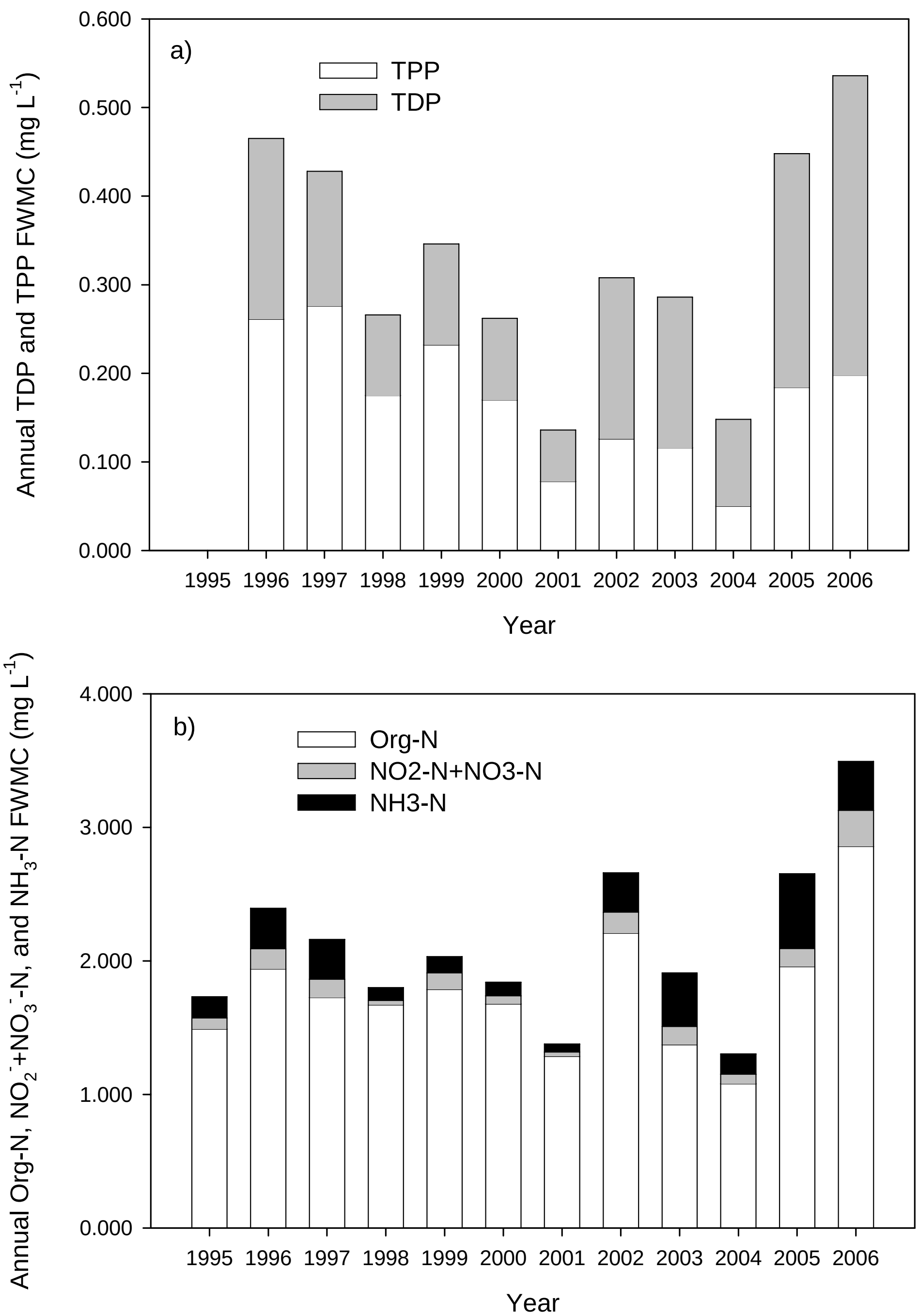


Figure 3.31. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Blindman River. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.20. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Blindman River. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg·L^{-1}$)	Minimum ($mg·L^{-1}$)	Year	Maximum ($mg·L^{-1}$)	Year
TP	0.297	0.136	2001	0.536	2006
TDP	0.152	0.058	2001	0.338	2006
TPP	0.175	0.050	2004	0.276	1997
TDP/TP	0.512	0.330	1999	0.663	2004
TN	1.973	1.305	2004	3.495	2006
Org N	1.702	1.079	2004	2.857	2006
NH_3-N	0.227	0.061	2001	0.560	2005
NO_2^-+NO_3^--N	0.130	0.032	2001	0.271	2006
DIN/TN	0.171	0.068	2001	0.282	2003
Annual stream volume (hm^3)	26.2	11.6	1995	64.1	1999

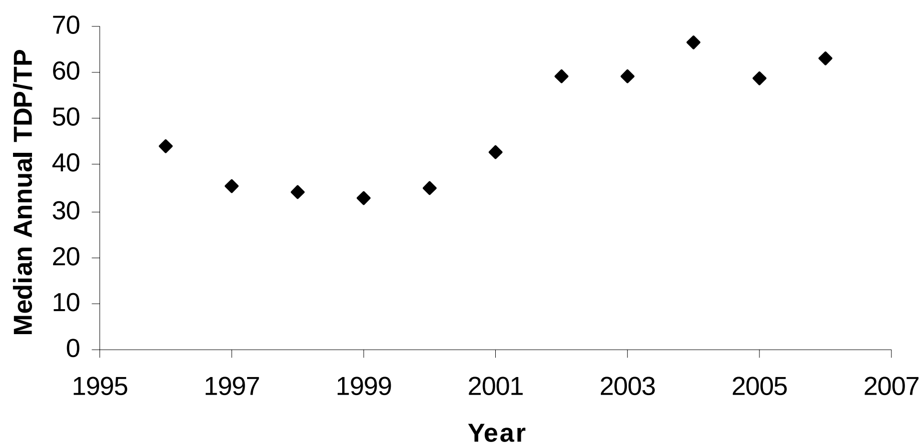


Figure 3.32. Median annual TDP/TP FWMCs in Blindman River during the study period (1996 to 2006).

There did not appear to be a trend in annual TP or TN FWMCs over time in Grande Prairie Creek (Figure 3.33). Flow weighted mean concentrations of N and P varied quite considerably form year to year but so did the magnitude and timing of peak discharge. There were some years with very little flow in the spring (e.g., 2000, 2004, and 2006), other years with flow only in the spring (e.g., 1999, 2002, 2003, and 2005), as well as years with summer peaks in discharge (e.g., 2004). The summer storm event in 2004 yielded the highest FWMCs of Org N and TPP. Concentrations in other years appeared to vary unpredictably. The ratio of TP and TN in the dissolved fractions also appeared to be influenced by flow. The lowest TDP/TP ratio was measured in 2004 and was likely a result of several high peaks in flow (Table 3.21). In contrast, the ratio of TN in the inorganic form was much higher in 1999 when peaks in NO_2^-+NO_3^--N and NH_3-N were observed, which likely corresponded to flow that was only measured in the spring. Information on management practices would also help to explain these observations.

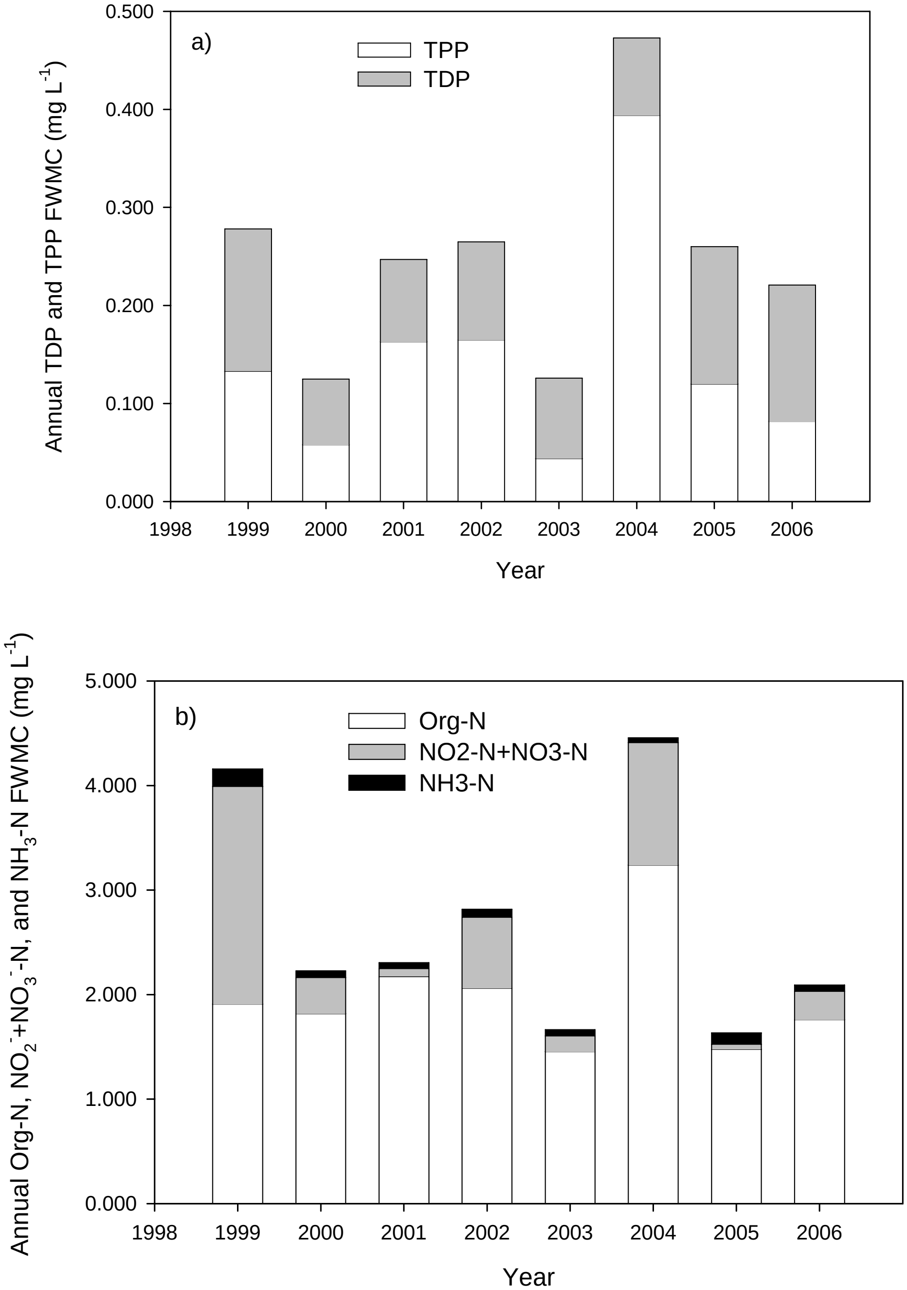


Figure 3.33. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Grande Prairie Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.21. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Grande Prairie Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg^{.}L^{-1}$)	Minimum ($mg^{.}L^{-1}$)	Year	Maximum ($mg^{.}L^{-1}$)	Year
TP	0.253	0.125	2000	0.473	2004
TDP	0.092	0.067	2000	0.145	1999
TPP	0.126	0.044	2003	0.394	2004
TDP/TP	0.363	0.167	2004	0.652	2003
TN	2.268	1.633	2005	4.513	2004
Org N	1.863	1.454	2003	3.238	2004
NH_3-N	0.063	0.045	2004	0.166	1999
NO_2^-+NO_3^--N	0.308	0.050	2005	2.083	1999
DIN/TN	0.171	0.058	2001	0.540	1999
Annual stream volume (hm^3)	7.4	1.5	2000	12.1	2005

An increasing temporal pattern in TP FWMCs was observed in Kleskun Main Drain from 1999 to 2006; however, an increasing monotonic trend was not found for N or the ratios of dissolved to total nutrients (Figure 3.34, Table 3.22). Similar to Grande Prairie Creek, NO_2^-+NO_3^--N FWMCs peaked in 1999 (Table 3.22), but the peak in NO_2^-+NO_3^--N did not appear to be related to stream flow. Ground-truthing and a better understanding of the agricultural practices in this watershed would assist in identifying sources of P contribution in the watershed. A statistical trend analysis would also be required to accurately assess temporal trends.

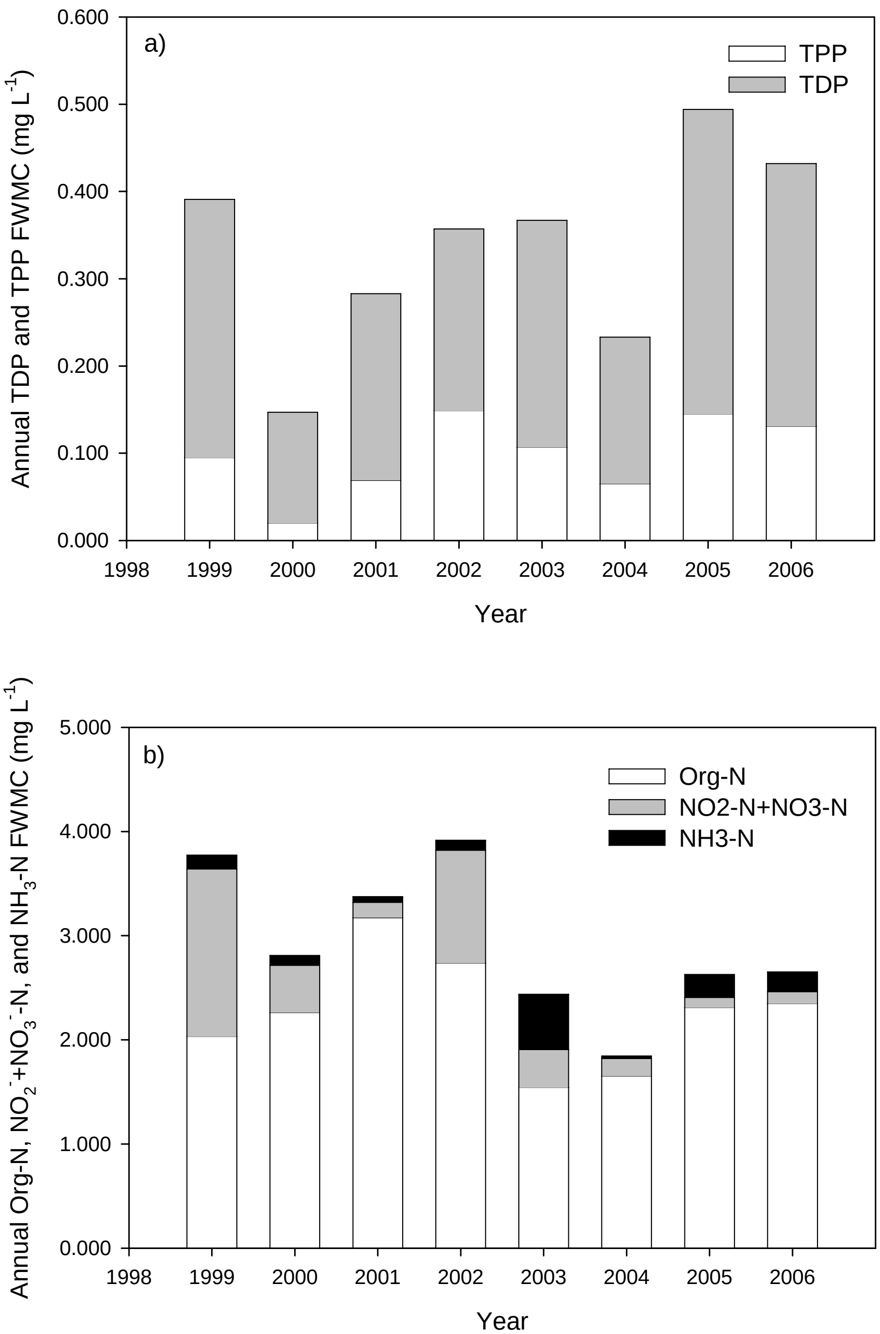


Figure 3.34. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Kleskun Drain. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.22. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Kleskun Drain. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg{\cdot}L^{-1}$)	Minimum ($mg{\cdot}L^{-1}$)	Year	Maximum ($mg{\cdot}L^{-1}$)	Year
TP	0.362	0.147	2000	0.494	2005
TDP	0.237	0.127	2000	0.349	2005
TPP	0.101	0.020	2000	0.149	2002
TDP/TP	0.654	0.582	2002	0.864	2000
TN	2.741	1.846	2004	3.931	2002
Org N	2.286	1.544	2003	3.172	2001
NH_3-N	0.115	0.024	2004	0.531	2003
NO_2^-+NO_3^--N	0.265	0.096	2005	1.605	1999
DIN/TN	0.157	0.060	2001	0.457	1999
Annual stream volume (hm^3)	1.7	0.04	2000	2.5	2002

There did not appear to be a monotonic temporal trend in TP, TDP, or TPP FWMC between 1996 and 2006 in Meadow Creek; however, annual TN FWMCs appeared to decrease (Figure 3.35). The decreasing TN trend did not appear to be related to flow, and agricultural intensity metrics (manure production and fertilizer and chemical expenses) decreased from 1996 to 2001 but increased again from 2001 to 2006. Annual total and organic fractions of P and N FWMCs were much higher in 1997 than other years (Table 3.23). The high FWMCs measured in 1997 were likely a reflection of increased runoff and movement of soil-bound P and Org N. Although annual stream volume was similar to other years of monitoring, a peak in stream flow was observed in March of 1997, while peaks in stream flow generally occurred in May in the stream (Chapter 2: Figure 2.17, Tables 2.5 to 2.7). This peak in flow in March may have been the result of a precipitation event that had a longer duration or higher intensity than other years which increased runoff and the amount of P (specifically particulate P) loading to the stream, as illustrated by the low ratio of TP as TDP in 1997. Moreover, a runoff event in the early spring may have carried more sediment with it if land cover was sparse or minimal compared to typical land cover in the late spring and summer. Similar FWMCs may have occurred in 1995 and 1996, but fewer samples were collected compared to 1997 and may not have accurately captured the peak flow events. The dissolved fraction of N was not lower with the higher flow event in 1997. In fact, the highest annual NH_3-N FWMC was measured that year. Land management data for this watershed would be useful in assessing contributions of N to the watershed.

The ratio of TP measured as TDP appeared to increase with time, with the exception of the annual value in 2000 (Figure 3.36). Annual flow volumes did not show any pattern with time in the stream and were not likely the main driver behind the increasing ratios of dissolved P. Annual stream volume in 2000 was lower than many other years. Although the annual flow volume was also low in 2001, the ratio of TP as TDP was not nearly as high as the amount measured in 2000. It is possible that a point source may have contributed to the high TP FWMC and high ratio of TDP in Meadow Creek in 2000. A statistical trend analysis would be required to properly asses the potential trends in Meadow Creek.

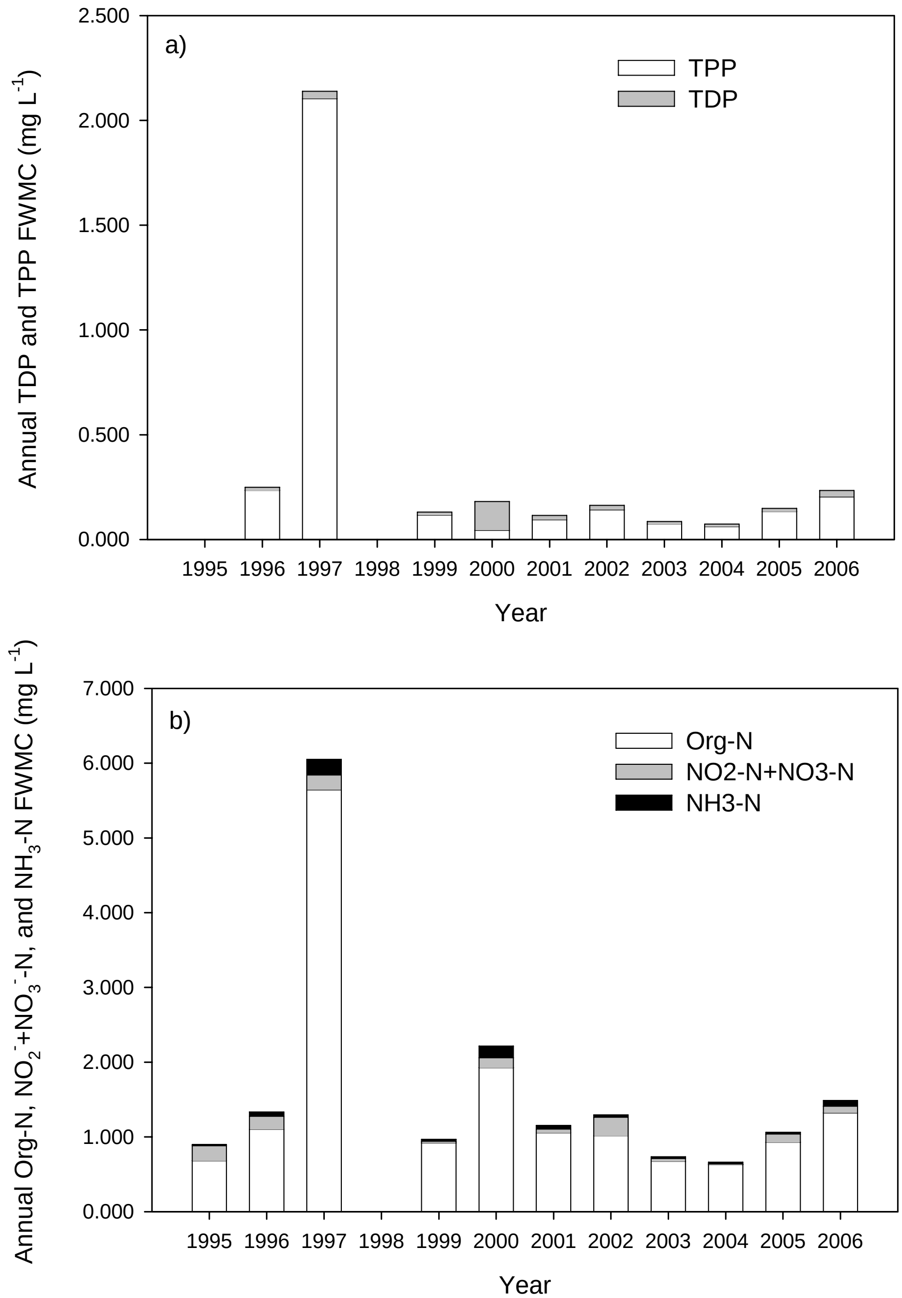


Figure 3.35. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Meadow Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.23. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Meadow Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg·L^{-1}$)	Minimum ($mg·L^{-1}$)	Year	Maximum ($mg·L^{-1}$)	Year
TP	0.148	0.073	2004	2.137	1997
TDP	0.018	0.011	2003	0.137	2000
TPP	0.125	0.044	2000	2.103	1997
TDP/TP	0.121	0.016	1997	0.757	2000
TN	1.154	0.662	2004	6.055	1997
Org N	1.017	0.630	2004	5.643	1997
NH_3-N	0.033	0.033	2002	0.210	1997
NO_2^-+NO_3^--N	0.110	0.010	2004	0.246	2002
DIN/TN	0.112	0.047	2004	0.244	1995
Annual stream volume (hm^3)	2.3	0.4	2000	10.8	2005

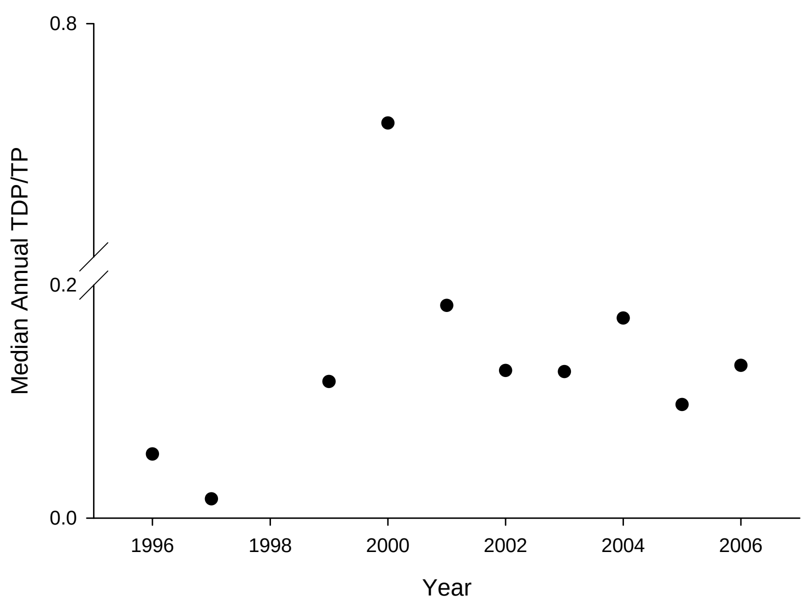


Figure 3.36. Median annual TDP/TP ratios in Meadow Creek from 1996 to 2006. Note that the TDP/TP value in 2000 was removed from this visual presentation of the data.

No observable trends with time in P or N FWMCs were found in Tomahawk Creek (Figure 3.37). Annual nutrient FWMCs were variable over the monitoring period (Table 3.24). However, there appeared to be a pattern of increasing TDP/TP ratios without a similar pattern in flow (Figure 3.38); a statistical trend analysis would be required to confirm the observation.

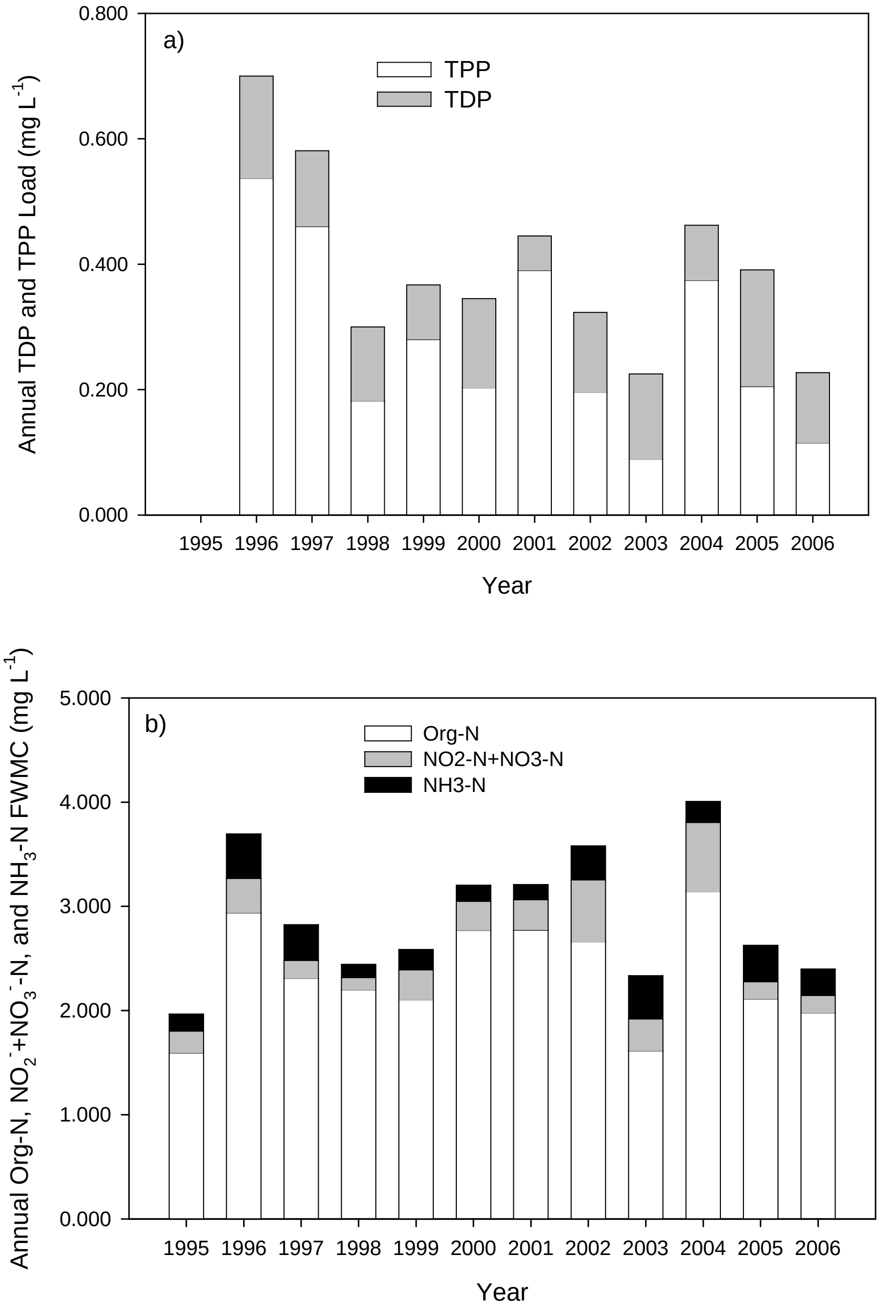


Figure 3.37. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Tomahawk Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.24. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Tomahawk River. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg \cdot L^{-1}$)	Minimum ($mg \cdot L^{-1}$)	Year	Maximum ($mg \cdot L^{-1}$)	Year
TP	0.356	0.201	1995	0.700	1996
TDP	0.121	0.055	2001	0.186	2005
TPP	0.205	0.089	2003	0.537	1996
TDP/TP	0.341	0.123	2001	0.605	2003
TN	2.727	1.968	1995	4.008	2004
Org N	2.254	1.593	1995	3.141	2004
NH_3-N	0.229	0.126	1998	0.429	1996
NO_2^-+NO_3^--N	0.282	0.118	1998	0.663	2004
DIN/TN	0.188	0.100	1998	0.309	2003
Annual stream volume (hm^3)	3.5	0.3	2006	12.8	1997

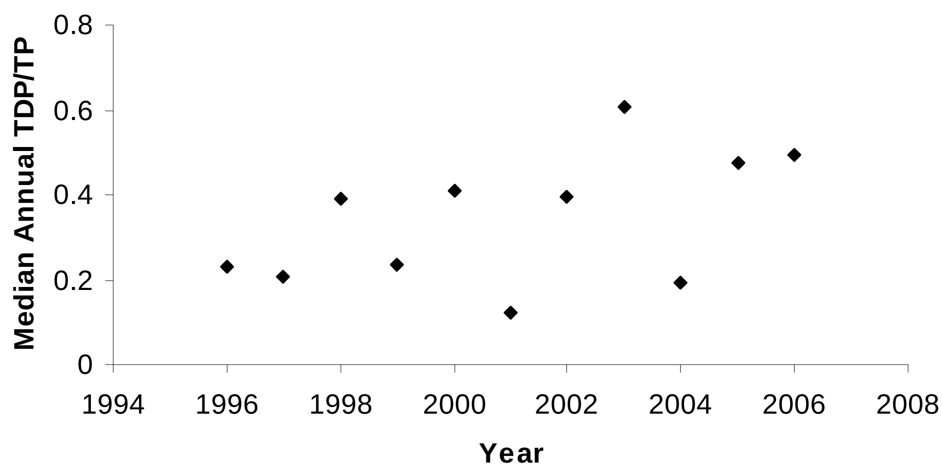


Figure 3.38. Median annual TDP/TP ratios in Tomahawk Creek from 1996 to 2006.

As in Meadow Creek, no temporal trend in annual TP or TN FWMCs was observed in Trout creek between 1996 and 2006 (Figure 3.39). Moreover, a similar spike in P and N FWMCs was observed in 1997 (Table 3.25). Again, annual stream volume was not any higher in 1997 than in other years. Flow records showed that a peak in flow occurred in March, while peaks generally occurred from May on in other years (Chapter 2: Figure 2.17). The similarities between Trout and Meadow Creeks reflect the influence of ecoregional characteristics (both streams are located in the Grassland ecoregion area). It was also observed that sampling in Trout Creek captured spring storm events in March and May of 1997, but precipitation events in June of 2005 may have been missed. It is possible that the very high nutrient FWMCs in 1997 in Trout Creek may not be exceptionally high and that other events that would have produced similar data in latter years were not captured.

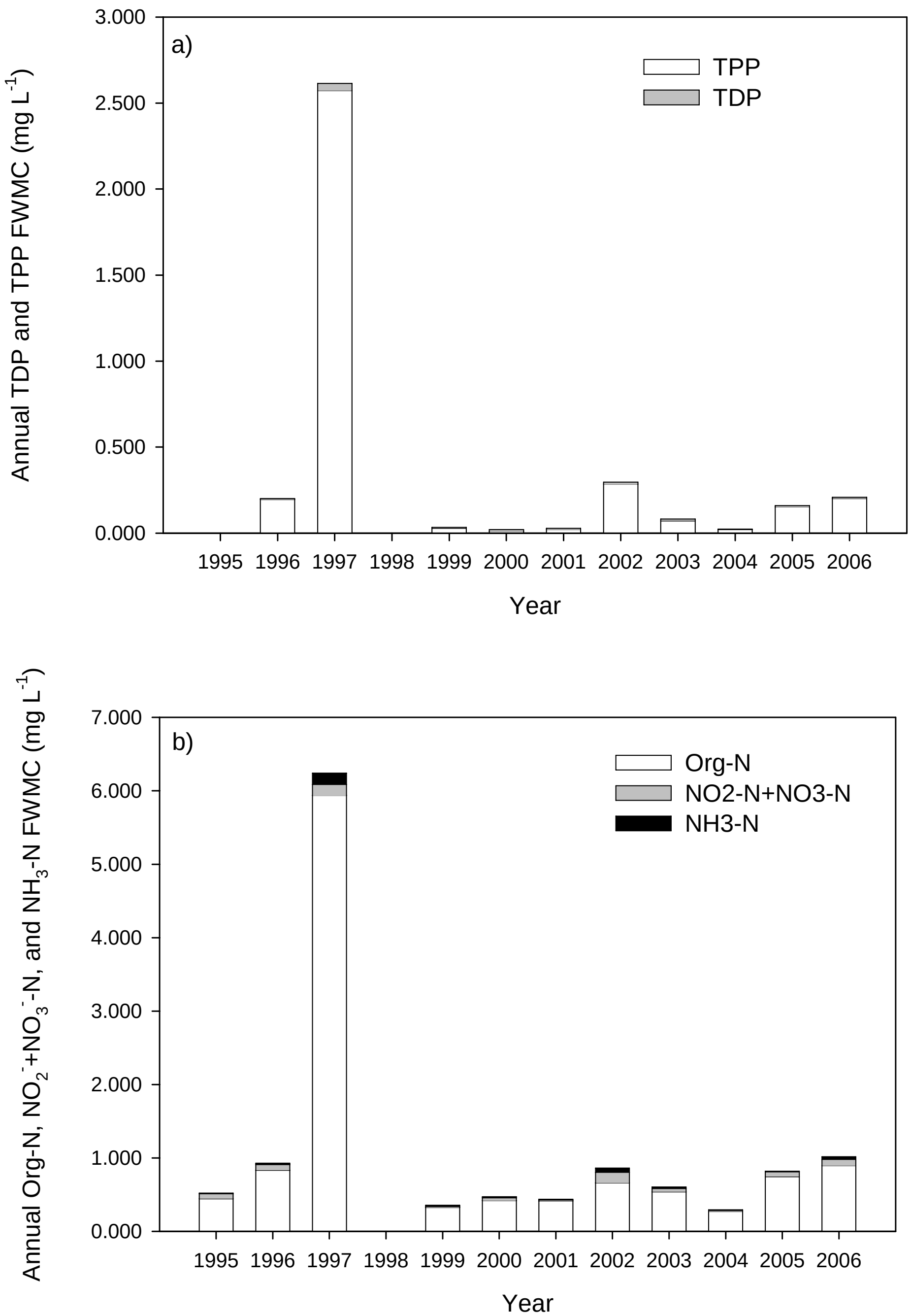


Figure 3.39. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Trout Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.25. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Trout Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg{\cdot}L^{-1}$)	Minimum ($mg{\cdot}L^{-1}$)	Year	Maximum ($mg{\cdot}L^{-1}$)	Year
TP	0.082	0.020	2000	2.614	1997
TDP	0.008	0.004	2004	0.041	1997
TPP	0.112	0.011	2000	2.573	1997
TDP/TP	0.093	0.016	1997	0.453	2000
TN	0.538	0.293	2004	6.248	1997
Org N	0.479	0.274	2004	5.938	1997
NH_3-N	0.014	0.007	1995	0.157	1997
$NO_2^-+NO_3^-$-N	0.042	0.011	2004	0.147	1997
DIN/TN	0.100	0.049	1997	0.233	2002
Annual stream volume (hm^3)	13.6	1.8	2000	74.0	2005

High Agricultural Intensity Watersheds (Buffalo, Haynes, Ray, Renwick, Strawberry, Stretton, Threehills, Wabash)

Buffalo Creek appeared to have increasing P and N FWMCs with time (Figure 3.40, Table 3.26). Interestingly, the agricultural intensity metric appeared to decrease in 2006 as a result of lower chemical and fertilizer sales percentiles; manure production did not appear to change among the Census years (Chapter 2: Figures 2.23 to 2.26). A statistical trend analysis would be required to confirm these observations.

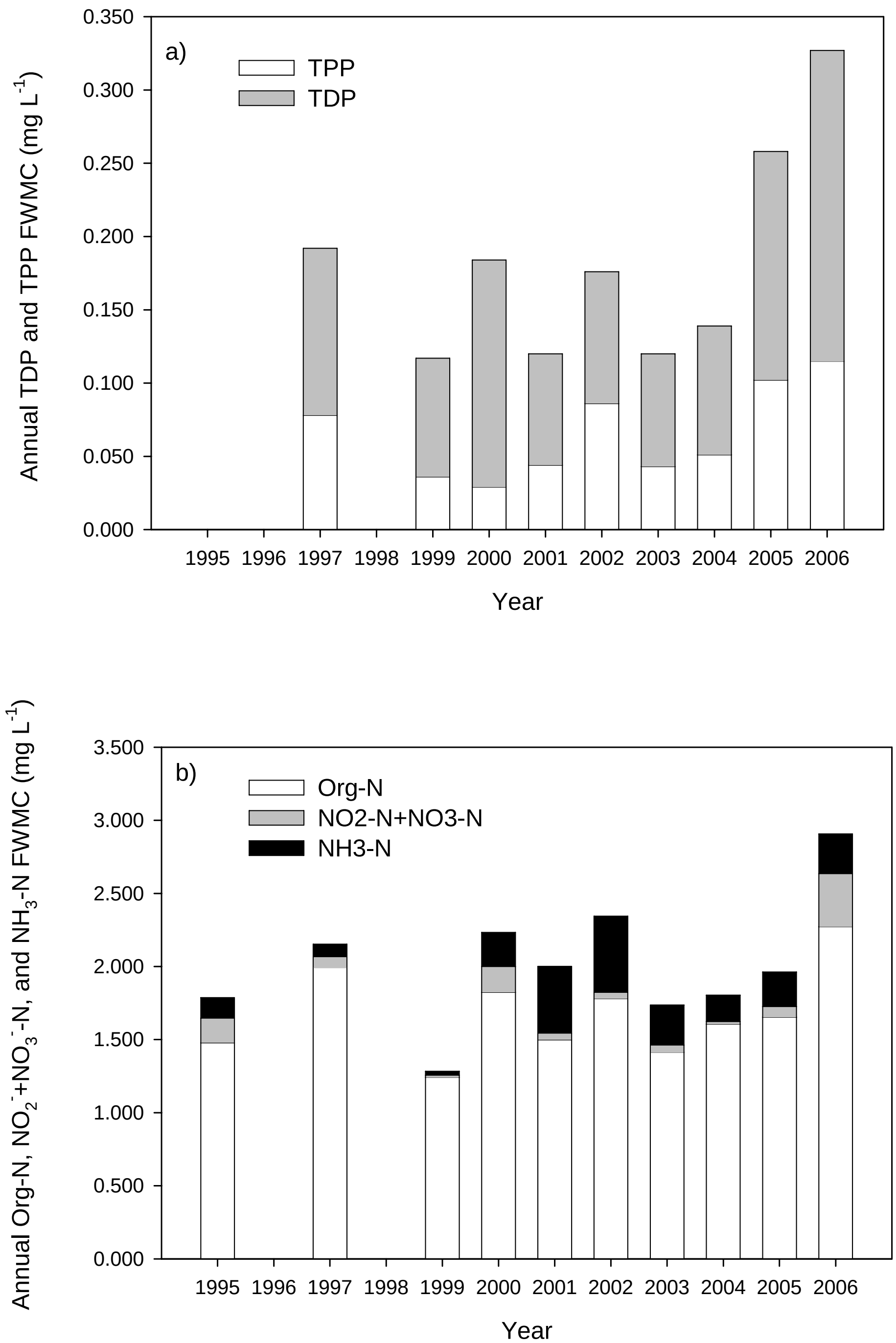


Figure 3.40. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Buffalo Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.26. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Buffalo Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg^{\cdot}L^{-1}$)	Minimum ($mg^{\cdot}L^{-1}$)	Year	Maximum ($mg^{\cdot}L^{-1}$)	Year
TP	0.180	0.117	1999	0.327	2006
TDP	0.090	0.076	2001	0.212	2006
TPP	0.051	0.029	2000	0.115	2006
TDP/TP	0.502	0.514	2002	0.841	2000
TN	1.842	1.284	1999	2.906	2006
Org N	1.606	1.243	1999	2.273	2006
NH_3-N	0.159	0.026	1999	0.520	2002
NO_2^-+NO_3^--N	0.137	0.014	1999	0.361	2006
DIN/TN	0.201	0.032	1999	0.251	2001
Annual stream volume (hm^3)	3.4	1.7	2002	11.0	1997

Temporal patterns in TP and TN FWMCs were not observed in Haynes Creek (Figure 3.41, Table 3.27), Ray Creek (Figure 3.42, Table 3.28), Strawberry Creek (Figure 3.43, Table 3.29) or Threehills Creek (Figure 3.44, Table 3.30). Interannual values were quite variable and did not correspond to changes in flow (e.g., Haynes Creek; Chapter 2, Figure 2.6) or apparent changes in agricultural intensity (e.g., Ray Creek; Chapter 2, Figure 2.23).

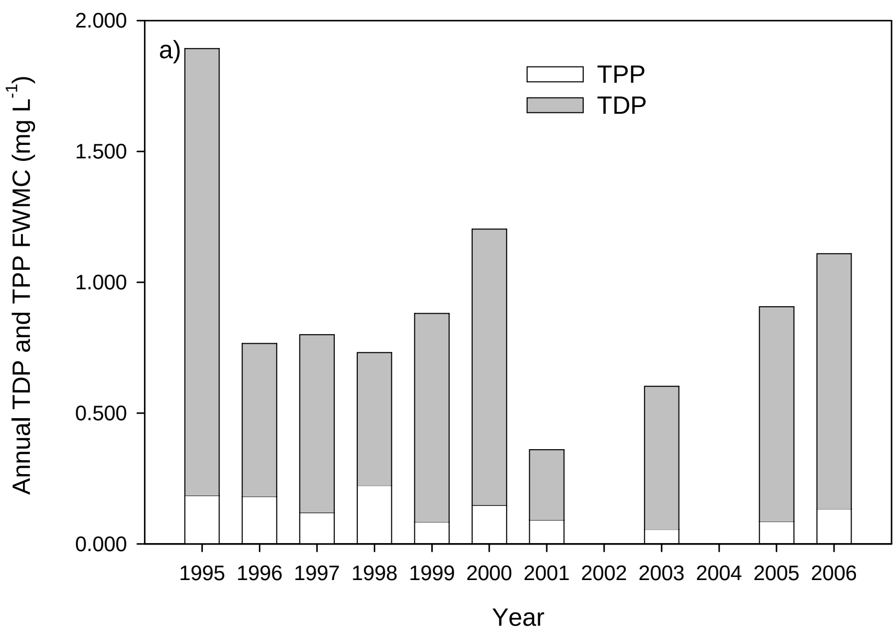


Figure 3.41. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Haynes Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

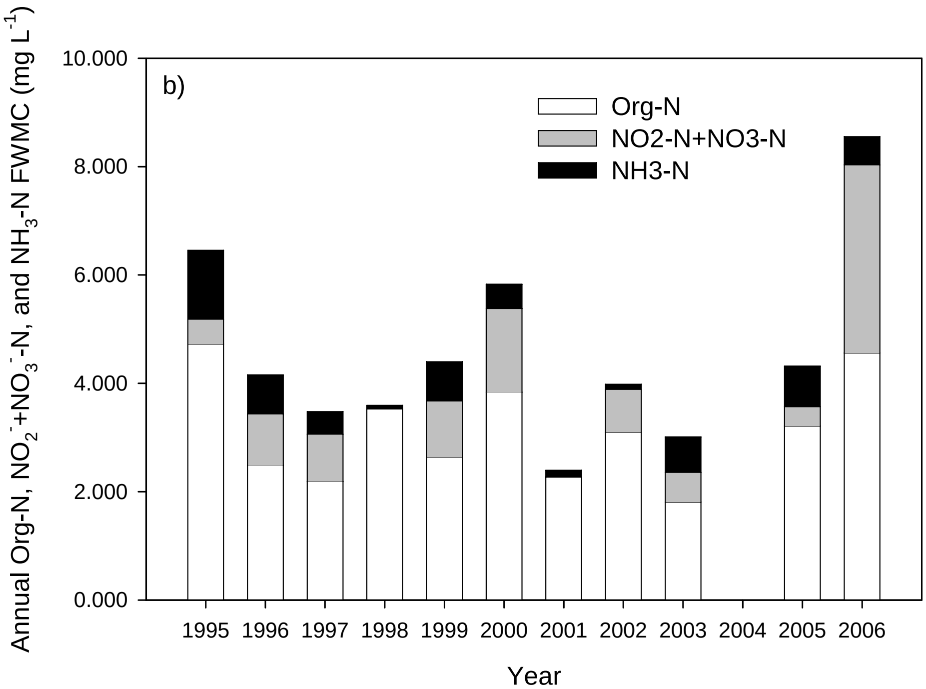


Figure 3.41 cont. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Haynes Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.27. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Haynes Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg \cdot L^{-1}$)	Minimum ($mg \cdot L^{-1}$)	Year	Maximum ($mg \cdot L^{-1}$)	Year
TP	0.799	0.360	2001	1.893	1995
TDP	0.738	0.269	2001	1.708	1995
TPP	0.127	0.056	2003	0.224	1998
TDP/TP	0.924	0.694	1998	0.907	2003
TN	4.159	2.392	2001	8.589	2006
Org N	3.098	1.806	2003	4.723	1995
NH_3-N	0.518	0.061	1998	1.275	1995
NO_2^-+NO_3^--N	0.788	0.010	2001	3.472	2006
DIN/TN	0.368	0.020	1998	0.465	2006
Annual stream volume (hm^3)	1.6	0.06	2001	5.6	1996

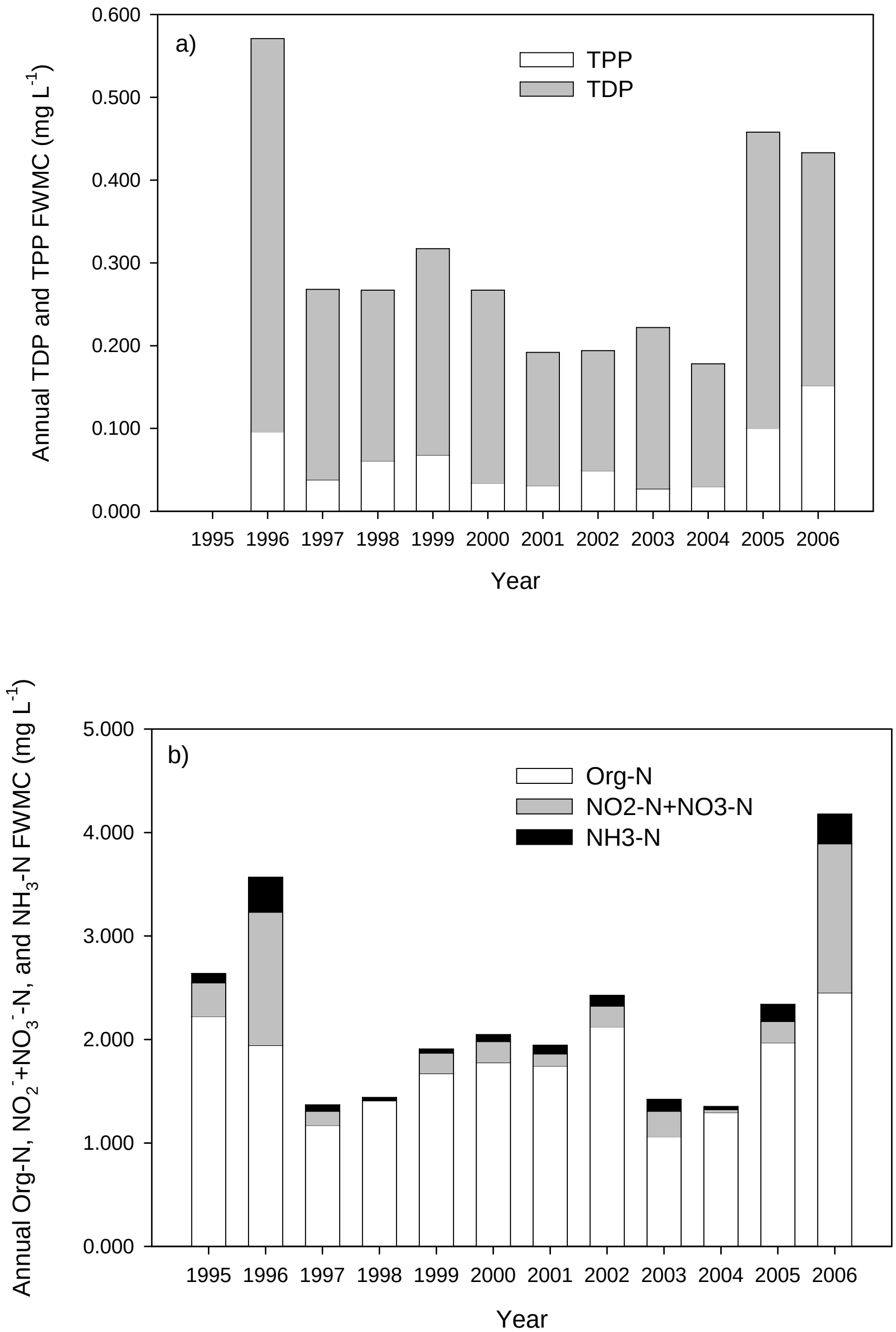


Figure 3.42. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Ray Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.28. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Ray Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg^{\cdot}L^{-1}$)	Minimum ($mg^{\cdot}L^{-1}$)	Year	Maximum ($mg^{\cdot}L^{-1}$)	Year
TP	0.268	0.178	2004	0.571	1996
TDP	0.230	0.145	2002	0.475	1996
TPP	0.049	0.027	2003	0.152	2006
TDP/TP	0.860	0.649	2006	0.878	2003
TN	1.995	1.353	2004	4.188	2006
Org N	1.760	1.062	2003	2.451	2006
NH_3-N	0.087	0.028	1998	0.337	1996
NO_2^-+NO_3^--N	0.201	0.005	1998	1.439	2006
DIN/TN	0.139	0.023	1998	0.455	1996
Annual stream volume (hm^3)	1.6	0.2	2001 and 2002	2.4	1997

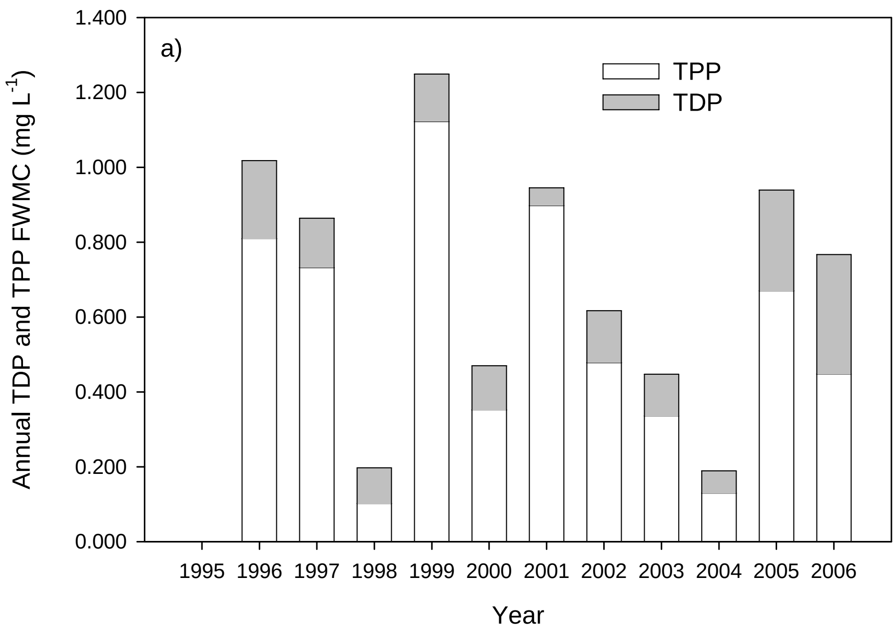


Figure 3.43. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Strawberry Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

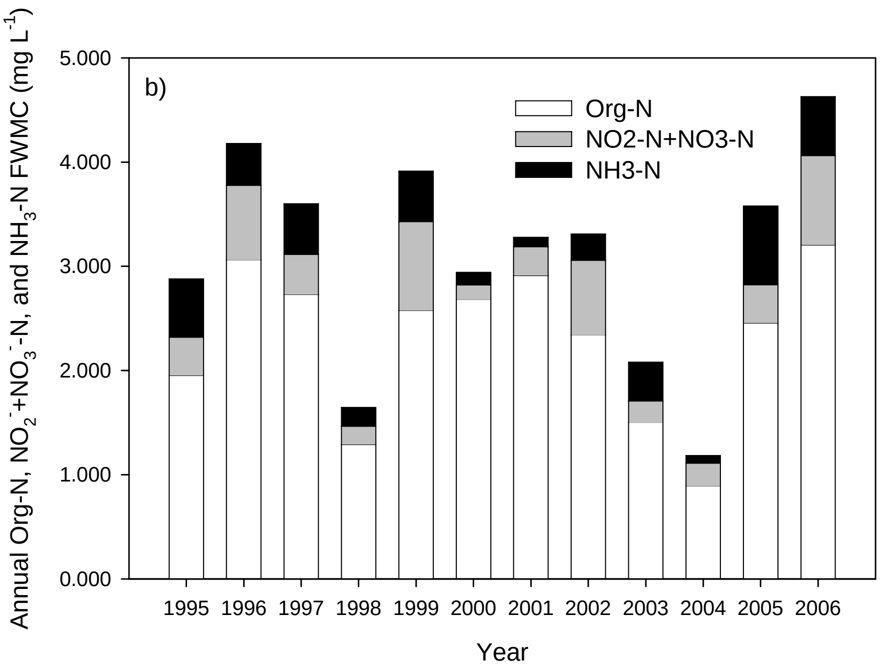
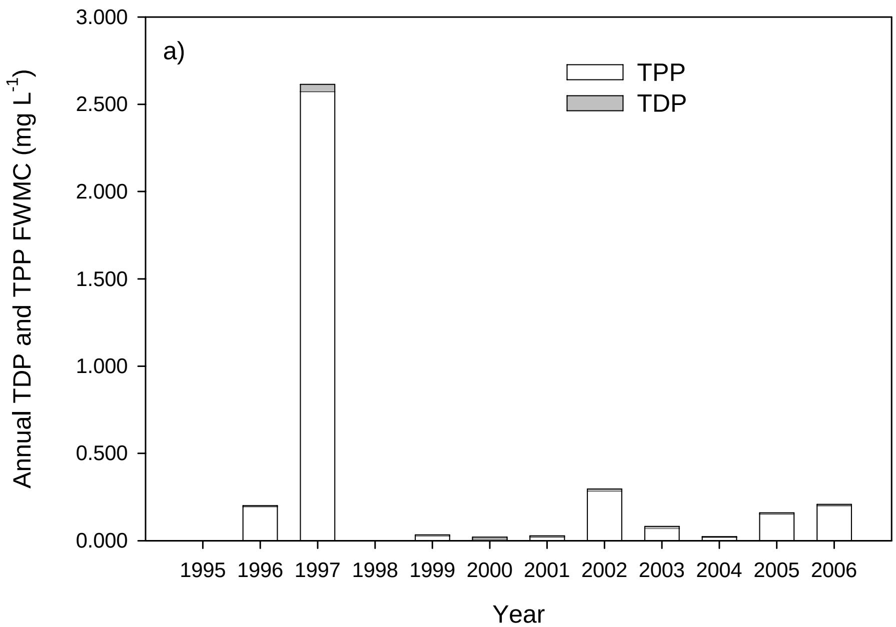


Figure 3.43 cont. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Strawberry Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.29. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Strawberry Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg·L^{-1}$)	Minimum ($mg·L^{-1}$)	Year	Maximum ($mg·L^{-1}$)	Year
TP	0.692	0.189	2004	1.249	1999
TDP	0.127	0.047	2001	0.319	2006
TPP	0.478	0.102	1998	1.122	1999
TDP/TP	0.184	0.049	2001	0.482	1998
TN	3.296	1.186	2004	4.628	2006
Org N	2.516	0.894	2004	3.203	2006
NH_3-N	0.387	0.075	2004	0.756	2005
NO_2^-+NO_3^--N	0.367	0.136	2000	0.859	2006
DIN/TN	0.271	0.088	2000	0.341	1999
Annual stream volume (hm^3)	12.3	5.3	2004	54.3	2000



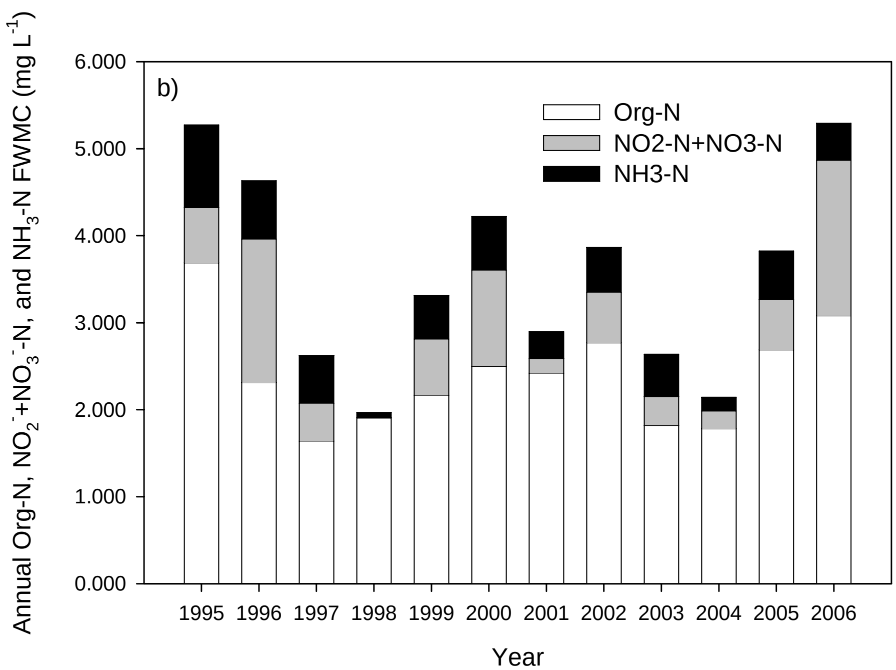
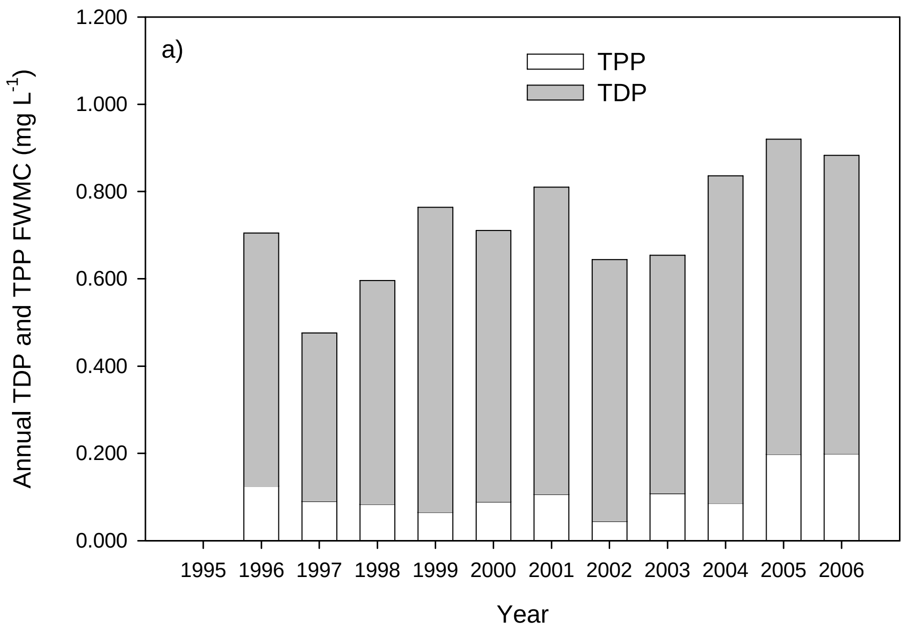


Figure 3.44. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Threehills Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.30. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Threehills Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg·L^{-1}$)	Minimum ($mg·L^{-1}$)	Year	Maximum ($mg·L^{-1}$)	Year
TP	0.593	0.278	1998	1.188	1995
TDP	0.443	0.180	1998	0.792	2005
TPP	0.102	0.074	1999	0.212	1999
TDP/TP	0.747	0.649	1998	0.891	1999
TN	3.571	1.969	1998	5.301	2006
Org N	2.368	1.639	1997	3.688	1995
NH_3-N	0.505	0.058	1998	0.952	1995
NO_2^-+NO_3^--N	0.579	0.006	1998	1.787	2006
DIN/TN	0.306	0.033	1998	0.500	1996
Annual stream volume (hm^3)	3.0	0.4	2001	8.4	1997

An increase in both P and N FWMCs with time was observed in Renwick Creek although the ratios of total P and N in the dissolved fraction did not appear to change (Figure 3.45, Table 3.31). The increase did not appear to be related to flow or Census of Agriculture data as metric percentiles remained similar from 1996 through 2006. A statistical trend analysis would be required to confirm the observations. Interestingly, the annual TDP FWMC was one of the top three values in 2001; however, the stream was only sampled in March and April of 2001. The annual FWMC in 2001 was likely influenced by sampling regime as dissolved nutrients were often highest in the spring with runoff and precipitation events. Although staff collecting samples in the watershed hadn't noticed any changes, it is possible that there may have been non-agricultural development (e.g., road construction that altered drainage or oil and gas activity) in the watershed upstream of the sampling site (out of site of the sampler). Ground truthing would be required to confirm the presence of non-agricultural activities in Renwick Creek in order to fully understand the changes in nutrient concentrations with time.



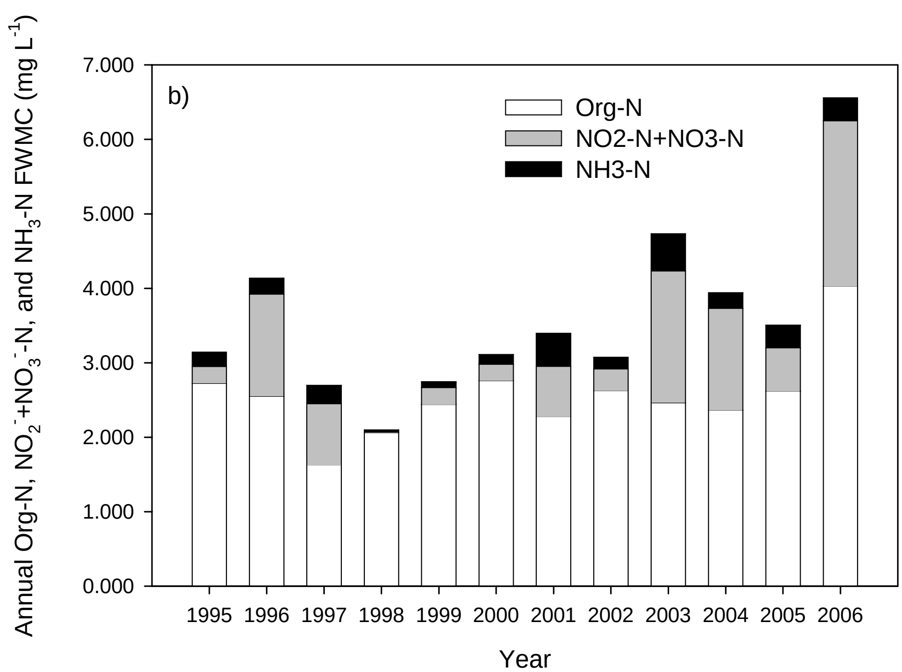


Figure 3.45. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Renwick Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.31. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Renwick Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg \cdot L^{-1}$)	Minimum ($mg \cdot L^{-1}$)	Year	Maximum ($mg \cdot L^{-1}$)	Year
TP	0.708	0.476	1997	0.920	2005
TDP	0.622	0.386	1997	0.750	2004
TPP	0.090	0.044	2002	0.199	2006
TDP/TP	0.878	0.775	2006	0.932	2002
TN	3.272	2.097	1998	6.566	2006
Org N	2.508	1.632	1997	4.032	2006
NH_3-N	0.213	0.029	1998	0.500	2003
NO_2^-+NO_3^--N	0.626	0.008	1998	2.217	2006
DIN/TN	0.290	0.017	1998	0.479	2003
Annual stream volume (hm^3)	0.7	0.03	2002	3.6	1997

There were no observable temporal trends in P or N FWMCs from 1995 to 2006 in Stretton Creek; however, temporal patterns were difficult to assess as four years of data were not available (2000-2003) (Figure 3.46, Table 3.32). The stream was dry for three of the years and there was not enough data for FLUX to calculate the FMWC or load in 2000. The watershed does not lend itself to an ideal location for examining trends in water quality as a result of intermittent flow.

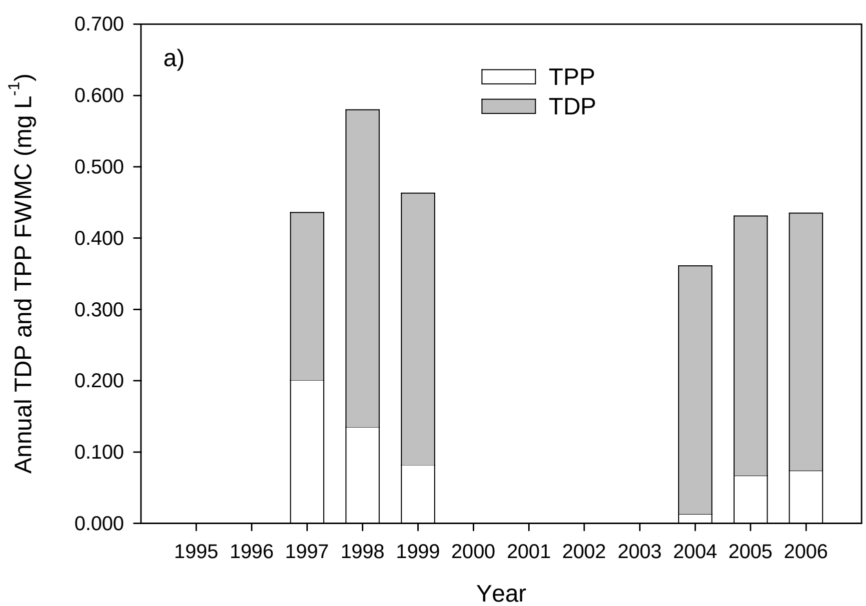


Figure 3.46. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Stretton Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

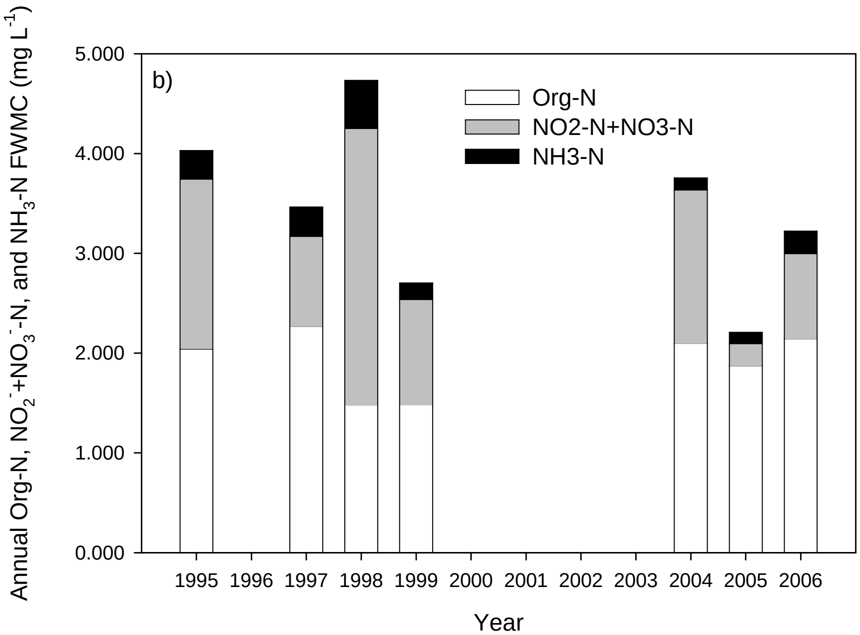


Figure 3.46 cont. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Stretton Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.32. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Stretton Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg\cdot L^{-1}$)	Minimum ($mg\cdot L^{-1}$)	Year	Maximum ($mg\cdot L^{-1}$)	Year
TP	0.436	0.361	2004	0.580	1998
TDP	0.362	0.235	1997	0.445	1998
TPP	0.078	0.013	2004	0.201	1997
TDP/TP	0.831	0.540	1997	0.964	2004
TN	3.477	2.209	2005	4.760	1998
Org N	2.039	1.479	1998	2.270	1997
NH_3-N	0.226	0.114	2005	0.483	1998
NO_2^-+NO_3^--N	1.052	0.221	2005	2.772	1998
DIN/TN	0.441	0.152	2005	0.684	1998
Annual stream volume (hm^3)	1.5	0.2	1998	2.3	2006

There appeared to be an increasing trend in TP and TN FWMCs and in the ratios of TDP/TP and DIN/TN with time in Wabash Creek (Figure 3.47). Again, a statistical trend analysis would be required to support the observation. Interestingly, the ratios of total nutrients in the dissolved fractions were highest in 2005 when annual flow volume was also highest. The median annual TP FWMC was also highest in 2005 and primarily consisted of TDP (Table 3.33). Total dissolved P also consisted of the majority of TP in 2002 and 2003, while the majority of TP was measured in the particulate fraction in the remaining years of the study, particularly in the first few years of monitoring.

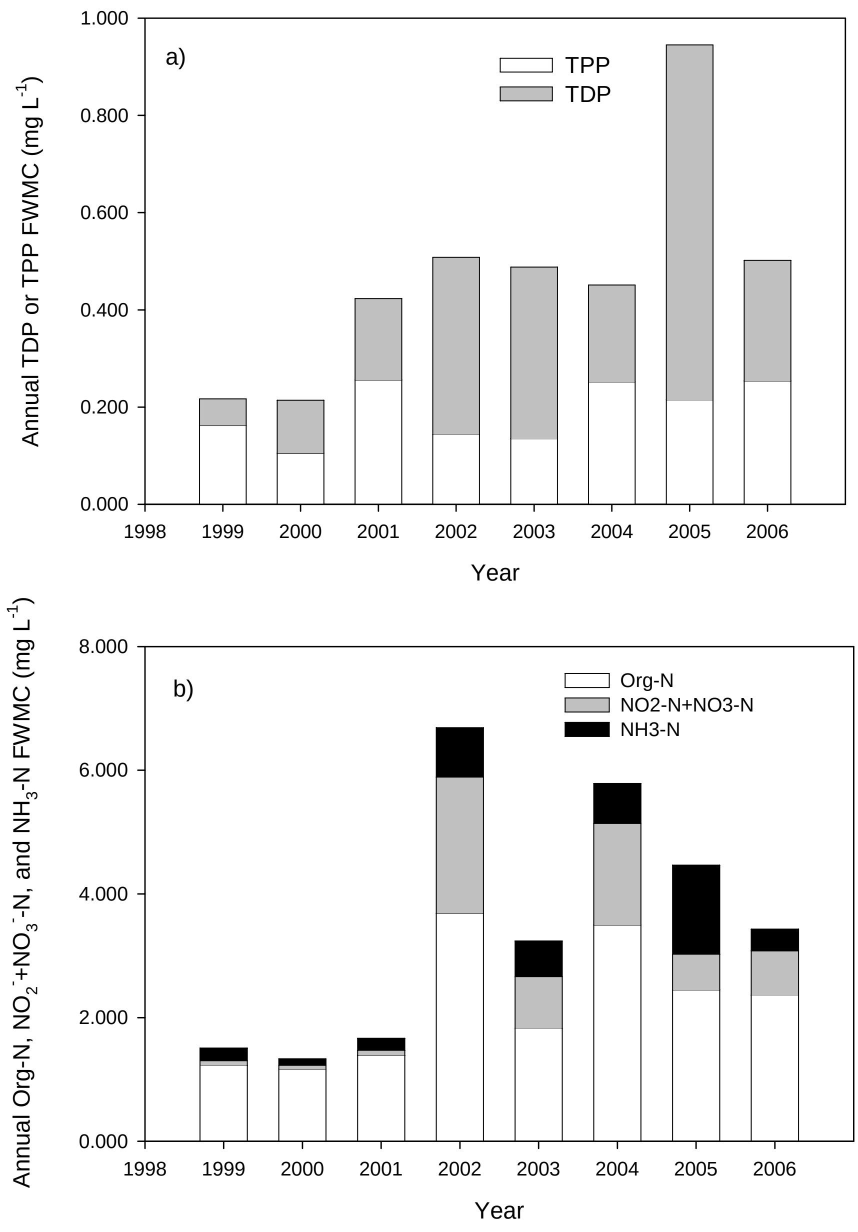


Figure 3.47. Annual TDP and TPP (a) and Org N, NO_2^{-}+NO_3^{-}-N, and NH_3-N (b) FWMCs in Wabash Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^{-}+NO_3^{-}-N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.33. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Wabash Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg·L^{-1}$)	Minimum ($mg·L^{-1}$)	Year	Maximum ($mg·L^{-1}$)	Year
TP	0.470	0.214	2000	0.945	2005
TDP	0.223	0.055	1999	0.730	2005
TPP	0.188	0.105	2000	0.256	2001
TDP/TP	0.475	0.254	1999	0.773	2005
TN	3.336	1.335	2000	6.708	2002
Org N	2.095	1.166	2000	3.683	2002
NH_3-N	0.464	0.105	2000	1.440	2005
NO_2^-+NO_3^--N	0.646	0.062	2000	2.207	2002
DIN/TN	0.348	0.125	2000	0.451	2005
Annual stream volume (hm^3)	1.5	0.06	2001	7.7	2005

Irrigated Watersheds (Battersea Drain, Crowfoot Creek, New West Coulee)

Initially, there appeared to be a temporal trend in annual TP and TN FWMCs (Figure 3.48) in Battersea Drain; however, the increasing pattern was driven by FWMCs in 2002 (0.609 $mg·L^{-1}$) and 2005 (1.432 $mg·L^{-1}$). Similarly, the fractions of P and N FWMCs were highest in 2005 (Org N and NO_2^-+NO_3^--N in 2002), with the lowest concentrations measured in 2000 (Table 3.34). Total annual flow volume was highest in 2005 (13.4 hm^3) but was not that much higher than the median annual flow volume between 1998 and 2006 (10.4 hm^3). It is possible that peaks in flow in certain months in 2002 and 2005 may have contributed to the much higher annual TP and TN FWMCs.

The ratio of TDP/TP and DIN/TN appeared to increase with time in Battersea Drain although a statistical trend analysis would be required to determine the significance of the increasing trend in TDP/TP and DIN/TN. The ratio of TP in the dissolved form was highest in 2002 (0.68) and 2005 (0.72), indicating that runoff and the resulting increased particulate concentrations were not the main drivers in the higher TP FWMC. Much of the TP was in particulate form in 1999 (0.07); there was no evidence of a heavy rain event associated with the high ratio of TPP in Battersea Drain that year (Donahue 2000). Although no precipitation event was observed in August of 1999 when the main portion of TPP was measured, irrigation water may have played a role in carrying particulate matter to the drain. The drain has also been channelized to straighten it out, which may have contributed increased sediment and particulate movement, though the exact dates are not known. Even with the data point removed for 1999, annual TDP/TP still appeared to increase. Although a high TPP ratio was observed in 1999, DIN was still higher that year than in several other years. The differing patterns in the ratios of dissolved P and N may have been a result of N transformations in agricultural soils before movement to the drain occurred or during in-stream processes. Since flow did not appear to be a major influence, the higher ratios of TP and TN in the dissolved fractions in 2002 and 2005 may have been driven by point sources (e.g., overflow of a catch basin and direct cattle access). Noteworthy, Battersea

Drain had the highest overall agricultural intensity rank, and fertilizer and chemical expenses percentiles in 1996 and 2006 of the three irrigated watersheds as well as the highest manure production percentile all three years. The source water for Battersea Drain comes from the Picture Butte Reservoir. It is unlikely that the source water flowing through the drain changed with time; however, the heavy rains in 2002 and 2005 may have contributed to degraded source water quality which subsequently flowed through the drain.

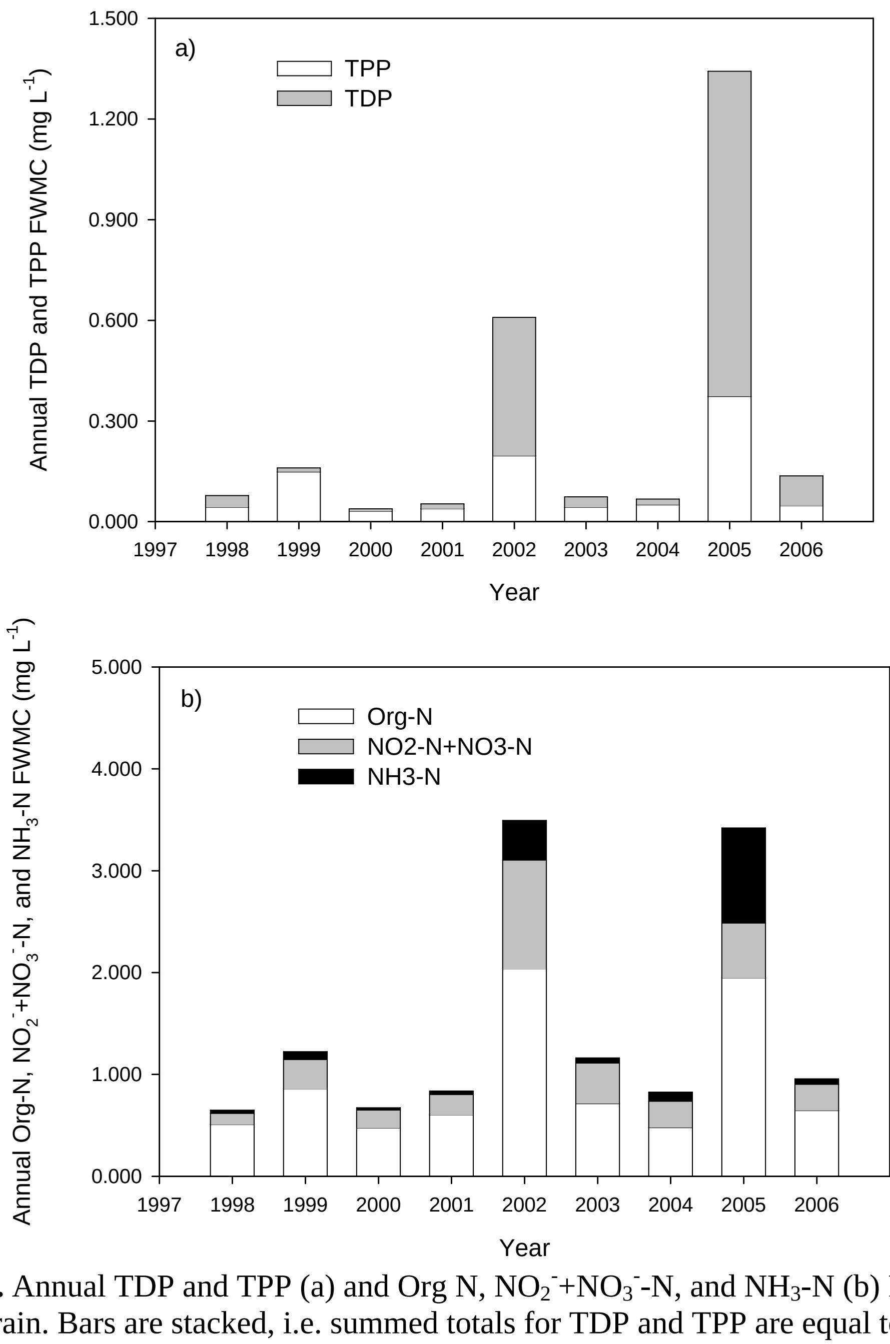
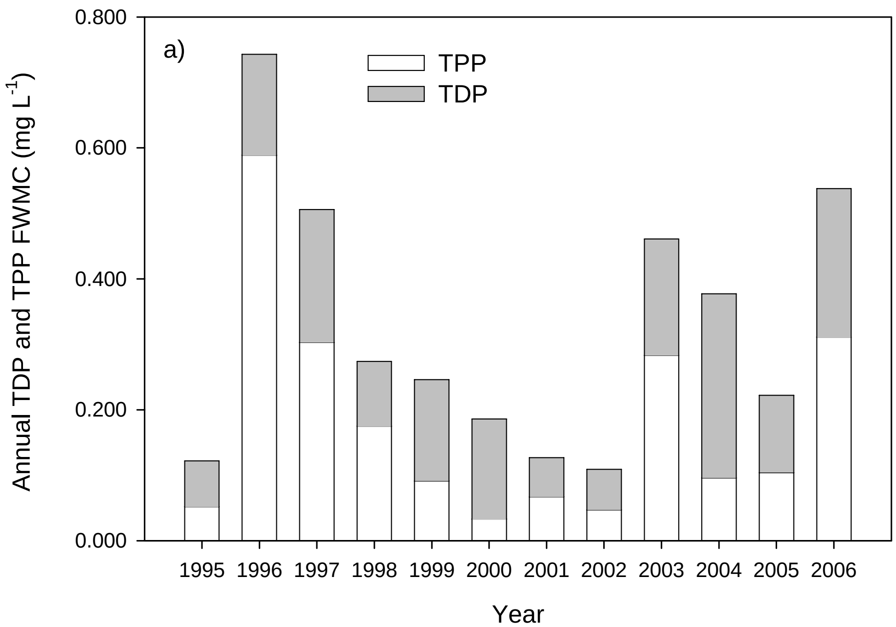


Figure 3.48. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Battersea Drain. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.34. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Battersea Drain. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg^{.}L^{-1}$)	Minimum ($mg^{.}L^{-1}$)	Year	Maximum ($mg^{.}L^{-1}$)	Year
TP	0.078	0.038	2000	1.342	2005
TDP	0.031	0.007	2000	0.969	2005
TPP	0.048	0.031	2000	0.373	2005
TDP/TP	0.396	0.073	1999	0.722	2005
TN	0.960	0.650	1998	3.498	2002
Org N	0.645	0.475	2000	2.036	2002
NH_3-N	0.055	0.021	2000	0.934	2005
NO_2^-+NO_3^--N	0.258	0.108	1998	1.067	2002
DIN/TN	0.325	0.216	1998	0.430	2005
Annual stream volume (hm^3)	10.4	0.3	1998	13.4	2005

Annual P and N FWMCs appeared to decrease with time (from 1996 to 2002) in Crowfoot Creek (Figure 3.49). However, concentrations increased in 2003 and did not exhibit any temporal patterns in the latter years of monitoring. A high ratio of TPP was measured in 1996 through 1998, 2003, and 2006 (TDP/TP between 0.21 and 0.42). Higher peaks in spring flow were recorded in 1996 and 1997 (the highest two years in the monitoring period) and also in 2003 and 2006. Particulate P FWMCs in these years were likely a result of increased runoff and movement of bound P from the soil to the stream. Interestingly, the majority of TP in 2004, which was the highest annual TP FWMC recorded in Crowfoot Creek, was in the dissolved fraction (TDP/TP =0.75). Agricultural practices may have influenced the nutrient FWMCs in the watershed in 2004. It is possible that research conducted in Crowfoot Creek (Ontkean et al 2003) may have increased producer awareness and the subsequent number of beneficial management practices implemented in the watershed.



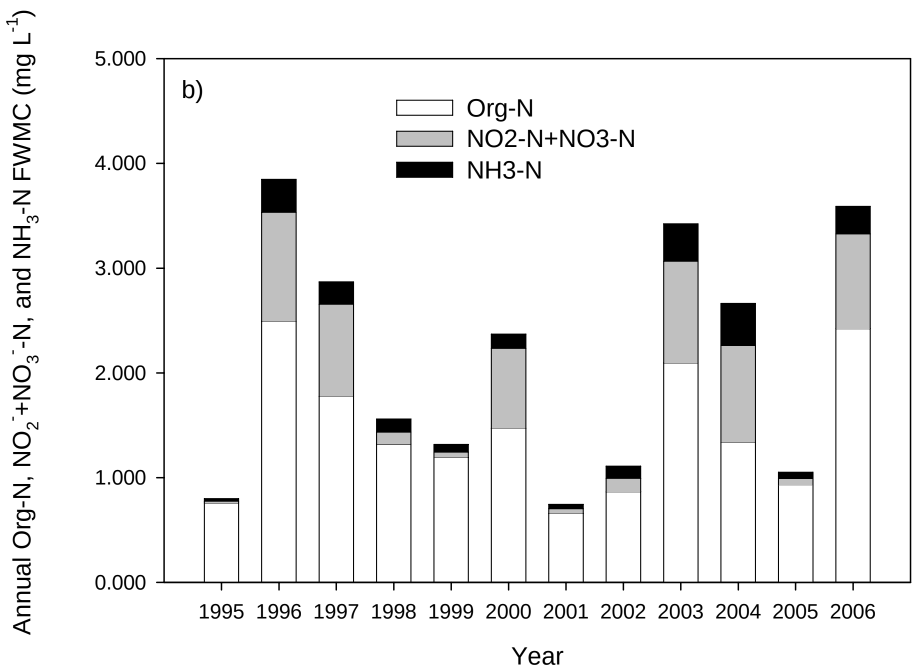


Figure 3.49. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in Crowfoot Creek. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.35. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in Crowfoot Creek. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg \cdot L^{-1}$)	Minimum ($mg \cdot L^{-1}$)	Year	Maximum ($mg \cdot L^{-1}$)	Year
TP	0.359	0.109	2002	0.742	1996
TDP	0.153	0.060	2001	0.281	2004
TPP	0.100	0.033	2000	0.589	1996
TDP/TP	0.591	0.207	1996	0.822	2000
TN	1.966	0.744	2001	3.848	1996
Org N	1.329	0.660	2001	2.492	1996
NH_3-N	0.132	0.025	1995	0.401	2004
NO_2^-+NO_3^--N	0.446	0.017	1995	1.041	1996
DIN/TN	0.273	0.053	1995	0.498	2004
Annual stream volume (hm^3)	32.3	19.1	2002	70.9	1997

No monotonic trend was observed in annual FWMCs with time for any forms of phosphorus in New West Coulee; however, TP FWMCs were slightly lower in 1999 and 2000 than latter years (Figure 2.50(a)). Annual TN FWMCs in New West Coulee initially appeared to increase between 1999 and 2006 but were driven by high concentrations in 2002 and 2003 (Figure 2.50(b)).

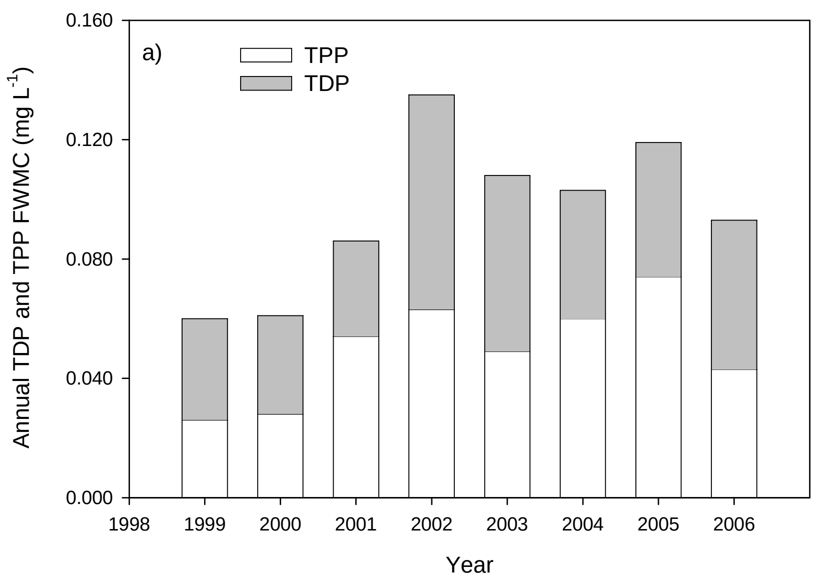


Figure 3.50. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in New West Coulee. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

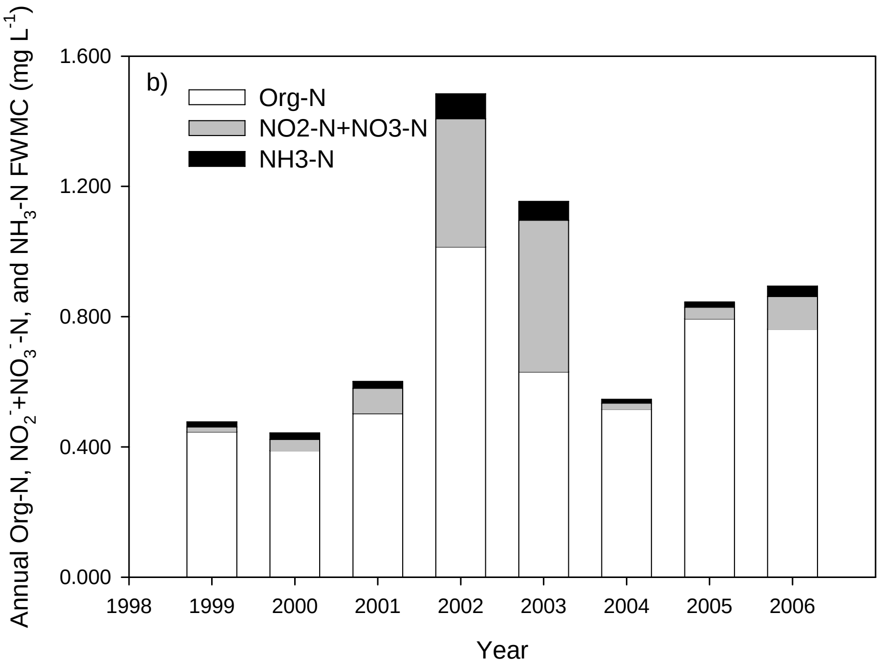


Figure 3.50 cont. Annual TDP and TPP (a) and Org N, NO_2^-+NO_3^--N, and NH_3-N (b) FWMCs in New West Coulee. Bars are stacked, i.e. summed totals for TDP and TPP are equal to TP and Org N, NO_2^-+NO_3^--N, and NH_3-N are equal to TN FWMCs for the year.

Table 3.36. Median (1999 to 2006), minimum, and maximum P and N FWMCs and median (1999 to 2006), minimum, and maximum annual stream volume (March to October) in New West Coulee. Year represents the year the minimum or maximum FWMC or flow occurred.

	Median ($mg \cdot L^{-1}$)	Minimum ($mg \cdot L^{-1}$)	Year	Maximum ($mg \cdot L^{-1}$)	Year
TP	0.098	0.060	1999	0.135	2002
TDP	0.044	0.032	2001	0.072	2002
TPP	0.052	0.026	1999	0.074	2005
TDP/TP	0.447	0.376	2001	0.562	1999
TN	0.724	0.443	2000	1.487	2002
Org N	0.573	0.389	2000	1.014	2002
NH_3-N	0.020	0.012	2004	0.076	2002
NO_2^-+NO_3^--N	0.057	0.015	1999	0.466	2003
DIN/TN	0.134	0.055	2004	0.454	2003
Annual stream volume (hm^3)	22.0	15.9	2006	29.6	2000

Seasonality of nutrient FWMC by agricultural intensity. Seasonality was examined for all forms of P and N FWMCs to assess whether nutrient concentrations varied during different times of the monitoring period (i.e., spring, summer, and fall). Box plots of monthly nutrient FWMCs (1999 to 2006) can be found in Appendix 6 for each agricultural intensity category.

Seasonality differed among agricultural intensity categories and nutrient forms. Low agricultural intensity streams showed no seasonal trend for phosphorus or nitrogen FWMCs at a significance level of $p<0.10$ (Table 3.37). In the moderate agricultural intensity watershed category, all parameters but monthly TDP FWMCs were not significantly different at $p<0.10$. While monthly TPP FWMCs in the high agricultural intensity streams were not significantly different ($p<0.1$), a significant difference was observed for monthly mean rank TP, TDP, and $NO_2^-+NO_3^-$-N FWMCs ($p<0.005$). Monthly mean rank TN, Org N, and NH_3-N FWMCs in the high agricultural intensity category were significantly different but at a lower level of significance (Table 3.37). Seasonality was not observed in any of the nutrient parameters in the irrigated watersheds with the exception of TN ($p<0.05$) and NH_3-N FWMCs ($p<0.10$). The lack of seasonal trends in the irrigated watersheds is likely a result of the controlled flows during the open water season.

Table 3.37. Kruskal-Wallis One-Way ANOVA (H) and Mann-Whitney significance (p) tests for seasonality of P and N FWMCs grouped by 1996 agricultural intensity categories.

Parameter	Low	Moderate	High	Irrigated
	(u, significance)			
TP	1.607, 0.978	6.539, 0.479	**22.936, 0.002**	2.937, 0.891
TDP	8.405, 0.298	**13.548, 0.060**	**22.719, 0.002**	1.791, 0.970
TPP	3.383, 0.847	3.106, 0.875	7.633, 0.366	3.706, 0.813
TN	2.849, 0.899	9.389, 0.226	**19.915, 0.006**	**14.687, 0.04**
Org N	2.040, 0.958	6.968, 0.432	**13.370, 0.064**	9.525, 0.217
$NO_2^-+NO_3^-$-N	9.825, 0.199	9.848, 0.197	**21.112, 0.004**	8.519, 0.289
NH_3-N	5.527, 0.596	7.863, 0.345	**15.657, 0.028**	**12.149, 0.096**

Note: **Bold** values are significant at $p<0.1$. The values represent the Kruskal-Wallis value and significance (H, p).

In the high agricultural intensity watersheds, N and P FWMCs were generally highest in the early spring (March and April) and lowest in the late spring (May/June) or early summer (July). Total dissolved P and NH_3-N FWMCs were significantly higher in March than in all other months of sampling, with $NO_2^-+NO_3^-$-N FWMCs significantly higher in March than months but April (Appendix 6: Figures A6.5 and A6.6). Organic N, TN, and TP FWMCs were still highest in March but were generally not significantly higher than FWMCs in the late summer (August) and fall (September/October) (Appendix 6: Figures A6.5 and A6.6). It is possible that the seasonality observed in the high agricultural intensity watersheds, the majority of which are located in the Parkland Ecoregion, is a result of increased nutrients in the soil from fall/spring manure application exceeding crop requirements. There may also be lower soil N and P levels in the summer as a result of increased crop requirements and uptake and less precipitation events to promote movement of nutrients via overland runoff. Snowmelt is also the dominant form of precipitation in the Parkland Ecoregion.

Nutrient FWMCs in moderate agricultural intensity watersheds were also highest in the early spring but were slightly different for TDP FWMCs. Mean rank TDP FWMCs were significantly higher in April than all other months of sampling but March, with FWMCs also higher in March than in October (Appendix 6: Figure A6.3). Again, mean rank FWMCs were not statistically different among months for the other nutrient parameters.

While FWMCs in the moderate agricultural intensity watersheds were highest in April, seasonal patterns for TN and NH_3-N in the irrigated watersheds were similar to the high agricultural intensity watersheds. Mean rank FWMCs were significantly higher in March than in all other months of sampling with the exception of April (Appendix 6: Figure A6.7 and A6.8). Increased soil N and P levels as a result of fall or spring manure application in excess of crop requirements may explain the seasonal trend observed in the irrigated watersheds; however, a seasonal pattern was not observed for P.

Mass Transport of Phosphorus and Nitrogen

Previous data collected through the CAESA agreement found that regardless of agricultural intensity, streams with high flows and low concentrations could contribute more mass than streams with low flows but high concentrations (Anderson et al. 1998b). Research outside of Alberta has reported similar findings where the magnitude of phosphorus loads were dependent on the magnitude and frequency of discharge and the flow pathway (Heathwaite and Dils 2000; Gentry et al. 2007). This association with stream flow has implications on the management of current and future development, specifically in areas of high runoff (Anderson et al. 1998b).

Nutrient loads were examined to assess potential downstream impacts from the AESA watersheds. Ecoregional characteristics were taken into consideration as nutrient loads are highly influenced by stream discharge. Furthermore, the frequency and timing of flood peaks and low flow events are often similar within a region as the area is often influenced by the same weather patterns.

Temporal loading in the AESA watersheds generally coincided with flow peaks: lower nutrient loading was observed during low flow periods; higher loading was measured during high flow periods. Detailed descriptions of temporal loading patterns for each watershed can be found in Appendix 7. Temporal patterns for a few streams have been described below.

Total P and TN loads in Grande Prairie Creek appeared to increase from 1999 through 2006 although a substantial drop in nutrient loads was observed in 2006 (Appendix 7: Figure A7.2). It appeared that TP and TN loads in Grande Prairie Creek were influenced by stream flow, sampling regime, and a possible change in agricultural intensity. Nitrogen and P loading were influenced by stream flow as low annual flow volumes coincided with low N and P loads in 2000 and 2006 and higher annual stream flow volume in 2002 and 2004 coincided with higher loading (Appendix 7: Table A7.2). Nutrient loads were not as high as would have been expected from flows in 2002. This was likely a result of under-sampling and misrepresentation of water quality in 2002. Only nine samples were collected in 2002 compared to the average 13 collected most years, and sampling occurred just before and just after peak flows occurred. Therefore, times of peak loading may not have be accurately captured in 2002. The annual TP and TN loads were lower in 2000 than all other years (Appendix 7: Figure A7.2, Table A7.23). The highest flows, and resulting loads, typically occurred in April or May in Grande Prairie Creek (Chapter 2: Results and Discussion, Hydrology- Grande Prairie Creek). However, peak flow in 2000 occurred in September with very low flows earlier on in the sampling season (March to August). Total N and P loads in September 2000 were much higher than the other months of sampling (March through August and October) and accounted for the majority of the annual load that year. The TP load in 2004 was interesting in that stream flow volumes in 2004 were not as high as previous years; however, the load was the highest observed (Appendix 7: Table A7.2). As in 2000, higher flows occurred in September in the watershed, which increased loading in the fall. Loading was also high in July as a result of precipitation events. High TN and TP loading in 2004 may also have been related to a change in agricultural intensity (Chapter 2: Table 2.11). However, it is not certain whether a rapid cause-effect relationship between increased agricultural intensity and increased loading would occur as there did not appear to be decreased loading in response to a decrease in agricultural intensity earlier in the monitoring period, and the manure production percentiles decreased with time (Chapter 2: Figure 2.24, Table 2.11).

Total phosphorus and TN loads appeared to decrease from 1995 through 2006 in Paddle River; however, this decrease may be attributed to the high loads observed in 1997 (Appendix 7: Table A7.5) and declining stream flow throughout the monitoring period (Appendix 7: Table A7.23). Although sampling was flow biased, there were only 6 samples collected in 1996 compared to 26 samples in 1997. As a result of under sampling, the data collected in 1996 may not be representative of the actual loading that occurred.

There appeared to be a slight declining trend in loading in Meadow Creek (Appendix 7: Figure A7.17); however, peak N and P loading in 1997 created the appearance of the decrease in loading. Although the highest loading occurred in 1997, flow volumes in 1997 were lower than those in 1995 and 1996. Furthermore, fewer samples were collected in 1995 and 1996 even though flow volumes were higher, resulting in inadequate assessment of peak flows. Any temporal trend was thus difficult to assess. Years with lower loading corresponded to years with low annual flow volumes.

The Battersea Drain appeared to have an increasing trend in TP and TN loading (Appendix 7: Figure A7.22). Flow patterns in 2002 and 2005 indicated prominent peaks even though they were not reflected in the annual flow volumes (Chapter 2: Figure 2.19, Tables 2.5, 2.6, and 2.7). These two peaks were related to higher than average amounts of precipitation (Chapter 2: Table 2.4). The Lethbridge Northern Irrigation District regulates the irrigation water but cannot control rainfall. Addition of water through precipitation combined with a lack of withdrawals (i.e., operators do not irrigate when it rains) likely resulted in the higher flows in 2002 and 2005. Generally, annual loads were lowest in 1998 and highest in 2005 (Appendix 7: Table A7.22). Annual stream flow volume steadily increased from 1999 to 2006 in Battersea Drain, and loads were likely influenced by this flow pattern. The increasing trend may also have been influenced by an increase in agricultural intensity (Chapter 2: Table 2.11) or a change in management practices within the watershed.

Export Coefficients for Phosphorus and Nitrogen in Dryland Watersheds

Phosphorus and N export coefficients were compared by watershed location within ecoregion areas and by agricultural intensity categories in addition to looking at their relationship with agricultural intensity metrics and seasonal trends. Exports permit the comparison of nutrient loads among watersheds of different sizes; however, they tend to be highly influenced by stream discharge as they are not normalized for flow like FWMCs. Exports are primarily a function of soil type, land use, landscape characteristics, and the amount, timing, and intensity of precipitation. Therefore, exports were examined among agricultural intensity and ecoregional categories to understand the various influences on nutrient exports in the small, agricultural watersheds. Relationships with agricultural intensity metrics were examined as the metrics can be used to link the type and intensity of agriculture in a watershed with changes in water quality. Lastly, seasonal patterns were examined to assess whether nutrient exports varied during different times of the monitoring period (i.e., spring, summer, and fall).

Total Phosphorus (TP) export coefficients. Total P export coefficients were most influenced by factors such as climate and landscape, which vary by Ecoregion, rather than agricultural intensity. Mean rank median annual TP export coefficients were significantly higher in the Boreal ($U=1315$, $p<0.005$) and Parkland ($U=282$, $p<0.005$) ecoregion areas than in the Grassland ecoregion (Figure 3.51). Total P export coefficients in the Continental divide appeared to be similar to those measured in the Boreal and Parkland ecoregion areas. There was no significant difference in mean rank TP export coefficients between the Boreal and Parkland ecoregion areas ($U=1825$, $p=0.11$). A significant difference was not observed among annual TP export coefficients when grouped by agricultural intensity ($H=1.236$, $p=0.539$, 2df) (Figure 3.52).

Median annual TP export coefficients differed among the dryland watersheds, with similarities among streams located in similar ecoregion areas (Figure 3.53). The highest median annual TP export coefficient was observed in Blindman Creek (0.214 $kg \cdot ha^{-1} yr^{-1}$), a moderate intensity watershed in the Boreal ecoregion area. The next three highest median annual TP export coefficients were also observed in streams located in the Boreal ecoregion area (Rose Creek, Strawberry Creek, and Kleskun Drain with exports of 0.197, 0.169, and 0.161 $kg \cdot ha^{-1} yr^{-1}$, respectively). These watersheds covered all of the dryland agricultural intensity categories (Rose Creek, low; Blindman Creek and Kleskun Drain, moderate; Strawberry Creek, high). Interestingly, the three AESA streams within the Grassland ecoregion area (Trout Creek, Meadow Creek, and Prairie Blood Coulee) had the lowest median annual TP export coefficients (0.020, 0.017, and 0.012 $kg \cdot ha^{-1} yr^{-1}$, respectively). Watersheds in the Parkland ecoregion area had TP export coefficients between 0.022 and 0.139 $kg \cdot ha^{-1} yr^{-1}$ (Buffalo, Haynes, Ray, Renwick, Stretton, Threehills, and Wabash Creeks). As discussed previously, all of the watersheds in the Parkland ecoregion were classified as high intensity watersheds based on the 1996 Census of Agriculture data (Anderson et al. 1999).

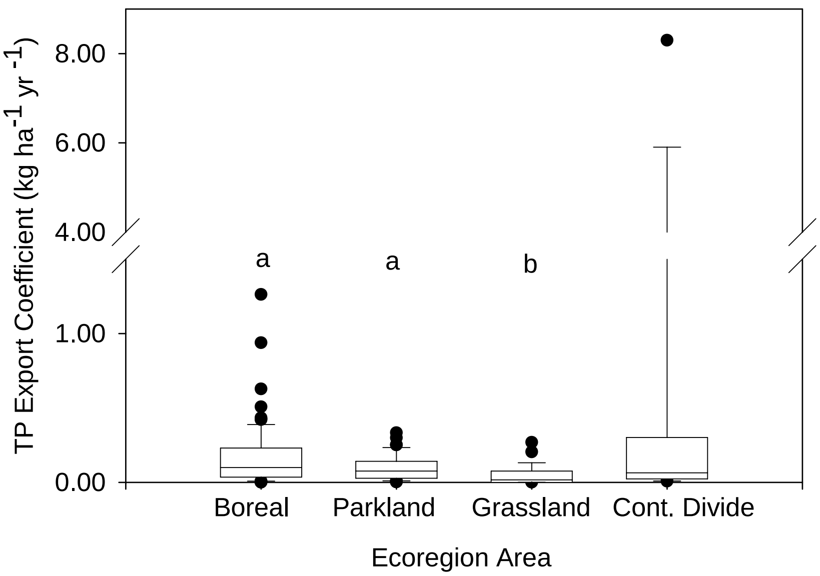


Figure 3.51. Box plots of median annual TP export coefficients (1999-2006) for the Boreal, Grassland, and Parkland ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the Grassland ecoregion area and other two ecoregion areas were statistically different at the $p<0.005$ significance level.

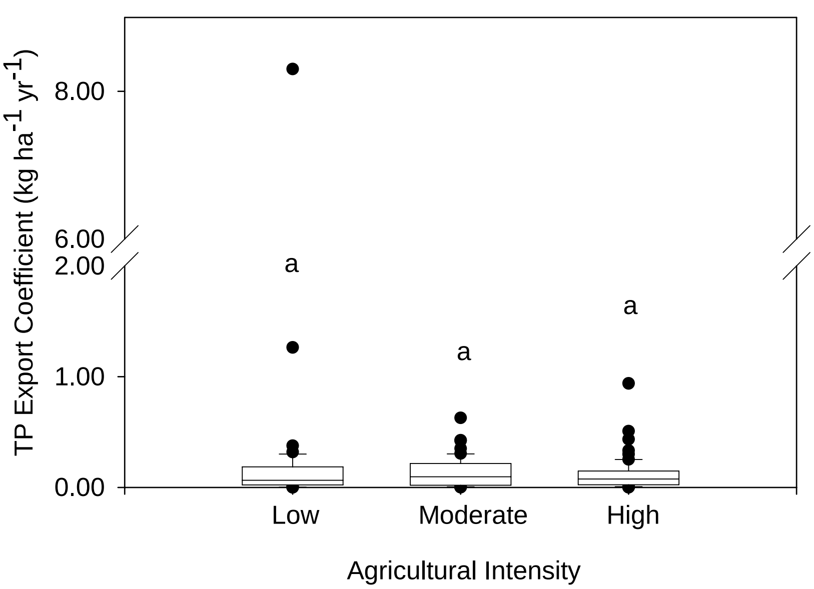


Figure 3.52. Box plots of median annual TP export coefficients (1999-2006) in the three dryland agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

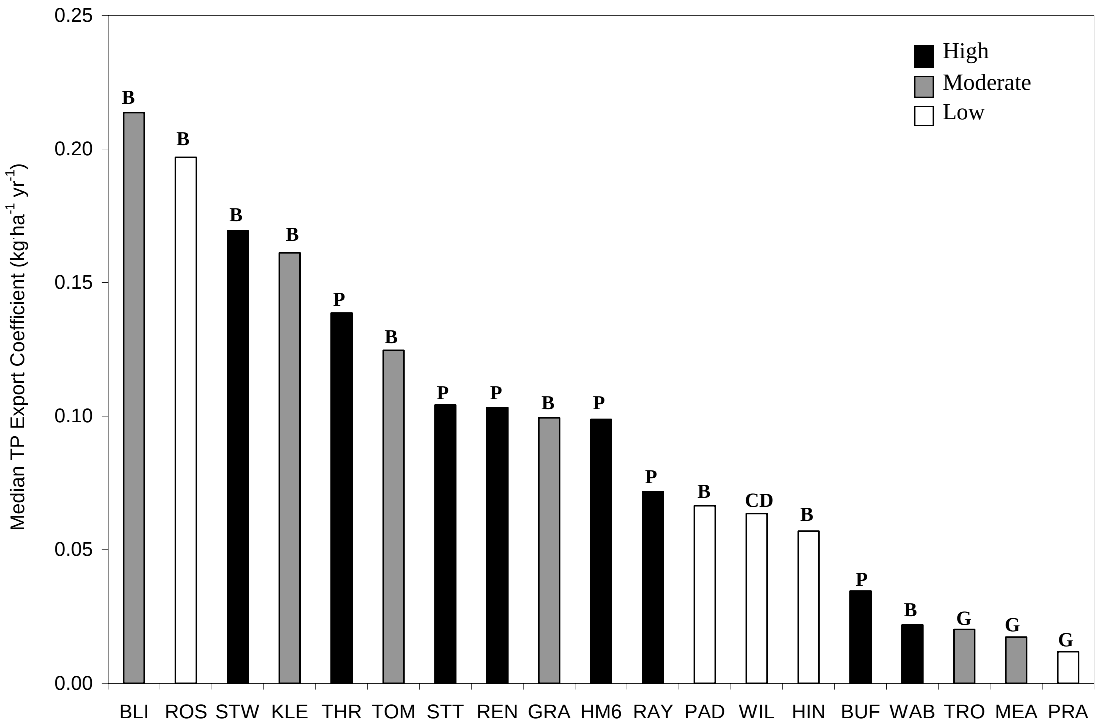


Figure 3.53. Median annual TP export coefficients for each dryland AESA watershed (1999-2006). Location by ecoregion area for each stream is illustrated as follows: Boreal (B), Parkland (P), and Grassland (G). For full stream names see Table 3.5 (pg.3-12).

Although median TP export coefficients appeared to be most influenced by ecoregional factors such as climate and topography, some exceptions were noted. A significant difference in TP export coefficients was not observed among streams located in the Parkland or Grassland ecoregion areas or among high agricultural intensity watersheds as tested by Kruskal-Wallis One-Way ANOVA ($p>0.05$) (Table 3.38). However, there were differences among streams within the Boreal ecoregion area ($p<0.05$) and within the low ($p<0.01$) and moderate ($p<0.005$) agricultural intensity categories.

Even though TP export coefficients of watersheds located in the Boreal ecoregion area were not all similar, agricultural intensity did not appear to be the dominant factor influencing these watersheds either. Mann-Whitney statistics run between individual streams located in the Boreal ecoregion area showed a significant difference between the three highest TP exporters (Blindman River, Rose Creek, and Strawberry Creek) and the two lowest exporters (Hines and Wabash Creeks) (Table 3.39). Kleskun Drain, Tomahawk Creek, and Grande Prairie Creek were not significantly different from the top three or bottom three exporters. Although the majority of streams with the highest exports in the Boreal ecoregion area were moderate agricultural intensity watersheds, the differences observed in TP export coefficients in the region were not influenced by agricultural intensity. Rose Creek, Paddle River, and Hines Creek were all low agricultural intensity watersheds with differing exports. Furthermore, no significant difference was found between low, moderate, and high agricultural intensity watersheds within the Boreal ecoregion area (H=2606, p=271, 2df).

Table 3.38. Kruskal-Wallis One-Way ANOVA statistics for median annual TP export coefficients (1999 to 2006) among dryland streams grouped by ecoregion area and by agricultural intensity.

Ecoregion Area/Agricultural Intensity	Kruskal-Wallis Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	19.463	**0.013**	8
Parkland	4.260	0.336	5
Grassland	0.218	0.165	2
Low	14.526	**0.006**	4
Moderate	12.112	**0.033**	5
High	11.422	0.121	7

Note: **Bold** values indicate a significant difference at $p<0.05$.

Wabash Creek had a much lower TP export coefficient than other streams in the Boreal ecoregion area and, as discussed later, a much lower ratio of TDP than other high agricultural intensity streams. The watershed is located north of Edmonton and is not in close proximity to the other AESA watersheds. Wabash Creek received a lower amount of annual precipitation than other watersheds in the Boreal ecoregion area (Chapter 2: Table 2.4), which is likely a result of lower snowfall amounts.

Furthermore, annual TP export coefficients from moderate agricultural intensity streams were grouped by location by ecoregion area: the Boreal and Grassland ecoregion areas. Mean rank annual TP export coefficients in Meadow Creek were significantly lower than the four remaining watersheds in the moderate agricultural intensity category (Table 3.40). Export coefficients were also lower in Trout Creek than those measured in Blindman River, Grande Prairie Creek, and Kleskun Drain.

Differences in TP export coefficients among the low agricultural intensity watersheds were likely a result of climatic and topographic variability across the group of streams. Annual TP export coefficients were significantly higher in Rose Creek than in all other low intensity watersheds (Table 3.41). There was no significant difference in mean rank TP export coefficients between the remaining low agricultural intensity streams with the exception of Prairie Blood Coulee. Annual TP export coefficients in Prairie Blood Coulee were significantly lower than those measured in Paddle River and Willow Creek. Prairie Blood Coulee generally receives lower annual precipitation than the rest of the low agricultural intensity watersheds, excluding those in the northern part of the province (Chapter 2: Table 2.4). The higher TP export coefficients measured in Rose Creek may be attributed to higher flow volumes and a larger drainage area. For example, Rose Creek and Paddle River are both located in the Western Alberta Upland sub-ecoregion of the Boreal. Although they may have similar landscape and agricultural intensity features, Rose Creek has a much larger effective drainage basin (approximately twice as large) which influences the larger flow volumes and loads (approximately three times higher).

Table 3.39. Mann-Whitney statistics comparing median annual TP export coefficients (1999 to 2006) among nine watersheds located in the Boreal ecoregion area. For full stream names, see Table 3.5 (pg. 3-12).

	BLI	GRA	HIN	KLE	PAD	ROS	STW	TOM	WAB
A.I:	(M)	(M)	(L)	(M)	(L)	(L)	(H)	(M)	(H)
BLI	-								
GRA	45.000 0.172	-							
HIN	**59.000** **0.005**	48.000 0.093	-						
KLE	41.000 0.345	30.000 0.834	18.000 0.115	-					
PAD	50.000 0.059	35.000 0.753	21.000 0.248	38.000 0.529	-				
ROS	33.000 0.916	18.000 0.141	**5.000** **0.001**	25.000 0.462	**19.000** **0.025**	-			
STW	32.000 1.00	22.000 0.294	**21.000** **0.037**	24.000 0.401	19.000 0.172	34.000 0.834	-		
TOM	42.000 0.294	26.000 0.529	**16.000** **0.014**	30.000 0.834	23.000 0.345	42.000 0.294	38.000 0.529	-	
WAB	**43.000** **0.009**	50.000 0.059	37.000 0.600	49.000 0.074	46.000 0.141	**55.000** **0.016**	**54.000** **0.021**	**51.000** **0.046**	-

Note: **Bold** values indicate a significant difference at p<0.05. The top value is the Mann-Whitney statistic (U); the bottom value is the significance level. The degrees of freedom for all Mann-Whitney statistics is 1.

Table 3.40. Mann-Whitney statistics for annual TP export coefficients from 1999 to 2006 for Trout and Meadow Creeks compared to the other four moderate agricultural intensity watersheds.

Moderate Agricultural Intensity Watershed	Trout Mann-Whitney Statistic (U)	Trout Level of Significance (p)	Meadow Mann-Whitney Statistic (U)	Meadow Level of Significance (p)
Blindman River	79.000	**0.017**	90.000	**0.001**
Grande Prairie Creek	62.000	**0.002**	52.000	**0.036**
Kleskun Drain	55.000	**0.006**	56.000	**0.012**
Tomahawk Creek	47.000	0.115	14.000	**0.009**

Note: **Bold** values indicate a significant difference at $p<0.05$. The degrees of freedom for all Mann-Whitney statistics is 1.

Table 3.41. Mann-Whitney Statistics for annual TP export coefficients from 1999 to 2006 for Rose Creek and Prairie Blood Coulee compared to other low agricultural intensity watersheds.

Low Agricultural Intensity Watershed	Rose Creek Mann-Whitney Statistic (U)	Rose Creek Level of Significance (p)	Prairie Blood Coulee Mann-Whitney Statistic (U)	Prairie Blood Coulee Level of Significance (p)
Hines Creek	5.0	**0.001**	47	0.115
Paddle River	19	**0.025**	77	**0.006**
Prairie Blood Coulee	2.0	**0.000**	-	-
Rose Creek	-	-	2.0	**0.000**
Willow Creek	42	0.294	12	0.036

Note: **Bold** values indicate a significant difference at $p<0.05$. The degrees of freedom for all Mann-Whitney statistics is 1.

Total P export coefficients for the AESA watersheds were compared with data from the CAESA study (Anderson et al. 1998b) and other studies in Alberta and outside of the province (Table 3.42). Ranges for TP export coefficients in the AESA watersheds were broad with overlap between agricultural intensity categories, as was found in the CAESA study. However, low agricultural intensity CAESA watersheds were reported as exporting more TP than the moderate and high intensity streams, whereas moderate agricultural intensity AESA watersheds generally exported the most TP. Differences in export coefficients between the AESA and CAESA streams are likely a result of differences in annual precipitation as well as changes within each agricultural intensity category. Note that the watersheds under low, moderate, and high agricultural intensity categories differed between the AESA and CAESA studies. Total P export coefficients for the AESA watersheds were similar to those reported for other studies in Alberta but were often lower than those reported for agricultural areas outside of the province. These differences are likely a result of variations in geology, landscape factors, and climatic conditions such as precipitation amount, duration, and intensity.

Table 3.42. Comparison of TP export coefficients from the AESA stream survey with data from other studies.

	Number of Streams	Land Cover (majority)	Median TP Export Coefficient ($kg \cdot ha^{-1} yr^{-1}$)	Source
Alberta				
AESA Streams (1999-2006)	19	Low, Mod., and High	0.079 (0.182 mean)	This study
	8	High Ag. Intenstiy	0.083	
	6	Moderate Ag. Intensity	0.096	
	5	Low Ag. Intensity	0.064	
CAESA Streams (1995+1996)				Anderson et al. (1998b)
	9	High Ag. Intenstiy	0.0661	
	12	Moderate Ag. Intensity	0.1092	
	6	Low Ag. Intensity	0.1988	
Haynes Creek (M6) (1995+1996)	1	High Ag. Intenstiy	0.1355	
Sakwatamau (1995 and 1996)	1	Forest	0.069 and 2.1911	
Wabamun (1981)	9	Agriculture	0.11	Mitchell (1985)
Wabamun (1981)	5	Forest	0.09	
Baptiste (1977-78)	4	Agriculture	0.13	Trew et al (1987)
Baptiste (1991-1995)	4*	Agriculture	0.11	Cooke (1996)
Baptiste (1994 and 1995)	1	Agriculture	0.14 and 0.12	Cooke and Prepas 1998
Baptiste (1994 and 1995)	1	Agriculture	0.82 and 0.57	
Baptiste (1994 and 1995)	1	Agriculture	0.57 and 0.34	
Baptiste (1977-78)	4	Forest	0.13	Trew et al (1987)
Baptiste (1991-1995)	4*	Forest	0.11	Cooke (1996)
Baptiste (1994 and 1995)	1	Forest	0.13 and 0.09	Cooke and Prepas 1998
Baptiste (1994 and 1995)	1	Forest	0.22 and 0.05	
Lac La Nonne (1981)	1	Agriculture	0.25	Mitchell and Hamilton (1982)
Lesser Slave Lake Basin (1991-1992)				Noton 1998
Driftpile River	1		0.11	
South Heart River	1		0.054	
Swan River	1		0.17	
Assineau River	1		0.08	
Marten Creek	1		0.17	

Table 3.42 Continued.

	Number of Streams	Land Cover (majority)	Median TP Export Coefficient ($kg \cdot ha^{-1} yr^{-1}$)	Source
Ontario				
Ave. annual export of 34 watersheds		Forest	0.048	Dillon and Kirchner 1975
(forest and pasture) over 20 months		Pasture	0.11	
Lake Simcoe (1990-1998)		Vegetable polder	1.09 (mean)	Winter et al. 2002
		Urban land	0.65 (mean)	
		Mixed agriculture	0.11 to 0.27	
		Forest	0.06 to 0.07	
Streams Outside of Canada				
State of Wisconsin (1973-1982)**	3	Forest	0.112 (mean)	Clesceri et al. 1986
State of Wisconsin (1973)	6	Agriculture	0.262 (mean)	
Ohio (1994-1998)				Vanni et al. 2001
Four Mile Creek		Agriculture	~ 1.5 (mean)	
Little Four Mile Creek		Agriculture	~ 1.2 (mean)	
Marshall's Creek		Agriculture	~ 1.7 (mean)	
Wagon Train Watershed (Nebraska)	13	Fallow	0.243 (mean)	Elrashidi et al. 2005
		Cropland	0.217 (mean)	
		Grassland	0.19 (mean)	
Luxembourg				Salvia-Castellvi et al. 2005
Teischelt (1993/94 and 1994/95)	1	Forest	0.8 and 0.9	
Kuebefiels (1993/94 and 1994/95)	1	Agriculture	0.5 and 0.8	
Sure (1989-1991)	1	Mixed	0.8 and 0.9	
Wiltz (1999-2001)	1	Mixed	1.0 and 1.2	
Bavigne (1989/90)	1	Mixed	0.07	
Surbich and Syrbach (1995/96)	2	Mixed	0.06	
Sweden (1998/1993-2006)	27	Agriculture	0.06 to 0.92 (range of means)	Kyllmar et al. 2006

*Different streams were sampled in 1977-78 and 1991-95

**stream 1: 1975-1977; stream 2: 1973/75-1982; stream 3: 1973

Total Dissolved Phosphorus (TDP) export coefficients. Total dissolved phosphorus export coefficients showed the same pattern with location by ecoregion area as TP although export coefficients in the Continental Divide were more similar to those measured in the Grassland ecoregion area. (Figure 3.54). Kruskal Wallis and Mann Whitney statistics showed mean rank TDP export coefficients were significantly higher in Parkland (U=96, p<0.005) and Boreal (U=1491, p<0.005) streams than in those streams located in the Grassland ecoregion area (Figure 3.54). However, no significant difference in mean rank TDP export coefficients was observed between the Parkland and Boreal ecoregion areas (U=1227, p=0.094). Noteworthy, streams in the Parkland ecoregion generally had higher median annual TDP export coefficients as a group, but the two highest median annual TDP export coefficients were measured in two watersheds in the Boreal ecoregion area (Blindman River and Kleskun Drain).

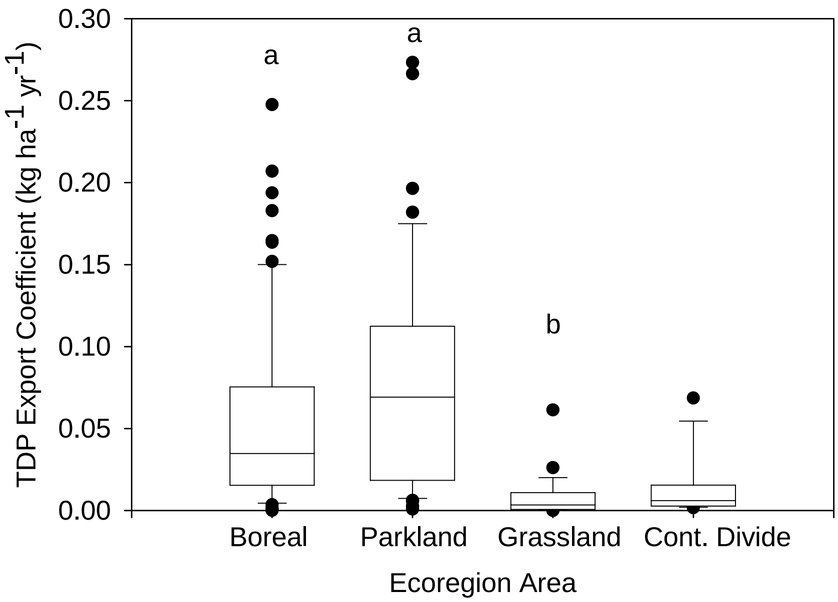


Figure 3.54. Box plots of median annual TDP export coefficients (1999 to 2006) for the Boreal, Grassland, and Parkland ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the Grassland ecoregion area and other two ecoregion areas were statistically different at the p<0.005 significance level.

Similar to the findings of the CAESA study (Anderson et al. 1999), the highest TDP export coefficients were observed in streams with high agricultural intensity (Figure 3.55). Overall, there was a stepwise trend of increasing export coefficients with increasing agricultural intensity. Annual TDP export coefficients were significantly lower in the low agricultural intensity category than in the high agricultural intensity watersheds (H=689, p=0.001). However, mean rank export coefficients in the moderate agricultural intensity group were not significantly higher than the low (U=786, p=0.145) or significantly lower than the high agricultural intensity groups (U=1157, p=0.136).

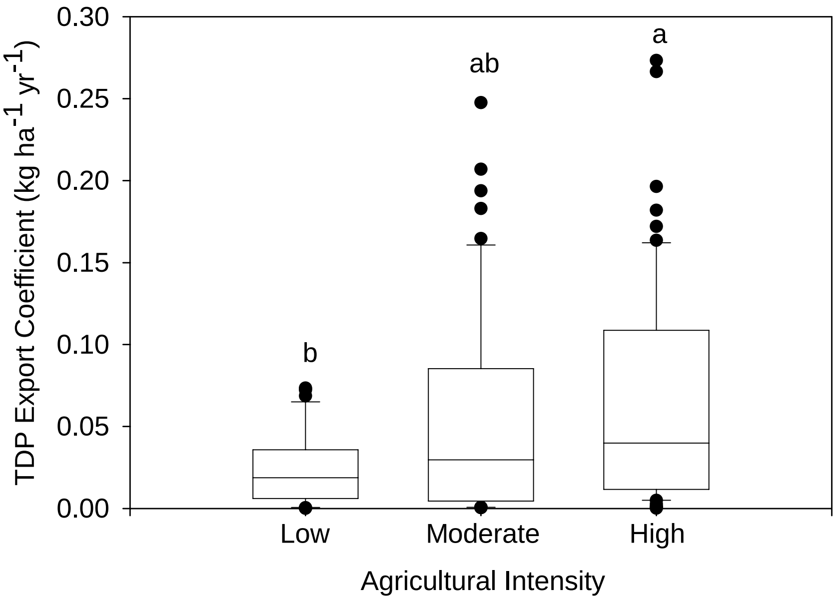


Figure 3.55. Box plots of median annual TDP export coefficients (1999 to 2006) in the three dryland agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the low and high agricultural intensity categories were statistically different at the $p<0.005$ level.

Regional climatic and topographical factors were not necessarily the main factors influencing median annual TDP export coefficients, as was observed for TP. The streams with the five highest median annual export coefficients (1999-2006) were Blindman Creek (0.1302 $kg \cdot ha^{-1} yr^{-1}$), Kleskun Drain (0.1199 $kg \cdot ha^{-1} yr^{-1}$), and Threehills (0.1057 $kg \cdot ha^{-1} yr^{-1}$), Stretton (0.0926 $kg \cdot ha^{-1} yr^{-1}$) and Haynes Creeks (0.0882 $kg \cdot ha^{-1} yr^{-1}$) (Figure 3.56). The four lowest median annual TDP export coefficients were measured in Trout (0.002 $kg \cdot ha^{-1} yr^{-1}$), Meadow (0.003 $kg \cdot ha^{-1} yr^{-1}$), and Willow Creeks (0.006 $kg \cdot ha^{-1} yr^{-1}$) and Prairie Blood Coulee (0.007 $kg \cdot ha^{-1} yr^{-1}$). As initially illustrated in Figure 3.54, streams monitored in the Grassland ecoregion (Trout, Meadow, and Prairie Blood Coulee) had much lower TDP export coefficients than streams monitored in other ecoregion areas. Moreover, the four streams with the lowest TDP export coefficients were all located in the southwestern part of the province and consisted of watersheds under both low and moderate agricultural intensity activities. With the exception of Buffalo Creek, streams in the Parkland ecoregion had higher TDP export coefficients than the other AESA streams; however, Blindman Creek and Kleskun Drain (Boreal) had the first and second highest median annual TDP export coefficients, respectively. It is important to note that all of the watersheds in the Parkland ecoregion were classified as high agricultural intensity watersheds based on the 1996 Census of Agriculture data (Anderson et al. 1999). As a result, it was difficult to separate the effects of agricultural intensity and ecoregion characteristics in the streams located in the Parkland ecoregion area.

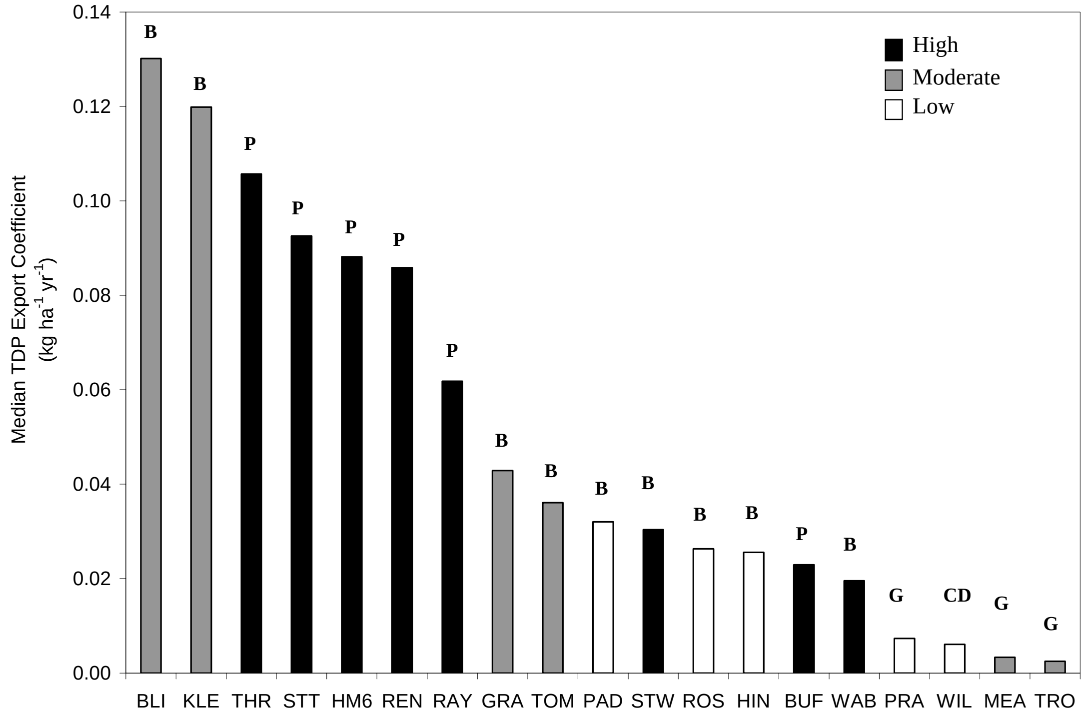


Figure 3.56. Median annual TDP export coefficients for each dryland AESA watershed (1999 to 2006). Location by ecoregion area for each stream is illustrated as follows: Boreal (B), Parkland (P), and Grassland (G). For full stream names, see Table 3.5 (pg. 3-12).

Since it appeared that TDP export coefficients were influenced by ecoregional characteristics and agricultural intensity, Kruskal-Wallis One-Way ANOVA statistics were run on median annual TDP export coefficients among streams grouped by ecoregion area and by agricultural intensity (Table 3.43). A significant difference in TDP export coefficients was not found among streams in the Grassland or Parkland ecoregion areas or among watersheds in the high agricultural intensity or low agricultural intensity categories. However, TDP export coefficients were significantly different among streams in the Boreal ecoregion area ($p<0.05$) and streams in the moderate agricultural intensity category ($p<0.005$).

Table 3.43. Kruskal-Wallis One-Way ANOVA statistics for median annual TDP export coefficients (1999 to 2006) among dryland streams grouped by ecoregion area and by agricultural intensity.

Ecoregion Area/Agricultural Intensity	Kruskal-Wallis Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	16.615	**0.034**	8
Parkland	5.583	0.349	5
Grassland	0.545	0.761	2
Low	8.409	0.078	4
Moderate	28.115	**0.000**	5
High	10.532	0.160	7

Note: **Bold** values indicate a significant difference at $p<0.05$.

Unlike TP export coefficients, a significant difference was observed among mean rank TDP export coefficients in streams of different agricultural intensities within the Boreal ecoregion area (H=10.214, p=0.006, 1df). Mann-Whitney tests on untransformed data showed that annual TDP export coefficients were significantly higher in moderate agricultural intensity streams in the Boreal ecoregion area than in low (U=206, $p<0.005$) or high agricultural intensity streams (U=152, $p<0.05$) within the ecoregion area. No difference was found at the $p<0.05$ level between mean rank TDP export coefficients in the low and high agricultural intensity streams.

Furthermore, Mann-Whitney tests between streams in the Boreal ecoregion area showed that the median annual TDP export coefficient in Blindman River was significantly higher than TDP export coefficients in all other watersheds in the Boreal ecoregion area with the exception of Kleskun Drain and Hines Creek (Table 3.44). The mean rank TDP export coefficient for Wabash Creek was significantly lower than Blindman River, Kleskun Drain, and Rose Creek (Table 3.45). Annual TDP export coefficients in the remaining watersheds in the Boreal were not statistically different from each other (H=6.182, p=0.403).

Table 3.44. Mann-Whitney statistics between median annual TDP export coefficients (1999 to 2006) in the Blindman River and those in eight other watersheds in the Boreal ecoregion area.

Stream (A.I.)	Mann-Whitney Statistic (U)	Level of Significance (p)
Grande Prairie Creek (M)	51.000	**0.046**
Hines Creek (L)	58.000	0.056
Kleskun Drain (M)	35.000	0.753
Paddle River (L)	63.000	**0.041**
Rose Creek (L)	78.000	**0.005**
Strawberry Creek (H)	69.000	**0.039**
Tomahawk Creek (M)	53.000	**0.027**
Wabash Creek (H)	55.000	**0.016**

Note: **Bold** values indicate a significant difference at $p<0.05$. All statistical comparisons had 1 degree of freedom.

Table 3.45. Mann-Whitney statistics between median annual TDP export coefficients (1999 to 2006) in Wabash Creek and three streams in the Boreal ecoregion area with significantly higher exports ($p<0.05$).

Stream	Mann-Whitney Statistic (U)	Level of Significance (p)
Blindman River	55.000	0.016
Kleskun Drain	51.000	0.046
Rose Creek	61.000	0.002

Note: All statistical comparisons had 1 degree of freedom.

The data show a clear distinction between TDP export coefficients from moderate agricultural intensity streams in the Boreal ecoregion area to those moderate agricultural intensity watersheds in the Grassland ecoregion area. Export coefficients in Trout and Meadow Creeks were significantly lower than the other moderate agricultural intensity watersheds that were all located in the Boreal ecoregion area (Table 3.46). Moderate agricultural intensity watersheds located in the Boreal ecoregion area did not have statistically different TDP export coefficients (Blindman Creek, Grande Prairie Creek, Kleskun Drain, and Tomahawk Creek). Thus, TDP export coefficients in the Grassland streams were likely influenced by climate and landscape factors typical for the region rather than agricultural intensity. Also, the majority of Trout and Meadow Creeks were covered in grassland (67 and 74%, respectively) with only a small percentage of each watershed covered in cropland or forage (Chapter 2: Table 2.9). In contrast, land cover for the moderate agricultural intensity watersheds in the Boreal ecoregion area ranged from 4 to 48% cropland and 13 to 49% forage.

Table 3.46. Mann-Whitney statistics for annual TDP export coefficients from 1999 to 2006 for Trout and Meadow Creeks compared to the other four moderate agricultural intensity watersheds.

Moderate Agricultural Intensity Watershed	Trout Kruskal-Wallis Statistic (U)	Trout Level of Significance (p)	Meadow Kruskal-Wallis Statistic (U)	Meadow Level of Significance (p)
Blindman River	88.000	**0.000**	80.000	**0.000**
Grande Prairie Creek	62.000	**0.002**	61.000	**0.002**
Kleskun Drain	55.000	**0.006**	56.000	**0.012**
Tomahawk Creek	84.000	**0.001**	4.000	**0.001**

Note: **Bold** values indicate a significant difference at $p<0.05$. The degrees of freedom for all Mann-Whitney statistics is 1.

Total Particulate Phosphorus (TPP) export coefficients. Median annual TPP export coefficients appeared to be most influenced by ecoregional characteristics that contributed to overland runoff and movement of soil bound P to surface water. In contrast to TP and TDP, the median annual TPP export coefficients were significantly higher in the Boreal ecoregion area than in both the Parkland ($U=2302$, $p<0.005$) and Grassland ecoregion areas ($U=1235.5$, $p<0.005$) (Figure 3.57). There was no significant difference in mean rank TPP export coefficients between the Parkland and Grassland ecoregion areas ($U=472$, $p=0.67$) (Figure 3.58). Export coefficients in the Continental Divide appeared similar to those in the Boreal ecoregion area. Annual TPP export coefficients were not significantly different between low and moderate agricultural intensity streams ($U=898$, $p=0.603$). However, mean rank TPP export coefficients were significantly lower in high agricultural intensity streams than in low ($U=1463$, $p=0.029$) or moderate agricultural intensity watersheds ($U=1863$, $p=0.003$). Those watersheds draining low or moderate agricultural intensity land had moderate to high runoff potential, while the high agricultural watersheds were classified as having low to moderate runoff potential. These patterns support the assumption that TPP export coefficients were most influenced by ecoregional characteristics, specifically runoff depth.

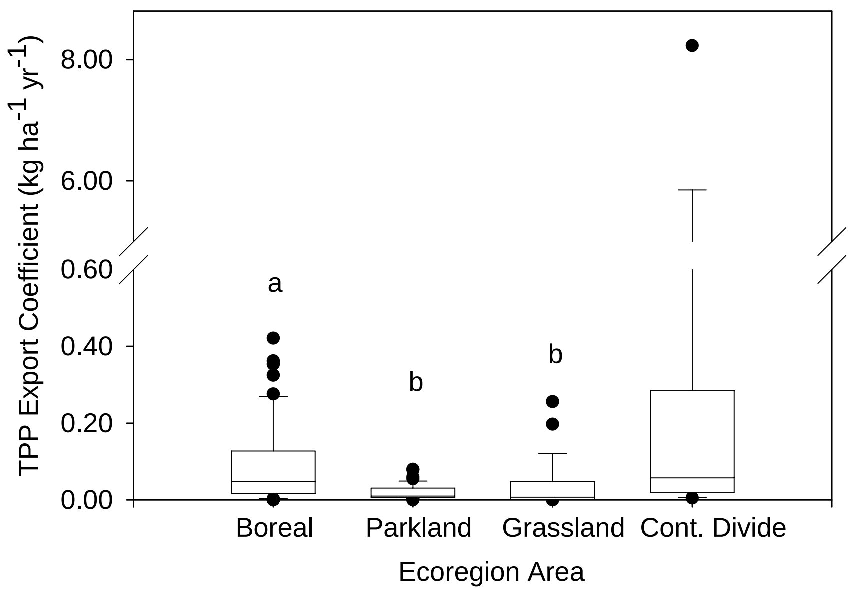


Figure 3.57. Box plots of median annual TPP export coefficients (1999 to 2006) for the Boreal, Grassland, and Parkland ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the Boreal ecoregion area and other two ecoregions were statistically different at the $p<0.005$ significance level. Note: an outlier (0.8422 kg. $ha^{-1}yr^{-1}$) in the Boreal ecoregion area is not shown.

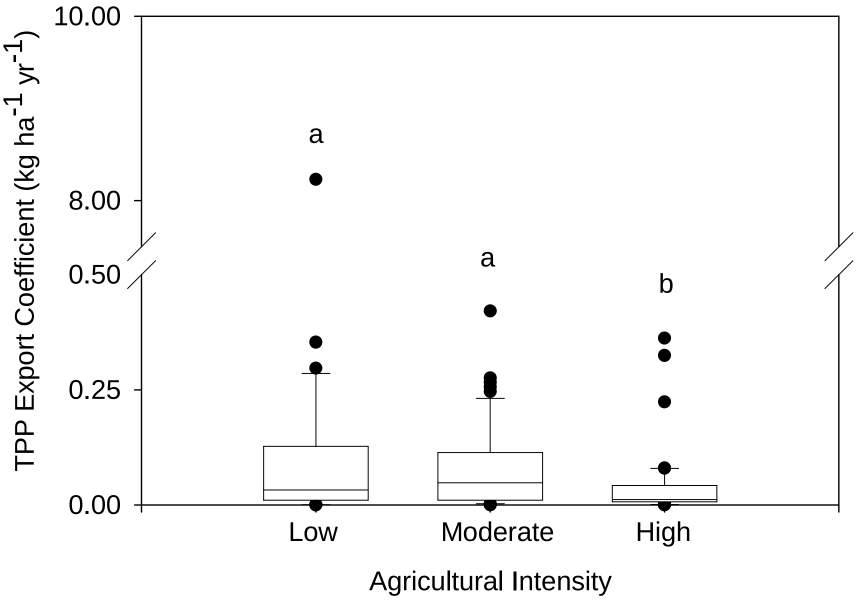


Figure 3.58. Box plots of median annual TPP export coefficients (1999 to 2006) in the three dryland agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the low and high agricultural intensity watersheds were significantly different at the $p<0.005$ level. Note: an outlier (1.2257 kg.ha^{-1}yr^{-1}) in the low agricultural intensity category is not shown.

Total particulate phosphorus export coefficients for the dryland watershed are illustrated in Figure 3.59. The top five exporters of TPP were Rose (0.1519 kg·ha^{-1}yr^{-1}), Strawberry (0.1518 kg·ha^{-1}yr^{-1}), Tomahawk (0.0898 kg·ha^{-1}yr^{-1}), Blindman (0.0835 kg·ha^{-1}yr^{-1}) and Willow Creeks (0.0574 kg·ha^{-1}yr^{-1}). All but Willow Creek (Continental Divide) were located in the Boreal ecoregion. The bottom five TPP exporters were Prairie Blood Coulee (0.0015 kg·ha^{-1}yr^{-1}) and Wabash (0.0073 kg·ha^{-1}yr^{-1}), Buffalo (0.0092 kg·ha^{-1}yr^{-1}), Ray (0.0099 kg·ha^{-1}yr^{-1}), and Renwick Creeks (0.0100 kg·ha^{-1}yr^{-1}). With the exception of Prairie Blood Coulee (low agricultural intensity), the seven streams with the lowest TPP exports were all high agricultural intensity streams and had low to moderate runoff potential. Wabash Creek was the only stream in the Boreal ecoregion area that did not have a higher TPP export coefficient than streams in the Grassland or Parkland ecoregion areas. The lower TPP export coefficients in this watershed were likely a result of the moderate runoff potential compared to the high runoff potential noted for the majority of the Boreal streams. The three streams within the Grassland ecoregion area had median annual TPP export coefficients scattered among export coefficients measured in the Parkland ecoregion streams.

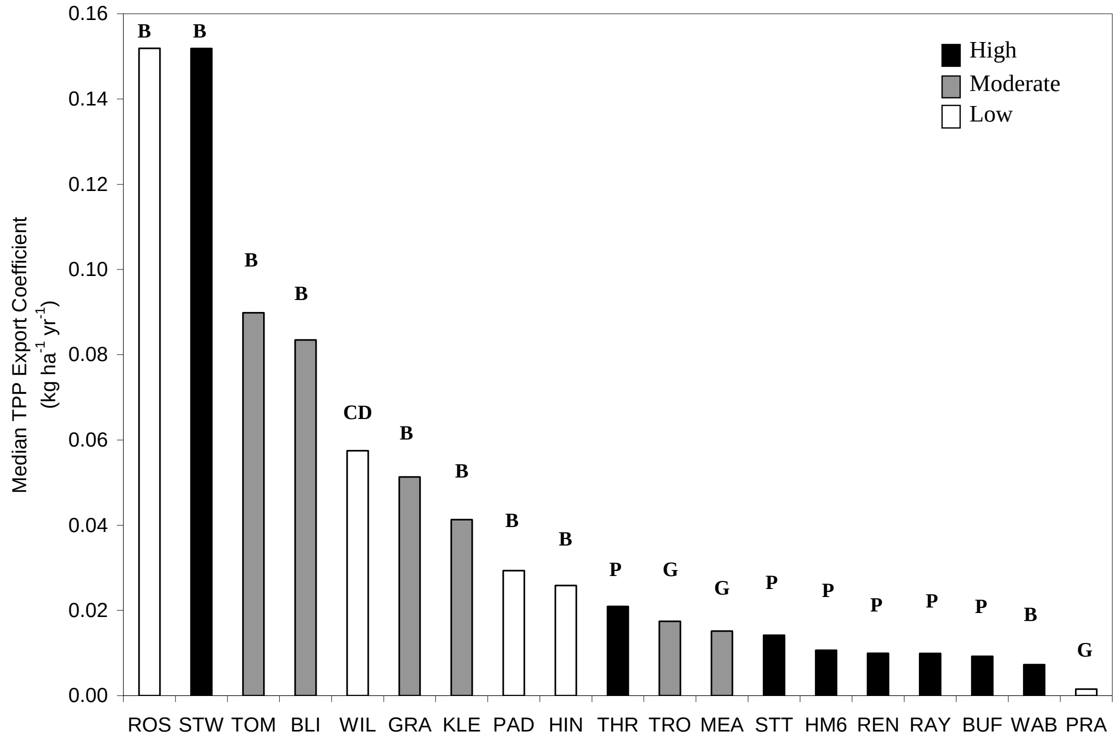


Figure 3.59. Median annual TPP export coefficients for each dryland AESA watershed (1999 to 2006). Location by ecoregion area for each stream is illustrated as follows: Boreal (B), Parkland (P), and Grassland (G). For full stream names, see Table 3.5 (pg 3-12).

Median annual TPP export coefficients were not significantly different between streams within the Parkland or Grassland ecoregion areas or among the moderate agricultural intensity watersheds (Table 3.47). However, significant differences were found between streams within the Boreal ecoregion area and low and high agricultural intensity categories.

Table 3.47. Kruskal-Wallis One-Way ANOVA statistics for median annual TPP export coefficients (1999 to 2006) among dryland streams grouped by ecoregion area and by agricultural intensity.

Ecoregion Area/Agricultural Intensity	Kruskal-Wallis Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	25.574	**0.001**	8
Parkland	1.811	0.875	5
Grassland	3.605	0.165	2
Low	17.245	**0.002**	4
Moderate	9.305	0.199	5
High	16.999	**0.017**	7

Note: **Bold** values indicate a significant difference at $p<0.05$.

Mann-Whitney tests showed that annual TPP export coefficients were significantly different among streams in the Boreal ecoregion area (H=25.574, p=0.001, 8df), but an obvious pattern was not present. Rose Creek had the highest median annual TPP export coefficient and had significantly higher mean rank TPP export coefficients than all other streams in the Boreal ecoregion area (Table 3.48) with the exception of Blindman River and Strawberry and Tomahawk Creeks (p>0.05). Annual TPP export coefficients were significantly higher in Blindman River than in Hines (U=57, p=0.009) or Wabash Creeks (U=59, p=0.005). Median annual TPP export coefficients in Hines Creek were significantly lower than those in all streams in the Boreal ecoregion area with the exception of Kleskun Drain, Paddle River, and Wabash Creek (Table 3.48). Furthermore, export coefficients in Wabash Creek were significantly lower than all TPP export coefficients in all streams in the Boreal with the exception of Hines Creek. Note that there was not a significant difference between the streams when grouped by agricultural intensity within the Boreal ecoregion area as was observed for TDP (H=0.896, p=0.639, 2df).

Table 3.48. Mann-Whitney statistics comparing median annual TPP export coefficients (1999 to 2006) among nine watersheds located in the Boreal ecoregion area. For full stream names, see Table 3.5 (pg 3-12)

	BLI	GRA	HIN	KLE	PAD	ROS	STW	TOM	WAB
A.I.:	(M)	(M)	(L)	(M)	(L)	(L)	(H)	(M)	(H)
BLI	-								
GRA	42 0.294	-							
HIN	57 **0.009**	44 0.208	-						
KLE	47 0.534	37 0.600	23 0.345	-					
PAD	57 0.283	33 0.534	23 0.131	25 0.183	-				
ROS	27 0.160	17 **0.026**	5 **0.001**	9 **0.004**	14 **0.013**	-			
STW	34 0.409	24 0.099	14 **0.013**	19 **0.039**	22 0.069	45 **0.007**	-		
TOM	43 0.934	27 0.160	10 **0.005**	21 0.058	26 0.137	53 0.457	50 0.620	-	
WAB	59 **0.005**	54 **0.021**	43 0.248	47 0.115	51 **0.046**	61 **0.002**	60 **0.003**	57 **0.009**	-

Note: **Bold** values indicate a significant difference at p<0.05. The top value is the Mann-Whitney statistic (U); the bottom value is the significance level. The degrees of freedom for all Mann-Whitney statistics is 1.

Interestingly, mean rank TPP export coefficients were significantly different between streams ranked as draining high agricultural intensity land (H=16.999, p=0.017, 7df), which was also the only parameter that showed a difference between export coefficients in the high agricultural intensity streams. A Kruskal-Wallis test for Buffalo, Haynes, Ray, Renwick, Stretton, Threehills, and Wabash Creeks showed no significant difference (H=2811, p=0.832, 6df). Thus, Strawberry Creek had significantly higher annual TPP export coefficients than the other high agricultural intensity watersheds and was the only stream with significantly different mean rank export coefficients within the high agricultural intensity category. Noteworthy, Strawberry Creek was the only high agricultural intensity watershed ranked as having a high runoff potential and also had a much larger effective drainage basin size (Chapter 2: Table 2.1). Higher overland runoff volumes in Strawberry Creek likely resulted in the higher TPP export coefficients observed. Thus, beneficial management practices aimed at reducing particulate P contributions by overland runoff would be more effective in a high agricultural stream such as Strawberry Creek than high agricultural intensity watersheds with lower volumes of overland runoff.

Statistical differences among TPP export coefficients from low agricultural intensity watersheds were not accounted for by differences in runoff potential. Annual TPP export coefficients in Rose Creek (moderate runoff potential) were significantly higher than those in all other low agricultural intensity watersheds with the exception of Willow Creek (U=41, p=0.345) (Table 3.49). Mean rank exports in Willow Creek (high runoff potential) were only significantly higher than those measured in Prairie Blood Coulee (high runoff potential) (U=7, p=0.009). Mean rank TPP export coefficients were significantly lower in Prairie Blood Coulee than all watersheds but Hines Creek.

Table 3.49. Mann-Whitney statistics between median annual TPP export coefficients (1999 to 2006) in the five low agricultural intensity watersheds.

Stream	Hines Creek	Paddle River	Prairie Blood Coulee	Rose Creek	Willow Creek
Hines Creek	-				
Paddle River	23.0 0.131	-			
Prairie Blood Coulee	50.0 0.059	71.0 **0.006**	-		
Rose Creek	5.00 **0.001**	14.0 **0.013**	2.00 **0.001**	-	
Willow Creek	17.0 0.115	0.462 0.540	7.00 **0.009**	41.0 0.345	-

Note: **Bold** values indicate a significant difference at p<0.05. The top value is the Mann-Whitney statistic (U); the bottom value is the significance level. The degrees of freedom for all Mann-Whitney statistics is 1.

Upon further investigation, it was confirmed that TPP export coefficients were generally influenced by runoff depth (Figure 3.60). As runoff depth increased, TPP export coefficients also increased (r_s= 0.73, p<0.001). This observation held true for the majority of watersheds regardless of their location by ecoregion or their agricultural intensity classification although some exceptions were observed. Willow Creek had the highest median annual runoff depth but did not follow the same pattern as was observed in the other watersheds; it is not included in Figure 3.60 in order to better illustrate the patterns in the data. Although a general increase in TPP export coefficients was observed as runoff depth increased, Willow Creek had a much lower TPP export coefficient than would have been expected (runoff depth = 151.67 mm; TPP export coefficient = 0.0575 $kg \cdot ha^{-1} yr^{-1}$).

Strawberry, Rose, and Tomahawk Creeks also had median annual TPP export coefficients that did not appear to follow the same relationship observed between TPP export coefficients and runoff depth in the other streams. All three streams, located in the Boreal ecoregion area, had much higher median TPP export coefficients than would have been expected from their respective runoff depths. The Spearman rank-order correlation coefficient increased to 0.83 ($p<0.001$) when Rose, Strawberry, Tomahawk, and Willow Creeks were removed from the analysis. These differences in TPP export coefficients and runoff depth were not likely influenced by agricultural intensity as the three streams represented low (Rose), moderate (Tomahawk), and high (Strawberry) agricultural intensity.

The 1991 land cover map of Strawberry Creek showed the watershed was covered by a small percentage of cropland compared to other high agricultural intensity watersheds and had a steeper slope around the mouth of the watershed, which was also the site of sample collection. Although the median annual TPP export coefficient was 0.152 $kg \cdot ha^{-1} \cdot yr^{-1}$ (1999-2006), annual TPP export coefficients ranged between 0.012 and 0.842 $kg \cdot ha^{-1} \cdot yr^{-1}$. Export coefficients were highest in years with high annual flow volume and lowest in years with low annual flow volume.

Based on land cover, agricultural intensity percentile, and location by ecoregion, Paddle River and Rose Creek would be expected to annually export similar amounts of TPP. However, Rose Creek had higher annual TPP loads (two times higher) as a result of higher flows (three times higher) and a larger drainage basin size (two times higher) than Paddle River.

Furthermore, median annual TPP export coefficients generally increased with increasing effective drainage area for watersheds in the Boreal ecoregion area, although there was only a weak, positive correlation ($r_s = 0.30$, $p>0.05$). The size of the effective drainage area was likely an important factor influencing export coefficients in the Boreal ecoregion area, specifically in Rose, Tomahawk, and Strawberry Creeks where export coefficients were higher than expected from the relationship with runoff depth. However, a relationship was not observed between TPP export coefficients and drainage area for the Parkland or Grassland watersheds. It appeared that the size of the drainage basin was a factor in TPP export coefficients when runoff potential was generally high, as in the Boreal ecoregion area, while TPP export coefficients in low runoff potential areas were more influenced by runoff depth.

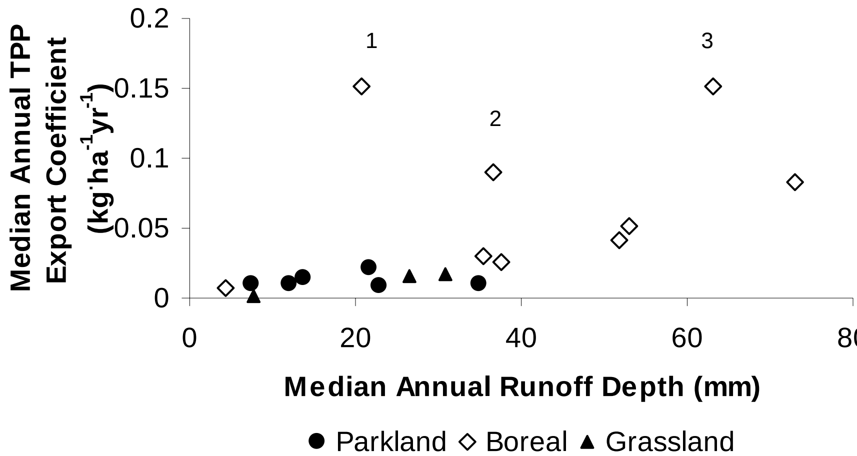


Figure 3.60. Median annual TPP export coefficients (1999 to 2006) compared to median annual runoff depth for the 18 dryland AESA watersheds. Note that Willow Creek was not included in the graph as discussed in the text. Strawberry, Tomahawk, and Rose Creeks are represented by the numbers 1, 2, and 3, respectively.

Total Dissolved P to Total P export coefficient ratios (TDP/TP). When grouped by ecoregion area, mean rank TDP/TP was significantly higher in the Parkland ecoregion than in the Boreal ($U=4039$, $p<0.005$) and Grassland ($U=1468$, $p<0.005$) ecoregion areas (Figure 3.61). Furthermore, TDP/TP was significantly higher in the Boreal than Grassland ecoregion area ($U=41583$, $p<0.005$). The median TDP/TP ratio in the Continental Divide appeared to be lower than those in the other ecoregion areas.

When grouped by agricultural intensity, the mean rank TDP/TP ratios in the high agricultural watersheds were significantly higher than those observed in low ($U=13471$, $p<0.005$) or moderate ($U=17014$, $p<0.005$) agricultural intensity streams (Figure 3.62). No significant difference was observed between low and moderate agricultural intensity watersheds ($U=41866$, $p=0.941$).

As illustrated in Figure 3.63, the ratio of TDP/TP exported from each watershed was not equivalent. Agricultural intensity appeared to influence the ratio of TP in the dissolved form when grouped by location by ecoregion area, and ecoregional characteristics appeared to influence TDP/TP exports in certain watersheds when grouped by agricultural intensity. Overall, higher ratios of TDP/TP were observed in streams located in the Parkland ecoregion, which were all high agricultural intensity watersheds. Total phosphorus export coefficients in the Boreal ecoregion were influenced by both TDP and TPP export coefficients (both were positively correlated with TP export coefficients). However, TPP was the dominant form of P in the TP export coefficients, suggesting that TP export in the Boreal ecoregion area was more influenced by flow and runoff. In contrast, the higher ratio of dissolved P in the Parkland ecoregion suggested a greater influence from agricultural sources (TDP was 0.71 to 0.90 of the TP export in watersheds located in the Parkland ecoregion).

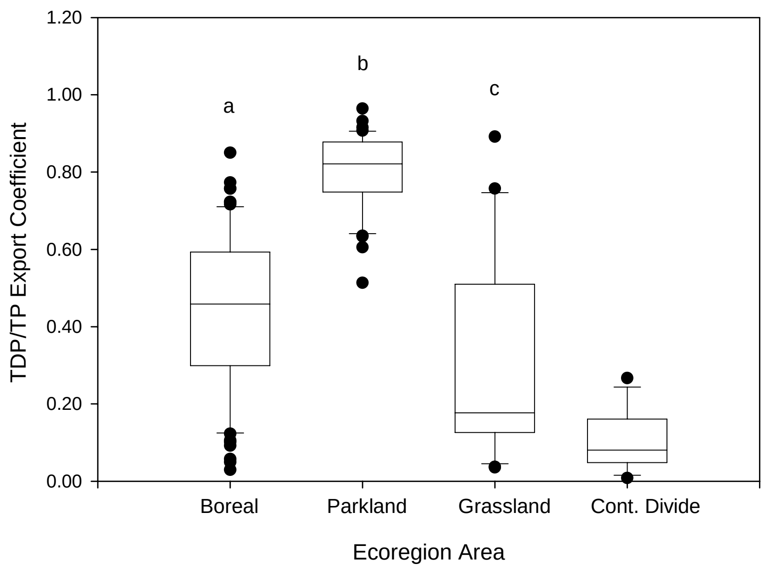


Figure 3.61. Box plots of median annual TDP/TP exports (1999 to 2006) for the Boreal, Grassland, and Parkland ecoregion areas. Medians of box plots with different letters are significantly different from one another at the 0.005 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

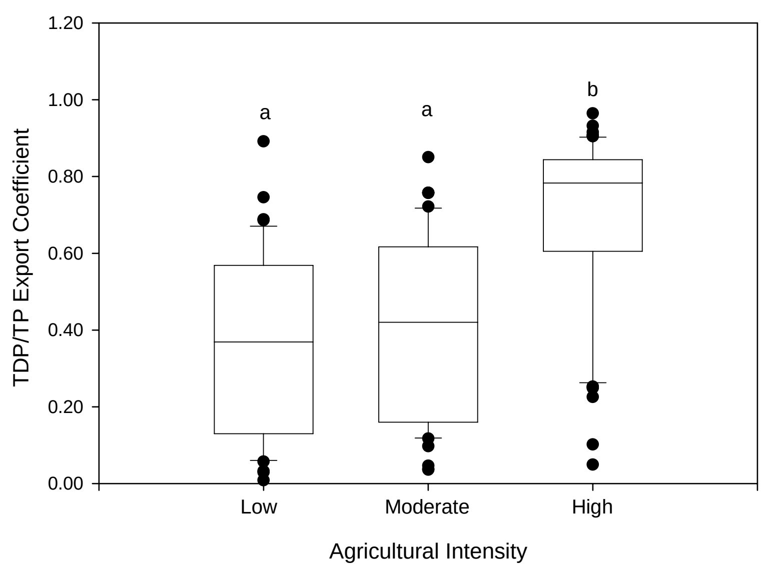


Figure 3.62. Box plots of median annual TDP/TP exports (1999 to 2006) in the three dryland agricultural intensity categories. Medians of box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Medians of box plots with different letters are significantly different at the $p<0.005$ level.

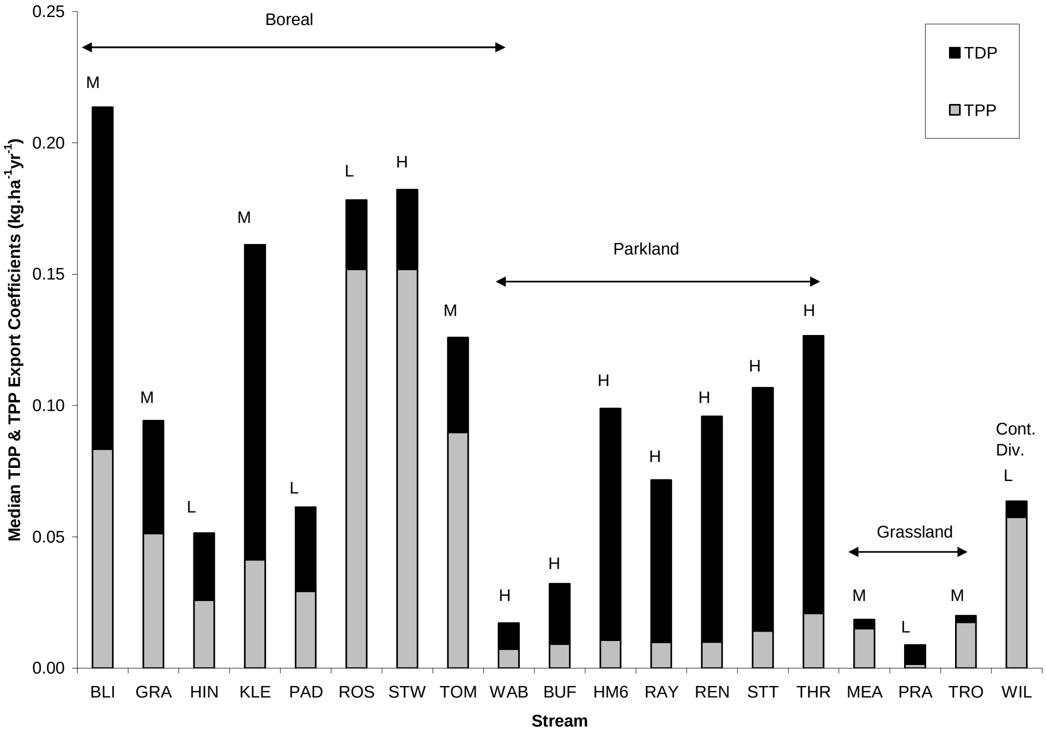


Figure 3.63. Median annual TDP and TPP export coefficients for each dryland AESA watershed (1999 to 2006). Agricultural intensity for each stream is illustrated as follows: Low (L), Moderate (M), and High (H). For full stream names, see Table 3.5 (pg. 3-12).

Although the high agricultural intensity Parkland watersheds had a higher ratio of TDP/TP compared to the other watershed groups, significant differences were observed among streams in each category (Table 3.50) A statistical difference was observed between streams in all categories, but these statistical differences were the result of a few watersheds that stood out (e.g., Buffalo Creek, Prairie Blood Coulee, and Kleskun Drain).

Buffalo Creek was the only stream with a significantly different ratio of TDP/TP in the Parkland ecoregion. When annual exports for Buffalo Creek were included in the Kruskal Wallis One-Way ANOVA for the Parkland ecoregion, the mean rank TDP/TP ratio was found to be significantly lower in Buffalo Creek than the remaining five watersheds in the Parkland ecoregion area (Table 3.51). However, there was not a significant difference between mean rank TDP/TP exports for the remaining streams in the Parkland ecoregion when data for Buffalo Creek was removed (H=7.959, p=0.93, 4df). Buffalo Creek appeared to be influenced by differences in landscape and topography as all watersheds within the Parkland ecoregion drained high agricultural intensity land. It was also noted in the AESA site selection report (Anderson et al. 1999) that not all of the drainage area in Buffalo Creek may contribute to the stream.

Table 3.50. Kruskal-Wallis One-Way ANOVA statistics for median annual ratios of TDP/TP exports (1999 to 2006) among dryland streams grouped by ecoregion area and by agricultural intensity.

Ecoregion / Ag. Intensity	Kruskal-Wallis Test Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	38.300	**0.000**	8
Parkland	19.516	**0.002**	5
Grassland	11.060	**0.040**	2
High	42.682	**0.000**	7
Moderate	28.661	**0.000**	5
Low	25.063	**0.000**	4

Note: **Bold** values indicate a significant difference at $p<0.05$.

Table 3.51. Mann-Whitney statistics between median annual TDP/TP ratios (1999 to 2006) in Buffalo Creek and the five other watersheds in the Parkland ecoregion.

Stream	Mann-Whitney Statistic (U)	Level of Significance (p)
Haynes Creek	1.000	**0.003**
Ray Creek	7.000	**0.009**
Renwick Creek	3.000	**0.002**
Stretton Creek	2.000	**0.017**
Threehills Creek	8.000	**0.012**

Note: **Bold** values indicate a significant difference at $p<0.05$. The degrees of freedom for all Mann-Whitney statistics is 1.

The ratio of TDP/TP in Prairie Blood Coulee was significantly higher than the mean rank exports in Trout (U=67, p=0.003) or Meadow (U=10, p=0.012) Creeks, while the latter two Grassland watersheds had similar TDP/TP ratios (U=38, p=0.859). Although Prairie Blood Coulee was classified as a low agricultural intensity dryland watershed and Meadow and Trout Creeks were classified as moderate agricultural intensity dryland watersheds, pivot circles were captured in orthophotos taken of Prairie Blood Coulee in 1991. The potential use of irrigation in the watershed would alter its behavior with respect to the influence of agricultural intensity on water quality. The presence of irrigation in the watershed could have contributed to the increased ratio of TP in the dissolved fraction.

Noteworthy, mean rank TDP/TP exports from Kleskun Drain were significantly higher than dissolved P ratios in watersheds located in the Boreal ecoregion as well as those measured in watersheds under moderate agricultural intensity (TDP/TP of 0.74 compared to range of 0.05 to 0.61 for other Boreal watersheds). The original agricultural intensity ranking calculated from the 1996 Census of Agriculture data listed Kleskun Drain as a high agricultural intensity watershed. However, it was assumed that the Kleskun Hills, not included in the effective drainage area, would likely contribute to runoff and flow in the watershed, thereby decreasing or diluting potential nutrient inputs from agricultural lands. It is possible that the Kleskun Hills did not contribute as much to drainage or stream flow as previously thought and that the watershed behaved more as a high agricultural intensity stream rather than one under moderate intensity agriculture. Further investigation into specific land management practices in the area would be required to understand the reasons for the differences in TDP/TP exports compared to other watersheds in the same region.

Although several studies have shown that dissolved phosphorus was the dominant form in forested regions with particulate phosphorus being the main from in agricultural areas, the AESA water quality data support previous studies in Alberta that found TDP increased with agricultural intensity (Anderson et al. 1998b; Cooke and Prepas 1998). It was thought that the contribution of nutrients from agricultural land was sufficient to mask any trends in exports as a result of runoff depth. It was also noted that all of the high agricultural intensity watersheds in the Parkland ecoregion area would likely have a similar runoff depth as runoff depth is influenced by factors characteristic of an ecoregion.

Total Nitrogen (TN) export coefficients. Annual TN export coefficients were grouped by ecoregion area and by agricultural intensity to investigate whether group medians were significantly different from each other and identify which factors influenced export coefficients. Boreal streams had higher export coefficients than watersheds located in the Parkland ($U = 1959$, $p = 0.018$, $df = 1$) and Grassland ecoregion areas ($U = 1321$, $p = 0$) (Figure 3.64). The streams with the five highest median annual TN export coefficients were also all in the Boreal: Blindman River (1.412 kg ha^{-1} yr^{-1}), Kleskun Drain (1.172 kg ha^{-1} yr^{-1}), and Tomahawk (1.123 kg ha^{-1} yr^{-1}), Grande Prairie (1.110 kg ha^{-1} yr^{-1}), and Rose Creeks (1.069 kg ha^{-1} yr^{-1}) (Figure 3.66). Trout Creek (0.158 kg ha^{-1} yr^{-1}), Prairie Blood Coulee (0.155 kg ha^{-1} yr^{-1}), and Meadow Creek (0.142 kg ha^{-1} yr^{-1}), all in the Grassland ecoregion area, had the lowest median annual TN export coefficients. Median TN export coefficients in the Grassland ecoregion were also statistically lower than TN export coefficients measured in the Parkland ecoregion ($U = 329.5$, $p = 0.015$). Median annual TN export coefficients in the Continental Divide appeared to be similar to those measured in the Parkland ecoregion. A significant difference was not found between mean rank TN export coefficients when grouped by agricultural intensity ($H=4.16$, $p=0.125$, 2df) (Figure 3.65), which was also observed for TP export coefficients grouped by agricultural intensity.

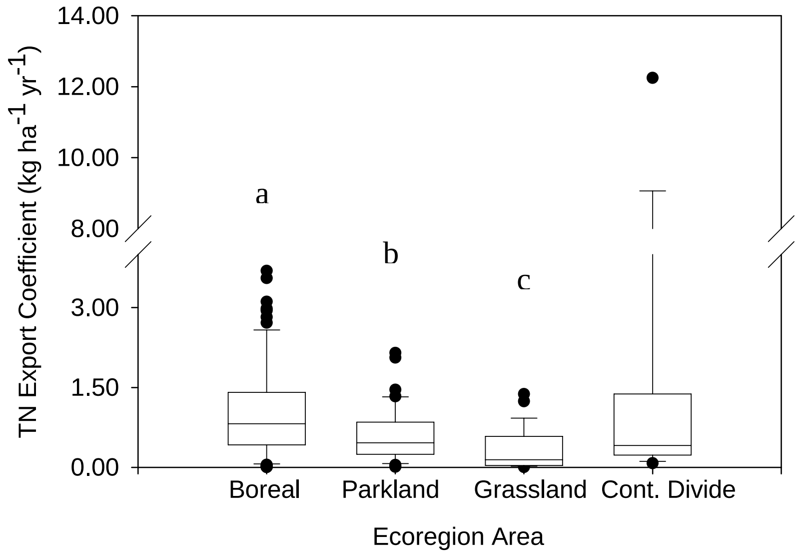


Figure 3.64. Box plots of median annual TN export coefficients (1999 to 2006) for the Boreal, Grassland, and Parkland ecoregion areas. Box plots with different letters are significantly different from one another at the 0.02 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the Boreal and Grassland ecoregion areas were different at the $p<0.005$ significance level.

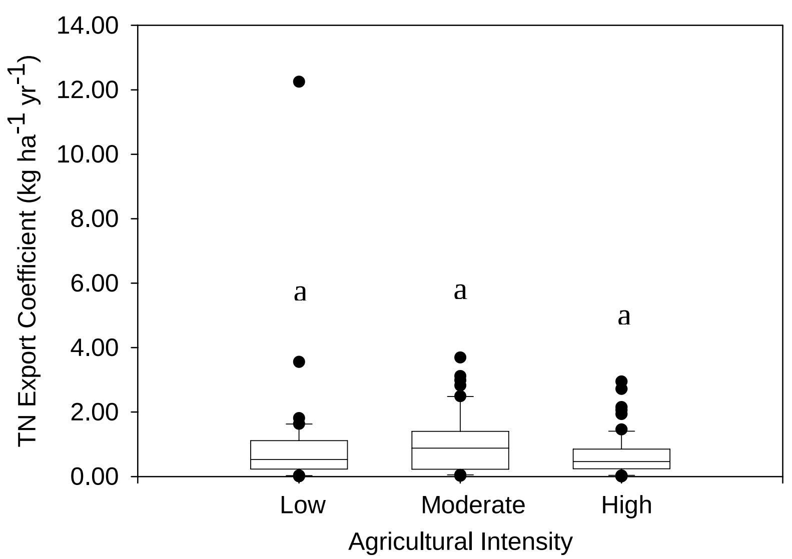


Figure 3.65. Box plots of median annual TN export coefficients (1999 to 2006) in the three dryland agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

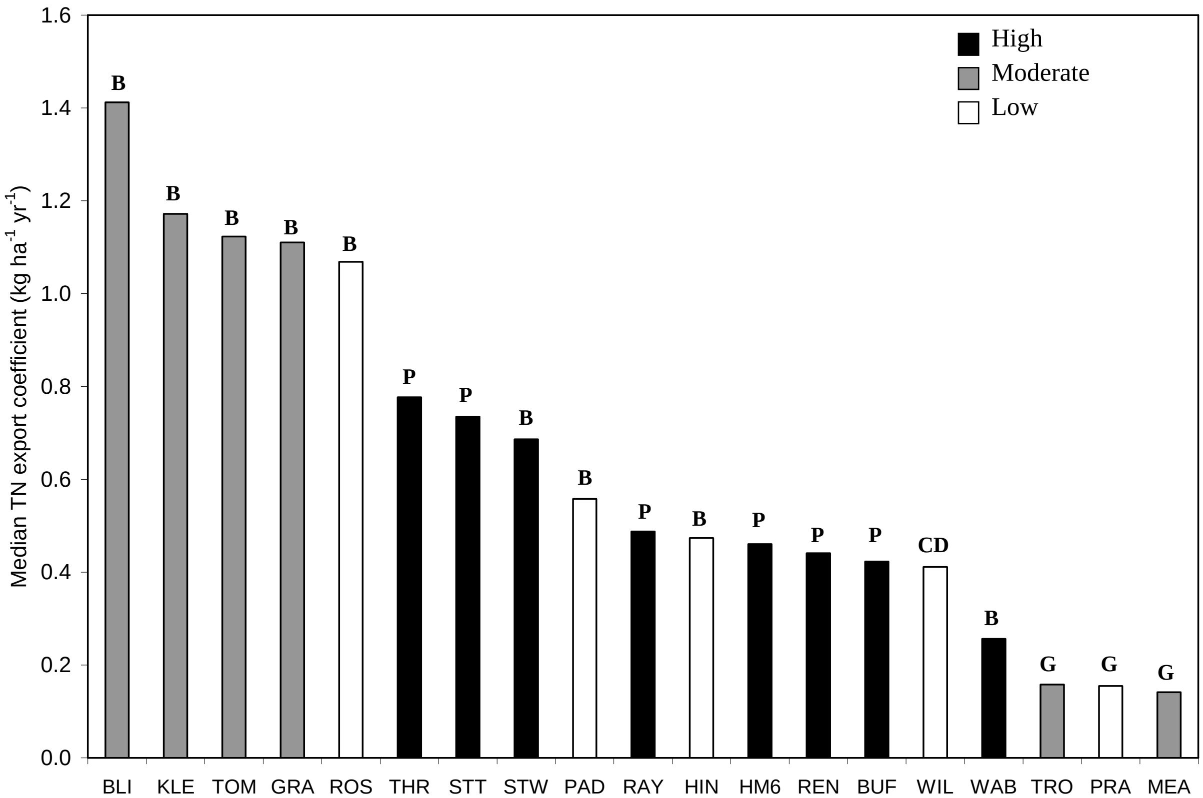


Figure 3.66. Median annual TN export coefficients for each dryland AESA watershed (1999 to 2006). Location by ecoregion area for each stream is illustrated as follows: Boreal (B), Parkland (P), and Grassland (G). For full stream names see, Table 3.5 (pg. 3-12).

Kruskal-Wallis One-Way ANOVA tests for TN export coefficients among streams within ecoregional or agricultural intensity categories were similar to TP Kruskal-Wallis statistics (Table 3.52); no significant difference was found among streams within the Parkland or Grassland ecoregion areas or in the high agricultural intensity category. Significant differences in TN export coefficients were observed among streams in the Boreal ecoregion area and low and moderate agricultural intensity groups.

Mean Rank TN export coefficients were significantly lower in Hines and Wabash Creeks than in the majority of the other streams in the Boreal ecoregion area (Table 3.53). Overall, a significant difference was not found in mean rank TN export coefficients among the remaining watersheds in the Boreal ($p>0.05$), although annual TN export coefficients in Paddle River were lower than a few of the streams or were borderline significantly different at the $p<0.05$ level (data not shown).

Table 3.52. Kruskal-Wallis One-Way ANOVA statistics for median annual TN export coefficients (1999 to 2006) among dryland streams grouped by ecoregion area and by agricultural intensity.

Ecoregion Area/Agricultural Intensity	Kruskal-Wallis Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	20.011	**0.010**	8
Parkland	2.950	0.769	5
Grassland	0.575	0.773	2
Low	9.664	**0.046**	4
Moderate	16.962	**0.005**	5
High	7.196	0.409	7

Note: **Bold** values indicate a significant difference at $p<0.05$.

Table 3.53. Mann-Whitney statistics comparing median annual TN export coefficients (1999 to 2006) among nine watersheds located in the Boreal ecoregion area. For full stream names, see Table 3.5 (pg. 3-12).

	BLI	GRA	HIN	KLE	PAD	ROS	STW	TOM	WAB
A.I.:	(M)	(M)	(L)	(M)	(L)	(L)	(H)	(M)	(H)
BLI	-								
GRA	41 0.345	-							
HIN	60 **0.003**	51 **0.046**	-						
KLE	39 0.462	34 0.834	15 0.074	-					
PAD	62 0.137	52 0.509	22 0.069	48 0.741	-				
ROS	55 0.589	40 0.537	8 **0.002**	42 0.643	21 **0.037**	-			
STW	64 0.217	50 0.877	31 0.190	47 0.939	37 0.396	56 0.537	-		
TOM	52 0.758	43 0.700	14 **0.009**	43 0.149	26 0.090	45 0.817	36 0.355	-	
WAB	58 **0.006**	54 **0.021**	38 0.529	51 **0.046**	46 0.141	58 **0.006**	51 **0.046**	57 **0.009**	-

Note: **Bold** values indicate a significant difference at $p<0.05$. The top value is the Mann-Whitney statistic (U); the bottom value is the significance level. The degrees of freedom for all Mann-Whitney statistics is 1.

As for P, TN export coefficients were split within the moderate agricultural intensity category between those watersheds in different ecoregion areas; however, ecoregional characteristics did not have the same influence within the low agricultural intensity category. Those watersheds located in the Boreal ecoregion area (Blindman River, Kleskun Drain, and Tomahawk and Grande Prairie Creeks) had significantly higher annual TN export coefficients than Trout and Meadow Creeks, which were located in the Grassland ecoregion area. Among the low agricultural intensity watersheds, Rose Creek had significantly higher annual TN export coefficients than the other four low agricultural intensity streams (Table 3.54). With the exception of TN export coefficients being higher in Paddle River than in Prairie Blood Coulee, a significant difference was not found among mean rank TN export coefficients in the remaining low agricultural intensity watersheds ($p>0.05$).

Table 3.54. Mann-Whitney statistics comparing median annual TN export coefficients (1999 to 2006) among five watersheds located in the low agricultural intensity category. For full stream names, see Table 3.5 (pg. 3-12).

	HIN	PAD	PRA	ROS	WIL
HIN	-				
PAD	22 0.069	-			
PRA	37 0.600	71 **0.026**	-		
ROS	8 **0.002**	21 **0.037**	9 **0.003**	-	
WIL	27 0.600	37 0.916	17 0.115	45 0.172	-

Note: **Bold** values indicate a significant difference at $p<0.05$. The top value is the Mann-Whitney statistic (U); the bottom value is the significance level. The degrees of freedom for all Mann-Whitney statistics is 1.

Overall, TN export coefficients varied among agricultural intensity categories with more similarities found among streams located in the same ecoregion areas; however, not all streams fit into the pattern of export coefficients expected with each ecoregion area. Hines Creek, Paddle River, and Wabash Creek seemed to have lower TN export coefficients than other watersheds in the Boreal ecoregion area.

Wabash Creek had the lowest TN export coefficient (0.256 kg ha^{-1} yr^{-1}) within the Boreal ecoregion area. The low export coefficient may be a result of the watershed's location within agricultural zone 8, which was noted to be of lower agricultural development (Anderson et al. 1999). It could also be due to the low flows and precipitation in Wabash Creek (Chapter 2: Table 2.4). The Wabash watershed is unique among the AESA watersheds in that it has a municipal influence (Depoe and Westbrook 2003). This municipal discharge did not appear to influence nutrient exports in the watershed; however, it should be noted that municipal discharge occurred in the fall when the number of samples collected was lower and less frequent. Thus, fall sampling may have ended before the municipal discharge occurred, or sampling events may not have coincided with the discharge, thereby missing the event.

Hines Creek, Wabash Creek, and Blindman River were similar in size with respect to the effective drainage area of each watershed (Chapter 2: Table 2.1). However, their annual export coefficients were very different regardless of the nutrient or its form. Significantly lower P and N export coefficients were measured in Wabash Creek than in Blindman River. Hines Creek also had significantly lower export coefficients than Blindman River for many parameters. Nutrient loads (Appendix 7) were much lower in Hines and Wabash Creeks than in Blindman River as were the total annual flow volumes (Appendix 7). The median annual flow volume was 17 times higher in Blindman River than in Wabash Creek. Thus, the loads in Blindman River would have been expected to be higher. Export coefficients were lower in Hines and Wabash Creeks compared to Blindman River because smaller loading values were divided by an effective drainage area of similar size. Stream flow and runoff would have been influenced by climatic conditions in the area and landscape effects such as soil type, land cover, and topography (Elrashidi et al. 2005).

Although TN export coefficients were significantly different between Paddle River and Rose Creek, both low agricultural intensity watersheds are located within the Western Alberta Upland ecoregion. Paddle River had a much lower median TN export coefficient (0.558 kg ha^{-1} yr^{-1}) than Rose Creek (1.069 kg ha^{-1} yr^{-1}). The higher export coefficients observed in Rose Creek may be a result of the higher flow volumes recorded in the watershed. When looking at FWMCs for the two watersheds (Figures 3.26 and 3.29, pgs. 3-46 and 3-51, respectively), annual data were not higher in Rose Creek than in Paddle River, supporting the idea that higher runoff and flow in Rose Creek increased N exports from the stream.

Total N export coefficients for the AESA watersheds were compared with data from the CAESA study (Anderson et al. 1998b) and other studies in Alberta and outside of the province (Table 3.55). Median annual TN export coefficients from high agricultural intensity AESA watersheds were similar to those reported for high agricultural intensity watersheds in the CAESA study. Ranges for TN export coefficients in the AESA watersheds were broad with overlap between watershed groups, as was found in the CAESA study. However, low agricultural intensity CAESA watersheds were reported as exporting more TN than the moderate and high intensity streams. In contrast, moderate agricultural intensity AESA watersheds generally exported the most TN. Note that the watersheds under low, moderate, and high agricultural intensity categories differed between the AESA and CAESA studies. Total N export coefficients for the AESA watersheds were similar to those reported for other studies in Alberta but were generally lower than those reported from areas outside of the province. Annual TN export coefficients for the AESA watersheds were generally lower than TN export coefficients found in the literature for forested watersheds (Table 3.55). These differences are likely a result of variations in geology, landscape factors, and climatic conditions such as precipitation amount, duration, and intensity.

Table 3.55. Comparison of TN export coefficients from the AESA stream survey with data from other studies.

	Number of Streams	Land Cover (majority)	Median TN Export Coefficient ($kg \cdot ha^{-1} yr^{-1}$)	Source
Alberta				
AESA Streams (1999-2006)	19	Low, Mod., and High	0.574 (0.864 mean)	This study
	8	High Ag. Intenstiy	0.461	
	6	Moderate Ag. Intensity	0.877	
	5	Low Ag. Intensity	0.526	
CAESA Streams (1995+1996)				Anderson et al. (1998b)
	9	High Ag. Intenstiy	0.4814	
	12	Moderate Ag. Intensity	0.6796	
	6	Low Ag. Intensity	1.3026	
Haynes Creek (M6) (1995+1996)	1	High Ag. Intenstiy	0.7124	
Sakwatamau (1995 and 1996)	1	Forest	0.4763 and 5.2692	
Wabamun (1981)	9	Agriculture	0.85	Mitchell (1985)
Wabamun (1981)	10	Forest	0.66	
Baptiste (1977-78)	4	Agriculture	0.023	Trew et al (1987)
Baptiste (1977-78)	4	Forest	1.61	
Lac La Nonne (1981)	1	Agriculture	1.62	Mitchell and Hamilton (1982)
Ontario				
Experimental lakes watershed		Forest	0.9	Alvarez-Cobelas et al. 2008
Lake Simcoe (1990-1998)		Vegetable polder	25.4 (mean)	Winter et al. 2002
		Mixed agriculture	2.2 to 7.9 (range)	
		Forest	1.7 to 2.7 (range)	
Streams Outside of Canada				
State of Wisconsin (1973-1982)*	3	Forest	3.72 (mean)	Clesceri et al. 1986
State of Wisconsin (1973)	6	Agriculture	6.69 (mean)	
Ohio (1994-1998)				Vanni et al. 2001
Four Mile Creek		Agriculture	~ 40 (mean)	
Little Four Mile Creek		Agriculture	~ 55 (mean)	
Marshall's Creek		Agriculture	~ 30 (mean)	
Sweden (1998/1993-2006)	27	Agriculture	2.4 to 40.9 (range of means)	Kyllmar et al. 2006

*stream 1: 1975-1977; stream 2: 1973/75-1982; stream 3: 1973

Nitrite-N + Nitrate-N (NO_2^--N+NO_3^--N) export coefficients. Median annual NO_2^-N+NO_3^--N export coefficients were influenced by both agricultural and ecoregional factors. Mean Rank NO_2^-N+NO_3^--N export coefficients were significantly higher in the Boreal and Parkland ecoregion areas than in the Grassland ecoregion (Boreal: U = 1161.5, p = 0.012, df = 1; Parkland: U = 312.5, p = 0.008, df = 1) (Figure 3.67). However, there was no significant difference between mean rank export coefficients in the Boreal and Parkland ecoregion areas (U = 1445.5, p = 0.554, df = 1). Export coefficients in the Continental Divide appeared to be more similar to those in the Boreal ecoregion area than the Grassland ecoregion. The same pattern was noted for TDP export coefficients grouped by ecoregion area. Annual NO_2^-N+NO_3^--N export coefficients were significantly lower in the low agricultural intensity watersheds than in the moderate (U=578, p=0.001) or high (U=712, p=0) agricultural intensity streams (Figure 3.68). A significant difference between NO_2^-N+NO_3^--N export coefficients in the high and moderate agricultural intensity categories was not observed (U=1384, p=0.841).

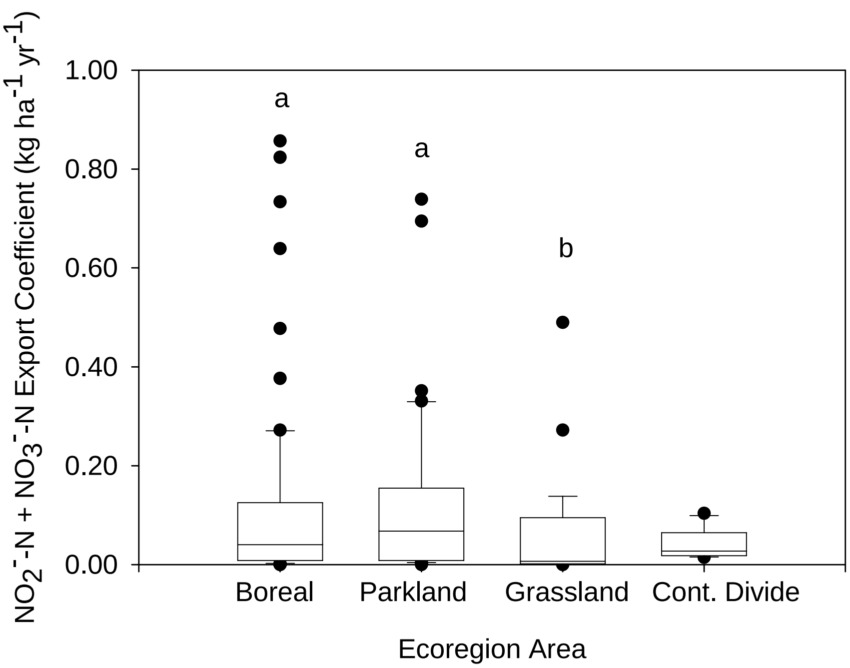


Figure 3.67. Box plots of median annual NO_2^--N+NO_3^--N export coefficients (1999 to 2006) for the Boreal, Grassland, and Parkland ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the Grassland ecoregion area and the Boreal and Parkland ecoregion areas were statistically different at the p<0. 05 and 0.01 significance levels, respectively.

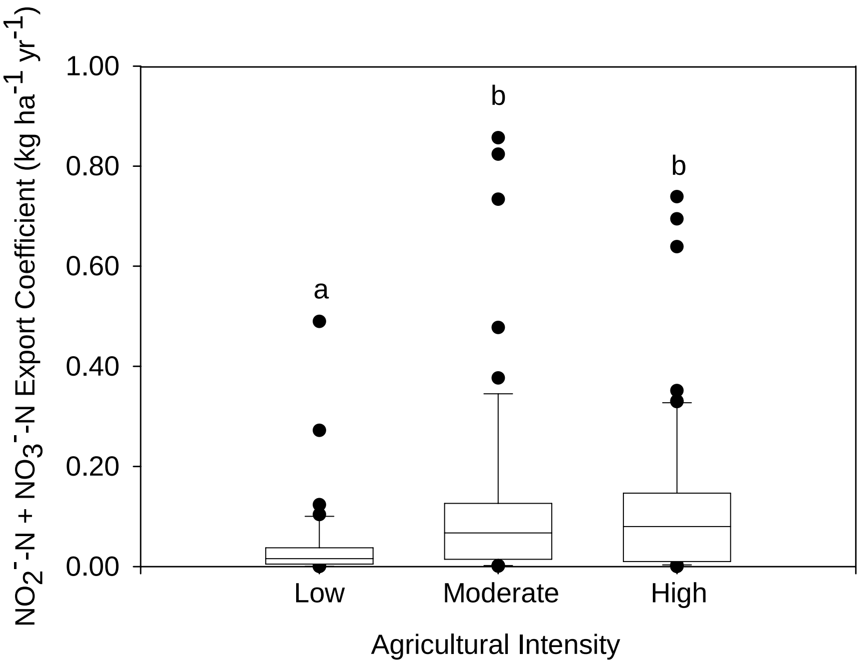


Figure 3.68. Box plots of median annual NO_2^--N+NO_3^--N export coefficients (1999 to 2006) in the three dryland agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots were significantly different between the low agricultural intensity watersheds and the moderate and high agricultural intensity groups at the $p<0.005$ level.

There did not appear to be a clear pattern among annual NO_2^--N+NO_3^--N export coefficients and agricultural intensity or location by ecoregion (Figure 3.69). Stretton (0.212 kg ha^{-1} yr^{-1}), Tomahawk (0.127 kg ha^{-1} yr^{-1}), and Threehills Creeks (0.124 kg ha^{-1} yr^{-1}) had the highest median annual NO_2^--N+NO_3^--N export coefficients. Prairie Blood Coulee (0.007 kg ha^{-1} yr^{-1}), Meadow Creek (0.007 kg ha^{-1} yr^{-1}), and Hines Creek (0.003 kg ha^{-1} yr^{-1}) had the lowest median annual NO_2^--N+NO_3^--N export coefficients. The three streams with the highest NO_2^--N+NO_3^--N export coefficients were classified in 1996 as either high or moderate agricultural intensity watersheds, while the three streams with the lowest export coefficients were classified as draining watersheds under low or moderate agricultural intensity. Although two of the three streams located in the Grassland ecoregion area had the lowest annual export coefficients with the third Grassland stream in the bottom five, watersheds located in the Boreal and Parkland ecoregion areas were mixed among having the highest and mid-range NO_2^--N+NO_3^--N export coefficients. Kruskal Wallis One-Way ANOVA tests showed that mean rank annual NO_2^--N+NO_3^--N export coefficients were not significantly different among streams within the high agricultural intensity category ($H=8.902$, $p=0.260$, 7df) or within the Parkland ($H=7.465$, $p=0.188$, 5df) or Grassland ecoregion areas ($H=0.02$, $p=0.99$, 2df). However, statistical analysis showed a significant difference in annual NO_2^--N+NO_3^--N export coefficients among streams within the Boreal ecoregion area ($H=29.356$, $p=0$, 8df) and the low ($H=13.268$, $p=0.010$, 4df) and moderate ($H=14.412$, $p=0.013$, 5df) agricultural intensity categories.

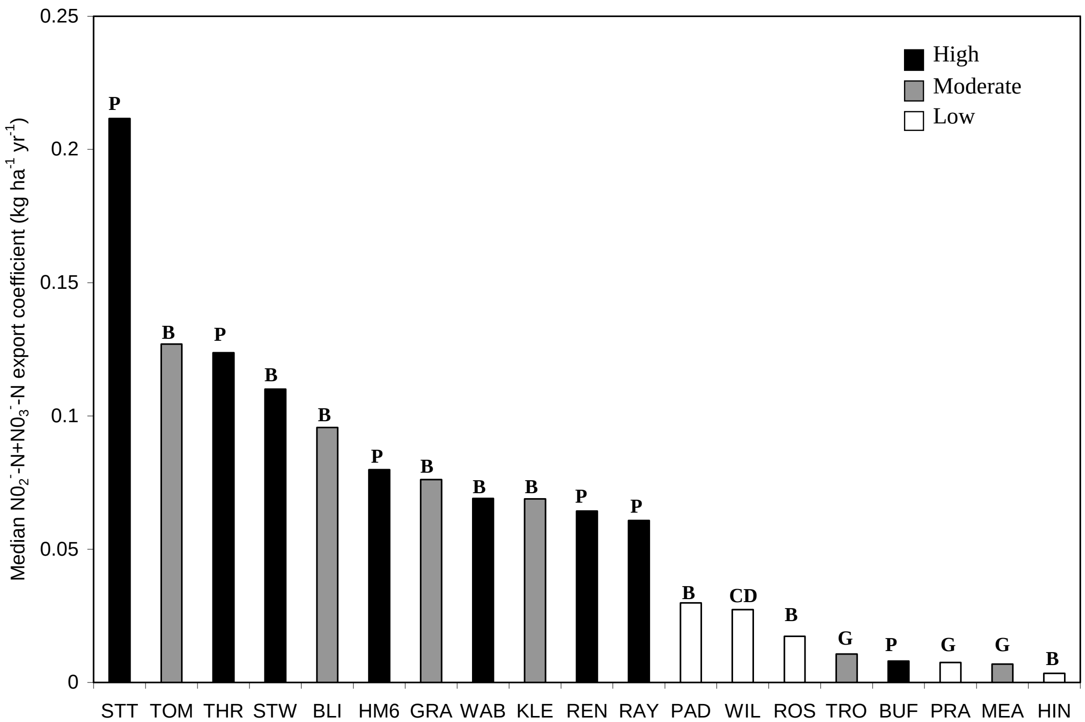


Figure 3.69. Median annual NO_2^--N+NO_3^--N export coefficients for each dryland AESA watershed (1999 to 2006). Location by ecoregion area for each stream is illustrated as follows: Boreal (B), Parkland (P), and Grassland (G). For full stream names, see Table 3.5 (pg. 3-12).

Agricultural intensity influenced annual NO_2^--N+NO_3^--N export coefficients within the Boreal ecoregion area. Mann-Whitney tests between streams showed that mean rank NO_2^--N+NO_3^--N export coefficients in Hines Creek were significantly lower than those measured in all other watersheds in the Boreal ecoregion area, while export coefficients in Paddle River and Rose Creek were significantly lower than export coefficients measured in the remaining streams in the Boreal ecoregion area with the exception of Kleskun Drain (Table 3.56). There was no difference between NO_2^--N+NO_3^--N export coefficients in the remaining streams in the Boreal ecoregion area, all of which were moderate and high agricultural intensity streams. Furthermore, Mann-Whitney tests run on NO_2^--N+NO_3^--N export coefficients by agricultural intensity category within the Boreal ecoregion area showed low agricultural intensity streams (Hines Creek, Rose Creek, Paddle River) had significantly lower mean rank export coefficients than streams under moderate ($H=94$, $p=0$) or high ($H=293$, $p=0.005$) agricultural intensity. There was no difference between NO_2^--N+NO_3^--N export coefficients in moderate and high agricultural intensity watersheds in the ecoregion area ($p>0.05$).

As observed for P, ecoregional characteristics played a stronger role in NO_2^--N+NO_3^--N export coefficients among watersheds draining moderate agricultural intensity land. A statistical difference was not observed between the two Grassland streams, Trout and Meadow Creeks. However, mean rank NO_2^--N+NO_3^--N export coefficients in both Grassland streams were significantly lower than those in the remaining moderate agricultural intensity watersheds, emphasizing the differences in nutrient export coefficients between the Boreal and Grassland ecoregion areas.

Table 3.56. Kruskal-Wallis One-Way ANOVA statistics for median annual NO_2^--N+NO_3^--N export coefficients (1999-2006) among dryland streams grouped by ecoregion and by agricultural intensity.

Ecoregion Area/Agricultural Intensity	Kruskal-Wallis Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	29.356	**0.000**	8
Parkland	7.465	0.188	5
Grassland	0.020	0.990	2
Low	13.268	**0.010**	4
Moderate	14.412	**0.013**	5
High	8.902	0.260	7

Note: **Bold** values indicate a significant difference at p<0.05.

Among the low agricultural intensity watersheds, Hines Creek was the stream with mean rank NO_2^--N +NO_3^--N export coefficients that were significantly lower than export coefficients in all other low agricultural intensity watersheds with the exception of Prairie Blood Coulee (Table 3.57). Mann-Whitney statistics did not show any difference among the other low agricultural intensity watersheds. Hines Creek did not necessarily have lower manure production, fertilizer sales, or chemical sales percentiles (Chapter 2: Figures 2.23 to 2.26, Table 2.11). Over half of the landscape surrounding Hines Creek is covered in trees and shrubs with only 1% forage, and 41% of the watershed is not classified as it falls outside of Alberta's White Zone (Appendix 1). Rose Creek and Paddle River also have high percentages of tree and shrub cover (72 and 67%, respectively) and little or no cropland. Interestingly, Hines Creek has the second highest effective drainage area of the low agricultural intensity streams with the lowest median annual load from 1999 to 2006 (Appendix 7). Median annual stream volume was not necessarily low, but the lower load may have resulted from different management practices in the watershed compared to other low agricultural intensity streams or from differences in the intensity or duration of precipitation or rainfall events. Stream flow historically peaks in the spring (April) with smaller peaks in June and August in Hines Creek (Chapter 2: Figures 2.14 and 2.15). In contrast, stream flow peaks in Rose Creek and Paddle River are longer, lasting from spring to mid summer. Stream flow in Willow Creek also peaks for a longer duration in the late spring/early summer. These differences in stream flow that arise from climatic conditions specific to the Ecoregion the watershed is located in may affect the timing and quantity of nutrient loading and subsequent exports.

Stretton Creek had the highest median annual NO_2^--N+NO_3^--N export coefficient as well as the highest median annual TDP export coefficient. However, the calculated value may be misleading as samples were not collected between 2000 and 2003 when the stream did not flow, removing any potentially lower annual export values from the median annual value. Although the stream had the highest median annual NO_2^--N+NO_3^--N export coefficient, mean ranks were not significantly different from any of the other high agricultural intensity watersheds (H=8.902, p=0.260, 7df) or from those streams located in the Parkland ecoregion (H=7.465, p=0, 5df).

Table 3.57. Mann-Whitney statistics comparing median annual $NO2^-$-N+$NO3^-$-N export coefficients (1999 to 2006) among five watersheds located in the low agricultural intensity category. For full stream names, refer to Table 3.5 (pg. 3-12).

	HIN	PAD	PRA	ROS	WIL
HIN	-				
PAD	5 **0.001**	-			
PRA	24 0.401	53 0.457	-		
ROS	4 **0.001**	61 0.316	39 0.487	-	
WIL	0 **0.001**	28 0.674	24, 0.401	14 0.059	-

Note: **Bold** values indicate a significant difference at $p<0.05$. The top value is the Mann-Whitney statistic (U); the bottom value is the significance level. The degrees of freedom for all Mann-Whitney statistics is 1.

Ammonia-N (NH_3-N) export coefficients. Median annual NH_3-N export coefficients exhibited similar patterns to TDP; mean rank NH_3-N export coefficients for all of the dryland watersheds were significantly different among ecoregion areas (H = 17.815, p = 0, 2df) (Figure 3.70). Export coefficients in the Boreal and Parkland ecoregion areas were significantly higher than NH_3-N export coefficients in the Grassland ecoregion (Boreal: U = 1439, p = 0; Parkland: U = 186.5, p = 0). As with NO_2^--N+NO_3^--N, there was no significant difference between mean rank NH_3-N export coefficients in the Boreal and Parkland ecoregion areas (U = 1607, p = 0.733). Export coefficients in the Continental Divide were similar to those in the Grassland ecoregion. Export coefficients were not significantly different between low and moderate agricultural intensity watersheds (U=770, p=0.111) or moderate and high agricultural intensity watersheds (U=1325, p=0.569) (Figure 3.71). However, mean rank NH_3-N export coefficients were significantly lower in the low agricultural intensity category than in the high one (U=815, p=0.009).

Annual export of NH_3-N was less than all other N parameters regardless of the stream (Figure 3.72). The top three exporting streams were Blindman River (0.144 kg ha^{-1} yr^{-1}), Threehills Creek (0.110 kg ha^{-1} yr^{-1}), and Tomahawk Creek (0.072 kg ha^{-1} yr^{-1}). The lowest median annual NH_3-N export coefficients were measured in Trout Creek (0.006 kg ha^{-1} yr^{-1}), Meadow Creek (0.005 kg ha^{-1} yr^{-1}), and Prairie Blood Coulee (0.005 kg ha^{-1} yr^{-1}). Note that the three streams with the lowest median annual NH_3-N export coefficients are all located in the grassland ecoregion area. Similar to NO_2^--N+NO_3^--N export coefficients, the top three streams with the highest NH_3-N export coefficients were classified in 1996 as high and moderate agricultural intensity watersheds. The three watersheds with the lowest NH_3-N export coefficients were classified as having moderate or low agricultural intensity.

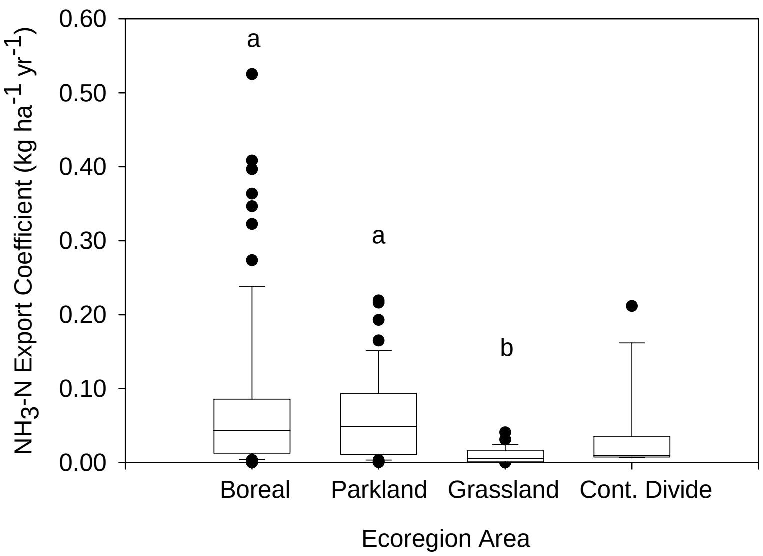


Figure 3.70. Box plots of median annual NH_3-N export coefficients (1999 to 2006) for the Boreal, Grassland, and Parkland ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the Grassland and other ecoregion areas were statistically different at the $p<0.005$ significance level.

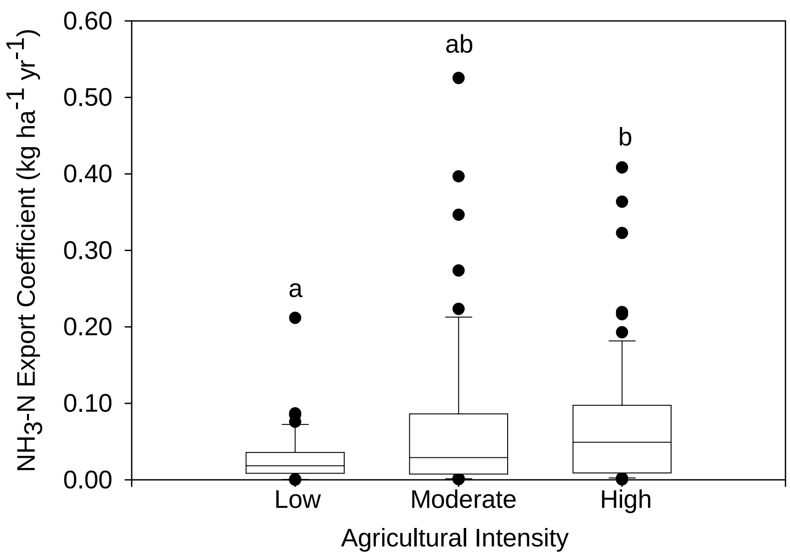


Figure 3.71. Box plots of median annual NH_3-N export coefficients (1999 to 2006) in the three dryland agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots for low and high agricultural intensity watersheds were statistically different at the $p<0.01$ significance level.

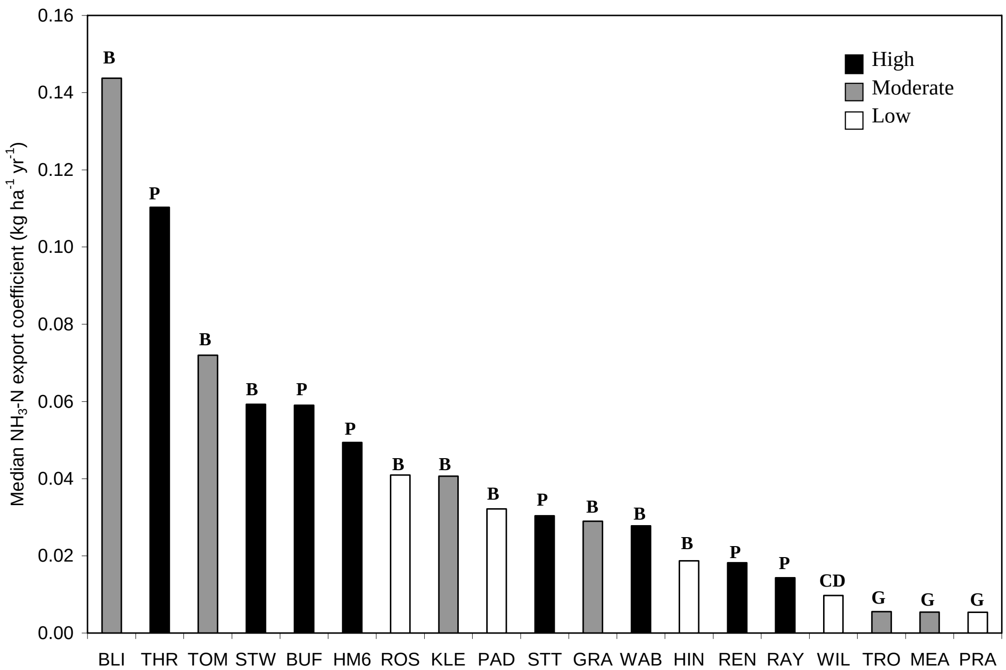


Figure 3.72. Median annual NH_3-N export coefficients for each dryland AESA watershed (1999 to 2006). Location by ecoregion area for each stream is illustrated as follows: Boreal (B), Parkland (P), and Grassland (G). For full stream names, see Table 3.5 (pg. 3-12).

Kruskal-Wallis One-Way ANOVA statistics showed no significant difference among streams within the Parkland or Grassland ecoregion areas or within the high agricultural intensity category (Table 3.58). As for NO_2^--N+NO_3^--N export coefficients, a significant difference was observed among NH_3-N export coefficients in streams within the Boreal ecoregion area and within the low ($p<0.01$) and moderate ($p<0.005$) agricultural intensity categories. No clear pattern was found among NH_3-N export coefficients in the Boreal ecoregion area although export coefficients in the low agricultural intensity streams in the region were significantly lower than those measured in watersheds under moderate agricultural intensity ($U=225$, $p=0.008$, 1df). No significant difference was found among the different agricultural intensity categories within the Boreal ecoregion area ($p>0.05$).

Table 3.58. Kruskal-Wallis One-Way ANOVA statistics for median annual NH_3-N export coefficients (1999 to 2006) among dryland streams grouped by ecoregion area and by agricultural intensity.

Ecoregion Area/Agricultural Intensity	Kruskal-Wallis Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	20.267	**0.009**	8
Parkland	8.146	0.148	5
Grassland	1.145	0.564	2
Low	10.798	**0.029**	4
Moderate	24.779	**0.000**	5
High	9.652	0.209	7

Note: **Bold** values indicate a significant difference at p<0.05.

Organic Nitrogen (Org N) export coefficients. Annual Org N export coefficients showed similar patterns by ecoregion area and agricultural intensity to TN and TP. Median Org N export coefficients in the Boreal ecoregion area were significantly higher than median export coefficients in both the Parkland (U=3022, p<0.005) and Grassland ecoregion areas (U=1626 p<0.005) (Figure 3.73). The top 5 annual Org N exporters were Blindman Creek (1.163 kg ha^{-1} yr^{-1}), Rose Creek (1.016 kg ha^{-1} yr^{-1}), Grande Prairie Creek (0.010 kg ha^{-1} yr^{-1}), Kleskun Drain (0.954 kg ha^{-1} yr^{-1}), and Tomahawk Creek (0.892 kg ha^{-1} yr^{-1}) - all located in the Boreal ecoregion area (Figure 3.75). Mean rank Org N export coefficients were also significantly higher in the Parkland streams than Grassland watersheds (U=487, p<0.01). The three streams with the lowest annual Org N export coefficients were the three Grassland watersheds (Prairie Blood Coulee, 0.102 kg ha^{-1} yr^{-1}; Meadow Creek, 0.130 kg ha^{-1} yr^{-1}; and Trout Creek, 0.142 kg ha^{-1} yr^{-1}) (Figure 3.75). As for TN, median Org N export coefficients in the Continental Divide appeared to be similar to those measured in the Parkland ecoregion. Note that a significant difference was not observed among mean rank Org N export coefficients when grouped by agricultural intensity (U=5.93, p=0.052) (Figure 3.74), which supported the assumption that Org N export coefficients were most influenced by ecoregional characteristics. Organic N export coefficients were higher than NO_2^- N+NO_3^--N and NH_3-N export coefficients.

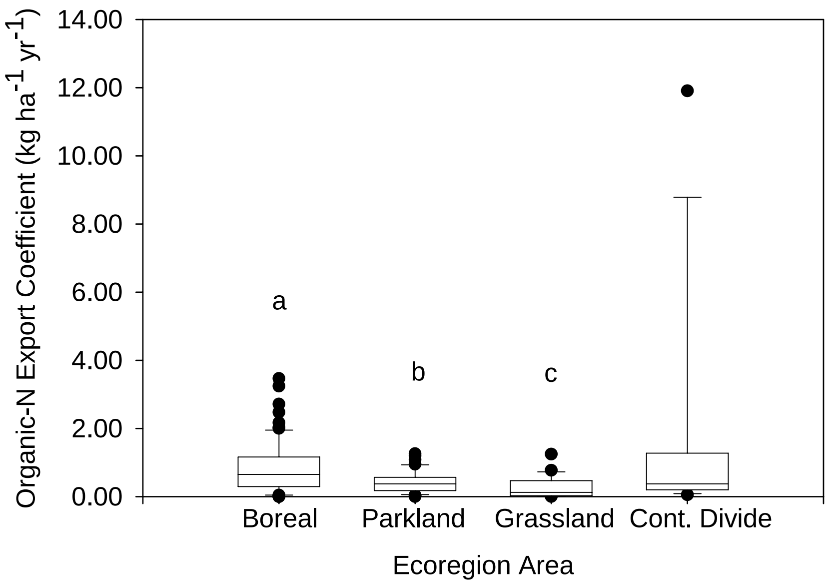


Figure 3.73. Box plots of median annual Org N export coefficients (1999 to 2006) for the Boreal, Grassland, and Parkland ecoregion areas. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Box plots between the Boreal and other two ecoregion areas were statistically different at the $p<0.005$ significance level.

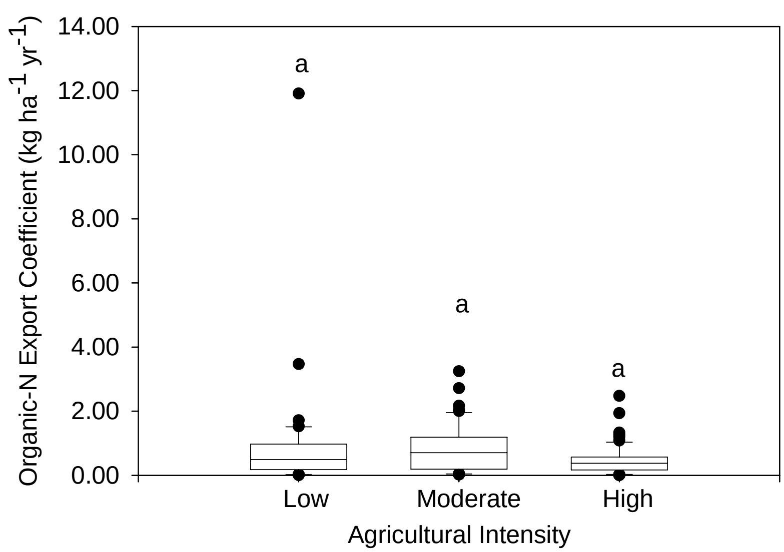


Figure 3.74. Box plots of median annual Org N export coefficients (1999 to 2006) in the three dryland agricultural intensity categories. Box plots with the same letter are not significantly different from one another at the 0.05 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

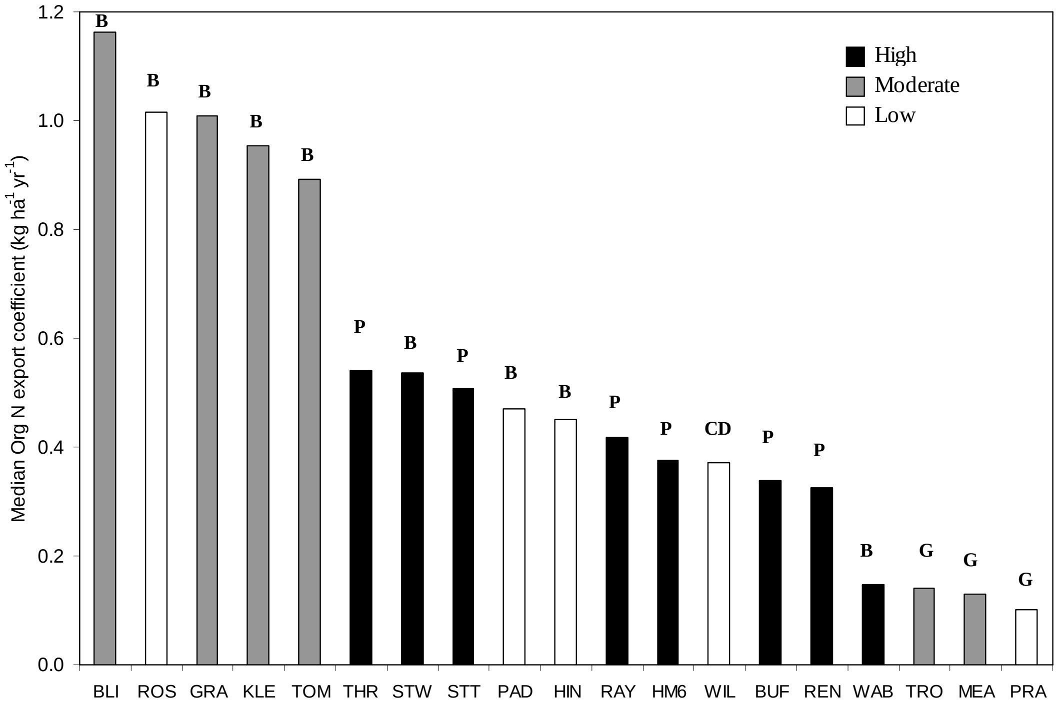


Figure 3.75. Median annual Org N export coefficients for each dryland AESA watershed (1999 to 2006). Location by ecoregion area for each stream is illustrated as follows: Boreal (B), Parkland (P), and Grassland (G). For full stream names, see Table 3.5 (pg. 3-12).

Similar to the dissolved fractions of N, statistical analyses showed mean rank Org N export coefficients were not significantly different among streams within the Parkland or Grassland ecoregion areas or within the high agricultural intensity category (Table 3.59). Annual Org N export coefficients were significantly different among streams with in the Boreal ecoregion area as well as within the low or moderate agricultural intensity groups.

Table 3.59. Kruskal-Wallis One-Way ANOVA statistics for median annual Org N export coefficients (1999 to 2006) among dryland streams grouped by ecoregion area and by agricultural intensity.

Ecoregion Area/Agricultural Intensity	Kruskal-Wallis Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	21.854	**0.005**	8
Parkland	3.614	0.606	5
Grassland	0.905	0.636	2
Low	11.407	**0.022**	4
Moderate	15.657	**0.008**	5
High	0.919	0.193	7

Note: **Bold** values indicate a significant difference at p<0.05.

Annual Org N export coefficients did not appear to be influenced by agricultural intensity within the Boreal ecoregion area. Export coefficients in Wabash Creek were significantly lower than all other streams in the Boreal with the exception of those measured in Hines Creek (Table 3.60). Moreover, Hines Creek had significantly lower annual Org N export coefficients than the remaining seven streams in the Boreal ecoregion area.

Unlike many of the other parameters, mean rank Org N export coefficients were not significantly different between the two ecoregion areas covered by the moderate agricultural intensity watersheds (Table 3.61). Both Meadow and Trout Creeks had Org N export coefficients significantly lower than those measured in Blindman River, Grande Prairie Creek, and Tomahawk Creek. However, no significant difference was found between mean rank Org N export coefficients measured in Trout and Meadow Creeks and Kleskun Drain.

Median Org N export coefficients were correlated with median annual runoff depth as shown in Figure 3.76. Specifically, a positive relationship was observed in the Boreal ecoregion area (Figure 3.77), which supports the statistical differences in export coefficients observed among the Boreal watersheds. The Spearman rank-order correlation coefficient for all dryland watersheds was 0.63 ($p<0.01$). The correlation coefficient was strongest for median annual Org N export coefficients from streams located in the Boreal ecoregion area ($r_s = 0.88$, $p<0.01$) although a positive relationship was also found for Org N export coefficients in the Parkland ecoregion ($r_s = 0.54$, $p>0.05$). Note that the relationship between runoff depth and Org N export coefficients by ecoregion area appeared to be much stronger than for TPP export coefficients.

Table 3.60. Mann-Whitney statistics comparing median annual Org N export coefficients (1999 to 2006) among nine watersheds located in the Boreal ecoregion area. For full stream names see, Table 3.5 (pg. 3-12).

	BLI	GRA	HIN	KLE	PAD	ROS	STW	TOM	WAB
A.I.:	(M)	(M)	(L)	(M)	(L)	(L)	(H)	(M)	(H)
BLI	-								
GRA	40 0.345	-							
HIN	**58** **0.006**	47 0.115	-						
KLE	41 0.345	36 0.067	16 0.093	-					
PAD	60 0.186	48 0.071	23 0.083	44 1.000	-				
ROS	48 1.000	33 0.247	**9** **0.003**	32 0.217	**19** **0.025**	-			
STW	64 0.217	49 0.939	33 0.247	45 0.817	39 0.487	60 0.355	-		
TOM	55 0.589	42 0.643	**16** **0.014**	41 0.589	27 0.105	49 0.939	36 0.355	-	
WAB	**62** **0.002**	**54** **0.021**	49 0.074	**54** **0.021**	**52** **0.036**	**62** **0.002**	**52** **0.036**	**60** **0.003**	-

Note: **Bold** values indicate a significant difference at $p<0.05$. The top value is the Mann-Whitney statistic (U); the bottom value is the significance level. The degrees of freedom for all Mann-Whitney statistics is 1.

Table 3.61. Mann-Whitney statistics for annual Org N export coefficients from 1999 to 2006 for Trout and Meadow Creeks compared to the four other moderate agricultural intensity watersheds.

Moderate Agricultural Intensity Watershed	Trout Kruskal-Wallis Statistic (U)	Trout Level of Significance (p)	Meadow Kruskal-Wallis Statistic (U)	Meadow Level of Significance (p)
Blindman River	86	**0.003**	90	**0.001**
Grande Prairie Creek	52	**0.036**	55	**0.016**
Kleskun Drain	46	0.141	48	0.093
Tomahawk Creek	83	**0.007**	10	**0.003**

Note: **Bold** values indicate a significant difference at $p<0.05$. The degrees of freedom for all Mann-Whitney statistics is 1.

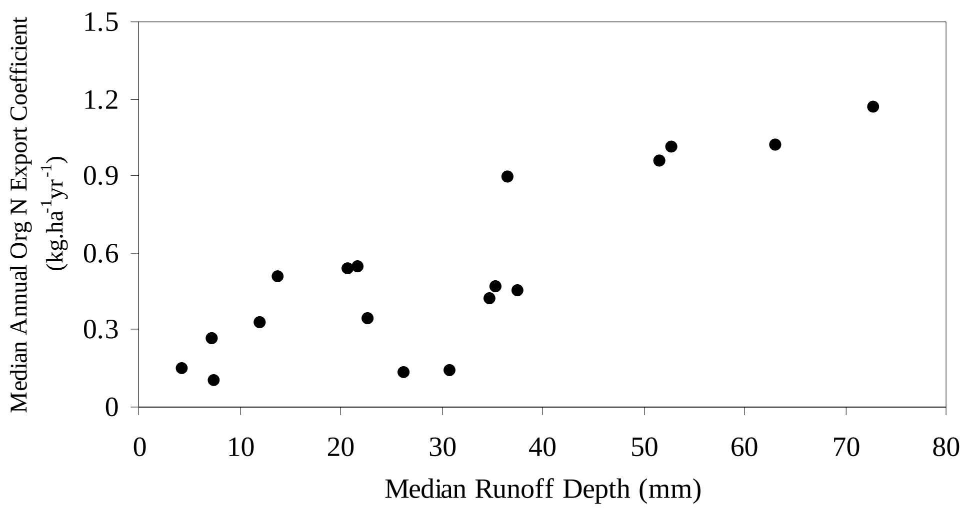


Figure 3.76. Median annual Org N export coefficients (1999 to 2006) compared to median annual runoff depth for the 18 dryland AESA watersheds.

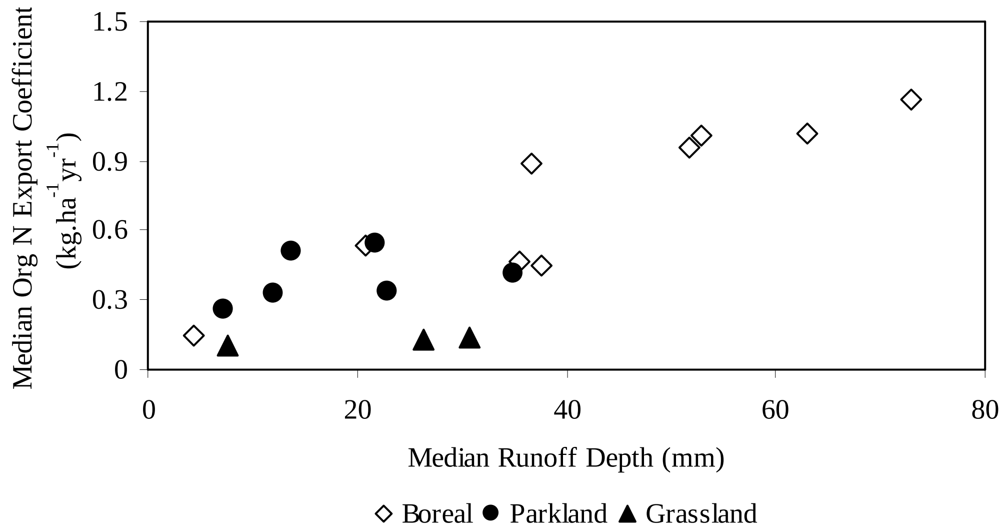


Figure 3.77. Median annual Org N export coefficients (1999 to 2006) compared to median annual runoff depth for the 18 dryland AESA watersheds grouped by ecoregion area.

Dissolved Inorganic Nitrogen to Total Nitrogen export coefficient ratios (DIN/TN). When grouped by ecoregion area, the ratio of DIN to TN in the Boreal was significantly higher than the Grassland ecoregion (U = 39681 p = 0.015) and significantly lower than the Parkland ecoregion (U = 36388 p = 0) (Figure 3.78). Dissolved inorganic N to TN in the Parkland ecoregion was also significantly higher than the ratio measured in the Grassland watersheds (U = 9597 p = 0). Since all of the watersheds located in the Parkland ecoregion were classified as draining land under high agricultural intensity, the data support the conclusion from CAESA that the higher the agricultural intensity, the higher the ratio of dissolved nutrients (Anderson et al. 1998b). The ratio of DIN/TN in the Continental Divide was similar to the ratio in the Grassland ecoregion.

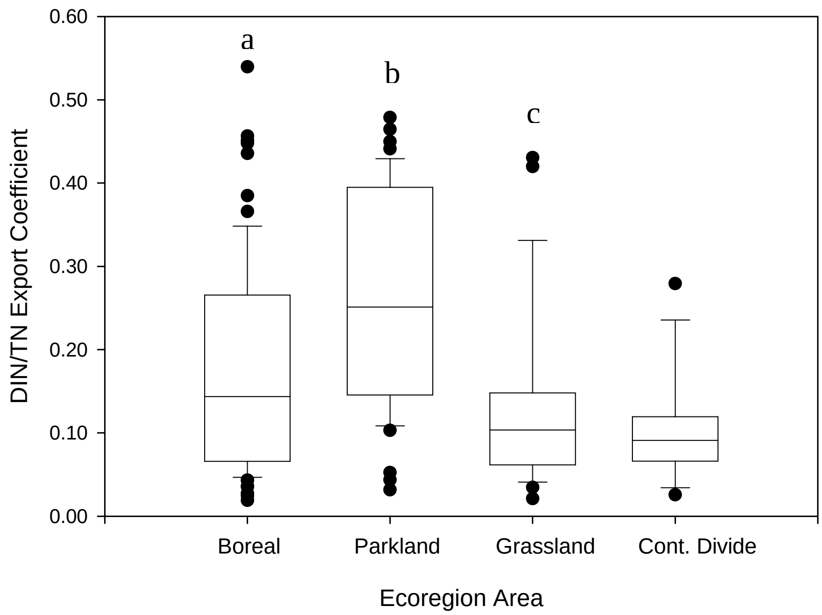


Figure 3.78. Box plots of median annual ratios of DIN/TN exports (1999 to 2006) for the Boreal, Grassland, and Parkland ecoregion areas. Medians of box plots with different letters are significantly different from one another at the 0.005 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Median ratios of DIN/TN were significantly different between the Boreal and Grassland ecoregion areas at $p<0.05$.

Ratios of DIN/TN showed a stepwise trend with increasing agricultural intensity similar to TDP exports when grouped by agricultural intensity (Figure 3.79). High agricultural intensity watersheds had significantly higher ratios than moderate (U=77032, $p<0.005$) or low (U=75557, $p<0.005$) agricultural intensity categories with the lowest ratios found in the low agricultural intensity watersheds (U=27729, $p<0.005$). High agricultural intensity watersheds had ratios ranging from 0.15 in Ray Creek to 0.40 in Wabash Creek. The ratio of DIN/TN in moderate agricultural intensity watersheds ranged from 0.09 in Meadow Creek to 0.18 in Tomahawk Creek. Ratios ranged from 0.05 (Hines Creek) to 0.12 (Paddle River) in low agricultural intensity watersheds.

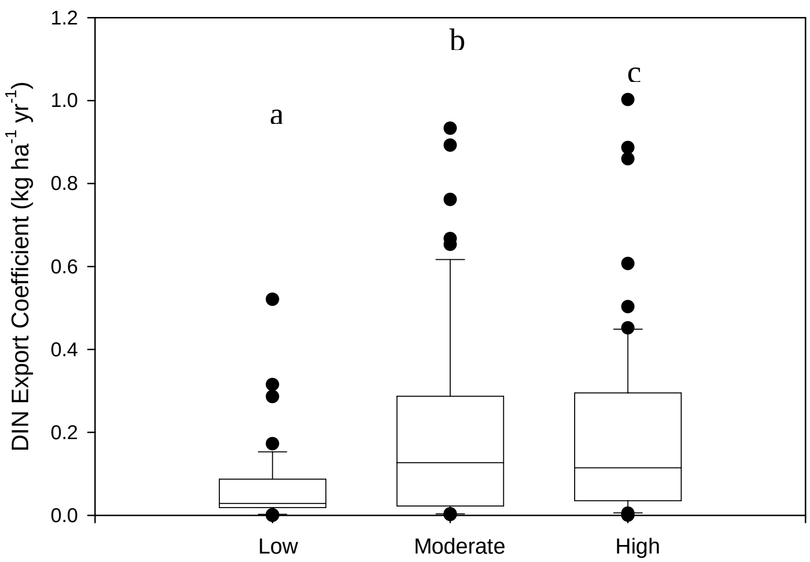


Figure 3.79. Box plots of median annual ratios of DIN/TN exports (1999 to 2006) in the three dryland agricultural intensity categories. Medians of box plots with different letters are significantly different from one another at the 0.005 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

As for TDP/TP, the ratio of TN as DIN differed among watersheds (Figure 3.80). Although the ratio of TN as DIN was statistically different among ecoregion areas and agricultural intensity groups, significant differences were also observed among streams within each category. Unlike TDP/TP, a statistical difference was only observed among streams in the Boreal and Parkland ecoregions and the moderate agricultural intensity watersheds (Table 3.62). Note that agricultural intensity and ecoregion areas are not completely independent. The type of agricultural production found in an area can be influenced by characteristics typical of a specific Ecoregion, such as climate and soil type.

Interestingly, the ratio of DIN/TN was the only parameter with a significant difference among watersheds within the Parkland ecoregion area. A statistically significant difference was not observed among Buffalo, Haynes, Ray, Renwick, and Stretton Creeks ($H=8.381$, $p=0.079$). Threehills Creek had a higher ratio of DIN/TN than Buffalo ($U=10$, $p<0.05$) and Ray Creeks ($U=9$, $p<0.02$) but did not statistically differ from the other watersheds in the Parkland ecoregion area.

Ratios on DIN/TN in the Boreal ecoregion area were most influenced by agricultural intensity rather than climatic, soil type, or topographical features. The ratio of DIN/TN in the Boreal watersheds was variable with the lowest and highest ratios measured in Hines (0.050) and Wabash Creeks (0.400), respectively (Figure 3.80). Statistical analyses showed that mean rank DIN/TN ratios were not significantly different among streams in the Boreal when the three low agricultural intensity watersheds in the region were removed ($H=6.617$, $p=0.246$). The ratio of TN as DIN was significantly lower in Hines Creek than in all other watersheds within the Boreal ecoregion area ($p<0.02$) with the exception of Rose Creek ($U=35$, $p=0.753$) and Paddle River ($U=17$, $p=0.115$). Mean rank DIN/TN exports were also significantly lower in Rose Creek than all other watersheds in the Boreal, including Paddle River. Paddle River only had mean rank DIN/TN exports statistically lower than mean rank exports in Blindman River ($U=52$, $p<0.05$) and Tomahawk and Strawberry Creeks ($U=4$, $p<0.005$).

Although P and N parameters were generally split within the moderate agricultural intensity category between those streams located within the Boreal ecoregion area (Blindman River, Grande Prairie Creek, Kleskun Drain, and Tomahawk Creek) and those within the Grassland ecoregion area (Meadow and Trout Creeks), the ratio of DIN/TN was only significantly different between Tomahawk and Meadow Creek ($U=5.00$, $p<0.005$). It is possible that agricultural intensity influenced the ratio of TN in the dissolved form more than ecoregional influences; however, it is uncertain why a difference existed between Tomahawk and Meadow Creeks.

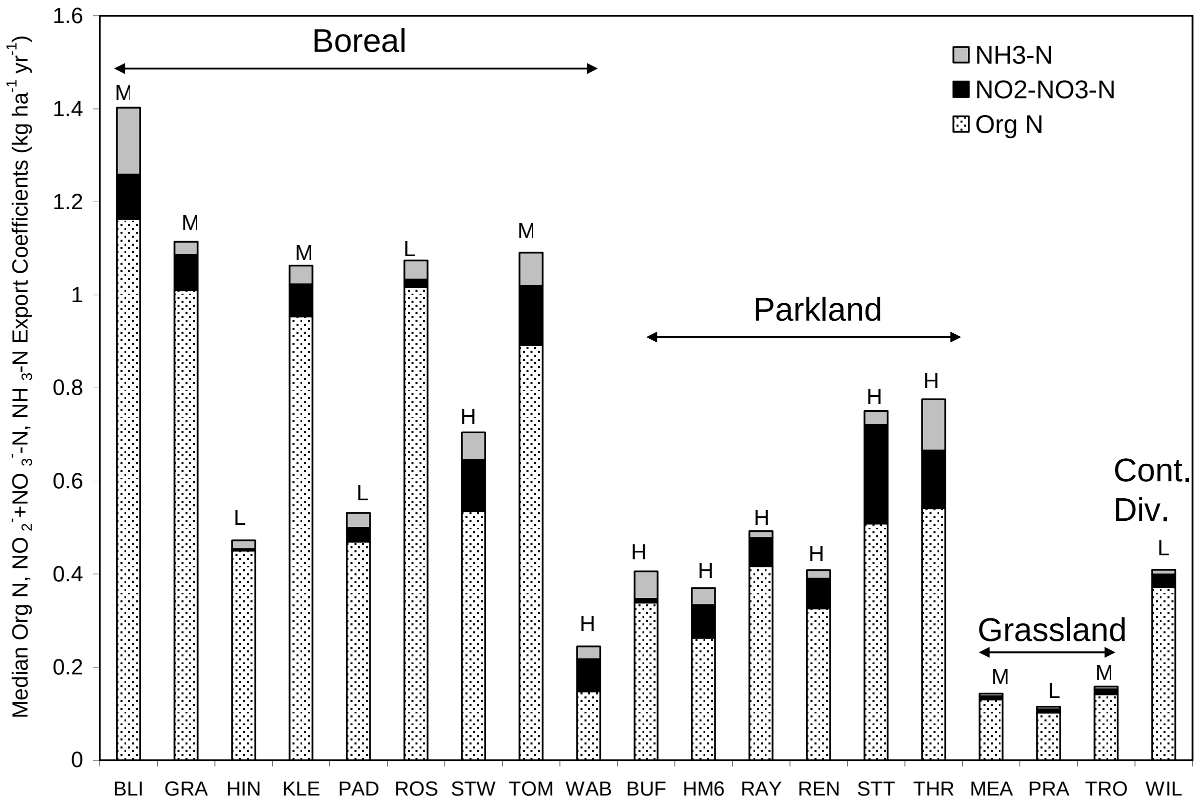


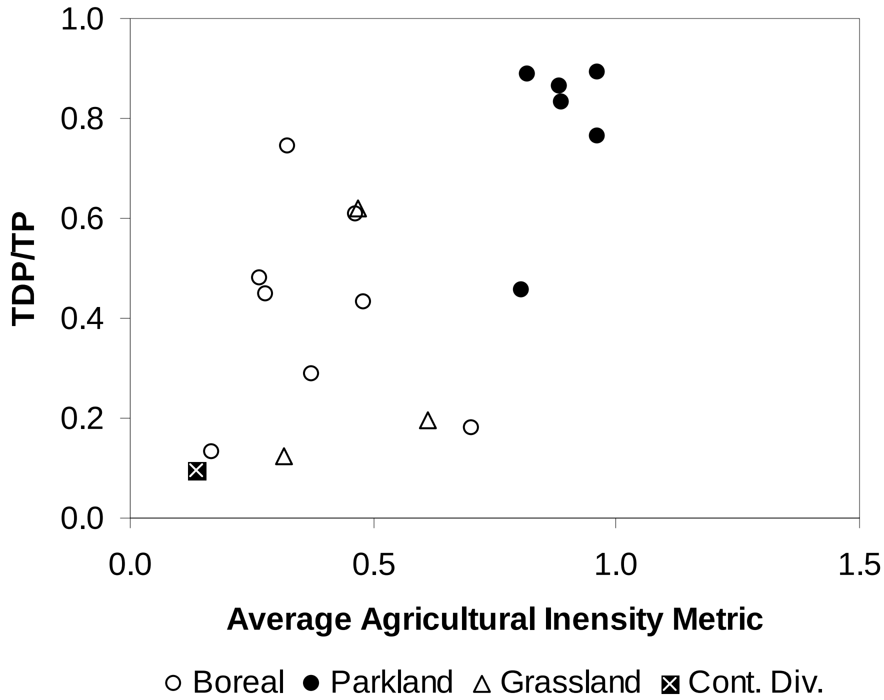
Figure 3.80. Median annual NO_2^-+NO_3^--N, NH_3-H, and Org N export coefficients for each dryland AESA watershed (1999 to 2006). Agricultural intensity for each stream is illustrated as follows: Low (L), Moderate (M), and High (H). For full stream names, see Table 3.5 (pg. 3-12).

Table 3.62. Kruskal-Wallis One-Way ANOVA statistics for median annual ratios of DIN/TN exports (1999 to 2006) among dryland streams grouped by ecoregion area and by agricultural intensity.

Ecoregion / Ag. Intensity	Kruskal-Wallis Test Statistic (H)	Level of Significance (p)	Degrees of Freedom
Boreal	38.408	**0.000**	8
Parkland	11.36	**0.045**	5
Grassland	0.140	0.932	2
High	14.032	0.051	7
Moderate	12.575	**0.028**	5
Low	7.074	0.132	4

Note: **Bold** values indicate a significant difference at $p<0.05$.

Correlations of P and N export coefficients with Agricultural Intensity (A.I.) metrics. In general, strong correlations were not observed between median annual P and N export coefficients for the AESA dryland streams and Census of Agriculture metrics averaged for the three years of data that were available (1996, 2001, and 2006). There was not a strong correlation between any of the agricultural intensity metrics with the total or dissolved fractions of N or P (Table 3.63). Interestingly, TPP export coefficients were strongly, negatively correlated with the A.I. intensity percentile (-0.57, $p<0.02$) and fertilizer and chemical sales percentiles (-0.66 and -0.65, respectively, $p<0.02$). The strongest correlations were observed with the ratios of TDP/TP and DIN/TN (Table 3.63). A very strong and positive correlation was found between TDP/TP and the average A.I. percentile (0.75, $p<0.02$, Figure 3.81) and average fertilizer and chemical sales percentiles (0.79 and 0.81, respectively, $p<0.02$). The ratio of DIN/TN was very strongly, positively correlated with all four metrics (Table 3.63). Note that DIN/TN was the only parameter that was strongly correlated with the average manure production percentile (0.77, $p<0.02$, Figure 3.81). Nitrogen is more available in manure than P and may account for the stronger correlation observed between DIN/TN and the average manure production percentile relative to that observed for TDP/TP. The stronger correlations between chemical sales and the ratios of TDP/TP and DIN/TN were unexpected considering that fertilizer and manure are nutrient sources. It is uncertain weather the metrics for nutrient sources are inadequate or if there is something inherent about the chemical sales percentiles (i.e., land use type) that is a better surrogate of nutrient use.



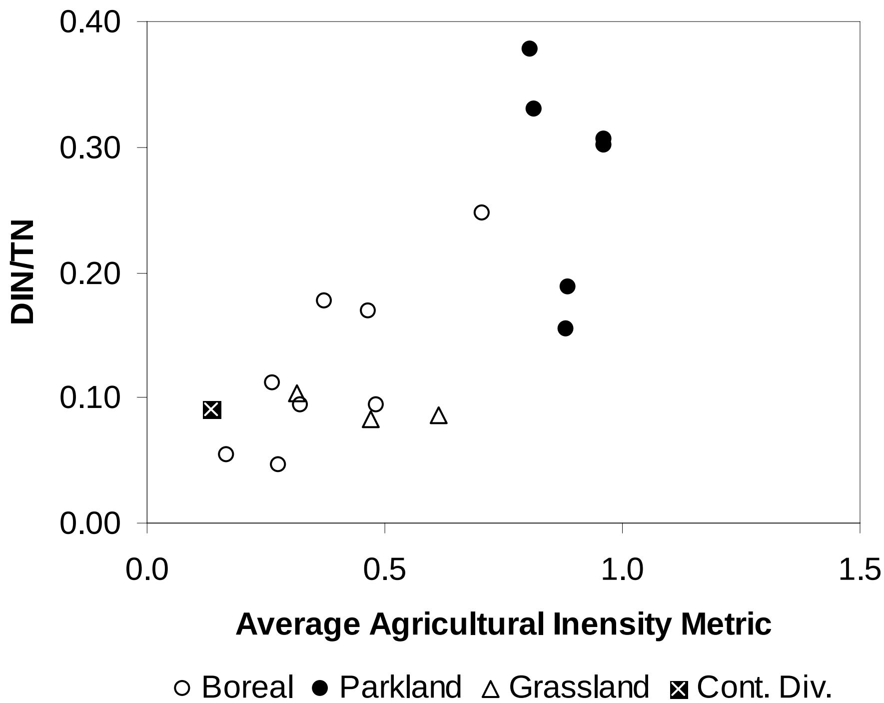


Figure 3.81. Average Agricultural Intensity (AI) Percentile (1999, 2001, and 2006) and the ratio of TDP/TP (a) and DIN/TN (b) for the AESA dryland agricultural watersheds.

Table 3.63. Spearman Rank Correlations between median annual P and N export coefficients (1999 - 2006) and average Census of Agriculture metrics (1996, 2001, and 2006) for 19 dryland AESA streams (n=19).

Median Anuual Export Parameter	Average Agricultural Intensity Percentile	Average Manure Production Percentile	Average Fertilizer Sales Percentile	Average Chemical Sales Percentile
TP	0.05	0.11	-0.03	-0.01
TDP	0.41	0.17	0.40	0.42
TPP	**-0.57**	-0.22	**-0.66**	**-0.65**
TDP/TP	**0.75**	0.38	**0.79**	**0.81**
TN	-0.13	-0.15	-0.14	-0.14
Org N	-0.23	-0.16	-0.24	-0.24
NO_2^-+NO_3^--N	0.42	0.46	0.27	0.31
NH_3-N	0.12	0.31	0.02	0.02
DIN/TN	**0.72**	**0.77**	**0.57**	**0.59**

Note: **Bold** denotes significance at $p<0.02$.

As agricultural practices and intensity may vary for different regions of the province, Spearman Rank Correlations were run between annual P and N export coefficients for the AESA streams and each average Census of Agriculture metric (1996, 2001, and 2006) grouped by ecoregion area (Table 3.64, Figure 3.81). Correlations differed for each ecoregion area, but the relationships were only significant for the Boreal watersheds. Overall, export coefficients in the Boreal ecoregion area were weakly correlated with the average A.I. percentile (Table 3.64). The only significant correlation observed was between DIN/TN and the overall agricultural intensity metric (0.77, $p<0.05$) and the manure production percentiles (0.97, $p<0.05$). A positive relationship was observed between NO_2^--N+NO_3^--N export coefficients and the overall agricultural intensity metric and manure production percentiles, but the correlations were not significant ($p>0.05$). The observed correlations in the Boreal ecoregion area were likely a result of the significant differences between export coefficients of streams within the ecoregion area and the factors that influenced these differences. Agricultural intensity is not uniform across watersheds in the Boreal ecoregion area. It appeared that nutrient export coefficients were most influenced by livestock production in the Parkland ecoregion area, as supported by the apparent positive relationships between nutrient export coefficients and the average manure production percentile. Surprisingly, nutrient export coefficients in the Parkland ecoregion area, all watersheds under high agricultural intensity, were not significantly correlated with the averaged agricultural intensity metric or manure production and fertilizer expenses percentiles ($p>0.05$, Table 3.64). It would be interesting to test if correlations became significant with a larger sample size. The relationships between nutrient exports and the agricultural intensity metrics in the grassland and irrigated grassland ecoregion areas have not been discussed as it is unclear whether any of the relationships are significant with such small samples sizes (Table 3.64).

Table 3.64. Spearman Rank Correlations between median annual P and N export coefficients (1999 to 2006) and average Census of Agriculture metrics (1996, 2001, 2006) for all dryland AESA watersheds grouped by ecoregion area.

	Boreal Ecoregion Area (n=9)				Parkland Ecoregion Area (n=6)				Grassland Ecoregion Area (n=3)			
Median Anuual Export Coefficient	Average Agricultural	Average Manure	Average Fertilizer	Average Chemical	Average Agricultural	Average Manure	Average Fertilizer	Average Chemical	Average Agricultural	Average Manure	Average Fertilizer	Average Chemical
TP	-0.15	0.13	-0.30	-0.40	**0.60**	**0.54**	-0.26	0.37	-0.50	0.50	-1.00	-1.00
TDP	0.00	-0.13	-0.08	-0.12	**0.66**	**0.77**	-0.31	0.31	0.50	-0.50	1.00	1.00
TPP	-0.15	0.12	-0.37	-0.43	**0.66**	**0.77**	-0.31	0.31	-0.50	0.50	-1.00	-1.00
TDP/TP	0.28	0.17	0.38	0.42	0.20	0.31	0.14	0.31	0.50	-0.50	1.00	1.00
TN	-0.07	-0.10	-0.20	-0.23	0.20	0.37	-0.20	-0.14	-1.00	-0.50	-0.50	-0.50
Org N	-0.20	-0.17	-0.25	-0.35	0.03	0.43	-0.37	-0.43	-0.50	0.50	-1.00	-1.00
NO_2^--N+NO_3^--N	**0.67**	**0.58**	0.37	0.27	0.43	**0.66**	-0.43	0.20	-1.00	-0.50	-0.50	-0.50
NH_3-N	0.02	0.43	-0.28	-0.42	0.26	**0.54**	**-0.60**	-0.20	-0.50	0.50	-1.00	-1.00
DIN/TN	**0.77**	**0.92**	**0.50**	0.40	0.26	**0.54**	**-0.60**	0.14	-0.50	0.50	-1.00	-1.00

Note: Bold values in the Boreal and Parkland ecoregion areas indicate a strong positive (>0.5) or negative (<-0.5) correlation between the parameter and the agricultural intensity metric. Correlations are significant (p<0.05) at 0.68 for n=9 and 0.89 for n=6.

Nutrient export and seasonality. Seasonality was examined for all forms of P and N exports to assess whether export coefficients varied during different times of the monitoring period (i.e., spring, summer, and fall). Box plots of monthly nutrient exports (1999 to 2006) can be found in Appendix 8 for each agricultural intensity category.

In general, seasonal trends in overall nutrient export in each watershed were influenced by climatic and geographic characteristics of each ecoregion area. Most forms of P and N export coefficients were highest in the early spring in both the Boreal and Parkland ecoregion areas. In contrast, streams in the Grassland ecoregion did not show seasonal differences in P or N export coefficients, while export coefficients in the Continental Divide were generally higher in June.

Phosphorus and N export coefficients in the Boreal ecoregion area exhibited a seasonal pattern (Appendix 8: Figures A8.1 to A8.3). In general, P and N export coefficients in the Boreal ecoregion area were highest in the early spring, specifically in April. Mean rank export coefficients for all forms of P and N in the Boreal ecoregion area were significantly higher ($p<0.005$) in April than in all months from May through October (Appendix 8: Figures A8.1 to A8.3). Export coefficients for all forms of P and the dissolved fractions of N were also significantly higher in March than in June, August, and September ($p<0.005$). This trend was not observed for TN or Org N.

Nutrient export coefficients in the Parkland ecoregion area showed seasonal trends with higher export coefficients observed in the early spring (March and April). Export coefficients for all forms of P and N in the Parkland ecoregion, with the exception of Org N, were statistically higher ($p<0.005$) in the early spring (March and April) than in all other months (Appendix 8: Figure A8.4 to A8.6). Mean rank Org N export coefficients were statistically higher in April than in May through October. Total and dissolved P export coefficients were also significantly higher in the late spring (May and June) than in the late summer (August) (Appendix 8: Figure A8.4). Total N export coefficients in the early spring (March) were significantly higher than in June, August, September, and October (Appendix 8: Figure A8.5). Snowmelt in the early spring may contribute to the observed seasonal trends in both the Parkland and Boreal ecoregion areas. Beneficial management practices focused on the seasonal export peaks may be required in order to reduce nutrient loading to downstream users.

Seasonality in nutrient export was not observed in the Grassland ecoregion area (Appendix 8: Figures A8.7 to A8.9). There was no significant difference among mean rank TP or TN export coefficients measured during any of the months of monitoring (KW = 3.858, p=0.796 for TP, KW=4.829, p=0.681 for TN). Moreover, no significant difference was observed in the Grassland ecoregion among months for any of the fractions of P or N (TDP: KW = 5.873, p=0.555; TPP: KW=4.175, p=0.759; Org N: KW=4.746, p=0.691; NO_2^--NO_3^--N: KW=4.373, p=0.736; NH_3-N: KW=4.716, p=0.695). Snowmelt may not contribute as much to spring runoff in the Grassland ecoregion, in comparison to the Boreal and Parkland ecoregion areas, as a result of chinook events that may decrease the volume of runoff from snowmelt in April and May.

A strong seasonal trend was not observed for P or N export coefficients in Willow Creek, the only stream located in the Continental Divide (Appendix 8: Figures A8.10 to A8.12). Median annual P and N export coefficients in June were higher than in other months; however, only mean rank TDP export coefficients were significantly higher in June than in all other months

(Appendix 8: Figure A8.10). In contrast, a significant difference in mean rank Org N export coefficients among months was not observed (Appendix 8: Figure A8.11). Other P and N parameters were statistically higher in June than in March with little or no difference in statistical significance between remaining months. Overall, export coefficients peaked in the late spring (June) and generally decreased over the summer and fall. The higher export coefficients in June could be a result of the climatic conditions in the Continental Divide. Snowmelt may not contribute as much to export coefficients in Willow Creek as chinook events during the winter may melt and sublimate snow, decreasing the final volume of runoff from snowmelt.

SUMMARY AND CONCLUSIONS

Objective 1: Compare instream nutrient concentrations to Canadian Water Quality Guidelines for the Protection of Aquatic Life (PAL) and livestock watering to asses the potential impact of agricultural activity on water quality as well as the impact of different intensities (low, medium, and high) and types (dryland versus irrigation) of agriculture.

a. Generally, the average TP and TN compliance from 1999 through 2006 was lowest in high agricultural intensity watersheds and highest in low agricultural intensity watersheds (low>moderate>high).
    - Annual variation among individual watersheds was large.

b. Watersheds in the high and irrigated agricultural intensities had lower average compliance with dissolved fractions of P and N (TDP, NO_2^--N, NO_3^--N, NH_3-N).
    - This is ecologically significant because dissolved fractions of P and N are more readily available for uptake by aquatic plants and algae than are particulate P or organic N and can contribute to high biomass levels in nutrient limited streams.
    - Reducing these dissolved fractions will help to decrease eutrophication which has been found to impact aquatic ecosystems and degrade surface water quality by creating toxic algal blooms, deplete oxygen, and decrease biodiversity (Carpenter et al. 1998).

c. Guidelines for NO_2^-, NO_2^-+NO_3^-, and NH_3-N were rarely exceeded.
    - Irrigated streams had the lowest NO_3^- compliance.
    - Rodvang et al. (2004) found manure applied to irrigated areas can have a greater effect on the leaching of nitrogen, specifically NO_3^-, into the groundwater.

d. Guidelines set for livestock watering were never exceeded.

Objective 2: Compare nutrient FWMCs and exports among agricultural intensity categories and ecoregion areas to assess whether agricultural and/or ecoregional characteristics influenced FWMCs and exports.

a. Overall P and N FWMC Conclusions

- Phosphorus and nitrogen annual FWMCs varied between streams and appeared to be influenced by agricultural intensity.
- Nitrogen FWMCs were always higher than P FWMCs regardless of the location or agricultural intensity of the watershed.
- TP FWMCs ranged from 0.008 to 1.300 mg L^{-1}
- TN FWMCs ranged from 0.120 to 8.600 mg L^{-1}
- A higher ratio of TP was found to be comprised of the dissolved fraction (TDP/TP ranged from 0.08 in Willow Creek to 0.892 in Haynes Creek) than was observed for TN (DIN/TN ranged from 0.046 in Rose Creek to 0.399 in Haynes Creek).

b. P and N FWMC by Agricultural Intensity

- Overall, agricultural intensity influenced N and P FWMCs (higher agricultural intensity, higher FWMC).
- The lowest and highest P and N FWMCs and ratios were found in streams draining watersheds under low and high agricultural intensity, respectively.
    - Rose and Willow Creeks had the lowest ratios of dissolved to total N and P, respectively, while Haynes Creek had the highest TDP/TP and DIN/TN ratios.
- All P and N parameters, with the exception of TPP and DIN/TN, were statistically highest in the high agricultural intensity category and lowest in the low agricultural intensity watersheds (L<M=I<H or L=I<M<H).
- No difference was found among agricultural intensity groups for TPP.
- DIN/TN showed a stepwise increase from low to high agricultural intensity with the highest FWMCs observed in the irrigated watersheds (L<M<H<I).

c. P and N FWMC by Ecoregion Area

- Nitrogen FWMCs were statistically higher in the Parkland ecoregion than in all other ecoregion areas for the majority of N fractions (NH_3-N, Org-N, TN).
    - Nitrogen FWMCs were higher in the Parkland ecoregion than all other ecoregion areas but were not statistically higher than the irrigated grasslands for NO_2^--N +NO_3^--N and DIN/TN.
    - Note that all watersheds in the Parkland ecoregion were under high agricultural intensity (according to 1996 agricultural intensity metrics), while all other ecoregion areas, with the exception of the irrigated grasslands, have a mix of agricultural intensities. The Irrigated Grasslands watersheds were all under high agricultural intensity.
- Phosphorus FWMCs were also statistically higher in the Parkland ecoregion than all other ecoregion areas for all fractions of P (TP, TDP, and TDP/TP) with the exception of TPP, which was statistically higher in the Boreal ecoregion area. The rest of the ecoregion areas did not statistically differ from one another.

- The Grassland ecoregion area (and Continental Divide) had lower P and N FWMCs – the ratios of dissolved to total (TDP/TP and DIN/TN) were statistically lower in the Grassland watersheds than all other ecoregion areas.

d. Overall P and N Export Coefficient Conclusions
  - Total P export coefficients ranged from 0.012 to 0.197 $kg \cdot ha^{-1} yr^{-1}$; TN export coefficients ranged from 0.03 to 12.0 $kg \cdot ha^{-1} yr^{-1}$.
  - Total phosphorus and nitrogen export coefficients were similar to values reported in other studies in Alberta and were also within the range of export coefficients measured in other studies in Canada, the United States, and Europe.
  - The highest TN and TP export coefficients were not necessarily measured in the same streams.
  - Nutrient export coefficients were primarily driven by factors characteristic of the ecoregion areas the watersheds were located in.
  - Overall, a clear pattern was not observed between agricultural intensity and nutrient export coefficients; however the ratios of TDP/TP and DIN/TN exported were influenced by agricultural intensity.

e. P and N Export Coefficients by Agricultural Intensity
  - Total P, TN, and Org N export coefficients were not influenced by agricultural intensity; a statistical difference was not observed among agricultural intensity categories.
  - Although not statistically significant among all agricultural intensity categories, TDP and NH_3-N export coefficients showed a stepwise trend of increasing export coefficients with increasing agricultural intensity, which was also reported in the CAESA study.
  - Watersheds within the moderate agricultural intensity category differed in nutrient export coefficients depending on the ecoregion area they were located in
    - Moderate agricultural intensity streams in the Grassland ecoregion had lower TP, TDP, NO_2^-+NO_3^--N, and TN export coefficients than moderate watersheds in the Boreal ecoregion area.
  - The ratio of DIN/TN showed a stepwise trend with increasing agricultural intensity (H>M>L); TDP/TP was higher in high agricultural intensity watersheds but was similar among low and moderate agricultural intensity streams.

f. P and N Export Coefficients by Ecoregion Area
  - Total nutrient export coefficients were generally highest in the Boreal ecoregion area.
    - Total particulate P export coefficients were influenced by runoff depth and were therefore highest in the Boreal ecoregion area with lower export coefficients measured in the Parkland and Grassland ecoregion areas.
    - Total N and Org N export coefficients were highest in the Boreal and lowest in the Grassland ecoregion area.

  - Total P, TDP, NO_2^-+NO_3^--N, and NH_3-N export coefficients were lowest in the Grassland ecoregion area with similar export coefficients found in both the Boreal and Parkland ecoregion areas.
- Variability in P and N export coefficients was observed within the Boreal ecoregion area; differences among watersheds were based on more specific ecoregion characteristics (i.e., differences in landscape and climate), not differences in agricultural intensity within the Boreal ecoregion area.
- The ratio of TDP/TP and DIN/TN was highest in the Parkland ecoregion and lowest in the Grassland ecoregion area. The higher ratios in the Parkland ecoregion were likely a result of higher agricultural intensity.
  - All of the watersheds in the Parkland ecoregion were all under high agricultural intensity (streams in the Boreal were under a mix of low, moderate, and high agricultural intensity; streams in the Grassland ecoregion were under a mix of low and moderate agricultural intensity).

Objective 3: Examine nutrient FWMC and export relationships with agricultural intensity metrics as a means to assess the impact of different intensities (low, moderate, and high) and types of agriculture on water quality (Objective 1 of the AESA stream survey).

a. Nutrient FWMC correlations with agricultural intensity metrics

- Overall, strong, positive correlations (0.50 to 0.75, $p<0.02$) were observed between agricultural intensity metrics and dissolved N fractions (NO_2^--N +NO_3^--N and NH_3-N FWMCs) as well as the ratios of TDP/TP and DIN/TN.
- Although strong, positive correlations were observed for many parameters with both fertilizer and chemical sales, only the ratio of DIN/TN was strongly correlated with the average manure production percentile (0.63, $p<0.02$).
  - The ratio of TDP/TP was strongly, positively correlated with fertilizer and chemical sales metrics overall but not with manure.
  - The ratio of DIN/TN was strongly, positively correlated with all 3 metrics overall (fertilizer expenses, chemical expenses, and manure production).
- The correlation between the overall agricultural intensity metric and the dissolved nutrient fractions supports use of the metric as an indicator for agricultural influence of nutrient concentrations.
  - Note that it was the dissolved fractions of N and P rather than the total that correlated most closely with the agricultural intensity metric at a provincial scale.
- In the Boreal ecoregion area, only TN and N fractions and ratios were strongly, positively correlated (0.75 to 0.92, $p<0.05$) with the average agricultural intensity percentile.
  - The agricultural intensity metric was more strongly correlated overall with N than with P in the Boreal ecoregion area, which may be a result of the higher but variable ratios of particulate P from runoff events.

- Total P and TN FWMCs in the Boreal were strongly, positively correlated with manure production percentiles ($p<0.05$).
    - TDP/TP was not correlated with any A.I. metrics, while DIN/TN was strongly correlated with manure production and fertilizer sales percentiles in the Boreal.
- Phosphorus FWMCs appeared to have a positive relationship with agricultural intensity percentiles in the Parkland ecoregion area, though TN was the only parameter significantly correlated with the average agricultural intensity metric (0.89, $p>0.05$).
    - Strong, positive correlations were observed between the fertilizer metric and TP, TDP, and Org N ($p<0.05$), while no significant correlations were observed between nutrient FWMCs and the manure production and fertilizer sales metrics.
    - The stronger correlation between chemical sales and nutrient concentrations in the Parkland ecoregion were unexpected considering that fertilizer and manure are nutrient sources. It is uncertain weather the metrics for nutrient sources are inadequate or if there is something inherent about the chemical sales percentiles (i.e., land use type) that is a better surrogate of nutrient use.

b. Export coefficient correlations with agricultural intensity metrics

- In general, strong correlations were not observed between median annual P and N export coefficients for the AESA dryland streams and Census of Agriculture metrics averaged for the three years of data that were available (1996, 2001, and 2006).
- The strongest correlations were observed between the metrics and the ratios of TDP/TP and DIN/TN.
    - A very strong and positive correlation was found between TDP/TP and the average agricultural intensity percentile (0.75, $p<0.02$) and average fertilizer and chemical sales percentiles (0.79 and 0.81, respectively).
    - The ratio of DIN/TN was very strongly, positively correlated with all agricultural intensity metrics ($p<0.02$). DIN/TN was the only parameter that was strongly correlated with the average manure production percentile (0.77).
- Nutrient correlations with agricultural intensity metrics varied among ecoregion areas, but the correlations were only significant for the Boreal watersheds.
    - DIN/TN in the Boreal ecoregion area was strongly, positively correlated with the agricultural intensity and manure production metrics (0.77 and 0.97, respectively; $p<0.05$).
    - Surprisingly, nutrient export coefficients in the Parkland Ecoregion were not significantly correlated with the averaged agricultural intensity metric or manure production and fertilizer expenses percentiles ($p>0.05$), even though all AESA watersheds located in the Parkland Ecoregion were under high intensity agriculture.

Objective 4: Identify changes in water quality with time (Objective 2 of the AESA stream survey).

a. Overall, temporal trends (1995 to 2006 or 1999 to 2006) in P and N FWMCs were not observed.
    - Many watersheds that exhibited inter-annual patterns in nutrient FWMCs were strongly influenced by flow.
    - Watersheds where a statistical trend analysis should be considered include Battersea Drain, Prairie Blood Coulee, Blindman River, Kleskun Drain, Meadow Creek, Tomahawk Creek, Buffalo Creek, Renwick Creek, and Wabash Creek. In each of these watersheds, an increasing or decreasing trend may emerge.

b. Generally, there were no temporal patterns in median annual loading values during the AESA monitoring period.
    - Median annual loading values were strongly correlated with annual flow volumes.
        - Typically, years with high flow volumes yielded high annual loading values.
        - If there was deviation from this pattern, there were three factors that may have contributed to the variation: sampling regime, seasonality of precipitation, and change of land management or land use.

Objective 5: Examine seasonal patterns to assess whether nutrient concentrations and exports vary during different times of the monitoring period (i.e., spring, summer, and fall).

a. Seasonality of FWMCs and mechanism for transport
    - In general, seasonal changes in water quality (March to October) were only observed in high agricultural intensity watersheds, which had higher nutrient FWMCs in the spring and early summer.
    - The lack of seasonal differences in N and P FWMCs for the other agricultural intensity watersheds was likely a reflection of the differences in regional characteristics such as soil type, topography, and climate.
        - The majority of watersheds classified in the high agricultural intensity category were located in the Parkland ecoregion area, while the remaining agricultural intensity groups covered multiple ecoregion areas.
        - Seasonality was generally not observed in the irrigated watersheds since flows are controlled.

b. Seasonality of P and N Export Coefficients and mechanism for transport
    - Seasonal trends in overall nutrient export in each watershed were influenced by the climatic and geographic characteristics representative of each ecoregion area.
    - Most forms of P and N export coefficients were highest in the early spring in the Boreal and Parkland ecoregion areas, particularly in April in the Boreal streams.

- In contrast, streams in the Grassland ecoregion area did not show seasonal differences in P or N export coefficient.
- The lowest P export coefficients occurred in March in the Continental Divide (Willow Creek). Significantly higher TN export coefficients were observed in June in Willow Creek.

Chapter 4: Bacteria

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INTRODUCTION

Bacteriological analysis of surface waters provides an indication of the presence of fecal material from warm-blooded animals (e.g., domestic animals, wildlife, and humans). Indicator bacteria, like fecal coliforms and *Escherichia coli* (*E. coli)*, originate in the digestive tract of animals and indicate the possible presence of pathogens that cause waterborne disease, though these bacteria do not necessarily cause illness themselves. Indicator bacteria are useful in the study of pathogen risk, as it is cost-prohibitive to study specific zoonotic pathogens (e.g., salmonella, listeria, campylobacter, *E. coli* O157:H7, *Cryptosporidium parvum*, giardia, and rotavirus) in long-term studies such as the AESA Stream Survey.

Fecal contamination of agricultural streams presents a potential health risk to those who come in contact with the water because of potential exposure to protozoan parasites, bacteria, or viruses. Fecal matter from livestock (e.g., swine, cattle, and poultry) or wildlife (e.g., birds, deer, and beaver) can enter streams with surface runoff, by sediment erosion, or by direct animal access to water (Collins 2004; Oliver et al. 2005; McDowell 2007). Agricultural practices such as spreading cattle or swine manure on frozen land or during wet periods amplify the likelihood of overland movement of fecal matter. However, the highest risk scenario for pathogen transfer to surface waters occurs during grazing, when cattle have direct access to water (Ferguson et al. 2003). Untreated human waste may also enter ditches and streams when individual septic systems leak or become overloaded during wet periods.

Most pathogens do not multiply outside of a host organism and have a limited lifetime in surface waters because conditions are sub-optimal for survival. The detection of fecal coliforms or *E. coli* in water is indicative of recent fecal contamination. The length of time these organisms will survive in the environment is dependent on environmental factors such as temperature, pH, sediment levels, and freezing and/or thawing processes (Verstraete and Voets 1976; Fujioka et al. 1981; Flint 1987). For example, low temperatures in surface water, which slow metabolism, generally prolong the survival of bacteria (Wang and Doyle 1998). In contrast, high temperatures during manure composting (temperature dependent upon substrate composition, moisture content, and duration of incubation) can decrease the likelihood of fecal bacteria and pathogen transmission (Turner 2002; Larney et al. 2003). Sediments in the bottom of streams often have higher concentrations of bacteria than the overlying water as a result of various environmental factors, such as increased organic matter, that affect survival of fecal bacteria (Gerba and McLeod 1976; Brettar and Hofle 1992). Fecal organisms are bound to sediments at the bottom of water sources until the sediments are disturbed by sources such as livestock or wildlife walking through the waterway (Sherer et al. 1988; McDowell 2007; Koirala et al. 2008). Therefore, bacteria die-off or resuspension in stream water impacts the concentrations observed (Jamieson et al. 2005).

Canadian guidelines for fecal coliform and *E. coli* exist to protect humans from potential health risks associated with recreation and irrigation uses. If guidelines are met, surface waters are considered safe for body contact during recreation or application to crops through irrigation. If guidelines are not met, the level of fecal contamination may pose a health risk. However, since indicator bacteria do not necessarily cause illness, more information is typically required on the source of bacteria to determine whether pathogens are present (Minnesota River Basin Data

Centre 2003). Whenever surface water is used as a drinking water source (e.g., dugout, lake, or river water for human consumption), it must be treated so that fecal coliform concentrations are 0 $CFU \cdot 100\ mL^{-1}$.

Alberta Context

Increased land pressure as a result of the intensity of livestock production in Alberta may lead to increased fecal contamination of surface water. The majority of agriculture in Alberta is accounted for by the beef cattle sector. Compared to other Canadian provinces, Alberta was the lead province in cattle and calf inventories in 2006 (6.3 million head), accounting for more than one-half (51.4 %) of the total estimated western Canadian herd (Statistics Canada 2006). Overall, large numbers of livestock across Alberta have the potential to significantly contribute bacteria into surface water through hydrologically linked pathways from manure spreading areas or by direct access to waterways.

On a provincial-scale, manure production tends to be greatest in the western portion of the white or agricultural zone (along the Highway 2 Edmonton-Calgary corridor) and in the Lethbridge area (Figure 4.1). Manure production values are based on the number of cattle and calves, pigs, hens, chickens, and other types of livestock. Livestock other than cows and calves are typically intensively managed, with manure often applied to land (via surface spreading or injection) in close proximity to the intensive livestock operation.

Efforts are currently being made to minimize the risks of fecal contamination to surface water bodies through encouraging best management practices, such as establishing proper grazing strategies and wintering sites, and by properly siting livestock corrals and manure storage sites and maintaining set back distances as outlined under the Agricultural Operation Practices Act (AOPA 2000). However, in cases where best management practices are not in place and natural conditions are favorable for overland movement of fecal bacteria into streams, contamination may be occurring. A sustainable livestock industry in Alberta is dependent on there being sufficient land base for manure application in regions where manure is generated.

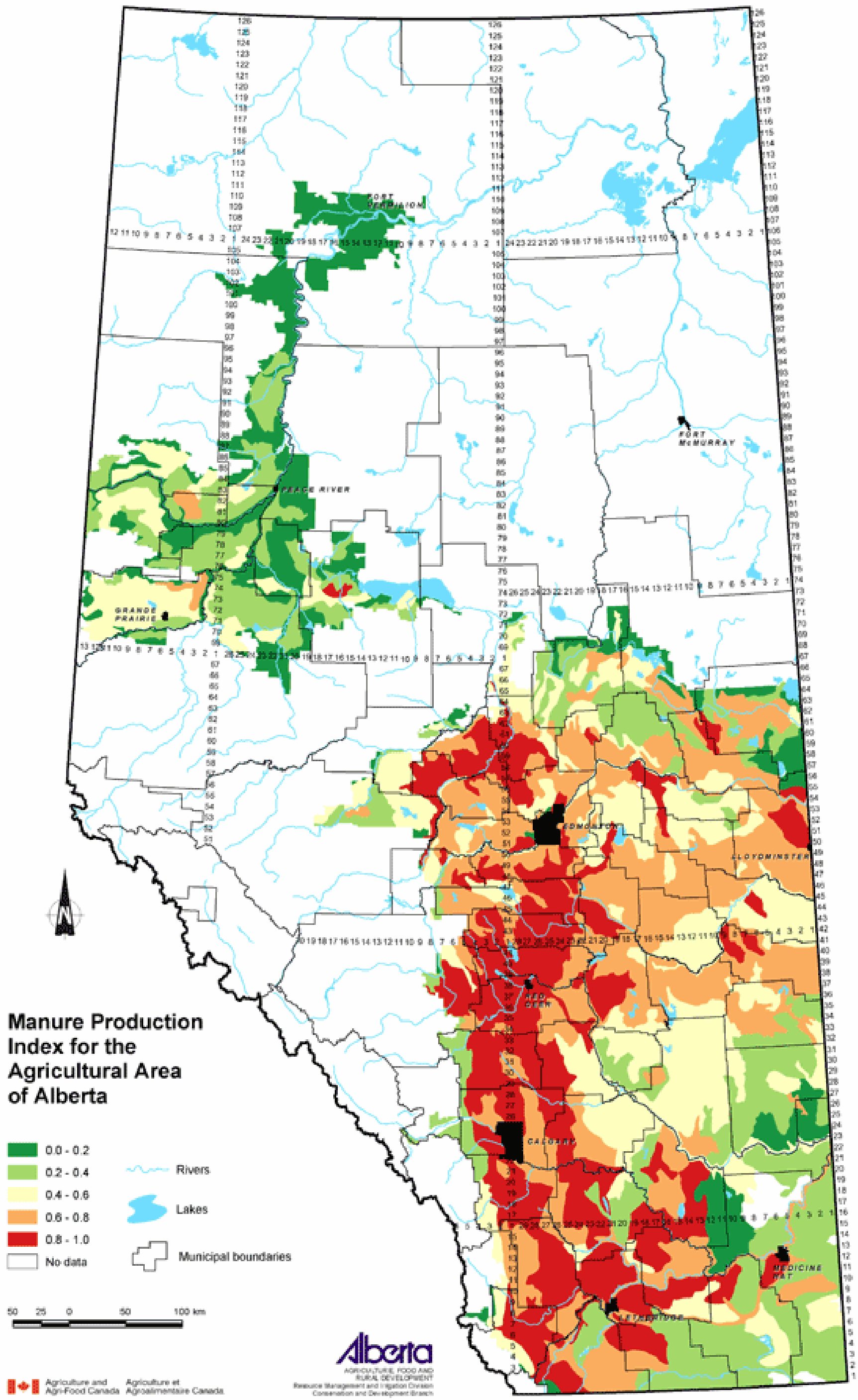

Figure 4.1. Distribution of manure production in Alberta's agricultural zone in 2001 (Source: Statistics Canada).

Objectives

As introduced in Chapter 1, the AESA Stream Survey was initiated to learn more about how stream water quality is impacted by low, moderate, and high intensity agriculture in Alberta and to track changes in water quality as the industry grows and agricultural management practices change.

The specific objectives of this chapter were to:

i. Assess compliance with Canadian Water Quality Guidelines;
ii. Assess differences in fecal bacteria levels among watersheds with varying levels of agricultural intensity (AI) and in different ecoregion areas;
iii. Examine inter annual patterns in fecal coliform and *E. coli*;
iv. Examine the relationship between manure production in a watershed and the number of fecal coliforms and *E. coli* detected in water;
v. Investigate the relationship between bacteria concentrations and possible transport mechanisms (e.g., flow, sediment transport, and precipitation); and
vi. Examine seasonal patterns in fecal coliform and *E. coli.*

MATERIALS AND METHODS

Sample Collection

Stream water samples for fecal bacteria analyses were collected at the same frequency as nutrient samples and followed a flow-proportionate sampling regime. Samples were collected twice per week during runoff periods, once per week as runoff subsided, and once every two weeks to once a month during baseflow conditions. The total number of samples collected for each stream varied from year to year as a function of stream flow and runoff.

Samples were collected from 1999 to 2006 from 23 streams (Chapter 2: Table 2.2). In addition, 16 watersheds from the original CAESA program were sampled and analyzed for bacteria in 1998. Any comparisons carried out among watersheds, agricultural intensity classification categories, or ecoregion categories were completed on data from 1999 to 2006 to ensure results could be attributed to the factors under investigation rather than the presence or lack of data for each stream. Years with no sample collection for a particular stream indicates that stream flow did not occur (Chapter 2: Table 2.8); however, the lack of sample collection was a result of logistic problems in some cases (see Chapter 2: Results and Discussion - Hydrology).

Laboratory Analyses

The Provincial Laboratories for Public Health in Edmonton and Calgary provided bacterial enumeration for fecal coliforms and *E. coli*. Samples were analyzed using membrane filtration followed by seven serial dilutions (APHA). The method detection limit (MDL) differed between laboratories, so the higher MDL (10 CFU·100 mL^{-1}) was applied to all samples to ensure consistency in the data set.

Data Management and Analyses of Bacteria Data

Data validation. All data were validated to ensure that *E. coli* concentrations did not exceed the fecal coliform concentrations. Concentrations below the lower method detection limit (MDL) were changed to ½ MDL. Concentrations greater than the upper method detection limit were set to equal the upper method detection limit.

Compliance with surface water quality guidelines. Ambient *E. coli* and fecal coliform concentrations were compared to the Alberta Surface Water Quality Guidelines (AENV 1999) for recreational and agricultural uses, respectively. Both guidelines were applied to all 23 AESA watersheds.

The irrigation guideline for fecal coliform bacteria (100 CFU·100 mL^{-1}) is primarily intended for irrigation water applied to produce that is consumed raw (e.g., lettuce, cabbage, and cauliflower), but it may also be applied to irrigation water used for other field crops.

The Health Canada recreation guideline for *E. coli* (200 CFU·100 mL^{-1}) applies to surface water bodies used for activities involving bodily contact with water (e.g., swimming, canoeing, and fishing). To correctly apply the recreation guideline, instantaneous bacteria concentrations should be compared to the trigger value of 400 CFU·100 mL^{-1}. If the trigger value is exceeded, the geometric mean of 5 samples collected in 30 days should fall below 200 CFU·100 mL^{-1} (Health and Welfare Canada 1992). The flow-biased sampling regime of the AESA stream survey was not set up to examine the geometric mean over 30 days of sampling. In this report, instantaneous bacteria concentrations were compared to the most restrictive value of 200 CFU·100 mL^{-1}.

Export coefficient and mass load calculations. Mass loads were determined from instantaneous daily discharge data and periodic sample data using version 4.5 and method #3 of the program FLUX (U.S. Army Corps of Engineers 1995). The data record from 1999 to 2006 for ambient *E. coli* and fecal coliform concentrations (CFU·100 mL^{-1}) and flow ($m^3 \cdot s^{-1}$) were used in FLUX. The same methods used to calculate annual nutrient loads were applied to the bacteria data (see Chapter 3: Materials and Methods – Load, FWMC, and export calculations). Mass loads were not reported in Haynes Creek M6 in 2004 and Stretton Creek in 2000 through 2003 as FLUX was not capable of accurately estimating the values with few samples or with very low or zero flow records. Loads were not calculated for Drain S-6 due to missing flow data.

Export coefficients were calculated by dividing the total mass load by the total active drainage area to express mass export of a watershed per unit time for comparison among watersheds (Chapter 2: Table 2.1). Export coefficients were not calculated for the irrigated streams (Battersea Drain, Crowfoot Creek, Drain S6, and New West Coulee) as discussed in Chapter 3 (see Chapter 3: Materials and Methods – Load, FWMC, and export calculations). The hydrology in watersheds with irrigation is altered due to irrigation canals that extend beyond the natural watershed boundary.

Statistical analyses. Any count below the method detection limit was included in the summary statistic calculations as ½ the MDL (or 5 CFU·100 mL^{-1}). All statistical analyses were completed in SYSTAT 10 (SPSS Inc. 2000). Summary statistics were computed for ambient and FWMC data for both *E. coli* and fecal coliforms. Data from 1999 to 2006 were used for statistical analyses.

In general, the bacteria concentration data set was highly skewed with a large proportion of censored data points (those values below the MDL). This limited the ability to log-normalize the data set, and consequently, all analyses conducted using bacteria concentrations were non-parametric.

The summary statistic used to report fecal bacteria data is very important. The geometric mean is often employed instead of an arithmetic mean due to the asymmetrical distribution of the bacteria data. The geometric mean tends to dampen the effect of very high or very low values which can bias the arithmetic mean. A monthly geometric mean (based on 5 samples collected over a 30-day period) is used to assess compliance with water quality guidelines. However, there is no universal agreement in the scientific community on which statistic (median, arithmetic mean, or geometric mean) should be consistently used in reporting bacteria data. All summary statistics were calculated and reported to better understand the data set.

The geometric mean was calculated in MS Excel using either the geometric mean function (GeoMean) or an array formula (=EXP(AVERAGE(LN(A1:A200)))). The geometric mean function (GeoMean) generated an error when applied to a long list of numbers necessitating the array formula.

Median annual fecal coliform and *E. coli* concentrations were not normally distributed. Too many median values of 5 (1/2 MDL) meant that log transformations would not normalize the data set. A nonparametric test (Kruskal Wallis One-Way ANOVA) was used on untransformed data followed by a Mann-Whitney test when applicable.

Unlike median annual data, annual geometric means could be normalized using a log 10 transformation, and parametric statistics were applied. Differences among streams, ecoregion areas, and agricultural intensity categories were assessed using ANOVA and Tukey's post hoc test.

Seasonality was examined using ambient fecal coliform and *E. coli* data (1999 to 2006) for all streams. Kruskal Wallis One-Way ANOVA and Mann-Whitney statistics were conducted on untransformed ambient fecal coliform and *E. coli* data as a suitable transformation to normalize the data was not found.

RESULTS AND DISCUSSION

Ambient Fecal Coliform and *E. coli* Concentrations

Overall. Fecal coliforms and *E. coli* were found to be highly correlated (Spearman rank correlation (r_s)=0.962). Summary tables, which include sample numbers, detection frequencies, medians, geometric and arithmetic means, interquartile ranges, and ratios of *E. coli* to fecal coliform concentrations, can be found in the Appendix (Tables A10.1, A10.2, A10.3a and b, Figure A10.1). Additional highlights are found in the text below.

Sample numbers - The majority of streams had between 13 and 19 samples collected per year, or a total of 100 or more samples over the 8 year monitoring period. The exceptions included Haynes Creek, Stretton Creek, Kleskun Drain, and Wabash Creek which had an average of 6 to 12 samples collected per year and a total of 47 to 92 samples collected (over the 1999 to 2006 monitoring period). Sampling frequencies were most consistent in the irrigation return flows where flows are controlled.

Detection frequency - The proportion of all samples with detectable levels of fecal bacteria ranged from 40% in Stretton Creek (high agricultural intensity) to 97% in Meadow Creek (moderate agricultural intensity) for *E. coli* and from 53% in Willow Creek (low agricultural intensity) to 99% in Meadow Creek for fecal coliforms. Detection frequencies were consistently higher for fecal coliforms as *E. coli* are a subset of fecal coliforms.

Median concentrations - Statistical calculations include censored data points; thus, median concentrations may better reflect detection frequencies in watersheds, particularly watersheds with lower or fewer occurrences of fecal bacteria. On a whole, all streams with the exception of two moderate agricultural intensity watersheds in the Grasslands ecoregion area (Meadow and Trout Creeks) had median *E. coli* concentrations <100 CFU·100 mL^{-1}. Furthermore, all streams but four (Trout, Meadow, Rose, and New West Coulee) had median fecal coliform concentrations <100 CFU·100 mL^{-1}.

Arithmetic mean concentrations - Mean concentrations of *E. coli* ranged from 20 (in Stretton Creek) to 1264 CFU·100 mL^{-1} (in Meadow Creek). Mean fecal coliform concentrations ranged from 23 CFU·100 mL^{-1} (in Stretton Creek) to 2221 CFU·100 mL^{-1} (in Meadow Creek). The three watersheds with the lowest concentrations of fecal coliforms or *E. coli* were Stretton, Willow, and Hines Creeks. The three watersheds with the highest concentrations of fecal coliforms and *E. coli* were Meadow Creek (both), Tomahawk Creek (both), Strawberry Creek (*E. coli* only) and Battersea Drain (fecal coliforms only). Arithmetic mean concentrations closely mirrored maximum concentrations (Spearman Rank Correlations (r_s)> 0.97).

Geometric mean concentrations - Geometric mean concentrations for *E. coli* ranged from 11 (in Stretton and Willow Creeks) to 448 CFU·100 mL^{-1} (in Meadow Creek) and from 12 (in Stretton Creek) to 660 CFU·100 mL^{-1} (in Meadow Creek) for fecal coliforms. The three watersheds with the lowest geometric means included Willow, Stretton, and Haynes Creeks, and the three with the highest means included Meadow Creek, Trout Creek and New West Coulee. Geometric mean concentrations closely mirrored median concentrations.

Interquartile ranges - Interquartile ranges and 75th percentiles were examined to gather information on the variability of instream bacteria concentrations and frequency of the higher bacteria concentrations among streams.

Interquartile ranges for *E. coli* varied from 15 to 980 (average = 148 CFU·100 mL^{-1}) and ranged from 15 to 1620 (average = 205 CFU·100 mL^{-1}) for fecal coliforms. The data imply that fecal bacteria concentrations could vary by an order of magnitude above or below the median in agricultural streams. The streams with higher variability included Rose Creek, Blindman River, Grande Prairie Creek, Tomahawk Creek, Trout Creek, Ray Creek, Strawberry Creek, and the four irrigated streams (Battersea Drain, Crowfoot Creek, Drain S6, and New West Coulee). The interquartile range for Meadow Creek was exceptionally high.

Generally speaking, streams with higher interquartile ranges (i.e., bacteria concentrations spanned a broader range) also had higher mean bacteria concentrations. However, this was not always the case. There were four exceptions: Battersea Drain, Kleskun Drain, Strawberry Creek, and Tomahawk Creek. In each of these streams the mean concentrations were higher than would be expected based on interquartile range or 75th percentile values. A few (<3) sampling dates with very high magnitude peaks (i.e., >10,000 CFU·100 mL^{-1}) skewed the mean upwards (Table 4.1).

Table 4.1. High magnitude peaks in fecal coliform and *E. coli* concentrations during the monitoring period, 1999 to 2006.

Watershed (agricultural intensity)	Date	Fecal coliform (CFU·100 mL^{-1})	*E. coli* (CFU·100 mL^{-1})	Comments
Kleskun Drain (moderate)	May 31, 2001	20,000	17,000	This was the only peak of this magnitude
Tomahawk Creek (moderate)	June 14, 1999	27,000	27,000	Two high magnitude peaks in spring/early summer
	May 26, 2006	42,000	40,000	
Strawberry Creek (high)	July 20, 2000	27,000	21,500	Two peaks mid-summer
	July 30, 2001	22,000	18,000	
Battersea Drain (irrigated)	June 10, 2002	34,000	10,000	Three peaks in June.
	June 9, 2005	21,000	15,000	
	June 15, 2006	15,000	10,000	

Means alone may not consistently be a reliable tool for assessing the risks due to pathogen levels in stream water, as highlighted in the above examples. Nine of the 23 AESA streams had fecal coliform or *E. coli* concentrations in the order of 10^4 CFU·100 mL^{-1} (Table A4.3a), but in the four streams noted above, this was the exception and not the rule. In the case of Tomahawk Creek, 2 of 156 samples were of very high magnitude, but they appear to be isolated events and represent <1.5% of samples over an eight year period. For the purpose of risk assessment, it may be useful to show the distribution of fecal bacteria in categories (e.g., <10, <100, <100, <1000, <10,000, <100,000, ≥100,000).

An initial risk assessment based on concentrations >1000 (i.e., 10x higher than the most stringent water quality guideline) revealed that in 6 of the 23 AESA streams (Willow Creek, Prairie Blood Coulee, Hines Creek, Stretton Creek, Wabash Creek, and Haynes Creek), fewer than 1% (≤ 1%) of samples had fecal coliform and *E. coli* concentrations >1000 CFU·100 mL^{-1}. Eight watersheds (the above plus Kleskun Drain and Buffalo Creek) had ≤ 1% of samples with *E. coli* concentrations >1000 CFU·100 mL^{-1}. Between 2 and 10% of samples in the rest of the AESA streams (with the exception of Meadow Creek) had fecal coliform and *E. coli* concentrations >1000 CFU·100 mL^{-1}. In Meadow Creek, 34% of fecal coliform and 29% of *E. coli* samples exceeded this threshold.

E. coli-to-Fecal coliform Ratio - In individual streams, the average proportion of fecal coliform comprised of *E. coli* ranged from 71 to 87% (Table A10.1). Median ratios were 100 for all streams except the three streams in the Grassland ecoregion area (Meadow Creek, Trout Creek, and Prairie Blood Coulee), the four streams in the Irrigated Grassland ecoregion area (Battersea Drain, Crowfoot Creek, Drain S-6, and New West Coulee), and Rose Creek (Boreal ecoregion area). Lower ratios may imply different sources of fecal contamination in the Fescue Grasslands and Mixed and Moist Mixed Grassland Ecoregions.

Compliance with surface water quality guidelines. Irrigation and recreation guidelines provide a benchmark with which to compare the bacteria data and assess general water quality.

Average annual percent compliance with the irrigation guideline (100 CFU·100 mL^{-1} fecal coliforms) ranged from 12% in Meadow Creek (moderate agricultural intensity) to 96% in Stretton Creek (high agricultural intensity). Compliance with the irrigation guideline was generally highest in watersheds with low and high agricultural intensity (Table 4.2). Since the guideline need only be applied under circumstances where the water is potentially used for irrigation, the watersheds where this information has greatest importance are Battersea Drain, Crowfoot Creek, Drain S6, and New West Coulee. It is important to clarify that the sampling location is at the mouth of these watersheds, and though irrigation agriculture is practiced within these basins, the source water used to irrigate crops is not the water sampled under the AESA Stream Survey. The sampling points may include both source water and some irrigation return flows. Compliance with irrigation guidelines would be directly applicable under circumstances where water from the irrigation return flow streams was used as source water for downstream users. Risks to human health may occur if water containing fecal bacteria in excess of irrigation guidelines was applied to crops that are eaten raw, such as lettuce, spinach, and tomatoes. Research has shown that risks may be lowered by waiting one to two weeks between irrigating and harvesting raw crops, as bacteria will desiccate and no longer be viable (Hutchison et al. 2008).

Average annual percent compliance with the recreation guideline ranged from 22% in Meadow Creek (moderate agricultural intensity) to 100% in Stretton Creek (high agricultural intensity) (Table 4.2). Again, compliance tended to be highest in watersheds with low or high intensity agriculture. High compliance with recreation guidelines suggests low risk to human health when in contact with water while undergoing activities like swimming, white water sports, sailing, canoeing, and fishing (CCME 1992). With the exception of three watersheds (Rose, Trout, and Meadow Creeks), mean annual compliance values were >75% for the 1999 to 2006 period. These values show that there is room for improvement but that compliance is generally high.

Table 4.2. Percent compliance with Alberta Surface Water Quality Guidelines for irrigation (fecal coliform) and recreation (*E. coli*) uses. Annual average and standard deviations reported.

	Fecal coliforms			*E. coli*		
	n[z]	% Compliance Irrigation (100 CFU 100 mL^{-1})	Std. Dev[y]	n[z]	% Compliance Recreation (200 CFU 100 mL^{-1})	Std. Dev[y]
Low agricultural intensity						
Hines Creek **(HIN)**	14	87	15	14	96	6
Paddle Creek **(PAD)**	19	80	8	19	92	6
Prairie Blood Coulee **(PRA)**	14	84	9	14	94	7
Rose Creek **(ROS)**	23	50	13	23	73	9
Willow Creek **(WIL)**	21	92	3	21	98	4
Moderate agricultural intensity						
Blindman Creek **(BLI)**	22	60	14	22	77	17
Grande Prairie Creek **(GRA)**	13	59	16	12	81	12
Kleskun Drain **(KLE)**	9	81	16	9	89	12
Meadow Creek **(MEA)**	19	12	16	18	22	18
Tomahawk Creek **(TOM)**	19	63	17	19	84	12
Trout Creek **(TRO)**	21	37	11	21	61	13
High agricultural intensity						
Buffalo Creek **(BUF)**	15	80	13	15	88	7
Haynes Creek M6 **(HM6)**	12	85	8	13	95	7
Ray Creek **(RAY)**	17	68	14	17	77	12
Renwick Creek **(REN)**	13	82	20	13	88	13
Stretton Creek **(STT)**	6	96	6	9	100	0
Strawberry Creek **(STW)**	17	66	15	16	80	13
Threehills Creek **(THR)**	17	78	10	17	92	6
Wabash Creek **(WAB)**	12	87	11	12	93	9
Irrigated watersheds						
Battersea Drain **(BAT)**	19	62	15	18	85	5
Crowfoot Creek **(CRO)**	19	53	12	19	76	9
Drain S6 **(DS6)**	17	62	10	16	76	13
New West Coulee **(NEW)**	18	50	11	18	77	16

[z] average number of samples collected per year from 1999 through 2006
[y] standard deviation

Although the majority of samples collected had bacteria levels below 200 CFU·100 mL^{-1}, when concentrations exceeded this range, they exceeded it by a lot. As such, compliance numbers should be examined together with the peak values in the future to provide more insight on the level of risk to human health. Minnesota River Basin Data Centre (2003) also suggests that where indicator bacteria imply a risk, it may be important to do additional work on the presence of pathogens since fecal coliforms and *E. coli* themselves do not necessarily cause illness.

There are currently no Canadian guidelines for bacteriological parameters for livestock watering. Some researchers have examined the relationship between clean water and cattle weight gain (Willms et al. 2002; Lardner et al. 2005), but the studies were not specific to bacteriological parameters. In these studies, weight gain was 23% higher in yearling heifers and 9% higher in steers that had access to clean water, compared to those cattle which watered directly from a stream or a dugout (Willms et al. 2002; Lardner et al. 2005).

Annual Ambient Geometric Means

By agricultural intensity. Fecal bacteria in Alberta's agricultural watersheds differed significantly among agricultural intensity categories (fecal coliforms: $F(3, 176)=27.149$, $p \leq 0.0001$; *E. coli*: $F(3, 176)=27.149$, $p \leq 0.0001$). However, unlike nutrient concentrations (Chapter 3: Comparisons of P and N FWMCs by Agricultural Intensity), fecal bacteria concentrations did not show an increasing pattern with agricultural intensity (Figure 4.2). In fact, annual geometric means for *E. coli* and fecal coliforms were significantly lower in both high and low intensity streams than in moderate intensity and irrigated streams (Tukey's post hoc, $p< 0.05$). The absence of a stepwise increase in bacteria concentrations from low to moderate to high agricultural intensity categories suggests that the sources and/or mechanisms of transport of fecal bacteria may not be the same as those for nutrients.

The same pattern among agricultural intensity categories was observed for mean rank annual median values (fecal coliforms: $H=57.592$, $p<0.0001$, *E. coli*: $H=52.126$, $p<0.0001$, df=3) (Appendix 11).

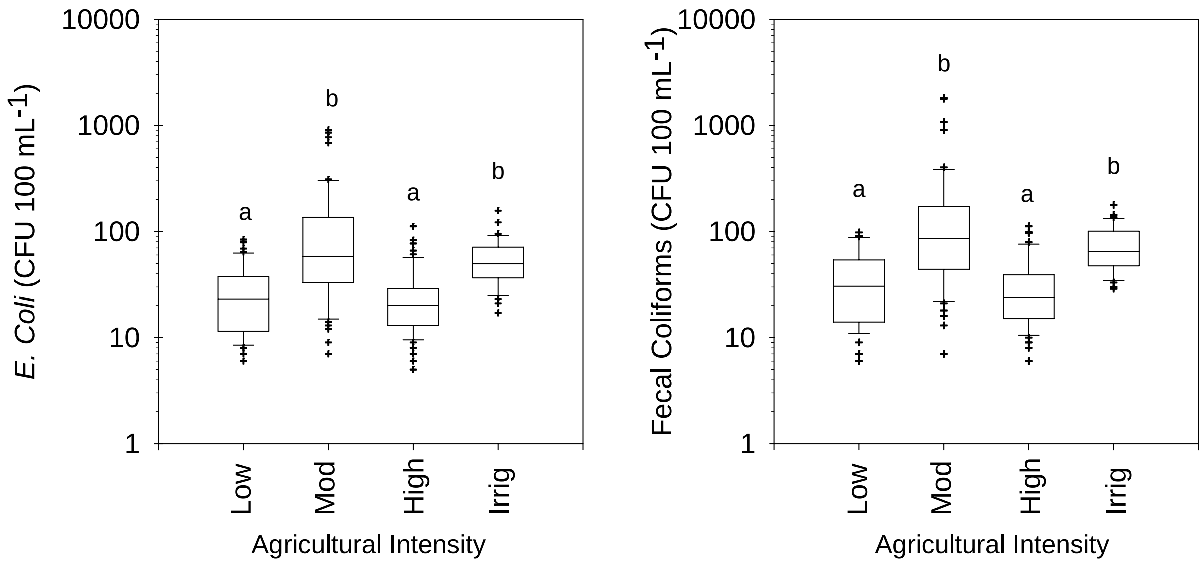

Figure 4.2. Boxplots of the annual geometric means of fecal coliform and *E. coli* for 23 AESA streams grouped by agricultural intensity where Mod represents moderate and Irrig represents irrigated watersheds. Boxplots stretch from the 25th percentile to the 75th percentile with the horizontal line in the middle of the box representing the median. Vertical lines represent 1.5 times the interquartile range, while crosshairs represent minima and maxima data points. Groups with the same letter are not significantly different (ANOVA and Tukey post hoc test, $p<0.05$).

Similarities and differences in fecal bacteria concentrations among streams within the same agricultural intensity category were explored. This was done to assess whether the agricultural intensity categories were a good indication of bacterial contamination or whether additional factors should be considered. Within both the low and moderate watershed intensity categories, there was one stream with significantly higher or lower annual geometric mean concentrations compared to other streams in the same category. In the low agricultural intensity grouping, significantly higher *E. coli* and fecal coliform concentrations were observed in Rose Creek (Figures 4.3a and 4.4a, Tukey's post hoc, $p<0.005$). In the moderate agricultural intensity category, significantly higher concentrations were observed in Meadow Creek (Figure 4.3b and 4.4b, Tukey's post hoc, $p\leq0.001$). In the high intensity category, concentrations in Strawberry Creek were significantly higher than in four streams (Haynes M6, Renwick, Stretton, and Wabash Creeks) (Tukey's post hoc, $p\leq0.01$) but similar to three others (Buffalo, Ray, and Threehills Creeks) (Figure 4.3c and 4.4c). Uniform bacteria concentrations were observed in the irrigated watersheds (Figure 4.3d and 4.4d, Tukey's post hoc, $p>0.05$).

The higher than expected *E. coli* and fecal coliform concentrations in Rose and Meadow Creeks indicate that there may be a need for management improvements in these watersheds. Furthermore, the higher concentrations may be the result of a poorly managed operation closer to the sampling site in Rose and Meadow Creeks. The other low and moderate agricultural intensity watersheds may also have some poorly managed sites, but if these operations are located further upstream of the sampling site, their contributions may be masked by dilution and therefore less noticeable at the outlet. Though the difference is not as dramatic as for Rose and Meadow Creeks, Strawberry Creek may also be a stream to investigate further. Manure production within a watershed is also an indicator to consider. As such, the relationship between manure production (tones per acre) in each basin and bacteria concentrations is explored later in this chapter (see Correlational Relationships).

Conversely, several AESA watersheds had notably lower annual geometric mean fecal bacteria concentrations than would be anticipated based on the agricultural intensity grouping. These watersheds include Willow Creek (low agricultural intensity), Kleskun Drain (moderate agricultural intensity) and Stretton Creek (high agricultural intensity). One would anticipate that management practices are excellent in these watersheds or that there are few livestock in the watershed. As mentioned previously, the relationship between manure production (tones per acre) in each basin and bacteria concentrations is explored later in this chapter (see Correlational Relationships).

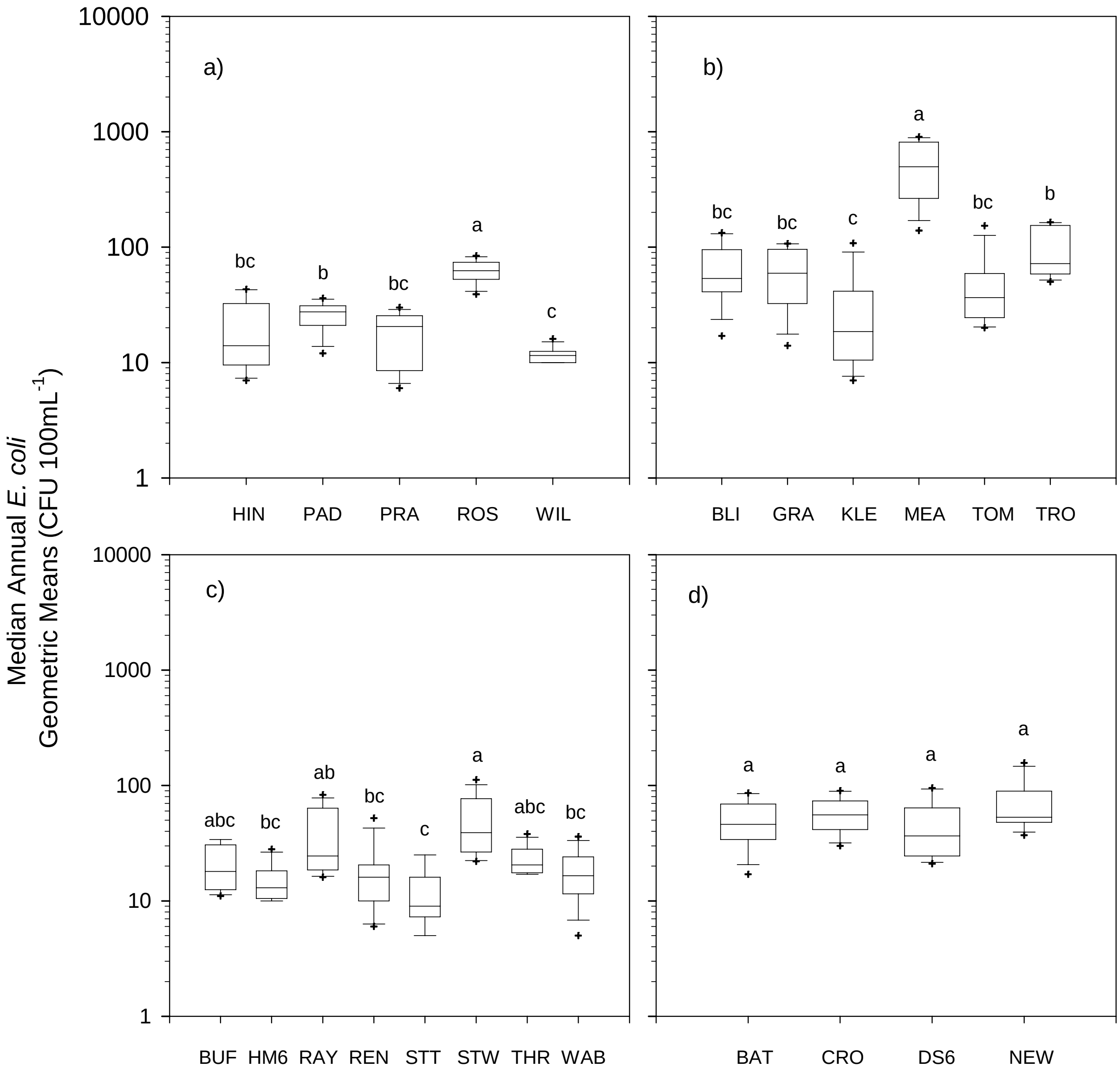

Figure 4.3. Boxplots of the annual geometric means of *E. coli* in watersheds with low (a), moderate (b), high (c), and irrigated (d) agriculture. In each plot, streams with the same letter are not significantly different from one another (ANOVA and Tukey post hoc test, p<0.05). See Table 4.2 (pg. 4-10) for full stream names.

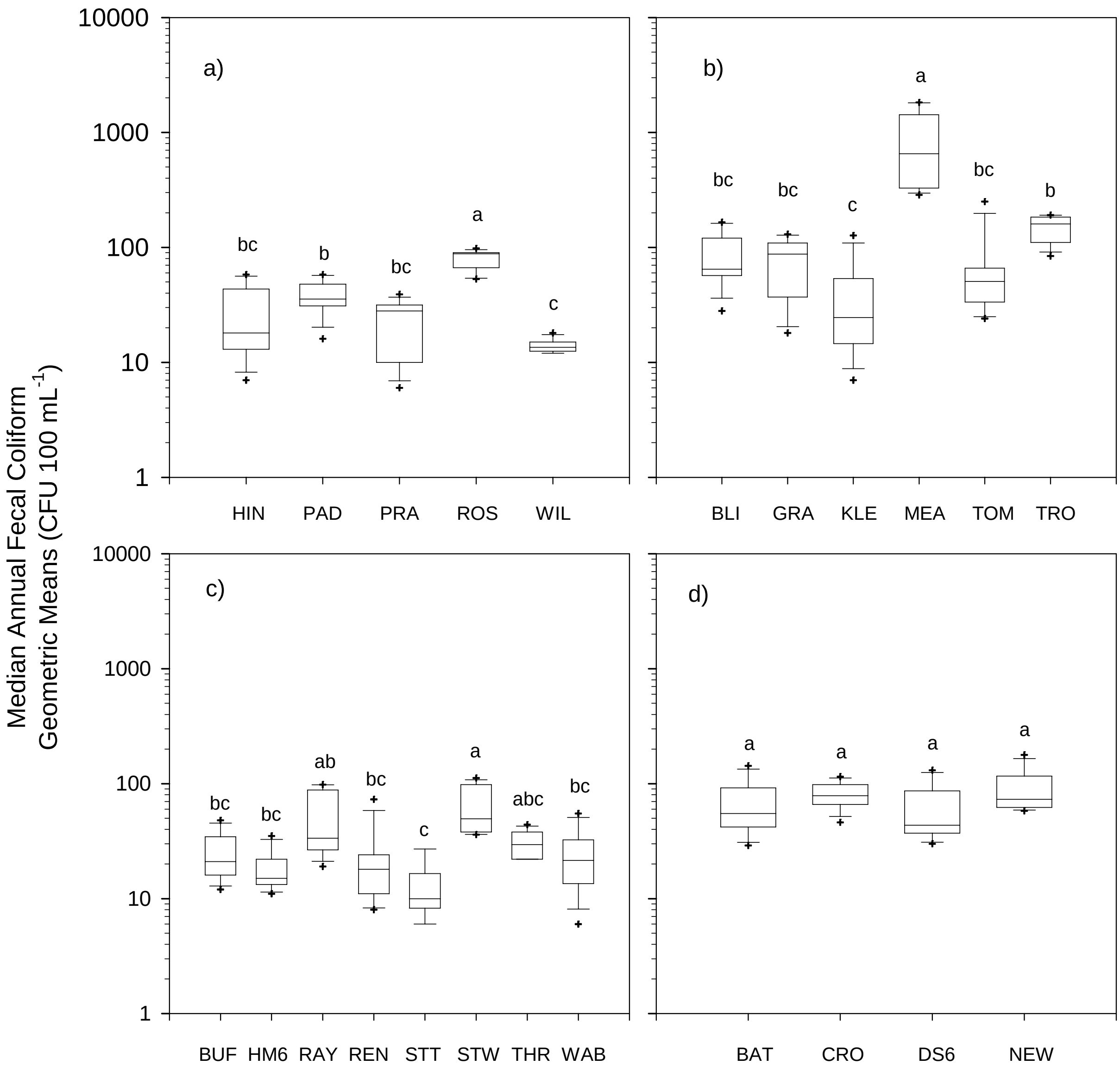

Figure 4.4. Boxplots of the annual geometric means of fecal coliforms in watersheds with low (a), moderate (b), high (c), and irrigated (d) agriculture. In each plot, streams with the same letter are not significantly different from one another (ANOVA and Tukey post hoc test, $p<0.05$). See Table 4.2 (pg. 4-10) for full stream names.

By ecoregion area. There was a significant difference in annual geometric means of fecal coliforms ($F(3,168) = 21.794$, $p \leq 0.0001$) and *E. coli* ($F(3,168) = 18.864$, $p \leq 0.0001$) among ecoregion areas. Ambient fecal coliform and *E. coli* concentrations were highest in the Grassland (Fescue Grasslands Ecoregion) and Irrigated Grassland ecoregion areas (comprised of the Moist Mixed and Mixed Grasslands Ecoregions). Annual geometric means in the Grassland watersheds were significantly higher than those in the Boreal ecoregion area (comprised of the Clear Hills Upland, Peace Lowland, Western Alberta Upland, and Boreal Transition Ecoregions) (Figure 4.5). However, a statistically significant difference was not observed between annual geometric means in the Irrigated Grassland and Boreal ecoregion areas. Fecal bacteria concentrations were significantly lower in the Parkland ecoregion area streams (Aspen Parkland Ecoregion) than all other watersheds. The Continental Divide was not included in the analysis as it only represents one stream (Willow Creek), but bacteria concentrations appeared to be similar to those in the Parkland ecoregion area. Similar patterns emerged when the Kruskal-Wallis and Mann-Whitney analyses were conducted on annual (data not shown).

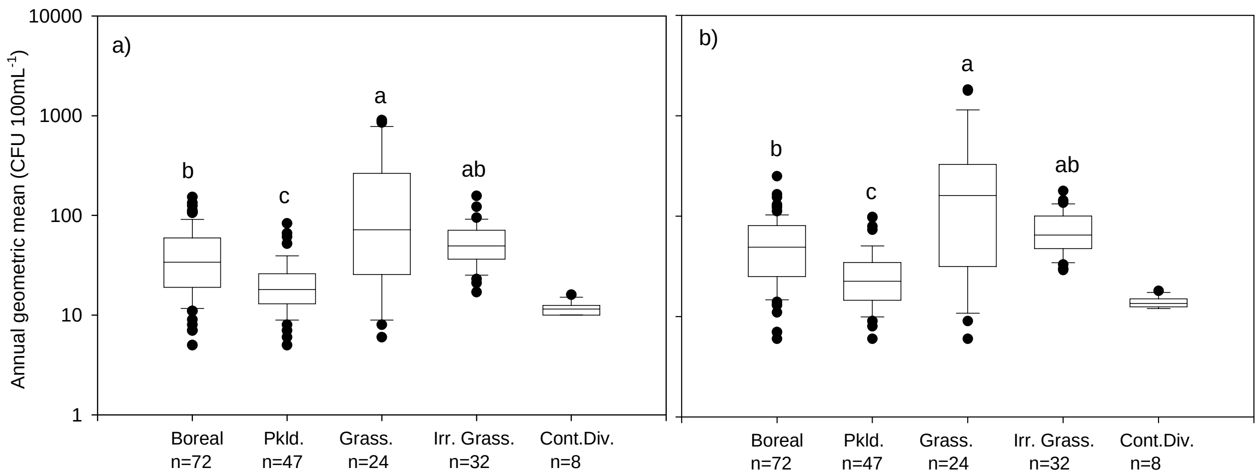

Figure 4.5. Boxplots of the annual geometric means of *E. coli* (a) and fecal coliform (b) bacteria for 23 AESA streams grouped by ecoregion area. Groups with the same letter are not significantly different from one another (ANOVA and Tukey post hoc test, $p<0.05$). See Chapter 2: Table 2.1 for the list of streams under each ecoregion area.

Annual geometric mean fecal bacteria concentrations were compared among watersheds within each ecoregion area to identify similarities and/or differences in microbial characteristics among streams in the same geographical area. Note that some ecoregion areas contain more than one agricultural intensity category but that agricultural intensity and ecoregion are not independent (Chapter 2, Figure 2.1.).

None of the three watersheds in the Grassland ecoregion area showed similar bacteria concentrations; there was a stepwise increase from Prairie Blood Coulee to Trout Creek to Meadow Creek (*E. coli*: $F(2,21)=58.282$, $p \leq 0.0001$, fecal coliform: $F(2,21)=63.026$, $p \leq 0.0001$). Note that Prairie Blood Coulee was classified as a low agricultural intensity watershed (but may have increased to a moderate agricultural intensity watershed), and Trout and Meadow Creeks were classified as moderate agricultural intensity watersheds. It is likely in this case that management practices within the individual watersheds (in terms of the prominent farm/industry type and the location of cattle operations and resulting risk of runoff) override ecoregional influences on fecal bacteria levels. This specifically applies to Meadow Creek where anomalously high bacteria levels have been observed. Additionally, Prairie Blood Coulee is a unique watershed in this ecoregion area in that it has a high percentage of crop land (81%), whereas Meadow and Trout Creeks are dominated by grassland (74 and 67%, respectively) (Chapter 2, Table 2.9). The differences in dominant land cover suggest that these watersheds are dominated by different farm industries (e.g., livestock versus commodities) which influence the observed bacteria concentrations.

Stronger similarities in bacterial concentrations were observed among watersheds in specific Ecoregions within the Boreal ecoregion area. Three of the eight streams (Hines Creek, Wabash Creek, and Kleskun Drain) in the Boreal ecoregion area had concentrations that were lower than one or more of the other streams. Two of the Boreal streams with lower bacteria concentrations were located in Peace Region (Hines Creek and Kleskun Drain). Also, higher concentrations were measured in the Boreal Transition and Western Alberta Upland Ecoregions.

In the Parkland and Irrigated Grassland ecoregion areas, the majority, if not all streams, showed uniform fecal bacteria levels. These ecoregion areas also contain watersheds with the same agricultural intensity classification.

By stream. There was little variability among stream *E. coli* and fecal coliform annual geometric means with the exception of Meadow creek (Figure 4.6). Meadow Creek had approximately 4 and 7x higher concentrations than the second highest annual geometric mean in Trout Creek for *E. coli* and fecal coliforms, respectively. Excluding Meadow Creek, median annual geometric means ranged from 9 to 72 CFU $100mL^{-1}$ for *E. coli* and from 10 to 160 CFU $100mL^{-1}$ for fecal coliforms. Meadow Creek had concentrations of 653 CFU $100mL^{-1}$ fecal coliforms and 496 CFU $100mL^{-1}$ *E. coli*.

Inter-annual patterns. Initial inspection of eight years of annual ambient concentrations showed no apparent provincial-scale increase or decrease in fecal coliform or *E. coli* concentrations with time (Figure 4.7). Annual geometric means for all 23 streams ranged from 6 to 1820 CFU 100 mL^{-1} fecal coliforms and 5 to 902 CFU 100 mL^{-1} for *E. coli*. The higher values, however, were only found in one stream: Meadow Creek. In the other watersheds, concentrations consistently ranged between 6 and 250 CFU 100 mL^{-1} for fecal coliforms and 5 and 160 CFU 100 mL^{-1} for *E. coli*. A Kruskal-Wallace analysis of variance using geometric means of ambient annual data (for all watersheds) indicated there was no significant difference among years for either fecal coliforms ($H = 8.811$; $p = 0.639$) or *E. coli* ($H = 7.863$; $p = 0.642$).

With the exception of a few watersheds, there were no inter-annual patterns in either fecal coliforms or *E. coli* from 1999 to 2006. Summary tables of annual median and geometric mean concentrations for fecal coliform and *E. coli* can be found in Tables 4.3 and 4.4. Watersheds that showed some inter-annual patterns are discussed further and include Meadow Creek, Ray Creek, and Blindman River. In these watersheds, fecal bacteria concentrations, NFR concentrations, and Census of Agriculture manure production percentiles were compared to investigate explanations for the changes in the fecal bacteria concentrations between 1999 and 2006. Non-filterable residue concentrations reflect the potential for sediment loading which has previously been described and documented as a transport mechanism for fecal bacteria (Jamieson et al. 2005).

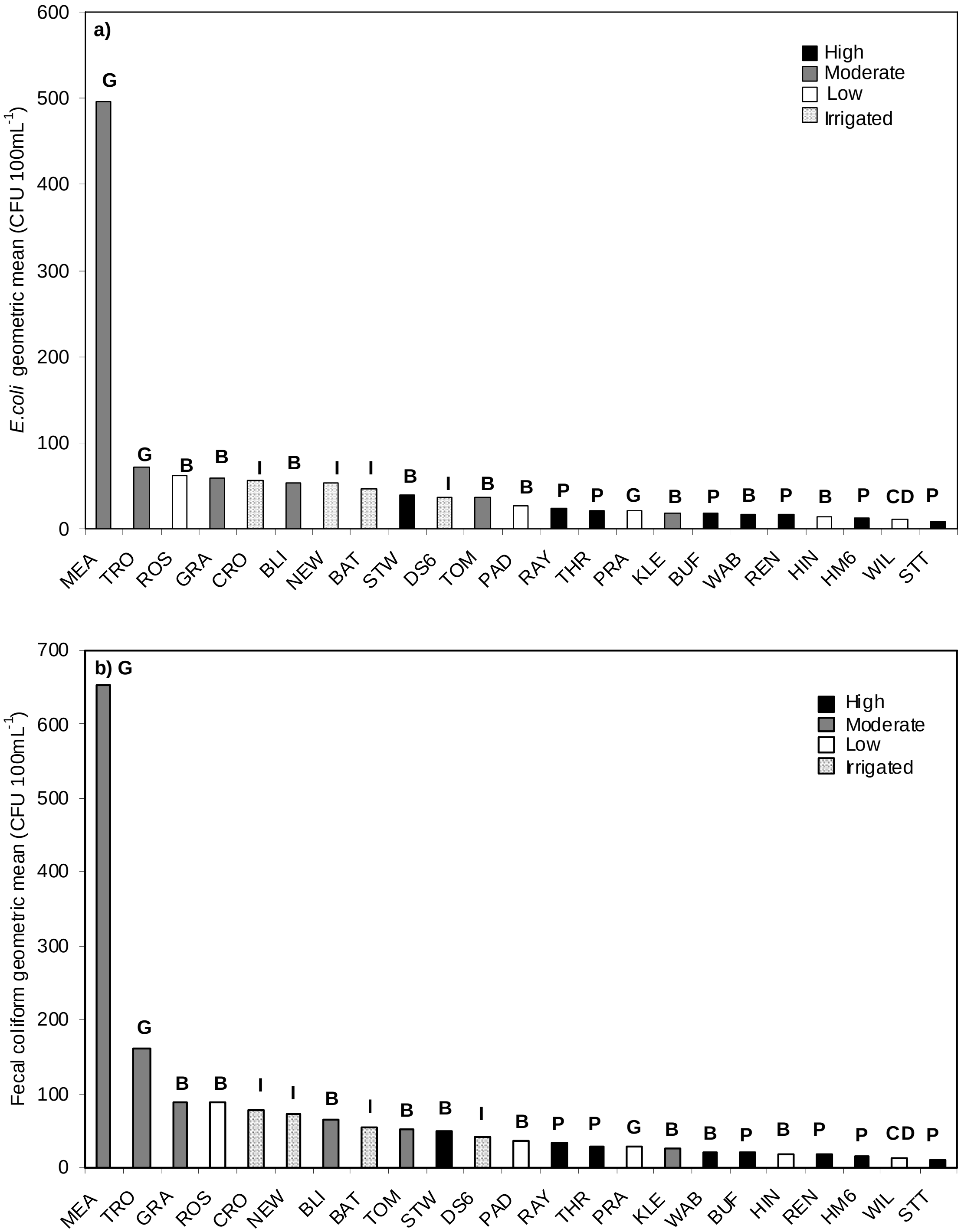

Figure 4.6. Median annual geometric means by agricultural intensity for fecal coliforms (a) and *E. coli* (b) in 23 AESA watersheds. Ecoregion areas are represented as B= boreal, P= parkland, G=grassland, and I = irrigated grassland. See Table 4.2 (pg. 4-10) for full stream names.

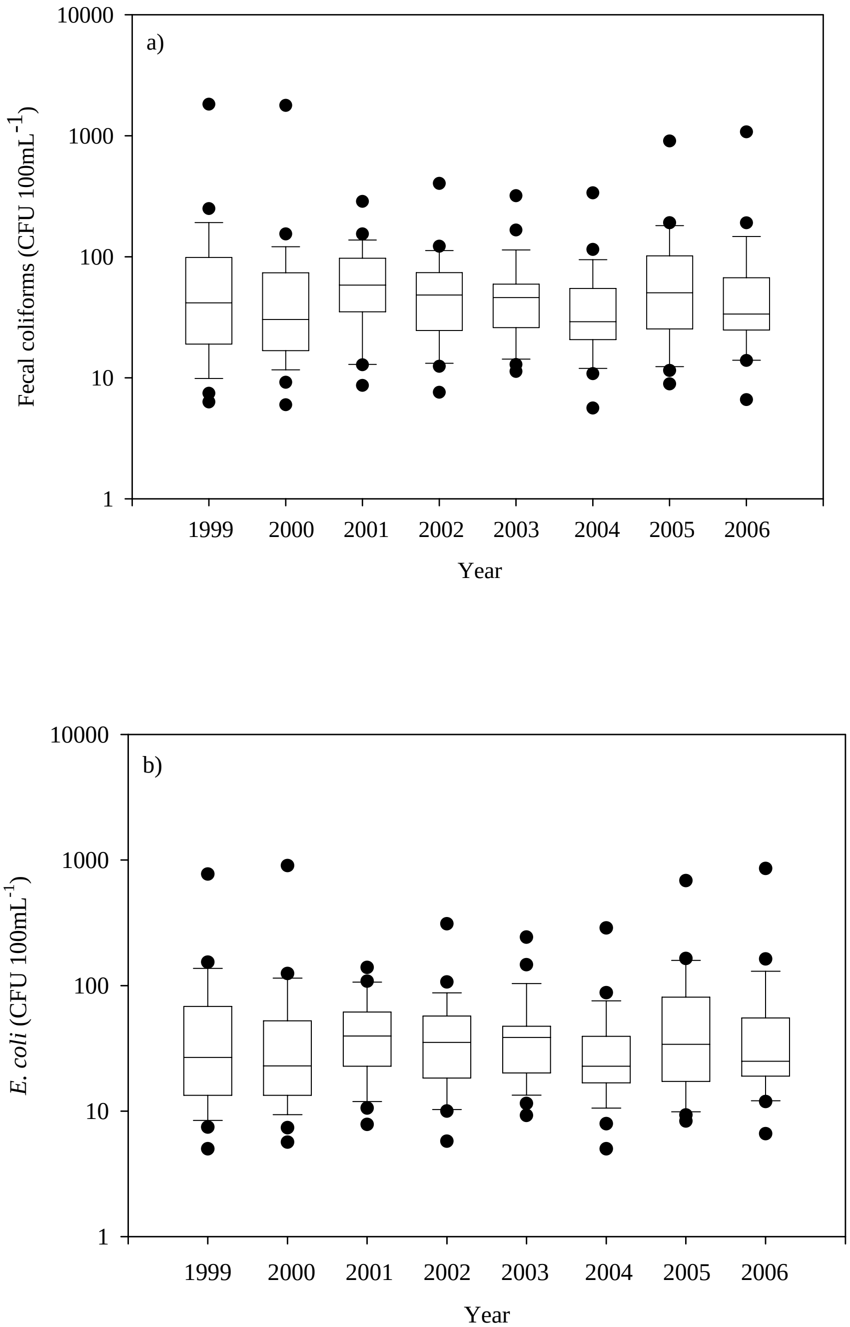

Figure 4.7. Boxplots for fecal coliform (a) and *E. coli* (b) geometric means for all streams from 1999 to 2006. Note the logarithmic scale on the y-axis.

Table 4.3. Annual median and geometric mean fecal coliform concentrations (CFU 100mL^{-1}), 1999 to 2006.

	1999		2000		2001		2002		2003		2004		2005		2006	
	Ambient	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean
Low Agricultural Intensity																
Hines Creek	5	7	10	18	40	35	65	52	60	58	8	11	20	18	14	15
Paddle Creek	50	38	35	30	40	55	20	33	50	41	10	16	60	58	40	32
Prairie Blood Coulee	23	26	5	6	5	9	20	31	9	11	18	32	39	30	53	39
Rose Creek	100	90	40	53	170	90	85	90	40	56	96	86	135	77	165	98
Willow Creek	15	18	5	13	6	13	8	12	5	16	5	12	8	14	5	14
Moderate Agricultural Intensity																
Blindman Creek	160	165	200	154	90	70	60	59	80	59	30	28	110	87	60	55
Grande Prairie Creek	90	84	20	18	110	130	100	122	110	91	40	48	100	96	10	26
Kleskun Drain	20	16	5	21	70	127	43	68	55	39	28	28	5	13	5	7
Meadow Creek	1800	1820	2300	1779	110	286	430	402	380	319	440	337	480	903	910	1074
Tomahawk Creek	170	250	65	58	35	40	70	74	70	51	10	24	50	50	10	27
Trout Creek	300	177	120	113	210	154	230	108	210	166	84	84	160	191	110	190
High Agricultural Intensity																
Buffalo Creek	10	15	5	12	5	19	10	17	15	30	10	23	50	48	30	39
Haynes Creek (M6)	10	24	10	16	10	14	10	13	20	35	-		5	11	10	15
Ray Creek	75	97	90	79	111	98	20	35	10	26	10	19	27	32	20	27
Renwick Creek	10	24	5	9	5	13	10	8	5	15	10	21	85	73	5	24
Stretton Creek	10	10	10	13							5	6	5	9	20	27
Strawberry Creek	110	99	105	112	60	97	20	36	100	57	50	39	35	37	35	42
Threehills Creek	20	22	30	36	60	44	10	25	15	22	20	22	45	40	20	34
Wabash Creek	5	6	20	19	45	41	30	24	10	13	70	55	20	24	10	14
Irrigation Agriculture																
Battersea Drain	170	143	28	29	67	49	56	72	70	53	52	35	96	112	82	57
Crowfoot Creek	60	46	195	92	96	65	115	67	119	87	120	115	84	104	85	70
Drain S6	33	42	43	33	49	62	34	45	46	41	25	30	60	131	220	111
New West Coulee	81	61	92	58	115	97	82	81	86	63	96	65	280	178	150	136

Table 4.4. Annual median and geometric mean *E. coli* concentrations (CFU $100mL^{-1}$), 1999 to 2006.

	1999		2000		2001		2002		2003		2004		2005		2006	
	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean	Median	Geometric Mean
Low Agricultural Intensity																
Hines Creek	5	7	10	14	30	23	40	42	45	43	5	8	10	11	14	14
Paddle Creek	40	27	30	28	20	36	10	24	30	28	10	12	36	34	10	18
Prairie Blood Coulee	11	18	5	6	5	8	13	25	5	9	15	26	33	23	40	30
Rose Creek	70	69	40	39	130	61	85	79	40	47	70	64	96	58	145	84
Willow Creek	8	10	5	10	5	11	7	10	5	16	5	12	8	12	5	13
Moderate Agricultural Intensity																
Blindman Creek	130	133	190	125	60	51	60	56	80	39	10	17	81	65	50	43
Grande Prairie Creek	80	64	10	14	100	106	80	107	110	85	36	39	80	55	10	26
Kleskun Drain	5	12	5	13	60	108	43	51	40	32	20	24	5	9	5	7
Meadow Creek	850	771	1100	902	72	139	330	309	290	242	350	287	455	684	690	855
Tomahawk Creek	130	153	45	53	25	34	60	65	50	39	10	20	20	28	10	21
Trout Creek	77	56	25	50	68	61	120	74	210	146	65	70	160	164	88	162
High Agricultural Intensity																
Buffalo Creek	5	11	5	12	5	16	5	13	13	27	8	20	50	34	30	34
Haynes Creek (M6)	10	20	5	13	10	13	5	10	10	28	-		5	10	8	12
Ray Creek	62	66	65	61	106	83	5	16	10	20	5	17	20	26	5	23
Renwick Creek	10	20	5	7	5	13	5	6	5	14	10	18	35	52	5	21
Stretton Creek	10	9	10	13	-		-		-		5	5	5	8	20	25
Strawberry Creek	75	77	105	112	60	77	20	30	80	47	20	31	15	23	15	22
Threehills Creek	15	17	30	30	45	38	5	18	10	18	10	17	25	26	15	23
Wabash Creek	5	5	20	18	20	27	20	21	10	11	40	36	10	15	10	12
Irrigation Agriculture																
Battersea Drain	90	82	15	17	43	39	35	56	47	44	39	29	73	86	75	48
Crowfoot Creek	35	30	108	53	62	47	46	36	65	60	99	87	65	90	85	58
Drain S6	24	26	10	23	34	40	19	35	46	38	9	21	56	95	170	88
New West Coulee	64	45	51	37	57	53	72	57	49	53	71	51	265	157	110	122

Although Meadow Creek had the highest fecal coliform and *E. coli* concentrations of all AESA watersheds, the concentrations were not as high or as variable from 2002 through 2003 (Figure 4.8a and b). These patterns were not related to NFR (Figure 4.8c) or a change in manure production; however, the box plots and data do demonstrate a lack of high fecal coliform maxima concentrations in 2002 through 2003. This difference between the maxima was considerable in May and June and differed by one to two orders of magnitude. It is possible that the maxima were not caught during sampling in these years or the sources (e.g., cattle, wildlife, and humans) were not contributing as intensely as in previous years. Bacteria numbers were often high in June, and only 2 samples were taken in June of 2002 and 2003 compared to the collection of 3 to 4 samples every June in other years. The fewer samples taken in June of 2002 and 2003 appear to be related to the sampling regime rather than a lack of precipitation or discharge (Chapter 2: Results and Discussion – Hydrology, Meadow Creek). Therefore, peaks in bacteria concentrations may have been missed.

Ray Creek had high fecal coliform and *E. coli* geometric means from 1999 through 2001, followed by a decrease from 2002 through 2006 (Figure 4.9a). This pattern paralleled declines in NFR, especially when compared to *E. coli* (Figure 4.9b and c). There was also a decrease in manure production in the watershed in 2001 (Chapter 2: Results and Discussion - Agricultural Intensity Classification in 1996, 2001 and 2006). The decrease in bacteria concentrations in the latter years of monitoring may suggest that the source of the high bacteria numbers in 1999 through 2001 had been removed.

In the Blindman River, fecal coliforms and *E. coli* decreased from 1999 through 2006 (Figure 4.10a and b). Maximum concentrations of these parameters were also highest in the earlier years of the study (i.e., 1999 and 2000). The decrease in fecal bacteria concentrations was not observed in the NFR concentrations (Figure 5.10c). The decrease in fecal bacteria concentrations over the monitoring period appeared to coincide with substantial decreases in manure production from 1996 to 2001 and again in 2006 (Chapter 2: Figure 2.24, Table 2.11).

Overall, bacteria patterns varied from 1999 through 2006, and there was no apparent provincial trend in the geometric means of all watersheds. Fecal coliform and *E. coli* concentrations in some watersheds appeared to be related to NFR concentrations or influenced by manure production, but the pattern was not universal. In fact, some streams with high manure production (e.g., Wabash Creek) often had very low concentrations of fecal bacteria, sometimes only slightly above the method detection limit.

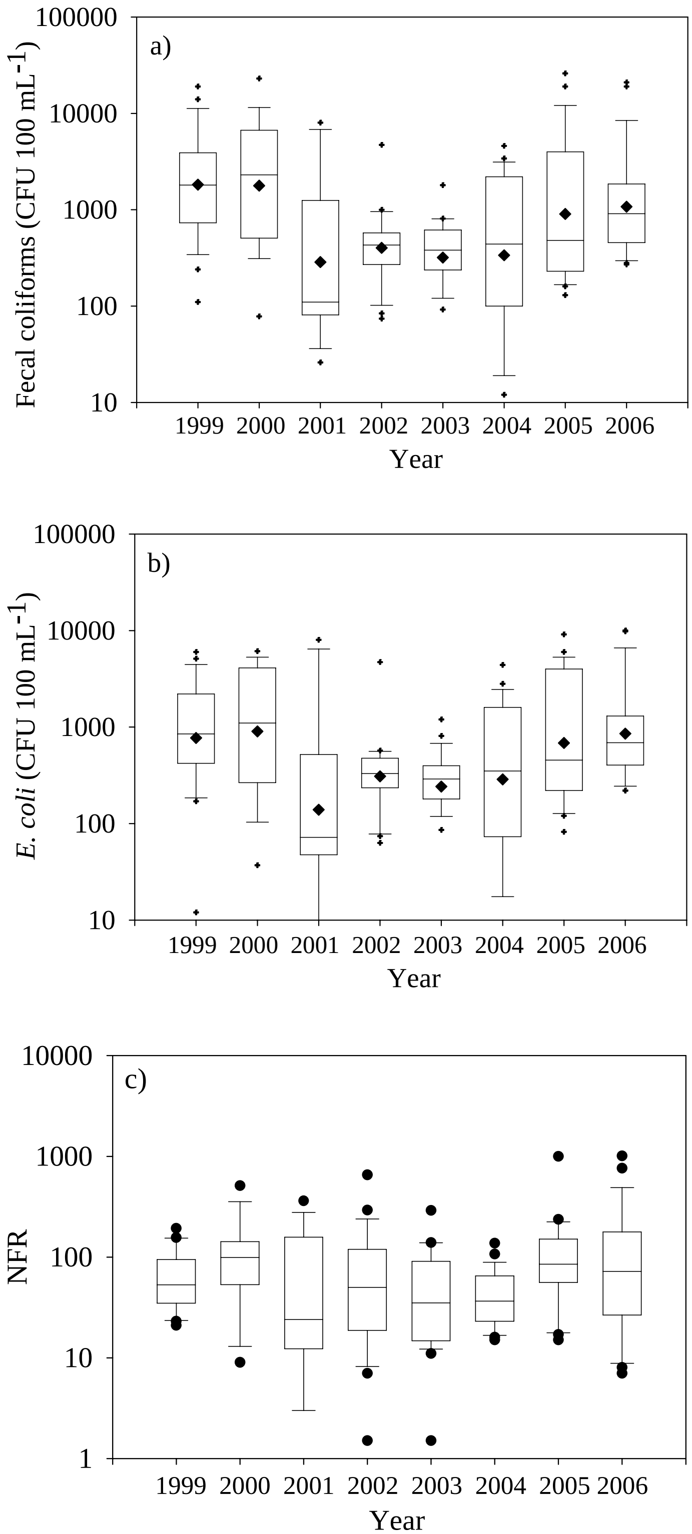

Figure 4.8. Boxplots of fecal coliforms (a), *E. coli* (b), and NFR (c) for Meadow Creek (1999 through 2006). Diamonds in (a) and (b) represent geometric means. Note the logarithmic y-axis scale.

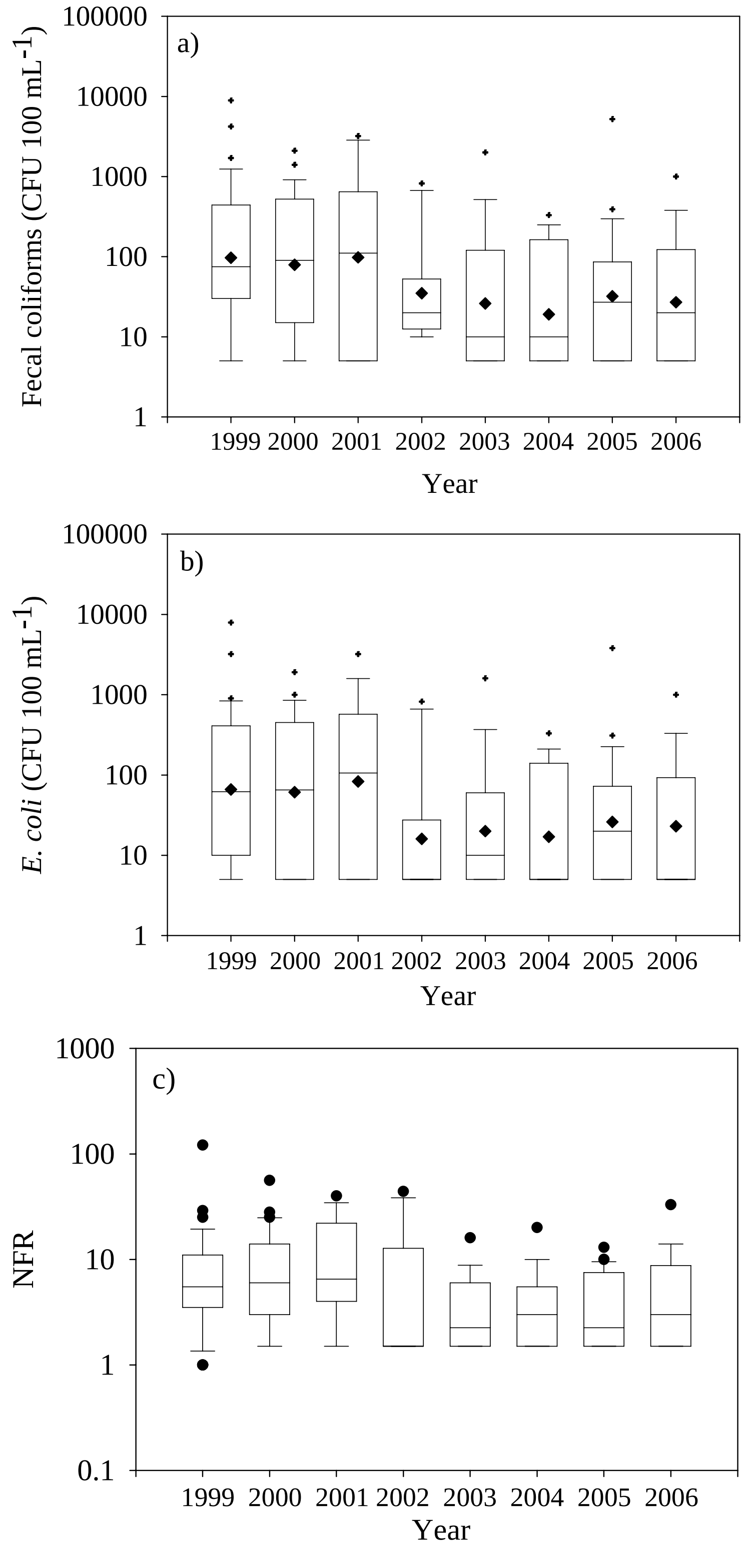

Figure 4.9. Boxplots of fecal coliforms (a), *E. coli* (b), and NFR (c) for Ray Creek (1999 through 2006). Diamonds in (a) and (b) represent geometric means. Note the logarithmic y-axis scale.

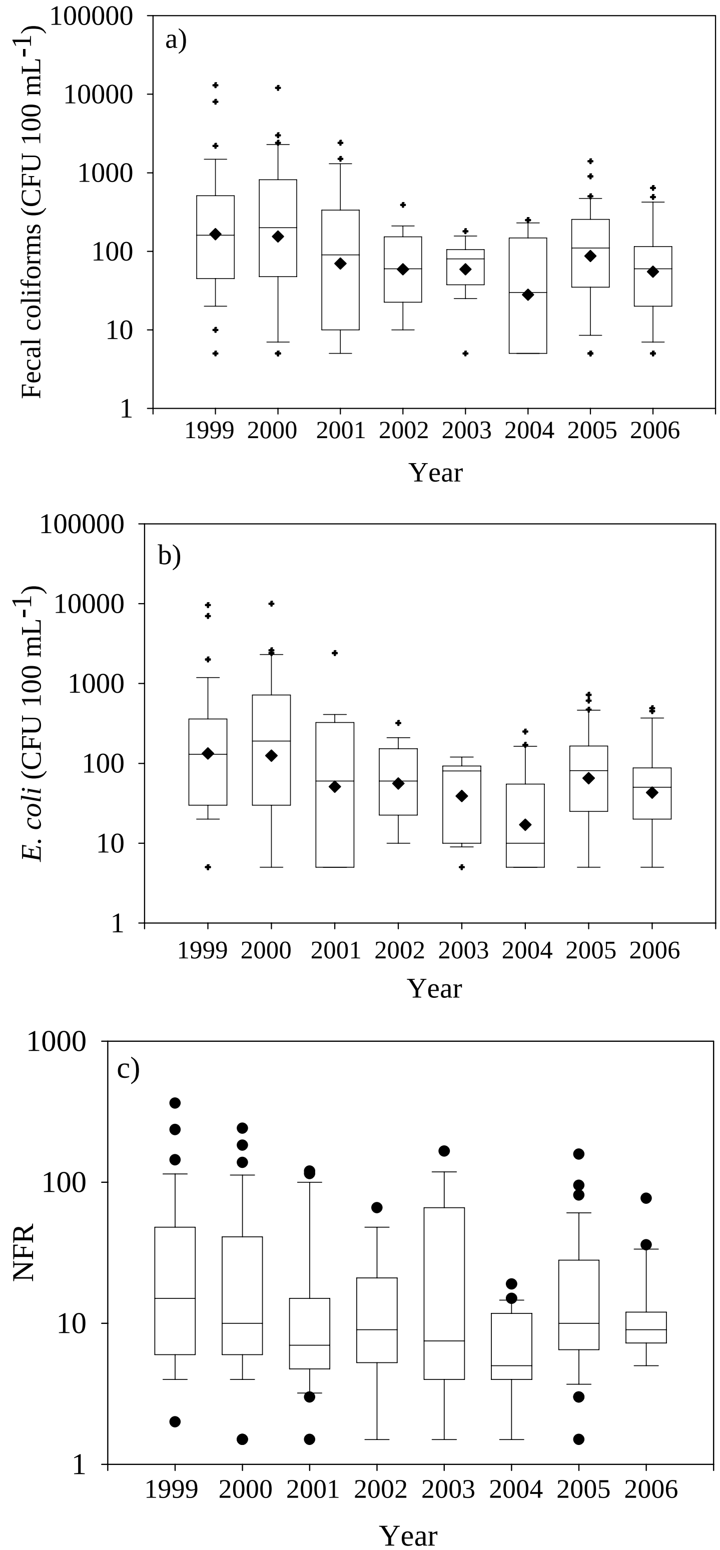

Figure 4.10. Box Plots of fecal coliforms (a), *E. coli* (b), and NFR (c) for the Blindman River from 1999 to 2006. Diamonds in (a) and (b) represent geometric means. Note the logarithmic y-axis scale.

Correlational Relationships

Fecal bacteria concentrations and agricultural intensity (AI) metrics. Spearman Rank Correlations were run between the median of annual geometric and arithmetic means of fecal coliforms and *E. coli* in 23 AESA streams (1999 to 2006) and the average overall agricultural intensity (AI) metric (1996, 2001, and 2006) as well as the individual metrics (e.g., fertilizer sales, chemical sales, and manure production percentiles). Correlations served as a means to examine the relationship between fecal contamination and agriculture. Correlation coefficients were similar for fecal coliforms and *E. coli*, so with the exception of Table 4.5, only fecal coliform data have been presented.

No significant correlations were found between the median of annual geometric means and the average agricultural intensity (r_s=-0.097, n=23, p>0.05), fertilizer sales (r_s =-0.098, n=23, p>0.05), chemical sales (r_s =-0.096, n=23, p> 0.05), or manure production (r_s = 0.024, n=23, p>0.05) percentiles. Correlations were slightly stronger when median annual arithmetic means were used in lieu of geometric means, particularly for the average manure production percentile (fecal coliform r_s=0.207, n=23, p>0.05), but they were not significant (Table 4.5).

Table 4.5. Spearman rank correlation coefficients (n=23) between average agricultural intensity, average manure production metrics (1999, 2001, 2006) and median annual geometric and arithmetic mean *E. coli* and fecal coliform concentrations. None of the correlations were significant (p>0.05).

	Average Agricultural Intensity		Average Manure Production	
	E. coli	Fecal coliforms	*E. coli*	Fecal coliforms
Geometric Mean (Median Annual)	-0.103	-0.097	0.020	0.024
Arithmetic Mean (Median Annual)	0.151	0.149	0.204	0.207

The poor relationship with the average AI percentile suggests that this was not an appropriate land-based metric for predicting fecal contamination in an agricultural watershed. Two of the three components of the AI metric are based on measures of cropping intensity (fertilizer and chemical expense percentiles), so the weak relationship was not entirely surprising. The relationship with the manure production percentile was also too weak to be deemed a reliable metric at a provincial scale (Table 4.5).

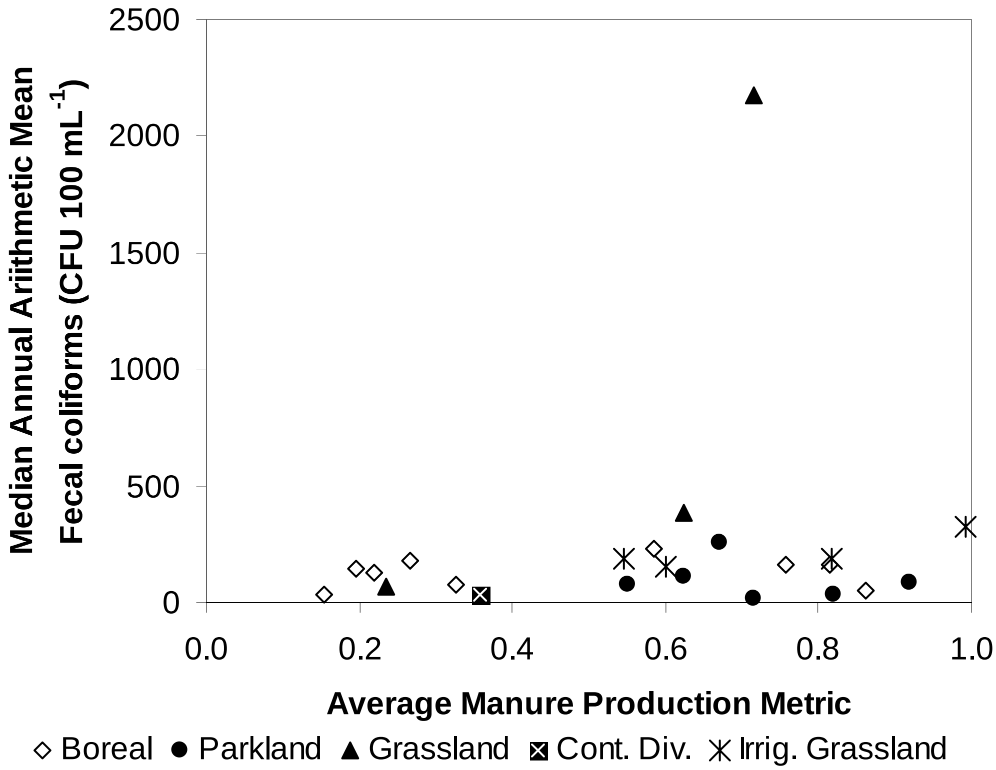

Figure 4.11. Average agricultural intensity metric (1999, 2001, 2006) and median annual arithmetic mean fecal coliform concentrations for 23 AESA watersheds by ecoregion area, where Cont.Div. = Continental Divide and Irrig. Grassland = Irrigated Grassland ecoregion area.

The relationship of median annual arithmetic mean fecal coliforms with manure production was explored on a regional (ecoregion area) scale to assess whether similarities in regional land uses or agricultural industries would strengthen the relationship. There appeared to be a weak relationship (Figure 4.11), but with such a small sample size, it is unclear whether that relationship is real. Additionally, none of the correlations were significant (Table 4.6).

Table 4.6. Spearman rank correlation coefficients for fecal coliforms by ecoregion area. Correlations were not significant (p>0.05).

	Average Manure Production			
	Boreal	Parkland	Grassland	Irrigated Grassland
Arithmetic Mean (Median Annual)	0.233[z] n=9	-0.257 n=6	1.000 n=3	0.800 n=4

[z]With Wabash removed r_s=0.571 (n=8)

Based on the average of the 1996, 2001, and 2006 manure production metrics (Statistics Canada 1996, 2001, and 2006), one would hypothesize that the highest bacteria concentrations would occur in two Irrigated Drains (Battersea Drain and New West Coulee) in the Irrigated Grassland ecoregion area, two high intensity watersheds in the Parkland ecoregion area (Haynes

M6 and Threehills Creeks), two high intensity watersheds in the Boreal ecoregion area (Wabash and Strawberry Creeks) and one moderate intensity watershed in the Boreal ecoregion area (Blindman River) (Chapter 2, Figure 2.24, Table 2.11). The average manure production metric for these seven watersheds was ≥ 0.75, which represents the highest agricultural intensity category. Based on the median annual fecal coliform arithmetic means in the AESA watersheds, only two of the seven watersheds with highest ranked average manure production also had among the highest bacteria concentrations: Battersea Drain and New West Coulee (Irrigated Grassland). Haynes Creek (M6) (high intensity, Parkland) and Wabash Creek (high intensity, Boreal) had among the lowest fecal coliform concentrations despite high manure production metrics. Different manure management practices and/or farm types in southern Alberta may provide some insight into this disparity. Likewise, hydrological regime may also need to be taken into consideration as stream flow in the Parkland ecoregion area is generally limited to the spring and early summer during spring melt. In contrast, Grassland streams often flow throughout the sampling season.

Fecal bacteria concentrations and physical and chemical parameters. Median annual ambient bacteria concentrations were correlated with physical and chemical parameters to explore broad scale relationships. The results using the median of annual medians are presented in this report (i.e., n=23), though correlations were also examined using annual medians for all years (1999 to 2006) (i.e., n= 23 x 8, more or less). The same relationships held true for both data sets (Table 4.7).

The processes (e.g., storm runoff) and pathways (e.g., ditches) that can lead to high suspended sediment concentrations in streams and rivers may also contribute to fecal matter entering surface waters (Minnesota River Basin Data Centre 2003). As such, it was hypothesized that annual runoff depth, annual precipitation, suspended sediment, and discharge would show positive relationships with fecal bacteria on a provincial scale.

A strong relationship was observed between the total suspended sediment (i.e., NFR) and median annual concentrations of fecal coliforms and *E. coli* for the 23 AESA streams (Table 4.7). A review of the data (Figure 4.12) showed that Meadow Creek drove the positive relationship; however, with Meadow Creek removed, the rank correlation coefficients did not change dramatically (fecal coliform: $r_s = 0.758$, *E. coli*: $r_s = 0.722$, $p<0.005$). The data suggest that, in general, agricultural streams in Alberta with higher suspended sediment are also more likely to have higher bacteria concentrations. The underlying cause for higher sediment levels may be natural (e.g., silty substrate) or the result of poor riparian or land management (e.g., eroding banks, bare soil in fields, ditches or gullies). There were only three streams - Meadow, Strawberry, and Tomahawk Creeks - where median annual NFR concentrations were greater than 30 mg L^{-1} (Figure 4.12); however, the bacteria concentrations varied greatly among these watersheds (e.g., 55 to 58 CFU 100 mL^{-1} for Strawberry and Tomahawk Creeks, respectively compared to 460 CFU 100 mL^{-1} for Meadow Creek).

The relationship between fecal bacteria and NRF was investigated for each ecoregion area. There appeared to be a stronger relationship between fecal bacteria and NFR in the southern part of the province (Grassland and Irrigated Grassland ecoregion areas) than in the north (Boreal and Parkland ecoregion areas) (Figure 4.13).

Table 4.7. Spearman Rank correlation coefficients between ambient fecal bacteria concentrations and physical parameters for median of annual medians (n=23) and annual medians (n=180).

	Fecal coliform	*E. coli*	Fecal coliform	*E. coli*
	n=23		n=180	
TSS/ NFR	**0.757**	**0.782**	**0.620**	**0.636**
pH	0.165	0.125	0.203	0.177
Conductivity	-0.252	-0.324	-0.198	-0.231

Bold denotes significance at $p<0.005$.

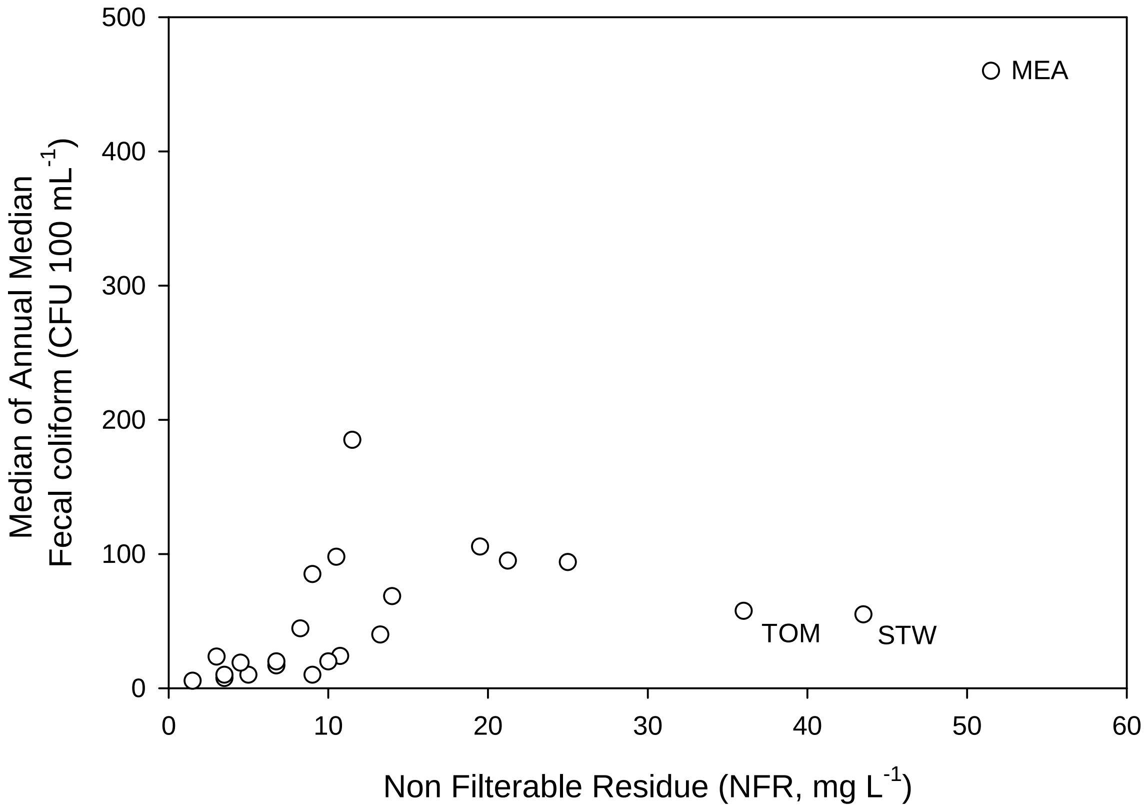

Figure 4.12. Scatter plot of TSS/NFR versus fecal coliform (median of annual median, n=23). Those watersheds greater than 30 mg L^{-1} NFR concentration are labeled as Tomahawk Creek (TOM), Strawberry Creek (STW), and Meadow Creek (MEA).

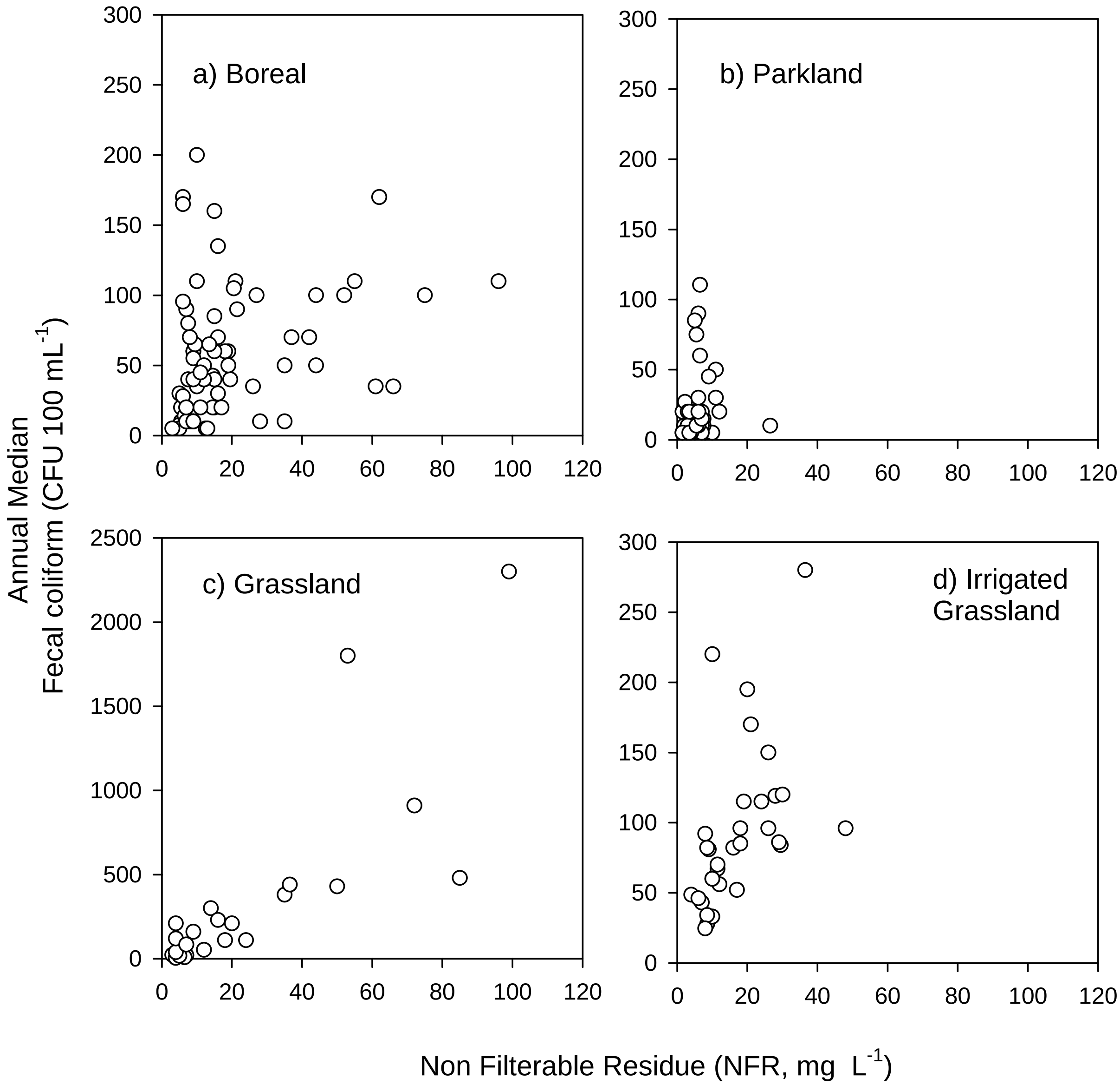

Figure 4.13. Scatter plot of TSS/NFR versus fecal coliform (annual medians) by ecoregion area: Boreal (a), Parkland (b), Grassland (c), and Irrigated Grassland (d). Note the different y-axis for the Grassland ecoregion area.

There was no significant correlation with runoff depth (Table 4.8). The analysis did not include the irrigated watersheds because of the inability to standardize based on watersheds area (see Chapter 3: Materials and Methods - Load, FWMC, and export calculations flow section). When two outliers, Meadow Creek and Willow Creek (Figure 4.14), were removed from the analysis, runoff depth was positively correlated with the fecal bacteria indicators (Table 4.9). Though a general relationship existed between the amount of water generated per unit area of the watershed (i.e., runoff depth) and fecal bacteria densities, there were clearly other factors at work.

Table 4.8. Spearman Rank correlation coefficients between ambient fecal bacteria concentrations and physical parameters (median of annual median). Irrigated watersheds have been excluded (n=19).

	Fecal coliform	*E. coli*
Runoff Depth	0.250 ($p>0.05$)	0.203 ($p>0.05$)

Table 4.9. Spearman Rank correlation coefficients between ambient fecal bacteria concentrations and physical parameters (median of annual median). Irrigated watersheds and Willow and Meadow Creeks have been excluded (n=17).

	Fecal coliform	*E. coli*
Runoff Depth	0.613 ($p<0.05$)	0.532 ($p<0.05$)

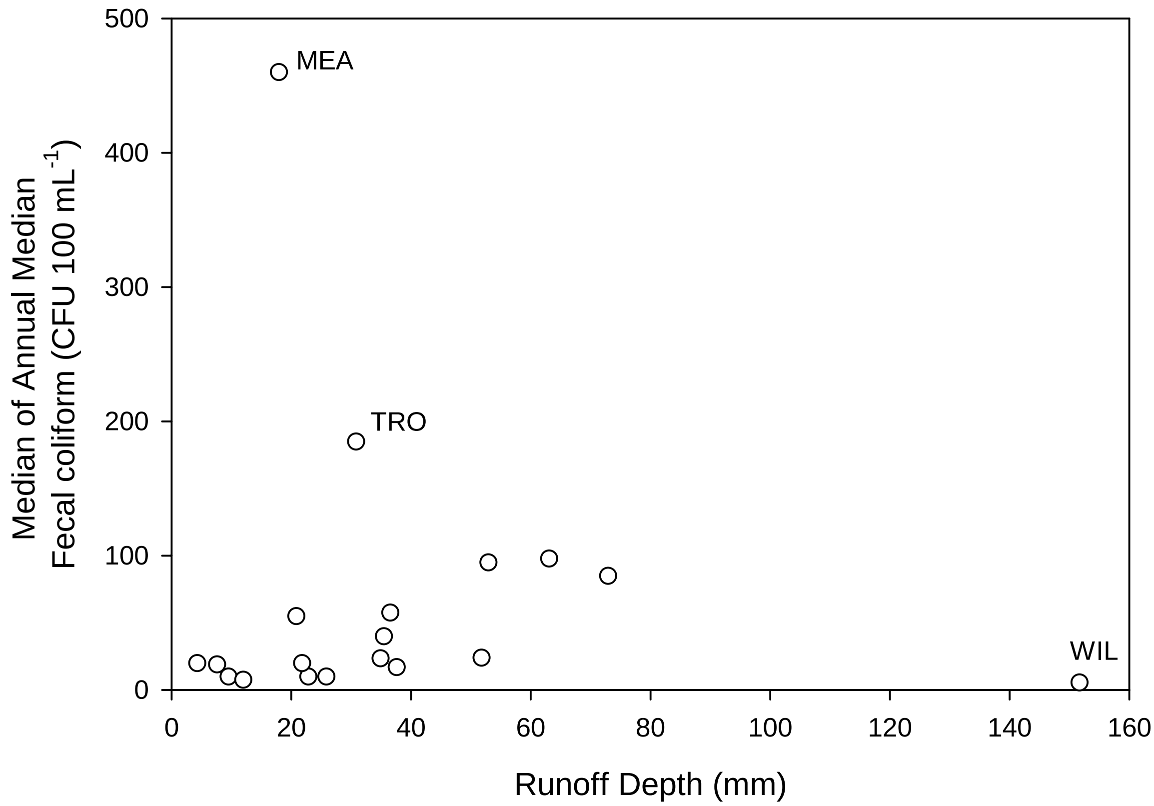

Figure 4.14. Scatter plot of runoff depth (mm) versus the median of annual fecal coliform means (1999 to 2006). The watersheds labeled in the figure are Meadow Creek (MEA), Trout Creek (TRO), and Willow Creek (WIL).

Correlations with nutrient chemistry were explored to assess whether high nutrient aquatic environments were also more likely to have high bacteria concentrations. Roszak and Colwell (1987) illustrated that bacteria survival is favored in high nutrient conditions and that fecal bacteria may also potentially reproduce in these conditions.

Overall, there did not appear to be any relationships between fecal bacteria and nutrient concentrations. Significant, negative correlations were observed between fecal coliforms and TDP and *E. coli* and TN and TDP (Table 4.10, Figure 4.15). The inverse relationship was unexpected as it suggests that higher fecal bacteria concentrations cannot be anticipated in watersheds with high nutrient concentrations. The observations support earlier conclusions that watersheds with higher agricultural intensity do not necessarily have high fecal bacteria concentrations. This has implications for planning and implementing watershed scale beneficial management practices, as the watersheds that would benefit most from reductions in fecal bacteria loads may not be the same as those that would benefit from reductions in nutrient loads.

Table 4.10. Spearman Rank correlation coefficients between ambient fecal bacteria concentrations and nutrient chemistry parameters (median of annual medians, n=23).

	Fecal coliform	*E. coli*
TN	-0.424	**-0.442**
DIN	-0.242	-0.239
N23	-0.137	-0.130
NH4	-0.320	-0.321
TP	-0.365	-0.382
TDP	**-0.463**	**-0.476**
TPP	0.140	0.150

Bold denotes significance at $p<0.05$.

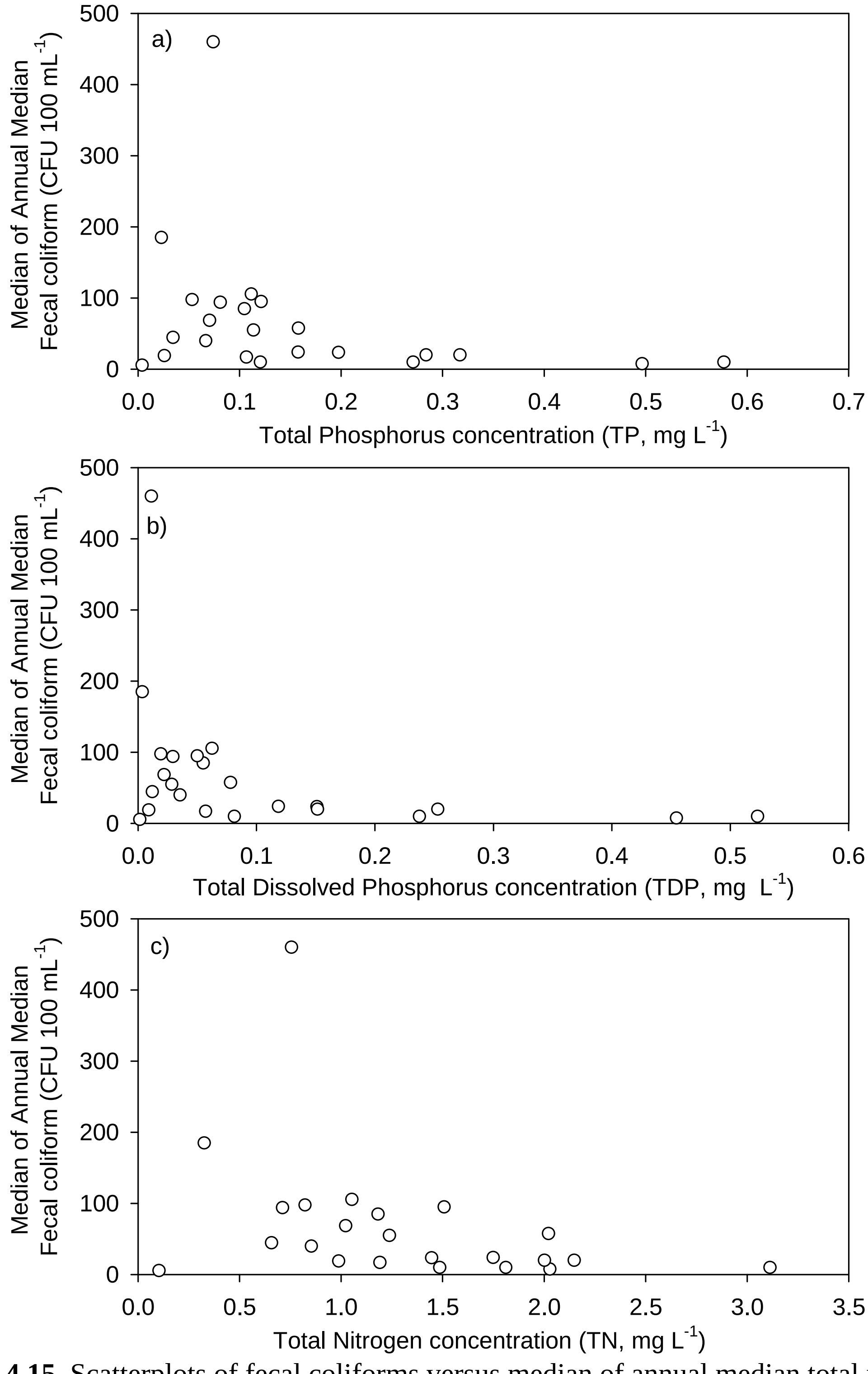

Figure 4.15. Scatterplots of fecal coliforms versus median of annual median total phosphorus (a), total dissolved phosphorus (b), and total nitrogen (c) concentrations in 23 AESA streams (1999 to 2006).

Seasonality of Ambient Fecal Coliforms and *E. coli* Concentrations

Bacteria concentrations change depending on the time of year (Tiedemann et al. 1987; Baxter-Potter and Gilliland 1988; Meals 1989). Some studies have shown that the highest fecal coliforms and other bacteria concentrations occur in the summer months (June through August) and lowest in the winter (December through February) and spring (March through May) (Tiedemann et al. 1987; Baxter-Potter and Gilliland 1988; Meals 1989). These authors have related the pattern to warmer temperature effects, a change in agricultural activity patterns, hydrologic phenomena, and a greater presence of livestock and wildlife activity (Tiedemann et al. 1987; Meals 1989). Lower stream flow during the summer enables cattle to spend more time in streams, which increases direct fecal coliform deposition to streams (Mostaghimi et al., 2000).

In the study streams, mean rank fecal coliform and *E. coli* counts were significantly higher during the summer months (June, July, and August) than all other months of sampling (Fecal coliforms: $H = 796$, $p < 0.005$; *E. coli*: $H = 742$, $p < 0.005$). Bacteria counts were also high in September and were significantly lower in the early spring (March and April). These seasonal patterns in fecal coliform and *E. coli* counts did not follow the same pattern observed for NFR, which was significantly higher in the early spring (March and April) and early summer (June and July) and lowest in October (Figure 4.16). The higher fecal bacteria counts in summer may be a result of more frequent cattle access, longer bacteria survival in warmer water temperatures, and increased transport in surface runoff during summer storm events.

Although in general bacteria counts showed higher numbers in the summer and lower values in the early spring, these seasonal patterns differed slightly by ecoregion area (Figure 4.17 a to d).

In the Boreal watersheds, mean rank fecal coliform and *E. coli* counts were significantly higher in July than all other months except June; high counts were also observed in August and September (Figures 4.17a and 4.18b). Fecal coliform and *E. coli* counts in the Boreal were not lowest in March; counts were similar to those observed in May, August, and September. The lowest counts were measured in April and October.

Mean rank fecal coliform and *E. coli* counts were significantly higher in July than all months but August in watersheds in the Parkland ecoregion area (Figures 4.17b and 4.18b). Counts were also high in June, August, and September. Mean rank bacteria counts were significantly lower in April than all other months, but values were also low in March and October.

Fecal coliform and *E. coli* followed similar patterns to one another in the Grassland ecoregion area with significantly higher counts observed in June as well as higher counts in May, July, August, and September (Figures 4.17c and 4.18c). The lowest counts were observed in March followed by April and October.

Fecal coliform counts in the irrigated watersheds most closely resembled the overall seasonal fecal coliform pattern with significantly higher values in June, July, and August than all other months of monitoring (Figure 4.17d). Higher counts were also observed in May and September. *Escherichia coli* counts in the irrigated watersheds were significantly higher in June than all other months, with the second highest counts measured in July and August. Bacteria counts were lowest in March, followed by April and October (Figure 4.18d).

Fecal coliform and *E. coli* counts followed similar patterns to one another in Willow Creek, the only watershed located in the Continental Divide. Significantly higher counts were measured in June than in all other months (data not shown). Higher counts were also observed in July, August, and September, whereas the lowest counts were measured in the spring (March, April, and May).

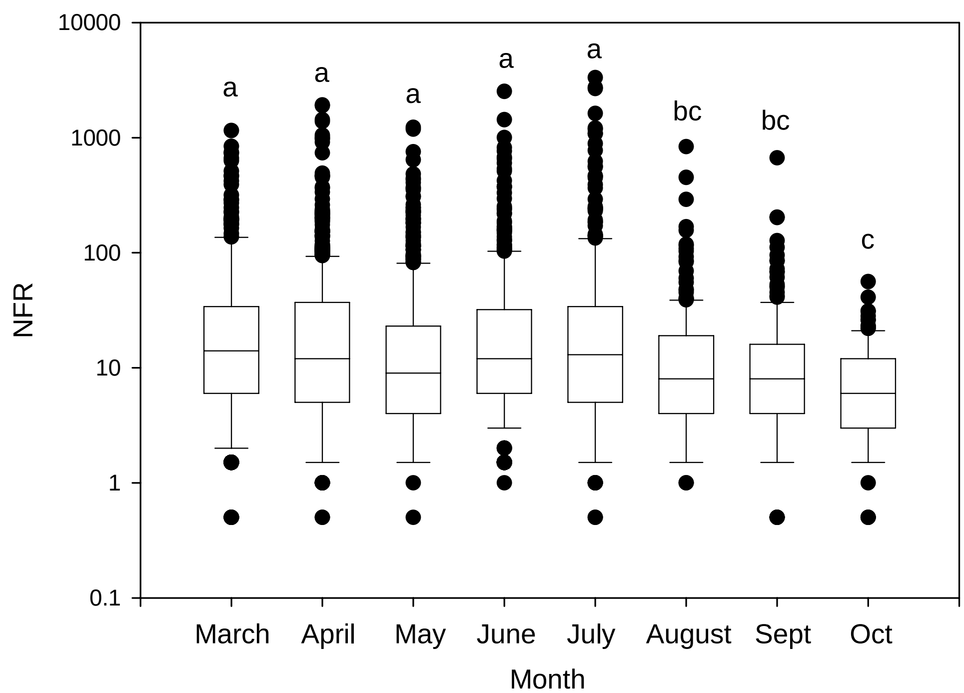

Figure 4.16. NFR box plot by month using all streams from 1999 through 2006. Box plots with different letters indicate a significant difference at the p<0.01 level (Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance test.

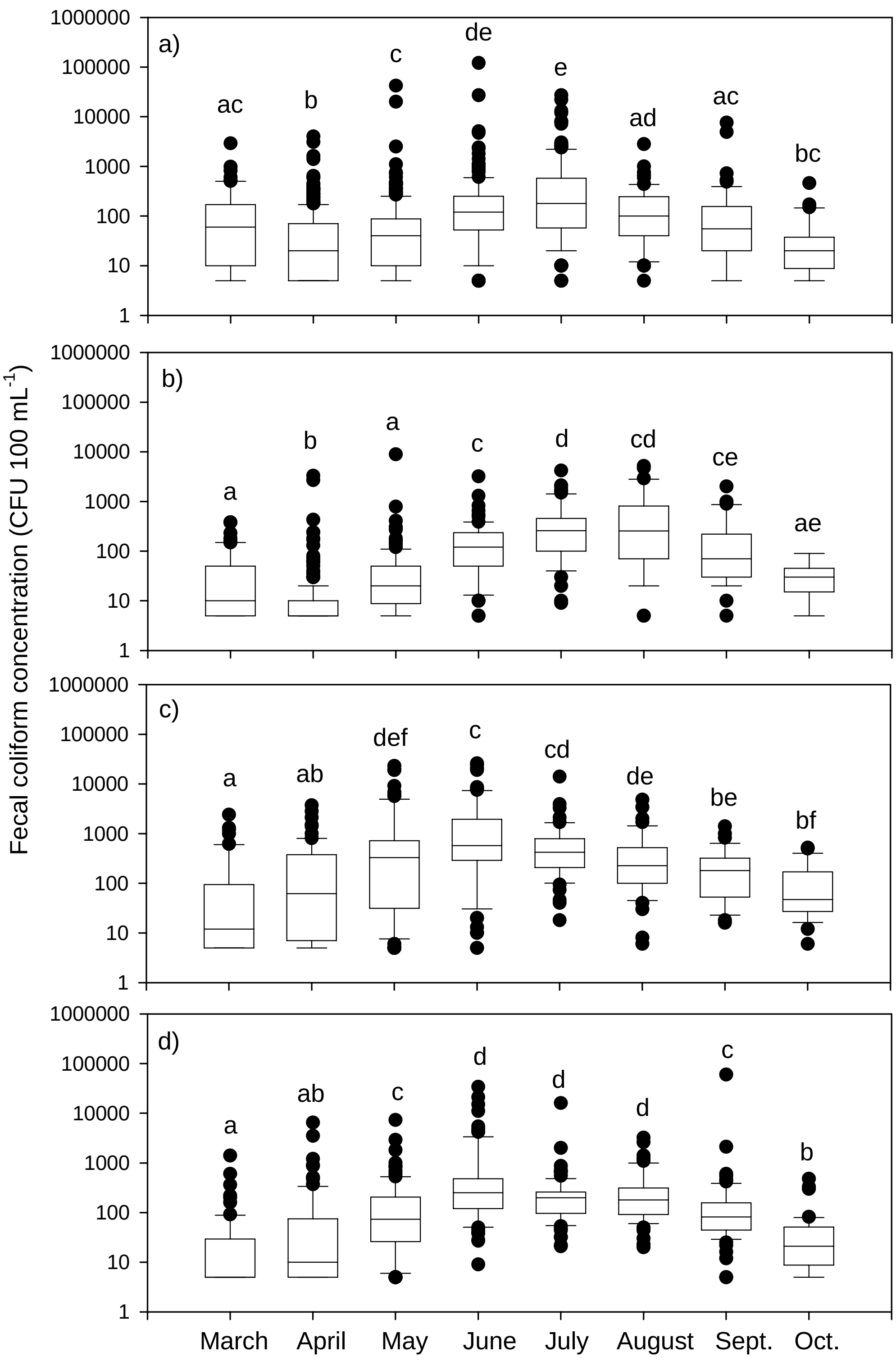

Figure 4.17. Fecal coliform concentrations by month (March through October) for Boreal (a), Parkland (b), Grassland (c), and Irrigated grassland (d) ecoregion areas. Box plots with different letters indicate a significant difference at the p<0.01 level (Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance test.

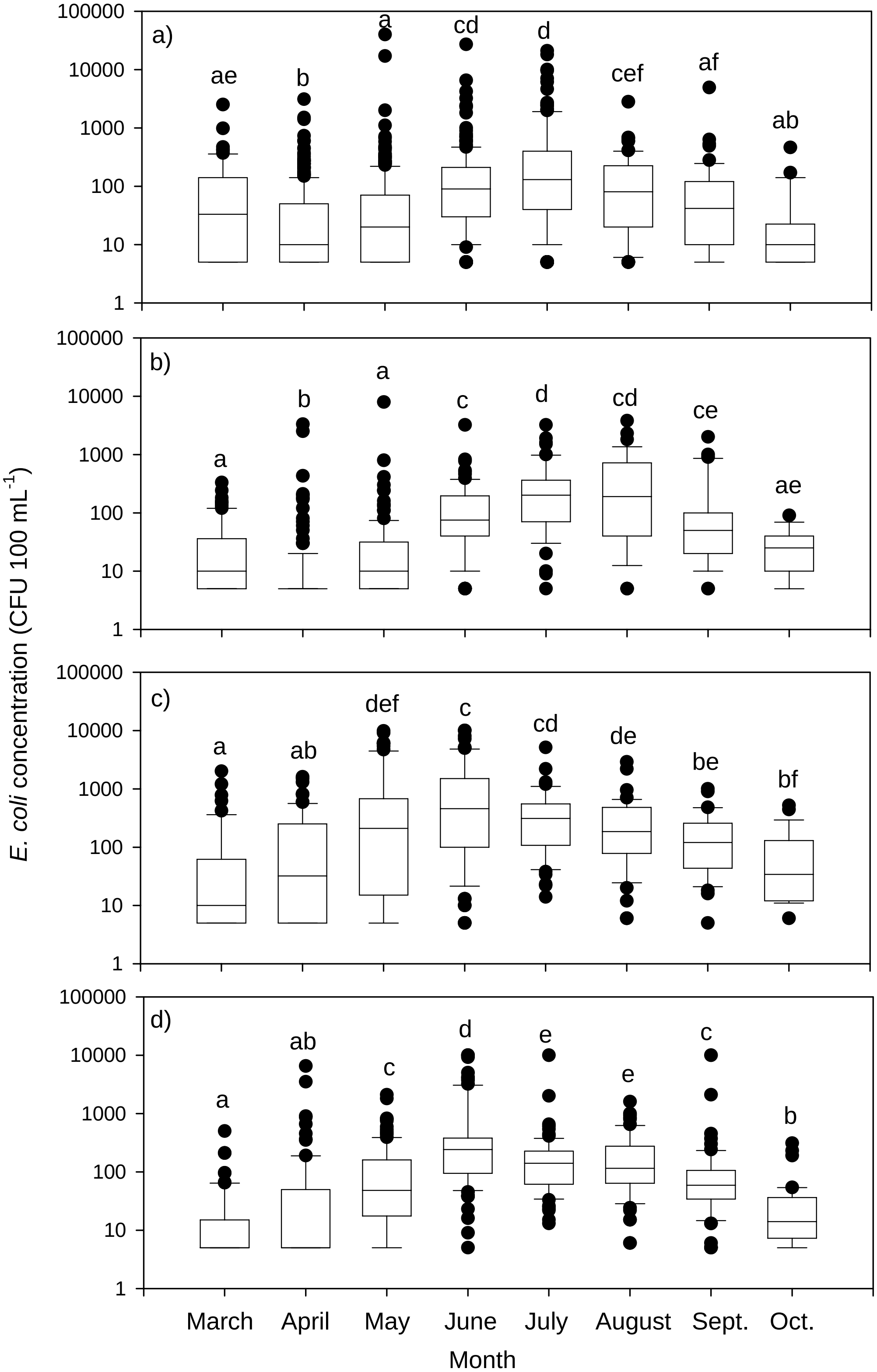

Figure 4.18. *E. coli* concentrations by month (March through October) for Boreal (a), Parkland (b),) Grassland (c), and Irrigated grassland (d) ecoregion areas. Box plots with different letters indicate a significant difference at the p<0.01 level (Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance test.

Although seasonal patterns were observed in the different ecoregion areas, not all watersheds within an ecoregion area necessarily showed seasonal trends in fecal bacteria counts. Unlike the rest of the watersheds located in the Boreal ecoregion area, Grande Prairie Creek, Kleskun Drain, and Hines Creek did not show statistical differences in bacteria counts among months (Table 4.11). Note that these three watersheds are located in the Peace Lowlands and Clearhills Uplands Ecoregions, whereas the rest of the watersheds in the Boreal ecoregion area are located further south in the Boreal Transition and Western Alberta Uplands Ecoregions. Seasonality was observed for fecal coliforms in Tomahawk Creek but not for *E. coli*. The differences in seasonal patterns within the Boreal ecoregion area may be explained by differences in climate, runoff patterns, and agricultural management practices among individual Ecoregions within the Boreal ecoregion area.

Table 4.11. Statistical output from Kruskal-Wallis tests for seasonal differences using all ambient fecal bacteria concentrations (1999 to 2006) in 23 AESA watersheds. Includes data from March to October (1999 through 2006). Degrees of freedom = 7 for all streams except Stretton (df=4).

	E.coli		Fecal coliform		NFR	
	Kruskal-Wallis Test Statistic	Probability	Kruskal-Wallis Test Statistic	Probability	Kruskal-Wallis Test Statistic	Probability
Boreal Forest						
Blindman River	52.5	**<0.0001**	58.7	**<0.0001**	32.2	**<0.0001**
Grande Prairie Creek	9.2	0.238	10.6	0.155	15.1	*0.034*
Hines Creek	11.1	0.134	11.1	0.133	21.2	**0.003**
Kleskun Drain	9.3	0.235	10.1	0.183	16.8	*0.018*
Paddle River	25.1	**<0.0001**	30.5	**<0.0001**	14.7	*0.04*
Rose Creek	77.7	**<0.0001**	84.6	**<0.0001**	23.0	**0.002**
Strawberry Creek	44.4	**<0.0001**	46.2	**<0.0001**	38.6	**<0.0001**
Tomahawk Creek	13.7	0.057	21.2	**0.003**	15.0	*0.036*
Wabash Creek	26.2	**<0.0001**	26.4	**<0.0001**	33.8	**<0.0001**
Parkland						
Buffalo Creek	73.3	**<0.0001**	72.4	**<0.0001**	8.9	0.261
Haynes Creek M6	31.8	**<0.0001**	28.0	**<0.0001**	15.7	*0.028*
Ray Creek	80.1	**<0.0001**	88.8	**<0.0001**	32.6	**<0.0001**
Renwick Creek	59.5	**<0.0001**	56.0	**<0.0001**	29.8	**<0.0001**
Stretton Creek	15.6	**0.004**	19.6	**<0.0001**	11.3	*0.024*
Threehills Creek	42.2	**<0.0001**	50.5	**<0.0001**	15.9	*0.027*
Northern Continental Divide						
Willow Creek	109.0	**<0.0001**	115.5	**<0.0001**	40.6	**<0.0001**
Grassland						
Meadow Creek	45.5	**<0.0001**	41.9	**<0.0001**	36.8	**<0.0001**
Prairie Blood Coulee	61.4	**<0.0001**	61.7	**<0.0001**	9.7	0.209
Trout Creek	90.4	**<0.0001**	101.7	**<0.0001**	30.2	**<0.0001**
Grassland (under irrigation)						
Battersea Drain	71.7	**<0.0001**	75.8	**<0.0001**	23.8	**0.001**
Crowfoot Creek	56.6	**<0.0001**	51.9	**<0.0001**	16.4	*0.021*
Drain S6	73.2	**<0.0001**	73.6	**<0.0001**	28.9	**<0.0001**
New West Coulee	61.4	**<0.0001**	62.7	**<0.0001**	47.5	**<0.0001**

Bold if significant at $p<0.005$
Italics if significant at $0.005<p<0.05$

Annual Export Coefficients

Actual annual export coefficients and loading values from 1999 to 2006 for the individual AESA streams are presented in Appendix 12 (Tables A12.1a and b).

By stream. Annual export coefficients for fecal bacteria (fecal coliform and *E. coli*) were calculated for all AESA watersheds except the four irrigated watersheds (19 of the 23 watersheds). An export coefficient is the estimate of the total amount of pollutant that passes a specific location during a specified interval of time (e.g., a year) divided by watershed area. While annual fecal bacteria exports for AESA streams provide an estimate of the total number of bacteria passing by the mouth of these small agricultural watersheds each year, the absolute number of bacteria that a downstream waterbody would receive is likely lower. A longitudinal study in a Minnesota watershed showed that as stream order (size) increases, fecal coliform concentrations generally decrease (Minnesota River Basin Data Centre 2003). The reasons for declining bacteria numbers included die-off, deposition of sediment (with associated bacteria), and dilution with downstream water that may have lower fecal coliform concentrations (Minnesota River Basin Data Centre 2003). As such, the values presented here likely over estimate the actual number of viable fecal bacteria (and thus, pathogens) exported from each system. However, they do provide an indication of the relative impacts each watershed may have on a receiving waterbody.

The highest fecal coliform export coefficients were observed in Tomahawk Creek, Rose Creek, and Blindman River, with all three watersheds located in the Boreal ecoregion area (median annual exports: 1860 to 2000 CFU ha^{-1} yr^{-1}, Figure 4.19). For *E. coli*, the streams with the highest exports were Rose Creek, Tomahawk Creek, and Trout Creek (median annual exports: 1450 to 1490 CFU ha^{-1} yr^{-1}, Figure 4.20). Trout Creek is located in the Grassland ecoregion area.

Trout and Meadow Creeks ranked very differently for fecal coliform compared to nutrient export coefficients. These two moderate intensity Grassland watersheds were among the top five watersheds with the highest fecal coliform exports. In contrast, they fell among the bottom three ranked watersheds for the majority of nutrient exports (TP, TDP, TN, Organic N, and NH_3-N; Chapter 3: Results and Discussion – Export coefficients for P and N in dryland watersheds).

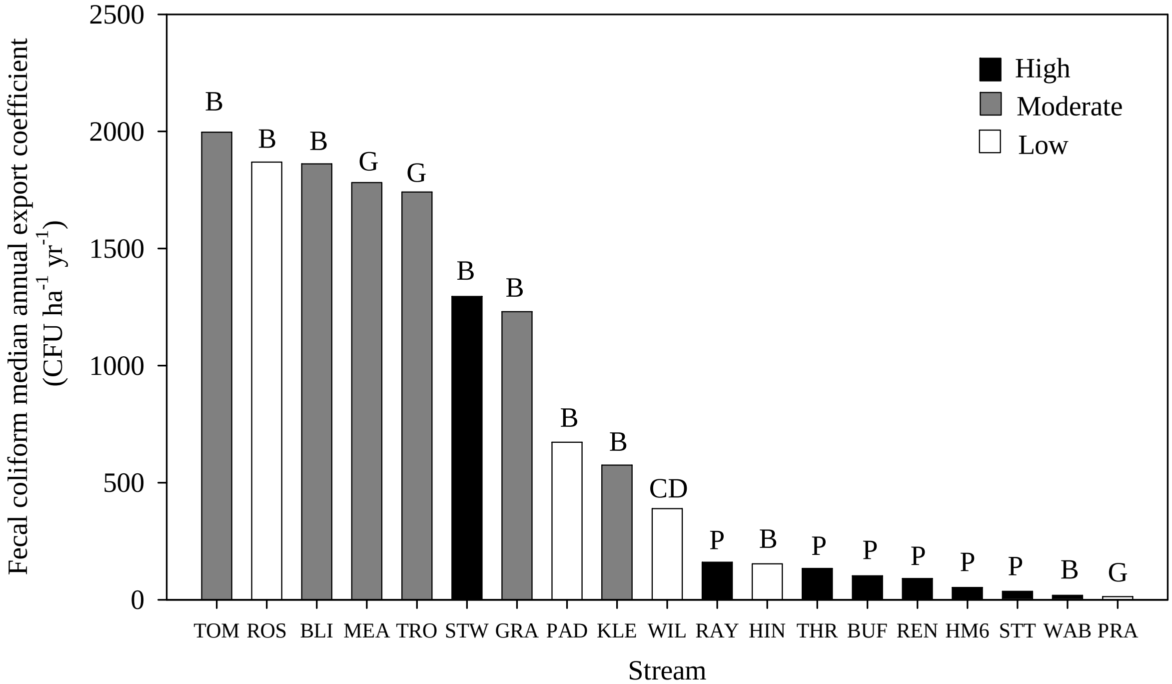

Figure 4.19. Median annual fecal coliform export coefficients for the 19 dryland AESA streams. Data are grouped by agricultural intensity and ecoregion where B= boreal, P= parkland, G=grassland, and CD= continental divide. See Table 4.2 (pg. 4-10) for full stream names.

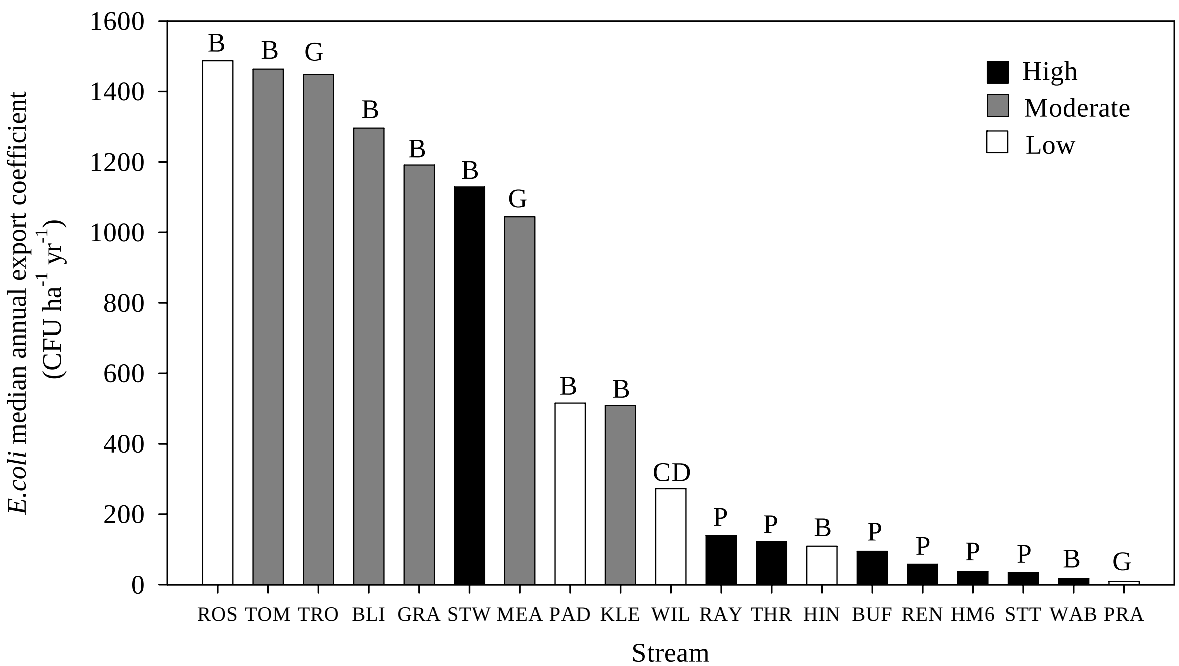

Figure 4.20. Median annual *E. coli* export coefficients for 19 dryland. Data are grouped by agricultural intensity and ecoregion where B= boreal, P= parkland, G=grassland, and CD= continental divide. See Table 4.2 (pg. 4-10) for full stream names.

By agricultural intensity. Export coefficients for *E. coli* were similar in low and high agricultural intensity watersheds and were significantly lower than *E. coli* exports in moderate agricultural intensity watersheds (Figure 4.21a, *E. coli*: $F(2, 144) = 16.600$, $p \leq 0.0001$). On the other hand, fecal coliform export coefficients were highest for the moderate agricultural intensity watershed category, followed by low and high agricultural intensity watershed categories (Figure 4.21b, fecal coliforms: $F(2, 144) = 18.687$, $p \leq 0.0001$).

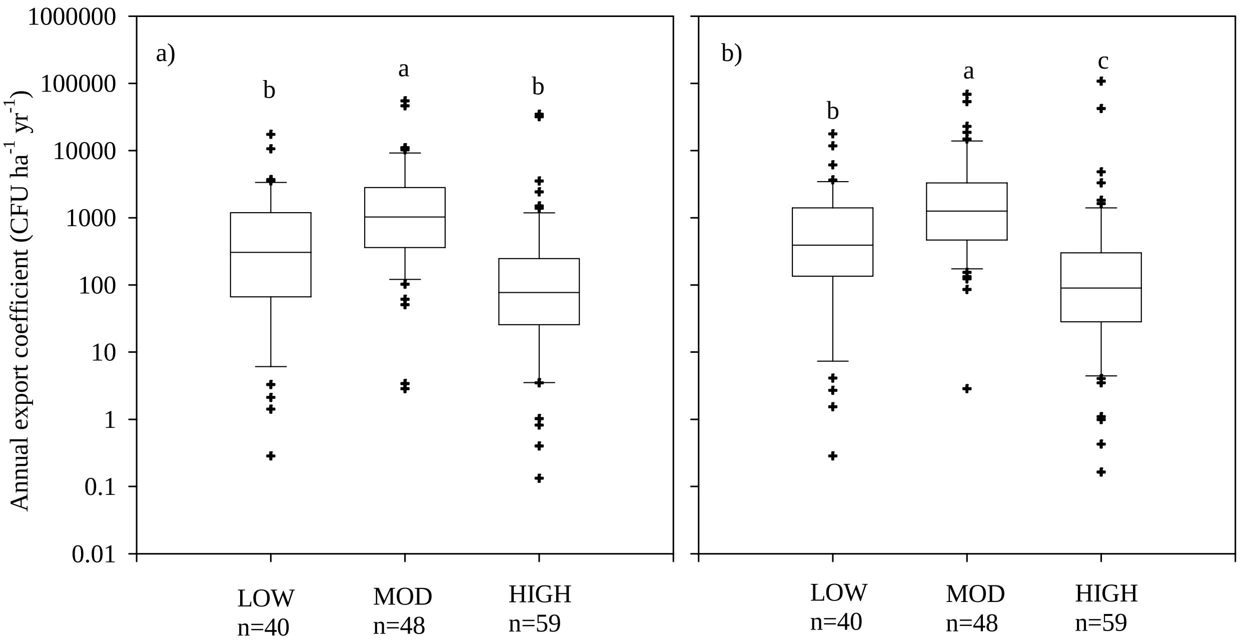

Figure 4.21. Box plots of annual export coefficients for *E. coli* (a) and fecal coliform (b) (1999 to 2006) in the three dryland agricultural intensity categories. Box plots with the same letter are not significantly different from one another at $p \leq 0.05$ (One-Way ANOVA, Tukey post hoc).

Annual export coefficients for *E. coli* were similar among watersheds within the low agricultural intensity category with the exception of Rose Creek, which had exports that were significantly higher than Hines Creek and Prairie Blood Coulee (Tukey post hoc, $p<0.01$, Figure 4.22a). Fecal coliform exports from Rose Creek were significantly higher than Willow Creek, Hines Creek, and Prairie Blood Coulee (Tukey post hoc, $p \leq 0.05$, Figure 4.22d).

Within the moderate agricultural intensity category, export coefficients generally did not vary significantly among streams (fecal coliform: $F(5,42)=2.236$, $p=0.068$, *E. coli*: $F(5,42)=2.188$, $p=0.074$, Figure 4.22b,e). One exception was that fecal coliform exports in Blindman River were significantly higher than those in Kleskun Drain (Tukey post hoc, fecal coliform: $p=0.047$, *E. coli*: $p=0.040$).

Export coefficients for fecal coliforms and *E. coli* differed among the high agricultural intensity watersheds (fecal coliforms: $F(7,51)=4.673$, $p \leq 0.0001$, *E. coli*: $F(7,51)=4.712$, $p \leq 0.0001$ Figure 4.22c,f). Fecal bacteria exports were greatest in Strawberry Creek but were only significantly higher than Haynes Creek, Renwick Creek, and Wabash Creek (Tukey post hoc, fecal coliform: p=0.047, *E. coli*: p=0.040). As noted previously, Strawberry Creek is the only high agricultural intensity watershed ranked as having a high runoff potential and also has a much larger effective drainage basin size (Chapter 2: Table 2.1).

The significantly higher export of fecal coliforms and *E. coli* in the moderate agricultural intensity watersheds was driven by Trout and Meadow Creeks. These patterns again emphasize the difference between bacteria and nutrient patterns as nutrient export coefficients (TP, TN, and Org-N) did not differ significantly among watershed categories with the exception of particulate phosphorus which was highest in low and moderate intensity watersheds (Chapter 3: Results and Discussion – Export coefficients for P and N in dryland watersheds).

By ecoregion area. Fecal coliform and *E. coli* export coefficients differed significantly among ecoregion areas (fecal coliforms: $F(2, 136)= 9.308$, $p \leq 0.0001$, *E. coli*: $F(2, 136)= 8.005$, p=0.001). Exports were significantly higher in streams in the Boreal and Grassland ecoregion areas than in Parkland watersheds (Tukey's post hoc, $p<0.05$). Visually, exports in the Continental Divide watershed were similar to those measured in the Boreal and Grassland ecoregion areas (Figure 4.23).

The ecoregion area rankings were different for fecal bacteria exports than for nutrient exports. Specifically, fecal bacteria exports from the Grassland ecoregion area were higher than would have been predicted based on nutrient exports. In contrast, total phosphorus exports were highest in the Boreal and Parkland ecoregion areas and lowest in Grassland watersheds (Chapter 3: Results and Discussion – Export coefficients for P and N in dryland watersheds). Total nitrogen exports were highest in the Boreal watersheds, moderate in the Parkland streams, and lowest in the Grassland watersheds. Higher bacteria exports from the Grassland streams were attributable to the high fecal bacteria concentrations in Trout and Meadow Creeks.

There were significant differences in annual fecal coliform and *E. coli* exports among streams in the Boreal ecoregion area (fecal coliform: $F(8,63)=7.029$, $p \leq 0.0001$, *E. coli*: fecal coliform: $F(8,63)=7.029$, $p \leq 0.0001$). Two watersheds (Wabash and Hines Creek) showed significantly lower export coefficients than the others. Specifically, exports from Wabash Creek were lower than Blindman River and Grande Prairie, Paddle, Rose, Strawberry, and Tomahawk Creeks (Tukey's post hoc, $p<0.05$). Exports from Hines Creek were significantly lower than those in Blindman River and Rose Creek for *E. coli* and Blindman River and Rose and Strawberry Creeks for fecal coliforms.

Within the Grassland ecoregion area, export coefficients from Meadow and Trout Creeks were significantly higher than Prairie Blood Coulee (fecal coliform: $F(2,21)=12.415$, $p \leq 0.0001$, *E. coli*: $(2,21)=10.438$, p=0.001).

There was no significant difference in export coefficients among watersheds in the Parkland ecoregion area (fecal coliform: $F(5,37)=2.220$, p=0.073, *E. coli*: $F(5,37)=2.217$, p=0.073). All watersheds in the Parkland ecoregion area are under high intensity dryland agriculture.

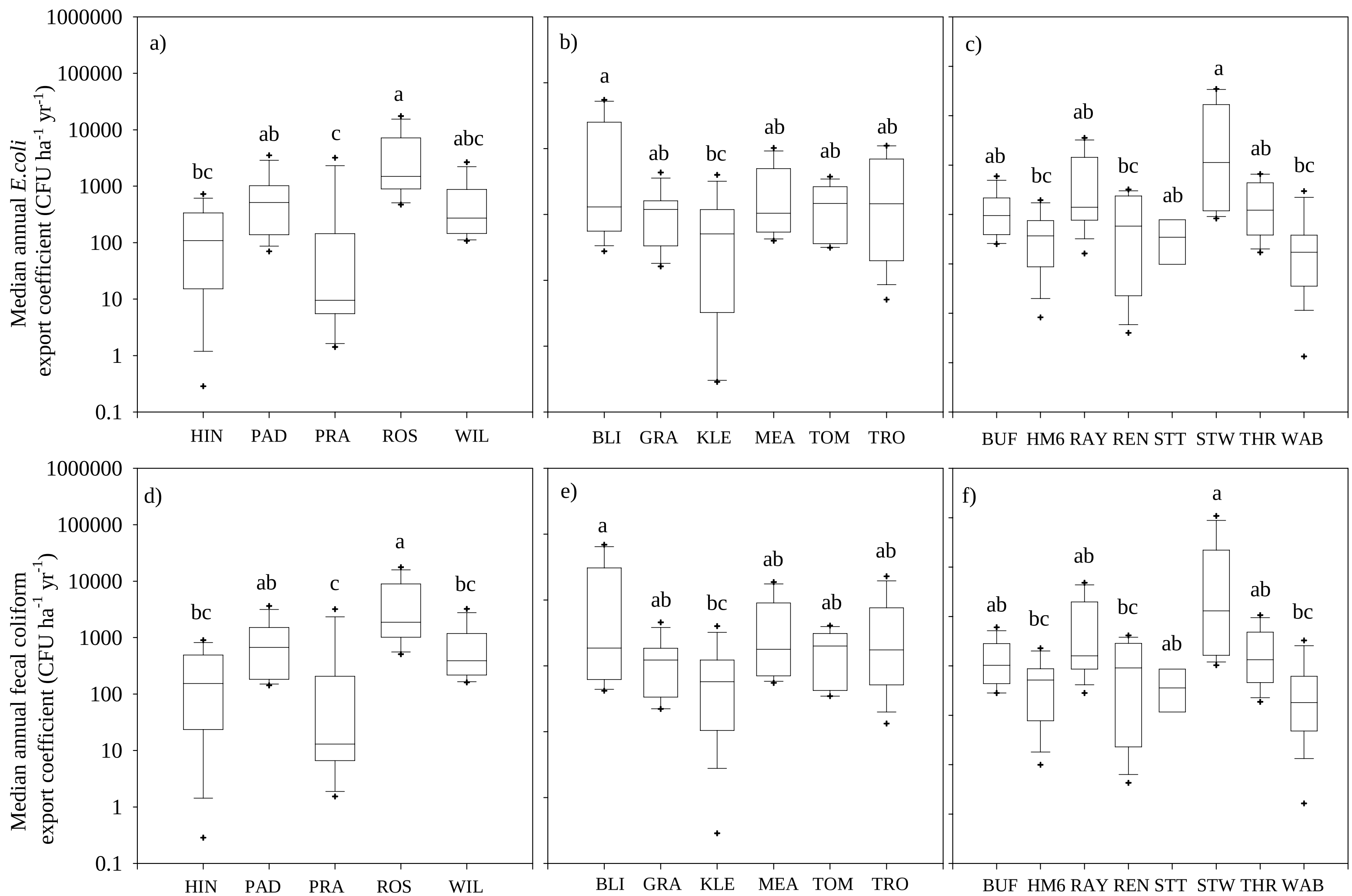

Figure 4.22. Boxplots of the median annual export coefficient for *E. coli* in watersheds with low (a), moderate (b), and high (c) intensity agriculture and fecal coliforms in watersheds with low (d), moderate (e), and high (f) intensity agriculture. In each plot, streams with the same letter are not significantly different (ANOVA and Tukey post hoc test, $p<0.05$). See Table 4.2 (pg. 4-10) for full stream names.

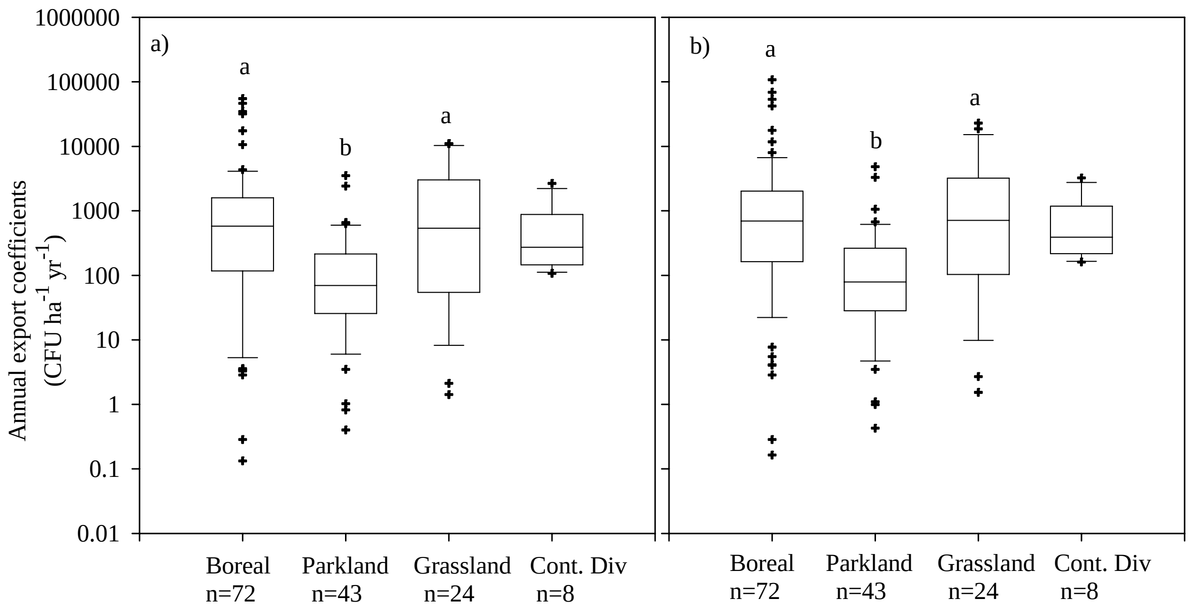

Figure 4.23. Boxplots of the annual export coefficients of *E. coli* (a) and fecal coliforms (b) for 23 AESA streams grouped by ecoregion area. Groups with the same letter are not significantly different (ANOVA and Tukey post hoc test, p<0.05).

Fecal Bacteria Loads

Annual loads of fecal bacteria (fecal coliform and *E. coli*) were calculated for 22 of the 23 AESA watersheds (all except Drain S-6). Annual load values (1999 to 2006) for the individual AESA streams are presented in Appendix 12 (Tables A12.1a and b). As noted for annual export coefficients, the annual bacteria load values presented here provide an estimate of the total number of bacteria passing by the mouth of these small agricultural watersheds in a year. However, the absolute number of bacteria received by a downstream waterbody would likely be lower due to die-off, deposition with sediment, and dilution with downstream water (Minnesota River Basin Data Centre 2003).

The highest median annual fecal coliform loads were observed in three watersheds located in the Boreal ecoregion area: Rose Creek (1.04×10^8 CFU year^{-1}), Strawberry Creek (7.63×10^7 CFU year^{-1}), and Blindman River (6.57×10^7 CFU year^{-1}) (Figure 4.24). In general, loads tended to be greatest from the dryland watersheds with the largest effective drainage areas. The two exceptions were Wabash and Hines Creeks which both have larger drainage areas (> 300 km^2) but relatively low fecal coliform loads.

All three irrigated watersheds (Battersea Drain, Crowfoot Creek, and New West Coulee) also had notably higher loads, with median annual fecal coliform loads ranging from 3.35×10^7 CFU year^{-1} (in Battersea Drain) to 6.36×10^7 CFU year^{-1} (in New West Coulee) (Figure 4.24). The volume of water travelling through irrigated return flow streams over the course of the open water season tends to be high relative to many dryland streams because of the constant supply.

With a larger number of confined feeding operations concentrated within the Lethbridge region in southern Alberta, there is a disproportionate amount of manure that is produced within these basins that is likely applied to local crops (due to the high cost of manure transportation).

Watersheds located in the Aspen Parkland Ecoregion, including Stretton Creek, Renwick Creek, and Haynes Creek M6, had the lowest fecal bacteria loads. Typically, watersheds with high agricultural intensity also had low loads, in part due to low flow volumes but also due to the type of agriculture (dryland cropping) practiced.

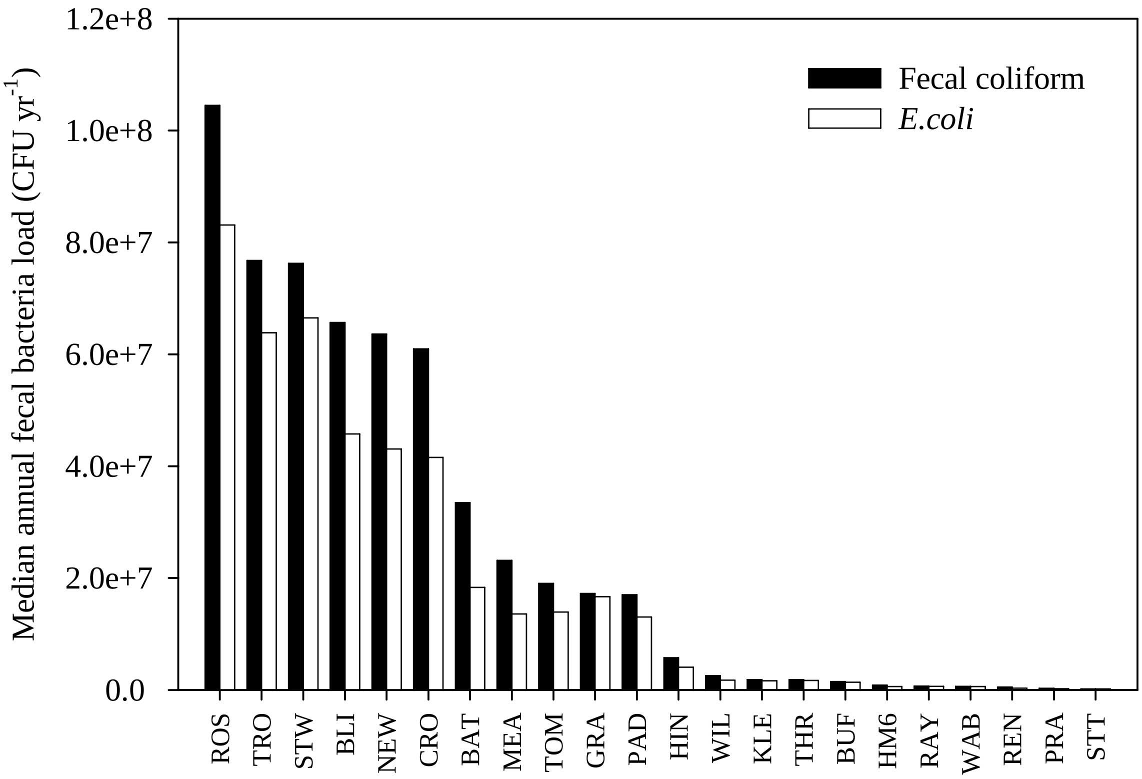

Figure 4.24. Median annual fecal coliform and *E. coli* loads for 22 of 23 AESA streams (all but Drain S6). See Table 4.2 (pg. 4-10) for full stream names.

SUMMARY AND CONCLUSIONS

Data collected during the AESA Stream Survey from 1999 through 2006 confirmed the presence of fecal bacteria in surface waters of agricultural watersheds across Alberta. Specific objectives outlined for this chapter are answered below.

Objective 1: Assess compliance with Canadian Water Quality Guidelines.

- Average annual percent compliance for the irrigation guideline (100 CFU·100 mL^{-1} fecal coliforms) ranged from 12% in Meadow Creek (moderate agricultural intensity) to 96% in Stretton Creek (high agricultural intensity).
    - Since the guideline is intended for circumstances where the water is potentially used for irrigation, the watersheds where the guideline has greatest importance are Battersea Drain, Crowfoot Creek, Drain S6, and New West Coulee, which had 62, 53, 62, and 50% compliance with the fecal coliform guideline for irrigation, respectively. It is important to clarify that the sampling location is at the mouth of these watersheds, and though irrigation agriculture is practiced within these basins, the source water used to irrigate crops is not the water sampled under the AESA program.
- Average annual percent compliance for the recreation guideline (200 CFU·100 mL^{-1} fecal coliforms) ranged from 22% in Meadow Creek (moderate) to 100% in Stretton Creek (high).
- Compliance tended to be highest in watersheds with low or high intensity agriculture.

Objective 2: Assess differences in fecal bacteria levels among watersheds with varying levels of agricultural intensity (AI) and in different ecoregion areas.

- Annual geometric means for *E. coli* and fecal coliforms were significantly lower in both high and low intensity streams than in moderate intensity and irrigated streams.
    - The absence of a stepwise increase from low to moderate to high intensity (as observed in dryland streams for total phosphorus and total nitrogen, Chapter 3) suggests that the sources and/or mechanisms for transport of these two types of agricultural pollutants (nutrients vs. fecal bacteria) may not be the same.
    - Based on agricultural intensity groupings, streams such as Rose, Meadow, and Strawberry had significantly higher annual geometric means, while Willow Creek, Kleskun Drain, and Stretton Creek had notably lower annual geometric means within each of the respective agricultural intensity classifications (low, moderate, and high agricultural intensity, respectively).
- Ambient fecal coliform and *E. coli* concentrations were highest in the Grassland ecoregion area, followed by the Irrigated Grassland ecoregion area, the Boreal ecoregion area, the Parkland ecoregion Area, and the Continental Divide.
    - Differences among ecoregions were not as strong with *E. coli* as with fecal coliforms.

Objective 3: Examine inter-annual patterns in fecal coliform and *E. coli*.

- Overall, there was no apparent provincial trend in the geometric means of all watersheds from 1999 through 2006.
- Meadow Creek, Ray Creek, and Blindman River demonstrated variability among sampling years. Possible contributing factors included changes in manure production and NFR levels.

Objective 4: Examine the relationship between manure production in a watershed and the number of fecal coliforms and *E. coli* detected in water.

- At a provincial scale, the manure production metric alone was not a good predictor of streams with the highest risk of fecal coliform contamination.
    - Fecal coliforms measured in high intensity streams in the Aspen Parkland Ecoregion were typically lower than would be expected based on the manure production values.
    - Highest bacteria concentrations were observed in the Fescue Grasslands Ecoregion, specifically in Trout and Meadow Creeks on the south-western edge of the agricultural white zone. Pasture land, cropping, and confined feeding operations are primary activities, and there is relatively low percent tree cover.

Objective 5: Investigate the relationship between bacteria concentrations and possible transport mechanisms (e.g., flow, sediment transport, and precipitation).

- A strong relationship was observed between the total suspended sediment (i.e., NFR) and median annual concentrations of fecal coliforms and *E. coli* (r_s = 0.775 and 0.782, respectively, $p<0.005$) for the 23 AESA streams.
    - These data suggest that, in general, agricultural streams in Alberta with higher suspended sediment are also more likely to have higher bacteria concentrations.
- The relationship with runoff depth was not as strong as with total suspended solids (r_s =0.250 and 0.203 for fecal coliforms and *E. coli*, respectively); however, the relationship was weaker as a result of patterns in Meadow Creek (high bacteria, low runoff depth) and Willow Creek (low bacteria, high runoff depth).
    - When these two streams were removed, the correlation coefficients more than doubled (r_s =0.613 and 0.532 ($p<0.05$) for fecal coliforms and *E. coli*, respectively).
- An inverse relationship was observed between nutrient concentrations and fecal coliforms and *E. coli* concentrations, the strongest being with total dissolved phosphorus (TDP) and total nitrogen (TN) (r_s=-0.442 to -0.476, $p<0.05$).
    - The inverse relationship was unexpected as it suggests that higher fecal bacteria concentrations cannot be anticipated in watersheds with high nutrient concentrations.

Objective 6: Examine seasonal patterns in fecal coliforms and *E. coli*.

- Overall, fecal coliform and *E. coli* counts were statistically highest in the summer months (June, July, and August) and lowest in the early spring (March and April).
    - Seasonal patterns for *E. coli* and fecal coliforms varied slightly by ecoregion area but generally followed the overall trend.
    - Seasonal patterns in fecal coliform and *E. coli* counts did not follow the same overall seasonal pattern observed for NFR, which was significantly higher in the early spring (March and April) and early summer (June and July) and lowest in October.
    - Higher fecal bacteria concentrations in the summer months may be a result of more frequent cattle access to surface water, longer bacteria survival in warmer water temperatures, and increased transport in surface runoff during summer storm events.
- Not all watersheds showed seasonal trends in fecal bacteria counts.
    - Seasonal patterns were not observed in watersheds in the northern Boreal ecoregion area (Grande Prairie Creek, Kleskun Drain, and Hines Creek).
    - Variability in seasonal patterns may have been a result of differences in climate and/or runoff patterns, land use, and agricultural management practices.

Chapter 5: Pesticides

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INTRODUCTION

Pesticides are used in agriculture to control pests and disease. Effective pest control allows for higher food production at lower cost and improves food quality and variety. Sales data for Alberta shows that the agriculture sector is the major user of pesticides. In 2003, a total of 9,264,488 kg of pesticide active ingredient was sold in or shipped to Alberta, and the majority of pesticide sales (96.2%) were in the agriculture sector (Byrtus 2007). The pesticides sold were predominantly herbicides (77.3%), while insecticides and fungicides comprised 4.7% and 3.4% of sales, respectively (Byrtus 2007).

Since pesticides are not naturally occurring in the environment, they present a unique challenge for agriculture (Anderson et al. 1997). Pesticide use can result in negative impacts to sensitive aquatic organisms, aquatic ecosystems, and human health because certain herbicides, insecticides, fungicides, and other pesticides persist in the environment longer than is required to kill target weeds and insects (Schulz 2004; Rice et al. 2007). Off-target movement of pesticides from soil to water results from direct or indirect transport mechanisms, including surface runoff and leaching, erosion of soil particles by wind or water, non-target drift from aerial or ground boom spraying, and deposition in precipitation. Other factors affecting the amount of pesticide transported include the slope of land and the interval between pesticide application and the occurrence of a runoff event. Generally, the shorter the interval between application and runoff, the greater the contaminant losses in surface runoff (Wauchope 1978). Potential pathways for non-target pesticide movement are illustrated in Figure 5.1; however, pesticide movement via atmospheric deposition (dry and wet) and long range transport and deposition are not depicted.

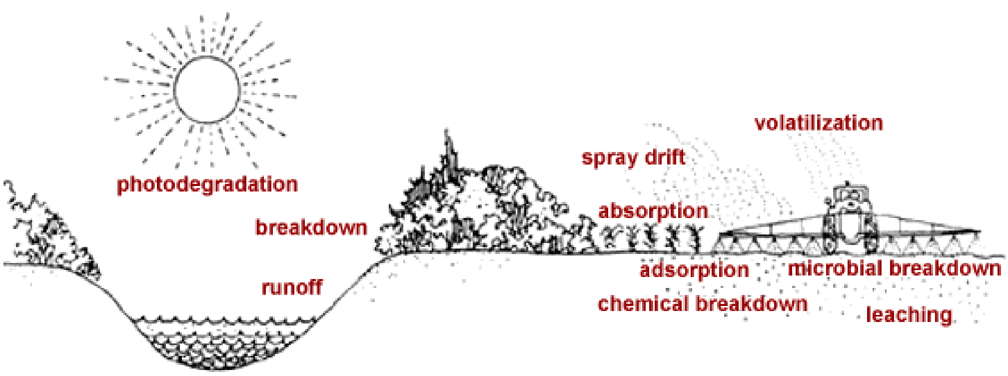


Figure 5.1. Pesticide pathways in the environment **(**Government of British Columbia).

Risks to water quality and human health are also dependent on pesticide characteristics that dictate their fate in the environment. Some pesticides are more likely to linger in the environment (e.g., organochlorines) while others are more likely to degrade (e.g., glyphosate). Characteristics that affect presence/absence in the environment include water solubility, soil adsorption, and persistence in the environment (i.e., half-life) (Government of British Columbia). Pesticides with high water solubility, low tendency to adsorb to soil particles, and long persistence or half-life have the highest potential to move into water.

Compounds with high solubility are more likely to be detected in water, even if they are not as commonly applied.

Assessments of streams in the United States, Canada, and Europe have shown that pesticides are frequently detected and often occur in mixtures (Anderson 2005; Chevre et al. 2006; Gilliom 2007). In agricultural applications, formulated products with multiple active ingredients or tank mixtures of two or more pesticides are commonly applied to crops, so the co-occurrence of multiple active ingredients in water is not surprising. Although pesticides tend to occur in low concentrations in the aquatic environment (Gilliom 2007), the occurrence of mixtures poses a toxicity risk to aquatic life. Studies have shown negative effects on invertebrate and amphibian populations (Bailey et al. 1997; Relyea 2008), as well as the nervous systems of fish (Tierney et al. 2008). The issue of pesticide mixtures in water is complex and is a challenge for regulators and ecotoxicologists alike (Lydy et al. 2004; Belden et al. 2007).

Spatial and Temporal Patterns in Pesticide Usage in Alberta

In Alberta, Census of Agriculture data are used to show regional patterns of pesticide use. The dollars spent on chemical expenses per acre provides a measure of the intensity of pesticide use and is used to define the chemical/pesticide component of the AESA agricultural intensity metric. The highest concentration of pesticide use occurs through central Alberta, as well as in the Lethbridge, Grande Prairie/Peace River, and Vegreville regions (Figure 5.2). From an ecoregional perspective, the high agricultural intensity watersheds are concentrated in the Aspen Parkland Ecoregion and irrigated portions of the Moist Mixed and Mixed Grassland Ecoregions. Of the 23 AESA watersheds, the highest chemical expenses coincide with watersheds ranked as having high agricultural intensity. These include Haynes Creek, Renwick Creek, and Battersea Drain (Figure 5.3). Watersheds with high chemical expenses also tend to have high fertilizer expenses and areas of crop land (Figure 5.3).

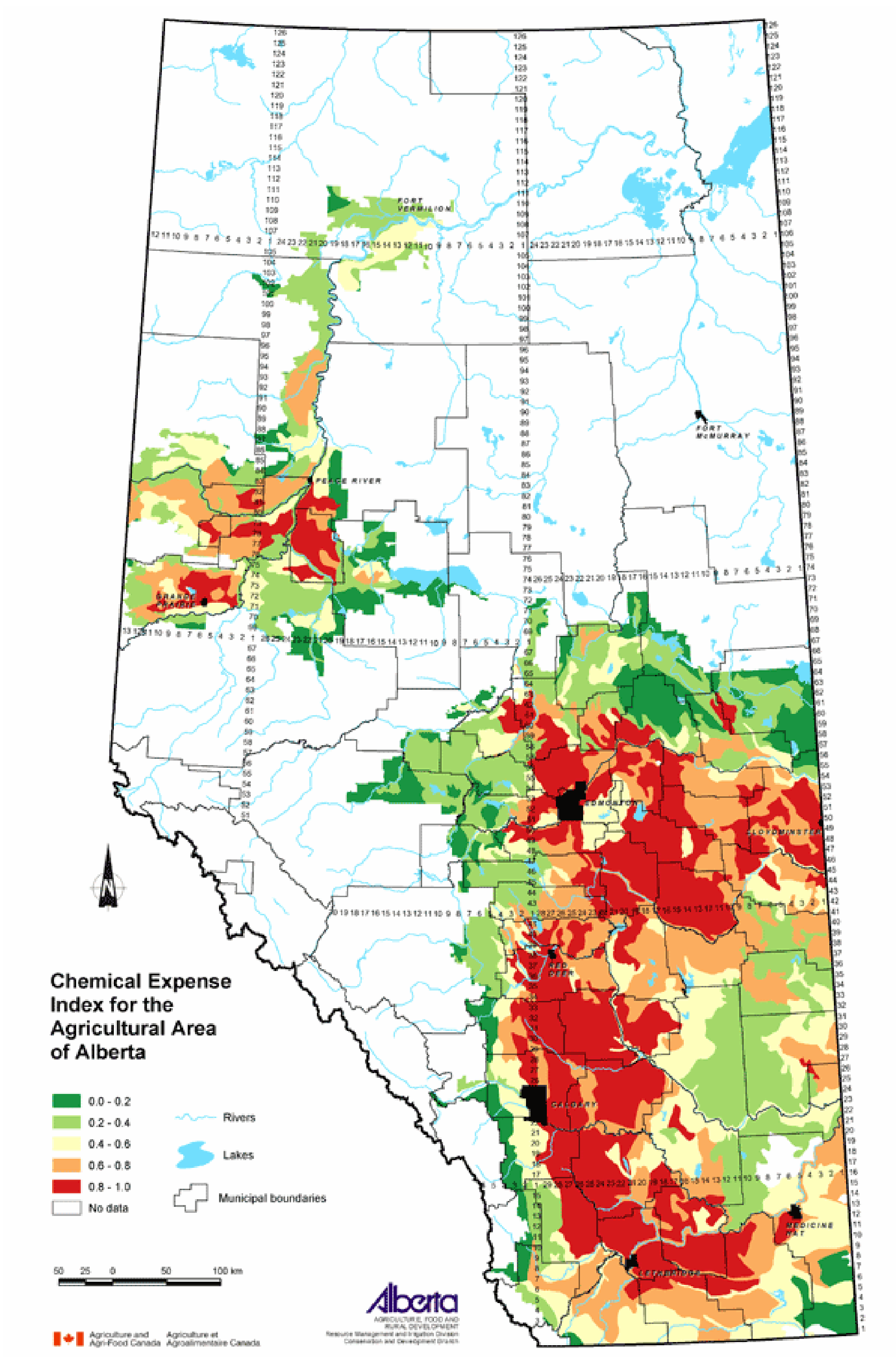


Figure 5.2. Map of the Chemical Expenses Index for the Agricultural Areas of Alberta in 2001.

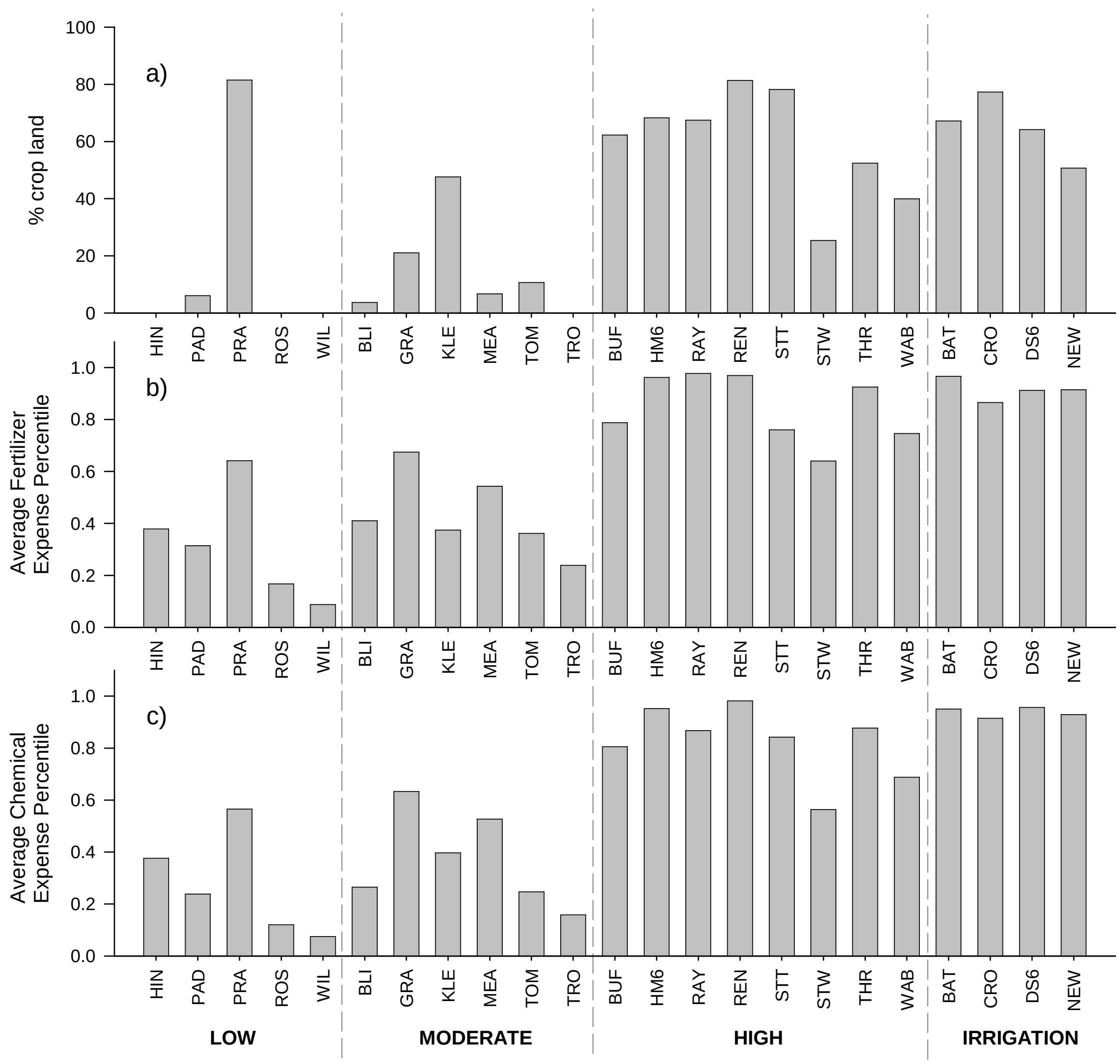


Figure 5.3. Land cover distribution (% crop land) and average fertilizer and chemical expenses percentiles for each of the AESA watersheds. Land cover information is based on 1991 data, and chemical and fertilizer expenses are based on the average of 1996, 2001, and 2006 Census of Agriculture data.

Census of Agriculture data are complemented by pesticide sales records that provide specific information on which pesticides are being sold, where, and in what quantities. Pesticide sales reviews are conducted every five years. The most recent review reported that the top 10 active ingredients in herbicide sold in the agricultural market are glyphosate, MCPA, 2,4-D, bromoxynil, triallate, ethalfluarlin, tralkoxydim, imazamethabenz, dicamba, and glufosinate (Byrtus 2007). The top two active ingredients in insecticides sold in the agricultural market are chlorpyrifos and carbaryl.

There have been some changes in sales of active ingredients between 1998 and 2003 (Table 5.1). The most notable is the jump in carbaryl sales (nearly 8000% increase) which coincided with issues with grasshopper populations. Substantial (>20%) increases in sales were observed from 1998 to 2003 for glyphosate, MCPA, bromoxynil, and glufosinate, and substantial decreases in sales were observed for triallate, ethalfluralin, and imazamethabenz-methyl. Pesticide active ingredients are anticipated to change over time as new products come on the market and specific pest outbreaks occur.

Table 5.1. Top 12 Agricultural active ingredient sales for 2003 and 1998 (Byrtus 2007)

Active Ingredient	Usage	2003 Sales (kg)	1998 Sales (kg)	% Change (1998 to 2003)
Glyphosate	Herbicide	3 333 994.5	2 627 599.3	**+26.9%**
MCPA*	Herbicide	1 096 848.9	884 937.5	**+23.9%**
2,4-D*	Herbicide	685 294.5	674 902.6	+1.5%
Bromoxynil*	Herbicide	354 906.6	268 105.3	**+32.4%**
Triallate*	Herbicide	197 221.4	693 269.3	**-71.6%**
Chlorpyrifos*	Insecticide	197 004.7	215 779.6	-8.7%
Ethalfluralin*	Herbicide	168 135.0	452 294.4	**-62.8%**
Tralkoxydim	Herbicide	141 226.1	126 323.5	+11.8%
Imazamethabenz*	Herbicide	138 551.4	173 679.2	**-20.2%**
Dicamba*	Herbicide	108 637.8	118 739.8	-8.5%
Glufosinate	Herbicide	106 689.6	63 400.8	**+68.3%**
Carbaryl	Insecticide	100 955.7	1 259.2	**+7917.4%**

Asterisks* denote compounds routinely monitored in the AESA WQ Program.
Bold values denote greater than 20% change in sales.

Provincially, agricultural pesticide use intensity has not changed much in Alberta in the 15-year period from 1988 to 2003. Overall, pesticide sales have increased slightly in this time; however, the pesticide use per unit area tends to remain the same (~ 0.8 kg ha^{-1}) (Byrtus 2007).

Objectives

As introduced in Chapter 1, the AESA Stream Survey was initiated in 1997 to learn more about how stream water quality is impacted by low, moderate, and high intensity agriculture in Alberta and to track changes in water quality as the industry grows and agricultural management practices change.

The specific objectives of this chapter were to:

i. Assess pesticide occurrence and concentration in Alberta's agricultural streams.
ii. Evaluate differences in pesticide occurrence and concentration among watersheds with low, moderate, and high intensity (dryland and irrigated) agriculture.
iii. Examine the relationship between measures of the intensity of pesticide use and presence of pesticide residues in surface water.
iv. Explore spatial and temporal trends in pesticide occurrence and concentration to assess risk to the environment and food safety.
v. Determine compliance of observed concentrations with Canadian Water Quality Guidelines and risk of cumulative effects using the Alberta Pesticide Toxicity Index.

MATERIALS AND METHODS

Field Methods

Sampling Protocol. Grab samples were collected from AESA streams during the open water season, from March through October. Pre-cleaned 1-L glass amber bottles were filled mid-stream until there was no headspace remaining. Samples were transported in coolers with ice to the laboratory within 24 hours of collection.

Sample collection followed a flow-biased regime where pesticide samples were collected more often during runoff or peak flow events (i.e., once per week during peak runoff, then once biweekly to once monthly as stream flow decreased). Due to cost, pesticide samples were collected less frequently than nutrient and fecal bacteria samples, approximately one pesticide sample for every two nutrient and bacteria samples.

Some pesticide monitoring was conducted in the AESA streams in 1997 and 1998. Forty-four samples were collected from 14 streams in 1997, and 117 samples were collected from 11 streams in 1998. All 23 streams were monitored from 1999 to 2006.

From 1999 to 2006, between 6 and 11 samples were generally collected annually per stream; however, sampling frequency varied depending on the amount of annual flow and the ecoregion where the watershed is located (Chapter 2: Precipitation and Hydrology section). From 1999 to 2006, an average of nine samples was collected per stream per year. The fewest number of samples collected in a single year was two (Crowfoot in 1999) and the most was 19 samples (Blindman and Paddle in 2005). In four of eight years, insufficient flow prevented sample collection in two streams (Stretton (2001 to 2003) and Haynes M6 (2004)). Refer to Appendix 17 (Table A17.2) for a summary of sampling frequency by year and stream.

QA/QC. Data quality was evaluated by field blanks, trip blanks, splits, and spikes. Refer to Anderson (2005) for more details on the QA/QC program.

The QA/QC data was not analyzed due to time constraints; however, it was anticipated that findings would be similar to earlier reports for Alberta data using the same field methods and laboratory. Anderson (2005) suggested that 'reported detection frequencies and concentrations are biased low' (i.e., concentrations and detections are higher in the environment).

Analytical Suite

The composition of the pesticide analytical suite was determined based on pesticide sales data (Byrtus 2000). The goal was to include the pesticide active ingredients that were most likely to be detected in the environment based on use and chemical properties; however, the suite was also dictated by analytical and funding constraints.

From 1999 to 2006, 40 active ingredients, breakdown products, and isomers (25 herbicides, 14 insecticides, and one fungicide) were routinely analyzed in the 23 AESA streams. In 2002, six compounds were added to the suite; in 2004, four compounds were added, bringing the totals to 46 and 50 compounds, respectively. In 2005, one additional compound (methomyl) was added. In 2006, funding from Alberta Environment enabled an upgrade to the extended pesticide scan, and 17 compounds were added to the suite bringing the total to 68 compounds (Table 5.2). A summary of pesticide mobility ratings for the compounds under study is found in Appendix 14.

Several pesticides, notably glyphosate and sulfonylurea compounds, were not included because of analytical costs and limitations. Glyphosate monitoring in a sub-set of AESA watersheds was conducted in 2005 and 2007, and findings are summarized in a separate report (Lorenz *in prep*).

Table 5.2. Pesticide analytical suite from 1999 to 2006.

Pesticides	
2,4-D	Fluazifop (added in 2006)
2,4-DB	Fluroxypy (added in 2006)
2,4-Dichlorophenol (added in 2002)	Gamma-Benzenehexachloride (Lindane)
4-Chloro-2-Methylphenol (added in 2002)	Hexaconazole (added in 2006)
Aldicarb (added in 2004)	Imazamethabenz-methyl
Aldrin (added in 2002)	Imazamox
Alpha-Benzenehaxachloride (Alpha-BHC)	Imazethapyr
Alpha-endosulfan	Iprodione (added in 2006)
Atrazine	Linuron (added in 2006)
Azinphosmethyl (Guthion)	Malathion
Bentazon (added in 2006)	MCPA
Bromacil	MCPB
Bromoxynil	MCPP
Carbathiin	Metalaxyl-m (added in 2006)
Chlorothalonil (added in 2006)	Methomyl (added in 2005)
Chlorpyrifos	Methoxychlor
Clodinafop Acid Metabolite (added in 2006)	Metolachlor (added in 2006)
Clodinafop-propargyl (added in 2006)	Metribuzin (added in 2006)
Clopyralid	Napropamide (added in 2004)
Cyanazine	Oxycarboxin (added in 2004)
Desethyl atrazine	Parathion (added in 2006)
Desisopropyl atrazine	Phorate
Diazinon	Picloram
Dicamba	Propiconazole (added in 2006)
Dichlorprop	Pyridaben
Diclofop-methyl	Quinclorac
Dieldrin (added in 2002)	Quizalofop (added in 2006)
Dimethoate	Simazine (added in 2002)
Disulfoton	Terbufos
Diuron	Thiamethoxam (added in 2006)
Ethalfluralin	Triallate
Ethion	Triclopyr (added in 2002)
Ethofumesate (added in 2006)	Trifluralin
Fenoxaprop-p-ethyl	Vinclozolin (added in 2004)

Laboratory analysis. One litre of unfiltered water was extracted and analyzed by mass spectrometry/gas chromatography - ion trap at the Alberta Research Council laboratory in Vegreville, AB. Additional details on methods are available in Anderson 2005.

Method detection limits (MDLs) ranged from L0.005 to L0.02 µg L^{-1} (parts per billion (ppb)) (Table 5.9). Results are expressed in µg active ingredient per litre. On occasions where a concentration below the MDL was reported, the reported value was converted to the MDL. This may result in discrepancies in detect versus non-detect data reported in the AESA annual technical reports where values below the MDL were reported (specifically malathion detections in 1999 and 2001). Dicamba detection limit changed in 2001 from L0.02 to L0.005 µg L^{-1}.

Data Analysis

For the purpose of comparing among agricultural intensity groupings, the data were limited to the eight-year monitoring period from 1999 to 2006.

For the purpose of examining year-to-year trends in pesticide detection, the data collected during 1998 were included for Battersea Drain, Blindman River, Crowfoot Creek, Haynes Creek, Ray Creek, Renwick Creek, Rose Creek, Stretton Creek, Strawberry Creek, Threehills Creek, and Tomahawk Creek since the sampling frequency was similar to the latter years. Samples from 1997 were not included in the discussion due to low sampling frequency.

Five metrics were used to examine spatial and temporal trends in the pesticide dataset (Table 5.3).

Table 5.3. Metrics used to evaluate pesticide data.

Metrics	Definition	Application and Considerations
Pesticide concentration	Actual concentration reported by the analytical laboratory for individual compounds	Summary statistics apply to measurable concentrations and should be evaluated in conjunction with detection frequencies to assess the extent of contamination
Pesticide detection frequency	Number of samples with at least one detection of an individual compound, divided by the number of samples analyzed	A measure of detection frequency for an individual pesticide
Total pesticide concentration	Sum of concentrations reported for all compounds detected in an individual sample	A measure of the potential cumulative impact
Total pesticide detection frequency	Number of samples with at least one pesticide detection per sample, divided by the number of samples analyzed	A measure of the presence/ absence of one or more pesticide residues.
Total number of detections per sample	Number of individual pesticides per sample	Evaluation of the presence of pesticide mixtures

Adapted from Anderson (2005)

Total concentration and total pesticide detection frequency were calculated using all available data each year (i.e., an increasing number of compounds over time). However, interannual comparisons only included the 40 compounds analyzed every year from 1998 to 2006.

Statistical comparisons among watershed agricultural intensity categories, years, and seasons were calculated using annual or monthly medians.

Censored data values. Statistics such as medians and percentiles for ‘measurable concentrations’ do not include censored data (i.e. ‘zero’ values). In the calculation of ‘total pesticide concentration’, censored data were replaced by ‘zero’.

Annual loads and export coefficients for pesticides were not calculated for this report. Refer to Anderson 2005 for mass transport values for AESA and other agricultural streams from 1997 to 2002.

Statistical Analysis. All statistical analyses were computed in SYSTAT 10 (SPSS Inc. 2000).

Prior to analysis, normality was first assessed using normal probability plots. Where the assumptions of the parametric test were met, one-way ANOVAs were used followed by the Tukey post-hoc comparison; otherwise the non-parametric Kruskal Wallis test was employed followed by the Mann Whitney U-test on all pairs of groups. All ANOVAs were completed using median annual data from 1999 to 2006 for each stream. Large proportions of censored data often required non-parametric analyses.

Figures of individual streams show summary statistics (median, mean, or maximum) for the entire data set and thus include a different number of samples for each stream. Generally medians are plotted in figures as they best represented the data overall.

Correlation Analyses. The relationship between the intensity of agriculture in a watershed (as % cropland and fertilizer and chemical expense percentiles) and pesticide occurrence was explored by Pearson Product-Moment Correlation for parametric data and Spearman Rank Correlation for non-parametric data.

Water Quality Guidelines. Water quality guidelines provide a consistent basis for assessing water quality conditions of Alberta’s streams and rivers. The province has derived or adopted surface water quality guidelines for the protection of three major water uses: freshwater aquatic life, agricultural water uses (irrigation and livestock watering), recreational use, and aesthetics. Pesticide concentrations in AESA streams were compared against guidelines in one or several of these categories. The most current provincial guidelines were used: Surface Water Quality Guidelines for Use in Alberta (AENV 1999). Several of Alberta’s guidelines were adopted from federal water quality guidelines, specifically Canadian Environmental Quality Guidelines (CEQGs) (CCME 1999a, 2002).

Drinking water guidelines are set by the federal government and applied to treated water used for human consumption. These guidelines are provided in this report for reference only.

Pesticide Toxicity Index. The Alberta Pesticide Toxicity Index (APTI) provides information on the relative toxicity of pesticide mixtures in surface waters and is used as a screening tool (Anderson 2008). The APTI evaluates observed pesticide concentrations in relation to the acute toxicity values (available from the USEPA ECOTOX database) based on two endpoints: i) EC_{50} (sublethal) algal and cladocerans ii) LC_{50} (mortality) invertebrates and fish. Data are also compared to a 'No Observable Effects Concentration" (NOEC) that is based on 1% of lowest LC_{50} or EC_{50}. The index output represents the samples and streams that are more or less likely to have toxic effects (i.e., risk) rather than a measure of actual toxicity (Anderson 2008).

The formula used to calculate the APTI is based on the work of Munn et al. (2006) (Anderson 2008) and is as follows:

$$PTI = \sum_{i=1}^{n} \frac{c_i}{EC_{xi}}$$

where c_i is the concentration of compound 'i'
n is the number of compounds detected
EC_{xi} is the effect endpoint associated with compound 'i' (e.g., LC_{50} or EC_{50})

There are three index categories:

- High Risk: Concentrations of one or more pesticide compounds exceed the most stringent EC_{50} and LC_{50} endpoint.
- Moderate Risk: Concentrations of one or more pesticide fall between the NOEC limit and the most stringent EC_{50} and LC_{50} endpoint.
- Low Risk: All pesticide concentrations fall below the NOEC (i.e. <1% of the lowest LC_{50} or EC_{50} values).

Daily data sets from each of the 23 AESA streams (1999 to 2006) were evaluated using the index to assess the pesticide toxicity risk in Alberta's agricultural streams and then compared among agricultural intensity categories.

The index has limitations, which are described by Anderson (2008). The limitation of applying the index to the AESA dataset is that the number of pesticides monitored increased from 40 to 68 compounds from 1999 to 2006, so the likelihood of a lower score is higher in the later years of monitoring when more compounds were analyzed.

Watershed abbreviation. For figure simplification, an abbreviation was often used for each watershed. Generally, the abbreviation made was the first three letters of the watershed although this is not always the case (Table 5.4).

Table 5.4. Watershed abbreviations used throughout the chapter.

Watershed	Abbreviation
Battersea Drain	BAT
Blindman River	BLI
Buffalo Creek	BUF
Crowfoot Creek	CRO
Drain S6	DS6
Grande Prairie Creek	GRA
Haynes Creek M6	HM6
Hines Creek	HIN
Kleskun Drain	KLE
Meadow Creek	MEA
New West Coulee	NEW
Paddle River	PAD
Prairie Blood Coulee	PRA
Ray Creek	RAY
Renwick Creek	REN
Rose Creek	ROS
Strawberry Creek	STW
Stretton Creek	STT
Threehills Creek	THR
Tomahawk Creek	TOM
Trout Creek	TRO
Wabash Creek	WAB
Willow Creek	WIL

RESULTS AND DISCUSSION

General Findings

At least one of the 40 routinely monitored pesticide compounds were detected in 990 of the 1627 samples collected from 1999 to 2006, an overall frequency of 61%. Total detection frequencies increased slightly when the additional 28 compounds (analyzed from 2002 onwards) were included in calculations. For the suite of 68 pesticides, one or more compounds were detected in 1041 of the 1627 samples or 64% (Table 5.5). The compounds that were added to the suite and accounted for the additional detections are listed in Table 5.6.

Table 5.5. Summary statistics for pesticide detection frequency for all samples collected from 1999 to 2006.

	Analytical Suite	
	40 compounds	68 compounds
Total number of samples	1627	1627
Number of samples with detections	990	1041
Detection Frequency (%)	60.8	64.0

Table 5.6. Number of compounds monitored and detected in each year of sampling.			
Monitoring Period	# Compounds Analyzed	# Compounds Detected	Newly added pesticides that were detected
1999 to 2001	40	24	
2002 to 2003	46	28	Simazine (H) Triclopyr (H) 2,4-Dichlorophenol (H-DP) 4-Chloro-2-Methylphenol (H-DP)
2004	50	30	Oxycarboxin (F) Vinclozolin (F)
2005	51	30	-
2006	68	37	Bentazon (H) Ethofumesate (H) Fluroxypyr (H) Metribuzin (H) Clodinafop Acid Metabolite (H-DP) Iprodione (F) Metalaxyl-M (F)

H= herbicide, H-DP= herbicide degradation product, F=fungicide

Regardless of how many compounds were analyzed (40 or 68), the median number of compounds detected per sample was two (Table 5.7). Summary statistics show that 75% of samples had ≤ 4 pesticide compounds. The maximum number of compounds detected per sample was 10 for the 40 compound suite and 12 for the 68 compound suite (Table 5.7).

Table 5.7. Summary statistics for the number of individual pesticides per sample for all samples collected from 1999 to 2006.

	Analytical Suite	
	40 compounds	68 compounds
n	990	1041
Minimum	1	1
25^{th} percentile	1	1
50^{th} percentile	2	2
75^{th} percentile	4	4
Maximum	10	12

n= number of samples with detections

The median total concentration of measurable levels of pesticide was 0.089 $\mu g\ L^{-1}$ and 0.098 $\mu g\ L^{-1}$ for the 40 and 68 compound suites, respectively (Table 5.8). The maximum concentration observed was 100 fold higher at 13.8 $\mu g\ L^{-1}$.

Table 5.8. Summary statistics for the total pesticide concentration for all samples collected from 1999 to 2006.

	Analytical Suite	
	40 compounds	68 compounds
n	990	1041
Minimum (μg L^{-1})	0.005	0.005
25th percentile (μg L^{-1})	0.027	0.030
50th percentile (μg L^{-1})	0.089	0.098
75th percentile (μg L^{-1})	0.309	0.319
Maximum (μg L^{-1})	13.8	13.8

n= number of samples with detections

Given the similarities in summary statistics for the 40 and 68 compound data sets, reported values in the remainder of the Chapter include all monitored (68) compounds (with the exception of interannual trends).

Pesticide Detection Frequency. Of the total 68 compounds analyzed, 37 pesticides or pesticide breakdown products were detected in measurable concentrations on at least one occasion. Of the 40 herbicides monitored, 29 were detected. Of the 20 insecticides monitored, four were detected, while four of the eight fungicides monitored were detected.

The eight most frequently detected compounds in agricultural streams were all herbicides. These eight herbicide active ingredients were each detected in ≥10% of analyzed samples: 2,4-D, MCPA, clopyralid, triclopyr, dicamba, picloram, imazamethabenz-methyl, and MCPP (Table 5.9). Detection frequencies were greatest for 2,4-D (49.5%) followed by MCPA (30.5%) and picloram (16.4%). The eight most ubiquitous compounds were all monitored from 1999 to 2006 with the exception of triclopyr, which was added to the analytical suite in 2002. Of the 21 remaining herbicides or breakdown products detected in AESA streams, 10 were detected in 1 to 9% of samples (triallate, bentazon, fluroxypyr, ethofumesate, bromoxynil, simazine, dichlorprop, 2,4-dichlorophenol, atrazine, and imazethapyr) and 11 were detected in less than 1% of samples (trifluralin, clodinafop acid metabolite, metribuzin, bromacil, diuron, imazamox, ethalfluralin, desisopropyl atrazine, 2,4-DB, 4-chloro2-methylphenol, and quinclorac).

The four insecticides detected were gammabenzenehexachloride (lindane), diazinon, chlorpyrifos, and alpha-benzenehexachloride (alpha-BHC) (Table 5.9). Detection frequencies for top detected insecticides were much lower than for herbicides (i.e. <1%). The most frequently detected insecticide was lindane, detected in 0.6% of samples (or 10 of 1627 samples), followed by diazinon (0.2%), chlorpyrifos (0.2%), and alpha-BHC (0.1%). All insecticides detected were monitored from 1999 to 2006.

The four fungicides detected in the AESA watersheds were iprodione (in 10 of 306 samples, or 3.3%), metalaxl-M (0.8%), vinclozolin (0.5%), and oxycarboxin (0.2%) (Table 5.9). These fungicide compounds were added to the analytical suite in 2002; thus, detection frequencies are based on four years of data.

Method detection limits have an influence on detection frequencies as a lower MDL (i.e.., 0.005 µg L^{-1}) will lead to higher detection frequencies than a 10-fold higher MDL (i.e., 0.05 µg L^{-1}). The approach in this report was to use the laboratory MDL to calculate detection frequencies; however, another approach would be to set a standard detection threshold for the calculation of detection frequency. For example, the United States Geological Survey applied a 0.1 µg L^{-1} threshold in their National Pesticide Assessment to account for variations in analytical sensitivity among different compounds (Gilliom 2007).

Pesticide Concentration. Pesticide concentration data need to be interpreted in the context of the detection frequency and compound toxicology to accurately assess risk. Median and mean concentrations are presented in Figure 5.4 and Table 5.9; a large difference in the two values suggests a relatively high peak concentration for the compound. Maximum concentrations observed are summarized in Table 5.9.

Of the 19 pesticide and breakdown products detected in >1% of samples, the herbicide imazamethabenz-methyl had the highest concentrations (median: 0.272 µg L^{-1} and mean: 0.440 µg L^{-1}) followed by the fungicide iprodione (median: 0.177 µg L^{-1} and mean: 0.183 µg L^{-1}), the herbicide simazine (median: 0.084 µg L^{-1} and mean: 0.364 µg L^{-1}), and the herbicide picloram (median: 0.065 µg L^{-1} and mean: 0.249 µg L^{-1}) (Figure 5.4a and b).

Several of the 18 pesticide and breakdown products detected in < 1% of samples were detected in concentrations higher than those detected in >1% of samples. However, occurrence was relatively rare. The pesticides to note include the fungicides metalaxyl-m and oxycarboxin and herbicides diuron, 2,4-DB, and 4-chloro-methylphenol (Figure 5.5a and b). All had median concentrations that exceeded 0.3 µg L^{-1}; however, none were detected on more than three occasions. Metalaxyl-m, oxycarboxin, and 4-chloro-methylphenol were each only detected in a single stream (Battersea Drain, Wabash Creek or New West Coulee, respectively). Diuron was detected in Rose and Wabash Creeks, and 2,4-DB was detected in Renwick Creek and New West Coulee.

Table 5.9. Summary of pesticide analyses from 1999 to 2006.

Pesticide	Year Added to Suite[Z]	MDL	Number of samples collected	Number of Detections	Detection Frequency (%)	Maximum Measurable Concentration ($ug\ L^{-1}$)	Median Measurable Concentration ($ug\ L^{-1}$)	Mean Measurable Concentration ($ug\ L^{-1}$)
HERBICIDES								
2,4-D	1997	L0.005	1627	805	49.5	8.534	0.031	0.139
2,4-DB	1997	L0.005	1627	2	0.1	0.665	0.335	0.335
2,4-DICHLOROPHENOL	2002	L0.01	994	15	1.5	0.469	0.045	0.070
4-CHLORO-2-METHYLPHENOL	2002	L0.01	994	1	0.1	2.876	2.876	2.876
ATRAZINE	1997	L0.005	1627	21	1.3	0.142	0.015	0.031
BENTAZON	2006	L0.005	306	17	5.6	0.599	0.035	0.119
BROMACIL	1997	L0.03	1627	4	0.2	0.297	0.158	0.169
BROMOXYNIL	1997	L0.005	1627	75	4.6	0.522	0.012	0.046
CLODINAFOPACIDMETABOLITE	2006	L0.02	304	1	0.3	0.024	0.024	0.024
CLODINAFOP-PROPARGYL	2006	L0.04	304	0				
CLOPYRALID (LONTREL)	1997	L0.02	1626	220	13.5	1.790	0.049	0.093
CYANAZINE	1997	L0.05	1627	0				
DESETHYLATRAZINE	1998	L0.05	1627	0				
DESISOPROPYLATRAZINE	1998	L0.05	1627	2	0.1	0.139	0.123	0.123
DICAMBA (BANVEL)	1997	L0.02/L0.005[Y]	1627	214	13.2	1.134	0.017	0.057
DICHLORPROP (2,4-DP)	1997	L0.005	1627	42	2.6	0.373	0.019	0.054
DICLOFOP-METHYL (HOEGRASS)	1997	L0.02	1627	0				
DIURON	1997	L0.2	1627	3	0.2	0.616	0.387	0.417
ETHALFLURALIN(EDGE)	1997	L0.005	1627	2	0.1	0.018	0.014	0.014
ETHOFUMESATE	2006	L0.005	306	15	4.9	0.399	0.047	0.088
FENOXAPROP-P-ETHYL	1998	L0.04	1627	0				
FLUAZIFOP	2006	L0.01	306	0				
FLUROXYPYR	2006	L0.01	306	16	5.2	0.409	0.033	0.066
IMAZAMETHABENZ-METHYL	1997	L0.05	1627	156	9.6	9.005	0.272	0.440
IMAZAMOX	1999	L0.02	1627	2	0.1	0.063	0.050	0.050
IMAZETHAPYR	1999	L0.02	1627	20	1.2	0.182	0.066	0.080
LINURON	2006	L0.02	306	0				
MCPA	1997	L0.005	1627	495	30.4	7.279	0.018	0.068
MCPB	1997	L0.02	1627	0				
MCPP (MECOPROP)	1997	L0.005	1627	160	9.8	2.068	0.019	0.052
METOLACHLOR	2006	L0.005	306	0				
METRIBUZIN	2006	L0.01	306	1	0.3	0.010	0.010	0.010
NAPROPAMIDE	2004	L0.02	641	0				
PICLORAM (TORDON)	1997	L0.005	1627	267	16.4	13.407	0.065	0.249
QUINCLORAC	1998	L0.005	1627	1	0.1	0.046	0.046	0.046
QUIZALOFOP	2006	L0.03	306	0				
SIMAZINE	2002	L0.01	1055	29	2.7	4.570	0.084	0.364
TRIALLATE (AVADEXBW)	1997	L0.005	1627	96	5.9	0.464	0.015	0.039
TRICLOPYR	2002	L0.01	1055	154	14.6	0.780	0.037	0.070
TRIFLURALIN (TREFLAN)	1997	L0.005	1627	10	0.6	0.187	0.012	0.029

Table 5.9. Summary of pesticide analyses from 1999 to 2006, cont'd.

Pesticide	Year Added to Suite	MDL	Number of samples collected	Number of Detections	Detection Frequency (%)	Maximum Measurable Concentration (ug L^{-1})	Median Measurable Concentration (ug L^{-1})	Mean Measurable Concentration (ug L^{-1})
INSECTICIDES								
ALDICARB	2004	L0.1	641	0				
ALDRIN	2002	L0.005	908	0				
ALPHA-BENZENEHEXACHLORIDE (ALPHA-BHC)	1997	L0.005	1627	2	0.1	0.008	0.007	0.007
ALPHA-ENDOSULFAN	1997	L0.005	1627	0				
CHLORPYRIFOS-ETHYL (DURSBAN)	1997	L0.005	1627	3	0.2	0.781	0.025	0.271
DIAZINON	1997	L0.005	1627	4	0.2	0.041	0.010	0.017
DIELDRIN	2002	L0.005	908	0				
DIMETHOATE (CYGON)	1998	L0.05	1627	0				
DISULFOTON (DI-SYSTON)	1997	L0.2	1627	0				
ETHION	1997	L0.1	1627	0				
GAMMA-BENZENEHEXACHLORIDE (LINDANE)	1997	L0.005	1627	10	0.6	0.030	0.019	0.019
GUTHION (AZINPHOS-METHYL)	1997	L0.2	1627	0				
MALATHION	1997	L0.05	1627	0				
METHOMYL	2005	L0.1	480	0	0.0			
METHOXYCHLOR	1997	L0.03	1627	0				
PARATHION	2006	L0.01	306	0				
PHORATE (THIMET)	1997	L0.005	1627	0				
PYRIDABEN	1998	L0.02	1627	0	0.0			
TERBUFOS	1997	L0.03	1627	0				
THIAMETHOXAM	2006	L0.05	241	0				
FUNGICIDES								
CARBATHIIN (CARBOXIN)	1997	L0.1	1627	0				
CHLOROTHALONIL	2006	L0.005	306	0				
HEXACONAZOLE	2006	L0.05	306	0				
IPRODIONE	2006	L0.02	306	10	3.3	0.365	0.177	0.183
METALAXYL-M	2006	L0.01	306	2	0.7	0.818	0.422	0.422
OXYCARBOXIN	2004	L0.05	641	1	0.2	0.404	0.404	0.404
PROPICONAZOLE	2006	L0.05	306	0				
VINCLOZOLIN	2004	L0.01	641	3	0.5	0.261	0.052	0.108

[Z] Year added to the suite is for reference purposes, all other columns represent data collected between 1999 and 2006.
[Y] Dicamba detection limit changed in 2001 from L0.02 to L0.005 µg L^{-1}

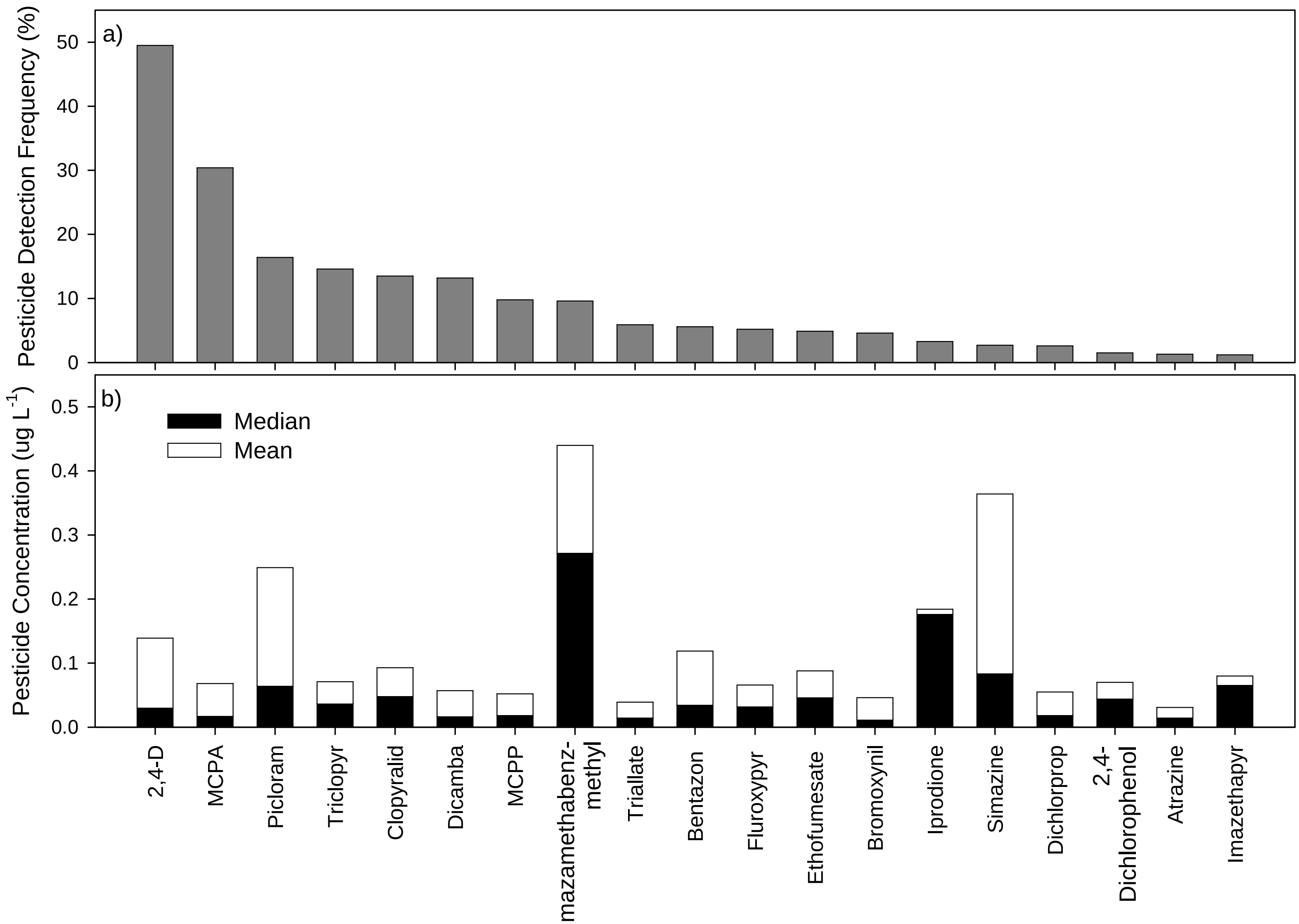


Figure 5.4. Pesticide detection frequency (a) and median/mean concentrations (b) for 19 pesticide compounds detected in >1 percent of samples collected from 1999 to 2006 in all AESA streams. Note than median and mean concentrations are not stacked.

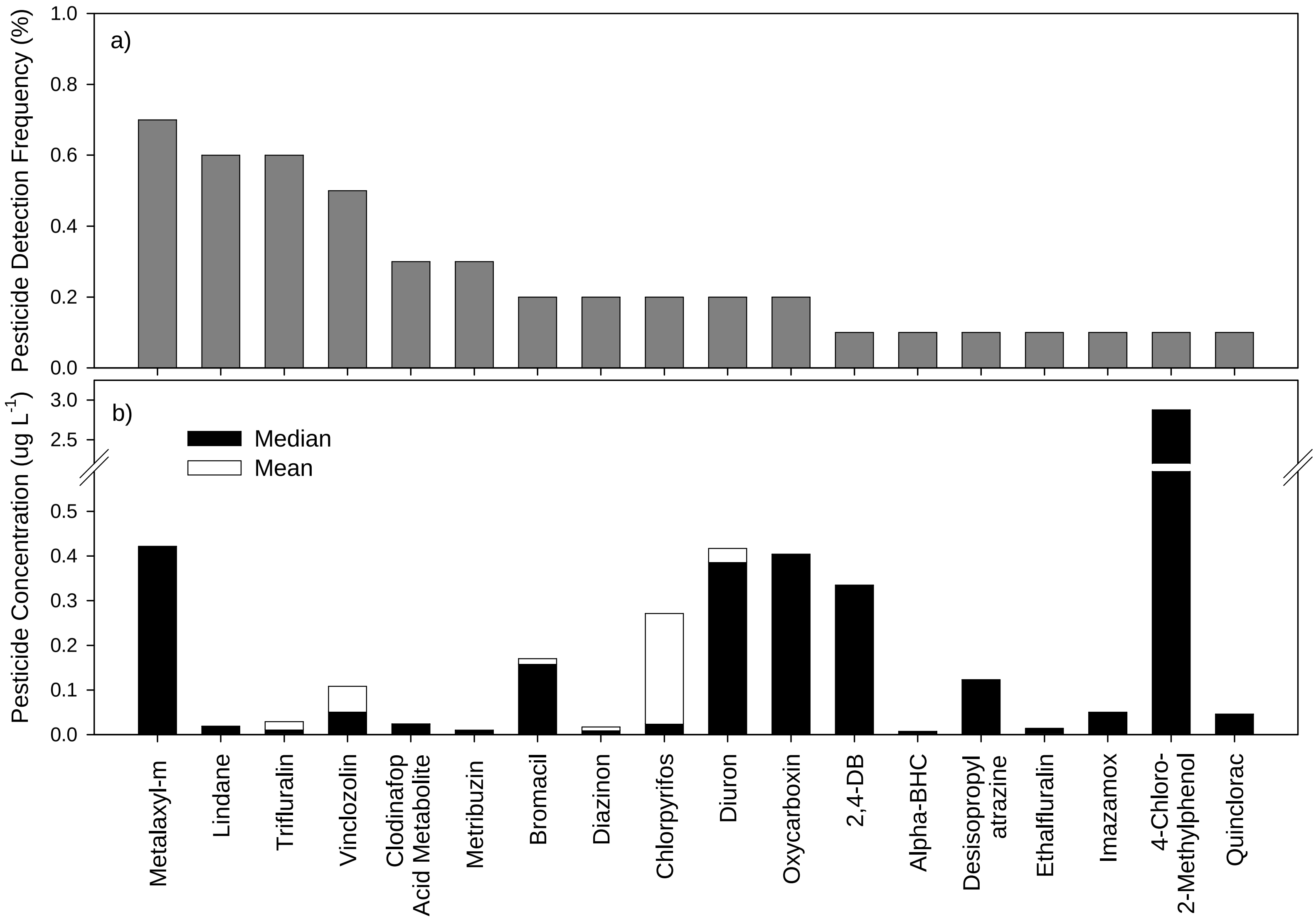


Figure 5.5. Pesticide detection frequency (a) and median/mean concentrations (b) for 18 pesticide compounds detected in <1 percent of samples collected from 1999 to 2006 in all AESA streams. Note than median and mean concentrations are not stacked.

Water Quality Guidelines

Water quality guidelines represent the acceptable level of a substance that can occur in a surface waterbody without causing adverse effects on the intended water use. Guidelines for pesticide compounds exist to protect aquatic life (PAL) in streams and agricultural uses (irrigation or livestock watering) of streams. Guidelines for human drinking water are not intended to be applied to untreated surface water but are included here to provide a benchmark where no other guideline existed. Three sets of water quality guidelines were reviewed to ensure that the most up-to-date and stringent guidelines were being used to assess water quality (Table 5.10). Data sources include Alberta's Surface Water Quality Guidelines (AENV 1999), Canadian Water Quality Guidelines (CCME 2005) and USEPA water quality criteria.

Sixteen of the 37 pesticides and breakdown products detected over the monitoring period (1999 to 2006) have a water quality guideline for PAL, agricultural uses and/or drinking water. Nine of the 16 compounds with guidelines were not met on one or more occasion. With the exception of MCPA, guidelines were exceeded only for a single use. No guideline for livestock watering was ever exceeded. Of the 23 AESA streams, 21 streams had one or more guidelines exceeded on at least one occasion.

Irrigation guidelines for MCPA and dicamba were exceeded most frequently (11.2% and 11.4% of samples, respectively), indicating potential for damage to sensitive plant species if stream water was used for irrigation purposes. Guidelines for the Protection of Aquatic Life (PAL) were exceeded for MCPA, 2,4-D, chlorpyrifos, lindane, and triallate but only in a small proportion of samples (0.2 to 0.5%).

The two compounds that exceeded guidelines most frequently also exceeded them by the greatest amount. Maximum concentrations of dicamba and MCPA were 189- and 291- fold higher than their irrigation guideline, respectively (Table 5.11). The insecticide chlorpyrifos also exceeded the PAL guideline by a large degree (223x), though infrequently. Water quality concerns with MCPA and dicamba are more chronic in nature, and non-compliance may have implications where stream water is applied to sensitive specialty crops.

Table 5.10. Water quality guidelines for pesticides ($\mu g\ L^{-1}$).

Pesticide Parameter	Protection of Aquatic Life[a]	Agricultural Uses[a] Irrigation Water	Agricultural Uses[a] Livestock Water	Drinking Water[b]
2,4-D	4	-	100	100
2,4-Dichlorophenol	-	-	-	900
Aldicarb	1	54.9	11	-
Atrazine	1.8	10	5	5
Bromacil	5	0.2	1100	-
Bromoxynil	5	0.33	11	5
Carbathiin	-	-	-	-
Chlorpyrifos	0.0035	-	24	90
Clopyralid	-	-	-	-
Cyanazine	2	0.5	10	10
Diazinon	-	-	-	20
Dicamba	10	0.006	122	120
Diclorprop	-	-	-	-
Diclofop-methyl	6.1	0.18	9	9
Dimethoate	6.2	-	3	20
Disulfoton	-	-	-	-
Diuron	-	-	-	150
Lindane	0.01	-	4	-
Glyphosate	65	-	280	280
Guthion	[0.01]	-	-	-
Imazamethabenz-methyl	-	-	-	-
Imazethapyr	-	-	-	-
Malathion	[0.1]	-	-	190
MCPA	2.6	0.025	25	-
MCPP	-	-	-	-
Methoxychlor	[0.03]	-	-	900
Phorate	-	-	-	2
Picloram	29	-	190	190
Simazine	10	0.5	10	10
Terbufos	-	-	-	1
Trillate	0.24	-	230	-
Triclopyr	-	-	-	-
Trifularlin	0.2	-	45	45

[a]Alberta Surface Water Quality Guidelines (AENV 1999)
[b]Summary of Guidelines for Canadian Drinking Water (PFSDW 2001)
[ ] Alberta adopted USEPA Water Quality Criteria

Table 5.11. Compliance with Canadian Water Quality Guidelines for the Protection of Aquatic Life or Irrigation.

Pesticide Compound	Type of Guideline	Percent samples where guideline exceeded (%)	Guideline Conc. (µg L^{-1})	Max. Conc. Detected (µg L^{-1})	Median Conc. (µg L^{-1})	Number of samples where guideline exceeded
Dicamba	IRRIG	11.4	0.006	1.134	0.018	185
MCPA	IRRIG	11.2	0.025	7.279	0.018	183
MCPA	PAL	0.2	2.6	7.279	0.018	1
Simazine	IRRIG	0.5	0.5	4.57	0.084	5
Gammabenzenehexachloride	PAL	0.5	0.01	0.03	0.015	8
Triallate	PAL	0.2	0.24	0.464	0.015	3
Chlorpyrifos	PAL	0.2	0.0035	0.781	0.015	3
Bromoxynil	IRRIG	0.2	0.33	0.522	0.011	3
Bromacil	IRRIG	0.1	0.2	0.297	0.158	1
2,4-D	PAL	0.2	4	8.534	0.031	4

When grouped by agricultural intensity category, the highest percentage of non-compliant samples occurred in irrigated watersheds followed by high intensity, moderate intensity, and low intensity watersheds (Figure 5.6).

In irrigated watersheds, non-compliance was observed for six compounds (dicamba, MCPA, simazine, chlorpyrifos, bromoxynil, and 2,4-D) (Figure 5.6). The greatest number of compounds with non-compliant concentrations was measured in New West Coulee. This included all compounds measure in New West Coulee with the exception of chlorpyrifos. Chlorpyrifos PAL guidelines were only exceeded in Battersea Drain. New West Coulee was the only stream of the 23 to have concentrations of simazine that exceeded irrigation guideline values. Crowfoot Creek had the highest proportion of samples of all AESA streams that did not comply with dicamba guidelines for irrigation use.

In high intensity watersheds, non-compliance was observed for six compounds (Figure 5.6). Haynes and Wabash Creeks experienced non compliance for the greatest number of individual compounds: dicamba, MCPA, lindane, and 2,4-D in Haynes Creek and dicamba, MCPA, bromacil, and triallate in Wabash Creek. A higher proportion of samples exceeded MCPA guidelines for irrigation in high intensity dryland watersheds than irrigated watersheds.

In moderate intensity watersheds, non-compliance was observed for four compounds (2,4-D, dicamba, triallate, and gammabenzehexachloride (lindane)) (Figure 5.6). Kleskun Drain and Grande Prairie Creek, both located in northern Alberta, were found to have the greatest number of non-compliant compounds. All guidelines were met in Trout Creek.

In the low intensity watersheds, non-compliance was observed for four compounds (Figure 5.6). Prairie Blood Coulee experienced non-compliance for three of these compounds (2,4-D, dicamba, and lindane). All guidelines were met in Hines Creek, while the remaining streams showed non-compliance for MCPA and/or dicamba.

The data were also examined to assess whether multiple guidelines were concurrently exceeded in samples. In eight of the 1627 samples collected from 1999 to 2006, three water quality guidelines were concurrently exceeded. In 62 of 1627 samples, two guidelines were concurrently exceeded. The streams that had the highest incidence of multiple exceedences were New West Coulee, Wabash Creek, and Crowfoot Creek.

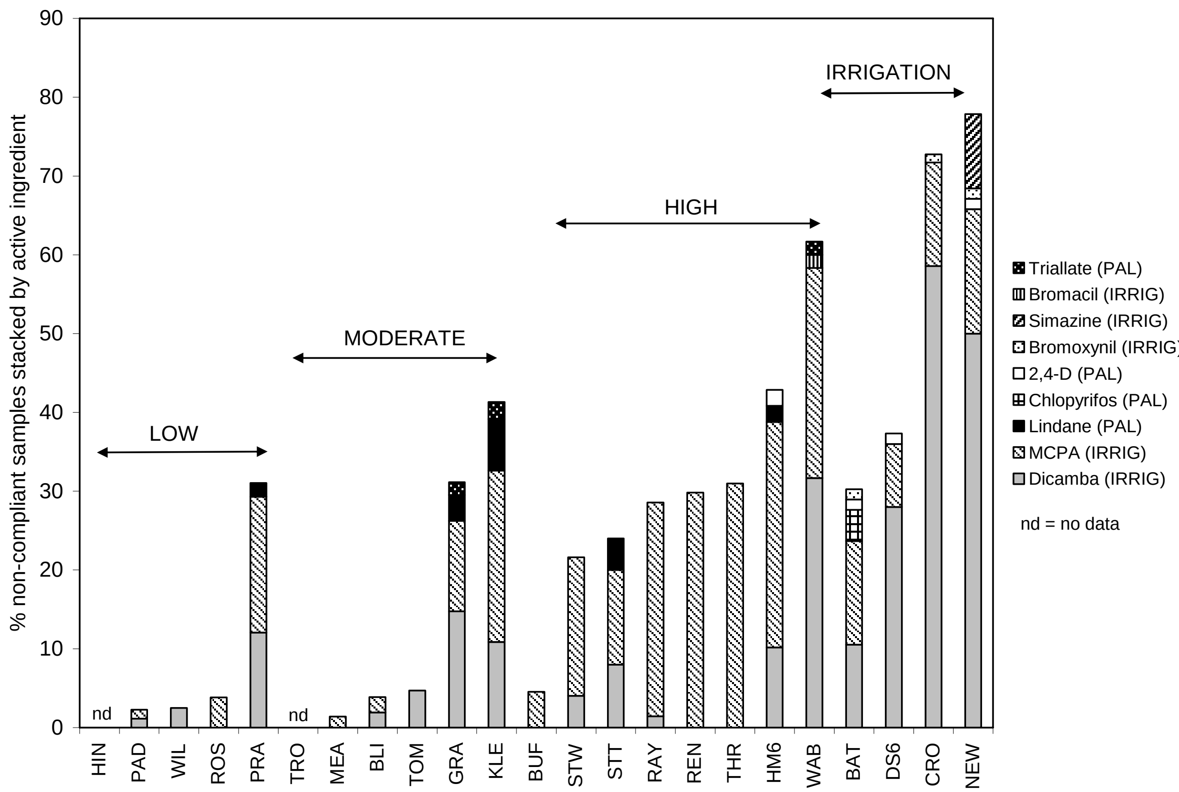


Figure 5.6. Stacked bar graph of percent non-compliance with water quality guidelines in 23 AESA watersheds. For full stream names refer to Table 5.4.

While 16 of the 37 pesticides detected had a water quality guideline, 21 compounds detected in Alberta's agricultural streams are currently without guidelines (Figure 5.7). Of the 21 compounds without guidelines

- Four were detected in ≥10% of samples (imazamethabenz-methyl, MCPP, clopyralid, and triclopyr), and
- Six were detected in >1 to 10% of samples (imazethapyr, dichlorprop, iprodione, ethofumesate, fluroxypyr, and bentazon).

The remaining 11 were detected in <1% of samples or the equivalent of 1 to 3 occasions over the monitoring period: quinclorac, 4-chloro-2-methylphenol, imazamox, ethalfluralin , 2,4-DB, alpha-benezenehexachloride, oxycarboxin, diuron, clodinafop acid metabolite, vinclozolin, and metalaxyl-M.

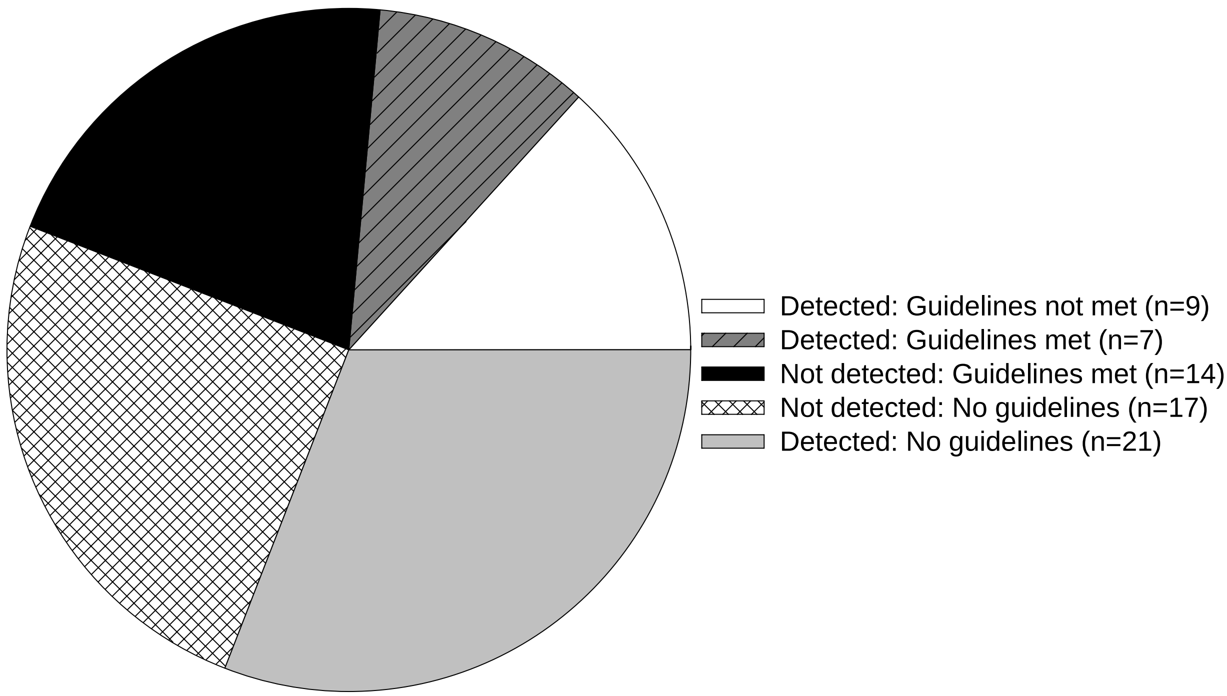


Figure 5.7. Comparison of pesticide detection and guideline compliance for the 68 compounds monitored from 1999 to 2006.

The four herbicides with more frequent detections (imazamethabenz-methyl, MCPP, clopyralid, and triclopyr) should be flagged as priority substances for guideline development in order to assess the risks for aquatic life, irrigation and/or livestock watering, and drinking water.

Alberta Pesticide Toxicity Index (APTI)

Four of the 23 AESA streams had a portion of samples that fell in the high risk category for either or both EC_{50} or LC_{50}. These include three irrigated drains (New West Coulee, Battersea Drain, and Crowfoot Creek) and one high intensity dryland stream (Threehills Creek) (Figure 5.8). The proportion of all samples that were classified as high risk ranged from 2.6% (Battersea Drain) to 6.6% (New West Coulee) for EC_{50} and from 1 (Crowfoot Creek) to 2.6% (Battersea Drain) for LC_{50}. Threehills Creek had 1.4% of samples that were rated high risk based on the LC_{50} endpoint. It is interesting that irrigated watersheds had a higher toxicity risk than high intensity dryland watersheds because the high intensity dryland watersheds had greater total concentrations. Total concentration alone is not a good measure of threat to aquatic life; it is important to know which compounds are present and in what concentrations and what their toxicity is to non-target aquatic species.

The two pesticide compounds that exceeded the endpoints for EC_{50} were simazine (New West Coulee) and chlorpyrifos (Battersea Drain), and they were chlorpyrifos (Battersea Drain) and diazinon (Crowfoot Creek, Threehills Creek) for LC_{50}. The algal EC_{50} endpoint for simazine was exceeded on five sampling dates from 2002 to 2006 in New West Coulee, either in March or April (13-Apr-02, 20-Mar-03, 15-Mar-04, 28-Mar-06, 13-Mar-06). The EC_{50} and LC_{50} endpoints for chlorpyrifos were both exceeded on two June sampling dates in Battersea Drain (10-Jun-99, 13-Jun-00). The invertebrate LC_{50} endpoint for diazinon was exceeded on one date in both Crowfoot Creek (8-Mar-01) and Threehills Creek (15-May-01).

These results suggest that improved management of simazine, chlorpyrifos, and diazinon should be a priority for these high-risk watersheds. Where possible, alternate pest control measures should be implemented to reduce risks to aquatic life.

Eleven streams in addition to New West Coulee, Battersea Drain, Crowfoot Creek, and Threehills Creek had occasions where moderate pesticide toxicity risk was observed for either LC_{50} and/or EC_{50} endpoints (Figure 5.8). In the majority of streams with moderate risk, both LC_{50} and EC_{50} endpoints were exceeded. In the irrigated streams, the four pesticide compounds that most frequently exceeded the NOEC limit were atrazine, triallate, dicamba and simazine. In Crowfoot Creek, diazinon concentrations also posed a risk. In the Peace Region (Grande Prairie Creek and Kleskun Drain), both lindane and triallate were present in concentrations that exceeded acceptable levels. In Wabash Creek, higher diuron concentrations contributed to higher toxicity risks. The pesticide active ingredients listed here were those that presented the higher risk, while additional compounds also exceeded NOEC on occasion.

Eight of the 23 streams had low pesticide toxicity risk for EC_{50} and LC_{50} endpoints, which suggests that pesticide presence did not threaten aquatic life. Streams with low risk included three of the five low agriculture watersheds (Hines Creek, Paddle River, Willow Creek), three of the six moderate agriculture watersheds (Blindman River, Meadow and Trout Creeks) and two of the eight high agriculture watersheds (Buffalo and Strawberry Creeks).

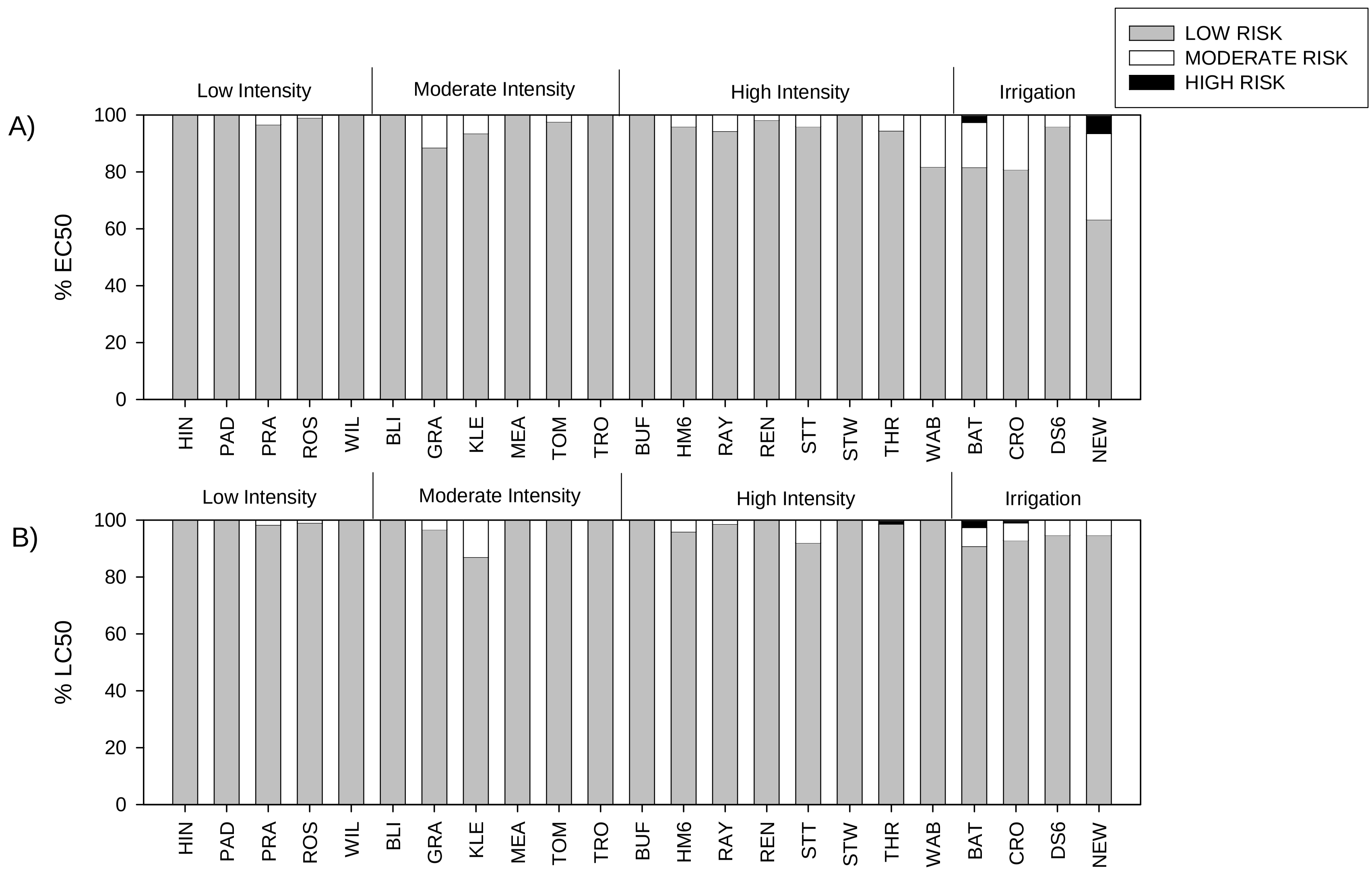


Figure 5.8. Distribution of Alberta Pesticide Toxicity Index (APTI) scores. APTI scores are based on lowest EC50 and L50 endpoints and include all available pesticide data from each of 23 AESA streams (n= 25 to 103). Refer to Table 5.4 for full stream names.

Pesticide Occurrences and Concentrations by Watershed Agricultural Intensity

Each of the 23 AESA watersheds had at least one pesticide compound detected in stream water during the monitoring period. Statistical analysis showed that pesticide detection frequency (H=75.936, $p \leq 0.0001$), total pesticide concentration ($F(3,165)=14.780$, $p \leq 0.0001$), and total number of compounds (H=47.708, $p \leq 0.0001$) increased significantly as agricultural intensity increased from low to high, thus suggesting that agricultural use of pesticides has a measurable impact on water quality.

Total pesticide detection frequency. Pesticide detection frequency increased in a stepwise fashion from low to moderate to high (dryland or irrigated) (Figure 5.9). Median annual detection frequency more than doubled from low to moderate intensity watersheds (21% to 56%, U=501.5, $p \leq 0.0001$) and nearly doubled again from moderate to high intensity dryland (94%, U=780.5, $p \leq 0.0001$) or irrigation watersheds (100%, U=116.0, $p \leq 0.0001$). These findings show that the likelihood of pesticides reaching surface waters is similar in high intensity agricultural watersheds, regardless of whether dryland or irrigated agriculture is practiced.

Among the 23 individual AESA watersheds, there was a wide range in pesticide detection frequencies (Figure 5.10). Detection frequencies were lowest in two low intensity watersheds (Willow Creek and Hines Creek) where only 7 and 8% of samples contained detectable levels of pesticides, respectively (Figure 5.10). In contrast, pesticides were detected in virtually every sample collected (99 to 100%) from New West Coulee and Crowfoot Creek (irrigated streams) and Kleskun Drain (moderate intensity). Other watersheds with detection frequencies >90% all had high intensity agriculture including Haynes Creek, Wabash Creek, and Stretton Creek, plus the irrigated Battersea Drain.

Of the low intensity watersheds, the highest pesticide detection frequency was observed in Prairie Blood Coulee (60%) (Figure 5.10). Of the moderate intensity watersheds, Kleskun Drain had the highest detection frequency (100%). Both of these watersheds also had the highest proportion of crop land in their respective watershed categories, according to 1991 land cover data (Chapter 2, Table 2.9).

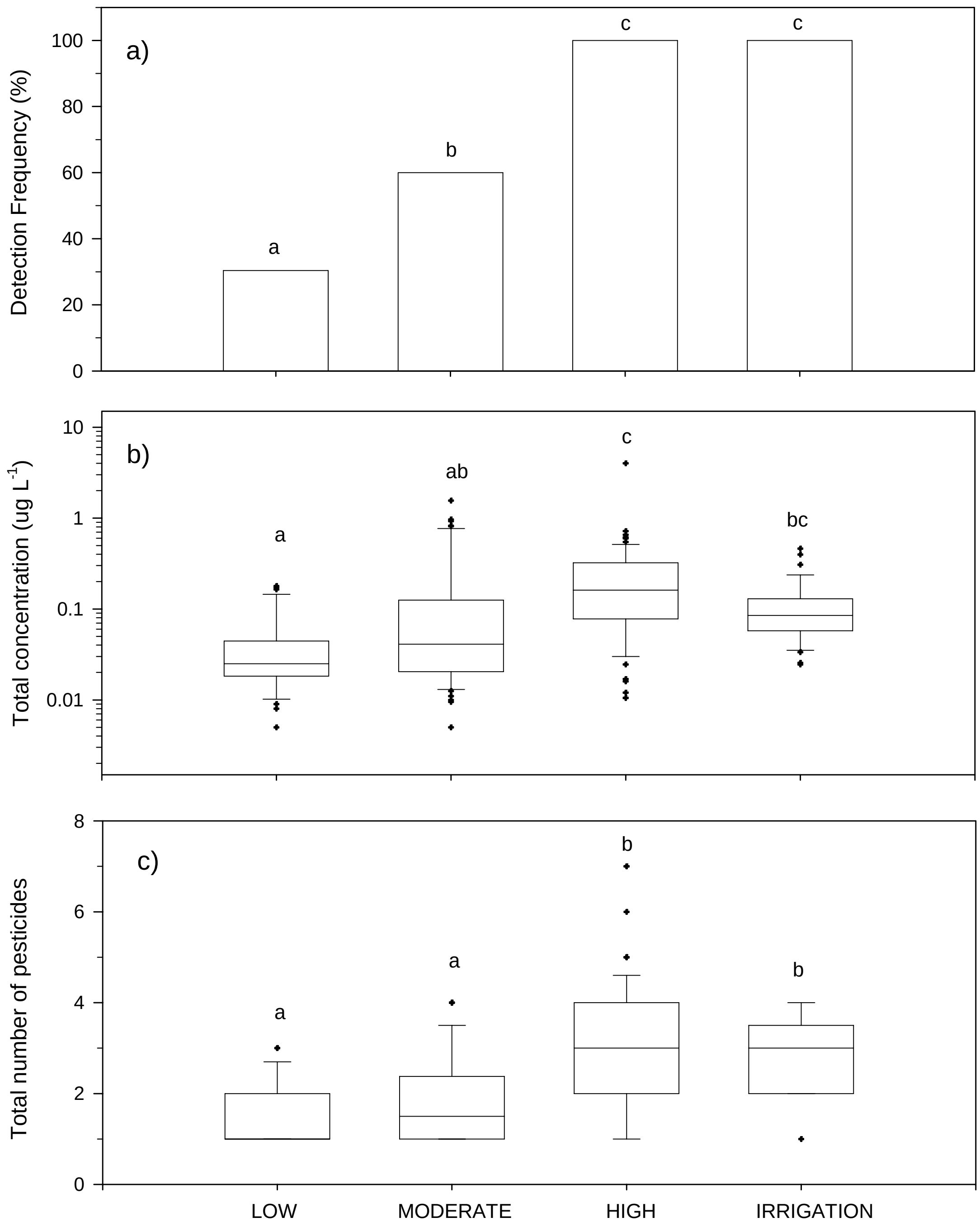


Figure 5.9. Total pesticide detection frequency (%) (a), total pesticide concentration (µg L^{-1}) (b), and total number of compounds detected (c) among watersheds with different agricultural intensities (low, moderate, high and irrigated). Box plots stretch from the 25th percentile to the 75th percentile with the horizontal line in the middle of the box representing the median. Vertical lines represent 1.5 times the interquartile range while crosshairs represent minima and maxima data points. The bar graph shows the median of annual medians. Bars or box plots with different letters above them are significantly different ($p<0.0001$). Kruskal Wallis analysis was used for (a) and (c) and a one-way ANOVA for (b).

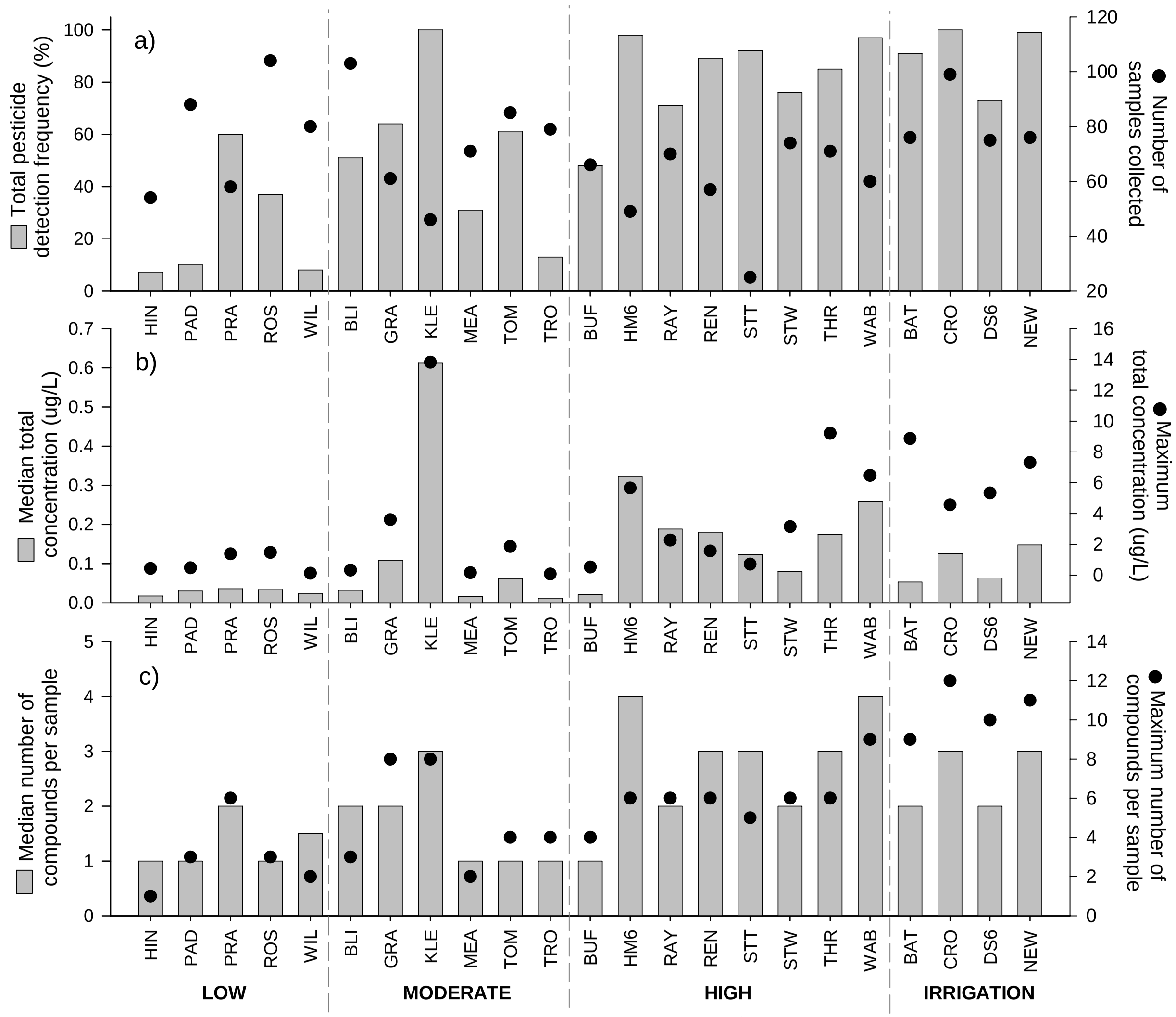


Figure 5.10. Detection frequency (%) (a), total pesticide concentration (µg L^{-1}) (b), and total number of compounds detected (c) in the 23 streams from 1999 to 2006. Summary statistics for each stream (total detection frequency, median, maximum) are calculated from the raw daily data set (68 compounds). Refer to Table 5.4 for full stream names.

Total pesticide concentration. Total pesticide concentrations followed a slightly different pattern than pesticide detection frequencies. Total concentrations were lowest in low agricultural intensity watersheds, highest in high intensity watersheds, and intermediate in both moderate and irrigated watersheds ($F(3,165)=14.780$, $p \leq 0.0001$)(Figure 5.10). Higher pesticide detections in high intensity watersheds were mirrored by higher total concentrations, suggesting that pesticides are found more frequently in these watersheds as well as at higher concentrations. Despite similar detection frequencies in high intensity dryland and irrigated streams, total pesticide concentrations were slightly lower in irrigated watersheds as they were not significantly different than either high or moderate intensity watersheds. Lower concentrations in irrigated watersheds may be due to dilution of irrigation return flow by cleaner irrigation source water and higher median flow volumes. Loading estimates from high intensity and irrigated watersheds would be useful in elucidating differences related to flow.

Among individual watersheds, median total pesticide concentrations based on all available data ranged from <0.02 µg L^{-1} in Hines, Trout, and Meadow Creeks to 0.613 µg L^{-1} in Kleskun Drain (Figure 5.10). Kleskun Drain clearly stood out as having the highest total concentration out of all AESA streams. The watershed with the second highest median total concentration was nearly half the value at 0.323 µg L^{-1} (Haynes Creek, high intensity). As described in Depoe and Westbrook (2003), non-agricultural influences in the watershed may have affected the water quality in Kleskun Drain. From 2000 to 2003, the highway that crosses the stream near the mouth of the watershed (Highway 34) was twinned, and pest control along the right-of-way presents a potential source of pesticides.

The highest maximum total pesticide concentrations were observed in high intensity and irrigated streams, with the exception of Kleskun Drain (moderate intensity). Maximum total concentrations ranged from 4.559 to 9.205 µg L^{-1} in Haynes Creek, Threehills Creek, Wabash Creek, Battersea Drain, Crowfoot Creek, Drain S-6, and New West Coulee. The highest maximum total concentration (13.814 µg L^{-1}) was observed in Kleskun Drain in July 2000.

Total number of detections per sample. The total number of pesticides detected per sample was similar in watersheds with low and moderate intensity agriculture and significantly higher in watersheds with high intensity or irrigated agriculture ($H=47.708$, $p \leq 0.0001$, Figure 5.11). Pesticide mixtures were more common in the high intensity watersheds regardless of whether dryland or irrigated agriculture was practiced. Within the low intensity category, the median number of compounds per sample (all data) was higher in Prairie Blood Coulee (two) compared to the rest of the low intensity streams (one) (Figure 5.10).

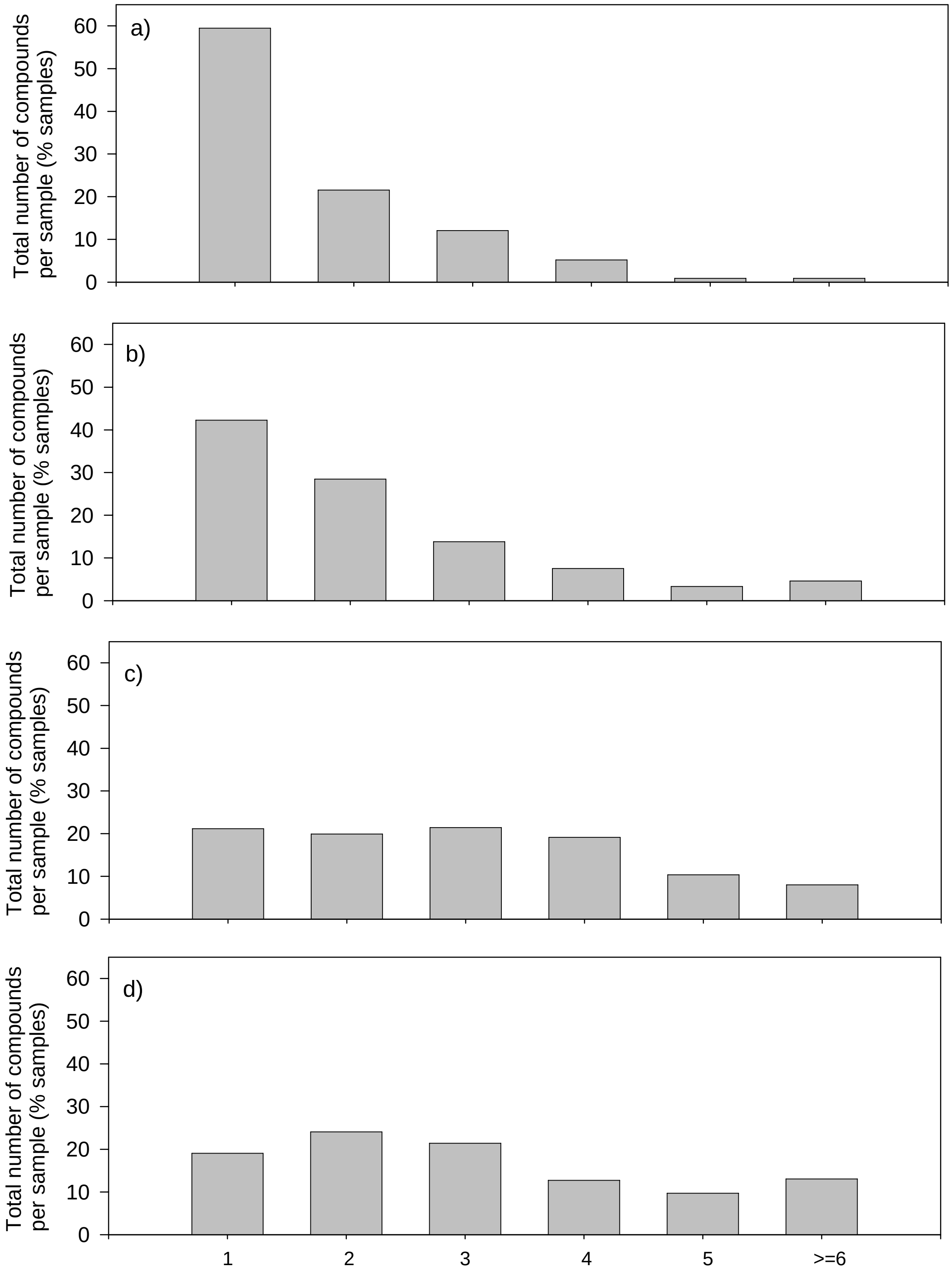


Figure 5.11. Histogram of the total number of compounds detected per sample (1,2,3,4,5 or ≥6) in watersheds with low (n=116) (a), moderate (n=239) (b), high (n=387) (c), and irrigated (n=299) (d) agricultural intensities. Plots show counts of pesticide mixtures from the raw daily data set.

In the moderate intensity category, the highest median number of compounds was observed in Kleskun Drain (three) compared to other streams (one or two). Of the high intensity dryland and irrigated streams, highest medians were observed in Haynes and Wabash Creeks (four compounds per sample) compared to three or fewer in the other streams.

For all data combined (n= 1627), the median number of compounds detected per sample was two. In low and moderate intensity watersheds, water samples predominantly contained only one pesticide (59 and 42%, respectively), and few samples (< 10%) contained greater than four pesticides (Figure 5.10). In the high intensity and irrigated watersheds there was nearly an equal number of samples with either one, two, or three pesticides detected and a notable shift in proportion of samples with four or more pesticides.

The higher diversity of compounds observed in high intensity and irrigated watersheds is reflected by the total number of pesticides detected per stream over the eight year monitoring period (Figure 5.12). In each of the four irrigated watersheds, a total of 15 or more compounds were detected. In high intensity watersheds, a total of ten or more compounds were detected in the majority of streams.

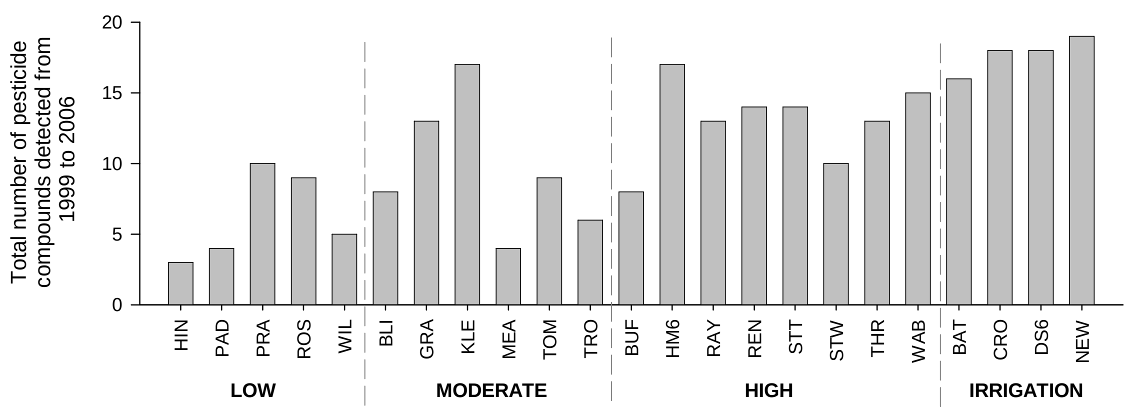


Figure 5.12. Total number of individual pesticides or degradation products detected in each of the AESA watersheds from 1999 to 2006. For full stream names see Table 5.4.

Relationship with Land Cover and Chemical and Fertilizer Expenses ($/acre)

The relationship between the intensity of agriculture in a watershed (as % cropland and fertilizer and chemical expense percentiles) and pesticide occurrence were explored using correlation analysis. Pearson rank correlations between total pesticide detection frequency and each of the three metrics were strong ($r \geq 0.75$) (Figure 5.13). Spearman rank correlations between total pesticide concentration and the three metrics were slightly weaker ($r_s = 0.59$ to 0.60) but were also similar for each metric. Weaker relationships with concentration data were expected as total concentration tended to be influenced by water management (irrigated versus dryland) as well as by the intensity of chemical use.

Strong relationships between pesticide occurrence and land-based metrics of agricultural intensity corroborate findings of Gilliom (2007). Their nation-wide pesticide survey of USA's streams from 1992 to 2001 found that pesticide detection frequency largely reflected the geographic distribution of land use, crops, and associated chemical use. The observed relationship in Alberta suggests that land-based watershed metrics may be used to predict the degree of pesticide contamination in small agricultural watersheds. Regression equations, with pesticide detection frequency in particular, could be applied as a management tool for small agricultural watersheds with similar runoff potential to the AESA watersheds.

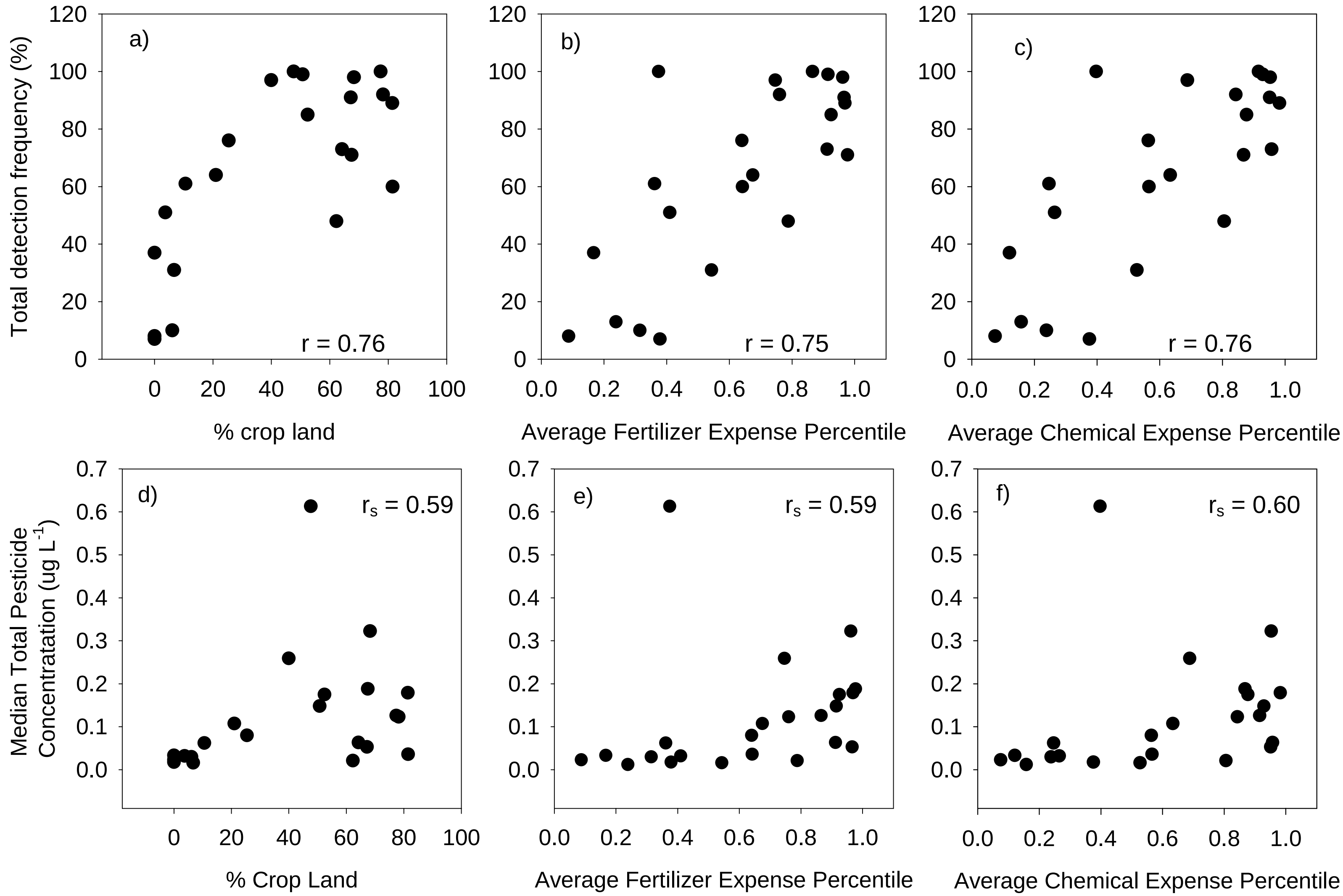


Figure 5.13. Correlations between total pesticide detection frequency and percent crop land(a), average fertilizer expense percentile (b), and chemical expense percentiles(c) (1996, 2001 and 2006 Ag Census data) and median total pesticide concentrations and percent crop land(d), average fertilizer expense percentile(e) and chemical expense percentiles(f) in 22 of 23 AESA streams (does not include Drain S-6). Pearson rank correlations were used for detection frequency and spearman rank correlations for pesticide concentration.

Relationship with Pesticide Sales Data

In this section we discuss pesticide findings in relation to sales data and physical properties. The details on interannual trends in specific pesticide compounds in individual AESA watersheds are included in the following section.

Frequent detections of the herbicides MCPA and 2,4-D in stream water are consistent with sales volumes; MCPA had the 2nd highest sales volume and 2,4-D had the 3rd highest sales volume in the 2003 review (~1,000,000 and 685,000 kg, respectively) (Byrtus 2007). Provincial sales volumes for dicamba and imazamethabenz-methyl were also high (108 639 kg and 138 551 kg, respectively), and these two herbicides were among the eight most ubiquitously detected (Figure 5.14). Triallate (197 221 kg) and bromoxynil (354 907 kg) sales volumes also support the detection frequencies of these two herbicides. Each of these six active ingredients were detected at least once in at least half (12) and up to all (23) of the AESA streams.

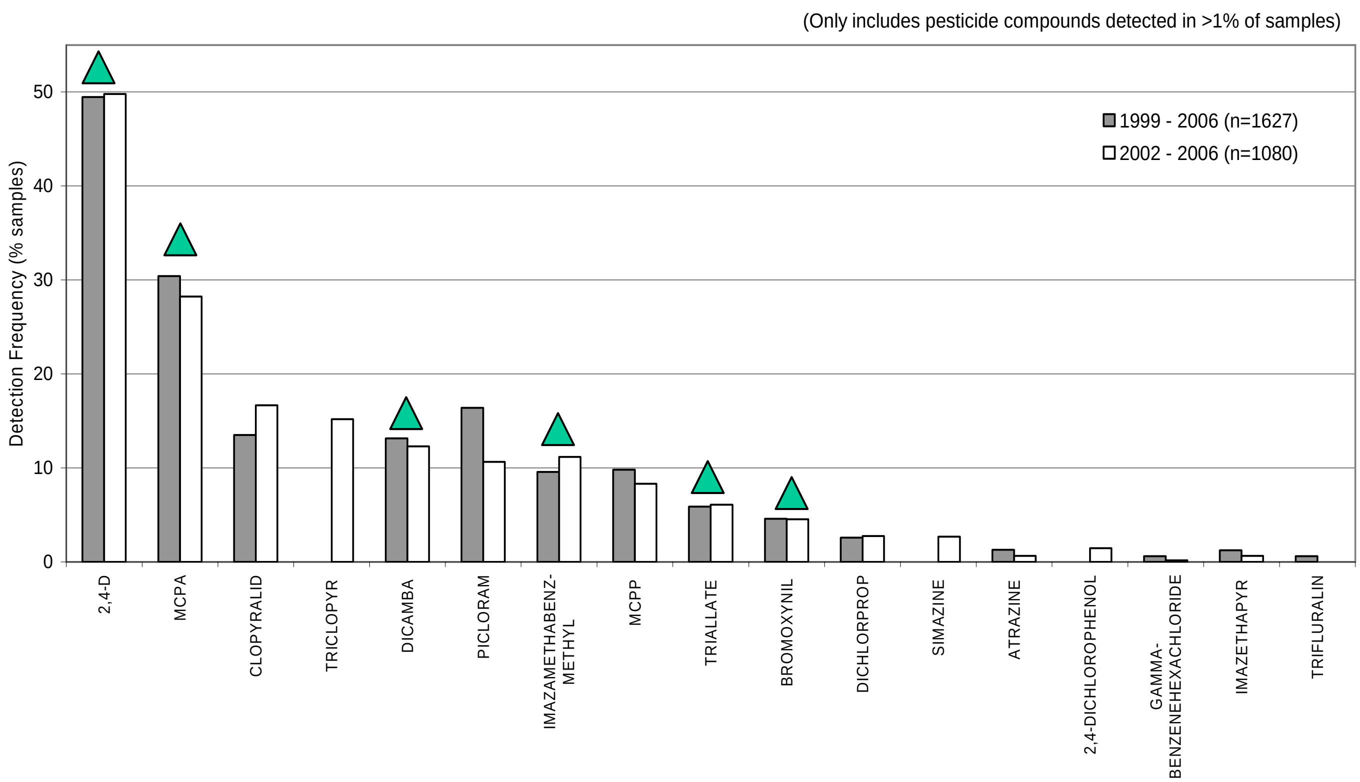


Figure 5.14. Pesticide detection frequency of pesticide compounds detected in >1% of samples. Triangles denote active ingredients with highest pesticide sales in 2003 (i.e., > 100,000 kg).

Anderson (2005) noted that pesticide characteristics related to mobility and persistence can override the influence of use (sales) patterns. Clopyralid, triclopyr, picloram, and MCPP were all are detected in similar frequencies as compounds with high sales and in a similar number of watersheds (16 to 19), despite lower sales volumes. All except MCPP are classified as having 'very high' mobility; thus, detections are likely related to physical characteristics and not directly based on sales (Table 5.12). Water solubilities for all four compounds were > 200,000 mg L^{-1}

(Table 5.12; Appendix 14). Another factor affecting movement is the type of application (soil surface-applied, soil-incorporated, or post-emergence).

Table 5.12. Characteristics of pesticide compounds or degradation products with 'very high' pesticide movement ratings.

Common name	Pesticide movement rating	Soil half-life (days)	Water solubility (mg L^{-1})	Sorption coefficient (soil Koc)
Bromacil acid	Very high	60	700	32
Bromacil lithium salt	Very high	60	700	32
Clopyralid amine salt	Very high	40	300,000	6
Dicamba salt	Very high	14	400,000	2
Imazamethabenz- methyl (p-isomer)	Very high	45	857	35
Imazethapyr	Very high	90	200,000	10
Metalaxyl	Very high	70	8400	50
Picloram salt	Very high	90	200,000	16
Triclopyr amine salt	Very high	46	2,100,000	20

Source: Wauchope et al. 1992; Augustijn-Beckers et al. 1994; Cotton 1995.

The high detection frequency of MCPP (mecoprop) may not be expected as it does not have high agricultural sales and the overall pesticide mobility ranking is high (vs. very high). Closer examination of the specific AESA watersheds with MCPP detections shows that both watersheds are under urban influence: Crowfoot Creek from the City of Calgary and Wabash Creek from the Town of Westlock. In both watersheds, MCPP is detected in nearly every sample (annual detection frequencies ranged from 46 to 100% in Crowfoot Creek and 56 to 100% for Wabash Creek). These findings show that urban influences (including golf courses, municipal treatment plants, and untreated road runoff) can also affect water quality in small rural watersheds in Alberta.

Imazamethabenz-methyl was one of the eight most ubiquitously detected compounds in addition to having some of the highest concentrations. In opposition to what was found for MCPP, the common occurrence of imazamethabenz-methyl may have been related to the large volume sold in 2003 (138 551 kg) combined with a very high pesticide mobility ranking.

The top selling herbicide in the province, glyphosate, was not included as one of the compounds analysed in the AESA Stream Survey because of the additional cost of analysis and poor analytical detection limits. In comparison to some of the compounds discussed above, glyphosate had annual sales of more than 3.3 million kg of active ingredient (based on 2003 sales). In 2007, a joint partnership was formed to investigate the presence of glyphosate in Alberta's agricultural watersheds (Lorenz, *in press*). Results from the study showed glyphosate was detected in all agricultural intensity watersheds and in all months sampled (March to October) with the exception of August. The maximum glyphosate concentration detected was 14µgL^{-1} and was measured in one of the high agricultural intensity streams. All glyphosate detections were below the guidelines for the protection of aquatic life (65µgL^{-1}), irrigation use (280µgL^{-1}), and livestock drinking water (280µgL^{-1}). For more information, see Lorenz, *in press*.

Spatial Distribution of Specific Pesticide Detections in Relation to Land Use

There were eight pesticides only detected in watersheds with irrigated agriculture and nine pesticides only detected in watersheds with dryland agriculture, reflecting the influence of the type of crops grown in each region (Table 5.13).

Irrigation

The eight compounds only detected in irrigated streams included four herbicides (simazine, ethofumesate, ethalfluralin, and metribuzin), two degradation products of herbicides (4-chloro-2-methylphenol – a degradation product of MCPA, MCPB and MCPP - and desipropylatrazine - a degradation product of atrazine), one insecticide (chlorpyrifos-ethyl), and one fungicide (metalaxyl-m) (Table 5.13). Atrazine was also detected primarily in irrigated streams but was also detected in Kleskun Drain.

Simazine and ethofumesate both had relatively high average detection frequencies in irrigated streams (14% and 18%, respectively). Detections of both compounds were greatest in New West Coulee (42% and 43%, respectively) (Table 5.13). Simazine was added to the pesticide suite in 2002, and ethofumesate was added to the pesticide suite in 2006. Both were detected in three of the four irrigated streams. Like atrazine, simazine is a triazine herbicide used for the control of broadleaf weeds in corn crops (Alberta Agriculture and Food 2007). Other applications of simazine include weed control in shelterbelts, woody ornamentals, and nursery stock or for industrial purposes (Alberta Agriculture and Food 2007). Ethofumesate is used for weed control in sugar beet crops. The fungicide metalaxyl-m was added to the analytical suite in 2006. It was only detected in Battersea Drain but in relatively high frequencies (9% of samples).

Dryland

Nine compounds were only detected in streams with dryland agriculture: four herbicides (diuron, quinclorac, bromacil, and trifluralin), one herbicide degradation product (clodinafop acid metabolite), two fungicides (iprodione and oxycarboxin), and one insecticide (gamma-benzenehexachloride (lindane)) and its degradation product (alpha- benzenehexachloride). Lindane, iprodione, and trifluralin were detected in 6, 5 and 4 streams, respectively, while the other compounds were detected in only one or two of the dryland streams.

Lindane was used as a seed treatment in canola until 1999 when manufacturers in Canada voluntarily withdrew its use. Producers were allowed to use existing stock until July 2001. It is still permitted for use on some cereal and vegetable crops, but this use is also under review. With the exception of Kleskun Drain, the majority of lindane detections in AESA streams were limited to 1999 and earlier (Table 5.14 and 5.15).

The fungicide iprodione was added in the analytical suite in 2006 and was detected in five of the eight high agricultural intensity watersheds. It was also found in relatively high frequencies in these streams (11 to 40%). Iprodione is used on canola and bean (kidney, snap, white) crops and is applied by spraying. The relatively frequent occurrence of iprodione in water in high intensity dryland watersheds suggests that it is a compound that should continue to be monitored and tracked over time in this region of the province.

Table 5.13. Detection frequencies of individual pesticide compounds in each of the AESA watersheds.

DETECTION FREQUENCY	Detected in both Dryland and Irrigated watersheds																				Detected in Dryland watersheds only									Detected in Irrigated watersheds only							
	2,4-D	MCPA	PICLORAM	MCPP	CLOPYRALID	DICAMBA	TRICLOPYR	BROMOXYNIL	TRIALLATE	IMAZAMETHABENZ-METHYL	DICHLORPROP	IMAZETHAPYR	2,4-DICHLOROPHENOL	FLUROXYPYR	ATRAZINE	DIAZINON	BENTAZON	2,4-DB	IMAZAMOX	VINCLOZOLIN	GAMMA-BENZENEHEXACHLORIDE	IPRODIONE	TRIFLURALIN	DIURON	ALPHA-BENZENEHEXACHLORIDE	BROMACIL	OXYCARBOXIN	CLODINAFOPACIDMETABOLITE	QUINCLORAC	SIMAZINE	ETHOFUMESATE	ETHALFLURALIN	CHLORPYRIFOS-ETHYL	4-CHLORO-2-METHYLPHENOL	METALAXYL-M	METRIBUZIN	DESISOPROPYLATRAZINE
Low Intensity Dryland																																					
Hines Creek	4	4					10																														
Paddle River	8	3				1	30																														
Prairie Blood Coulee	53	36	2	10	10	16		14		2	5										2																
Rose Creek	19	11	17		1		16	1				1									1			1													
Willow Creek	3	1	4	1		3																															
Moderate Intensity Dryland																																					
Blindman River	33	16	20	3	1	2	28	5																													
Grande Prairie Creek	25	51	10	11	36	16	38		20	16		8		14							3		2														
Kleskun Drain	74	37	80	4	70	13	7	2	4	15		7	4		2				2		9		7		2												
Meadow Creek	30	8	1					1																													
Tomahawk Creek	59	6	15	1	1	5	25	1					4																								
Trout Creek	8	4	5	1			5		1																												
High Intensity Dryland																																					
Buffalo Creek	44	20			2		10	5		3	3											17															
Haynes M6 Creek	90	67	41	12	71	10	36	2	2	53	2	6				2	13				2	25	4														
Ray Creek	36	46	13	3	17	3	3	3	7	40	1	1				1																					
Renwick Creek	68	63	35	2	37			14	16	32	2	4		7				2				20							2								
Stretton Creek	84	72	16	8		8	43		8	20	4	8								15	4	40	16														
Strawberry Creek	58	27	58	3	8	7	55	5		1															1												
Threehills Creek	51	65	15	1	28	6		7	11	45	1	3				1												8									
Wabash Creek	90	65	62	85	38	35	40		10	18	2		3									11		3		7	3										
High Intensity Irrigation																																					
Battersea Drain	89	42		5	17	16		8	24		8		2	14	7		23													11	9		4		9		
Crowfoot Creek	99	45	10	60	9	64	4	14	17	15	5	1	6	14	10	1			1													1					
Drain S-6	69	35	8	4	3	35	25	5	3		3		2	5	4		33			3										2	19	1					
New West Coulee	97	51	4	11	20	53		14	17		24		10	33	3		19	1												42	43			2		5	3
Number of streams with detections	**23**	**23**	**19**	**18**	**17**	**17**	**16**	**16**	**13**	**12**	**12**	**9**	**7**	**6**	**5**	**4**	**4**	**2**	**2**	**2**	**6**	**5**	**4**	**2**	**2**	**1**	**1**	**1**	**1**	**3**	**3**	**2**	**1**	**1**	**1**	**1**	**1**

Table 5.14. Detection frequency (%) of gamabenzenehexachloride (Lindane) in AESA watersheds where lindane was detected from 1997 to 2006.										
Watershed	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Crowfoot Creek	33	8								
Grande Prairie Creek			29							
Haynes Creek (M6)			14							
Kleskun Drain			25		20	17		14		
Prairie Blood Coulee				17						
Renwick Creek	50	13								
Rose Creek			7							
Stretton Creek			20							

Table 5.15. Maximum concentration (μg L^{-1}) of gamabenzenehexachloride (Lindane) in AESA watersheds where lindane was detected from 1997 to 2006.										
Watershed	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Crowfoot Creek	0.010	0.023								
Grande Prairie Creek			0.030							
Haynes Creek (M6)			0.017							
Kleskun Drain			0.027		0.029	0.021		0.009		
Prairie Blood Coulee				0.012						
Renwick Creek	0.011	0.010								
Rose Creek			0.010							
Stretton Creek			0.012							

Seasonal Patterns

Seasonally, the majority of samples were collected in April (n= 443) and May (n=300), and the fewest samples were collected in August, September, and October (range n= 89 to 107). An intermediate number of samples were collected in March, June, and July (n=167 to n=217) (Figure 5.15a).

Total pesticide detection frequency. The largest proportion of samples with one or more pesticide detections occurred in March (74%), April (72%), June (71%), and July (65%) (Figure 5.15). However, detection frequencies in these four months were not dramatically higher than in May (53%) or August to October (46 to 58%). Among all months, the largest difference in detection frequency was between March (74%) and October (46%).

Weak seasonality was observed in total pesticide detection frequency (H=14.116, p=0.049). Detection frequency was significantly higher in early spring with snow melt (March and April) and early summer during periods of application (June and July) than in late fall (October) (Mann-Whitney, $p<0.05$).

Total pesticide concentration. Similar to total pesticide detection frequency, median total concentrations were also highest in March, April, June, and July (Figure 5.15). Median concentrations in March (0.164 µg L^{-1}), April (0.149 µg L^{-1}), June (0.112 µg L^{-1}), and July (0.107 µg L^{-1}) all exceeded 0.1 µg L^{-1}. Median concentrations in the remaining months ranged from 0.040 µg L^{-1} (October) to 0.065 µg L^{-1} (August).

Total pesticide concentrations peaked in early spring with runoff and dropped in May, then peaked in summer during the application period and dropped in fall. Statistical analysis of monthly median total concentrations showed a significant difference among months (one-way ANOVA, $F(7,654)=4.232$, $p<0.0001$) with higher concentrations in March, April, and July (Tukey post hoc, $p<=0.05$) compared to October. Concentrations in March were also greater than May (Tukey post hoc, $p\leq 0.05$).

Total number of pesticides detected per sample. As anticipated, there was a similar seasonal pattern in the total number of pesticides per sample with peaks in spring and fall. The median number of compounds per sample was three in March and June and two in all other months (all data, Figure 5.15). Stream water samples with six or more pesticide compounds also typically occurred in spring or summer.

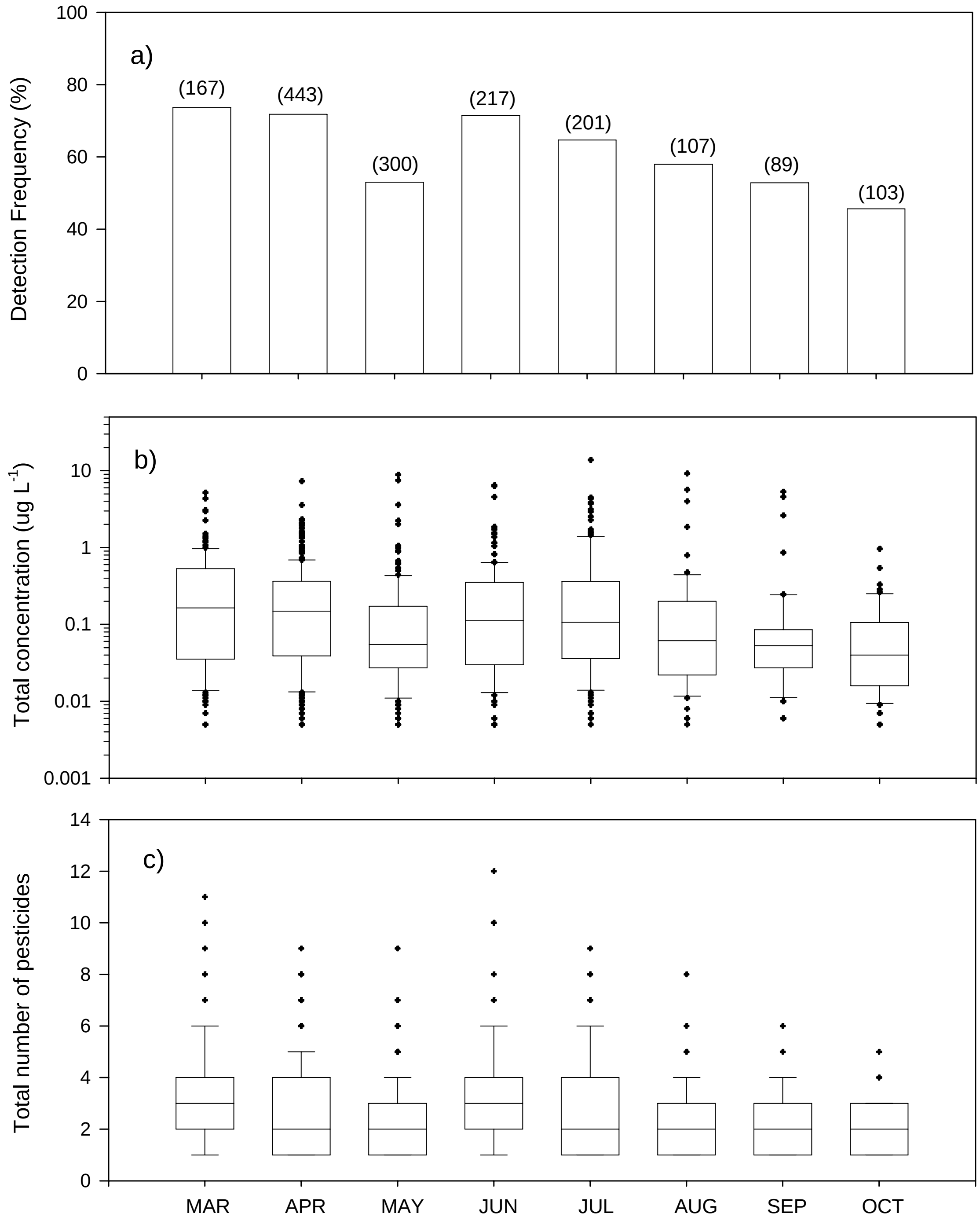


Figure 5.15. Total pesticide detection frequency (%) (a), total pesticide concentration (µg L^{-1}) (b), and total number of compounds detected (c) from March to October. All data are shown (68 compound analytical suite). Bracketed values in (a) are the total number of samples collected in each month.

Statistical analysis of the number of pesticides per sample showed a significant difference among months (H=35.119, $p \leq 0.0001$). Values were higher in early spring (March, April) and early to mid-summer (June and July) than in May, August, September, and October (Mann-Whitney, $p<0.05$). The most diverse pesticide mixtures were observed in March (11 compounds per sample) and June (12 compounds per sample).

Seasonal patterns were also explored within each of the agricultural intensity categories. Patterns were similar to those observed for the entire data set; however, there were no seasonal differences in low and moderate intensity categories.

Seasonal patterns for a subset of individual pesticide compounds are explored in the 'Individual compounds' section of the report.

Interannual Variation

Total pesticide detection frequency, concentration, and number of compounds per sample. Between 1999 and 2006, the highest number of pesticide samples were collected in 2005 (n=253), and the least were collected in 2002 (n=159) (Table 5.16).

Among years there was some variation in the total pesticide detection frequency with the highest proportion of pesticide occurrences in 2000 (70%) and fewest in 2005 (55%). In other years, detections ranged from 57 to 64% (Table 5.16). A potential downward trend in total detection frequency could be emerging for the suite of 40 pesticides monitored every year from 1999 to 2006. However, this pattern is dampened when all 68 compounds are included in the analysis.

Median and total pesticide concentrations were more variable among years than were detection frequencies or the number of compounds detected per sample (Table 5.16 and Figures 5.16 and 5.17). Total pesticide concentrations in moderate intensity watersheds were quite variable among years with median values showing 3-fold increases from 1999 to 2000 (0.031 µg L^{-1} to 0.113 µg L^{-1}, respectively) and again from 2001 to 2002 (0.037 µg L^{-1} to 0.124 µg L^{-1}, respectively) (Table 5.16). Higher variability in pesticide concentrations may be related to interannual fluctuations in moisture patterns and stream flow volumes.

Potential downward trends in total concentrations in high intensity and irrigated watersheds are halted with increases in values again in 2006 (Figure 5.17).

Table 5.16. Annual summary statistics for total pesticide detection frequency, median concentration, and median number of compounds per sample based on the 40 compound dataset.

	1999	2000	2001	2002	2003	2004	2005	2006
Number of samples								
ALL	179	230	163	159	198	218	253	227
LOW	40	49	43	45	40	50	62	55
MOD	52	60	47	46	50	62	72	56
HIGH	66	79	37	34	59	56	77	64
IRRIG	21	42	36	34	49	50	42	52
Total Pesticide Detection Frequency (%)								
ALL	63	70	64	57	60	60	55	59
LOW	23	39	33	24	18	14	19	24
MOD	50	67	55	46	44	52	47	38
HIGH	89	85	86	74	80	84	66	78
IRRIG	90	83	92	97	86	88	100	96
Total Pesticide Concentration (ug L^{-1})*								
ALL	0.102	0.122	0.083	0.123	0.116	0.082	0.035	0.074
LOW	0.017	0.029	0.030	0.021	0.049	0.046	0.019	0.015
MOD	0.031	0.113	0.037	0.124	0.034	0.026	0.020	0.045
HIGH	0.205	0.157	0.167	0.137	0.169	0.179	0.073	0.196
IRRIG	0.070	0.126	0.109	0.131	0.110	0.055	0.043	0.070
Number of Pesticides Per Sample*								
ALL	3	2	2	2	3	2	2	2
LOW	1	1	1.5	1	2	2	1.5	1
MOD	1	2	2	2	1	1.5	1	2
HIGH	3	3	3	3	3	3	3	3
IRRIG	3	3	3	3	3	2	2	2

* Median values

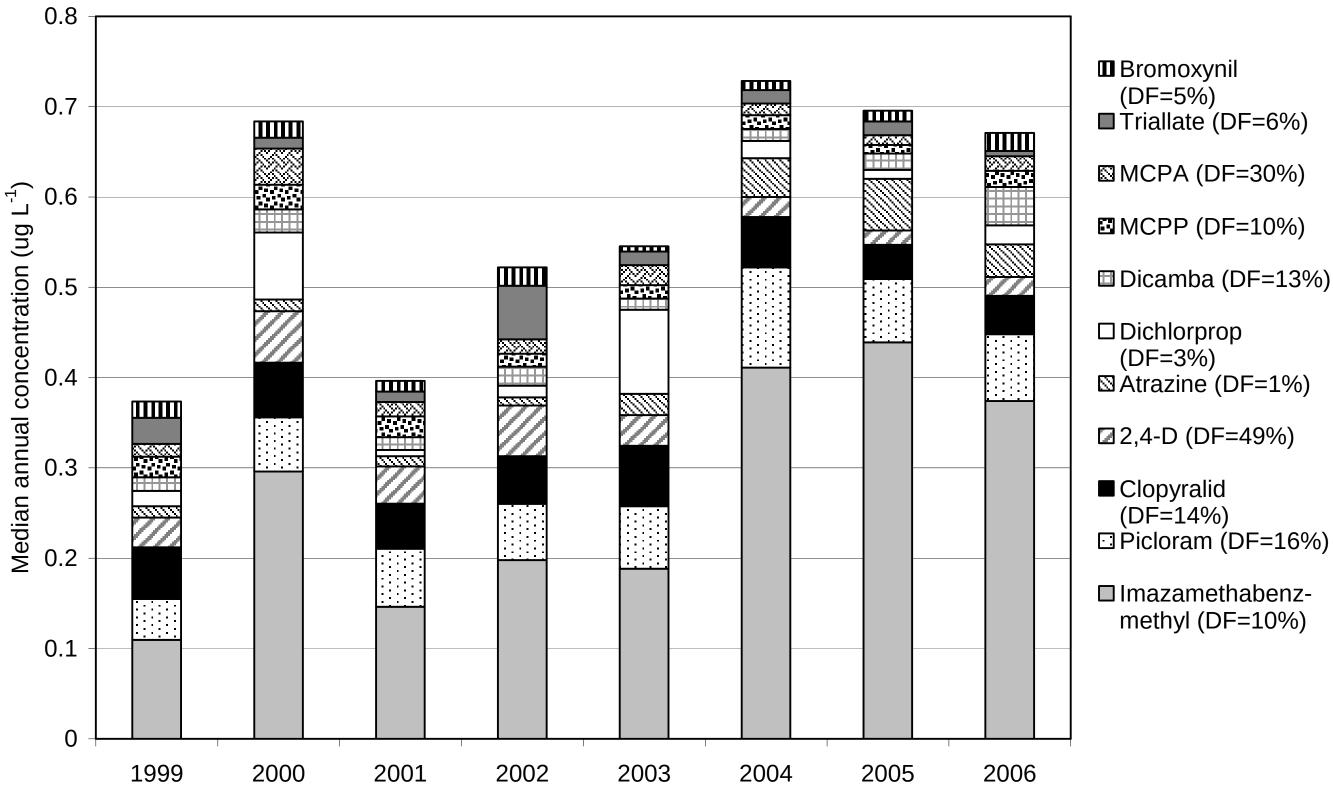
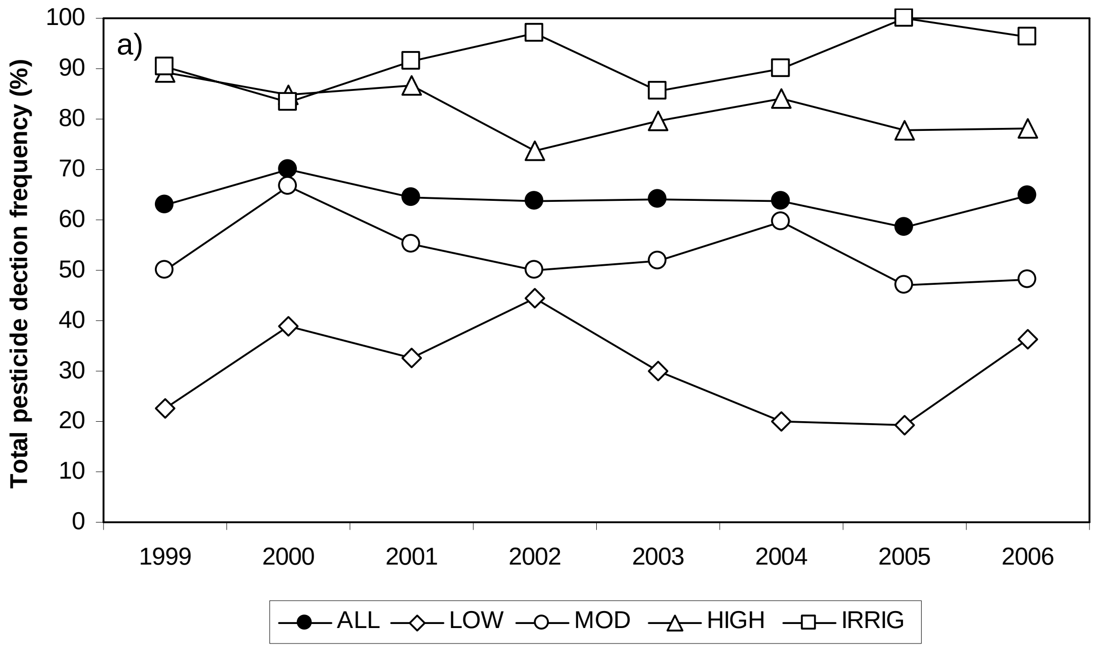


Figure 5.16. Interannual patterns in median annual concentrations of the 11 most ubiquitous compounds from 1999 to 2006. Compounds in the legend are in the order they appear in the bar graph with imazamethabenzmethyl on the bottom and bromoxynil on the top.



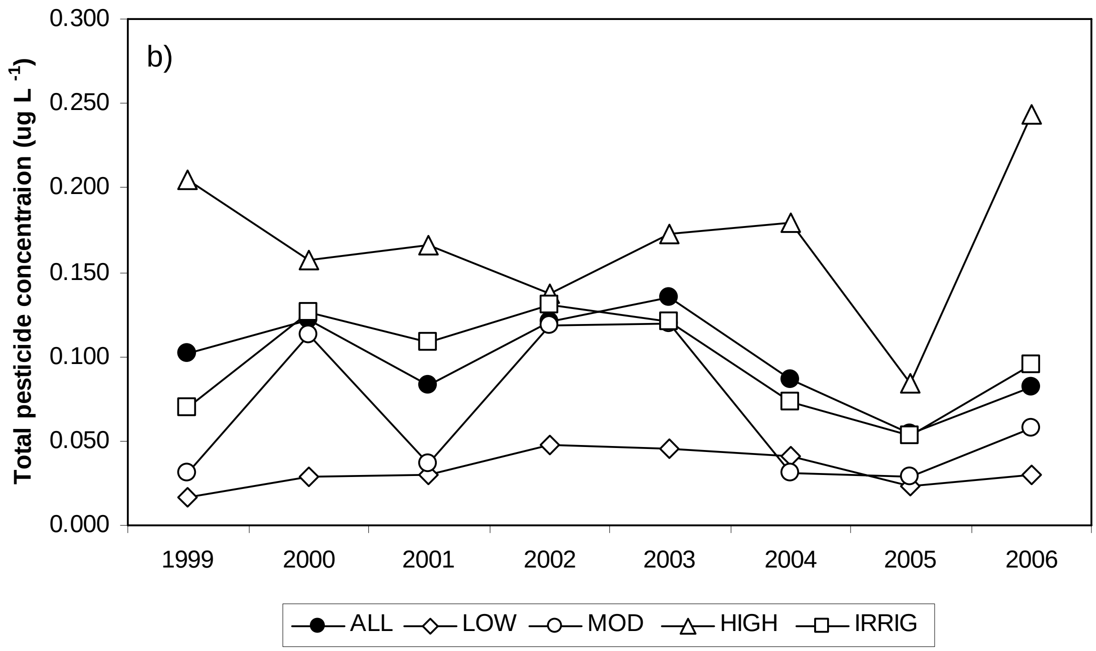


Figure 5.17. Interannual patterns in total pesticide detection frequency (%) (a) and total pesticide concentration (µg L^{-1}, median values) (b) from 1999 to 2006 for the 40 compound dataset. The data shown are for all AESA streams but are grouped by agricultural intensity category.

Individual Compound Discussion

The eight most ubiquitous compounds are outlined below.

2,4-D and MCPA. Both 2,4-D (2,4-dichlorophenoxyacetic acid) and MCPA (2-methyl-4-chlorophenoxyacetic acid) are phenoxy herbicides with similar modes of action. Other phenoxy herbicides detected in this study include MCPP and quinclorac. As mentioned earlier, MCPA had the 2nd highest sales and 2,4-D had the 3rd highest sales in 2003 (~1,000,000 and 685,000 kg, respectively) (Byrtus 2007). Both of these active ingredients are registered for non-agricultural as well as agricultural uses and were detected in all 23 of the AESA streams. In agriculture, these two herbicides are included in many pesticide mixtures. Common trade names include Grazon, Prestige, and FlaxMax DLX. For domestic uses, 2,4-D and mecoprop are the active ingredients in commonly used turf herbicides (Byrtus 2007).

The highest pesticide detection frequencies of 2,4-D were in irrigation return flows ($H = 85.715$, $p < 0.0001$, Figure 5.18), with the highest detections (>80%) in Crowfoot Creek, New West Coulee, and Battersea Drain (Figure 5.19). In dryland streams, 2,4-D detections increased as watershed agricultural intensity increased from low to moderate to high. Within the high intensity category, Wabash Creek and Haynes Creek also had detections of 2,4-D in ~ 80% of samples. With the exception of Battersea Drain, median concentrations of 2,4-D were also highest in these five streams ranging from 0.054 to 0.073 µg L^{-1}(Battersea Median is: 0.034 µg L^{-1}).

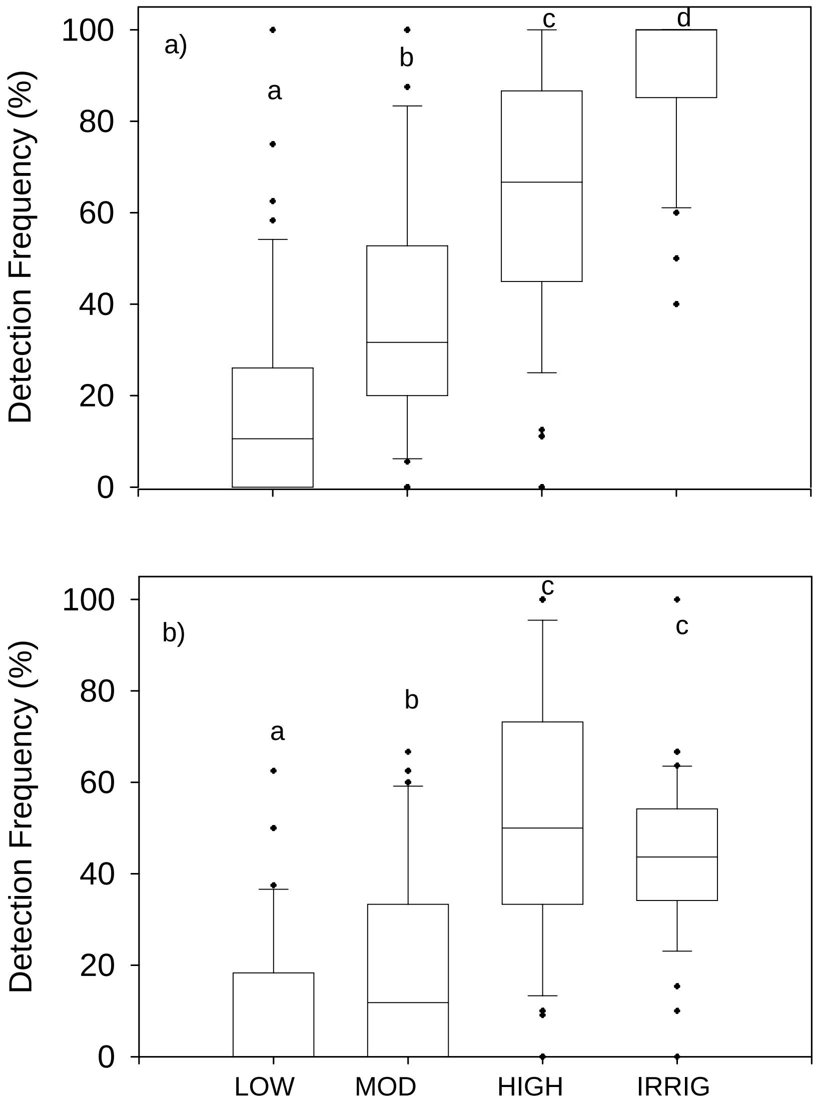


Figure 5.18. Box plots of annual 2,4-D (a) and MCPA (b) detection frequencies by watershed agricultural intensity category. Letters above box plots indicate significant differences among agricultural intensity categories. Different letters are significant from one another at p<0.05.

The lowest detection frequencies of 2,4-D were observed in two low intensity watersheds, Willow Creek and Hines Creek, where 2,4-D was detected in 3 and 4% of samples, respectively. The median concentration in these few samples was high and similar to five watersheds described earlier; however, the infrequent detections result in less of a water quality concern.

MCPA detection frequencies were highest in both high intensity dryland and irrigated streams (H= 68.611, p< 0.0001 Figure 5.18). The five streams with detection frequencies (~60%) were all high intensity watersheds: Stretton Creek, Haynes Creek, Wabash Creek, Threehills Creek, and Renwick Creek. All of the streams, with the exception of Wabash Creek, were located in the Aspen Parkland. Of the irrigation return flows, MCPA detections were highest in New West Coulee (~50%). The highest median concentration values were observed in low (Prairie Blood

Coulee), moderate (Kleskun Drain), and high intensity (Strawberry and Ray Creeks) streams, none of which had the highest detection frequencies (Figure 5.19).

From 1998 to 2003, pesticide sales volumes for 2,4-D remained virtually unchanged (+1.5%, Table 5.1, pg. 10), while MCPA sales volumes increased substantially (+23.9%, Table 5.1, pg. 10). From 1999 to 2006, detection frequencies did not differ significantly in any of the years of monitoring (2,4-D: H=1.842, p=0.968; MCPA: H=4.957, p=0.665) nor within any of the agricultural intensity categories (Figure 5.20).

In individual streams, a potential downward trend in 2,4-D concentrations was observed in Haynes, Ray, Renwick, and Wabash Creeks. Each stream had higher concentrations earlier in the monitoring record (1999 to 2001) (Figure 5.21). In contrast, MCPA concentrations did not show any notable increase in any of the AESA streams despite the increase in pesticide sales.

Seasonally, detection frequencies of 2,4-D were highest in spring and summer and significantly lower in fall (H=8.775, p=0.012); concentrations were only significantly higher in summer (H=10.3, p=0.006; Figure 5.34, pg. 5-69). MCPA detection frequencies were significantly higher in summer, followed by spring and fall (H=83.02, p<0.0001): concentrations were only statistically higher in summer (H=21.69, p<0.001). Of the eight most frequently detected compounds, MCPA was the only one to show a summer concentration peak.

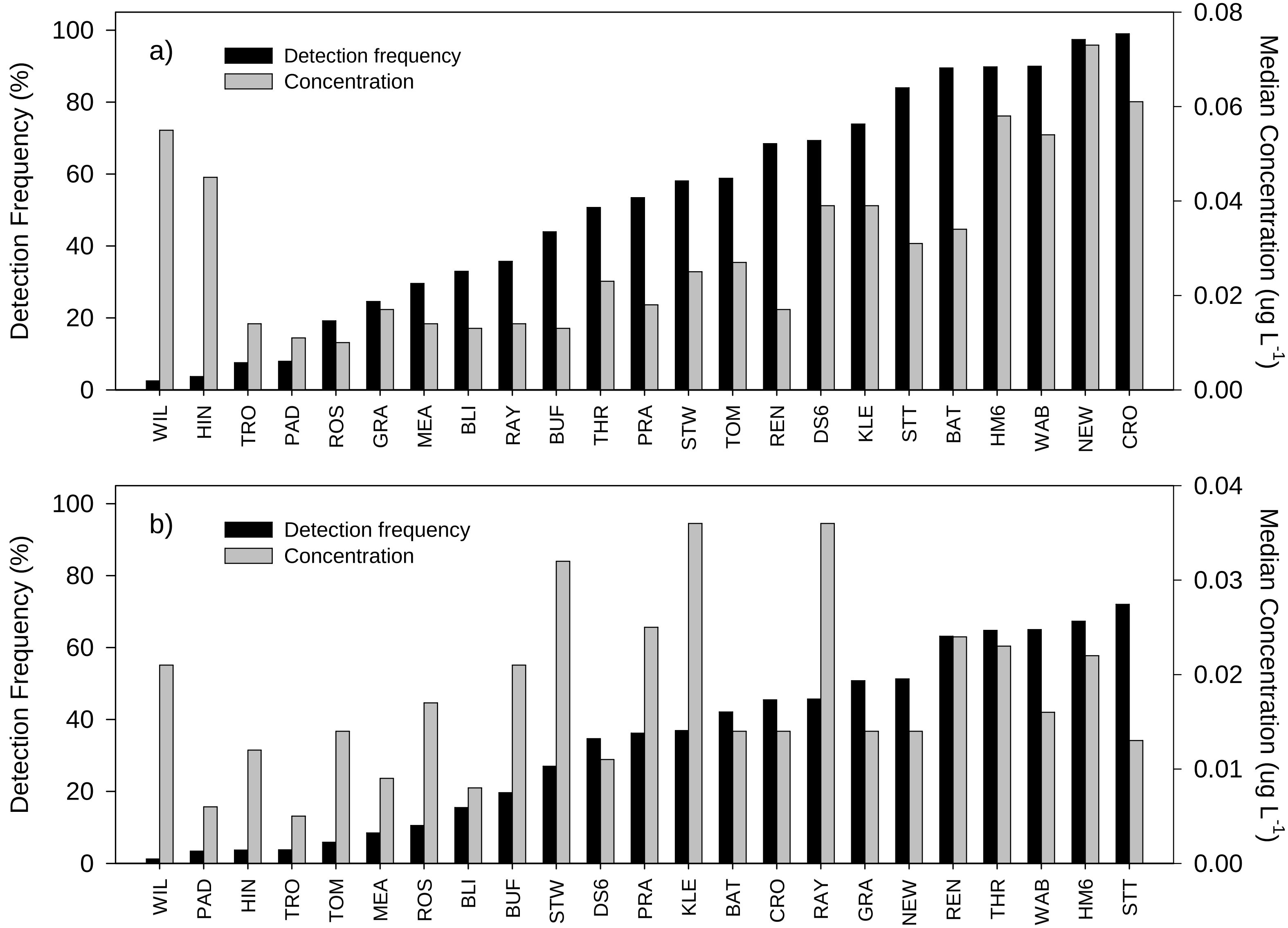


Figure 5.19. Patterns in 2,4-D (a) and MCPA (b) detection frequencies and pesticide concentrations in AESA watersheds. For full stream names see Table 5.4.

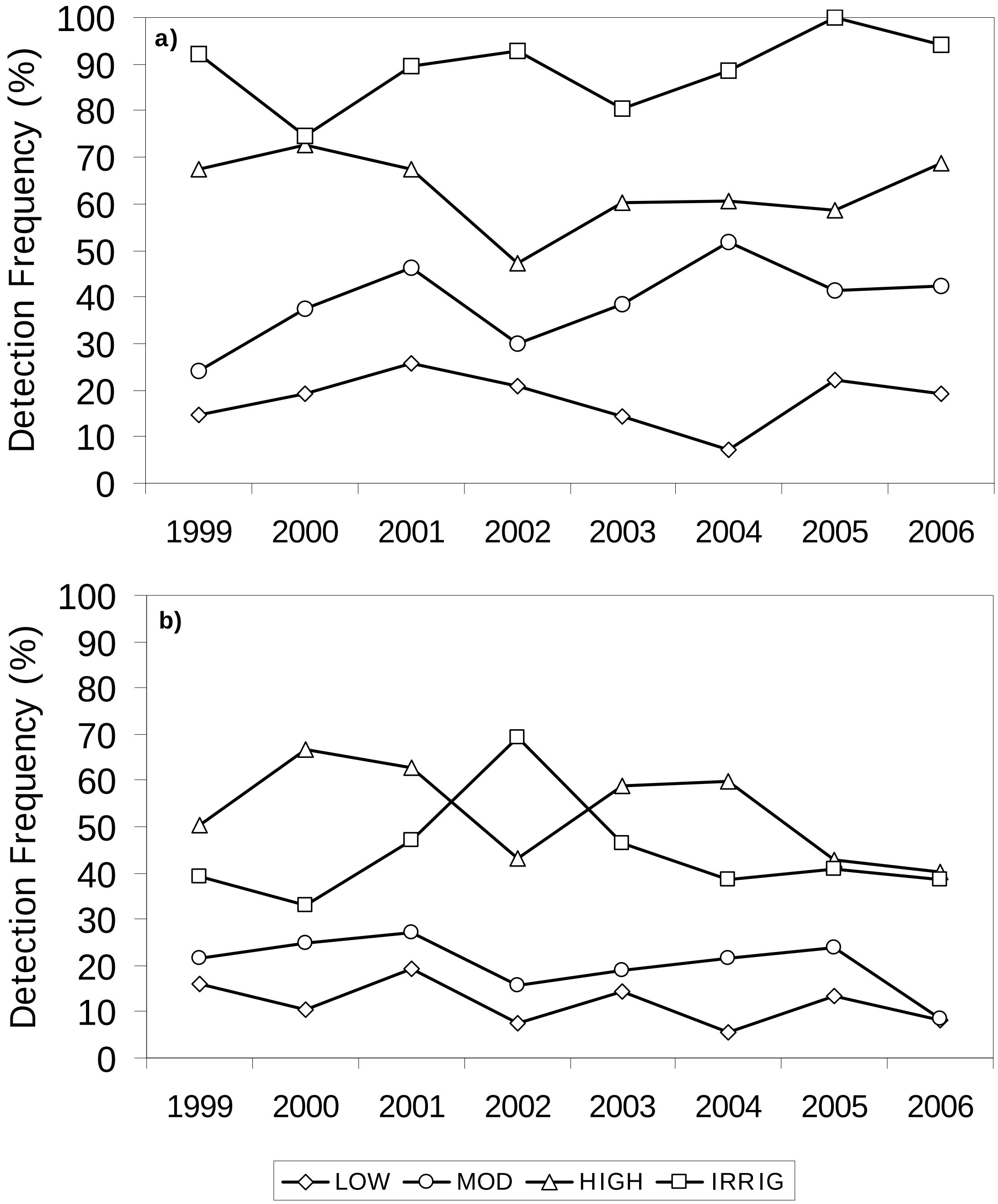


Figure 5.20. Changes in 2,4-D (a) and MCPA (b) detection frequencies over time.

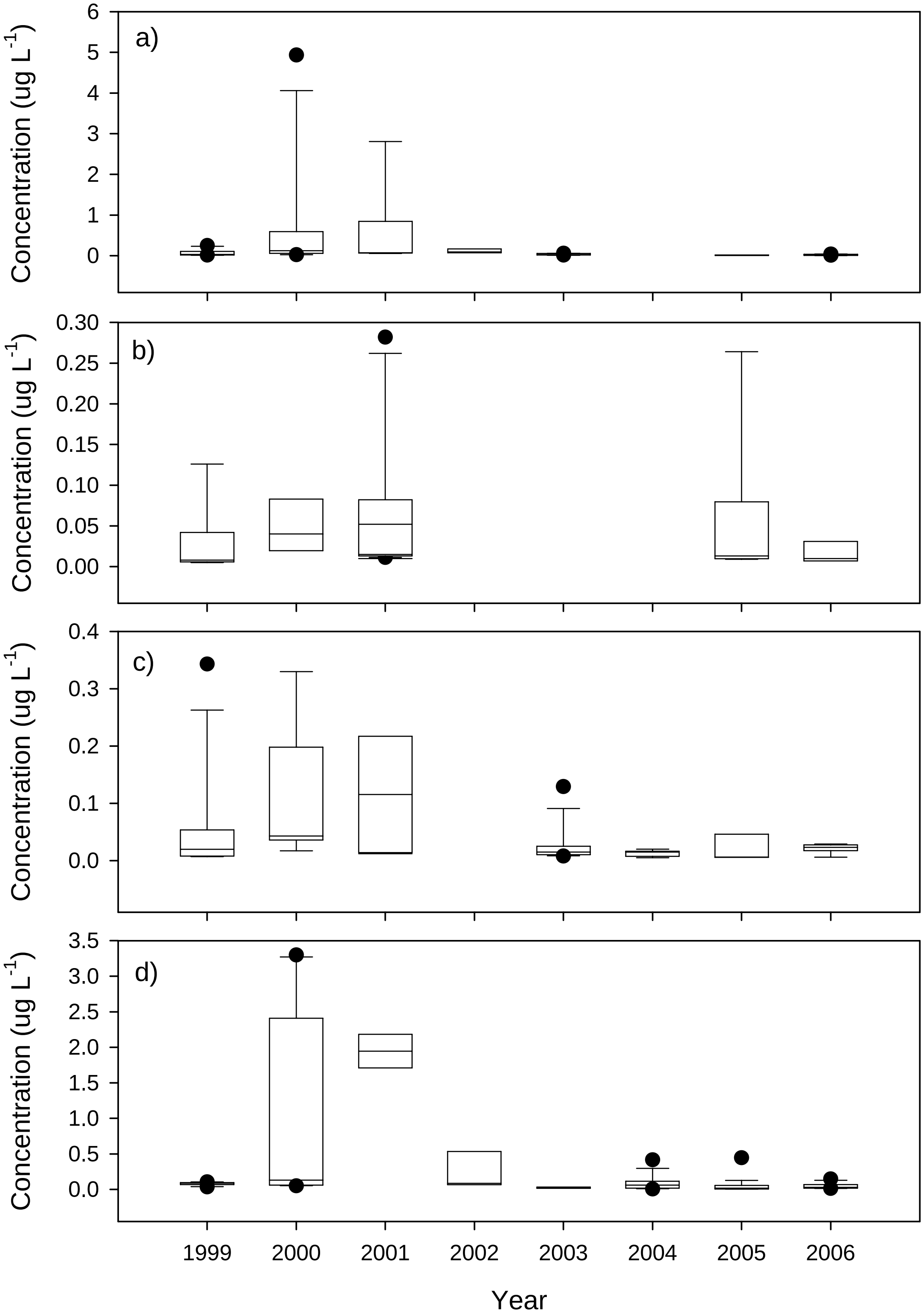


Figure 5.21. Concentrations of 2,4-D in Haynes Creek M6 (a), Ray Creek (b), Renwick Creek (c), and Wabash Creek (d) from 1999 to 2006.

Picloram and Triclopyr. Both picloram and triclopyr are carboxylic acid herbicides that are used to control weeds (chamomile, knapweed, Canada thistle, toadflax, and clover) and brush (alder, birch, maple, poplar, and spruce) on pasture and rangeland, roadsides, and utility rights of way (Alberta Agriculture and Food 2007). Both were widely detected in AESA watersheds, with picloram detections in 19 of 23 streams and triclopyr detections in 16 of 23 streams.

Picloram detections were significantly higher in moderate and high intensity agricultural watersheds than the low and irrigated watersheds (H=27.908, p<0.0001, Figure 5.22). Streams with highest detection frequencies of picloram typically had high intensity agriculture: Strawberry (58%), Haynes (41%), Renwick (35%), and Wabash Creeks (62%) (Figure 5.23). Detection frequencies and maximum concentration in Kleskun Drain were the highest of all streams at 80% and 13.407 µg L^{-1}, respectively. Road construction presents a likely source for picloram in this basin. However, it is notable that the highest median concentrations were seen in both watersheds in the Peace River Lowland Ecoregion, Kleskun Drain (0.564 µg L^{-1}) and Grande Prairie Creek (0.232 µg L^{-1}). Two streams with typically low pesticide detection, Blindman River and Rose Creek, had intermediate frequencies of picloram detections.

Triclopyr detection frequencies did not vary significantly among watershed categories (H= 4.014, p=0.260), though the highest frequencies tended to occur in several high intensity watersheds: Strawberry (55 %), Stretton (43 %), Wabash (40 %), and Haynes Creeks (36%). Both picloram and triclopyr were infrequently detected in irrigation return flows. This is likely attributable to land cover in the grassland ecoregion of the province and less need for brush control.

The median concentration of triclopyr observed for all detections was relatively low at 0.036 µg L^{-1}. The highest concentrations were observed in Kleskun Drain (median = 0.406 µg L^{-1}), though detections occurred only in 2002 and 2003. Other Boreal ecoregion watersheds had detectable levels of triclopyr in the 5 years where it was monitored: Strawberry Creek, Wabash Creek and Blindman River. Triclopyr is one of the most frequently used compounds in the commercial and industrial sector (Byrtus 2007) and may be routinely used for weed control along roadsides in this part of the province.

Triclopyr and picloram both showed significantly higher detection frequencies in spring compared to summer and fall (triclopyr: H=22.96, p<0.0001, picloram: H=18.44, p<0.0001, Figure 5.34, pg. 5-69). Picloram concentrations did not vary significantly among seasons, while triclopyr concentrations were highest in summer (H=6.91, p=0.032).

Interannual variation in picloram detections was significant (H=24.45, p=0.001, Figure 5.24a). By agricultural intensity category, interannual variation was only significant in high intensity watersheds (H=18.852, p=0.009). A decline in picloram detections was observed in the majority of high intensity watersheds from earlier to later in the monitoring record (Figure 5.24b). However, substantial increases in detections in 2006 in Strawberry and Wabash Creeks, and smaller increases in Stretton and Haynes Creeks, lead to questions about whether the pattern will continue.

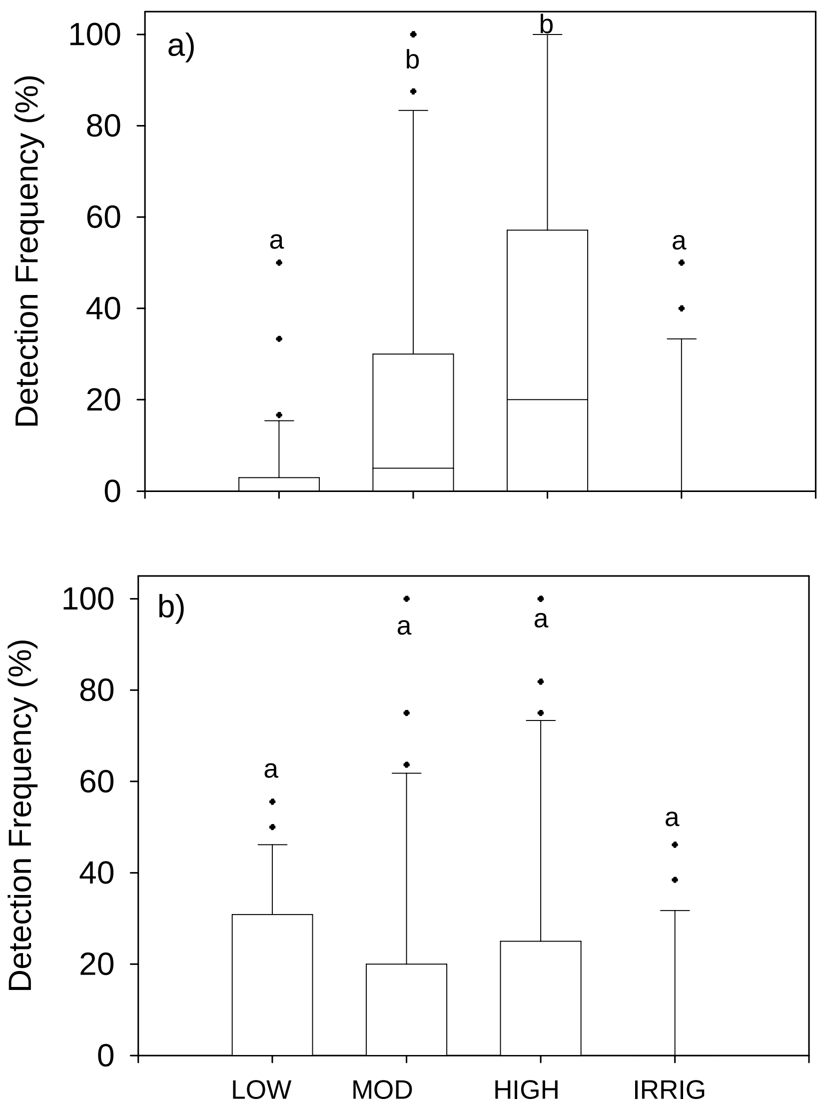


Figure 5.22. Box plots of annual picloram (a) and triclopyr (b) detection frequencies by watershed agricultural intensity category. Letters indicate significance differences between agricultural intensities ($p<0.05$).

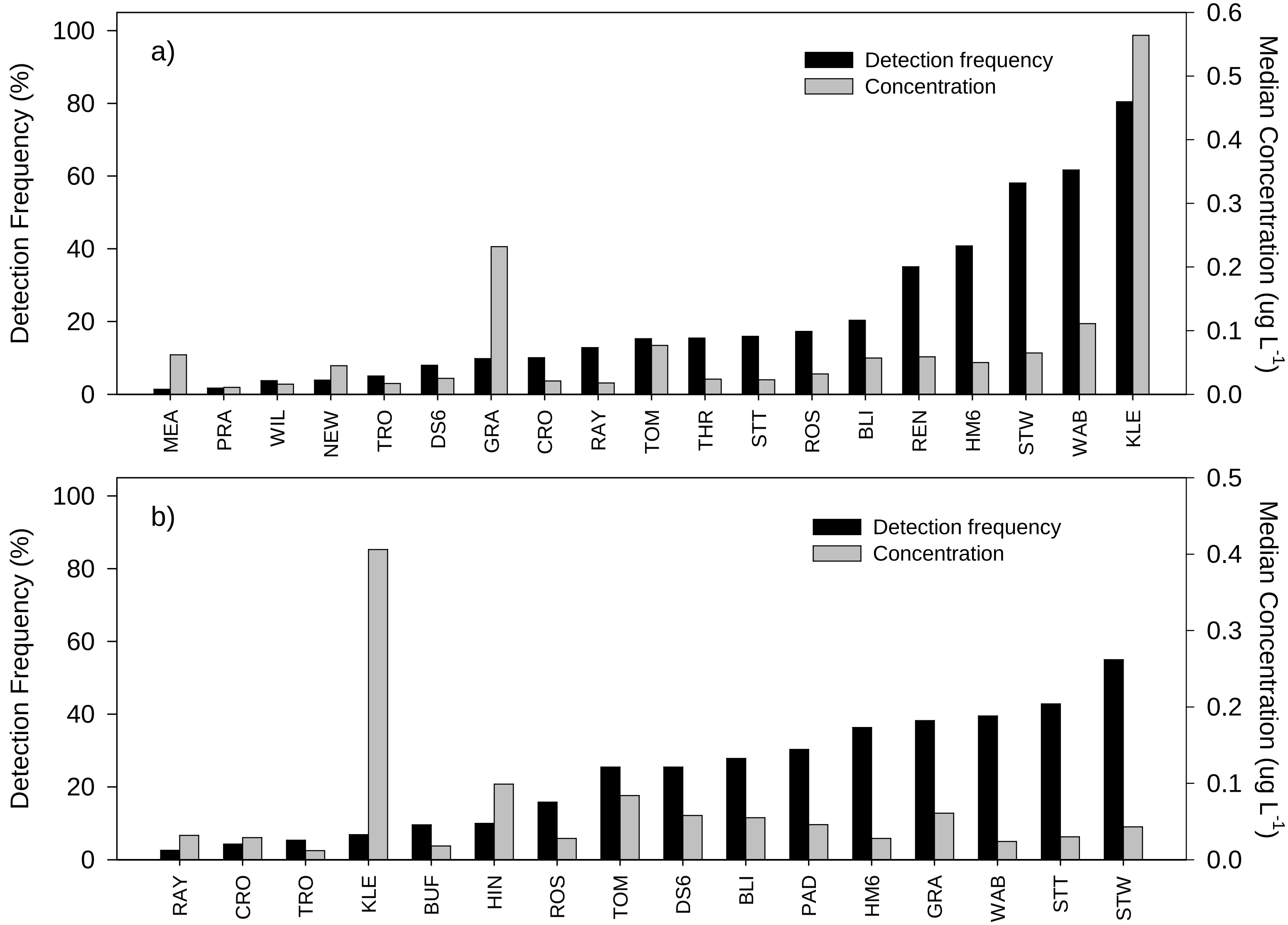
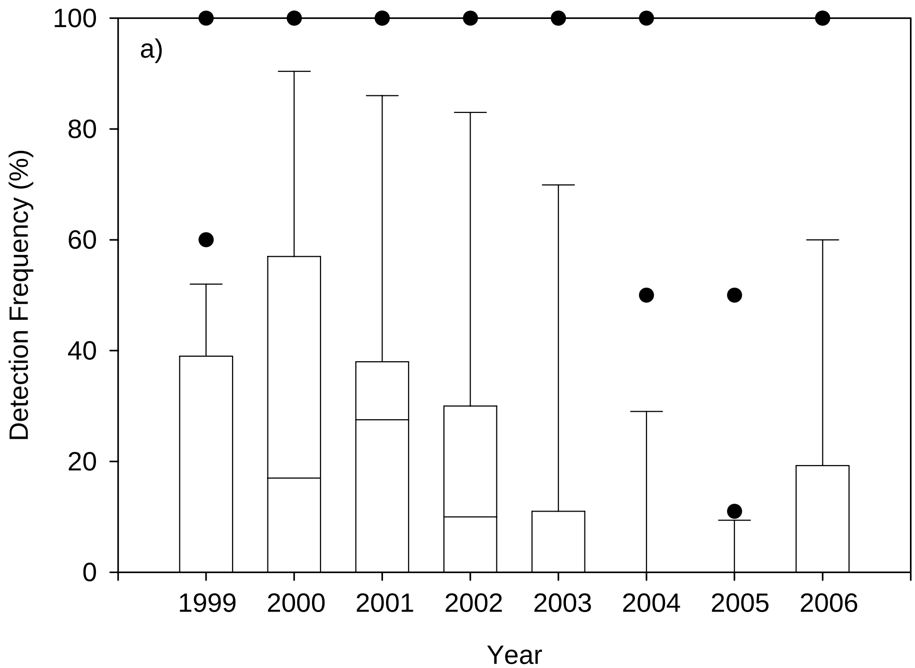


Figure 5.23. Patterns in picloram (a) and triclopyr (b) detection frequencies (primary y-axis) and pesticide concentrations (secondary y-axis) in AESA watersheds from 1999 to 2006.



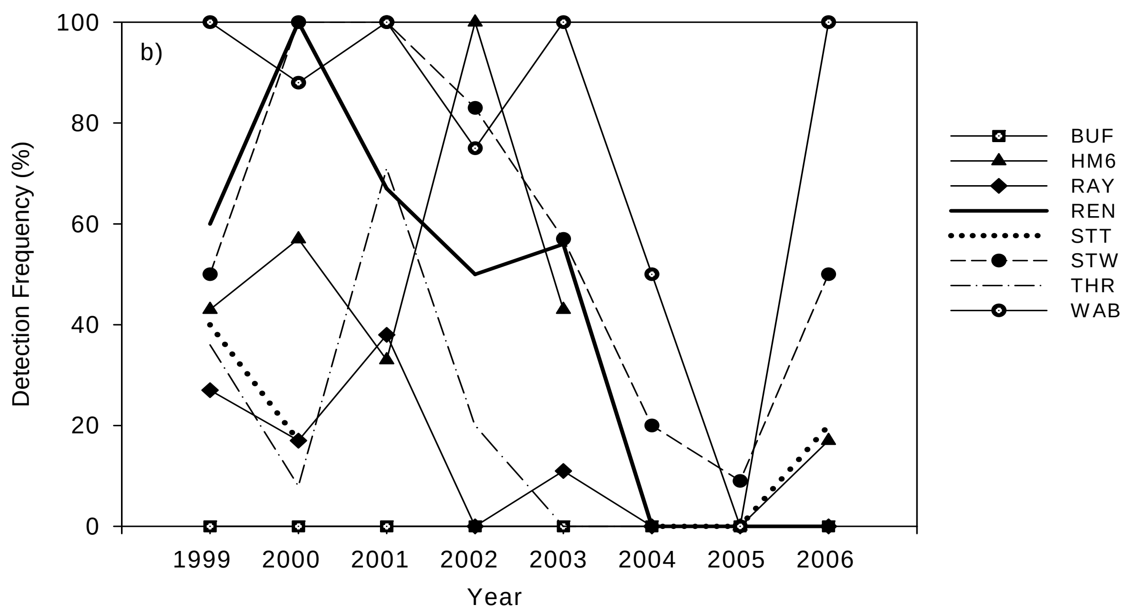


Figure 5.24. Patterns in the detection frequency of picloram over time in all watersheds (a) and in high intensity watersheds (b). BUF= Buffalo Creek; HM6= Haynes Creek M6; RAY= Ray Creek; REN= Renwick Creek; STT= Stretton Creek; STW= Strawberry Creek; THR= Threehills Creek; WAB= Wabash Creek.

Clopyralid. Clopyralid is classified as a carboxylic acid herbicide similar to picloram and triclopyr. It is used for broadleaf weed control in cereal, canola, and flax seed crops and is very mobile and moderately persistent in soils (Anderson 2005; Alberta Agriculture and Food 2007). Clopyralid can be applied in mixtures with MCPA to cereals and flax and to glyphosate-tolerant canola (Alberta Agriculture and Food 2007).

Clopyralid was detected in 17 of 23 AESA streams. Detection frequency was highest in high intensity watersheds followed by irrigated watersheds and moderate intensity watersheds (H=24.333, $p \leq 0.0001$, Figure 5.25). The highest observed clopyralid detection frequencies were in watersheds with either moderate or high intensity dryland agriculture. In both Haynes Creek and Kleskun Drain, detection frequencies were ~70%, which were notably higher than other streams. The next highest detection frequencies were in Wabash, Renwick, Grande Prairie, and Threehills Creeks with detections ranging from 28 to 38% (Figure 5.26).

The median concentration for all samples was 0.049 µg L^{-1}. The maximum concentration observed was 1.79 µg L^{-1} in Kleskun Drain (in April 2002). Median concentrations were highest for Kleskun Drain and Drain S-6. Detection frequency in Drain S6 was relatively low.

Detections of clopyralid were significantly higher in spring and summer than in fall (H=16.77, $p<0.0001$, Figure 5.34, pg.5-69). Concentrations did not differ significantly among seasons; though in Kleskun Drain there were notably higher concentrations in April 2002 and 2003 and October 2004.

Interannual changes in clopyralid detections were not statistically significant. No qualitative trends were observed in individual streams, with the exception of Wabash Creek (Figure 5.27) where clopyralid detections appear to have peaked in 2003 and concentrations appear to have been declining since 2001.

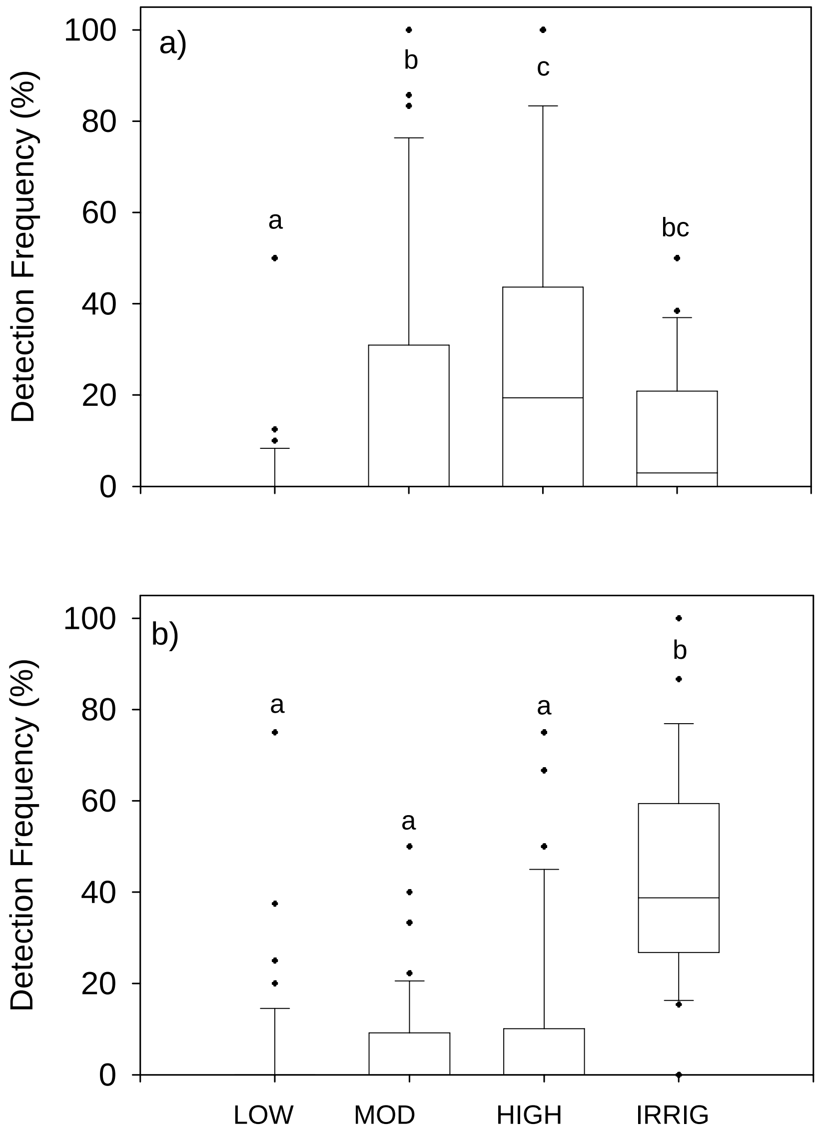


Figure 5.25. Box plots of annual clopyralid (a) and dicamba (b) detection frequencies by watershed agricultural intensity category. Letters indicate significance between agricultural intensities ($p<0.05$).

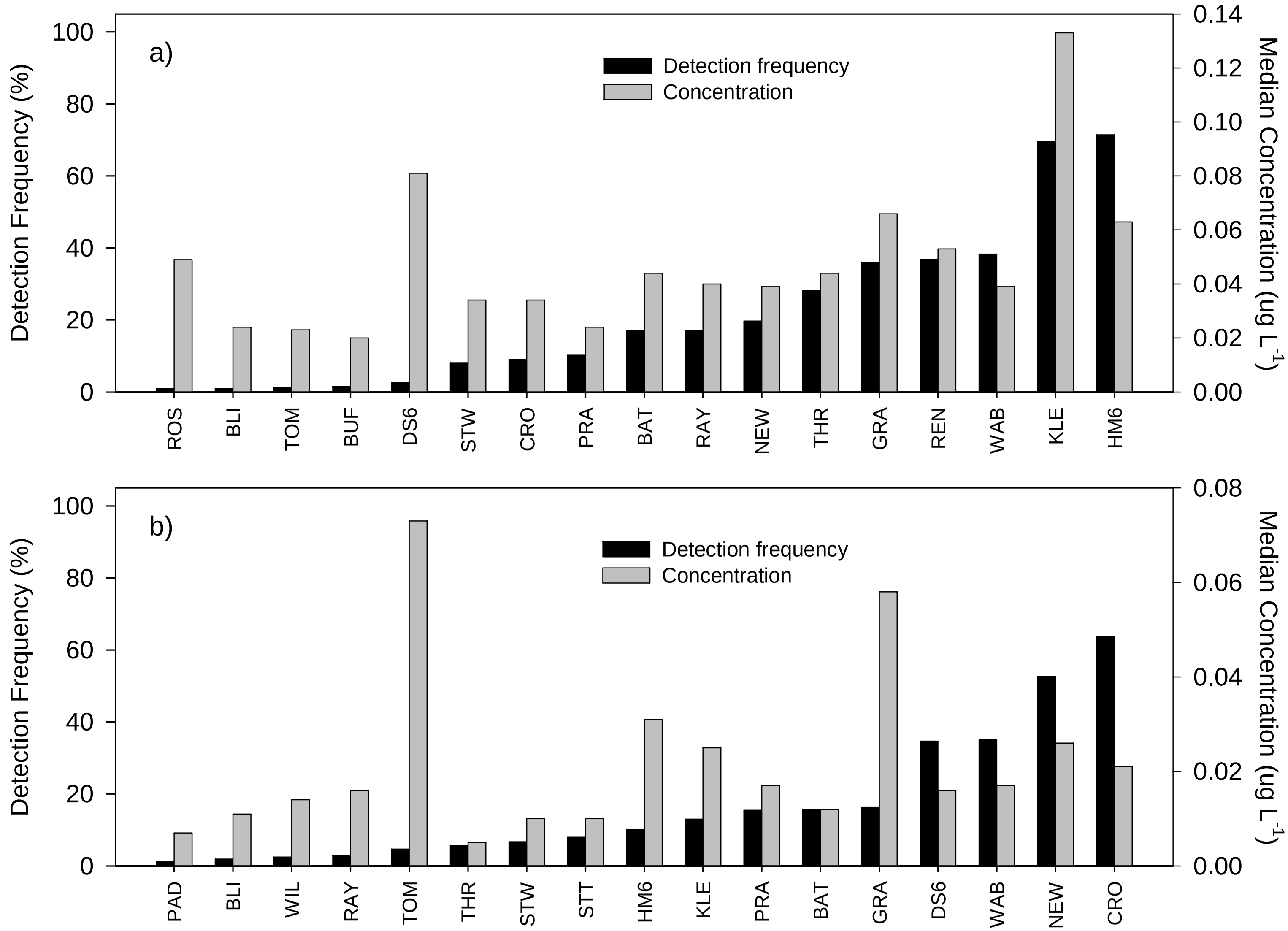


Figure 5.26. Patterns in clopyralid (a) and dicamba (b) detection frequencies (primary y-axis) and pesticide concentrations (secondary y-axis) in AESA watersheds. Watersheds on the x-axis are abbreviated. For full stream names see Table 5.4.

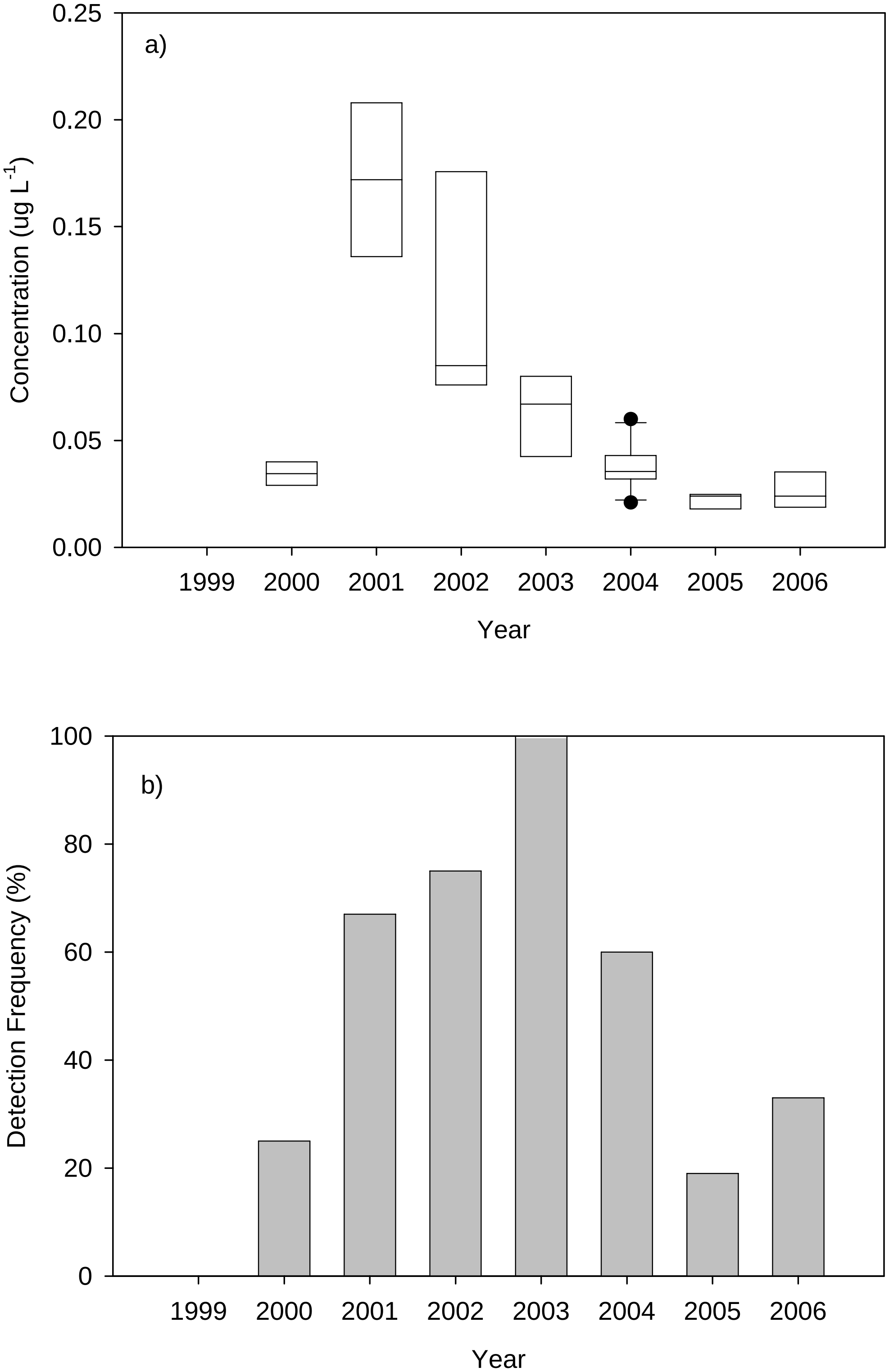


Figure 5.27. Interannual patterns in clopyralid concentrations (a) and detection frequency (b) in Wabash Creek.

Dicamba. Dicamba is a benzoic acid herbicide used in the agricultural sector for weed control in cereals, corn, and pastures but also has non-crop applications (Alberta Agriculture and Food 2007). Dicamba is registered in mixes with 2,4-D and MCPA for cereals, 2,4-D for pasture, rangeland and non-crop applications, and 2,4-D and glyphosate for chemical fallow/stubble/reduced tillage for annual weeds and roadside vegetation control (Alberta Agriculture and Food 2007). The pesticide movement rating is very high (Table 5.12., pg. 5-36).

Dicamba was detected in 17 of the 23 AESA streams. Pesticide detection frequency was significantly higher in irrigated watersheds than in any of the dryland watershed categories (H=67.425, $p \leq 0.0001$, Figure 5.25). The streams with high detection frequencies (>30%) include three irrigated streams (Crowfoot Creek, New West Coulee, and Drain S6) and one high intensity dryland stream (Wabash Creek) (Figure 5.26).

The median concentration for all samples was 0.017 µg L^{-1}, which is about twice the water quality guideline value for the protection of irrigation water (0.006 µg L^{-1}). Streams with the highest median concentrations include Tomahawk Creek, Grande Prairie Creek, Haynes Creek, New West Coulee, and Kleskun Drain. The maximum observed concentration was 1.134 µg L^{-1} in Tomahawk Creek in 2004. Detection frequencies need to be examined together with median concentrations, particularly in the case of Tomahawk Creek where detection frequencies were relatively low. In contrast, New West Coulee had high detection frequencies and high median and peak concentrations.

Detections of dicamba did not show strong seasonality; however, concentrations were significantly higher in summer months (H=7.016, p=0.030, Figure 5.34, pg. 5-69).

Interannual differences were not observed in dicamba detection frequencies for all streams combined or within agricultural intensity categories. Noteworthy patterns in dicamba detections and concentrations were observed in Crowfoot and Wabash Creeks (Figure 5.28 and 5.29). Both creeks are subject to urban influence; thus, patterns observed may also be influenced by changes in domestic uses (lawns, turf, and golf courses).

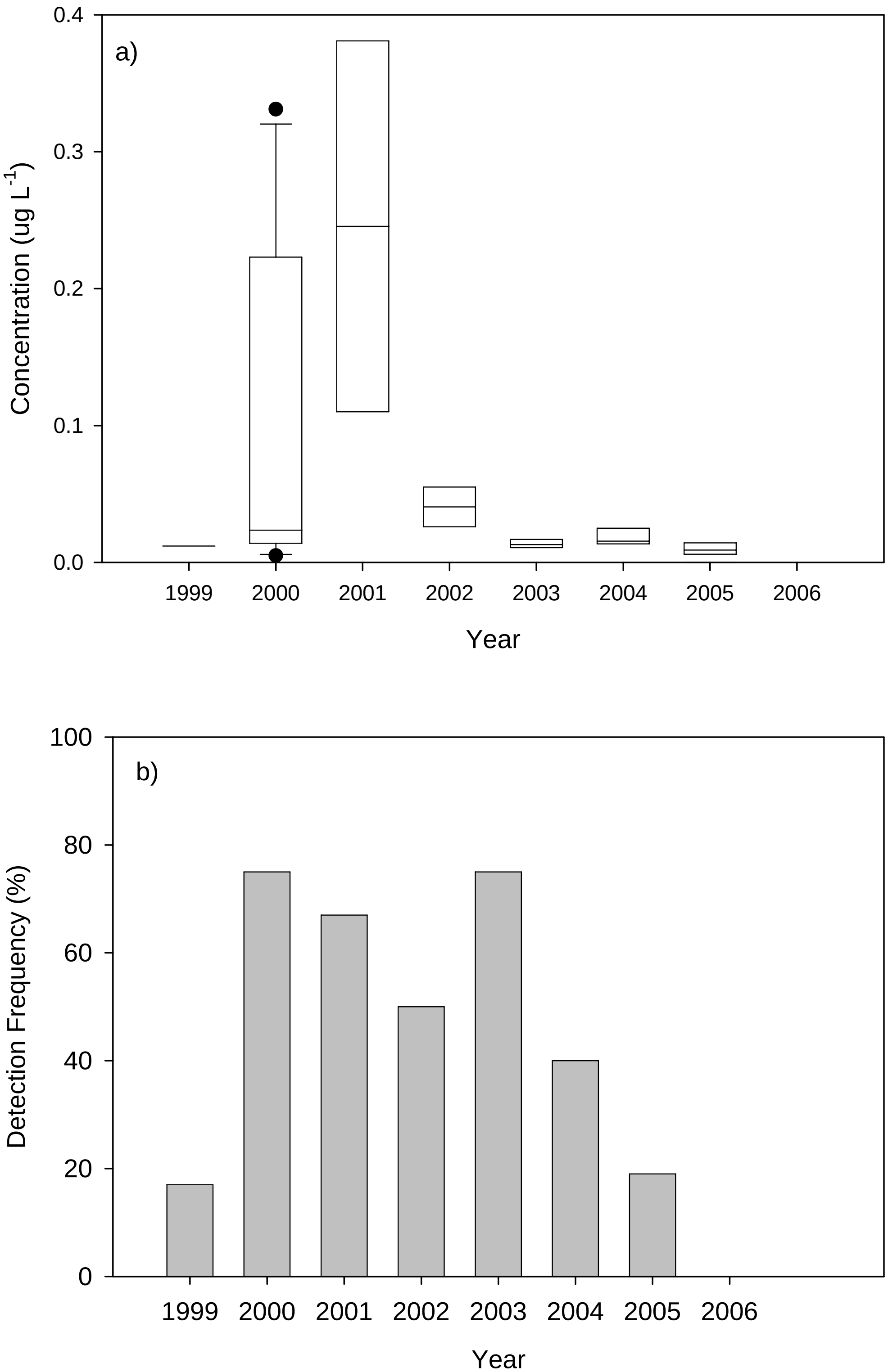


Figure 5.28. Interannual patterns in dicamba concentrations (a) and detection frequency (b) in Wabash Creek.

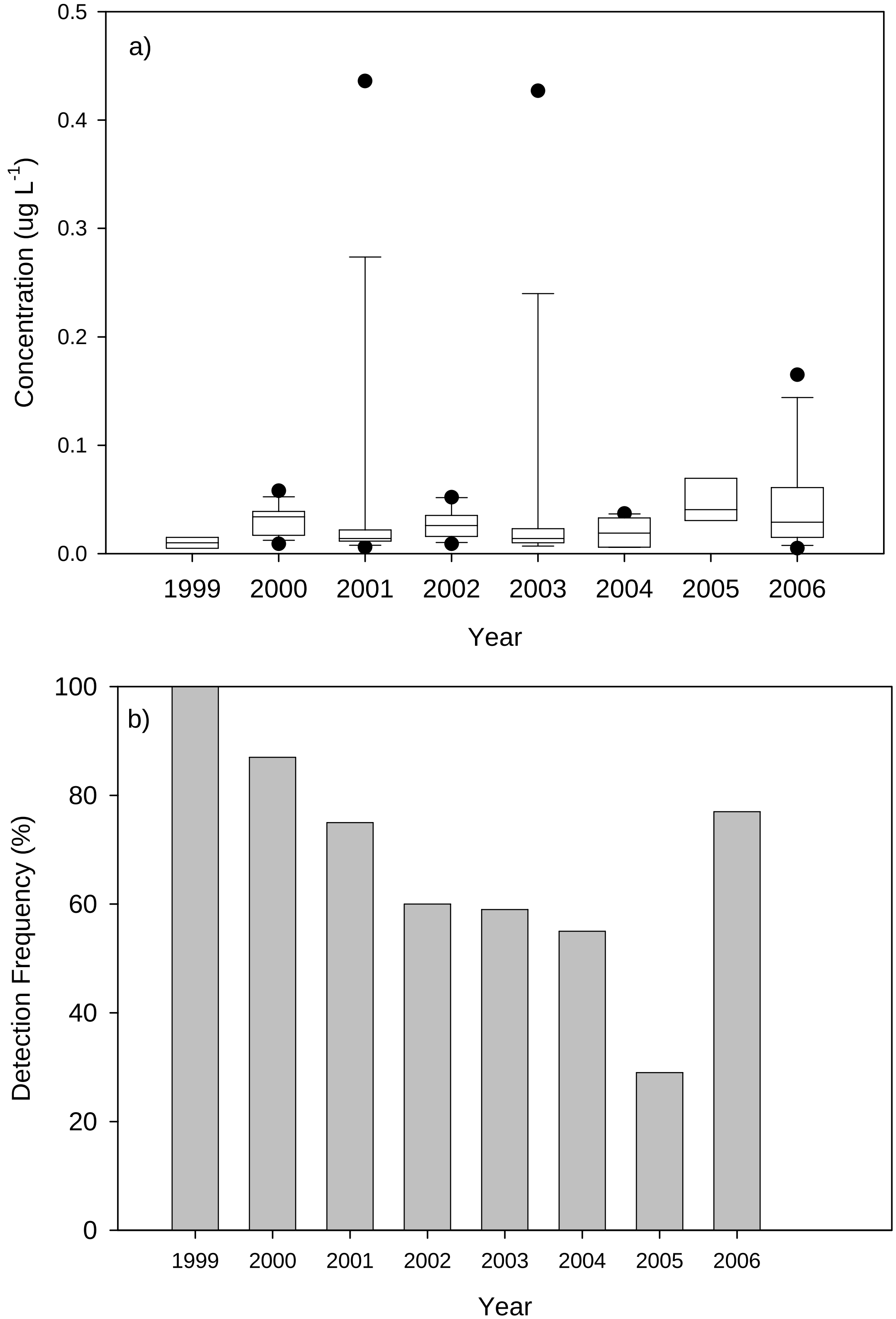


Figure 5.29. Interannual patterns in dicamba concentrations (a) and detection frequency (b) in Crowfoot Creek.

MCPP. MCPP (mecoprop, methylchlorophenoxypropionic acid) is a phenoxy herbicide used for weed control in cereal crops as well as for lawns and turf. MCPP does not readily adsorb to soils and is likely to be mobile in terrestrial environments (USEPA 2007). The pesticide movement rating is high (Table 5.12., pg. 5-36).

MCPP was detected in 18 of the 23 AESA streams. On average, the highest pesticide detection frequencies were in irrigated streams, followed by high, moderate, and low intensity watersheds ($H=20.539$, $p \leq 0.0001$, Figure 5.30). Higher values in the irrigated and high intensity watershed categories were driven by high detection frequencies in Crowfoot Creek (60%) and Wabash Creek (85%), respectively (Figure 5.31). Both streams are potentially under the influence of municipal discharges, which may account for the higher detections. In other streams with detections, MCPP occurred in <13% of samples (Haynes Creek, Grande Prairie Creek, New West Coulee, Prairie Blood Coulee, and 12 others) (Figure 5.31).

The median concentration for all samples was 0.019 µg L^{-1}. The two maximum concentrations observed in Wabash Creek in June 2000 (1.558 and 2.068 µg L^{-1}) appear to be isolated peaks. Low level concentrations of MCPP in Crowfoot and Wabash Creeks do not appear to show any seasonal or temporal patterns. No guidelines currently exist for this compound.

There was no interannual variation in MCPP detection frequency from 1999 to 2006; however, patterns in detections suggest potential declines in Crowfoot and Wabash Creek (Figure 5.32). MCPP detections and concentrations did not vary significantly among seasons (Figure 5.34, pg. 5-69).

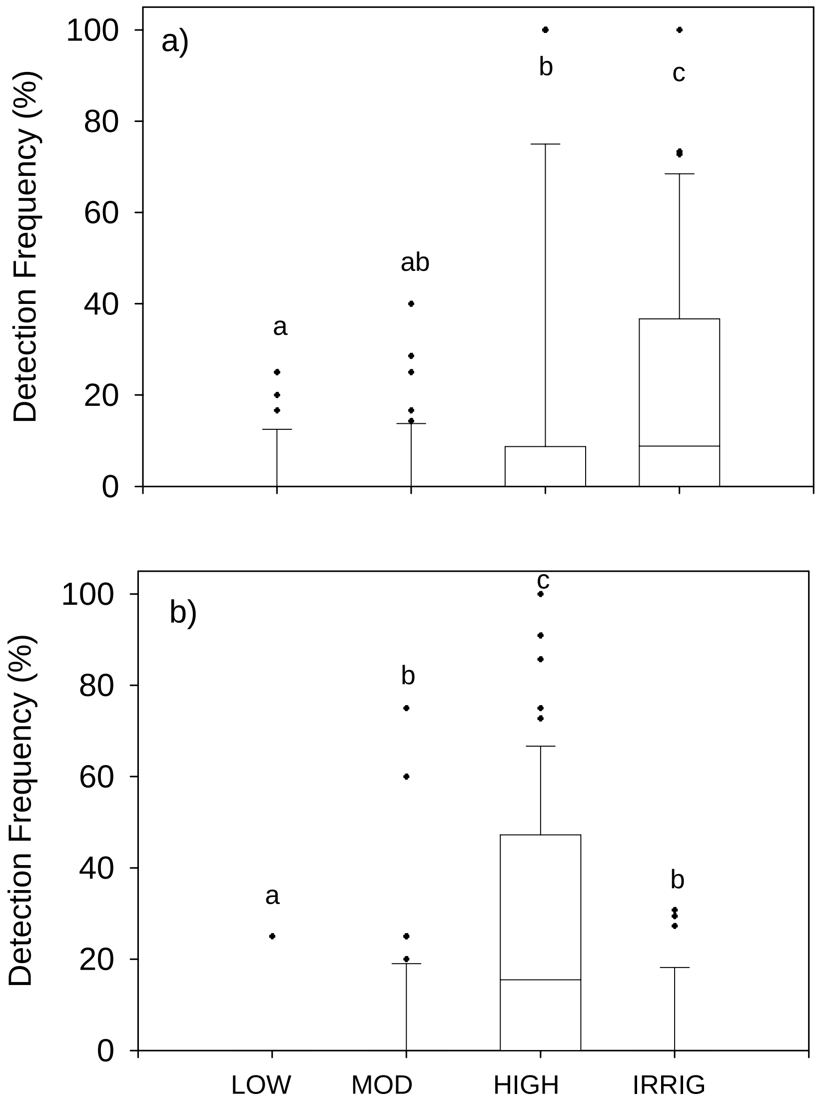


Figure 5.30. Box plots of annual MCPP (a) and imazamethabenz-methyl (b) detection frequencies by watershed agricultural intensity category. Letters denote significant differences among agricultural intensity categories ($p<0.05$).

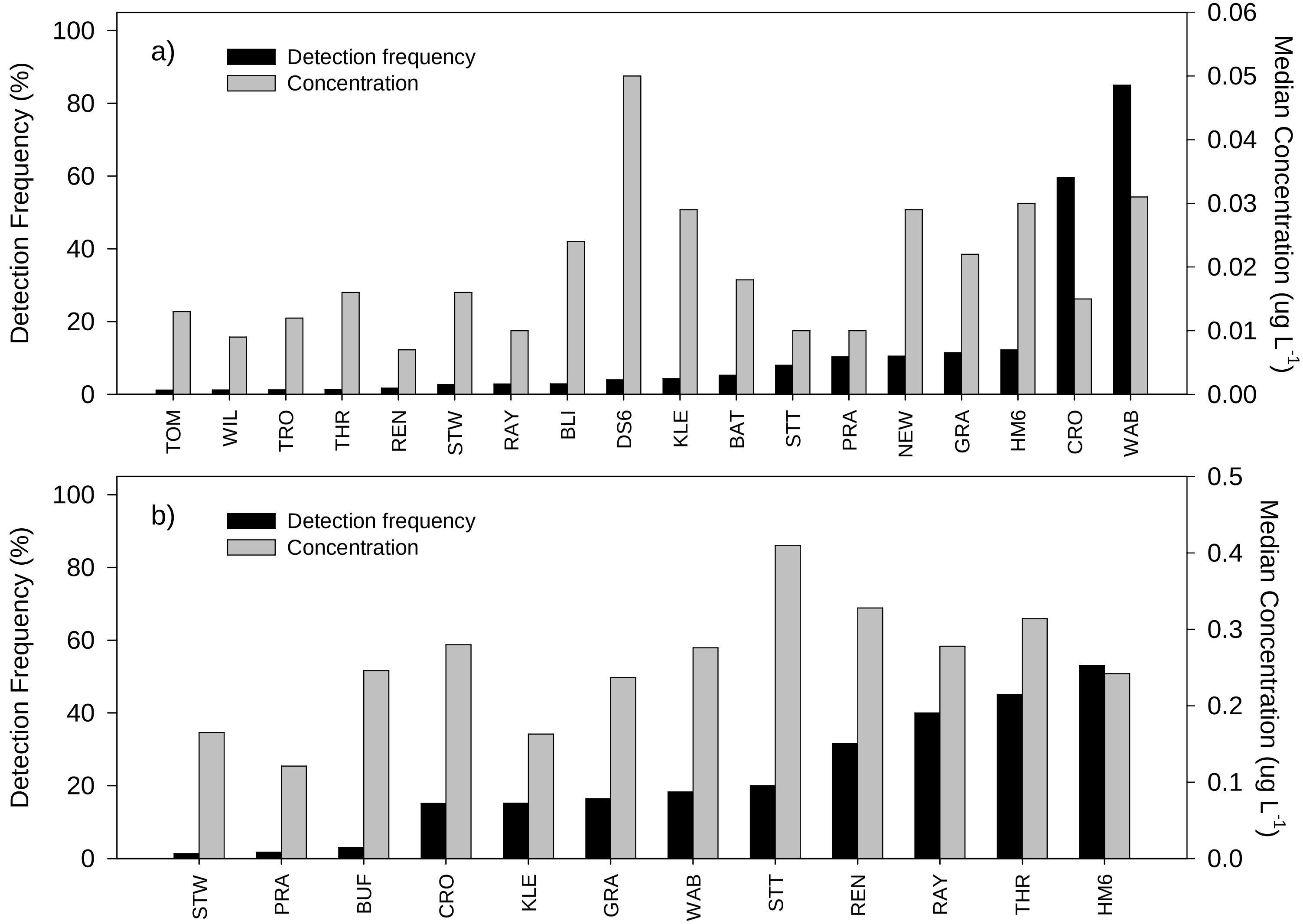


Figure 5.31. Patterns in MCPP (a) and imazamethabenz-methyl (b) detection frequencies (primary y-axis) and median pesticide concentrations (secondary y-axis) in AESA watersheds. For full watershed names see Table 5.4.

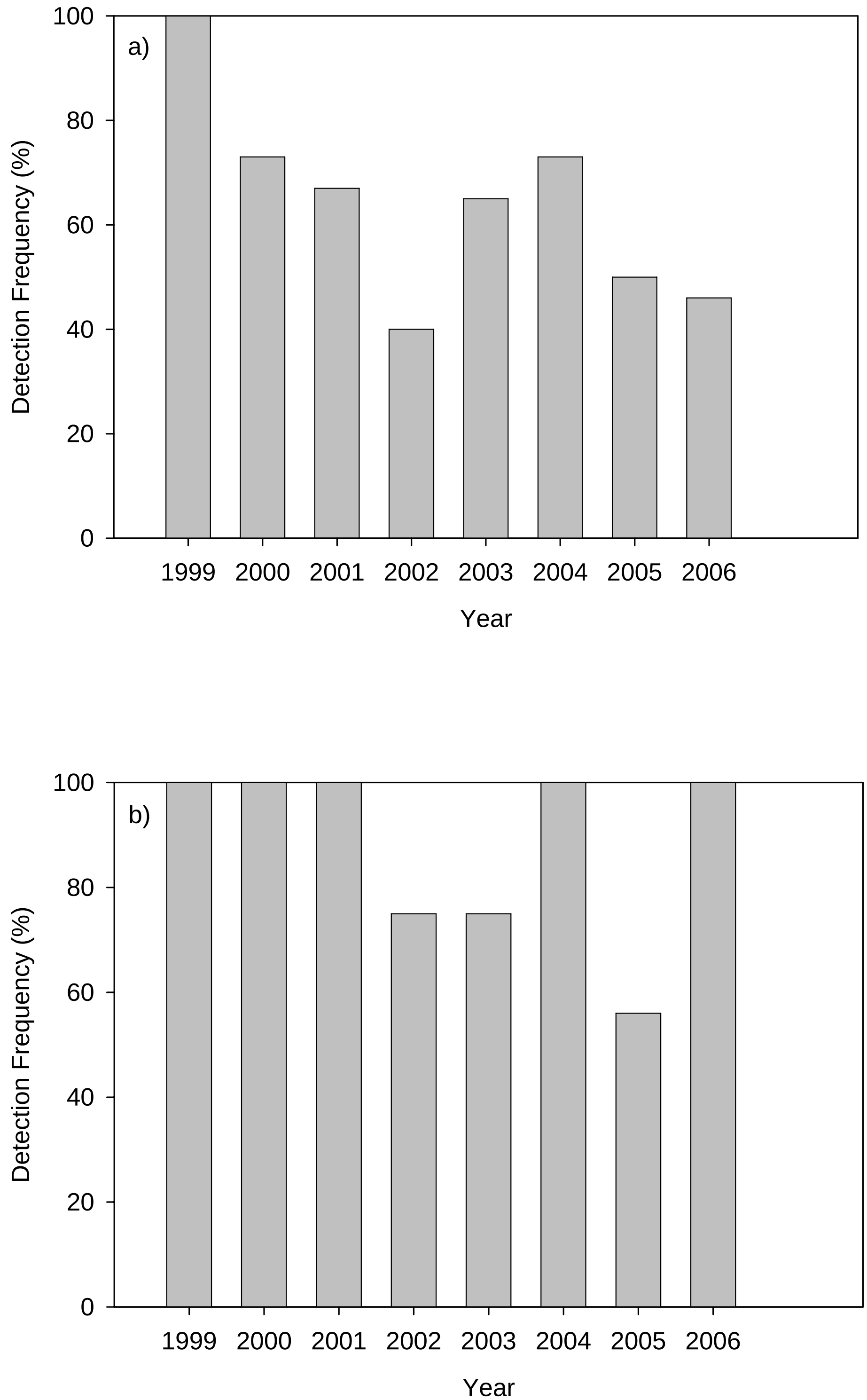


Figure 5.32. Interannual patterns in MCPP detection frequencies in Crowfoot Creek (a) and Wabash Creek (b).

Imazamethabenz-methyl. Imazamethabenz-methyl is an imidazolinone herbicide used to control stinkweed, wild mustard, wild oats, and buckwheat in annual rye grass, barley, wheat, and sunflower crops (Alberta Agriculture and Food 2007) and is used for agricultural purposes only (Anderson 2005). Other imidazolinone herbicides detected in AESA streams include imazethapyr and imazamox. The pesticide movement rating is for imazamethabenz-methyl very high (Table 5.12, pg. 5-36).

Imazamethabenz was detected in 12 of 23 streams. Detection frequencies were significantly higher in high intensity agricultural watersheds, followed by moderate intensity watersheds, then irrigated and low intensity watersheds (H= 48.860, $p \leq 0.0001$, Figure 5.30). It is noteworthy that only one stream with low intensity agriculture (Prairie Blood Coulee) and one stream with irrigated agriculture (Crowfoot Creek) had imazamethabenz detections (Figure 5.31). Streams with the highest detections are clustered together in the Aspen Parkland and include Haynes Creek, Threehills Creek, Ray Creek, and Renwick Creek.

The median concentration in all samples was 0.272 µg L^{-1}. Imazamethabenz concentrations were the highest of the eight commonly detected herbicide compounds. Median concentrations were similar among streams with detections (Figure 5.31), and no significant interannual changes were observed in imazamethabenz detection frequencies when all streams were analyzed together or among high agricultural intensity watersheds. However, a qualitative increase in imazamethabenz concentrations was observable from 2001 to 2005 (Figure 5.33). Therefore, imazamethabenz may be a compound to keep a closer eye on in the future.

Seasonally, detection frequencies were highest in spring when many of the high intensity watersheds were flowing (H=61.896, $p \leq 0.0001$) although concentration differences were not significant (Figure 5.34).

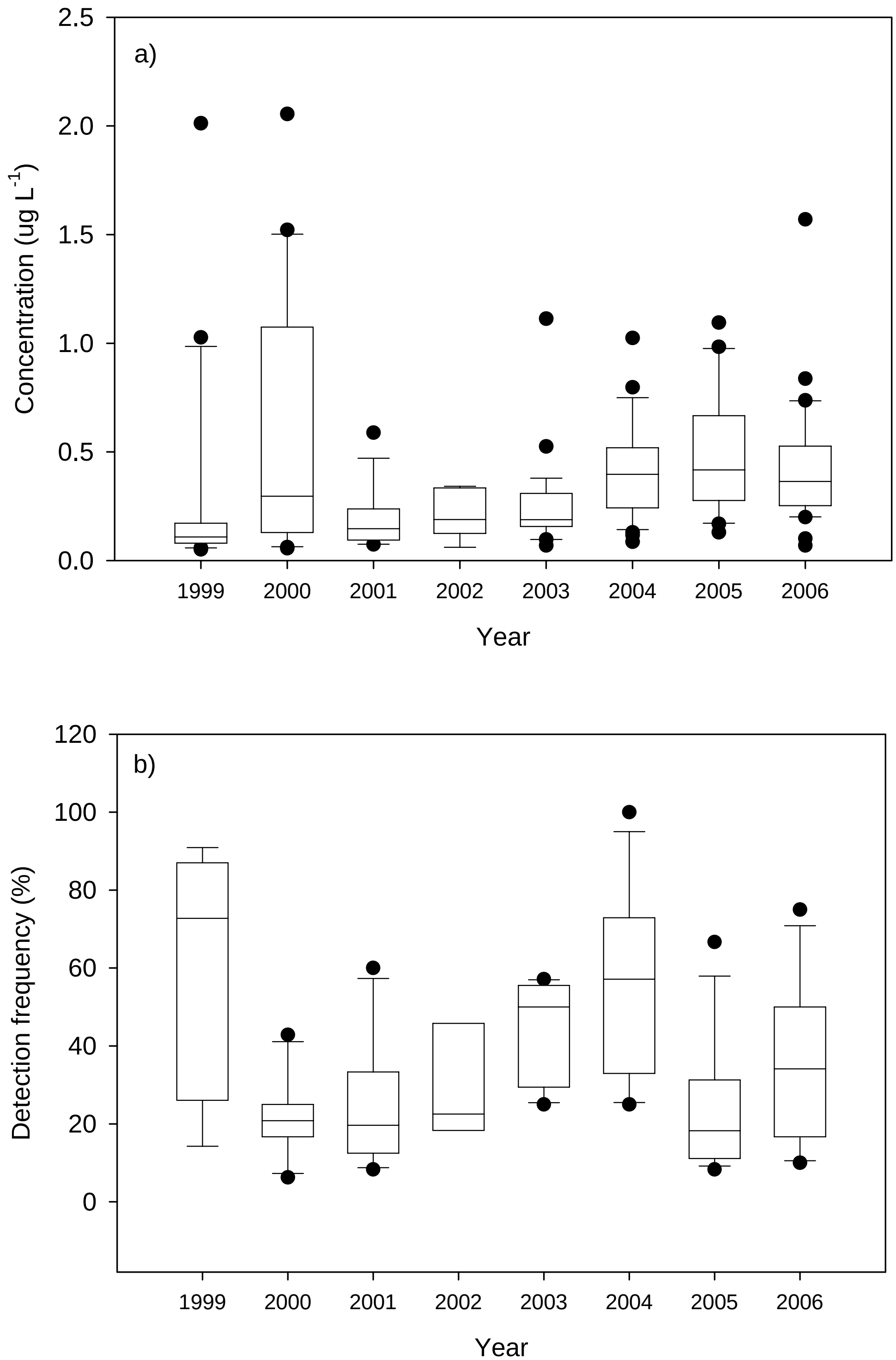


Figure 5.33. Interannual patterns in imazamethabenz-methyl concentrations (a) and detection frequency (b) from 1999 through 2006 in all AESA watersheds.

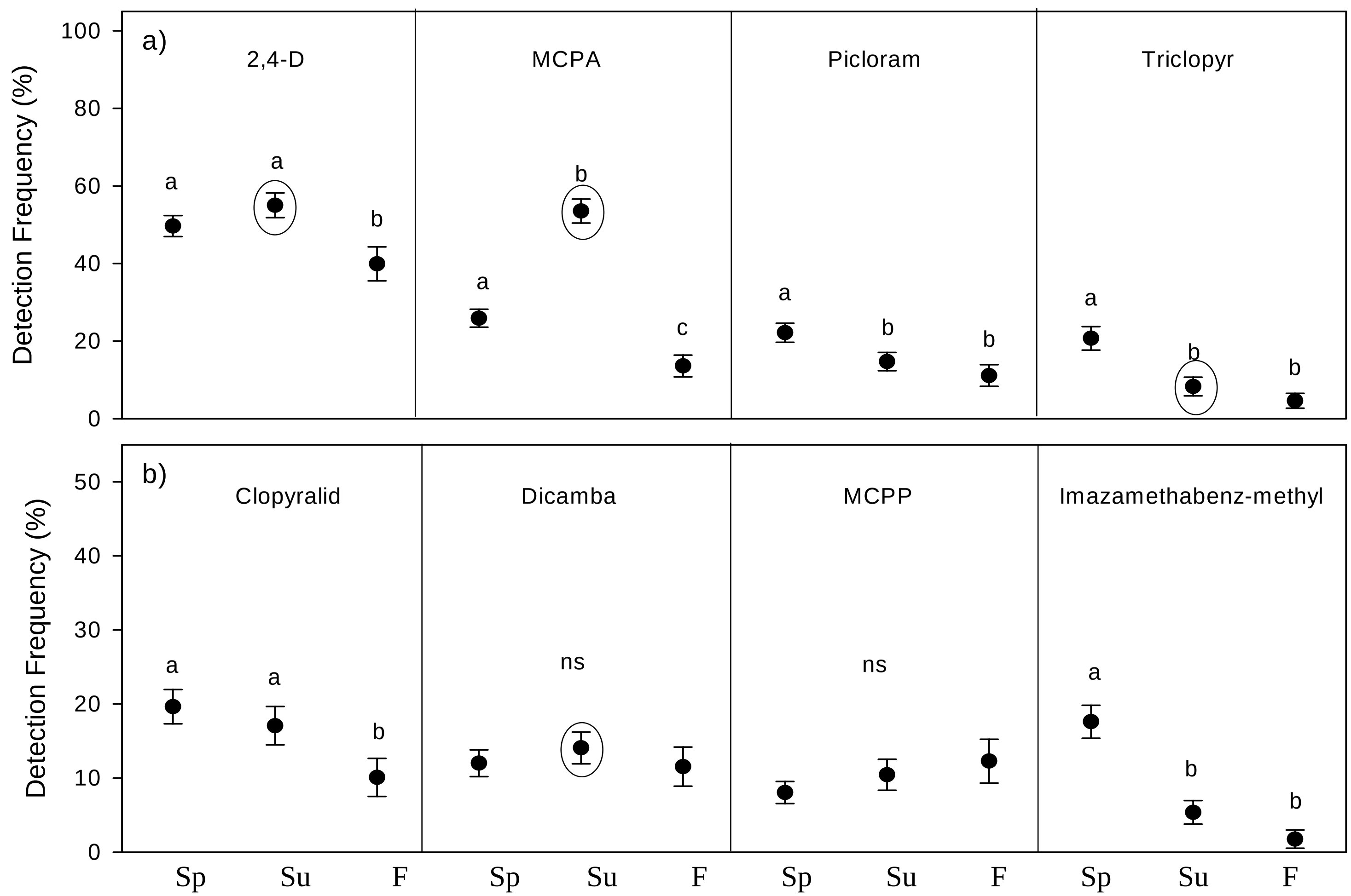


Figure 5.34. Seasonal detection frequency in the top eight detected pesticide compounds. Plots show mean annual detection frequency for all 23 watersheds (1999 to 2006) ±1 standard error. Letters represent statistically significant differences among seasons (Mann-Whitney, $p<0.05$) with ns representing no difference. Circles represent seasons with significantly higher median measurable concentrations (data not shown). Sp= Mar, Apr, May, SU=Jun, Jul, Aug, F =Sept, Oct.

SUMMARY AND CONCLUSIONS

The results of our study suggest that pesticides are commonly found in agricultural streams throughout the province and tend to be found with greater frequency and in greater concentrations with increasing agricultural intensity. Maximum concentrations are typically found in spring and in the summer application period. Concentrations occasionally exceed existing guidelines for either the protection of aquatic life or irrigation. In addition, there were multiple pesticides detected within a single sample. New tools such as the Alberta Pesticide Toxicity Index are useful in determining the risk of multiple co-occurring compounds; however, guidelines also need to be developed for some compounds, specifically those that are commonly applied and frequently detected or found in greater concentrations. The specific objectives of this chapter and future directions are addressed below.

Objective 1: Assess pesticide occurrences and concentrations in Alberta's agricultural streams.

- Low level concentrations of a variety of pesticides were commonly found in agricultural watersheds.
    - One or more pesticides were detected in 1041 of the 1627 samples collected from 1999-2006, or 64% of samples.
    - Thirty-seven of the 68 compounds monitored were detected. Of the total 68 compounds analyzed (from 1999-2006), detections included
        - 29 of 40 herbicides + breakdown products and isomers,
        - 4 of 20 insecticides + breakdown products and isomers, and
        - 4 of 8 fungicides.
    - The median number of compounds detected per sample was two.
    - The median total concentration of measurable pesticide in each sample was 0.098 $\mu g\ L^{-1}$.
- Herbicides were detected more frequently than insecticides or fungicides.
    - The top two detected herbicides, 2,4-D and MCPA, were detected in 46% and 31% of samples, respectively. Another six herbicide active ingredients were detected in ≥10% of analyzed samples (clopyralid, triclopyr, dicamba, picloram, imazethabenz-methyl, and MCPP). The remaining 21 compounds were detected in <10% of samples.
    - The top detected insecticide was gammabenzehexachloride (lindane), found in 0.6% of samples.
    - The top detected fungicide was iprodione, found in 3.3% of samples.
- The results confirm and support previously reported findings for Alberta's surface waters (Anderson 2005).

Objective 2: Evaluate differences in pesticide occurrence and concentration among watersheds with low, moderate, and high intensity (dryland and irrigated) agriculture

- Pesticide detection frequency, total pesticide concentration, and total number of compounds increased significantly as agricultural intensity increased from low to high, indicating that agricultural use of pesticides has a measurable impact on water quality.

    - Total detection frequency increased from 24% in low intensity to 80% in high intensity dryland watersheds; total detection frequency was 91% in high intensity irrigated watersheds.
    - Total concentrations were lowest in low agricultural intensity watersheds, highest in high intensity watersheds, and intermediate in both moderate and irrigated watersheds. Lower concentrations in irrigated watersheds may be due to dilution of irrigation return flow by cleaner irrigation source water and higher median flow volumes.
    - The total number of pesticides detected per sample was similar in watersheds with low and moderate intensity agriculture (predominantly one pesticide) and significantly higher in watersheds with high intensity or irrigated agriculture (equal number of samples with one, two, or three pesticides detected). Pesticide mixtures were more common in the high intensity watersheds regardless of whether dryland or irrigated agriculture was practiced.
- This study confirms findings found in Anderson et al. (1998b, c) that streams draining higher agricultural intensity watersheds have more frequent pesticide detections at higher concentrations.

Objective 3: Examine the relationship between measures of the intensity of pesticide use and presence of pesticide residues in surface water

- There was a strong correlation between the intensity of agriculture in a watershed (as % cropland and fertilizer and chemical expense percentiles) and total pesticide detection frequency.
- Similar correlations with total pesticide concentration were slightly weaker. However, this is likely because concentrations appear to be influenced by the type of water management (irrigated versus dryland) as well as by the intensity of chemical use.
- The findings suggest that land-based watershed metrics may be used to predict the degree of pesticide contamination in small agricultural watersheds.

Objective 4: Explore spatial and temporal trends in pesticide occurrence and concentration to assess risk to the environment and food safety.

Spatial trends

- Regional differences existed in compounds detected in streams as a result of the type of agriculture (irrigated versus dryland) practiced. There were eight pesticides detected only in watersheds with irrigated agriculture and nine pesticides detected only in watersheds with dryland agriculture (low, moderate, or high intensity).

Temporal trends: Interannual

- At a broad level, trends in pesticide detection frequency and total concentration reflect provincial scale agricultural census (2001 to 2006) and sales data (1998 – 2003): no change over time.
- Temporal patterns in certain individual active ingredients were observed with time either at provincial, regional, or site specific scales.
    - Imazamethabenz-methyl concentrations showed a qualitative increase from 2001 to 2006 at a provincial scale.

  - Picloram detection frequencies showed significant declines in high intensity dryland watersheds from 1999 to 2005 but increased again slightly in 2006.
  - Simazine detection frequencies increased in New West Coulee from 2002 to 2006.

Temporal trends: Seasonal

- Plots of monthly total concentrations by agricultural intensity category show that total concentrations (monthly medians) were highest in March in High agricultural intensity streams (with snowmelt runoff) and highest in June in Irrigated streams (following application). However, both watershed types showed a spring peak in March and a summer peak in June or July.
- Seasonal patterns in total concentrations were less pronounced in Moderate and Low agricultural streams.
- There is a lot of variability in moderate agricultural intensity stream concentrations.

Objective 5: Determine compliance of observed concentrations with Canadian Water Quality Guidelines and risk of cumulative effects using the Alberta Pesticide Toxicity Index.

- Irrigation guidelines for MCPA and dicamba were exceeded most frequently (11.2% and 11.4% of samples, respectively), indicating potential for damage to sensitive plant species if stream water was used for irrigation purposes.
- Guidelines for the Protection of Aquatic Life (PAL) were exceeded for 2,4-D, MCPA, chlorpyrifos, lindane, and triallate but only in a small proportion of samples (0.2 – 0.5%).
- Irrigated watersheds had a higher toxicity risk than high intensity dryland watersheds indicating total pesticide concentration alone is not a good measure of threat to aquatic life; it is important to know which compounds are present and in what concentrations.

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Appendices

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APPENDIX 1: 1991 LAND COVER

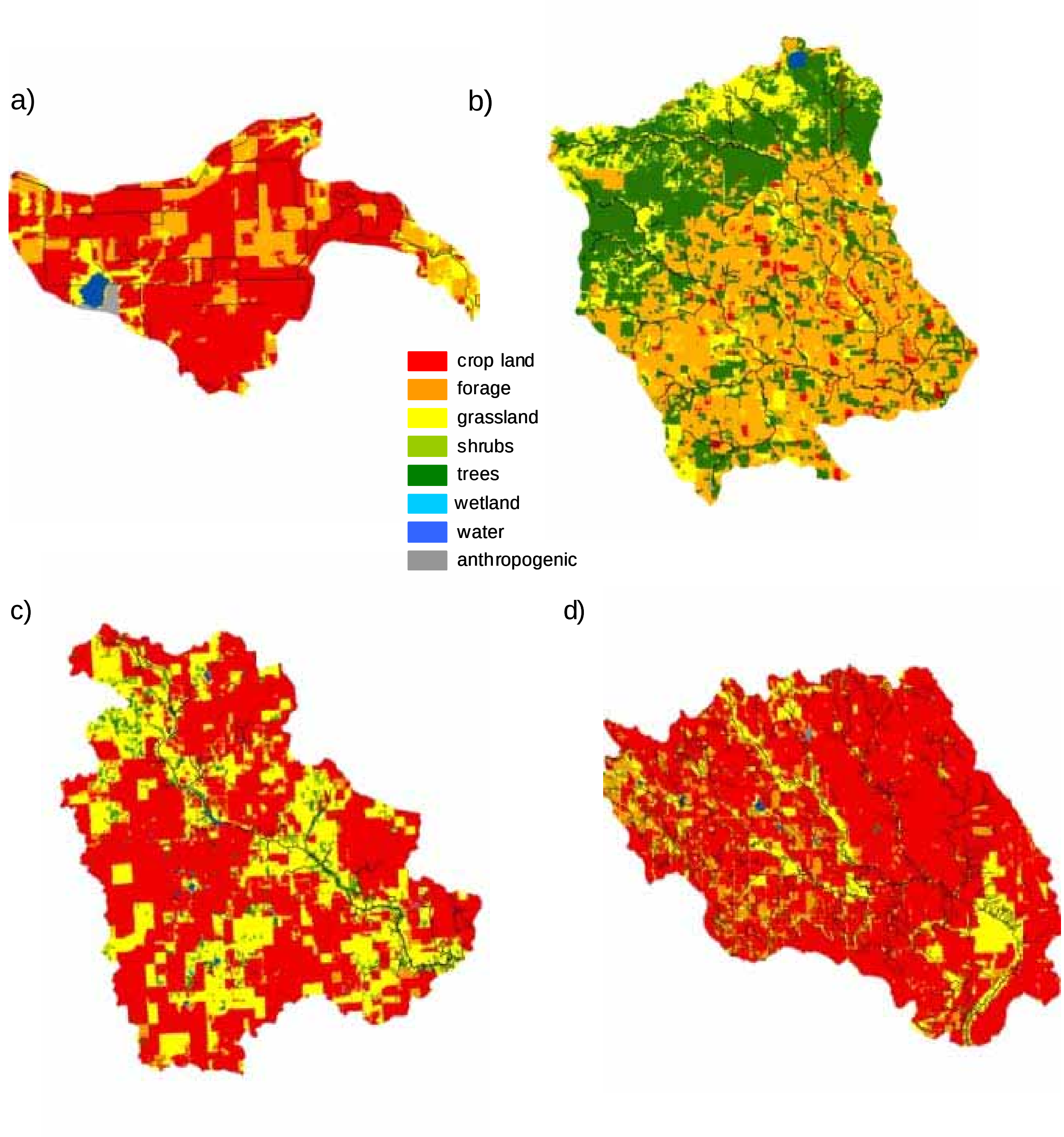


Figure A1.1. 1991 land cover for (a) Battersea Drain, (b) Blindman River, (c) Buffalo Creek, and (d) Crowfoot Creek. Black lines represent the watershed hydrology. As there is no scale for these watersheds, refer to Chapter 2 (Table 2.9) to see percentages of land cover for each watershed.

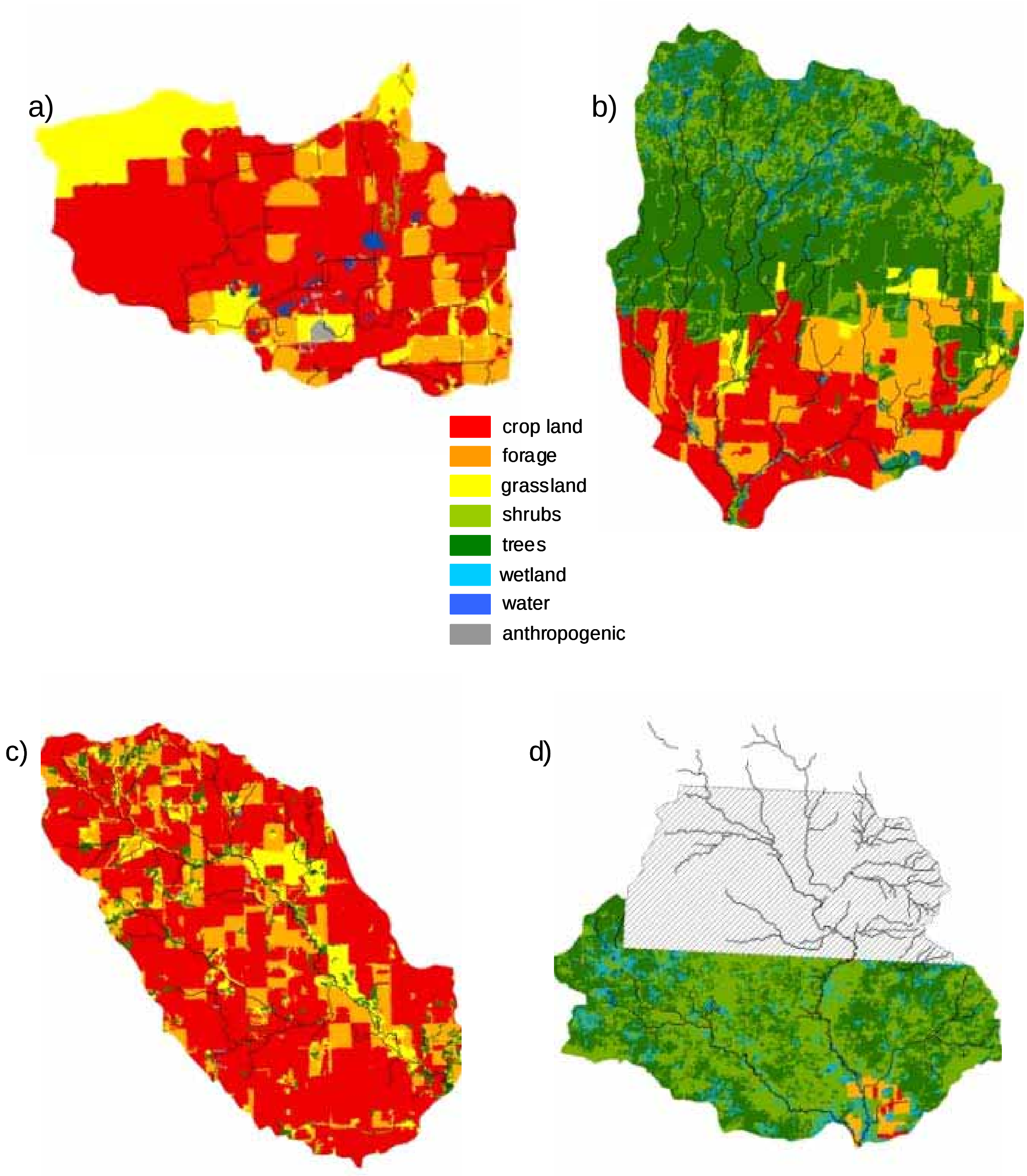


Figure A1.2. 1991 land cover for (a) Drain S6, (b) Grande Prairie Creek, (c) Haynes Creek, and (d) Hines Creek. Diagonal lines on Hines Creek represent the watershed area lying outside of Alberta's agricultural zone. Black lines represent the watershed hydrology. As there is no scale for these watersheds, refer to Chapter 2 (Table 2.9) to see percentages of land cover for each watershed.

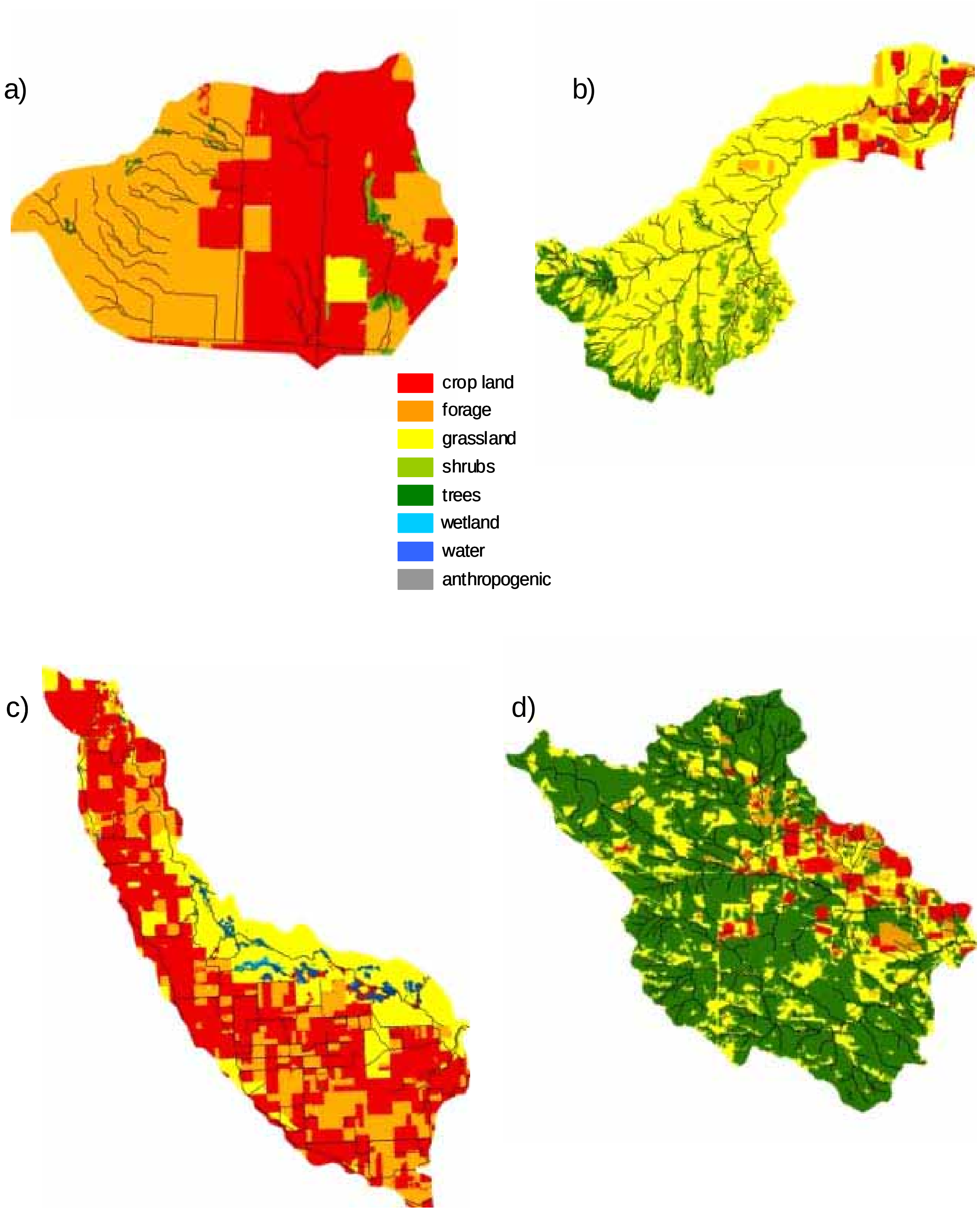


Figure A1.3. 1991 land cover for (a) Kleskun Drain, (b) Meadow Creek, (c) New West Coulee, and (d) Paddle River. Black lines represent the watershed hydrology. As there is no scale for these watersheds, refer to Chapter 2 (Table 2.9) to see percentages of land cover for each watershed.

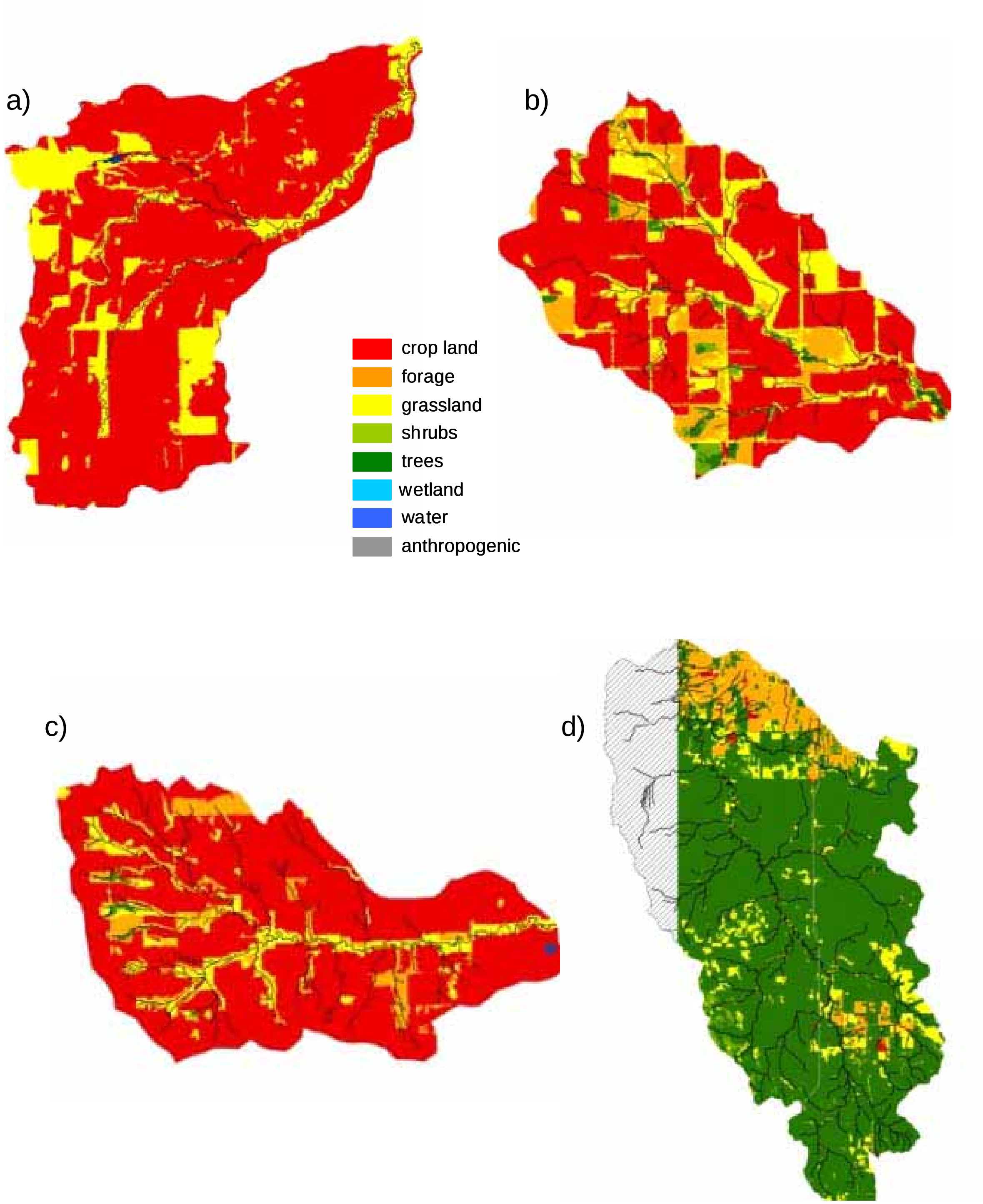


Figure A1.4. 1991 land cover for (a) Prairie Blood Coulee, (b) Ray Creek, (c) Renwick Creek, and (d) Rose Creek. Diagonal lines on the north-east corner of Rose Creek represent the area of the watershed that lies outside of Alberta's agricultural zone. Black lines represent the watershed hydrology. As there is no scale for these watersheds, refer to Chapter 2 (Table 2.9) to see percentages of land cover for each watershed.

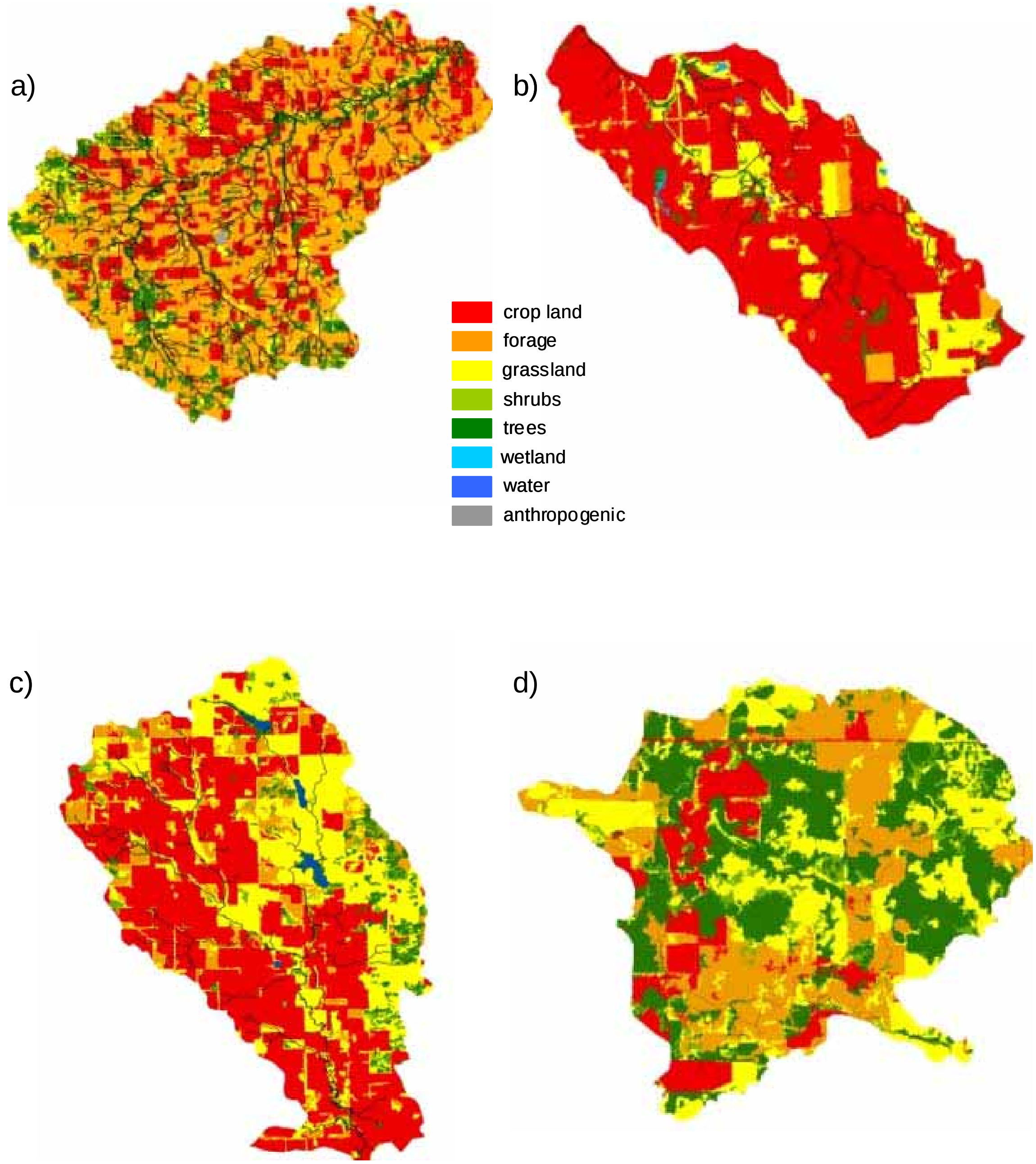


Figure A1.5. 1991 land cover for (a) Strawberry Creek, and (b) Stretton Creek, (c) Threehills Creek, and (d) Tomahawk Creek. Black lines represent the watershed hydrology. As there is no scale for these watersheds, refer to Chapter 2 (Table 2.9) to see percentages of land cover for each watershed.

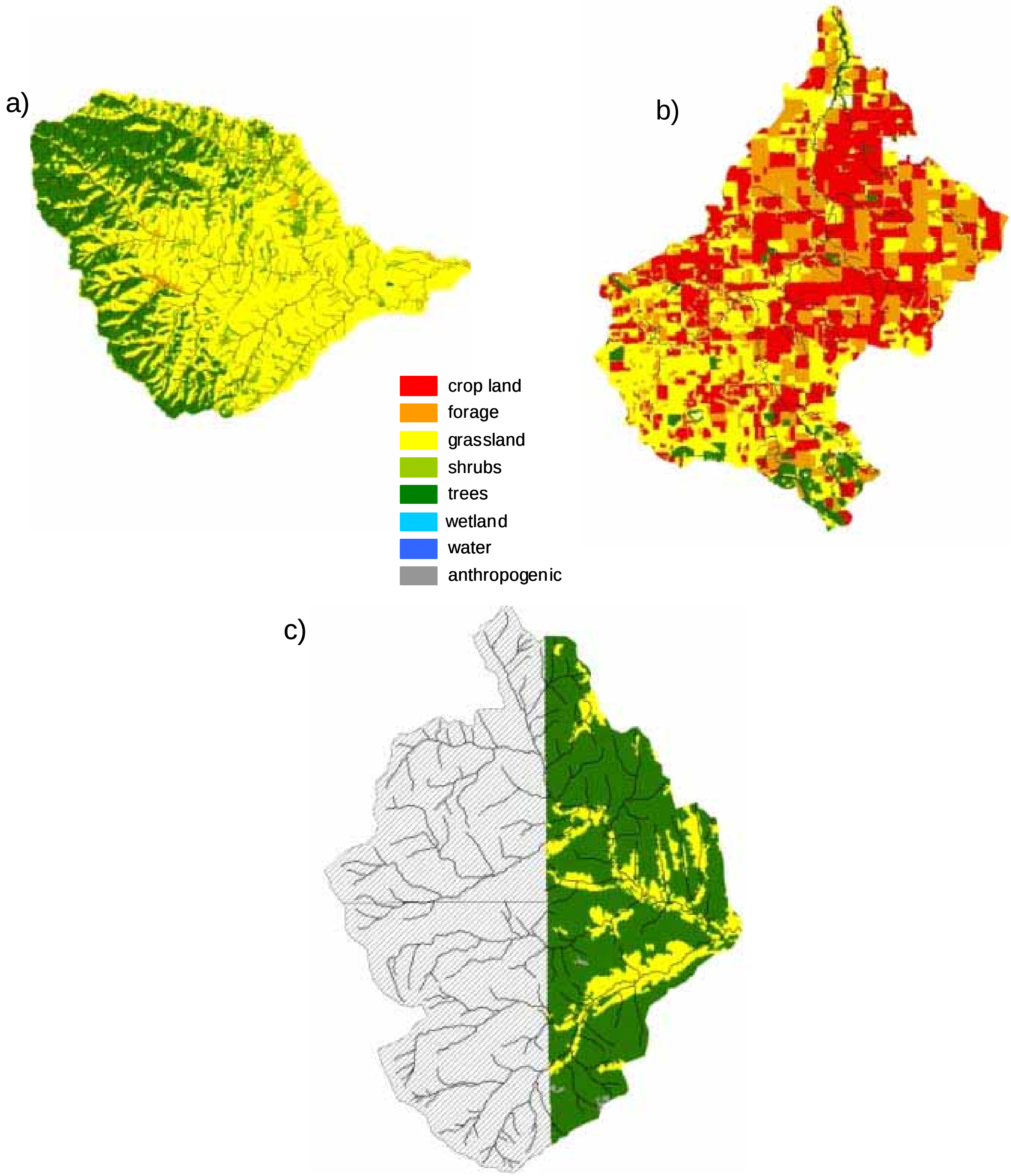


Figure A1.6. 1991 land cover for (a) Trout Creek, (b) Wabash Creek, and (c) Willow Creek. The diagonal lines on the eastern portion of Willow Creek represent the area of the watershed that lies outside of Alberta's agricultural zone. Black lines represent the watershed hydrology. As there is no scale for these watersheds, refer to Chapter 2 (Table 2.9) to see percentages of land cover for each watershed.

APPENDIX 2: PRECIPITATION 1995 TO 2006

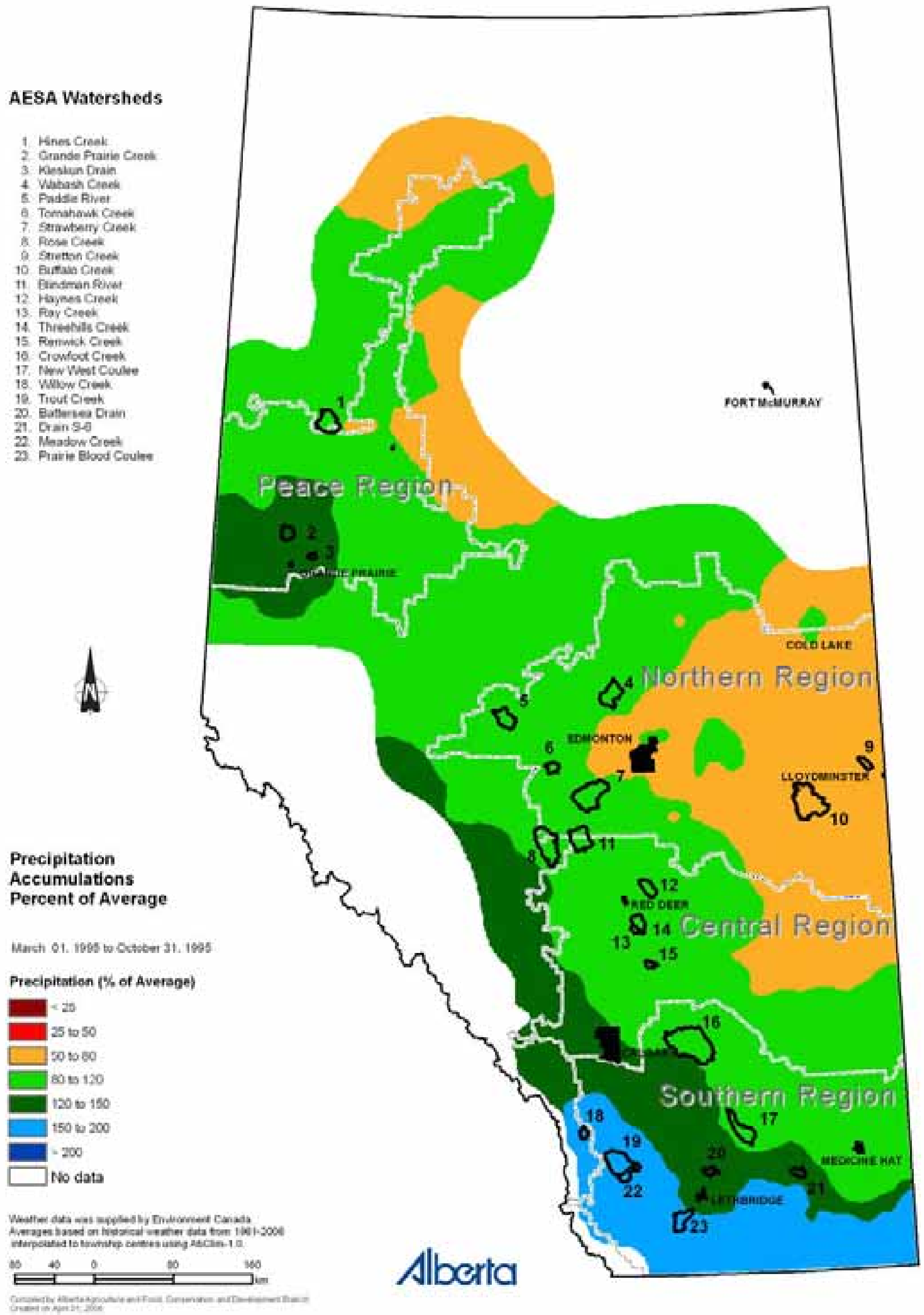


Figure A2.1. Precipitation accumulations in Alberta's agricultural zone from March 1, 1995 to October 31, 1995 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

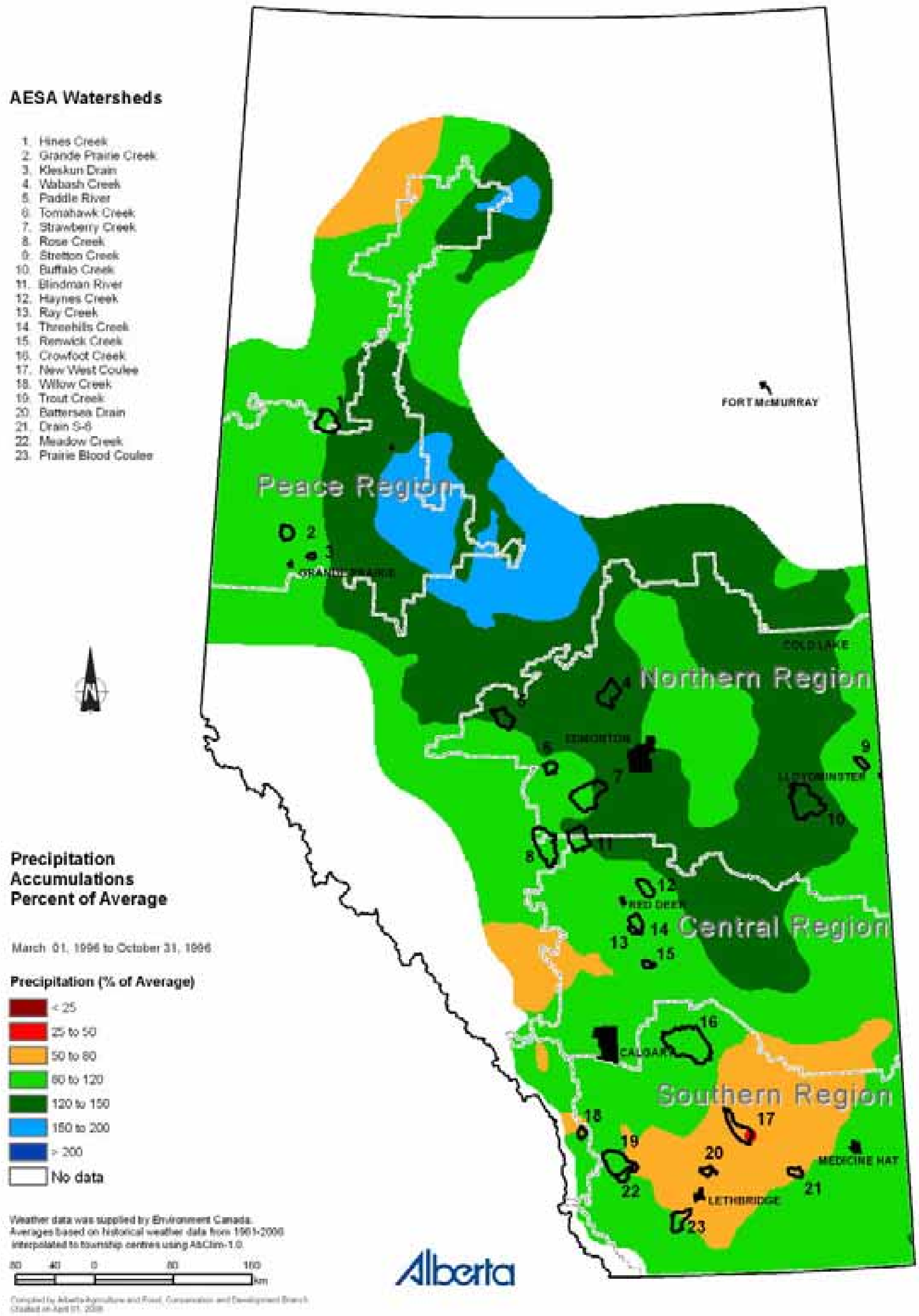


Figure A2.2. Precipitation accumulations in Alberta's agricultural zone from March 1, 1996 to October 31, 1996 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

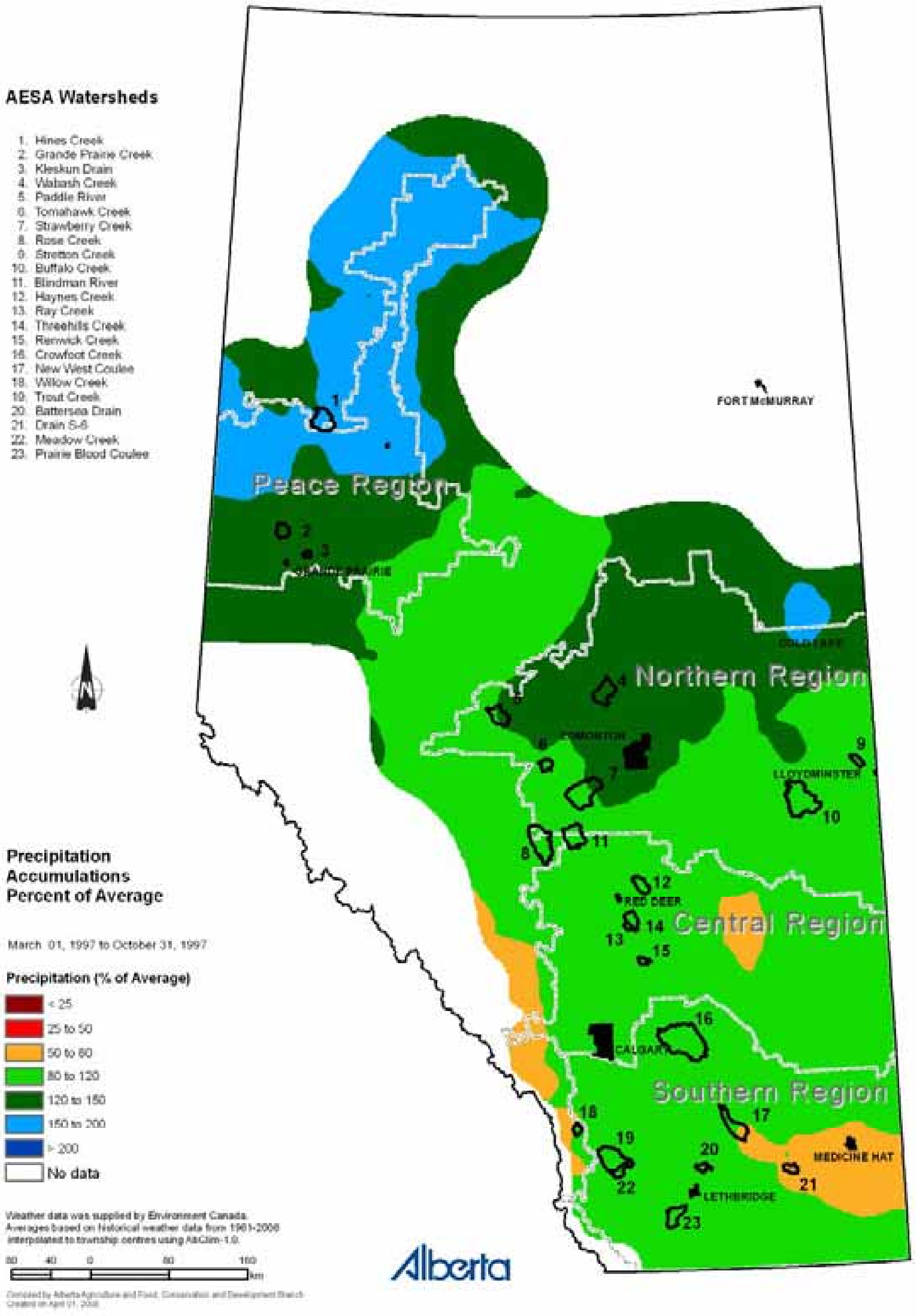


Figure A2.3. Precipitation accumulations in Alberta's agricultural zone from March 1, 1997 to October 31, 1997 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

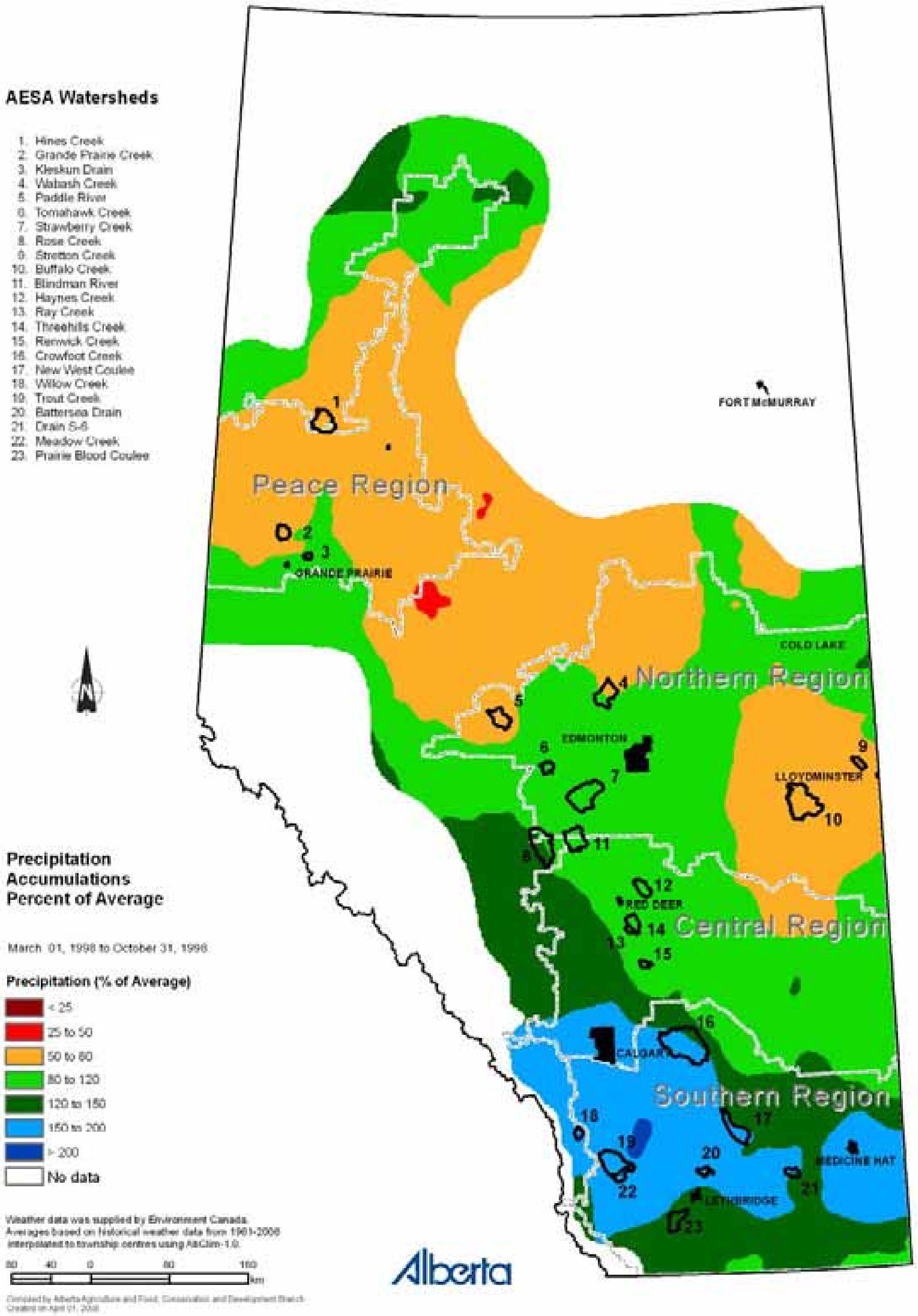


Figure A2.4. Precipitation accumulations in Alberta's agricultural zone from March 1, 1998 to October 31, 1998 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

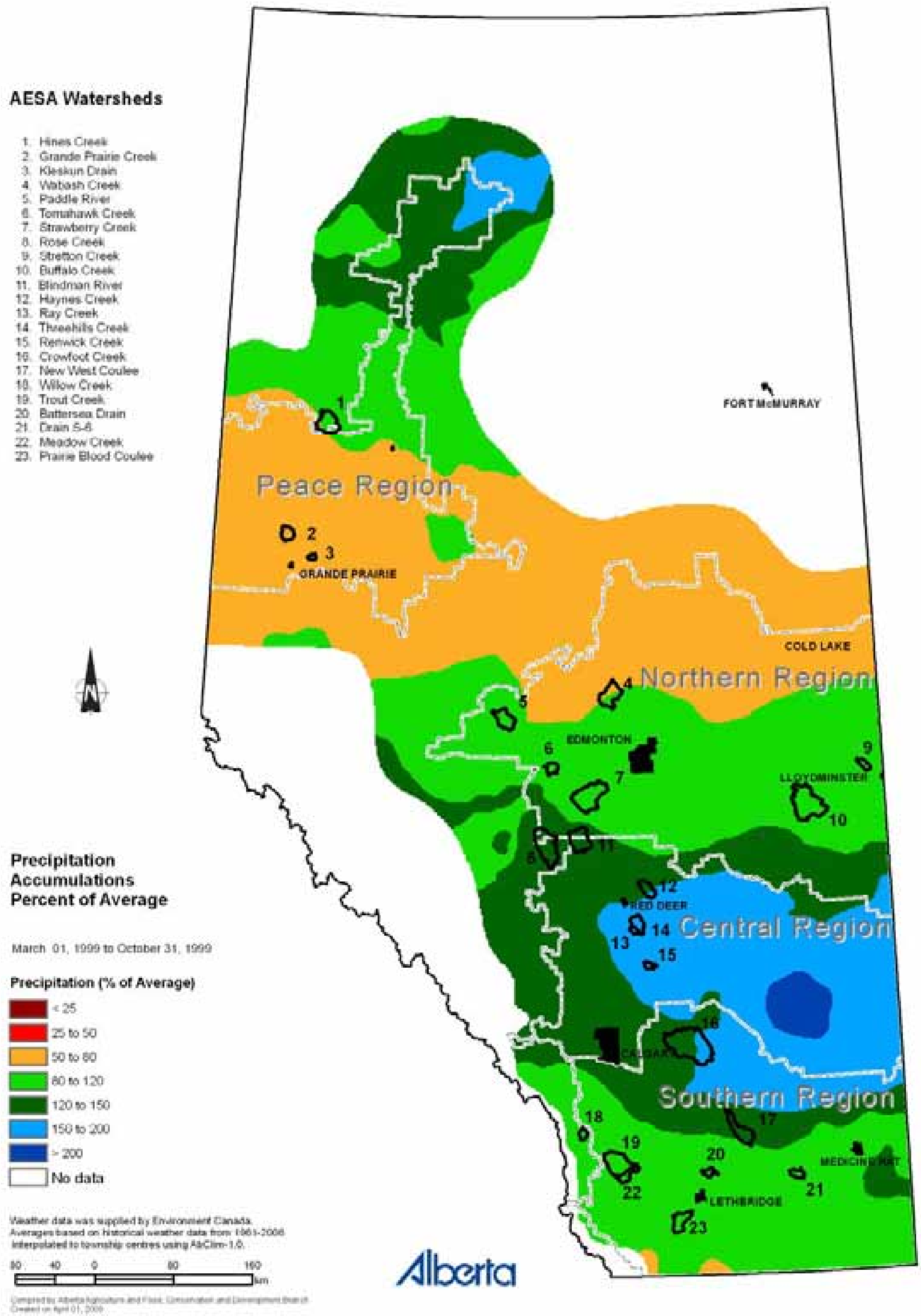


Figure A2.5. Precipitation accumulations in Alberta's agricultural zone from March 1, 1999 to October 31, 1999 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

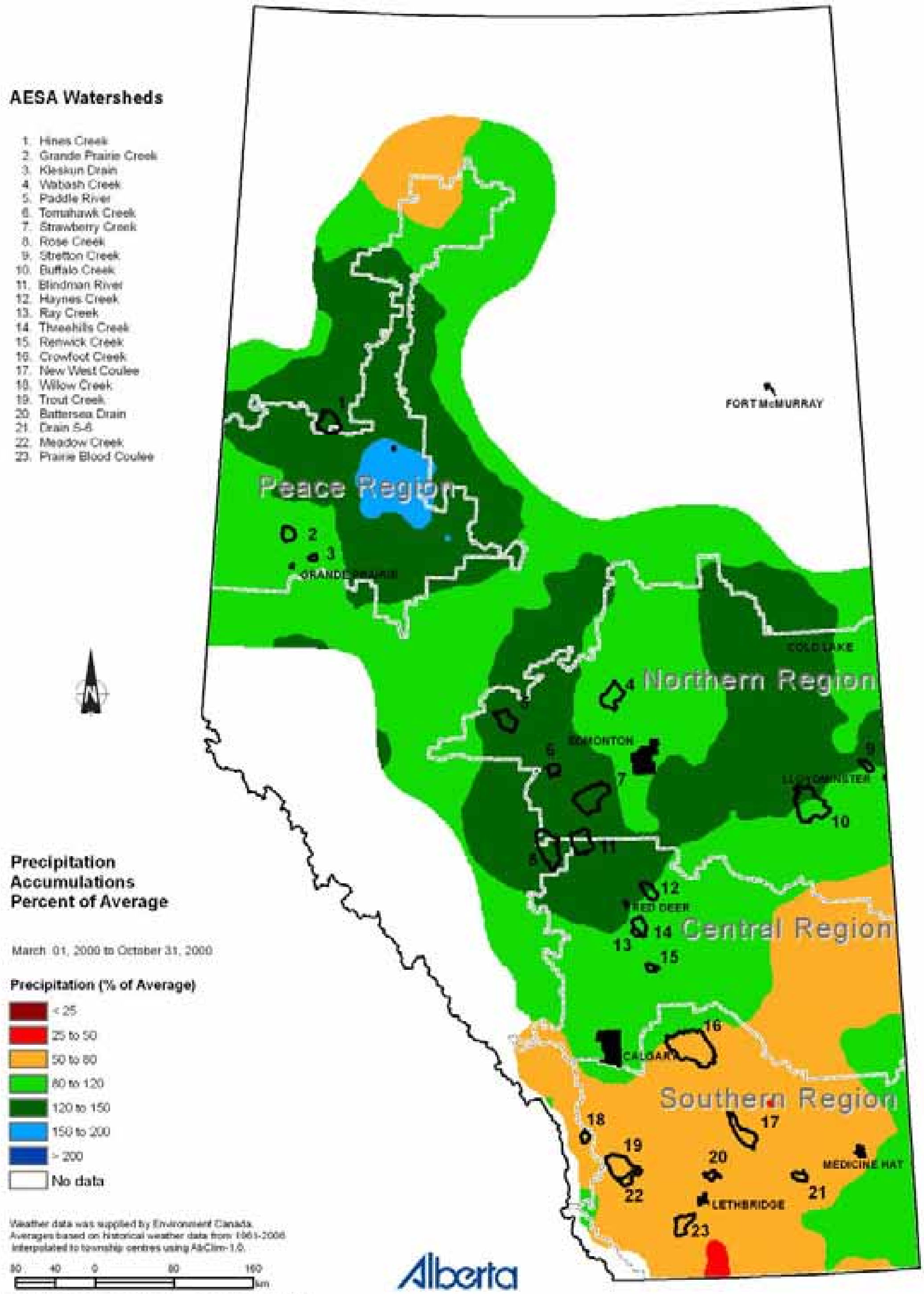


Figure A2.6. Precipitation accumulations in Alberta's agricultural zone from March 1, 2000 to October 31, 2000 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

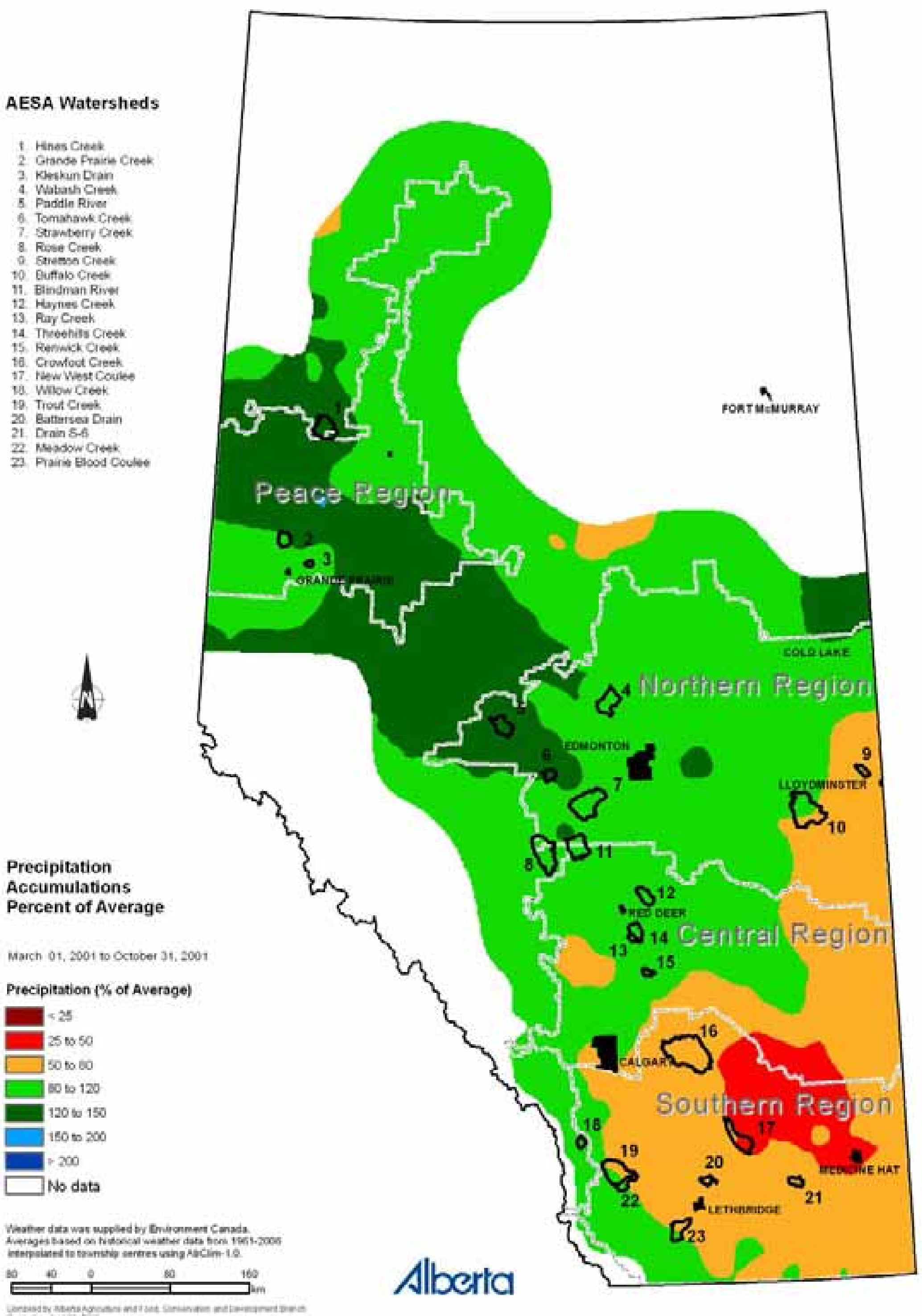


Figure A2.7. Precipitation accumulations in Alberta's agricultural zone from March 1, 2001 to October 31, 2001 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

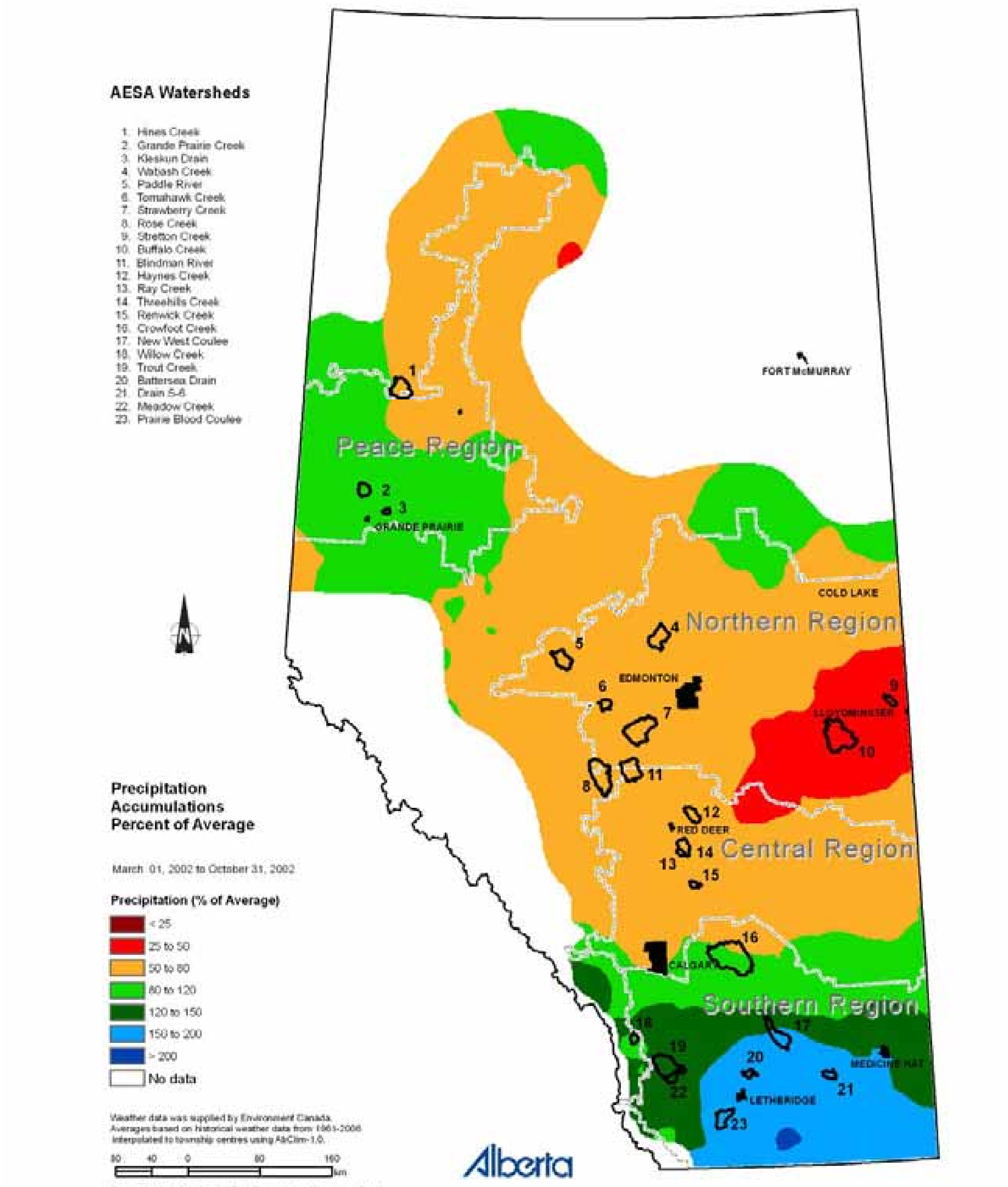


Figure A2.8. Precipitation accumulations in Alberta's agricultural zone from March 1, 2002 to October 31, 2002 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

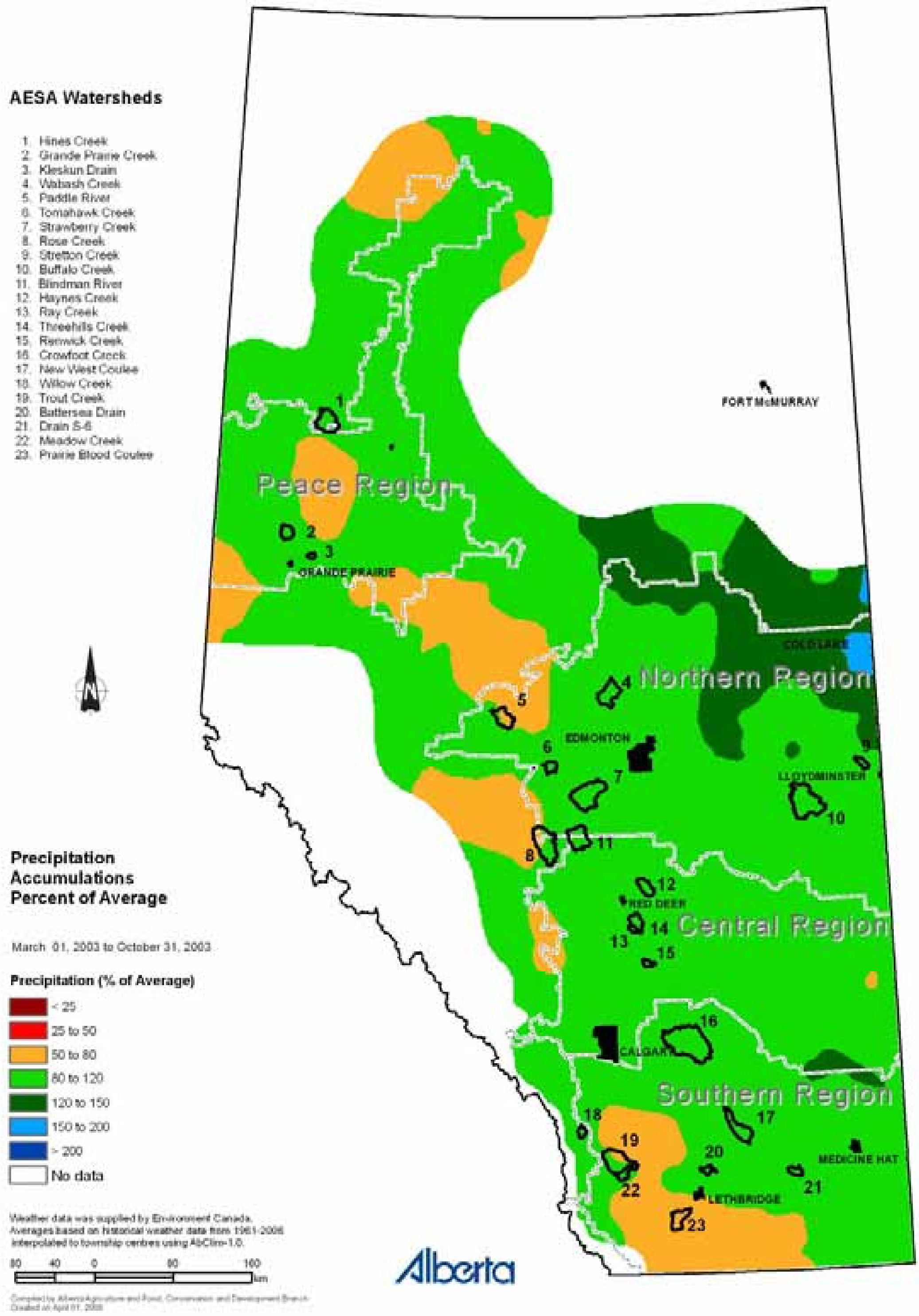


Figure A2.9. Precipitation accumulations in Alberta's agricultural zone from March 1, 2003 to October 31, 2003 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

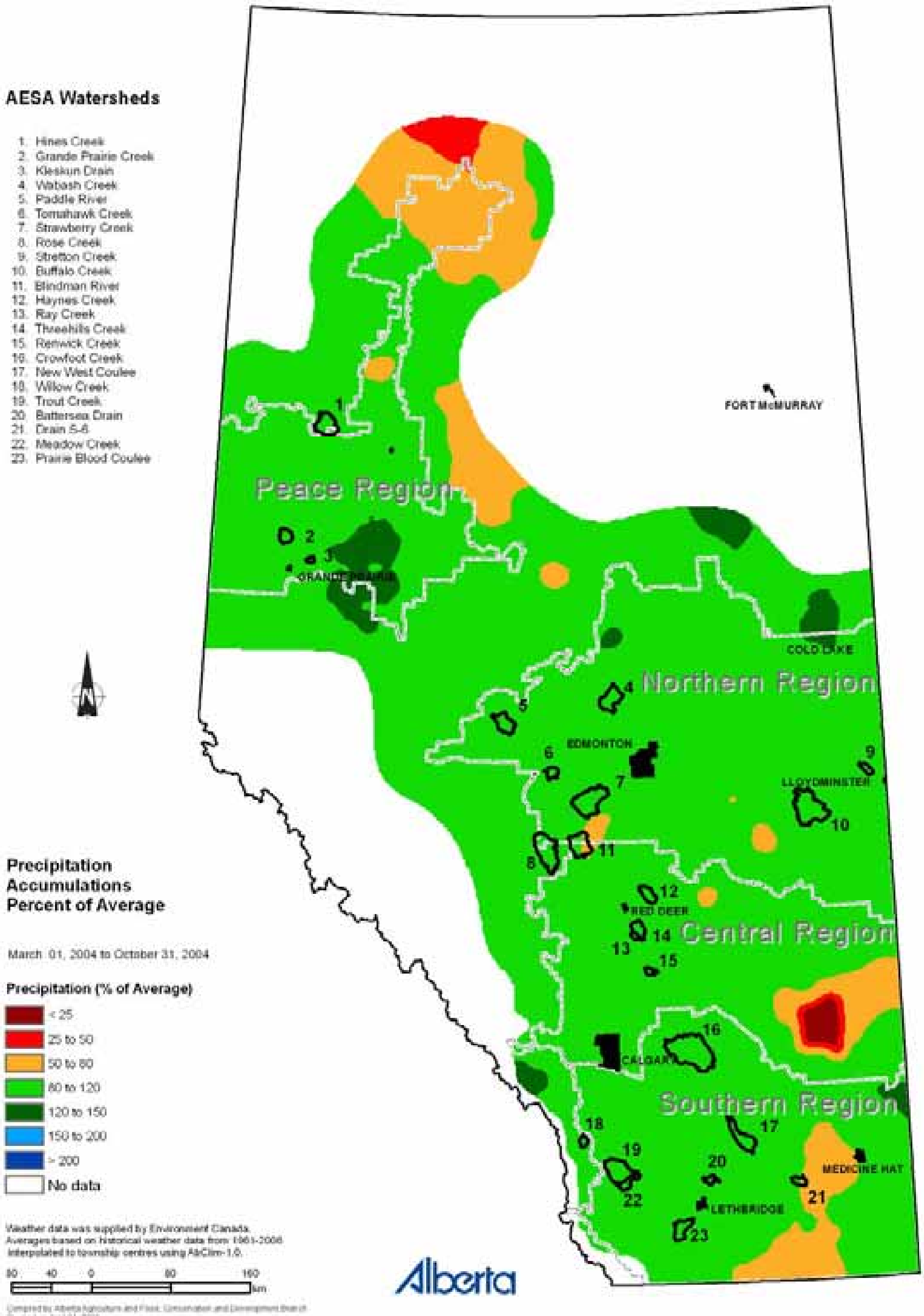


Figure A2.10. Precipitation accumulations in Alberta's agricultural zone from March 1, 2004 to October 31, 2004 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

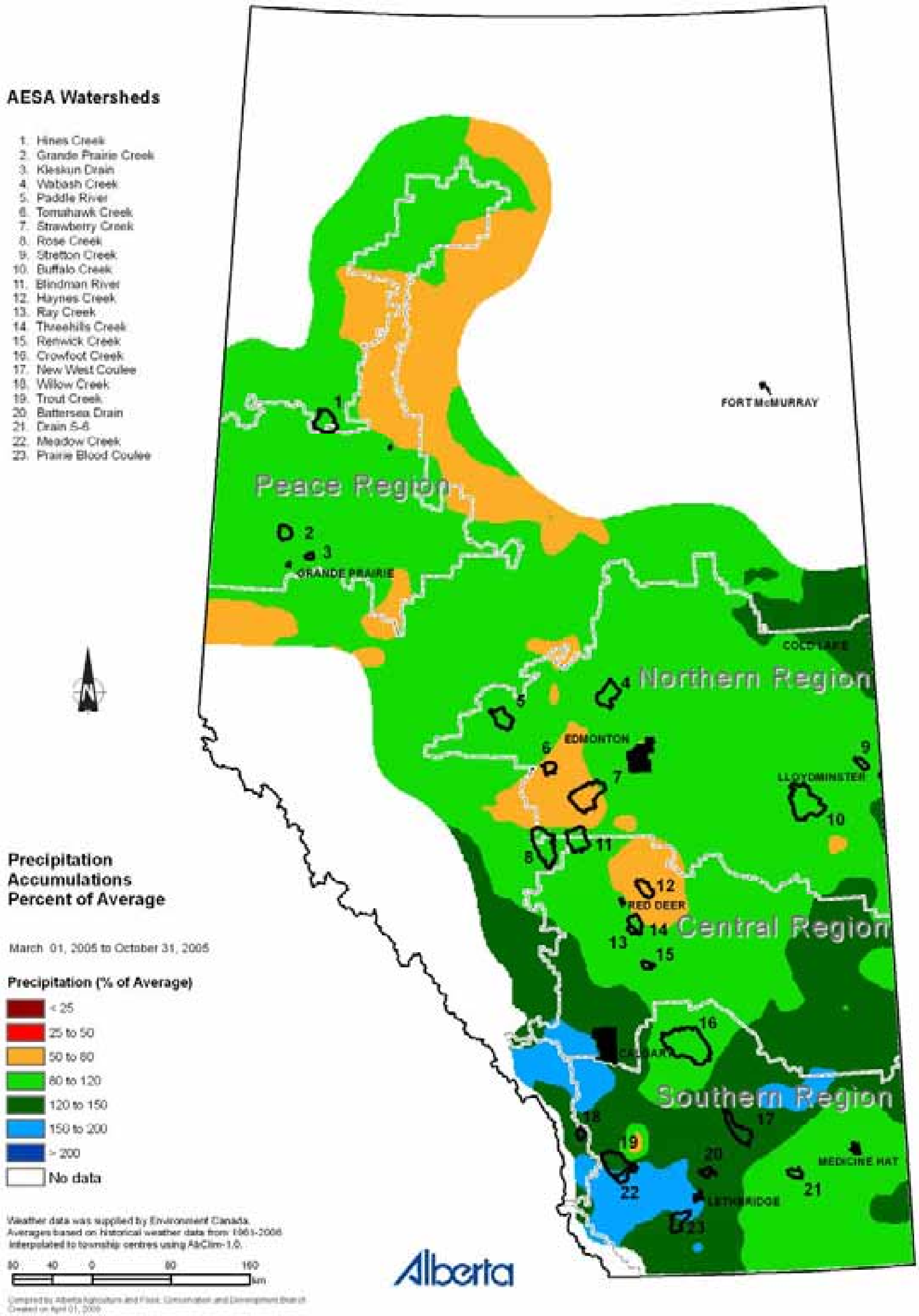


Figure A2.11. Precipitation accumulations in Alberta's agricultural zone from March 1, 2005 to October 31, 2005 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

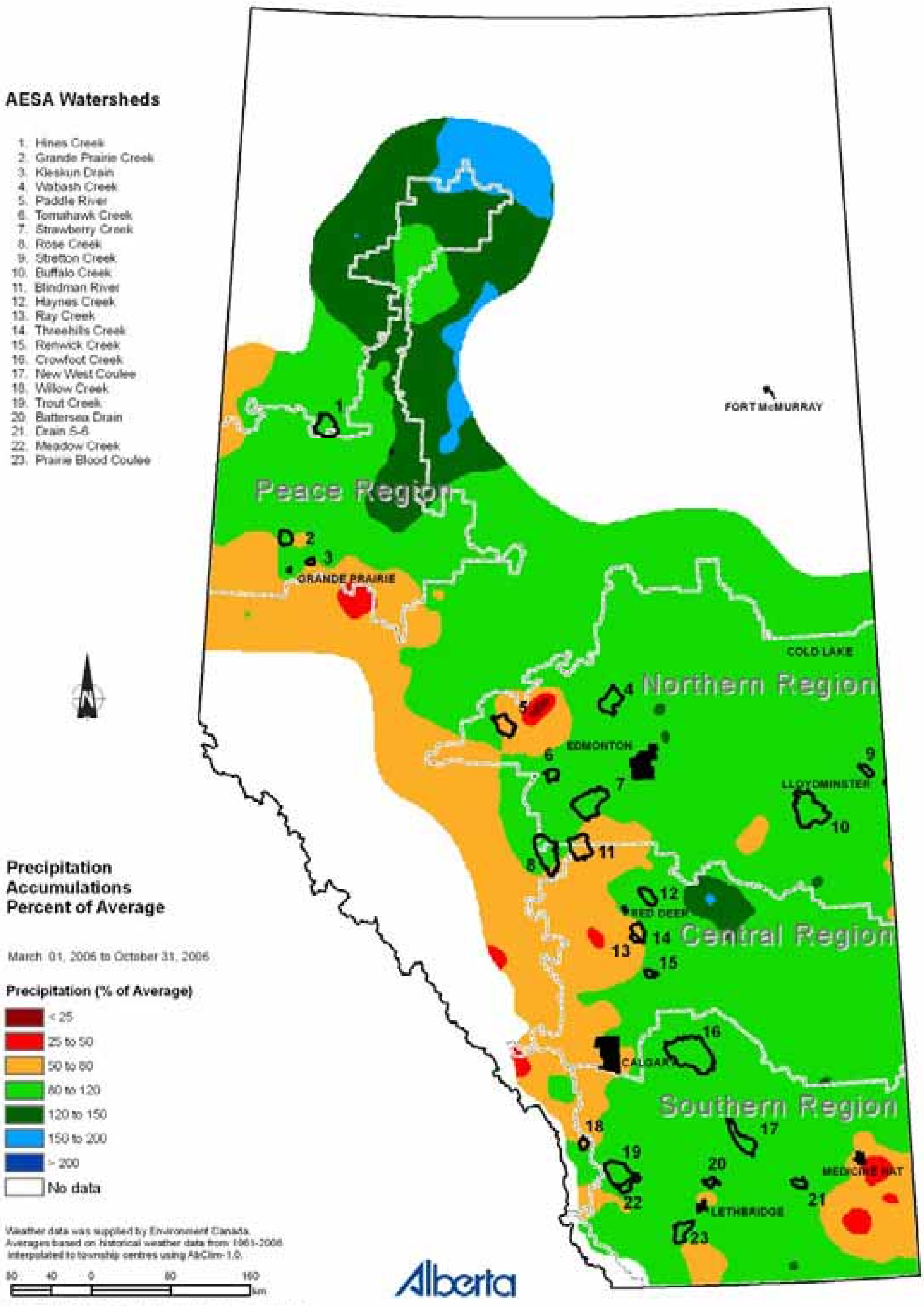


Figure A2.12. Precipitation accumulations in Alberta's agricultural zone from March 1, 2006 to October 31, 2006 as a percent of the average (1961 to 2006). Black outlines represent the watershed boundaries of the 23 AESA watersheds.

APPENDIX 3: LABORATORY METHODS

Table A3.1. ALS code and method for each parameter analyzed.

Parameter	ALS Code	ALS Method
Total Kjeldahl Nitrogen (TKN)	N-TOTKJ-ED	APHA-Norg C-Dig.-Auto-Colorimetry
Ammonia (NH_3-N)	NH4-ED	APHA 4500-NH3 F-Colorimetry
Nitrite-N (NO_2-N)	NO2-ED	APHA 4500-NO2B-Colorimetry
Nitrate-N (NO_3-N)	NO3-IC-CL	
Nitrite + Nitrate (NO_2-N + NO_3-N)		APHA 4500-NO3 E-Colorimetry
Total Phosphorus (TP)	PO4-T-COL-CL	APHA 4500-P B,E-Auto-Colorimetry
Total Dissolved Phosphorus (TDP)	PO4-TD-COL-CL	APHA 4500-P B,E-Auto-Colorimetry
Total Suspended Solids (TSS)	SOLIDS-TOTSUS-CL	APHA 2540 D- Gravimetric
pH	PH-CL	APHA 4500-H, 2510
Conductivity	EC-ED	APHA 4500-H, 2510

APPENDIX 4: METHOD DETECTION LIMITS

All values less than the method detection limit (MDL) were set at one-half the MDL, and parameters with multiple detection limits were set at one-half the highest MDL (Westbrook and McEachern 2002). Occasionally, NO_2^--N values exceeded NO_2^-+NO_3^--N values in a sample when the NO_2^-+NO_3^--N data were below the MDL. In the few instances where this occurred, the NO_2^-+NO_3^--N was set to equal the NO_2^--N value.

Table A4.1. Changes in the lowest method detection limit (LMDL) for nitrogen and phosphorus forms determined by ALS laboratories during CAESA and AESA surface water monitoring studies (1995-2006).

Parameter	1st MDL	2nd MDL	3rd MDL	4th MDL	Comments
NFR	*L1 (1995-1999)	L3 (1999-2006)			*All samples under L1 were >3 or <L1. Censored data changed to 1.5.
TP	L0.001 (1995-2006)	*L0.02 (2002, sporadic)			*No samples reported below L0.02. Censored data reported as 0.0005
TDP	L0.001 (1995-2006)	*L0.02 (2002 sporadic)			*Only 2 QC samples were analyzed with L0.02. Censored data reported as 0.0005.
TKN	L0.05 (1995-1997; 2000-2006)	*L0.01 (1998-1999)	**L0.2 (2000)		* 14 QC samples under L0.01. ** Four samples in 2000 under L0.2 changed to 1. Majority of data set analyzed at L0.05 and changed to ½ L0.05 (0.025).
NO_2^-+NO_3^--N	L0.001 (1996 only)	L0.005 (1997-2000)	L0.006 (2000-2006)	L0.01 (sporadic)	
NH_3-N	L0.005 (1999-2006)	*L0.001 (1996-1999)			* Only 4 QC samples under L0.001. Used ½ L0.005 for data range.

APPENDIX 5: NUTRIENT COMPLIANCE TO GUIDELINES

Table A5.1. Percent TP compliance to the Protection of Aquatic Life nutrient guideline for each stream by year (1999 to 2006).

Stream Year	1999	2000	2001	2002	2003	2004	2005	2006	Average
				% Compliance					
Low Agricultural Intensity									
Hines Creek	0	12	0	0	0	23	0	0	4
Paddle River	13	35	20	29	8	17	0	0	15
Prairie Blood Coulee	100	100	67	59	75	89	88	56	79
Rose Creek	35	48	65	47	40	63	47	67	51
Willow Creek	88	96	95	71	100	95	95	95	92
Moderate Agricultural Intensity									
Blindman River	0	7	29	20	0	5	0	0	8
Grande Prairie Creek	25	40	8	0	18	14	0	10	14
Kleskun Drain	0	7	0	10	13	9	0	0	5
Meadow Creek	0	23	45	48	57	36	27	39	35
Tomahawk Creek	0	0	0	0	7	0	3	0	1
Trout Creek	96	100	82	67	83	96	71	65	82
High Agricultural Intensity									
Buffalo Creek	21	0	0	8	13	0	4	0	6
Haynes Creek M1	0	0	0	0	0	0	0	0	0
Haynes Creek M6	0	0	0	0	0	no data	0	0	0
Ray Creek	3	4	8	0	21	0	0	0	5
Renwick Creek	0	8	0	0	0	0	0	0	1
Strawberry Creek	19	40	36	45	27	47	38	10	33
Stretton Creek	0	0	no data	no data	no data	0	0	0	0
Threehills Creek	0	0	8	0	0	0	0	0	1
Wabash Creek	0	11	0	0	0	0	43	0	7
Irrigation Streams									
Battersea Drain	33	78	56	13	15	36	5	10	31
Crowfoot Creek	7	11	0	0	9	10	0	5	5
Drain S-6	60	88	93	79	81	64	67	38	71
New West Coulee	64	64	36	44	25	16	10	19	35

Table A5.2. Percent TN compliance to the Protection of Aquatic Life nutrient guideline for each stream by year (1999 to 2006).

Stream	1999	2000	2001	2002	2003	2004	2005	2006	Average
				% Compliance					
Low Agricultural Intensity									
Hines Creek	0	18	8	7	0	50	44	17	18
Paddle River	71	81	60	57	62	100	33	74	67
Prairie Blood Coulee	77	67	78	41	38	84	41	0	53
Rose Creek	52	69	65	60	69	84	72	79	69
Willow Creek	100	100	95	100	100	100	95	95	98
Moderate Agricultural Intensity									
Blindman River	29	34	41	33	29	53	22	5	31
Grande Prairie Creek	8	15	0	11	27	21	0	20	13
Kleskun Drain	0	7	0	20	0	18	0	0	6
Meadow Creek	63	23	73	67	86	86	68	65	66
Tomahawk Creek	0	0	14	0	0	0	3	0	2
Trout Creek	100	95	100	90	91	100	90	87	94
High Agricultural Intensity									
Buffalo Creek	7	18	11	8	6	19	8	6	10
Haynes Creek M1	0	5	11	0	0	0	0	0	2
Haynes Creek M6	0	0	0	0	0	no data	0	0	0
Ray Creek	0	8	8	0	21	7	15	20	10
Renwick Creek	0	0	0	0	0	0	0	0	0
Strawberry Creek	27	37	27	55	45	76	44	10	40
Stretton Creek	0	9	no data	no data	no data	0	0	0	2
Threehills Creek	0	0	8	0	0	0	0	0	1
Wabash Creek	0	11	0	0	0	0	43	7	8
Irrigation Streams									
Battersea Drain	60	56	69	44	40	52	38	48	51
Crowfoot Creek	60	42	65	44	48	57	45	36	50
Drain S-6	100	94	93	79	44	77	93	86	83
New West Coulee	100	86	71	56	50	56	65	57	68

Table A5.3. Percent NO_2-N compliance to the Protection of Aquatic Life nutrient guideline for each stream by year (1999 to 2006).

Stream	1999	2000	2001	2002	2003	2004	2005	2006	Average
					% Compliance				
Low Agricultural Intensity									
Hines Creek	100	100	100	100	100	100	100	100	100
Paddle River	100	100	100	100	100	100	100	100	100
Prairie Blood Coulee	100	100	100	94	100	94	100	94	98
Rose Creek	100	100	100	100	100	100	100	100	100
Willow Creek	100	100	100	100	100	100	100	100	100
Moderate Agricultural Intensity									
Blindman River	100	100	100	100	100	100	100	100	100
Grande Prairie Creek	100	100	100	100	100	92	100	100	99
Kleskun Drain	100	100	100	90	100	100	100	100	99
Meadow Creek	100	100	100	100	100	100	100	100	100
Tomahawk Creek	100	96	93	90	100	100	100	100	97
Trout Creek	100	100	100	100	100	100	100	100	100
High Agricultural Intensity									
Buffalo Creek	100	100	100	100	92	100	96	94	98
Haynes Creek M1	91	91	88	83	90	83	82	78	86
Haynes Creek M6	18	24	100	100	100	no data	90	67	71
Ray Creek	100	100	82	100	93	100	100	87	95
Renwick Creek	100	100	60	83	82	73	95	87	85
Strawberry Creek	96	100	91	100	100	100	100	90	97
Stretton Creek	100	100	no data	no data	no data	60	100	100	92
Threehills Creek	91	92	91	86	86	100	90	89	90
Wabash Creek	100	100	100	29	75	77	87	87	82
Irrigation Streams									
Battersea Drain	87	94	81	88	50	88	81	71	80
Crowfoot Creek	100	78	94	100	78	81	100	86	90
Drain S-6	100	100	100	100	100	100	100	100	100
New West Coulee	100	100	100	88	85	96	100	90	95

Table A5.4. Percent NO_3-N compliance to the Protection of Aquatic Life nutrient guideline for each stream by year (1999 to 2006).

Stream	1999	2000	2001	2002	2003	2004	2005	2006	Average
	% Compliance								
Low Agricultural Intensity									
Hines Creek	100	100	100	100	100	100	100	100	100
Paddle River	100	100	100	100	100	100	100	100	100
Prairie Blood Coulee	100	100	100	100	100	100	100	94	99
Rose Creek	100	100	100	100	100	100	100	100	100
Willow Creek	100	100	100	100	100	100	100	100	100
Moderate Agricultural Intensity									
Blindman River	100	100	100	100	100	100	100	100	100
Grande Prairie Creek	92	89	100	88	100	100	100	100	96
Kleskun Drain	100	100	100	100	100	100	100	100	100
Meadow Creek	100	100	100	100	100	100	100	100	100
Tomahawk Creek	100	100	100	100	100	100	100	100	100
Trout Creek	100	100	100	100	100	100	100	100	100
High Agricultural Intensity									
Buffalo Creek	100	100	100	100	100	100	100	100	100
Haynes Creek M1	96	91	100	67	90	100	94	89	91
Haynes Creek M6	100	100	100	100	100	no data	100	78	97
Ray Creek	100	100	100	100	100	100	100	100	100
Renwick Creek	100	100	100	100	100	91	100	93	98
Strawberry Creek	96	100	100	100	100	100	100	100	100
Stretton Creek	100	100	no data	no data	no data	100	100	100	100
Threehills Creek	100	100	100	100	100	100	100	100	100
Wabash Creek	100	100	100	100	100	100	100	100	100
Irrigation Streams									
Battersea Drain	87	76	81	88	80	92	95	67	83
Crowfoot Creek	100	94	100	100	100	100	100	100	99
Drain S-6	100	100	100	100	100	100	100	100	100
New West Coulee	100	100	100	100	95	100	100	100	99

Table A5.5. Percent NH_3-N compliance to the Protection of Aquatic Life nutrient guideline for each stream by year (1999 to 2006).

Stream	1999	2000	2001	2002	2003	2004	2005	2006	Average
				% Compliance					
Low Agricultural Intensity									
Hines Creek	100	100	100	100	100	95	100	100	99
Paddle River	100	100	100	100	100	100	100	100	100
Prairie Blood Coulee	100	100	100	88	100	100	100	100	99
Rose Creek	100	100	100	100	100	100	100	100	100
Willow Creek	100	100	100	100	100	100	100	100	100
Moderate Agricultural Intensity									
Blindman River	97	100	100	100	100	100	100	100	100
Grande Prairie Creek	100	95	100	100	100	100	100	100	99
Kleskun Drain	100	100	100	100	100	100	100	100	100
Meadow Creek	100	85	100	100	100	95	95	96	96
Tomahawk Creek	100	100	100	100	100	100	100	100	100
Trout Creek	100	100	100	100	100	100	100	96	100
High Agricultural Intensity									
Buffalo Creek	100	91	44	75	94	94	96	94	86
Haynes Creek M1	100	100	94	94	89	72	78	94	90
Haynes Creek M6	100	90	56	100	85	no data	100	90	89
Ray Creek	97	96	92	100	100	100	100	100	98
Renwick Creek	100	100	100	100	100	100	100	100	100
Strawberry Creek	81	97	100	100	100	100	94	100	97
Stretton Creek	100	100	no data	no data	no data	100	100	100	100
Threehills Creek	94	92	92	100	86	100	100	89	94
Wabash Creek	100	89	100	88	100	92	87	93	94
Irrigation Streams									
Battersea Drain	87	100	88	81	90	68	71	67	82
Crowfoot Creek	100	100	100	94	100	95	95	95	97
Drain S-6	93	100	100	93	94	100	100	100	98
New West Coulee	100	100	100	100	95	84	100	95	97

APPENDIX 6: FWMC SEASONALITY BOXPLOTS

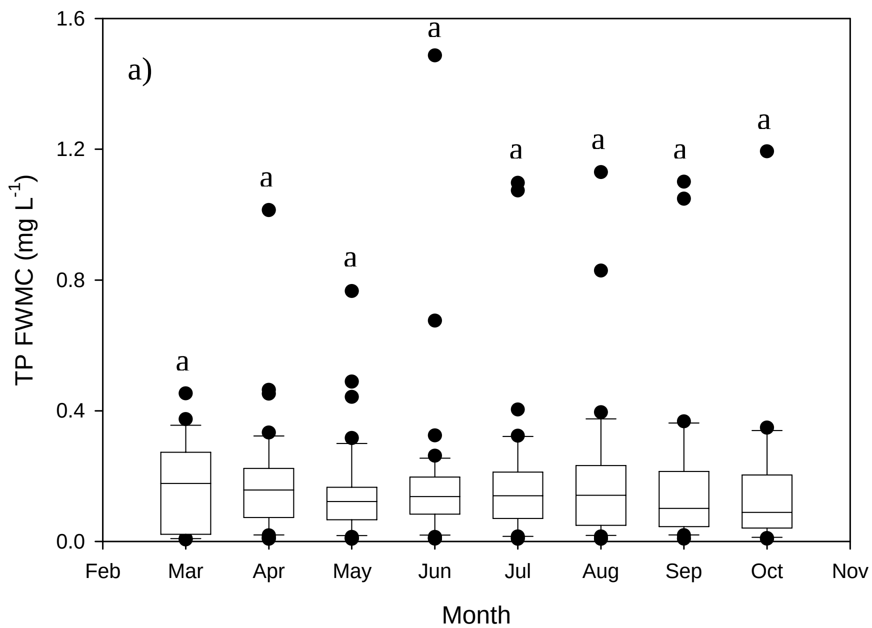
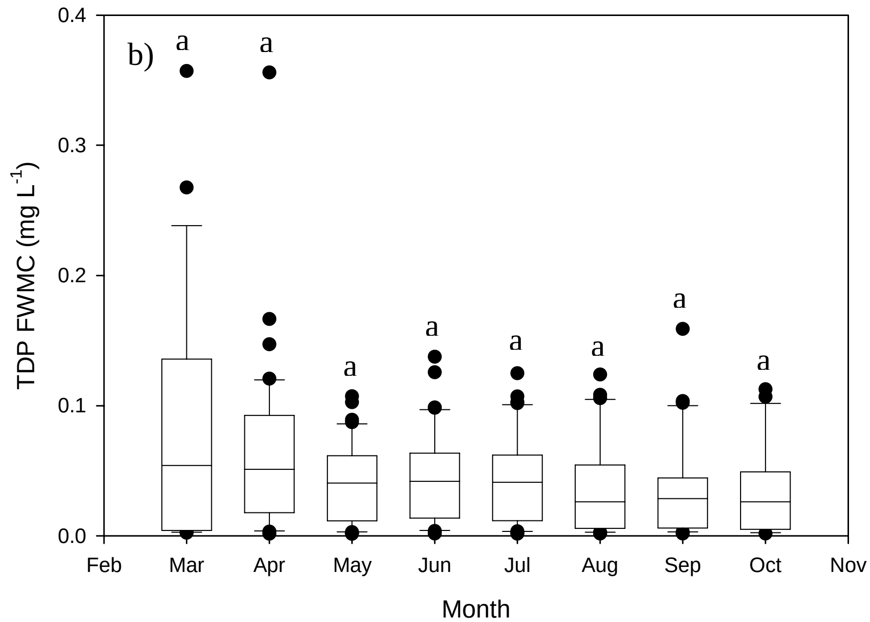


Figure A6.1. Seasonal trends in median monthly (1999-2006) TP (a) FWMCs for streams draining low agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.



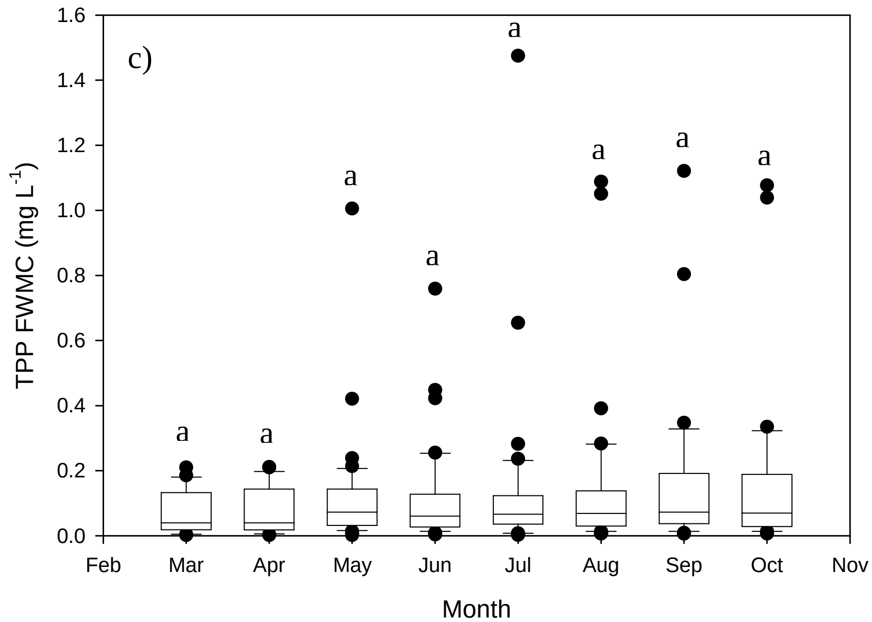


Figure A6.1. cont. Seasonal trends in median monthly (1999-2006) TDP (b) and TPP (c) FWMCs for streams draining low agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

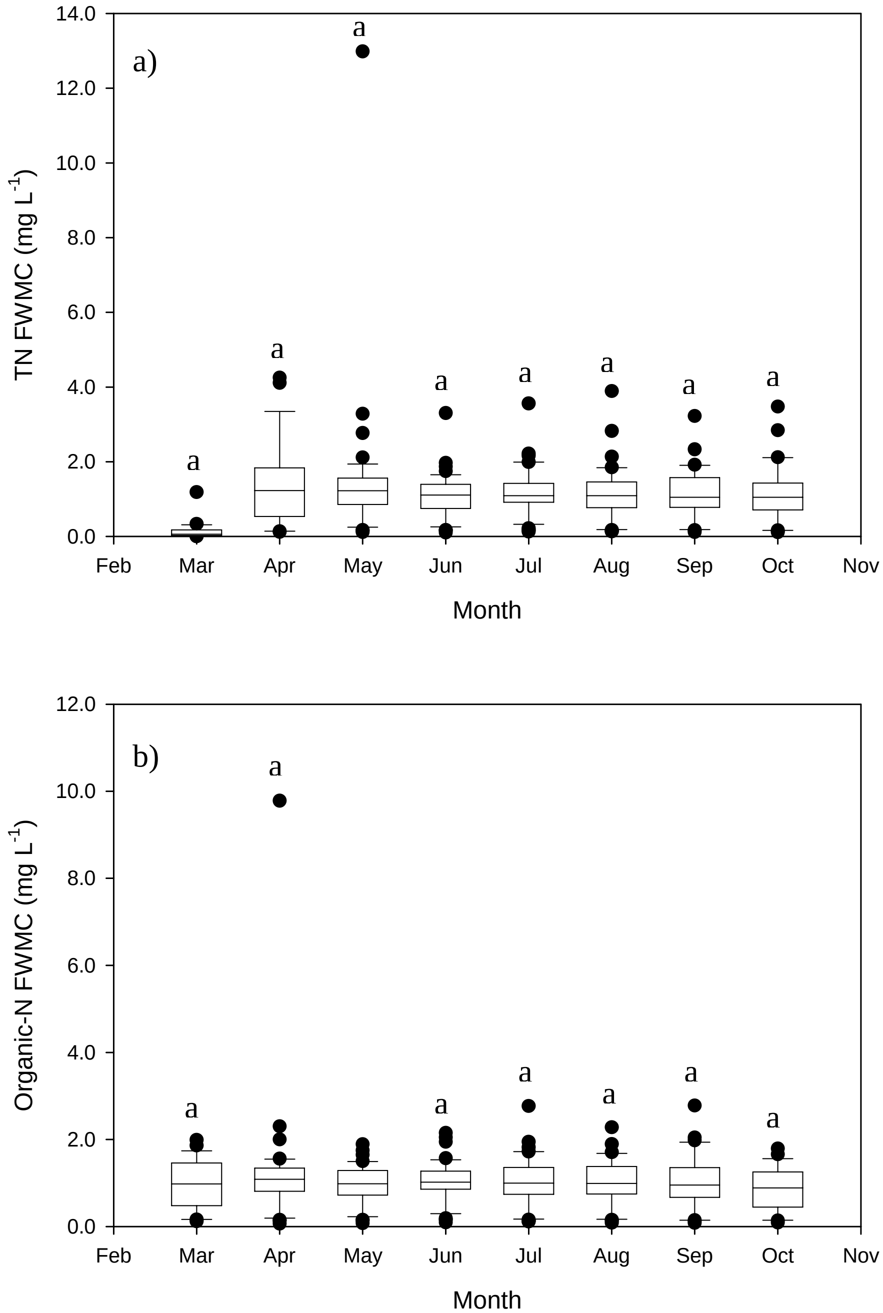
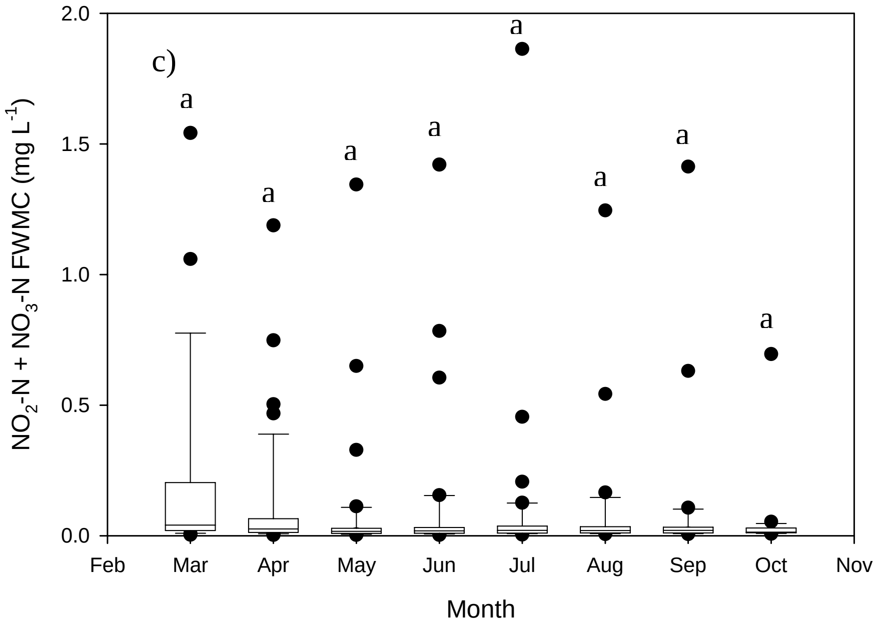


Figure A6.2. Seasonal trends in median monthly (1999-2006) TN (a) and Org N (b) FWMCs for streams draining low agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.



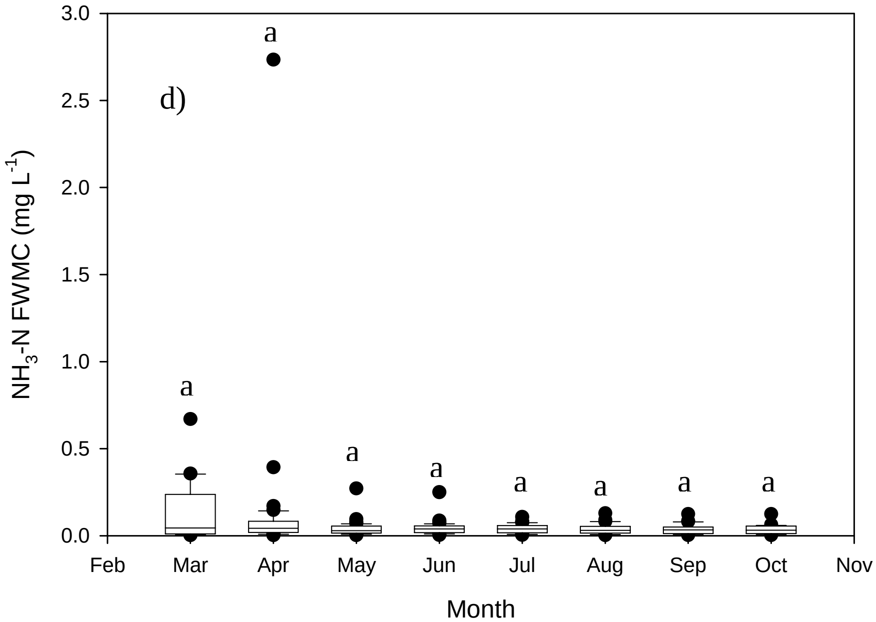


Figure A6.2. cont. Seasonal trends in median monthly (1999-2006) NO_2-NO_3-N (c) and NH_3-N (d) FWMCs for streams draining low agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

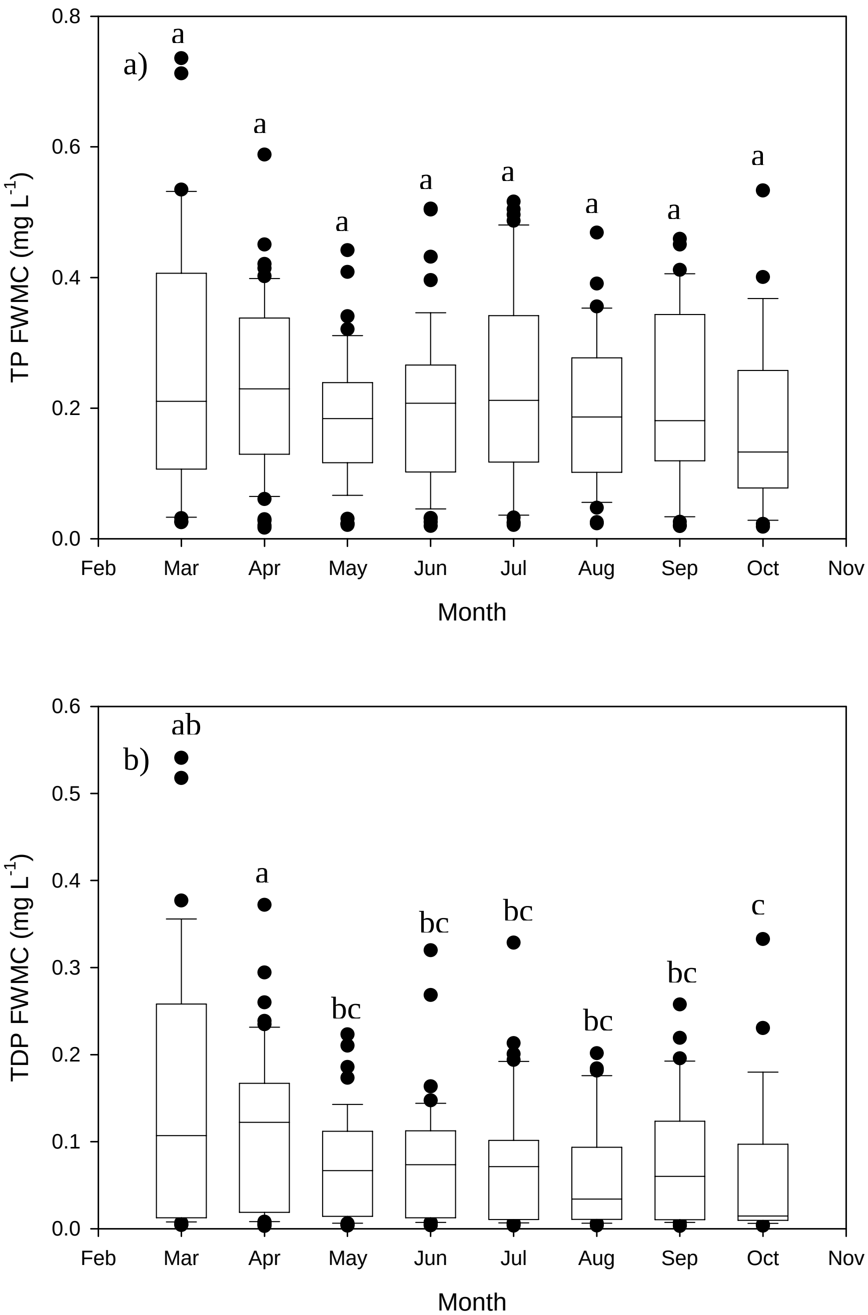


Figure A6.3. Seasonal trends in median monthly (1999-2006) TP (a) and TDP (b) FWMCs for streams draining moderate agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Medians of box plots with different letters were significantly different at the p<0.10 level.

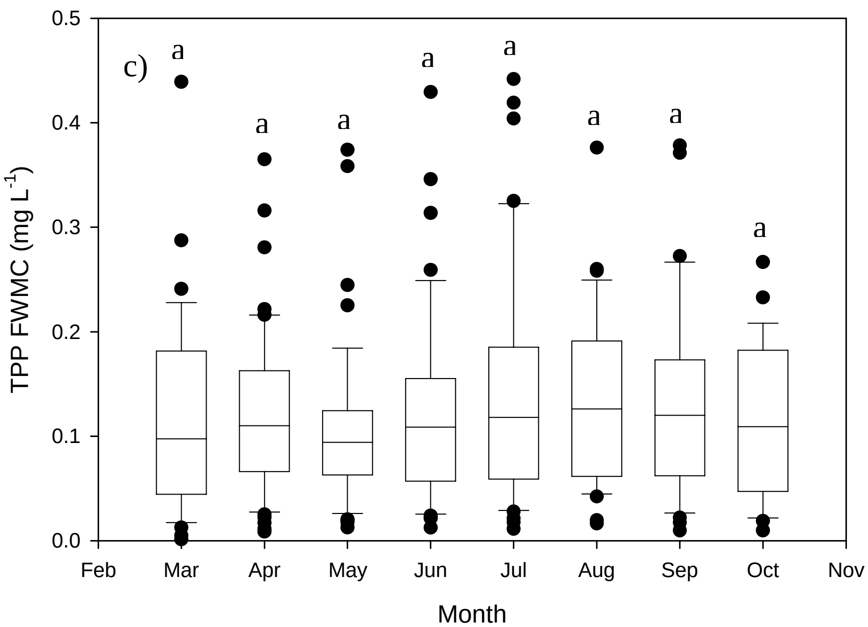
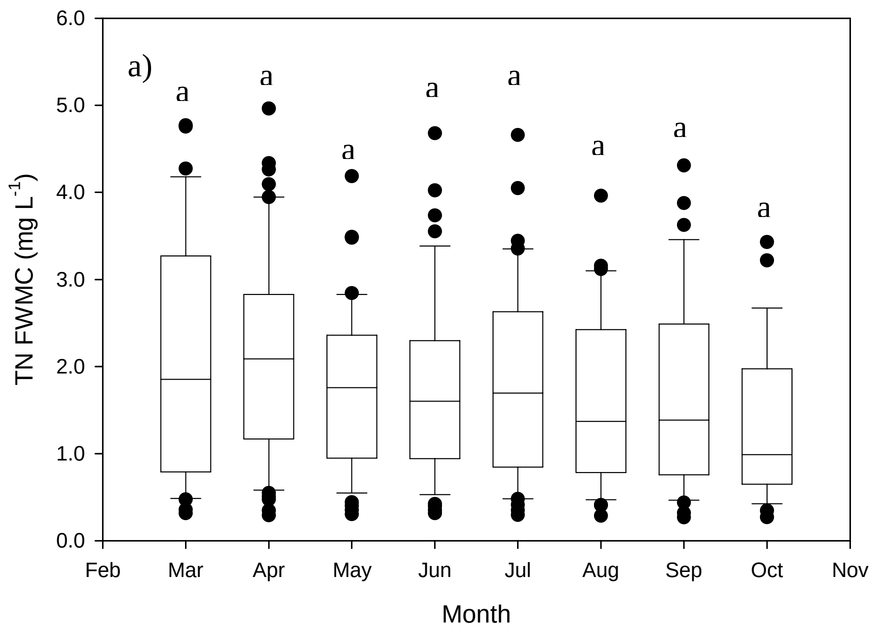


Figure A6.3 continued. Seasonal trends in median monthly (1999-2006) TPP (c) FWMCs for streams draining moderate agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.



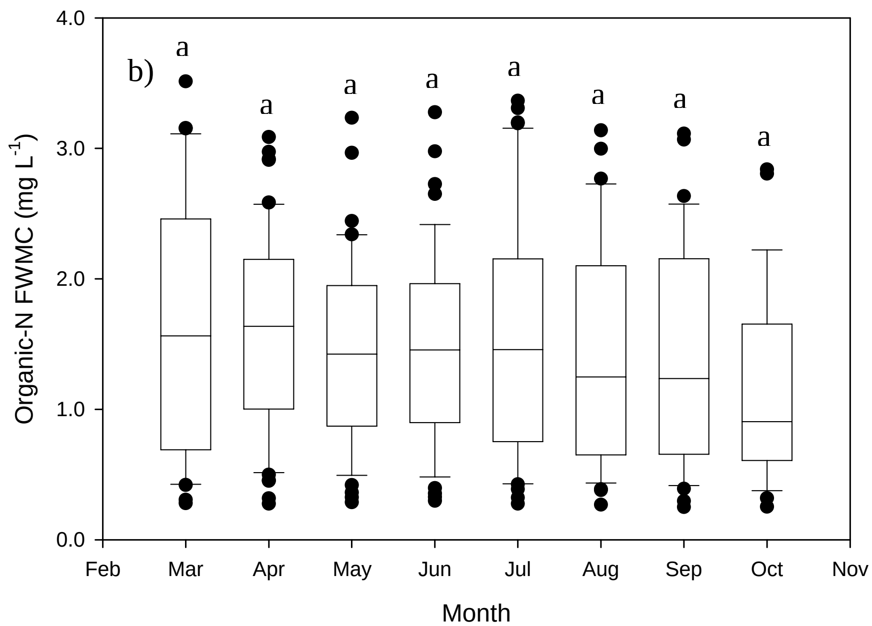
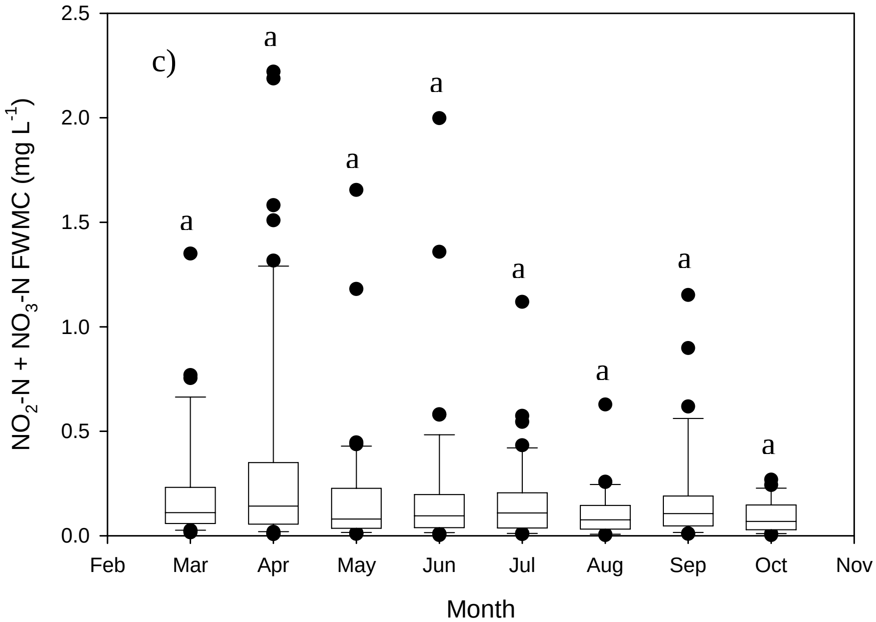


Figure A6.4. Seasonal trends in median monthly (1999-2006) TN (a) and Org N (b) FWMCs for streams draining moderate agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.



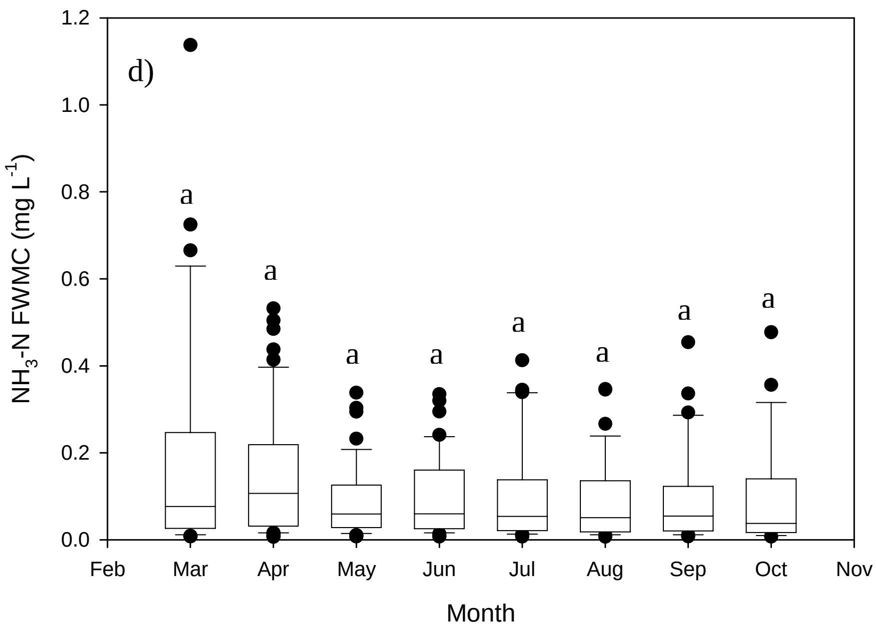


Figure A6.4 continued. Seasonal trends in median monthly (1999-2006) NO_2-NO_3-N (c) and NH_3-N (d) FWMCs for streams draining moderate agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

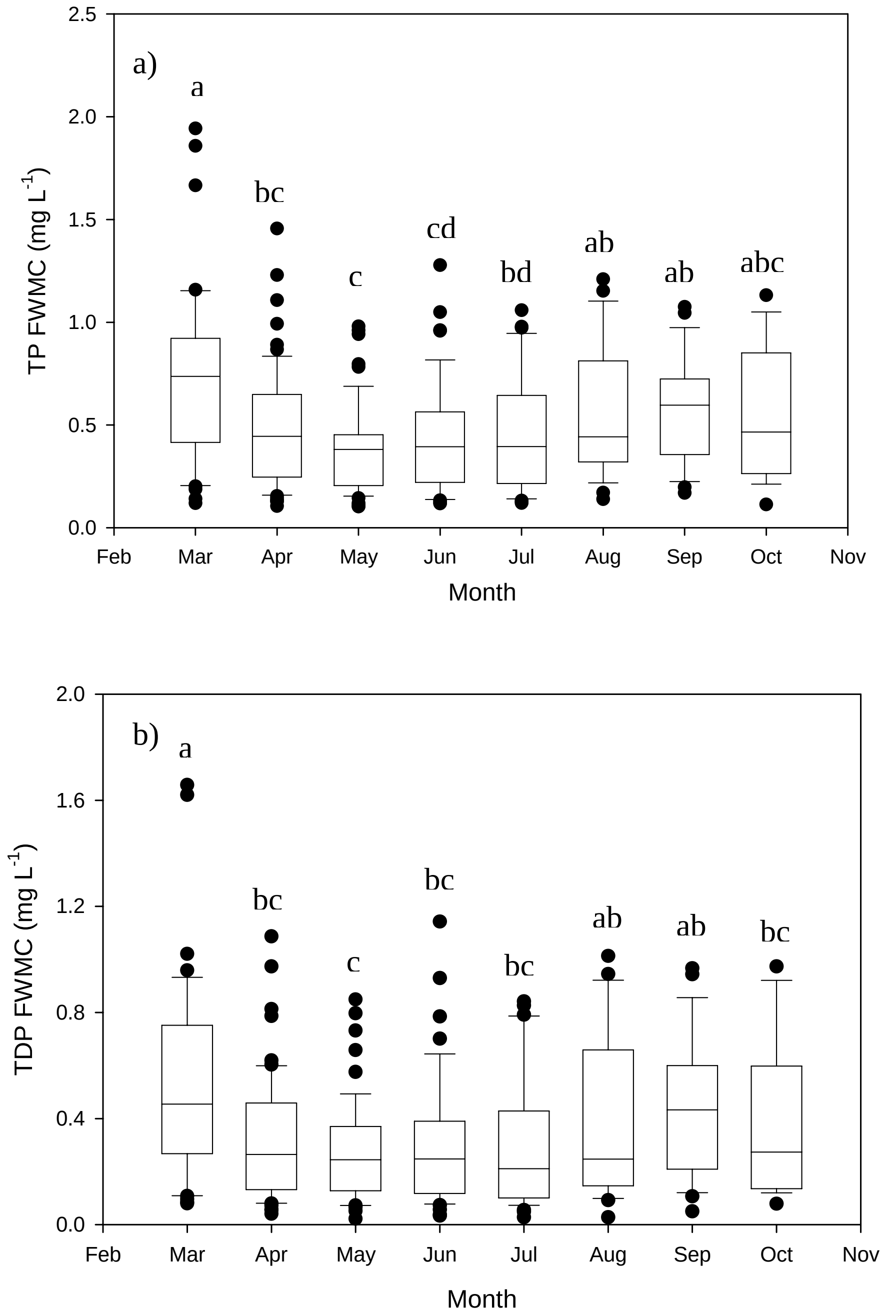


Figure A6.5. Seasonal trends in median monthly (1999-2006) TP (a) and TDP (b) FWMCs for streams draining high agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Medians of box plots with different letters were significantly different at the p<0.05 level.

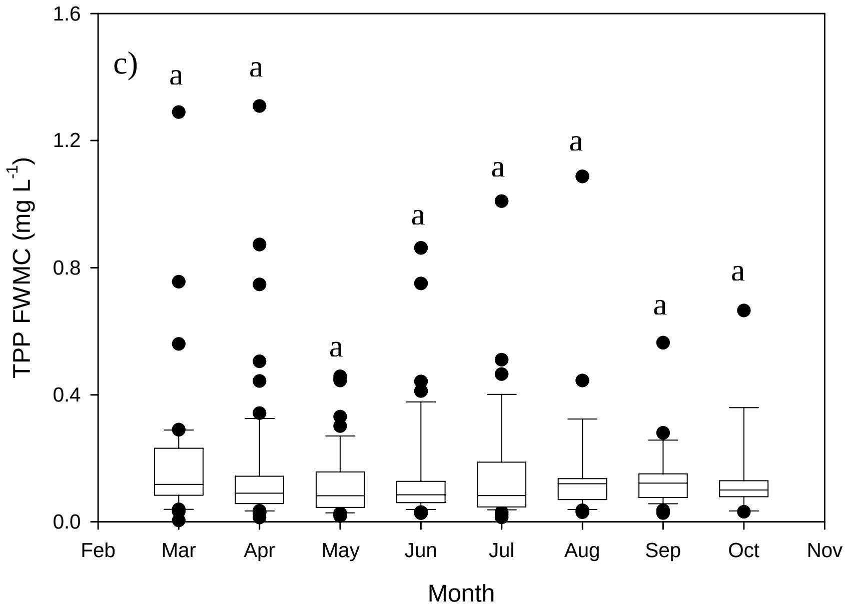


Figure A6.5 continued. Seasonal trends in median monthly (1999-2006) TPP (c) FWMCs for streams draining high agricultural intensity watersheds. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Medians of box plots with different letters were significantly different at the p<0.05 level.

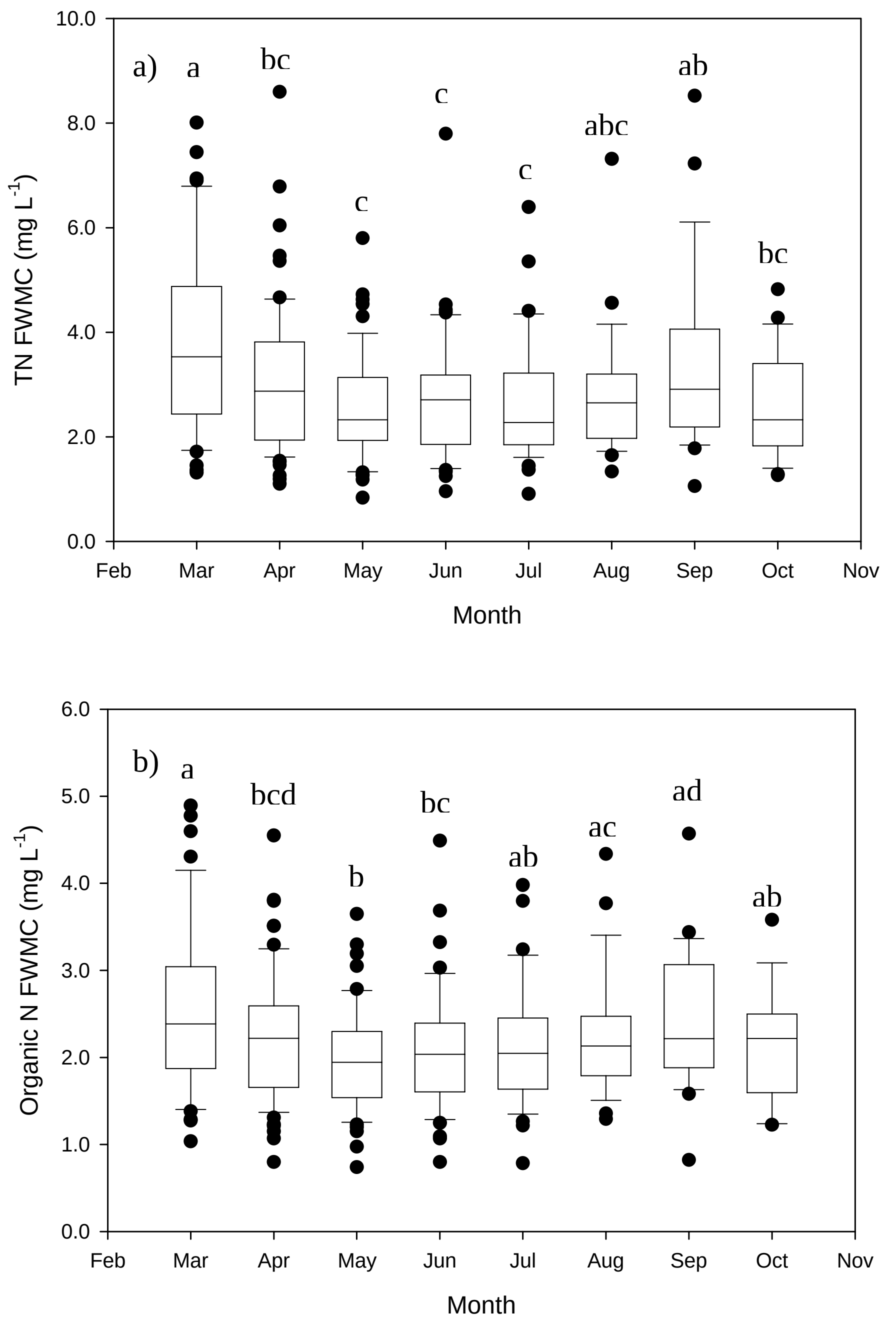


Figure A6.6. Seasonal trends in median monthly (1999-2006) TN (a) and Org N (b) FWMCs for streams draining high agricultural intensity watersheds. Medians of box plots with different letters were significantly different at the $p<0.05$ level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

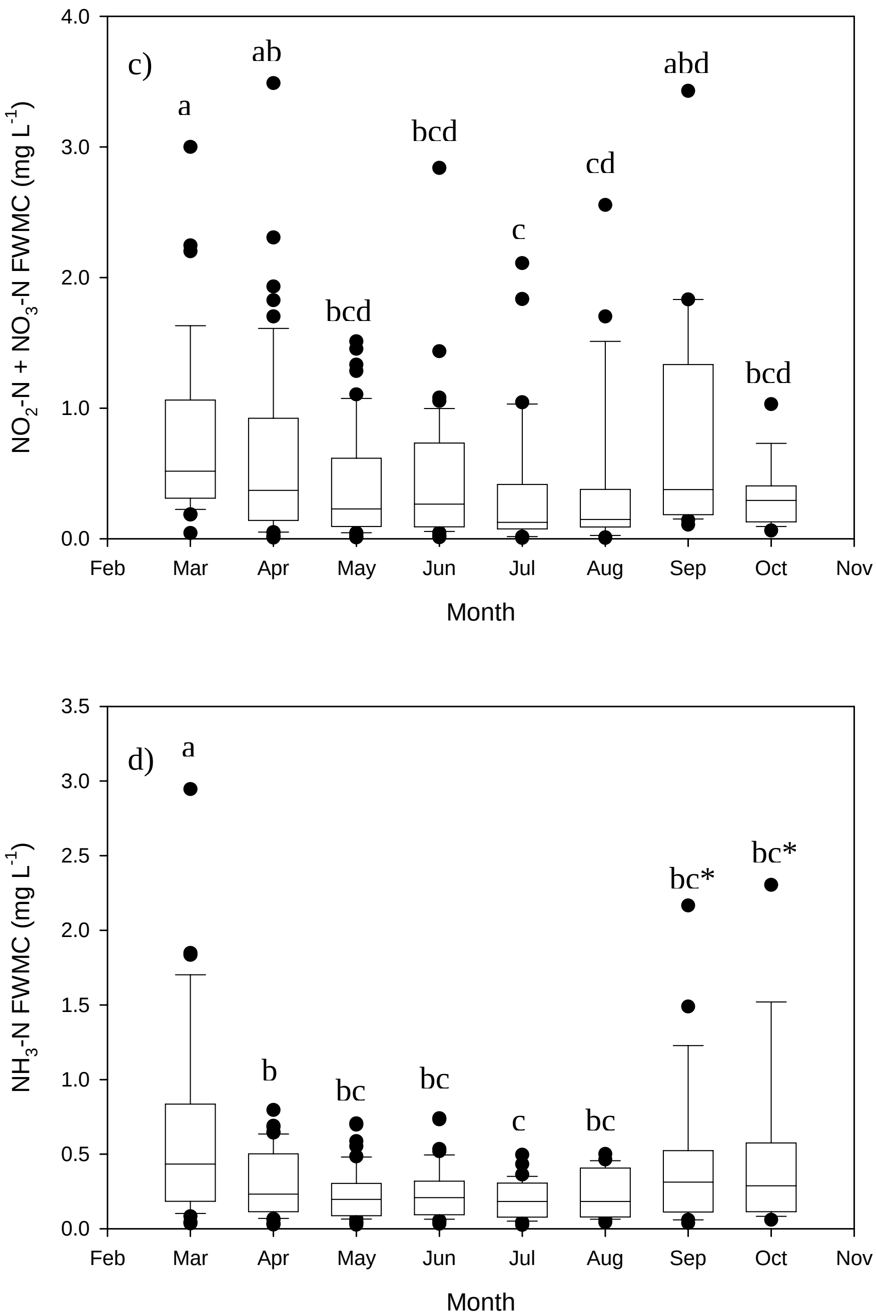


Figure A6.6 continued. Seasonal trends in median monthly (1999-2006) NO_2-NO_3-N (c) and NH_3-N (d) FWMCs for streams draining high agricultural intensity watersheds. Medians of box plots with different letters were significantly different at the $p<0.05$ level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data (* $p<0.1$).

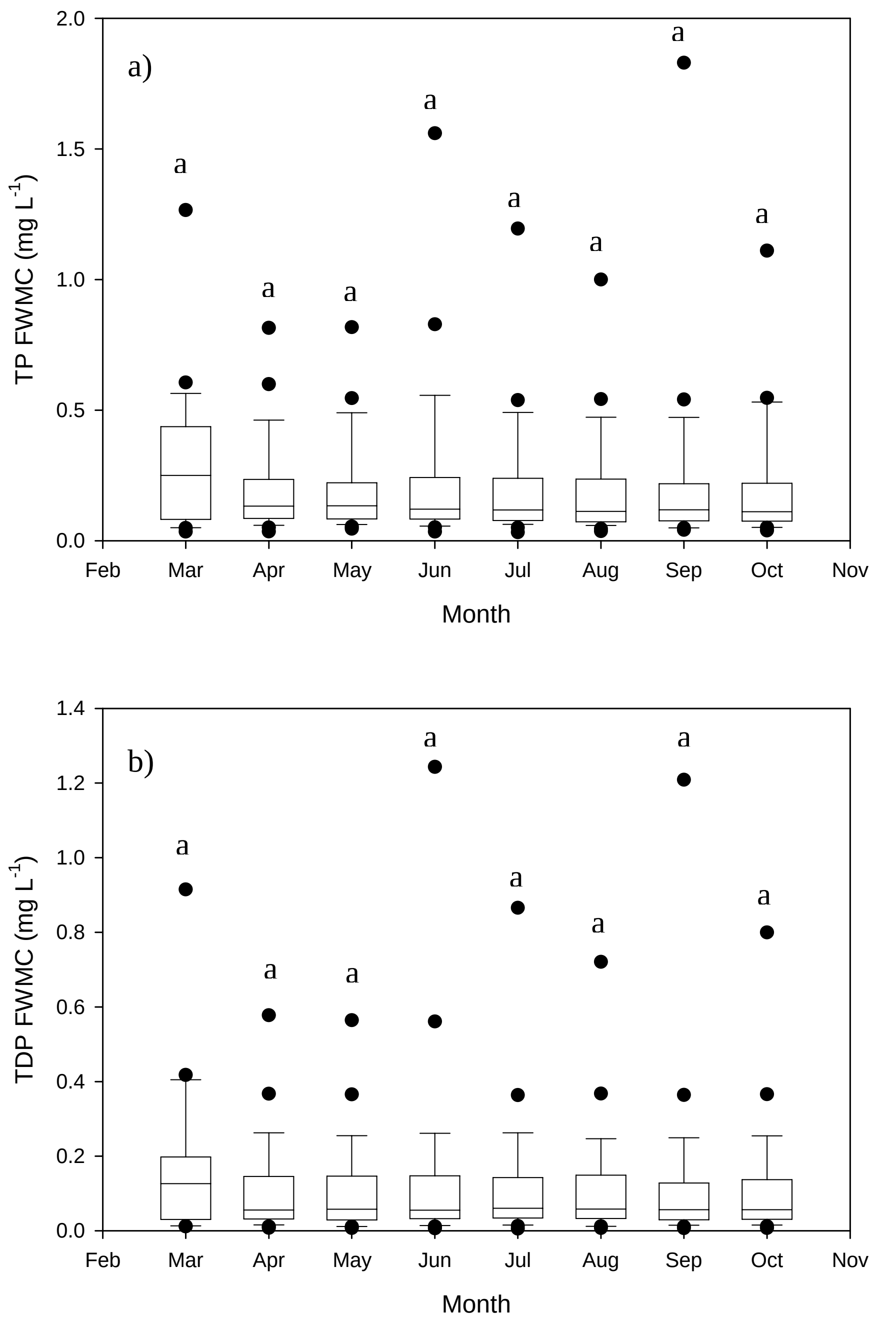


Figure A6.7. Seasonal trends in median monthly (1999-2006) TP (a) and TDP (b) FWMCs for watersheds draining irrigation return flows. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.

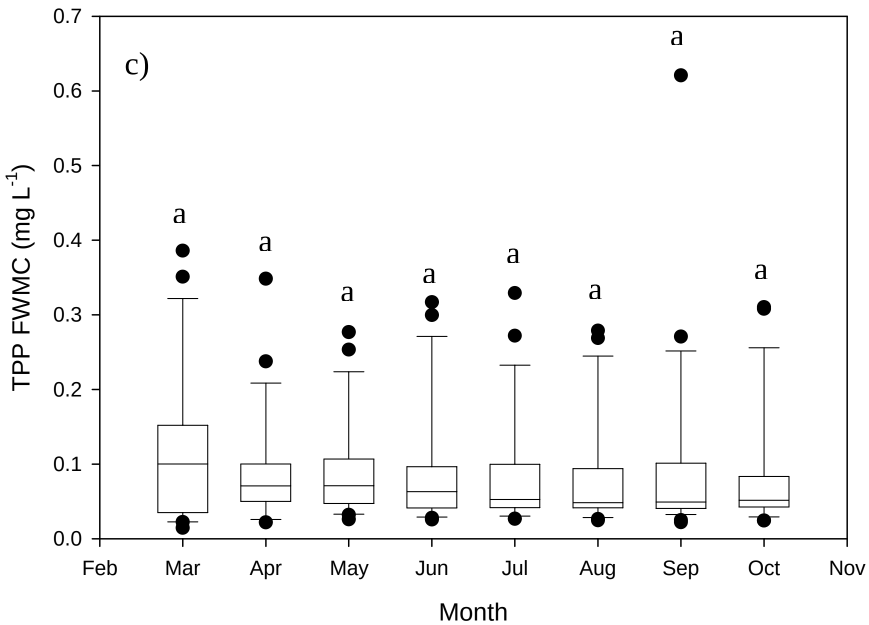
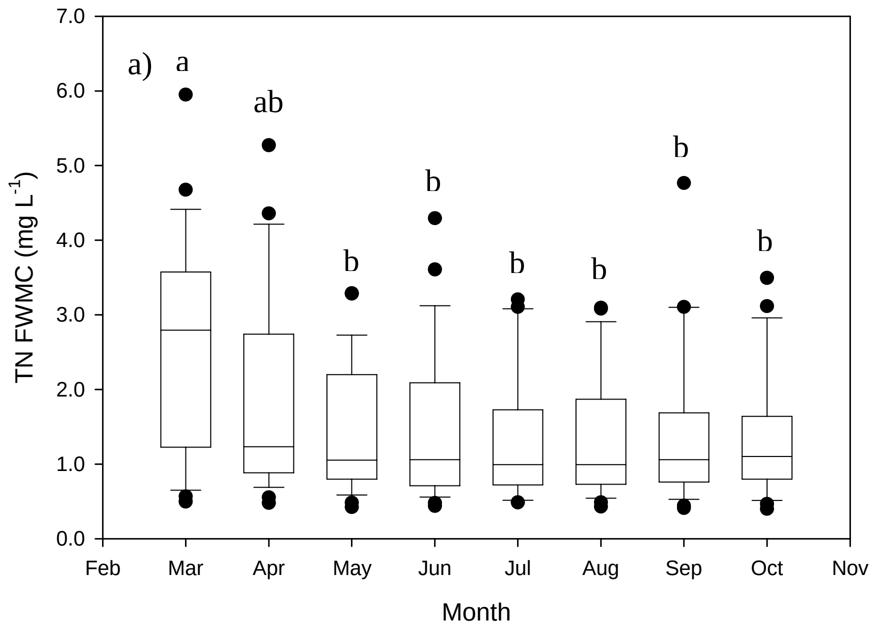


Figure A6.7 continued. Seasonal trends in median monthly (1999-2006) TPP (c) FWMCs for watersheds draining irrigation return flows. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data.



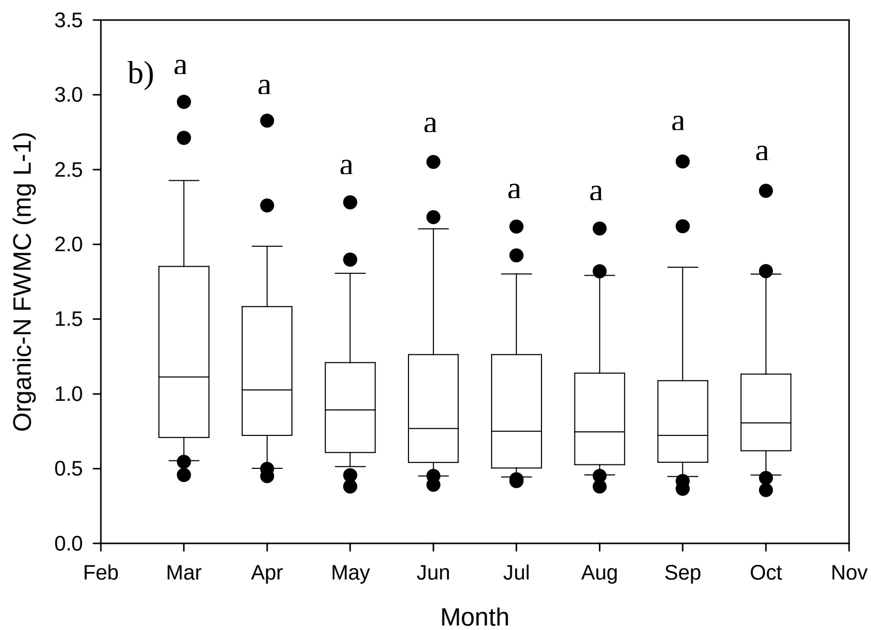


Figure A6.8. Seasonal trends in median monthly (1999-2006) TN (a) and Org N (b) FWMCs for watersheds draining irrigation return flows. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Medians of box plots with different letters were significantly different at the $p<0.05$ level.

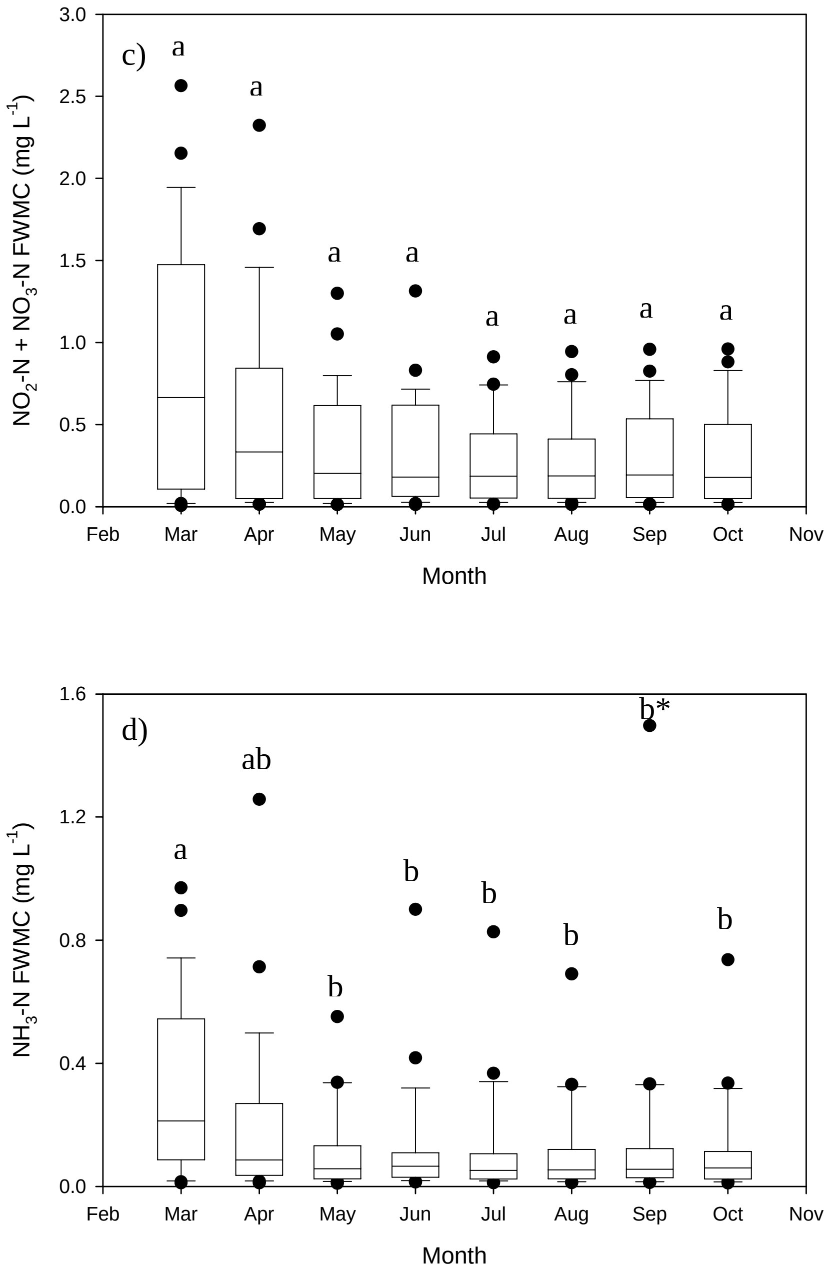


Figure A6.8 continued. Seasonal trends in median monthly (1999-2006) NO_2-NO_3-N (c) and NH_3-N (d) FWMCs for watersheds draining irrigation return flows. Medians of box plots with the same letter are not significantly different from one another at the 0.10 level as tested with Kruskal-Wallis One-Way ANOVA and Mann-Whitney significance tests on untransformed data. Medians of box plots with different letters were significantly different at the $p<0.05$ level (* $p<0.10$).

APPENDIX 7: MASS TRANSPORT AND TEMPORAL PATTERNS

Mass Transport of Phosphorus and Nitrogen

It should be noted that this section describes temporal patterns and does not include direct comparisons of nutrient loads among the watersheds of different sizes as a standardization factor for discharge and/or watersheds area is needed. These comparisons are made in the flow-weighted mean concentration (FWMC) section where instream concentrations were standardized by flow (units), and export coefficient section where the median annual loads were divided by effective drainage areas.

Temporal patterns by ecoregion. Annual loads are presented in this section for each individual stream and grouped by ecoregion in a north to south fashion for ease of discussion. Within an ecoregion the frequency and timing of flood peaks and low flow events are often similar, as the region is influenced by the same weather patterns. Discussions within an ecoregion generally work from northern watersheds to southern watersheds.

Boreal Ecoregion

Hines Creek- From 1999 through 2006 there was no apparent increase in phosphorus and nitrogen loads from Hines Creek (Figure A7.1). Among years annual stream volumes varied considerably, with low flow in 1999 (0.2 hm^3) and peak low in 2003 (22.6 hm^3). Similarly, the lowest and highest annual loads (in kg) for all nutrient parameters (Total N, organic N, NH_3-N, TP, TDP) except NO_2+NO_3-N and TPP were observed in 1999 and 2003, respectively (Table A7.1). NO_2+NO_3-N and TPP loads were also lowest in 1999, but peaked in 2001.

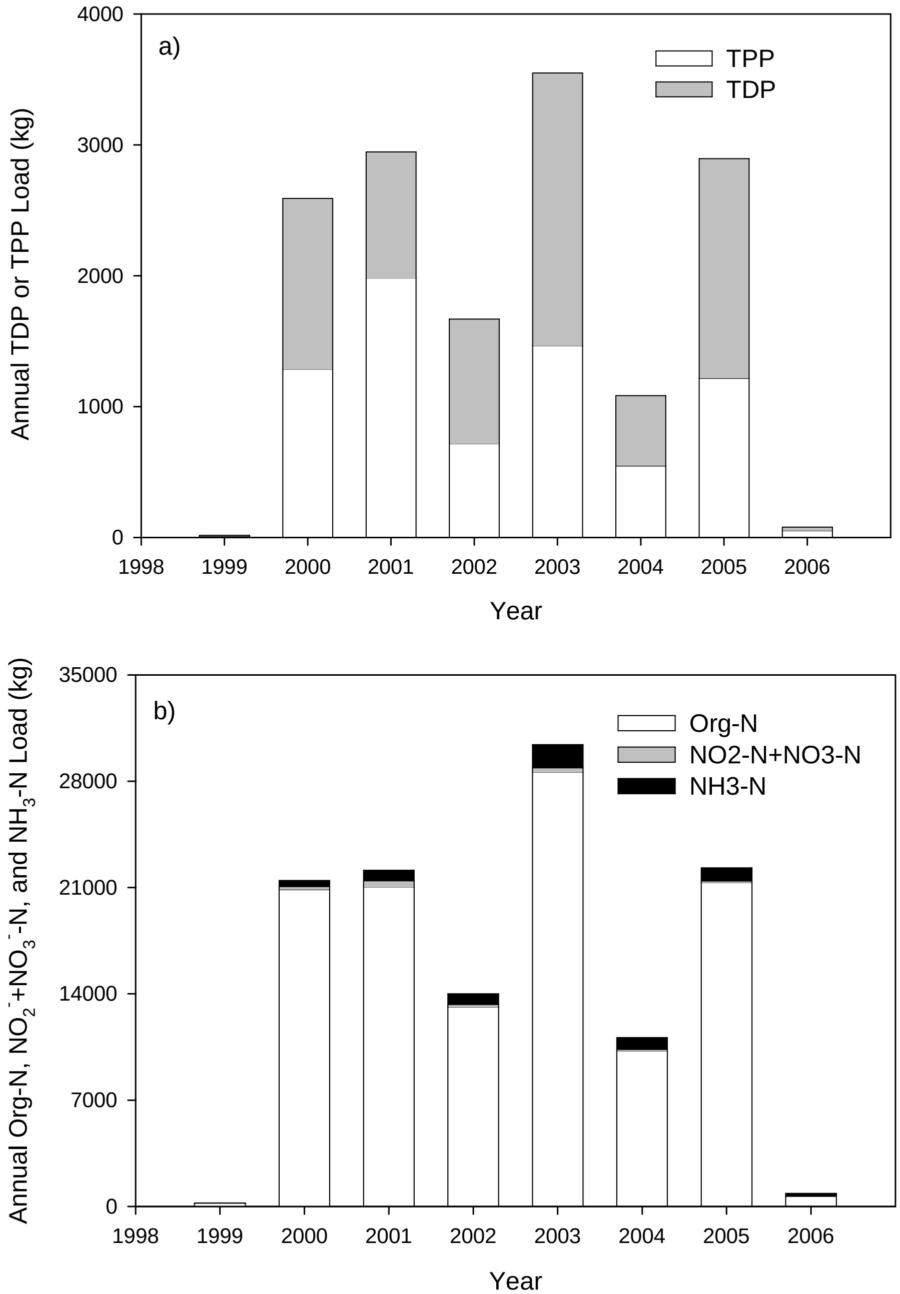
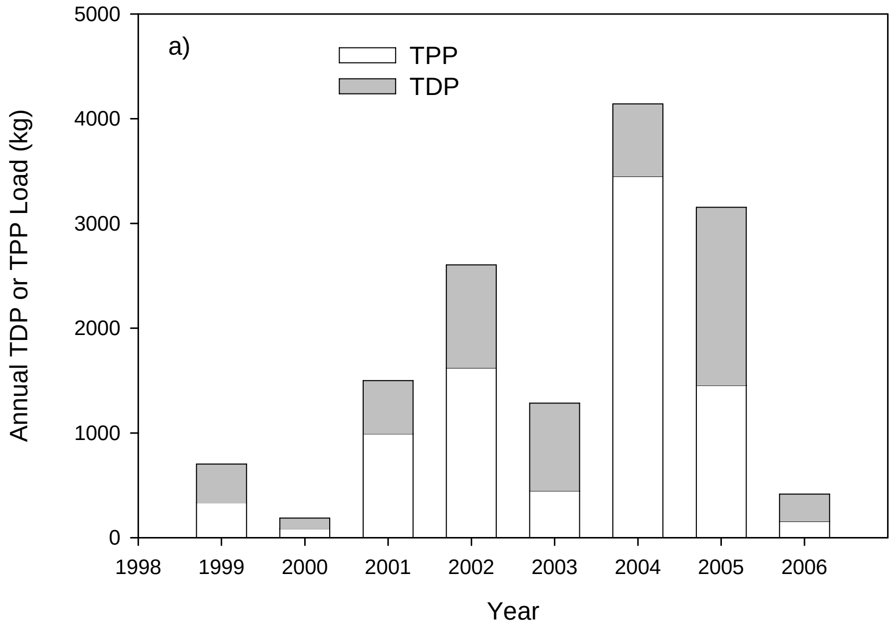


Figure A7.1. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1999 through 2006 in Hines Creek, a low agricultural intensity watershed in the Boreal ecoregion.

Table A7.1. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Hines Creek, 1999 to 2006.

	Median ($kg\ yr^{-1}$)	Minimum ($kg\ yr^{-1}$)	Year	Maximum ($kg\ yr^{-1}$)	Year
Total N	17725	221		30415	
Organic N	17000	217		28609	2003
NH_3-N	700	4		1532	
NO_2+NO_3-N	128	1	1999	3400	2001
Total P	2129	16		3550	2003
TPP	967	8		1983	2001
TDP	956	8		2084	2003
Annual stream volume (hm^3)	14.1	0.2	1999	22.6	2003

Grande Prairie Creek- Total P and TN loads in Grande Prairie Creek appeared to increase from 1999 through 2006 although a substantial drop in nutrient loads was observed in 2006 (Figure A7.2). It appeared that TP and TN loads in Grande Prairie Creek were influenced by stream flow, sampling regime, and change in agricultural intensity. Nitrogen and P loading were influenced by stream flow as low annual flow volumes coincided with low N and P load in 2000 and 2006 and higher annual stream flow volume in 2002 and 2004 coincided with higher loading (Table A7.2). Median annual loads were lower in 2000 and 2001, but these loads also incorporated samples collected in September and October. Increased sampling in the fall may have resulted in the lower P loads especially considering sampling in other years was primarily between April and June (See Chapter 3: Results and Discussion, Export Coefficient Seasonality). The TP load in 2004 was interesting in that stream flow volumes in 2004 were not as high as previous years; however, the load was the highest observed (Table A7.2). High TN and TP loading in 2004 may have been related to a change in agricultural intensity as agricultural intensity in the Grande Prairie watershed dropped from being of a moderate rank in 1996 to a low rank in 2001 then back up to a moderate rank in 2006.



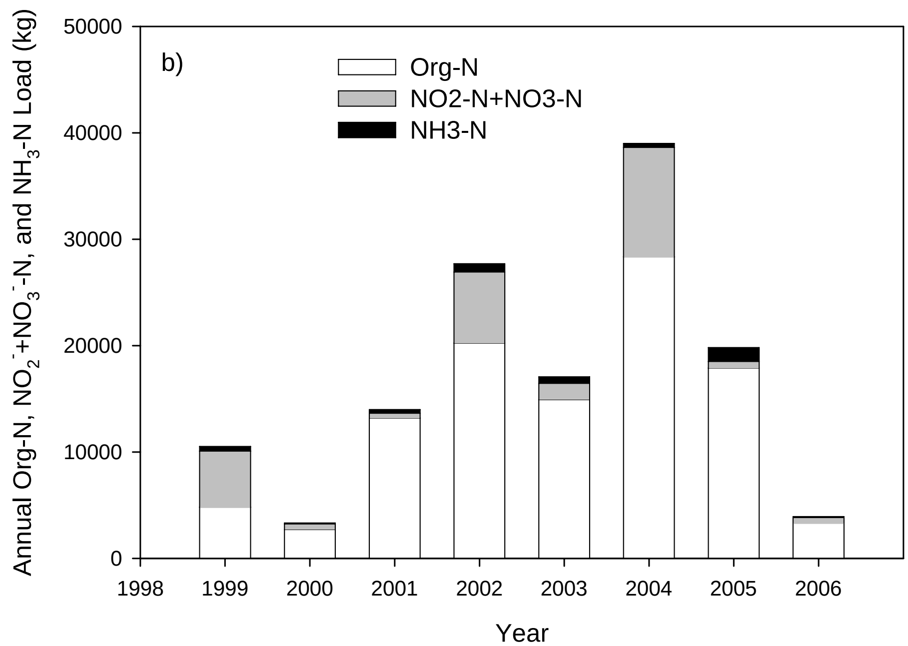


Figure A7.2. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1999 to 2006 in Grande Prairie Creek. Grande Prairie Creek is a moderate agricultural intensity watershed that lies in the boreal ecoregion.

Table A7.2. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Grande Prairie Creek, 1999 to 2006.

	Median ($kg\ yr^{-1}$)	Minimum ($kg\ yr^{-1}$)	Year	Maximum ($kg\ yr^{-1}$)	Year
Total N	15543	3337	2000	39505	2004
Organic N	14055	2717		28347	
NH_3-N	405	98		1312	2005
NO_2+NO_3-N	1065	461	2001	10269	
Total P	1391	187		4141	2004
TPP	719	87	2000	3448	
TDP	601	100		3154	2005
Annual stream volume (hm^3)	7.4	1.5	2000	12.1	2005

Kleskun Drain- There was no temporal TP or TN loading pattern observed in the Kleskun Main Drain watershed from 1999 through 2006 (Figure A7.3). The annual flow volumes in Kleskun Main Drain were also variable from 1999 through 2006, however, years of low N and P loading were correlated to years of low annual flow volumes in 2000 (0 hm^3) and 2006 (0.1 hm^3). Similarly, 2002 and 2003 were peak years in nitrogen loading and annual flow volume at 2.5 hm^3 and 2.4 hm^3 respectively (Table A7.3). The lack of a trend in Kleskun Main Drain is interesting as all agricultural intensity factors decreased from 1996 through 2001 and 2006 according to Census of Agriculture data. On the other hand, the agricultural intensity ranking based on Census of Agriculture data may not be accurate as the polygon used to determine the agricultural production in the watershed is not the same area as the actual watershed (Chapter 2, Table 2.1). To alleviate this concern, ground-truthing of the watershed and/or gaining a better understanding of the agricultural practices in the Kleskun Drain watershed would assist in the explanation of the observed loading. Total P loads were strongly correlated with TDP (0.93, $p<0.05$) and TPP (1.00, $p<0.05$) loads. Phosphorus loading did not appear to be influenced by peaks in stream flow; however, annual flow volumes were positively correlated with TP (0.98, $p<0.05$), TDP (0.91, $p<0.05$), and TPP (0.98, $p<0.05$).

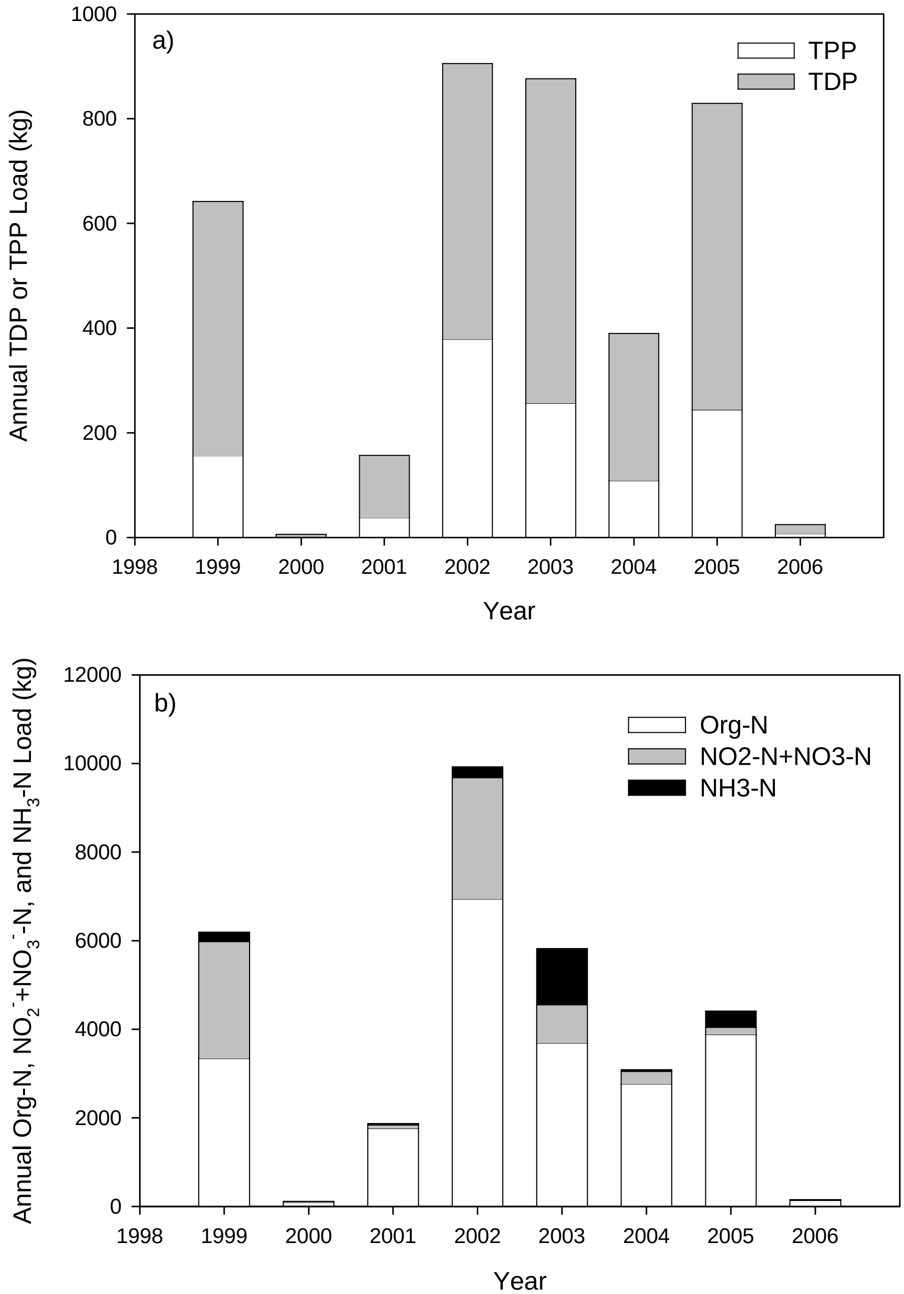
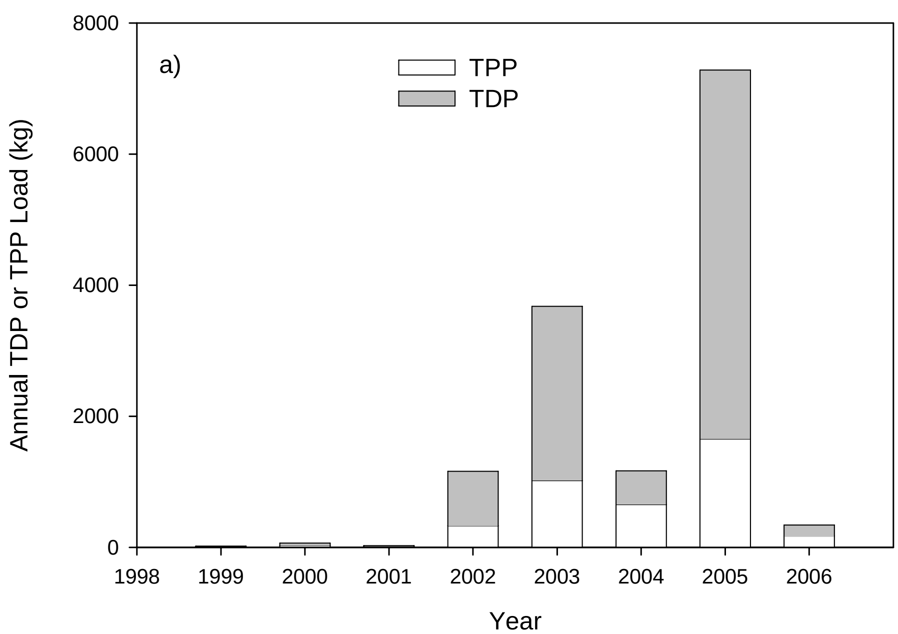


Figure A7.3. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1999 to 2006 in Kleskun Main Drain. Kleskun Main Drain is a moderate agricultural intensity watershed that lies in the boreal ecoregion.

Table A7.3. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Kleskun Main Drain, 1999 to 2006.					
	Median ($kg\ yr^{-1}$)	Minimum ($kg\ yr^{-1}$)	Year	Maximum ($kg\ yr^{-1}$)	Year
Total N	3750	1145	2000	9960	2002
Organic N	3052	92		6937	
NH_3-N	130	4		1269	2003
NO_2+NO_3-N	220	6	2006	2741	
Total P	515	6		905	2002
TPP	132	1	2000	379	
TDP	384	5		620	2003
Annual stream volume (hm^3)	1.7	0	2000	2.5	2002

Wabash Creek- Wabash Creek appeared to have a slight increasing trend in TP and TN from 1999 through 2006 (Figure A7.4) Total P and TN loads where highest with high annual flow volumes and lowest with low annual flow volumes (Table A7.4). With the correlation of nutrient loads to flow, the increasing pattern from 1999 to 2006 may be attributed to higher flows recorded in later monitoring years. Total P loads were strongly, positively correlated with both TDP (0.98, $p<0.05$) and TPP (1.00, $p<0.05$). Moreover, annual stream volume was also highly correlated with P loads at 0.98 for TP, TDP, and TPP ($p<0.05$). This was especially the case in 2003 and 2005 where extremely high TP loads were corresponding to the two highest annual flow volumes.



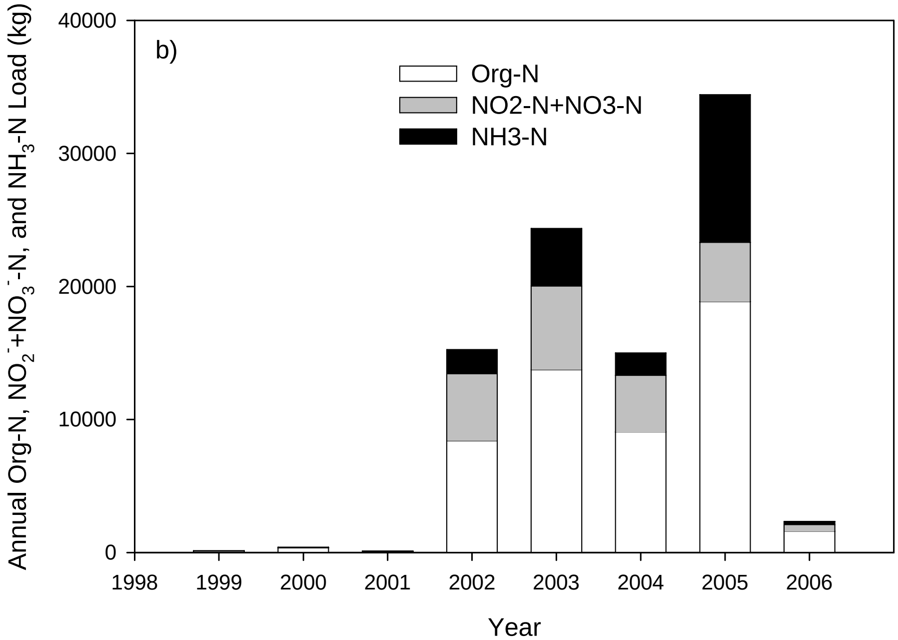
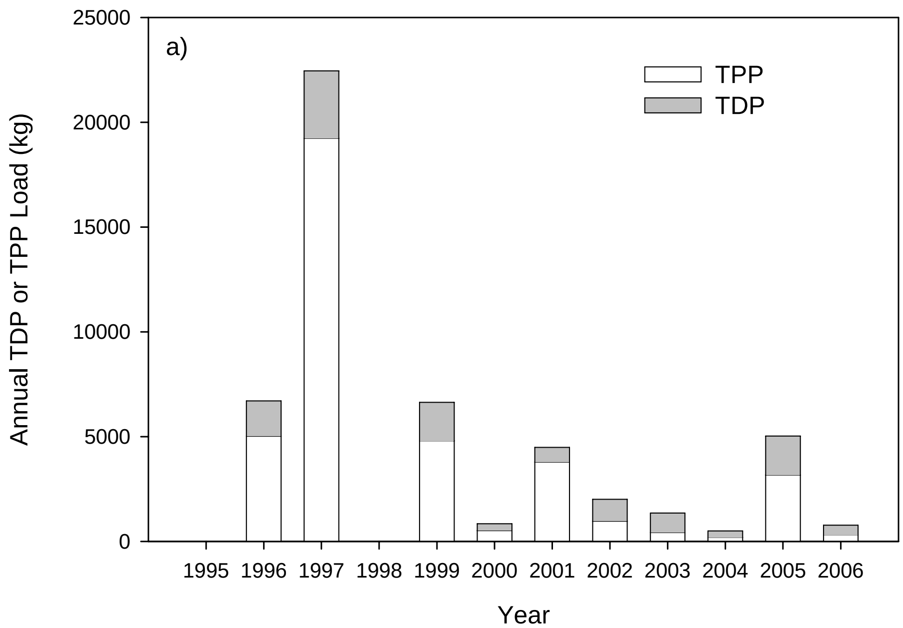


Figure A7.4. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1999 to 2006 in Wabash Creek. Wabash Creek is a high agricultural intensity watershed that lies in the boreal ecoregion.

Table A7.4. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Wabash Creek, 1999 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	8814	104	2001	34441	2005
Organic N	5002	87	2001	18864	2005
NH_3-N	955	12	2001	11098	2005
NO_2+NO_3-N	2375	5	2001	6282	2003
Total P	1159	20	1999	7282	2005
TPP	329	15	1999	1654	2005
TDP	514	5	1999	5627	2005
Annual stream volume (hm^3)	1.5	0.1	1999 and 2001	7.7	2005

Paddle River- Total phosphorus and TN loads appear to decrease from 1995 through 2006 in Paddle River (Figure A7.5), however this decrease may be attributed to the high loads observed in 1997 (Table A7.5) and declining stream flow through the monitoring period (Table A7.23). Total stream volume in 1997 was high at 46.0 hm^3 and likely contributed to the high loading observed. Relating the loads measured to stream flow we find annual flow volume in 1996 was similar to that in 1997, however the loading data in 1996 were clearly lower than 1997 (Table A7.5). Looking at sampling data it was apparent that although sampling was flow biased, there were only 6 samples taken in 1996 compared to 26 samples in 1997. Due to being under-sampled, 1996 data may not be representative of the actual loads that occurred. Assuming that the 1996 loads were higher than reported would support the apparent decrease in loading over time (Figure A7.5). A strong positive correlation existed between both stream flow and TP (0.86, $p<0.05$) and TPP (0.91, $p<0.05$) loads. Total dissolved P (0.88, $p<0.05$) and TPP (0.95, $p<0.05$) were both positively correlated with TP loading.



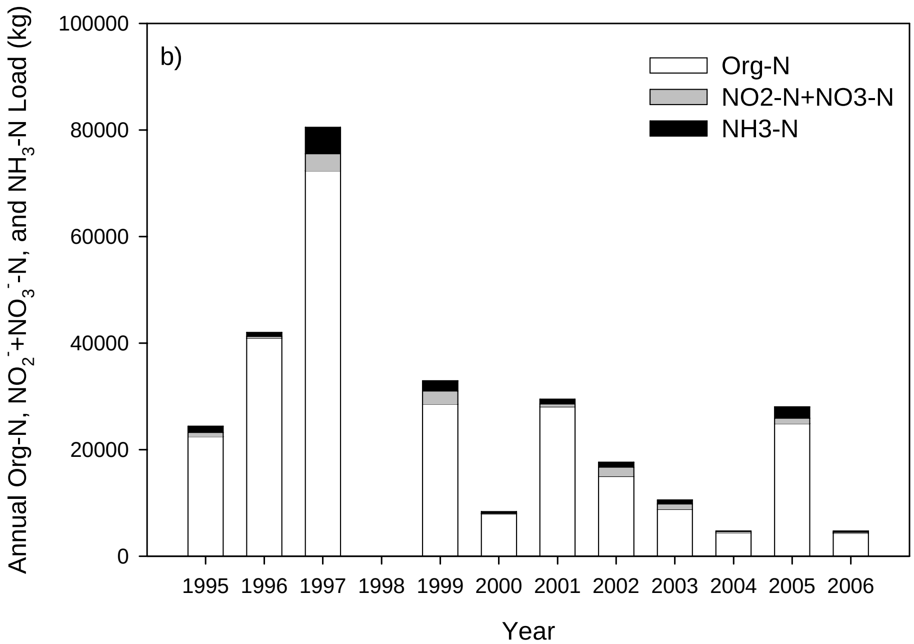


Figure A7.5. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Paddle River. Paddle River is a low agricultural intensity watershed that lies in the boreal ecoregion.

Table A7.5. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Paddle River, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	24410	4717	2004	80507	1997
Organic N	22455	4317	2006	72328	
NH_3-N	901	80	2004	4943	
NO_2+NO_3-N	775	115	2000	3225	
Total P	4486	502		22454	
TPP	2068	200	2004	19229	
TDP	987	302		3224	
Annual stream volume (hm^3)	16.4	4.4	2006	45.5	1997

Tomahawk Creek- Tomahawk Creek TP and TN loading data were variable but had a slight decreasing trend from 1995 through 2006 (Figure A7.6). This apparent decrease began with high annual flow volumes and nutrient loading in 1996 and 1997 then coincided with a decrease in annual flow volume (Table A7.6, Table A7.23). In addition to decreasing annual flow volumes, the apparent trend in decreasing P loads over the monitoring period may also be related to the decrease in agricultural intensity from a moderate rank to a low rank from 1996 to 2006 according to the Census of Agriculture (Chapter 2, Table 2.11). Spearman's rank correlations showed a strong correlation between TP and TPP (0.93, $p<0.05$) loads. Moreover, annual stream volume was correlated with TP (0.93, $p<0.05$) and TPP (0.83, $p<0.05$) loads.

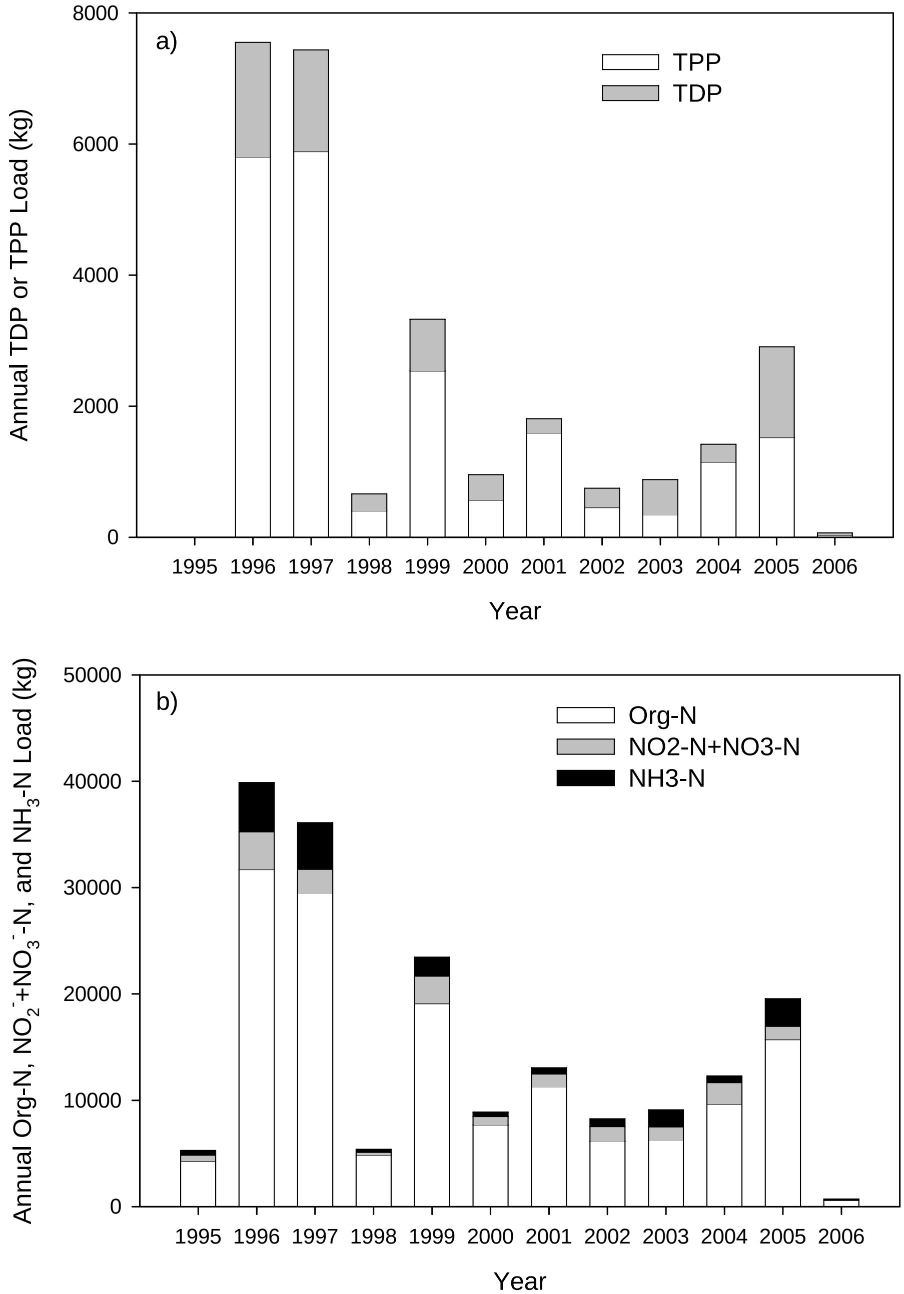
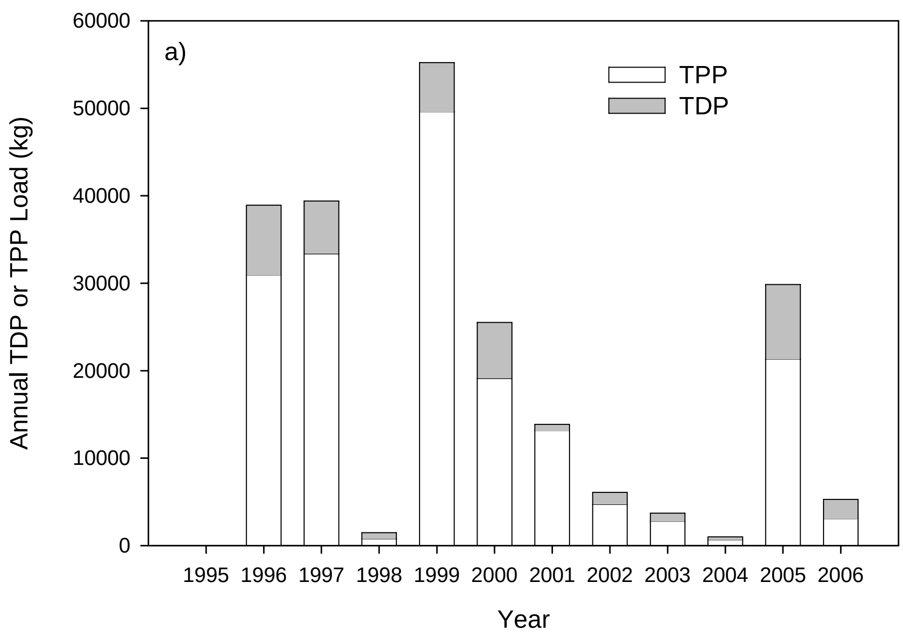


Figure A7.6. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Tomahawk Creek. Tomahawk Creek is a moderate agricultural intensity watershed that lies in the boreal ecoregion.

Table A7.6. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Tomahawk Creek, 1995 to 2006.					
	Median ($kg\ yr^{-1}$)	Minimum ($kg\ yr^{-1}$)	Year	Maximum ($kg\ yr^{-1}$)	Year
Total N	10704	708	2006	39903	1996
Organic N	8662	584		31694	
NH_3-N	685	76		4629	
NO_2+NO_3-N	1210	49		3546	
Total P	1188	67		7550	
TPP	1148	34		5884	1997
TDP	393	33		1753	1996
Annual stream volume (hm^3)	3.5	0.3	2006	12.8	1997

Strawberry Creek- There were no patterns in TP and TN loading in Strawberry Creek from 1995 to 2006 (Figure A7.7). Flows were correlated with loading as low flow volume coincided with the lowest annual loading in 2004, while high annual flow volumes coincided with the highest annual flow volumes in 1999 and 2005 (Table A7.7). A Spearman's rank correlation between annual stream volume and TP and TPP loads confirmed the observations that higher P loads occur with higher annual stream volumes as TP and TPP positively correlated at 0.91 ($p<0.05$). Further, TP loads were positively correlated with both TDP and TPP although a stronger correlation was observed for TP with TPP (1.00, $p<0.05$) than with TP and TDP (0.76, $p<0.05$).



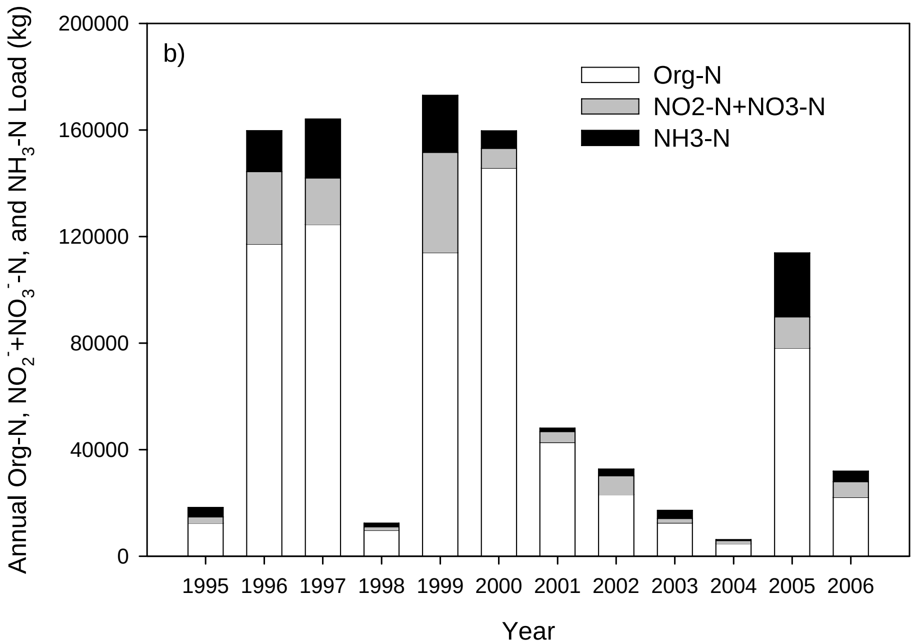


Figure A7.7. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Strawberry Creek. Strawberry Creek is a high agricultural intensity watershed that lies in the boreal ecoregion.

Table A7.7. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Strawberry Creek, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	40420	6248		173376	1999
Organic N	32929	4707		145674	2000
NH_3-N	3731	396	2004	24051	2005
NO_2+NO_3-N	6484	1132		37637	
Total P	9976	994		55229	1999
TPP	13168	684		49604	
TDP	2200	311		8548	2005
Annual stream volume (hm^3)	12.3	5.3	2004	54.3	2000

Blindman River- Between 1995 and 2006 there was no trend in TP or TN loading in the Blindman River (Figure A7.8). Flow volumes were low in 1995, 2002, 2004 and 2006 which corresponded with low TP and TN loading (Table A7.8). Given the association between annual flow volumes and nitrogen loading, we were uncertain why there was low NH_3-N and NO_2+NO_3-N loading in 2001 (Table A7.8). Total P loads were strongly, positively correlated with both TDP and TPP (0.93, $p<0.05$).

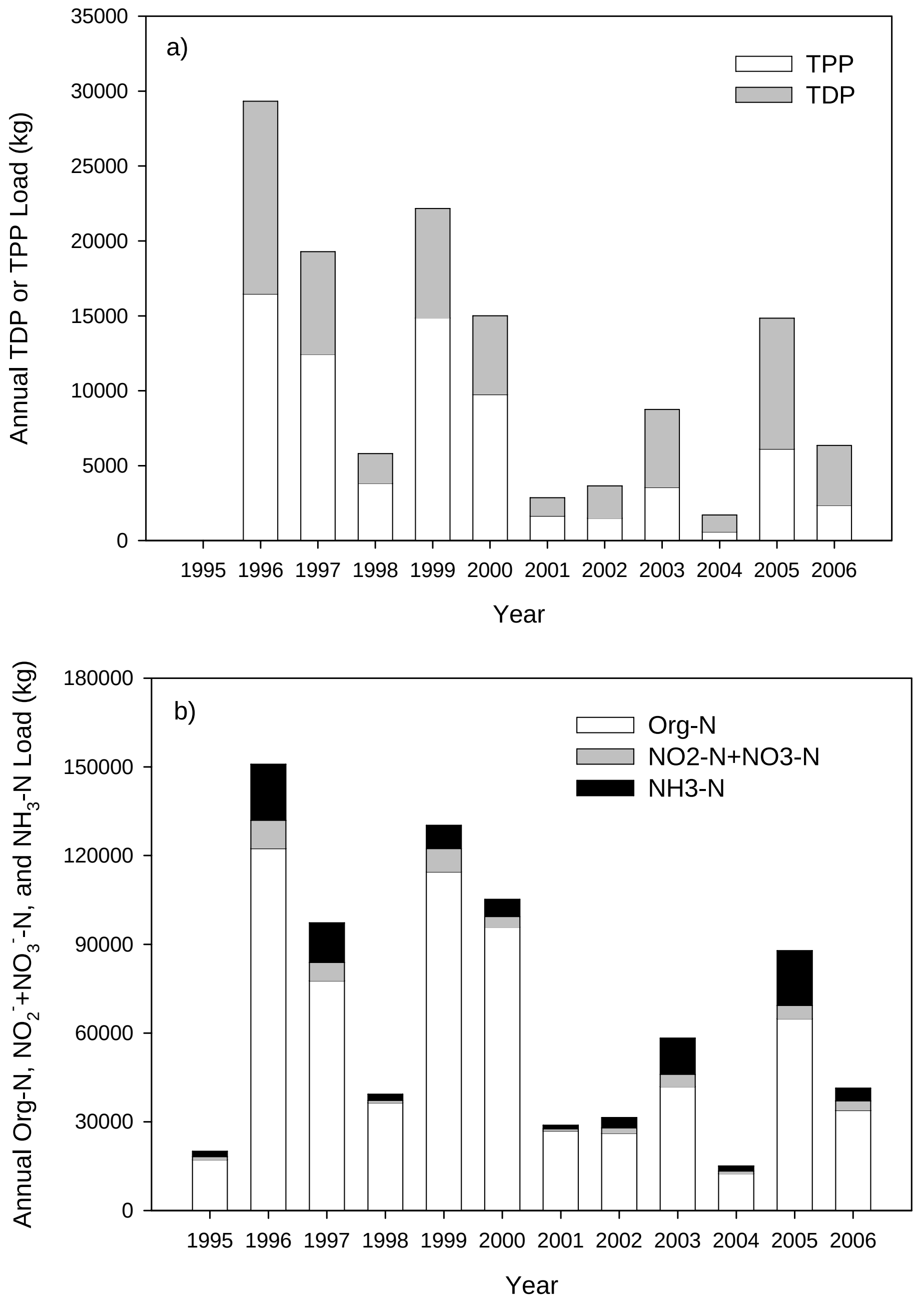
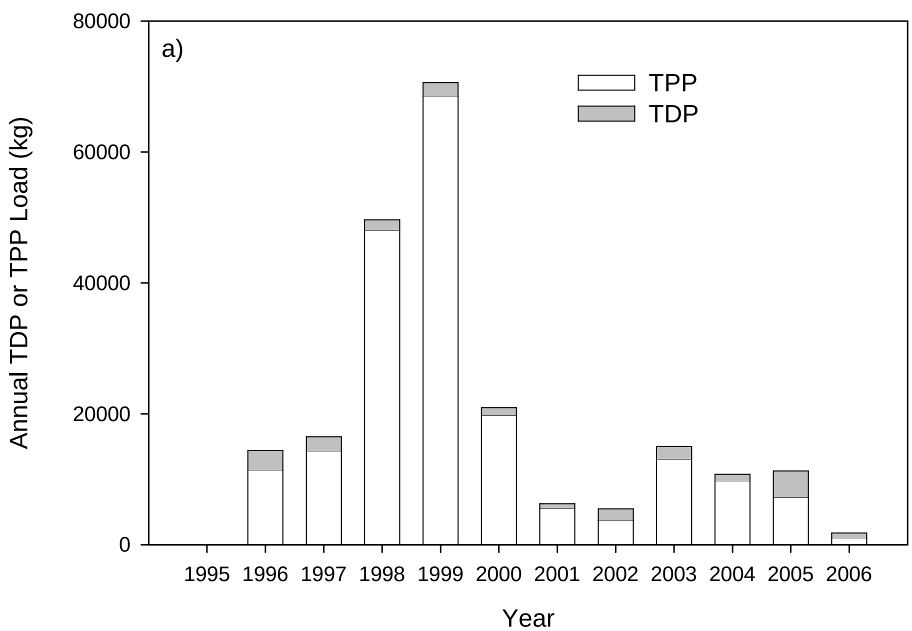


Figure A7.8. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Blindman River. Blindman River is a moderate agricultural intensity watershed that lies in the boreal ecoregion.

Table A7.8. Median, minimum, maximum load nutrient parameters and annual flow volumes in Blindman River, 1995 to 2006.					
	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	49846	15105	2004	150969	
Organic N	39165	12492		122365	
NH_3-N	5073	1282	2001	19015	1996
NO_2+NO_3-N	3376	672		9562	
Total P	7541	1703		29324	
TPP	3810	574	2004	16450	
TDP	5193	1130		12874	
Annual stream volume (hm^3)	26.2	11.6	1995 and 2004	63.1	1996

Rose Creek- Overall there appeared to be no trend in annual TP and TN loading in Rose Creek from 1995 through 2006 (Figure A7.9). Maxima and minima loading of N and P correlated with annual stream flow volumes (Table A7.9). Annual N and P loading appeared to increase from 1996 to 1999 but between 2000 and 2006 remained relatively constant at levels that were similar to loading prior to 1998. Phosphorus fractions were positively correlated with annual stream volume including TP (0.83, $p<0.05$), TDP (0.78, $p<0.05$), and TPP (0.77, $p<0.05$).



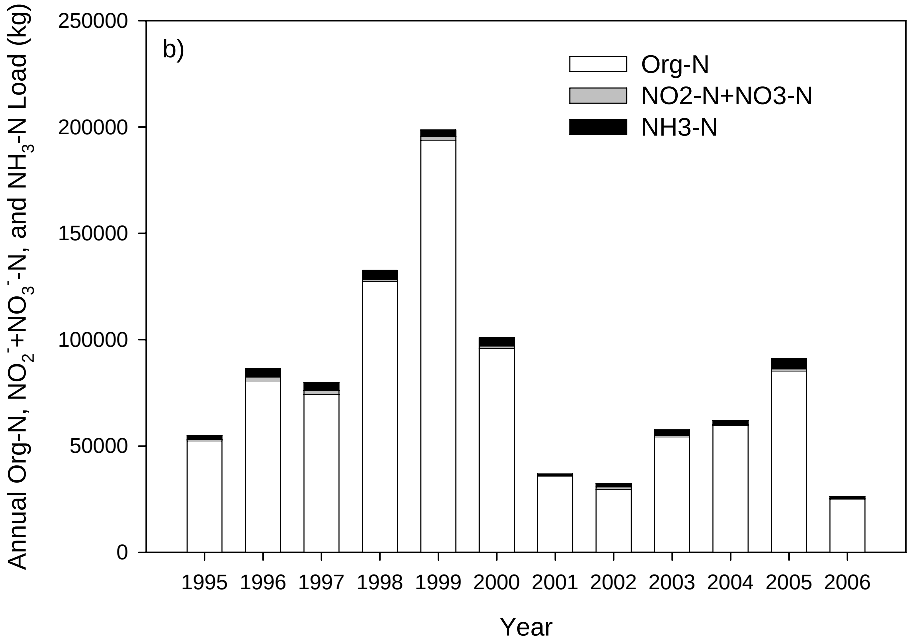


Figure A7.9. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Rose Creek. Rose Creek is a low agricultural intensity watershed that lies in the boreal ecoregion.

Table A7.9. Median, minimum, maximum load nutrient parameters and annual flow volumes in Rose Creek, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	70860	26160	2006	198646	1999
Organic N	66960	25029		193809	
NH_3-N	2973	667		4844	2005
NO_2+NO_3-N	968	298	2001	2142	1996
Total P	12820	1797	2006	70576	1999
TPP	11449	1016		68516	
TDP	1742	660	2001	4035	2005
Annual stream volume (hm^3)	46.4	28.0	2001	85.4	1999

Parkland Ecoregion

Buffalo Creek-Buffalo Creek TP and TN loading had no temporal patterns from 1995 through 2006 (Figure A7.10). There appeared to be an increase in TP and TN loading from 2002 through 2006 which may be attributed to an increase in flows over the same time period (Table A7.23). Maxima TP and TN values occurred in 1997 with the highest annual flow volume, while minima loading occurred in different years, both corresponding to low annual flow volumes (Table A7.10). Total phosphorus loads were positively correlated with TDP (0.95, $p<0.05$) and TPP

(0.79, p<0.05) loads. Further, P loading was positively correlated with annual stream volume including TP (0.95, p<0.05) and TDP (0.95, p<0.05).

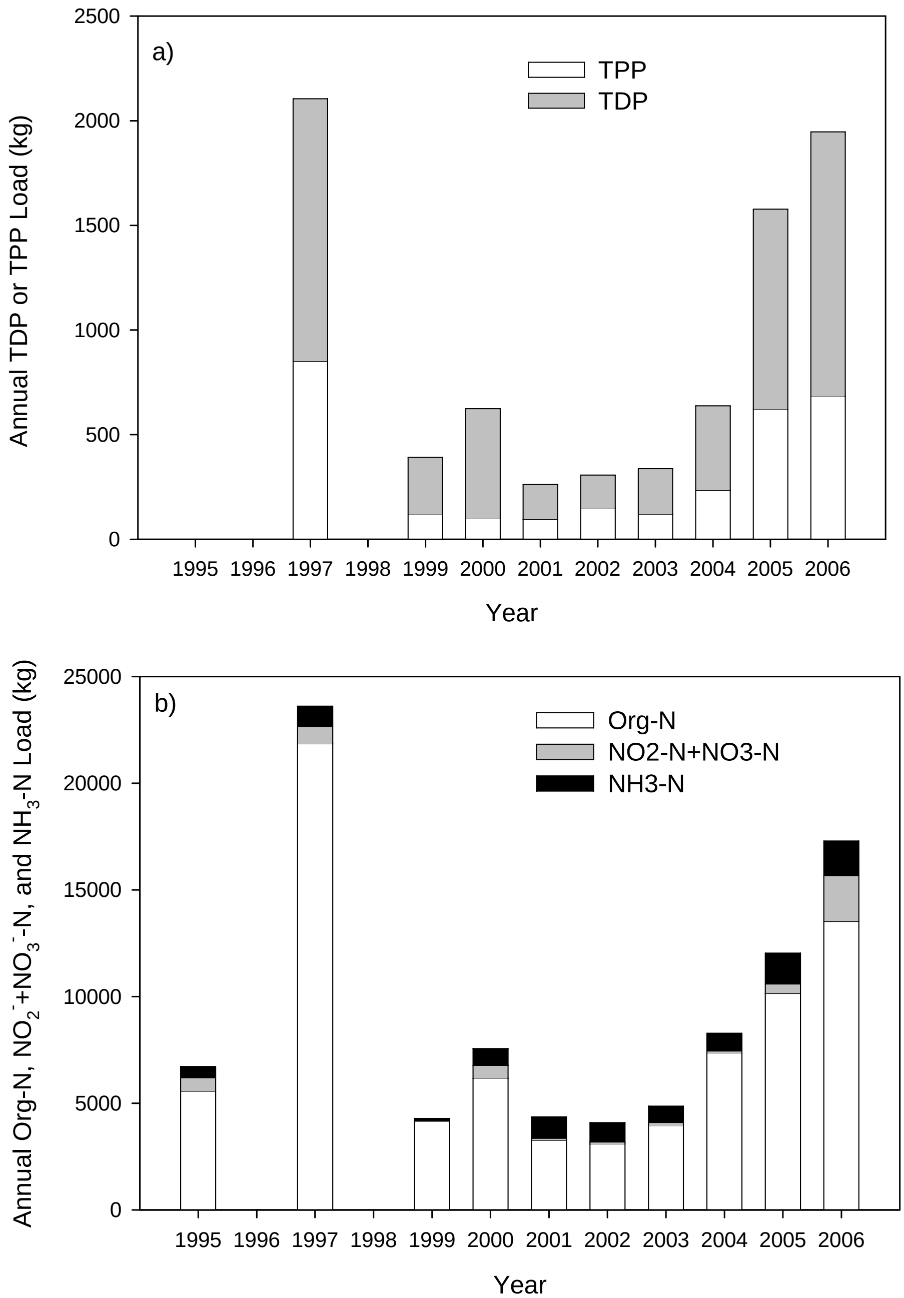
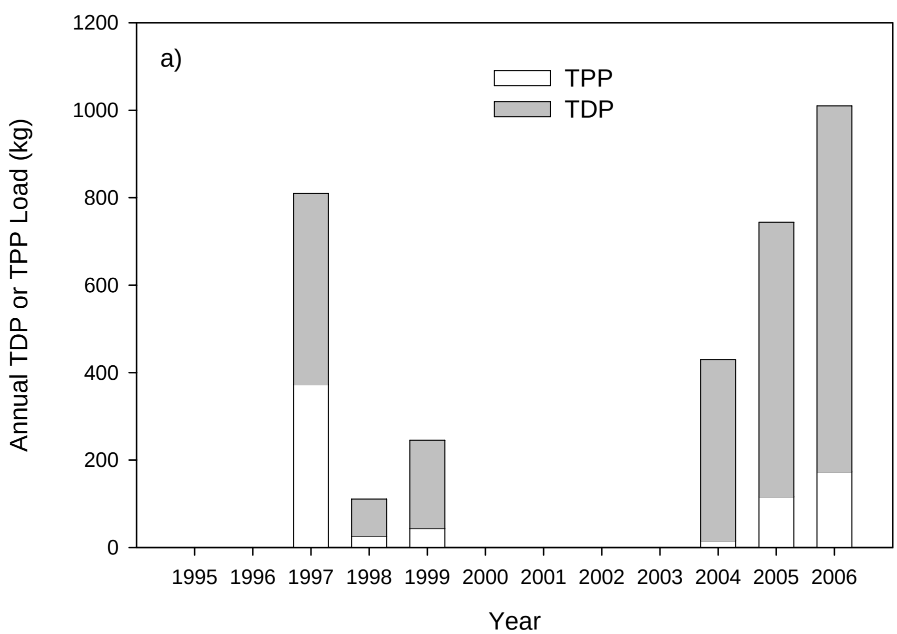


Figure A7.10. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Buffalo Creek. Buffalo Creek is a high agricultural intensity watershed that lies in the parkland ecoregion.

Table A7.10. Median, minimum, maximum load nutrient parameters and annual flow volumes in Buffalo Creek, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	7151	4089	2002	23002	1997
Organic N	5867	3107		22800	
NH_3-N	868	88	1999	1617	2006
NO_2+NO_3-N	289	48		2149	
Total P	630	282	2001	2104	1997
TPP	149	95		851	
TDP	404	158	2002	1263	2006
Annual stream volume (hm^3)	3.6	1.7	2002	11.0	1997

Stretton Creek- Stretton Creek was monitored from 1995 through 2006; however, there were no flows from 2000 through 2003, making it difficult to observe any loading patterns over the monitoring period (Figure A7.11). Few years had similar maxima in P and N parameters possibly due multiple years which had low flows in Stretton Creek (Table A7.11; Table A7.23). Spearman rank correlations showed a strong, positive correlation between TP and TDP (1.00, p=0.05). Annual stream volume was also positively correlated with TP (1.00, p=0.05) and TDP (1.00, p=0.05).



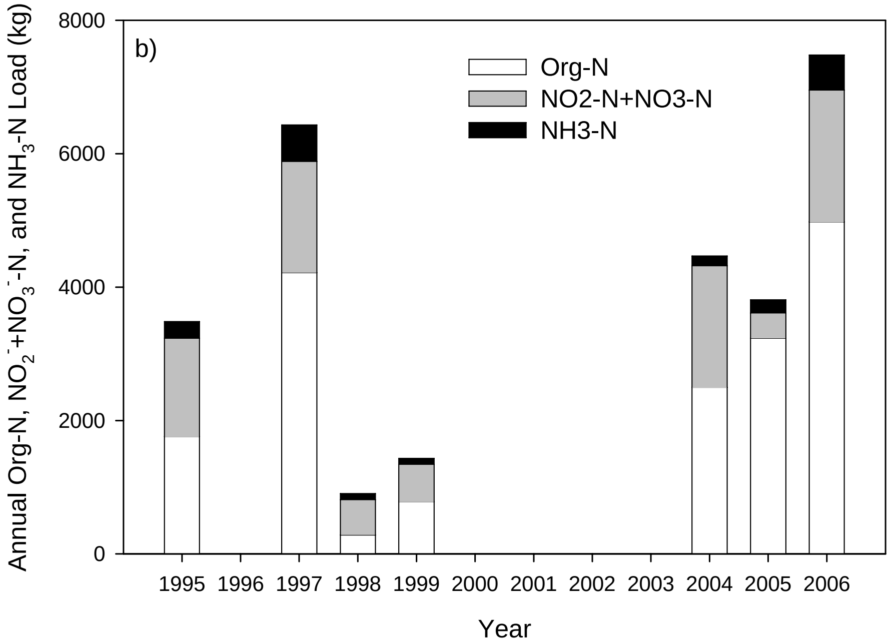


Figure A7.11. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Stretton Creek. Stretton Creek is a high agricultural intensity watershed that lies in the parkland ecoregion.

Table A7.11. Median, minimum, maximum load nutrient parameters and annual flow volumes in Stretton Creek, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	3810	909	1998	7494	2006
Organic N	2496	283		4977	
NH_3-N	196	88	1999	545	1997
NO_2+NO_3-N	1472	381	2005	1978	2006
Total P	501	110	1998	1010	
TPP	80	15	2004	373	1997
TDP	426	85	1998	837	2006
Annual stream volume (hm^3)	1.2	0.2	1998	2.3	2006

Haynes Creek M6- There were no patterns in TP and TN loading in Haynes Creek M6 from 1995 to 2006 (Figure A7.12). The load data, however, had a discrepancy in number of samples taken annually. For instance, 1995, 1998, 2001, and 2005 all had low annual flow volumes (0.3, 0.1, 0.1, and 0 hm^3, respectively), but the number of samples taken changed from 35, 23, 9 and 0, respectively. This may have influenced the loading data reported. In 2004 there was no flow, hence missing data in Figure A7.12. Even with the variable sampling, annual nutrient loading appeared to be influenced by annual stream volume where lowest loads occurred in years with lower flow, and highest loads occurred in years with higher flow (Table A7.23). This was reflected in the Spearman's rank correlations between annual stream volume and TP, TDP, and

TPP loads which were all strongly, positively correlated at 0.96 ($p<0.05$). TP loads were strongly correlated with both TDP and TPP, both at 1.00 ($p<0.05$).

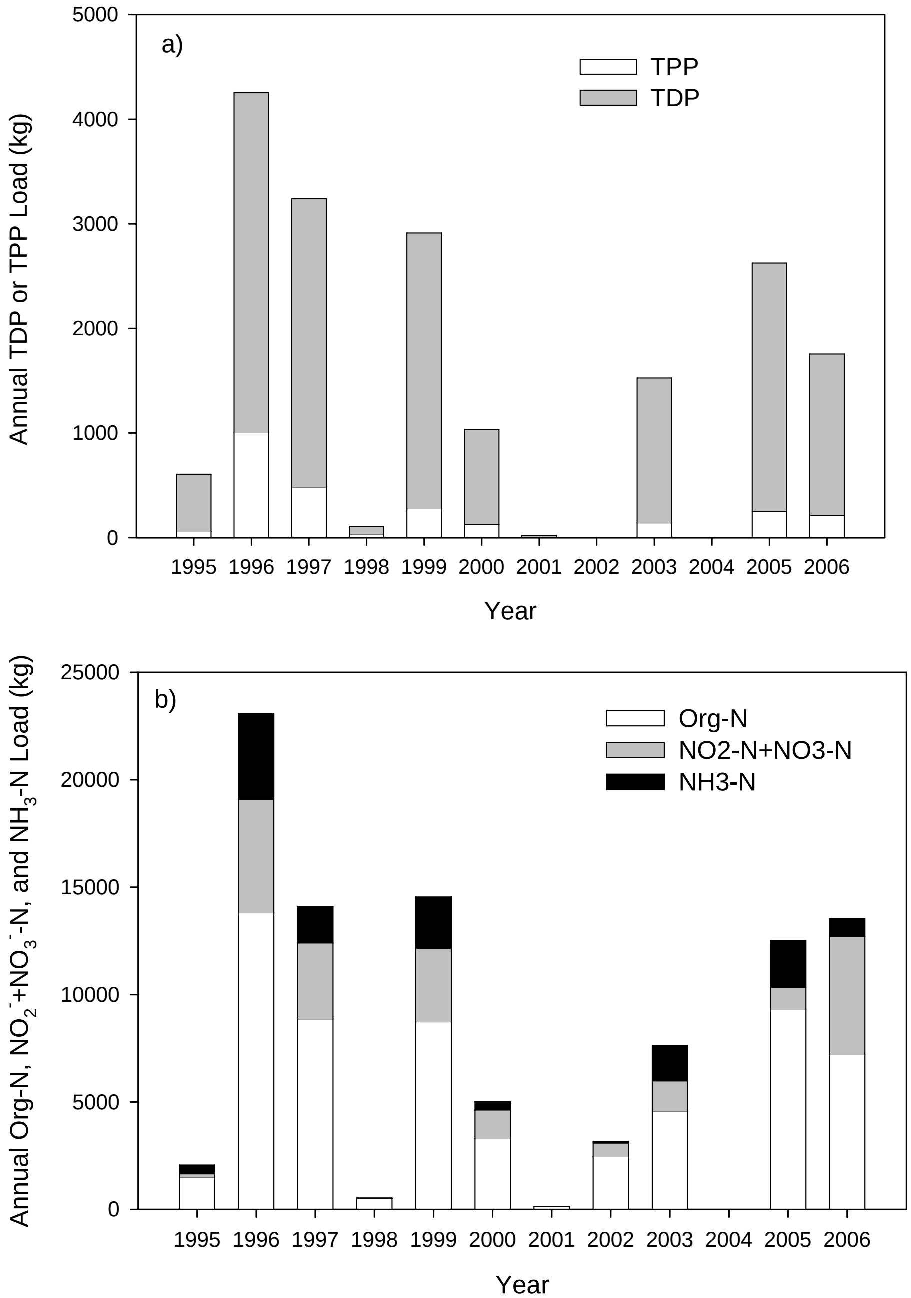
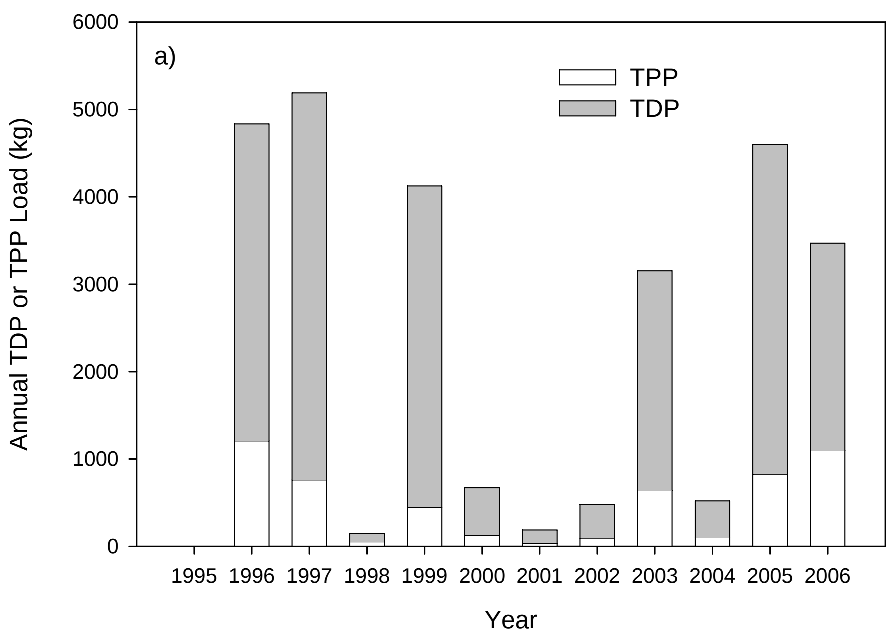


Figure A7.12. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Haynes Creek M6. Haynes Creek M6 is a high agricultural intensity watershed that lies in the parkland ecoregion.

Table A7.12. Median, minimum, maximum load nutrient parameters and annual flow volumes in Haynes Creek M6, 1995 to 2006.					
	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	7648	136		23093	1996
Organic N	4583	128		13812	
NH_3-N	819	7	2001	3985	
NO_2+NO_3-N	1324	1		5489	2006
Total P	1527	20		4252	
TPP	177	5		1006	1996
TDP	1464	15		3246	
Annual stream volume (hm^3)	1.2	0.1	1998 and 2001	5.6	1996

Threehills Creek- Threehills Creek had no observable pattern in TP and TN loading from 1995 through 2006 (Figure A7.13). Maxima and minima nutrient loads were related to annual flow volumes with high loads occurring with high annual flow volumes, and vice versa (Table A7.13; Table A7.23). Spearman rank correlations showed that annual stream volume was highly correlated with TP (0.82, $p<0.05$), TDP (0.85, $p<0.05$) and TPP (0.85, $p<0.05$). Total P loads were strongly, positively correlated with both TDP (0.98, $p<0.05$) and TPP (0.90, $p<0.05$) loads.



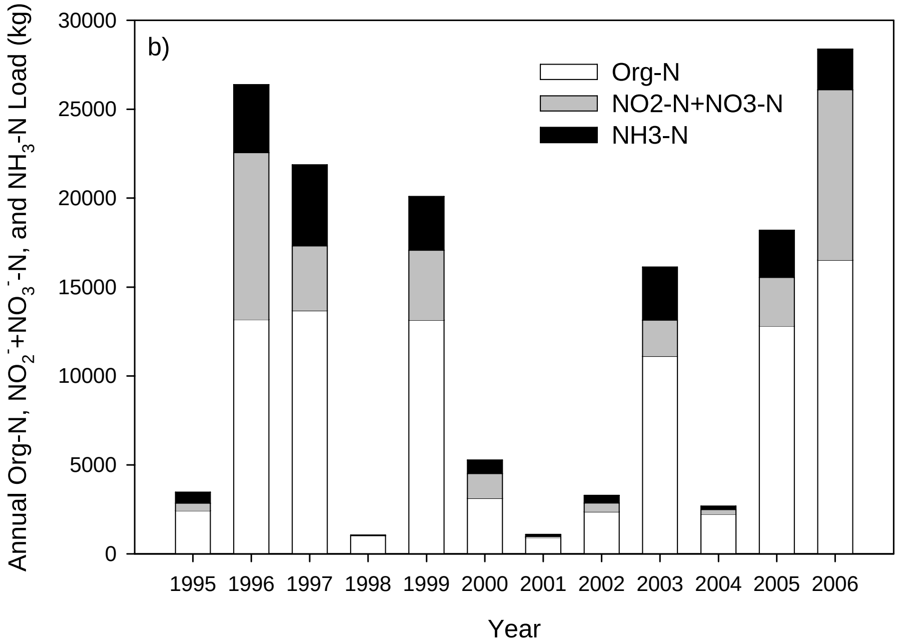


Figure A7.13. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Threehills Creek. Threehills Creek is a high agricultural intensity watershed that lies in the parkland ecoregion.

Table A7.13. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Threehills Creek, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	10719	1058	1998	28427	2006
Organic N	7120	917	2001	16516	
NH_3-N	1522	31		4558	1997
NO_2+NO_3-N	1707	3	1998	9583	2006
Total P	1968	149		5188	1997
TPP	449	34	2001	1207	1995
TDP	2374	97	1998	4428	1997
Annual stream volume (hm^3)	3.0	0.4	2001	8.3	1997

Ray Creek- Ray Creek did not have any observable patterns in TP and TN loading from 1995 to 2006 (Figure A7.14). Ranges in TP and TN loading were related to flow where years with high annual flow volumes were also years of high loading (Table A7.14; Table A7.23). Total P loads were highly correlated with TDP (1.00, p<0.05) and TPP (0.98, p<0.05) loads. This was also reflected in the Spearman rank correlations where annual stream volume was strongly, positively correlated with TP (1.00, p<0.05), TDP (1.00, p<0.05), and TPP (0.98, p<0.05).

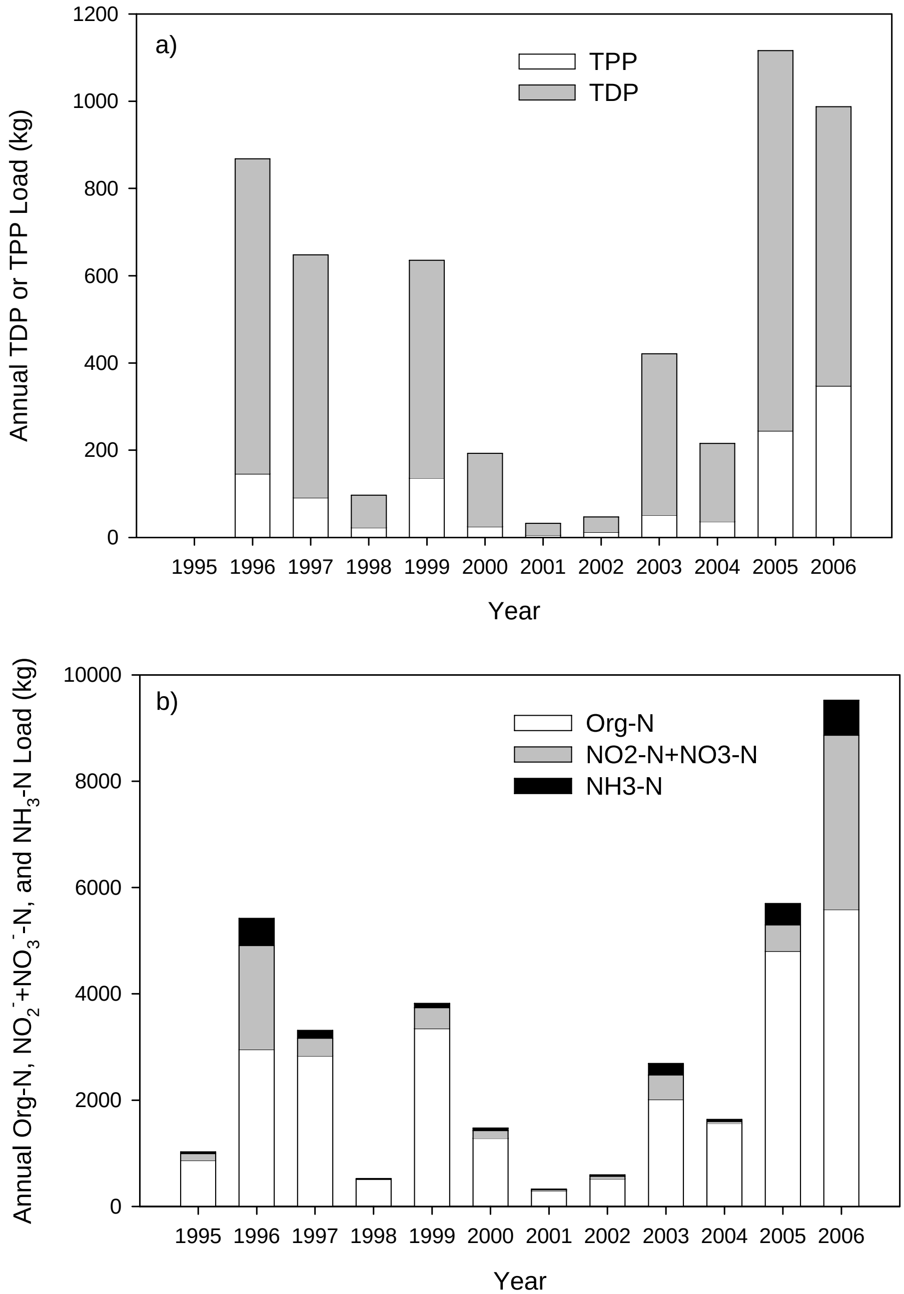
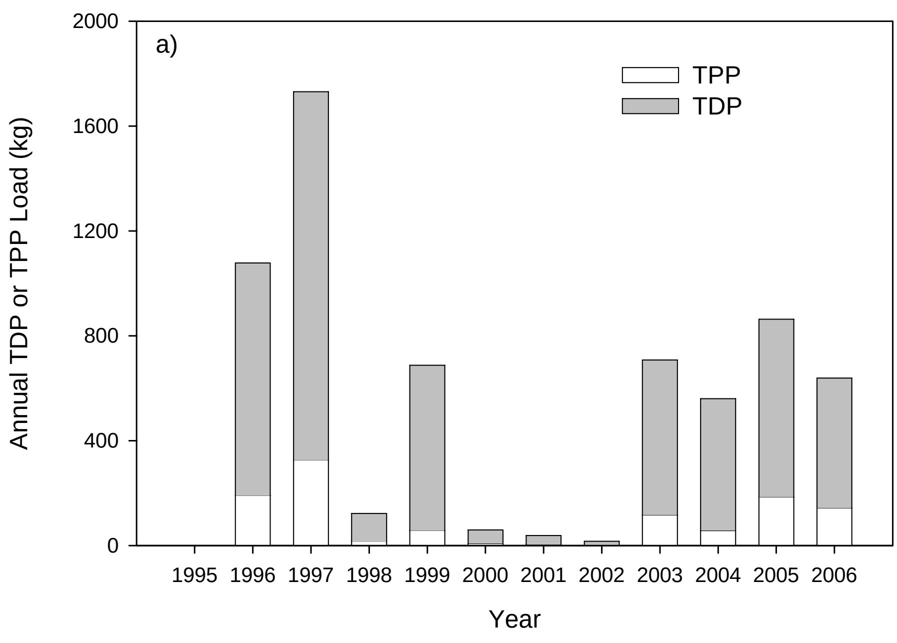


Figure A7.14. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Ray Creek. Ray Creek is a high agricultural intensity watershed that lies in the parkland ecoregion.

Table A7.14. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Ray Creek, 1995 to 2006.					
	Median ($kg\ yr^{-1}$)	Minimum ($kg\ yr^{-1}$)	Year	Maximum ($kg\ yr^{-1}$)	Year
Total N	2165	328	2001	9543	
Organic N	1788	294		5585	2006
NH_3-N	64	10	1998	656	
NO_2+NO_3-N	240	2		3280	
Total P	318	32	2001	1116	2005
TPP	51	5		347	2006
TDP	370	27		872	2005
Annual stream volume (hm^3)	1.4	0.2	2001 and 2002	2.4	2005

Renwick Creek- There was no temporal pattern in TP or TN loading observed in Renwick Creek from 1995 to 2006 (Figure A7.15). The loads observed were reflective of the annual flow volumes measured (Table A7.15). This is also supported by the strong, positive correlation between annual flow volume and TP (0.98, $p<0.05$), TDP (0.9, $p<0.053$), and TPP (0.92, $p<0.05$) loads.



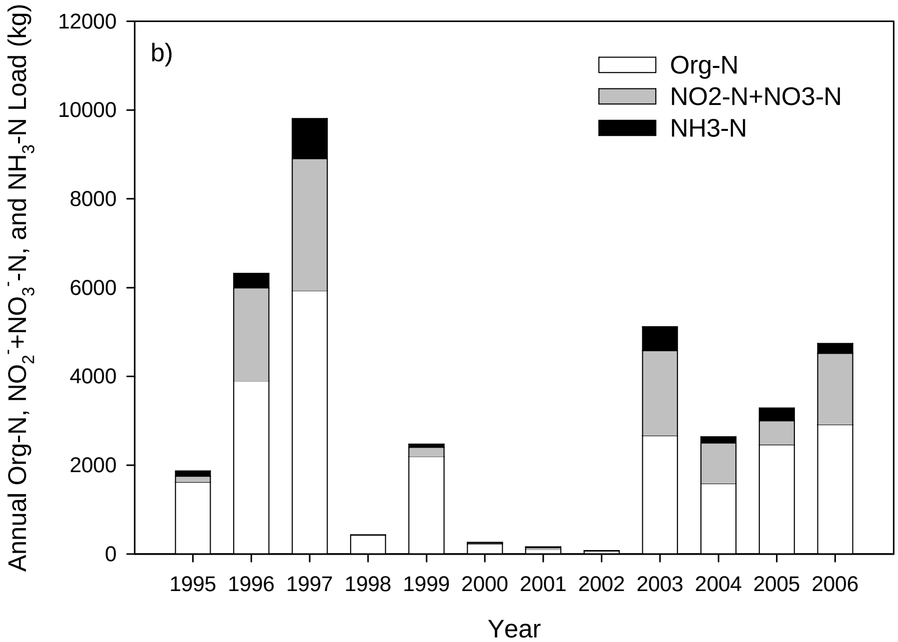


Figure A7.15. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Renwick Creek. Renwick Creek is a high agricultural intensity watershed that lies in the parkland ecoregion.

Table A7.15. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Renwick Creek, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	2560	78	2002	9824	1997
Organic N	1909	66	2002	5933	1997
NH_3-N	129	4	2002	900	1997
NO_2+NO_3-N	374	2	1998	2974	1997
Total P	600	16	2002	1731	1997
TPP	58	1	2002	327	1997
TDP	503	15	2002	1404	1997
Annual stream volume (hm^3)	0.7	0.0	2001 and 2002	3.6	1997

Grassland Ecoregion

Trout Creek- Between 1995 and 2006 there was no observable pattern in TP and TN loading for Trout Creek (Figure A7.16). The majority of loads had maxima in 1997 and minima in 2000 (Table A7.16). Comparing the loading data to annual flow volumes it is apparent that although the loading was highest in 1997, the annual flow volume in 1995 was 2 times as high, and flow volume in 2005 was 3 times as high (Table A7.23). Looking at the sampling regime, it was found that 1995 and 1996 were under-sampled, missing the peak flow time in both years. This could create the appearance of high loading in 1997. Even with the under-sampling, the general trend remains with higher load correlating to higher annual stream flow volumes. Spearman's Rank correlations between annual stream volume and TP, TDP, and TPP loads were all strongly correlated at 0.98 ($p<0.05$), supporting higher loading during years with more stream flow.

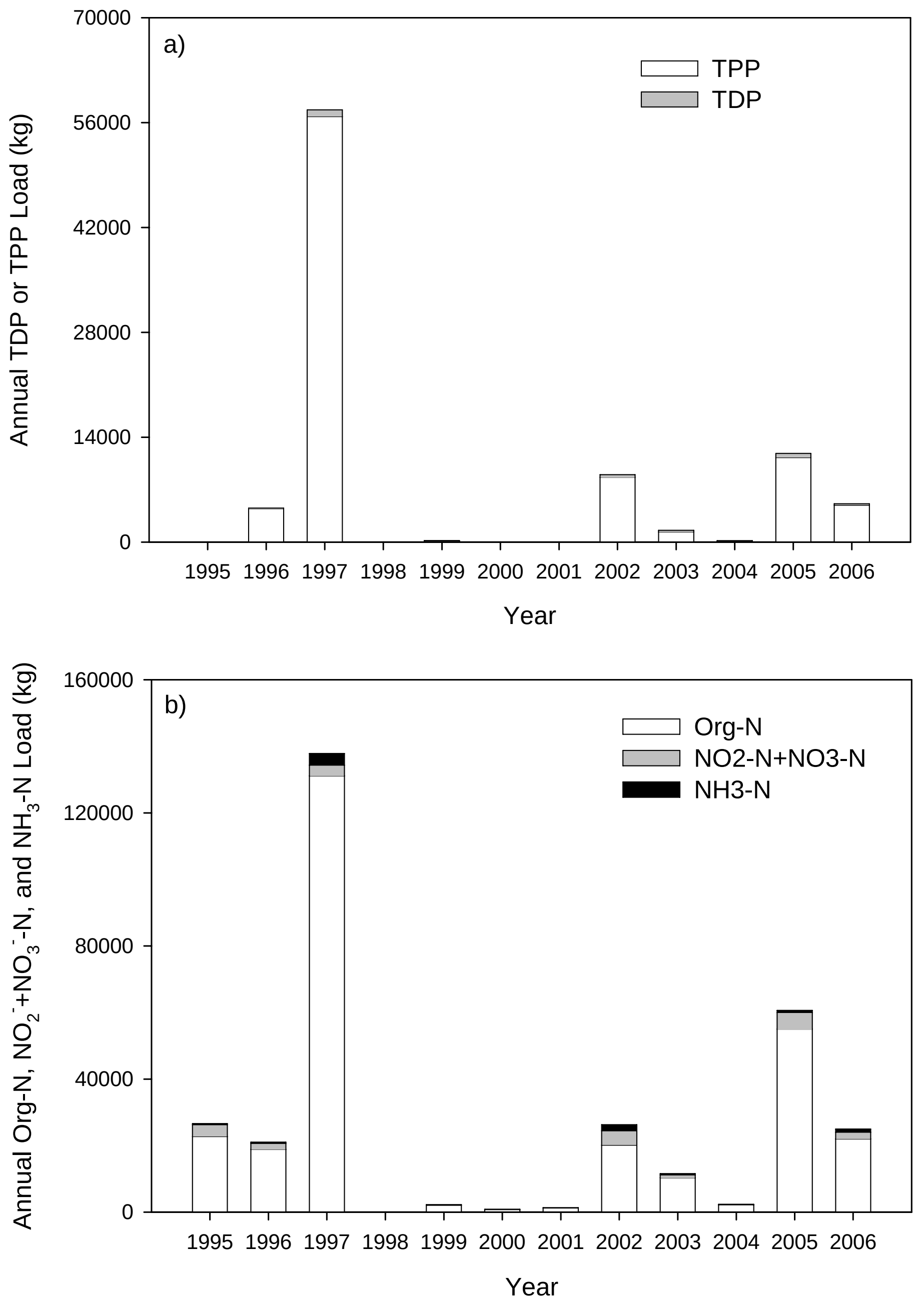
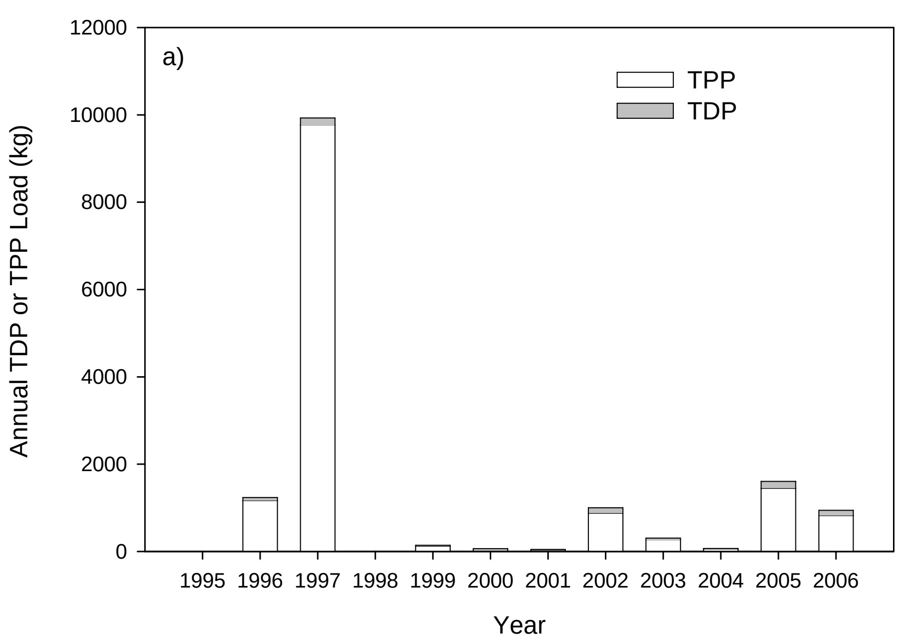


Figure A7.16. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Trout Creek. Trout Creek is a moderate agricultural intensity watershed that lies in the grassland ecoregion.

Table A7.16. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Trout Creek, 1995 to 2006.

	Median ($kg\ yr^{-1}$)	Minimum ($kg\ yr^{-1}$)	Year	Maximum ($kg\ yr^{-1}$)	Year
Total N	21055	826	2000	137917	1997
Organic N	18879	732	2000	131075	1997
NH_3-N	398	23	2000	3463	1997
NO_2+NO_3-N	1749	48	2001	4979	2005
Total P	3838	36	2000	57705	1997
TPP	2901	20	2000	56800	1997
TDP	151	16	2000	904	1997
Annual stream volume (hm^3)	22.1	1.8	2000	74.0	2005

Meadow Creek- When initially assessing TP and TN loading in Meadow Creek from 1995 through 2006 there appeared to be a slight declining trend (Figure A7.17), this declining trend however was created by the peak in N and P loading in 1997. Comparing the loading data to flow volumes it became apparent that the flow volumes in 1997 were lower than those of 1995 and 1996. Similar to Meadow Creek, it appeared that although flow volumes were higher in 1995 and 1996, there were fewer samples taken during these years and the loading data were under-sampled. Any temporal trend was thus difficult to assess. Flow volumes were highest in 2005 at 11 hm^3 and lowest in 2000 and 2001 at 0.4 hm^3 (Table A7.23). The low annual flow volumes corresponded to low annual loading; however, although there was a peak in annual flow volumes in 2005, nitrogen loads were not high (Table A7.17). Besides in 1997, loading also was high in 1996, 2002, 2005, and 2006. The apparent relationship between high loading and high annual flow volume was supported by Spearman's rank correlations where annual stream volume was positively correlated with TP, TDP, and TPP at 0.98, 0.76, and 1.00, respectively (all $p<0.05$).



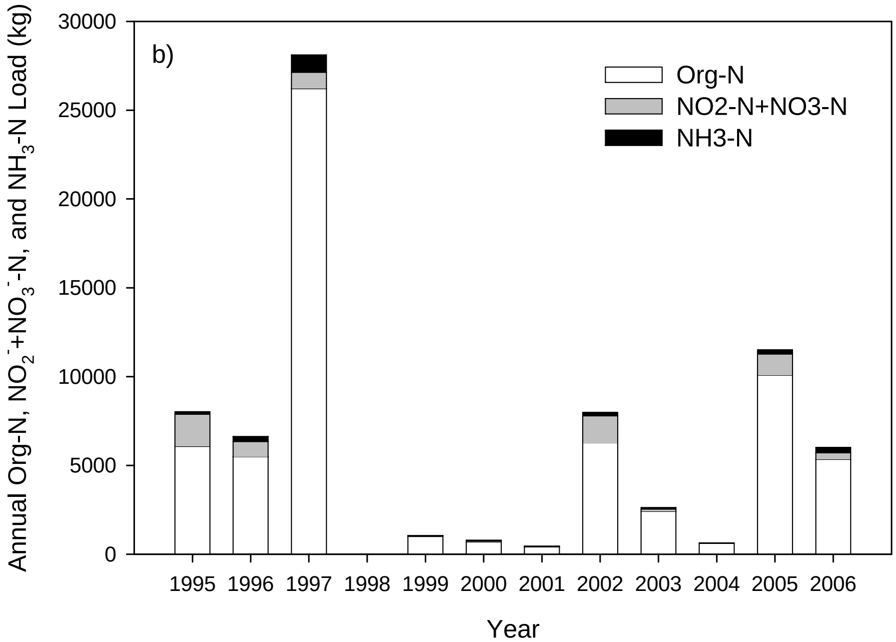
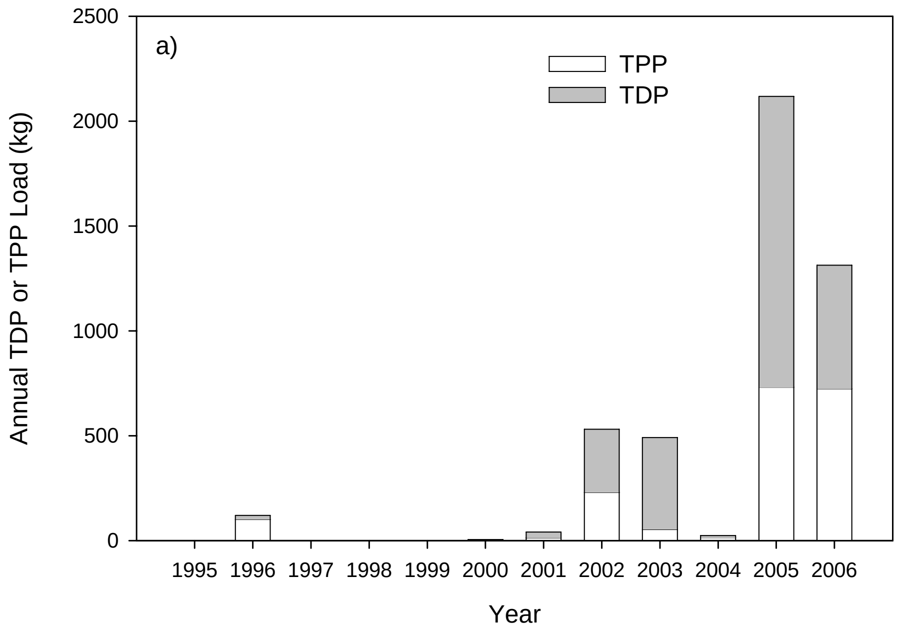


Figure A7.17. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Meadow Creek. Meadow Creek is a moderate agricultural intensity watershed that lies in the grassland ecoregion.

Table A7.17. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Meadow Creek, 1995 to 2006.

<table>
<tr><th></th><th>Median (kg yr^{-1})</th><th>Minimum (kg yr^{-1})</th><th>Year</th><th>Maximum (kg yr^{-1})</th><th>Year</th></tr>
<tr><td>Total N</td><td>6020</td><td>458</td><td rowspan="2">2001</td><td>28125</td><td rowspan="6">1997</td></tr>
<tr><td>Organic N</td><td>5340</td><td>418</td><td>26212</td></tr>
<tr><td>NH_3-N</td><td>145</td><td>19</td><td></td><td>976</td></tr>
<tr><td>NO_2+NO_3-N</td><td>369</td><td>10</td><td>2004</td><td>1812</td></tr>
<tr><td>Total P</td><td>943</td><td>46</td><td>2001</td><td>9929</td></tr>
<tr><td>TPP</td><td>544</td><td>16</td><td>2000</td><td>9767</td></tr>
<tr><td>TDP</td><td>59</td><td>8</td><td>2001</td><td>161</td><td></td></tr>
<tr><td>Annual stream volume (hm^3)</td><td>4.0</td><td>0.4</td><td>2000 and 2001</td><td>10.8</td><td>2005</td></tr>
</table>

Prairie Blood Coulee- Overall P loading in Prairie Blood Coulee appeared high in 2002, 2003, 2005, and 2006, creating an observable increasing trend in TP and TN loading from 1995 to 2006 (Figure A7.18). However, stream flow was also highest in 2002, 2003, 2005, and 2006 compared to previous years (Table A7.23). Flows in June 2005 were especially high in Prairie Blood Coulee at Lethbridge, which was said to have had only a 1 % chance of occurring in any given year, based on historical data (Alberta Environment 2005). Higher P loading occurred in years when annual flow volume and annual stream volume was strongly and positively correlated with TP (0.93, $p<0.05$), TDP (0.91, $p<0.05$), and TPP (0.98, $p<0.05$) loading. In contrast, annual flow volumes were measured as low as 0.1 hm^3 which occurred in 1999. These ranges in annual flow volumes were reflected in the ranges of the annual N loading (Table A7.18). The TP loads measured in Prairie Blood Coulee were equally positively correlated with TDP and TPP loads (0.98, $p<0.05$). Despite the discussion above, there are uncertainties surrounding Prairie Blood Coulee temporal trends. Firstly, it is ambiguous whether the higher N and P loading in the latter years of sampling were solely the result of increased stream flow or was also influenced by an increase in agricultural intensity in the watershed. The agricultural intensity classification data from the 2001 and 2006 Census of Agriculture shows an increase in agricultural intensity; however, the polygon used in 2006 is much larger than the polygon used in 2001. Secondly, the polygons used in 2001 and 2006 were of a larger size than the actual Prairie Blood Coulee watershed boundary used in the AESA project. Thirdly, when reviewing aerial photos of the Prairie Blood Coulee watershed, there appeared to be pivot circles, indicating irrigation activity.



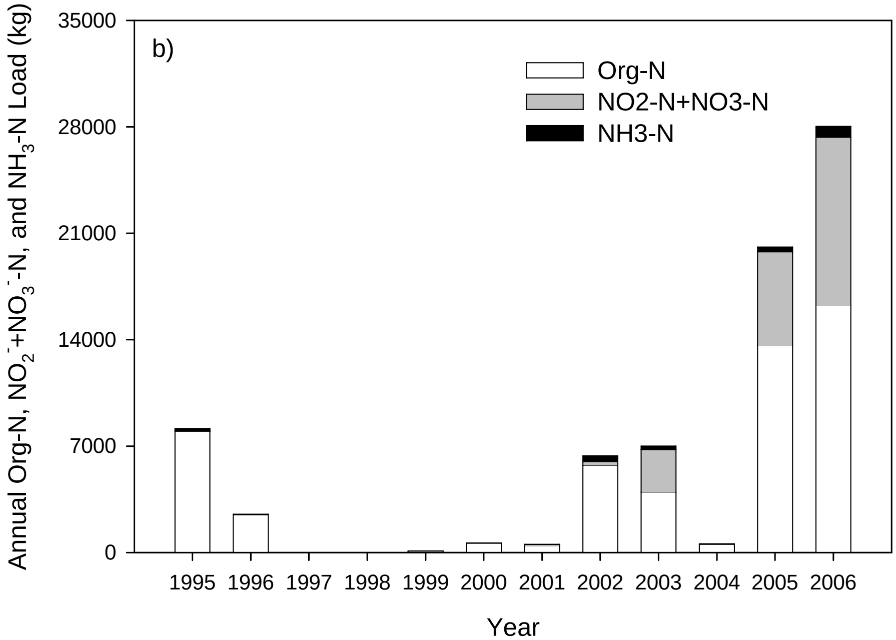


Figure A7.18. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Prairie Blood Coulee. Prairie Blood Coulee is a low agricultural intensity watershed that lies in the grassland ecoregion.

Table A7.18. Median, minimum, maximum load nutrient parameters and annual flow volumes in Prairie Blood Coulee, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	4443	101	1999	28034	2006
Organic N	3236	97		16256	
NH_3-N	90	3		703	
NO_2+NO_3-N	1065	461	2001	10269	2004
Total P	280	2	1999	2118	
TPP	53	1	1999 and 2000	732	2005
TDP	29	1	1999	1386	
Annual stream volume (hm^3)	2.8	0.1	1999	10.1	2005

Continental Divide Ecoregion

Willow Creek- From 1999 through 2006 there appeared to be an increasing trend in TP and TN loads in Willow Creek (Figure A7.19). This increase was more apparent when loads from 2005 were removed. Willow Creek had an extremely high flow volume in June 2005 due to a large amount of rain intensifying snowmelt in the headwaters, the same event shown in Prairie Blood Coulee in 2005. Alberta Environment (2005) estimated flooding levels during June 2005 in

Willow Creek at Claresholm, Alberta to only have a 1 % chance of occurring in any given year. Other than 2005, loading in Willow Creek appeared to be correlated to stream flow (Table A7.19). This was found to be true for P where annual stream volume was strongly, positively correlated with TP (0.83, $p<0.05$), TDP (0.91, $p<0.05$), and TPP (0.83, $p<0.05$) loads. TP loads were also highly correlated with TDP (0.98, $p<0.05$) and TPP (1.00, $p<0.05$) loads.

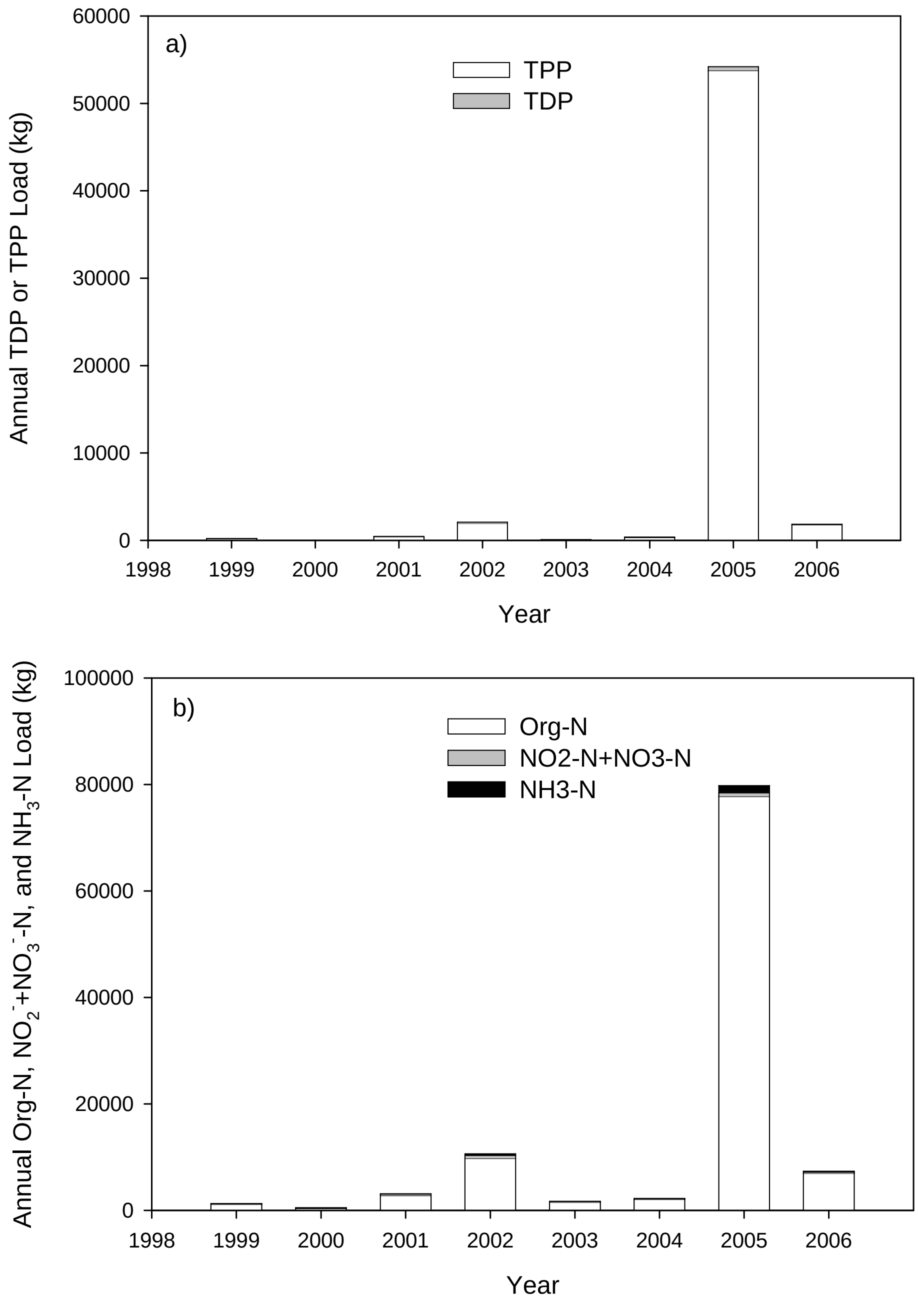
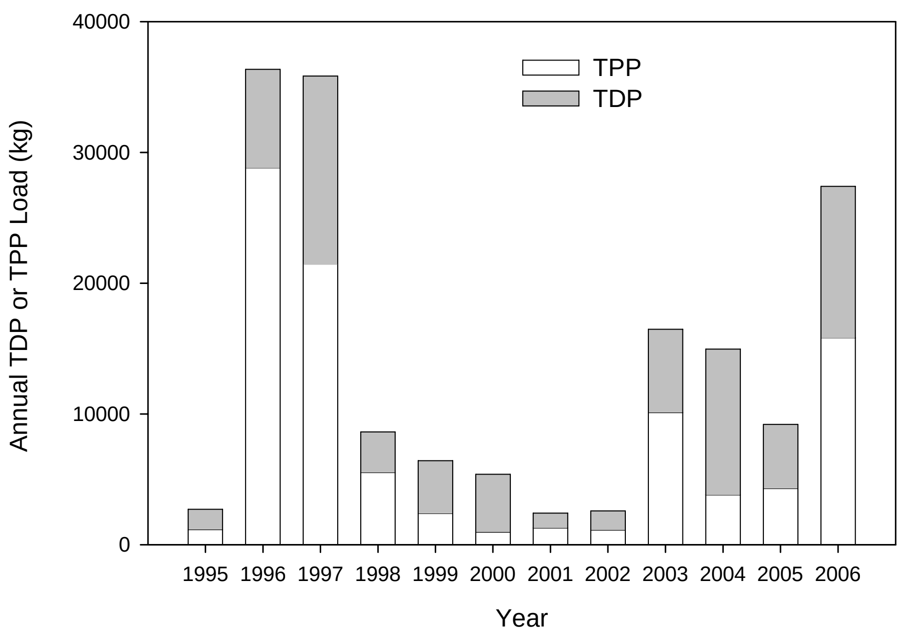


Figure A7.19. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Willow Creek. Willow Creek is a low agricultural intensity watershed and is the only AESA watershed that lies in the continental divide ecoregion.

Table A7.19. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Willow Creek, 1999 to 2006.					
	Median ($kg\ yr^{-1}$)	Minimum ($kg\ yr^{-1}$)	Year	Maximum ($kg\ yr^{-1}$)	Year
Total N	2686	498	2000	79955	
Organic N	2433	358		77737	
NH_3-N	178	12	2001	11098	2005
NO_2+NO_3-N	64	95		676	
Total P	415	43	2000	54191	
TPP	375	31		53743	
TDP	40	11		448	
Annual stream volume (hm^3)	9.9	4.1	2000	41.4	2005

Irrigated Grassland Ecoregion

Crowfoot Creek- The TP and TN loading in Crowfoot Creek from 1995 through 2006 were too variable for a temporal pattern to be observed (Figure A7.20). Generally, peak loads occurred with the highest annual flow volumes and low loads occurred with the lowest annual flow volumes (Table A7.20 and A7.23). This was supported by Spearman's rank correlations which showed a positive correlation between annual stream volume and TP (0.92, $p<0.05$), TDP (0.90, $p<0.05$), and TPP (0.75, $p<0.05$).



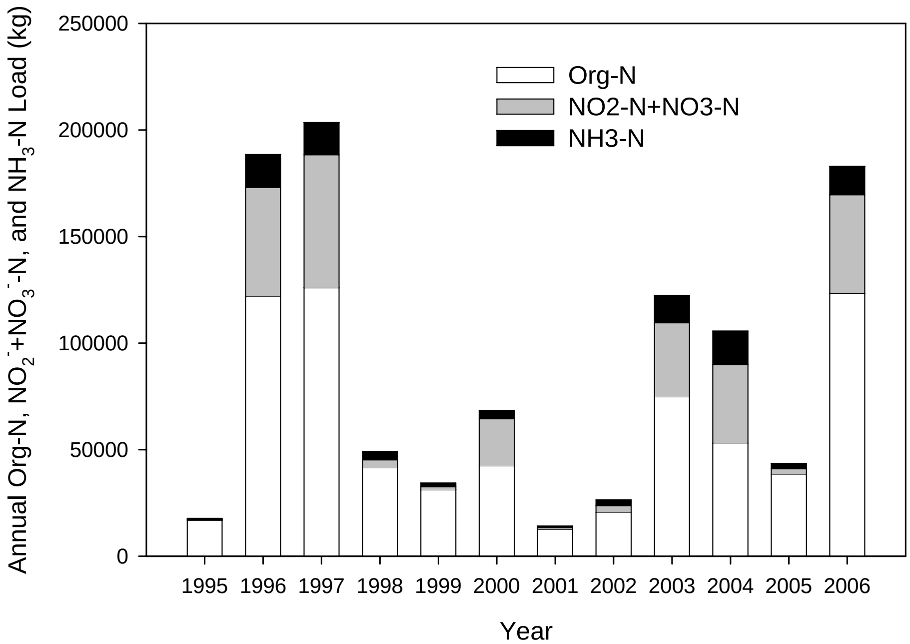
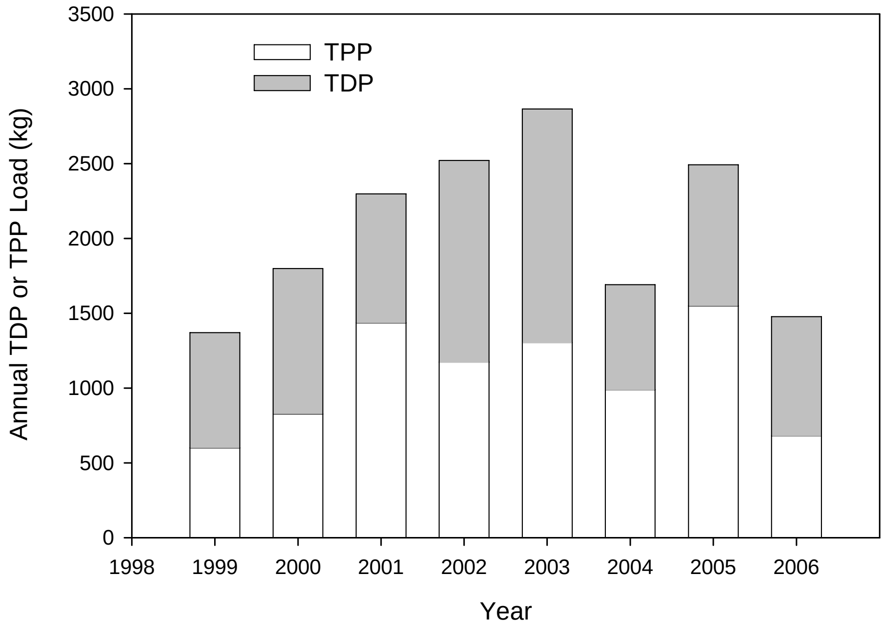


Figure A7.20. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Crowfoot Creek. Crowfoot Creek is a high but irrigated agricultural intensity watershed that lies in the grassland ecoregion.

Table A7.20. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Crowfoot Creek, 1995 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	58877	14210	2001	208679	1997
Organic N	42067	12596		125958	
NH_3-N	3980	560	1995	15900	2004
NO_2+NO_3-N	12779	383		62352	1997
Total P	3909	2417	2001	36356	1996
TPP	4050	958	2000	28830	
TDP	4665	1142	2001	14388	1997
Annual stream volume (hm^3)	33.6	19.1	2001	70.9	1997

New West Coulee- New West Coulee was monitored from 1999 through 2006 and there was no observable trend in TP and TN loading over this time (Figure A7.21). Annual stream flow volumes were steady from 1999 to 2006 with one peak in 2000 (Table A7.21 and A7.23). Generally, nutrient loading peaked in 2003 when flow volumes were high (Table A7.23). With the variability in loading and annual flow volumes, Spearman’s rank correlations did not show any correlation between P loading and annual stream volume.



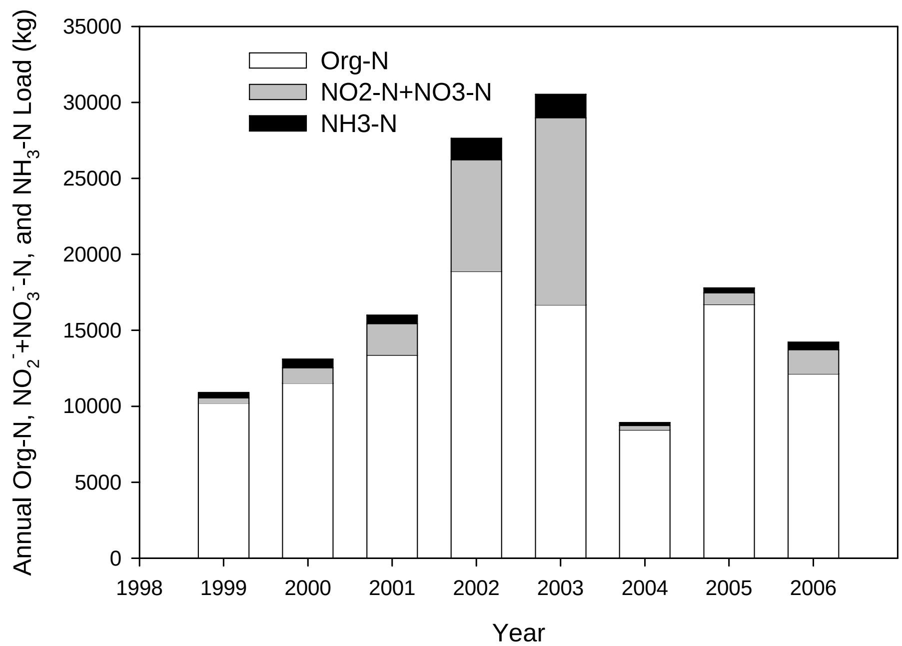
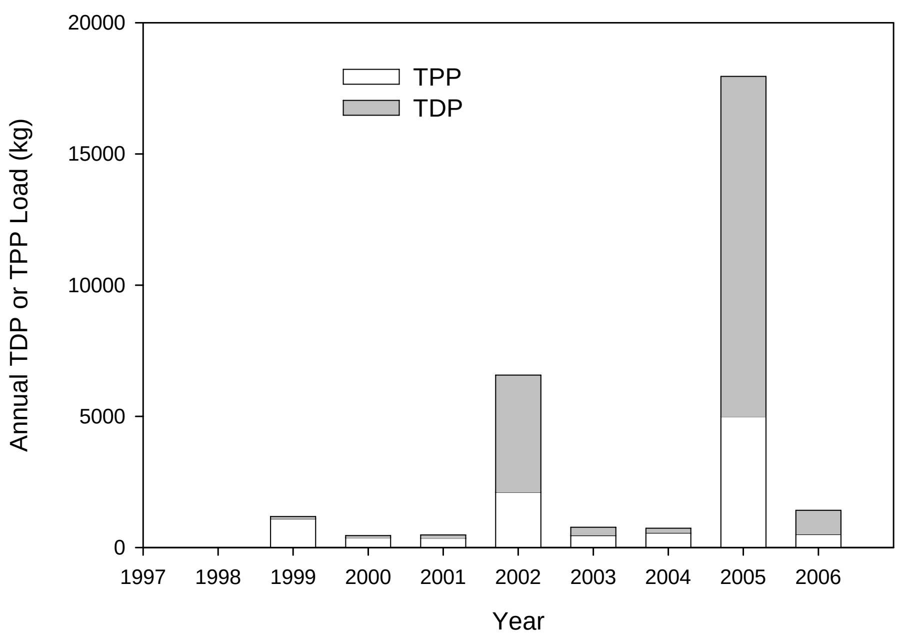


Figure A7.21. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in New West Coulee. New West Coulee is a high but irrigated agricultural intensity watershed that lies in the grassland ecoregion.

Table A7.21. Median, minimum, maximum load for nutrient parameters and annual flow volumes in New West Coulee, 1999 to 2006.

	Median ($kg\ yr^{-1}$)	Minimum ($kg\ yr^{-1}$)	Year	Maximum ($kg\ yr^{-1}$)	Year
Total N	15136	8929		30557	2003
Organic N	12738	8434	2004	18879	2002
NH_3-N	537	201		1542	
NO_2+NO_3-N	1297	289		12316	2003
Total P	2048	1370	1999	2864	
TPP	1081	682	2006	1549	2005
TDP	903	701	2004	1561	2003
Annual stream volume (hm^3)	22.0	15.9	2006	29.6	2000

Battersea Drain- The Battersea Drain, from 1998 through 2006, had an increasing trend in TP and TN loading (Figure A7.22). High loading years include 2002 and 2005, however even with these two years removed the TP and TN loading data appear to increase. Although not reflected in the annual flow volumes, flow patterns in 2002 and 2005 indicate prominent peaks despite being regulated by the St. Mary's Irrigation District. These two peaks were related to higher than average amounts of precipitation (Appendix 2). Generally, annual loads were lowest in 1998 and highest in 2005 (Table A7.22). The increasing trend could be influenced by an alteration of management practices within the watershed as the Census of Agricultural data indicate the agricultural intensity rating of the Battersea Drain remained high from 1996 through 2006; however, land use data are not available to confirm this observation.



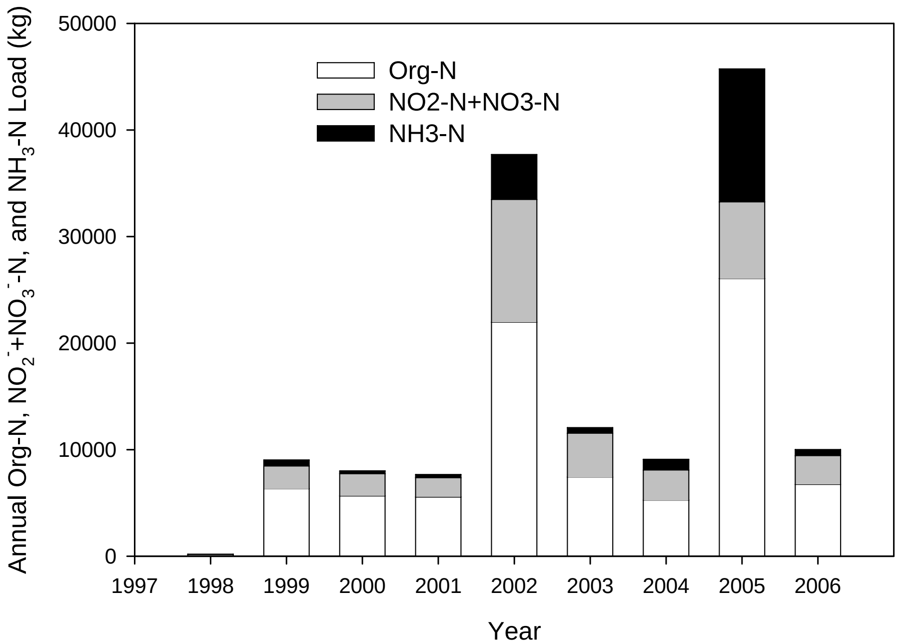


Figure A7.22. Annual loads (kg yr^{-1}) of phosphorus (a) and nitrogen (b) from 1995 to 2006 in Battersea Drain. Battersea Drain is a high but irrigated agricultural intensity watershed that lies in the grassland ecoregion.

Table A7.22. Median, minimum, maximum load for nutrient parameters and annual flow volumes in Battersea Drain, 1998 to 2006.

	Median (kg yr^{-1})	Minimum (kg yr^{-1})	Year	Maximum (kg yr^{-1})	Year
Total N	9108	198	1998	45745	2005
Organic N	6350	155		26058	
NH_3-N	570	10		12489	
NO_2+NO_3-N	2688	33		11512	2002
Total P	774	24		17957	2005
TPP	499	13		358	2001
TDP	184	11		12967	2005
Annual stream volume (hm^3)	10.4	0.3	1998	13.4	2005

Table A7.23. Annual stream volumes for each watershed during the CAESA and AESA monitoring periods. Stream flow data were provided by Water Survey of Canada and annual stream volumes were calculated by FLUX (U.S. Army Corps of Engineers 1995).

	Annual Stream Flow Volume (hm^3)											
Watershed	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Battersea Drain				0.3	7.4	11.9	9.2	10.8	10.4	11.0	13.4	10.4
Blindman River	11.6	63.1	45.0	21.8	64.1	57.1	21.0	11.8	30.5	11.6	33.1	11.8
Buffalo Creek	3.8	5.3	11.0	2.6	3.3	3.4	2.2	1.7	2.8	4.6	6.1	6.0
Crowfoot Creek	22.2	49.0	70.9	31.6	26.2	28.9	19.1	23.9	35.7	39.7	41.4	50.9
Grande Prairie Creek					2.5	1.5	6.1	9.8	10.3	8.8	12.1	1.9
Hines Creek					0.2	21.2	17.0	10.6	22.6	11.1	22.3	0.5
Haynes M1 Creek	0.1	1.0	1.0	0.1	1.2	0.3	0.1	0.7	1.2	0.0	0.7	0.5
Haynes M6 Creek	0.3	5.6	4.1	0.1	3.3	0.9	0.1	0.8	2.5	0.0	2.9	1.6
Kleskun Main Drain					1.6	0.0	0.6	2.5	2.4	1.7	1.7	0.1
Meadow Creek	8.9	5.0	4.6		1.1	0.4	0.4	6.2	3.6	1.0	10.8	4.0
New West Coulee					22.9	29.6	26.6	18.6	26.5	16.4	21.0	15.9
Paddle River	16.4	42.1	45.5		22.3	9.3	19.7	8.1	8.7	6.9	16.6	4.4
Prairie Blood Coulee	5.1	2.7			0.1	0.6	0.5	6.2	2.8	0.6	10.1	7.9
Ray Creek	0.4	1.5	2.4	0.4	2.0	0.7	0.2	0.2	1.9	1.2	2.4	2.3
Renwick Creek	0.6	1.5	3.6	0.2	0.9	0.1	0.0	0.0	1.1	0.7	0.9	0.7
Rose Creek	40.9	77.4	67.6	52.0	85.4	67.0	28.0	29.8	37.4	33.1	81.0	29.1
Stretton Creek	0.9		1.9	0.2	0.5					1.2	1.7	2.3
Strawberry Creek	6.4	38.2	45.6	7.6	44.2	54.3	14.7	9.9	8.3	5.3	31.8	6.9
Threehills Creek	0.7	5.7	8.3	0.5	6.1	1.3	0.4	0.9	6.1	1.3	4.8	5.4
Tomahawk Creek	2.7	10.8	12.8	2.2	9.1	2.8	4.1	2.3	3.9	3.1	7.4	0.3
Trout Creek	51.1	22.7	22.1		6.4	1.8	3.1	30.4	19.2	8.0	74.0	24.5
Wabash Creek					0.1	0.3	0.1	2.3	7.5	2.6	7.7	0.7
Willow Creek					7.8	4.1	8.1	24.1	9.8	12.6	41.4	10.0

APPENDIX 8: NUTRIENT EXPORT SEASONALITY BOX PLOTS

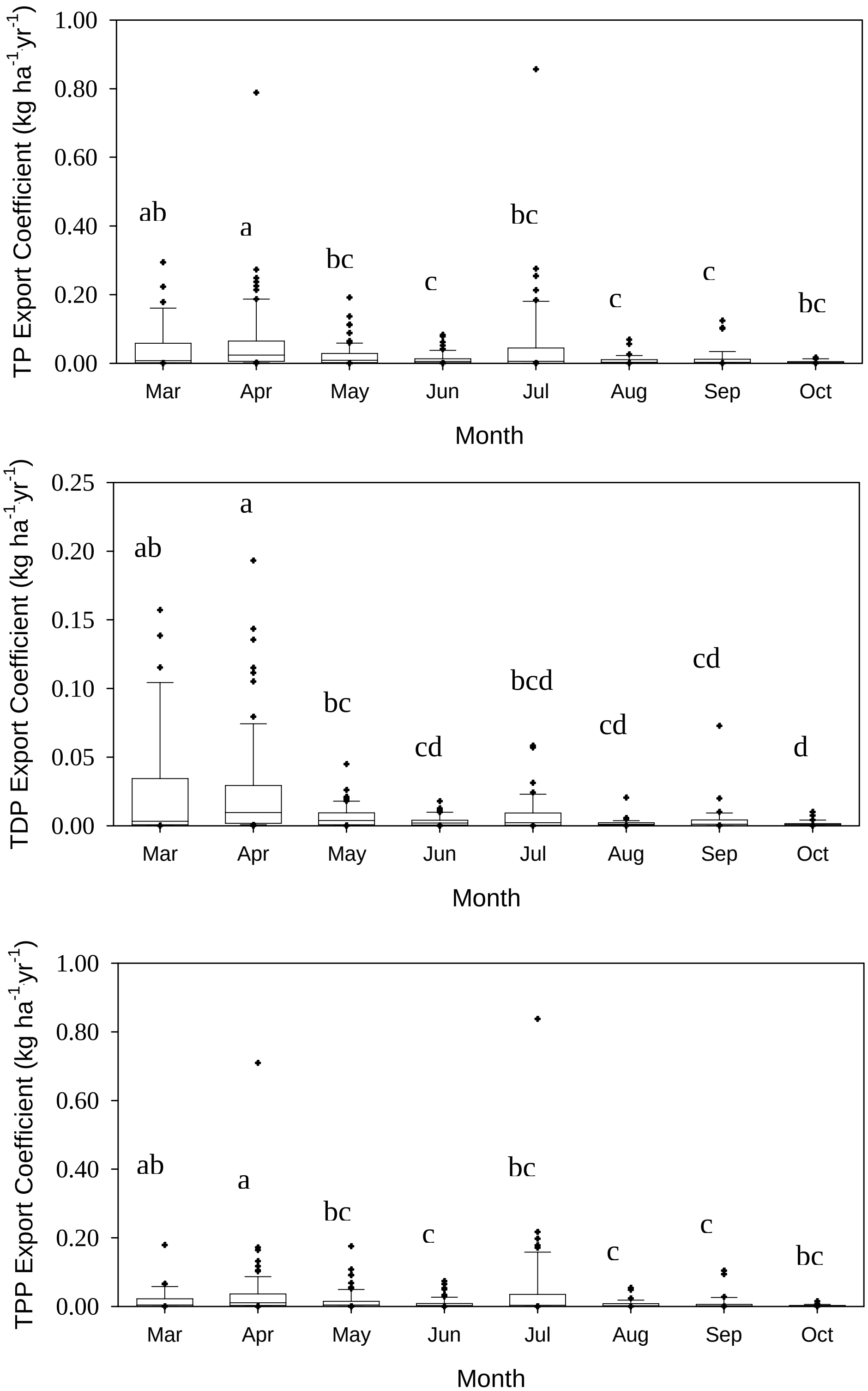


Figure A8.1. Seasonal trends in TP (a), TDP (b), and TPP* (c) exports in the Boreal ecoregion for median monthly data (1999-2006). Significance level at 0.005 (* p<0.01).

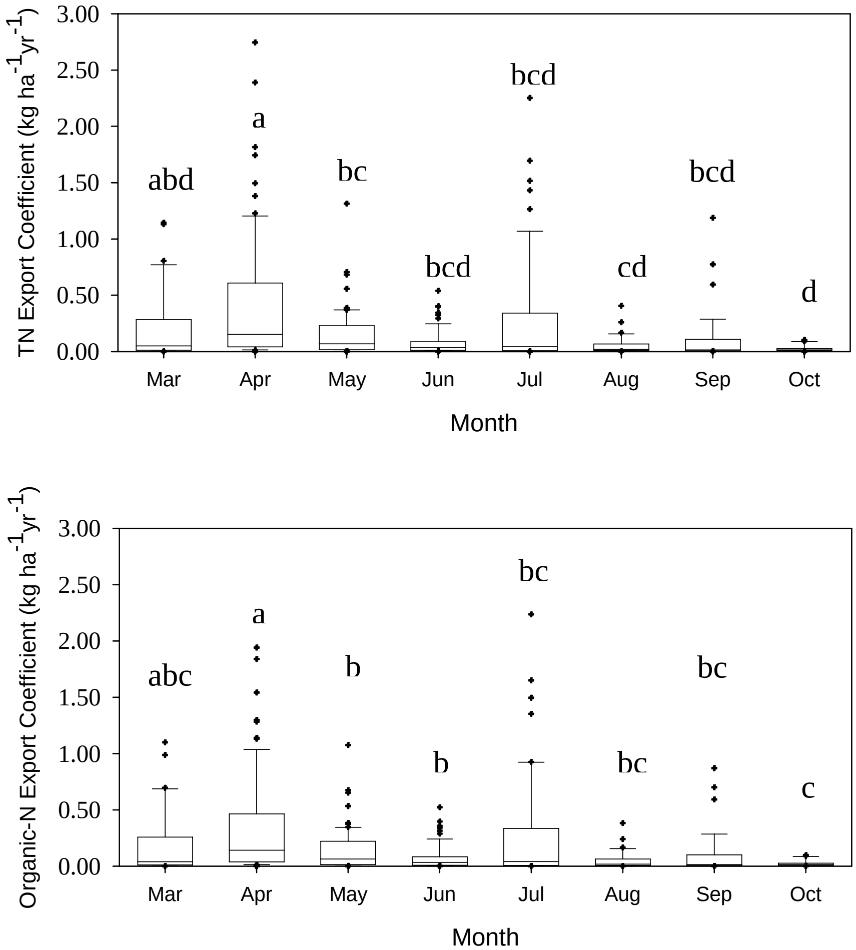


Figure A8.2. Seasonal trends in TN (a) and Org N (b) exports in the Boreal ecoregion for median monthly data (1999-2006). Significance level at 0.005.

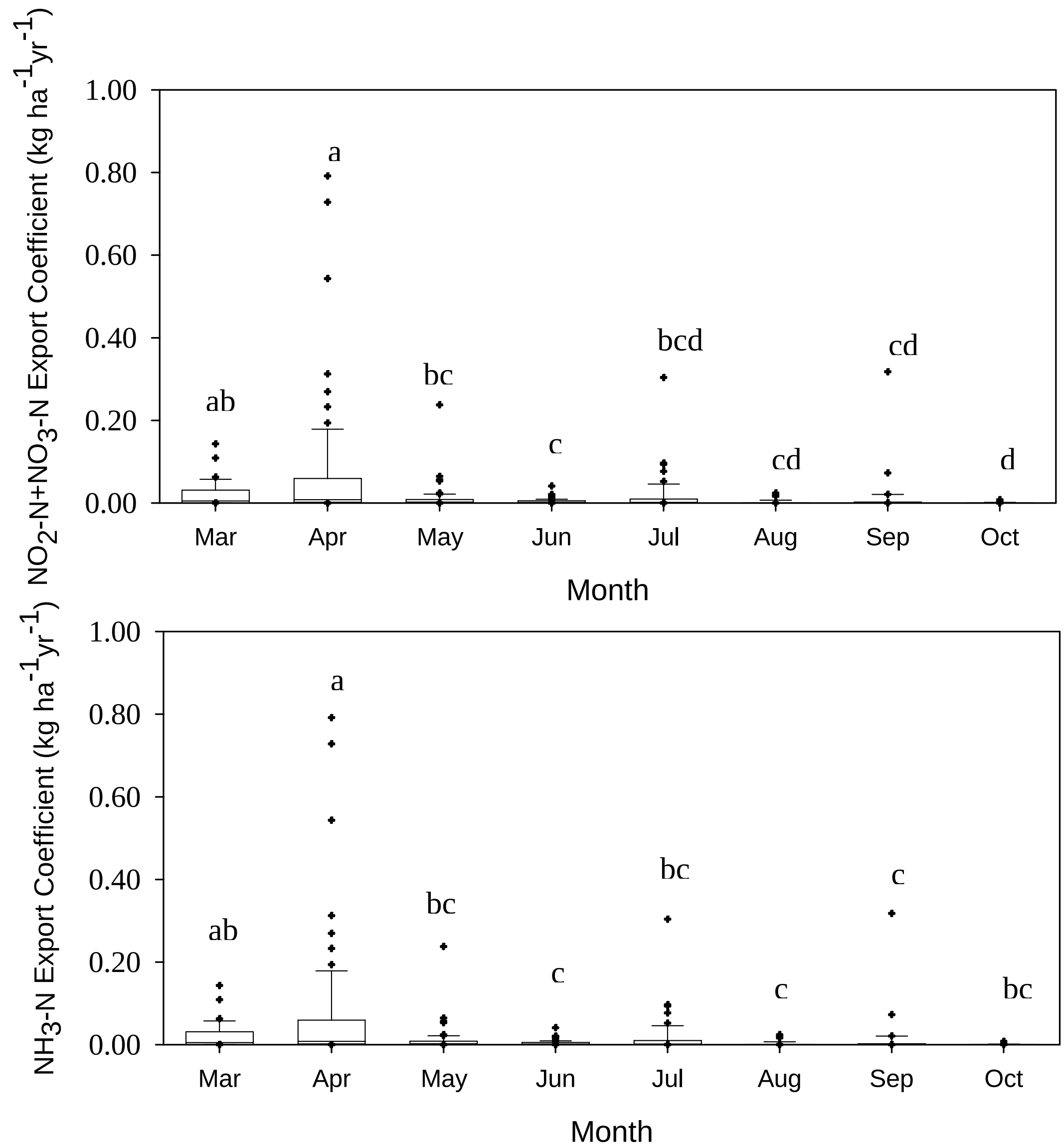


Figure A8.3. Seasonal trends in NO_2-NO_3-N (a) and NH_3-N (b) exports in the Boreal ecoregion for median monthly data (1999-2006). Significance level at 0.005.

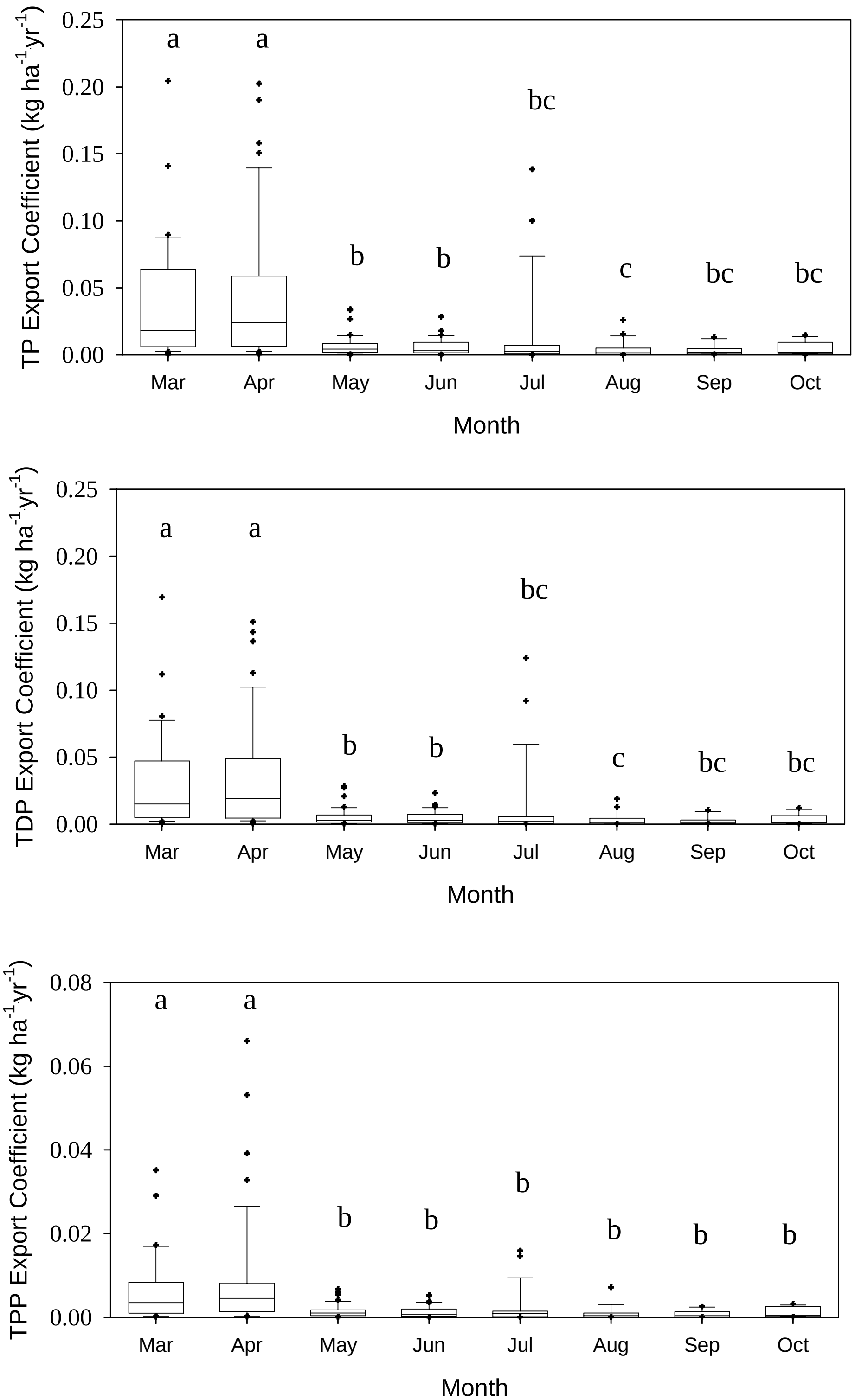


Figure A8.4. Seasonal trends in TP (a), TDP (b), and TPP (c) exports in the Parkland ecoregion for median monthly data (1999-2006). Significance level at 0.005.

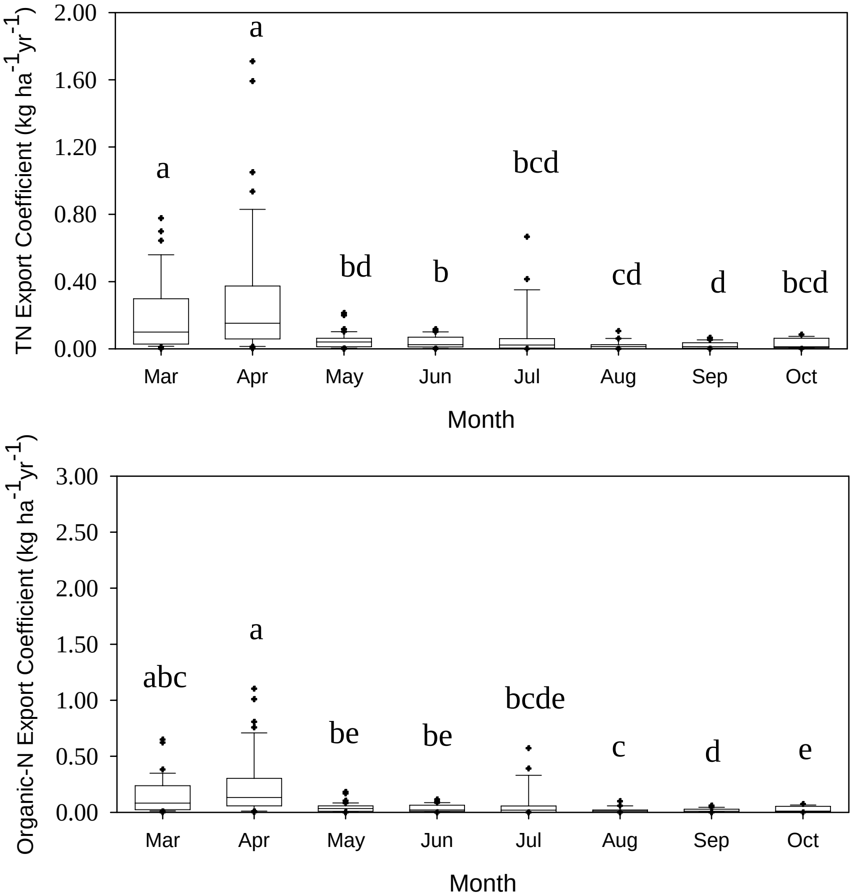
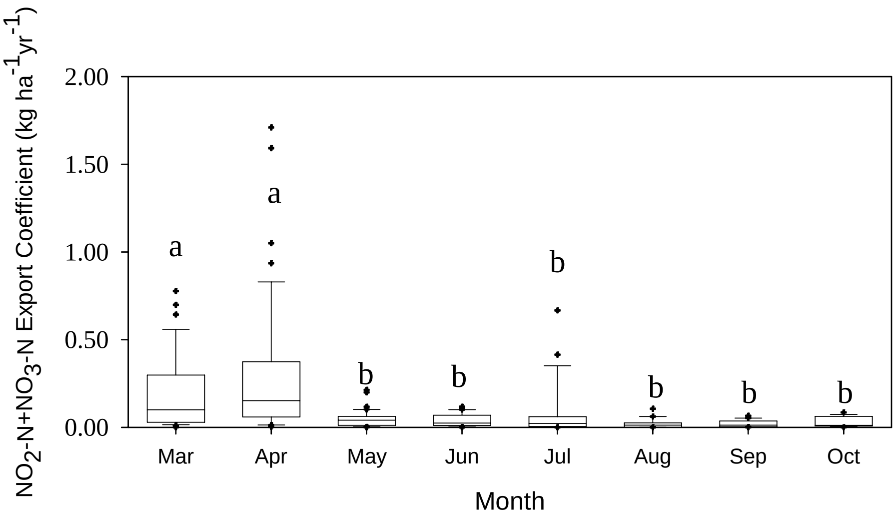


Figure A8.5. Seasonal trends in TN (a) and Org N (b) exports in the Parkland ecoregion for median monthly data (1999-2006). Significance level at 0.005.



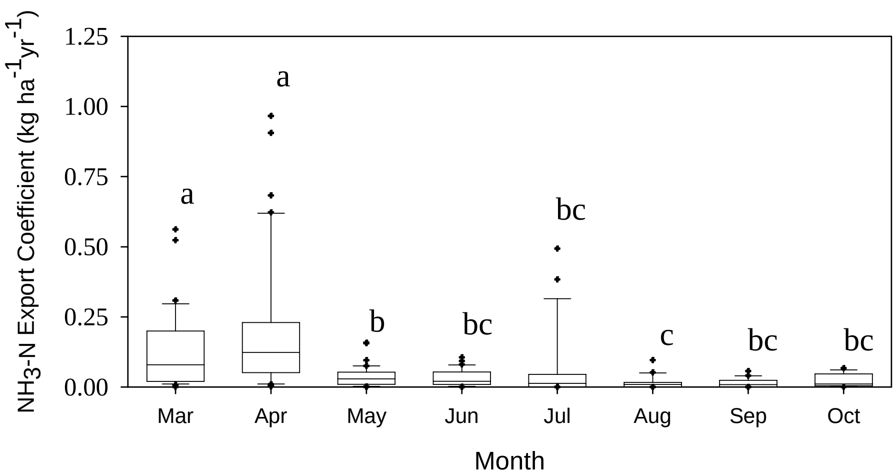


Figure A8.6. Seasonal trends in NO_2-NO_3-N (a) and NH_3-N (b) exports in the Parkland ecoregion for median monthly data (1999-2006). Significance level at 0.005.

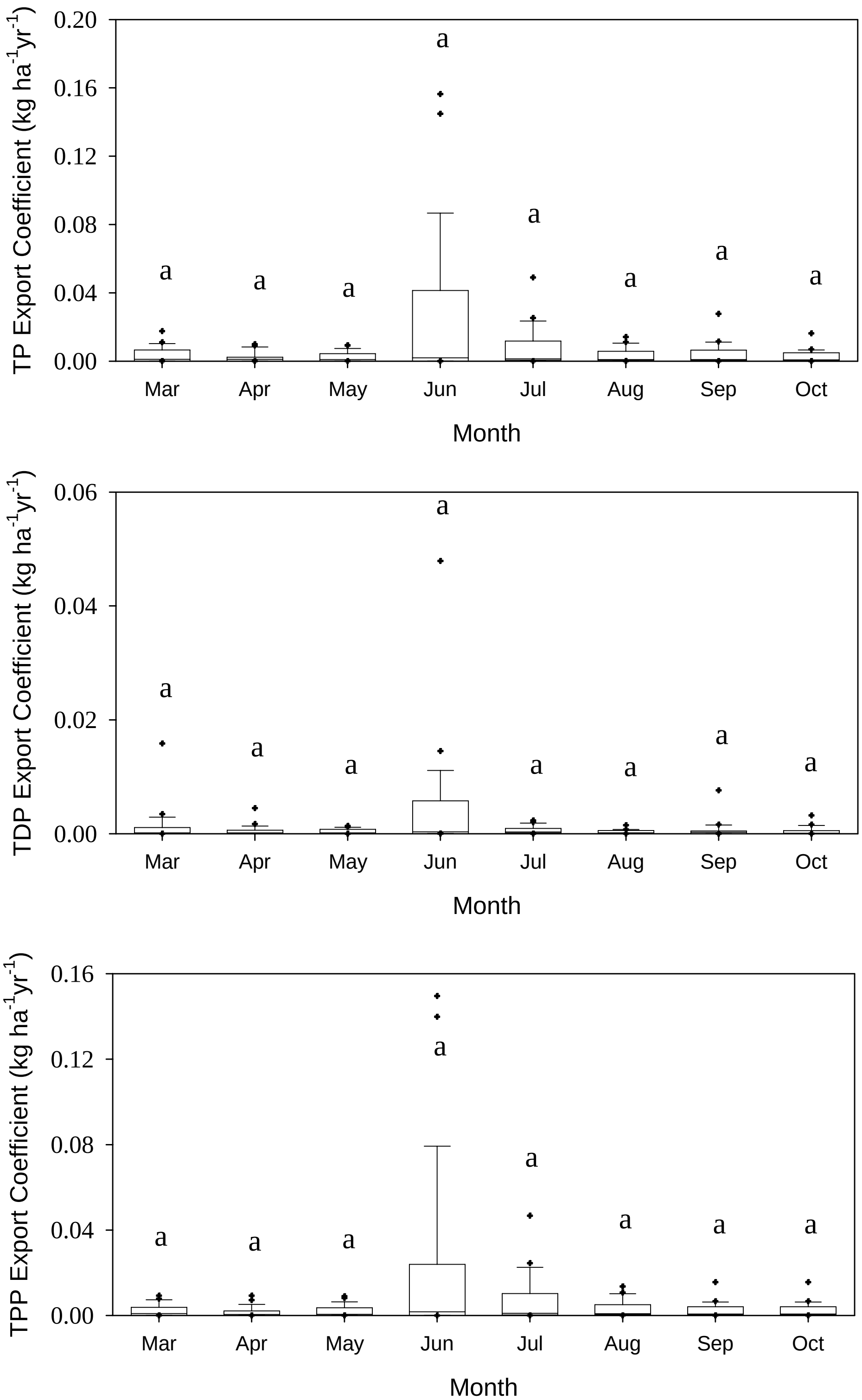
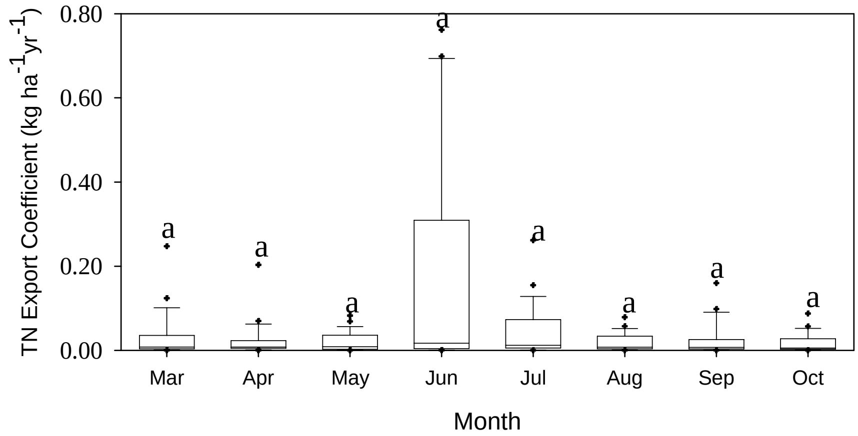


Figure A8.7. Seasonal trends in TP (a), TDP (b), and TPP (c) exports in the Grassland ecoregion for median monthly data (1999-2006). Significance level at 0.005.



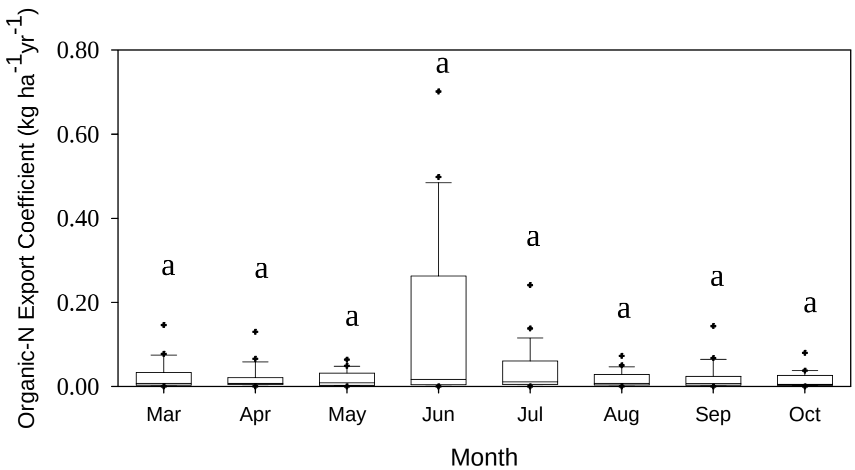


Figure A8.8. Seasonal trends in TN (a) and Org N (b) exports in the Grassland ecoregion for median monthly data (1999-2006). Significance level at 0.005.

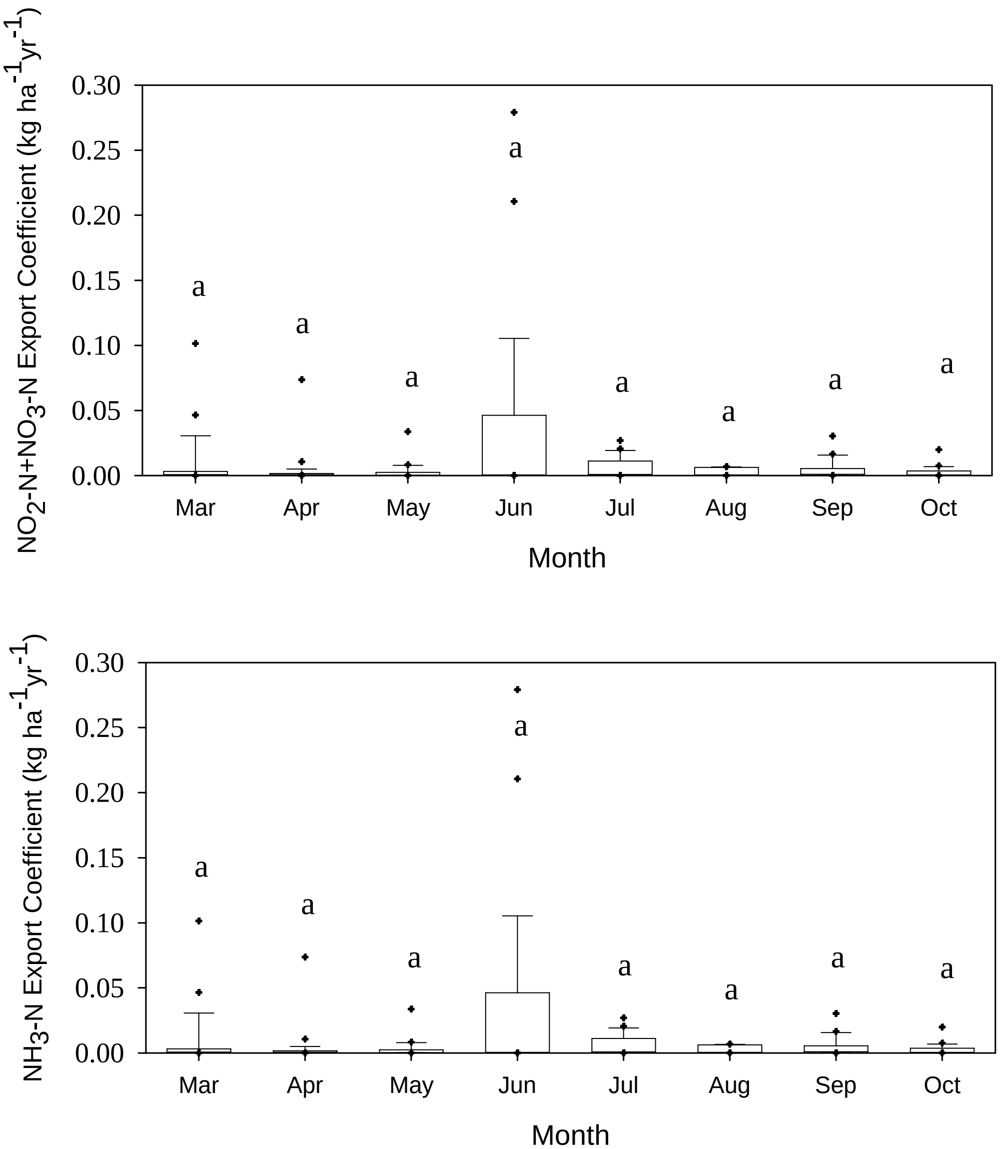


Figure A8.9. Seasonal trends in NO_2-NO_3-N (a) and NH_3-N (b) exports in the Grassland ecoregion for median monthly data (1999-2006). Significance level at 0.005.

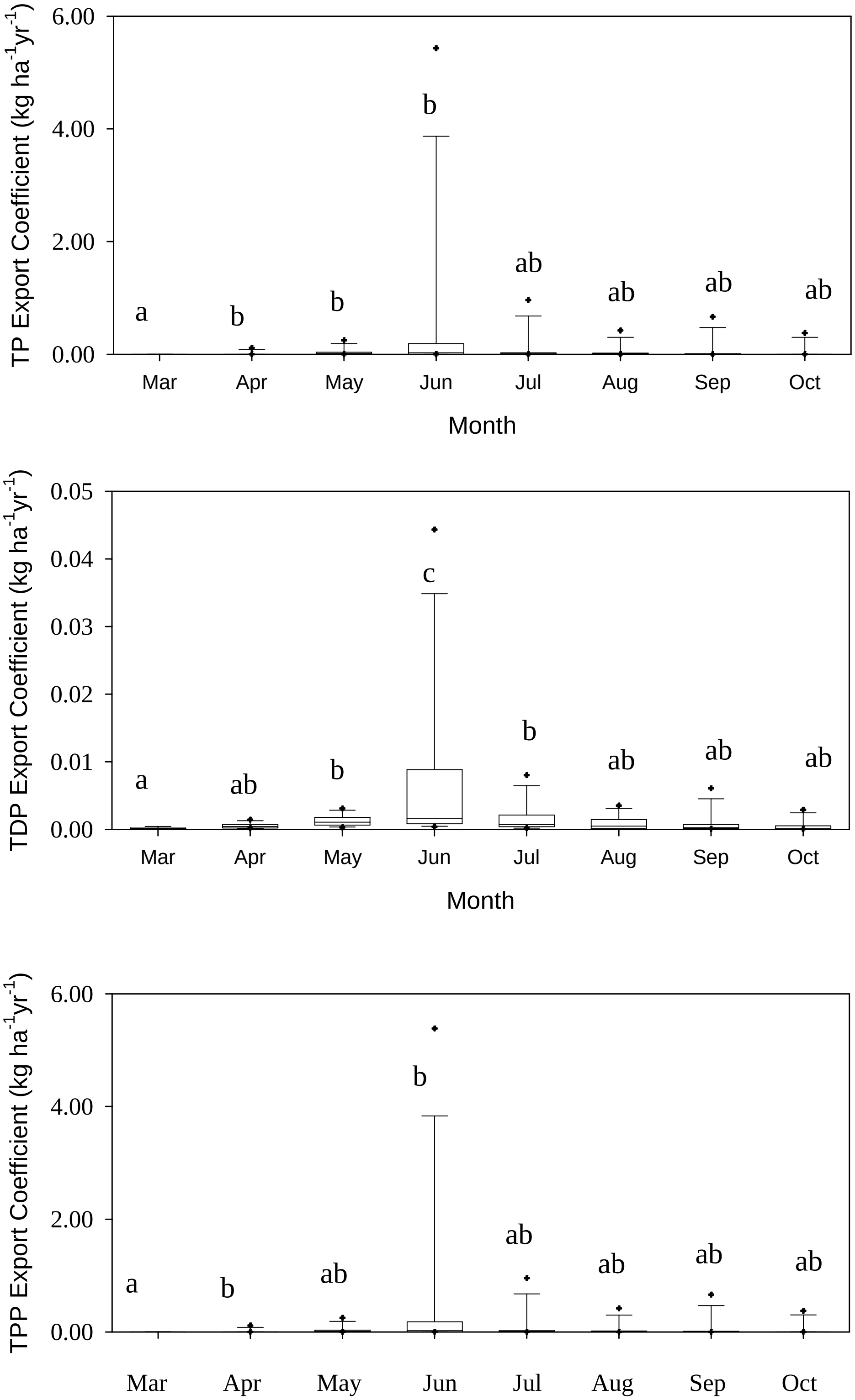


Figure A8.10. Seasonal trends in TP (a), TDP (b), and TPP (c) exports in the Continental Divide ecoregion for median monthly data (1999-2006). Significance level at 0.01.

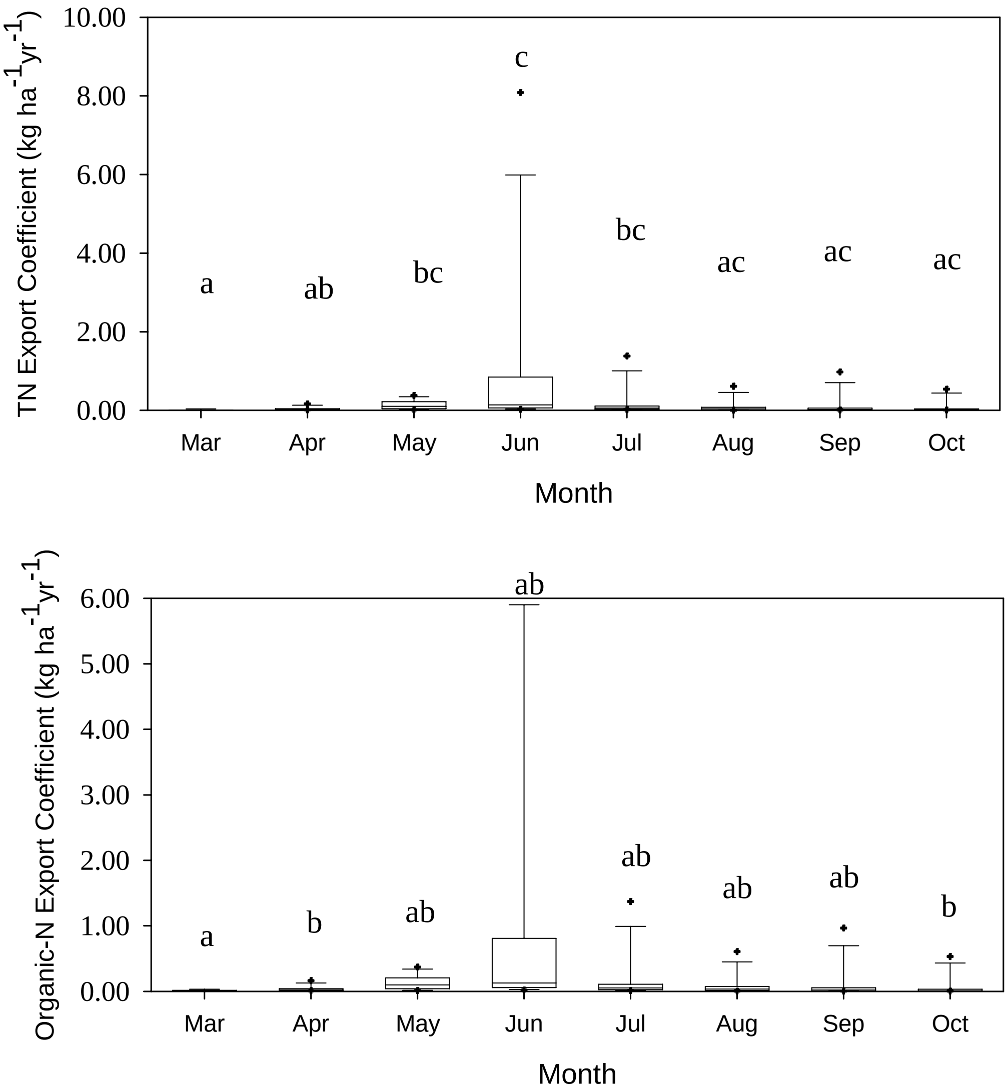


Figure A8.11. Seasonal trends in TN (a) and Org N (b) exports in the Continental Divide ecoregion for median monthly data (1999-2006). Significance level at 0.01.

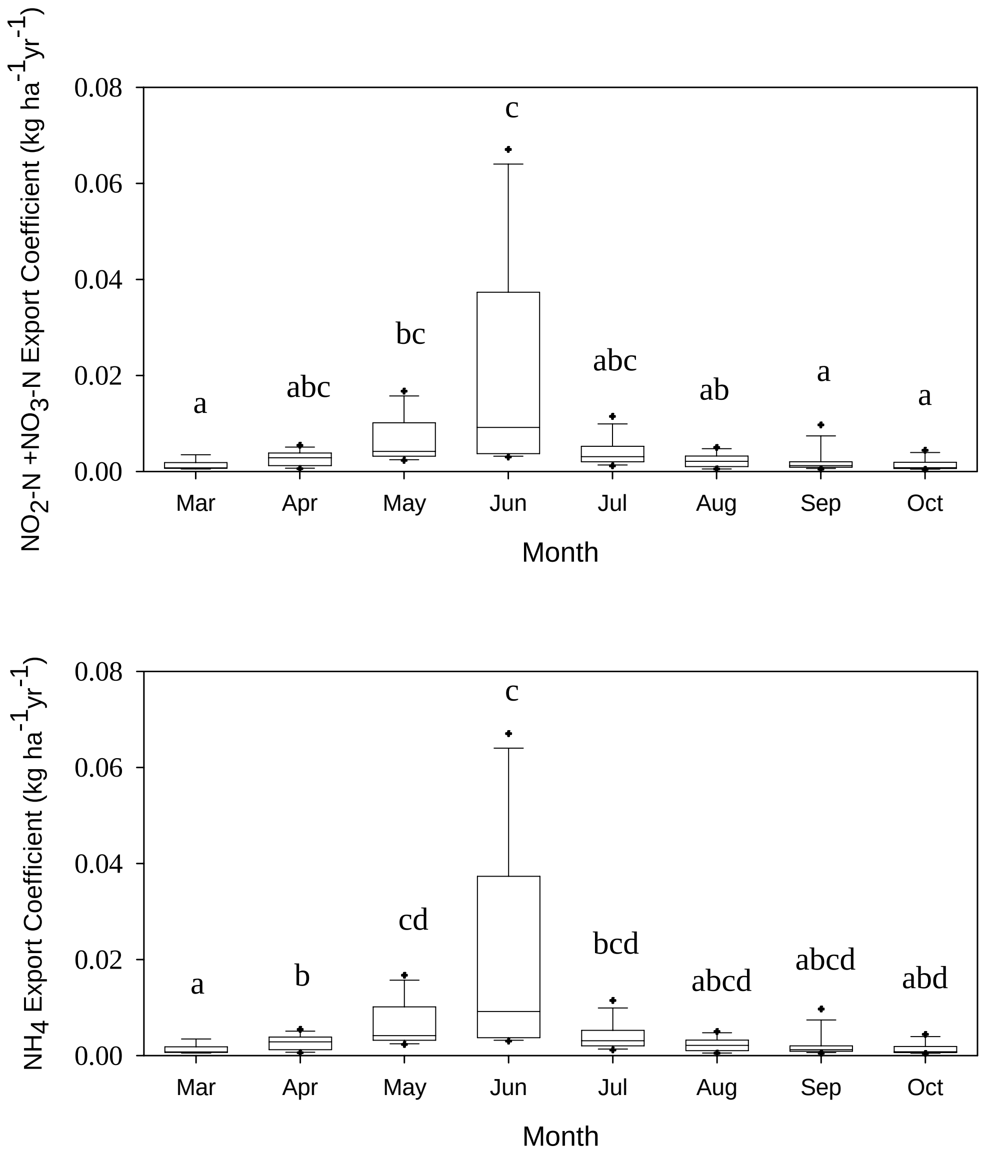


Figure A8.12. Seasonal trends in NO_2-NO_3-N (a) and NH_3-N (b) exports in the Continental Divide ecoregion for median monthly data (1999-2006). Significance level at 0.01.

APPENDIX 9: SUMMARY STATISTICS FOR AMBIENT DATA, FWMC, MASS LOADING, AND EXPORT COEFFICIENTS

Table A9.1. Summary statistics for instream concentrations (mg L^{-1}) from 1999 to 2006.

AESA Watershed	Number of Years	Summary Statistic	Total Nitrogen TN	Organic Nitrogen ORGN	Nitrite-N + Nitrate-N N23	Ammonia-N NH4	Total Phosporus TP	Total Dissolved Phosphorus TDP	Total Particulate Phosphorus TPP
Battersea Drain		N of cases	151	151	152	152	152	151	151
Battersea Drain		Minimum	0.233	0.160	0.003	0.003	0.005	0.001	0.000
Battersea Drain	8	Maximum	17.380	8.850	10.200	4.850	4.860	3.430	2.020
Battersea Drain		Median	0.999	0.602	0.156	0.043	0.069	0.018	0.047
Battersea Drain		Mean	2.591	1.108	1.222	0.264	0.234	0.145	0.089
Blindman River		N of cases	179	179	179	179	179	179	179
Blindman River		Minimum	0.133	0.120	0.003	0.003	0.020	0.009	0.000
Blindman River	8	Maximum	6.540	4.398	0.698	1.950	0.982	0.655	0.517
Blindman River		Median	1.165	1.099	0.012	0.031	0.107	0.058	0.045
Blindman River		Mean	1.552	1.318	0.065	0.168	0.177	0.104	0.072
Buffalo Creek		N of cases	119	119	119	119	119	119	119
Buffalo Creek		Minimum	0.833	0.530	0.003	0.003	0.036	0.018	0.000
Buffalo Creek	8	Maximum	4.684	4.642	0.653	1.120	0.920	0.740	0.217
Buffalo Creek		Median	1.473	1.322	0.016	0.060	0.115	0.082	0.030
Buffalo Creek		Mean	1.678	1.438	0.049	0.191	0.146	0.100	0.046
Crowfoot Creek		N of cases	154	155	154	155	155	155	155
Crowfoot Creek		Minimum	0.133	0.115	0.003	0.003	0.038	0.013	0.000
Crowfoot Creek	8	Maximum	8.190	4.210	4.100	2.140	4.600	0.600	4.410
Crowfoot Creek		Median	1.003	0.883	0.008	0.033	0.116	0.066	0.052
Crowfoot Creek		Mean	1.580	1.145	0.281	0.156	0.222	0.114	0.108
Drain S6		N of cases	134	134	134	134	134	134	134
Drain S6		Minimum	0.028	0.019	0.003	0.003	0.004	0.001	0.000
Drain S6	8	Maximum	3.248	2.918	1.030	0.429	0.764	0.511	0.253
Drain S6		Median	0.682	0.514	0.058	0.059	0.034	0.012	0.020
Drain S6		Mean	0.811	0.606	0.133	0.071	0.059	0.027	0.033
Grande Prairie Creek		N of cases	104	104	104	104	104	104	104
Grande Prairie Creek		Minimum	0.343	0.314	0.003	0.003	0.019	0.007	0.001
Grande Prairie Creek	8	Maximum	7.700	4.041	4.300	0.628	0.720	0.630	0.598
Grande Prairie Creek		Median	1.544	1.405	0.017	0.038	0.119	0.053	0.049
Grande Prairie Creek		Mean	2.066	1.564	0.427	0.074	0.166	0.089	0.077
Hines Creek		N of cases	115	115	115	115	115	115	115
Hines Creek		Minimum	0.103	0.069	0.003	0.003	0.033	0.004	0.004
Hines Creek	8	Maximum	4.251	3.330	0.279	1.440	1.160	0.357	1.153
Hines Creek		Median	1.193	1.130	0.003	0.037	0.109	0.060	0.049
Hines Creek		Mean	1.314	1.174	0.017	0.123	0.163	0.073	0.090
Haynes Creek M6		N of cases	94	94	95	95	95	93	93
Haynes Creek M6		Minimum	1.403	1.309	0.003	0.012	0.123	0.113	0.000
Haynes Creek M6		Maximum	10.110	5.436	4.370	3.050	2.150	1.870	0.360
Haynes Creek M6		Median	3.041	2.642	0.010	0.107	0.630	0.562	0.058
Haynes Creek M6		Mean	3.289	2.796	0.288	0.235	0.680	0.600	0.086
Kleskun Drain		N of cases	74	74	74	74	74	74	74
Kleskun Drain		Minimum	0.315	0.257	0.003	0.003	0.030	0.011	0.000
Kleskun Drain	8	Maximum	4.990	4.214	2.660	0.574	0.959	0.630	0.329
Kleskun Drain		Median	1.857	1.677	0.047	0.045	0.148	0.114	0.047
Kleskun Drain		Mean	2.167	1.825	0.241	0.102	0.208	0.143	0.065
Meadow Creek		N of cases	152	152	152	152	152	151	152
Meadow Creek		Minimum	0.063	0.058	0.003	0.003	0.003	0.001	0.000
Meadow Creek	8	Maximum	5.557	4.686	0.512	0.514	0.782	0.468	0.726
Meadow Creek		Median	0.758	0.717	0.003	0.020	0.076	0.012	0.060
Meadow Creek		Mean	0.995	0.909	0.049	0.037	0.104	0.020	0.084
New West Coulee		N of cases	144	144	144	144	144	143	144
New West Coulee		Minimum	0.243	0.224	0.003	0.003	0.006	0.001	0.000
New West Coulee		Maximum	7.300	4.650	4.300	1.830	1.340	0.501	0.866
New West Coulee		Median	0.693	0.627	0.007	0.017	0.078	0.028	0.038
New West Coulee		Mean	1.202	0.902	0.175	0.125	0.126	0.052	0.075
Paddle River		N of cases	156	156	156	156	156	154	156
Paddle River		Minimum	0.373	0.256	0.003	0.003	0.021	0.012	0.000
Paddle River	8	Maximum	4.758	3.982	0.778	0.654	0.744	0.542	0.575
Paddle River		Median	0.846	0.800	0.005	0.027	0.073	0.037	0.035
Paddle River		Mean	1.046	0.945	0.048	0.053	0.130	0.065	0.066

Table A9.1. Cont. Summary statistics for instream concentrations (mg L^{-1}) from 1999 to 2006.

AESA Watershed	Number of Years	Summary Statistic	Total Nitrogen TN	Organic Nitrogen ORGN	Nitrite-N + Nitrate-N N23	Ammonia-N NH4	Total Phosporus TP	Total Dissolved Phosphorus TDP	Total Particulate Phosphorus TPP
Prairie Blood Coulee		N of cases	110	110	110	110	110	110	110
Prairie Blood Coulee		Minimum	0.333	0.323	0.003	0.003	0.003	0.001	0.000
Prairie Blood Coulee	8	Maximum	28.479	21.500	3.340	6.000	0.941	0.741	0.258
Prairie Blood Coulee		Median	0.974	0.940	0.003	0.017	0.024	0.009	0.015
Prairie Blood Coulee		Mean	1.690	1.268	0.316	0.106	0.057	0.034	0.023
Ray Creek		N of cases	141	141	141	141	141	141	141
Ray Creek		Minimum	0.803	0.707	0.003	0.003	0.033	0.024	0.000
Ray Creek	8	Maximum	6.070	4.384	1.910	0.971	1.310	1.090	0.332
Ray Creek		Median	1.513	1.402	0.003	0.034	0.192	0.149	0.027
Ray Creek		Mean	1.777	1.585	0.116	0.075	0.240	0.195	0.045
Renwick Creek		N of cases	108	108	108	108	108	107	107
Renwick Creek		Minimum	1.463	1.403	0.003	0.003	0.047	0.046	0.000
Renwick Creek	8	Maximum	8.690	4.864	3.360	1.160	1.530	1.350	0.317
Renwick Creek		Median	2.147	1.951	0.003	0.042	0.485	0.446	0.039
Renwick Creek		Mean	2.562	2.164	0.290	0.108	0.546	0.492	0.057
Rose Creek		N of cases	182	182	182	182	182	181	181
Rose Creek		Minimum	0.028	-0.034	0.003	0.003	0.013	0.001	0.000
Rose Creek	8	Maximum	5.769	5.698	0.731	1.250	2.570	0.800	2.546
Rose Creek		Median	0.833	0.773	0.003	0.021	0.049	0.020	0.026
Rose Creek		Mean	1.008	0.934	0.023	0.051	0.136	0.043	0.093
Stretton Creek		N of cases	49	49	49	49	49	49	49
Stretton Creek		Minimum	0.303	0.274	0.003	0.009	0.061	0.023	0.003
Stretton Creek	4	Maximum	5.880	3.515	1.650	0.875	1.000	0.924	0.146
Stretton Creek		Median	1.713	1.615	0.022	0.044	0.319	0.278	0.048
Stretton Creek		Mean	2.182	1.700	0.371	0.110	0.351	0.298	0.053
Strawberry Creek		N of cases	132	132	132	132	132	132	132
Strawberry Creek		Minimum	0.263	0.245	0.003	0.003	0.011	0.001	0.000
Strawberry Creek	8	Maximum	6.850	4.950	3.030	1.560	3.150	0.680	3.044
Strawberry Creek		Median	1.123	1.074	0.032	0.042	0.098	0.024	0.063
Strawberry Creek		Mean	1.689	1.334	0.197	0.158	0.262	0.069	0.193
Threehills Creek		N of cases	145	145	145	145	145	145	145
Threehills Creek		Minimum	1.043	0.760	0.003	0.003	0.062	0.043	0.000
Threehills Creek	8	Maximum	7.910	4.980	2.940	1.940	1.630	1.370	0.270
Threehills Creek		Median	2.013	1.858	0.003	0.047	0.308	0.260	0.053
Threehills Creek		Mean	2.414	1.989	0.227	0.198	0.380	0.312	0.068
Tomahawk Creek		N of cases	154	154	154	154	154	154	154
Tomahawk Creek		Minimum	0.823	0.530	0.003	0.003	0.023	0.016	0.000
Tomahawk Creek	8	Maximum	5.590	5.184	2.010	0.988	0.995	0.416	0.938
Tomahawk Creek		Median	2.075	1.842	0.068	0.068	0.162	0.081	0.072
Tomahawk Creek		Mean	2.284	1.968	0.166	0.149	0.214	0.104	0.110
Trout Creek		N of cases	168	168	168	168	168	168	168
Trout Creek		Minimum	0.063	0.058	0.003	0.003	0.003	0.001	0.000
Trout Creek	8	Maximum	3.285	2.997	0.227	0.129	0.580	0.071	0.573
Trout Creek		Median	0.333	0.316	0.003	0.008	0.021	0.004	0.016
Trout Creek		Mean	0.446	0.404	0.027	0.014	0.048	0.006	0.041
Wabash Creek		N of cases	100	100	100	100	100	100	100
Wabash Creek		Minimum	0.852	0.809	0.003	0.002	0.039	0.031	0.000
Wabash Creek	8	Maximum	8.400	5.580	2.970	6.720	2.360	1.660	1.630
Wabash Creek		Median	2.259	1.813	0.017	0.065	0.293	0.134	0.129
Wabash Creek		Mean	2.712	1.916	0.338	0.458	0.420	0.251	0.169
Willow Creek		N of cases	167	167	167	167	167	167	167
Willow Creek		Minimum	0.028	-0.048	0.003	0.001	0.001	0.001	0.000
Willow Creek	8	Maximum	4.373	4.275	0.198	0.075	3.200	0.132	3.180
Willow Creek		Median	0.113	0.098	0.008	0.003	0.005	0.002	0.002
Willow Creek		Mean	0.207	0.178	0.021	0.008	0.039	0.004	0.035

Table A9.2. Summary statistics for FWMC (mg L^{-1}) from 1999 to 2006.

AESA Watershed	Number of Years	Summary Statistic	Total Nitrogen	Organic-N	Nitrite-N + Nitrate-N	Ammonia-N	Total Phosphorus	Total Dissolved Phosphorus	Total Particulate Phosphorus
			TN	Org-N	NO_2^--N +NO_3^--N	NH_3-N	TP	TDP	TPP
Battersea Drain		Median	1.062	0.680	0.272	0.066	0.105	0.024	0.049
Battersea Drain		Mean	1.575	0.970	0.397	0.207	0.310	0.194	0.116
Battersea Drain	8	Minimum	0.672	0.475	0.176	0.021	0.038	0.007	0.031
Battersea Drain		Maximum	3.498	2.036	1.067	0.934	1.342	0.969	0.373
Battersea Drain		St. Dev.	1.177	0.643	0.295	0.317	0.457	0.342	0.120
Blindman River		Median	1.973	1.732	0.130	0.224	0.297	0.142	0.150
Blindman River		Mean	2.160	1.778	0.124	0.258	0.309	0.164	0.146
Blindman River	8	Minimum	1.305	1.079	0.032	0.061	0.136	0.058	0.049
Blindman River		Maximum	3.495	2.858	0.271	0.560	0.536	0.338	0.241
Blindman River		St. Dev.	0.737	0.572	0.074	0.176	0.137	0.095	0.065
Buffalo Creek		Median	1.982	1.630	0.048	0.255	0.157	0.089	0.048
Buffalo Creek		Mean	2.035	1.661	0.097	0.275	0.180	0.117	0.063
Buffalo Creek	8	Minimum	1.284	1.243	0.014	0.026	0.117	0.076	0.029
Buffalo Creek		Maximum	2.906	2.272	0.361	0.520	0.327	0.212	0.115
Buffalo Creek		St. Dev.	0.479	0.312	0.118	0.155	0.076	0.051	0.033
Crowfoot Creek		Median	1.814	1.265	0.446	0.127	0.234	0.154	0.094
Crowfoot Creek		Mean	2.027	1.372	0.481	0.181	0.283	0.154	0.129
Crowfoot Creek	8	Minimum	0.744	0.660	0.043	0.041	0.109	0.060	0.033
Crowfoot Creek		Maximum	3.594	2.422	0.972	0.401	0.538	0.281	0.311
Crowfoot Creek		St. Dev.	1.132	0.613	0.443	0.141	0.158	0.076	0.107
Grande Prairie Creek		Median	2.268	1.863	0.308	0.063	0.253	0.092	0.127
Grande Prairie Creek		Mean	2.679	1.986	0.603	0.080	0.249	0.104	0.145
Grande Prairie Creek	8	Minimum	1.633	1.454	0.050	0.045	0.125	0.067	0.044
Grande Prairie Creek		Maximum	4.513	3.238	2.083	0.166	0.473	0.145	0.394
Grande Prairie Creek		St. Dev.	1.096	0.565	0.706	0.040	0.109	0.032	0.110
Hines Creek		Median	1.310	1.236	0.011	0.054	0.142	0.059	0.063
Hines Creek		Mean	1.258	1.161	0.012	0.085	0.137	0.065	0.071
Hines Creek	8	Minimum	0.998	0.923	0.003	0.018	0.098	0.048	0.049
Hines Creek		Maximum	1.687	1.369	0.023	0.355	0.173	0.092	0.116
Hines Creek		St. Dev.	0.242	0.177	0.007	0.111	0.028	0.018	0.025
Haynes Creek (M6)		Median	4.321	3.098	0.788	0.518	0.880	0.808	0.089
Haynes Creek (M6)		Mean	4.496	3.06	1.108	0.472	0.803	0.744	0.100
Haynes Creek (M6)	7 (6)	Minimum	2.392	1.805	0.010	0.095	0.360	0.269	0.056
Haynes Creek (M6)		Maximum	8.589	4.562	3.472	0.751	1.203	1.056	0.147
Haynes Creek (M6)		St. Dev.	1.984	0.937	1.152	0.272	0.307	0.291	0.034
Kleskun Drain		Median	2.741	2.287	0.265	0.115	0.362	0.237	0.101
Kleskun Drain		Mean	2.939	2.258	0.503	0.168	0.338	0.240	0.098
Kleskun Drain	8	Minimum	1.846	1.544	0.096	0.024	0.147	0.127	0.020
Kleskun Drain		Maximum	3.931	3.172	1.605	0.531	0.494	0.349	0.149
Kleskun Drain		St. Dev.	0.714	0.535	0.552	0.161	0.112	0.074	0.045
Meadow Creek		Median	1.108	0.973	0.071	0.030	0.140	0.018	0.105
Meadow Creek		Mean	1.198	1.06	0.087	0.051	0.141	0.033	0.109
Meadow Creek	8	Minimum	0.662	0.631	0.010	0.021	0.073	0.011	0.044
Meadow Creek		Maximum	2.218	1.927	0.246	0.155	0.233	0.137	0.203
Meadow Creek		St. Dev.	0.494	0.412	0.078	0.046	0.052	0.043	0.051
New West Coulee		Median	0.724	0.573	0.057	0.021	0.098	0.044	0.052
New West Coulee		Mean	0.807	0.632	0.143	0.031	0.096	0.046	0.050
New West Coulee	8	Minimum	0.443	0.389	0.015	0.012	0.060	0.032	0.026
New West Coulee		Maximum	1.487	1.014	0.466	0.076	0.135	0.072	0.074
New West Coulee		St. Dev.	0.367	0.211	0.181	0.023	0.026	0.014	0.017
Paddle River		Median	1.347	1.145	0.052	0.068	0.201	0.093	0.095
Paddle River		Mean	1.338	1.191	0.077	0.069	0.196	0.081	0.115
Paddle River	8	Minimum	0.687	0.645	0.012	0.012	0.073	0.035	0.029
Paddle River		Maximum	2.167	1.838	0.216	0.129	0.302	0.129	0.215
Paddle River		St. Dev.	0.471	0.389	0.068	0.040	0.087	0.038	0.075

Table A9.2. Cont. Summary statistics for FWMC (mg L^{-1}) from 1999 to 2006.

AESA Watershed	Number of Years	Summary Statistic	Total Nitrogen	Organic-N	Nitrite-N + Nitrate-N	Ammonia-N	Total Phosphorus	Total Dissolved Phosphorus	Total Particulate Phosphorus
			TN	Org-N	NO_2^--N +NO_3^--N	NH_3-N	TP	TDP	TPP
Prairie Blood Coulee		Median	1.110	1	0.105	0.028	0.087	0.054	0.026
Prairie Blood Coulee		Mean	1.615	1.167	0.405	0.042	0.098	0.063	0.035
Prairie Blood Coulee	8	Minimum	0.705	0.68	0.004	0.012	0.009	0.005	0.002
Prairie Blood Coulee		Maximum	3.545	2.056	1.399	0.089	0.210	0.154	0.092
Prairie Blood Coulee		St. Dev.	0.978	0.43	0.536	0.031	0.077	0.057	0.031
Ray Creek		Median	1.995	1.76	0.201	0.094	0.245	0.214	0.042
Ray Creek		Mean	2.205	1.761	0.329	0.112	0.283	0.221	0.061
Ray Creek	8	Minimum	1.353	1.062	0.028	0.030	0.178	0.145	0.027
Ray Creek		Maximum	4.188	2.451	1.439	0.288	0.458	0.358	0.152
Ray Creek		St. Dev.	0.888	0.442	0.454	0.083	0.111	0.074	0.044
Renwick Creek		Median	3.453	2.543	0.626	0.257	0.787	0.692	0.098
Renwick Creek		Mean	3.890	2.699	0.917	0.266	0.778	0.666	0.112
Renwick Creek	8	Minimum	2.746	2.28	0.219	0.079	0.644	0.546	0.044
Renwick Creek		Maximum	6.566	4.032	2.217	0.500	0.920	0.750	0.199
Renwick Creek		St. Dev.	1.243	0.561	0.770	0.150	0.103	0.069	0.057
Rose Creek		Median	1.411	1.35	0.016	0.055	0.268	0.028	0.248
Rose Creek		Mean	1.459	1.39	0.019	0.050	0.309	0.035	0.274
Rose Creek	8	Minimum	0.900	0.862	0.011	0.023	0.062	0.018	0.035
Rose Creek		Maximum	2.326	2.269	0.036	0.074	0.826	0.058	0.802
Rose Creek		St. Dev.	0.465	0.465	0.008	0.016	0.236	0.015	0.240
Stretton Creek		Median	2.969	1.986	0.952	0.145	0.433	0.362	0.071
Stretton Creek		Mean	2.976	1.9	0.915	0.157	0.423	0.363	0.059
Stretton Creek	4	Minimum	2.209	1.484	0.221	0.114	0.361	0.348	0.013
Stretton Creek		Maximum	3.757	2.144	1.535	0.226	0.463	0.381	0.082
Stretton Creek		St. Dev.	0.667	0.302	0.544	0.051	0.043	0.013	0.031
Strawberry Creek		Median	3.296	2.516	0.321	0.313	0.692	0.123	0.463
Strawberry Creek		Mean	3.117	2.321	0.453	0.340	0.703	0.149	0.554
Strawberry Creek	8	Minimum	1.186	0.894	0.136	0.075	0.189	0.047	0.130
Strawberry Creek		Maximum	4.628	3.202	0.859	0.756	1.249	0.319	1.122
Strawberry Creek		St. Dev.	1.072	0.76	0.305	0.249	0.340	0.096	0.325
Threehills Creek		Median	3.571	2.461	0.580	0.494	0.550	0.439	0.104
Threehills Creek		Mean	3.529	2.403	0.675	0.446	0.602	0.484	0.118
Threehills Creek	8	Minimum	2.146	1.782	0.164	0.157	0.415	0.334	0.074
Threehills Creek		Maximum	5.301	3.079	1.787	0.613	0.966	0.792	0.205
Threehills Creek		St. Dev.	0.999	0.458	0.540	0.147	0.169	0.146	0.047
Tomahawk Creek		Median	2.916	2.387	0.289	0.230	0.356	0.120	0.204
Tomahawk Creek		Mean	2.995	2.394	0.344	0.256	0.348	0.117	0.232
Tomahawk Creek	8	Minimum	2.335	1.613	0.165	0.146	0.225	0.055	0.089
Tomahawk Creek		Maximum	4.008	3.14	0.663	0.416	0.463	0.186	0.390
Tomahawk Creek		St. Dev.	0.603	0.515	0.185	0.099	0.089	0.041	0.110
Trout Creek		Median	0.538	0.479	0.042	0.014	0.057	0.008	0.050
Trout Creek		Mean	0.608	0.534	0.052	0.022	0.106	0.008	0.099
Trout Creek	8	Minimum	0.293	0.274	0.011	0.008	0.020	0.004	0.011
Trout Creek		Maximum	1.018	0.897	0.142	0.059	0.296	0.011	0.285
Trout Creek		St. Dev.	0.264	0.217	0.045	0.018	0.104	0.003	0.103
Wabash Creek		Median	3.336	2.095	0.646	0.464	0.470	0.223	0.189
Wabash Creek		Mean	3.539	2.2	0.775	0.540	0.468	0.278	0.190
Wabash Creek	8	Minimum	1.335	1.167	0.062	0.105	0.214	0.055	0.105
Wabash Creek		Maximum	6.708	3.683	2.207	1.440	0.945	0.730	0.256
Wabash Creek		St. Dev.	2.047	0.984	0.787	0.438	0.227	0.212	0.061
Willow Creek		Median	0.283	0.256	0.020	0.010	0.043	0.004	0.039
Willow Creek		Mean	0.517	0.485	0.020	0.012	0.214	0.005	0.209
Willow Creek	8	Minimum	0.123	0.088	0.010	0.004	0.009	0.002	0.007
Willow Creek		Maximum	1.929	1.876	0.033	0.033	1.308	0.011	1.297
Willow Creek		St. Dev.	0.607	0.596	0.008	0.009	0.446	0.003	0.443

Table A9.3. Summary statistics for mass transport (kg) data from 1999 to 2006.

AESA Watershed	Number of Years	Summary Statistic	Total Nitrogen	Organic-N	Nitrite-N + Nitrate-N	Ammonia-N	Total Phosphorus	Total Dissolved Phosphorus	Total Particulate Phosphorus
			TN	Org-N	NO_2^--N +NO_3^--N	NH_3-N	TP	TDP	TPP
Battersea Drain		Median	9571	6546	2764	574	977	253	526
Battersea Drain		Mean	17431	10625	4298	2491	3698	2394	1304
Battersea Drain	8	Minimum	7678	5258	1815	251	457	82	358
Battersea Drain		Maximum	45755	26058	11513	12489	17957	12967	4990
Battersea Drain		St. Dev.	15222	8365	3400	4245	6110	4523	1603
Blindman River		Median	49846	37849	3376	5073	7541	4595	2947
Blindman River		Mean	62325	52048	3340	6915	9412	4377	5036
Blindman River	8	Minimum	15106	12492	672	1282	1704	1130	574
Blindman River		Maximum	130316	114476	7880	18536	22164	8741	14858
Blindman River		St. Dev.	41179	36414	2338	5881	7261	2794	4974
Buffalo Creek		Median	6221	5163	119	868	508	337	135
Buffalo Creek		Mean	7850	6463	454	930	760	495	266
Buffalo Creek	8	Minimum	4089	3107	48	88	262	158	95
Buffalo Creek		Maximum	17301	13528	2150	1618	1947	1263	684
Buffalo Creek		St. Dev.	4707	3732	714	466	641	407	244
Crowfoot Creek		Median	56051	40482	12545	3361	7808	4665	3095
Crowfoot Creek		Mean	74591	49584	18420	6746	10604	5635	4969
Crowfoot Creek	8	Minimum	14210	12596	821	790	2417	1142	958
Crowfoot Creek		Maximum	183072	123390	46110	15900	27395	11574	15821
Crowfoot Creek		St. Dev.	58216	35485	18777	6154	8563	3931	5308
Grande Prairie Creek		Median	15543	14055	1066	406	1391	601	719
Grande Prairie Creek		Mean	16993	13182	3230	509	1749	682	1067
Grande Prairie Creek	8	Minimum	3337	2717	461	98	187	100	87
Grande Prairie Creek		Maximum	39505	28347	10269	1313	4141	1701	3448
Grande Prairie Creek		St. Dev.	12170	9115	3739	396	1415	507	1124
Hines Creek		Median	17725	16999	128	701	2129	956	967
Hines Creek		Mean	15312	14513	152	644	1853	944	909
Hines Creek	8	Minimum	221	217	1	4	16	8	8
Hines Creek		Maximum	30415	28609	400	1532	3550	2084	1983
Hines Creek		St. Dev.	10807	10313	134	471	1355	742	698
Haynes Creek (M6)		Median	7648	4583	1325	820	1527	1464	177
Haynes Creek (M6)		Mean	7960	5102	1899	1072	1474	1477	169
Haynes Creek (M6)	7 (6)	Minimum	136	129	1	7	20	15	5
Haynes Creek (M6)		Maximum	14591	9302	5489	2387	2912	2635	276
Haynes Creek (M6)		St. Dev.	5708	3428	1906	996	1068	961	99
Kleskun Drain		Median	3750	3052	220	130	516	384	132
Kleskun Drain		Mean	3961	2824	849	274	479	330	149
Kleskun Drain	8	Minimum	115	92	6	4	6	5	1
Kleskun Drain		Maximum	9961	6937	2742	1269	905	620	379
Kleskun Drain		St. Dev.	3368	2232	1169	424	384	257	136
Meadow Creek		Median	1843	1708	89	71	225	44	197
Meadow Creek		Mean	3889	3353	414	120	523	66	456
Meadow Creek	8	Minimum	458	419	10	19	46	8	16
Meadow Creek		Maximum	11523	10073	1515	307	1605	156	1449
Meadow Creek		St. Dev.	4161	3531	599	113	587	60	530
New West Coulee		Median	15136	12738	1297	537	2048	903	1081
New West Coulee		Mean	17410	13485	3216	689	2064	994	1070
New West Coulee	8	Minimum	8929	8435	289	201	1370	701	600
New West Coulee		Maximum	30557	18880	12317	1542	2864	1561	1549
New West Coulee		St. Dev.	7775	3617	4334	505	551	303	352
Paddle River		Median	14114	11880	755	814	1683	811	742
Paddle River		Mean	17071	15249	910	903	2706	933	1773
Paddle River	8	Minimum	4717	4317	115	80	502	302	200
Paddle River		Maximum	32965	28571	2451	2145	6642	1858	4799
Paddle River		St. Dev.	11648	10464	837	762	2342	625	1845

Table A9.3. Cont. Summary statistics for mass transport (kg) data from 1999 to 2006.

AESA Watershed	Number of Years	Summary Statistic	Total Nitrogen	Organic-N	Nitrite-N + Nitrate-N	Ammonia-N	Total Phosphorus	Total Dissolved Phosphorus	Total Particulate Phosphorus
			TN	Org-N	NO_2^--N +NO_3^--N	NH_3-N	TP	TDP	TPP
Prairie Blood Coulee		Median	3496	2300	168	122	266	165	34
Prairie Blood Coulee		Mean	7920	5164	2544	208	566	345	221
Prairie Blood Coulee	8	Minimum	101	98	1	3	2	1	1
Prairie Blood Coulee		Maximum	28034	16256	11064	704	2118	1386	732
Prairie Blood Coulee		St. Dev.	10549	6395	4075	252	772	478	322
Ray Creek		Median	2165	1788	270	64	318	274	44
Ray Creek		Mean	3225	2425	610	185	456	349	107
Ray Creek	8	Minimum	328	294	20	14	32	27	5
Ray Creek		Maximum	9543	5585	3280	656	1116	872	347
Ray Creek		St. Dev.	3108	1959	1097	232	418	304	126
Renwick Creek		Median	2560	1892	374	106	600	499	58
Renwick Creek		Mean	2349	1529	655	162	447	375	72
Renwick Creek	8	Minimum	78	66	7	4	16	15	1
Renwick Creek		Maximum	5129	2918	1914	541	863	677	186
Renwick Creek		St. Dev.	2028	1217	754	185	349	289	70
Rose Creek		Median	59758	56794	968	2288	11004	1472	8492
Rose Creek		Mean	75697	72358	861	2470	17757	1670	16087
Rose Creek	8	Minimum	26161	25029	298	667	1797	660	1016
Rose Creek		Maximum	198646	193810	1646	4844	70576	4035	68516
Rose Creek		St. Dev.	56495	55271	447	1452	22158	1088	21973
Stretton Creek		Median	4140	2864	1192	171	587	521	80
Stretton Creek		Mean	4302	2873	1186	239	607	520	87
Stretton Creek	4	Minimum	1435	786	381	88	245	202	15
Stretton Creek		Maximum	7495	4977	1978	525	1010	837	173
Stretton Creek		St. Dev.	2495	1737	832	196	338	274	71
Strawberry Creek		Median	40420	32929	6483	3490	9976	1788	8946
Strawberry Creek		Mean	72920	55357	9568	7912	17568	3259	14310
Strawberry Creek	8	Minimum	6248	4707	1132	396	994	311	684
Strawberry Creek		Maximum	173376	145675	37637	24051	55229	8548	49604
Strawberry Creek		St. Dev.	66315	51851	11836	9355	18499	3137	16281
Threehills Creek		Median	10719	7121	1707	1522	1912	1459	288
Threehills Creek		Mean	11910	7778	2559	1558	2151	1729	422
Threehills Creek	8	Minimum	1097	917	62	118	187	153	34
Threehills Creek		Maximum	28427	16516	9583	3024	4599	3772	1097
Threehills Creek		St. Dev.	10125	6215	3132	1294	1857	1531	399
Tomahawk Creek		Median	10704	8662	1210	686	1188	344	856
Tomahawk Creek		Mean	11922	9552	1303	1061	1514	490	1024
Tomahawk Creek	8	Minimum	709	584	49	76	67	33	34
Tomahawk Creek		Maximum	23471	19076	2591	2606	3328	1386	2538
Tomahawk Creek		St. Dev.	7062	5827	764	855	1116	426	832
Trout Creek		Median	6967	6263	470	246	887	110	770
Trout Creek		Mean	16286	14220	1559	501	3505	170	3334
Trout Creek	8	Minimum	826	733	48	23	36	16	20
Trout Creek		Maximum	60668	55006	4980	1804	11831	549	11281
Trout Creek		St. Dev.	20770	18577	2031	624	4651	191	4469
Wabash Creek		Median	8814	5002	2375	956	750	341	251
Wabash Creek		Mean	11563	6528	2568	2402	1717	1231	486
Wabash Creek	8	Minimum	104	87	5	12	20	5	15
Wabash Creek		Maximum	34441	18864	6282	11098	7282	5628	1654
Wabash Creek		St. Dev.	13025	7162	2679	3815	2557	1986	591
Willow Creek		Median	2686	2433	178	64	415	40	375
Willow Creek		Mean	13355	12792	275	264	7411	97	7314
Willow Creek	8	Minimum	498	358	95	44	43	11	31
Willow Creek		Maximum	79955	77740	676	1382	54191	448	53743
Willow Creek		St. Dev.	27132	26441	226	460	18919	148	18776

Table A9.4. Summary statistics for export coefficient (kg ha^{-1} yr^{-1}) data from 1999 to 2006.

AESA Watershed	Number of Years	Summary Statistic	Total Nitrogen	Organic-N	Nitrite-N + Nitrate-N	Ammonia-N	Total Phosphorus	Total Dissolved Phosphorus	Total Particulate Phosphorus
			TN	Org-N	NO_2^--N +NO_3^--N	NH_3-N	TP	TDP	TPP
Blindman River		Median	1.41205	1.072	0.09565	0.1437	0.2136	0.13015	0.08345
Blindman River		Mean	1.7655625	1.474	0.094625	0.1959125	0.2666375	0.123975	0.14265
Blindman River	8	Minimum	0.4279	0.354	0.019	0.0363	0.0483	0.032	0.0163
Blindman River		Maximum	3.6917	3.243	0.2232	0.5251	0.6279	0.2476	0.4209
Blindman River		St.Dev.	1.1665553	1.032	0.066243679	0.1666157	0.20567421	0.07917106	0.14091209
Buffalo Creek		Median	0.42315	0.351	0.0081	0.05905	0.0345	0.02295	0.00925
Buffalo Creek		Mean	0.5340375	0.44	0.03085	0.0633	0.051725	0.03365	0.0181
Buffalo Creek	8	Minimum	0.2782	0.211	0.0033	0.006	0.0178	0.0107	0.0065
Buffalo Creek		Maximum	1.177	0.92	0.1462	0.1101	0.1324	0.0859	0.0466
Buffalo Creek		St.Dev.	0.3202164	0.254	0.048564655	0.03169367	0.04360664	0.02767867	0.01659845
Grande Prairie Creek		Median	1.11025	1.004	0.07615	0.029	0.0994	0.0429	0.05135
Grande Prairie Creek		Mean	1.213775	0.942	0.2307375	0.0363375	0.1249	0.048675	0.076225
Grande Prairie Creek	8	Minimum	0.2383	0.194	0.033	0.007	0.0134	0.0071	0.0062
Grande Prairie Creek		Maximum	2.8218	2.025	0.7335	0.0938	0.2958	0.1215	0.2463
Grande Prairie Creek		St.Dev.	0.869263	0.651	0.26705352	0.02831107	0.10107321	0.03621526	0.08027677
Hines Creek		Median	0.4739	0.455	0.0034	0.01875	0.05695	0.02555	0.02585
Hines Creek		Mean	0.4093875	0.388	0.0040625	0.0172125	0.04955	0.0252375	0.0243125
Hines Creek	8	Minimum	0.0059	0.006	0	0.0001	0.0004	0.0002	0.0002
Hines Creek		Maximum	0.8132	0.765	0.0107	0.041	0.0949	0.0557	0.053
Hines Creek		St.Dev.	0.2889273	0.276	0.003577284	0.01260334	0.0362256	0.01983719	0.0186711
Haynes Creek (M6)		Median	0.4607	0.276	0.0798	0.0494	0.092	0.0882	0.01065
Haynes Creek (M6)		Mean	0.4795429	0.307	0.114385714	0.06458571	0.08878571	0.08896667	0.01016667
Haynes Creek (M6)	7 (6)	Minimum	0.0082	0.008	0	0.0004	0.0012	0.0009	0.0003
Haynes Creek (M6)		Maximum	0.879	0.56	0.3307	0.1438	0.1754	0.1588	0.0167
Haynes Creek (M6)		St.Dev.	0.3438503	0.207	0.114808092	0.05998648	0.06431421	0.05791648	0.00601121
Kleskun Drain		Median	1.17195	0.954	0.06885	0.04065	0.1612	0.11985	0.0413
Kleskun Drain		Mean	1.2377375	0.883	0.26525	0.08555	0.1496	0.103125	0.04645
Kleskun Drain	8	Minimum	0.0358	0.029	0.002	0.0012	0.0019	0.0016	0.0003
Kleskun Drain		Maximum	3.1127	2.168	0.8568	0.3966	0.2829	0.1938	0.1183
Kleskun Drain		St.Dev.	1.0524229	0.698	0.365210135	0.13254828	0.11989379	0.0803644	0.04244055
Meadow Creek		Median	0.14175	0.131	0.0068	0.00545	0.0173	0.0034	0.01515
Meadow Creek		Mean	0.2991625	0.258	0.0317875	0.009225	0.0402125	0.0051125	0.035075
Meadow Creek	8	Minimum	0.0352	0.032	0.0007	0.0015	0.0035	0.0006	0.0012
Meadow Creek		Maximum	0.8864	0.775	0.1165	0.0236	0.1235	0.012	0.1114
Meadow Creek		St.Dev.	0.3200908	0.272	0.046073619	0.00868262	0.0451379	0.00459019	0.0407339
Paddle River		Median	0.5579	0.47	0.02985	0.03215	0.0665	0.03205	0.0293
Paddle River		Mean	0.6747625	0.603	0.0359875	0.0356625	0.106925	0.036875	0.070075
Paddle River	8	Minimum	0.1864	0.171	0.0045	0.0031	0.0198	0.0119	0.0079
Paddle River		Maximum	1.303	1.129	0.0969	0.0848	0.2625	0.0734	0.1897
Paddle River		St.Dev.	0.4603851	0.414	0.033112209	0.03012512	0.0925809	0.02472395	0.07294416
Prairie Blood Coulee		Median	0.15465	0.102	0.0074	0.0054	0.01175	0.0073	0.00155
Prairie Blood Coulee		Mean	0.35045	0.229	0.1125375	0.009175	0.025025	0.01525	0.0098125
Prairie Blood Coulee	8	Minimum	0.0045	0.004	0	0.0001	0.0001	0	0.0001
Prairie Blood Coulee		Maximum	1.2404	0.719	0.4895	0.0311	0.0937	0.0613	0.0324
Prairie Blood Coulee		St.Dev.	0.4667616	0.283	0.180303585	0.01115805	0.03416961	0.02115771	0.01421342

Table A9.4. Cont. Summary statistics for export coefficient (kg ha^{-1} yr^{-1}) data from 1999 to 2006.

AESA Watershed	Number of Years	Summary Statistic	Total Nitrogen	Organic-N	Nitrite-N + Nitrate-N	Ammonia-N	Total Phosphorus	Total Dissolved Phosphorus	Total Particulate Phosphorus
			TN	Org-N	NO_2^--N +NO_3^--N	NH_3-N	TP	TDP	TPP
Ray Creek		Median	0.48765	0.403	0.0608	0.01435	0.0717	0.06185	0.00985
Ray Creek		Mean	0.7263125	0.546	0.1374375	0.041675	0.1027	0.078575	0.024125
Ray Creek	8	Minimum	0.0738	0.066	0.0044	0.0032	0.0073	0.0061	0.0012
Ray Creek		Maximum	2.1493	1.258	0.7387	0.1478	0.2513	0.1964	0.0782
Ray Creek		St.Dev.	0.6999555	0.441	0.247006703	0.05226263	0.09419818	0.06835748	0.02842332
Renwick Creek		Median	0.44065	0.326	0.06435	0.01825	0.1032	0.08585	0.01
Renwick Creek		Mean	0.4043375	0.263	0.1127	0.0278875	0.076875	0.0644625	0.012425
Renwick Creek	8	Minimum	0.0134	0.011	0.0013	0.0007	0.0028	0.0026	0.0002
Renwick Creek		Maximum	0.8827	0.502	0.3294	0.0931	0.1486	0.1166	0.032
Renwick Creek		St.Dev.	0.3489985	0.21	0.129805833	0.03178578	0.06001121	0.04969438	0.01203919
Rose Creek		Median	1.06905	1.016	0.0173	0.04095	0.19685	0.02635	0.1519
Rose Creek		Mean	1.35415	1.294	0.0154	0.0442	0.3176625	0.029875	0.287775
Rose Creek	8	Minimum	0.468	0.448	0.0053	0.0119	0.0322	0.0118	0.0182
Rose Creek		Maximum	3.5536	3.467	0.0294	0.0867	1.2625	0.0722	1.2257
Rose Creek		St.Dev.	1.0106354	0.989	0.008005891	0.02599703	0.39637205	0.01947018	0.39307562
Stretton Creek		Median	0.7353	0.509	0.21165	0.03035	0.10425	0.09255	0.01415
Stretton Creek		Mean	0.7642	0.51	0.210575	0.0424	0.107875	0.0924	0.015425
Stretton Creek	4	Minimum	0.255	0.14	0.0677	0.0157	0.0436	0.0358	0.0027
Stretton Creek		Maximum	1.3312	0.884	0.3513	0.0932	0.1794	0.1487	0.0307
Stretton Creek		St.Dev.	0.4431949	0.309	0.147872859	0.03476272	0.06009933	0.04861927	0.01267159
Strawberry Creek		Median	0.68625	0.559	0.1101	0.05925	0.16935	0.03035	0.1519
Strawberry Creek		Mean	1.2380375	0.94	0.1624625	0.134325	0.2982875	0.055325	0.24295
Strawberry Creek	8	Minimum	0.1061	0.08	0.0192	0.0067	0.0169	0.0053	0.0116
Strawberry Creek		Maximum	2.9436	2.473	0.639	0.4083	0.9377	0.1451	0.8422
Strawberry Creek		St.Dev.	1.1258738	0.88	0.200943353	0.15881872	0.31408348	0.05325665	0.27641824
Threehills Creek		Median	0.7768	0.516	0.1237	0.1103	0.13855	0.1057	0.0209
Threehills Creek		Mean	0.8630625	0.564	0.185425	0.1129	0.15585	0.1253125	0.030575
Threehills Creek	8	Minimum	0.0795	0.066	0.0045	0.0085	0.0135	0.0111	0.0025
Threehills Creek		Maximum	2.0599	1.197	0.6944	0.2191	0.3332	0.2733	0.0795
Threehills Creek		St.Dev.	0.7336526	0.45	0.226944359	0.09379516	0.13453532	0.11089983	0.02891499
Tomahawk Creek		Median	1.12315	0.909	0.127	0.07195	0.12465	0.0361	0.08985
Tomahawk Creek		Mean	1.250975	1.002	0.1367375	0.1113375	0.15885	0.051425	0.10745
Tomahawk Creek	8	Minimum	0.0744	0.061	0.0052	0.0079	0.007	0.0035	0.0035
Tomahawk Creek		Maximum	2.4629	2.002	0.2719	0.2735	0.3492	0.1454	0.2663
Tomahawk Creek		St.Dev.	0.7410822	0.611	0.080131961	0.08973375	0.11713999	0.04469262	0.08732038
Trout Creek		Median	0.158	0.142	0.01065	0.00555	0.0201	0.0025	0.01745
Trout Creek		Mean	0.3693	0.322	0.03535	0.01135	0.079475	0.0038875	0.0756
Trout Creek	8	Minimum	0.0187	0.017	0.0011	0.0005	0.0008	0.0004	0.0004
Trout Creek		Maximum	1.3757	1.247	0.1129	0.0409	0.2683	0.0125	0.2558
Trout Creek		St.Dev.	0.4709977	0.421	0.046050438	0.0141591	0.10548945	0.0043423	0.10134766
Wabash Creek		Median	0.25625	0.145	0.06905	0.0278	0.0218	0.00995	0.0073
Wabash Creek		Mean	0.3361375	0.19	0.074625	0.0698375	0.049925	0.0357875	0.0141375
Wabash Creek	8	Minimum	0.003	0.003	0.0001	0.0004	0.0006	0.0001	0.0004
Wabash Creek		Maximum	1.0012	0.548	0.1826	0.3226	0.2117	0.1636	0.0481
Wabash Creek		St.Dev.	0.3786343	0.208	0.077885516	0.11090754	0.07434665	0.05772497	0.01719717
Willow Creek		Median	0.41125	0.373	0.02735	0.00975	0.06355	0.00605	0.05745
Willow Creek		Mean	2.0451375	1.959	0.0421875	0.0404875	1.1349375	0.01485	1.1200625
Willow Creek	8	Minimum	0.0763	0.055	0.0145	0.0068	0.0065	0.0017	0.0048
Willow Creek		Maximum	12.2442	11.905	0.1035	0.2117	8.2988	0.0686	8.2302
Willow Creek		St.Dev.	4.1550105	4.049	0.034656454	0.07048324	2.89720554	0.02267113	2.87528425

APPENDIX 10: FECAL BACTERIA SUMMARY STATISTICS

Table A10.1. Summary statistics for all fecal bacteria samples collected from AESA streams from 1999 to 2006.

	E.coli						Fecal coliforms						% fecal coliforms comprised of *E.coli*	
	N of cases	% N above MDL	Median	Geometric Mean	Arithmetic Mean	Maximum	N of cases	% N above MDL	Median	Geometric Mean	Arithmetic Mean	Maximum	Mean	Median
Low Intensity														
Hines Creek	111	59	10	16	36	300	113	71	20	21	46	470	81	100
Paddle Creek	151	75	20	26	82	1400	153	86	40	38	123	4000	75	100
Prairie Blood Coulee	110	63	12	18	66	1000	110	73	15	23	89	1400	83	98
Rose Creek	179	84	80	60	187	4900	180	88	110	78	224	4900	80	92
Willow Creek	163	47	5	11	25	470	163	53	8	14	32	470	86	100
Moderate Intensity														
Blindman Creek	177	82	60	64	350	10000	178	89	90	84	429	13000	81	100
Grande Prairie Creek	100	77	60	47	162	2800	101	85	80	59	183	2900	82	100
Kleskun Drain	70	46	14	21	385	17000	71	58	20	27	537	20000	84	100
Meadow Creek	150	97	410	448	1264	10000	150	99	520	660	2221	26000	73	76
Tomahawk Creek	152	74	38	42	638	40000	154	82	60	57	699	42000	79	100
Trout Creek	165	86	100	89	370	10000	165	93	200	143	563	24000	71	82
High Intensity														
Buffalo Creek	116	53	10	21	94	1000	118	64	20	26	114	1600	84	100
Haynes Creek (M6)	92	54	10	15	51	970	92	68	10	18	60	1100	87	100
Ray Creek	134	57	24	36	288	7900	135	70	50	48	359	8900	82	100
Renwick Creek	102	45	10	19	133	2500	103	60	10	23	166	2900	86	100
Stretton Creek	45	40	5	11	20	160	47	55	10	12	23	180	87	100
Strawberry Creek	130	79	40	51	575	21000	131	89	60	66	659	27000	78	100
Threehills Creek	136	55	20	22	101	3300	136	76	20	29	130	4700	82	100
Wabash Creek	96	57	10	16	37	310	98	67	20	20	52	400	80	100
Irrigation														
Battersea Drain	150	76	45	44	352	10000	151	81	70	58	1028	60000	78	84
Crowfoot Creek	152	81	68	57	225	9200	153	89	92	80	281	11000	76	83
Drain S6	132	69	32	39	322	10000	132	75	49	54	486	16000	80	87
New West Coulee	142	87	75	66	227	6500	142	89	110	87	275	6500	81	87

Table A10.2. Summary statistics for ambient fecal bacteria data collected from 23 streams from 1999 to 2006.

	N	N above MDL (%)	Median [z]	Geometric Mean [z]	Arithmetic Mean[y]	Min	25^{th} Quartile	75^{th} Quartile	Maximum
			(CFU·100 mL^{-1})						
Fecal coliforms [y]	2976	2342 (79%)	50	50	417	<MDL	10	190	*60 000*
E. coli	2955	2064 (69%)	32	38	283	<MDL	<MDL	150	40 000

[z] *Calculations include censored data (1/2 MDL = 5* CFU·100 mL^{-1}*)*

[y] *Does not include fecal coliform data point from Strawberry Creek on June 12, 2000 (120 000 CFU 100* mL^{-1} *)*

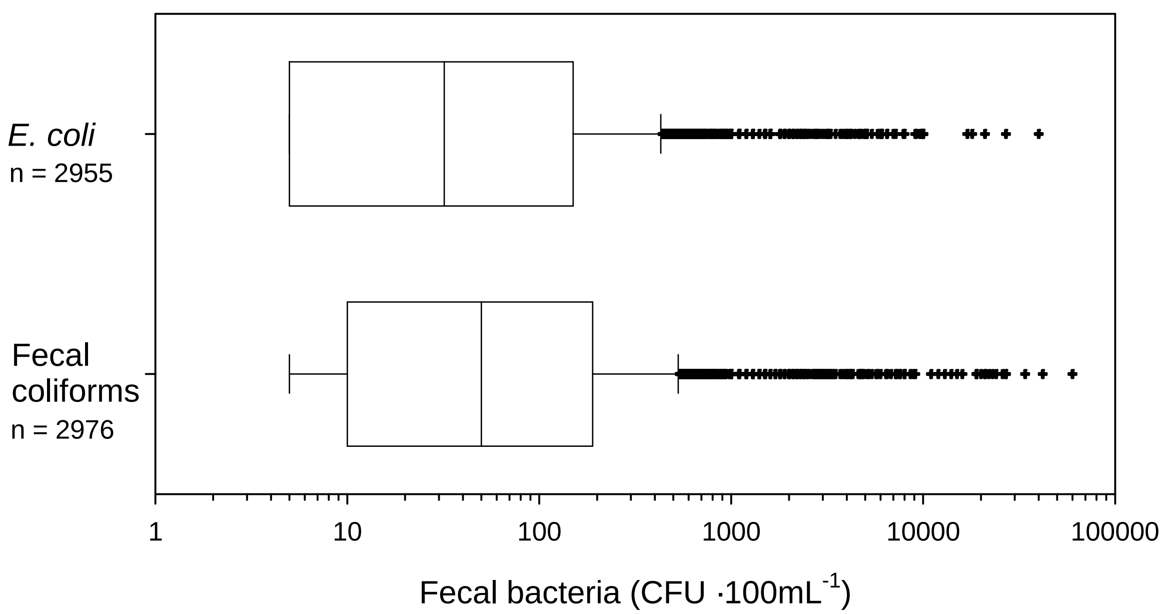


Figure A10.1. Box plots showing all ambient fecal coliform and *E. coli* data collected from 23 AESA streams from 1999 to 2006. Dotted lines: Fecal coliform (ASWQG for recreation: 100 CFU·100 mL^{-1}) and *E. coli* (ASWQG for irrigation: 200 CFU·100 mL^{-1}).

Table A10.3a. *E. coli* summary statistics for each stream, 1999 to 2006.

	N of cases	% N above MDL	% N LMDL	25th percentile	Median	75th Percentile	95th Percentile	Maximum	IQR	Geometric Mean	Arithmetic Mean
Battersea Drain	150	76	24	8	45	118	736	10000		44	352
Blindman River	177	82	18	20	60	210	972	10000	190	64	350
Buffalo Creek	116	53	47	5	10	53	490	1000	48	21	94
Crowfoot Creek	152	81	19	15	68	203	554	9200	188	57	225
Drain S6	132	69	31	5	32	193	858	10000	188	39	322
Grande Prairie Creek	100	77	23	10	60	145	533	2800	135	47	162
Hines Creek	111	59	41	5	10	40	153	300	35	16	36
Haynes Creek M6	92	54	46	5	10	37	210	970	32	15	51
Kleskun Drain	70	46	54	5	14	50	833	17000	45	21	385
Meadow Creek	150	97	3	220	410	1200	5620	10000	980	448	1264
New West Coulee	142	87	13	22	75	198	568	6500	176	66	227
Paddle Creek	151	75	25	10	20	70	435	1400	60	26	82
Prairie Blood Coulee	110	63	37	5	12	49	320	1000	44	18	66
Ray Creek	134	57	43	5	24	195	1000	7900	190	36	288
Renwick Creek	102	45	55	5	10	45	714	2500	40	19	133
Rose Creek	179	84	16	20	80	210	526	4900	190	60	187
Strawberry Creek	130	79	21	10	40	160	1820	21000	150	51	575
Stretton Creek	45	40	60	5	5	20	98	160	15	11	20
Threehills Creek	136	55	45	5	20	70	258	3300	65	22	101
Tomahawk Creek	152	74	26	9	38	140	1225	40000	131	42	638
Trout Creek	165	86	14	21	100	380	1180	10000	359	89	370
Wabash Creek	96	57	43	5	10	33	173	310	28	16	37
Willow Creek	163	47	53	5	5	22	81	470	17	11	25

Table A10.3b. Fecal coliform summary statistics for all AESA watersheds, 1999 to 2006.

	N of cases	% N above MDL	% N LMDL	25th percentile	Median	75th Percentile	95th Percentile	Maximum	IQR	Geometric Mean	Arithmetic Mean
Battersea Drain	151	81	19	12	70	165	900	60000		1028	58
Blindman River	178	89	11	30	90	250	1415	13000	220	84	429
Buffalo Creek	118	64	36	5	20	90	590	1600	85	26	114
Crowfoot Creek	153	89	11	28	92	250	832	11000	222	80	281
Drain S6	132	75	25	6	49	243	1770	16000	237	54	486
Grande Prairie Creek	101	85	15	20	80	170	590	2900	150	59	183
Haynes Creek M6	92	68	32	5	10	40	269	1100	35	18	60
Hines Creek	113	71	29	5	20	50	182	470	45	21	46
Kleskun Drain	71	58	42	5	20	72	1700	20000	67	27	537
Meadow Creek	150	99	1	280	520	1900	8875	26000	1620	660	2221
New West Coulee	142	89	11	38	110	250	861	6500	212	87	275
Paddle Creek	153	86	14	10	40	90	432	4000	80	38	123
Prairie Blood Coulee	110	73	27	5	15	71	430	1400	66	23	89
Ray Creek	135	70	30	5	50	240	1790	8900	235	48	359
Renwick Creek	103	60	40	5	10	86	891	2900	81	23	166
Rose Creek	180	88	12	20	110	270	702	4900	250	78	224
Strawberry Creek	131	89	11	20	60	198	2335	120000	178	69	1563
Stretton Creek	47	55	45	5	10	20	97	180	15	12	23
Threehills Creek	136	76	24	9	20	93	270	4700	84	29	130
Tomahawk Creek	154	82	18	10	60	168	1280	42000	158	57	699
Trout Creek	165	93	7	42	200	500	2000	24000	458	143	563
Wabash Creek	98	67	33	5	20	50	248	400	45	20	52
Willow Creek	163	53	47	5	8	35	120	470	30	14	32

Table A10.4. Number of fecal coliform samples and an average for each AESA watershed from 1999 to 2006.

Watershed	1999	2000	2001	2002	2003	2004	2005	2006	Average
Battersea Drain	15	18	16	15	20	25	21	21	19
Blindman Creek	34	29	17	15	13	19	32	19	22
Buffalo Creek	13	11	9	12	16	16	24	17	15
Crowfoot Creek	15	18	17	16	22	21	22	22	19
Drain S6	15	16	14	14	15	22	15	21	17
Grande Prairie Creek	11	20	13	9	11	13	15	9	13
Haynes Creek M6	21	21	9	7	13		11	10	13
Hines Creek	7	16	12	14	12	22	18	22	14
Kleskun Drain	7	13	7	10	8	10	11	5	9
Meadow Creek	18	13	11	20	21	22	22	23	19
New West Coulee	14	14	14	15	19	25	20	21	18
Paddle River	23	26	15	14	13	12	33	19	19
Prairie Blood Coulee	13	9	9	17	8	19	17	18	14
Ray Creek	30	23	12	7	14	15	19	15	17
Renwick Creek	25	11	5	5	11	13	18	15	13
Rose Creek	31	29	17	14	15	18	32	24	23
Strawberry Creek	26	30	11	11	11	17	16	10	17
Stretton Creek	11	11				6	10	9	9
Threehills Creek	30	21	12	7	14	16	18	18	17
Tomahawk Creek	23	26	14	11	15	17	31	17	19
Trout Creek	22	19	17	19	21	23	21	23	21
Wabash Creek	9	18	4	8	8	13	23	15	12
Willow Creek	25	23	21	19	16	20	21	19	21

APPENDIX 11: MEDIAN ANNUAL BACTERIA CONCENTRATIONS BY AGRICULTURAL INTENSITY

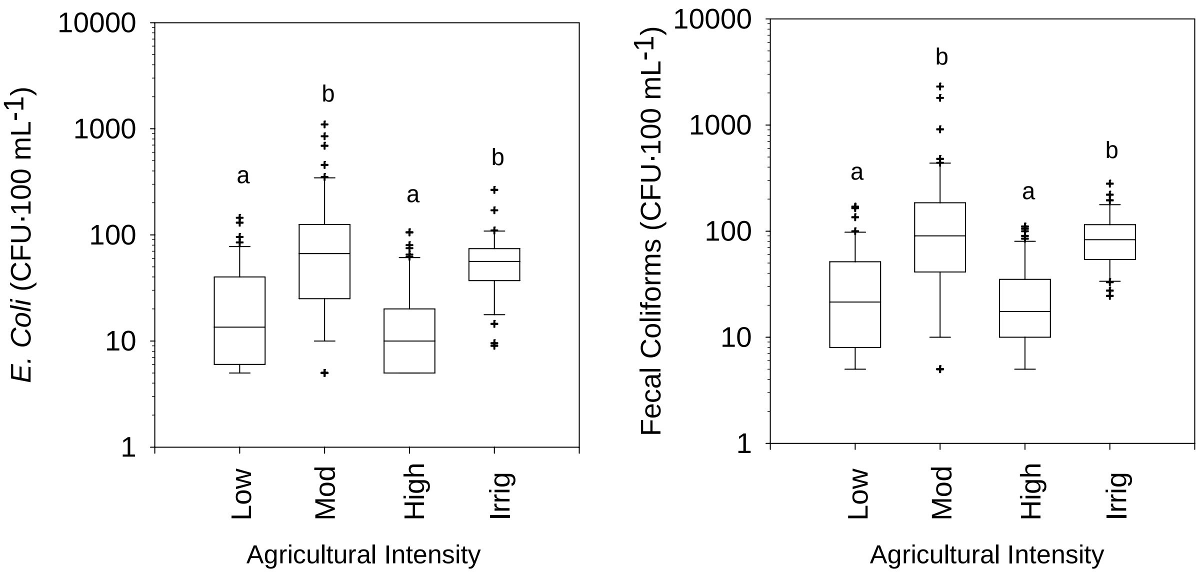


Figure A11.2. Box plots of median annual fecal coliform and *E. coli* for 23 AESA streams grouped by agricultural intensity. Groups with the same letter are not significantly different (Kruskal Wallis ANOVA and Mann-Whitney U test, $p<0.05$)

APPENDIX 12: FECAL BACTERIA LOADS AND EXPORT COEFFICIENTS, 1999 TO 2006

Table A12.1a. Annual *E. coli* load and export values for all AESA streams from 1999 through 2006. Irrigated watersheds do not have exports as the effective drainage area could not be determined.

	1999		2000		2001		2002		2003		2004		2005		2006	
	Load (CFU.yr^{-1})	Export Coeff. (CFU ha^{-1} yr^{-1})	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.
Low Agricultural Intensity																
Hines Creek	1.1E+04	0.3	1.3E+07	349.5	1.2E+07	323.0	5.8E+06	154.9	2.7E+07	726.1	1.0E+06	27.2	2.4E+06	63.3	1.2E+05	3.3
Paddle Creek	1.7E+07	678.6	1.4E+07	560.6	8.9E+07	3526.6	1.2E+07	470.2	3.2E+06	125.9	1.8E+06	69.9	3.5E+07	1369.8	3.8E+06	150.0
Prairie Blood Coulee	4.8E+04	2.1	3.2E+04	1.4	2.1E+05	9.4	1.3E+06	58.0	2.2E+05	9.6	2.0E+05	8.9	7.2E+07	3187.7	5.2E+06	231.3
Rose Creek	5.9E+08	10605.0	2.1E+08	3701.6	7.2E+07	1294.4	2.6E+07	471.2	3.3E+07	595.7	9.8E+08	17454.0	9.4E+07	1680.3	6.7E+07	1196.9
Willow Creek	1.1E+06	169.5	7.0E+05	107.4	8.0E+05	122.2	1.9E+06	286.1	1.7E+06	258.6	7.7E+06	1183.1	1.7E+07	2656.0	3.7E+06	566.5
Mean	*1.2E+08*	*2291.1*	*4.7E+07*	*944.1*	*3.5E+07*	*1055.1*	*9.4E+06*	*288.1*	*1.3E+07*	*343.2*	*2.0E+08*	*3748.6*	*4.4E+07*	*1791.4*	*1.6E+07*	*429.6*
Median	*1.1E+06*	*169.5*	*1.3E+07*	*349.5*	*1.2E+07*	*323.0*	*5.8E+06*	*286.1*	*3.2E+06*	*258.6*	*1.8E+06*	*69.9*	*3.5E+07*	*1680.3*	*3.8E+06*	*231.3*
Moderate Agricultural Intensity																
Blindman Creek	1.9E+09	54841.0	1.6E+09	46463.5	1.4E+08	3895.0	1.7E+07	475.2	2.3E+07	644.1	9.7E+06	275.8	6.1E+07	1734.8	3.0E+07	857.4
Grande Prairie Creek	2.3E+06	162.4	6.2E+06	444.4	2.1E+07	1488.2	2.1E+07	1482.3	1.3E+07	899.7	6.1E+07	4331.5	2.4E+07	1729.7	3.1E+06	223.1
Kleskun Drain	3.3E+05	102.4	1.1E+04	3.4	1.3E+07	3988.4	4.2E+06	1311.9	3.4E+06	1051.1	2.9E+06	913.7	2.0E+05	61.2	9.2E+03	2.9
Meadow Creek	1.3E+07	1005.3	5.2E+06	398.3	6.3E+06	483.6	4.1E+07	3147.3	1.4E+07	1083.2	7.7E+06	591.6	8.8E+07	6744.8	1.3E+08	10225.0
Tomahawk Creek	3.6E+07	3737.6	1.7E+07	1821.2	2.4E+07	2547.9	3.0E+06	312.7	3.8E+06	398.5	2.6E+07	2734.5	1.1E+07	1106.0	3.1E+06	321.2
Trout Creek	1.0E+07	228.7	2.2E+06	51.0	7.4E+06	168.3	1.3E+08	2880.0	9.1E+07	2072.2	3.6E+07	824.5	4.8E+08	10977.1	4.9E+08	11025.8
Mean	*3.3E+08*	*10012.9*	*2.8E+08*	*8197.0*	*3.5E+07*	*2095.2*	*3.5E+07*	*1601.6*	*2.5E+07*	*1024.8*	*2.4E+07*	*1611.9*	*1.1E+08*	*3725.6*	*1.1E+08*	*3775.9*
Median	*1.2E+07*	*617.0*	*5.7E+06*	*421.4*	*1.7E+07*	*2018.0*	*1.9E+07*	*1397.1*	*1.3E+07*	*975.4*	*1.8E+07*	*869.1*	*4.3E+07*	*1732.2*	*1.7E+07*	*589.3*
High Agricultural Intensity																
Buffalo Creek	7.4E+05	50.6	4.0E+05	27.5	1.0E+06	67.8	3.7E+05	25.2	1.8E+06	121.9	3.8E+06	255.7	2.6E+06	176.2	8.7E+06	592.6
Haynes Creek (M6)	1.1E+06	64.3	2.4E+05	14.7	1.4E+04	0.8	1.1E+05	6.6	3.2E+06	195.0	-	-	6.1E+05	36.9	1.3E+06	78.7
Ray Creek	1.6E+07	3523.0	3.7E+05	83.3	5.9E+05	133.3	7.2E+04	16.2	3.1E+05	69.8	6.5E+05	145.7	1.1E+07	2426.5	2.0E+06	439.9
Renwick Creek	1.3E+06	220.7	5.9E+03	1.0	2.0E+04	3.5	2.3E+03	0.4	1.4E+06	249.5	2.3E+05	38.9	1.9E+06	321.1	4.5E+05	76.9
Stretton Creek	5.0E+04	8.9	-	-	-	-	-	-	-	-	5.9E+04	10.6	3.3E+05	58.4	5.5E+05	97.2
Strawberry Creek	8.8E+07	1496.0	1.9E+09	31873.6	2.0E+09	34636.8	6.3E+06	107.1	7.6E+06	129.2	4.9E+06	83.0	8.2E+07	1389.1	5.1E+07	868.9
Threehills Creek	3.3E+06	239.8	3.7E+05	27.1	2.4E+05	17.0	8.5E+05	61.9	8.7E+06	633.9	6.8E+05	49.3	9.1E+06	658.4	2.5E+06	181.9
Wabash Creek	4.6E+03	0.1	2.1E+05	6.0	1.2E+05	3.6	9.8E+05	28.4	9.7E+05	28.2	1.6E+06	47.7	1.0E+07	296.4	1.2E+05	3.5
Mean	*1.4E+07*	*700.4*	*2.7E+08*	*4576.2*	*2.9E+08*	*4980.4*	*1.2E+06*	*35.1*	*3.4E+06*	*203.9*	*1.7E+06*	*90.1*	*1.5E+07*	*670.4*	*8.3E+06*	*292.5*
Median	*1.2E+06*	*142.5*	*3.7E+05*	*27.1*	*2.4E+05*	*17.0*	*3.7E+05*	*25.2*	*1.8E+06*	*129.2*	*6.8E+05*	*49.3*	*5.8E+06*	*308.7*	*1.6E+06*	*139.6*
Irrigation Agriculture																
Battersea Drain	2.2E+07	-	7.3E+06	-	7.2E+06	-	3.1E+08	-	1.5E+07	-	1.5E+07	-	2.1E+08	-	1.0E+08	-
Crowfoot Creek	2.5E+07	-	4.7E+07	-	3.6E+07	-	2.9E+07	-	3.5E+07	-	7.5E+07	-	6.1E+08	-	6.1E+07	-
Drain S6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
New West Coulee	2.2E+07	-	2.9E+07	-	3.0E+07	-	1.3E+08	-	8.3E+07	-	3.4E+07	-	5.2E+07	-	8.6E+07	-
Mean	*2.3E+07*	-	*2.8E+07*	-	*2.5E+07*	-	*1.6E+08*	-	*4.4E+07*	-	*4.1E+07*	-	*2.9E+08*	-	*8.2E+07*	-
Median	*2.2E+07*	-	*2.9E+07*	-	*3.0E+07*	-	*1.3E+08*	-	*3.5E+07*	-	*3.4E+07*	-	*2.1E+08*	-	*8.6E+07*	-

Table A12.1b. Annual fecal coliform load and export values for all AESA streams from 1999 through 2006. Irrigated watersheds do not have exports as the effective drainage area could not be determined.

	1999		2000		2001		2002		2003		2004		2005		2006	
	Load (CFU.yr^{-1})	Export Coeff. (CFU ha^{-1} yr^{-1})	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.	Load	Export Coeff.
Low Agricultural Intensity																
Hines Creek	1.1E+04	0.29	1.4E+07	363.9	2.3E+07	617.8	6.7E+06	179.3	3.4E+07	901.5	1.6E+06	42.9	4.8E+06	127.3	1.5E+05	4.1
Paddle Creek	1.8E+07	713.2	1.6E+07	632.2	9.2E+07	3644.3	2.6E+07	1033.2	5.0E+06	196.1	3.6E+06	142.2	5.0E+07	1985.5	4.3E+06	169.0
Prairie Blood Coulee	6.1E+04	2.7	3.5E+04	1.5	3.3E+05	14.4	1.7E+06	74.1	2.4E+05	10.6	2.6E+05	11.5	7.2E+07	3193.8	7.7E+06	339.5
Rose Creek	6.5E+08	11716.2	3.4E+08	6113.1	8.1E+07	1442.0	2.8E+07	508.6	3.8E+07	674.9	9.9E+08	17669.2	1.3E+08	2295.8	7.6E+07	1360.5
Willow Creek	2.7E+06	411.5	1.1E+06	160.9	1.2E+06	176.5	2.4E+06	366.1	1.7E+06	258.6	1.1E+07	1665.4	2.1E+07	3234.7	4.6E+06	700.9
Mean	*1.4E+08*	*2568.8*	*7.4E+07*	*1454.4*	*3.9E+07*	*1179.0*	*1.3E+07*	*432.3*	*1.6E+07*	*408.3*	*2.0E+08*	*3906.2*	*5.5E+07*	*2167.4*	*1.9E+07*	*514.8*
Median	*2.7E+06*	*411.5*	*1.4E+07*	*363.9*	*2.3E+07*	*617.8*	*6.7E+06*	*366.1*	*5.0E+06*	*258.6*	*3.6E+06*	*142.2*	*5.0E+07*	*2295.8*	*4.6E+06*	*339.5*
Moderate Agricultural Intensity																
Blindman Creek	2.4E+09	68952.7	1.9E+09	53378.4	2.8E+08	7998.0	1.7E+07	486.1	2.7E+07	756.9	1.5E+07	418.3	9.1E+07	2570.2	4.1E+07	1151.9
Grande Prairie Creek	3.1E+06	221.4	6.3E+06	448.1	2.3E+07	1643.2	2.1E+07	1526.1	1.3E+07	933.7	6.4E+07	4560.8	2.9E+07	2056.2	3.1E+06	223.1
Kleskun Drain	3.9E+05	123.2	2.7E+05	85.8	1.3E+07	4001.4	4.4E+06	1359.7	3.5E+06	1089.9	3.2E+06	995.6	4.9E+05	153.3	9.2E+03	2.9
Meadow Creek	2.7E+07	2097.6	9.7E+06	747.7	7.1E+06	549.9	4.2E+07	3236.4	1.9E+07	1464.1	8.6E+06	660.7	1.9E+08	14770.7	2.4E+08	18660.9
Tomahawk Creek	3.9E+07	4087.6	2.3E+07	2396.5	2.4E+07	2557.7	3.3E+06	346.5	4.7E+06	498.4	3.5E+07	3642.5	1.5E+07	1597.3	3.3E+06	346.9
Trout Creek	3.0E+07	675.7	5.9E+06	132.7	1.6E+07	354.3	1.5E+08	3360.4	1.1E+08	2466.4	4.5E+07	1015.3	5.2E+08	11869.7	1.0E+09	22849.7
Mean	*4.2E+08*	*12693.0*	*3.2E+08*	*9531.5*	*6.1E+07*	*2850.7*	*3.9E+07*	*1719.2*	*2.9E+07*	*1201.6*	*2.8E+07*	*1882.2*	*1.4E+08*	*5502.9*	*2.2E+08*	*7205.9*
Median	*2.9E+07*	*1386.7*	*8.0E+06*	*597.9*	*1.9E+07*	*2100.4*	*1.9E+07*	*1442.9*	*1.6E+07*	*1011.8*	*2.5E+07*	*1005.5*	*6.0E+07*	*2313.2*	*2.2E+07*	*749.4*
High Agricultural Intensity																
Buffalo Creek	8.6E+05	58.5	4.1E+05	28.2	1.1E+06	73.0	4.2E+05	28.6	1.9E+06	131.2	4.5E+06	307.7	3.7E+06	253.5	8.8E+06	600.8
Haynes Creek (M6)	1.5E+06	90.0	2.7E+05	16.0	1.6E+04	1.0	8.3E+04	5.0	3.8E+06	226.8	-	-	8.6E+05	51.8	1.3E+06	79.3
Ray Creek	2.1E+07	4815.7	4.4E+05	99.5	7.5E+05	169.0	1.3E+05	28.2	3.2E+05	72.5	6.7E+05	149.9	1.5E+07	3305.9	2.7E+06	602.8
Renwick Creek	1.7E+06	301.0	6.4E+03	1.1	2.0E+04	3.5	2.5E+03	0.4	1.6E+06	267.8	2.6E+05	45.4	2.4E+06	411.0	7.9E+05	135.9
Stretton Creek	7.1E+04	12.6	-	-	-	-	-	-	-	-	6.0E+04	10.6	3.3E+05	58.7	6.4E+05	113.1
Strawberry Creek	1.1E+08	1821.9	6.3E+09	107749.2	2.5E+09	42179.1	9.7E+06	165.2	9.5E+06	160.5	6.0E+06	102.5	9.6E+07	1626.1	5.7E+07	963.5
Threehills Creek	4.0E+06	293.4	4.4E+05	31.8	2.6E+05	18.7	8.5E+05	61.9	9.3E+06	675.6	8.3E+05	60.0	1.5E+07	1060.8	2.8E+06	204.3
Wabash Creek	5.7E+03	0.2	2.7E+05	7.7	1.9E+05	5.5	1.2E+06	36.1	9.8E+05	28.4	3.0E+06	86.3	1.1E+07	326.8	1.4E+05	4.0
Mean	*1.7E+07*	*924.2*	*9.1E+08*	*15419.1*	*3.6E+08*	*6064.2*	*1.8E+06*	*46.5*	*3.9E+06*	*223.3*	*2.2E+06*	*108.9*	*1.8E+07*	*886.8*	*9.2E+06*	*338.0*
Median	*1.6E+06*	*191.7*	*4.1E+05*	*28.2*	*2.6E+05*	*18.7*	*4.2E+05*	*28.6*	*1.9E+06*	*160.5*	*8.3E+05*	*86.3*	*7.5E+06*	*368.9*	*2.0E+06*	*170.1*
Irrigation Agriculture																
Battersea Drain	4.8E+07	-	1.8E+07	-	1.1E+07	-	9.6E+08	-	1.7E+07	-	1.9E+07	-	1.5E+09	-	1.5E+08	-
Crowfoot Creek	3.9E+07	-	6.8E+07	-	4.3E+07	-	4.5E+07	-	5.4E+07	-	8.7E+07	-	7.1E+08	-	6.8E+07	-
Drain S6	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-
New West Coulee	3.1E+07	-	4.5E+07	-	6.5E+07	-	1.8E+08	-	8.7E+07	-	4.3E+07	-	6.3E+07	-	9.0E+07	-
Mean	*3.9E+07*	-	*4.4E+07*	-	*4.0E+07*	-	*4.0E+08*	-	*5.3E+07*	-	*4.9E+07*	-	*7.5E+08*	-	*1.0E+08*	-
Median	*3.9E+07*	-	*4.5E+07*	-	*4.3E+07*	-	*1.8E+08*	-	*5.4E+07*	-	*4.3E+07*	-	*7.1E+08*	-	*9.0E+07*	-

APPENDIX 13: DETECTION FREQUENCY AND MAXIMUM CONCENTRATION OF 42 PESTICIDE COMPOUNDS FOR ALL WATERSHEDS 1999 TO 2006.

Table A13.1. Detection frequency and maximum concentration of 42 pesticide compounds in Battersea Drain, 1998 to 2006.

	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1998	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide Compound	Detection Frequency (%)									Mean	Maximum (µg L^{-1})								
Number of samples analyzed	**13**	**6**	**10**	**7**	**6**	**11**	**13**	**10**	**13**		**13**	**6**	**10**	**7**	**6**	**11**	**13**	**10**	**13**
2,4-D	92	83	60	100	100	91	92	100	92	90	1.759	0.202	0.093	0.356	0.27	8.534	0.141	0.347	0.342
MCPA	54	67	10	43	100	64	23	30	38	48	0.102	0.133	0.070	7.279	0.182	0.043	0.010	0.041	0.028
TRIALLATE(AVADEXBW)	0	0	10	43	33	18	15	60	15	22			0.005	0.014	0.014	0.018	0.009	0.037	0.006
DICAMBA(BANVEL)	23	17	20	29	50	18	15	0	0	19	0.048	0.023	0.015	0.037	0.159	0.085	0.005		
CLOPYRALID(LONTREL)	0	0	0	14	50	36	0	0	38	15				0.069	0.049	0.095			0.386
DICHLORPROP(2,4-DP)	23	17	0	14	33	9	0	10	0	12	0.110	0.009		0.032	0.007	0.310		0.021	
BROMOXYNIL	15	17	10	14	17	18	0	0	0	10	0.050	0.030	0.007	0.007	0.007	0.348			
ATRAZINE	0	50	0	0	17	9	0	0	0	8		0.142			0.009	0.014			
MCPP(MECOPROP)	15	0	10	0	0	27	0	0	0	6	0.011		0.011			0.109			
CHLORPYRIFOS-ETHYL(DURSBAN)	8	33	10	0	0	0	0	0	0	6	0.007	0.781	0.025						
2,4-DB	0	0	0	0	0	0	0	0	0	0									
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0									
BROMACIL	0	0	0	0	0	0	0	0	0	0									
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0	0									
DIAZINON	0	0	0	0	0	0	0	0	0	0									
DIURON	0	0	0	0	0	0	0	0	0	0									
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0									
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0	0									
PICLORAM(TORDON)	0	0	0	0	0	0	0	0	0	0									
PYRIDABEN	0	0	0	0	0	0	0	0	0	0									
QUINCLORAC	0	0	0	0	0	0	0	0	0	0									
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0									
IMAZETHAPYR	0	0	0	0	0	0	0	0	0	0									
IMAZAMOX		0	0	0	0	0	0	0	0	0									
SIMAZINE					17	18	0	30	0	13					0.370	0.210		0.024	
TRICLOPYR					0	0	0	0	0	0									
2,4-DICHLOROPHENOL					0	9	0	0	0	2						0.074			
4-CHLORO-2-METHYLPHENOL					0	0	0	0	0	0									
BENTAZON									8	8									0.051
FLUROXYPYR									0	0									
ETHOFUMESATE									8	8									0.187
METALAXYL-M									15	15									0.818
CLODINAFOP ACID METABOLITE									0	0									
IPRODIONE									0	0									
METRIBUZIN									0	0									
OXYCARBOXIN								0	0	0									
VINCLOZOLIN								0	0	0									
METHOMYL								0	0	0									
GLYPHOSATE(ROUNDUP)					33	0	0			11					0.395				
AMINOMETHYLPHOSPHONICACID					0	0	0			0									
GLUFOSINATE					0	0	0			0									
Number of compounds detected*	7	7	7	7	9	11	4	5	7	15									

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.2. Detection frequency and maximum concentration of 42 pesticide compounds in Blindman River, 1997 to 2006.

Pesticide compound	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)										Mean	Maximum (µg L^{-1})									
Number of samples analyzed	**3**	**17**	**18**	**14**	**10**	**10**	**9**	**13**	**19**	**10**		**3**	**17**	**18**	**14**	**10**	**10**	**9**	**13**	**19**	**10**
2,4-D	0	18	6	29	50	30	22	38	47	50	29		0.029	0.033	0.026	0.091	0.078	0.01	0.016	0.105	0.029
MCPA	33	12	28	7	10	0	22	23	16	10	16	0.006	0.018	0.028	0.041	0.008		0.014	0.007	0.010	0.013
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0	0	0										
DICAMBA(BANVEL)	0	0	0	0	0	0	11	0	0	10	2							0.013			0.008
CLOPYRALID(LONTREL)	0	0	0	0	0	10	0	0	0	0	1						0.024				
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0	0	0										
BROMOXYNIL	0	0	11	0	0	0	11	8	5	0	4			0.008				0.006	0.008	0.005	
ATRAZINE	0	0	0	0	0	0	0	0	0	0	0										
MCPP(MECOPROP)	0	0	6	14	0	0	0	0	0	0	2			0.011	0.027						
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0	0										
2,4-DB	0	0	0	0	0	0	0	0	0	0	0										
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0	0										
BROMACIL	0	0	0	0	0	0	0	0	0	0	0										
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0	0										
DIAZINON	0	0	0	0	0	0	0	0	0	0	0										
DIURON	0	0	0	0	0	0	0	0	0	0	0										
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0	0										
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0	0										
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0	0	0										
PICLORAM(TORDON)	0	18	11	57	30	30	0	0	11	30	19		0.105	0.031	0.156	0.193	0.180			0.070	0.073
PYRIDABEN		0	0	0	0	0	0	0	0	0	0										
QUINCLORAC		0	0	0	0	0	0	0	0	0	0										
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0	0										
IMAZETHAPYR		0	0	0	0	0	0	0	0	0	0										
IMAZAMOX			0	0	0	0	0	0	0	0	0										
SIMAZINE						0	0	0	0	0	0										
TRICLOPYR						20	11	46	32	20	26						0.060	0.074	0.195	0.158	0.091
2,4-DICHLOROPHENOL						0	0	0	0	0	0										
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0	0										
BENTAZON							0			0	0										
FLUROXYPYR							0			0	0										
ETHOFUMESATE							0			0	0										
METALAXYL-M							0			0	0										
CLODINAFOP ACID METABOLITE							0			0	0										
IPRODIONE							0			0	0										
METRIBUZIN							0			0	0										
OXYCARBOXIN								0	0	0	0										
VINCLOZOLIN								0	0	0	0										
METHOMYL									0	0	0										
GLYPHOSATE(ROUNDUP)										0	0										
AMINOMETHYLPHOSPHONICACID										0	0										
GLUFOSINATE										0	0										
Number of compounds detected*	1	3	5	4	3	4	5	4	5	5	8										

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.3. Detection frequency and maximum concentration of 42 pesticide compounds in Buffalo Creek, 1997 to 2006.

Pesticide Compound	1997	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs Mean	1997	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)										Maximum Concentration ($\mu g\ L^{-1}$)								
Number of samples Analyzed	**2**	**4**	**6**	**4**	**7**	**14**	**9**	**12**	**10**		**2**	**4**	**6**	**4**	**7**	**14**	**9**	**12**	**10**
2,4-D	100	25	67	25	0	43	56	58	50	47	0.048	0.012	0.029	0.057		0.09	0.04	0.045	0.021
MCPA	0	0	17	25	0	29	22	33	10	15			0.035	0.104		0.085	0.022	0.023	0.021
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0	0									
DICAMBA(BANVEL)	0	0	0	0	0	0	0	0	0	0									
CLOPYRALID(LONTREL)	0	0	0	0	0	0	0	8	0	1								0.020	
DICHLORPROP(2,4-DP)	0	0	0	25	0	0	0	8	0	4				0.007				0.007	
BROMOXYNIL	0	0	0	25	0	7	0	8	0	4				0.012		0.006		0.006	
ATRAZINE	0	0	0	0	0	0	0	0	0	0									
MCPP(MECOPROP)	0	0	0	0	0	0	0	0	0	0									
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0									
2,4-DB	0	0	0	0	0	0	0	0	0	0									
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0									
BROMACIL	0	0	0	0	0	0	0	0	0	0									
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0									
DIAZINON	0	0	0	0	0	0	0	0	0	0									
DIURON	0	0	0	0	0	0	0	0	0	0									
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0									
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	8	10	2								0.169	0.323
PICLORAM(TORDON)	0	0	0	0	0	0	0	0	0	0									
PYRIDABEN		0	0	0	0	0	0	0	0	0									
QUINCLORAC		0	0	0	0	0	0	0	0	0									
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0									
IMAZETHAPYR		0	0	0	0	0	0	0	0	0									
IMAZAMOX		0	0	0	0	0	0	0	0	0									
SIMAZINE					0	0	0	0	0	0									
TRICLOPYR					0	0	0	25	20	9								0.020	0.105
2,4-DICHLOROPHENOL					0	0	0	0	0	0									
4-CHLORO-2-METHYLPHENOL					0	0	0	0	0	0									
BENTAZON					0	0			0	0									
FLUROXYPYR					0	0			0	0									
ETHOFUMESATE					0	0			0	0									
METALAXYL-M					0	0			0	0									
CLODINAFOP ACID METABOLITE						0			0	0									
IPRODIONE					0	0			20	7									0.224
METRIBUZIN					0	0			0	0									
OXYCARBOXIN							0	0	0	0									
VINCLOZOLIN							0	0	0	0									
METHOMYL								0	0	0									
GLYPHOSATE(ROUNDUP)					0	10	0		22	8						0.201			5.649
AMINOMETHYLPHOSPHONICACID					0	0	0		11	3									1.148
GLUFOSINATE					0	0	0		0	0									
Number of compounds detected*	1	1	2	4	0	3	2	7	5	8									

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.4. Detection frequency and maximum concentration of 42 pesticide compounds in Crowfoot Creek, 1997 to 2006.

	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide Compound	Detection Frequency (%)										Mean	Maximum Concentration (µg L^{-1})									
Number of samples analyzed	**3**	**12**	**2**	**15**	**12**	**15**	**17**	**11**	**14**	**13**		**3**	**12**	**2**	**15**	**12**	**15**	**17**	**11**	**14**	**13**
2,4-D	100	83	100	100	92	100	100	100	100	100	98	1.006	0.502	0.021	1.407	0.35	0.75	1.187	0.484	0.162	1.684
MCPA	67	50	0	40	50	53	47	55	43	38	44	0.066	0.082		0.094	0.065	0.081	0.053	0.075	0.058	0.783
TRIALLATE(AVADEXBW)	67	17	0	13	0	0	41	27	29	8	20	0.407	0.026		0.108			0.119	0.033	0.037	0.005
DICAMBA(BANVEL)	67	25	100	87	75	60	59	55	29	77	63	0.405	0.319	0.015	0.058	0.436	0.052	0.427	0.037	0.093	0.165
CLOPYRALID(LONTREL)	0	8	0	0	17	27	6	9	0	8	7		0.038			0.040	0.053	0.020	0.034		0.080
DICHLORPROP(2,4-DP)	0	8	0	0	0	13	0	9	0	15	5		0.021				0.025		0.012		0.373
BROMOXYNIL	33	25	0	7	8	13	18	18	14	23	16	0.007	0.058		0.008	0.005	0.029	0.015	0.019	0.024	0.502
ATRAZINE	67	50	50	33	25	0	0	9	0	0	23	0.026	0.047	0.009	0.027	0.015			0.016		
MCPP(MECOPROP)	100	67	100	73	67	40	65	73	50	46	68	0.064	0.036	0.024	0.133	0.032	0.070	0.039	0.019	0.045	0.040
CHLORPYRIFOS-ETHYL(DURSBAN)	0	8	0	0	0	0	0	0	0	0	1		0.015								
2,4-DB	0	0	0	0	0	0	0	0	0	0	0										
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0	0										
BROMACIL	0	0	0	0	0	0	0	0	0	0	0										
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0	0										
DIAZINON	0	0	0	0	8	0	0	0	0	0	1					0.041					
DIURON	0	0	0	0	0	0	0	0	0	0	0										
ETHALFLURALIN(EDGE)	33	0	0	7	0	0	0	0	0	0	4	0.006			0.009						
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	33	8	0	0	0	0	0	0	0	0	4	0.010	0.023								
IMAZAMETHABENZ-METHYL	0	0	0	0	8	0	29	27	14	31	11					0.074		1.113	0.678	0.439	0.404
PICLORAM(TORDON)	0	25	50	40	0	0	0	0	0	23	14		0.058	0.005	0.025						0.040
PYRIDABEN		0	0	0	0	0	0	0	0	0	0										
QUINCLORAC		0	0	0	0	0	0	0	0	0	0										
TRIFLURALIN(TREFLAN)	33	0	0	0	0	0	0	0	0	0	3	0.007									
IMAZETHAPYR		0	0	0	0	0	0	0	0	8	1										0.025
IMAZAMOX			0	0	0	0	0	0	0	8	1										0.063
SIMAZINE						0	0	0	0	0	0										
TRICLOPYR						0	0	0	0	23	5										0.074
2,4-DICHLOROPHENOL						0	6	9	0	15	6							0.061	0.014		0.076
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0	0										
BENTAZON						0	0	0		0	0										
FLUROXYPYR						0	0	25		15	10								0.022		0.409
ETHOFUMESATE						0	0	0		0	0										
METALAXYL-M						0	0	0		0	0										
CLODINAFOP ACID METABOLITE						0	0	0		0	0										
IPRODIONE						0	0	0		0	0										
METRIBUZIN						0	0	0		0	0										
OXYCARBOXIN								0	0	0	0										
VINCLOZOLIN								0	0	0	0										
METHOMYL									0	0	0										
GLYPHOSATE(ROUNDUP)																					
AMINOMETHYLPHOSPHONICACID																					
GLUFOSINATE																					
Number of compounds detected*	10	12	5	9	9	7	9	12	7	15	21										

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.5. Detection frequency and maximum concentration of 42 pesticide compounds in Drain S6, 1999 to 2006.

	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide compound	Detection Frequency (%)								Mean	Maximum Concentration (µg L^{-1})							
Number of samples analyzed	**7**	**8**	**9**	**7**	**10**	**13**	**8**	**13**		**7**	**8**	**9**	**7**	**10**	**13**	**8**	**13**
2,4-D	86	50	67	71	40	62	100	85	70	0.205	0.104	0.014	0.176	0.144	0.144	4.8	0.32
MCPA	57	38	33	57	30	15	50	23	38	0.105	0.041	0.018	0.039	0.031	0.007	0.272	0.013
TRIALLATE(AVADEXBW)	0	0	0	14	0	0	13	0	3				0.006			0.104	
DICAMBA(BANVEL)	29	38	33	43	30	46	25	31	34	0.043	0.029	0.012	0.016	0.023	0.067	0.079	0.092
CLOPYRALID(LONTREL)	14	0	0	0	10	0	0	0	3	0.053				0.109			
DICHLORPROP(2,4-DP)	14	0	0	0	0	0	13	0	3	0.016						0.057	
BROMOXYNIL	14	0	0	0	0	8	13	8	5	0.018					0.008	0.082	0.020
ATRAZINE	0	0	11	0	10	0	13	0	4			0.132		0.033		0.057	
MCPP(MECOPROP)	14	0	0	0	10	0	13	0	5	0.015				0.050		0.095	
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0								
2,4-DB	0	0	0	0	0	0	0	0	0								
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0								
BROMACIL	0	0	0	0	0	0	0	0	0								
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0								
DIAZINON	0	0	0	0	0	0	0	0	0								
DIURON	0	0	0	0	0	0	0	0	0								
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	13	0	2							0.018	
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0								
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0								
PICLORAM(TORDON)	0	0	33	29	10	0	0	0	9			0.030	0.640	0.041			
PYRIDABEN	0	0	0	0	0	0	0	0	0								
QUINCLORAC	0	0	0	0	0	0	0	0	0								
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0								
IMAZETHAPYR	0	0	0	0	0	0	0	0	0								
IMAZAMOX	0	0	0	0	0	0	0	0	0								
SIMAZINE				0	10	0	0	0	2					0.371			
TRICLOPYR				0	0	38	25	46	22						0.248	0.051	0.165
2,4-DICHLOROPHENOL				0	0	0	13	0	3							0.469	
4-CHLORO-2-METHYLPHENOL				0	0	0	0	0	0								
BENTAZON				0	50	100		8	39					0.030	0.308		0.487
FLUROXYPYR				0	50	0		0	13					0.073			
ETHOFUMESATE				0	0	80		0	20						0.106		
METALAXYL-M				0	0	0		0	0								
CLODINAFOP ACID METABOLITE				0	0	0		0	0								
IPRODIONE				0	0	0		0	0								
METRIBUZIN				0	0	0		0	0								
OXYCARBOXIN						0	0	0	0								
VINCLOZOLIN						11	0	0	4						0.011		
METHOMYL							0	0	0								
GLYPHOSATE(ROUNDUP)				0	0	0			0								
AMINOMETHYLPHOSPHONICACID				0	0	0			0								
GLUFOSINATE				0	0	0			0								
Number of compounds detected*	7	3	5	5	10	8	11	6	18								

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.6. Detection frequency and maximum concentration of 42 pesticide compounds in Grande Prairie Creek, 1999 to 2006.

	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide Compound	Detection Frequency (%)								Mean	Maximum Concentration (µg L^{-1})							
Number of samples analyzed	**7**	**12**	**8**	**5**	**6**	**8**	**9**	**6**		**7**	**12**	**8**	**5**	**6**	**8**	**9**	**6**
2,4-D	0	25	25	20	17	38	22	50	25		0.041	1.164	0.023	0.005	0.017	0.006	0.111
MCPA	57	50	63	60	33	63	44	33	50	0.187	0.031	0.083	0.065	0.024	0.028	0.031	0.055
TRIALLATE(AVADEXBW)	29	25	13	40	33	25	0	0	21	0.029	0.464	0.012	0.080	0.020	0.015		
DICAMBA(BANVEL)	14	33	13	0	0	13	0	50	15	0.044	0.213	0.007			0.006		0.071
CLOPYRALID(LONTREL)	29	42	50	40	17	50	4	33	33	0.128	0.221	0.052	0.069	0.109	0.261	0.023	0.330
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0								
BROMOXYNIL	0	0	0	0	0	0	0	0	0								
ATRAZINE	0	0	0	0	0	0	0	0	0								
MCPP(MECOPROP)	29	25	0	0	0	13	0	17	10	0.040	0.030				0.008		0.006
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0								
2,4-DB	0	0	0	0	0	0	0	0	0								
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0								
BROMACIL	0	0	0	0	0	0	0	0	0								
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0								
DIAZINON	0	0	0	0	0	0	0	0	0								
DIURON	0	0	0	0	0	0	0	0	0								
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0								
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	29	0	0	0	0	0	0	0	4	0.030							
IMAZAMETHABENZ-METHYL	14	25	13	20	0	25	11	17	16	0.227	1.521	0.146	0.189		0.311	0.453	0.247
PICLORAM(TORDON)	0	17	25	20	0	0	0	17	10		0.316	2.364	0.054				0.148
PYRIDABEN	0	0	0	0	0	0	0	0	0								
QUINCLORAC	0	0	0	0	0	0	0	0	0								
TRIFLURALIN(TREFLAN)	14	0	0	0	0	0	0	0	2	0.007							
IMAZETHAPYR	29	0	0	20	0	13	0	17	10	0.163			0.077		0.070		0.054
IMAZAMOX	0	0	0	0	0	0	0	0	0								
SIMAZINE				0	0	0	0	0	0								
TRICLOPYR				60	100	50	0	0	42				0.235	0.433	0.061		
2,4-DICHLOROPHENOL				0	0	0	0	0	0								
4-CHLORO-2-METHYLPHENOL				0	0	0	0	0	0								
BENTAZON						0		0	0								
FLUROXYPYR						100		0	50						0.054		
ETHOFUMESATE						0		0	0								
METALAXYL-M						0		0	0								
CLODINAFOP ACID METABOLITE						0		0	0								
IPRODIONE						0		0	0								
METRIBUZIN						0		0	0								
OXYCARBOXIN						0	0	0	0								
VINCLOZOLIN						0	0	0	0								
METHOMYL							0	0	0								
GLYPHOSATE(ROUNDUP)																	
AMINOMETHYLPHOSPHONICACID																	
GLUFOSINATE																	
Number of compounds detected*	9	8	7	8	5	10	4	8	13								

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.7. Detection frequency and maximum concentration of 42 pesticide compounds in Hines Creek, 1999 to 2006.

	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide Compound	Detection Frequency (%)								Mean	Maximum Concentration($\mu g\ L^{-1}$)							
Number of samples analyzed	**3**	**6**	**5**	**9**	**7**	**8**	**8**	**8**		**3**	**6**	**5**	**9**	**7**	**8**	**8**	**8**
2,4-D	0	0	20	0	0	0	13	0	4			0.081				0.008	
MCPA	33	0	20	0	0	0	0	0	7	0.014		0.010					
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0								
DICAMBA(BANVEL)	0	0	0	0	0	0	0	0	0								
CLOPYRALID(LONTREL)	0	0	0	0	0	0	0	0	0								
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0								
BROMOXYNIL	0	0	0	0	0	0	0	0	0								
ATRAZINE	0	0	0	0	0	0	0	0	0								
MCPP(MECOPROP)	0	0	0	0	0	0	0	0	0								
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0								
2,4-DB	0	0	0	0	0	0	0	0	0								
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0								
BROMACIL	0	0	0	0	0	0	0	0	0								
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0								
DIAZINON	0	0	0	0	0	0	0	0	0								
DIURON	0	0	0	0	0	0	0	0	0								
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0								
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0								
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0								
PICLORAM(TORDON)	0	0	0	0	0	0	0	0	0								
PYRIDABEN	0	0	0	0	0	0	0	0	0								
QUINCLORAC	0	0	0	0	0	0	0	0	0								
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0								
IMAZETHAPYR	0	0	0	0	0	0	0	0	0								
IMAZAMOX	0	0	0	0	0	0	0	0	0								
SIMAZINE				0	0	0	0	0	0								
TRICLOPYR				33	14	0	0	0	10				0.431	0.018			
2,4-DICHLOROPHENOL				0	0	0	0	0	0								
4-CHLORO-2-METHYLPHENOL				0	0	0	0	0	0								
BENTAZON				0		0		0	0								
FLUROXYPYR				0		0		0	0								
ETHOFUMESATE				0		0		0	0								
METALAXYL-M				0		0		0	0								
CLODINAFOP ACID METABOLITE				0		0		0	0								
IPRODIONE				0		0		0	0								
METRIBUZIN				0		0		0	0								
OXYCARBOXIN						0	0	0	0								
VINCLOZOLIN						0	0	0	0								
METHOMYL							0	0	0								
GLYPHOSATE(ROUNDUP)																	
AMINOMETHYLPHOSPHONICACID																	
GLUFOSINATE																	
Number of compounds detected*	1	0	2	1	1	0	1	0	3								

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.8. Detection frequency and maximum concentration of 42 pesticide compounds in Haynes Creek M1, 1999 to 2006.

Pesticide Compound	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1998	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)									Mean	Maximum Concentration (µg L^{-1})								
Number of samples analyzed	**5**	**8**	**13**	**5**	**4**	**6**	**3**	**8**	**4**		**5**	**8**	**13**	**5**	**4**	**6**	**3**	**8**	**4**
2,4-D	80	38	54	80	75	67	100	63	75	70	0.415	0.179	0.817	0.17	0.028	0.41	0.04	0.065	0.034
MCPA	80	25	54	80	50	33	100	50	25	55	0.069	0.296	1.878	0.030	0.010	0.035	0.077	0.051	1.231
TRIALLATE(AVADEXBW)	20	25	0	0	0	0	0	0	0	5	0.011	0.006							
DICAMBA(BANVEL)	0	0	8	0	0	0	0	0	0	1			0.011						
CLOPYRALID(LONTREL)	0	38	46	60	100	83	67	75	75	60		0.152	2.717	0.086	0.198	0.238	0.388	0.138	0.819
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0	0									
BROMOXYNIL	0	25	0	0	0	0	33	0	0	6		0.082					0.005		
ATRAZINE	0	0	0	0	0	0	0	0	0	0									
MCPP(MECOPROP)	0	0	8	0	25	0	0	0	0	4			0.025		0.030				
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0									
2,4-DB	0	0	0	0	0	0	0	0	0	0									
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0									
BROMACIL	0	0	0	0	0	0	0	0	0	0									
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0	0									
DIAZINON	0	0	0	0	0	0	0	0	0	0									
DIURON	0	0	0	0	0	0	0	0	0	0									
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	40	38	0	0	0	0	0	0	0	9	0.025	0.025							
IMAZAMETHABENZ-METHYL	0	75	92	80	75	100	100	38	75	71		6.287	5.250	0.203	0.586	0.517	1.204	0.722	0.864
PICLORAM(TORDON)	40	50	77	40	100	50	33	0	75	52	0.017	0.056	0.136	0.332	0.093	0.226	0.015		0.163
PYRIDABEN	0	0	0	0	0	0	0	0	0	0									
QUINCLORAC	0	0	0	0	0	0	0	0	0	0									
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0									
IMAZETHAPYR	0	0	0	0	0	0	0	0	0	0									
IMAZAMOX		0	0	0	0	0	0	0	0	0									
SIMAZINE					0	0	0	0	0	0									
TRICLOPYR					25	67	100	25	0	43					0.011	0.026	0.490	0.047	
2,4-DICHLOROPHENOL					0	0	0	0	0	0									
4-CHLORO-2-METHYLPHENOL					0	0	0	0	0	0									
BENTAZON						0	0		0	0									
FLUROXYPYR						25	100		0	42						0.042	0.460		
ETHOFUMESATE						0	0		0	0									
METALAXYL-M						0	0		0	0									
CLODINAFOP ACID METABOLITE						0	0		0	0									
IPRODIONE						0	0		25	8									0.178
METRIBUZIN						0	0		0	0									
OXYCARBOXIN								0	0	0									
VINCLOZOLIN								0	0	0									
METHOMYL								13	0	6								0.100	
GLYPHOSATE(ROUNDUP)					50	0	33			28					1.105		0.859		
AMINOMETHYLPHOSPHONICACID					0	0	0			0									
GLUFOSINATE					0	0	0			0									
Number of compounds detected*	5	8	7	5	7	7	8	6	6	14									

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.9. Detection frequency and maximum concentration of 42 pesticide compounds in Haynes Creek M6, 1997 to 2006.

	1997	1998	1999	2000	2001	2002	2003	2005	2006	All Yrs	1997	1998	1999	2000	2001	2002	2003	2005	2006
Pesticide compound	Detection Frequency (%)									Mean	Maximum Concentration (µg L^{-1})								
Number of samplers analyzed	**5**	**9**	**7**	**14**	**6**	**3**	**7**	**6**	**6**		**5**	**9**	**7**	**14**	**6**	**3**	**7**	**6**	**6**
2,4-D	20	67	86	100	83	100	100	50	100	78	0.021	0.195	0.248	4.933	2.81	0.191	0.06	0.02	0.042
MCPA	40	56	43	79	67	67	57	67	83	62	0.030	0.268	0.242	0.365	0.124	0.008	0.040	0.019	0.044
TRIALLATE(AVADEXBW)	20	0	14	0	0	0	0	0	0	4	0.011		0.005						
DICAMBA(BANVEL)	0	0	0	21	33	0	0	0	0	6				0.087	0.032				
CLOPYRALID(LONTREL)	0	0	14	64	83	100	100	83	83	59			0.035	0.106	0.132	0.150	0.223	0.084	0.118
DICHLORPROP(2,4-DP)	0	44	14	0	0	0	0	0	0	7		0.025	0.018						
BROMOXYNIL	0	33	14	0	0	0	0	0	0	5		0.033	0.005						
ATRAZINE	0	0	0	0	0	0	0	0	0	0									
MCPP(MECOPROP)	40	11	0	36	17	0	0	0	0	11	0.016	0.009		0.184	0.038				
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0									
2,4-DB	0	0	0	0	0	0	0	0	0	0									
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0									
BROMACIL	0	0	0	0	0	0	0	0	0	0									
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0									
DIAZINON	0	0	14	0	0	0	0	0	0	2			0.005						
DIURON	0	0	0	0	0	0	0	0	0	0									
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	14	0	0	0	0	0	0	2			0.017						
IMAZAMETHABENZ-METHYL	40	11	86	43	0	67	57	67	67	49	1.285	1.230	0.186	1.332		0.342	0.310	0.958	1.570
PICLORAM(TORDON)	20	0	43	57	33	100	43	0	17	35	0.014		0.038	0.156	0.050	0.075	0.073		0.026
PYRIDABEN		11	0	0	0	0	0	0	0	1		0.025							
QUINCLORAC		0	0	0	0	0	0	0	0	0									
TRIFLURALIN(TREFLAN)	0	0	29	0	0	0	0	0	0	3			0.005						
IMAZETHAPYR		0	29	7	0	0	0	0	0	4			0.062	0.039					
IMAZAMOX			0	0	0	0	0	0	0	0									
SIMAZINE						0	0	0	0	0									
TRICLOPYR						0	100	0	17	29							0.039		0.039
2,4-DICHLOROPHENOL						0	0	0	0	0									
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0									
BENTAZON							50		0	25							0.015		
FLUROXYPYR							0		0	0									
ETHOFUMESATE							0		0	0									
METALAXYL-M							0		0	0									
CLODINAFOP ACID METABOLITE							0		0	0									
IPRODIONE							0		33	17									0.339
METRIBUZIN							0		0	0									
OXYCARBOXIN								0	0	0									
VINCLOZOLIN								0	0	0									
METHOMYL								0	0	0									
GLYPHOSATE(ROUNDUP)							0			0									
AMINOMETHYLPHOSPHONICACID							0			0									
GLUFOSINATE							0			0									
Number of compounds detected*	6	7	12	8	6	5	7	4	7	18									

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.10. Detection frequency and maximum concentration of 42 pesticide compounds in Kleskun Drain, 1999 to 2006.

Pesticide Compound	Detection Frequency (%) 1999	2000	2001	2002	2003	2004	2005	2006	All Yrs Mean	Maximum Concentration (µg L⁻¹) 1999	2000	2001	2002	2003	2004	2005	2006
Number of samples analyzed	**4**	**8**	**5**	**6**	**6**	**7**	**6**	**4**		**4**	**8**	**5**	**6**	**6**	**7**	**6**	**4**
2,4-D	25	50	100	50	83	100	83	100	74	0.009	1.972	0.359	0.83	0.241	0.457	0.027	0.142
MCPA	0	38	60	33	33	43	67	0	34		0.074	0.068	0.013	0.052	0.173	0.093	
TRIALLATE(AVADEXBW)	0	0	20	17	0	0	0	0	5			0.034	0.412				
DICAMBA(BANVEL)	0	13	40	33	17	0	0	0	13		0.190	0.027	0.048	0.012			
CLOPYRALID(LONTREL)	50	13	60	100	83	86	83	100	72	0.066	0.134	0.232	1.790	1.439	0.873	0.243	0.133
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0								
BROMOXYNIL	0	0	0	0	0	14	0	0	2						0.011		
ATRAZINE	0	0	0	0	0	14	0	0	2						0.070		
MCPP(MECOPROP)	0	0	40	0	0	0	0	0	5			0.051					
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0								
2,4-DB	0	0	0	0	0	0	0	0	0								
ALPHA-BENZENEHEXACHLORIDE	0	0	20	0	0	0	0	0	3			0.008					
BROMACIL	0	0	0	0	0	0	0	0	0								
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0								
DIAZINON	0	0	0	0	0	0	0	0	0								
DIURON	0	0	0	0	0	0	0	0	0								
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0								
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	25	0	20	17	0	14	0	0	9	0.027		0.029	0.021		0.009		
IMAZAMETHABENZ-METHYL	0	0	60	17	0	0	0	75	19			0.163	0.146				0.837
PICLORAM(TORDON)	25	88	80	83	100	100	50	100	78	0.073	13.407	1.947	2.580	0.819	2.085	0.494	1.285
PYRIDABEN	0	0	0	0	0	0	0	0	0								
QUINCLORAC	0	0	0	0	0	0	0	0	0								
TRIFLURALIN(TREFLAN)	25	25	0	0	0	0	0	0	6	0.013	0.013						
IMAZETHAPYR	50	0	0	0	0	14	0	0	8	0.182					0.079		
IMAZAMOX	0	0	0	0	0	14	0	0	2						0.037		
SIMAZINE				0	0	0	0	0	0								
TRICLOPYR				17	17	0	0	0	7				0.780	0.031			
2,4-DICHLOROPHENOL				0	0	14	0	0	3						0.023		
4-CHLORO-2-METHYLPHENOL				0	0	0	0	0	0								
BENTAZON				0	0	0		0	0								
FLUROXYPYR				0	0	0		0	0								
ETHOFUMESATE				0	0	0		0	0								
METALAXYL-M				0	0	0		0	0								
CLODINAFOP ACID METABOLITE				0	0	0		0	0								
IPRODIONE				0	0	0		0	0								
METRIBUZIN				0	0	0		0	0								
OXYCARBOXIN						0	0	0	0								
VINCLOZOLIN						0	0	0	0								
METHOMYL							0	0	0								
GLYPHOSATE(ROUNDUP)																	
AMINOMETHYLPHOSPHONICACID																	
GLUFOSINATE																	
Number of compounds detected*	6	6	10	9	6	10	4	4	17								

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.11. Detection frequency and maximum concentration of 42 pesticide compounds in Meadow Creek, 1997 to 2006.

	1997	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1997	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide Compound	Detection Frequency (%)									Mean	Maximum Concentration (µg L^{-1})								
Number of samples analyzed	**2**	**5**	**6**	**5**	**10**	**10**	**12**	**10**	**13**		**2**	**5**	**6**	**5**	**10**	**10**	**12**	**10**	**13**
2,4-D	0	40	33	60	20	20	50	30	8	29		0.052	0.094	0.023	0.057	0.014	0.154	0.015	0.009
MCPA	50	20	33	20	0	0	0	10	8	16	0.005	0.011	0.034	0.012				0.005	0.007
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0	0									
DICAMBA(BANVEL)	0	0	0	0	0	0	0	0	0	0									
CLOPYRALID(LONTREL)	0	0	0	0	0	0	0	0	0	0									
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0	0									
BROMOXYNIL	0	20	0	0	0	0	0	0	0	2		0.011							
ATRAZINE	0	0	0	0	0	0	0	0	0	0									
MCPP(MECOPROP)	0	0	0	0	0	0	0	0	0	0									
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0									
2,4-DB	0	0	0	0	0	0	0	0	0	0									
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0									
BROMACIL	0	0	0	0	0	0	0	0	0	0									
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0									
DIAZINON	0	0	0	0	0	0	0	0	0	0									
DIURON	0	0	0	0	0	0	0	0	0	0									
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0									
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0	0									
PICLORAM(TORDON)	0	0	0	0	10	0	0	0	0	1					0.062				
PYRIDABEN		0	0	0	0	0	0	0	0	0									
QUINCLORAC		0	0	0	0	0	0	0	0	0									
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0									
IMAZETHAPYR		0	0	0	0	0	0	0	0	0									
IMAZAMOX		0	0	0	0	0	0	0	0	0									
SIMAZINE					0	0	0	0	0	0									
TRICLOPYR					0	0	0	0	0	0									
2,4-DICHLOROPHENOL					0	0	0	0	0	0									
4-CHLORO-2-METHYLPHENOL					0	0	0	0	0	0									
BENTAZON					0	0	0		0	0									
FLUROXYPYR					0	0	0		0	0									
ETHOFUMESATE					0	0	0		0	0									
METALAXYL-M					0	0	0		0	0									
CLODINAFOP ACID METABOLITE					0	0	0		0	0									
IPRODIONE					0	0	0		0	0									
METRIBUZIN					0	0	0		0	0									
OXYCARBOXIN							0	0	0	0									
VINCLOZOLIN							0	0	0	0									
METHOMYL								0	0	0									
GLYPHOSATE(ROUNDUP)																			
AMINOMETHYLPHOSPHONICACID																			
GLUFOSINATE																			
Number of compounds detected*	1	3	2	2	2	1	1	2	2	4									

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.12. Detection frequency and maximum concentration of 42 pesticide compounds in New West Coulee, 1999 to 2006.

	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide Compound	Detection Frequency (%)								Mean	Maximum Concentration (µg L^{-1})							
Number of samples analyzed	**6**	**9**	**8**	**6**	**11**	**13**	**10**	**13**		**6**	**9**	**8**	**6**	**11**	**13**	**10**	**13**
2,4-D	100	89	100	100	91	100	100	100	97	0.101	0.258	0.721	5.46	0.45	0.324	0.172	0.587
MCPA	33	44	63	67	45	62	40	54	51	0.034	0.177	0.205	0.033	0.178	0.080	0.036	0.040
TRIALLATE(AVADEXBW)	0	11	38	33	45	15	0	0	18		0.009	0.027	0.072	0.026	0.018		
DICAMBA(BANVEL)	33	22	75	67	55	46	40	77	52	0.015	0.061	0.205	0.970	0.045	0.029	0.163	0.362
CLOPYRALID(LONTREL)	0	0	25	50	36	15	0	31	20			0.237	0.053	0.074	0.051		0.044
DICHLORPROP(2,4-DP)	17	0	38	50	18	23	10	38	24	0.020		0.253	0.237	0.093	0.053	0.024	0.122
BROMOXYNIL	0	22	25	17	9	23	0	15	14		0.133	0.522	0.067	0.244	0.015		0.059
ATRAZINE	0	11	0	0	0	0	0	8	2		0.005						0.036
MCPP(MECOPROP)	0	33	25	0	0	8	10	8	10		0.086	0.039			0.042	0.019	0.005
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0								
2,4-DB	0	11	0	0	0	0	0	0	1		0.665						
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0								
BROMACIL	0	0	0	0	0	0	0	0	0								
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	15	2								0.139
DIAZINON	0	0	0	0	0	0	0	0	0								
DIURON	0	0	0	0	0	0	0	0	0								
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0								
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0								
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0								
PICLORAM(TORDON)	0	33	0	0	0	0	0	0	4		0.065						
PYRIDABEN	0	0	0	0	0	0	0	0	0								
QUINCLORAC	0	0	0	0	0	0	0	0	0								
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0								
IMAZETHAPYR	0	0	0	0	0	0	0	0	0								
IMAZAMOX	0	0	0	0	0	0	0	0	0								
SIMAZINE				17	27	38	40	69	38				0.760	0.793	0.668	0.224	4.570
TRICLOPYR				0	0	0	0	0	0								
2,4-DICHLOROPHENOL				0	18	0	10	15	9					0.077		0.045	0.050
4-CHLORO-2-METHYLPHENOL				0	9	0	0	0	2					2.876			
BENTAZON				0	0	20		23	11						0.016		0.249
FLUROXYPYR				0	50	80		15	36					0.030	0.036		0.059
ETHOFUMESATE				0	0	60		46	27						0.071		0.399
METALAXYL-M				0	0	0		0	0								
CLODINAFOP ACID METABOLITE				0	0	0		0	0								
IPRODIONE				0	0	0		0	0								
METRIBUZIN				0	0	20		0	5						0.010		
OXYCARBOXIN						0	0	0	0								
VINCLOZOLIN						0	0	0	0								
METHOMYL							0	0	0								
GLYPHOSATE(ROUNDUP)				17	0	8			8				0.219		0.242		
AMINOMETHYLPHOSPHONICACID				0	0	0			0								
GLUFOSINATE				0	0	0			0								
Number of compounds detected*	4	9	8	8	11	13	7	14	19								

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.13. Detection frequency and maximum concentration of 42 pesticide compounds in Paddle River, 1997 to 2006.

	1997	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1997	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide Compound	Detection Frequency (%)									Mean	Maximum Concentration (µg L^{-1})								
Number of samples analyzed	**2**	**10**	**11**	**11**	**8**	**9**	**7**	**19**	**13**		**2**	**10**	**11**	**11**	**8**	**9**	**7**	**19**	**13**
2,4-D	0	10	18	9	13	11	0	0	8	8		0.011	0.013	0.014	0.03	0.008			0.007
MCPA	50	0	0	0	0	11	0	5	8	8	0.005					0.224		0.005	0.006
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0	0									
DICAMBA(BANVEL)	0	0	9	0	0	0	0	0	0	1			0.007						
CLOPYRALID(LONTREL)	0	0	0	0	0	0	0	0	0	0									
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0	0									
BROMOXYNIL	0	0	0	0	0	0	0	0	0	0									
ATRAZINE	0	0	0	0	0	0	0	0	0	0									
MCPP(MECOPROP)	0	0	0	0	0	0	0	0	0	0									
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0									
2,4-DB	0	0	0	0	0	0	0	0	0	0									
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0									
BROMACIL	0	0	0	0	0	0	0	0	0	0									
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0									
DIAZINON	0	0	0	0	0	0	0	0	0	0									
DIURON	0	0	0	0	0	0	0	0	0	0									
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0									
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0	0									
PICLORAM(TORDON)	0	0	0	0	0	0	0	0	0	0									
PYRIDABEN		0	0	0	0	0	0	0	0	0									
QUINCLORAC		0	0	0	0	0	0	0	0	0									
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0									
IMAZETHAPYR		0	0	0	0	0	0	0	0	0									
IMAZAMOX		0	0	0	0	0	0	0	0	0									
SIMAZINE					0	0	0	0	0	0									
TRICLOPYR					50	56	43	0	38	37					0.315	0.234	0.050		0.052
2,4-DICHLOROPHENOL					0	0	0	0	0	0									
4-CHLORO-2-METHYLPHENOL					0	0	0	0	0	0									
BENTAZON							0		0	0									
FLUROXYPYR							0		0	0									
ETHOFUMESATE							0		0	0									
METALAXYL-M							0		0	0									
CLODINAFOP ACID METABOLITE							0		0	0									
IPRODIONE							0		0	0									
METRIBUZIN							0		0	0									
OXYCARBOXIN							0	0	0	0									
VINCLOZOLIN							0	0	0	0									
METHOMYL								0	0	0									
GLYPHOSATE(ROUNDUP)									0	0									
AMINOMETHYLPHOSPHONICACID									0	0									
GLUFOSINATE									0	0									
Number of compounds detected*	1	1	2	1	2	3	1	1	3	4									

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.14. Detection frequency and maximum concentration of 42 pesticide compounds in Prairie Blood Coulee, 1999 to 2006.

Pesticide Compound	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)								Mean	Maximum Concentration ($\mu g\ L^{-1}$)							
Number of samples analyzed	**6**	**6**	**4**	**8**	**4**	**10**	**8**	**12**		**6**	**6**	**4**	**8**	**4**	**10**	**8**	**12**
2,4-D	50	33	100	63	50	20	75	58	56	0.033	0.048	0.021	0.103	0.036	0.055	0.097	0.031
MCPA	33	17	50	38	50	20	63	33	38	0.025	0.057	0.009	0.459	0.809	0.776	0.076	0.051
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0								
DICAMBA(BANVEL)	0	0	75	38	25	20	0	0	20			0.027	0.245	0.005	0.510		
CLOPYRALID(LONTREL)	0	0	50	13	50	10	0	0	15			0.026	0.021	0.323	0.030		
DICHLORPROP(2,4-DP)	0	0	0	13	0	0	25	0	5				0.007			0.010	
BROMOXYNIL	17	17	0	25	0	0	25	17	13	0.005	0.018		0.267			0.041	0.014
ATRAZINE	0	0	0	0	0	0	0	0	0								
MCPP(MECOPROP)	0	17	25	25	0	20	0	0	11		0.012	0.006	0.017		0.041		
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0								
2,4-DB	0	0	0	0	0	0	0	0	0								
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0								
BROMACIL	0	0	0	0	0	0	0	0	0								
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0								
DIAZINON	0	0	0	0	0	0	0	0	0								
DIURON	0	0	0	0	0	0	0	0	0								
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0								
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	17	0	0	0	0	0	0	2		0.012						
IMAZAMETHABENZ-METHYL	0	0	0	0	25	0	0	0	3					0.121			
PICLORAM(TORDON)	0	0	0	13	0	0	0	0	2				0.011				
PYRIDABEN	0	0	0	0	0	0	0	0	0								
QUINCLORAC	0	0	0	0	0	0	0	0	0								
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0								
IMAZETHAPYR	0	0	0	0	0	0	0	0	0								
IMAZAMOX	0	0	0	0	0	0	0	0	0								
SIMAZINE				0	0	0	0	0	0								
TRICLOPYR				0	0	0	0	0	0								
2,4-DICHLOROPHENOL				0	0	0	0	0	0								
4-CHLORO-2-METHYLPHENOL				0	0	0	0	0	0								
BENTAZON				0	0	0		0	0								
FLUROXYPYR				0	0	0		0	0								
ETHOFUMESATE				0	0	0		0	0								
METALAXYL-M				0	0	0		0	0								
CLODINAFOP ACID METABOLITE				0	0	0		0	0								
IPRODIONE				0	0	0		0	0								
METRIBUZIN				0	0	0		0	0								
OXYCARBOXIN						0	0	0	0								
VINCLOZOLIN						0	0	0	0								
METHOMYL							0	0	0								
GLYPHOSATE(ROUNDUP)																	
AMINOMETHYLPHOSPHONICACID																	
GLUFOSINATE																	
Number of compounds detected*	3	5	5	8	5	5	4	3	10								

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.15. Detection frequency and maximum concentration of 42 pesticide compounds in Ray Creek, 1997 to 2006.

Pesticide Compound	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)										Mean	Maximum Concentration ($\mu g\ L^{-1}$)									
Number of samples analyzed	**2**	**9**	**11**	**12**	**8**	**5**	**9**	**8**	**9**	**8**		**2**	**9**	**11**	**12**	**8**	**5**	**9**	**8**	**9**	**8**
2,4-D	100	67	45	33	75	0	11	13	56	38	44	0.088	0.024	0.126	0.114	0.282		0.01	0.015	0.264	0.038
MCPA	100	56	45	42	63	20	44	63	44	38	51	0.061	0.089	0.213	0.190	0.132	0.013	0.085	0.054	0.295	0.031
TRIALLATE(AVADEXBW)	50	0	36	0	0	0	0	0	0	13	10	0.087		0.146							0.015
DICAMBA(BANVEL)	0	0	9	0	0	0	0	0	11	0	2			0.027						0.005	
CLOPYRALID(LONTREL)	0	0	36	0	0	60	22	0	33	0	15			0.170			0.034	0.109		0.048	
DICHLORPROP(2,4-DP)	0	0	9	0	0	0	0	0	0	0	1			0.010							
BROMOXYNIL	50	11	18	0	0	0	0	0	0	0	8	0.006	0.007	0.060							
ATRAZINE	0	0	0	0	0	0	0	0	0	0	0										
MCPP(MECOPROP)	50	0	9	8	0	0	0	0	0	0	7	0.006		0.005	0.014						
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0	0										
2,4-DB	0	0	0	0	0	0	0	0	0	0	0										
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0	0										
BROMACIL	0	0	0	0	0	0	0	0	0	0	0										
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0	0										
DIAZINON	0	0	0	0	13	0	0	0	0	0	1					0.008					
DIURON	0	0	0	0	0	0	0	0	0	0	0										
ETHALFLURALIN(EDGE)	50	0	0	0	0	0	0	0	0	0	5	0.039									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0	0										
IMAZAMETHABENZ-METHYL	0	11	73	17	25	0	56	75	22	38	32		0.100	0.889	2.055	0.588		0.525	0.506	1.095	0.650
PICLORAM(TORDON)	0	0	27	17	38	0	11	0	0	0	9			0.012	0.034	0.374		0.018			
PYRIDABEN		0	0	0	0	0	0	0	0	0	0										
QUINCLORAC		0	0	0	0	0	0	0	0	0	0										
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0	0										
IMAZETHAPYR		0	9	0	0	0	0	0	0	0	1			0.051							
IMAZAMOX			0	0	0	0	0	0	0	0	0										
SIMAZINE						0	0	0	0	0	0										
TRICLOPYR						0	0	0	0	13	3										0.032
2,4-DICHLOROPHENOL						0	0	0	0	0	0										
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0	0										
BENTAZON							0	0		0	0										
FLUROXYPYR							0	0		0	0										
ETHOFUMESATE							0	0		0	0										
METALAXYL-M							0	0		0	0										
CLODINAFOP ACID METABOLITE							0	0		0	0										
IPRODIONE							0	0		0	0										
METRIBUZIN							0	0		0	0										
OXYCARBOXIN								0	0	0	0										
VINCLOZOLIN								0	0	0	0										
METHOMYL									0	0	0										
GLYPHOSATE(ROUNDUP)						25	25	0		0	13						0.268	1.067			
AMINOMETHYLPHOSPHONICACID						0	0	0		0	0										
GLUFOSINATE						0	0	0		0	0										
Number of compounds detected*	6	4	11	5	5	2	5	3	5	5	14										

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.16. Detection frequency and maximum concentration of 42 pesticide compounds in Renwick Creek, 1997 to 2006.

Pesticide Compound	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)										Mean	Maximum Concentration (µg L^{-1})									
Number of samples analyzed	**2**	**8**	**10**	**5**	**3**	**4**	**9**	**7**	**9**	**10**		**2**	**8**	**10**	**5**	**3**	**4**	**9**	**7**	**9**	**10**
2,4-D	100	100	80	100	67	25	100	71	44	50	74	0.082	4.834	0.343	0.33	0.217	0.012	0.129	0.02	0.086	0.029
MCPA	100	50	60	100	100	100	67	57	44	40	72	0.327	0.202	0.277	0.148	0.018	0.165	0.137	0.148	1.292	0.188
TRIALLATE(AVADEXBW)	0	13	10	0	0	0	89	0	0	0	11		0.016	0.006				0.069			
DICAMBA(BANVEL)	0	0	0	0	0	0	0	0	0	0	0										
CLOPYRALID(LONTREL)	0	13	20	0	0	75	67	43	33	40	29		0.068	0.085			0.204	0.186	0.035	0.083	0.118
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	11	0	1									0.006	
BROMOXYNIL	50	50	20	0	0	0	0	43	22	10	20	0.016	0.022	0.037					0.016	0.093	0.044
ATRAZINE	0	0	0	0	0	0	0	0	0	0	0										
MCPP(MECOPROP)	0	13	0	0	0	0	0	0	0	10	2		0.005								0.007
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0	0										
2,4-DB	0	0	10	0	0	0	0	0	0	0	1			0.005							
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0	0										
BROMACIL	0	0	0	0	0	0	0	0	0	0	0										
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0	0										
DIAZINON	0	0	0	0	0	0	0	0	0	0	0										
DIURON	0	0	0	0	0	0	0	0	0	0	0										
ETHALFLURALIN(EDGE)	50	0	0	0	0	0	0	0	0	0	5	0.009									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	50	13	0	0	0	0	0	0	0	0	6	0.011	0.010								
IMAZAMETHABENZ-METHYL	0	13	30	0	33	0	56	57	11	40	24		0.190	0.110		0.218		0.357	0.797	0.390	0.737
PICLORAM(TORDON)	0	25	60	100	67	50	56	0	0	0	36		0.279	0.392	0.211	0.289	0.011	0.056			
PYRIDABEN		0	0	0	0	0	0	0	0	0	0										
QUINCLORAC		0	0	0	0	0	0	0	0	10	1										0.046
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0	0										
IMAZETHAPYR		0	0	0	0	0	0	29	0	0	3								0.146		
IMAZAMOX			0	0	0	0	0	0	0	0	0										
SIMAZINE						0	0	0	0	0	0										
TRICLOPYR						0	0	0	0	0	0										
2,4-DICHLOROPHENOL						0	0	0	0	0	0										
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0	0										
BENTAZON							0	0		0	0										
FLUROXYPYR							0	0		10	3										0.103
ETHOFUMESATE							0	0		0	0										
METALAXYL-M							0	0		0	0										
CLODINAFOP ACID METABOLITE							0	0		0	0										
IPRODIONE							0	0		30	10										0.236
METRIBUZIN							0	0		0	0										
OXYCARBOXIN								0	0	0	0										
VINCLOZOLIN								0	0	0	0										
METHOMYL									0	0	0										
GLYPHOSATE(ROUNDUP)						25	57	29		22	33						0.309	2.626	1.266		3.181
AMINOMETHYLPHOSPHONICACID						0	0	0		11	3										1.032
GLUFOSINATE						0	0	0		0	0										
Number of compounds detected*	5	9	8	3	4	4	6	6	6	9	16										

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.17. Detection frequency and maximum concentration of 42 pesticide compounds in Rose Creek, 1997 to 2006.

Pesticide Compound	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs Mean	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)											Maximum Concentration (µg L^{-1})									
Number of samples analyzed	**8**	**15**	**15**	**14**	**12**	**10**	**10**	**13**	**17**	**13**		**8**	**15**	**15**	**14**	**12**	**10**	**10**	**13**	**17**	**13**
2,4-D	0	20	13	29	0	30	10	15	24	31	17		0.019	0.01	0.128		0.05	0.015	0.007	0.029	0.076
MCPA	13	27	13	36	17	0	10	8	0	0	12	0.010	0.032	0.017	1.005	0.011		0.016	0.006		
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0	0	0										
DICAMBA(BANVEL)	0	0	0	0	0	0	0	0	0	0	0										
CLOPYRALID(LONTREL)	0	0	7	0	0	0	0	0	0	0	1			0.049							
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0	0	0										
BROMOXYNIL	0	7	0	0	0	0	10	0	0	0	2		0.008					0.005			
ATRAZINE	0	0	0	0	0	0	0	0	0	0	0										
MCPP(MECOPROP)	0	0	0	0	0	0	0	0	0	0	0										
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0	0										
2,4-DB	0	0	0	0	0	0	0	0	0	0	0										
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0	0										
BROMACIL	0	0	0	0	0	0	0	0	0	0	0										
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0	0										
DIAZINON	0	0	0	0	0	0	0	0	0	0	0										
DIURON	0	0	0	0	8	0	0	0	0	0	1					0.387					
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0	0										
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	7	0	0	0	0	0	0	0	1			0.010							
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0	0	0										
PICLORAM(TORDON)	0	27	0	50	33	10	10	15	6	15	17		0.160		0.327	0.076	0.014	0.049	0.039	0.028	0.075
PYRIDABEN		0	0	0	0	0	0	0	0	0	0										
QUINCLORAC		0	0	0	0	0	0	0	0	0	0										
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0	0										
IMAZETHAPYR		0	7	0	0	0	0	0	0	0	1			0.079							
IMAZAMOX			0	0	0	0	0	0	0	0	0										
SIMAZINE						0	0	0	0	0	0										
TRICLOPYR						30	0	0	6	46	16						0.079			0.010	0.394
2,4-DICHLOROPHENOL						0	0	0	0	0	0										
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0	0										
BENTAZON							0			0	0										
FLUROXYPYR							0			0	0										
ETHOFUMESATE							0			0	0										
METALAXYL-M							0			0	0										
CLODINAFOP ACID METABOLITE							0			0	0										
IPRODIONE							0			0	0										
METRIBUZIN							0			0	0										
OXYCARBOXIN								0	0	0	0										
VINCLOZOLIN								0	0	0	0										
METHOMYL									0	0	0										
GLYPHOSATE(ROUNDUP)						0	22	0		25	12							0.330			6.356
AMINOMETHYLPHOSPHONICACID						0	0	0		0	0										
GLUFOSINATE						0	0	0		8	2										7.801
Number of compounds detected*	1	4	5	3	3	3	4	3	3	3	9										

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.18. Detection frequency and maximum concentration of 42 pesticide compounds in Stretton Creek, 1997 to 2006.

	1997	1998	1999	2000	2004	2005	2006	All Yrs	1997	1998	1999	2000	2004	2005	2006
Pesticide Compound	Detection Frequency (%)							Mean	Maximum Concentration (µg L^{-1})						
Number of samples analyzed	**1**	**5**	**5**	**6**	**3**	**6**	**5**		**1**	**5**	**5**	**6**	**3**	**6**	**5**
2,4-D	100	80	100	67	100	83	80	87	0.049	0.076	0.086	0.054	0.028	0.199	0.04
MCPA	100	20	80	100	100	50	40	70	0.018	0.028	0.020	0.079	0.010	0.019	0.013
TRIALLATE(AVADEXBW)	0	0	20	17	0	0	0	5			0.007	0.011			
DICAMBA(BANVEL)	0	0	0	0	67	0	0	10					0.010		
CLOPYRALID(LONTREL)	100	0	0	0	0	0	0	14	0.026						
DICHLORPROP(2,4-DP)	0	0	20	0	0	0	0	3			0.024				
BROMOXYNIL	0	0	0	0	0	0	0	0							
ATRAZINE	0	0	0	0	0	0	0	0							
MCPP(MECOPROP)	0	0	20	17	0	0	0	5			0.007	0.012			
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0							
2,4-DB	0	0	0	0	0	0	0	0							
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0							
BROMACIL	0	0	0	0	0	0	0	0							
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0							
DIAZINON	0	0	0	0	0	0	0	0							
DIURON	0	0	0	0	0	0	0	0							
ETHALFLURALIN(EDGE)	100	0	0	0	0	0	0	14	0.015						
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	20	0	0	0	0	3			0.012				
IMAZAMETHABENZ-METHYL	100	80	0	17	100	0	20	45	0.212	0.540		0.479	0.657		0.268
PICLORAM(TORDON)	0	0	40	17	0	0	20	11			0.077	0.016			0.017
PYRIDABEN		0	0	0	0	0	0	0							
QUINCLORAC		0	0	0	0	0	0	0							
TRIFLURALIN(TREFLAN)	100	20	80	0	0	0	0	29	0.025	0.068	0.187				
IMAZETHAPYR		75	40	0	0	0	0	19		0.409	0.053				
IMAZAMOX			0	0	0	0	0	0							
SIMAZINE					0	0	0	0							
TRICLOPYR					0	100	0	33						0.047	
2,4-DICHLOROPHENOL					0	0	0	0							
4-CHLORO-2-METHYLPHENOL					0	0	0	0							
BENTAZON							0	0							
FLUROXYPYR							0	0							
ETHOFUMESATE							0	0							
METALAXYL-M							0	0							
CLODINAFOP ACID METABOLITE							0	0							
IPRODIONE							40	40							0.202
METRIBUZIN							0	0							
OXYCARBOXIN					0	0	0	0							
VINCLOZOLIN					0	17	20	12						0.261	0.052
METHOMYL						0	0	0							
GLYPHOSATE(ROUNDUP)					33			33					0.942		
AMINOMETHYLPHOSPHONICACID					0			0							
GLUFOSINATE					0			0							
Number of compounds detected*	6	5	9	6	4	4	6	16							

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.19. Detection frequency and maximum concentration of 42 pesticide compounds in Strawberry Creek, 1997 to 2006.

Pesticide Compound	Detection Frequency (%) 1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs Mean	Maximum Concentration (µg L^{-1}) 1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Number of samples analyzed	**3**	**7**	**12**	**16**	**6**	**6**	**7**	**10**	**11**	**6**		**3**	**7**	**12**	**16**	**6**	**6**	**7**	**10**	**11**	**6**
2,4-D	100	71	58	69	83	67	57	40	27	83	66	0.05	0.536	0.671	2.075	0.078	0.063	0.05	0.071	0.017	0.139
MCPA	33	29	17	50	33	0	71	10	9	17	27	0.163	0.040	0.054	0.760	0.019		0.054	0.053	0.008	0.071
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0	0	0										
DICAMBA(BANVEL)	0	0	8	6	0	50	0	0	0	0	6			0.024	0.019		0.010				
CLOPYRALID(LONTREL)	0	0	8	6	33	0	0	0	0	33	8			0.026	0.042	0.044					0.323
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0	0	0										
BROMOXYNIL	33	14	8	13	0	0	0	10	0	0	8	0.009	0.008	0.009	0.018				0.020		
ATRAZINE	0	0	0	0	0	0	0	0	0	0	0										
MCPP(MECOPROP)	0	0	0	6	0	0	0	10	0	0	2				0.022				0.010		
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0	0										
2,4-DB	0	0	0	0	0	0	0	0	0	0	0										
ALPHA-BENZENEHEXACHLORIDE	0	0	8	0	0	0	0	0	0	0	1			0.005							
BROMACIL	0	0	0	0	0	0	0	0	0	0	0										
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0	0										
DIAZINON	0	0	0	0	0	0	0	0	0	0	0										
DIURON	0	0	0	0	0	0	0	0	0	0	0										
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0	0										
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0	0										
IMAZAMETHABENZ-METHYL	0	0	0	6	0	0	0	0	0	0	1				0.165						
PICLORAM(TORDON)	67	57	50	100	100	83	57	20	9	50	59	0.055	0.406	0.158	0.291	0.322	0.105	0.157	0.072	0.013	0.099
PYRIDABEN		0	0	0	0	0	0	0	0	0	0										
QUINCLORAC		0	0	0	0	0	0	0	0	0	0										
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0	0										
IMAZETHAPYR		0	0	0	0	0	0	0	0	0	0										
IMAZAMOX			0	0	0	0	0	0	0	0	0										
SIMAZINE						0	0	0	0	0	0										
TRICLOPYR						67	57	20	82	50	55						0.084	0.034	0.048	0.152	0.079
2,4-DICHLOROPHENOL						0	0	0	0	0	0										
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0	0										
BENTAZON							0			0	0										
FLUROXYPYR							0			0	0										
ETHOFUMESATE							0			0	0										
METALAXYL-M							0			0	0										
CLODINAFOP ACID METABOLITE							0			0	0										
IPRODIONE							0			0	0										
METRIBUZIN							0			0	0										
OXYCARBOXIN								0	0	0	0										
VINCLOZOLIN								0	0	0	0										
METHOMYL									0	0	0										
GLYPHOSATE(ROUNDUP)										100	100										1.813
AMINOMETHYLPHOSPHONICACID										0	0										
GLUFOSINATE										0	0										
Number of compounds detected*	4	4	7	8	4	4	4	6	4	5	10										

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.20. Detection frequency and maximum concentration of 42 pesticide compounds in Threehills Creek, 1997 to 2006.

Pesticide Compound	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs Mean	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)											Maximum Concentration (µg L^{-1})									
Number of samples analyzed	**6**	**9**	**11**	**12**	**7**	**5**	**9**	**9**	**8**	**10**		**6**	**9**	**11**	**12**	**7**	**5**	**9**	**9**	**8**	**10**
2,4-D	67	56	45	58	71	40	11	56	63	60	53	0.077	0.041	0.077	0.109	0.09	0.008	0.024	0.043	0.069	0.071
MCPA	50	56	91	58	86	40	44	78	50	60	61	0.135	0.146	0.229	0.395	0.134	0.011	0.129	0.077	0.091	0.149
TRIALLATE(AVADEXBW)	17	0	36	0	0	20	0	11	25	0	11	0.074		0.071			0.096		0.020	0.042	
DICAMBA(BANVEL)	0	0	0	8	0	0	11	11	13	0	4				0.005			0.006	0.005	0.005	
CLOPYRALID(LONTREL)	0	0	9	0	43	100	44	22	25	30	27			0.182		0.047	0.100	0.207	0.085	0.041	0.071
DICHLORPROP(2,4-DP)	17	11	0	8	0	0	0	0	0	0	4	0.006	0.006		0.074						
BROMOXYNIL	17	22	18	0	0	0	0	22	13	0	9	0.006	0.005	0.081					0.012	0.005	
ATRAZINE	0	0	0	0	0	0	0	0	0	0	0										
MCPP(MECOPROP)	0	0	0	8	0	0	0	0	0	0	1				0.016						
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0	0										
2,4-DB	0	0	0	0	0	0	0	0	0	0	0										
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0	0										
BROMACIL	0	0	0	0	0	0	0	0	0	0	0										
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0	0										
DIAZINON	0	0	0	0	14	0	0	0	0	0	1					0.012					
DIURON	0	0	0	0	0	0	0	0	0	0	0										
ETHALFLURALIN(EDGE)	33	0	0	0	0	0	0	0	0	0	3	0.029									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0	0										
IMAZAMETHABENZ-METHYL	17	11	91	25	14	0	44	67	38	50	36	1.853	0.096	9.005	1.140	0.099		0.363	2.934	0.983	0.406
PICLORAM(TORDON)	0	11	36	8	71	20	0	0	0	0	15		0.009	0.052	0.011	0.536	0.024				
PYRIDABEN		0	0	0	0	0	0	0	0	0	0										
QUINCLORAC		0	0	0	0	0	0	0	0	0	0										
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0	0										
IMAZETHAPYR		0	18	0	0	0	0	0	0	0	2			0.054							
IMAZAMOX			0	0	0	0	0	0	0	0	0										
SIMAZINE						0	0	0	0	0	0										
TRICLOPYR						0	0	0	0	0	0										
2,4-DICHLOROPHENOL						0	0	0	0	0	0										
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0	0										
BENTAZON							0	0		0	0										
FLUROXYPYR							0	0		0	0										
ETHOFUMESATE							0	0		0	0										
METALAXYL-M							0	0		0	0										
CLODINAFOP ACID METABOLITE							0	0		10	3										0.024
IPRODIONE							0	0		0	0										
METRIBUZIN							0	0		0	0										
OXYCARBOXIN								0	0	0	0										
VINCLOZOLIN								0	0	0	0										
METHOMYL									0	0	0										
GLYPHOSATE(ROUNDUP)						0	25	0		0	6							0.966			
AMINOMETHYLPHOSPHONICACID						0	0	0		0	0										
GLUFOSINATE						0	0	0		0	0										
Number of compounds detected*	7	6	8	7	6	5	5	7	7	5	14										

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.21. Detection frequency and maximum concentration of 42 pesticide compounds in Tomahawk Creek, 1997 to 2006.

	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Pesticide Compound	Detection Frequency (%)										Mean	Maximum Concentration ($\mu g\ L^{-1}$)									
Number of samples analyzed	**3**	**13**	**12**	**12**	**10**	**5**	**8**	**9**	**18**	**11**		**3**	**13**	**12**	**12**	**10**	**5**	**8**	**9**	**18**	**11**
2,4-D	67	69	58	75	20	60	88	78	56	45	62	0.062	0.089	0.326	0.467	0.039	0.276	1.668	0.672	0.571	0.053
MCPA	0	23	8	8	0	0	25	0	6	0	7		0.013	0.014	0.019			0.018		0.009	
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0	0	0										
DICAMBA(BANVEL)	0	0	0	8	0	0	0	22	6	0	4				0.007				1.134	0.008	
CLOPYRALID(LONTREL)	0	0	0	0	10	0	0	0	0	0	1					0.023					
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0	0	0										
BROMOXYNIL	0	0	8	0	0	0	0	0	0	0	1			0.005							
ATRAZINE	0	0	0	0	0	0	0	0	0	0	0										
MCPP(MECOPROP)	0	0	0	0	0	0	13	0	0	0	1							0.013			
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0	0										
2,4-DB	0	0	0	0	0	0	0	0	0	0	0										
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0	0										
BROMACIL	0	0	0	0	0	0	0	0	0	0	0										
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0	0										
DIAZINON	0	0	0	0	0	0	0	0	0	0	0										
DIURON	0	0	0	0	0	0	0	0	0	0	0										
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0	0										
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0	0										
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0	0	0										
PICLORAM(TORDON)	100	31	17	83	10	0	0	0	0	0	24	0.032	0.094	0.384	0.131	0.077					
PYRIDABEN		0	0	0	0	0	0	0	0	0	0										
QUINCLORAC		0	0	0	0	0	0	0	0	0	0										
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0	0										
IMAZETHAPYR		0	0	0	0	0	0	0	0	0	0										
IMAZAMOX			0	0	0	0	0	0	0	0	0										
SIMAZINE						0	0	0	0	0	0										
TRICLOPYR						0	75	0	0	64	28							0.186			0.387
2,4-DICHLOROPHENOL						0	0	11	6	0	3								0.046	0.012	
4-CHLORO-2-METHYLPHENOL						0	0	0	0	0	0										
BENTAZON							0			0	0										
FLUROXYPYR							0			0	0										
ETHOFUMESATE							0			0	0										
METALAXYL-M							0			0	0										
CLODINAFOP ACID METABOLITE							0			0	0										
IPRODIONE							0			0	0										
METRIBUZIN							0			0	0										
OXYCARBOXIN								0	0	0	0										
VINCLOZOLIN								0	0	0	0										
METHOMYL									0	0	0										
GLYPHOSATE(ROUNDUP)																					
AMINOMETHYLPHOSPHONICACID																					
GLUFOSINATE																					
Number of compounds detected*	2	3	4	4	3	1	4	3	4	2	9										

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.22. Detection frequency and maximum concentration of 42 pesticide compounds in Trout Creek, 1997 to 2006.

Pesticide Compound	1997	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs	1997	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)									Mean	Maximum Concentration (µg L^{-1})								
Number of samples analyzed	**2**	**6**	**8**	**9**	**10**	**11**	**13**	**10**	**12**		**2**	**6**	**8**	**9**	**10**	**11**	**13**	**10**	**12**
2,4-D	0	17	13	22	0	0	8	10	0	8		0.026	0.023	0.01			0.017	0.005	
MCPA	0	17	13	11	0	0	0	0	0	4		0.005	0.006	0.005					
TRIALLATE(AVADEXBW)	0	0	13	0	0	0	0	0	0	1			0.009						
DICAMBA(BANVEL)	0	0	0	0	0	0	0	0	0	0									
CLOPYRALID(LONTREL)	0	0	0	0	0	0	0	0	0	0									
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0	0									
BROMOXYNIL	0	0	0	0	0	0	0	0	0	0									
ATRAZINE	0	0	0	0	0	0	0	0	0	0									
MCPP(MECOPROP)	0	0	13	0	0	0	0	0	0	1			0.012						
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0	0									
2,4-DB	0	0	0	0	0	0	0	0	0	0									
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0	0									
BROMACIL	0	0	0	0	0	0	0	0	0	0									
DESISOPROPYLATRAZINE		0	0	0	0	0	0	0	0	0									
DIAZINON	0	0	0	0	0	0	0	0	0	0									
DIURON	0	0	0	0	0	0	0	0	0	0									
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0	0									
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0	0									
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0	0									
PICLORAM(TORDON)	0	0	0	33	10	0	0	0	0	5				0.022	0.005				
PYRIDABEN		0	0	0	0	0	0	0	0	0									
QUINCLORAC		0	0	0	0	0	0	0	0	0									
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0	0									
IMAZETHAPYR		0	0	0	0	0	0	0	0	0									
IMAZAMOX		0	0	0	0	0	0	0	0	0									
SIMAZINE					0	0	0	0	0	0									
TRICLOPYR					10	0	0	0	17	5					0.068				0.012
2,4-DICHLOROPHENOL					0	0	0	0	0	0									
4-CHLORO-2-METHYLPHENOL					0	0	0	0	0	0									
BENTAZON						0	0		0	0									
FLUROXYPYR						0	0		0	0									
ETHOFUMESATE						0	0		0	0									
METALAXYL-M						0	0		0	0									
CLODINAFOP ACID METABOLITE						0	0		0	0									
IPRODIONE						0	0		0	0									
METRIBUZIN						0	0		0	0									
OXYCARBOXIN							0	0	0	0									
VINCLOZOLIN							0	0	0	0									
METHOMYL								0	0	0									
GLYPHOSATE(ROUNDUP)																			
AMINOMETHYLPHOSPHONICACID																			
GLUFOSINATE																			
Number of compounds detected*	0	2	4	3	2	0	1	1	1	6									

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.23. Detection frequency and maximum concentration of 42 pesticide compounds in Wabash Creek, 1999 to 2006.

Pesticide Compound	Detection Frequency (%) 1999	2000	2001	2002	2003	2004	2005	2006	All Yrs Mean	Maximum Concentration (µg L^{-1}) 1999	2000	2001	2002	2003	2004	2005	2006
Number of samples analyzed	**6**	**8**	**3**	**4**	**4**	**10**	**16**	**9**		**6**	**8**	**3**	**4**	**4**	**10**	**16**	**9**
2,4-D	100	88	67	100	100	90	88	89	90	0.107	3.299	2.184	0.981	0.046	0.415	0.444	0.146
MCPA	67	88	67	75	100	90	44	33	70	0.010	0.441	0.037	0.246	0.103	0.048	0.027	0.033
TRIALLATE(AVADEXBW)	0	0	0	50	50	10	0	11	15				0.340	0.037	0.010		0.008
DICAMBA(BANVEL)	17	75	67	50	75	40	19	0	43	0.012	0.331	0.381	0.055	0.018	0.033	0.016	
CLOPYRALID(LONTREL)	0	25	67	75	100	60	19	33	47		0.040	0.208	0.206	0.082	0.060	0.025	0.039
DICHLORPROP(2,4-DP)	0	0	0	0	0	10	0	0	1						0.122		
BROMOXYNIL	0	0	0	0	0	0	0	0	0								
ATRAZINE	0	0	0	0	0	0	0	0	0								
MCPP(MECOPROP)	100	100	100	75	75	100	56	100	88	0.036	2.068	0.180	0.259	0.045	0.138	0.091	0.044
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0								
2,4-DB	0	0	0	0	0	0	0	0	0								
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0								
BROMACIL	17	13	33	25	0	0	0	0	11	0.124	0.191	0.065	0.297				
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0								
DIAZINON	0	0	0	0	0	0	0	0	0								
DIURON	0	0	33	0	0	10	0	0	5			0.616			0.249		
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0								
AMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0								
IMAZAMETHABENZ-METHYL	0	0	0	25	0	50	25	11	14				0.061		0.590	0.394	0.303
PICLORAM(TORDON)	100	88	100	75	100	50	0	100	77	0.429	1.147	1.355	0.182	0.280	0.155		0.783
PYRIDABEN	0	0	0	0	0	0	0	0	0								
QUINCLORAC	0	0	0	0	0	0	0	0	0								
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0								
IMAZETHAPYR	0	0	0	0	0	0	0	0	0								
IMAZAMOX	0	0	0	0	0	0	0	0	0								
SIMAZINE				0	0	0	0	0	0								
TRICLOPYR				25	75	20	63	11	39				0.010	0.057	0.013	0.090	0.029
2,4-DICHLOROPHENOL				0	0	10	0	0	2						0.019		
4-CHLORO-2-METHYLPHENOL				0	0	0	0	0	0								
BENTAZON								0	0								
FLUROXYPYR								0	0								
ETHOFUMESATE								0	0								
METALAXYL-M								0	0								
CLODINAFOP ACID METABOLITE								0	0								
IPRODIONE								11	11								0.365
METRIBUZIN								0	0								
OXYCARBOXIN						0	0	11	4								0.404
VINCLOZOLIN						0	0	0	0								
METHOMYL							0	0	0								
GLYPHOSATE(ROUNDUP)								0	0								
AMINOMETHYLPHOSPHONICACID								0	0								
GLUFOSINATE								0	0								
Number of compounds detected*	6	7	8	10	8	12	7	10	15								

*Does not include Glyphosate, AMPA or Glufosinate

Table A13.24. Detection frequency and maximum concentration of 42 pesticide compounds in Willow Creek, 1999 to 2006.

Pesticide Compound	1999	2000	2001	2002	2003	2004	2005	2006	All Yrs Mean	1999	2000	2001	2002	2003	2004	2005	2006
	Detection Frequency (%)									Maximum Concentration ($\mu g\ L^{-1}$)							
Number of samples analyzed	**6**	**12**	**11**	**10**	**10**	**12**	**10**	**9**		**6**	**12**	**11**	**10**	**10**	**12**	**10**	**9**
2,4-D	0	17	0	0	0	0	0	0	2		0.102						
MCPA	0	0	9	0	0	0	0	0	1			0.021					
TRIALLATE(AVADEXBW)	0	0	0	0	0	0	0	0	0								
DICAMBA(BANVEL)	0	8	9	0	0	0	0	0	2		0.019	0.008					
CLOPYRALID(LONTREL)	0	0	0	0	0	0	0	0	0								
DICHLORPROP(2,4-DP)	0	0	0	0	0	0	0	0	0								
BROMOXYNIL	0	0	0	0	0	0	0	0	0								
ATRAZINE	0	0	0	0	0	0	0	0	0								
MCPP(MECOPROP)	0	8	0	0	0	0	0	0	1		0.009						
CHLORPYRIFOS-ETHYL(DURSBAN)	0	0	0	0	0	0	0	0	0								
2,4-DB	0	0	0	0	0	0	0	0	0								
ALPHA-BENZENEHEXACHLORIDE	0	0	0	0	0	0	0	0	0								
BROMACIL	0	0	0	0	0	0	0	0	0								
DESISOPROPYLATRAZINE	0	0	0	0	0	0	0	0	0								
DIAZINON	0	0	0	0	0	0	0	0	0								
DIURON	0	0	0	0	0	0	0	0	0								
ETHALFLURALIN(EDGE)	0	0	0	0	0	0	0	0	0								
GAMMA-BENZENEHEXACHLORIDE(LINDANE)	0	0	0	0	0	0	0	0	0								
IMAZAMETHABENZ-METHYL	0	0	0	0	0	0	0	0	0								
PICLORAM(TORDON)	0	17	9	0	0	0	0	0	3		0.029	0.016					
PYRIDABEN	0	0	0	0	0	0	0	0	0								
QUINCLORAC	0	0	0	0	0	0	0	0	0								
TRIFLURALIN(TREFLAN)	0	0	0	0	0	0	0	0	0								
IMAZETHAPYR	0	0	0	0	0	0	0	0	0								
IMAZAMOX	0	0	0	0	0	0	0	0	0								
SIMAZINE				0	0	0	0	0	0								
TRICLOPYR				0	0	0	0	0	0								
2,4-DICHLOROPHENOL				0	0	0	0	0	0								
4-CHLORO-2-METHYLPHENOL				0	0	0	0	0	0								
BENTAZON						0		0	0								
FLUROXYPYR						0		0	0								
ETHOFUMESATE						0		0	0								
METALAXYL-M						0		0	0								
CLODINAFOP ACID METABOLITE						0		0	0								
IPRODIONE						0		0	0								
METRIBUZIN						0		0	0								
OXYCARBOXIN						0	0	0	0								
VINCLOZOLIN						0	0	0	0								
METHOMYL							0	0	0								
GLYPHOSATE(ROUNDUP)																	
AMINOMETHYLPHOSPHONICACID																	
GLUFOSINATE																	
Number of compounds detected*	0	4	3	0	0	0	0	0	5								

*Does not include Glyphosate, AMPA or Glufosinate

APPENDIX 14: PESTICIDE MOBILITY TABLE

Table A14.1. Pesticide mobility table.

Common Name	Pesticide Movement Rating	Soil Half-life (days)	Water Solubility (mg/l)	Sorption Coefficient (soil Koc)
2,4-D acid	Moderate	10	890	20
2,4-DB acid	Very Low	5	46	440
Aldicarb	High	30	6000	30
Aldrin	Very Low	365	0.027	5000
Atrazine	High	60	33	100
Azinphos-methyl	Low	10	29	1000
Bentazon sodium salt	High	20	2,300,000	34
Bromacil acid	Very High	60	700	32
Bromacil lithium salt	Very High	60	700	32
Bromoxynil butyrate ester	Very Low	7	27	1079
Bromoxynil octanoate ester	Extremely Low	7	0.08	10,000
Chlorothalonil	Low	30	0.6	1380
Chlorpyrifos	Very Low	30	0.4	6070
Chlorpyrifos-methyl	Very Low	7	4	3000
Clopyralid amine salt	Very High	40	300,000	6
Cyanazine	Low	14	170	190
Diazinon	Low	40	60	1000
Dicamba salt	Very High	14	400,000	2
Dichlorprop (2,4-DP) ester	Low	10	50	1000
Diclofop-methyl	Extremely Low	30	0.8	16,000
Dieldrin	Extremely Low	1000	0.2	12,000
Dimethoate	Moderate	7	39,800	20
Disulfoton	Low	30	25	600
Diuron	Moderate	90	42	480
Endosulfan	Extremely Low	50	0.32	12,400
Ethalfluralin	Very Low	60	0.3	4000
Ethion	Extremely Low	150	1.1	10,000
Ethofumesate	Moderate	30	50	340
Fenoxaprop-ethyl	Extremely Low	9	0.8	9490
Fluazifop-butyl	Very Low	21	2	3000
Fluazifop-p-butyl	Very Low	15	2	5700
Imazamethabenz-methyl(m-isomer)	High	45	1370	66
Imazamethabenz-methyl(p-isomer)	Very High	45	857	35
Imazethapyr	Very High	90	200,000	10
Iprodione	Low	14	13.9	700
Lindane	Moderate	400	7	1100
Linuron	Moderate	60	75	400
Malathion	Extremely Low	1	130	1800
MCPA dimethylamine salt	High	25	866,000	20
MCPA ester	Low	25	5	1000
MCPB sodium salt	High	14	200,000	20
Mecoprop (MCPP) dimethylamine salt	High	21	660,000	20
Metalaxyl	Very High	70	8400	50
Methomyl	High	30	58,000	72
Methoxychlor	Extremely Low	120	0.1	80,000
Metolachlor	High	90	530	200
Metribuzin	High	40	1220	60
Napropamide	Moderate	70	74	700
Oxycarboxin	Moderate	20	1000	95
Parathion (ethyl parathion)	Very Low	14	24	5000
Phorate	Low	60	22	1000
Picloram salt	Very High	90	200,000	16
Propiconazole	Moderate	110	110	650
Quizalofop-ethyl	Moderate	60	0.31	510
Simazine	High	60	6.2	130
Terbufos	Very Low	5	5	500
Triallate	Low	82	4	2400
Triclopyr amine salt	Very High	46	2,100,000	20
Triclopyr ester	Low	46	23	780
Trifluralin	Very Low	60	0.3	8000
Vinclozolin	Moderate	20	1000	100

Source: Wauchope et al. 1992; Augustijn-Beckers et al. 1994; Cotton 1995.

APPENDIX 15: MEDIAN CONCENTRATION AND NUMBER OF SAMPLES FOR DRYLAND AND/OR IRRIGATED WATERSHEDS

Table A15.1. Median concentration (µg L^{-1}) plus number of samples in parenthesis for a variety of compounds detected in both dryland and irrigated watersheds.

Watershed	2,4-D	MCPA	Picloram	MCPP	Clopyralid	Dicamba	Triclopyr	Bromoxynil	Triallate	Imazamethabenz-methyl	Dichlorprop	Imazethapyr	2,4-Dichlorophenol	Fluroxypyr	Atrazine	Diazinon	Bentazon	2,4-DB	Imazamox	Vinclozolin
Low intensity dryland																				
Hines Creek	0.045 (2)	0.012 (2)					0.099 (4)													
Paddle River	0.011 (7)	0.006 (3)				0.007 (1)	0.046 (17)													
Prairie Blood Coulee	0.018 (31)	0.025 (21)	0.011 (1)	0.010 (6)	0.024 (6)	0.017 (9)		0.013 (8)		0.121 (1)	0.008 (3)									
Rose Creek	0.010 (20)	0.017 (11)	0.032 (18)		0.049 (1)		0.028 (10)	0.005 (1)				0.079 (1)								
Willow Creek	0.055 (2)	0.021 (1)	0.016 (3)	0.009 (1)		0.014 (2)														
Moderate intensity dryland																				
Blindman River	0.013 (34)	0.008 (16)	0.057 (21)	0.024 (1)	0.011 (2)	0.055 (17)	0.006 (5)													
Grande Prairie Creek	0.017 (15)	0.014 (31)	0.232 (6)	0.022 (7)	0.066 (22)	0.058 (10)	0.061 (13)		0.022 (12)	0.237 (10)		0.077 (5)		0.054 (1)						
Kleskun Drain	0.039 (34)	0.036 (17)	0.564 (37)	0.029 (2)	0.133 (32)	0.025 (6)	0.406 (2)	0.011 (1)	0.223 (2)	0.163 (7)		0.105 (3)	0.023 (1)		0.070 (1)					
Meadow Creek	0.014 (21)	0.009 (6)	0.062 (1)					0.011 (1)												
Tomahawk Creek	0.027 (50)	0.014 (5)	0.077 (13)	0.013 (1)	0.023 (1)	0.073 (4)	0.084 (13)	0.005 (1)					0.029 (2)							
Trout Creek	0.014 (6)	0.005 (3)	0.017 (4)	0.012 (1)			0.012 (3)		0.009 (1)											

Table A15.1. cont.

Watershed	2,4-D	MCPA	Picloram	MCPP	Clopyralid	Dicamba	Triclopyr	Bromoxynil	Triallate	Imazamethabenz-methyl	Dichlorprop	Imazethapyr	2,4-Dichlorophen,	Fluroxypyr	Atrazine	Diazinon	Bentazon	2,4-DB	Imazamox	Vinclozolin
High intensity dryland																				
Buffalo Creek	0.013 (29)	0.021 (13)			0.020 (1)		0.018 (5)	0.006 (3)		0.246 (2)	0.007 (2)									
Haynes Creek (M6)	0.058 (44)	0.022 (33)	0.050 (20)	0.030 (6)	0.063 (35)	0.031 (5)	0.028 (8)	0.005 (1)	0.005 (1)	0.242 (26)	0.018 (1)	0.046 (3)				0.005 (1)	0.015 (1)			
Ray Creek	0.014 (25)	0.036 (32)	0.018 (9)	0.010 (2)	0.040 (12)	0.016 (2)	0.032 (1)	0.056 (2)	0.077 (5)	0.278 (28)	0.010 (1)	0.051 (1)				0.008 (1)				
Renwick Creek	0.017 (39)	0.024 (36)	0.059 (20)	0.007 (1)	0.053 (21)			0.019 (8)	0.011 (9)	0.328 (18)	0.006 (1)	0.135 (2)		0.103 (1)				0.005 (1)		
Stretton Creek	0.031 (21)	0.013 (18)	0.023 (4)	0.010 (2)		0.010 (2)	0.030 (6)		0.009 (2)	0.410 (5)	0.024 (1)	0.053 (2)								0.157 (2)
Strawberry Creek	0.025 (43)	0.032 (20)	0.065 (43)	0.016 (2)	0.034 (6)	0.010 (5)	0.043 (22)	0.014 (4)		0.165 (1)										
Threehills Creek	0.023 (36)	0.023 (46)	0.024 (11)	0.016 (1)	0.044 (20)	0.005 (4)		0.012 (5)	0.031 (8)	0.314 (32)	0.074 (1)	0.048 (2)				0.012 (1)				
Wabash Creek	0.054 (54)	0.016 (39)	0.111 (37)	0.031 (51)	0.039 (23)	0.017 (21)	0.024 (17)		0.015 (6)	0.276 (11)	0.122 (1)		0.019 (1)							
High intensity irrigation																				
Battersea Drain	0.034 (68)	0.014 (32)		0.018 (4)	0.044 (13)	0.012 (12)		0.009 (6)	0.011 (18)		0.015 (6)		0.074 (1)	0.026 (3)	0.014 (5)		0.030 (5)			
Crowfoot Creek	0.061 (98)	0.014 (45)	0.021 (10)	0.015 (59)	0.034 (9)	0.021 (63)	0.029 (3)	0.013 (14)	0.019 (17)	0.280 (15)	0.012 (5)	0.025 (1)	0.045 (4)	0.027 (3)	0.013 (10)	0.041 (1)			0.063 (1)	
Drain S6	0.039 (52)	0.011 (26)	0.025 (6)	0.050 (3)	0.081 (2)	0.016 (26)	0.058 (13)	0.019 (4)	0.055 (2)		0.037 (2)		0.469 (1)	0.073 (1)	0.057 (3)		0.035 (7)			0.011 (1)
New West Coulee	0.073 (74)	0.014 (39)	0.045 (3)	0.029 (8)	0.039 (15)	0.026 (40)		0.059 (11)	0.014 (13)		0.023 (18)		0.045 (5)	0.030 (7)	0.021 (2)		0.045 (4)	0.665 (1)		
# of streams with detections	**23**	**23**	**19**	**18**	**17**	**17**	**16**	**16**	**13**	**12**	**12**	**9**	**7**	**6**	**5**	**4**	**4**	**2**	**2**	**2**

Table A15.2. Median concentration (μg L^{-1}) of pesticide compounds detected in either dryland or irrigation watersheds. Number of samples collected is in parenthesis.

	Detected in dryland watersheds only									Detected in irrigated watersheds only							
Watershed	Gamma-benzenehexachloride	Iprodione	Trifluralin	Diruon	Alpha-benzenehexachloride	Bromacil	Oxycarboxin	Clodinafopacidmeta-bolite	Quinclorac	Simazine	Ethofumesate	Ethalfluralin	Chlorpyrifos-ethyl	4-Chloro-2-Methylphenol	Metalaxl-M	Metribuzin	Desisopropylatrazine
Low intensity dryland																	
Hines Creek																	
Paddle River																	
Prairie Blood Coulee	0.012 (1)																
Rose Creek	0.010 (1)			0.387 (1)													
Willow Creek																	
Moderate intensity dryland																	
Blindman River																	
Grande Prairie Creek	0.026 (2)		0.007 (1)														
Kleskun Drain	0.024 (4)		0.013 (3)		0.008 (1)												
Meadow Creek																	
Tomahawk Creek																	
Trout Creek																	

Table A15.2. Continued.

	Detected in dryland watersheds only									Detected in irrigated watersheds only							
Watershed	Gamma-benzene-hexachloride	Iprodione	Trifluralin	Diruon	Alpha-benzenehexa-chloride	Bromacil	Oxycarboxin	Clodinafop-acidmetabolite	Quinclorac	Simazine	Ethofumesate	Ethalfluralin	Chlorpyrifos-ethyl	4-Chloro-2-Methylphenol	Metalaxl-M	Metribuzin	Desisopropyl-atrazine
High intensity dryland																	
Buffalo Creek		0.149 (2)															
Haynes Creek (M6)	0.017 (1)	0.245 (2)	0.005 (2)														
Ray Creek																	
Renwick Creek		0.133 (3)							0.046 (1)								
Stretton Creek	0.012 (1)	0.122 (2)	0.019 (4)														
Strawberry Creek					0.005 (1)												
Threehills Creek								0.024 (1)									
Wabash Creek		0.365 (1)		0.433 (2)		0.158 (4)	0.404 (1)										
High intensity irrigation																	
Battersea Drain										0.052 (6)	0.097 (2)		0.025 (3)		0.422 (2)		
Crowfoot Creek												0.009 (1)					
Drain S6										0.371 (1)	0.065 (4)	0.018 (1)					
New West Coulee										0.094 (22)	0.025 (9)			2.876 (1)		0.010 (1)	0.123 (2)
# of streams with detections	**6**	**5**	**4**	**2**	**2**	**1**	**1**	**1**	**1**	**3**	**3**	**2**	**1**	**1**	**1**	**1**	**1**

APPENDIX 16: PESTICIDE GUIDELINES

Table A16.1. Protection of Aquatic life, agriculture, drinking water and consumption guideline values of a variety of pesticide compounds.

			PAL[z]	AGR[y]	AGR[y]	DRINK	DRINK	US source			
										Human health for the consumption of:	
Parameter Name	Most stringent-Canadian	Most stringent-All	Freshwater (µg L^{-1})	Irrigation (µg L^{-1})	Livestock (µg L^{-1})	MAC[x] (µg L^{-1})	AO or OG[w] (mg L^{-1})	CMC[v] (µg L^{-1})	CCC[u] (µg L^{-1})	Water and organism (µg L^{-1})	Organism only (µg L^{-1})
Aldicarb	1	1	1	54.9	11	9					
Aldrin (+Dieldrin)	~~0.004~~[t]	0.000049	~~0.004~~[t]			0.7		3		0.000049	0.00005
Atrazine	1.8	1.8	1.8	10	5	5					
Atrazine metabolites (DW)	5	5				5					
Azinphos-methyl (Guthion)	20	0.01				20			0.01		
Bromacil	0.2	0.2	5	0.2	1100						
Bromoxynil	0.33	0.33	5	0.33	11	5					
Dichlorophenol, 2,4-	900	77				900	0.3			77	290
Chlorothalonil	0.18	0.18	0.18	5.8	170						
Chlorpyrifos	0.0035	0.0035	0.0035	ID[s]	24	90		0.083	0.041		
Cyanazine	0.5	0.5	2	0.5	10	10					
Diazinon	20	0.17				20		0.17	0.17		
Dicamba	0.006	0.006	10	0.006	122	120					
*2,4-Dichlorophenoxyacetic	100	100				100					
Alpha-Endosulfan		0.056						0.22	0.056	62	89
Endosulfan	0.02	0.02	0.02								
Phenoxy herbicides	4	4	4		100					100	
Diclofop-methyl	0.18	0.18	6.1	0.18	9	9					
Dieldrin (+Aldrin)	~~0.004~~[t]	0.000052	~~0.004~~[t]			0.7		0.24	0.056	0.000052	0.000054
Dimethoate	3	3	6.2	ID[s]	3	20					
Diuron	150	150				150					

Table A16.1. Continued

			PAL[z]	AGR[y]	AGR[y]	DRINK	DRINK	US Source			
										Human health for the consumption of:	
Parameter Name	Most stringent-Canadian	Most stringent-All	Freshwater ($\mu g\ L^{-1}$)	Irrigation ($\mu g\ L^{-1}$)	Livestock ($\mu g\ L^{-1}$)	MAC ($\mu g\ L^{-1}$)	AO or OG ($mg\ L^{-1}$)	CMC ($\mu g\ L^{-1}$)	CCC ($\mu g\ L^{-1}$)	Water and organism ($\mu g\ L^{-1}$)	Organism only ($\mu g\ L^{-1}$)
Glyphosate	65	65	65		280	280					
Hexachlorocyclohexane	0.01	0.01	0.01		4						
Linuron	0.071	0.071	7	0.071	ID[s]						
Malathion	190	0.01				190			0.01		
Methylchlorophenoxyacetic acid	0.025	0.025	2.6	0.025	25	TBA[r]					
Methoxychlor	900	0.03				900			0.03	100	
Metolachlor	7.8	7.8	7.8	28	50	50					
Metribuzin	0.5	0.5	1	0.5	80	80					
Parathion	50	0.013				50		0.065	0.013		
Phorate	2	2				2					
Picloram	29	29	29	ID[s]	190	190					
Simazine	0.5	0.5	10	0.5	10	10					
Terbufos	1	1				1					
Triallate	0.24	0.24	0.24	ID[s]	230						
Trifluralin	0.2	0.2	0.2	ID[s]	45	45					

[z]PAL is the Protection of Aquatic life
[y]AGR is agriculture
[x]MAC is maximum acceptable concentration
[w]AO is aesthetic objectives and OG is operational guidance values
[v]CMC is criteria maximum concentration
[u]CCC is criterion continuous concentration
[t]Crossed out values indicate withdrawn guidelines which are no longer recommended (CCME 2007)
[s]ID is insufficient data
[r]TBA is to be announced

APPENDIX 17: SAMPLE NUMBERS FOR NUTRIENTS, BACTERIA, AND PESTICIDES

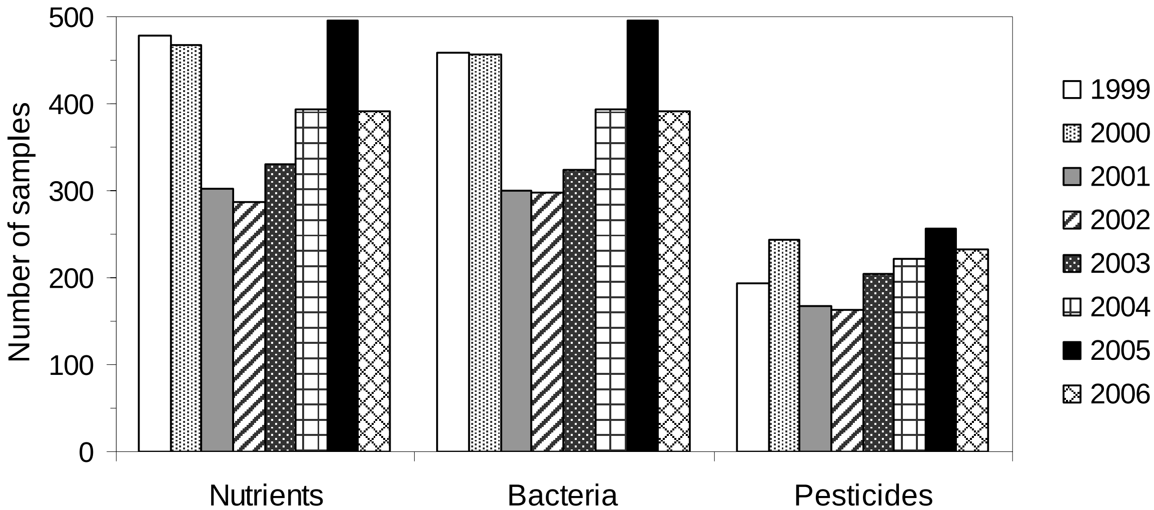


Figure A17.1. Number of nutrient, bacteria, and pesticide samples for all AESA watersheds from 1999 to 2006.

Table A17.2. Pesticide sample numbers by stream and year.

Stream	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006
Battersea Drain		13	6	10	7	8	11	13	10	13
Blindman River	3	19	19	14	11	12	10	13	19	10
Buffalo Creek	2		4	6	4	8	14	9	12	12
Crowfoot Creek	3	12	2	15	12	15	17	11	14	13
Drain S6			7	8	9	8	10	13	8	13
Grande Prairie Creek			7	12	8	5	6	8	9	6
Haynes Creek	5	17	15	27	11	7	13	3	14	10
Hines Creek			3	6	5	9	7	8	8	8
Kleskun Drain			4	8	5	6	6	7	6	4
Meadow Creek	2		5	6	5	10	10	12	10	13
New West Coulee			6	9	8	8	11	13	10	13
Paddle River	2		13	11	11	9	10	8	19	13
Prairie Blood Coulee			6	6	4	8	4	10	8	12
Ray Creek	2	9	11	12	8	5	9	8	9	8
Renwick Creek	2	8	11	5	3	4	9	7	9	10
Rose Creek	8	15	15	15	13	12	11	15	17	13
Strawberry Creek	3	7	12	17	6	6	7	10	11	6
Stretton Creek	1	5	5	6				3	6	5
Threehills Creek	6	9	11	12	7	5	9	10	8	10
Tomahawk Creek	3	13	12	13	10	5	9	10	18	11
Trout Creek	2		6	8	9	10	11	13	10	12
Wabash Creek			6	8	3	4	4	11	16	9
Willow Creek	2	11	6	12	11	10	10	12	10	9

APPENDIX 18: NFR, PH, AND TEMPERATURE SUMMARY STATISTICS, 1999 TO 2006.

Table A18.1. Non-filterable residue (NFR) summary statistics from 1999 to 2006 for median annual ambient data, mass loading, FWMC, and export coefficients.

AESA Watershed	Number of Years	Summary Statistic	Ambient (mg L^{-1})	Load (kg yr^{-2})	FWMC mg L^{-1}	Export Coefficient (kg ha^{-1} yr^{-1})
Battersea Drain		N of cases		152	152	
Battersea Drain		Mean	15	406362	40.105	
Battersea Drain	8	Median	14	234010	21.642	N/A
Battersea Drain		Min	9	129907	14.154	
Battersea Drain		Max	26	974382	114.105	
Blindman River		N of cases		179	179	
Blindman River		Mean	9	2953855	74.437	83.679
Blindman River	8	Median	9	2035497	71.010	57.663
Blindman River		Min	5	104253	9.003	2.953
Blindman River		Max	15	9405790	146.826	266.453
Buffalo Creek		N of cases		119	119	
Buffalo Creek		Mean	11	97091	22.363	6.605
Buffalo Creek	8	Median	9	41365	11.454	2.814
Buffalo Creek		Min	3	25874	8.162	1.760
Buffalo Creek		Max	27	280715	45.762	19.096
Crowfoot Creek		N of cases		155	155	
Crowfoot Creek		Mean	22	3254805	81.461	
Crowfoot Creek	8	Median	20	1409883	43.697	N/A
Crowfoot Creek		Min	10	426518	22.341	
Crowfoot Creek		Max	30	12279380	241.042	
Drain S6		N of cases				
Drain S6		Mean	8			
Drain S6	8	Median	8	N/A	N/A	N/A
Drain S6		Min	4			
Drain S6		Max	10			
Grande Prairie Creek		N of cases		104	104	
Grande Prairie Creek		Mean	25	838801	100.454	59.914
Grande Prairie Creek	8	Median	21	473598	54.188	33.828
Grande Prairie Creek		Min	6	28565	15.156	2.040
Grande Prairie Creek		Max	55	2686338	273.298	191.881
Hines Creek		N of cases		114	114	
Hines Creek		Mean	8	227355	16.270	6.079
Hines Creek	8	Median	7	137802	9.298	3.685
Hines Creek		Min	4	782	4.139	0.021
Hines Creek		Max	19	521637	46.510	13.948
Haynes Creek M6		N of cases		95	95	
Haynes Creek M6		Mean	5	16827	9.264	1.014
Haynes Creek M6	7	Median	5	20307	7.494	1.223
Haynes Creek M6		Min	4	333	3.044	0.020
Haynes Creek M6		Max	7	32209	18.069	1.940
Kleskun Drain		N of cases		74	74	
Kleskun Drain		Mean	10	105518	83.390	32.974
Kleskun Drain	8	Median	11	62482	80.261	19.526
Kleskun Drain		Min	3	528	9.206	0.165
Kleskun Drain		Max	16	286883	183.742	89.651
Meadow Creek		N of cases		152	152	
Meadow Creek		Mean	57	735085	165.763	56.545
Meadow Creek	8	Median	52	195639	147.213	15.049
Meadow Creek		Min	24	40800	47.494	3.138
Meadow Creek		Max	99	2306366	330.652	177.413
New West Coulee		N of cases		144	144	
New West Coulee		Mean	24	1199392	56.056	
New West Coulee	8	Median	25	1162475	58.468	N/A
New West Coulee		Min	8	379272	16.590	
New West Coulee		Max	48	2384009	113.259	
Paddle River		N of cases		156	156	
Paddle River		Mean	13	1991412	120.117	78.712
Paddle River	8	Median	13	638791	74.753	25.249
Paddle River		Min	7	55914	8.143	2.210
Paddle River		Max	19	6296911	319.700	248.890

Table A18.1, cont. Non-filterable residue (NFR) summary statistics from 1999 to 2006 for median annual ambient data, mass loading, FWMC, and annual export coefficients.

AESA Watershed	Number of Years	Summary Statistic	Ambient (mg L^{-1})	Load (kg yr^{-2})	FWMC mg L^{-1}	Export Coefficient (kg ha^{-1} yr^{-1})
Prairie Blood Coulee		N of cases		110	110	
Prairie Blood Coulee		Mean	6	143567	19.566	6.353
Prairie Blood Coulee	8	Median	5	10127	6.999	0.448
Prairie Blood Coulee		Min	3	401	2.790	0.018
Prairie Blood Coulee		Max	12	536512	67.848	23.739
Ray Creek		N of cases		141	141	
Ray Creek		Mean	4	12363	8.872	2.784
Ray Creek	8	Median	3	7707	8.294	1.736
Ray Creek		Min	2	1367	2.892	0.308
Ray Creek		Max	7	40592	20.281	9.142
Renwick Creek		N of cases		108	108	
Renwick Creek		Mean	4	8670	12.196	1.492
Renwick Creek	8	Median	4	8080	11.182	1.391
Renwick Creek		Min	2	154	3.557	0.026
Renwick Creek		Max	7	24083	22.262	4.145
Rose Creek		N of cases		182	182	
Rose Creek		Mean	18	17405160	297.751	311.362
Rose Creek	8	Median	11	9260943	298.626	165.670
Rose Creek		Min	6	649884	22.370	11.626
Rose Creek		Max	75	68236140	798.919	1220.682
Stretton Creek		N of cases		38	38	
Stretton Creek		Mean	3	11146	11.416	1.980
Stretton Creek	4	Median	4	11424	6.553	2.029
Stretton Creek		Min	2	5609	3.251	0.996
Stretton Creek		Max	5	16128	29.306	2.865
Strawberry Creek		N of cases		132	132	
Strawberry Creek		Mean	45	16044878	689.865	272.409
Strawberry Creek	8	Median	44	13099361	473.088	222.400
Strawberry Creek		Min	15	734094	139.365	12.463
Strawberry Creek		Max	96	38294140	2165.919	650.155
Threehills Creek		N of cases		145	145	
Threehills Creek		Mean	7	77542	18.054	5.619
Threehills Creek	8	Median	7	36189	11.353	2.622
Threehills Creek		Min	6	2655	6.706	0.192
Threehills Creek		Max	12	259667	48.422	18.816
Tomahawk Creek		N of cases		154	154	
Tomahawk Creek		Mean	36	1178692	288.885	123.682
Tomahawk Creek	8	Median	36	1465277	203.174	153.754
Tomahawk Creek		Min	14	11582	39.227	1.215
Tomahawk Creek		Max	62	2097941	586.606	220.141
Trout Creek		N of cases		168	168	
Trout Creek		Mean	12	6106289	171.473	138.465
Trout Creek	8	Median	12	633774	49.479	14.371
Trout Creek		Min	4	17376	9.928	0.394
Trout Creek		Max	20	20064980	659.498	454.988
Wabash Creek		N of cases		100	100	
Wabash Creek		Mean	11	120050	30.249	3.490
Wabash Creek	8	Median	10	58154	19.976	1.691
Wabash Creek		Min	7	769	8.918	0.022
Wabash Creek		Max	16	487651	64.822	14.176
Willow Creek		N of cases		167	167	
Willow Creek		Mean	3	6037128	183.173	924.522
Willow Creek	8	Median	2	433765	47.252	66.426
Willow Creek		Min	1	22667	5.563	3.471
Willow Creek		Max	10	42707170	1030.484	6540.149

Table A18.2. Annual pH summary statistics for each watershed from 1999 to 2006. N/A indicates data were not available for that year.

		1999	2000	2001	2002	2003	2004	2005	2006
Battersea Drain	N of cases	15	18	16	16	20	25	21	21
	Min	7.7	8.1	8.0	7.3	6.9	8.1	7.8	8.2
	Max	8.8	8.8	8.5	8.5	9.0	8.7	8.7	9.0
	Median	8.2	8.3	8.2	8.0	8.4	8.4	8.3	8.6
	Mean	8.2	8.4	8.2	8.0	8.3	8.4	8.3	8.5
Blindman River	N of cases	34	29	17	15	14	19	32	19
	Min	7.1	7.4	7.4	7.2	6.9	7.6	7.6	7.9
	Max	8.4	8.3	8.6	8.2	8.5	8.5	8.4	8.4
	Median	8.0	7.9	8.0	7.8	7.8	8.3	8.2	8.2
	Mean	7.8	7.9	8.0	7.7	7.8	8.2	8.1	8.2
Buffalo Creek	N of cases	14	11	9	12	16	16	24	17
	Min	8.1	7.9	8.1	7.9	8.0	8.3	7.9	8.1
	Max	8.6	8.3	8.5	8.5	8.6	8.5	8.5	8.6
	Median	8.3	8.1	8.3	8.2	8.4	8.4	8.4	8.4
	Mean	8.3	8.1	8.3	8.2	8.4	8.4	8.3	8.4
Crowfoot Creek	N of cases	14	19	17	15	23	21	22	22
	Min	7.7	7.5	7.7	7.6	6.8	7.8	8.0	8.0
	Max	8.7	9.0	8.6	8.5	8.5	8.5	8.6	8.7
	Median	8.3	8.2	8.1	8.0	8.3	8.3	8.3	8.4
	Mean	8.2	8.1	8.1	8.0	8.1	8.2	8.3	8.4
Drain S6	N of cases	15	16	15	13	16	22	15	21
	Min	8.0	7.9	8.0	7.8	7.3	8.0	8.2	8.1
	Max	9.1	8.7	8.2	8.8	8.6	8.5	8.3	8.5
	Median	8.2	8.2	8.1	8.0	8.3	8.3	8.3	8.3
	Mean	8.3	8.2	8.1	8.1	8.2	8.2	8.3	8.3
Grande Prairie Creek	N of cases	12	20	13	9	11	14	15	10
	Min	7.2	7.0	7.3	6.9	7.1	7.6	7.3	7.7
	Max	8.3	8.7	8.0	8.1	8.3	8.5	8.3	8.4
	Median	7.7	7.9	7.7	7.4	7.7	8.0	8.0	8.2
	Mean	7.7	7.8	7.7	7.5	7.7	8.0	7.9	8.1
Hines Creek	N of cases	8	17	12	14	12	22	18	12
	Min	7.7	7.1	7.3	7.2	6.7	7.4	7.6	7.9
	Max	8.0	8.1	8.0	8.0	8.1	8.2	8.2	8.3
	Median	7.9	7.5	7.6	7.4	7.7	8.0	7.8	8.1
	Mean	7.9	7.5	7.6	7.5	7.6	8.0	7.8	8.1
Haynes Creek M6	N of cases	22	21	9	8	13		12	10
	Min	7.0	7.2	7.7	7.0	7.0		7.7	7.8
	Max	8.4	8.4	8.5	7.8	8.4	N/A	8.5	8.3
	Median	7.8	7.9	8.2	7.4	8.2		8.1	8.1
	Mean	7.8	7.8	8.1	7.4	8.0		8.1	8.1
Kleskun Drain	N of cases	8	14	7	10	8	11	11	5
	Min	6.9	6.8	7.0	6.8	6.6	7.0	7.4	7.0
	Max	8.0	7.4	7.5	7.3	8.0	8.0	8.0	7.9
	Median	7.3	7.2	7.2	7.1	7.2	7.9	7.8	7.8
	Mean	7.3	7.2	7.2	7.1	7.3	7.7	7.7	7.7
Meadow Creek	N of cases	19	13	11	21	21	22	22	23
	Min	8.2	8.1	8.0	7.9	8.0	8.3	8.4	8.4
	Max	8.7	8.6	8.5	8.4	9.0	8.8	8.6	8.6
	Median	8.5	8.5	8.5	8.3	8.5	8.6	8.5	8.5
	Mean	8.5	8.4	8.4	8.3	8.5	8.5	8.5	8.5

Table A18.2, cont. Annual pH summary statistics for each watershed from 1999 to 2006. N/A indicates data were not available for that year.

		1999	2000	2001	2002	2003	2004	2005	2006
New West Coulee	N of cases	14	14	14	16	20	25	20	21
	Min	7.9	8.0	8.0	7.5	6.8	7.5	8.1	8.1
	Max	8.4	8.6	8.3	8.3	8.5	8.5	8.4	9.0
	Median	8.2	8.2	8.1	8.0	8.4	8.3	8.3	8.4
	Mean	8.2	8.3	8.1	8.0	8.1	8.2	8.3	8.4
Paddle River	N of cases	22	26	15	14	13	12	33	19
	Min	7.4	7.8	7.5	7.3	7.5	8.2	7.7	8.2
	Max	8.4	8.3	8.3	8.3	8.4	8.5	8.4	8.4
	Median	8.1	8.1	8.1	8.0	8.0	8.4	8.2	8.3
	Mean	8.0	8.1	8.0	7.9	8.0	8.4	8.2	8.3
Prairie Blood Coulee	N of cases	13	9	9	17	8	19	17	18
	Min	7.8	7.7	8.0	7.4	7.4	8.1	8.2	8.2
	Max	8.3	8.2	8.5	8.3	8.5	8.4	8.4	8.6
	Median	8.1	8.1	8.3	8.2	8.4	8.3	8.4	8.4
	Mean	8.1	8.0	8.3	8.1	8.2	8.3	8.3	8.4
Ray Creek	N of cases	32	26	12	7	14	15	20	15
	Min	7.0	7.7	7.7	7.5	7.3	7.8	7.7	7.9
	Max	8.5	8.3	8.5	8.3	8.6	8.5	8.5	8.5
	Median	8.1	8.2	8.2	8.1	8.3	8.3	8.2	8.3
	Mean	8.1	8.1	8.2	8.1	8.2	8.2	8.2	8.3
Renwick Creek	N of cases	27	12	5	6	11	13	19	15
	Min	7.4	7.7	7.5	7.7	6.9	7.9	7.6	7.8
	Max	8.6	8.3	8.4	8.4	8.6	8.4	8.5	8.6
	Median	8.2	8.2	7.8	8.2	8.1	8.3	8.2	8.3
	Mean	8.1	8.1	7.9	8.1	7.9	8.2	8.2	8.3
Rose Creek	N of cases	31	29	17	15	15	19	32	24
	Min	7.5	7.6	7.6	7.3	7.0	7.5	7.7	7.9
	Max	8.5	8.3	8.4	8.3	8.5	8.5	8.4	8.5
	Median	8.0	8.0	8.1	7.7	8.0	8.3	8.1	8.3
	Mean	7.9	8.0	8.1	7.8	7.9	8.2	8.1	8.3
Stretton Creek	N of cases	12	11				6	10	10
	Min	7.3	7.3				7.8	7.5	7.6
	Max	8.3	8.3	N/A	N/A	N/A	8.4	8.3	8.3
	Median	8.1	8.1				8.2	8.0	8.2
	Mean	7.9	7.9				8.1	8.0	8.1
Strawberry Creek	N of cases	26	30	11	11	11	17	16	10
	Min	7.6	7.5	6.9	5.0	6.8	7.7	7.6	7.9
	Max	8.5	8.4	8.4	8.3	8.6	8.5	8.6	8.5
	Median	8.2	8.2	8.2	8.1	8.2	8.4	8.3	8.4
	Mean	8.2	8.1	8.0	7.6	7.9	8.3	8.2	8.3
Threehills Creek	N of cases	33	24	12	8	14	16	20	18
	Min	7.2	7.6	7.5	7.4	7.1	7.6	7.7	7.9
	Max	8.6	8.5	8.6	8.2	8.5	8.6	8.5	8.7
	Median	8.0	8.1	8.2	7.8	8.2	8.4	8.3	8.4
	Mean	8.0	8.1	8.1	7.8	8.0	8.3	8.2	8.3
Tomahawk Creek	N of cases	23	26	14	11	15	17	31	17
	Min	7.0	7.5	6.9	6.8	7.3	7.3	7.6	7.9
	Max	8.0	8.1	8.3	8.0	8.1	8.4	8.2	8.3
	Median	7.6	7.7	7.7	7.5	7.7	8.1	8.1	8.2
	Mean	7.6	7.7	7.6	7.4	7.7	8.0	8.0	8.2

Table A18.2, cont. Annual pH summary statistics for each watershed from 1999 to 2006. N/A indicates data were not available for that year.

		1999	2000	2001	2002	2003	2004	2005	2006
Trout Creek	N of cases	23	19	17	21	21	23	21	23
	Min	8.3	8.1	8.0	7.9	7.4	8.2	8.4	8.3
	Max	8.6	8.8	8.8	8.4	8.6	8.6	8.6	8.6
	Median	8.4	8.3	8.3	8.3	8.5	8.5	8.5	8.5
	Mean	8.4	8.3	8.4	8.2	8.4	8.4	8.5	8.5
Wabash Creek	N of cases	10	18	5	8	8	13	23	15
	Min	7.5	7.3	7.1	6.9	7.3	7.0	7.4	7.6
	Max	8.0	8.2	8.2	7.6	8.2	8.3	8.4	8.3
	Median	7.8	7.7	7.4	7.4	7.6	8.0	8.1	8.1
	Mean	7.8	7.7	7.5	7.3	7.7	8.0	8.0	8.0
Willow Creek	N of cases	25	23	21	21	16	20	21	19
	Min	8.1	7.9	7.9	7.8	7.9	8.1	7.9	8.2
	Max	8.4	8.4	8.2	8.4	8.5	8.4	8.5	8.4
	Median	8.3	8.2	8.1	8.1	8.3	8.3	8.3	8.3
	Mean	8.3	8.2	8.1	8.1	8.3	8.3	8.3	8.3

Table A18.3. Annual temperature (°C) summary statistics for each watershed from 1999 to 2006. N/A indicates data were not available for that year.

		1999	2000	2001	2002	2003	2004	2005	2006
Battersea Drain	N of cases	15	18	15	16	18	25	21	19
	Min	1	0	4	0	-1	-1	0	2
	Max	20	21	19	25	24	22	20	26
	Median	11	13	13	14	12	11	11	14
	Mean	11	11	12	13	12	10	12	13
Blindman River	N of cases	33	29	15	15	14	19	32	19
	Min	0	1	1	0	0	0	0	0
	Max	20	22	25	19	26	22	22	24
	Median	10	14	15	9	8	10	13	9
	Mean	10	11	13	9	9	9	11	10
Buffalo Creek	N of cases	14	11	9	12	16	16	24	17
	Min	1	0	5	0	0	2	-2	1
	Max	16	17	17	18	21	19	23	19
	Median	6	7	10	6	5	11	9	7
	Mean	7	8	10	8	8	10	10	9
Crowfoot Creek	N of cases	14	19	17	15	22	21	20	22
	Min	0	0	0	1	0	1	1	5
	Max	20	22	20	20	21	20	21	23
	Median	10	9	12	10	11	9	10	9
	Mean	11	9	10	9	10	9	10	12
Drain S6	N of cases	15	16	15	14	16	21	14	20
	Min	6	6	4	7	-1	3	9	3
	Max	20	24	19	19	18	22	18	22
	Median	14	14	13	12	11	12	10	11
	Mean	14	14	13	13	10	11	12	11
Grande Prairie Creek	N of cases	11	18	13	9	10	11	11	10
	Min	1	0	0	0	3	0	0	1
	Max	22	20	18	16	18	19	16	16
	Median	6	9	11	5	5	11	9	6
	Mean	6	9	9	6	9	10	7	8
Hines Creek	N of cases	8	16	12	14	12	22	18	11
	Min	7	0	0	0	0	0	0	0
	Max	19	17	17	17	17	18	18	17
	Median	12	6	12	4	3	6	7	9
	Mean	13	7	8	6	6	6	7	7
Haynes Creek M6	N of cases	21	18	8	8	13		11	10
	Min	1	-1	0	3	2		0	4
	Max	22	22	25	10	20	N/A	13	18
	Median	10	12	13	6	11		8	8
	Mean	10	11	12	6	10		6	10
Kleskun Drain	N of cases	8	14	7	9	7	9	10	5
	Min	1	0	1	0	0	0	0	1
	Max	10	17	15	15	15	16	17	9
	Median	3	10	7	6	12	9	8	7
	Mean	4	9	8	7	9	7	8	6
Meadow Creek	N of cases	18	13	9	21	21	22	22	23
	Min	0	-4	-1	1	-5	0	-4	1
	Max	26	21	17	19	23	25	24	20
	Median	15	7	8	11	7	12	12	11
	Mean	15	9	8	11	8	11	13	11

Table A18.3, cont. Annual temperature (°C) summary statistics for each watershed from 1999 to 2006. N/A indicates data were not available for that year.

		1999	2000	2001	2002	2003	2004	2005	2006
New West Coulee	N of cases	14	13	13	16	19	25	20	20
	Min	1	6	5	2	-1	-1	4	0
	Max	21	20	21	23	23	23	21	24
	Median	13	13	13	13	11	10	12	13
	Mean	13	13	14	12	10	10	13	12
Paddle River	N of cases	20	26	14	14	12	12	33	19
	Min	0	1	1	1	0	0	1	0
	Max	19	20	21	20	22	18	19	22
	Median	7	10	15	5	7	9	9	8
	Mean	8	10	12	7	8	9	9	9
Prairie Blood Coulee	N of cases	13	8	9	17	8	18	16	18
	Min	1	0	7	-1	-3	1	5	3
	Max	20	14	20	28	16	22	26	23
	Median	11	9	14	19	11	12	14	16
	Mean	11	7	13	16	9	13	15	14
Ray Creek	N of cases	29	24	12	7	14	15	20	15
	Min	1	1	0	6	1	1	0	1
	Max	24	22	23	15	21	22	17	20
	Median	10	11	12	9	14	5	6	7
	Mean	11	10	10	9	11	9	7	8
Renwick Creek	N of cases	27	11	5	6	11	13	19	15
	Min	1	1	0	8	1	0	0	1
	Max	24	13	9	12	16	22	21	21
	Median	11	8	2	10	7	7	10	10
	Mean	12	8	4	10	8	8	8	11
Rose Creek	N of cases	28	29	16	15	15	18	32	24
	Min	0	1	1	0	0	0	0	0
	Max	19	18	23	18	24	21	20	22
	Median	11	12	17	6	3	9	10	13
	Mean	10	10	13	8	8	9	9	10
Stretton Creek	N of cases	11	11				6	10	10
	Min	1	2				3	-2	2
	Max	15	16	N/A	N/A	N/A	13	15	16
	Median	5	7				7	5	8
	Mean	7	8				7	6	8
Strawberry Creek	N of cases	25	30	10	11	11	16	16	10
	Min	0	0	0	0	0	0	0	0
	Max	20	22	27	17	22	18	16	19
	Median	11	16	11	5	6	9	6	3
	Mean	11	13	11	7	7	8	6	6
Threehills Creek	N of cases	32	23	12	8	14	16	20	18
	Min	1	2	0	5	1	0	0	1
	Max	24	20	25	14	21	21	18	20
	Median	10	13	12	9	10	8	7	8
	Mean	11	11	12	9	10	9	8	10
Tomahawk Creek	N of cases	19	24	13	11	13	17	31	16
	Min	0	0	0	1	0	0	0	0
	Max	21	22	22	16	22	19	19	19
	Median	10	13	16	6	9	11	10	8
	Mean	10	12	13	7	9	9	10	8

Table A18.3, cont. Annual temperature (°C) summary statistics for each watershed from 1999 to 2006. N/A indicates data were not available for that year.

		1999	2000	2001	2002	2003	2004	2005	2006
Trout Creek	N of cases	23	19	16	21	21	23	20	23
	Min	0	-4	0	2	-4	-1	-4	1
	Max	23	22	23	17	21	25	22	18
	Median	14	9	10	10	6	10	10	10
	Mean	13	10	11	11	7	11	11	10
Wabash Creek	N of cases	10	18	5	8	7	12	22	15
	Min	0	1	1	0	1	0	1	1
	Max	10	20	18	9	16	22	18	19
	Median	8	10	16	5	5	12	8	8
	Mean	6	10	11	4	8	10	9	8
Willow Creek	N of cases	25	23	21	21	15	20	21	19
	Min	1	-5	0	0	-5	-1	-2	-2
	Max	17	14	11	13	14	15	16	16
	Median	8	5	6	4	4	6	5	6
	Mean	8	6	5	4	5	6	6	6