University of Alberta

Agricultural Impacts on Surface Water Quality in the Crowfoot Creek Watershed

by

Gerald R. Ontkean ©

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Gerald Ontkean

127 Purdue Court
Lethbridge, Alberta
T1K 4S1

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University of Alberta

Faculty of Graduate Studies and Research

The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled *Agricultural Impacts on Surface Water Quality in the Crowfoot Creek Watershed* submitted by *Gerald R. Ontkean* in partial fulfillment of the requirements for the degree of *Master of Science* in *Water and Land Resources*.

D S Chanasyk

D. S. Chanasyk (supervisor)

M. J. Dudas

M.J. Dudas

F E Hicks

F.E. Hicks

D. Rodney Bennett

D.R. Bennett

Date: Sept 20/00

ABSTRACT

Concerns regarding agricultural impacts on water quality resulted in a study examining total nitrogen (TN), total phosphorus (TP) and faecal coliform bacteria (FC) within the Crowfoot Creek watershed in southern Alberta. TN and TP concentrations were highest in the spring and generally decreased over the year, with some increases due to rainfall. TN exceeded Alberta Water Quality Guidelines during spring, early summer and event periods, while TP exceeded guidelines throughout the monitoring period. TN was dominated by organic nitrogen and TP was dominated by dissolved P. FC counts increased with increased presence of livestock. Counts were lowest in the spring and fall, and greatest during the summer. Particulate P was lowest and FC highest in reaches having grassland adjacent to watercourses. Concentrations of all parameters frequently exceeded guidelines and indicated a negative impact of agricultural practices on water quality within the sub-basins. Irrigation return flow water had a positive impact on water quality.

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1. INTRODUCTION

Continued growth of the world's population has resulted in the need for continually increased food and fibre production. Enhanced production and/or conversion of marginal land to agricultural uses must occur to accomplish this increase with finite arable land resources. In either case, atmospheric, soil and water resources will be affected (Gish and Sadeghi 1993). Alberta's agriculture industry is under increasing pressure to remain globally competitive while providing responsible stewardship of our natural resources. Agriculture must be seen as an industry which produces high quality, safe products in a clean environment.

Public concern regarding the quality of water is increasing. In the United States, agriculture has been identified as the largest contributor to nonpoint source pollution of surface and groundwater systems (Geleta et al. 1994; Tim and Jolly 1994). The United States Environmental Protection Agency (USEPA 1976) identified agricultural nonpoint runoff of sediment and agricultural chemicals as impairing the quality of 55% of surveyed river length and 58% of surveyed lake area. Worldwide, 30 to 50% of the earth's land is currently affected by nonpoint source degradation from erosion, fertilizers, pesticides, organic matter and sewage sludge (Sharpley and Meyer 1994).

In Canada, most of the attention given water quality problems has centred on the Great Lakes region (Whillans et al. 1986; Paterson and Lindwall 1992). As recently as 1992, the Canadian prairies were not considered as a high risk area regarding water quality problems due to low precipitation and less intensive development (Paterson and Lindwall 1992). There is, however, a lack of data available to determine the general impact of agriculture on water quality in Alberta (Paterson 1992).

Several studies in Alberta have suggested that agriculture may be contributing to the degradation of water quality. A study under the Environmental Sustainability Initiative (Miller et al. 1992) documented the presence of agriculturally derived nutrients and herbicides in both surface and ground water in southern Alberta. Additional studies carried out in southern Alberta (Greenlee et al. 1995; Greenlee and Lund 1995) indicated that several water quality parameters in irrigation return flows at times exceeded the Canadian Drinking

Water Standards for human/livestock consumption and irrigation use (CCREM 1987). These parameters included salinity, TDS, nitrate-nitrogen and total and faecal coliforms. Phosphate-phosphorus concentrations were above the maximum desirable concentration for flowing water (USEPA 1976), although these concentrations were detected infrequently.

Based on studies carried out in several irrigation districts, while irrigation source water is generally acceptable for irrigation, there was a change in water quality between the source and return flow streams with concentrations of salinity, total phosphorus and bacteria increasing, and nitrate and nitrite decreasing (Madawaska Consulting 1997). The report of the Bow River Water Quality Task Force (1991) listed agricultural runoff as a potential source of contamination in the Bow River downstream of the City of Calgary. Hence, water users have become increasingly concerned about the potential for point and nonpoint source contamination of the river by agricultural enterprises along the river and its tributaries.

The Crowfoot Creek watershed was identified by the Bow River Water Quality Task Force (1991) as an area of concern. This watershed is situated in an area of medium to intensive irrigated and dryland agricultural production and receives irrigation return flows from the Western Irrigation District. The task force identified nutrients, coliform bacteria and metals as the main contaminants in the Bow River within the reach where Crowfoot Creek discharges. The levels of nutrients and faecal coliform bacteria at the mouth of Crowfoot Creek usually exceeded various water quality guidelines (Madawaska Consulting 1995).

This project was initiated to determine if agricultural practices are contributing to increases in total and dissolved phosphorus, total nitrogen and faecal coliform bacteria in four sub-basins of the Crowfoot Creek watershed and, if so, to attempt to identify land uses having an effect.

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2. PROJECT AND SITE CHARACTERIZATION

2.1 Project Background

The Bow River Water Quality Task Force (1991) identified nutrients, coliform bacteria and metals as the main contaminants in the reach of the Bow River receiving discharge from the Crowfoot Creek watershed. A review of water quality at the outlet of the Crowfoot Creek watershed conducted by Madawaska Consulting (1995) indicated levels of nutrients and feacal coliform bacteria in excess of some water quality guidelines. The authors recommended a water sampling program along Crowfoot Creek to evaluate loadings of various contaminants.

In 1996, the Irrigation Branch of Alberta Agriculture, Food and Rural Development (AAFRD) began a four-year study of the Crowfoot Creek watershed, with the objective of determining the agricultural impacts on water quality in the watershed (AAFRD 1996). This thesis project was developed as part of the AAFRD study.

2.2 Watershed Description

2.2.1 Location

The Crowfoot Creek watershed is located approximately 85 km east of the City of Calgary, in Wheatland County, and covers an area of approximately 1600 km^2. The watershed includes the area drained by Parflesh and Chancellor Creeks into Crowfoot Creek, as well as the area drained by Crowfoot Creek itself (Fig. 2.1).

2.2.2 Watershed Hydrology

Crowfoot Creek contains two branches, the West Branch and the North Branch (Fig. 2.1). The West Branch of Crowfoot Creek originates in two small branches in Township 24, Range 23. Water in the West Branch consists of snowmelt runoff in the spring, return flow spillwater from the Western Irrigation District (WID) irrigation system and surface runoff. The main irrigation district inputs are from Secondary Canal B and three major irrigation return flows, and constitute the majority of the water in this branch during the irrigation season. A total of 5993 ha of land are under irrigation in this basin.

The North Branch of Crowfoot Creek originates north of the village of Standard in Township 25, Range 22. Water in this branch is generally from the same sources as for the West Branch. Only one major irrigation return flow enters this branch, while springs north of the Town of Standard add a minor amount of water to it.

Other notable hydrologic features on these two branches include two Ducks Unlimited ponds. The Hilton Project is located on the West Branch in Sections 16, 17 and 20 in Township 24, Range 23. The Dallas Project is located on the North Branch in Section 26 in Township 24, Range 22. An additional source of flow on the North Branch is an annual sewage effluent release from the Town of Standard, which takes place during the last week of October and lasts four to five days.

The confluence of the West and North Branches of Crowfoot Creek is located in Sec. 29-23-21-W4. As with the West and North Branches, snowmelt runoff, WID return flows and surface runoff account for the water in the section of the creek below the confluence. Two major irrigation return flows enter the creek between the confluence and the Bow River. In addition, Parflesh Creek empties into Crowfoot Creek in SW 25-23-21-W4 and Chancellor Creek joins Crowfoot Creek in SE 19-23-20-W4. Both of these creeks typically flow only during years with high spring runoff. Crowfoot Creek discharges into the Bow River on the Siksika Reserve in Township 21, Range 20.

2.2.3 Climate

The Crowfoot Creek watershed is located within the Prairies Ecozone, moist mixed grassland ecoregion. This area is closely related with semiarid moisture conditions and Dark Brown Chernozemic soils. The mean annual temperature is approximately 2.5 °C, but can reach as high as 5 °C in some areas of the ecozone. Mean summer temperature is 15.5 °C and the mean winter temperature is -11 °C. Annual precipitation ranges from 350 - 400 mm (Ecological Stratification Working Group 1997).

Mean annual temperature, using 30-year (1961-1990) averages from Environment Canada stations located at Calgary International Airport and Gleichen, were 3.8 and 3.7 °C, respectively (Table 2.1). The 30-year average annual precipitation at the same two sites was 398 and 355 mm, respectively (Table 2.2) (AAFRD 2000).

2.2.4 Topography and Watershed Drainage

The Crowfoot Creek watershed consists of topographic highs in the north and west regions of the basin, with elevation decreasing to the junction of Crowfoot Creek and the Bow River. Elevations range from approximately 1085 to 885 m at the Bow River (Fig. 2.2). Eight major sub-basins have been defined and consist of mainly 2 to 5% slopes, with 6 to 9% slopes as a minor to significant inclusion (Fig. 2.3). Slopes greater than 10% are found along the north boundary of the watershed, the south-west edge of the watershed and along the lower end of the Crowfoot Creek valley, below the confluence of the West and North Branches of the Crowfoot.

Surface runoff from the topographic highs in the west flows to the north and northeast, while surface runoff from the north flows to the south and southeast. Crowfoot Creek drains from the northwest to the southeast.

2.2.5 Geology

2.2.5.1 Bedrock Geology

The bedrock geology of the Crowfoot Creek basin consists of two major types (Fig. 2.4), the Horseshoe Canyon Formation and the Scollard Formation. Both are subgroups of the Edmonton Formation.

The Horseshoe Canyon Formation or lower Edmonton Formation is the oldest formation in the Edmonton group. It is composed of a flat-lying sequence of many thin and interfingering layers of fresh and brackish-water sandstone, siltstone, mudstone, shale, carbonaceous shale and coal. The predominant color is gray to light gray or greenish gray.

The Scollard Formation is also referred to as the upper Edmonton Formation. It is lithologically very similar to the Horseshoe Canyon Formation. This formation has been included as a member of the Paskapoo Formation as well (Rodvang 1997; EUB-AGS 1999).

2.2.5.2 Surficial Geology

The majority of the Crowfoot Creek basin is overlain by silt and clay lacustrine and draped moraine deposits (Shetsen 1987). The area to the north and east of the North Branch of Crowfoot Creek, south of the Town of Standard and east to the edge of the basin is dominantly fine sediment lacustrine deposits (Fig. 2.5). The southeast corner of the basin is stagnation moraine with undulating to hummocky topography. Relief ranges from less than 3 to 15 m.

The area between the North and West branches of Crowfoot Creek consists of draped moraine with flat to undulating topography. The area to the south and west of Crowfoot Creek is mainly fine textured lacustrine deposits with areas of draped and hummocky moraine. The dominant surface expression is gently undulating.

2.2.6 Hydrogeology

The dominant groundwater flow is from topographic highs in the basin towards Crowfoot Creek (Rodvang 1997). In the western section of the basin, canal seepage is estimated to account for 18% of the annual recharge while water applied by irrigation accounts for approximately 6% of the annual recharge (Rodvang 1997).

2.2.7 Vegetation/Crops

Grasses are dominated by speargrass and wheatgrass. There is a variety of deciduous shrubs including buckbrush, chokecherry, wolf willow and saskatoon. Scrubby aspen, willow, cottonwood and box-elder are found in river valleys and terraces. Saline areas support alkali grass, wild barley, red sampire and sea blite. Cereal grains and oilseeds are grown, both under irrigation and in grain-fallow dryland rotations (Ecological Stratification Working Group 1997).

2.2.8 Soils

Soils in the study area are dominantly Orthic Dark Brown Chernozemic soils formed on either medium textured lacustrine material or morainal deposits (Fig. 2.6a and 2.6b).

Regosolic Dark Brown Chernozemic and Gleyed Dark Brown Chernozemic soil also comprise significant portions of several areas. Minor inclusions of Black Solodized Solonetz and Solonetzic Brown Chernozemic soils are present. Depressional areas contain Orthic and Orthic Humic Gleysolic soils. Other parent materials include coarse sediments, gravelly coarse sediments and medium-textured moraine overlying softrock (CAESA-Soil Inventory Project Working Group 1998).

2.3 Site Selection and Characterization

2.3.1 Site Selection

Of the 30 sites monitored in the Crowfoot Creek watershed study, nine were included in this project. These sites were numbers: 10, 11, 13, 14, 16, 19, 22, 23 and 24 (Fig. 2.1). The Technical Implementation Committee, created by AAFRD and its partners to develop the monitoring plan for the Crowfoot Creek watershed study, selected Sites 10, 11, 13, 14 and 16 early in 1996. Site 19 was selected in late spring of 1996 and the remaining sites were determined in the fall of 1996.

2.3.2 Landcover Classification and Mapping

The landcover in each basin was divided into four principal groups. These groups - natural and improved grassland, annual crops, forage crops and summerfallow - represented the majority of the landcover conditions that were likely to be found in the study area.

Initially, landcover was collected using aerial photography and field trips. Black and white 1:15,000 enlargements from 1:30,000 aerial photographs taken on August 4, 1991 were obtained. These photographs covered the areas adjacent to Crowfoot Creek and several of its tributaries. Landcover was determined by direct visual examination of the area in July and August 1996, with the information recorded onto the aerial photographs. The information from the photographs was digitized and a computer map of the area created. This method proved very time consuming and labour intensive.

A second, less time-consuming method involved the use of satellite imagery. An image collected by the LANDSAT 5 TM satellite, using bands 3, 4 and 5, on August 5, 1997, was purchased by AAFRD. The scene description was track 41, frame 24. The land cover classification was carried out by Terrain Resources in Lethbridge, Alberta and involved performing an unsupervised classification of the satellite image.

2.3.3 Crowfoot Creek Sub-basins

Sub-basins 10 and 19 are on the Crowfoot Creek. Sub-basins 14 and 16 drain into Crowfoot Creek and are used as WID return flow channels. Sub-basin 10 consists of the area draining past monitoring Site 10 located in SE19-24-23-W4 (Fig. 2.3). Inflow into this sub-basin occurs primarily at Site 13. Site 13, while being considered as the inflow to Sub-basin 10, is also a WID irrigation system turnout. The site is located approximately 50 m downstream from where water is diverted from Secondary B Canal into the Stonehouse ditch and does not have a drainage area associated with it. During spring runoff, no water passes this site as the gate from Secondary B canal is not open. During the irrigation season, the volume of water passing this site is determined by the amount of water being released into the Stonehouse ditch for irrigation and other uses. There is one irrigation system withdrawal just upstream of the monitoring site. When demand for water stops, or the irrigation district stops diverting water into their system, flows at this site often drop to zero. Several other irrigation return flows spill water into the channel upstream of Site 10. These flows are of various magnitudes and were not monitored.

Sub-basin 10 has dominantly gentle slopes of 2 to 5% with minor inclusions of 6 to 9% slopes. The creek itself is relatively narrow and shallow for most of the length of this sub-basin. Sub-basin 10 drains an area of approximately 4,453 ha (Table 2.3) made up primarily of 48% annual crops and 41% native grassland. Much of the area adjacent to the creek is in grass and cattle are pastured here for much of the year with open access to the water (Fig. 2.7). Due to the gentle slopes along the creek, cropland is cultivated close to the edge of the creek channel. Additionally, there are several large wetland areas along this reach of the creek.

Sub-basin 19 is the sub-basin that drains past Site 19 located in SE1-24-23-W4 (Fig. 2.3). Inflow is monitored at Site 11, the downstream end of the Hilton Project Ducks

Unlimited pond. Several smaller irrigation district return flows contribute water to this sub-basin. Sub-basin 19 drains approximately 6,073 ha with annual crops and native grassland making up approximately 77 and 12% of the landcover, respectively (Table 2.3). Much of the grassland is along the creek in areas too difficult to farm due to the tortuous nature of the creek (Fig. 2.8). Slopes in this sub-basin generally range from 2 to 5%, with some areas of 6 to 9%. Several small, steeply-sloping areas of 10 to 15% are also present.

2.3.4 WID Return Flow Sub-basins

Sub-basin 14 drains past Site 14, located in the NE8-24-23-W4 (Fig. 2.3). Inflows to this sub-basin are from various WID spills and return flows, primarily Site 22, while the outflow enters the Hilton Project Ducks Unlimited wetland. Sub-basin 14 is 902 ha in size. Slopes in the lower end of the sub-basin slope towards the north and range from 2 to 5%, while slopes in the upper portions range mainly from 6 to 9%, with inclusions of 9 to 15% slopes. Landcover is comprised of 51% annual crops, 26% native grass and 22% summerfallow (Table 2.3). Much of the grassland is adjacent to the creek and immediately upstream of the monitoring site (Fig. 2.9). The steeper portions of the sub-basin are fenced and grassed. Cattle are resident in this basin during the summer months and have open access to the creek.

Sub-basin 16 includes the water passing Site 16 in SW28-23-22-W4 (Fig. 2.3). As with Sub-basin 14, inflows are from WID spills and return flows, primarily Sites 23 and 24. Outflow enters Crowfoot Creek downstream of Site 19. Sub-basin 16 drains an area of approximately 4,207 ha. Slopes range from 2 to 5% with inclusions of 6 to 9% in the upper slope areas. The landcover is comprised of 49% annual crops, 22% native grassland and 19% summerfallow (Table 2.3). The grassland areas are located mainly in the upper portion of the sub-basin, with the cropland and summerfallow distributed throughout the sub-basin (Fig. 2.10).

2.4 Site Instrumentation

Instrumentation at all monitoring sites was installed and operational by the end of

July 1996. Monitoring of flows and water quality was carried out until the end of October 1996. Flows and water quality were monitored from the beginning of spring runoff until the end of October in 1997 to 1999.

2.4.1 Precipitation

Rainfall was monitored at six locations in the basin. Tipping bucket rain gauges connected to dataloggers were used at Sites 5, 10, 15, 16, 18 and at the AAFRD meteorological station in NE7-24-22-W4, located approximately four kilometers north and three kilometers east of Site 19 (Fig. 2.3). Rainfall data were collected at Sites 5, 10, 15 and at the meterological station during the summer of 1996. All of the rain gauges were operational for the final three years of the study.

2.4.2 Flow Determinations

The discharge passing any given monitoring site was calculated using the velocity and stream cross-sectional area method (Gordon et al. 1997; Gray and Wigham 1970; Linsley et al. 1975). This method requires the determination of the cross-sectional area of the stream and the average stream velocity. Stream velocity was measured using an electronic digitmeter (Scientific Instruments Inc. CMD 9000 Digitmeter) connected to a current meter by a wading rod. A Scientific Instruments Top Setting Wading Rod and either a Model 1205 "mini" or Model 1210 "maxi" current meter were used in this study. Measurements were taken at suitable locations near the monitoring site.

During periods of high flow, such as spring runoff, flow velocity measurements were difficult to obtain. Entering the creek to perform cross-sectional metering safely was not possible due to water depth and flow velocity. In these instances, alternative monitoring sites were used to allow use of a Swoffer datalogging current meter (Swoffer Instruments Inc. Model 3000-1514). At the alternate sites, some form of structure was used to facilitate standing above the water. Most of the sites used road crossing bridges, but some sites had old telephone poles laid across the banks to stand on. The alternate sites were near the usual metering sites.

Flow velocities were metered at each site beginning with spring runoff and continuing until the end of October. Sites were flow metered as often as possible in order to obtain flow velocity data relating to as many flow stages, or depths, as possible. The discharges calculated from these measurements were used to develop a stage-discharge curve for each site. This curve describes the relationship between depth of water and discharge for a specific monitoring site. The greater the range of stages that have associated flow velocities, the better the stage-discharge curve relationship. The curve was developed using a power curve program.

The stage-discharge curve was used in association with the depth or stage data obtained from sites instrumented with stilling wells and dataloggers or from staff gauge readings. Stream depth gauging was automated using a Lakewood Model CP-XA (1996) datalogger with ROM upgrade and calibrated float potentiometer installed in the stilling well (Fig. 2.11). The dataloggers provided the stage at each site in 20-min intervals. Using the stage-discharge relationship, a discharge was determined for each 20-min datalogger reading as well as for mean daily flows.

2.4.3 Sampling Periods

Three distinct sampling periods were considered during the study. Spring runoff was the period from the beginning of snowmelt runoff to its completion. Spring runoff was monitored in 1997, 1998 and 1999. The study did not begin in time to monitor spring runoff in 1996.

The period from the end of spring runoff until the end of October was termed the post-spring runoff (PSRO) period. This period was monitored during all four years of the study. In 1996, monitoring began in July, missing approximately the first three and one half months of the period. During the majority of this time, irrigation water was being diverted into the WID infrastructure. The WID began diverting water into its infrastructure around mid-May in 1997 and during the first week of May in 1998 and 1999.

Events sampled during the study were selected based on the extent of the storm and the ability, both in terms of manpower and budget, to carry out the more intensive sampling pattern. In order to observe how the entire basin reacted to runoff producing storms, only general rainfalls of high amounts were considered for event sampling. Six events over the four years of the study were sampled, including one storm in 1996 (September 17), 1997 (May 25) and 1998 (June 29 - July 4) and three in 1999 (May 14, June 2 - 3, July 13). These events were sampled more intensively than the regular sampling pattern, usually on a daily basis. Several storms were not large scale enough to affect the entire basin and were not sampled, but may have had a localized effect on water quality.

Water quality samples were collected weekly during the period following spring runoff until the end of October. Samples were collected more frequently during spring runoff and rainfall events. Sampling was conducted daily at the beginning of spring runoff and reduced in frequency as the runoff decreased. Daily sampling was carried out during rainfall events. Water quality samples were collected if there was flowing water present at the site.

2.4.4 Water Quality Sampling

Samples for water quality analysis were collected at the same locations as the flow was measured. Samples were collected using an ISCO Model 3700 automated sampler or as a grab sample. The grab samples were used at sites where an ISCO sampler was not practical, such as low flow sites where a water sample was not collected every sampling period. Grab samples were used at all sites during spring runoff and at times when the ISCO samplers were not used due to freezing temperatures. Samples collected for determination of faecal coliform bacteria were collected as grab samples only.

During a sampling period the ISCO sampler was programmed to collect a water sample every 2 h over a 24-h period. The samplers were set to begin sample collection in the morning. At this time, a 9400-ml Nalgene bottle was placed into the sampler to hold the sample. In 1996 the bottles were washed after every use. For the remainder of the study, a plastic bag was inserted into each bottle to hold the water. These bags were disposed after each use. Ice packs were placed around the bottle inside the sampler to keep the sample as cool as possible in order to slow chemical and biological activity in the collected water. At the end of the 24-h sampling period the bottle was removed from the sampler and manually agitated in order to place sediments back into suspension. A 2000-ml portion of this sample was then poured off into a 2000-ml Nalgene bottle. The 2000-ml bottle was triple rinsed with sample water before being filled and placed into a cooler with ice packs for transportation to the field laboratory.

When collecting grab samples, the water sample was collected on the same day that the samples were collected from the ISCO samplers. Grab samples were collected by filling a 2000-ml Nalgene bottle which had been triple rinsed with creek water prior to sample collection. The grab samples were also transported in coolers with ice packs to reduce transformations in the sample.

Samples for coliform bacteria determination were collected in 250-ml plastic bottles obtained from the Provincial Laboratory for Public Health. The bottles contained sodium thiosulfate powder which served as a preservative. These bottles were not triple rinsed before being filled, instead the bottle was filled to at least the 250-ml fill line and placed in a cooler for transportation.

During event sampling, water samples were collected by both sampling methods. When a runoff event was anticipated, the ISCO samplers were set for event sampling. In this mode the sampler was triggered by the datalogger and took a sample. The trigger was activated by an increase or decrease in stage in the water course. The change required to trigger the sampler was in proportion to the flow at the site allowing both the rising and falling limbs of the hydrograph to be sampled. If there was not sufficient time to set the samplers or the site did not have an ISCO sampler installed, grab samples were collected. All coliform samples were taken as grab samples.

2.4.5 Water Quality Analysis

After collection, the 2000 ml water samples were transported to a field laboratory. All 125 ml and 500 ml sample bottles used were triple rinsed with the sample water that was to be placed into them. Samples were stored at 4 °C until transport. Transportation to the Lethbridge laboratory was accomplished in coolers with ice packs. In Lethbridge, samples were stored at 4 °C until analysis. Analyses were usually performed within ten days, 30 days at maximum.

The 2000-ml field bottles were shaken to resuspend sediments and one 500-ml and one 125-ml sample of unfiltered water were poured from them. The 500-ml sample was analyzed for total suspended solids. The 125-ml sample was preserved with 2 ml of 5% sulphuric acid and analyzed for total Kjeldahl nitrogen and total phosphorus.

Two 125-ml filtered samples were also required. Filtration was carried out using 0.45-μm filter paper. If the sample contained large amounts of sediment, the filter paper would plug and several filter papers were required in order to obtain the required sample volume. During spring runoff and event flows, large amounts of fine sediments resulted in very slow filtering using filter paper. For these samples, high capacity filters were used in order to speed the filtering process and to reduce the handling of the filtering equipment.

One of the two filtered samples was analyzed for pH, electrical conductivity, calcium, magnesium, sodium, potassium, chloride, sulfate, bicarbonate, carbonate, total dissolved solids, nitrite, nitrate, ammonia-nitrogen and dissolved phosphorus. The second filtered sample was analyzed for dissolved phosphorus. The dissolved phosphorus sample was preserved with 2 ml of 5% sulphuric acid.

The analytical methods used by the Soil and Water Laboratory, Irrigation Branch, Alberta Agriculture, Food and Rural Development in Lethbridge to determine water quality parameters are listed in Tables 2.4 and 2.5.

The samples for faecal coliform bacteria determination were sent by courier to the Provincial Laboratory for Public Health in Calgary and were analyzed within 24 h using the membrane filtration method (Greenberg et al. 1992).

2.5 Data Analysis

Time series graphs of the parameters examined for all years of the study were produced for each of the study sites. These graphs showed temporal variability in water quality attribute concentrations during the monitoring season allowing a visual comparison between sites. Bar graphs were used to illustrate flow-weighted mean concentrations (FWMCs) and mass loads of various water quality parameters.

FWMCs and mass loads were calculated using the computer model FLUX (Walker 1996). FLUX estimates the nutrient or other water quality component loading passing a tributary sampling station over a given time. FLUX uses six techniques to map the flow-concentration relationship developed from the sample record onto the entire flow record. Total mass discharge and the associated error statistics are produced. If the data are stratified properly, the six methods should give load estimates that are not significantly different from each other. The data can be stratified based on the flow or time period. For this study, stratification was based on time period allowing specific time periods to be analyzed separately and compared to others. Periods of interest included spring runoff, post-spring runoff, and rainfall event periods. The uncertainty in the estimate was illustrated by the coefficient of variance or CV. The calculation method used was the one with lowest CV for each particular site (Greenlee et al. 2000).

Statistical analyses were carried out using WQHYDRO (Aroner 1995). Data values less than detection limits were assigned a value of one-half the detection limit in order for them to be graphed and used in statistical analyses and flow models. Non-parametric tests were used to compare median concentrations among sites due to the non-normality of the data. Non-parametric tests do not assume a particular form of distribution and are resistant to outliers. The Kruskill-Wallis test was used to perform multiple site comparisons. The Wilcoxon-Mann-Whitney rank sum test was used for two-site comparisons.

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Table 2.1. Seasonal temperature summary (degrees C)[z]

Station	Year	JAN.	FEB.	MAR.	APR.	MAY	JUN.	JUL.	AUG.	SEP.	OCT.	NOV.	DEC.	MEAN
Calgary International Airport	1999	-7.9	-1.5	-0.8	4.8	8.6	11.8	13.6	15.8	10.2	5.4	1.3	-0.6	5.1
	1998	-13.5	-2.9	-4.5	6.3	12.2	12.5	17.7	17.8	12.7	6.9	-2.5	-8.4	4.5
	1997	-12.4	-2.8	-4.0	2.2	9.3	13.8	16.0	15.8	13.0	4.8	-1.5	-2.7	4.3
	1996	-15.8	-5.8	-5.9	4.8	6.5	13.3	16.4	16.8	9.1	4.2	-9.7	-14.4	1.6
	1995	-7.3	-5.1	-2.9	3.3	9.3	14.1	15.5	13.6	11.9	4.1	-4.8	-11.0	3.4
	1994	-9.3	-13.5	2.4	5.8	10.5	13.3	17.4	16.1	13.0	4.5	-3.3	-5.8	4.3
	1993	-11.4	-7.1	-0.3	4.8	11.5	12.8	13.3	13.8	10.3	6.3	-2.9	-1.9	4.1
	1992	-1.1	-2.5	3.1	6.3	9.5	14.8	14.2	14.0	9.5	5.3	-1.3	-11.8	5.0
	1991	-8.8	0.9	-3.4	5.4	9.4	13.0	16.4	18.0	11.5	2.1	-2.5	-2.0	5.0
30 Year Average		-9.7	-6.4	-2.6	4.1	9.7	14.0	16.4	15.7	10.6	5.7	-3.0	-8.4	3.8
Gleichen	1999	-10.8	-1.8	-1.0	5.1	8.6	12.0	13.9	15.3	10.4	5.6	*	*	*
	1998	-15.8	-2.9	-4.4	6.4	12.2	13.4	18.5	17.9	13.5	7.4	-2.0	-9.5	4.5
	1997	-15.9	-5.5	-5.1	2.1	9.0	14.1	17.4	16.9	12.5	5.7	-1.9	-4.5	3.7
	1996	-19.0	-9.3	-7.0	4.8	7.6	14.3	17.4	16.8	9.6	4.2	-9.4	-16.1	1.2
	1995	-10.6	-5.5	-2.8	3.2	10.0	15.0	16.9	14.3	12.1	3.9	-5.6	-13.8	3.1
	1994	-12.9	-15.9	2.3	5.9	10.8	14.2	18.1	17.0	12.9	4.6	-3.2	-7.4	3.9
	1993	-14.2	-9.8	-0.4	4.9	11.8	13.6	13.9	14.3	10.2	5.2	-4.5	-4.0	3.4
	1992	-2.0	-3.6	2.8	7.4	10.0	15.3	14.4	14.7	9.7	5.0	-2.7	-14.1	4.7
	1991	-11.6	0.5	-3.5	6.2	10.0	13.8	17.0	19.1	11.9	1.6	-3.9	-4.9	4.7
30 Year Average		-11.9	-8.2	-2.9	4.7	10.7	15.1	17.3	16.7	11.1	5.8	-3.7	-10.0	3.7

[z] These are verified data from Environment Canada's climate archives; * missing data.

Table 2.2. Seasonal precipitation summary (mm)[z]

		JAN.	FEB.	MAR.	APR.	MAY	JUN.	JUL.	AUG.	SEP.	OCT.	NOV.	DEC.	TOTAL
Calgary International Airport	1999	11.3	0.0	6.4	72.8	52.8	95.4	103.8	88.4	9.1	3.6	12.4	1.8	456.8
	1998	15.6	4.0	59.4	41.2	86.4	110.4	132.2	18.0	26.0	11.4	14.1	19.0	537.7
	1997	18.5	3.6	17.1	12.6	100.7	138.4	16.9	57.8	47.8	14.8	0.6	6.3	435.1
	1996	27.8	3.0	35.4	18.5	51.4	59.2	41.9	21.0	46.4	23.4	30.1	18.1	376.2
	1995	2.8	2.1	7.3	31.8	71.9	43.4	133.4	34.2	27.9	14.4	22.7	22.9	414.8
	1994	10.6	9.6	8.1	12.6	62.5	68.4	38.0	84.4	10.4	31.4	13.9	5.2	355.1
	1993	5.8	12.5	17.8	6.5	61.9	118.4	87.0	92.3	24.3	9.0	10.4	3.6	449.5
	1992	2.2	3.6	7.9	24.6	46.2	177.2	76.2	41.5	48.1	1.5	38.8	14.0	494.9
	1991	7.4	14.9	21.0	7.1	96.1	113.2	29.6	64.2	25.9	15.8	9.6	1.8	406.6
30 Year Average		12.1	9.8	14.4	25.0	52.8	76.8	69.9	48.7	48.1	15.5	11.6	13.1	398.0
Gleichen	1999	13.0	0.0	2.0	31.0	79.0	95.0	94.0	54.0	10.0	4.0	*	*	*
	1998	4.0	0.0	50.0	49.0	35.0	113.0	96.0	37.0	21.0	2.0	*	12.0	*
	1997	13.0	6.0	57.0	15.0	56.0	49.5	21.0	54.0	41.0	5.0	0.0	0.0	317.5
	1996	15.2	1.0	14.0	16.0	30.5	47.0	49.0	44.0	64.0	8.5	20.0	21.5	330.7
	1995	5.0	2.5	20.5	40.0	78.0	37.0	78.0	39.0	17.0	18.0	15.5	28.0	378.5
	1994	17.0	3.0	4.5	0.0	96.0	30.5	33.0	36.0	7.0	36.5	9.5	1.5	274.5
	1993	1.0	13.0	16.0	10.0	46.0	68.0	76.6	118.0	37.0	1.0	8.5	5.0	400.1
	1992	2.0	7.0	0.0	24.0	57.0	116.0	73.0	26.0	43.5	7.0	9.0	7.0	371.5
	1991	5.0	9.0	9.0	4.0	91.5	85.5	18.0	57.0	3.0	21.0	5.0	0.0	308.0
30 Year Average		15.8	10.9	15.9	26.1	47.8	62.6	51.3	37.7	41.9	12.8	16.0	15.9	354.7

[z] These are verified data from Environment Canada's climate archives; * missing data.

Table 2.3. Landcover types and percentages within the sub-basins

Sub-basin	Landcover	Hectares	% of sub-basin
Basin 10	Annual crops	2123	48
	Grassland	1815	41
	Summerfallow	148	3
	Improved pasture	290	7
	Other	78	1
Total		4454	
Basin 19	Annual crops	4657	77
	Grassland	738	12
	Summerfallow	394	6
	Improved pasture	235	4
	Other	50	1
Total		6074	
Basin 14	Annual crops	461	51
	Grassland	237	26
	Summerfallow	199	22
	Improved pasture	5	1
	Other	0	0
Total		902	
Basin 16	Annual crops	2076	49
	Grassland	966	23
	Summerfallow	813	19
	Improved pasture	351	8
	Other	0	0
Total		4206	

Table 2.4. Water analysis methods for salinity parameters

Parameter	Abbreviation	Detection Limit	Units	Method (unless otherwise specified, methods are from Greenberg et al. (1992))
pH	none	0.1	none	
electrical conductivity	EC	0.01	ds/m	Conductivity bridge
calcium	Ca	0.0005	mmol$_c$/L	Atomic absorption (Method 3500-Ca B)
magnesium	Mg	0.0005	mmol$_c$/L	Atomic absorption (Method 3500-Mg B)
sodium	Na	0.1	mmol$_c$/L	Flame photometry (Method 3500-Na D)
potassium	K	0.05	mmol$_c$/L	Flame photometry (Method 3500-K D)
sulphate	SO_4	0.03	mmol$_c$/L	Turbimetric (Method 4500-SO4 E)
chloride	Cl	0.11	mmol$_c$/L	Automated ferricyanide (Method 4500-Cl E)
carbonate	CO_3	0.06	mmol$_c$/L	Titration (Method 2320 B)
bicarbonate	HCO_3	0.03	mmol$_c$/L	Titration (Method 2320 B)
total dissolved solids	TDS		mg/L	Summation
total suspended solids	TSS		mg/L	Filtration and drying (Method 2540 D)

Table 2.5. Water analysis methods for nutrients

Parameter	Abbreviation	Detection Limit mg L^{-1}	Method (unless otherwise specified, methods are from Greenberg et al. (1992))
nitrogen as nitrate	NO_3-N	0.04	Automated hydrazine reduction (Method 4500-NO3 H)
nitrogen as nitrite	NO_2-N	0.04	Colorimetric (Method 4500-NO2 B)
ammonia	NH_3	0.04	Automated phenate (Method 4500-NH3 H)
ortho-phosphate	PO_4-P	0.01	Automated ascorbic acid reduction (Method 4500-P F)
dissolved phosphorus	DP (as PO_4-P)	0.01	Semi-micro Kjeldahl digest (Method 4500-N C) followed by automated ascorbic acid reduction (Method 4500-P F)
total phosphorus	TP (as PO_4-P)	0.01	Semi-micro Kjeldahl digest (Method 4500-N C) followed by automated ascorbic acid reduction (Method 4500-P F)
total Kjeldahl nitrogen	TKN	0.04	Semi-micro Kjeldahl digest (Method 4500-N C) followed by automated phenate (Method 4500-NH3 H)

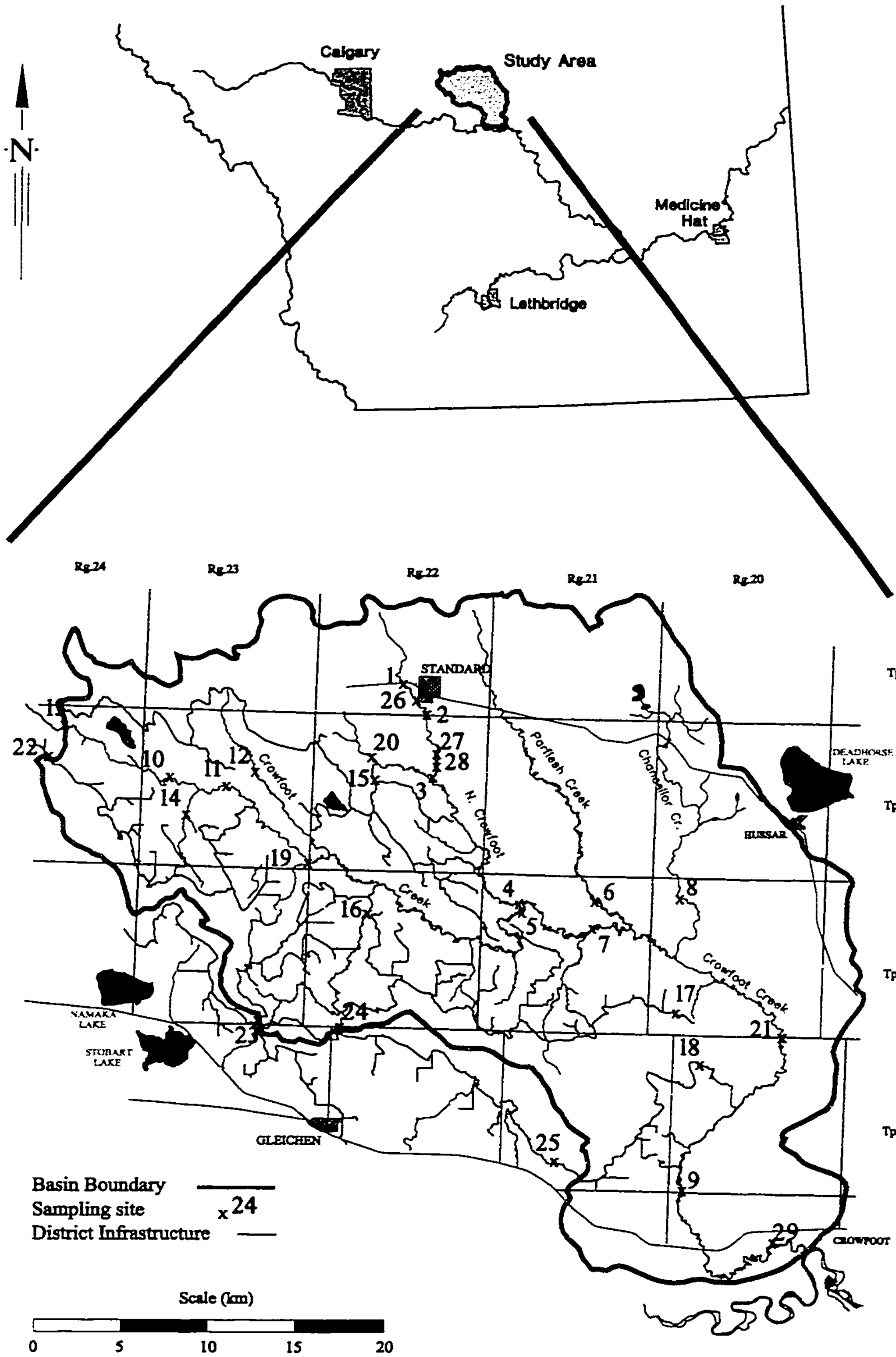

Figure 2.1. Crowfoot Creek watershed.

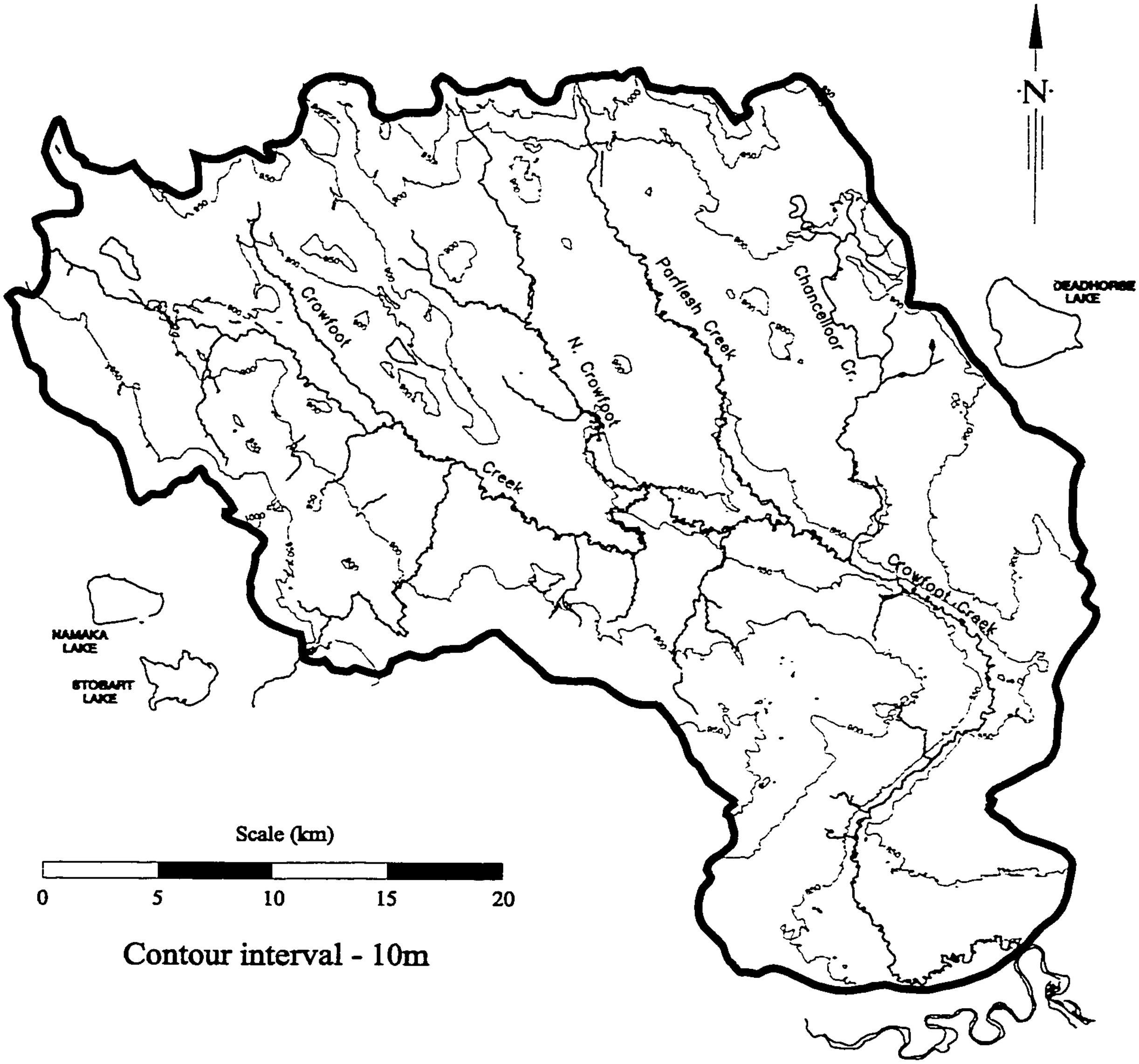

Figure 2.2. Crowfoot Creek watershed topography.

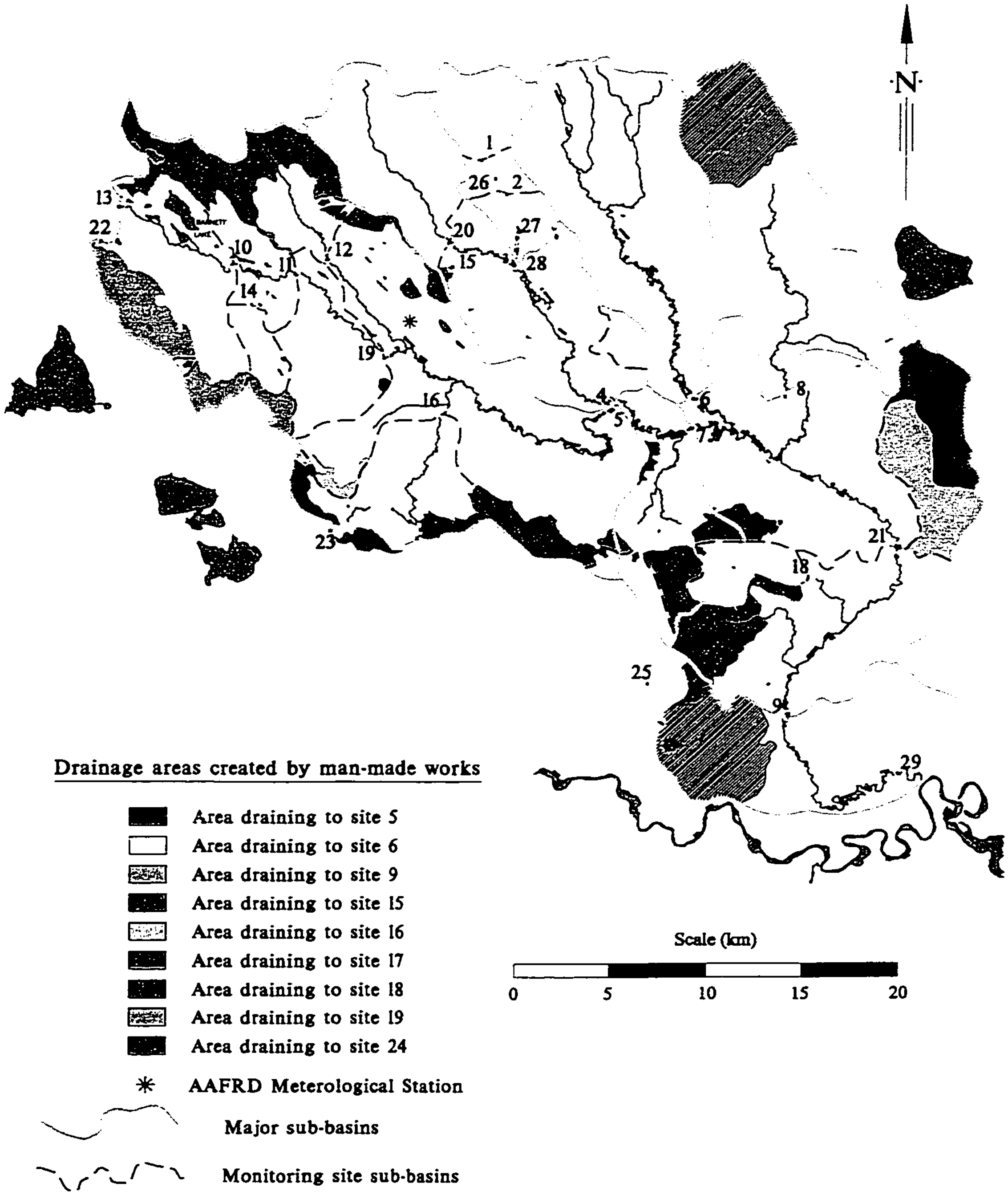

Figure 2.3. Crowfoot Creek watershed sub-basins and drainage.

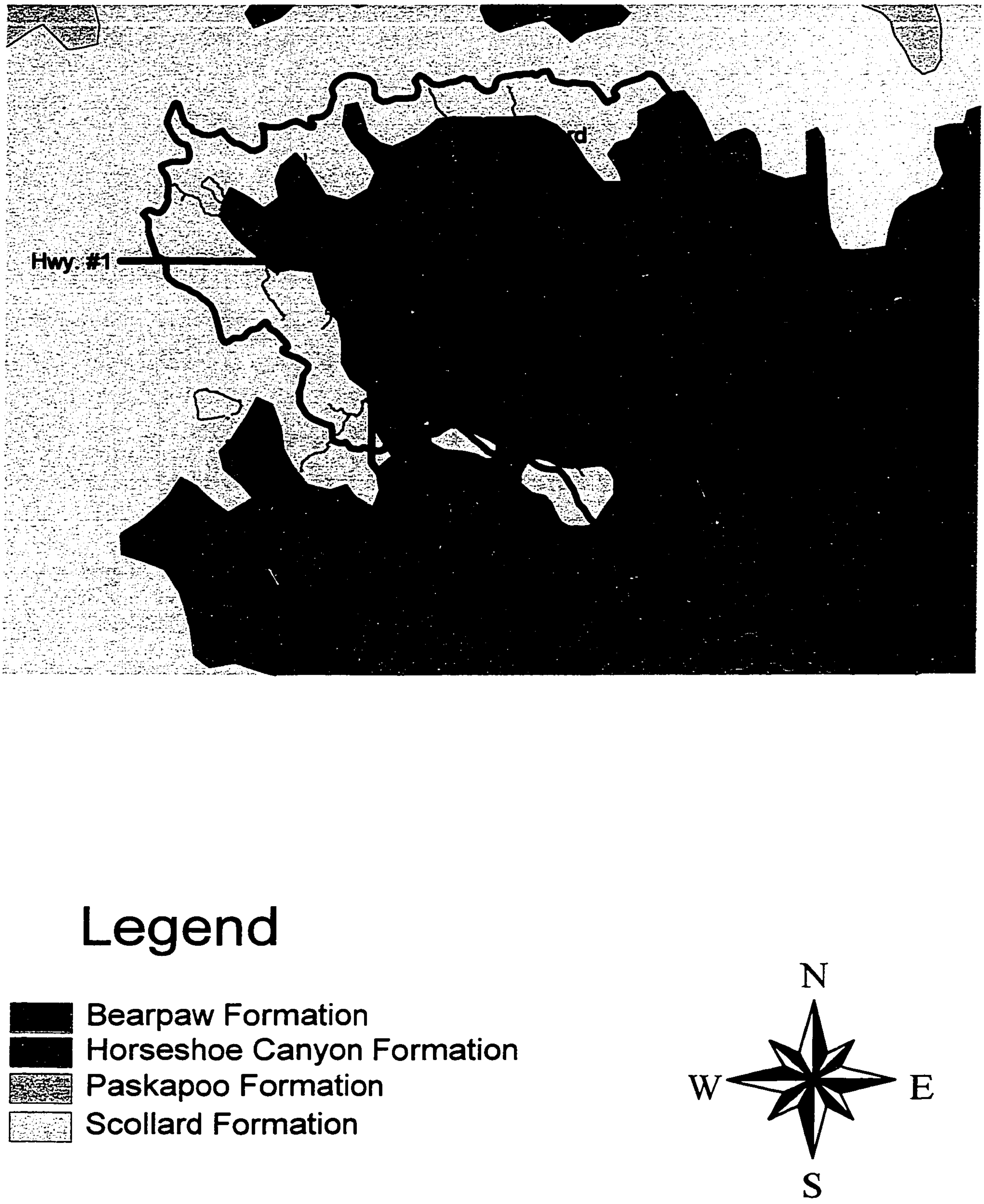

Figure 2.4. Bedrock geology.

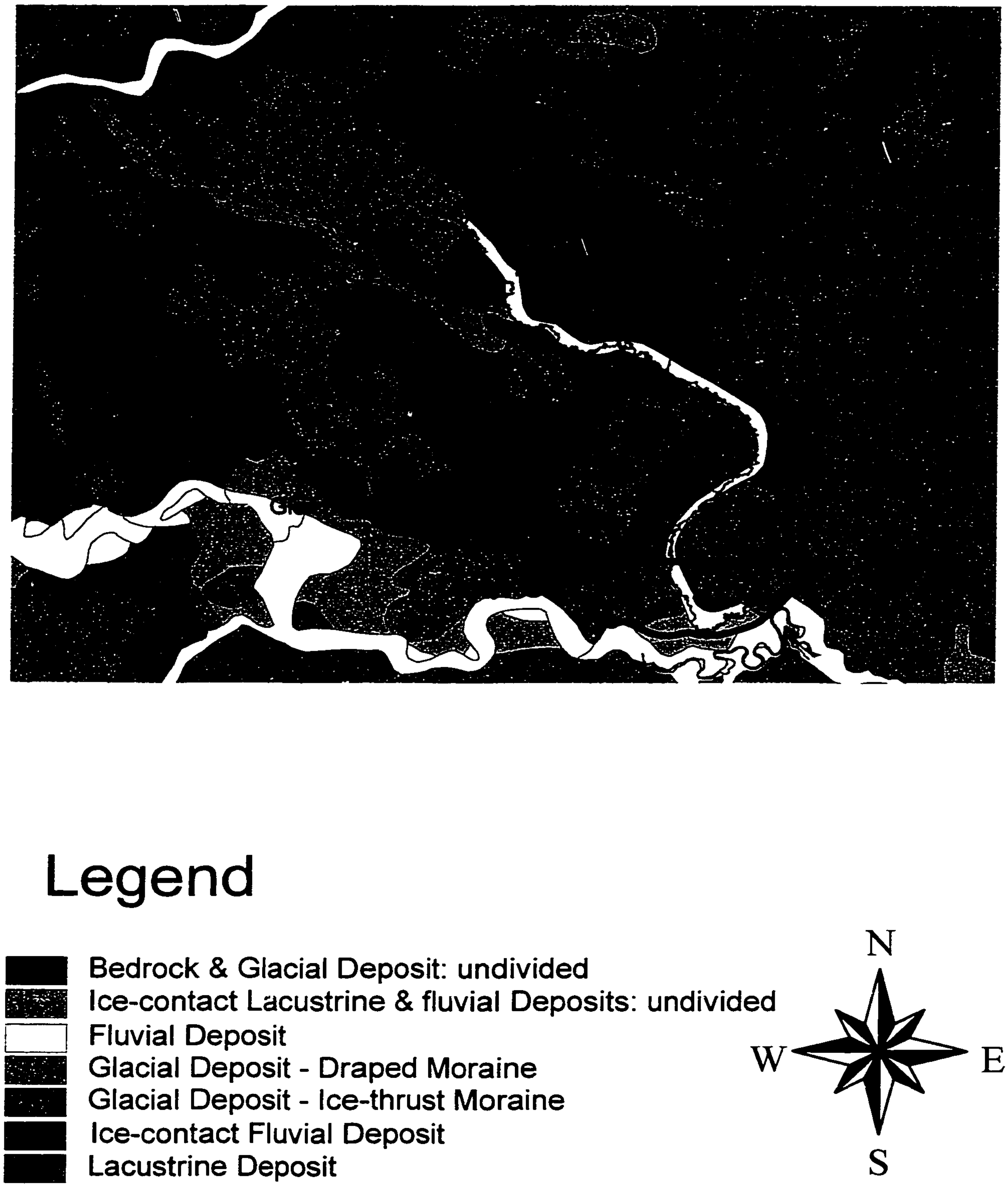

Figure 2.5. Surfici l geology.

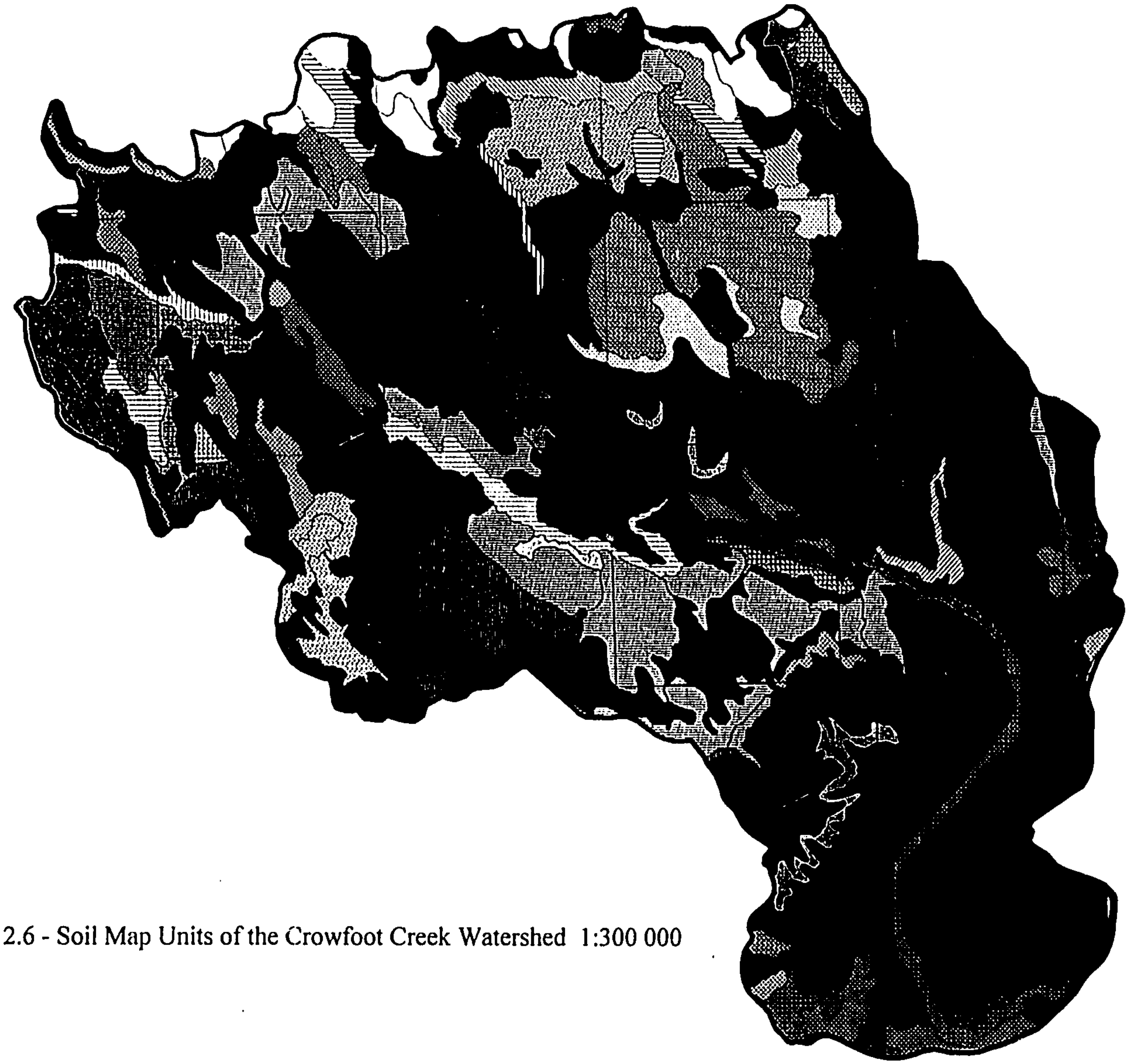

Figure 2.6 - Soil Map Units of the Crowfoot Creek Watershed 1:300 000

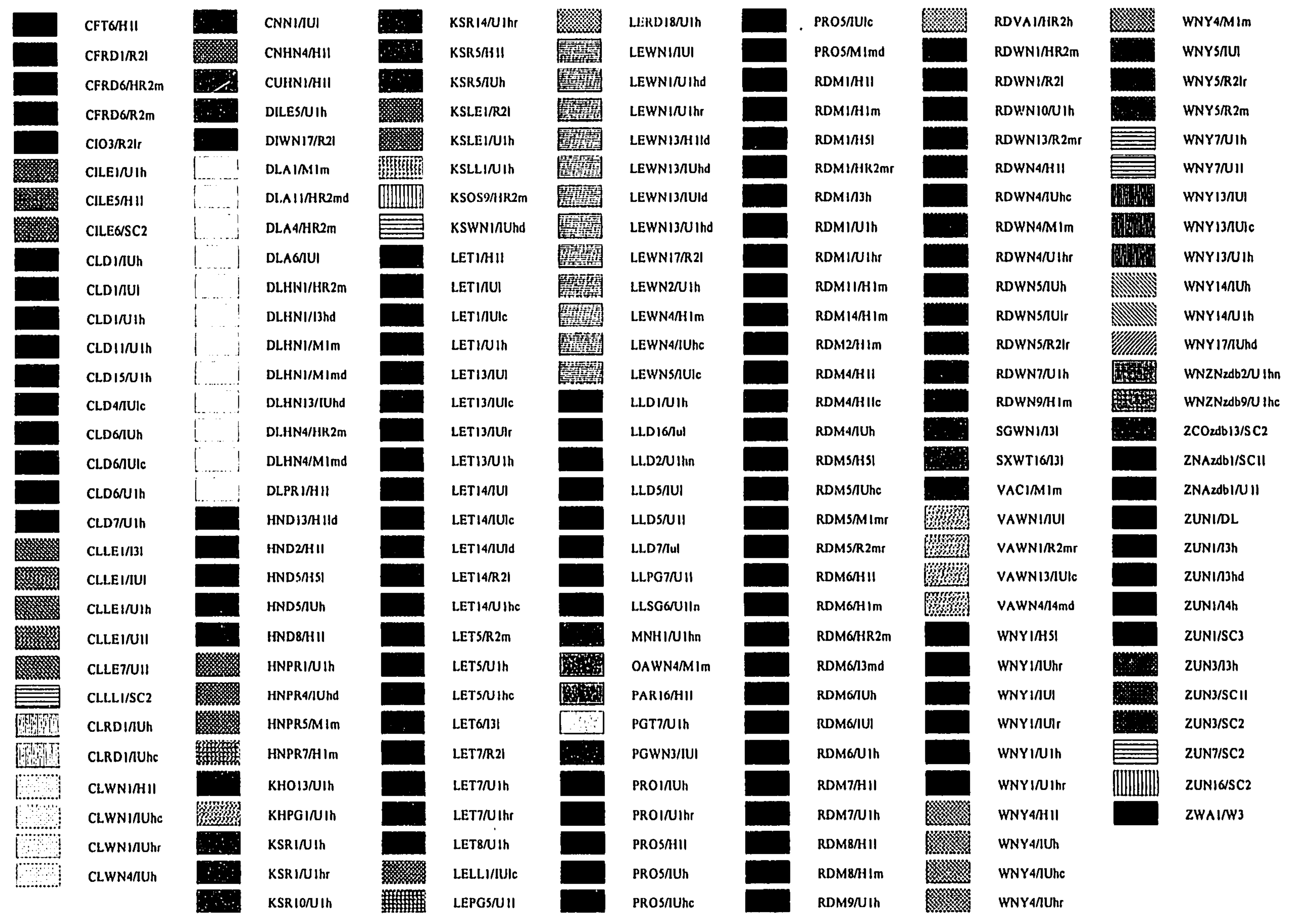

Figure 2.6a - Soil Map Units for the Crowfoot Creek Watershed

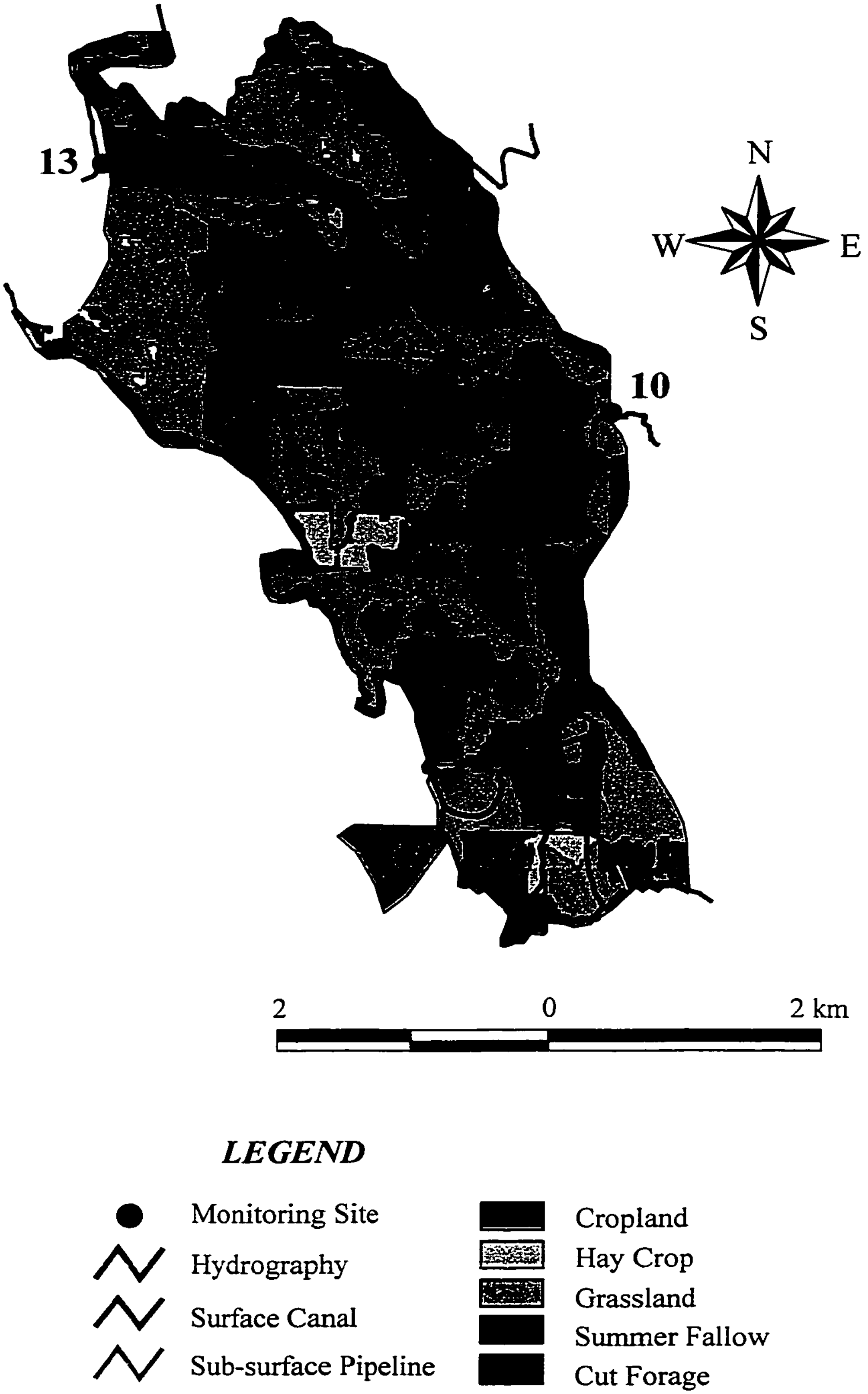

Figure 2.7. Sub-basin 10 landcover.

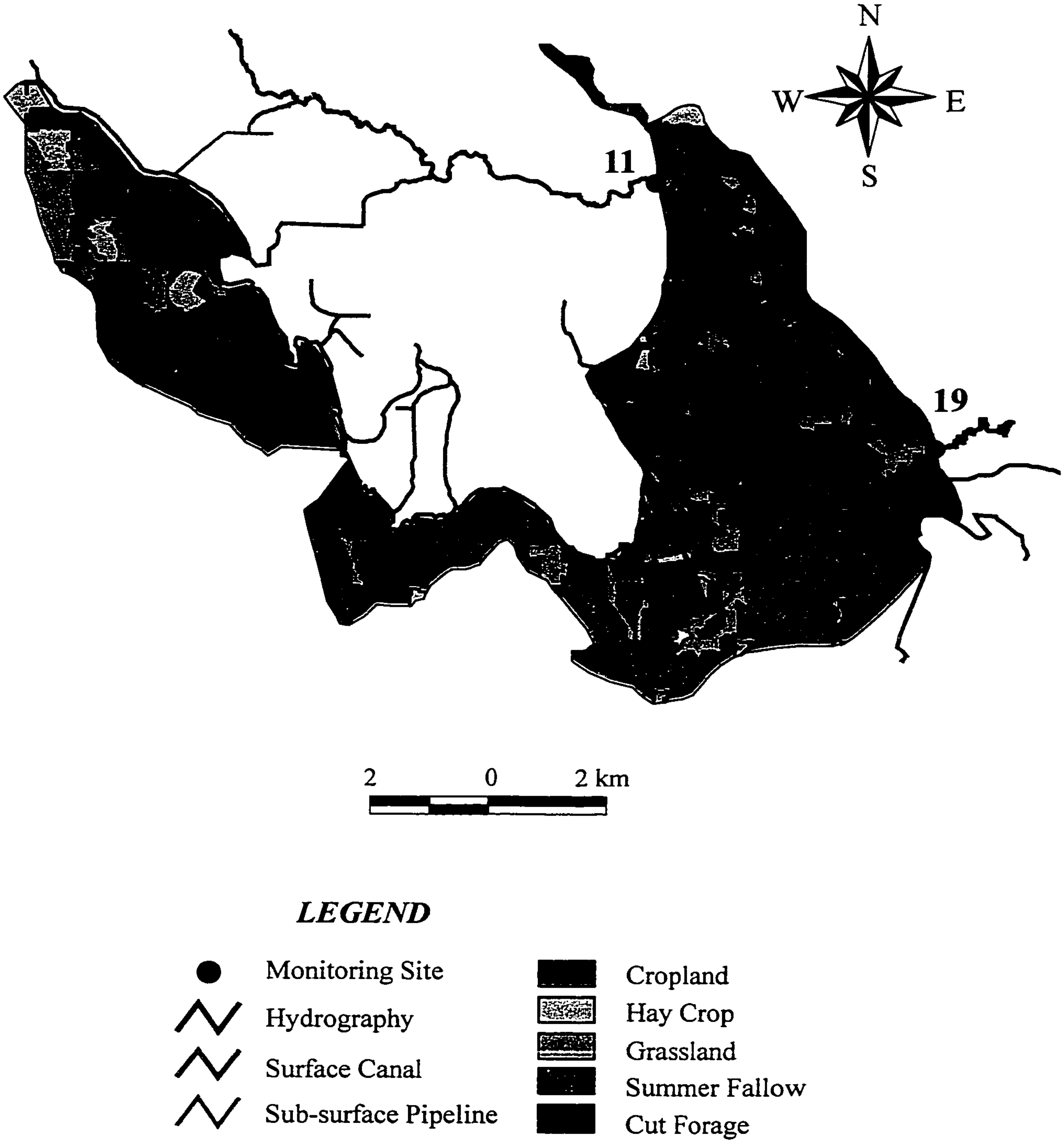

Figure 2.8. Sub-basin 19 landcover.

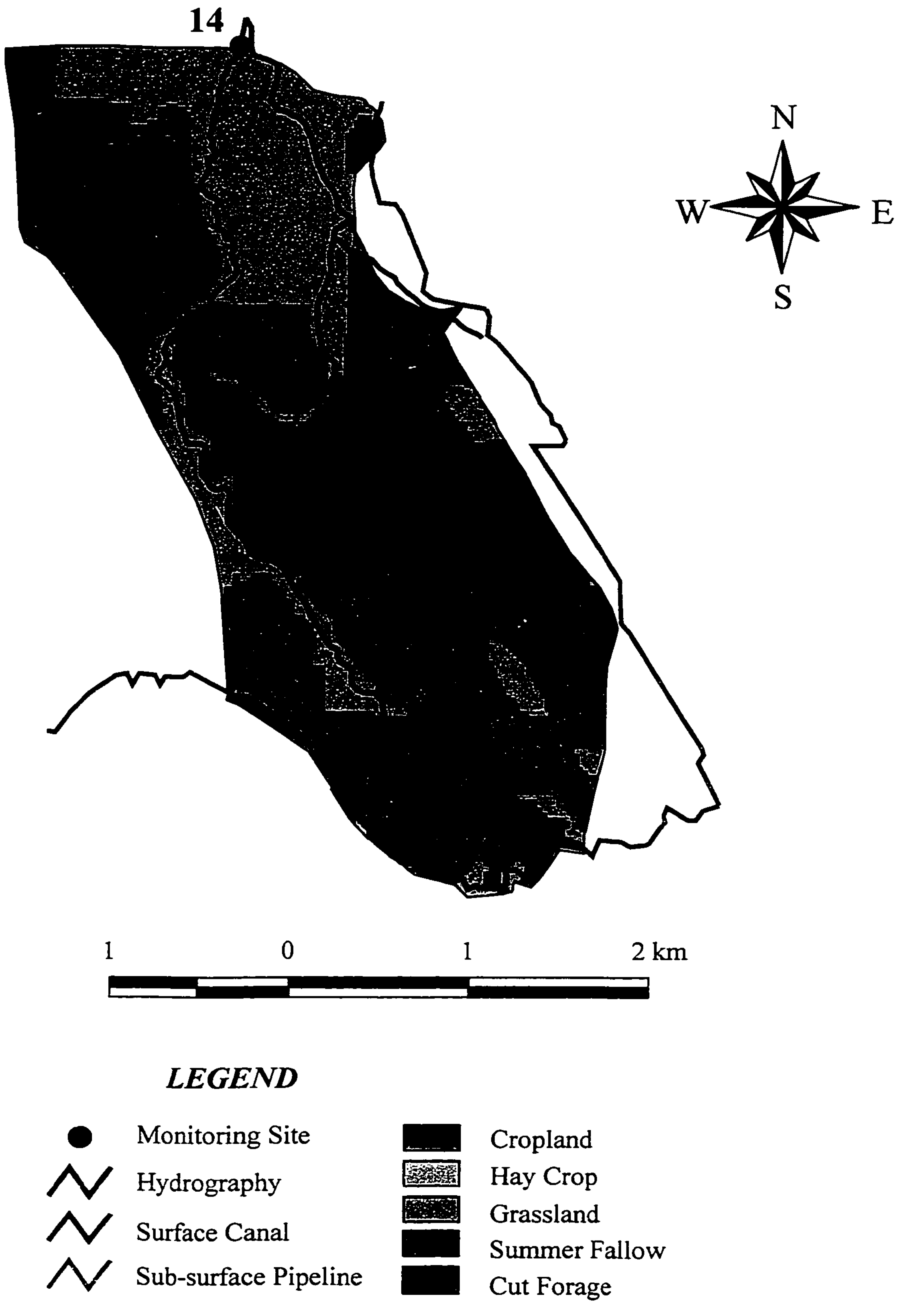

Figure 2.9. Sub-basin 14 landcover.

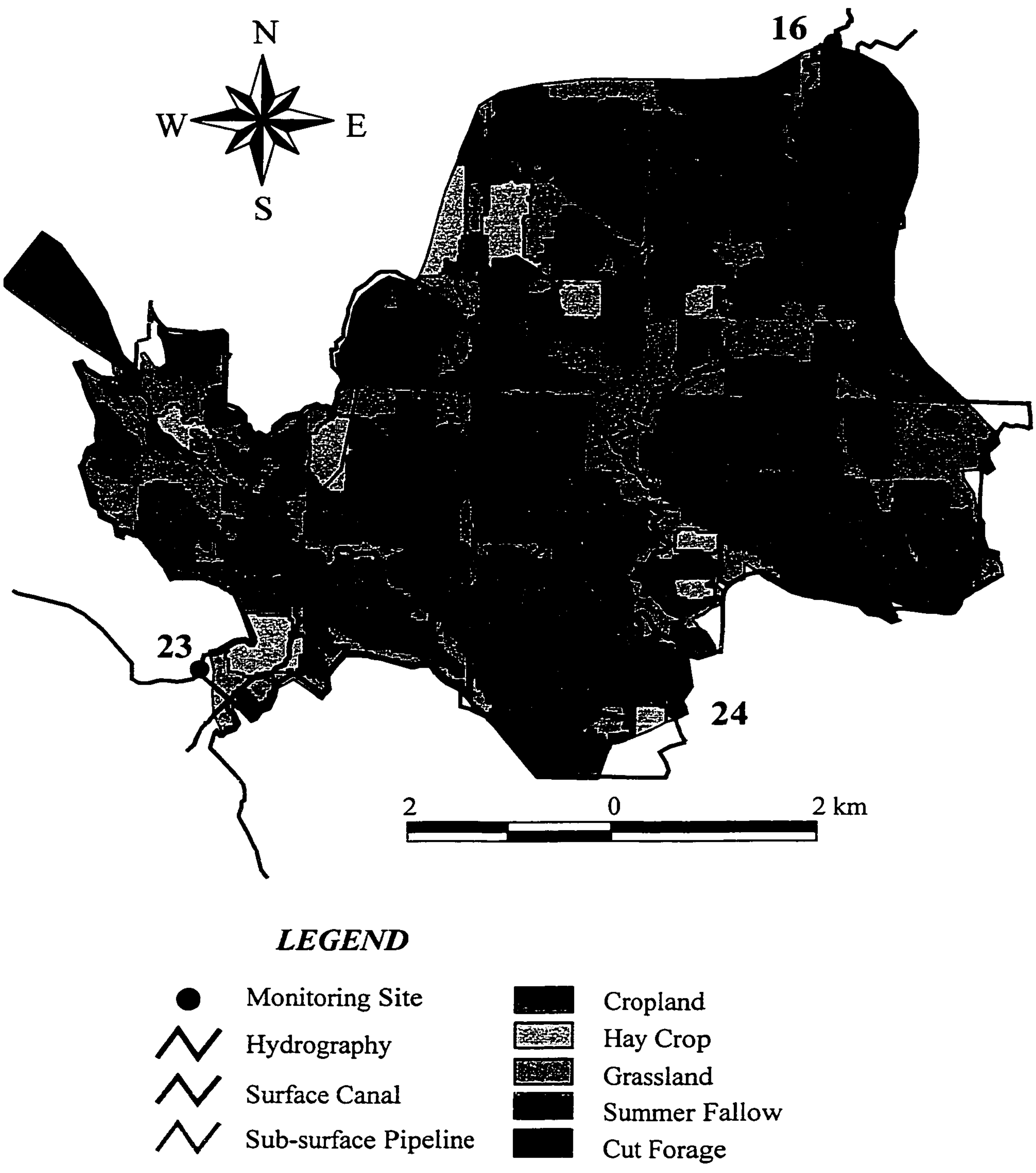

Figure 2.10. Sub-basin 16 landcover.

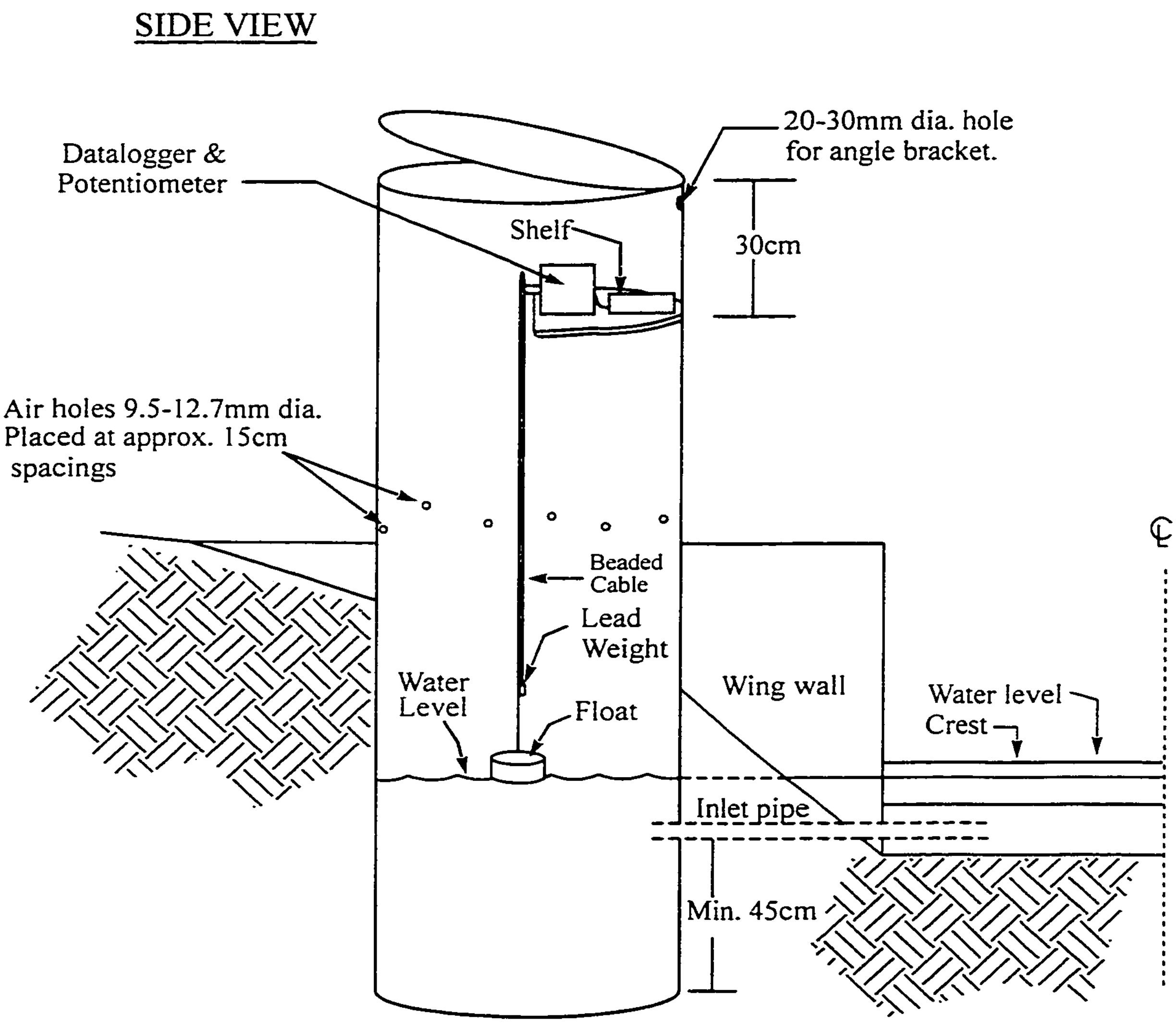

Figure 2.11. Typical hydrometric stilling well.

3. HYDROLOGY OF IRRIGATED PRAIRIE WATERSHEDS

3.1 Introduction

Only a small percentage of the water flowing in streams is the result of precipitation landing directly on the stream; the majority of streamflow comes from runoff from rain or snowmelt on upland surfaces (Gordon et al. 1997). Spring snowmelt and yearly rainfall are the primary hydrological forces on the prairies. Melting snow in the mountains results in flow for a portion of the year in the major rivers on the prairies. In watersheds that are distant from the mountains, flows due to snowmelt are limited to the spring of each year. Flows after spring runoff are primarily the result of rainfall events, although groundwater discharge is also a contributing factor. Many prairie streams are intermittent or ephemeral and may stop flowing during some parts of the year.

Prairie spring runoff flows are short duration, high volume flows resulting from snowmelt runoff. Rainfall may contribute to these flows in some instances. Flows increase and decrease rapidly depending on the speed of snowmelt in response to variations in temperature and solar radiation. Water from snowmelt may evaporate, infiltrate, flow into surface storage or flow overland into the surface drainage system.

Snowmelt runoff accounts for more than 85% of the total runoff from agricultural watersheds in western Canada (Nicholaichuk 1967). About 80% of the streamflow in the major rivers and water stored on the prairies comes from the snowpack. Snowpacks on the prairies are typically thin, ranging from 50 to 150 mm of water equivalent, and aerially variable due to wind action. The wind tends to blow the snow into swales and gullies as well as into the lee of vegetation. The snow is usually dry with low density, but becomes denser when packed by wind action (Meier 1990). Snow deposition can be managed to some degree by agricultural cropping practices such as varying stubble height and the use of windbreaks.

During spring, conditions are favorable for large amounts of surface runoff. The soil is usually frozen, resulting in reduced rates of infiltration (Granger et al. 1984). Surface depressions will hold some of the runoff water and, with reduced rates of infiltration and sublimation, will fill quickly. Once the depressions are filled, overland flow can begin almost immediately and snowmelt floods can occur (Meier 1990). Depending on distribution of the snow and the antecedent soil moisture, from 50 to 80% of the snowpack will contribute to runoff (Chanasyk and Woytowich 1986). Kane et al. (1991) in studies in the arctic found that 50 to 66% of the water from the snowpack became runoff, 20 to 34% was evaporation and 10 to 19% was held in storage. The magnitude of the snowmelt runoff flows will be affected by any factors that affect the process of snowmelt.

Rainfall runoff accounts for the remainder of the runoff from agricultural lands. Rainfall on the prairies occurs in several forms. While there are periods of long duration, low intensity rainfall, there are also high intensity, short duration storms. These latter storms are potentially more damaging due to the increased risk of soil erosion associated with them. The more intense the storm, the greater the soil loss (Hargrave and Shaykewich 1997). Many factors can influence the shape of the hydrograph during a storm. The rising limb is affected most strongly by climatic factors such as rainfall intensity and duration, distribution of rainfall, direction of storm movement and type of precipitation and storm (Gray and Wigham 1970).

Both short and long duration storms have the potential to cause surface runoff and erosion. The short duration, high intensity storms can result in the rapid exceedence of the soil's infiltration rate, resulting in surface runoff beginning sooner than from a low intensity storm. High intensity storms will also cause the breakdown of the surface structure of the soil, again resulting in reduced infiltration. Long duration, low intensity storms may also result in surface runoff, but only if the storm duration is great enough to cause the infiltration rate to be exceeded.

Soil moisture conditions will influence the amount of runoff. If the antecedent moisture content of the soil is low, then infiltration is high. The point where the infiltration rate is exceeded and surface runoff begins may not be reached. Conversely, a high antecedent moisture content at the time of the storm will allow the point where the infiltration rate is exceeded and surface runoff begins to be reached sooner. This may also result in greater erosion than from the low intensity storm (Hargrave and Shaykewich 1997).

The degree of vegetative cover will influence the amount of runoff. A bare or sparsely covered soil surface, such as found prior to the seeding of crops in the spring or after harvest in the fall, is more vulnerable to surface runoff. A well developed cover of living or dead plant material will reduce the velocity of raindrops striking the soil surface. This will help to maintain infiltration rates by maintaining the aggregation of the soil particles for a longer time. If surface runoff begins, the plant cover, as well as plant residue, will reduce the velocity of the runoff allowing an increased period for infiltration. The reduced velocity will also reduce the energy available for movement of the soil particles. If velocity is reduced, some of the sediment entrained in the runoff water will be deposited.

After harvest, fields are susceptible not only due to reduced cover, but due to soil compaction. If the fields have not been cultivated, high traffic areas will be present due to machinery traffic. Infiltration in these areas will be reduced resulting in surface runoff starting sooner than in adjacent areas.

Streamflows may also be regulated by diversion canals and/or dams, but the effects of these structures are difficult to determine with the effects dependent on the degree to which regulated releases are returned to the river farther downstream (Gordon et al. 1997). In some areas, flow regulation can change the seasonal distribution of flow, reduce the incidence and severity of floods and decrease long-term average flows. Low-flow behavior is also impacted by increasing the duration and frequency of low-flow extremes, including periods of zero flow (Gordon et al. 1997). In arid areas, regulated flows during non-precipitation periods during the irrigation season will be in excess of natural flows due to additions of irrigation return flow water (Karchenko and Maddock 1982).

Return flows to streams vary due to changing demands on irrigation infrastructure. Precipitation will result in greater volumes of water being returned to the receiving stream as irrigation use decreases. The amount of water being diverted by the irrigation district will be reduced during these periods. Dry periods will result in greater volumes of water being diverted for irrigation use, but also greater volumes withdrawn from the infrastructure for use. Reservoirs can influence sediment movement, stream temperatures and water quality.

The objective of this study was to quantify flows and their spatial and temporal distribution for a portion of Crowfoot Creek watershed and two major irrigation district return flow locations within it. Flows were considered during three periods: spring runoff, post-spring runoff (PSRO) and event periods. Differences in the flows between the Crowfoot Creek and WID return flow sites were also examined. Flow patterns and quantities were examined to relate flows to water quality parameters.

3.2 Materials and Methods

Methods of data collection related to water quantity and precipitation, as well as the construction and operation of the flow monitoring stations used in this study were described in Chapter 2.

3.3 Results and Discussion

Flow occurred throughout the year in some portions of the Crowfoot Creek watershed due to springs and other forms of groundwater discharge. The majority of flow occured from the beginning of spring runoff, usually in mid to late-March, until the end of September when the WID stopped delivering water to the irrigation district. After spring runoff and prior to the irrigation district diverting water into the irrigation infrastructure, flows decreased to baseflow levels. After irrigation diversion stoped in late September, flow levels returned to baseflow levels (Fig. 3.1).

3.3.1 Spring Runoff

Total precipitation for the November to March period during each year of the study was presented in Table 2.2. The snowpack in 1997 was extensive and resulted in large amounts of water being released into the surface drainage system of the Crowfoot Creek watershed. Spring runoff began in Sub-basins 10 and 19 around March 20, 1997; March 26, 1998 and March 15, 1999. In 1997, spring flow volumes recorded at the Water Survey of Canada station at the exit of the Crowfoot Creek watershed were in the top 5% of flows recorded since 1953. Runoff volumes exiting both Sub-basins 10 and 19 during spring runoff in 1997 comprised 60% of the total flow volume recorded during the monitoring period (Tables 3.1 and 3.2).

Snowmelt runoff in the other two years of the study was not as pronounced. Snowcover in both 1998 and 1999 was lower than in 1997. While total precipitation from November to March in 1998 was similar to that recorded in 1997, most fell during a late winter storm in March (Table 2.2). Much of the moisture from this storm sublimated, leaving little to run off. Due to the scattered snowcover in the spring of 1998 and 1999, the soil was likely exposed to higher temperatures, resulting in thawing of the soil surface. High infiltration rates may have resulted, allowing additional moisture to infiltrate rather than run off. Runoff volumes from Sub-basins 10 and 19 in 1998 made up 26 and 2%, respectively, of the total monitored flow volume. In 1999, runoff volumes recorded at Sub-basins 10 and 19 were 15 to 2%, respectively, of the total monitored runoff volume (Tables 3.1 and 3.2).

Flows generated from within the sub-basin were calculated as the difference between the inflows and outflows of the sub-basin. Negative values indicated large withdrawals of water between inflow and outflow during the season, as well as little contribution from spring runoff. There was no spring runoff at Site 13, the inflow into Sub-basin 10, in any of the three years (Table 3.3). Consequently, all of the water flowing out of Sub-basin 10 was generated from within the sub-basin (Table 3.4). Volumes in 1997 were much greater than in 1998 or 1999.

In Sub-basin 19, the flow generated from within the sub-basin was the difference in spring runoff flows between Site 11 (Table 3.5) and Site 19 during spring runoff. In 1997, 50% of the flow generated from within Sub-basin 19 flowed out during spring runoff (Table 3.6). In 1998 and 1999, spring runoff flows were much lower and -7% and 2% of the flow passed out of the sub-basin during spring runoff. The "negative" runoff during 1998 indicates this sub-basin contributed little in the way of surface runoff that spring. Water that entered the sub-basin from upstream may have filled several of the wetland areas along this reach of Crowfoot Creek. In 1999, while more water passed out of the sub-basin than entered, little runoff occurred.

Peak flows in Sub-basins 10 and 19 during spring runoff showed the same pattern as flow volumes over the three years. In 1997, peak flows ranged from 4.331 m^3 s^{-1} in Sub-basin 10 to 5.094 $m^3 s^{-1}$ in Sub-basin 19. In 1998 and 1999, peak flows were well under 1 $m^3 s^{-1}$ (Tables 3.1 and 3.2). The high flows in 1997 indicate large amounts of surface runoff occurred. In 1998 and 1999, the amount of surface runoff was reduced, as the snow cover was considerably less and was located primarily in sheltered areas such as in draws and along the creek itself where the snow collected due to wind action. Melting in these sheltered areas occurred more slowly than in areas exposed to the sun and peak flows were, therefore, lower. The amount of actual surface runoff area was limited primarily to these sheltered areas as well. In 1999, surface runoff was made up primarily of the snow and ice melting out of the creek channel.

The WID return flow sites followed a pattern similar to that for the Crowfoot Creek sites during spring runoff. As with Sub-basin 10, neither of the WID sub-basins had monitored flow entering from upstream. In 1997 in Sub-basin 14, 53% of the total flow volume passed the site in the spring, while at Sub-basin 16, 76% of the flow volume was produced during springmelt. In 1998 and 1999, the percent of the total flow volume in the spring was considerably lower (Tables 3.7 and 3.8).

As at the Crowfoot Creek sites, peak flows in the WID sub-basins during springmelt were indicative of the amount of runoff produced. In 1997, peak flows ranged from 1.995 $m^3 s^{-1}$ in Sub-basin 14 to 3.556 $m^3 s^{-1}$ in Sub-basin 19. In 1998 and 1999, peak flows did not exceed 0.764 $m^3 s^{-1}$.

3.3.2 Post-spring Runoff (PSRO)

In prairie watersheds, the majority of flow occurs during spring runoff. The post-spring runoff period would normally have low flows of groundwater, some periods of rainfall-runoff flows and in some cases flows cease. In the Crowfoot Creek watershed, however, the creek is used as a receiving channel for return flows from the WID. As a result, flow in the creek is almost continuous from the time that water is diverted to the irrigation system in May until the end of September when diversion stops. Flows decrease to a baseflow level during the period between the end of spring runoff and the start of diversion into the WID infrastructure and again in October after the WID has stopped diverting water to the infrastructure. Kuhnke (1986) described a similar pattern of flows in other irrigation districts in southern Alberta, with flows increasing from May through the summer months until diversions decrease in late summer.

While irrigation return flows dominate the post-spring runoff period, the volume and timing of the flows are dictated by consumer demand for water. Changes in demand generally correspond with the pattern of water demand by irrigated crops. Periods of high precipitation or low crop demand result in a decreased demand for water, while periods of low precipitation or high crop demand result in increased demand for water. The greater the demand for water, the greater the amount of water diverted into the irrigation infrastructure for use.

Overall, the pattern of flows during the PSRO period indicates an increase in the mean daily flow (MDF) from May until mid- to late July as the growing season progresses. MDF then decreases during August, but then increases again in September (Fig. 3.1). Demand generally increases as the growing season progresses. By mid- to late summer, crops have matured to the point that further irrigation is not required. In September, flows will increase in response to late season irrigation, water storage for domestic use and the filling of dugouts for livestock over winter.

The overall pattern of the MDF gives a view of the flow pattern over a longer term of weeks and months. The long term variation in flow is influenced by factors such as changes in crop water demand due to growth stage and variation in temperatures, timing of seeding and rainfall timing and amounts. The various peaks and valleys in the hydrographs are a reflection of the short term changes in demand for water. These peaks can be affected by individual management of the water.

In general, PSRO flows in the Crowfoot Creek sub-basins varied gradually over time and in response to demand for irrigation water. Increases and decreases in MDF occurred over a period of weeks, with peaks and valleys in the flows indicative of short term variation. Each year, however, will produce an individual use pattern (Fig. 3.1). This pattern is affected most strongly by the amount and timing of precipitation. In 1997, the large spring runoff supplied early growing season moisture. This was followed by a significant rainfall in late

May. Flows then remained fairly constant until August when they decreased. In 1998, the low amount of spring snowmelt moisture resulted in an immediate water requirement in the spring. Flows at the study sites increased steadily until late June. At this time a major rainfall occurred reducing demand. Following the event, flows decreased until September. The exception to this was in Sub-basin 10, where flows increased in August and then remained steady until late September (Fig. 3.2). Springmelt in 1999 also supplied little soil moisture and demand for irrigation water was high. However, rainfall from May through mid-July reduced the irrigation requirement. While flows tended to increase until mid-July, high flows in this time period were the result of the large rainfalls in addition to the normal irrigation flows. Flows from mid-July to the end of the season tended to decrease.

Total PSRO flows passing out of Sub-basin 10 accounted for 40 to 85% of the total flows during the three full years of monitoring (Table 3.1). In 1996, there was no flow data prior to July; hence, all the recorded water was from the PSRO period. However, when the inflows from Site 13 (Table 3.3) were subtracted from the total, more water flowed into Sub-basin 10 in 1998 and 1999 than flowed out (Table 3.4). The opposite was true for 1996 and 1997. This appeared to be an indication of a high demand for water within this sub-basin in 1998 and 1999, with more water withdrawn for irrigation and other uses than passed out of the sub-basin.

PSRO flows in Sub-basin 19 accounted for 40 to 98% of the total monitored flows during the three full years of monitoring (Table 3.2). Half of the volume was generated from within the sub-basin during the post-spring runoff period in 1997 (Table 3.6). For the remaining two years, from 98 to 107% of the flow was generated from within the sub-basin. The 107% flow is due to the negative flow volumes in the spring of 1998. The water generated from within the sub-basins can come from WID irrigation flows that are not monitored, as well as surface runoff during precipitation.

Peak flows during post-spring runoff in the four years ranged from 0.049 $m^3 s^{-1}$ to 2.103 $m^3 s^{-1}$ at Site 10 and from 0.467 $m^3 s^{-1}$ to 2.529 $m^3 s^{-1}$ at Site 19 (Tables 3.1 and 3.2). Peak flows were associated with rainfall events, indicating that surface runoff occurred during some of the rainfall events recorded in the watershed.

Flow at the WID return flow sites followed a pattern similar to that at the Crowfoot Creek sites. As at the Crowfoot Creek sites in 1997, PSRO volumes made up a smaller percentage of the total flow volume due to the high spring runoff volumes. In 1998 and 1999, post-spring runoff flows dominated the flow pattern at the WID sites (Tables 3.7 and 3.8).

During PSRO at the WID sites, all but one peak flow was associated with rainfall. This flow occurred late in the irrigation season and may have been associated with high flows observed at the end of the irrigation season. Peak flows over the four years at Site 14 ranged from 0.210 $m^3\ s^{-1}$ to 0.269 $m^3\ s^{-1}$. At Site 16, the peaks flows ranged from 0.239 $m^3\ s^{-1}$ to 1.042 $m^3\ s^{-1}$ (Tables 3.7 and 3.8). Flows observed at the WID return flow sites appeared to vary more in the short term than at the Crowfoot Creek sites (Figs. 3.1 and 3.3). Withdrawals and inputs due to irrigation and other uses had a more noticeable impact on the smaller stream volume in these channels.

3.3.3 Rainfall Events

As mentioned, most peak flows in any of the study years were associated with precipitation or snowmelt (Figs. 3.4 and 3.5). Rainfall effects included sharp increases in the flow, with peak flows occurring soon after. As initially, the only additional water entering Crowfoot Creek or the WID return flow channels came from surface runoff, it appeared that surface runoff occurred during many of the larger rainstorms. Surface runoff was also the result of rainstorms of lower intensity, but high overall amount. If much of the sub-basin had recently been irrigated, a rainstorm of smaller total amount could cause runoff to occur due to the high antecedent moisture content of the soil. Some of the lower intensity rainstorms may have been responsible for peak flows occurring later in an event or shortly after the event was completed.

High flows may also have been the result of farmers reducing withdrawals of water for irrigation. The high flows resulting from surface runoff may decrease more slowly than in a natural setting due to the increased volume of water being sent through the irrigation return flow channels. Additionally, when flows level off after the rainfall event, they may initially do so at a level higher than when the rainfall began. As the irrigation district adjusts the amount of water diverted to the system, these flows vary again.

Understanding the cause and timing of high flows is important when an examination of the water quality is carried out. If high flows were the result of surface runoff, water quality would be impacted differently than if the high flows were the result of an irrigation district adjustment of the flow.

3.4 Conclusions

Spring runoff flows were short duration flows generated by snowmelt runoff in the spring, while PSRO flows occurred from the end of the spring runoff period until the end of October. PSRO flows varied in response to demand on the WID irrigation infrastructure based on crop growth and the amount and timing of rainfall. Flow variability appeared greater in the WID return flow sub-basins than the Crowfoot Creek sub-basins due to the greater impact of on-farm irrigation management on the former flows.

Rainfall events were often associated with sharp increases in flows. Increases in flows were initially a result of surface water entering Crowfoot Creek or the WID return flow channels. As the storm progressed, the increased volume of water may also be attributed to less water being used for irrigation and being passed on to the creek via irrigation return flow channels. Peak flows occurred soon after significant amounts of precipitation.

In 1997, the large volume of runoff due to snowmelt dominated the flows for that year, but in the remaining years of the study, irrigation district flows in the PSRO period were the dominant source of water within the sub-basins studied. Some rainfall events during the PSRO period resulted in surface runoff in all years of the study.

3.5 References

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Table 3.1. Total flow parameters at Site 10

Year	Spring runoff volume (m^3)	% of total flow	Peak Flow ($m^3\ s^{-1}$) (Date)	Post-spring runoff volume (m^3)	% of total flow	Peak flow ($m^3\ s^{-1}$) (Date)	Total flow volume (m^3)
1996	No data	No data	No data	567,512	100%	0.187 (Aug 6)	567,512
1997	1,857,540	60	4.331 (Mar 21)	1,258,510	40	2.103 (May 26)	3,116,050
1998	217,797	26	0.680 (Apr 1)	621,870	74	0.318 (July 4)	839,667
1999	15,236	15	0.031 (Mar 25)	85,416	85	0.049 (June 26)	100,652

Table 3.2. Total flow parameters at Site 19

Year	Spring runoff volume (m^3)	% of total flow	Peak Flow ($m^3\ s^{-1}$) (Date)	Post-spring runoff volume (m^3)	% of total flow	Peak flow ($m^3\ s^{-1}$) (Date)	Total flow volume (m^3)
1996	No data	No data	No data	2,186,182	100	0.467 (July 26)	2,186,182
1997	5,782,208	60	5.094 (Mar 24)	3,931,519	40	1.635 (May 29)	9,713,726
1998	115,933	2	0.276 (Mar 30)	4,600,235	98	2.529 (June 12)	4,716,168
1999	77,801	2	0.131 (Mar 15)	4,700,278	98	2.266 (July 18)	4,778,080

Table 3.3. Total flow parameters at Site 13

Year	Spring runoff (m^3)	Peak flow (m^3 s^{-1}) (Date)	Post-spring runoff (m^3)	Peak flow (m^3 s^{-1}) (Date)	Total volume (m^3)
1996	0	0	470,441	0.246 (July 15)	470,441
1997	0	0	665,011	0.232 (July 14)	665,011
1998	0	0	1,127,113	0.333 (May 25)	1,127,113
1999	0	0	529,484	0.207 (Aug 4)	529,484

Table 3.4. Within-basin flow parameters at Site 10

Year	Spring runoff volume (m^3)	% of total flow	Post-spring runoff volume (m^3)	% of total flow	Total flow (m^3)
1996	No data	No data	97,071	100	97,071
1997	1,857,540	76	593,499	24	2,451,039
1998	217,797		-505,243		-287,446
1999	15,236		-444,068		-428,832

Table 3.5. Total flow parameters at Site 11

Year	Spring runoff (m^3)	Peak flow (m^3 s^{-1}) (Date)	Post-spring runoff (m^3)	Peak flow (m^3 s^{-1}) (Date)	Total volume (m^3)
1996	No data	No data	1,515,730	0.433 (Aug 8)	1,515,730
1997	3,659,699	5.351 (Mar 26)	1,814,673	0.323 (May 29)	5,474,372
1998	309,907	0.372 (Mar 30)	1,774,491	2.563 (July 5)	2,084,397
1999	26,155	0.071 (Mar 15)	2,474,777	0.676 (July 18)	2,500,933

Table 3.6. Within-basin flow parameters at Site 19

Year	Spring runoff volume (m^3)	% of total flow	Post-spring runoff volume (m^3)	% of total flow	Total flow (m^3)
1996	No data	No data	670,452	100	670,452
1997	2,122,509	50	2,116,846	50	4,239,355
1998	-193,974	-7	2,825,744	107	2,631,771
1999	51,646	2	2,225,501	98	2,277,147

Table 3.7. Total flow parameters at Site 14

Year	Spring runoff (m^3)	% of total flow	Peak flow (m^3 s^{-1}) (Date)	Post-spring runoff (m^3)	% of total flow	Peak flow (m^3 s^{-1}) (Date)	Total volume (m^3)
1996	No data	No data	No data	471,175	No data	0.226 (Aug 5)	471,175
1997	1,039,129	53	1.995 (Mar 26)	926,916	47	0.210 (July 1)	1,966,045
1998	27,943	4	0.069 (Apr 1)	641,699	96	0.224 (July 6)	669,642
1999	4,462	0.4	0.018 (Mar 15)	1,075,887	99.6	0.269 (July 15)	1,080,349

Table 3.8. Total flow parameters at Site 16

Year	Spring runoff (m^3)	% of total flow	Peak flow (m^3 s^{-1}) (Date)	Post-spring runoff (m^3)	% of total flow	Peak flow (m^3 s^{-1}) (Date)	Total volume (m^3)
1996	No data	No data	No data	1,183,194	No data	0.512 (Sept 18)	1,183,194
1997	5,177,634	76	3.566 (Mar 26)	1,663,347	24	0.239 (Oct 2)	6,840,981
1998	466,600	11	0.764 (Apr 2)	3,693,833	89	1.042 (July 5)	4,160,433
1999	110,666	4	0.241 (Mar 17)	2,768,393	96	0.977 (July 2)	2,879,059

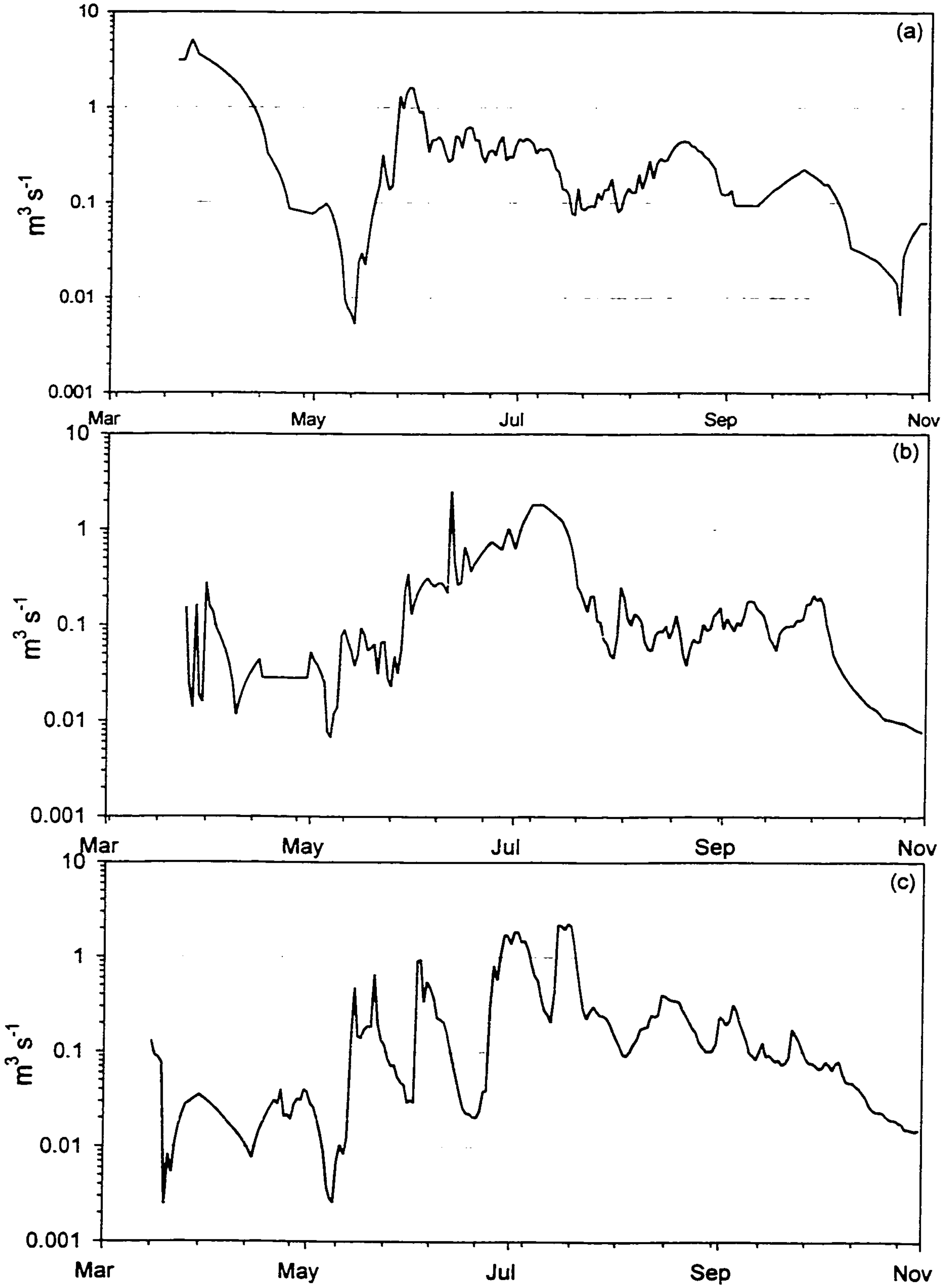

Figure 3.1. Flow hydrographs for Site 19 in a)1997 b)1998 c)1999.

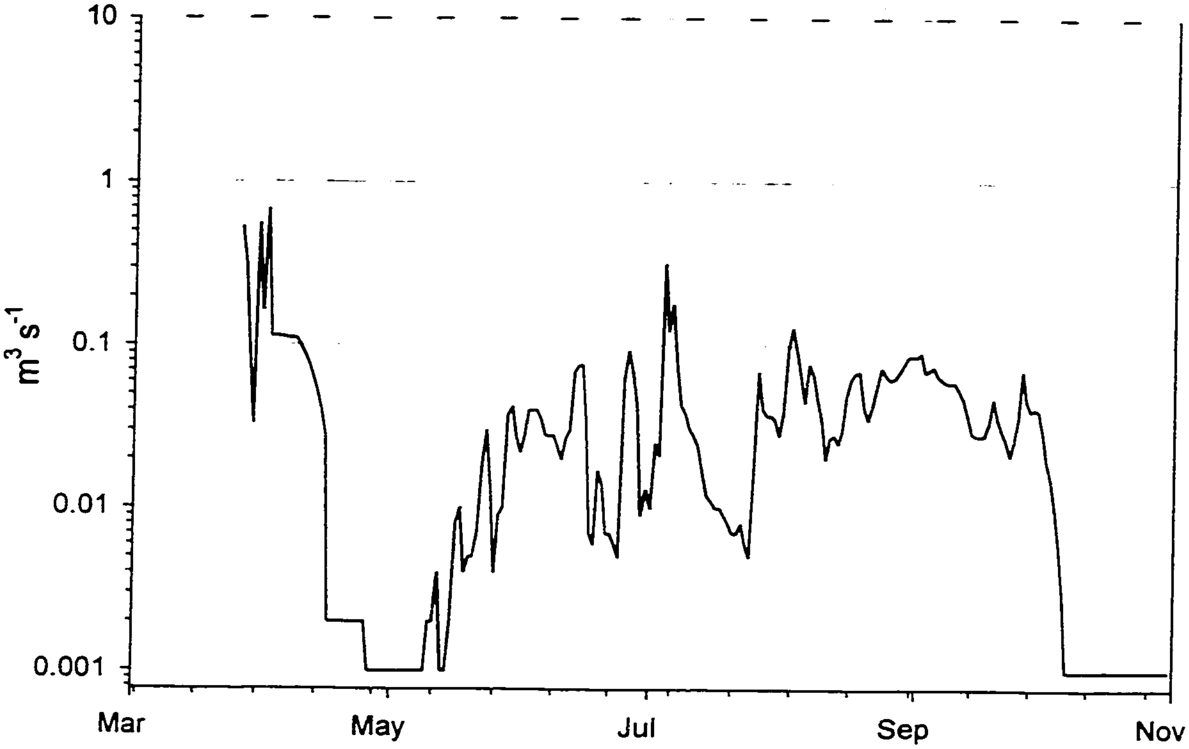

Figure 3.2. Flow hydrograph for Site 10 - 1998.

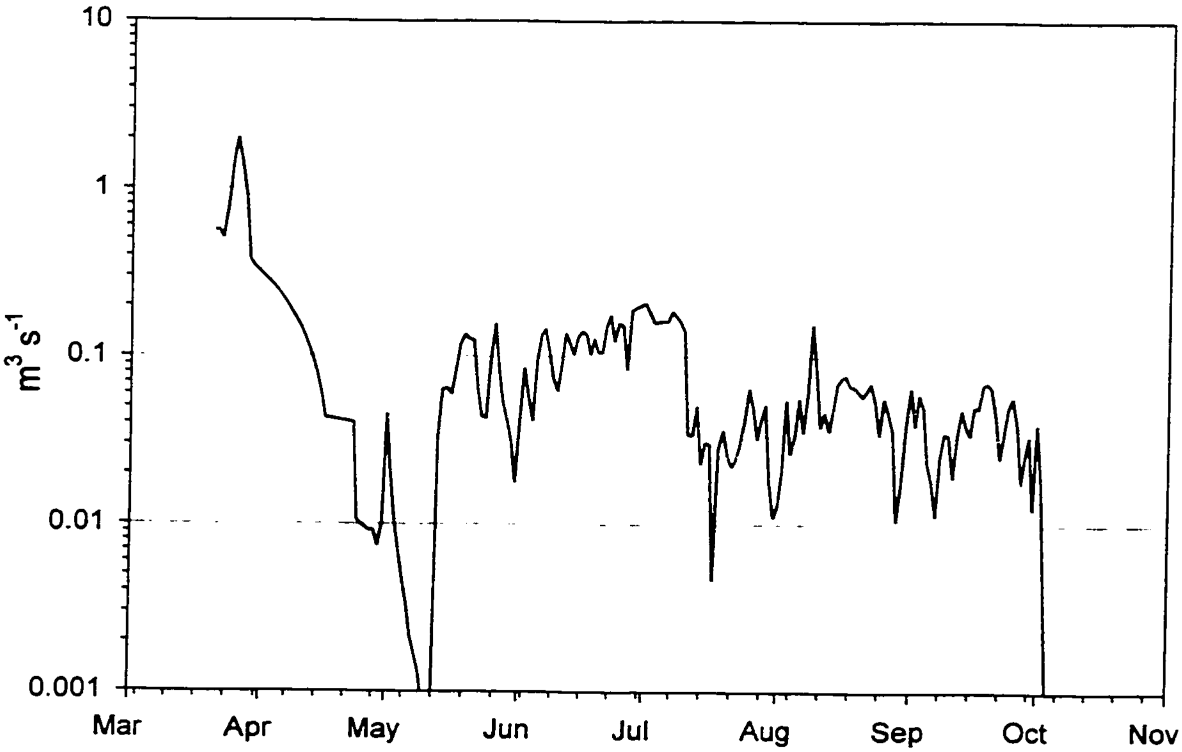

Figure 3.3. Flow hydrograph for Site 14 - 1997.

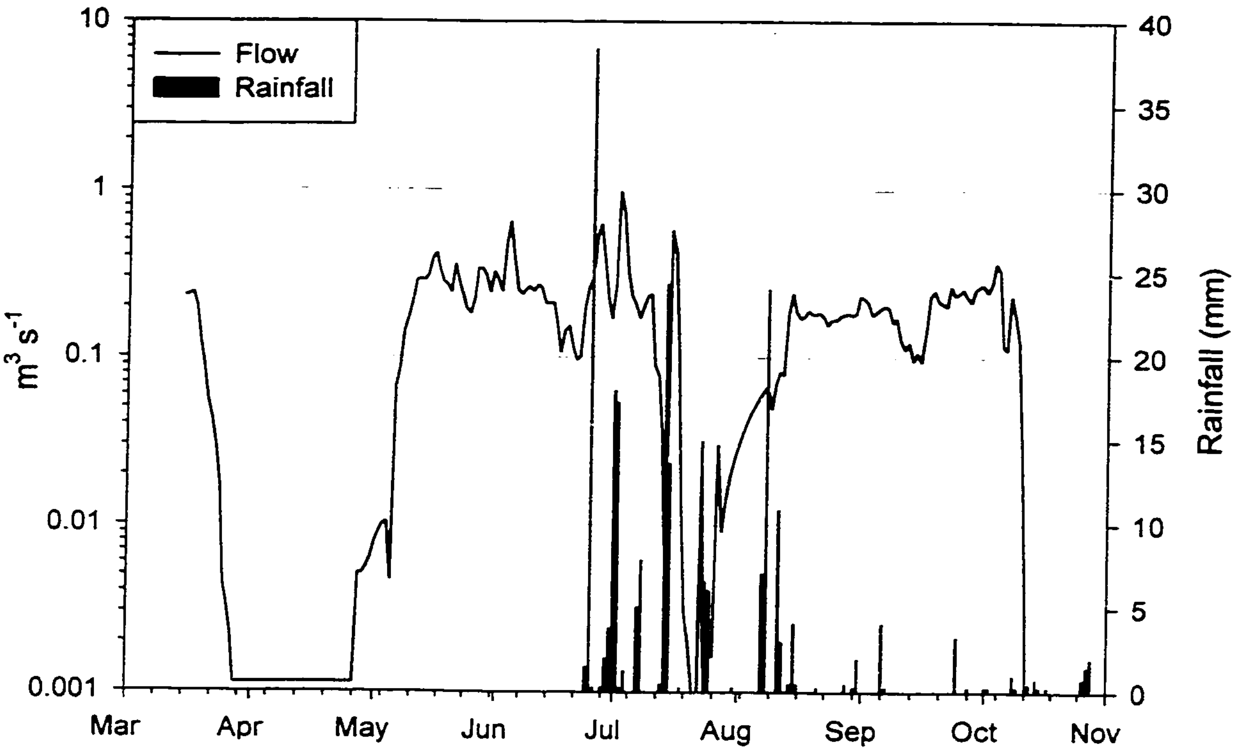

Figure 3.4. Flow hydrograph and rainfall for Site 16 - 1999.

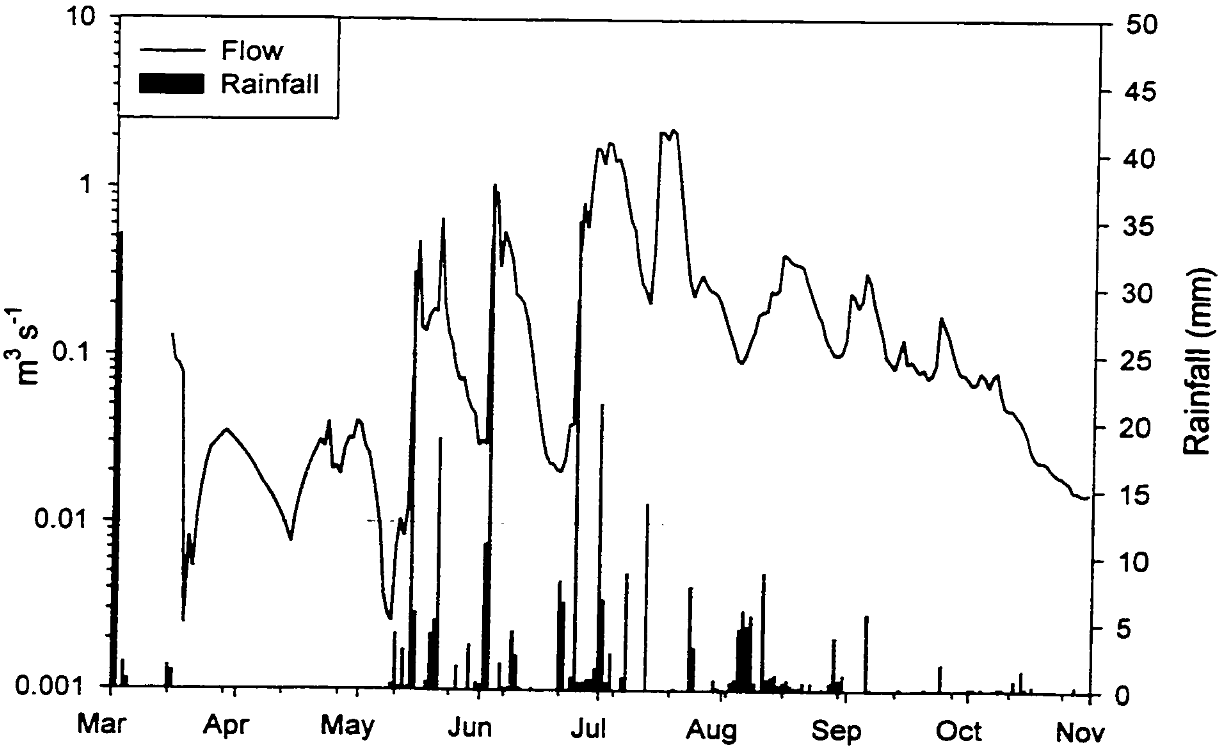

Figure 3.5. Flow hydrograph and rainfall for Site 19 - 1999.

4. IMPACT OF NUTRIENTS ON SURFACE WATER QUALITY IN AN IRRIGATED PRAIRIE WATERSHED I: PHOSPHORUS

4.1 Introduction

Elevated concentrations of nutrients such as phosphorus (P) and nitrogen (N) can accelerate aquatic weed and algal growth in surface waters (Cross and Cooke 1996). The enrichment of water with nutrients is referred to as eutrophication and its impacts are seen in increased aquatic plant growth, oxygen depletion, release of phosphorus from sediments, pH variability, plant species and food chain effects and production of toxins (Sharpley et al. 1994; Cross and Cooke 1996). While eutrophication is a natural phenomenon, it can reduce functional water use and shorten lake life. Additionally, nutrients such as ammonia may be toxic to fish.

Crop lands are potentially a major contributor of nutrients to aquatic systems since a requirement for good crop growth is nutrient availability. Lake water concentrations of P between 0.01 and 0.02 mg L^{-1} are considered critical values above which eutrophication is increased. These values are an order of magnitude greater than the concentrations of P required in the soil for plant growth (Sharpley 1997). The increasing intensity of land use for crop production is reflected in higher nutrient levels in rivers and lakes (Cooper 1993; Heathwaite 1995). Runoff from feedlot and grazing operations is also a contributor of N and P to surface water. In Alberta, many lakes and other surface water bodies are naturally highly eutrophic and sensitive to further enrichment by P (Howard et al. 1999).

4.1.1 Transportation Mechanisms

The two main pathways of nutrient transport from agricultural land are surface and subsurface flow. Surface flow will transport nutrients attached to sediment particles as well as dissolved soluble nutrients. Subsurface flow will transport mainly soluble nutrients (Heathwaite 1995).

Phosphorus can be transported in either soluble or particulate form. Transport of soluble P is a result of the desorption, dissolution and extraction of P from soil and plant material (Sharpley and Halvorson 1994). These processes take place in a thin layer of surface soil and the P is transported with the surface runoff water. Any P that leaches into the soil profile is attenuated in the subsoil by chemical precipitation of calcium phosphates.

Particulate P includes all P adsorbed to soil particles and organic matter. Transport occurs during surface runoff as these materials are moved over the soil surface and into the water (Sharpley and Halvorson 1994). Due to the selective transport of organic matter and clay-sized particles during surface runoff, particulate P makes up a large percentage of the P removed from a watershed (Marston 1989).

4.1.2 Phosphorus

Phosphorus (P) is a nonmetallic element which can exist in organic and inorganic forms and be present in dissolved or particulate species. Interest in the P content in water stems from the major role P plays in biological metabolism. P is an essential element in plant growth and necessary for crop production (McNeely et al. 1979; Sharpley 1997). Since P occurs in lower concentrations than other major nutrients required by biota, it most commonly limits biological productivity (Wetzel 1983). P is more likely to adsorb to soil particles and is, therefore, less likely to leach into groundwater and is more likely to contaminate surface water than groundwater (Cross and Cooke 1996).

Organic particulates and dissolved organic compounds are released from organic matter in plants and residues. Particulate P is that fraction removed by a 0.45 μm filter. Dissolved inorganic phosphorus is released from available phosphorus. The inorganic particulate component is made up of minerals derived from geochemical weathering of parent material, principally apatite (Wetzel 1983; Marston 1989).

Inputs of P can include animal manure, phosphate detergents, soil erosion, drainage from agricultural land, domestic and industrial sources and atmospheric deposition (Wetzel 1983; CCREM 1987; Cross and Cooke 1996).

Dissolved P includes:

- Orthophosphate (PO_4). This is the most biologically available form of P;
- Polyphosphates. Complex organic phosphates with a high molecular weight;
- P attached to organic colloids; and

- P contained in low molecular weight organic compounds.

Particulate P includes:

- P found in organisms including DNA, RNA, vitamins, ATP and ADP;
- P adsorbed onto mineral phases of rock and soil;
- P adsorbed onto detritus; and
- P that is biologically bound.

Sharpley et al. (1994) stated that particulate P contains the bulk of the P transported from conventionally tilled land, estimated to range from 75 to 95%. Runoff from grass and forest land carries less sediment and is dominated by dissolved P. Cooke and Prepas (1997) found that much of the P from agricultural lands in Alberta is in the form of dissolved P. Particulate P is considered a long term source of P in the water, while dissolved P is immediately bioavailable (Sharpley et al. 1987).

The exchange of phosphorus between water and sediments is a major component of the phosphorus cycle in natural waters (Wetzel 1983). Phosphorus in the sediments occurs mainly as apatite, nonapatite inorganic phosphorus, organic phosphorus and orthophosphate associated with hydrated ferric oxides. Under aerobic conditions, phosphorus is associated with the sediments, especially iron and mangenese. Under anaerobic conditions, reduction of these compounds takes place releasing the phosphorus, as well as the iron and mangenese back into the water where it is available for uptake by plants (Wetzel 1983; CCREM 1987).

In natural surface water, phosphate concentrations between 0.01 and 0.05 mg L^{-1} are typical (CCREM 1987). Values range from 0.001 to over 200 mg L^{-1}. There is generally a large spatial and temporal variation in phosphorus concentrations in Canadian surface waters. Elevated levels can occur due to drainage through areas high in phosphatic rock content, or rich in organic matter (CCREM 1987).

The Alberta Surface Water Quality Guidelines (Alberta Environment 1999) have set 0.05 mg L^{-1} as the maximum allowable concentration of total phosphorus in surface water in Alberta.

4.1.3 Contribution of Phosphorus From Agricultural Land

Amounts of P transported from uncultivated land are considered background and cannot be reduced. These levels may be sufficient to cause eutrophication of surface water. Due to the limited information on the losses of P from areas before cultivation, it is difficult to quantify the increase in P loss following the introduction of cultivation (Sharpley and Halvorson 1994; Abrams and Jarrell 1995). Intensive agricultural production is a major source of phosphorus load to lakes from watersheds with a high agricultural land use (Logan 1982). The risk of P export from the land to the water increases with an increasing soil test P level (Howard et al. 1999). Losses are influenced by the rate, time and method of application, form of fertilizer, amount and time of rainfall after application and vegetative cover (Sharpley and Halvorson 1994).

Several studies have examined the impact on P loss under various management schemes. Studies by Logan (1982), Sharpley (1993) and Sharpley et al. (1995) found that losses of bioavailable P from no-till fields were lower than from conventionally tilled fields. While conservation tillage methods were found to reduce total P losses, the level of bioavailable P losses increased. Sharpley et al. (1987) found that the respective amounts of N and P transported from crop rotation watersheds were 1.7 and 1.8 times greater than fertilized continuous wheat and 2.3 and 4.6 times greater than from unfertilized native grass.

Smith et al. (1993) reported that cover crops reduced the loss of particulate nutrients, but in some cases soluble P loss increased, due partly to the contribution of P from vegetative material. The application of animal manures can result in an increase in surface soil accumulations of P (Smith et al.1993; Howard et al. 1999).

The objective of this chapter was to determine if phosphorus concentrations and mass loads varied spatially or temporally within selected sub-basins along Crowfoot Creek and two irrigation return flow channels, and to determine if water quality within these sub-basins is affected by various types of agricultural landcover. Compliance with water quality guidelines was evaluated. The impact of irrigation flows on water quality in Crowfoot Creek was also examined.

4.2 Methods

Methods of data collection and analysis were presented in Chapter 2.

4.3 Results and Discussion

4.3.1 Spring Runoff - Crowfoot Creek Sub-basins

In Sub-basin 10, no data were collected during spring runoff at Site 13 (inflow) due to lack of a drainage basin to contribute runoff. At the outflow, Site 10, median values of total phosphorus (TP) were fairly consistent, ranging from 0.494 to 0.642 mg L^{-1} over the three spring runoff periods (Table 4.1). Flow-weighted mean concentrations (FWMC) were slightly higher in 1998 than 1997, and significantly higher in 1999 than 1998 (Fig. 4.1).

The trend in mass load followed an inverse pattern to that of FWMC, with the load in 1997 being higher than in 1998 or 1999 (Fig. 4.2). Although the FWMCs in 1997 and 1998 were similar, flow volume in 1997 was more than eight times that in 1998 (Table 4.1), resulting in much higher mass loads. In 1999, flow volumes were much lower than in the previous two years, so even with the highest FWMC of the three spring runoffs, the mass load of TP was the lowest.

Total dissolved phosphorus (TDP) made up the majority of the TP detected in Sub-basin 10 during spring runoff (Table 4.2): 80, 87 and 93% of the TP mass load in 1997, 1998 and 1999, respectively (Table 4.3).

In Sub-basin 19, median TP concentrations decreased between Site 11 (inflow) and Site 19 (outflow) in all three years. The decrease in concentration was not significant during spring runoff in 1997 or 1998; however, there was a significant difference in 1999 (Table 4.1). FWMCs of TP decreased between the inflow and outflow sites during spring runoff in all three years, with the largest decrease occurring in 1999 and the smallest in 1997 (Fig. 4.1). Mass loads increased between inflow and outflow during spring runoff in 1997, indicating a net export of TP. During spring runoff in 1998 and 1999, the mass load decreased within Sub-basin 19, indicating a net import of TP.

At Site 11, both the median concentrations and FWMCs of TP increased in each of the three years (Table 4.1; Fig. 4.1), while mass loads decreased over the same period (Fig.

4.2). At Site 19, median concentrations and FWMCs were higher in 1998 than 1997, and lowest in 1999. Mass loads at this site decreased each year, likely in response to the reduced volume of runoff.

Median concentrations of TDP were higher at Site 11 than at Site 19 in all three years, significantly in 1999 (Table 4.2). FWMCs were also higher at Site 11 than 19 in all three years of the study, as were mass loads. The proportion of the TP load made up of TDP was similar to Sub-basin 10. The percentages of TDP over the three years were 81, 90 and 89% at Site 11 and 87, 86 and 81% at Site 19, indicating little change in TDP within the sub-basin (Table 4.3).

Primary sources of phosphorus that may be transported during spring runoff are sediments and dissolved forms of P removed from the soil, soil organic matter and plant material. Sources of P in the creek and wetlands include P attached to sediments as well as dissolved forms which result from the decay of plant material and desorption from particulate matter (Peverly 1985; Yan et al. 1998). Dissolved forms of P dominate in the Crowfoot Creek sub-basins.

Variable runoff volumes (Table 4.1) produced over the three years of the study may have affected the amount of P that was transported to the creek as well as the amount of P exported from the sub-basins. Due to the extensive snow cover in 1997, a larger percentage of each sub-basin may have contributed to surface runoff, increasing the area that could potentially contribute P to the water via surface runoff. Increased erosion during spring runoff (Chanasyk and Woytowich 1986) could result in the transportation of additional P into the creek. Snow was observed to cover a smaller portion of the sub-basin area in 1998. As a result, there was likely less moisture available for runoff in 1998 than in 1997. The amount of snow-covered area was lowest in 1999 and was located mainly in sheltered areas and within the creek channel itself.

The greater volumes of runoff in 1997 and 1998 compared to 1999 resulted in a lower FWMC of TP at Sites 10 and 11. FWMCs at Site 19 were lower than at the other two sites, perhaps due to the impact of the numerous wetlands along the reaches above Sites 10 and 11. The nutrient content of the wetlands would be expected to increase due to the decay of plants over the winter months. As the runoff water moved through these areas, nutrients were removed. With high volumes of water, the FWMCs would be reduced. If the volume of water was not sufficient to flush the wetland areas, then the FWMCs would remain higher as in 1998 and 1999. The median values of TP in 1999 at Sites 10 and 11 were the highest of the three spring runoff periods.

TP decreased within Sub-basin 19 in all three years. The higher P concentrations at Site 11 compared to Site 19 may have been due to plant decay in the Hilton wetland upstream of Site 11 over the previous fall and winter. Both median concentrations and FWMCs of TP increased at Site 11 over the three years. The increase may have been the result of the reduced volume of runoff passing through the wetland in successive years. This resulted in reduced flushing of P out of the wetland. Higher concentrations of P may, therefore, have been maintained for a longer period of time.

At Site 19, the median concentrations and FWMCs decreased during 1999 after increasing from 1997 to 1998. There were fewer wetland areas within Sub-basin 19 compared to Sub-basin 10 and the area upstream of Site 11, resulting in a decrease in internal loading due to plant decay and sediment release. The concentrations of TP within Sub-basin 19 decreased due to dilution as the runoff water entering the creek between Sites 11 and 19 likely contained a lower concentration of P than the inflow water at Site 11. During springmelt in 1999, inputs from within Sub-basin 19 were reduced again as there was little surface runoff to transport P from the sub-basin land surface. The reduced volume of runoff entering the sub-basin at Site 11 introduced less P. In combination with the reduced inflow, the small amount of surface runoff diluted the water entering the sub-basin resulting in a significantly lower concentration of P within Sub-basin 19 in 1999.

Mass loads at the Crowfoot Creek monitoring sites varied with flow volume. In 1997, the large volume of water passing out of the sub-basins resulted in large mass loads. The loads decreased in the remaining two years as did the flow volumes. There was a net import of P into Sub-basin 19 in 1997, while in 1998 and 1999 there was a net export of P. There was not a large difference in the FWMCs between Site 11 and Site 19 in 1997, but the volume of water passing Site 19 was much higher. In 1998 and 1999, the FWMCs at Site

11 were much larger than those at Site 19. In 1998, combined with a higher flow, loads were higher at Site 11 than at Site 19. In 1999, flow volumes were higher at Site 19, but loads at the inflow and outflow sites were almost equal.

The percentage of TP made up by TDP is 80% or more at all of the Crowfoot Creek sites. The high percentage of TDP detected is in agreement with other studies in Alberta (Cooke and Prepas 1997; Anderson et al. 1998); however, studies in the United States indicate that particulate P is the dominant form of P removed from agricultural areas (Sharpley et al 1987; Sharpley et al. 1994; Douglas et al. 1998). The variations in TDP between years and sites may be due to several factors, including the decrease in runoff flow volume, the contribution of wetland areas through the decay of plant material and the landcover associated with each sub-basin.

In 1997, the high runoff volumes may have caused greater erosion of the soil surface and re-suspended sediments within the creek channel, resulting in more particulate matter being suspended in the water column being sampled. The greater the percentage of particulate P, the lower the percentage of TDP. In 1998 and 1999, reduced runoff volume may have been the result of reduced surface runoff. Less surface runoff will likely result in less sediment transport. There may also have been reduced scouring of the channel and a greater opportunity for instream sediments to settle out of the water column in the wetland areas due to reduced flow velocities.

TDP concentrations may have been impacted by the release of P in wetlands and sloughs along the creek in this area. Studies of highly organic soils detected high concentrations of reactive P during the spring thaw period (Perverly 1982). Flow volumes in the latter two years may not have been sufficient to flush the accumulated TDP out of the wetlands above Site 11. Within Sub-basin 19, there are fewer wetland areas and in association with dilution by runoff, the percentage of TDP decreased slightly each year.

The landcover in each sub-basin may also have had an effect on the composition of P in the water. In Sub-basin 10, grassland made up just over 40% of the landcover, with a large percentage of the area adjacent to the creek in grass cover. Grass strips act as a buffer strip and likely caused deposition of some sediment and related constituents before they reach the creek (Schmitt et al. 1999). The water reaching the creek will be dominated by the dissolved form of P. The grassland area also contains faecal material left by grazing cattle. As this material also contains high amounts of DP (Cooke and Prepas 1997), additional DP may be added to the surface runoff water as it flows through these areas (Jawson et al. 1982).

In contrast, the landcover of Sub-basin 19 was dominated by annual cropland. Areas adjacent to the creek were also dominated by cropland with lesser areas of grassland. The proximity of the cropland to the creek increased the potential for increased amounts of particulate matter, with P attached, to be transported into the creek. The grassland areas in Sub-basin 19 were not as wide or continuous as those in Sub-basin 10 and hence did not provide as large a buffer.

Sub-basin 10, over the three years of spring runoff, had an increasing percentage of TDP. Less surface runoff in 1998 and 1999 than in 1997, and the large grassland buffer area, may have affected this increase. In Sub-basin 19, the percentage of TDP decreased all three years, significantly in 1999. The decrease may have been due to the high inflow concentrations of TDP from Site 11 being diluted by water with lower levels of TDP within the sub-basin, as well as by the introduction of more particulate matter from within the sub-basin.

4.3.2 Spring Runoff - WID Return Flow Sub-basins

Neither Sub-basin 14 or 16 have inflow data for the spring runoff period. In Sub-basin 14, the median concentration of TP varied slightly from 0.418 to 0.356 to 0.337 mg L^{-1} over the three years (Table 4.1). The FWMCs over the same period remained fairly constant as well: 0.449 to 0.435 to 0.365 mg L^{-1} (Fig. 4.1). In Sub-basin 16, the median TP concentrations were more variable: 0.482 and 0.408 mg L^{-1} in 1997 and 1998, and 0.170 mg L^{-1} in 1999 (Table 4.1). The FWMCs also reflected this variation (Fig. 4.1).

Mass loads of TP in the WID return flow sub-basins were controlled by flow volume. High flow volumes in 1997 resulted in high mass loads in both WID sub-basins (Fig. 4.2). In 1998 and 1999, the flows and FWMC in Sub-basin 14 were similar, therefore, the mass loads did not vary. In Sub-basin 16 in 1998, while flows were similar to those in 1999, the

FWMCs were higher and as a result the mass loads were higher than in 1999.

TDP makes up the greatest portion of TP in the WID return flow sub-basins. In Sub-basin 14, TDP made up 89, 91 and 83 % of TP in 1997, 1998 and 1999, respectively, a trend similar to that for the Crowfoot Creek sub-basins (Table 4.3). In Sub-basin 16, the levels were 76, 69 and 69% respectively, slightly lower than the Crowfoot Creek sub-basins and Sub-basin 14 (Table 4.3).

As in the Crowfoot Creek sub-basins, sources of P during the spring consist of particulate and dissolved forms transported into the drainage channel by surface runoff and P in the drainage channel. The similar FWMCs in Sub-basin 14 over the three years indicated that runoff in this sub-basin carried a consistent amount of P and high volumes of water did not reduce the FWMC. Slopes in the upper reaches of this sub-basin were steep and snowmelt runoff flowed into the drainage ways fairly quickly, reducing the time available for infiltration. Reduced infiltration resulted in more water carrying P reaching the drainage channels, but less contact time between the water and the soil. Infiltrating water may reach the surface at some time, but some of the P may be retained in the soil and removed from the runoff water.

In Sub-basin 16, median concentrations and FWMC of TP were similar in the first two years when there was more surface runoff during snowmelt, but lower in 1999 when spring runoff consisted of more channel melt and less runoff from the surrounding agricultural land. This sub-basin has gentle overall slopes which may have allowed greater contact time between the soil and water and potentially removed more P.

Both WID sub-basins were dominated by annual cropland (Table 2.3), with the difference between the two being the location of the various landcovers in relation to the drainage channels. In Sub-basin 14, more of the area adjacent to the drainage channel was in grass cover than in Sub-basin 16 (Figs. 2.9 and 2.10). The main drainage channel of Sub-basin 14 was grassed to the top of the slopes in the steeper sloped areas. The lower, flatter areas had a greater percentage of cropland, but some portions of the drainage channel in these areas had grassland adjacent to them, especially just upstream of the monitoring site. The grassed areas likely acted as buffer strips, decreasing the amount of particulate matter reaching the creek.

In Sub-basin 16 there was more annual cropland cultivated to the edge of the return flow channel. The lower percentage of TDP exported from Sub-basin 16 compared to Sub-basin 14 may be due to the reduced amount of grass buffer areas adjacent to the drainage channel. As a result, more particulate matter may be able to enter the channel. Even though both sub-basins were dominated by cropland, the location of the various landcovers in relation to the drainage pathways may have influenced the effect these areas had on the overall water quality in the two sub-basins.

The FWMCs in the Crowfoot Creek sub-basins were much higher than those in the WID sub-basins, likely due to the buildup of nutrients in the sloughs and wetland areas of the Crowfoot Creek sub-basins over the fall and winter months. The WID sub-basins did not have as many of these areas along their length. Mass load values during spring runoff in the study sub-basins were dominated by the volume of water leaving the sub-basin.

4.3.3 Post-spring Runoff - Crowfoot Creek Sub-basins

4.3.3.1 Total Phosphorus

In Sub-basin 10, the TP concentrations at the inflow and outflow sites decreased during the post-spring runoff (PSRO) period with the exception of the event periods (Fig. 4.3). Concentrations were highest during the period between the end of spring runoff and the beginning of diversion by the WID and during the last month of the monitoring period. There was a significant increase of the median concentrations of TP between the inflow and outflow sites during the PSRO period in all years (Table 4.4).

The FWMCs followed a similar pattern to the median concentrations with values at Site 13 lower than those at Site 10 (Fig. 4.4). The TP FWMCs at Site 13 were similar to the levels in other WID canals delivering water to the Crowfoot Creek watershed. FWMCs at Site 13 was less than 0.10 mg L^{-1} in all years of the study, while at Site 10 the FWMCs were eightfold higher in 1997, fourfold higher in 1998 and fivefold higher in 1999 than the Site 13 values.

Within the PSRO period, FWMCs immediately after spring runoff tended to be higher than other time periods, with the exception of event periods. The lowest FWMCs were usually found in periods without a significant rainfall event or later in the PSRO period.

Mass loads of TP in Sub-basin 10 were influenced by the volume of water passing through the sub-basin. Due to larger volumes of water passing out of the sub-basin, the mass loads at Site 10 were larger than those at Site 13 in 1997 and 1998 (Fig. 4.5). In 1999, more water entered the sub-basin than flowed out, resulting in a net import of phosphorus into the sub-basin.

In Sub-basin 19, TP concentration decreased over the monitoring period in each year, except during rainfall events (Fig. 4.6). As in Sub-basin 10, the greatest concentration occurred during the period between the end of spring runoff in mid-April and the diversion of water by the WID in mid-May. There was also a slight increase in the concentration of TP at the end of the monitoring season. TP concentration decreased between inflow and outflow, but the decrease was not significant in any year of the study (Table 4.4).

FWMCs at Sites 11 and 19 were similar in 1997 and 1999, while in 1998 the FWMCs at Site 19 were obviously lower. The greater FWMC at Site 11 in 1999 was due to the high concentration of TP with a lower volume of water compared to Site 19 (Fig. 4.4). Mass loads increased between Site 11 and 19 as a function of the increased volume of water (Fig. 4.5). The PSRO values in 1998 for this sub-basin may have been affected by the large rainfall event in July.

The temporal trends in FWMC in Sub-basin 19 were similar to those in Sub-basin 10. The FWMCs in the time period following spring runoff were the highest except for those during the event period. The lowest FWMCs were during time periods where there had not been many significant rainfall events.

As in Sub-basin 10, the mass loads were dependent on the volume of water passing through the sub-basin. The greater volume of water exiting the sub-basin past Site 19 compared to entering the sub-basin at Site 11, resulted in higher mass loads in all years of the study and a net export of P from this sub-basin.

The temporal variation in the concentration of TP in the Crowfoot Creek sub-basins was a result of the amount of WID water being returned to Crowfoot Creek. During the

PSRO period, water from the WID system was the dominant source of water in the creek. In the period immediately after spring runoff, flows in the creek were low and were generated by the last snowmelt runoff from sheltered areas and groundwater discharge.

The levels of TP in the early stage of the monitoring period indicated that a base level of phosphorus existed in the creek. Sources may include P released from sediments in the channel as well as release from decaying plant material. The addition of irrigation return flow water with a lower concentration of phosphorus resulted in dilution and a gradual decrease in the overall concentration of P over the monitoring period. The cessation of return flows into the creek late in the PSRO period corresponded with an increase in TP concentration.

4.3.3.2 Total Dissolved Phosphorus

In Sub-basin 10, temporal trends of total dissolved phosphorus (TDP) were similar to the trends for TP. At Sites 13 and 10, high concentrations of TDP in the period following spring runoff gradually decreased until the end of the season, with the exception of event periods. The increase in median values between inflow and outflow sites was statistically significant in 1997, 1998 and 1999 (Table 4.5). TDP was not determined in 1996.

The FWMCs for TDP followed the same pattern as TP, with concentrations increasing greatly between inflow and outflow. The FWMCs for the period between spring runoff and the input of irrigation water tended to be the highest, while the periods without significant rainfall had the lowest FWMCs. This held true in 1997 and 1999. In 1998, the time period used in the FLUX model to examine spring runoff data extended beyond the period following spring runoff into the early portion of WID flows, resulting in a FWMC that was similar to the periods without rainfall. The percentage of TP as TDP at the inflow site ranged from 45 to 80% over the three years of monitoring (Table 4.6); at the outflow site, the percentage of TDP exceeded 87% in all three years.

Mass loads of TDP followed the same pattern as TP. There was a net export of TDP from this sub-basin in 1997 and 1998, while in 1999 there was an net import of TDP due to differences in flow volume.

The presence of TDP in Sub-basin 19 showed trends opposite those of Sub-basin 10. Concentrations decreased during the monitoring period, excepting event periods, with a slight increase at the end of the PSRO period. Inflow median values were significantly higher than outflow values in all years of the study (Table 4.5). The source of the TDP entering Sub-basin 19 may be the Ducks Unlimited wetland. The FWMCs over the PSRO period indicated a higher concentration of TDP at Site 11 than Site 19.

TDP made up a lower amount of TP load at Site 19 than at Site 11 (Table 4.6). In 1997, the difference was approximately 25%, while in 1998 and 1999 the differences were 4 and 6%, respectively.

The tendency for the decrease in TDP concentrations at Site 10 over the PSRO period was due to the dilution of P by better quality inflow water. As with TP, the creek was flushed out over the PSRO period, and the concentration of TDP decreased. In Sub-basin 10, the increase in the percentage of TDP within the sub-basin may be due to a number of reasons. The velocity of water in the canal supplying water to Site 13 was greater than that between Sites 13 and 10. The decrease in velocity as the water was diverted into Crowfoot Creek may have resulted in particulate matter settling out of the water, increasing the percentage of TDP. Several wetlands along this reach of the creek would have the same effect as the water velocity decreased within these areas. The wetlands were also an additional source of DP resulting from the breakdown of plant material.

Within Sub-basin 19, the decrease in TDP may be due to essentially the opposite reasons as the increase in Sub-basin 10. The water velocity was generally higher, the result of more water being passed through this sub-basin. More particulate material was suspended in the water. Fewer wetlands also resulted in less opportunities for material to settle out of the water and less input of nutrients from within wetlands.

4.3.4 Post-spring Runoff - WID Return Flow Sub-basins

4.3.4.1 Total Phosphorus

During the PSRO period, levels of TP in the water entering the Crowfoot Creek watershed from the WID irrigation infrastructure were fairly consistent from year to year.

Concentrations tended to decrease during the PSRO period (Fig. 4.7). The exception was in 1999 when TP increased at all sites in the latter part of the irrigation season. Median levels of total phosphorus for all three study years were less than 0.1 mg L^{-1} at all inflow sites (Table 4.7), with little variation from site-to-site or year-to-year. Site 23 exhibited the greatest variation.

TP FWMCs at the inflow sites were close to the AWQG of 0.05 mg L^{-1}. Several of the sites had FWMCs below this level (Fig. 4.8). Site 23 in 1997 and 1999, and Site 24 in 1998 and 1999 had FWMC in exceedance of the guideline during the monitoring period, but the level of exceedance was small as none of the FWMC exceeded 0.07 mg L^{-1}.

Mass loads were similar among sites, with the exception of Site 24. In both 1998 and 1999, the mass load was much higher at Site 24 than at the other inflow sites due to higher flow volumes (Fig. 4.9).

At the return flow sites, Sites 14 and 16, TP concentrations were highest early in the PSRO period with levels decreasing as the monitoring period continued (Fig. 4.10). Rainfall events sometimes resulted in increased concentrations. Median concentrations of TP in Sub-basin 14 increased significantly between inflow and outflow in all years (Table 4.7). In Sub-basin 16, median concentrations increased between inflow and outflow in all years. The increase was significant in all years between Site 24 and Site 16, and in 1998 and 1999 between Site 23 and Site 16. In 1997, the median TP value increased, but not significantly.

FWMCs of TP increased greatly within each of the sub-basins in all years of the study (Fig. 4.8). In Sub-basin 14, the inflow FWMC was consistent over the three years monitored, while outflow FWMCs were similar in the first two years of the study and higher in 1999. In Sub-basin 16, the inflows at Sites 23 and 24 were similar, as were the outflows.

Periods of time without significant rainfall had the lowest FWMCs. In 1997, the period following spring runoff had high FWMCs, while in 1998 this did not hold true. In 1999, the number of events in the early part of the monitoring season may have resulted in the FWMCs remaining high until August.

Mass loads over the PSRO period varied between inflow and outflow (Fig. 4.9). In Sub-basin 14 in 1997, more TP was imported than exported. In 1998, import and export were approximately equal, while in 1999 more TP was exported than imported. In Sub-basin 16, there was a greater amount of TP imported than exported into this sub-basin in 1997 and 1999; in 1998, the import and export values were similar.

The WID inflow water had low levels of TP compared to the concentrations detected at the return flow sites. The water quality deteriorated rapidly between inflow and return flow with P originating from the same sources as in the Crowfoot sub-basins. The flushing of the return flow channels by irrigation water resulted in the decrease in TP concentrations over time.

Concentrations of TP were lower in the WID return flow sub-basins compared to the Crowfoot Creek sub-basins, likely due to less inputs from other return flow sources as well as the lack of wetlands as a constant source of P in the water.

4.3.4.2 Total Dissolved Phosphorus

Temporal trends of TDP at the WID inflow sites in 1997 and 1998 indicated a decrease over the PSRO period. In 1999, there was no clear trend over the monitoring period and the overall TDP levels appeared higher than in 1997 and 1998. Increasing median values over the three years at the inflow sites tended to support this view, especially at Sites 23 and 24 (Table 4.8).

Median TDP values increased in all three years at all three inflow sites. The increase at Site 22 was smaller than at Sites 23 and 24, where median TDP concentrations increased twofold between 1997 and 1998, and more than sixfold between 1997 and 1999. Increases in median TDP concentrations at Site 22 increased more than threefold between 1997 and 1999. FWMCs of TDP were below 0.02 mg L^{-1} at all of the inflow sites during 1997 and 1998. As with the median concentrations, FWMCs increased in 1999 and were between 0.02 and 0.04 mg L^{-1} at all three sites.

Mass loads of TDP at Site 22 did not vary much over the three years of monitoring. Loads at Sites 23 and 24 were consistent, with the exception of 1999 when mass loads were much higher than the previous years.

The percentage of TP made up of TDP increased at the inflow sites each year (Table

4.6). At Sites 23 and 24, the percentage of TDP increased by two to four times between 1998 and 1999. This followed smaller increases at both sites between 1997 and 1998. At Site 22, the percentage of DP increased, but by a smaller amount. All three sites had similar percentages of DP in 1999.

Temporal concentrations of TDP at the return flow sites tended to decrease from the beginning of the monitoring period to the end. Elevated concentrations during this period were often associated with rainfall events. Median concentrations at Sites 14 and 16 increased slightly over the three years.

The FWMCs were similar at both return flow sites in 1997 and 1998, ranging from 0.05 to 0.06 mg L^{-1}. As with TP, there was an increase in FWMCs in 1999 to between 0.07 and 0.08 mg L^{-1}. Mass loads again were dependent on flow volume with the loads at Site 16 being more than twice as much as at Site 14. The percentage of TP made up by TDP at the return flow sites was consistently around 50% (Table 4. 6).

Comparison of TDP between the inflow and outflow sites indicated a significant increase in TDP concentration within the WID sub-basins in all three years (Table 4.6). The FWMCs also indicated a large increase in concentration within the sub-basins. Mass load imported into Sub-basin 14 was similar to that being exported. In Sub-basin 16, more TDP was exported than imported in 1997 and 1998, while in 1999 the reverse was true.

The increase in TDP at Sites 23 and 24 in 1999 was likely due to a change in the content of the source water. The water at these sites originated from A canal. In 1999, the Town of Strathmore was given permission to spill sewage effluent into A canal, potentially adding DP to the system. The remaining site, Site 22, was on B canal.

Other causes of the increase in TDP were similar to those found in the Crowfoot Creek sub-basins. The change in flow velocity between the inflow and outflow sites resulted in less particulate matter in the water, and in a greater percentage of P made up by TDP.

4.3.5 Event Period Flows

Flow-weighted mean concentrations and mass loads derived from the FLUX model cover a period of time that exceeded the period when precipitation was actively falling. This was required in order for the model to achieve the best fit for the data provided. The extended time period resulted in larger mass loads than may have occurred during the rainfall.

4.3.5.1 Crowfoot Creek Sub-basins

Event period FWMCs of TP in the Crowfoot Creek sub-basins were generally greater than during the PSRO period at inflow and outflow sites (Table 4.9). Outflow FWMCs were greater than inflow during all events in Sub-basin 10. In Sub-basin 19, outflow FWMC was greater than inflow in 1997 and during the first event in 1999. The event in 1998 showed a decrease in concentration while the remaining events varied little. The decrease in concentration was likely due to dilution of the high concentrations by WID return flows into the creek upstream of Site 19. The potentially higher inflow concentrations were the result of large flow volumes during this event, which may have flushed out the Hilton wetland, resulting in elevated TP levels. The percentage of PSRO TP mass load exported during event periods ranged from 6 to 72% in Sub-basin 10 and from 7 to 74% in Sub-basin 19 (Table 4.10).

The FWMCs of TDP were also generally higher during event periods compared to PSRO periods in the Crowfoot Creek sub-basins (Table 4.9). Mass loads of TDP comprised over 70% of the TP load during event periods at Site 10. At Site 19, TDP made up at least 47% of the TP event load (Table 4.11).

Surface runoff entering the creek during rainfall events was the only new source of water in the creek. Increases in the concentration of various forms of TP during events indicated additional P was entering Crowfoot Creek with the runoff. This was in agreement with Greenlee et al. (2000) who observed that elevated constituent concentrations were closely related to surface runoff during and immediately after major precipitation events.

The sources of nutrients in runoff can be difficult to determine (Schepers and Francis 1982). Additional P can come from particulate matter carried by surface runoff (Hargrave and Shaykewich 1997; Douglas et al. 1998), P in dissolved forms released from soil particles and soil organic matter (Ahuja et al. 1982) or leached from living and dead plant material

(Schepers and Francis 1982; Schreiber and McDowell 1985). Other sources are resuspended sediment in the creek itself, as well as nutrient released by decaying plant material in instream wetlands (Peverly 1985).

The high percentage of P export in association with rainfall events was supported by studies which indicate that much of the erosion from agricultural land and nutrient load could be transported in a few events (Edwards and Owens 1991; Hargrave and Shaykewich 1997). The high percentage of TDP exported during events agreed with studies in Alberta, but was in opposition to several papers from the United States that found particulate P to be the dominant form of phosphorus in runoff water (Sharpley et al. 1987; Sharpley et al. 1994; Douglas et al. 1998).

The data from the individual rainfall events indicated considerable variability in response to events. The largest percentage mass load loss of TP during an event from Sub-basin 10 occurred during May 1997. In Sub-basin 19, the event resulting in the largest percentage mass load loss of TP was in July 1998, with the second largest event in May 1997 (Table 4.10).

Many factors can influence the amount of surface runoff and the increase in concentration during a rainfall event. In this study, the precipitation data indicated that the level of antecedent moisture may have had an effect on runoff. In Sub-basin 10 in 1997, a 21.6-mm rainfall occurred five days before the sampled rainfall event, increasing the antecedent soil moisture. The spring runoff had also been large that year, possibly resulting in high soil moisture levels immediately following spring runoff. In 1999, the spring runoff was much less than in 1997 and there had been only 8 mm of rainfall in the two weeks before the sampled storm. The lower antecedent moisture condition may have resulted in more rainfall infiltrating rather than running off. In 1997, Sub-basin 19 had a slightly smaller rainfall total of 17 mm during the event, but similar results to Sub-basin 10.

In May, the soil is more vulnerable to erosion due to rainfall. Many crops have not yet been seeded, but the fields may have been cultivated. Seeded crops have not yet produced a full canopy to help protect the soil by reducing raindrop impact. Phosphorus fertilizer applied to fields at this time is available to be carried away with surface runoff since crops have not had an opportunity to utilize it. There is also a greater depth of interaction between the rain water and the soil, resulting in a greater amount of nutrients being available for transport in surface runoff.

During the rainfall event in July 1998, the intensity of the rainfall may have been responsible for the high runoff. There was also considerable precipitation preceding this storm. At Site 10, a total of 36 mm of precipitation had fallen the week before 55 mm of precipitation on July 4, 1998. In Sub-basin 19, rainfall resulted in 38 mm of precipitation preceding the 33 mm rainfall during the sampled event. The high antecedent moisture would result in surface runoff occurring sooner. Additional soil moisture would have been provided by irrigation.

High mass loads during the July 1998 event were also due to the high flow volumes in the watershed. The high flow volumes were the result of large amounts of water being passed through the Crowfoot Creek watershed via the WID infrastructure in response to a large storm in the Calgary region. The runoff from this storm passed through Chestermere Reservoir and into the WID system to eventually spill into Crowfoot Creek.

The 1999 events accounted for between 6 and 27% of the mass load exported from Sub-basin 10, and 24 and 32% exported from Sub-basin 19. These events seemed to indicate the effect of increasing soil moisture. The initial rainfall in May 1999 occurred with little preceding rainfall. Rainfall occurred regularly during the period between events and the percentage of TP exported with each event increased.

Landcover did not appear to have any obvious effect on the TP mass loads. Examination of TDP, however, indicated some potential effects. The two sub-basins had differing responses in TDP as a result of rainfall events. In Sub-basin 10, the percentage of TP as TDP stayed fairly constant during rainfall events (Table 4.11), while in Sub-basin 19 the percentage decreased during most events.

The difference between the sub-basins may be related to the timing of the rainfall and the dominant landcover in each sub-basin. The large percentage of grassland adjacent to the creek in Sub-basin 10 may have resulted in less particulate matter reaching the creek during event runoff, however, dissolved forms of P would still be transported to the creek. In Sub-basin 19, there is more cropland and cultivated area near the creek, and as a result, the percentage of TDP is not as high as more particulate matter would reach the creek. TDP percentages in Sub-basin 19 during the events in May resulted in less than 50% TDP, while the June event had 78% TDP. The two events in July had over 90% TDP. The increasing canopy cover would reduce raindrop impact and the crop growth would reduce overland flow velocities. Higher amounts of particulate matter would be left in the field, while more dissolved forms of P would continue on to the creek.

4.3.5.2 WID Return Flow Sub-basins

Due to the nature of the flows and concentrations at the inflow sites, the FLUX model was unable to delineate the event sampling periods at the inflow sites with sufficient accuracy. Mass loads and FWMCs were not calculated for event periods at the inflow sites.

The WID return flow sub-basins reacted in a similar fashion to the Crowfoot Creek sub-basins. Generally, the FWMCs of TP and TDP increased at Sites 14 and 16 during rainfall events (Table 4.9). The percent of the TP mass load exported during events ranged from 8 to 49% in Sub-basin 14, and 9 to 36% in Sub-basin 16 (Table 4.10). The percentage of event TP mass load exported as TDP ranged from 53 to 78% in Sub-basin 14, and 30 to 73% in Sub-basin 16 (Table 4.11).

As with the Crowfoot Creek sub-basins, individual rainfall events had varying effects on P response. The rainfall event in July 1998 resulted in the largest removal of TP and TDP from the sub-basins. A total of 49 and 36% of the TP, and 49 and 45% of the TDP for the PSRO period was exported from Sites 14 and 16, respectively. The May 1997 event resulted in the second highest removal of TP and TDP from Site 14, however, Site 16 did not have as large a response to this event.

In Sub-basin 14, the percentage of TDP in relation to the time of year did not give clear results. The events in 1999 followed the trend of increasing TDP with increasing crop cover. However, in July 1998, the percentage of TDP was similar to the value for May 1999. The percentage of TDP during the events in May 1997 and 1999 did not increase from the overall PSRO percentage (Table 4.11) Earlier in the year, it would be expected that there might be lower percentages of TDP due to particulate matter being carried into the channel from cropland. However, in Sub-basin 14, there was a large amount of grassland that could act as a buffer strip. In 1999, the higher levels later in June and July could not be explained with the available data.

In Sub-basin 16, the relationship between TDP and rainfall events was a bit more clear cut. The TDP percentages in May of 1997 and 1999 were the lowest of the event periods. In 1999, the remaining events increased throughout the year. The 1998 event, however, had a lower TDP concentration than did the July 1999 event. A possible explanation for this was that the intensity and distribution of the rainfall event, and lack of areas for the particulate matter to settle out, resulted in high particulate matter levels.

4.6 Conclusions

Concentrations of phosphorus within the study sub-basins varied depending on the time of year, amount of runoff and landcover. In the Crowfoot Creek sub-basins during spring runoff, concentrations of P were increased due to the contributions from wetlands along this portion of Crowfoot Creek. In years with extensive surface runoff, P was transported from the surrounding land surfaces. In years with little or no runoff, concentrations reflected the inputs from internal sources such as wetlands. Mass loads were a reflection of the amount of runoff, with high runoff years producing large mass loads and years with little runoff producing small mass loads.

The concentrations of P within the WID sub-basins were lower than those in the Crowfoot Creek sub-basins, likely due to the reduction in wetland areas within the WID sub-basins. The amount of runoff in the WID sub-basins did not seem to affect the concentration of P to the same degree as in the Crowfoot Creek sub-basins, as the levels remained fairly constant over the duration of the study. Mass loads were a reflection of the amount of runoff. With the exception of 1997, mass loads were smaller in the WID sub-basins due to lesser water volumes and lower concentrations of P.

Landcover in both the Crowfoot Creek and WID sub-basins appeared to affect the percentage of P in dissolved form. TDP decreased with increasing cropped land and summerfallow within and adjacent to the creek. Years with greater amounts of runoff also produced a lesser percentage of TDP.

Concentrations of P tended to decline during the PSRO period, likely due to dilution and flushing of P in the creek and return flow channels over the course of the PSRO period by WID water with lower concentrations of P. Concentrations tended to increase slightly at the end of the monitoring period as flows decreased, indicating that a natural level of P existed in both Crowfoot Creek and WID sub-basins. TP concentrations exceeded the AWQG of 0.5 mg L^{-1} during the majority of the monitoring period.

TP concentration increased significantly between the inflow water from the WID infrastructure and monitoring sites within three of the studied sub-basins, indicating the presence of increased amounts of P within these sub-basins. In Sub-basin 19, there was no significant change in TP concentrations.

Within the same three sub-basins, TDP increased significantly as well, while there was a significant decrease in TDP concentrations in Sub-basin 19. The increase in the percentage of TDP was likely due to the decrease in water velocity resulting in deposition of particulate P. In Sub-basin 19, water with a high concentration of TDP entered the sub-basin from the Hilton wetland. As this water traveled within the sub-basin, the higher velocity and inputs of water from other WID spills added more particulate matter to the water, decreasing the percentage of TDP.

Mass loads at the Crowfoot Creek sites tended to be higher at the outflow than the inflow sites due to greater flow and concentration. In 1999, however, low water demand resulted in Sub-basin 10 having a net import of P. The WID sub-basins had a net decrease in mass load within the sub-basins due to withdrawals of water for irrigation use.

Overall, the concentrations of P were lower in the WID sub-basins than the Crowfoot Creek sub-basins, indicating that the input of the return flow water into Crowfoot Creek tended to reduce the concentration of P in the creek.

Rainfall events generally resulted in an increase in concentration of TP and TDP in the sub-basins. Large percentages of the total PSRO mass load of P were also removed during events. The magnitude of the increase was affected by antecedent moisture conditions and the timing of rainfall in relation to landcover development. Events that occurred when high soil moisture conditions were present generally resulted in greater increases in TP concentration. Rainfall early in the season, prior to the development of an extensive leaf canopy, resulted in increases in TP concentration.

The amount of TDP removed appeared to be affected by the time of year, landcover and stage of plant development. Events later in the season had increased amounts of TDP as there was less soil erosion due to more extensive crop growth. There were also greater amounts of TDP in sub-basins with more grassland adjacent to the creek. The grass likely acted as a buffer and filtered the particulate P out before it reached the creek.

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Table 4.1. Spring runoff total phosphorus summary statistics and median comparisons

	TP				
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
			1997		
Site 10[z]	0.225	0.586	1.163	0.228	1.87
Site 11[y]	0.364	0.566a	0.888	0.133	3.27
Site 19[y]	0.358	0.488a	1.106	0.177	5.80
Site 14[z]	0.110	0.418	0.570	0.161	0.92
Site 16[z]	0.071	0.482	0.677	0.184	3.84
			1998		
Site 10[z]	0.170	0.494	1.536	0.503	0.23
Site 11[y]	0.348	0.971a	4.641	1.123	0.17
Site 19[y]	0.164	0.528a	0.964	0.212	0.10
Site 14[z]	0.061	0.355	0.956	0.264	0.03
Site 16[z]	0.055	0.408	1.013	0.250	0.10
			1999		
Site 10[z]	0.221	0.642	1.738	0.608	0.03
Site 11[y]	0.258	1.141a	2.084	0.575	0.03
Site 19[y]	0.154	0.341b	0.621	0.119	0.09
Site 14[z]	0.054	0.337	0.584	0.156	0.03
Site 16[z]	0.057	0.170	0.409	0.105	0.12

[z] - no statistical analysis carried out.
[y] - median values in each sub-basin each year followed by the same letter are not significantly different ($p < 0.05$).

Table 4.2. Spring runoff total dissolved phosphorus summary statistics and median comparisons

	TDP				
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
			1997		
Site 10^z	0.203	0.428	1.116	0.219	1.87
Site 11^y	0.324	0.466a	0.721	0.111	3.27
Site 19^y	0.285	0.460a	1.02	0.165	5.80
Site 14^z	0.083	0.409	0.497	0.134	0.92
Site 16^z	0.031	0.354	0.525	0.153	3.84
			1998		
Site 10^z	0.113	0.390	1.441	0.488	0.23
Site 11^y	0.250	0.838a	4.172	1.04	0.17
Site 19^y	0.135	0.447a	0.877	0.199	0.10
Site 14^z	0.026	0.289	0.896	0.263	0.03
Site 16^z	0.043	0.272	0.765	0.206	0.10
			1999		
Site 10^z	0.148	0.565	1.619	0.564	0.03
Site 11^y	0.171	1.031a	1.797	0.502	0.03
Site 19^y	0.098	0.204b	0.481	0.109	0.09
Site 14^z	0.049	0.271	0.485	0.138	0.03
Site 16^z	0.026	0.092	0.228	0.062	0.12

^{z} - no statistical analysis carried out.
^{y} - median values in each sub-basin each year followed by the same letter are not significantly different ($p < 0.05$).

Table 4.3. Percent of total phosphorus as total dissolved phosphorus - spring runoff

	1997	1998	1999
Site 10	80	87	93
Site 11	81	90	89
Site 19	87	86	81
Site 14	89	91	83
Site 16	76	69	69

Table 4.4. Crowfoot Creek sub-basins post-spring runoff total phosphorus summary statistics and median comparisons[zy]

	TP				
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
			1996		
Site 13	0.005	0.212a	0.751	0.197	0.56
Site 10	0.170	0.299b	0.976	0.201	0.57
Site 11	0.143	0.290a	0.812	0.174	1.52
Site 19	0.005	0.222a	0.656	0.171	2.19
			1997		
Site 13	0.005	0.066a	0.345	0.103	0.67
Site 10	0.063	0.261b	0.991	0.191	1.25
Site 11	0.069	0.163a	0.605	0.170	2.21
Site 19	0.052	0.138a	0.724	0.151	3.92
			1998		
Site 13	0.023	0.056a	0.309	0.070	1.12
Site 10	0.084	0.293b	0.719	0.176	0.59
Site 11	0.069	0.258a	0.605	0.170	1.90
Site 19	0.085	0.173a	0.752	0.179	4.59
			1999		
Site 13	0.023	0.108a	0.268	0.069	0.53
Site 10	0.093	0.353b	1.483	0.295	0.07
Site 11	0.088	0.360a	2.35	0.385	2.20
Site 19	0.072	0.289a	0.621	0.134	4.69

[z] - statistical comparisons were carried out only between the inflow and outflow of each sub-basin.
[y] - median values in each sub-basin each year followed by the same letter are not significantly different ($p < 0.05$).

Table 4.5. Crowfoot Creek sub-basins post-spring runoff total dissolved phosphorus summary statistics and median comparisons[zy]

	TDP				
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
			1996[x]		
Site 13	No data	No data	No data	No data	0.56
Site 10	No data	No data	No data	No data	0.57
Site 11	No data	No data	No data	No data	1.52
Site 19	No data	No data	No data	No data	2.19
			1997		
Site 13	0.005	0.028a	0.303	0.089	0.67
Site 10	0.046	0.197b	0.547	0.132	1.25
Site 11	0.045	0.136a	0.590	0.152	2.21
Site 19	0.024	0.075b	0.322	0.089	3.92
			1998		
Site 13	0.005	0.021a	0.221	0.056	1.12
Site 10	0.066	0.189b	0.648	0.172	0.59
Site 11	0.109	0.231a	0.819	0.219	1.90
Site 19	0.047	0.129b	0.725	0.174	4.59
			1999		
Site 13	0.005	0.076a	0.227	0.062	0.53
Site 10	0.052	0.323b	0.809	0.189	0.07
Site 11	0.067	0.342a	0.674	0.160	2.20
Site 19	0.052	0.246b	0.416	0.113	4.69

[z] - statistical comparisons were carried out only between the inflow and outflow of each sub-basin.

[y] - median values in each sub-basin each year followed by the same letter are not significantly different ($p < 0.05$).

[x] - TDP was not determined in 1996.

Table 4.6. Percent of total phosphorus as total dissolved phosphorus during post-spring runoff at all study sites

	Crowfoot Creek Sub-basins		
	1997	1998	1999
Site 13	50	45	81
Site 10	N/D	88	88
Site 11	82	92	95
Site 19	57	89	89
	WID Return flow sub-basins		
	1997	1998	1999
Site 22	31	57	63
Site 14	56	55	49
Site 23	14	25	56
Site 24	14	17	60
Site 16	51	46	57

Table 4.7. WID return flow sub-basins post-spring runoff total phosphorus summary statistics and median comparisons [zy]

	TP				
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
			1996		
Site 22[x]	No data	No data	No data	No data	No data
Site 14	0.078	0.204	0.645	0.143	0.47
Site 23[x]	No data	No data	No data	No data	No data
Site 24[x]	No data	No data	No data	No data	No data
Site 16	0.031	0.204	0.751	0.158	1.18
			1997		
Site 22	0.012	0.038a	0.102	0.022	5.40
Site 14	0.025	0.056b	0.325	0.077	1.05
Site 23	0.029	0.058ab	0.146	0.031	3.36
Site 24	0.023	0.052a	0.118	0.027	5.25
Site 16	0.029	0.077b	0.241	0.044	3.00
			1998		
Site 22	0.011	0.022a	0.789	0.154	2.12
Site 14	0.035	0.083b	0.283	0.057	0.64
Site 23	0.015	0.040a	0.147	0.033	1.42
Site 24	0.012	0.042a	0.139	0.035	5.64
Site 16	0.021	0.086b	0.318	0.072	3.70
			1999		
Site 22	0.019	0.031a	0.077	0.017	2.57
Site 14	0.031	0.094b	0.345	0.084	1.05
Site 23	0.028	0.065a	0.208	0.037	2.41
Site 24	0.024	0.054a	0.105	0.021	4.50
Site 16	0.017	0.097b	0.654	0.121	2.76

[z] - statistical comparisons were carried out only between the inflow and outflow of each sub-basin.

[y] - median values in each sub-basin each year followed by the same letter are not significantly different ($p < 0.05$).

[x] - Inflow sites in 1996 did not have sufficient samples for statistical analysis.

Table 4.8. WID return flow sub-basins post-spring runoff total dissolved phosphorus summary statistics and median comparisons[zy]

TDP					
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
1996[x]					
Site 22	No data	No data	No data	No data	No data
Site 14	No data	No data	No data	No data	0.47
Site 23	No data	No data	No data	No data	No data
Site 24	No data	No data	No data	No data	No data
Site 16	No data	No data	No data	No data	1.18
1997					
Site 22	0.005	0.005a	0.250	0.052	5.40
Site 14	0.005	0.034b	0.21	0.043	1.05
Site 23	0.005	0.005a	0.074	0.015	3.36
Site 24	0.005	0.005a	0.067	0.013	5.25
Site 16	0.005	0.027b	0.079	0.018	3.00
1998					
Site 22	0.005	0.008a	0.169	0.031	2.12
Site 14	0.023	0.051b	0.145	0.034	0.64
Site 23	0.005	0.011a	0.032	0.007	1.42
Site 24	0.005	0.010a	0.037	0.008	5.64
Site 16	0.01	0.032b	0.207	0.046	3.70
1999					
Site 22	0.005	0.016a	0.073	0.017	2.57
Site 14	0.025	0.056b	0.256	0.053	1.05
Site 23	0.012	0.034a	0.072	0.019	2.41
Site 24	0.005	0.031a	0.066	0.015	4.50
Site 16	0.011	0.042b	0.403	0.082	2.76

[z] - statistical comparisons were carried out only between the inflow and outflow of each sub-basin.

[y] - median values in each sub-basin each year followed by the same letter are not significantly different ($p < 0.05$).

[x] - TDP not determined in 1996.

Table 4.9. Total phosphorus and total dissolved phosphorus event and post-spring runoff (PSRO) flow-weighted mean concentrations (FWMC) - all study sites

	TP (mg L^{-1})							
	1997		1998		1999			
	Event	PSRO	Event	PSRO	Event 1	Event 2	Event 3	PSRO
Site 13	N/D	0.049	0.169	0.073	0.110	0.129	0.190	0.099
Site 10	0.589	0.424	0.659	0.288	0.400	0.652	0.610	0.491
Site 11	0.385	0.240	0.616	0.521	0.350	0.439	0.438	0.378
Site 19	0.452	0.258	0.437	0.316	0.470	0.430	0.410	0.318
Site 14	0.194	0.107	0.200	0.104	0.213	0.154	0.206	0.153
Site 16	0.129	0.117	0.185	0.114	0.130	0.212	0.410	0.134

	TDP (mg L^{-1})							
	1997		1998		1999			
	Event	PSRO	Event	PSRO	Event 1	Event 2	Event 3	PSRO
Site 13	N/D	0.025	0.119	0.033	0.079	0.092	0.164	0.080
Site 10	0.502	0.432	0.641	0.252	0.412	0.462	0.550	0.433
Site 11	0.294	0.193	0.574	0.525	0.340	0.405	0.430	0.362
Site 19	0.214	0.148	0.393	0.281	0.240	0.340	0.396	0.284
Site 14	0.103	0.060	0.109	0.058	0.113	0.108	0.160	0.075
Site 16	0.039	0.060	0.1047	0.053	0.053	0.151	0.303	0.077

N/D - no data

Table 4.10. Percent of post-spring runoff mass load exported during event periods

	TP		
	1997	1998	1999
	% of PSRO load (ML event/ML PSRO)[z]	% of PSRO load (ML event/ML PSRO)	% of PSRO load (ML event/ML PSRO)
Site 13	No data	46 (26/56)	Event 1 - 8 (4/53) Event 2 - 15 (8/53) Event 3 - 28 (15/53)
Site 10	72 (379/530)	40 (69/170)	Event 1 - 6 (2/34) Event 2 - 24 (8/34) Event 3 - 27 (9/34)
Site 11	18 (96/531)	83 (893/1080)	Event 1- <1(3/833) Event 2 - 4 (30/833) Event 3 - 25 (207/833)
Site 19	57 (576/1010)	74 (1070/1452)	Event 1 - 7 (110/1494) Event 2 - 12 (182/1494) Event 3 - 32 (486/1494)
Site 14	31 (35/113)	49 (33/67)	Event 1 - 8 (13/161) Event 2 - 10 (17/161) Event 3 - 8 (14/161)
Site 16	9 (32/352)	36 (152/422)	Event 1 - 18 (68/371) Event 2 - 19 (72/371) Event 3 - 12 (44/371)

[z] - ML - mass load in kg.

Table 4.11. Percent of event TDP as TDP (ML TDP event/ML TDP PSRO)[z]

	1997 % of PSRO load	1998 % of PSRO load	1999 % of PSRO load
Site 13	N/D	70	Event 1 - 71 Event 2 - 72 Event 3 - 85
Site 10	85	97	Event 1 - 86 Event 2 - 70 Event 3 - 89
Site 11	76	93	Event 1- 97 Event 2 - 92 Event 3 - 99
Site 19	47	90	Event 1 - 51 Event 2 - 78 Event 3 - 97
Site 14	53	54	Event 1 - 53 Event 2 - 70 Event 3 - 78
Site 16	30	58	Event 1 - 40 Event 2 - 72 Event 3 - 73

[z] - ML - mass load in kg.

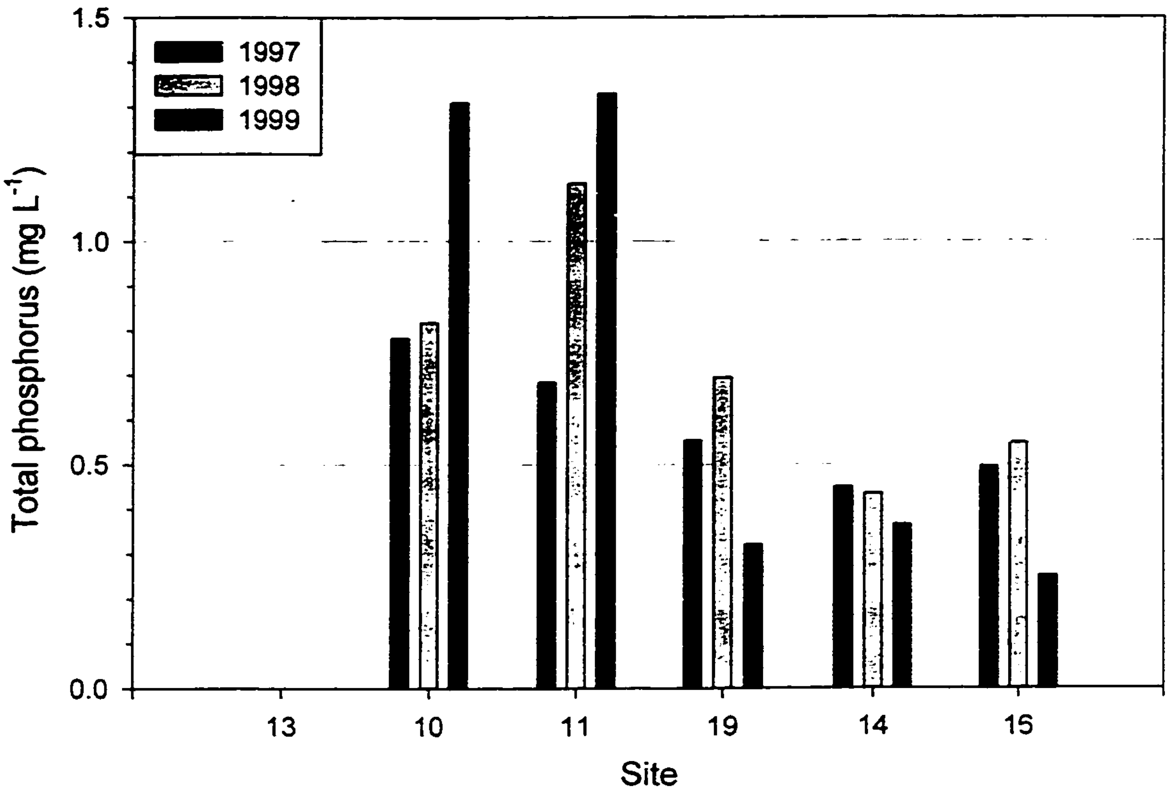

Figure 4.1. Total phosphorus flow-weighted mean concentrations during spring runoff.

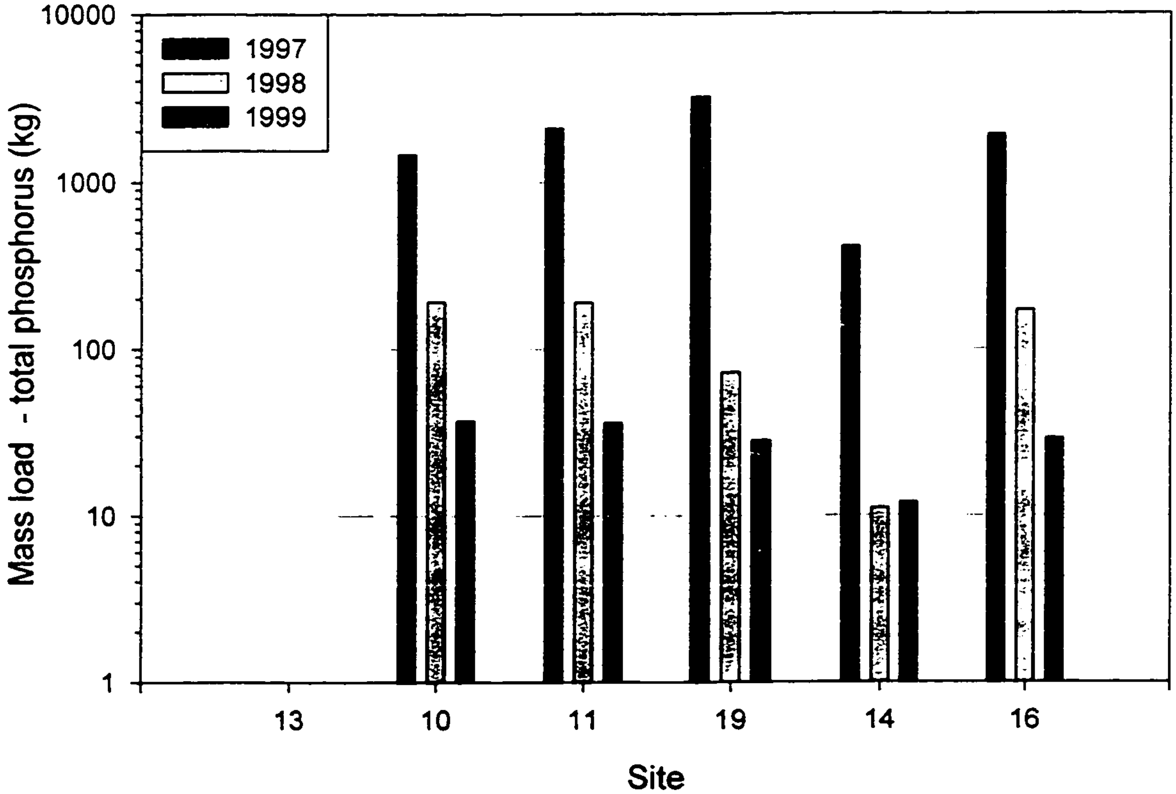

Figure 4.2. Total phosphorus mass load during spring runoff.

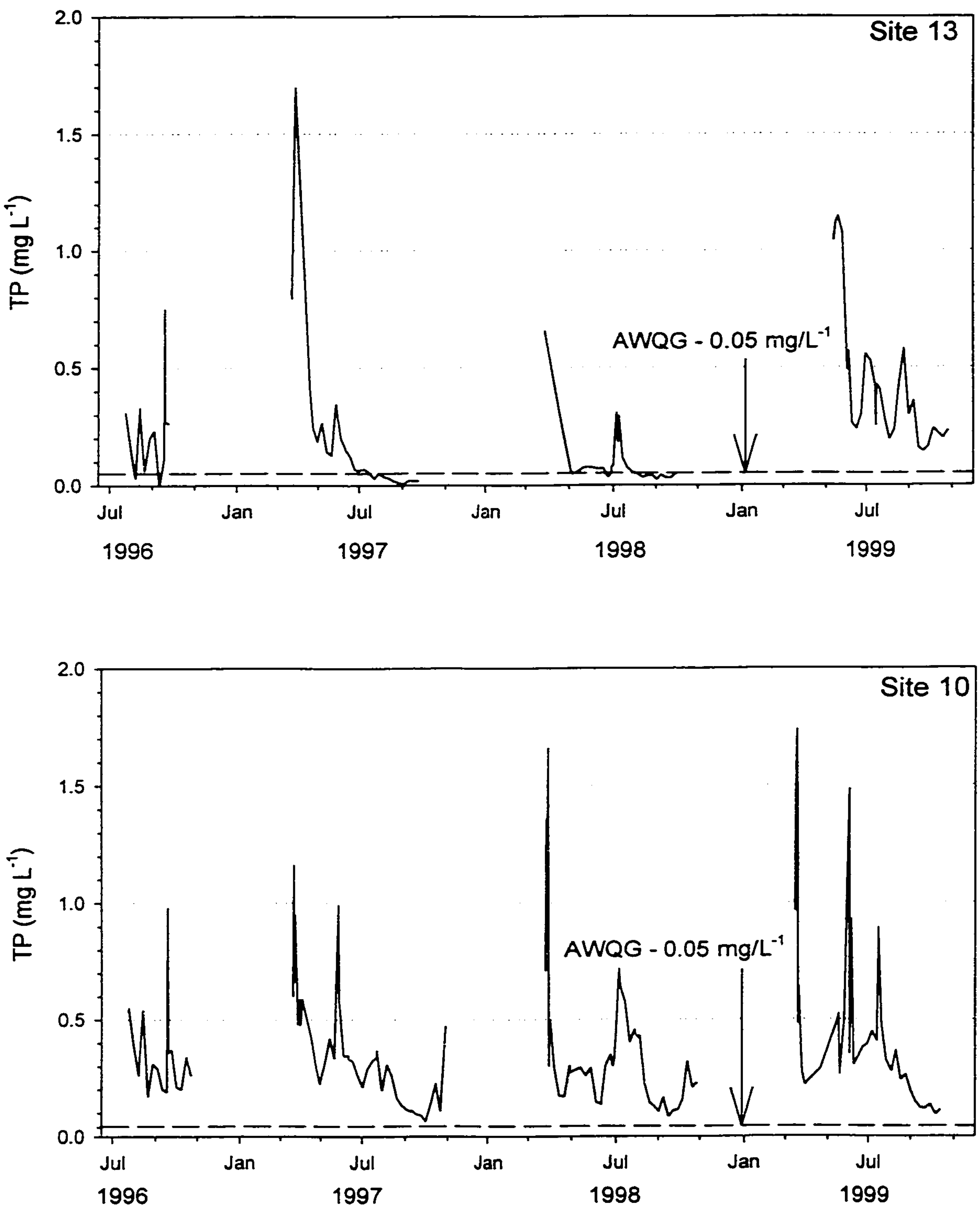

Figure 4.3. Total phosphorus in Sub-basin 10 from 1996 to 1999.

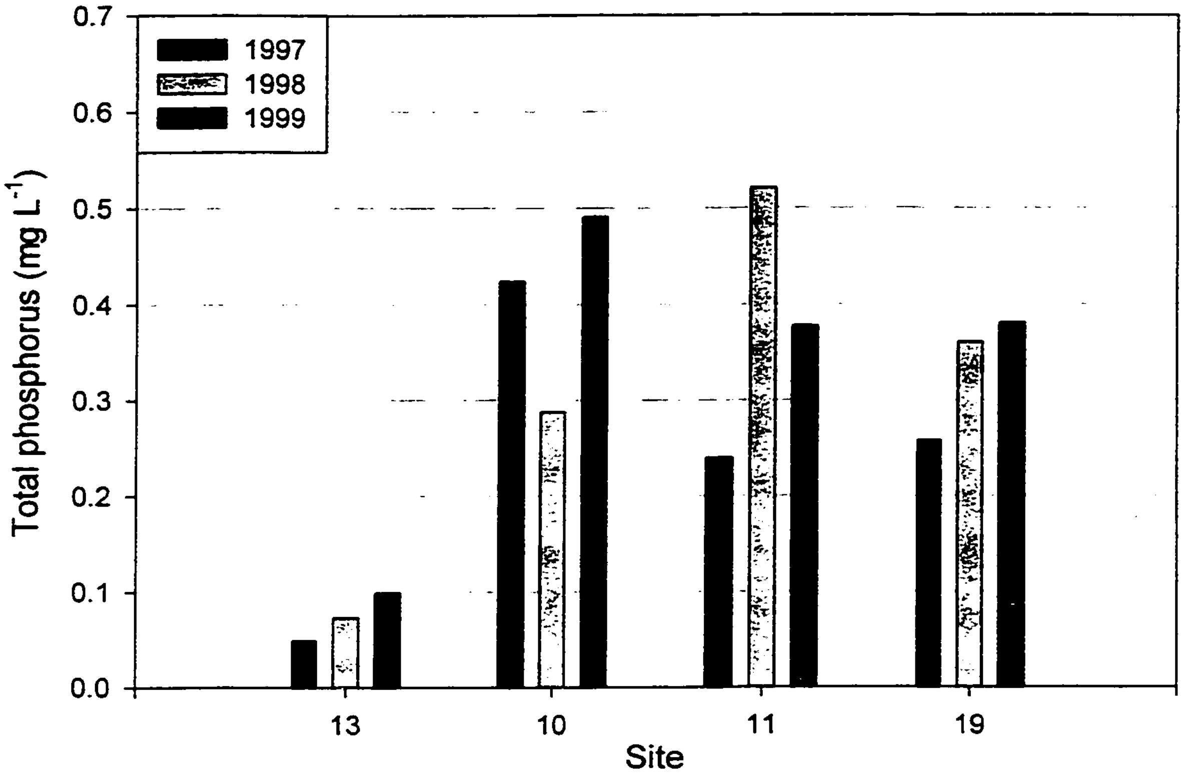

Figure 4.4. Total phosphorus flow-weighted mean concentrations during post-spring runoff.

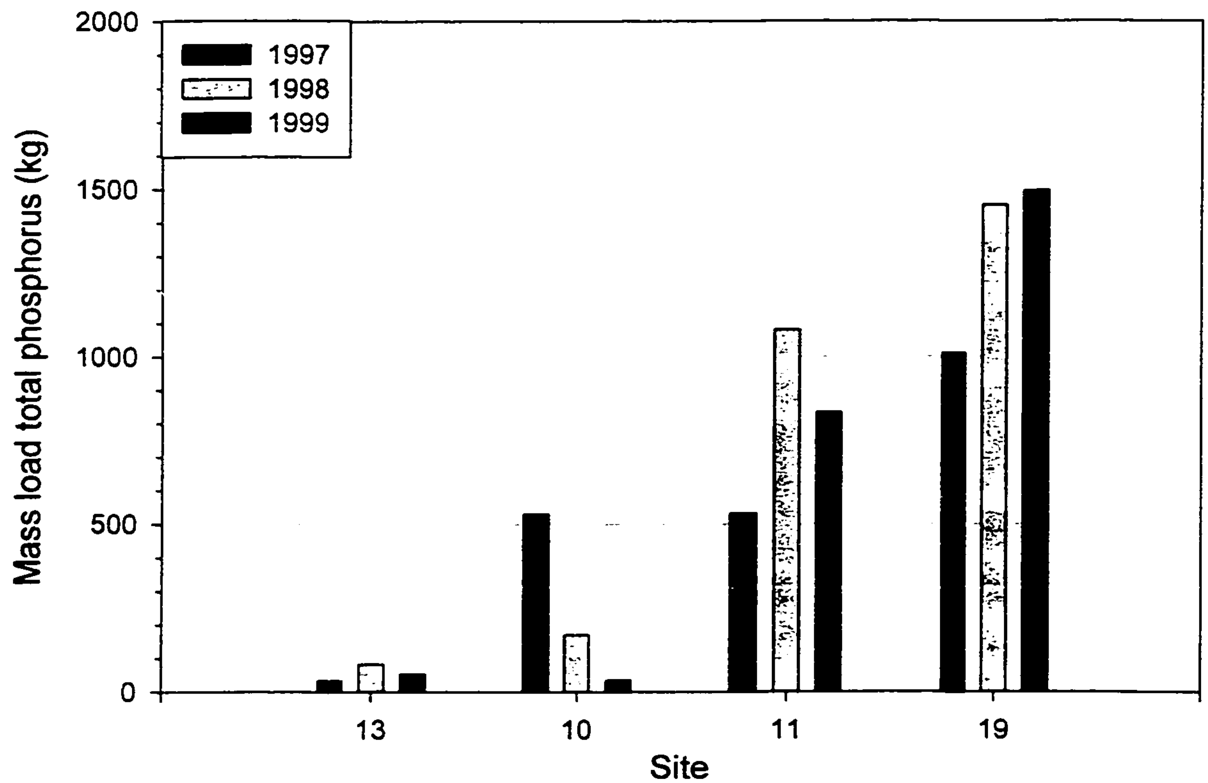

Figure 4.5. Total phosphorus mass load during post-spring runoff.

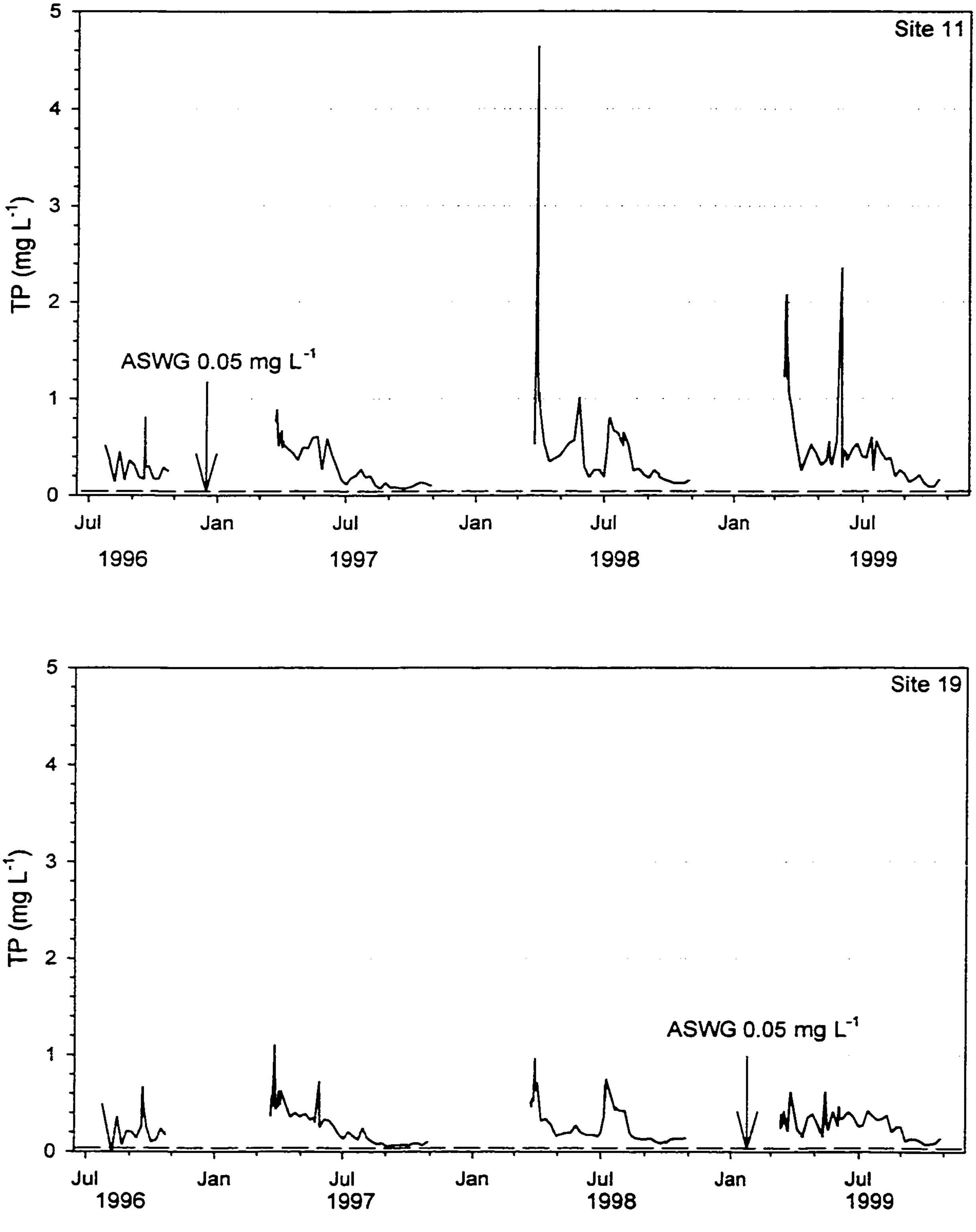

Figure 4.6. Total phosphorus in Sub-basin 19 from 1996 to 1999.

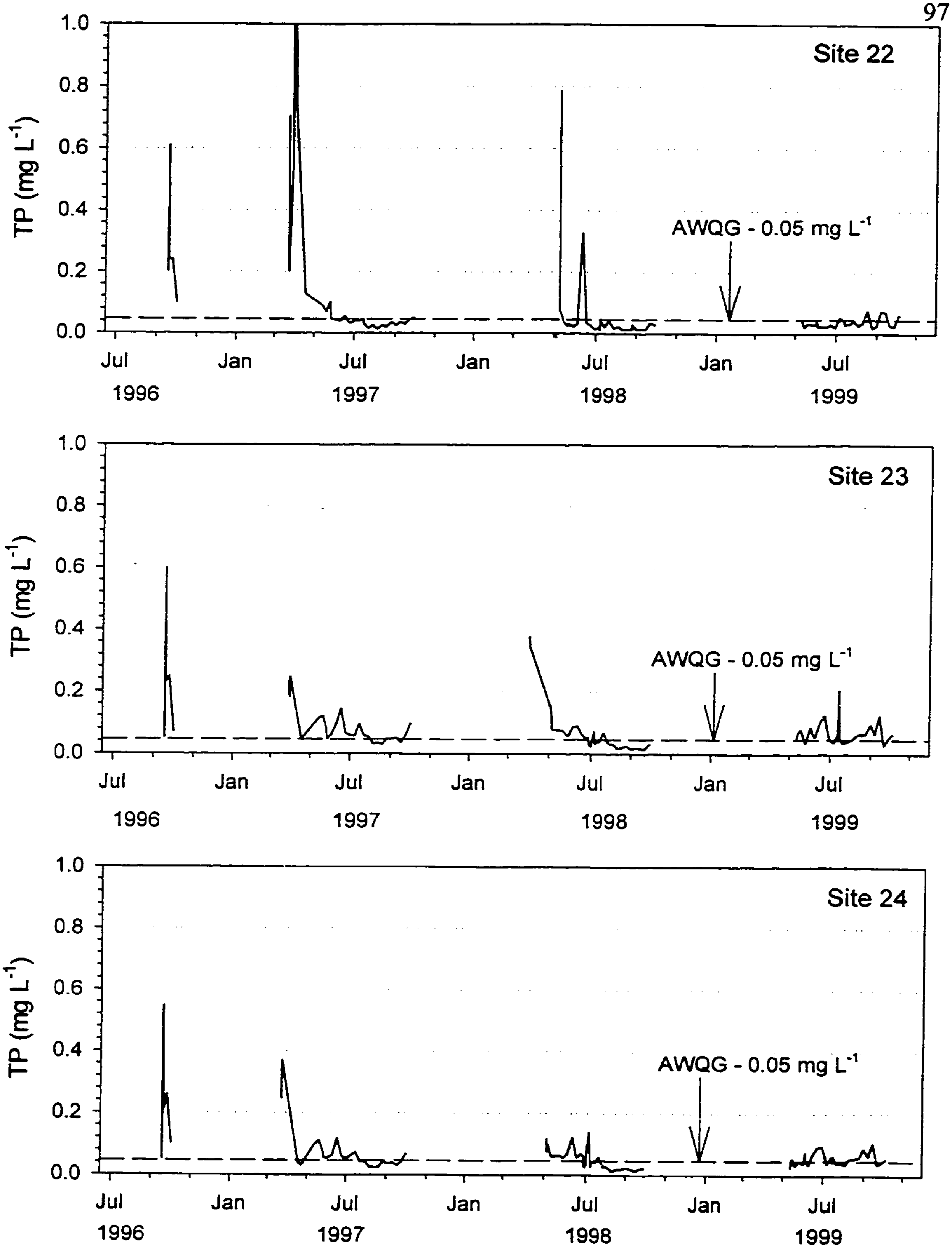

Figure 4.7. Total phosphorus at the inflow sites of the WID return flow sub-basins from 1996 to 1999.

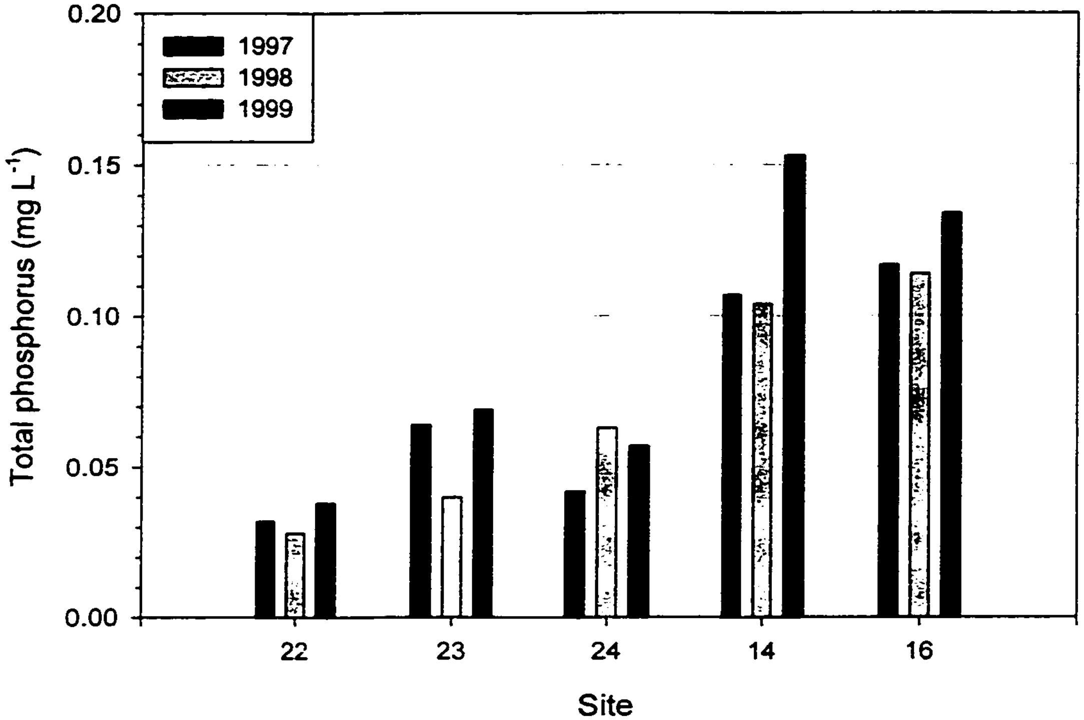

Figure 4.8. Total phosphorus flow-weighted mean concentrations during post-spring runoff at WID return flow sub-basins.

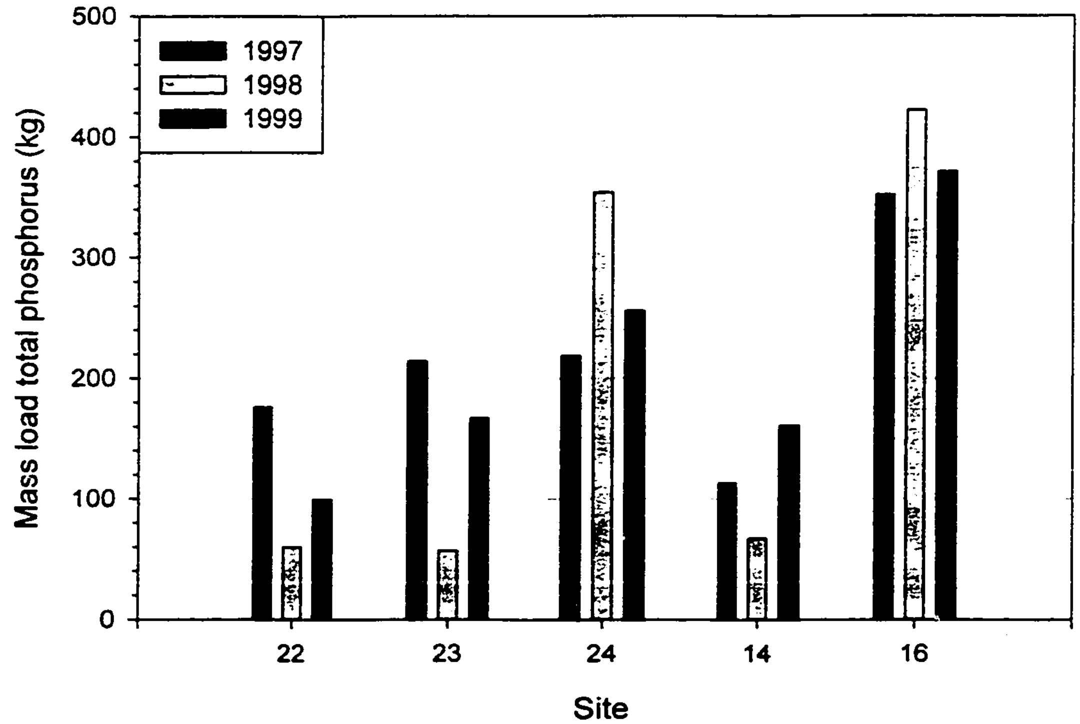

Figure 4.9. Total phosphorus mass load during post-spring runoff at WID return flow sub-basins.

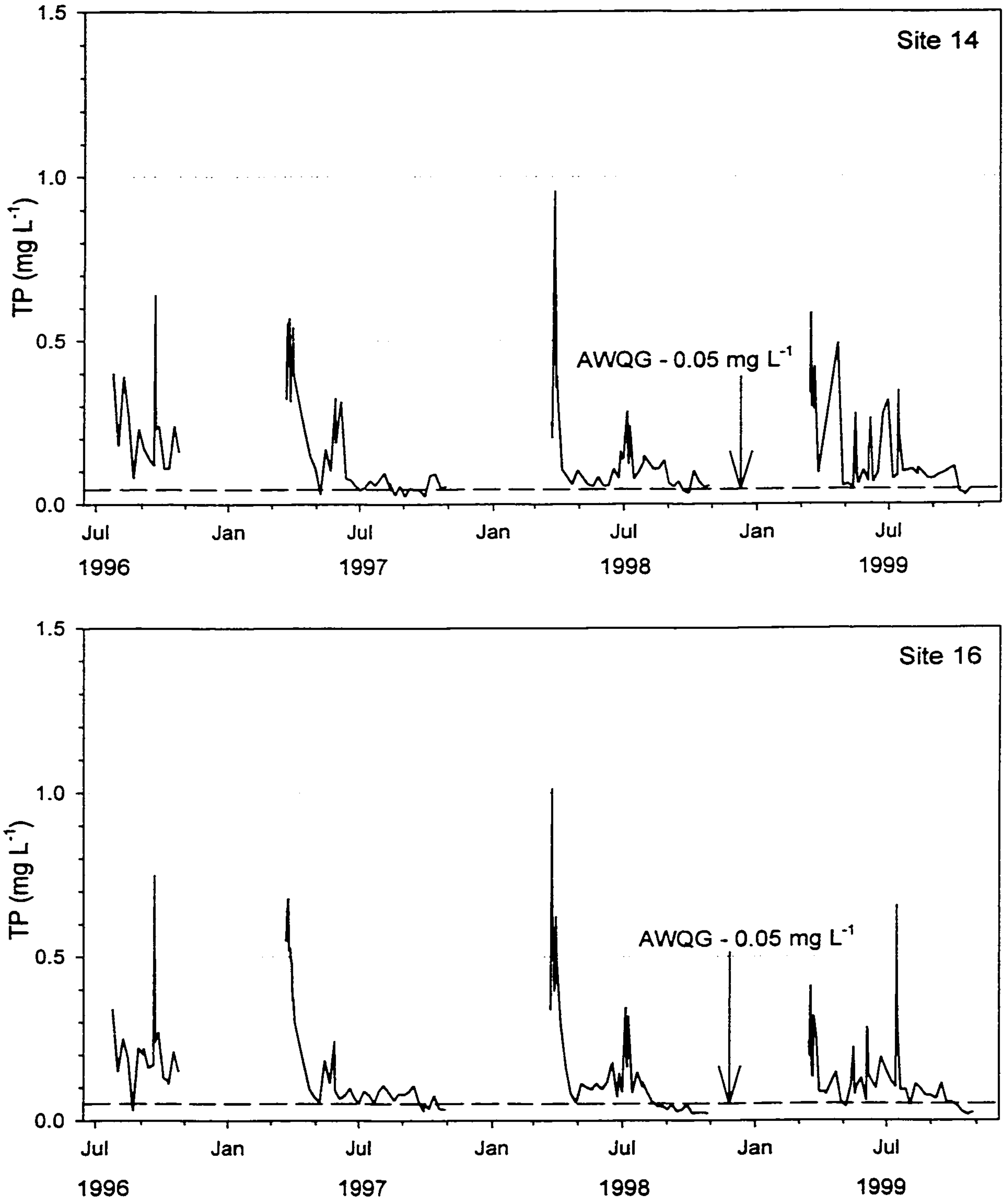

Figure 4.10. Total phosphorus at the outflow sites of the WID return flow sub-basins from 1996 to 1999.

5. IMPACT OF NUTRIENTS ON SURFACE WATER QUALITY IN AN IRRIGATED WATERSHED II: NITROGEN

5.1 Introduction

See Section 4.1 in Chapter 4.

5.1.1 Transportation Mechanisms

The two main pathways of nutrient transport from agricultural land are surface and subsurface flow. Surface flow will transport nutrients attached to sediment particles as well as dissolved soluble nutrients. Subsurface flow will transport mainly soluble nutrients (Heathwaite 1995).

A major fraction of N lost by surface runoff is organic N, which is associated with silt-sized particles and sediments (Muchovej and Rechcigl 1994). Nitrogen in the forms of ammonia and ammonium can be transported by either surface or subsurface runoff. Ammonium salts are highly soluble and more easily transported by leaching. Ammonia is generally transported adsorbed to sediments. The anionic species, nitrate and nitrite, are highly soluble and transported primarily by subsurface flow (Nommik and Vahtras 1982; Cross and Cooke 1996).

5.1.2 Nitrogen

Nitrogen occurs naturally in soil and water in a variety of forms. Inorganic N is present primarily in the form of nitrate (NO_3^-), nitrite (NO_2^-), ammonium (NH_4^+) and molecular nitrogen. Naturally occurring, organic N, is primarily in the form of amines and amides. Nitrogen compounds are present in cellular constituents, as nonliving particulate matter, soluble organic compounds and as inorganic ions in solution (Brezonik 1972). Potential agricultural sources of N are nitrogenous fertilizers, runoff from feedlots and manure storage piles, heavy applications of manure to the land and legumes in crop rotations (Greenlee et al. 2000). The concentrations of the various nitrogen compounds can range up to 100 mg L^{-1} in surface waters (McNeely et al. 1979).

Nitrogen forms are interrelated by the series of reactions known as the nitrogen cycle. The main processes in the cycle are mineralization, nitrification, denitrification, immobilization and volatilization (Brezonik 1972; Cross and Cooke 1996):

1. Mineralization - transforms organic nitrogen to ammonium ions (NH_4^+).
2. Nitrification - conversion of ammonium by soil microbes and bacteria to nitrite and nitrate.
3. Denitrification - conversion of nitrates into nitrogen gas (N_2).
4. Imobilization - conversion of ammonium and nitrate to organic nitrogen in microorganisms. This form can be mineralized back to ammonium.
5. Volatilization - loss of N_2 gas to the atmosphere. This can occur from fertilizers and manure.

5.1.2.1 Ammonia

Ammonia in water is generated by heterotrophic bacteria as a primary end product of the decomposition of organic matter. Decomposition can be directly from proteins or from other nitrogenous organic compounds. Ammonia is a major excretory product of aquatic animals, but the amount produced is small compared to the amount generated by decomposition. The form of ammonia in water is mainly NH_4^+ and undissociated NH_4OH. Ammonia is strongly sorbed to particulate and colloidal particles, especially in lakes with high pH and high concentrations of dissolved organic matter. Small amounts of ammonia are fixed from the atmosphere by cyanobacteria (Wetzel 1983).

In the soil, ammonia is fixed by bacteria associated with legumes such as Nitrosomonas and Nitrobacter into first nitrite and then nitrate (Cross and Cooke 1996). As in water, ammonia is strongly adsorbed by soil particles and moved into surface water by erosion. Commercial fertilizers made of highly soluble ammonia and ammonium salts, when applied in excess of crop requirements, may be transported into the aquatic system by rainfall or irrigation (McNeely et al. 1979; CCREM 1987).

Ammonia is present in large quantities in decomposing organic matter and in smaller quantities in the air, soil, and water. The concentration of ammonia under well oxygenated conditions is low, due to the conversion of ammonia to nitrite and nitrate. The energy needed for plants to assimilate ammonia is low, making it the primary nitrogen source in the aquatic system (CCREM 1987).

Concentrations of ammonia and ammonium compounds in natural water are usually below 0.1 mg L^{-1}, with higher concentrations considered an indication of anthropogenic input and organic pollution. Environmental concentrations of total ammonia (ionized and unionized) in Western Canadian surface waters range from 0.014 to 2.00 mg L^{-1} (McNeely et al. 1979; CCREM 1987).

The Canadian Water Quality Guideline relating to fresh water aquatic life is 1.37 mg L^{-1} at 10 °C with a pH of 8. The unionized portion of the total ammonia is the toxic component. Since both total ammonia and unionized ammonia values vary with temperature and pressure, only total ammonia is given in the CCREM guidelines.

5.1.2.2 Nitrate and Nitrite

Nitrogen present in the soil and water in reduced and organic forms is converted by bacteria into nitrite and nitrate through the process of nitrification (Hem 1989). While ammonia cations are adsorbed onto mineral surfaces, these anionic species are easily transported by water.

Nitrite is produced biochemically by incomplete nitrification of ammonia under aerobic conditions and by denitrification under anaerobic conditions (CCREM 1987). Nitrite is usually present under conditions of low or limited oxygen supply and is, therefore, seldom present in large concentrations in surface waters as it is readily transformed to nitrate (McNeely et al. 1979; CCREM 1987; Hem 1989). This transformation is facilitated by the Nitrosomonas sp. bacteria. Levels seldom exceed 1 mg L^{-1} and are usually in the order of 1 ug L^{-1} (McNeely et al. 1979; CCREM 1987).

The continuation of the nitrification process involves the transformation of nitrite to nitrate. This portion of the process is facilitated by bacteria as well, primarily Nitrobacter sp. As with nitrite, this process takes place under aerobic conditions with denitrification taking place under anaerobic conditions. Surface waters contain at least a trace of nitrates. While levels can exceed 100 mg L^{-1} in severely polluted waters, levels are rarely in excess of 5 mg L^{-1} and are often less than 1 mg L^{-1} (McNeely et al. 1979; CCREM 1987). Nitrate concentrations may fluctuate, with higher concentrations being present in the winter due to groundwater impacts, and in the spring due to contributions from overland flow (CCREM 1987).

Canadian Water Quality Guidelines for nitrite include: drinking water, 1.0 mg L^{-1}; livestock watering, 10 mg L^{-1}; freshwater aquatic life, 0.02 mg L^{-1} (CCREM 1987).

Excessive concentrations of nitrate in drinking water can cause methemoglobinemia, or "blue baby" syndrome in children. The drinking water guideline for nitrate is 10 mg L^{-1} and the livestock watering guideline is 100 mg L^{-1} of nitrate-N.

5.1.2.3 Organic Nitrogen

Organic nitrogen is the organically bound form of nitrogen. It includes all organic compounds such as proteins, polypeptides, amino acids and urea. Dissolved organic nitrogen (DON) in fresh water makes up more than 50% of the total soluble nitrogen. Over one-half of the DON is in the form of amino N compounds. Simple amino acids are substrates that are used by bacteria. They have high rates of decomposition and their instantaneous concentration in fresh water is low. The amount of DON in fresh water is five to ten times the amount of particulate organic nitrogen (PON). PON is contained in plant and animal material (McNeely et al. 1979; Wetzel 1983).

Total nitrogen is considered to be the sum of NO_3^-+ NO_2^-+TKN. The Alberta Surface Water Quality Guideline (AENV 1999) for this parameter is 1.0 mg L^{-1}.

5.1.3 Contribution of Nitrogen From Agricultural Land

Losses of fertilizer nitrogen are influenced by the rate, time and method of application, form of fertilizer, amount and time of rainfall after application and vegetative cover (Sharpley and Halvorson 1994). Smith et al. (1993) reported that cover crops reduced the loss of particulate nutrients. The application of animal manures can result in an increase in surface soil accumulations of P (Smith et al.1993; Howard et al 1999). The conversion of ammonia to nitrate results in a form of nitrogen that can be easily leached into the soil by infiltrating water and away from the areas of surface runoff. As a result, there are low levels of nitrate in surface runoff (Sharpley et al. 1987). In this study, 22 to 93% of the total nitrogen transported was particulate nitrogen. Both the concentration of total nitrogen and the total nitrogen load were higher on wheat and rotationally cropped areas compared to native grassland areas. Studies carried out on erodible soils in Mississippi found that 99% of the nitrogen was lost with the eroded material in situations of high erosion. Under a no-till system, the particulate losses of nitrogen ranged from 11 to 50% of the total nitrogen lost (Marston 1989).

The objective of this chapter was to determine if nitrogen concentrations and mass loads varied spatially or temporally within selected sub-basins along Crowfoot Creek and two irrigation return flow channels, and to determine if water quality within these sub-basins was affected by various types of agricultural landcover. Compliance with water quality guidelines was assessed. The impact of irrigation flows on water quality in Crowfoot Creek was also examined.

5.2 Methods

See Chapter 2 for the methods of data collection and analysis.

5.3 Results and Discussion

5.3.1 Spring Runoff - Crowfoot Creek Sub-basins

In Sub-basin 10, the median concentrations and FWMC of total nitrogen (TN) responded in a similar manner over the three years of spring runoff. Median concentrations increased slightly between 1997 and 1998, then decreased slightly in 1999 (Table 5.1). FWMC increased slightly each year (Fig. 5.1). Mass loads exported from the sub-basin over the three years were very different due to the difference in flows each spring. The largest loads were observed in 1997, then decreased over the remaining two years (Fig. 5.2).

The median concentrations at the inflow and outflow sites in Sub-basin 19 increased from 1997 to 1998, then decreased in 1999 (Table 5.1). A comparison of the median values between the inflow and outflow sites showed a decrease in TN each year. The decrease was not significant in 1997 or 1998, but was in 1999. Median values at all of the Crowfoot Creek monitoring sites in all years exceeded the AWQG of 1.0 mg L^{-1} .

The FWMCs at the inflow and outflow sites also increased from 1997 to 1998, and decreased in 1999 (Fig. 5.1). Comparison of the FWMCs between the inflow and outflow sites indicated little change in FWMC within Sub-basin 19 in 1997 and 1998. However, the FWMC decreased approximately 2 mg L^{-1} within the sub-basin in 1999.

Mass loads at the inflow and outflow sites of Sub-basin 19 decreased each year (Fig. 5.2). Loads in 1997 were much higher than in 1998 or 1999 due to the greater flow volume in that year. The mass load of TN exported in 1997 was over 20,000 kg, but was less than 200 kg in 1999. However, comparison of mass loads at the inflow and outflow sites indicated little change within the sub-basin. There was a net export of nitrogen from the sub-basin during spring runoff in 1997 and 1999, and a net import in spring runoff 1998.

Sources of nitrogen in the sub-basins included sediments, breakdown products from plant decay and applied fertilizer. Most nitrogen in the soil exists in organic form (Marston 1989). This was true of the Crowfoot Creek sub-basins during spring runoff as the nitrogen fraction was dominated by organic nitrogen, with ammonium nitrogen making up the next largest percentage. Water color at the monitoring sites during the spring was often amber in color, indicative of high organic matter content. The dominance of organic and ammonium nitrogen is in agreement with studies carried out on land that has received manure or has had cattle present (Kirchmann 1994; Cooke and Prepas 1997; Anderson et al. 1998).

The dominance of organic nitrogen may be due to environmental conditions during spring runoff. The activity of the bacteria required to break down organic nitrogen to ammonia, nitrite and nitrate is temperature dependent and likely declined during the period of springmelt (Wetzel 1983; Anderson et al. 1998). The wetlands along the creek, especially in Sub-basin 10, would be sources of organic N due to the decay of plant material. The loss of nitrogen through plant uptake would be reduced at this time of year. Nitrite and nitrate were both present in low concentrations at the Crowfoot Creek sites. Nitrite concentrations were below 0.01 mg L^{-1} with a peak of 0.03 mg L^{-1} in 1997, while nitrate concentrations did not exceed 0.20 mg L^{-1} during spring runoff.

In Sub-basin 10, the similarity in the concentrations of TN over the three years indicated there was a base level of N independent of the amount of runoff. In 1999, the low runoff compared to the previous two years would likely not have provided as much N as in the previous two years. The FWMC, however, was highest in 1999. The wetlands were a likely source of this pool of N, with the surrounding landcover contributing to it.

In Sub-basin 19, the concentration of TN decreased between inflow and outflow in all three years. This does not necessarily mean there was not a contribution of N from within this sub-basin during spring runoff. The water entering the sub-basin at Site 11 has high levels of TN, probably due to inputs from the Hilton Project wetland upstream. The decrease in TN concentration may be due to the dilution of N by snowmelt water entering the creek within the sub-basin. This water likely had a lower concentration of nitrogen.

Landcover affects were difficult to determine. Sites 10 and 11, with wetlands upstream of them, generally had higher FWMCs compared to Site 19. In 1999, during the small runoff, Sites 10 and 11 still had high FWMCs, while the FWMC at Site 19 was lower, indicating that inputs from the surrounding landcover had been removed. There was no clear trend between cultivated and grassed areas.

5.3.2 Spring Runoff - WID Return Flow Sub-basins

Median concentrations of TN in Sub-basin 14 were the highest in 1998 and lowest in 1999. In Sub-basin 16, median concentrations of TN were similar in 1997 to 1998, but lower in 1999 (Table 5.1). The FWMCs at both sites increased from 1997 to 1998, however, the lowest values were observed in 1999 (Fig. 5.1). Median concentrations exceeded the AWQG of 1.0 mg L^{-1} in all years at both sites monitored, with the exception of Site 16 in 1999, which was just below the guideline (0.927 mg L^{-1}). Mass loads at both sites were highest in 1997 and decreased sharply in 1998 (Fig. 5.2). At Site 14 in 1999, loads were similar to those in 1998. At Site 16, mass loads decreased substantially again from 1998 to 1999.

Sources of nitrogen in the WID sub-basins were similar to those in the Crowfoot Creek sub-basins, with the main difference being the decreased extent of wetland areas within the WID sub-basins. The extent of surface runoff appeared to have had an effect on the FWMCs in these sub-basins. The large runoff volume in 1997 may have reduced the TN concentration, while in 1998 the lower surface runoff did not cause as much dilution with higher FWMCs the result. In 1999, runoff was limited largely to the creek and sheltered areas. With the reduced transport of N from the surrounding land, the FWMC was the lowest of the three years as there was less nitrogen transported into the drainage channel.

The effect of landcover in each sub-basin on the concentration of TN was difficult to ascertain. The higher overall concentration of TN in Sub-basin 16 may have been due to a greater sediment load being able to reach the creek. Suspended sediment loads were much higher at Site 16 than 14 (data not presented). The additional N associated with Sub-basin 16 may have been due to the greater amount of annual crop land in the sub-basin. Sub-basin 14 had more grassland near the drainage channel acting as a buffer strip, resulting in the deposition of a portion of the sediment load from runoff. In 1999, when there was little surface runoff, Sub-basin 14 had higher median concentrations and FWMCs of TN than Sub-basin 16.

In comparison with the Crowfoot Creek sub-basins, Sub-basin 14 tended to have lower median and FWMCs of TN. Median concentrations in Sub-basin 16 were slightly higher than the Crowfoot Creek sub-basins in 1997, slightly lower in 1998 and much lower in 1999. The FWMC's were similar to much lower than the Crowfoot Creek sub-basins.

5.3.3 Post-spring Runoff - Crowfoot Creek Sub-basins

Median concentrations of TN increased significantly between inflow and outflow of Sub-basin 10 in each year of the study (Table 5.2). There was also more variability in the concentrations at Site 10 than at Site 13. The Alberta Water Quality Guideline (AWQG) for TN is 1.0 mg L^{-1}. Median values at Site 13 were less than the guideline in all years monitored. Median values at the outflow, Site 10, exceeded the guideline in 1998 and 1999. TN concentrations were high early in the PSRO period (Fig. 5.3), then decreased except during event periods. TN concentrations increased late in the monitoring period in 1997 and 1998 as flows decreased. In 1999, flows ceased late in the season, therefore, no samples were collected, and concentration change was not detected.

In Sub-basin 19, median concentrations of TN between the inflow and outflow decreased significantly in 1997 and 1998, while in 1999 the decrease was not significant (Table 5.2). Median values at Site 11 exceeded AWQG during 1998 and 1999. Outflow concentrations exceeded AWQG in 1999 only. Temporal trends of TN in Sub-basin 19 were similar to those in Sub-basin 10 (Fig. 5.4).

The FWMCs in Sub-basin 10 also indicated an increase in TN levels within the sub-basin (Fig. 5.5). The FWMC of TN entering Sub-basin 10 at Site 13 did not exceed 0.7 mg L^{-1}, a concentration similar to the FWMC of TN at other inflow sites in the watershed. FWMCs did not exceed 0.5 mg L^{-1} at any of these sites. The FWMCs of TN exiting Sub-basin 10 were two to three times higher than inflow levels, with values ranging from approximately 1 mg L^{-1} in 1998 to over 1.5 mg L^{-1} in 1997 and 1999. The FWMCs of TN were high in the period following spring runoff and decreased during the monitoring season. The lowest values occurred during periods without a major rainfall and tended to increase in the last month of the monitoring period. This increase was related to the cessation of WID return flows into Crowfoot Creek. The lack of dilution resulted in TN concentrations increasing.

The FWMCs in Sub-basin 19 did not vary to a great degree between inflow and outflow sites (Fig. 5.5). In 1997 and 1999, there was a small variation in FWMCs, while in 1998 there was an obvious decrease between inflow and outflow. The temporal trends of the FWMCs of TN were similar to those in Sub-basin 10.

The magnitude of mass loads that entered and exited the Crowfoot Creek sub-basins were controlled by the volume of water moving through the sub-basin. The high flows at Site 10 compared to Site 13 in 1997 resulted in higher mass loads and a net export of N. In 1998, there was also a net export of TN from Sub-basin 10, while in 1999 there was a net import (Fig. 5.6). In Sub-basin 19, there was a net export of TN in each of the years.

The increase in TN concentrations within Sub-basin 10 indicated a source or sources of nitrogen within the sub-basin. The FWMCs of TN during periods with no major rainfall were higher than the FWMCs of TN entering the sub-basin, indicating a base level of nitrogen within the creek. As during the spring, sources of nitrogen may have been decaying plant material, release from sediment or soluble forms of N transported in the water. The high concentrations early in the PSRO period were likely due to a flushing out of the material contained in wetlands along the creek, as well as in the creek channel itself. Flows were low early in the PSRO period as the WID had not yet begun to divert water. As flows increased, the concentration of TN decreased as the better quality irrigation water diluted the water in the creek.

In Sub-basin 19, the decrease in median concentrations and FWMCs may have been due to dilution of the inflow water which had high concentrations of TN. The WID return flow water was of better quality, lower TN concentrations, than the water in Crowfoot Creek. The inflow water cames out of the Hilton Wetland and may have had elevated levels of TN. Nitrogen may also have been lost from the water by natural processes such as volatilization and dentrification (Wetzel 1983). A decrease in the concentration does not necessarily indicate a lack of inputs of N within this sub-basin.

5.3.4 Post-spring Runoff - WID Sub-basins

Median concentrations of TN were less than the AWQG at the three inflow sites (Sites 22, 23 and 24) (Table 5.3). TN concentrations among the inflow sites were consistent, with median concentrations not significantly different among Sites 22, 23 and 24 during 1997 and 1998. In 1999, the median concentration of TN at Site 23 was significantly greater than Site 22 or 24. The concentrations of TN at the inflow sites tended to decrease over the PSRO period, with the exception of some of the event periods (Fig. 5.7). Concentrations of TN at the return flow sites (Sites 14 and 16) also tended to decrease over the PSRO period, with the exception of event periods (Fig. 5.8). Median concentrations were below 1.0 mg L^{-1}.

Median concentrations tended to increase between the inflow and outflow sites (Table 5.3). This held true for Sub-basin 14, with significant increases in all years. In Sub-basin 16, the median concentration between Sites 23 and 16 decreased in 1997 and increased in 1998 and 1999. Between Sites 24 and 16, the median concentrations were similar in 1997 and increased in 1998 and 1999. None of the changes in median concentration in Sub-basin 16 were significant. Graphs of the FWMCs at the inflow sites illustrate the similarity in values (Fig. 5.9). The concentrations were all less than 0.45 mg L^{-1}, with a variation of approximately 0.15 mg L^{-1} over all years. The FWMCs at the return sites were greater than those at the inflow sites in all years.

As in the Crowfoot Creek sub-basins, there was an obvious increase in TN concentrations within both the WID sub-basins. Similar to the Crowfoot Creek sub-basins, sources of N in the WID sub-basins included sediment release, plant decay and soluble forms of N transported by water. Mass loads decreased between inflow and outflow in all years in Sub-basin 14, indicating a net import of N (Fig. 5.10). A similar situation occurred in Sub-basin 16. The sum of the loads entering the sub-basin at Sites 23 and 24 was greater than at Site 16, indicating a net decrease in N within the sub-basin. The nitrogen mass lost was likely applied to the land by irrigation during the growing season, as well as lost through other processes such as volatilization.

5.3.5 Event Flow Periods

Flow-weighted mean concentrations and mass loads derived from the FLUX model for event periods covered a time frame that exceeded the period when precipitation was actively falling. This was required in order for the model to achieve the best fit of the data provided. The extended time period resulted in larger mass loads than may have occurred during sampled rainfall events.

5.3.5.1 Crowfoot Creek Sub-basins

Concentrations of TN at the Crowfoot Creek sub-basin monitoring sites were generally highest during major rainfall events. FWMCs of TN at the inflow and outflow sites were also higher during event periods than during time periods without significant rainfall. Event FWMCs were also higher than the overall PSRO FWMCs (Table 5.4).

The greatest increases in FWMC of TN occurred during events earlier in the PSRO monitoring period. At Site 10, increases were associated with events in May 1997 and 1999 (Event 1) with increases of 0.720 and 0.580 mg L^{-1}, respectively, over the PSRO FWMC. At Site 19, events in May (Event 1) and June (Event 2) 1999 resulted in increases of TN of

0.391 and 0.810 mg L^{-1}, respectively.

Mass loads also responded to rainfall events. In Sub-basin 10, mass loads exported during individual event periods comprised from 9 to 74% of the total load of nitrogen exported from the sub-basin during the PSRO period (Table 5.5). Three of the five events resulted in mass load losses of between 22 and 26%, indicating a fairly consistent movement of nitrogen during events. In 1999, the three sampled events accounted for 53% of nitrogen exported during PSRO.

Mass loads exported from Sub-basin 19 also indicated a rainfall event effect. Individual events accounted for the export of from 8 to 61% of the total mass load of TN during the PSRO period. In a fashion similar to Sub-basin 10, three of five events in Sub-basin 19 resulted in losses of between 8 and 32% of the total mass load. The three events in 1999 resulted in 54% of the total mass load exported in that year, similar to Sub-basin 10.

Surface runoff entering the creek during rainfall events was the only new source of water in the creek. Increases in the concentration of TN during events indicated that nitrogen in various forms was entering the creek. The composition of TN was dominated by organic nitrogen. This in agreement with Jacobs and Gilliam (1985) who found that the greatest losses of N were in organic form, especially during the summer months.

The sources of nitrogen in runoff can be difficult to determine (Schepers and Francis 1982). Additional N can come from particulate matter carried by surface runoff (Hargrave and Shaykewich 1997; Douglas et al. 1998), N in dissolved forms released from soil particles and soil organic matter (Ahuja et al. 1982), or leached from living and dead plant material (Schepers and Francis 1982; Schreiber and McDowell 1985). Other sources are re-suspended sediment in the creek itself as well as release by decaying plant material contained in instream wetlands (Peverly 1985).

The export of nitrogen in association with rainfall events was supported by studies which report that much of the erosion from agricultural land and nutrient load can be transported in a few events (Edwards and Owens 1991; Hargrave and Shaykewich 1997). The data from the individual rainfall events indicated considerable variability in response to events. The events in May 1997 and July 1998 resulted in the largest export of mass load from Sub-basins 10 and 19, respectively. The effect of antecedent moisture conditions on the amount of surface runoff appeared to have had the same effect on TN transport as on P transport (Section 4.3.5).

The sampled events that occurred in May 1997 and May 1999 resulted in different percentages of mass load removed from the watershed. In Sub-basin 10 in May 1997, a 21 mm rainfall occurred five days before the sampled rainfall event, increasing the antecedent soil moisture. Additionally, the spring runoff had also been quite large and may have resulted in high soil moisture levels. In comparison, spring runoff in 1999 was much less than in 1997, and there had been only 8 mm of rainfall in the two weeks prior to the sampled event. The lower antecedent moisture content may have resulted in a greater amount of moisture infiltrating than running off. A similar situation was present in Sub-basin 19 with similar results.

As discussed in the chapter on phosphorus, the soil was more vulnerable to erosion due to rainfall in May. Many crops have either not been seeded or just recently seeded and the crop canopy is not sufficient to protect the soil surface from raindrop impact. Nitrogen fertilizers placed with the crop at that time were available to be carried away in surface runoff in particulate or dissolved forms. The exposed surface of the soil allowed a greater depth of interaction between the rain water and the soil, resulting in a greater amount of nutrient being made available for transport in surface runoff (Ahuja et al. 1982; Ahuja and Lehman 1983).

In 1998, the high storm intensity of July 4 may have been responsible for the high runoff. Between 55 and 33 mm of rain fell on that day at Sites 10 and 19, respectively. The antecedent soil moisture conditions preceding this storm were high with 36 mm of precipitation falling the week before the sampled event at Site 10 and 38 mm at Site 19. High antecedent soil moisture levels may have resulted in surface runoff being initiated sooner. Soil moisture levels would also have been augmented by irrigation at that time of the year.

High mass loads during the July 1998 event were also generated by the high flow volumes in the watershed. The high flow volumes were the result of large amounts of water being passed through the Crowfoot Creek watershed via the WID infrastructure in response to a large storm in the Calgary region. The runoff from this storm passed from the City of Calgary storm system into Chestermere Lake. From there, the water passed into the WID system to eventually spill into Crowfoot Creek.

In 1999, the three sampled events also appeared to be controlled to some degree by the antecedent soil moisture content. The initial rainfall in May occurred with low antecedent moisture conditions. Rainfall occurred regularly during the period between events and the percentage of TN exported with the later events was higher than the preceding events. This was especially noticeable at Site 19 where the percentage increased from 15 to 32%.

The impact of landcover on TN concentration was difficult to ascertain. There did not appear to be a significant increase in the concentrations of nitrate, nitrite or ammonia during event periods. The conversion of ammonia to nitrate and its subsequent ability to infiltrate resulted in the removal of this form of nitrogen from the soil surface where it was most vulnerable to transport (Sharpley et al. 1987). Organic N remained the dominant form of nitrogen, and could be transported from any of the landcovers in the Crowfoot Creek sub-basins.

5.3.5.2 WID Sub-basins

Due to the nature of the flows and concentrations at the inflow sites, the FLUX model was unable to delineate the event sampling periods at the WID inflow sites with sufficient accuracy. Mass loads and FWMCs were not calculated for event periods at these sites. The FWMCs at the return flow sites in both WID sub-basins increased over PSRO levels during all events (Table 5.4). In Sub-basin 14, the greatest increases occurred in relation to events early in the PSRO period. Sub-basin 16 did not have as clear a pattern during event periods. The sampled events in May 1997 and July 1998 resulted in small increases in FWMC and in 1999, the event FWMC increased with each event.

The percentage of the total PSRO mass load of TN removed by individual events ranged from 7 to 41% in Sub-basin 14 and 8 to 24% in Sub-basin 16 (Table 5.5). The percentages were more consistent over the sampled events, with the majority of the events resulting in less than 25% of the total nitrogen load being removed in any one storm. The lack of wetlands to contribute nitrogen during rainfall events may be a reason for the consistency of the TN removals during the events. As with the Crowfoot Creek sub-basins, the dominant form of nitrogen was organic.

5.3 Conclusions

Concentrations of TN in the study sub-basins during the spring runoff period indicated a base level of nitrogen, independent of runoff additions. Both high and low runoff water volumes resulted in increases in TN concentrations. Concentrations were greatest during spring runoff, with median concentrations and FWMCs exceeding the AWQG of 1.0 mg L^{-1} at all monitored sites. FWMCs at the monitoring sites increased during spring runoff, however, median concentrations of TN varied little. Organic and ammonia forms of nitrogen were dominant during spring runoff.

The Crowfoot Creek sub-basins generally had higher TN concentrations than the WID sub-basins, possibly due to the presence of numerous wetland areas along the Crowfoot sub-basins which would have increased concentrations of N from the decay of plant material and inputs from the surrounding landcover. The TN FWMCs in Sub-basin 19 tended to decrease during spring runoff as the high concentrations entering the sub-basin were diluted by runoff water with lower TN concentrations. The impact of landcover on TN during spring runoff was difficult to ascertain, although Sub-basin 10, dominated by grassland adjacent to the creek, tended to have greater concentrations of TN than Sub-basin 19 which was dominated by cropland.

Within the four study sub-basins during the PSRO period, TN concentrations decreased from early May until baseflow conditions were achieved during October, when TN concentrations tended to increase. The initial decrease in concentration was associated with the input of irrigation return flow water from the WID infrastructure. The WID inflow water had lower concentrations of TN compared to the water in Crowfoot Creek. During the PSRO period, irrigation water likely flushed accumulated nitrogen out of wetlands and the watercourse, diluting the nitrogen and decreasing its concentration. In October, the WID generally ceased delivery of water and concentrations rose. Concentrations of TN in WID inflow water are usually below the AWQG of 1.0 mg L^{-1}. In contrast, TN concentrations in Crowfoot Creek and the WID return flow channels exceeded the AWQG early in the PSRO period, but decreased and tended to remain below the guideline for much of the remainder of the period, except during rainfall events. Organic nitrogen remained the dominant form of nitrogen during the PSRO period, with ammonia, nitrite and nitrate present in low concentrations.

Median concentrations and FWMCs increased within both WID sub-basins. In the Crowfoot Creek sub-basins, TN concentrations generally increased in Sub-basin 10 and decreased in Sub-basin 19 during the PSRO period. The increases in concentration indicated that sources of nitrogen, such as wetland areas within the sub-basin, runoff from the surrounding land and natural sources, were present within the sub-basins. The decrease in concentration in Sub-basin 19 during PSRO was likely due to dilution of the high levels of TN entering the sub-basin from the wetland upstream. While inputs of nitrogen likely occurred within the sub-basin, their concentration appeared to be lower than that in the creek. This would account for the decrease in concentration. The primary input of water within this sub-basin was from WID return flows which tended to have a lower concentration of TN than water in Crowfoot Creek.

Both median concentrations and FWMCs of TN were greater in the Crowfoot Creek sub-basins than in the WID return flow sub-basins. Perhaps the higher concentrations in the Crowfoot Creek sub-basins were due to the presence of wetlands along Crowfoot Creek, which have sources of N from plant decay.

Mass loads were indicative of the volume of water passing through each sub-basin. Moving downstream, the volume of water would be expected to be greater. This was tempered slightly by the fact that irrigation water, not natural flows, was the source of water in Crowfoot Creek during the PSRO period. Withdrawals and varying demands on water within the sub-basins resulted in varying flow volumes entering and leaving the sub-basins. In the Crowfoot Creek sub-basins, mass loads generally increased with downstream position, indicating that there was a net export of nitrogen from these sub-basins. The only exception to this was in Sub-basin 10 in 1999 when there was a net import of N in Sub-basin 10, an indication that more water was removed between inflow and outflow than was returned to the creek within this sub-basin. The N that is lost is likely applied to the fields with irrigation water.

Both WID return flow sub-basins had a net import of N during the PSRO period. These sub-basins had water entering from only a few sources and had much more water removed for irrigation and other requirements. As a result, even with an increase in concentration between inflow and outflow, the mass load was lower due to the withdrawals of water.

The FWMCs were greater during rainfall events than during the overall PSRO period. This was consistent in most of the sampled events in all of the sub-basins. The greatest increases occurred during the earliest events in the PSRO period and during periods of high antecedent soil moisture conditions.

Event FWMCs in the Crowfoot sub-basins generally exceeded the AWQG for TN of 1.0 mg L^{-1}, while FWMCs in the WID sub-basins were generally less than, but close to, this guideline. Individual sample concentrations tended to exceed guideline values. Large percentages of the mass load for the overall PSRO period were exported during these events and this, in association with the increased concentration, indicated that the majority of the impact on water quality from nitrogen during the PSRO period occurred during surface runoff due to rainfall.

The impact of various landcovers on the amount of N lost during runoff within the PSRO period was difficult to determine. Neither cropland or grassland cover seemed to result in increased amounts of TN in the water.

While there was a natural level of nitrogen within the sub-basins, high amounts of runoff in the spring and flushing by irrigation return flow water during the PSRO period reduced the concentrations of TN. Low spring runoff water volumes and event flows during the PSRO period resulted in elevated concentrations of TN. The large spring runoff volumes and input of WID water during the year, while resulting in lower TN concentrations, also resulted in a higher mass load of nitrogen being passed out of the watershed and into the Bow River. The elevated TN concentrations during rainfall events indicated there was a contribution of nitrogen from the surrounding agricultural land.

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Table 5.1. Spring runoff total nitrogen summary statistics and median comparisons for all study sites

	TN				
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
			1997		
Site 10[z]	1.244	2.577	5.294	0.994	1.87
Site 11[y]	1.151	2.560a	3.900	0.684	3.27
Site 19[y]	1.114	2.380a	13.88	2.802	5.80
Site 14z	0.667	1.975	3.088	0.707	0.92
Site 16[z]	0.689	2.657	4.130	1.016	3.84
			1998		
Site 10[z]	1.256	2.797	5.780	1.560	0.23
Site 11[y]	1.534	3.906a	7.903	1.648	0.17
Site 19[y]	1.092	3.283a	5.934	1.413	0.10
Site 14[z]	0.706	2.112	5.626	1.352	0.03
Site 16[z]	0.794	2.623	5.989	1.317	0.10
			1999		
Site 10[z]	1.311	2.303	5.399	1.542	0.03
Site 11[y]	1.838	3.597a	5.417	1.090	0.03
Site 19[y]	1.261	1.736b	4.263	0.840	0.09
Site 14[z]	0.625	1.666	2.611	0.562	0.03
Site 16[z]	0.463	0.927	2.793	0.636	0.12

[z] - no statistical anlysis carried out.
[y] - median values in each sub-basin each year followed by the same letter are not significantly different (<0.05).

Table 5.2. Post-spring runoff total nitrogen summary statistics and median comparisons for the Crowfoot Creek sub-basin[zy]

	TN				
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
		1996			
Site 13	0.175	0.292a	0.575	0.126	0.56
Site 10	0.420	0.662b	1.060	0.195	0.57
Site 11	0.371	0.698a	0.899	0.115	1.52
Site 19	0.202	0.513b	1.522	0.297	2.19
		1997			
Site 13	0.192	0.474a	1.929	0.477	0.67
Site 10	0.397	0.855b	3.977	0.737	1.25
Site 11	0.532	0.990a	1.610	0.236	2.21
Site 19	0.318	0.701b	1.729	0.344	3.92
		1998			
Site 13	0.174	0.465a	1.253	0.232	1.12
Site 10	0.395	1.039b	1.625	0.347	0.59
Site 11	0.637	1.050a	1.717	0.289	1.90
Site 19	0.471	0.735b	1.349	0.233	4.59
		1999			
Site 13	0.211	0.749a	1.473	0.376	0.53
Site 10	0.371	1.237b	2.676	0.657	0.07
Site 11	0.577	1.254a	2.154	0.384	2.20
Site 19	0.474	1.212a	3.788	0.686	4.69

[z] - statistical comparisons were carried out only between the inflow and outflow of each sub-basin.

[y] - median values in each sub-basin each year followed by the same letter are not significantly different (<0.05).

Table 5.3. WID return flow sub-basins post-spring runoff total nitrogen summary statistics and median comparisons [zy]

TN					
	Min mg L^{-1}	Median mg L^{-1}	Max mg L^{-1}	SD mg L^{-1}	Flow Volume (x 10^6 m^3)
1996[x]					
Site 22	No data	No data	No data	No data	No data
Site 14	0.099	0.398	0.934	0.196	0.47
Site 23	No data	No data	No data	No data	No data
Site 24	No data	No data	No data	No data	No data
Site 16	0.219	0.478	0.914	0.181	1.18
1997					
Site 22	0.171	0.404a	1.179	0.224	5.40
Site 14	0.214	0.454b	1.578	0.311	1.05
Site 23	0.221	0.474a	1.803	0.356	3.36
Site 24	0.219	0.406a	0.692	0.154	5.25
Site 16	0.223	0.405a	1.347	0.236	3.00
1998					
Site 22	0.139	0.300a	0.715	0.158	2.12
Site 14	0.159	0.509b	1.402	0.273	0.64
Site 23	0.143	0.356a	0.913	0.190	1.42
Site 24	0.105	0.364a	1.037	0.221	5.64
Site 16	0.147	0.470a	0.802	0.196	3.70
1999					
Site 22	0.173	0.319a	0.967	0.160	2.57
Site 14	0.232	0.578b	1.307	0.330	1.05
Site 23	0.202	0.418a	1.059	0.225	2.41
Site 24	0.200	0.383a	1.344	0.245	4.50
Site 16	0.174	0.525a	2.055	0.455	2.76

[z] - statistical comparisons were carried out only between the inflow and outflow of each sub-basin.

[y] - median values in each sub-basin each year followed by the same letter are not significantly different ($p < 0.05$).

[x] - Inflow sites in 1996 did not have sufficient samples for statistical analysis.

Table 5.4. Total nitrogen event and post-spring runoff (PSRO) flow-weighted mean concentrations (FWMC) at all study sites

	TN (mg L^{-1})							
	1997		1998		1999			
	Event	PSRO	Event	PSRO	Event 1	Event 2	Event 3	PSRO
Site 13	N/D	0.344	0.752	0.521	0.852	0.736	1.03	0.647
Site 10	2.37	1.650	1.54	1.03	2.19	1.89	1.67	1.61
Site 11	1.19	0.965	1.42	1.47	1.7	1.53	1.36	1.28
Site 19	1.28	0.889	1.06	0.927	2.20	2.34	1.73	1.39
Site 14	0.964	0.624	0.826	0.520	1.16	0.929	0.697	0.672
Site 16	0.764	0.720	0.658	0.598	0.820	1.23	1.80	0.693

N/D - no data.

Table 5.5. Percent of TN post-spring runoff mass load exported during event periods at all study sites

	TN		
	1997	1998	1999
	% of PSRO load (ML event/ML PSRO)[z]	% of PSRO load (ML event/ML PSRO)	% of PSRO load (ML event/ML PSRO)
Site 13	No data	20 (114/585)	Event 1 - 9 (32/343) Event 2 - 13 (44/343) Event 3 - 23 (80/343)
Site 10	74 (1527/2064)	26 (162/608)	Event 1 - 9 (10/113) Event 2 - 22 (24/113) Event 3 - 22 (25/113)
Site 11	12 (299/2494)	74 (2065/2809)	Event 1- <1 (15/2824) Event 2 - 4 (103/2824) Event 3 - 23 (643/2824)
Site 19	47 (1635/3484)	61 (2601/4258)	Event 1 - 8 (512/6546) Event 2 - 15 (992/6546) Event 3 - 32 (2056/6546)
Site 14	26 (172/656)	41 (136/333)	Event 1 - 10 (70/705) Event 2 - 14 (102/705) Event 3 - 7 (47/705)
Site 16	8 (190/2159)	24 (541/2214)	Event 1 - 22 (424/1914) Event 2 - 22 (416/1914) Event 3 - 10 (189/1914)

[z] - ML - mass load in kg.

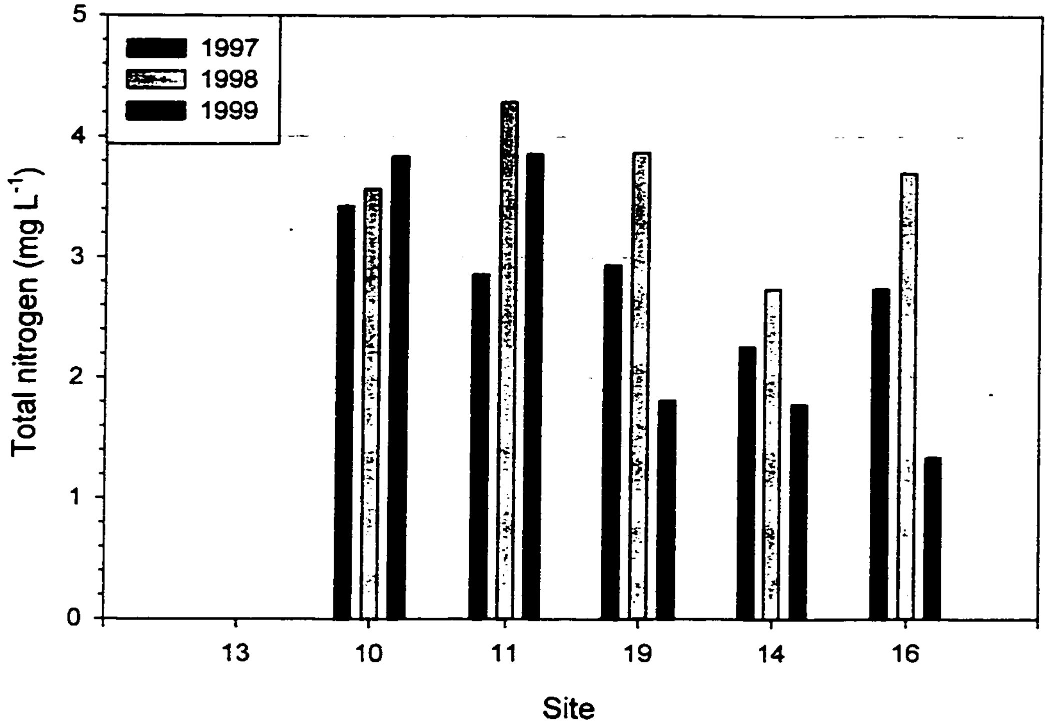


Figure 5.1. Total nitrogen flow-weighted mean concentrations during spring runoff.

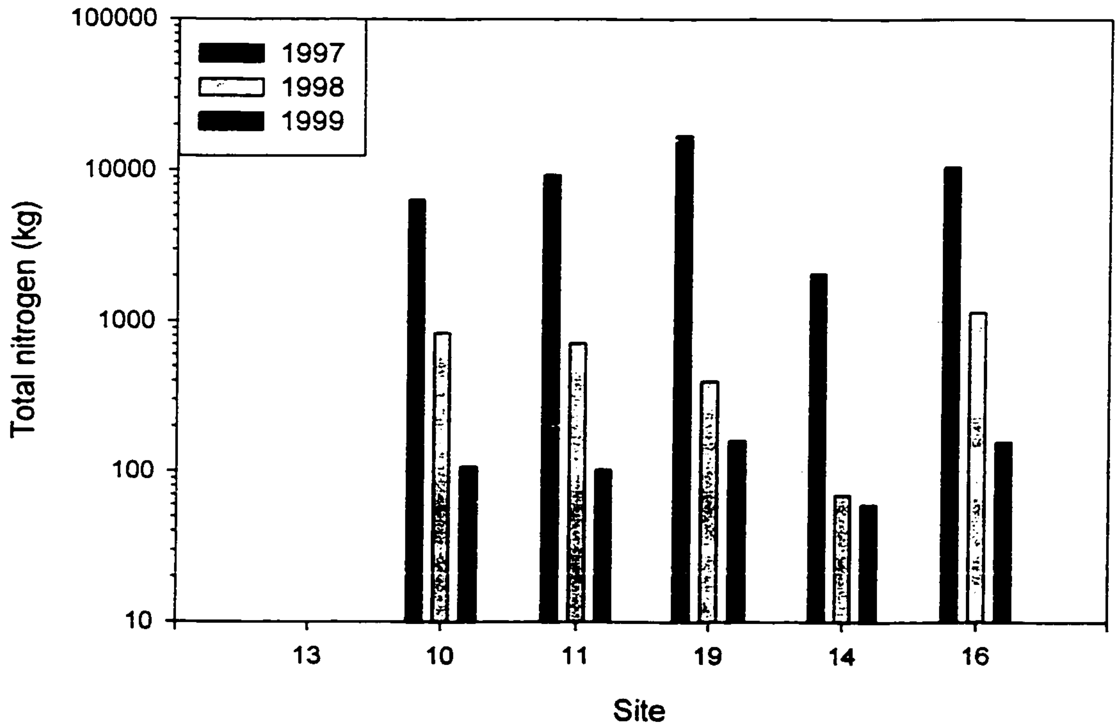


Figure 5.2. Total nitrogen mass load during spring runoff.

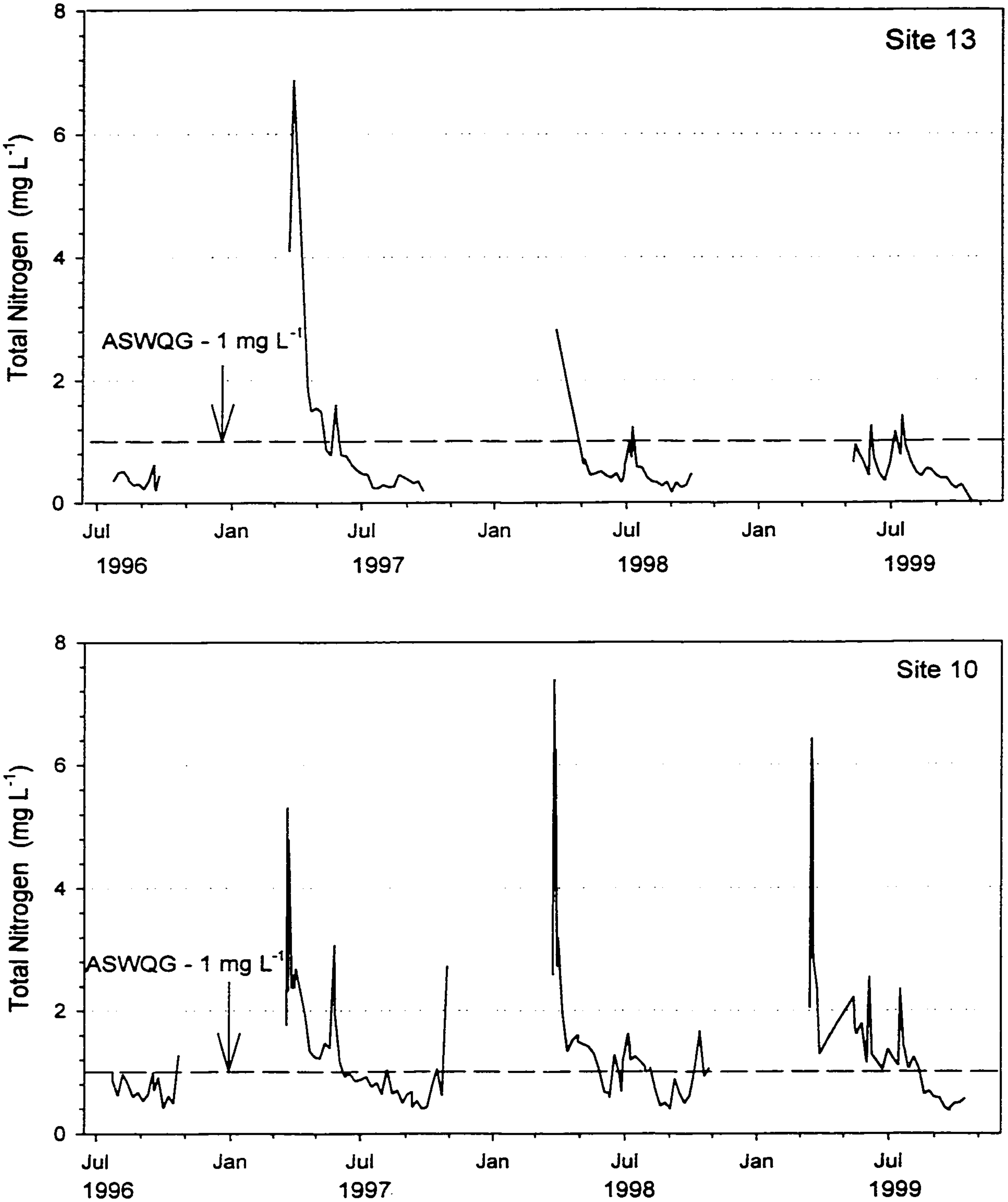
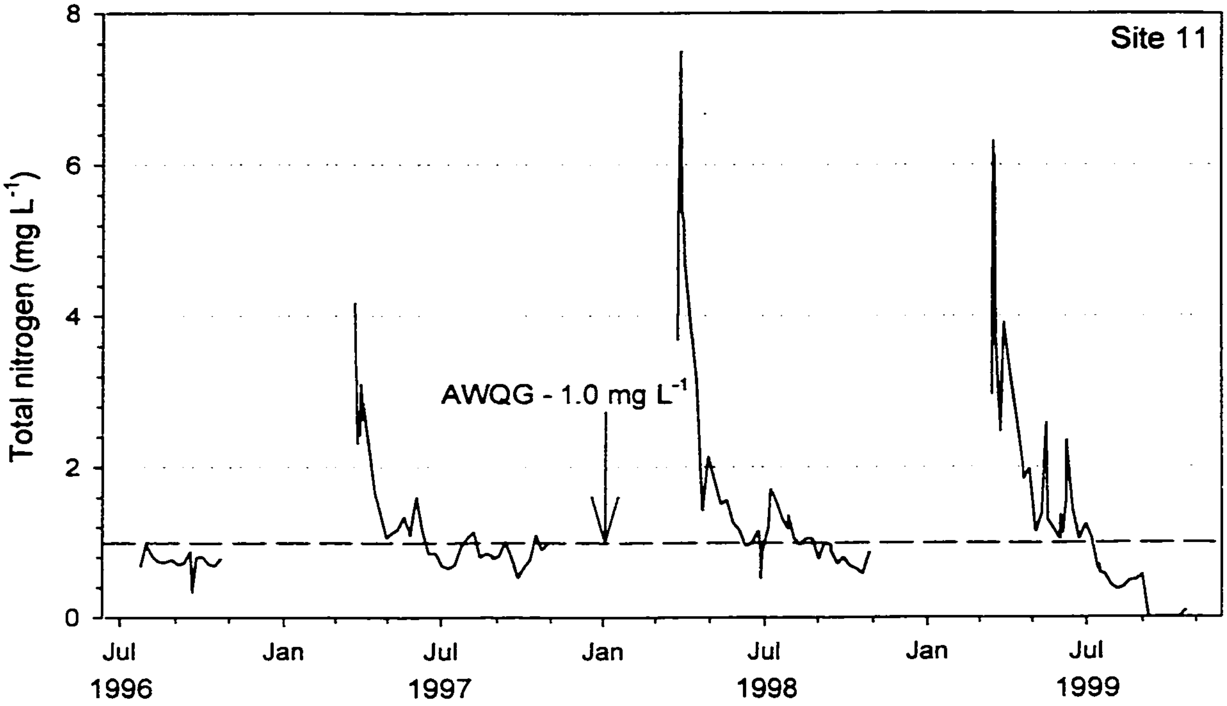


Figure 5.3. Total nitrogen at the inflow and outflow sites of Sub-basin 10 from 1996 to 1999.



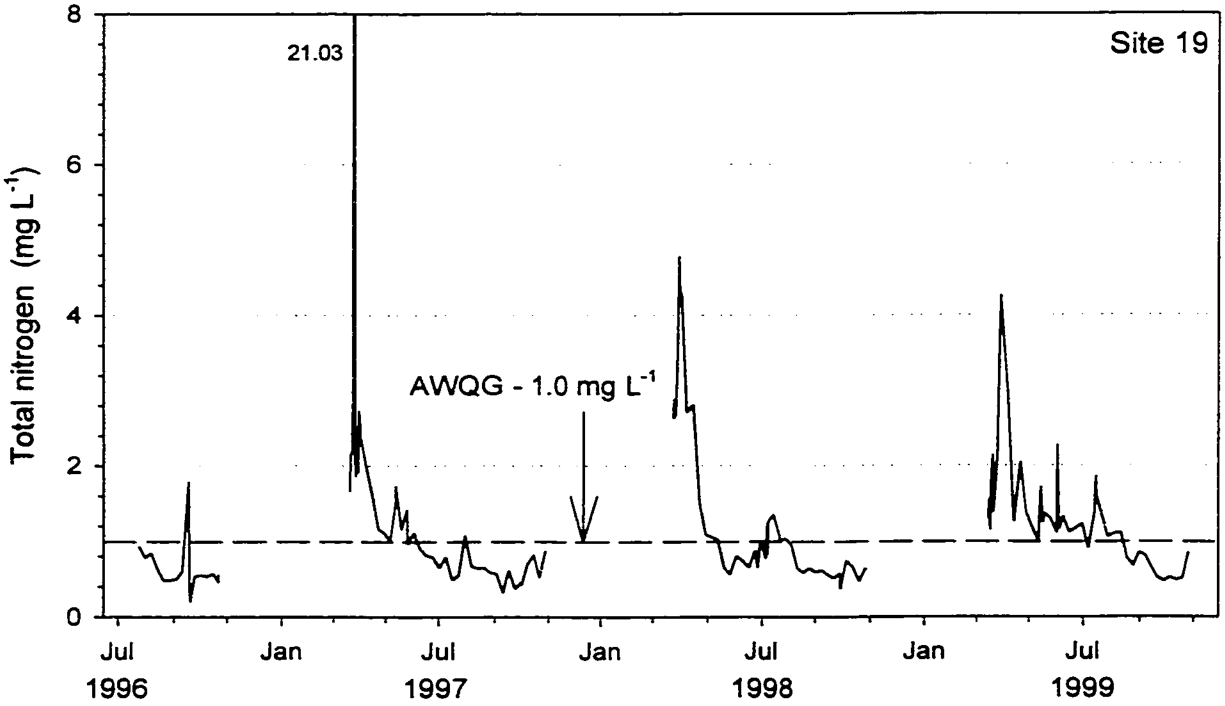


Figure 5.4. Total nitrogen at the inflow and outflow sites of Sub-basin 19 from 1996 to 1999.

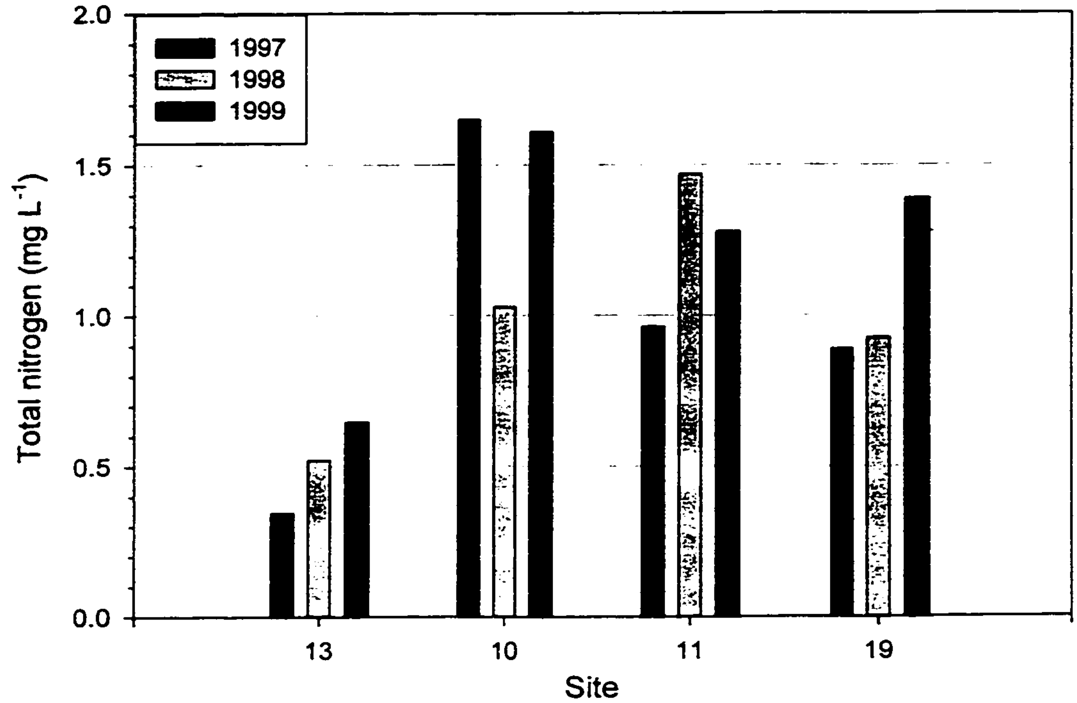


Figure 5.5. Total nitrogen flow-weighted mean during post-spring runoff at the Crowfoot Creek sub-basins.

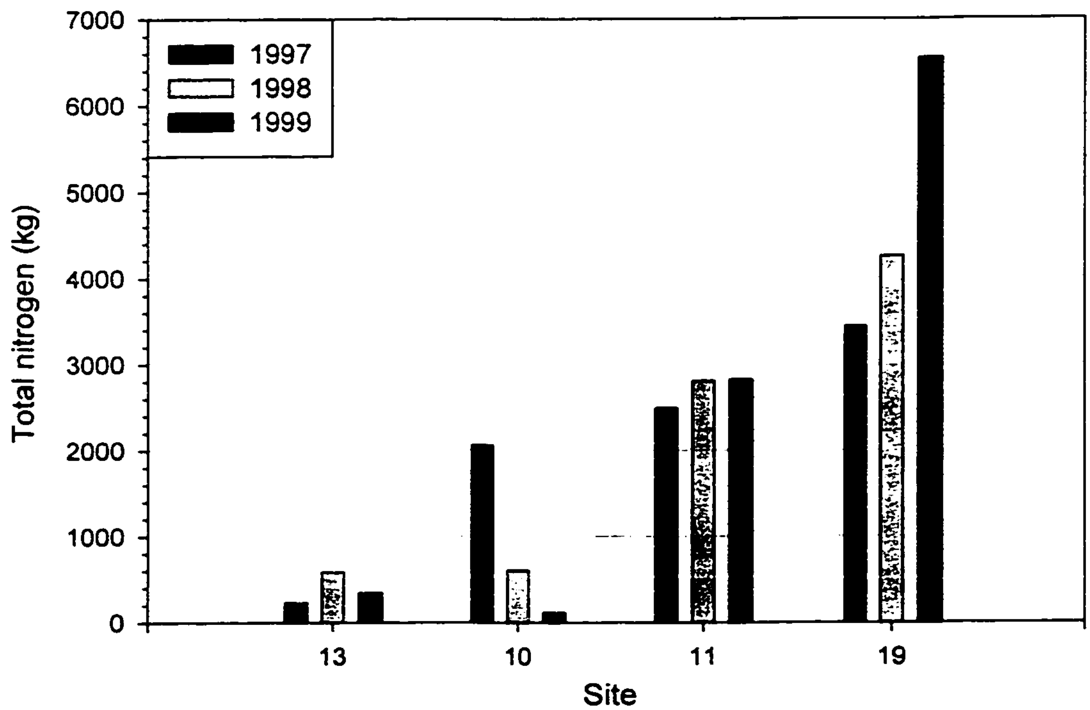


Figure 5.6. Total nitrogen mass load during post-spring runoff at the Crowfoot Creek sub-basins.

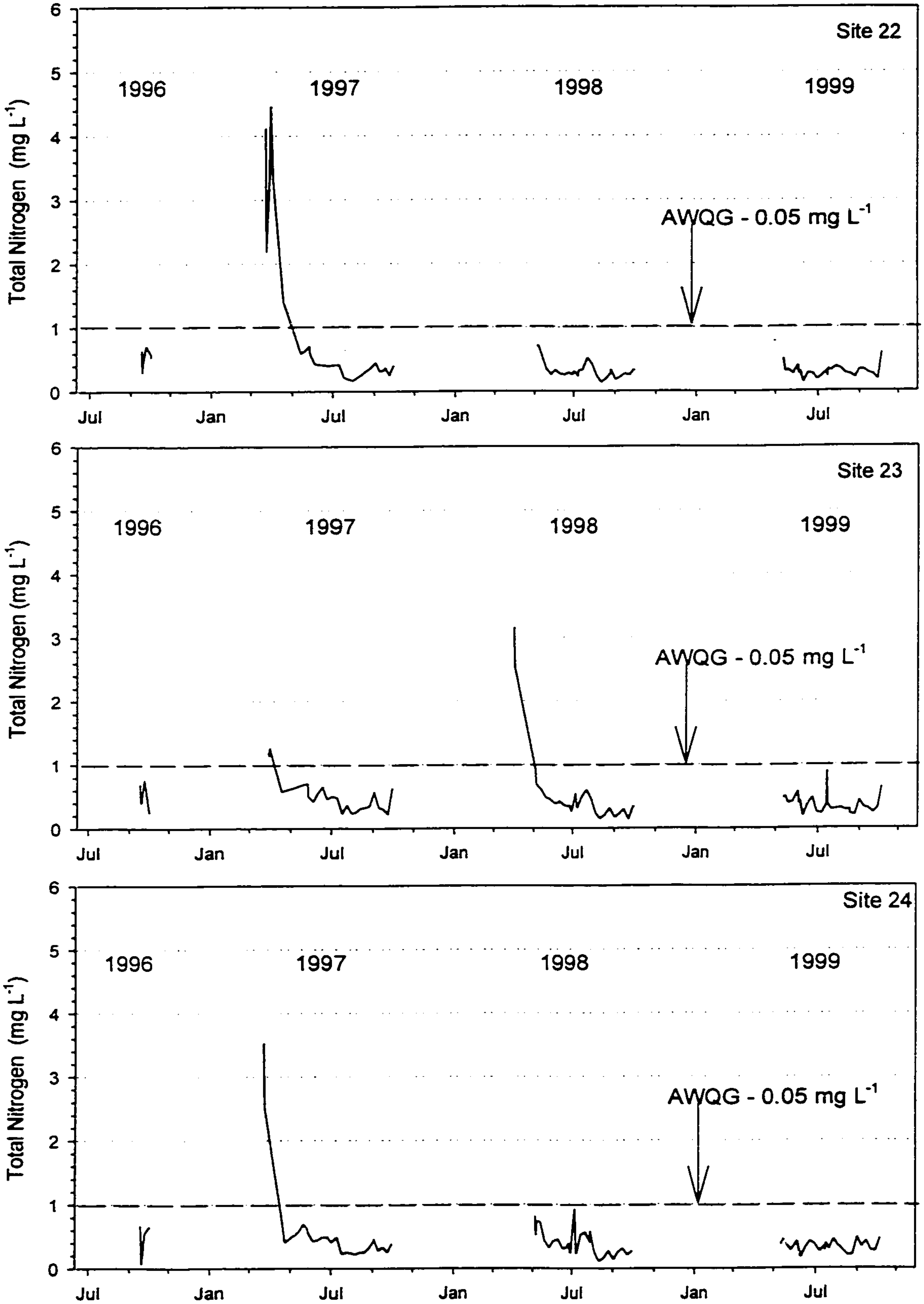


Figure 5.7. Total nitrogen at the inflow sites for the WID return flow sub-basins.

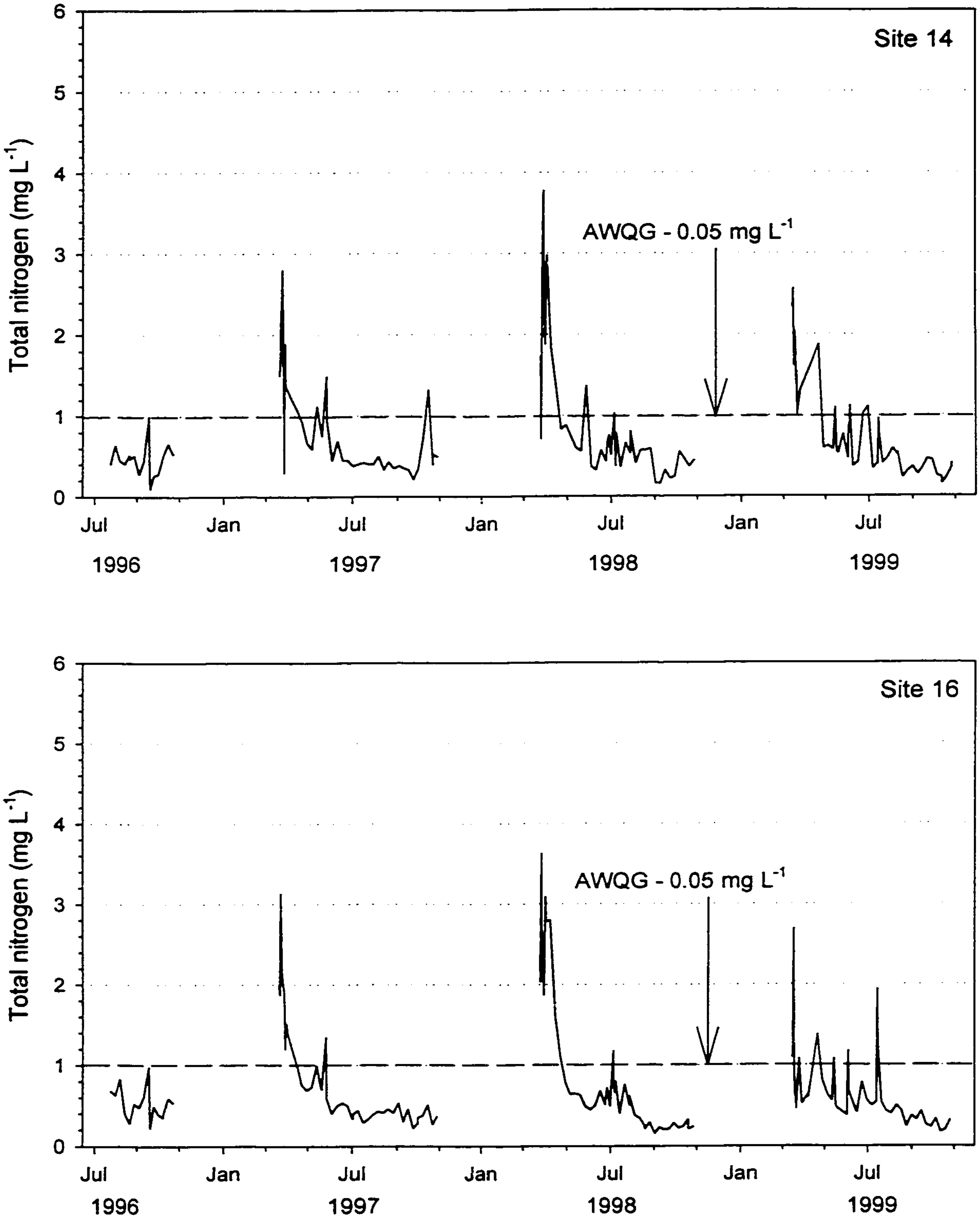


Figure 5.8. Total nitrogen at the outflow sites for the WID return flow sub-basins from 1996 to 1999.

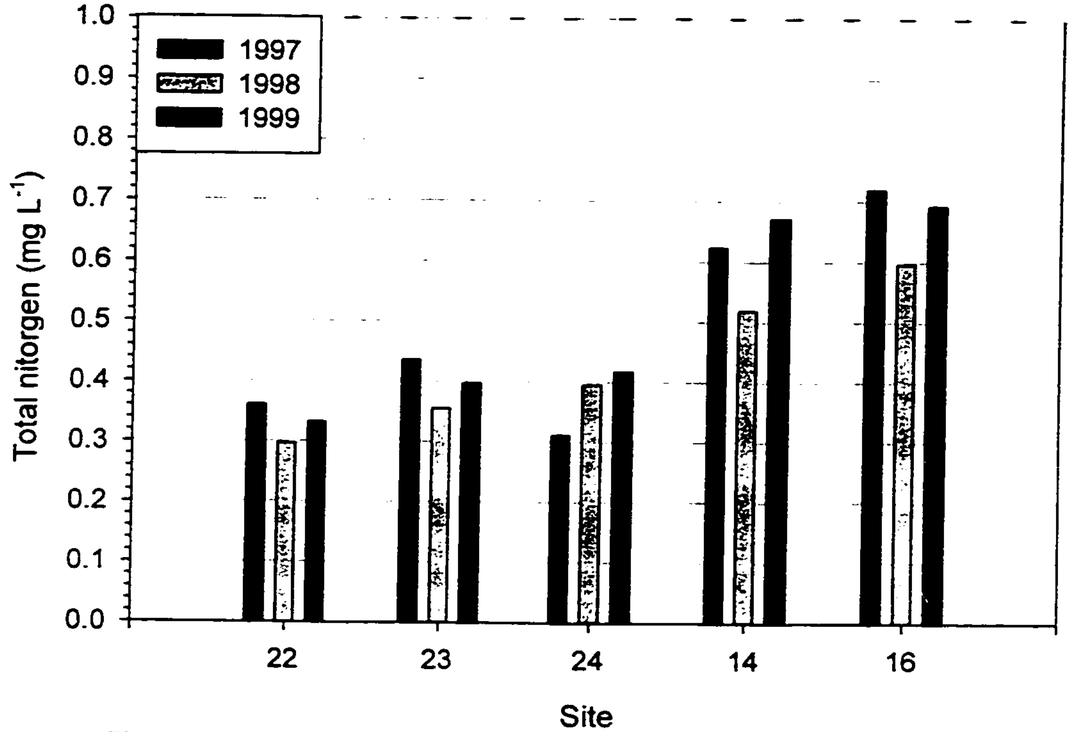


Figure 5.9 - Total nitrogen flow-weighted mean concentrations during post-spring runoff at WID return flow sub-basins.

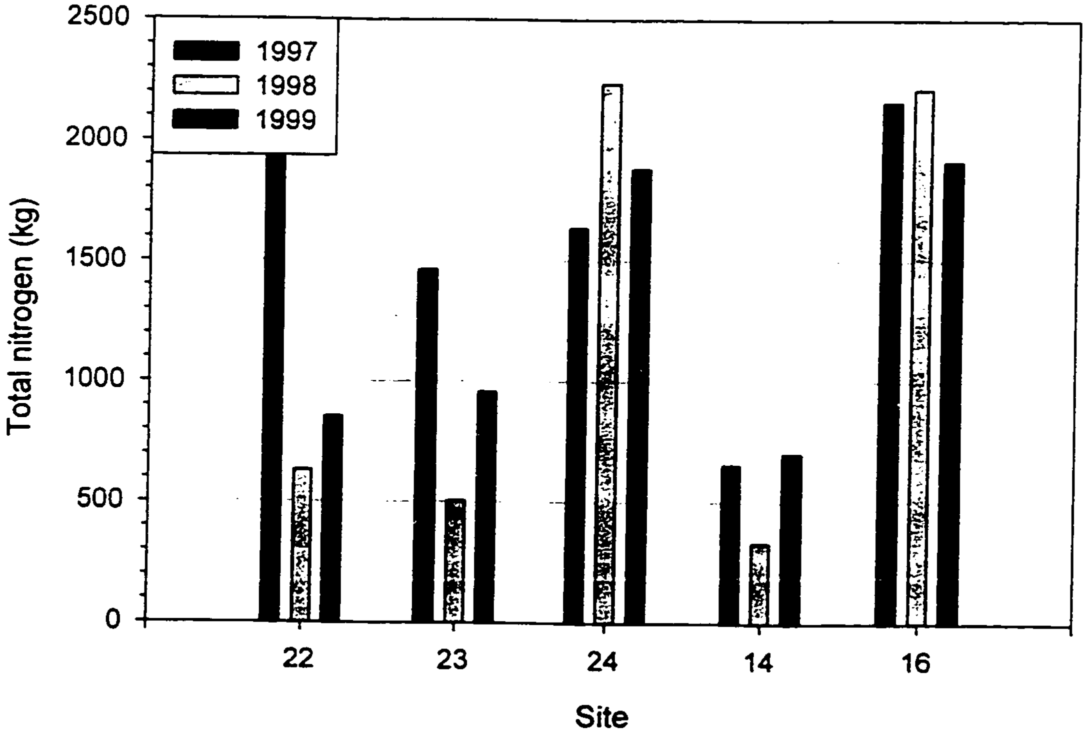


Figure 5.10 - Total nitrogen mass load during post-spring runoff at the WID return flow sub-basins.

6. MICROBIOLOGICAL IMPACTS ON SURFACE WATER QUALITY IN AN IRRIGATED WATERSHED

6.1 Introduction

In grassland areas, pathogens and indicator bacteria derived from warm-blooded animals reach streams by direct feacal discharges, movement with surface runoff and transport via sediment and waste particles (Gary and Adams 1985). The degree of contamination of surface water by pathogens depends primarily on the survival of the organisms and the degree of retention of the organisms by the soil (Gerba et al. 1975).

6.1.1 Factors Affecting Bacterial Survival

In general, temperature, pH, moisture, nutrient supply and solar radiation have the greatest effect on enteric (intestinal) bacterial survival (Gerba et al., 1975; Crane and Moore 1986), with temperature most often mentioned as affecting the survival of enteric bacteria. Several studies have shown that bacterial survival is inversely related to temperature below 15 °C. Studies of high elevation, cold water streams have shown both low and high counts of feacal coliforms (Thelin and Gifford 1983). In a laboratory study *E. coli* survived and multiplied in a low temperature, low nutrient situation (Thelin and Gifford 1983). Elevated temperatures, when combined with drying conditions will increase die-off rates (Gerba et al. 1975; Thelin and Gifford 1983; Crane and Moore 1986).

Moisture conditions are of great importance in determining bacterial survival in the soil. Increased soil moisture content resulted in increased survival rates at several temperatures (Crane and Moore 1986). Survival time in sandy soils was low compared to higher moisture retaining soils such as loams (Gerba et al. 1975).

Nutrient content of the soil and surface water will also affect the survival rate of organisms. In soil, the die-off of bacteria has been related to the organism's inability to lower its metabolism under low nutrient availability. The level of organic matter can, therefore, influence organism survival time (Crane and Moore 1986). In an aquatic environment, the nutrient content of the water will also affect the die-off rate of the bacteria, with high nutrient levels leading to increased bacterial survival (Gerba et al. 1975).

Extremes in pH increase die-off rates. Neutral to slightly alkaline conditions are the most suitable for extended bacterial survival. Ultraviolet light has also been observed to be bactericidal (Gerba et al. 1975; Crane and Moore 1983).

6.1.2 Immobilization in Soil and Water Systems

Viable organisms can be immobilized in the soil by the filtering action of the soil, adsorption to soil constituents, sedimentation and natural die-off (Gerba et al. 1975). Leaching of organisms will be limited by the soil pore size. In unsaturated soils under normal field conditions, water filled pore spaces are generally too narrow to allow the passage of bacteria and viruses deep into the soil profile. Conditions of unsaturated flow lead to the retention of large amounts of bacteria at or near the soil surface. Under saturated conditions, macropores would be available to transport organisms farther into the soil profile. (Gerba et al. 1975; Reddy et al. 1981; Reneau et al. 1989).

Bacteria and viruses can be retained in the soil. Both are amphoterically charged colloidal particles and, in most soil and water systems, the pH of the solution will result in a negatively charged particle (Gerba et al. 1975; Burge and Enkiri 1978; Burge and Marsh 1978; Reneau et al. 1989). The adsorption process is influenced by clay content and type, ionic species and ionic strength. One theory of the adsorption mechanism states that the amount of adsorption is dependent on the concentration and type of cations present in solution with maximum adsorption thought to be approximately 10 times greater for divalent than monovalent cations (Gerba et al. 1975). The organisms are linked to the clay particles via a clay-cation-organism bridge. The mutual repulsion between the clay particle and the organism is overcome by this bridging. A reduction in cation concentration will lead to a breakdown of the bridge effect and desorption of the organism will take place. This can occur when solutions of lower ionic strength, such as rainwater, are flushed through the system, or in the presence of organic matter. The organic matter will compete with the organisms for adsorption sites as well as extract the organisms from adsorbed positions.

A second theory states that as multivalent cations fix themselves to the soil particle, the net charge of the particle is reduced. If the charge is reduced sufficiently, the solid and the organism will move close enough to be affected by van der Waals forces (Gerba et al. 1975).

Due to the retention of organisms by clay particles and the filtering effect of the soil pore space, the majority of organisms are retained near the soil surface. This increases the potential for pollution of surface runoff water (Reddy et al. 1981). Erosion and surface runoff from a watershed must be considered as processes that may result in the contamination of surface waters with pathogenic organisms (Burge and Marsh 1978).

6.1.3 Indicator Bacteria

The need to examine the microbiological parameters of water quality as it relates to biological parameters has been recognized since 1855. At that time, Snow and Budd related outbreaks of typhoid fever and cholera to water contaminated with faecal wastes (Sloat and Ziel 1987). Since that time, numerous studies have shown that agricultural runoff can contain feacal bacteria in concentrations that exceed water use guidelines (Howell et al. 1995).

Many types of bacteria occur naturally in surface water and have no impact on human health. However, some types of pathogenic bacteria and viruses enter the aquatic environment as a result of human activity and can pose health threats (Saffran 1996). Pathogenic species include *Salmonella*, *Shigella*, *Yersinia enterocolitica*, *Klebsiella pneumoniae*, *Vibrio cholerae* and *Campylobacter jejuni*. Diseases caused by these organisms are spread by water contaminated with faecal material from humans and other warm-blooded animals (Sloat and Ziel 1987).

Bacteriological water quality is determined through the use of "indicator bacteria". There are several reasons for this. The indicator bacteria are usually present in greater numbers, are easier to isolate and are safer to work with than the pathogenic organisms. If the indicator organisms are absent, the probability of the existence of bacterial pathogens in the water is small. Conversely, if the indicator organisms are present in certain amounts, there is a greater likelihood that pathogens are also present (Thelin and Gifford 1983; Sloat and Ziel 1987).

The indicator organism used to indicate the presence of enteric bacteria must be exclusively of faecal origin and consistently present in fresh faecal waste. Organisms used as bacterial indicators of pollution include coliform bacteria, faecal streptococci (enterococci) and sulfite-reducing clostridia (i.e. *Clostridium perfringens*) (Sloat and Zeil 1987). Coliform bacteria are commonly used as indicators of anthropogenic impacts on water quality (Saffran 1996).

Coliform bacteria belong to the Enterobacteriaceae family and consist of aerobic and facultative anaerobic, gram-negative, non-spore forming, rod-shaped bacteria which ferment lactose to produce gas and acid within 48 h at 35 °C. A problem with the use of total coliform counts as an indicator of the presence of enteric bacteria is that a high total coliform count can be the result of inclusion of bacteria of non-faecal origin such as *Klebisella* and *Citrobacter*. These organisms do not necessarily pose a health risk (Sloat and Zeil 1987; Saffran 1996).

In 1904, a test was developed to determine the presence of coliforms of faecal origin. Faecal coliforms are thermotolerant bacteria which ferment lactose to produce acid and gas at 44.5 or 45 °C and generally originate from the gastrointestinal tract of warm-blooded animals (Sloat and Zeil 1987; Saffran 1996). Faecal coliform bacteria are, therefore, a more useful indicator of water contamination by sewage than total coliforms, which may originate from non-faecal sources such as the soil and organic matter (Thelin and Gifford 1983; Saffran 1996). Faecal coliforms can indicate contamination by feces of livestock, wildlife and waterfowl as well as humans (Saffran 1996). The microbiological quality standards for most natural water sources are based on faecal coliform counts (Sloat and Zeil 1987).

Because some bacteria identified by the faecal coliform test can live and reproduce in organic rich waters, a specific faecal coliform indicator is used. *Eschericia coli* (*E. coli*) is the indicator bacteria used to assess the level of faecal contamination from animal or human waste due to its high correlation with rates of gastrointestinal illness in humans. *E. coli* make up 90-100% of the coliform organisms in human and animal waste. In sewage and contaminated water samples, this percentage drops to 59% (Sloat and Zeil 1987; Saffran

1996).

Although *E. coli* has been the indicator of choice, their enumeration has in the past required complicated, time consuming and expensive techniques (CCREM 1987). Total coliform and faecal coliform counts were more easily used as indicators of faecal contamination. Recent refinement of methods to enumerate *E. coli* have led to re-consideration of its use in water quality monitoring (CCREM 1987; Sloat and Zeil 1987).

6.1.4 Water Quality Standards

The Canadian Water Quality Guidelines (CCREM 1987) and the Surface Water Quality Guidelines for Use in Alberta (AENV 1999) present guidelines for both faecal and total coliforms and *E. coli*. The following guideline values were used. For treated drinking water, the guideline is 0 counts of faecal coliforms per 100 ml and 10 counts of total coliforms per 100 ml of water. No surface water should be used for consumption without treatment.

The Alberta Surface Water Quality Guideline (AENV 1999) for irrigation is 200 faecal coliforms per 100 ml for the irrigation of vegetable crops. The value is derived by calculating the geometric mean of at least five samples over 30 days. The CWQG is 100 faecal coliforms per 100 ml and covers all irrigated crops.

ASWQG and CWQG for contact recreation (200 counts per 100 ml) use a geometric mean of not less than 5 samples within a 30-day period to determine guideline compliance. A value of 400 counts of fecal coliform bacteria per 100 ml can be used as an instantaneous measure of water quality relating to contact recreation if the data for the calculation of the geometric mean do not meet the criteria. Values exceeding this level may require more detailed sampling (Health and Welfare Canada 1992). The 400 faecal coliform per 100 ml value was used in this thesis. The definition of potable water is water that can be treated for use as a drinking water supply. The CWQG for potable water is 1000 faecal coliform per 100 ml.

6.1.5 Contributions From Agricultural Land

Numerous studies have indicated the presence of cattle grazing on grassland have an adverse impact on the levels of coliform bacteria found in surface water. Doran and Linn (1979) also concluded that the water quality from pasture and rangeland often exceeded water quality guidelines. They also cautioned that wildlife could also have an adverse effect on water quality. Gary and Adams (1985) stated that the levels of indicator bacteria in surface water were low, except when sheep and cattle had recently been grazing in the area. Howell et al. (1995) observed that when faecal bacteria were present, rainfall rapidly moved them from the soil surface into well and spring water. This was in an area of continuous cattle grazing. However, Buckhouse and Gifford (1976) reported that, although there were levels of bacteria present to be transported, there was no impact on surface water by faecal coliforms from grazing use. The latter study was undertaken in an arid area of south-eastern Utah with little runoff.

Studies in irrigation system flows in Alberta indicated some impact from the surrounding area. Some levels of faecal coliforms in exceedence of water quality guidelines were observed in the inflow and return flows of several irrigation districts (Madawaska Consulting 1997). Greenlee et al. (2000) found that, for faecal coliforms, the water quality at sites entering four watersheds examined in the Lethbridge Northern and Bow River Irrigation Districts met guidelines more often than water sampled at the outlets. The increase indicated a source of faecal coliforms within these irrigated basins.

The objective of this chapter was to determine if microbiological water quality varied spatially or temporally along a selected reach of Crowfoot Creek and at several irrigation return flow points, and to determine if agricultural practices were affecting the quality. Faecal coliform levels were examined for compliance with various guidelines. Variability in the concentrations of indicator bacteria and relative loadings was also assessed.

6.2 Methods

Methods of data collection and analysis were presented in Chapter 2.

6.3 Results and Discussion

6.3.1 Spring Runoff - Crowfoot Creek Sub-basins

Concentrations of faecal coliform bacteria (FC) exported from Sub-basin 10 during the three spring runoff periods followed the same general trends. Several peaks were observed with counts decreasing to low levels by the end of the runoff period.

Median FC counts per 100 ml were very consistent over the three years, with counts less than 40 detected. These median values did not exceed the CWQG for irrigation (100 counts 100 ml^{-1}), contact recreation (400 counts 100 ml^{-1}) or potable water (1000 counts 100 mi^{-1}) (Table 6.1). Peak FC counts were much higher, ranging from 310 to 1000 counts 100 ml^{-1} over the three spring runoff periods.

The FLUX model was adapted to calculate "pseudo" flow-weighted mean concentrations and mass loads. The values produced were not a weight per unit volume for concentration or a weight for mass load, but give an indication of the variation of FC counts due to flow variation as well as an indication of the numbers of organisms passing out of the sub-basins. The coefficient of variation for the FC data calculated with the model was often higher than the level of variation desired when modeling standard FLUX parameters such as nutrients. The higher variation is due to the processes by which bacteria enter the surface water. While many of the substances that affect water quality enter surface water via overland flow related to snowmelt and precipitation, coliform bacteria can be deposited in the water any time an animal has access to the water. As the FLUX model develops a relationship between concentration and flow, this open access to the water weakens the relationship, resulting in more variation in the FLUX estimates of concentration and load.

The FWMCs for Sub-basin 10 were variable over the three years (Fig. 6.1). Values in 1997 and 1999 were much higher than 1998, the result of consistently higher FC counts detected during those years. As with the nutrient parameters, mass load was flow dependent, with FC loads in 1997 higher than those in 1998 or 1999 (Fig. 6.2).

In Sub-basin 19, the temporal trends at the inflow and outflow sites also peaked and then decreased to low levels by the end of the spring runoff period. Statistical comparison of the median count values between inflow and outflow revealed no significant differences between the sites in any of the spring runoff periods (Table 6.1). Median values were well below irrigation, contact recreation and potable water guidelines. Peak count values exceeded irrigation guidelines at Sites 11 and 19 in 1997, while peak values were below the guidelines in 1998 and 1999.

FWMCs at the inflow, Site 11, were higher than the outflow, Site 19, in 1997 and 1999 (Fig. 6.1). In 1998, the FWMC was slightly higher at the outflow. The large difference in FWMCs compared to median values was due to the influence of a few very high values during the monitoring period. Mass loads were highest in 1997 during the large runoff, while loads in the remaining years were much lower, due to the reduced runoff volume (Fig. 6.2).

The presence of FC bacteria in the spring may have been from natural sources such as beavers or birds, or introduced sources such as cattle, hogs, horses or sheep. Faecal coliform bacteria can survive long periods of time in the environment, as much as a year or more when contained in faecal material (Buckhouse and Gifford 1976; Theilin and Gifford 1983). A study by Putz et al. (1982) found no significant die-off of microorganisms in acclimatized effluent discharged into an ice-covered river. Natural sources of FC contributing to contamination are indistinguishable from introduced sources using faecal coliform analysis (Doran and Linn 1979). The high peak FC counts at Site 10 compared to the other two Crowfoot Sites may be an indication of FC input from introduced sources at this site. The low median values and lack of a significant change in FC counts between Sites 11 and 19 indicated little impact on FC numbers along this reach.

The high levels of FC exiting Sub-basin 10 may have been due to the presence of cattle in the sub-basin. Cattle were stocked on the grassland areas near the creek, generally during the period from May to October. Several studies have indicated that the presence of cattle in a watershed can have an adverse impact on biological water quality and that FC can survive long periods of time in the environment, even in cold temperatures (Buckhouse and Gifford 1976; Gary and Adams 1985; Howell et al. 1995). As a result, the FC detected may be from cattle grazed during the previous season, or natural wildlife in the area. Cattle were not observed in close proximity to the creek in Sub-basin 19 during spring runoff, therefore, median and peak FC counts tended to be lower than in Sub-basin 10.

Peak FC values occurred early in the runoff period in 1997 and 1999. In 1998, the peak occurred later in the runoff period and was not very distinct. The large runoff in 1997 may have contributed to a flushing of the area and high FC values. In 1999, the early peak may have been due to the initial melt flushing the FC out, while the late peak or lack of a distinct peak in 1998 may have been a result of fewer cattle in the sub-basins.

Decreasing FWMC between sites in Sub-basin 19 indicated that any contribution of organisms to the creek in this sub-basin was exceeded by their natural die-off, resulting in the decreasing concentrations. The small increase in FWMC in 1998 was likely due to an input of bacteria along the reach in that year. The overall low levels of FC in 1998 made it difficult to assess the actual variation in that reach.

Overall, the median values of FC detected at the Crowfoot Creek sites were very low. While the high peak values indicated a contribution from the surrounding land, the levels rapidly decreased and there was little negative impact on the FC levels in the creek. The pattern of management of the livestock within the sub-basins influenced the degree of impact from bacteria. If cattle were not wintered within the sub-basin, there would have been fewer faecal coliforms to contaminate the surface water (Doran and Linn 1979). The management of the sub-basins over the winter was not directly monitored, but the field logs indicated that cattle were not present around the monitoring sites during the spring runoff period. The low FC counts were indicative of the lack of direct impact on the creek by animals during this period and the high peak values indicated some adverse impact from the surrounding land. This impact may be due to the contributions of wildlife as well as livestock (Doran and Linn 1979).

6.3.2 Spring Runoff - WID Sub-basins

Temporal trends observed in the WID sub-basins were similar to those observed in the Crowfoot Creek sub-basins. FC peaks occurred early in the spring runoff period and decreased to low levels by the end of the runoff period. Median faecal coliform counts at Sites 14 and 16 were below CWQG for irrigation, while peak values exceeded irrigation, contact recreation and potable water guidelines (Table 6.1). The median values tended to be slightly higher at the WID sites than at Sites 11 and 19 on Crowfoot Creek. Median values at Site 10 were higher than at the WID sites in 1997 and 1999.

The FWMCs at Site 14 increased each year, while at Site 16 the FWMCs increased from 1997 to 1998, but were the lowest in 1999 (Fig. 6.1). As with the Crowfoot Creek sites, the variability of the FWMCs was likely due to several high FC counts during the spring runoff period as the median values indicated that at least half of the samples taken had low FC concentrations. Mass loads, due to the higher flow volumes, were greater at Site 16 than at Site 14 in 1997 and 1998 (Fig. 6.2). In 1999, FC loads at Site 16 were similar to, although slightly lower than, Site 14.

As at the Crowfoot Creek sites, the WID return flow sites had contributions of faecal coliforms from both natural and introduced sources. Sub-basin 14 had a cattle population during the majority of the time from May to September or October. In 1998, cattle were observed in this sub-basin as early as April 1, perhaps resulting in the high FC counts near the end of April. The faecal material may contain viable organisms through the winter and release them with the spring runoff. There were also cattle present in Sub-basin 16, although not in visual range of the site. Field logs did not indicate cattle near the site during the spring runoff period.

6.3.3 Post-spring Runoff - Crowfoot Creek Sub-basins

Temporal trends at the Crowfoot Creek sites showed an increase in FC counts during the early part of the PSRO monitoring period, which may be related to the start of diversion into the WID irrigation infrastructure. FC counts stayed fairly constant from May to September, with the highest values occurring during the summer, then decreasing from the end of September to the end of the PSRO period (Figs. 6.3 and 6.4). Peak values were often associated with rainfall events.

In Sub-basin 10, the median counts at the inflow and outflow sites were fairly consistent over the PSRO period (Table 6.2). Median values increased between the inflow and outflow sites in all years of the study. In 1998 and 1999, this increase was statistically significant. Median inflow counts were at or below 100 counts 100 ml^{-1} at Site 13. At Site

10, median values were between 200 and 300 counts 100 ml^{-1} in all years, in excess of the irrigation guideline.. The magnitude of peak FC counts varied widely between years and most were associated with rainfall events (Table 6.2). Peak values were extremely high and exceeded the potable water guideline.

Median FC counts in Sub-basin 19 were also fairly consistent over the PSRO period (Table 6.2). A significant increase in FC between the inflow and outflow of Sub-basin 19 occurred in all years of the study. Median values at Site 11 were well below 100 counts 100 ml^{-1}. At Site 19, median counts exceeded the irrigation guideline of 100 counts 100 ml^{-1} in 1996 and 1997, but in 1998 and 1999 median values were below 100 counts 100 ml^{-1}. As in Sub-basin 10, the peak values were extremely high and exceeded the potable water guideline by a large amount.

In Sub-basin 10, the FWMCs were similar at the inflow from year to year, decreasing slightly over the study period (Fig. 6.5). At the outflow site, FWMCs increased greatly each year. Due to the differences in the flow volume, the mass load at Site 13 was fairly constant in 1997 and 1998, but decreased by half in 1999. At Site 10, the mass load was consistent over the study years, with decreasing flow volume being offset by the increasing number of FC (Fig. 6.6).

The FWMCs at Site 11 increased greatly from 1997 to 1998, but decreased in 1999, while at Site 19 the FWMC decreased by approximately half from 1997 to 1998 (Fig. 6.5). In 1999, the FWMC was much higher, over 3,000 counts 100 ml^{-1}. Between inflow and outflow sites, the FWMC increased in 1997 and 1999, and decreased in 1998.

Mass loads in Sub-basin 19 during the PSRO period were flow dependent. In 1997, flows were much higher at Site 19 than Site 11 (Fig. 6.6). In 1998, while flows were higher at Site 19, the FWMC was much higher at Site 11, resulting in similar mass loads between the sites. In 1999 at Site 19, flows and FWMC resulting in higher loads than at Site 11.

At all Crowfoot Creek sites, median levels of FC were higher during the PSRO period compared to spring runoff. Cattle were generally turned out to graze in the Crowfoot Creek watershed anytime after the beginning of May and were removed in September or October. Additionally, cattle were often turned out onto stubble to graze in the fall, increasing the area occupied (Personal Communication - Brenda Ralston, June 12, 2000). Placement of cattle in the sub-basins corresponded to the increase in FC counts, while their removal resulted in a decrease in count magnitude.

This response was similar to that from other studies carried out on grazing areas. Howell et al. (1995) found that faecal coliforms were present in surface waters and increased when cattle were present. Pastures with cattle had higher faecal coliform counts than those without. They also determined that the FC concentrations remained high for as long as a month after the cattle were removed. Gary and Adams (1985) observed that when cattle were moved into an area near a monitoring site, higher densities of FC were detected. When the cattle were removed, the densities decreased. As stated in the section on spring runoff, indicator organisms were present in areas that did not have introduced hosts (Doran and Linn 1979). As a result, the FC numbers in these sub-basins likely had a portion of natural inputs. However, Buckhouse and Gifford (1976) found that there was little danger of contamination in dry areas with ephemral streams unless faecal material was deposited directly in the stream. Contaminated overland flow did not travel sufficient distances to adversely impact any water bodies or streams.

There was a significant increase in FC counts in Sub-basin 10, indicating an input of FC, with cattle a possible source. Field logs indicated the presence of cattle in this area from early June to the end of the monitoring period.

The FC counts at the inflow to Sub-basin 19 were low due to the presence of the Hilton wetland upstream of this site. The retention of FC in this wetland likely resulted in significant die-off of FC due to exposure to UV light, increased temperature and other water quality parameters. There were also not a large number of cattle in this area to add FC to the creek. As a result, low FC counts were observed at the wetland outlet, Site 11. FC counts increased significantly within the sub-basin. Although this sub-basin was dominated by annual cropland, areas bordering the creek and other drainage channels were sometimes fenced for use as grazing areas. These areas were often too rough to be farmed, or were a means of eliminating odd-shaped parts of a field. The cattle had grass for feed and open access to the water, allowing direct input of faecal material into the water. During high flows, if the water overtopped the banks of the channel, faecal material from the adjacent land would be washed downstream.

The FWMCs were influenced by the amount of FC carried into the creek by rainfall events and by the presence of animals in the creek. Many of the rainfall events resulted in elevated levels of FC. The high count values related to the storms affected the FWMC for the entire season. The FC counts at Sites 10 and 19 were examples of this during the rainfall in July 1999. The large inputs of FC caused the overall FWMC to be very high. As a result of the higher flows and higher FWMCs, the mass loads at both sub-basin outflow sites were higher than the inflows, indicating an increase in the amount of FC bacteria in both Crowfoot Creek sub-basins during the PSRO period. This level was primarily due to the presence of livestock in the sub-basins.

6.3.4 Post-spring Runoff - WID Return Flow Sub-basins

Temporal trends at the inflow and outflow sites of the WID return flow sub-basins were similar to those at the Crowfoot Creek sub-basins (Figs. 6.7 and 6.8). Count levels increased in May as cattle were put into grazing areas and decreased in the fall when the cattle were removed. All of the inflow and outflow sites had erratic patterns with many peaks and valleys.

Inflow sites (Sites 22, 23 and 24) have median count levels below 100 counts 100 ml^{-1} in all three years. Peak values at these sites exceeded 100 counts 100 ml^{-1} with a maximum of 2,700 counts 100 ml^{-1} at Site 23 in 1999. Outflow sites (Sites 14 and 16) have median values exceeding 200 counts 100 ml^{-1} in all years, exceeding irrigation guidelines. Peak values at these sites exceeded 2,000 counts 100 ml^{-1}, with a maximum of 29,000 counts 100 ml^{-1} at Site 14 in 1998, well over agricultural and recreational guidelines. Statistical comparison of the median values between inflow and outflow sites in each sub-basin showed a significant increase in FC counts in each year (Table 6.3).

The FWMCs also indicated a large increase in FC within the WID return flow sub-basins (Fig. 6.9). Inflow site FWMCs were fairly consistent over the three years, with values below 200 counts 100 ml^{-1}. FWMCs at the outflow sites followed a similar pattern. At Site

14, the FWMCs increased each year. In 1998, the concentration was approximately one third higher than in 1997, while in 1999 the FWMC was more than three times as high. At Site 16, the magnitude of the counts was not as high as Site 14, but FWMCs in 1999 were much higher than in the previous two years.

Mass loads of FC were flow dependent and reflected the volume of water passing each inflow monitoring site (Fig. 6.10). Trends in mass loads at Sites 14 and 16 were opposite to those of the FWMCs, due to the higher volume of water passing out of Sub-basin 16 compared to Sub-basin 14. The number of organisms passing out of Sub-basin 14 was lower than Sub-basin 16, however, the concentration was higher.

As with the Crowfoot Creek sub-basins, the level of impact of FC on water quality in the WID return flow sub-basins was related to the presence of cattle. In Sub-basin 14 during the PSRO period, there was a large amount of grassland adjacent to the channel, which allowed cattle open access to the water. In Sub-basin 16, there were also some large areas of grass that were grazed during the PSRO period, as well as several reaches of the return flow channel that had been fenced on both sides of the channel. These areas tended to be too rough for cultivation and were fenced off for grazing. The cattle had grass and water and grazed these areas during the PSRO period, increasing the potential for the addition of FC. During periods of high flow, water overtopping the banks of the channel would wash faecal material deposited on the sides of the channel into the stream.

6.3.5 Event Flows - Crowfoot Creek Sub-basins

Rainfall events during the PSRO period had varying effects on the levels of faecal coliform bacteria detected in Crowfoot Creek. FC counts from individual samples collected during and soon after rainfall events tended to be higher than counts in samples taken prior to the rainfall, as well as the median values for the site.

A comparison of the FWMCs for the event periods, as calculated by the FLUX model and the FWMC for the PSRO period, gave mixed results (Table 6.4). FWMCs for some of the events were less than or similar to FWMCs for the PSRO period. Examples of this are in May 1997 and May (Event 1) and June (Event 2)1999. In 1999, PSRO period FWMCs were elevated due to high FC counts detected during the third event in July. The event in July 1998 resulted in higher FWMCs than the PSRO level in Sub-basin 10. In Sub-basin 19, the inflow counts were slightly higher, while the outflow was slightly lower. The greatest impact on FWMCs was during a rainfall event in July 1999. FWMCs at the inflows to both sub-basins increased slightly, while the outflows increased substantially. Individual sample faecal coliform counts during this event reached 62,000 and 33,000 counts at Sites 10 and 19, respectively.

Peak concentrations at all of the Crowfoot Creek sites were often, but not always, related to rainfall events. The FC counts from the samples collected during the events were not always the highest values at that site in that monitoring period. There were occasions when high FC counts were recorded during non-event related periods. The ability of animals carrying the FC bacteria to access the water at any time, not just during events, may have resulted in elevated FC counts.

Examination of the FLUX data did not indicate the same tendencies as observed with the nutrient parameters. The FWMCs during the nonevent periods were not always lower than during the events. There appeared to be a decrease in the FWMCs during the last month of the monitoring period at the sites where this month was examined by the FLUX model. This decrease corresponded with the removal of cattle from the sub-basin in September and October.

The lack of correlation between higher FWMCs during events and lower FWMCs during non-event periods was due to the access of cattle during non-event periods. FC counts increased during non-event periods as cattle accessed the water. As a result, the FWMCs that are derived using the FLUX model were higher than would be expected in non-event periods.

6.3.6 Event Flows - WID Return Flow Sub-basins

While the highest concentrations of FC did not always correspond with rainfall events, FC counts in these sub-basins tended to increase in response to rainfall (Table 6.4). In 1997 and 1998, peak FC levels at Site 14 occurred during non-event periods. In 1999, each of the sampled events at Site 14 had elevated FC counts over the samples taken outside the event. At Site 16, peak concentrations of FC in 1997 occurred late in the season, while the event in May did not seem to have a large impact on FC levels. The FC levels detected during the sampled events in 1998 and 1999 were usually higher than pre-event levels and indicated a response to the rainfall and surface runoff. There were several high FC counts that were not related to rainfall events. As in the Crowfoot Creek sub-basins, this was related to the open access that cattle in the sub-basins had to the water.

FC count levels during events generally exceeded the guideline for irrigation and contact recreation. In 1999, levels at Site 14 exceeded the guideline for potable water during all events, while at Site 16 FC levels exceeded the potable water guideline once, as well as falling just below it during another event.

Comparing the WID sub-basins to the Crowfoot Creek sub-basins gave a clear indication of which sub-basin had better faecal coliform levels. Examination of the landcovers indicated that the sub-basins with grassed areas near the watercourse and resident cattle populations during the summer had the highest levels of FC contamination.

6.4 Conclusions

During the spring runoff period, levels of coliform were generally fairly low, however, peak values exceeded water quality guidelines for irrigation, contact recreation and potable water uses. Concentrations exiting Sub-basin 10 were higher than those from other sub-basins and may be related to the presence of cattle in this sub-basin at some time during the winter or before spring runoff began. In Sub-basin 19, the decrease in concentration between inflow and outflow, and the low magnitude of the values, indicated that few faecal coliform bacteria were added to the creek during the spring runoff period in this sub-basin. In the WID sub-basins, FC concentrations were also very low and indicated a small adverse impact related to these bacteria during spring runoff.

Cattle were not as prevalent in the study areas during spring runoff compared to the rest of the monitoring period. As a result, the overall faecal coliform bacteria levels were low during the spring runoff period and may be related primarily to non-agricultural, indigenous wildlife. The longevity of FC bacteria in the environment may also have been responsible for some of the FC detected. Mass loads also seemed to support the theory of decreasing faecal coliform numbers in the spring, as there was a net reduction in the mass load within Sub-basin 19.

Faecal coliform counts increased within all the sub-basins during the PSRO period in all years, indicating a source of bacteria within the sub-basins. Inflow water tended to meet most agricultural water quality guidelines. Mass load values indicated a net export of bacteria from each of the sub-basins during the PSRO period. Increases in FC during the early part of the PSRO period, likely resulted from the introduction of cattle into the sub-basin in the early summer, while decreasing values in the fall corresponded to their removal. FC levels were much greater during the PSRO period than in the spring, with median values often ten-fold higher.

The greatest increases in bacterial numbers occurred in sub-basins with the greatest presence of cattle near to, or along, the watercourse. Sub-basins 10, 14 and 16 had cattle present during the majority of the PSRO period, had larger median values and showed greater increases in concentration than Sub-basin 19, which had fewer cattle in close proximity to the creek. Site 11, the inflow to Sub-basin 19, had very low counts of FC due to the mitigating effect of the Hilton wetland. The increased detention time in the wetland likely resulted in the die-off of many of the FC bacteria.

Rainfall events often resulted in elevated FC counts, but not all of the peak values occurred during rainfall events. The ability of the cattle to access the watercourses at any time resulted in increased contamination during non-event periods. The longevity of the FC, possibly over several months, also meant there was potential for FC contamination after the cattle had been removed from the sub-basins.

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Table 6.1 - Spring runoff faecal coliform summary statistics and median comparisons for all study sites.

	Faecal coliform (counts 100 ml^{-1})				
	Min	Median	Max	SD	Flow Volume (x 10^6 m^3)
			1997		
Site 10*	2	38	1000	349	1.87
Site 11[y]	1	10a	240	78	3.27
Site 19[y]	1	4a	130	38	5.80
Site 14*	1	12	112	37	0.92
Site 16*	1	23	91	32	3.84
			1998		
Site 10*	7	37	310	90	0.23
Site 11[y]	1	7a	11	3	0.17
Site 19[y]	2	12a	57	16	0.10
Site 14*	1	67	3700	1103	0.03
Site 16*	4	32	430	129	0.10
			1999		
Site 10*	4	33	620	219	0.03
Site 11[y]	1	3a	64	25	0.03
Site 19[y]	1	2a	73	20	0.09
Site 14*	8	19	150	42	0.03
Site 16*	2	8	480	135	0.12

* - no statistical analysis carried out.
y - median values followed by the same letter are not significantly different ($p < 0.05$).

Table 6.2 - Crowfoot Creek sub-basins post-spring runoff faecal coliform summary statistics and median comparisons[y,z].

	Faecal coliform (counts 100 ml^{-1})				
	Min	Median	Max	SD	Flow Volume (x 10^6 m^3)
1996*					
Site 13	35	93	1700	663	0.56
Site 10	27	230	3600	1281	0.57
Site 11	7	28	39	13	1.52
Site 19	24	150	670	221	2.19
1997					
Site 13	21	100a	640	153	0.67
Site 10	13	210a	2500	630	1.25
Site 11	1	17a	140	33	2.21
Site 19	4	130b	650	176	3.92
1998					
Site 13	8	69a	270	62	1.12
Site 10	5	205b	6900	1347	0.59
Site 11	1	32a	1100	211	1.90
Site 19	4	80b	460	115	4.59
1999					
Site 13	8	50a	2500	664	0.53
Site 10	28	280b	62000	13036	0.07
Site 11	2	18a	470	112	2.20
Site 19	6	69b	33000	6022	4.69

* - Due to reduced sample numbers and different sampling pattern, statistical analysis was not carried out for this year.

y - median values followed by the same letter are not significantly different ($p<0.05$).

z - statistical comparisons were carried out only between the inflow and outflow of each sub-basin.

Table 6.3 - WID return flow sub-basins post-spring runoff faecal coliform summary statistics and median comparisons[y,z].

Faecal coliform (counts 100 ml^{-1})					
	Min	Median	Max	SD	Flow Volume (x 10^6 m^3)
1996*					
Site 22	No data	No data	No data	No data	No data
Site 14	230	1640	3640	1426	0.47
Site 23	No data	No data	No data	No data	No data
Site 24	No data	No data	No data	No data	No data
Site 16	128	400	2000	636	1.18
1997					
Site 22	8	22a	636	139	5.40
Site 14	80	580b	6000	1280	1.05
Site 23	9	47a	200	42	3.36
Site 24	12	56a	180	35	5.25
Site 16	23	430b	4300	835	3.00
1998					
Site 22	5	42a	210	45	2.12
Site 14	10	775b	29000	5396	0.64
Site 23	4	44a	520	96	1.42
Site 24	15	80b	1500	297	5.64
Site 16	11	275c	2000	483	3.70
1999					
Site 22	6	46a	1200	244	2.57
Site 14	8	750b	20000	3915	1.05
Site 23	2	28a	2700	610	2.41
Site 24	3	58a	850	182	4.50
Site 16	27	380b	6000	1367	2.76

[y] - median values followed by the same letter are not significantly different ($p < 0.05$)

[z] - statistical comparisons were carried out only between the inflow and outflow of each sub-basin.

* - Due to reduced sample numbers and diffemt sampling pattern, statistical analysis was not carried out for this year.

Table 6.4 - Faecal coliform event and post-spring runoff (PSRO) flow-weighted mean concentrations (FWMC).

	Faecal coliform (counts 100 ml^{-1})							
	1997		1998		1999			
	Event	PSRO	Event	PSRO	Event 1	Event 2	Event 3	PSRO
Site 13	N/D	115	104	71	72	90	71	61
Site 10	196	351	4242	932	667	731	35057	7828
Site 11	22	15	394	307	8	57	59	54
Site 19	288	205	88	104	41	20	15292	3335
Site 14	381	590	833	876	1385	4178	6877	2068
Site 16	455	385	905	453	213	978	4756	818

N/D - no data

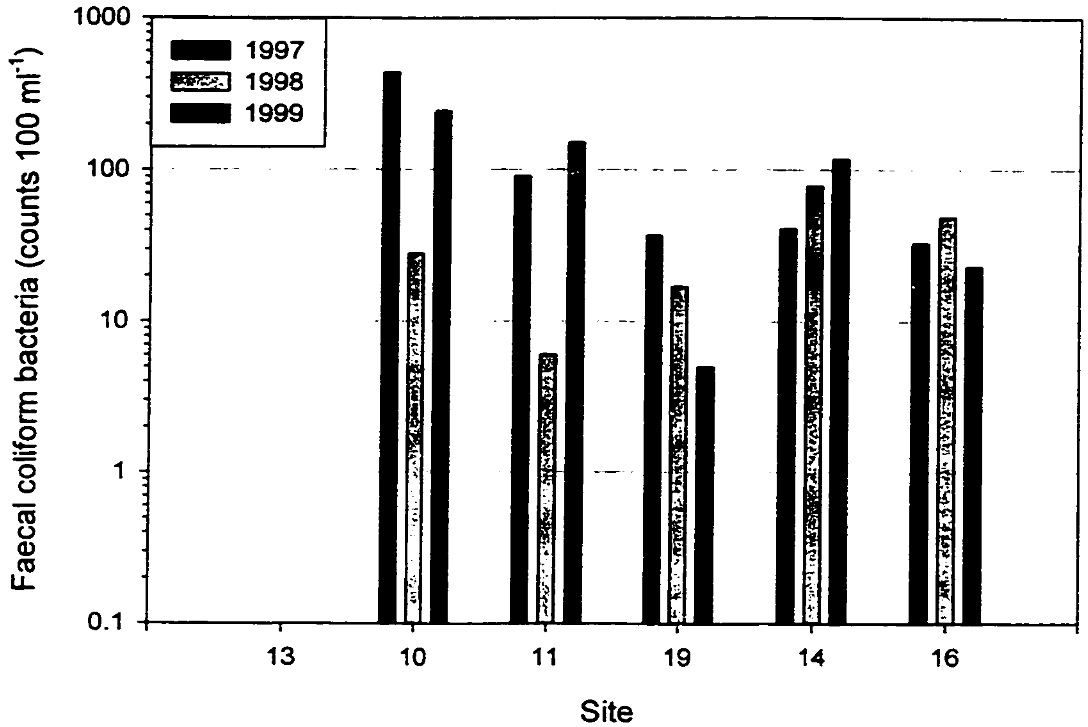

Figure 6.1. Faecal coliform bacteria flow-weighted mean concentrations during spring runoff.

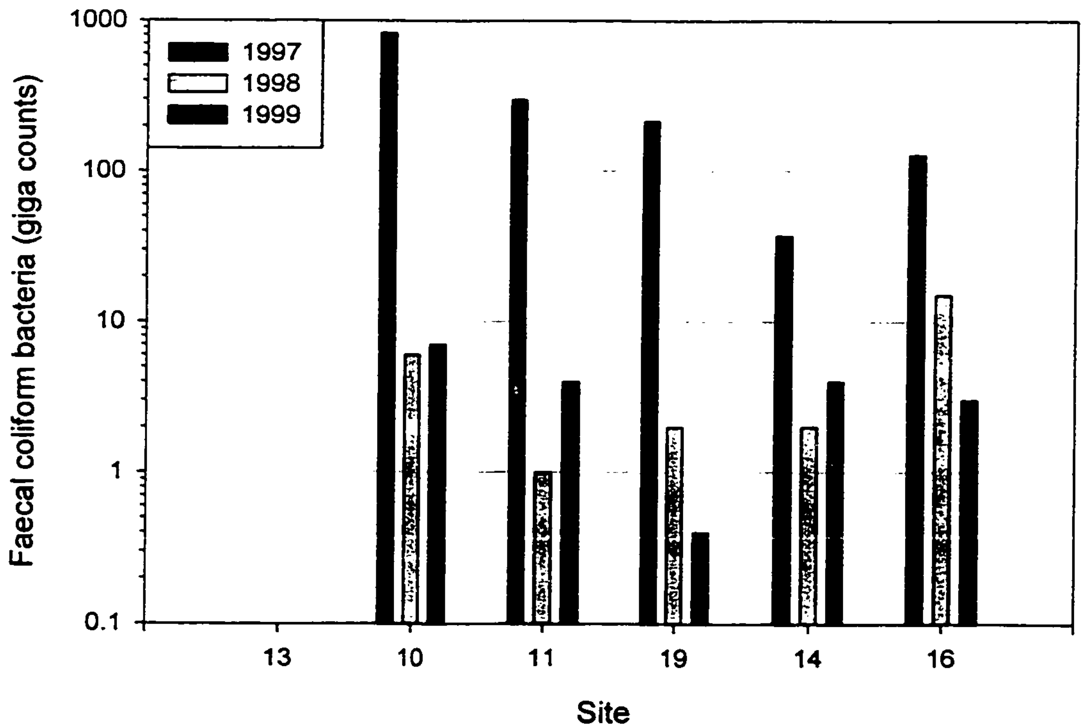

Figure 6.2. Faecal coliform bacteria mass load during spring runoff.

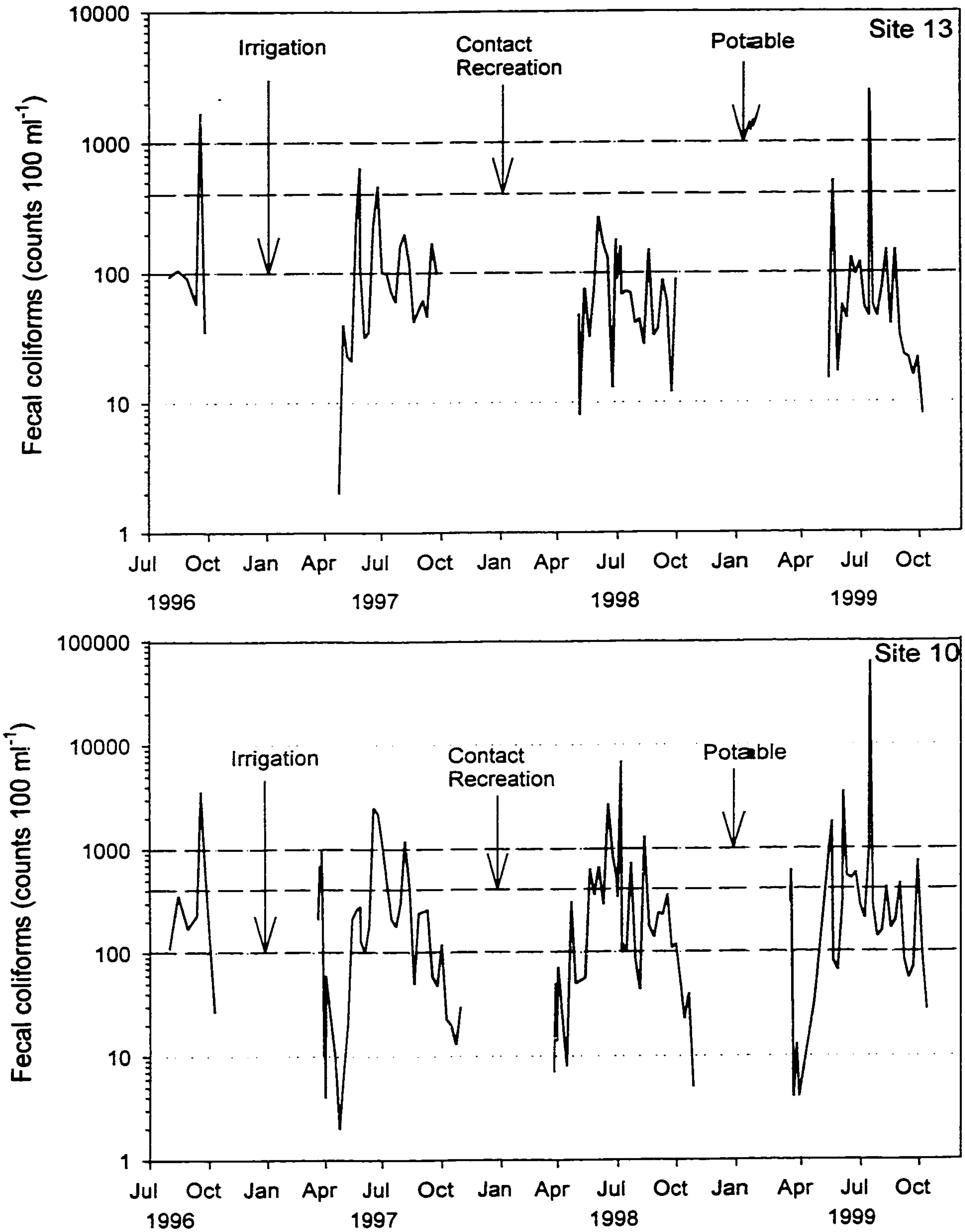

Figure 6.3. Faecal coliform bacteria counts for the inflow and outflow sites Sub-basin 10 from 1996 to 1999.

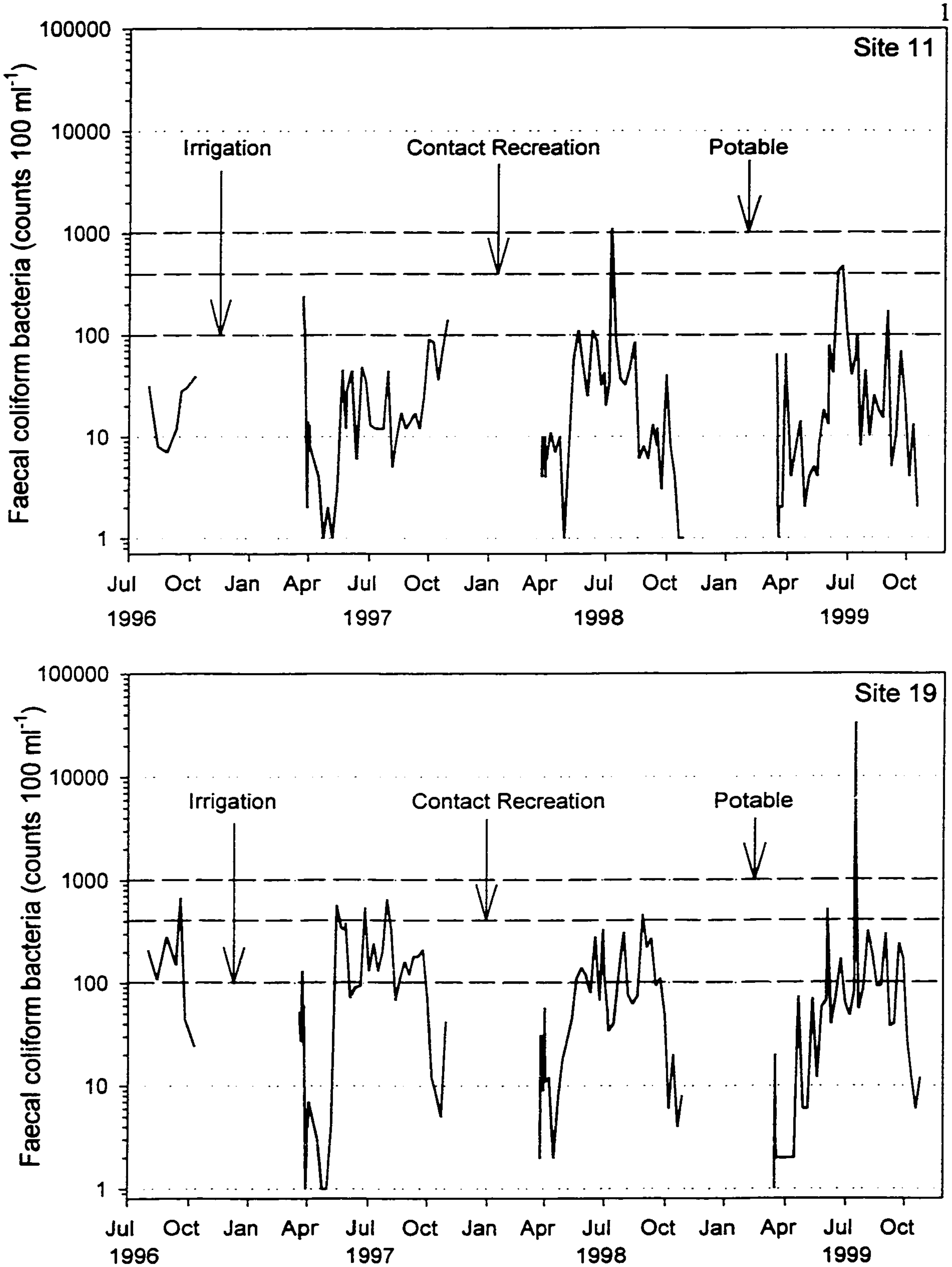

Figure 6.4. Faecal coliform bacteria counts for the inflow and outflow sites Sub-basin 19 from 1996 to 1999.

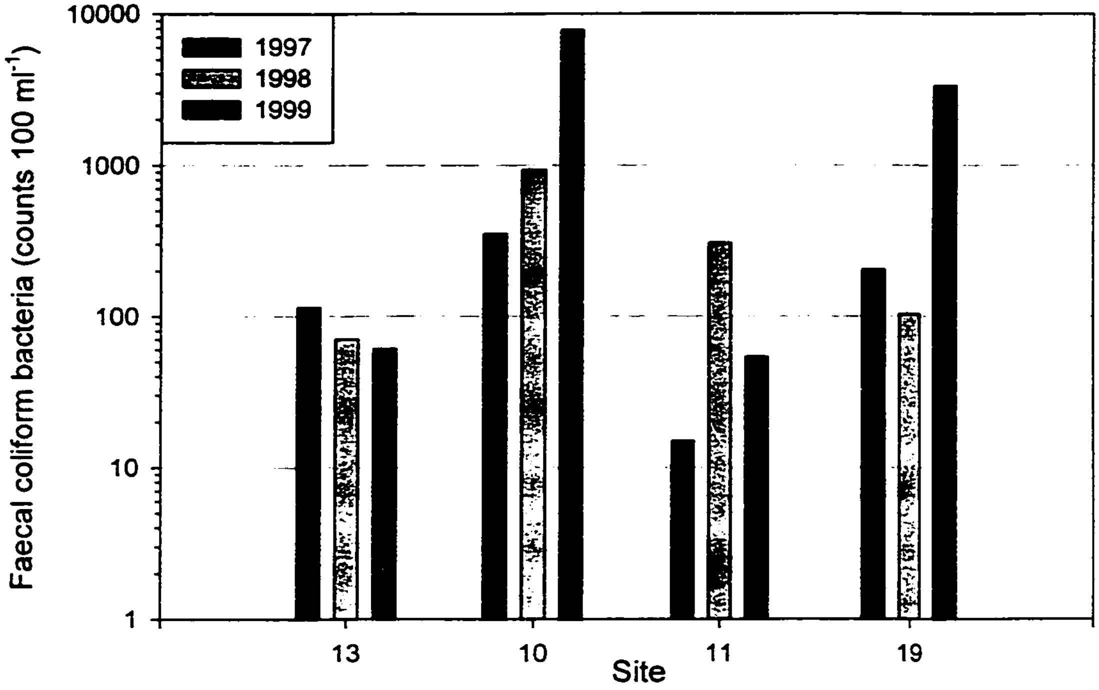

Figure 6.5. Faecal coliform bacteria flow-weighted mean counts during post-spring runoff at the Crowfoot Creek sites.

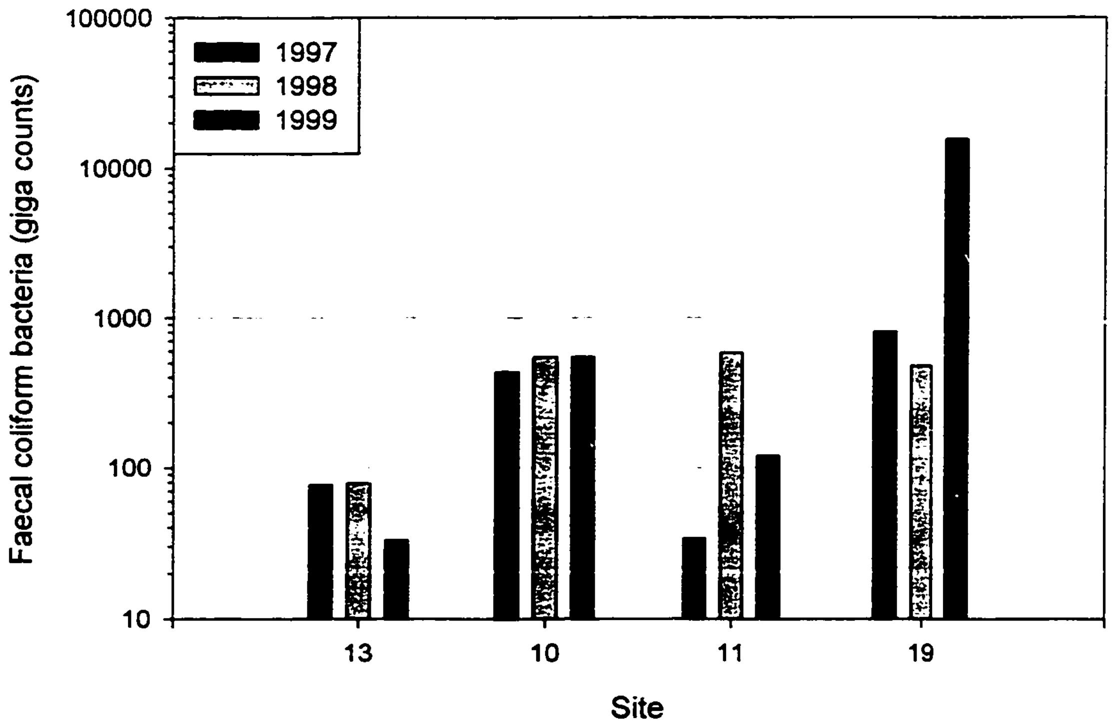

Figure 6.6. Faecal coliform bacteria mass load during post-spring runoff at the Crowfoot Creek sites.

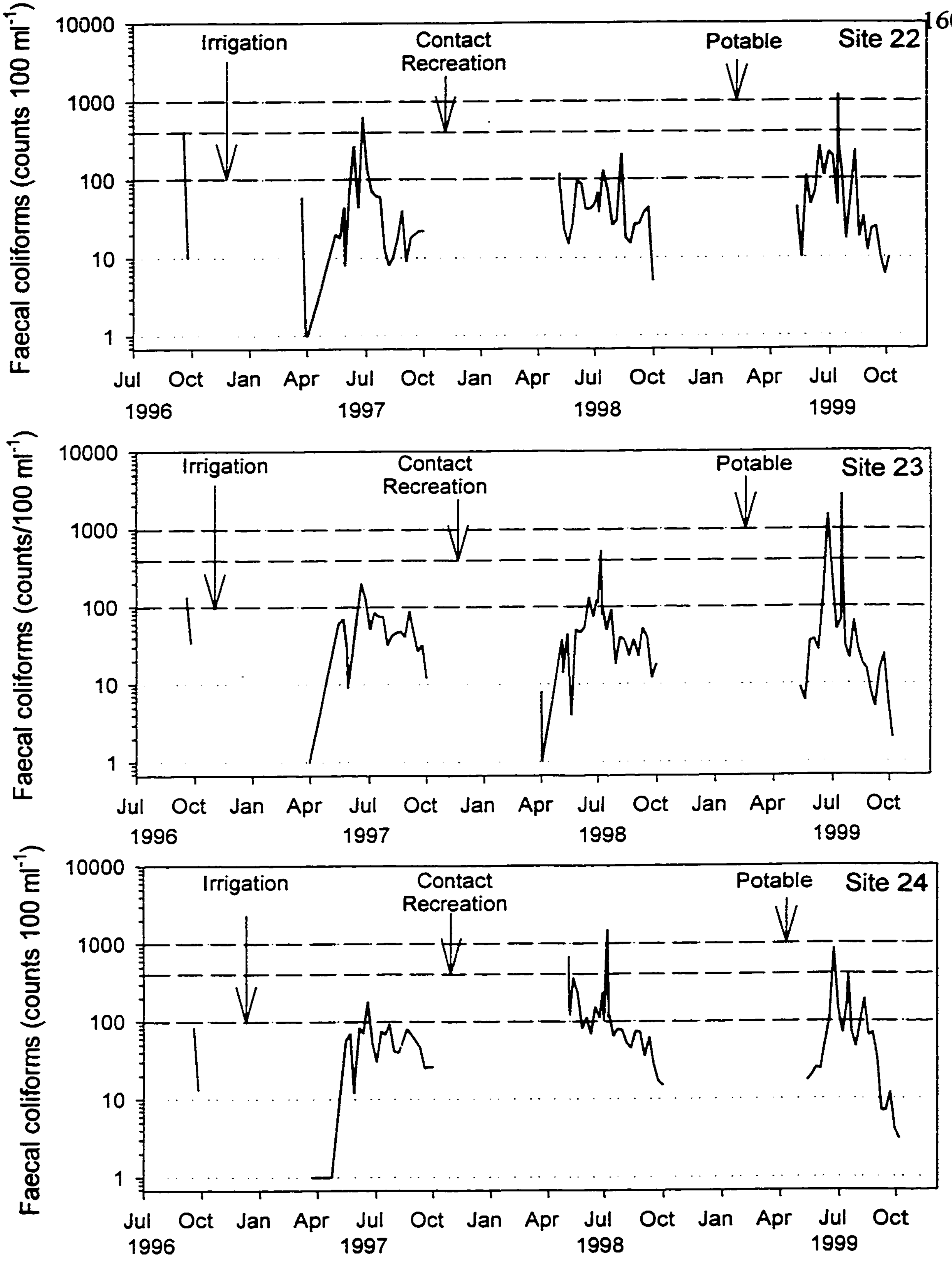

Figure 6.7. Faecal coliform bacteria counts at the inflow sites of the WID return flow sub-basins from 1996 to 1999.

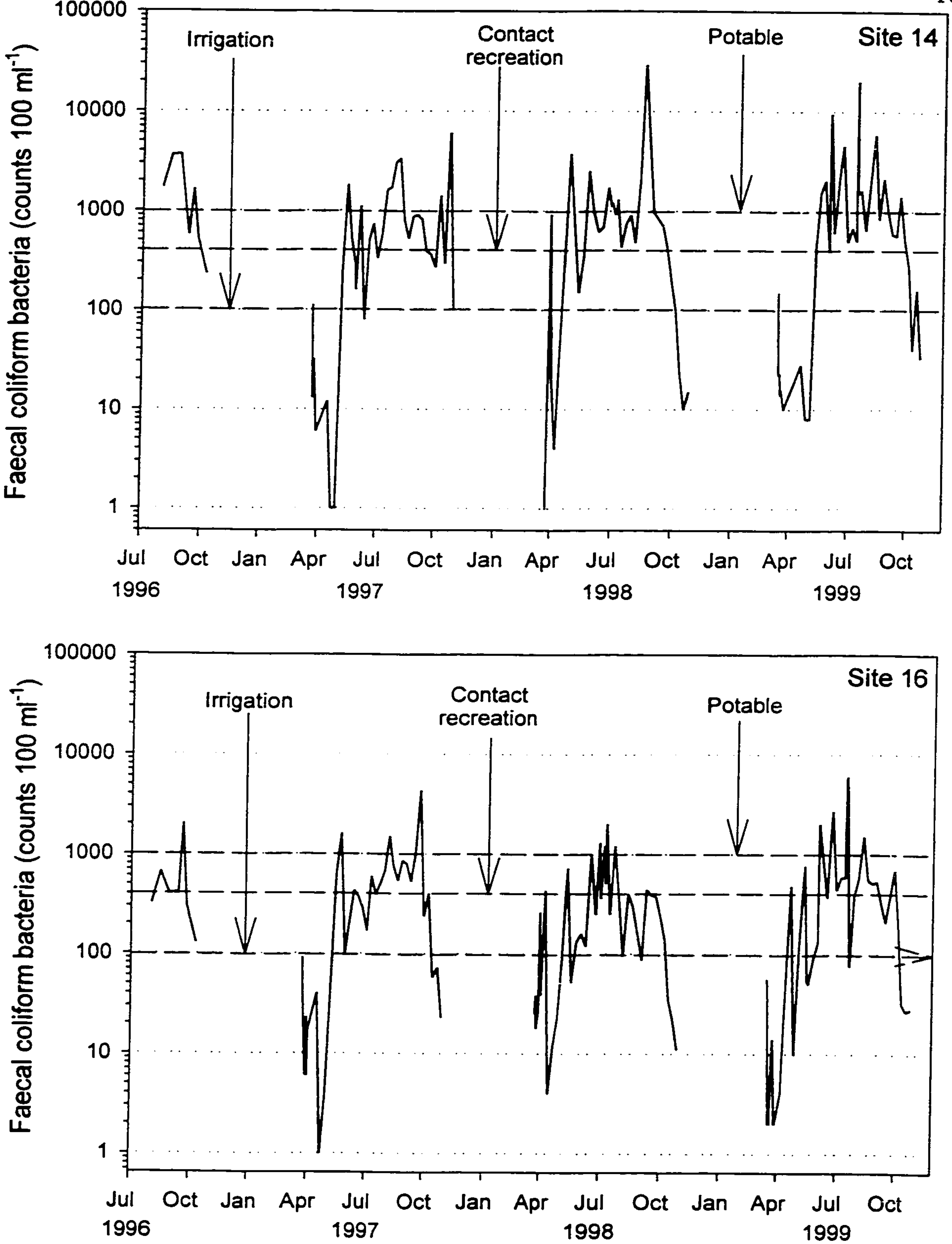

Figure 6.8. Faecal coliform bacteria counts at the outflow sites of the WID return flow sub-basins from 1996 to 1999.

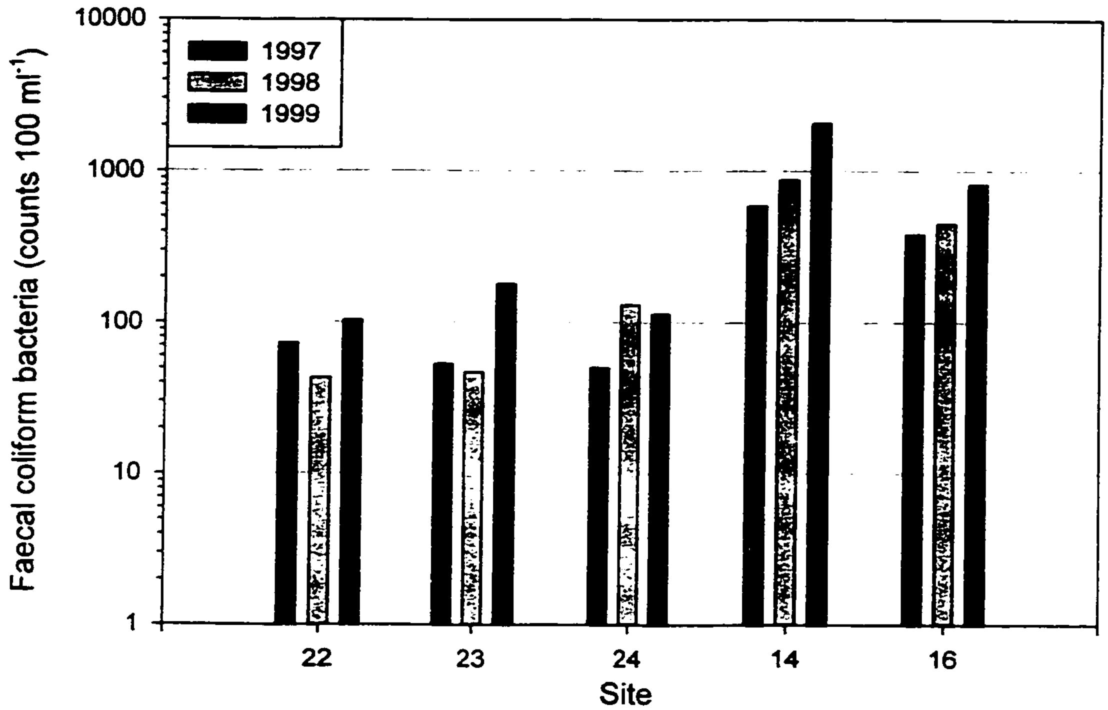

Fig. 6.9. Faecal coliform bacteria flow-weighted mean concentration during the post-spring runoff period at the WID sub-basin sites.

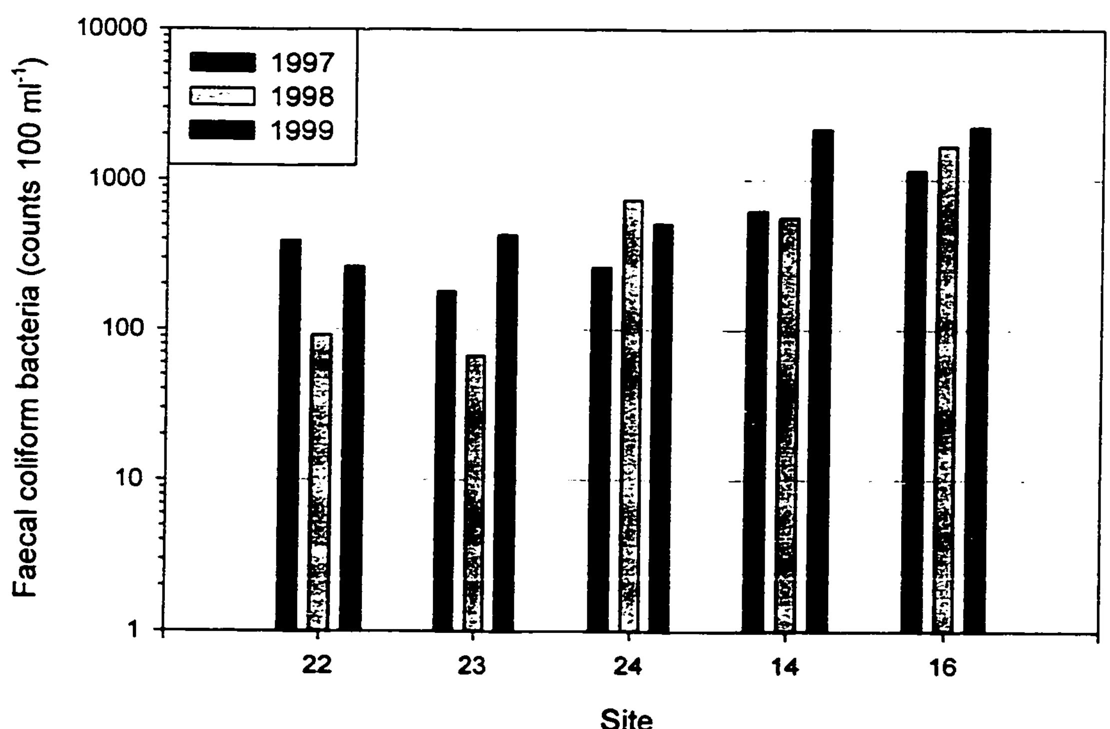

Fig 6.10. Faecal coliform bacteria mass load during the post-spring runoff period at the WID return flow sub-basins.

7. SYNTHESIS

Increasing global concerns regarding the quality of water have resulted in a heightened scrutiny of industrial and agricultural practices that may affect it. In Alberta, several studies have indicated that agriculture may be adversely impacting water quality. The Crowfoot Creek watershed, which drains into the Bow River east of Calgary, has been identified as a source of high levels of nutrients and coliform bacteria to the Bow River. Four sub-basins of the Crowfoot Creek watershed were studied to assess the level of impact that agriculture may be having on the quality of water leaving the watershed.

In most prairie watersheds, the majority of flow occurs during the spring snowmelt period, while flows are low or nonexistent during the remainder of the year. In irrigated watersheds, however, flow patterns differ from this norm as irrigation return flows maintain a level of flow that is greater than otherwise would be expected from the period after spring runoff ceases until the end of the irrigation season. This additional volume of water can affect both the concentration of various parameters in the watercourse as well as the mass load of the parameter being transported.

Several factors affect the volume of water passing through the sub-basins within the Crowfoot Creek watershed. Snowcover varied widely over the three springs during which data were collected. Large amounts of snowcover, as in 1997, provided a larger percentage of the water passing through the individual sub-basins during the entire year. This large runoff was not typical and it was more likely that the volume of water that passed out of the watershed in the spring made up a smaller percentage of the total volume. Runoff in 1998 was likely more typical of springmelt periods in the Crowfoot Creek watershed, however, flows in that year may also have been below normal. Flows in 1999 were likely well below normal.

Irrigation return flows were more likely to provide the majority of the water passing through the watershed over the course of the year and did so in two of three years. Post-spring runoff flows generally comprised more than 75% of the total monitored flow in the three complete years of monitoring. The magnitude of flows during the PSRO periods was affected by the irrigation demand on the irrigation system by the users. In years of high demand, more water was diverted through the WID infrastructure, resulting in greater volumes of water during the PSRO period. In years of low irrigation demand, such as 1999, less water was diverted for use.

Large volumes of water entered Crowfoot Creek and the WID sub-basins during the PSRO period via surface runoff from rainfall. Prediction of the timing and magnitude of these flows was complicated by variability of rainfall across the sub-basins, antecedent moisture content of the soil, type of landcover and the amount of irrigation prior to the event.

Nitrogen and phosphorus concentrations in both Crowfoot Creek and WID return flow sub-basins were affected by natural as well as agricultural sources. During spring runoff in these sub-basins, there was a base level of both N and P in the watercourse which may have been affected by inputs from the surrounding agricultural land during the previous season, as well as by natural processes. In years that surface runoff occurred in the spring, concentrations of nutrients in the watercourse were impacted by the additional nutrients carried in the surface runoff. Comparison of years of high and low spring runoff indicated that nutrients were being transported out of the sub-basins regardless of the overall volume of runoff.

The concentrations of TN and TP in the Crowfoot Creek and WID sub-basins in the spring almost always exceeded the AWQG guidelines of 1.0 and 0.05 mg L^{-1}, respectively. The TP guidelines were exceeded by significant amounts, with median values ten-fold higher in all of the sub-basins. Median TN values exceeded the guideline by as much as two-fold, with only the WID sub-basins meeting or slightly exceeding the guideline.

Nutrient sources during the spring were likely decaying plant material that increased in concentration in numerous sloughs and wetlands over the winter months. This was especially true in the Crowfoot Creek sub-basins. In the WID sub-basins, the concentrations of the nutrients were lower as there are fewer wetlands in these sub-basins. The high nutrient concentrations during spring runoff generally occurred when runoff water flushed nutrients from within the sub-basins. Faecal material from cattle, that may have been transported into the watercourse with surface runoff or deposited directly by livestock during the previous year, was an additional source of N and P in all of the sub-basins.

The high runoff volumes in spring 1997 resulted in large amounts of nutrients being introduced into Crowfoot Creek and WID watercourses through increased transport of N and P from the surrounding land. However, the resulting concentrations were lower, as the increased runoff volume diluted the parameters. In the remaining years, the reduced flow volumes resulted in increased concentrations as nutrient concentrations were not diluted to the same magnitude. Trends in the mass loads were an inverse of those for concentrations, with high volumes resulting in greater loads than the low volumes.

During the PSRO period, input of WID return flow water had a dilution effect on water in the sub-basins similar to the snowmelt runoff in the spring. Concentrations of N and P were high in the early part of the PSRO period, but decreased as the addition of the WID water diluted these levels throughout the period. Regardless of the flow, concentrations of N and P increased between inflow sites, made up of WID water, and the exit sites in three of the sub-basins during PSRO. The remaining sub-basin showed a decrease in concentration between inflow and outflow. Concentrations of N and P entering this sub-basin were very high and inputs from within this sub-basin may not have been great enough to maintain this concentration.

The overall patterns within the various sub-basins indicated that, as in the spring, there was a base level of nutrients present. Water passing through the sub-basins generally increased in concentration. In Sub-basin 19, which is farther downstream, composition of the water varied due to the reduced number of wetlands and other sources of nutrients within the stream. This resulted in a decrease in concentration as the WID water flushed out the accumulated nutrients. Some of the increase in concentration may also be due to agricultural inputs such as fertilizer and crop residue that have been transported to the watercourse at a previous time.

Levels of TP exceeded the AWQG almost all of the time in all of the sub-basins, with median values of approximately twice guideline values. TN concentrations exceeded the AWQG early in the period, but decreased to nearer the guideline value for much of the PSRO period. This pattern changed during rainfall events. Additional water entered the creek via surface runoff during these periods, carrying N and P in both particulate and dissolved forms. The timing and intensity of the events affected the magnitude of the impact on the receiving stream. Events occurring earlier in the PSRO period appeared to have a greater impact on N and P concentrations than those later in the season. The antecedent moisture conditions of the soil may also have affected the amount of runoff produced, resulting in greater amounts of surface runoff.

Concentrations of N and P detected during event periods within the PSRO period were higher than values for the overall PSRO period and exceeded AWQG for these parameters during most of the events. Mass loads transported out of the sub-basins during events tended to be a significant portion of the load transported during the overall PSRO period. Hence, a large impact on water quality was associated with rainfall runoff events. As the Crowfoot Creek watershed is dominated by agricultural land use, the adverse impact on water quality appeared to be related to agricultural practices.

Temporally, nutrient water quality in the Crowfoot Creek watershed was poorest from spring to early summer. Concentrations tended to decrease during the year with the addition of WID water, reaching their lowest values late in the year. Concentrations increased with the cessation of spills of WID water in the fall. Water withdrawn for use early in the year likely contained concentrations of TN and TP in excess of the relevant AWQG.

Nutrient concentrations in the WID sub-basins were lower compared to those in the Crowfoot Creek sub-basins. In the spring, wetlands along the Crowfoot Creek likely provided additional nutrients, resulting in elevated nutrient concentrations compared to WID sub-basins. These nutrients were also present during the PSRO period and contributed additionally during rainfall events when high flows resulted in a flushing out of wetland areas. While wetlands can be useful in the removal of nutrients from the water, they can also be a source of nutrients. The processes controlling nutrient form and removal within wetlands are complex and can vary greatly with changing temporal conditions.

Generally, WID return flows would be expected to improve the quality of water in Crowfoot Creek. Water flowing into the WID sub-basins was of good quality, with the AWQG for TP and TN met or only slightly exceeded in most cases. Water quality deteriorated within the WID sub-basins due to inputs from within the sub-basin. The ease with which runoff entered the watercourse within the sub-basin significantly affected this deterioration. This was not the case with all irrigation return flow channels due to differences in construction. Constructed channels have banks that impede the movement of surface runoff into them, resulting in less impact from the surrounding land. WID water returned to Crowfoot Creek from this type of channel may have concentrations of TP and TN more similar to concentrations in the water entering the sub-basin. Addition of better quality water had a positive effect on the receiving stream.

In the spring, loads contributed by the WID system were small, as flows were low, while in the PSRO period, loads were higher, as were the flows. Overall, mass loads decreased within the WID sub-basins as water and nutrients were removed for irrigation.

When examining water quality, it is important to decide what factor will be used for determining impact: either the concentration of the parameter or the load of material that is passed out of the watershed. While the mass load indicates how much N and P is being transported out of the watershed on a weight basis, concentration is the value used to assess the risk associated with the various uses of the water.

Spill water from the WID sub-basins into Crowfoot Creek generally had low concentrations of nutrients. While the added volume of water provided by WID return flows lowered nutrient concentrations through dilution, the resulting mass load of nutrient exported from the watershed was greater, perhaps resulting in water quality problems in downstream watersheds. A reduced volume of WID return flow water resulted in higher concentrations of nutrients which increased the risk for users within the watershed.

The impact of landcover on nutrients in the studied sub-basins was variable. Examination of TP indicated the dominant form was dissolved P, and the percentage of total P varied with the source water as well as the time of year and amount of runoff that entered the creek.

Inflow water from the WID infrastructure, with its relatively high velocity, tended to carry more particulate P. Once this water entered the sub-basins and the velocity decreased, the particulate matter settled out, leaving the dissolved forms as dominant. Sub-basins with greater percentages of grassland adjacent to the watercourse appeared to have greater percentages of TDP, while sub-basins that were cropped closer to the edge of the watercourse had lower percentages of TDP.

During events, landcover appeared to affect the percentage of TDP through its ability to protect the surface from raindrop impact. Under reduced raindrop impact, the soil maintained its aggregation, reducing the amount of soil transported to the watercourse by surface runoff. This process did not, however, have the same effect on the dissolved component of P as this fraction was still transported in surface runoff water.

Altering agricultural practices may help in preventing nutrient loss to surface water. Reduced tillage and increased residue cover are two methods that help to reduce the impact of rainfall on the soil. Increased residue also helps to improve the infiltration rate of the soil, reducing the amount of surface runoff. By keeping surface runoff to a minimum, the amount of nutrients removed in this manner will also be reduced. The movement of nutrients into the soil can, however, pose a danger to shallow groundwater. Grass buffer strips near the watercourse will also aid in the reduction of particulate transport, but will not have as great an effect on reducing the movement of the dissolved nutrient fraction. Another method of reducing the impacts of nutrients on surface water is more intensive nutrient input management, specifically of chemical fertilizers. Fertilizers are most often applied in the spring during seeding. The soil is vulnerable to erosion during this time and the additional nutrients in the soil increase the potential for them to be transported to watercourses. Application of fertilizers to the levels required by the crop based on soil tests would be one method of controlling excess nutrients in the soil.

Microbiological impacts on water quality in the study sub-basins appeared to be related to the presence or absence of livestock within the sub-basins. Faecal coliform levels were very low during the spring when livestock were generally not as prevalent within the sub-basins. As with nutrients, there was a base level of FC within the sub-basins due to both indigenous wildlife, such as beavers and birds, and introduced sources such as livestock. Cattle that were present prior to spring runoff in some of the sub-basins did not seem to contribute to a consistently high concentration of FC. Peak values in the spring may have been due to runoff flushing FC from manure into the stream.

During the PSRO period, faecal coliform counts increased early in the period in response to the increased presence of cattle in the sub-basins. Greater wildlife activity also resulted in increased impacts on faecal coliform counts. Counts remained high until fall, when cattle were removed from the watershed. The magnitude of the FC counts was affected by the number of cattle and their proximity to water within the sub-basins. Three of the sub-basins had a resident cattle population with open access to the water during the PSRO period and generally had greater concentrations of FC than the other sub-basin. Counts exceeded the CWQG for irrigation, contact recreation or potable water uses on many occasions.

In contrast to nutrient impacts on the water quality, FC counts did not vary as directly with surface-runoff-causing events as did nutrient concentrations. As long as animals had access to the watercourse, the potential for contamination existed. When bacteria or nutrients are on the land surface, they usually require a hydrologic process, such as surface runoff, to move them into the water. Such occurrences were readily observed. When cattle moved into a watercourse, they did not do so at any specific time or under any specific conditions. Therefore, increases in count levels occurred even during non-runoff periods. Rainfall events did, however, result in increases in FC counts, indicating that the faecal materials deposited in the areas affected by surface runoff contributed to water quality degradation. The degree of degradation was high enough to result in exceedence of guidelines for irrigation, contact recreation and potable water.

The response of FC to the presence of cattle in the sub-basins indicated that cattle contributed to degradation of water quality in the sub-basins. Current grazing management practices utilized available water and feed in the sub-basins. The grazing of cattle in riparian areas and along the irrigation infrastructure has been conducted for many years as these areas have an abundant supply of food and water. Proximity of these areas to water, and the ability of FC to survive for extended periods of time outside their hosts, has resulted in this practice becoming a source of surface water contamination. Keeping the cattle a distance from the water through offsite watering, for example, may reduce impacts. Grassland adjacent to the watercourses appeared to reduce the amount of particulate nutrients reaching the water, however, livestock often grazed there, resulting in high concentrations of faecal coliform bacteria.

The Bow River Water Quality Task Force identified nutrients and coliform bacteria, among other parameters, as having an impact on the Bow River within the reach where the Crowfoot Creek watershed discharges. A study of nutrient and coliform bacteria within four sub-basins of the Crowfoot Creek watershed carried out over four years indicated that levels of TP, TN and faecal coliform bacteria often exceeded AWQG and CWQG for various agricultural, recreational and environmental uses.

Concentrations of these constituents were influenced by the hydrology of the watershed, with spring runoff and rainfall events resulting in increases in concentration and the removal of large mass loads of the constituents. Non-event flows had high levels of TN in the spring and early portion of the summer, while TP concentrations exceeded guidelines throughout the monitoring period. Faecal coliform counts increased with the increased presence of cattle in the sub-basins. Concentrations of all parameters studied increased within the sub-basins. While some sources of these constituents are natural, such as plant decay in wetlands, the dominant agricultural nature of the Crowfoot Creek watershed suggests that agricultural practices within the watershed contribute nutrients and faecal coliform bacteria, resulting in degradation of water quality in the Crowfoot Creek watershed. Landcover effects were difficult to ascertain. Grasslands adjacent to the watercourse appeared to be better at reducing the amount of particulate P in streams when contrasted with areas of annual crop or summerfallow. However, grazed areas were also likely sources of faecal coliform bacteria.

The impact of WID return flows on water quality was generally positive. The return flows were of better overall quality than flows in Crowfoot Creek, so dilution of the water resulted in lower concentrations. Mass loads within the WID sub-basins decreased as water was removed and applied to the land by irrigation. The mass of the constituents exported was greater in the WID sub-basins due to the increased flow volume provided by irrigation return flows into Crowfoot Creek.

Areas for further study include:

- examination of the hydrological patterns within an irrigated prairie watershed and the effects on water quality constituent concentrations;
- the degree to which wetlands in watersheds act as sources/sinks of nutrients;
- delineation of contributing areas within watersheds to better understand contributions from specific landcovers on water quality and how best to manage them and;
- the degree of impact of grazing cattle on the microbiological quality of water.