CHARACTERIZATION OF THE URBAN RUNOFF QUALITY FROM THE CITY OF SASKATOON TO THE SOUTH SASKATCHEWAN RIVER

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By

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ABSTRACT

A major upgrade to the wastewater treatment plant in Saskatoon, Canada significantly improved the final effluent quality. Consequently, the relative impact of the city's urban runoff on the receiving stream, the South Saskatchewan River, has increased. Moreover, at the inception of the study, pending amendments to provincial legislation governing urban runoff were such that urban runoff would no longer be automatically exempt from regulation. In response to this impending change, which has since been made, Saskatchewan Environment initiated a study to examine the water quality of the urban runoff in Saskatoon, because little had been done to date involving the water quality of urban runoff in Saskatchewan.

The field program was conducted in 2001 and 2002 to collect representative urban runoff water quality and flow rate data from four different land uses: newer residential, older residential, commercial, and industrial. Three characterizations of the water quality were developed on the basis of the data collected: Site Mean Concentration (SMC), multiple variable regression models, and the unit load. The SMC results indicate that the average water quality parameter concentrations in Saskatoon are greater than those from NURP, the updated U.S. nationwide urban runoff database, and from Vancouver, Canada, but are similar to those from Wisconsin. The regression analyses indicate that the rainfall depth is the most frequently significant parameter in the prediction of event loads. The unit load analyses indicate that the commercial catchment produces the most pollutant load per unit area. Comparison of the methods indicates that the SMC can be used to estimate longer term urban runoff loads, in lieu of the more complex regression method.

Heavy metals, pesticides, and fecal coliforms were detected in the urban runoff at concentrations that exceed guideline values. Further investigation is recommended.

In comparison to the loads discharged by local point sources, urban runoff contributes larger total suspended solids (TSS) and total Kjeldahl nitrogen (TKN) loads to the South Saskatchewan River. The load of COD to the river is comparable to that of the Saskatoon Wastewater Treatment Plant (WWTP). The total phosphorus load contributed

by urban runoff is slightly smaller than that of the WWTP. Considering the relative load of TSS from urban runoff to the WWTP and the potential for other, more toxic pollutants to adsorb to the TSS, sediment controls should be implemented at all levels of development. Further examination of urban runoff with specific emphasis on spring and winter runoff is recommended.

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LIST OF SYMBOLS/TERMINOLOGY

5min	5 minute average, maximum intensity rainfall
ante	Antecedent dry period
AO	Aesthetic objective
aveint	Average rainfall intensity
BOD_5	Five day biochemical oxygen demand
CCA	Chromated copper arsenate
CFU	Colony forming unit
C_i	Concentration during interval i
Cl^-	Chloride
COD	Chemical oxygen demand
COS	City of Saskatoon
CWA	U.S. Clean Water Act
depth	Depth of rainfall
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DOF	Degrees of freedom
dur	Duration of rainfall
EMC	Event mean concentration
EMPA	Environmental Management and Protection Act (Saskatchewan)
EPA	U.S. Environmental Protection Agency
FC	Fecal coliform
IETD	Inter-event time definition
IMAC	Interim maximum acceptable concentration
L	Load discharged
MAC	Maximum acceptable concentration
max	Maximum instantaneous intensity
MCL	Maximum contaminant level
MCLG	Maximum contaminant level goal
MPE	Median percentage calibration error
M_T	Total mass discharged

NO_2-N	Nitrite as nitrogen
NO_3-N	Nitrate as nitrogen
NPDES	National Pollutant Discharge Elimination System (U.S. EPA Program)
NURP	Nationwide Urban Runoff Program
NWRI	National Water Research Institute
OP	Ortho phosphorous
PAH	Polycyclic aromatic hydrocarbon
PCB	Polychlorinated bi-phenyl
pesticide	Used in the general sense to refer to collectively refer to herbicides, insecticides, fungcides, etc.
SE	Saskatchewan Environment
SMC	Site mean concentration
SWQO	Surface water quality objectives
TC	Total coliform
TDS	Total dissolved solids
TKN	Total Kjeldahl nitrogen
TN	Total nitrogen
TP	Total phosphorous
TSS	Total suspended solids
U.S.	United States of America
USGS	United Stated Geological Survey
V_i	Volume discharged during interval i
V_T	Total volume discharged during event
WWTP	Wastewater Treatment Plant

All mass per volume concentrations are reported as mg/L unless otherwise noted.

CHAPTER 1 INTRODUCTION

1.1 BACKGROUND

Historically, rainfall-runoff pollution was essentially ignored, as it was frequently viewed as clean rain (Makepeace et al., 1995). Rainfall-runoff was mainly a problem of flooding and public nuisance (i.e. quantity of runoff). As such, the focus of rainfall-runoff management has primarily been the disposal of runoff and the structures required to economically and safely accomplish the task of disposal (Makepeace et al., 1995). However, rain falling upon a catchment accumulates pollutants from the air, roadways, and sewers and is transformed into a municipal wastewater (Chambers et al., 1997). Urban runoff can contain suspended solids, nutrients (principally nitrogen and phosphorous), microbes (including pathogens), organic compounds (petroleum products and BOD causing compounds) and toxic substances (heavy metals and pesticides) (WEF/ASCE, 1998; Alberta Environment Protection, 1999a; Makepeace et al., 1995; Marsalek and Schroeter, 1988). Urban runoff is a source of pollutant input to receiving streams.

Some of the earliest co-ordinated studies of urban runoff were started in 1972 when the United States (U.S.) Congress passed the Clean Water Act (CWA). The CWA created programs that located and examined discharges to surface water that had poor water quality. Amendments to CWA in 1987 introduced further regulations for controlling the quality of urban runoff discharges under a program called the National Pollutant Discharge Elimination System (NPDES). NPDES is a system of permitting discharges from all sources that exceed certain minimum flow quantity and/or quality requirements, including urban runoff. After more than 30 years of work on urban runoff, the U.S. Environmental Protection Agency (EPA) still considers urban runoff to be a “major stressor” on the waterways of the U.S. (Mays, 2001).

Urban runoff has been identified by Canadian agencies as a source of pollutant input to receiving streams. Coincident with the inception of this project in 2001, the

National Water Research Institute (NWRI) of Environment Canada published a report entitled "Threats to Sources of Drinking Water and Aquatic Ecosystem Health in Canada", which identified urban runoff as a threat (Environment Canada, 2001). The report recognizes the need to further examine the effect of urban runoff on Canadian waterways, including the need to collect new data in order to acquire a better understanding of urban runoff and its associated processes (i.e. pollutant build-up and washoff). Much of the data currently used was collected 20 years ago, subsequent to which chemical detection capabilities have improved, new products have been introduced, and attitudes towards the environment have changed. The report emphasizes the need for drinking water source protection, with the improvement of urban runoff water quality being one such means.

The Province of Alberta has begun to mandate total loading limits from some urban areas (Deong, 2001). This mandate means that municipal discharge permits now encompass all municipal discharges within the corporate limits, including urban runoff. It places the onus upon the municipality to determine the total load that is discharged to the receiving stream. The Province of Ontario, in addition to regulations, has produced guides for designers and municipalities to aid in the management of urban runoff from both a quantity and a quality perspective (OMOE, 2001).

Urban runoff is a non-point pollution source; it is generated from the general landscape, and not one specific location. In contrast, point source pollution is pollution discharged from a specific area and is generally heavily regulated while required to produce relatively clean effluent prior to discharge to the environment (Makepeace et al., 1995). Examples of point source pollutant sources include wastewater treatment facilities, manufacturing facilities, and chemical production facilities.

Historically, the impact of treated municipal effluent on receiving streams has been greatly reduced as sewage treatment technologies have advanced and new infrastructure is constructed. An example is the Saskatoon wastewater treatment plant (WWTP). Table 1.1 presents average effluent concentrations from pre- and post-upgrade. In the 1980's pre-upgrade, the WWTP utilized primary treatment only, producing a final effluent with average water quality parameters. The treatment process was upgraded to a coupled biological nutrient removal - activated sludge process in 1996 (i.e. advanced tertiary treatment). The reduction in effluent concentrations is approximately 10-fold across each of the three parameters, which is a significant improvement in effluent quality.

Table 1.1 – COS WWTP effluent average annual concentrations.

Parameter	Pre-upgrade *	Post-upgrade **
BOD_5 (mg/L as O_2)	83	8.5
TSS (mg/L)	74	5.9
TP (mg/L)	5.2	0.37

* average between 1985 to 1989 (Chambers et al., 1997)

** 2001 data (after COS, 2003)

The impact of urban runoff from the COS on the water quality of the South Saskatchewan River has not been well documented. Further, little specific information is known about pollutant input to Saskatchewan waterways from urban runoff. Some of the first work documenting the characteristics of urban runoff was conducted by COS in 1984 and 1985 (Munch and Keller, 1985). This work focused on acquiring a broad understanding of the composition of urban runoff discharged to the South Saskatchewan River and the accumulation of pollutants in backwater areas within the river channel. The study primarily collected grab samples during dry weather flows. One rainfall event was examined, in which a time series of samples was collected. The study began, in the first season, by examining the majority of outfalls within the city and, in the second season, focused on locations that had unsatisfactory water quality results. Mass load estimates for use in the determination of the impact upon the South Saskatchewan River based on this data are not possible, as an intensive rainfall-runoff event sampling program, which was not done, is required to provide sufficient data to permit an estimate of the mass load (Ellis and Hvitved-Jacobsen, 1996).

In the period since the work of Munch and Keller (1985), COS has upgraded the WWTP. Intuitively, with the WWTP upgrade yielding a significant improvement in effluent quality, the impact of the municipal sewage effluent has been reduced and the relative impact of urban runoff has likely increased. It still, however, remains essentially as an unknown. Currently, COS maintains an urban runoff monitoring program under which dry weather samples are collected from key outfalls and the remaining outfalls are visually inspected. This program is mainly intended to detect cross-connections to the storm sewers from the sanitary sewer system.

Based on the lack of urban runoff water quality data and increased focus on urban runoff water quality, Saskatchewan Environment (SE) initiated this study to provide a first look at urban runoff in the Province of Saskatchewan based on Saskatoon. The resulting project, which forms the basis of this thesis, is a joint initiative of the University of Saskatchewan Department of Civil and Geological Engineering (U of S), COS, SE, and the Meewasin Valley Authority (MVA).

1.2 PROJECT OBJECTIVES

This project is intended to provide preliminary water quality information, which inherently includes aspects of water quantity, and to characterize the urban runoff discharged from the City of Saskatoon to the South Saskatchewan River. The following sub-objectives further define this purpose:

- Undertake a review of literature germane to the work;
- Conduct an urban runoff water quality sampling and analysis program;
- Characterize the urban runoff based on common accepted practice in terms of the site mean concentration (SMC), multiple variable regression, and the unit load (mass discharged per unit area);
- Examine the urban runoff characterizations for water quality variations attributable to land use type and rainfall parameters;
- Estimate the total annual loading of urban runoff pollutants to the South Saskatchewan River; and
- Compare the estimated total annual urban runoff loading to the discharges of point sources such as the COS WWTP and some local industries.

1.3 SCOPE OF THE RESEARCH

The project was specifically focussed on gathering data that are representative of specific land use types. As such, outfalls or catchments that have known environmental concerns were deliberately avoided in the selection of the monitoring locations. Each of the four locations represents a specific and different primary land use type: newer construction residential, older construction residential, light industrial, and commercial. The study did not make use of any computer modelling programs. Specific assessment of the impact of the discharges upon the South Saskatchewan River is not made. The water quality analysis is focussed primarily on basic water quality parameters, with strategic samples being analyzed for additional pollutants, such as heavy metals and pesticides.

CHAPTER 2 LITERATURE REVIEW

2.1 GENERAL BACKGROUND OF URBAN RUNOFF

Runoff and specifically urban runoff has only been recognized as a source of pollutant input to receiving waters in the last 30 to 40 years, starting with the U.S. CWA of 1972 (WEF/ASCE, 1998). The CWA initially focussed on immediately cleaning up point sources and gaining an understanding of non-point sources. The non-point source examinations, however, did not fair well, because of a lack of experience examining a "dynamic system subject to the vagaries of urban hydrology" (WEF/ASCE, 1998). The difficulties resulted in a more concerted effort to understand urban runoff - the Nationwide Urban Runoff Program (NURP) - which was conducted in the early 1980's by the U.S. EPA. NURP examined 2300 station-storms at 81 urban sites in 28 metropolitan areas (Smullen et al., 1999). NURP used a flow weighted mean concentration, called the Event Mean Concentration (EMC), to characterize the water quality parameters for each rainfall-runoff event. Catchments were characterized using an average of the EMC's called the Site Mean Concentration (SMC). NURP also included a Priority Pollutant Study, which examined samples for a group of 129 CWA classified toxic chemicals (Adams and Papa, 2000). Two of the more significant conclusions of NURP are (Smullen et al., 1999):

- The characterizations, which were thought to be land use dependent, showed large variability between catchments. Based on the variability, the SMC's were determined to not be significantly different and thus it was concluded that land use is not a significant variable with respect to the SMC; and
- EMC's are log-normally distributed, which means that the geometric mean is the appropriate measure of central tendency for determining of the SMC.

In 1987, the CWA was amended to introduce the NPDES (WEF/ASCE, 1998). Urban runoff, other municipal discharges, and point sources that exceed stipulated flow quantity and quality requirements, are permitted under NPDES. Stormwater permitees are required to collect characterization and loading data, and to formulate a plan to manage the stormwater discharge to reduce the pollution discharged.

Environment Canada and the Ontario Ministry of the Environment jointly began to examine urban runoff as early as 1973 as part of the Research Program for the Abatement of Municipal Pollution under provisions of the Canada-Ontario Agreement on Great Lakes Water Quality (M.M. Dillon Ltd., 1979). Research projects were established that examined water quantity (Marsalek, 1977) and water quality (Mills, 1977). The water quantity projects examined the volume, flow rates, and time to peak of urban runoff in Burlington, Ontario and modelled the quantity data using the computer programs SWMM and STORM (Marsalek, 1977). The water quantity modelling efforts were met with good fit to the observed data. The water quality project examined the urban runoff from two catchments in the Borough of East York, Toronto, Ontario (Mills, 1977). The project was, however, fraught with instrumentation problems, leading to the abandonment of one of the two catchments. The project refocused on gathering water quality data from the remaining catchment and had reasonable results.

Through the NWRI, Environment Canada continues urban runoff research and has identified urban runoff as a threat to drinking water supplies and aquatic ecosystem health (Environment Canada, 2001). One of the significant issues identified by NWRI is the lack of current urban runoff data, as most of the large studies collecting data were completed in the 1980's. Since that time, analytical detection capabilities have improved, new products (chemical and other) have been developed and/or identified, and attitudes towards the environment have changed. To date, however, a nationwide program has not been instituted in Canada.

2.2 REGULATORY FRAMEWORK IN SASKATCHEWAN

Environmental regulation in Canada is not exclusive to either the Federal or Provincial governments (Irvine, 2002). In general, the governance of water quality falls under provincial jurisdiction, while federal jurisdiction generally focuses on specific items of nationwide interest, for example water quality as it relates to fisheries.

The current Saskatchewan Environmental Management and Protection Act (EMPA) 2002 came into force October 1, 2002 (Government of Saskatchewan, 2002a). EMPA includes changes that reflect suggestions of the North Battleford Water Inquiry, incorporate the former Ozone-Depleting Substances Control Act, as well as sections specifically dealing with contaminated sites and spills. The management of stormwater was also changed. Under the former EMPA, 1983-84, stormwater was automatically exempted from regulation (Saskatchewan Environment, 2002a). In the current EMPA, stormwater, which is defined as "rainwater or water resulting from the melting of snow or ice", is no longer automatically exempted from regulation (Government of Saskatchewan, 2002a) and must be permitted. The Guidelines for Sewage Works Design (Saskatchewan Environment, 2002b) contains a section pertaining to stormwater, which includes design suggestions and basic requirements for detention basins, outfalls, and drainage channels. Recently, Saskatchewan Environment published Stormwater Guidelines (Saskatchewan Environment, 2006), which are intended as high level technical guidance. The Stormwater Guidelines refer to the Saskatchewan Surface Water Quality Objectives for discharge water quality guidance. The water quality of stormwater discharges, however, remains unregulated.

Without specific regulation and as indicated by the Stormwater Guidelines (Saskatchewan Environment, 2006), the water quality of urban runoff discharges are guided by the Saskatchewan Surface Water Quality Objectives (SWQO) (SERM, 1997). Saskatchewan Environment "…has not set 'across the board' effluent standards…" (SERM, 1997) and instead examines each case, considering both the pollutant and the receiving water body. The document does, however, set out basic objectives for effluent discharges. These objectives, which are mainly qualitative, apply to both the effluent and the mixing zone. The objectives include items such as:

- avoidance of concentrations or combinations of substances that are acutely toxic;
- avoiding sludge deposits that affect aquatic life or waterfowl;
- free of debris that accumulates on the surface;
- avoiding nutrient concentrations that would lead to the eutrophication of the receiving water; and

- consumption of no more than 30%, in the case of diffuse discharges, and 10%, in the case of point discharges, of the assimilative capacity of the receiving water, based on the receiving water's minimum average 7 day flow rate with a 10 year return period (7Q10).

In addition to the general objectives, mixing zones have additional objectives, including (SERM, 1997):

- minimizing the size of the zone;
- imposing limited use for the mixing zone; and
- toxicity limits (96 hr LC50) for indigenous aquatic life should not be exceeded.

For the receiving water, the SWQO also include General and Specific objectives. The General Surface Water Quality Objectives "concern basic quality characteristics, which will afford a minimum degree of protection of all beneficial uses of surface water bodies…" (SERM, 1997). These general objectives apply to all waters outside of the mixing zones. The general objectives, summarized in Appendix A, are comprised of quantitative values as well as some qualitative descriptions.

The Specific Guidelines are based on specific uses such as cattle watering, varying degrees of water based recreation, aquatic life, and wildlife. These guidelines include specific quantitative values. Potable water supply is included in the specific guidelines section; however, it only includes qualitative statements regarding good watershed management and references the Water Regulations, 2002, (Government of Saskatchewan, 2002b) for quantitive values.

2.3 WATER QUALITY

Water pollution has two types of impact: acute and chronic (Thomann and Mueller, 1987; Harremoes, 1988; Adams and Papa, 2000; Bannerman et al., 1996; Chambers et al., 1997; Ellis and Hvitved-Jacobsen, 1996). Acute impacts produce immediate physiological effects caused by 'high' concentrations. In general, acute impacts are experienced in the order of hours after exposure. Acute toxicity limits are determined in laboratory assays. Water quality regulations or objectives are generally set several orders of magnitude smaller than the acute toxicity limit. Chronic impacts are caused by long term exposure to generally lower concentrations of a pollutant. Chronic effects are experienced after months or years of exposure. Numerous researchers have found that the effects of urban runoff are generally chronic in nature and that the pollutants are bioaccumulative (Adams and Papa, 2000; Bannerman et al., 1996; Chambers et al., 1997; Ellis and Hvitved-Jacobsen, 1996).

Many water quality parameters are of interest when examining urban runoff water quality and they have been studied to varying degrees. Some studies focus on only one parameter (Charbeneau and Barrett, 1998), while others focus on significant numbers of pollutants; for example, the United States Geological Survey (USGS) study of the mid-1980's examined over 150 water quality parameters (Smullen et al., 1999). NURP used 10 parameters to characterize the quality of urban runoff. These parameters are total suspended solids (TSS), five-day biochemical oxygen demand (BOD_5), chemical oxygen demand (COD), total phosphorous (TP), soluble phosphorus, total Kjeldahl nitrogen (TKN), nitrate plus nitrite ($NO_{2,3}$), extractable copper, extractable lead, and extractable zinc. Alberta Environment Protection (1999a) has compiled a list of parameters they suggest as an initial set of study parameters to which other parameters should be added considering both the catchment and receiving water characteristics. The list includes TSS, total dissolved solids (TDS), BOD_5, TP, total nitrogen (TN), Nitrate (NO_3), Chloride (Cl), Lead (Pb), Zinc (Zn), total coliforms (TC), E.Coli, and fecal coliforms (FC).

Makepeace et al. (1995) authored a literature review of as much literature as possible from the previous 25 years. In total, 140 articles were reviewed, mainly originating from the U.S., with several from Canada and Europe. The review examined concentrations of parameters and compared them to guidelines, where guidelines exist, for both aquatic and human health. Based on the work, a list of 28 "most critical stormwater contaminants" was established and is broken into two parts: those affecting human health (primarily drinking water related), and those affecting aquatic ecosystem health. The lists are summarized in Table 2.1.

Table 2.1 – Most critical stormwater contaminants (after Makepeace et al., 1995).

Human Health	Aquatic Health
Total suspended solids, aluminum, chloride, chromium, iron, lead, mercury, total polycyclic aromatic hydrocarbons (PAH's), benzo(a)pyrene, tetrachlorethylene, fecal coliforms, fecal streptococci, and Enterococci.	Total solids, total suspended solids, aluminum, beryllium, cadmium, chloride, chromium, copper, iron, lead, mercury, nitrogen, silver, zinc, dissolved oxygen, polychlorinated biphenyl (PCB's), bis(2-ethylhexyl) phthalate, γ-BHC, chlordane, heptachlor, and heptachlor epoxide.

The following discussion is a review of some of the water quality parameters reported to be significant in urban runoff and examined in this work. An expanded discussion of a variety of water quality parameters related to urban runoff is found in Appendix B.

2.3.1 Total Suspended Solids

The effects of TSS are one of the most documented effects of urban runoff (WEF/ASCE, 1998). TSS are of concern in urban runoff for a number of reasons. TSS cloud the water, which make it aesthetically displeasing, and prevent light from penetrating the water column (Peavy et al., 1985). More significantly, TSS provide adsorption sites for chemical and biological contaminants. These contaminants include heavy metals (Sansalone and Buchburger, 1997; Sansalone and Buchburger, 1996; Sutherland and Jelen, 1995), pesticides and herbicides (Marsalek and Schroeter, 1988), and live biological cells, which can include pathogens (Peavy et al., 1985; Sutherland and Jelen, 1995). Further, TSS in stormwater have been found to be highly correlated (positive correlation) with both nutrients and heavy metals (Adams and Papa, 2000).

TSS are the residue remaining on a 0.45 micron glass fibre filter after drying at 103-105°C (APHA et al., 1992). TSS consist of inorganic mineral soils (clay, silt, sand), organic particles (plant fibres, biological floc) and/or immiscible liquids (Peavy et al., 1985). Canadian and Saskatchewan aquatic guidelines for TSS permit an increase of 10 mg/L or 10% over the background concentration for waters with TSS greater than 100 mg/L (CCREM, 1987; SERM, 1997). In urban runoff, Makepeace et al. (1995) report a range of TSS concentrations from 1.0 to 36,200 mg/L, and EMC's ranging from 4 to 1223 mg/L. Cordery (1977) suggests urban runoff TSS results are misleadingly small because larger solids, such as litter and rocks, are too big and heavy and do not get collected by sampling equipment.

TSS impact aquatic habitat and life. Ellis and Hvitved-Jacobsen (1996) found that TSS discharged from stormwater systems are generally poor bedding material for plants and animals. In the receiving stream, TSS settle out and cover the native soils and alter their physical structure (i.e. particle size and composition), which hampers native organisms by altering their preferred habitat and exposing the organisms to whatever contaminants are attached to the TSS. Chambers et al. (1997) suggest that the changes caused by urban runoff TSS alter the aquatic food web structure and impair otherwise healthy aquatic populations.

Polychlorinated biphenyls (PCB's) are one example of the relationship of TSS to toxic pollutants. PCB's are only slightly soluble in water, however PCB's adsorb to sediments very easily (Makepeace et al., 1995). Thus PCB's spilled into sediment laden urban runoff are likely to contaminate the sediments and the area where the sediment eventually settles.

Hall et al. (1999) state the case of TSS very clearly: "… suspended sediments transported during storm events, with their plethora of chemical contaminants, are toxic." The NURP final report strongly suggests that when TSS concentrations are high, urban runoff controls should be implemented as opposed to advanced wastewater treatment (Makepeace et al., 1995).

2.3.2 Dissolved Oxygen

Dissolved Oxygen (DO) is essential for the survival and growth of fish and other aquatic species (Peavy et al., 1985; CCREM, 1987) as well as in the self-purification of receiving streams (CCREM, 1987). Further, the end-products of degradation in an anoxic/anaerobic environment are also quite aesthetically displeasing and often toxic to aquatic life. In urban runoff, DO has been reported to range from 0.0 mg/L to 14.0 mg/L (fully saturated) (Makepeace et al., 1995). The aquatic life protection guideline for DO is not less than 5 mg/L.

BOD_5 and COD are measures of the oxygen required to stabilize a sample, biochemically and using a strong oxidizing agent, respectively (Snoeyink and Jenkins, 1980; APHA et al., 1992). Both BOD_5 and COD reduce the amount of dissolved oxygen in the water. In urban runoff, BOD_5 has a reported range of 1.0 to 7,700 mg/L and COD has a reported range of 7 to 2,200 mg/L (Makepeace et al., 1995). Note that the BOD_5 and COD results are independent of each other, and have been pooled from several data sources. The apparent anomaly between the BOD_5 and COD (i.e. BOD_5 greater than the COD) is a reflection of the different data sources. In terms of BOD_5 and COD, urban runoff can approach concentrations similar to those of untreated wastewater.

2.3.3 Nutrients

Nitrogen and Phosphorus are the nutrients commonly of concern in surface water (Peavy et al., 1985). Nitrogen has many forms in water. In general, nitrogen is the growth limiting nutrient in marine (salt water) environments (Chambers et al., 1997). The main forms are nitrate (NO_3^-), nitrite (NO_2^-), ammonia (NH_3), and organic nitrogen (APHA et al., 1992; Snoeyink and Jenkins, 1980). Nitrate is a nutrient important in plant growth. Organic nitrogen and ammonia are measured together as total Kjeldahl nitrogen (TKN). Specific Canadian surface water quality guidelines for nitrate have not been set, except for a statement to avoid nuisance growth (SERM, 1997; Makepeace et al., 1995). Nitrate and Nitrite are discussed further in Appendix B.

Ammonia is the nitrogen form normally of concern in surface waters because it is very toxic to aquatic life (Makepeace et al., 1995; Francis-Floyd and Watson, 1996). Depending upon the target species, ammonia has aquatic acute toxicities ranging from 0.083 to 1.1 mg/L and chronic toxicity occurring at 0.0017 mg/L for salmonids (Makepeace et al., 1995). The Canadian freshwater aquatic guideline is 0.019 mg/L. The aquatic toxicity of ammonia increases with increasing pH and temperature. Ammonia is important in drinking water because of its interference with chlorine disinfection, although a Canadian drinking water guideline has not been established (CCME, 2003). Ammonia is found both in the dissolved state in the water column and sorbed to sediments (CCREM, 1987). It has been reported to range from 0.01 to

4.30 mg/L in urban runoff. Duncan (1997) found that nitrogen forms were significantly correlated (95% confidence) to annual rainfall across all land use types, which suggests that nitrogen deposition is related to atmospheric processes. Sources of nitrogen include fertilizers, industrial cleaning operations, feed lots, animal excrement, and combustion of fossil fuels (CCREM, 1987; EPA, 2003; Makepeace et al., 1995).

Total phosphorus (TP) is an important aquatic plant nutrient and has been frequently studied in relation to urban runoff (Makepeace et al., 1995; Duncan, 1997; Environment Canada, 2001; Chambers et al., 1997; Bannerman et al., 1996). TP is generally the growth limiting nutrient in freshwater plant systems (Chambers et al., 1997; CCREM, 1987) and has been found in concentrations ranging from 0.01 to 7.30 mg/L (Makepeace et al., 1995). TP is found adsorbed to sediments, in aquatic organisms, and in the dissolved state. The Province of Alberta has set a surface water objective for TP of 0.05 mg/L (Alberta Environment, 1999b). Elevated levels of phosphorus in freshwater are associated with prolific plant growth and accelerated eutrophication (CCREM, 1987). For example, downstream of the City of Saskatoon WWTP and prior to its upgrade in 1996, elevated nutrient concentrations were determined to be the major cause of excessive aquatic plant growth that contributed to water quality impairment (Chambers and Prepas, 1994). Phosphorous in urban runoff is derived from sources such as fertilizers, industrial waste, detergents, tree leaves, lubricants, and organic and inorganic chemical decomposition (Makepeace et al., 1995; CCREM, 1987).

2.3.4 Chloride

Chloride is an emerging issue in runoff, especially in cold weather climates. Road salts, due to the presence of chloride, have recently been classified as ‘toxic’ under the Canadian Environmental Protection Act (Environment Canada and Health Canada, 2001). However, road salts are not being banned. The designation is intended to recognize and highlight the potential environmental harm from the typically large quantities applied to roadways. Chloride, in urban runoff, is primarily derived from the de-icing of roads and sidewalks, tire ballasting, dust control, manufacturing of chemicals, wastewater treatment, fertilizers, and insecticides (Makepeace et al., 1995;

Environment Canada and Health Canada, 2001). It has been found in runoff in concentrations ranging from 0.30 to 82,000 mg/L. The Canadian drinking water guideline aesthetic objective (AO) for chloride is less than 250 mg/L and is primarily to prevent taste (CCME, 2003). Guidelines for irrigation of crops range from 100 to 700 mg/L depending on specific crop sensitivity (CCREM, 1987). Fruit crops are generally more sensitive than vegetable crops. Research suggests that 5% of fresh water aquatic species would be affected (LC50) at chronic chloride concentrations of 210 mg/L (Environment Canada and Health Canada, 2001). Acute fresh water toxicities are suggested to be in the range of 1400 mg/L.

Chloride accumulates in deep storage compartments of oil and grit separators (e.g. Stormceptor), creating a high density layer of chloride rich water (Health Canada and Environment Canada, 2001; Marsalek and Schaefer, 2003). This chemically induced stratification can also apply to other stormwater BMP's such as constructed wetlands and detention ponds. Issues associated with this stratification include reduced removal of fine sediments due to reduced effective volume, shock loadings should the dense layer wash out, and leaching of sediment bound heavy metals in the bottoms. In detention ponds (i.e. urban lakes), stratification can prevent the turn-over (mixing) of lakes, thereby restricting the distribution of DO and nutrients. Concentrations have been measured as high as 4,000 mg/L in natural ponds and wetlands, 2,000 to 5,000 mg/L in urban detention ponds, and 36,500 mg/L in manhole type oil and grit separators (Environment Canada and Health Canada, 2001; Marsalek and Schaefer, 2003). In aquatic plant habitat impacted by chloride, cattails and common reed grass are observed to readily invade.

2.3.5 Metals

Many metals have been detected in urban runoff (Makepeace et al., 1995; Bannerman et al., 1996; Munch and Keller, 1985; Marsalek and Schroeter, 1988). Based on a review of 140 studies, Makepeace et al. (1995) found lead, copper, zinc, cadmium, nickel, arsenic, and beryllium were found to be of "greatest concern" because the concentrations found exceeded guideline values by 10 times or more. Metals pose a toxicity concern for both acute and chronic exposure for aquatic and human life (CCME, 2003; Rice et. al., 2002; Makepeace et al., 1995; EPA, 2003). Changes in other parameters, such as hardness, alkalinity, and pH, can affect the toxicity of many metals. Many metals are also positively correlated with TSS load. Common sources of metals include roadway runoff (e.g. vehicle corrosion, engine drips, driving surface wear, vehicle component wear, road salting), preservative treated wood, electrical waste, corrosion of galvanized steel, and fossil fuel combustion (CCME, 2003; Rice et. al., 2002; Makepeace et al., 1995; EPA, 2003; CCREM, 1987; Health Canada, 2003a). Expanded discussion of the aforementioned metals is provided in Appendix B.

2.3.6 Organic Chemical Parameters

Many organic chemical contaminants have been identified in stormwater in numerous studies (Makepeace et al., 1995; Marsalek and Schroeter, 1988; Adams and Papa, 2000; Bannerman et al., 1996). NURP alone examined over 100 such parameters, many of which have toxicity concerns and are associated with TSS (Makepeace et al., 1995). Organic chemical parameters are derived from sources such as pesticides, petroleum fractions, dry cleaning agents, industrial degreasers, dyes, preservatives, and plastics manufacturing (Health Canada, 2003b; Makepeace et al., 1995; EPA, 2003; CCREM, 1987). (Note that, herein, the term pesticide is used as a generic term and includes herbicides, fungicides, insecticides, and fumigants.) More information regarding organic chemical parameters is contained in Appendix B.

2.3.7 Micro-organisms

"Stormwater has been shown to be a possible major source of microbial pollution to receiving bodies of water" (Makepeace et al., 1995). Some micro-organisms are pathogenic to humans; disease outbreaks in both Walkerton, Ontario and North Battleford, Saskatchewan were directly linked to micro-organism contamination of the drinking water source (O'Connor, 2002; Stirling et al., 2001). The pathogenic risk associated with micro-organism contamination is the concern in urban runoff. Micro-organism contamination is a common cause of beach closures (CCREM, 1987; Chambers et al., 1997; Adams and Papa, 2000; OMOE, 1998). Micro-organism contamination has caused the closure of shell fish beds (Krenkel and Novotny, 1980). Micro-organisms can bind to sediments (Makepeace et al., 1995; Glasner and McKee, 2002), which provides considerably lengthened survival times and greater resistance to disinfection (Ellis and Hvitved-Jacobsen, 1996; Makepeace et al., 1995). Sediment bound micro-organisms can become re-suspended into the water column, thereby increasing the pathogenic risk during periods when the bottom sediments are disturbed (Glasner and McKee, 2002), during periods of increased flow, or during wading. Total coliforms (TC) and fecal coliforms (FC) are commonly used as indicators of micro-organism contamination. These indicators and other issues regarding micro-organisms are discussed further in Appendix B.

2.4 POLLUTANT BUILDUP AND WASH-OFF

Buildup and wash-off are the general terms used to describe the complex processes involved in pollutant deposition on and removal from a catchment by rainfall-runoff. Buildup includes the processes of deposition and accumulation on the catchment surface. Wash-off includes the removal of pollutants from the catchment surface as well as from the atmosphere. The buildup and wash-off processes are idealized in Figure 2.1.

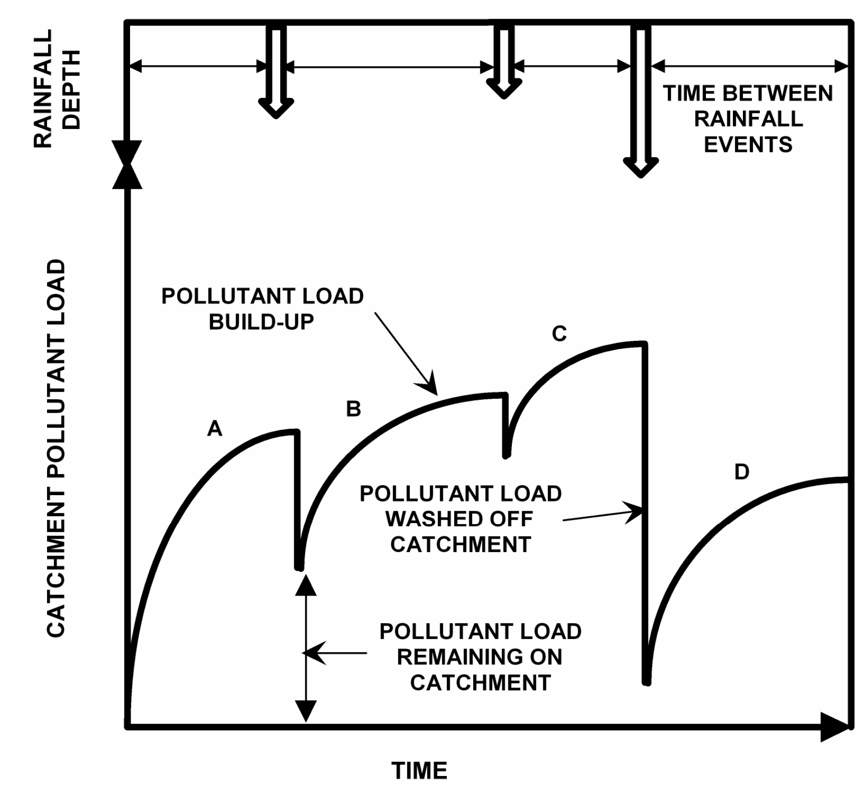


Figure 2.1 – Idealized representation of the build-up and wash off processes (after LeBouthillier et al., 2000).

The curves (A, B, C, and D) in Figure 2.1 represent the build-up of pollutant mass on the catchment prior to a rainfall-runoff event. The height of the nearly vertical lines between the curves represents the pollutant removed from the catchment during a rainfall-runoff event. The point where the subsequent buildup curve begins represents the amount of pollutant remaining on the catchment after the rainfall-runoff event. Build-up is generally thought to have a decreasing rate of increase (Charbeneau and Barrett, 1998; LeBouthillier et al., 2000; Akan and Houghtalen, 2003). When a rainfall event occurs, runoff is generated and some amount of pollutant is removed from the catchment. The rainfall events illustrated in Figure 2.1 do not show complete wash-off.

Two general approaches to the computation of buildup and wash-off are used: lumped time and continuous time (Akan and Houghtalen, 2003). The lumped time approach examines both buildup and wash-off together, while the continuous time approach examines buildup and wash-off separately. Lumped time approaches generally produce a mass load for a given period and are less complex than continuous time models. Lumped time methods are examined further in subsequent sections as characterization parameters.

Continuous time models examine buildup and wash-off separately, generally produce a concentration output, and are more complex than lumped time models. They are generally empirical (Akan and Houghtalen, 2003), but mechanistic or semi-mechanistic models have been proposed (Sutherland and Jelen, 1995). Empirical buildup models generally utilize power law or exponential relationships (Charbeneau and Barrett, 1998; Akan and Houghtalen, 2003) and are usually modelled as a function of several variables including the antecedent dry period, land use, traffic density, and ultimate (maximum potential) catchment pollutant load (Charbeneau and Barrett, 1998; Adams and Papa, 2000; Akan and Houghtalen, 2003). Empirical wash-off models generally utilize an exponential decay relationship over a relatively short period of time (Charbeneau and Barrett, 1998; Akan and Houghtalen, 2003). Wash-off is thought to be influenced by rainfall characteristics (intensity, depth, etc.), catchment topography, particle characteristics, and catchment surface type and condition (new pavement, old pavement, grass, open field, etc.) (Butler and Davies, 2000).

2.5 CHARACTERIZATION PARAMETERS

2.5.1 Event/Site Mean Concentration

The event mean concentration (EMC) was selected in the 1980's by the U.S. EPA as the primary method for evaluation of urban runoff pollutant loading and has become the predominant method for reporting runoff event water quality (EPA, 1982; WEF/ASCE, 1998; Charbeneau and Barrett, 1998; Ellis and Hvitved-Jacobsen, 1996; Novotny, 1992). The EMC characterizes an individual event, while the average EMC or Site Mean Concentration (SMC) is used to characterize the catchment. The SMC can then be used to further evaluate urban runoff pollutant loading. The SMC method has been adopted by some municipal jurisdictions in Canada, such as the City of Calgary under the approval of Alberta Environment (Deong, 2001).

The EMC is calculated by dividing the total pollutant mass washed off by the total volume of water discharged during the event, as shown in [2.1]

[2.1] $$\mathrm{EMC} = \frac{M_T}{V_T} = \frac{\sum_{i=1}^{n} C_i \cdot V_i}{\sum_{i=1}^{m} V_i}$$

where M_T is the total mass washed off, V_T is the total volume discharged, C_i is the pollutant concentration at interval i, V_i is the volume discharged in interval i, n is the total number of intervals of concentration data, and m is the total number of intervals during the runoff event. Note the difference between the numerator and denominator, wherein the numerator only covers the sampling period while the denominator covers the entire event. Ideally, n and m are the same. EMC and SMC have units of concentration (mg/L) and have been found to be independent of runoff volume (Marsalek and Schroeter, 1988). This is not surprising considering that runoff volume is used in the calculation.

The SMC is a measure of central tendency of the EMC and is determined based upon the underlying distribution of the EMC's (Adams and Papa, 2000). It has been generally accepted that the distribution of EMC's does not fit the normal distribution, and that the log-normal distribution is usually the best fit (Van Buren et al., 1997; U.S.

EPA, 1982; Ellis and Hvitved Jacobsen, 1996; WEF/ASCE 1998; Marsalek, 1990; Smullen, 1999). The measure of central tendency for the log-normal distribution is the geometric mean (i.e. the mean of the log transformed data). Instead of the geometric mean, some researchers (e.g., Bannerman et al., 1996) use the median value of the data.

One of the conclusions of NURP was that land use has no effect on SMC, mainly because the site to site variability of the SMC's was large among similar land uses (Bannerman, 1996; Duncan, 1997; Marsalek and Schroeter, 1998; Smullen et al., 1999). Smullen et al. (1999) report that, in spite of this finding, urban runoff investigators continue to use land use specific values. A criticism of this NURP conclusion is that the definitions of the land use types were not specific enough and open to too much interpretation (Bannerman et al., 1996). The result of this non-specificity is variability in the data that may have caused variation in the SMC's that masked the true similarities between the land uses. Bannerman et al. (1996) found statistically significant differences between SMC's for different land uses and suggested that SMC's are affected by land use.

The SMC can be used to estimate the annual load discharged (Marsalek, 1990; Charbeneau and Barrett, 1998; Deong, 2001). Often referred to as the "simple method", the load is determined as the product of the SMC and the volume discharged as shown in [2.2], viz.

[2.2] $$L = SMC \cdot V_T$$

where L is the load discharged, and V_T is the volume discharged. In the absence of measured data, the volume can be determined by using rainfall records (Charbeneau and Barrett; Novotny, 1992; Marsalek, 1991). Using the rainfall depth from local gauges, and recorded runoff volume data, a runoff coefficient, C, can be developed as formulated in [2.3]

[2.3] $$C = \frac{V_T}{d \cdot A}$$

where d is the rainfall depth over the catchment and A is the catchment area. The volume discharged can be estimated by rearranging equation [2.3]. Marsalek (1991) supports this method of estimation of the urban runoff pollutant load and refers to a long term study that, considering all other sources of error, achieved reasonable accuracy using this method.

2.5.2 Unit Load

The unit load is also used to characterize pollutant discharges (Novotny, 1992; Marsalek, 1990). The unit load (U) is the total annual mass of pollutant (M_{annual}) discharged divided by the area of the catchment from which it originated. It has units of kg/ha/year and is formulated in [2.5] as

$$[2.5] \quad U = \frac{M_{annual}}{A}$$

Depending on the available data, M_{annual} can also be the average over several years. To determine the annual load using the unit load, the only information that is required is the catchment area. This parameter, different than the SMC, does depend on the catchment type, because it incorporates all characteristics of the catchment within one parameter (Novotny, 1992). The incorporation of all of the effects into one parameter is also the cause of some skepticism, namely that the unit load is too simple and masks some of the inherent variability in the pollutant accumulation and washoff and hydrological processes (WEF/ASCE, 1998). The unit loads have, however, been used with some success in pollutant load estimation (Marsalek, 1991), but the SMC method is generally preferred (WEF/ASCE, 1998).

2.6 RAINFALL ANALYSIS

In the usage of the SMC and determination of the unit load, the volume of runoff is required. The runoff volume can be determined from direct measurements at or near storm sewer outfalls or estimated from rainfall data (Marsalek, 1991). Analysis of rainfall data requires a precise definition of a statistical rainfall event. A minimum rainfall amount (depth) and the amount of time between discrete rainfall measurements are sufficient to make the judgement that the events are separate. The inter-event time definition (IETD) is used in the determination of the minimum time between events (Adams and Papa, 2000; Adams et al., 1986).

Adams and Papa (2000) present two methods for determination of the IETD. The first method uses a statistical correlation method which requires the processing of a significant amount of rainfall data. The second is a simpler approximate method that uses a plot of the number of rainfall events per year versus the inter-event time. Using the shape of the plotted curve, the IETD is the time when the rate of change of the number of events per year begins to stabilize with a change in inter-event time. A generalized IETD plot for two cities is presented in Figure 2.2. Based on the shape of the curve, a range of potential IETD's for each city is indicated. Adams and Papa (2000) suggest that an IETD of between one and six hours is appropriate for most urban areas, but that the selection should also be based upon the catchment and the intended use of the data. For example, larger catchments have a slower response to rainfall input and more time is required to return to baseline conditions. Therefore, in this case a longer IETD should be chosen (Adams et al., 1986). For planning level estimations, a longer IETD should be chosen because it filters out the smaller less significant rainfall events.

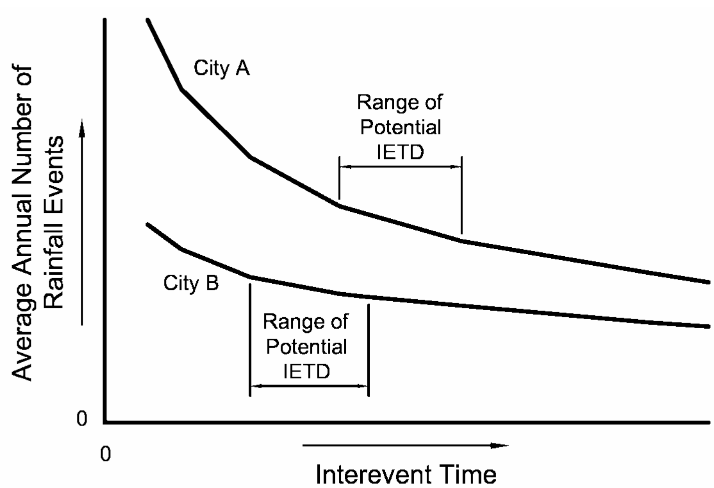


Figure 2.2 – Generalized IETD variation with annual number of rainfall events (after Adams and Papa, 2000).

Also important in defining a statistical rainfall event is the minimum depth of rainfall. Adams et al. (1986) suggest a minimum rainfall depth of 0.5 mm based on experience from Vancouver monitoring work. NURP suggests a minimum of 0.1 inches (2.5 mm) however, this is intended to assist U.S. municipalities to economically gather data from the most representative rainfall events.

2.7 REPORTED CHARACTERIZATIONS

NURP examined 81 catchments representing various land uses in 28 metropolitan areas throughout the U.S. in the late 1970's and early 1980's. NURP reported SMC's based on all urban sites that were monitored. The reported SMC characterizations are shown in Table 2.2. The results were also reported as annual unit loads which are shown in Table 2.3.

Table 2.2 – Reported SMC's (mg/L).

Location/Study	Comment	TSS	TKN	TP	BOD_5	COD
NURP, 1983[a]	USA, All urban	100	1.50	0.33	9	65
Smullen et al., 1999	USA, All urban	78.4	1.73	0.32	14.1	52.8
Duncan, 1997	World wide, All urban	155	2.63[b]	0.32		
	One std dev range	51-468	1.38-5.01	0.13-0.79		
Choe et al., 2002	Korea, Residential	414	6.81	2.85	125	226
	Commercial	276	14.1	1.88	169	501
	Mixed Industrial	106	5.07	1.93	43	80
Macdonald, 2003	Vancouver, all urban	44	0.8	0.14	5	34
Ellis and Hvitved-Jacobsen, 1996	Demark, all urban	30-100	2	0.5	5	40-60
Bannerman et al., 1996	Wisconsin, all urban	237	1.8[b]	0.45	18	69

[a] – as reported by Novotny, 1992
[b] – TN

Table 2.3 – Reported unit loads (kg/ha/yr).

Location/Study	Comment	TSS	TKN	TP	BOD_5
Marsalek, 1978	Low-Med density residential	390	9[b]	1.6	34
	Commercial	560	11.2[b]	3.4	90
	Industrial	672	7.8[b]	2.2	34
NURP, 1984[a]	Residential	550	5.8	1.3	
	Commercial	1460	15.4	3.4	
	All urban	640	3.6	6.6	

[a] – as reported in Raymond, 1997
[b] – TN

Smullen et al. (1999) undertook to update the characterizations presented in NURP. The project data set utilized the original NURP data, USGS data as well as

NPDES monitoring data. The results generally showed reductions from the NURP values. They concluded that significant differences exist between the original NURP characterizations and the pooled results. The updated values are shown in Table 2.2.

Choe et al. (2002) reported SMC's from several catchments with specific land uses in Chongju, Republic of Korea. They note that their characterizations are somewhat different from others reported in the literature. They attribute the difference to a lack of street sweeping and heavy industrialization in Korea.

Similar to Smullen et al. (1999), Duncan (1997) collected and pooled numerous data sets from around the world to produce SMC's for TSS, TN, and TP for all urban land uses, as shown in Table 2.2. Duncan (1997) also reported the standard deviation of the log transformed data to indicate the range of variability of the data used. Table 2.2 reports these values transformed to real space (i.e. log base 10).

Marsalek (1978) reported annual unit load characterizations, shown in Table 2.3, for a variety of unswept, urban Ontario catchments. The work was carried out as part of the Pollution from Land Use Activities Reference Group of the International Joint Commission in the early assessments of pollutant loading to the Great Lakes. The author notes that, for planning newly developed land (i.e. prior to final landscaping development), the unit load for TSS should be increased to 1700 kg/ha/yr from the value reported in Table 2.3.

CHAPTER 3 STUDY SITES

3.1 General Physical and Climatic Conditions of the City of Saskatoon

Saskatoon is the largest city in Saskatchewan with a population of more than 196,811 persons (Statistics Canada, 2001). It is located at 52° 07' N, 106° 38' W, on the banks of the South Saskatchewan River. It has a plan area of approximately 154 km^2 and geodetic elevations ranging from approximately 474 m at the river to approximately 512 m at the east boundary of the City.

The climate is continental and dry. The City receives an average of 2,380 hours of sunshine and an average of 350 mm of precipitation annually, 265 mm of which falls as rain (Environment Canada, 2003). The summer period (May to September) has a mean temperature of 15.8 °C.

3.2 Site Selection and Characteristics

Each catchment was chosen on the basis of the primary land use that it represented, namely newer residential, older residential, commercial and light industrial. The various catchments were also chosen because of their relatively large size, thus making them less susceptible to the effects of clandestine or incidental spills or point sources, which may otherwise compromise the study. Moreover, the trunk sewers from large catchments have larger (deeper) flows on a more consistent basis, thus making it easier to collect samples. The catchments and specific monitoring locations are shown in Figure 3.1. The shaded area of the catchment indicates the portion of the catchment that was monitored. The arrows point to the monitoring locations. The catchment characteristics are summarized in Table 3.1. By COS convention, catchment areas in Saskatoon are named for the area that they drain or the roadway that they run under just before discharging to the river. As is the case throughout COS, the selected catchments have separate sewer systems and do not include any intentional interconnections between the sanitary and storm sewer systems. The study catchments were chosen to avoid existing BMP's so that the true characteristics of the land use were observed. During the 2002 data collection season, an existing BMP was noted in the Taylor Street catchment that had not been previously identified. The BMP, an equalization storage pond, is intended to provide flooding protection to low lying homes and is discussed further in Section 3.2.4. The Taylor Street catchment was the only study catchment to have a BMP. COS also operates a street sweeping program in the spring to remove sand and gravel applied to the streets during the winter. The program is intended to improve aesthetics, reduce the amount of blowing dust, and for safety (i.e. traction to asphalt).

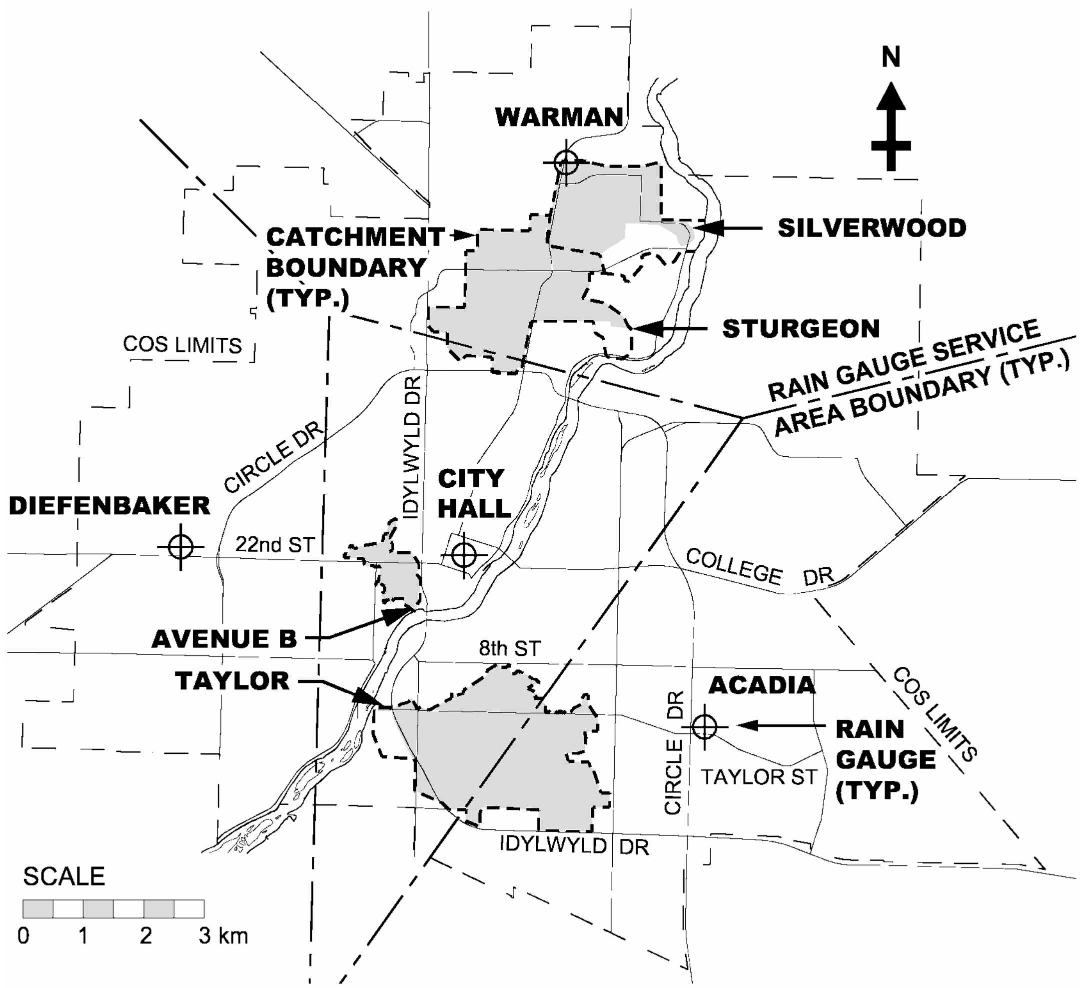


Figure 3.1 – Study catchments, monitoring locations and rain gauges. The shaded areas indicate the portion of the catchment monitored. The arrows indicate the specific monitoring locations.

Table 3.1 – Catchment characteristics.

Catchment Name	Catchment Area Monitored (ha)	Relative Area (% of COS)	Catchment Type
Avenue B	74.6	0.49	Commercial
Silverwood	241	1.57	Newer residential
Sturgeon	420	2.73	Light industrial
Taylor	616	4.10	Older residential
Total	1352	8.89	

Within each catchment, specific monitoring locations (manholes) were chosen based on the accessibility for equipment, safety during servicing, and hydraulic characteristics. To simplify development of stage-discharge curves, the ideal pipe has a straight alignment in plan view and the same slope on the effluent side of the manhole as on the influent side. None of the locations had the same pipe slope into and out of the manhole. In most catchments, separate, but sequential, manholes were used for water quality sampling and water quantity monitoring because of physical space limitations.

At each location, the slopes of the storm sewer invert were determined using as-constructed drawings, provided by COS, and confirmed with field surveys. Almost invariably, the surveyed elevations were different from that shown on the as-constructed drawings. The elevations determined by an optical level survey of the Taylor Street location were substantially different from the as-constructed drawings (200 mm and greater), which was confirmed (with good agreement) using a total station. The horizontal distances between manholes were taken from the as-constructed drawings. The calculated pipe slopes are shown in Table 3.2 along with the slopes determined from the as-constructed drawings. The locations noted in Table 3.2 are listed in a downstream direction (i.e. the first point is the highest in elevation). The slopes and distances noted are between the point for which they are listed and the previous point in the list.

Table 3.2 – Pipe slopes and distances between manholes.

Location	Slope		Distance
	Survey (% m/m)	COS as-constructed (% m/m)	(m)
Ave. B			
Ave. B & 19th St. MH	-	-	-
WQ sampler and depth monitor MH	0.66%	0.66%	176
Outfall invert	0.52%	0.62%	63
Silverwood			
WQ sampler MH	-	-	-
Depth monitor MH	1.96%	1.29%	13
D/S MH	1.08%	1.29%	74
Sturgeon Drive			
WQ sampler MH	-	-	-
Depth monitor MH	2.38%	2.11%	28
D/S MH	1.67%	1.71%	210
Taylor Street			
St. George MH	-	-	-
On-ramp MH	0.42%	0.43%	189
COS depth monitor MH (Herman)	0.49%	0.43%	88
St. Charles MH	0.41%	0.44%	92

3.2.1 Avenue B Catchment

The Avenue B catchment is situated west of the downtown core and is primarily commercial development. Land uses within the catchment include restaurants, warehousing, commercial resellers, and small vehicle dealerships (some with gravel lots). Development in the area began as early as 1900. The catchment generally slopes to the east-south-east at about 0.4%. An example of typical land use and development are shown in Figure 3.2.

Figure 3.2 – An example of typical land use and development within the Avenue B catchment (300 Block Avenue C South looking north).

The Avenue B monitoring manhole is located at the intersection of Avenue B and 18th Street, within the former A.L. Cole power station site, as shown in Figure 3.3. An area of 74.6 ha or 0.49% of COS was monitored. The monitoring manhole has a rim to invert depth of approximately 10.9 m, which is greater than the maximum 8 m lift of the sampler pump. Therefore, at this site, the sampling equipment was hung approximately 3.5 m down from the rim, as shown in Figure 3.4. The concrete storm sewer pipe is 1.37 m in diameter. Figure 3.5 shows the site with the sampler sitting adjacent to the manhole.

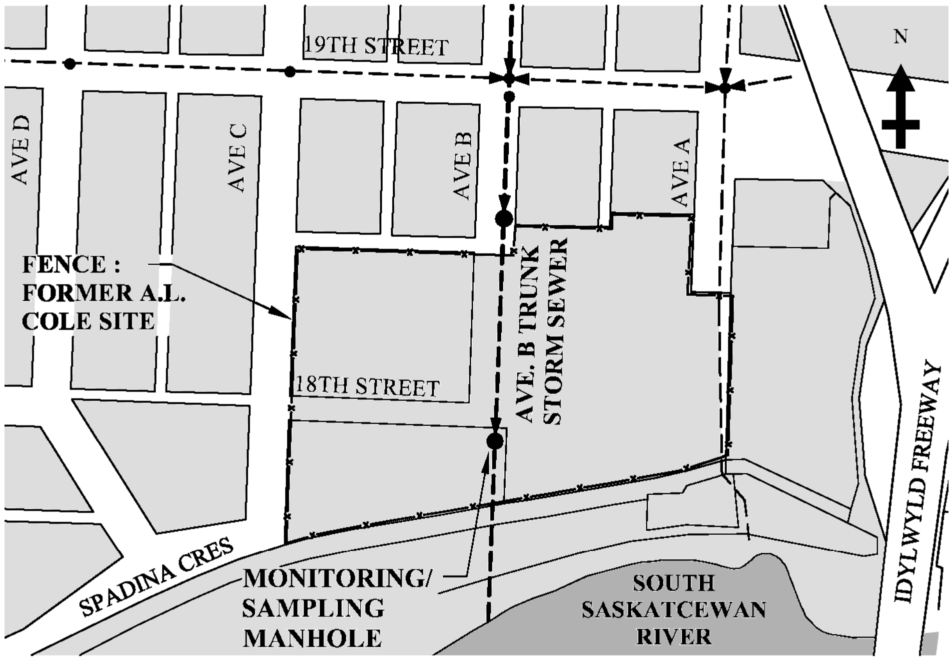


Figure 3.3 – Avenue B monitoring location site plan.

Figure 3.4 – Avenue B monitoring location manhole with equipment installed.

Figure 3.5 – General view of the Avenue B location with sampling equipment.

3.2.2 Silverwood Catchment

The Silverwood catchment is located in the north portion of the City on the west side of the South Saskatchewan River. It is primarily newer residential development that was developed in the early to mid 1980's. The catchment is generally graded to the east at 0.1% in the western portion and 0.7% to 1.0% to the east in the eastern portion where the catchment descends into the river valley. A typical view of the catchment is shown in Figure 3.6.

The Silverwood monitoring location is located in the park adjacent to Whiteswan Drive and the COS WWTP. A site plan is shown in Figure 3.7. COS maintains a permanent ultrasonic flow depth monitor in the storm sewer of the catchment. An area of 241 ha or 1.57% of COS was monitored for both flow and water quality. The rim to invert depth of the flow monitor manhole is approximately 3.9 m. The plan alignment of the influent and effluent pipes is straight. The sampling manhole was chosen directly upstream of the depth monitor manhole. The sampling manhole has a rim to invert depth of approximately 3.9 m. The storm sewer pipe at both manholes is 1.5 m diameter reinforced concrete. Figure 3.8 shows the sampling manhole during sample retrieval. Figure 3.9 shows the manhole with the sampling equipment installed.

Figure 3.6 – Example of typical development in the Silverwood catchment (Neusch Cres. looking north west).

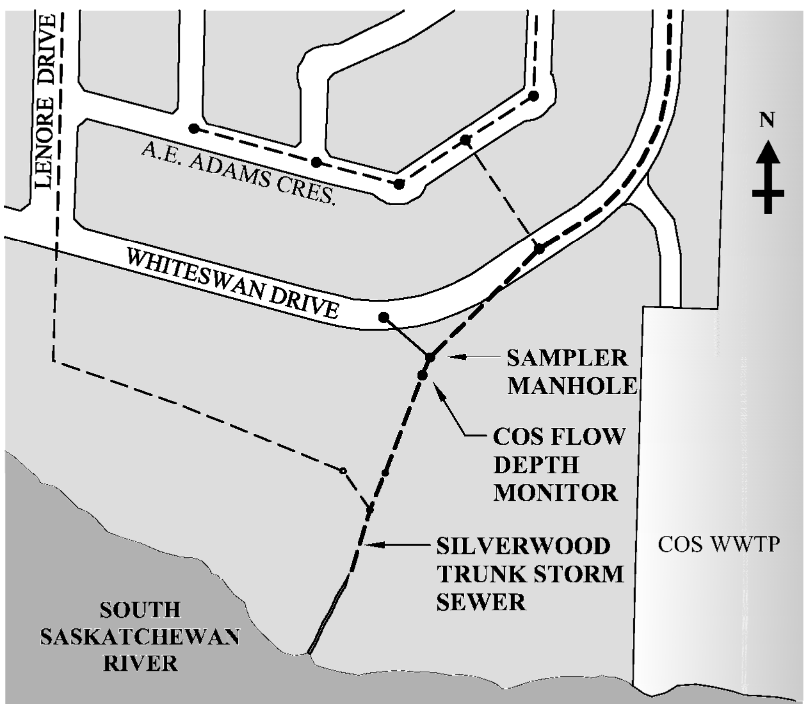


Figure 3.7 – Silverwood catchment monitoring location site plan.

Figure 3.8 – Silverwood trunk water quality monitoring location during sample retrieval with Gordon Liang conducting on-site tests.

Figure 3.9 – Silverwood trunk water quality monitoring location – manhole with equipment installed.

3.2.3 Sturgeon Drive Catchment

The Sturgeon Drive catchment is located in the north industrial portion of the city on the west side of the river. The catchment is primarily light industrial. Typical activities within the catchment include metal fabrication, auto body repair, asphalt oil processing, storage compounds for trucking companies (gravel lots), and warehousing. An example of the land uses and developments are shown in Figure 3.10. The catchment does not appear to have a definite general slope akin to the other catchments.

The Sturgeon Drive trunk monitoring manhole is located at the intersection of Nahanni Drive and Nahanni Drive, as shown in the site plan, Figure 3.11. An area of 420 ha or 2.73% of COS was monitored at this location. The 2001 water quality and flow monitoring manhole location is shown in Figure 3.12. Note that the background of Figure 3.12 is not representative of the land uses of this catchment. The manhole has a rim to invert depth of approximately 5.9 m. The storm sewer pipe is 1.37 m diameter reinforced concrete. Figures 3.13 and 3.14 show the manhole with the sampling equipment installed. To allow installation of the sampler, COS removed the top ladder rung from this manhole.

Figure 3.10 – An example of typical development in the Sturgeon Drive catchment (3200 block Northridge Drive, looking east).

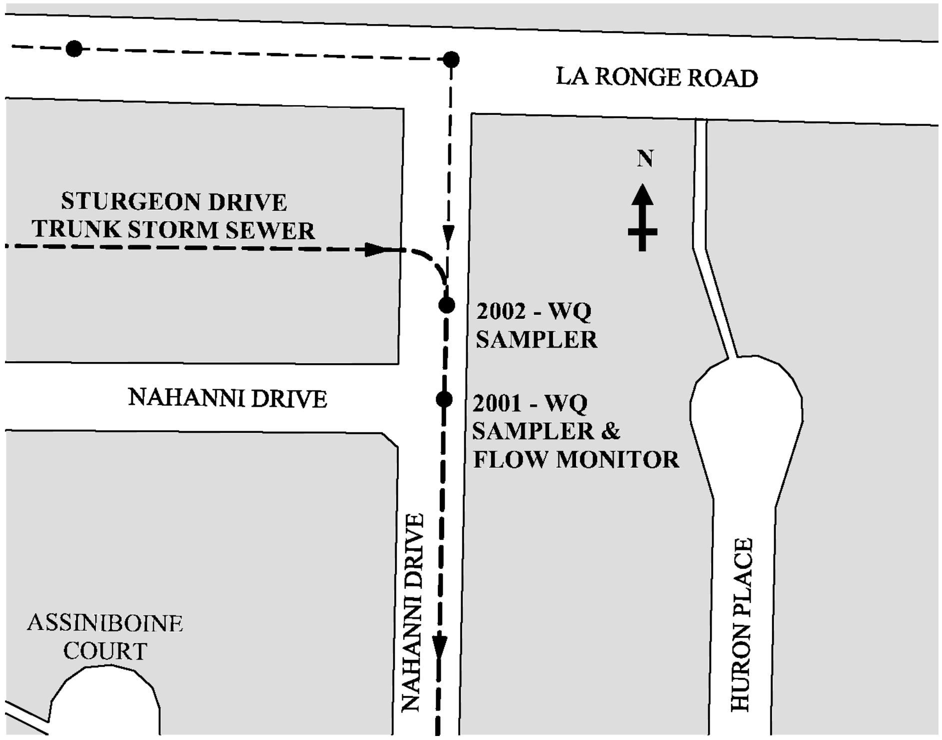


Figure 3.11 – Sturgeon Drive monitoring location site plan.

Figure 3.12 – Sturgeon Drive trunk monitoring location (intersection of Nahanni Drive and Nahanni Drive looking NE).

Figure 3.13 – Sturgeon Drive trunk 2001 water quality monitoring location manhole with equipment installed.

Figure 3.14 – Close-up view of Sturgeon Drive monitoring manhole with equipment installed.

3.2.4 Taylor Street Catchment

The Taylor Street catchment is located in the southern portion of the City on the east side of the river. The catchment is primarily older residential, with some small strip malls scattered throughout the catchment, typical of residential development. The majority of the catchment was constructed during the 1950's, except for a small portion (less than 5% by area) in the south east corner, which was constructed as late as 2000. An example of the catchment is shown in Figure 3.15. The catchment generally slopes to the west at 0.4% to 0.5%. After selection and instrumentation of this catchment, a small equalization storage pond was found connected to the catchment storm sewer. The pond, shown in Figure 3.16, is intended to provide some storm sewer surcharge protection for low lying areas. Considering the small size of the pond in relation to the catchment size and the ponds intended function (i.e. quantity issue not quality issue), the effect of the pond on the characterizations is presumed to be minimal.

The Taylor Street sampling manhole is located on Taylor Street near the on-ramp to the Idylwyld Freeway and is shown in Figure 3.17. A siteplan of the monitoring location if shown in Figure 3.18 The sampling manhole has a rim to invert depth of approximately 5.3 m. Figure 3.19 shows the manhole with the sampling equipment installed. The flow monitoring manhole is directly downstream of the sampling site and has a rim to invert depth of approximately 5.4 m. At both manholes, the storm sewer is 1.52 m diameter corrugated metal pipe. Upstream and downstream of the sampling manhole, COS has lined the pipe with stainless steel to reduce the pipe friction, because insufficient capacity and surcharging have been an issue. The storm sewer pipe downstream of the COS flow depth monitor manhole was not lined at the time of the study.

Figure 3.15 – An example of typical development in the Taylor Street catchment (2100 block Haultain Avenue, looking south).

Figure 3.16 – Equalization storage pond in the Taylor Street catchment (Glasgow Park, looking west).

Figure 3.17 – General view of the Taylor Street sampling location (500 block Taylor Street West, looking west).

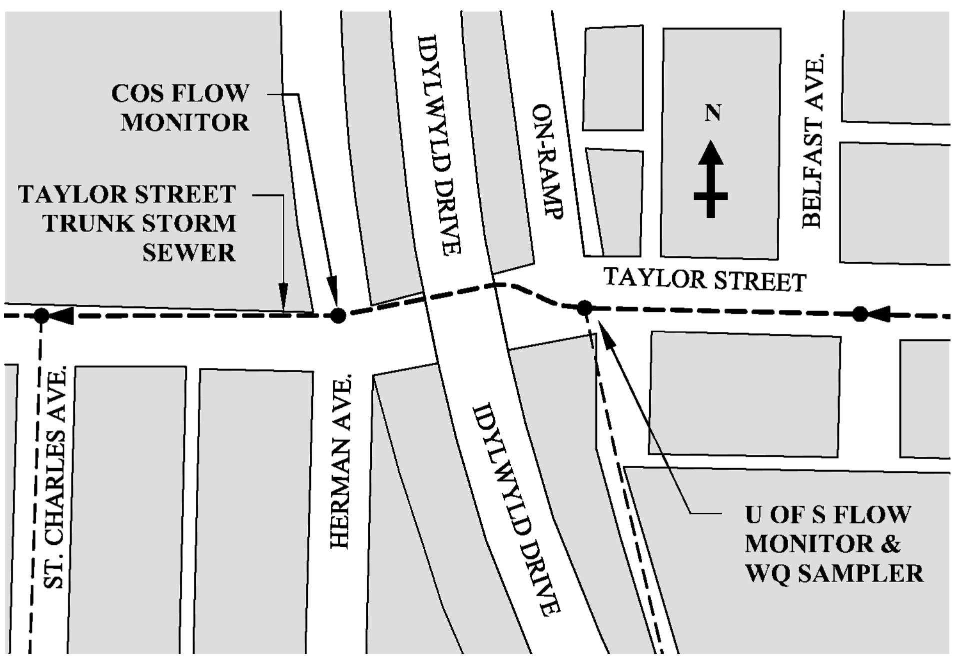


Figure 3.18 – Taylor Street monitoring location site plan.

Figure 3.19 – Taylor Street manhole with sampling equipment installed.

CHAPTER 4 METHODOLOGY

4.1 DATA COLLECTION EQUIPMENT

4.1.1 Water Quality Sample Collection

The use of automated samplers is strongly recommended in the literature as they provide samples over the duration of a runoff event and the ability to initiate the sampling sequence without being on site (Alley, 1977; Ellis and Hvitved-Jacobsen, 1996). Ellis and Hvitved-Jacobsen (1996) conclude that spot or grab sampling of runoff flows is inappropriate for stormwater studies, and that "intensive monitoring programs are needed…". To this end, each monitoring location was outfitted with an automated water sampler, which was used to draw samples from the trunk sewer. Two samplers were provided by the University of Saskatchewan Department of Civil and Geological Engineering (ISCO 6700 and ISCO 1680) and two were provided by SE (American Sigma Streamline 800SL). All of the samplers use a 12 volt, 7 to 18 amp-hour lead acid gel cell (non-spill) battery or nickel cadmium (NiCad) battery. The NiCad batteries were abandoned early in the program in favour of the more reliable lead acid batteries. The samplers were also moved from location to location to fill in gaps in the data due to equipment malfunctions.

The ISCO 6700 sampler is digitally controlled and has an add-on module for discharge measurement. The sampler is capable of drawing 24 - 1 L samples of various types, including even time based, flow proportioned, uneven time based, or split programming that combines two of the above types of sampling. The sampler monitors the depth of flow using an ultrasonic sensor. The flow data are stored digitally in the sampler's memory until downloaded using a laptop computer. The ISCO 6700 was used at Sturgeon Drive and Taylor Street in 2001 and at Avenue B in 2002. The sampler is shown in Figure 4.1.

Figure 4.1 – ISCO 6700 sampler.

The ISCO 1680 sampler has analog controls and manual data acquisition. It is capable of drawing 28 – 500 mL samples based on an even time basis or flow pulse input (flow proportioned). The time interval or number of flow pulses prior to collecting the first sample can be set to be different than the remaining samples. If the initial interval is set to a small value, then a sample is collected when the sampler is enabled. If the initial interval is set to a large value, then the first sample is delayed. A short (one pulse) initial interval was used so that a sample was collected as soon as the sampler was enabled. An external trigger mechanism was used to enable the sampler when a preset depth of flow was detected and to provide an electrical pulse every 0.1 minutes. The triggers are described following the sampler descriptions. The preset depth of flow was chosen to minimize false triggers and yet to trigger as early in the runoff event as possible. The ISCO 1680 was used at Silverwood in 2001 and at Taylor Street in 2002. The sampler was specifically used in the Taylor Street catchment because the greater number of sample bottles allowed a longer monitoring window while maintaining the same sample interval. The sampler is shown in Figure 4.2.

Figure 4.2 – ISCO 1680 sampler.

The two American Sigma Streamline 800 SL samplers are digitally controlled, with some digital data storage capacity. However, the digitally stored data must be recorded manually from the LCD display. The samplers can draw 24 - 1 L samples and can collect samples based on an even time basis or flow pulse input. A major drawback of these samplers is that they do not collect a sample when enabled and do not have an option to use a different time for the initial sample. Therefore, one interval must elapse prior to the collection of the first sample, thus delaying the first sample further from the start of the flow event. These samplers were used at Avenue B, Silverwood and Taylor Street in 2001 and at Sturgeon Drive and Silverwood in 2002. One of these samplers is shown in Figure 4.3.

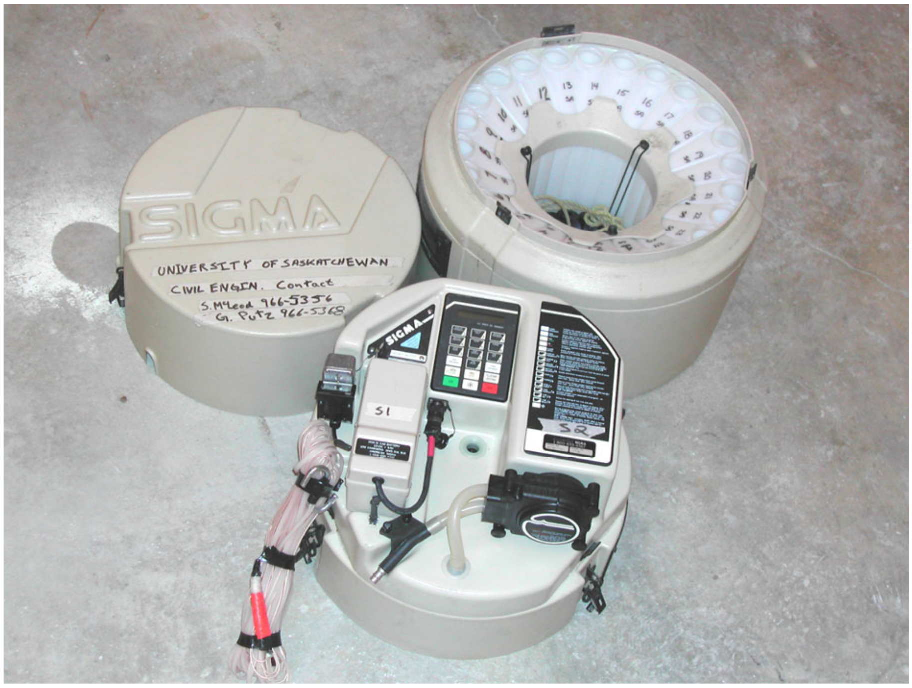


Figure 4.3 – American Sigma 800 SL sampler.

As noted previously, the ISCO 1680 and the two Sigma samplers require additional equipment to trigger (initiate) the sampling cycle. The sampling process is triggered (enabled) when water touches contacts on the end of a long wire suspended at a site specific depth within the manhole. After triggering, the trigger generates an electrical pulse, akin to a flow meter pulse, every 0.1 minutes. The sampler records the number of pulses and collects a sample after a predetermined number of pulses. The sample interval time is set by setting the number of pulses between samples. The trigger has an electronic timer that begins when the unit is triggered. The time that the sampler triggered is determined by subtracting the time on the trigger from the time when the samples are retrieved.

The triggers had problems. After several malfunctions, the problems were investigated with the assistance of the Engineering Shops electronics technician. Based on the investigations, modifications were made that significantly improved the reliability and functionality of the triggers. The problems and modifications are described further in Appendix C.

All of the samplers collect water samples using a peristaltic pump, which provides for large suction heads. All of the samplers use 10 mm PVC intake tubing with a weighted strainer on the end. The strainer prevents large particles from plugging the pump system. In addition, all of the samplers were outfitted with a suspension harness comprised of a large hook, chains, and linkage mechanisms, as shown in Figure 4.4.

Figure 4.4 – Typical suspension harness setup.

Significant difficulty was experienced with the strainers supplied with the samplers. The strainer on the left in Figure 4.5 was supplied with the ISCO 6700. The supplied strainers, especially those for the Sigma samples (not shown in the figure), are small and appear to be intended for low velocity flow such as lakes or small streams. In the higher velocity storm sewer flow, the small strainers are pushed to the surface by the force of the flow. With the strainer on the surface of the flow, a significant amount of air is drawn by the pump, which causes erratic water sample volumes. The difficulties were investigated using the flumes in the Hydrotechnical Laboratory at the U of S. Based on the investigation, new strainers were designed and manufactured (right most in Figure 4.5). The new strainers functioned substantially better than the previous strainers and the previous problems were alleviated. Further explanation is contained in Appendix C.

Figure 4.5 – Suction strainers used with samplers in successive redesigns (left to right).

4.1.2 Flow Monitoring

Flow depth data were collected using the add-on flow module for the ISCO 6700 sampler and by COS using custom built equipment. In both cases, the depth of flow was measured using an ultrasonic flow head and the data stored locally until being downloaded to a computer.

The depth measurement on the ISCO 6700 was calibrated on a regular basis to ensure accuracy of the data collected. On most occasions, the calibration was performed using a weighted measuring tape, measuring from the top of the manhole to the pipe invert and to the top of the flow. On the ISCO 6700, depths of flow were recorded at five minute intervals.

The COS equipment was operated and maintained by COS staff. COS staff collected the data from the depth monitor and provided raw data to the project. Datum was logged as a discrete measurement once every five minutes and downloaded from the logger once per week. In 2001, COS flow monitors were placed at Silverwood in May and at Taylor Street late in the season. During the 2001 season, COS staff experienced significant difficulty maintaining the monitors in an operational state, which resulted in the flow data being sporadic. In the 2002 season, the same two locations were instrumented by COS. The Sturgeon Drive catchment was intended to be instrumented by COS, but the Silverwood equipment and others in the COS's fleet malfunctioned and replacements did not arrive in time to be of use to the project. As a result, the Sturgeon Drive catchment was not instrumented in 2002 for flow monitoring and the Silverwood catchment was only partially instrumented with flow monitoring in 2002.

4.1.3 Rainfall Data

Rainfall data were provided by COS. COS maintains a small number of gauging stations throughout the city. The gauges are located at City Hall, Acadia Drive fire hall, Warman Road fire hall, and Diefenbaker Drive fire hall as indicated on Figure 3.1. The gauges are tipping bucket style and have a bucket capacity of 0.2 mm per tip. As with the flow monitors, COS staff had difficulty maintaining the rain gauges in an operational state and the data are sporadic.

4.2 SAMPLE COLLECTION AND RETRIEVAL

In preparation for sample collection, the water samplers were outfitted with a fully charged battery, a set of clean bottles, and suspended in the storm sewer manhole. The samplers began collecting samples when the water level in the pipe reached a predetermined depth, usually between 50 to 150 mm. The depths were initially selected on the basis of a small set of flow depth data collected early in the 2001 season and were refined as necessary to trigger as early in the event as possible, while minimizing false triggers.

In general, the sampling cycles were 30 to 140 minutes in duration, depending on the catchment and sampler. The cycles were based on hydrographs collected early in the project for each catchment. As soon as practical, the samples were recovered, which was generally within 10 to 12 hours after the beginning of the rainfall event. An example of sampler recovery is shown in Figure 4.6. Upon removal of the sampler from the manhole, the samples were capped for transport, removed from the sampler, and replaced with fresh bottles. The water samples were then returned to the U of S for pre-processing and packaging for shipment to the participating laboratories (discussed in Section 4.3) for analysis.

Figure 4.6 – Typical sampler recovery by Erin McCaig (left) and Kevin Sturgeon.

4.3 WATER QUALITY ANALYSIS

Several laboratories were utilized, to varying degrees, in the analysis of the water samples, including the U of S Environmental Engineering Lab, Saskatchewan Health Provincial Water Laboratory (Regina), COS Water Treatment Plant Lab, and the Saskatchewan Research Council (SRC) Analytical Laboratory. Analyses at the Provincial Lab and at the SRC Analytical Lab were provided by SE. Analyses at the other two labs were provided by their respective organizations.

Although several laboratories were utilized, the available laboratory resources were limited. Due to this limitation, a decision was taken early in the project to perform only a limited number of water quality analyses. It was decided that a few basic tests that are relatively easy to reliably conduct would be carried out on the discrete samples collected by the water quality samplers and that more comprehensive analyses would be performed on a composite sample made from the discrete samples as described in subsequent paragraphs.

The discrete samples were automatically collected by the samplers on a timed basis. Each of these samples represented the water quality at one point (or short interval) during a runoff event. The samples were collected at an interval of one to five minutes depending on the catchment. The discrete samples were intended to provide an indication of how the overall water quality changed during the runoff event. Discrete samples were analyzed for parameters that are reasonably quick and simple to assess, and are an indicator of overall water quality. Specifically, the discrete samples were analyzed for total suspended solids (TSS), conductivity, and pH. The sample temperature was also measured whenever it was possible to get to the site within a short time after the final sample had been withdrawn from the trunk sewer, which proved to be difficult to accomplish in most cases and therefore few events have a temperature profile. In the early part of 2001, the turbidity of the discrete samples was also measured. However, it became too time consuming to correctly perform the test and it was discontinued in July 2001. The difficulties were related to keeping the solids in suspension while the reading was being taken.

Composite samples were chosen for the majority of the water quality analyses because of the limited availability of laboratory resources. A composite sample also provided sufficient volume for performing a variety of water quality tests, including biological analyses and occasionally herbicide/pesticide scans and metals analyses.

In urban runoff sampling, flow weighted (proportioned) composite samples are generally preferred (EPA, 1982). Several difficulties are encountered in the collection of flow-weighted composite samples. Firstly, if the composite sample is created automatically by the sampler, a calibrated flow meter must be connected to the sampler to tell the sampler when to collect the sample. Alternatively, the flow weighted composite can be manually created by taking a volume proportionate to the flow rate (at the time when the sample was collected) from each of the timed samples. For both of these methods, the flow rate is required immediately to determine the composition of the composite sample. Only the ISCO 6700 has the capability to measure the flow rate. Discharge data from COS were supplied on a weekly or biweekly basis. Therefore, this method was not an option.

Secondly, flow-weighted composites are directed at a certain size of event, as they require a reference volume to determine when to collect the sample (i.e. after a specified volume of flow has passed, then collect a sample). In events somewhat smaller than the reference volume, the samples will not provide an accurate representation of the flow because the samples are too greatly spaced relative to the size of the event. In events somewhat larger than the reference volume, the number of discrete sample containers will be expended before the event is complete. Again, the samples will not provide an accurate representation of the runoff quality because, relative to the length of the event, the samples will be too tightly bunched near the beginning of the event. An appropriate reference volume can be chosen, however a substantial amount of data must be reviewed to avoid the aforementioned difficulties. Based on the foregoing discussion, time weighting was chosen as the method to create the composite samples.

Aliquots of equal volume were removed from each of the discrete samples and combined to create a time weighted event composite sample. Shih et al. (1994) conducted practical and theoretical comparisons of the two sample types and found that a time-composite sample comprised of eight or more samples produces an acceptable approximation of a flow weighted composite sample. They further concluded that a composite sample comprised of 30 timed samples shows no difference from a flow weighted composite sample. Therefore, for this work, a time weighted composite sample created using 24 or 28 samples was taken to provide a representative composite sample. This method of creating a composite sample is also supported by Smullen et al. (1999), who list the time weighted composite method as an alternative to flow weighted composite samples.

The composite samples were shipped to the Saskatchewan Health Provincial Water Laboratory in Regina, where a full spectrum of standard water quality tests, as listed in Table 4.1, was conducted. In 2001, selected samples were also collected and analyzed for herbicides and pesticides, as listed in Table 4.2, commonly found in Saskatchewan (as determined by the Provincial Lab). The composite samples were placed in coolers, packed with ice, and shipped to the Provincial Laboratory in Regina using the provincial inter-office mail system. The mail was collected twice daily from SE's office at Innovation Place, at approximately 9:00 a.m. and 2:00 p.m. In 2002, due to concerns over shipping delays, coliform analysis was provided by COS at the COS Water Treatment Plant Laboratory, where the analysis could be performed within the limited analysis window of 24 hours from sample collection.

Table 4.1 – Water quality parameters analyzed as part of the study.

Total Dissolved Solids	Total Hardness
Suspended Solids (fixed)	Chloride
Suspended Solids (volatile)	Potassium (ICP)
Suspended Solids (total)	Turbidity
Conductivity	Biochemical Oxygen Demand
Sulphate	Dissolved Organic Carbon
pH	Preserved Ammonia - N
Total Alkalinity	Nitrate - N
Bicarbonate	Total Kjeldahl Nitrogen
Sodium (ICP)[a]	Phosphorous (total)
Magnesium (ICP)	Phosphorous (ortho)
Calcium (ICP)	Chemical Oxygen Demand
Total coliform	Fecal coliform

[a] – inductively coupled plasma

Table 4.2 – Pesticides analyzed as part of the study.

Trifluralin	2,4-DB
Triallate	Dichlorprop
Diclofop	MCPA
Mecoprop (MCPP)	Dicamba
2,4-D	Bromoxynil
Trifluralin	

Throughout the program, selected samples were also analyzed for heavy metals content. This analysis was provided by SE at the SRC Analytical Laboratory in Saskatoon. The heavy metals included in the analysis are listed in Table 4.3.

Table 4.3 – Heavy metals analyzed as part of the study.

Mercury	Cobalt	Silver
Aluminum	Copper	Sodium
Antimony	Iron	Strontium
Arsenic	Lead	Tin
Barium	Manganese	Titanium
Beryllium	Molybdenum	Vanadium
Bismuth	Nickel	Zinc
Boron	Phosphorus	Zirconium
Cadmium	Potassium	
Calcium	Selenium	
Chromium	Silicon (soluble)	

Quality assurance and quality control (QA/QC) samples were employed to provide confidence in the water quality analysis results. In total, 12 QA/QC samples were submitted. These samples included blank samples (i.e. "pure" water), spiked samples (i.e. known concentration) and replicate samples (i.e. same sample submitted twice), which were sent to the Provincial Laboratory and COS laboratories. U of S equipment was routinely calibrated and checked against standard solutions. It has been suggested in the literature that the largest source of error when dealing with urban runoff samples is inadvertent sample switching in the laboratory (Kavelaars, 1998), likely due to the large numbers of samples handled. QA/QC samples and careful handling aid in the detection and control of this error. The results of the QA/QC work are reported in Section 5.4.2.

CHAPTER 5 DATA PRESENTATION AND ANALYSIS OF RESULTS

5.1 GENERAL DATA SYNOPSIS

The field sampling program generated samples acceptable for water quality analyses from 73 station events. A station event is one rainfall-runoff event at one monitoring station. Thus, in the case of this project, a rainfall covering the entire city, and consequently all four catchments, could produce as many as four station events. Table 5.1 provides a summary of the amount of data collected.

Table 5.1 – 2001 and 2002 collected data summary.

Location	Rainfall events	Events sampled	Quality data	Flow data	Useable events	Baseflow samples
Ave. B	49 (27[a])	16	15	10	9	11
Taylor	49	27	26	21	8	11
Silverwood	48	20	19	9	8	10
Sturgeon	48	10	6	4	2[b]	11
Total	194	73	66	44	27	43

[a] 2002
[b] Result only TSS

The "Rainfall events" column in Table 5.1 lists the number of rainfall events occurring during the monitoring period or the potential number of rainfall events from which data may have been collected. In the Avenue B catchment, only events from 2002 were useable because of equipment constraints. The "Events sampled" column represents the number of events for which water quality samples were collected and sent for analysis to the Provincial Lab in Regina. The "Quality data" column shows the number of events for which water quality results were returned from the Provincial Lab. The "Flow data" column shows the number of events for which flow rate data are available. "Useable events" are those events for which both water quality concentration data and flow rate data were available, thus permitting calculation of an event load. The result of two samples in the Sturgeon Drive catchment only pertains to TSS. The number of baseflow samples collected over the period is listed in the "Baseflow samples" column.

The low recovery rates evident in Table 5.1 are indicative of the difficulty experienced with the collection of samples and the coordination of multiple data sources. Further discussion of the data shown in Table 5.1 is combined with the discussion of Figures 5.1 to 5.4 that show the coverage of the flow rate and rainfall data for each of the catchments.

In 2001, sampling equipment was installed at the end of May and removed in late August. After the equipment was removed, a thorough cleaning and inspection were performed in preparation for the next season. In the 2002 season, sampling equipment was installed in early May and removed in late August. Gaps in the plotted data (primarily the flow data) of Figures 5.1 to 5.4 are caused by missing data due equipment malfunction.

Figure 5.1 shows the Avenue B catchment flow rate and the City Hall rainfall data. In the 2001 season, flow monitoring equipment was unavailable for the Avenue B catchment. Consequently, event loads were unable to be determined for 2001. In 2002, flow monitoring equipment was installed at the monitoring location allowing 67 days of flow data to be collected, 16 rainfall-runoff events to be sampled, and nine event loads to be determined. The poor rate of recovery (i.e. 9 of 16) is mainly due to flow monitoring equipment malfunction.

Figure 5.2 shows the Taylor Street catchment flow and the City Hall rainfall data for 2001 and 2002. The catchment had flow monitoring for 55 days in 2001 and 85 days in 2002. During the two seasons, a total of 27 rainfall-runoff events were sampled allowing eight event loads to be determined. The poor recovery rate (i.e. 8 of 27) is a result of sporadic malfunctions of both the flow monitoring and water quality sampling equipment.

Figure 5.3 shows the Silverwood catchment flow and the Warman Road firehall rainfall record for 2001 and 2002. The catchment had flow monitoring for 31 days in 2001 and 27 days in 2002. During the two seasons, a total of 20 rainfall-runoff events were sampled allowing for eight event loads to be determined. The COS flow monitor at this location proved especially difficult to maintain in an operational condition and was the cause of the minimal periods of available flow data. The large spike in the flow in Figure 5.3b was caused by a short duration intense thunderstorm that is not well represented by the available rainfall data.

Figure 5.4 shows the Sturgeon Drive catchment flow and the Warman Road firehall rainfall record for 2001 and 2002. The catchment had flow monitoring for 12 days in 2001 and 20 days in 2002. The availability of flow metering equipment for this location was a significant problem caused by malfunctions of the other monitors in the City's fleet. During the two seasons, a total of 10 events were sampled, but only two event loads were able to be determined. Errors in the creation of the composite samples early in 2001 rendered the composite sample water quality results unusable. Only the discretely measured TSS results are useable.

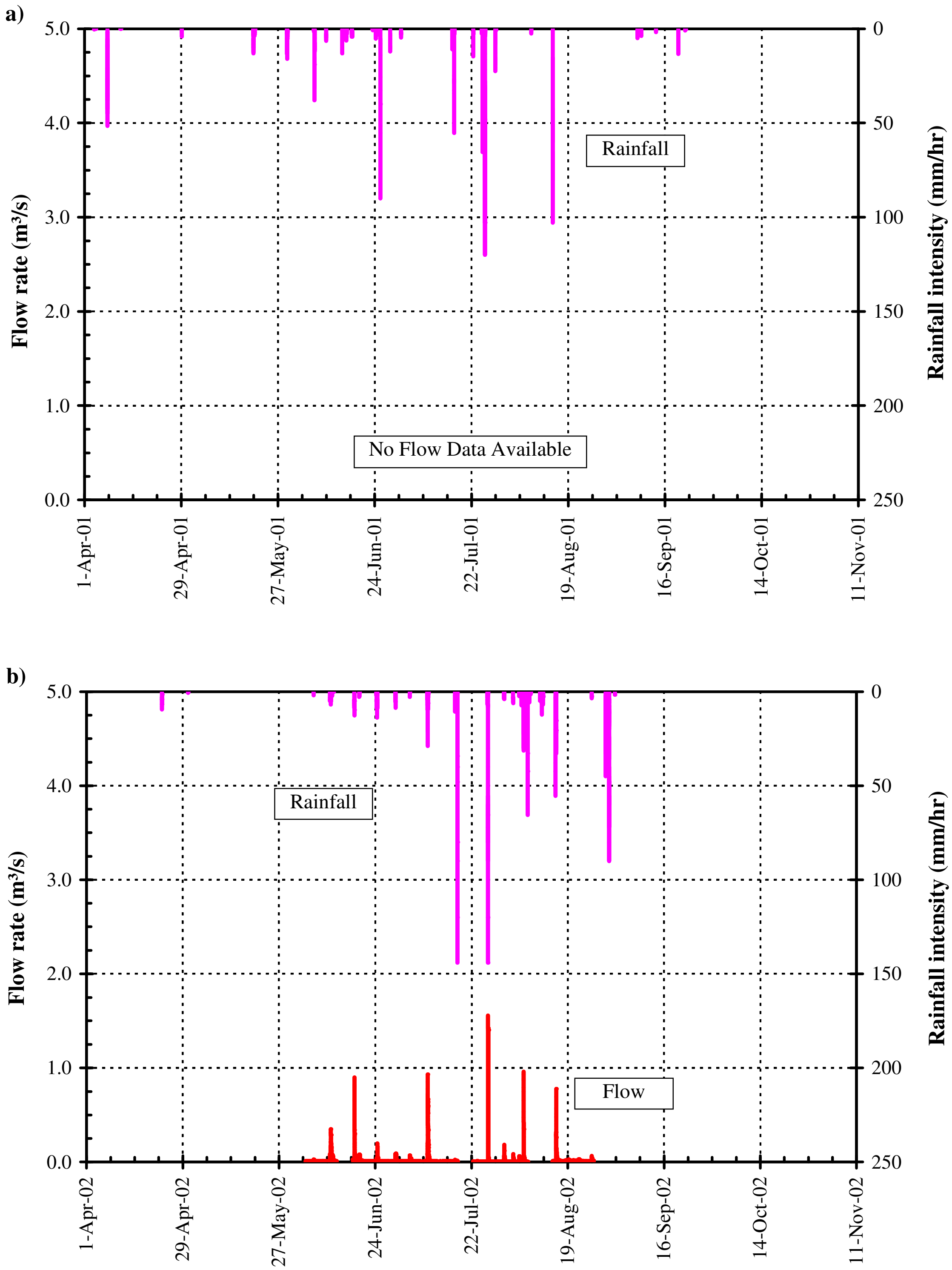


Figure 5.1 – Avenue B catchment flow and City Hall rainfall data: a) 2001; b) 2002.
(Data gaps were caused by equipment malfunctions.)

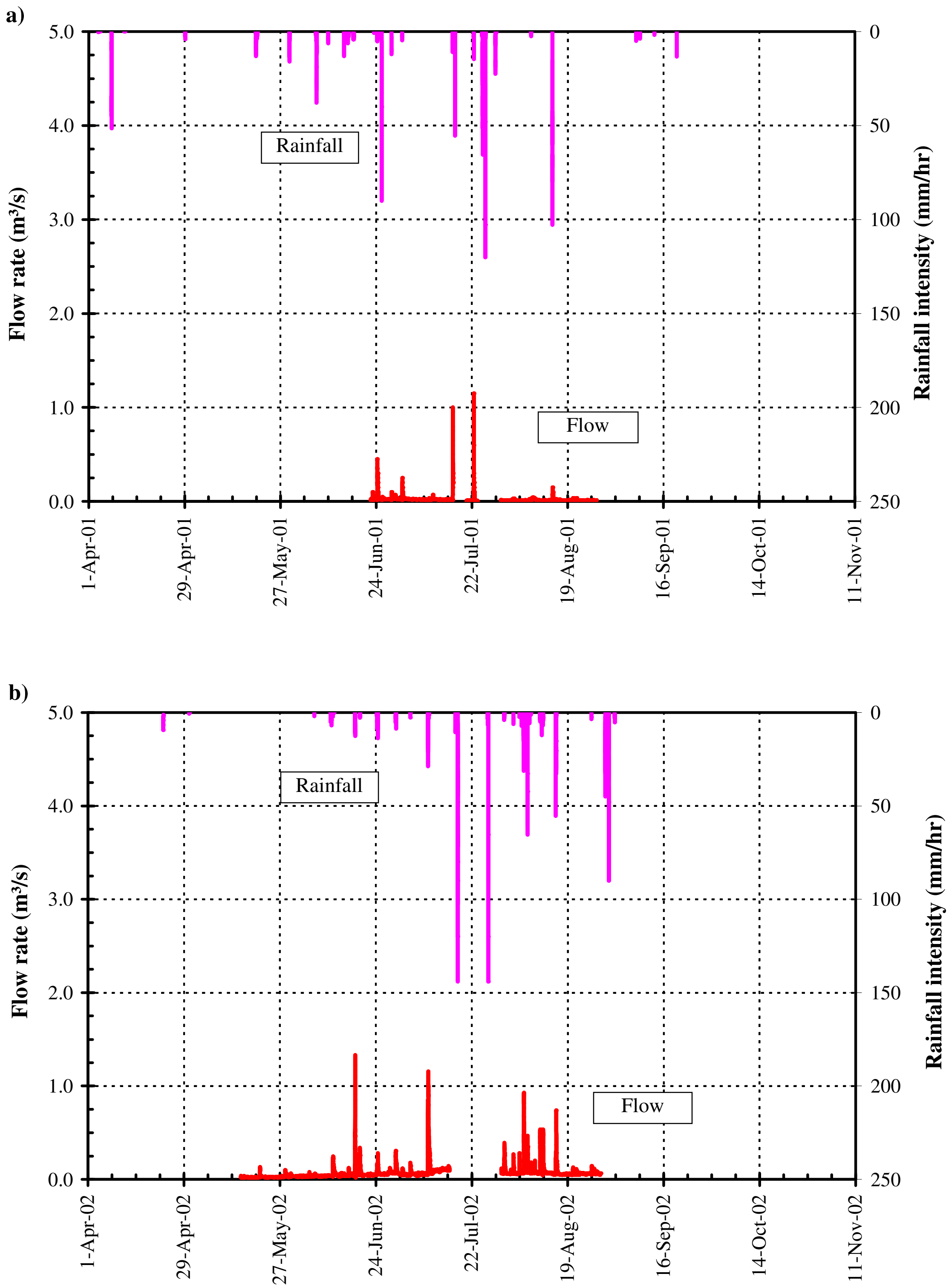
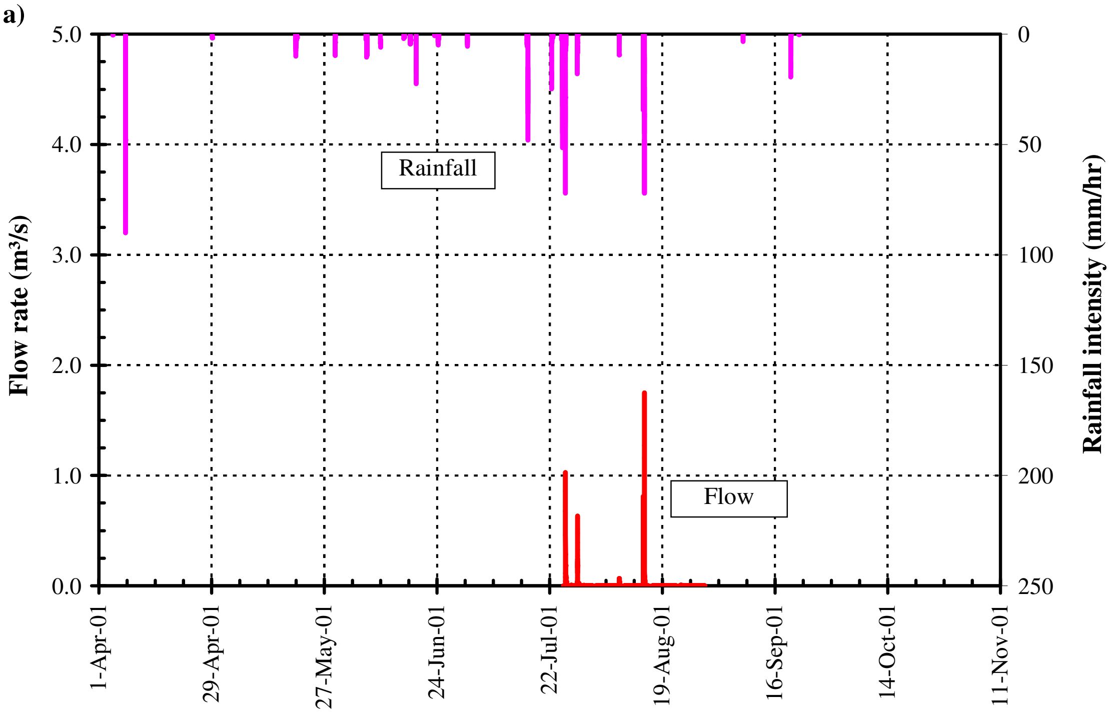


Figure 5.2 – Taylor Street catchment flow and City Hall rainfall data: a) 2001; b) 2002. (Data gaps were caused by equipment malfunctions.)

a)



b)

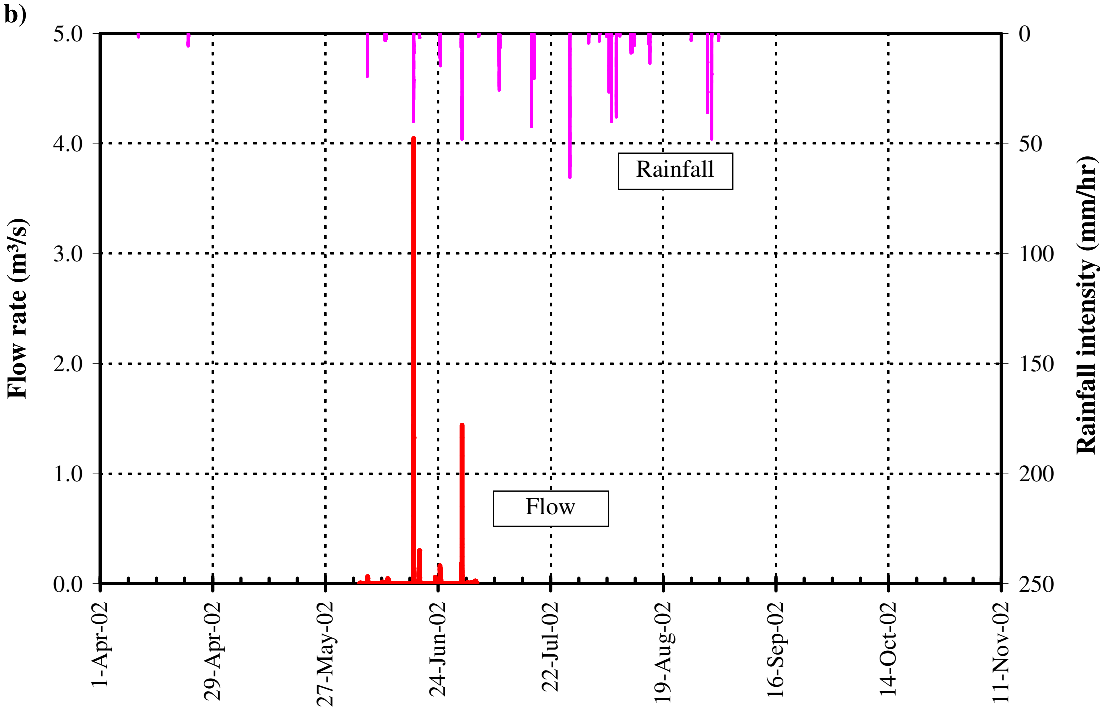


Figure 5.3 – Silverwood catchment flow and Warman Road fire hall rainfall data: a) 2001; b) 2002. (Data gaps were caused by equipment malfunctions.)

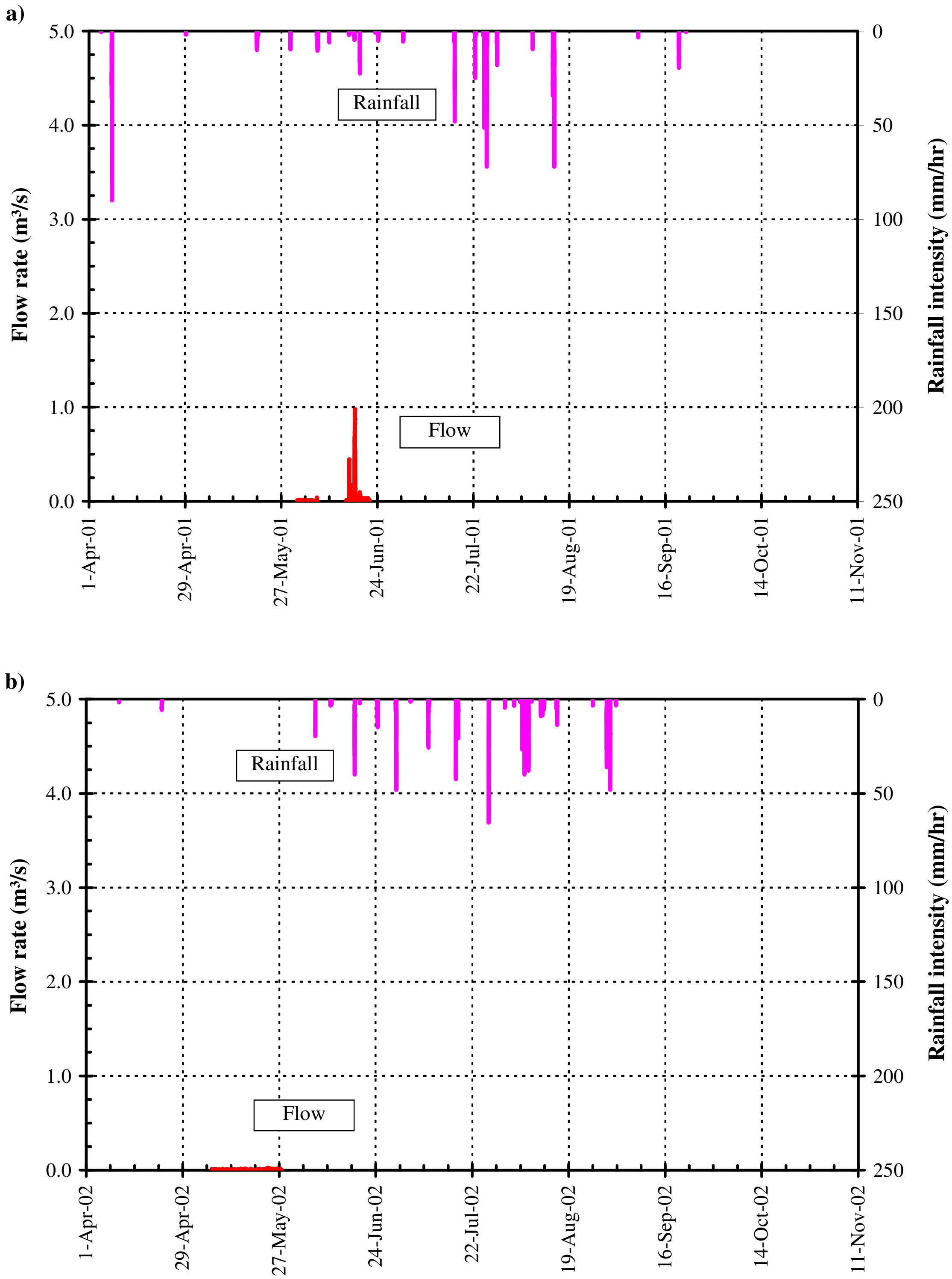


Figure 5.4 – Sturgeon Drive catchment flow and Warman Road fire hall rainfall data: a) 2001; b) 2002. (Data gaps were caused by equipment malfunctions.)

5.2 RAINFALL ANALYSIS

Raw rainfall data from each COS operated gauge, described in Section 4.1.3 and shown in Figure 3.1, were provided to the project by COS. Each datum point is comprised of the date and time when the tip occurred. The rainfall intensity is determined by dividing the depth represented by each tip by the time between subsequent tips, as shown by

$$\text{intensity}=\frac{\text{depth}}{\text{time}_i-\text{time}_{i-1}} \qquad [5.1]$$

where time_i is the time of tip i (hours), time_{i-1} is the time of the previous tip (hours) and depth is the depth of rainfall (mm) between tips, in this case 0.2 mm. When calculated from raw rainfall data, this intensity (mm/hr) is herein referred to as the instantaneous intensity. Other rainfall intensities can be computed using various forms of averaging of the same rainfall depth data.

The Thiessen polygon method was used to define the boundaries of the rain gauge service areas and, where necessary, to combine data from multiple gauges. The Thiessen polygon method defines the rain gauge service area based on equal distances between the gauges. Within a given catchment, the Thiessen weighting for a particular rain gauge is the ratio of the rain gauge service area within the catchment to the total area of the catchment. The Taylor Street and Sturgeon Drive catchments are split by the boundary between two polygons. Rainfall parameters for these two catchments were determined by combining data from two different gauges. The resulting polygon areas are illustrated in Figure 3.1 and the Thiessen weightings are summarized in Table 5.2. The remaining two catchments are completely within a single rain gauge service area.

Table 5.2 – Catchment area distribution and polygon weighting.

Catchment/	Rain gauge	Area (ha)	Weighting
Taylor	Acadia Drive	293	0.476
	City Hall	323	0.524
	Total	650	
Sturgeon	City Hall	25	0.059
	Warman Road	395	0.941
	Total	420	

Malfunctions of the gauges resulted in periods without data. In July 2002, the Acadia Drive gauge began to malfunction and data were unavailable for the rest of the year. Data from the Diefenbaker rain gauge were not available beginning in the first week of August 2002 through the remainder of the year. Data from the City Hall gauge were substituted for these two gauges.

A common time base is required to combine the data from multiple rain gauges. A five minute interval was chosen. A five minute interval is long enough to provide reasonable averaging, but is not so long that it masks significant changes in the intensity. A five minute time interval was used by Raymond (1997) in Saskatoon and Gromaire-Mertz (1999) in Paris, France in the processing of rainfall data with respect to urban runoff studies. The depth of rain falling in a given five minute interval was determined for each gauge by linearly proportioning the rainfall depth between gauge tips. The rainfall depth from each gauge was multiplied by the Thiessen polygon weighting and added to the total depth for the five minute interval for the catchment. The intensity over the interval was then calculated using [5.1].

Parameters such as the inter-event time and minimum rainfall depth are necessary to distinguish individual events. The minimum storm depth chosen for this project is 0.6 mm. Adams et al. (1986) suggest a minimum of 0.5 mm based on experience in the Vancouver area, while NURP (EPA, 1983) suggests a minimum of 2.5 mm (0.1 inches). The NURP minimum is recommended to municipalities because of the NURP conclusion that significant washoff does not occur until 2.5 mm of rainfall depth has fallen. However, that conclusion was made for the United States, where the majority of the area receives significantly more annual rainfall than Saskatoon (Driver and Tasker, 1990; Environment Canada, 2001). Due to the different annual rainfall amounts, different amounts of washoff are likely to occur, and therefore the depth of rainfall required before significant washoff begins is not known. The minimum rainfall depth of 0.6 mm represents three tips of the 0.2 mm rain gauge, which eliminates very small events that may or may not produce runoff and possible erroneous tips of the gauge. Events that did not meet the minimum event requirements were removed from the dataset.

The IETD establishes the minimum time required to elapse between gauge tips to consider the tips as part of separate rainfall events. The IETD for the project was determined based on data from the Acadia Drive rain gauge using al of the available 2001 and 2002 data and the approximate graphing method proposed by Adams et al. (1986). The plot, shown in Figure 5.5, was generated by varying the inter-event time from 30 minutes to 12 hours and plotting the number of rainfall events corresponding to each inter-event time. Shown for comparison in the figure is the same relationship for Regina, which has been estimated from Adams and Papa (2000).

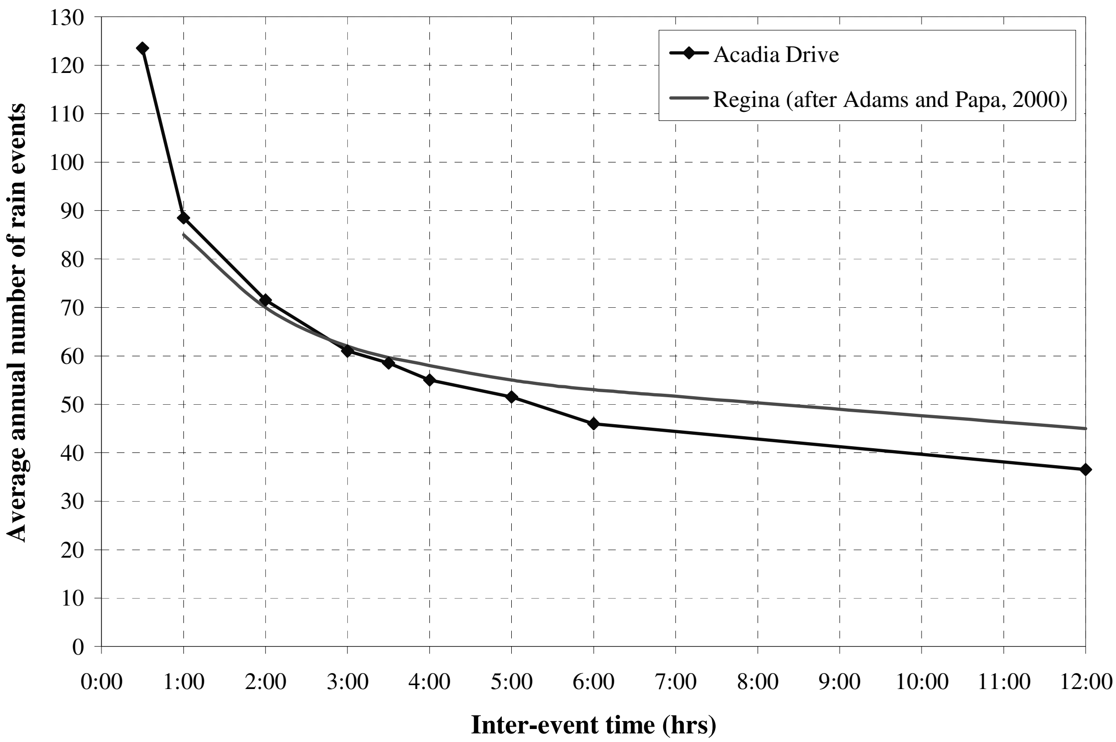


Figure 5.5 – IETD selection plot and comparison to Regina data.

Using the approximate graphing method, the IETD is the inter-event time where the slope of the curve begins to stabilize. In Figure 5.5, stabilization begins to occur at about three hours and continues for the rest of the plot. The slightly larger reduction between five and six hours occurs because of the limited data for the very dry 2001 season. As shown in Figure 5.5, the IETD plot determined using the Acadia Drive gauge data agrees well with the plot for Regina. Adams and Papa (2000) suggest that the choice of IETD must also be related to the size of the catchment and the intended use of the data. Larger catchments take longer to respond to rainfall at the hydraulically far end of the pipe network (i.e. longer time of concentration), thus indicating a longer time between the rainfall events is required for the event observations (mainly flow rate) to be independent of previous events. When the intended use of the data is large scale estimates, a longer IETD is also suggested. In the case of this work, the catchments are generally substantial in size and the intended use of the data is that of planning level estimates of load. Both of these factors indicate that a longer IETD is appropriate. For this work, the IETD was chosen to be five hours, which fits within the guidelines for IETD's suggested by Adams and Papa (2000) of one to six hours.

Several rainfall parameters can be used to describe the rainfall. These parameters are event average intensity (mm/hr), antecedent dry period (days), event duration (minutes), maximum five minute average intensity (mm/hr), and maximum instantaneous intensity (mm/hr). The average intensity is the total rainfall depth divided by the duration of the rainfall event. The antecedent dry period is the time elapsed between the last tip of the previous event and the first tip of the current event. The event duration is the elapsed time from the first tip to the last tip of the event. The maximum five minute average intensity is the maximum intensity that occurred in an event determined when a five minute averaging period is used. The maximum instantaneous intensity is the maximum intensity calculated from the raw data recorded by the rain gauge. Table 5.3 presents a summary of the rainfall parameters for 2001 and 2002. Further rainfall data are found in Appendix D.

The Acadia Drive gauge had the longest average duration, the longest average antecedent dry period, the largest average depth, the smallest average intensity, and the smallest average maximum instantaneous intensity. The largest maximum intensity was from the City Hall rain gauge. In general, the City Hall, Diefenbaker and Warman Road gauges had similar rainfall parameters. The minimum duration from the Acadia Drive rain gauge of 36 min. is a result of applying the minimum rainfall requirements and a sporadically functioning gauge.

Table 5.3 – Summary of rainfall parameters for each catchment for 2001 and 2002.

Rain Gauge		Duration (min)	Antecedent period (days)	Depth (mm)	Average intensity (mm/hr)	Maximum instantaneous intensity (mm/hr)	Maximum five minute average intensity
Acadia Drive	Average	407	7.7	4.0	0.6	5	3.35
	Max	1727	38	22	1.6	60	30.7
	Min	36	0.59	0.60	0.09	0.09	0.09
City Hall	Average	278	5.4	5.7	2.9	24	11.5
	Max	1436	36	32	54	144	71.3
	Min	3.5	0.22	0.60	0.08	0.08	0.08
Diefenbaker	Average	240	6.3	5.0	2.7	18	9.26
	Max	1592	44	26	17	120	59.7
	Min	3.5	0.23	0.60	0.20	0.91	0.91
Warman Road	Average	211	6.3	5.2	3.7	20	10.8
	Max	1362	44	23	32	90	33.9
	Min	3.4	0.25	0.60	0.21	0.38	0.38

Environment Canada reports data for three rain gauges within Saskatoon. The gauges are located at the Saskatoon airport, the Saskatchewan Research Council (SRC) at Innovation Place, and the Saskatoon Water Treatment Plant (WTP). Table 5.4 presents the thirty year annual Normal rainfall depths for each gauge and the depth of rainfall recorded at each of the gauges in 2001 and 2002. The year 2001 was the driest year in over one hundred years of rainfall record (Environment Canada, 2003). The average rainfall in 2001 was 54.3% of the normal rainfall, while 2002 had nearly normal rainfall (110% of normal).

Table 5.4 – Rainfall Data 2001 and 2002 (Environment Canada, 2003).

Gauge Location	Normals 1971-2000 (mm)	Annual Rainfall 2001 (mm)	2002 (mm)
Airport	265.2	135.9	259.5
WTP	280.4	160.9	341.6
SRC	252.3	136.6	276.2
Average	266.0	144.5	292.4

5.3 HYDRAULIC ANALYSIS

The depth of flow at each storm sewer flow monitoring station was determined using ultrasonic depth sounding equipment. The time and depth of flow were recorded by a data logger at five minute intervals. Location specific stage discharge curves were developed to convert the recorded depths of flow to volumetric discharge or flow rate. Manning's equation [5.2] was chosen to generate the rating curves, and can be stated as

[5.2] $$Q = \frac{1}{n} A R^{2/3} S^{1/2}$$

where Q is the volumetric flow rate (m^3/s), n is Manning's roughness coefficient, A is the cross-sectional area of the flow (m^2), R is the hydraulic radius (m) and S is the slope of the channel (m/m). The hydraulic radius is the cross-sectional area (A) of the flow divided by the wetted perimeter (P). The area and the wetted parameter are determined by geometric relations of the pipe and the depth of flow, which are shown in Appendix E.

Insitu roughness measurements were attempted at each site using a salt tracer. All salt tracer tests were performed only for baseflow conditions. Based upon the dilution of the salt tracer, the flow rate in the storm sewer was calculated. During the tracer test, the depth of the flow in the storm sewer was measured and recorded. Therefore, all of the variables in Manning's equation were known except for the roughness. The variables in Manning's equation were rearranged and the roughness was determined. The details of each monitoring location and determination of the rating curve are discussed in Appendix E. The rating curves for Avenue B, Taylor Street (U of S, and COS monitors), Silverwood, and Sturgeon Drive catchments are presented in Figures 5.6 to 5.10, respectively. The COS flow monitor in the Taylor Street catchment was installed in a different location than the U of S flow monitor, thus two rating curves are required. The complete set of flow depth data are contained on the data CD in Appendix L.

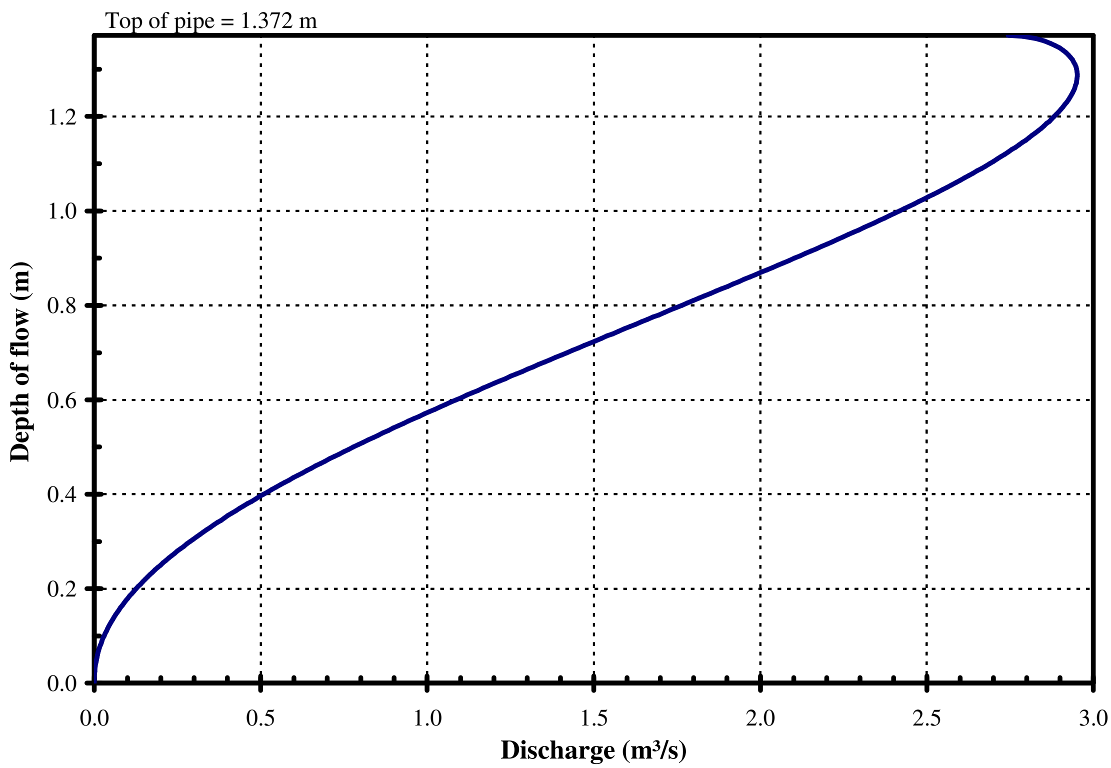


Figure 5.6 – Avenue B stage-discharge curve.

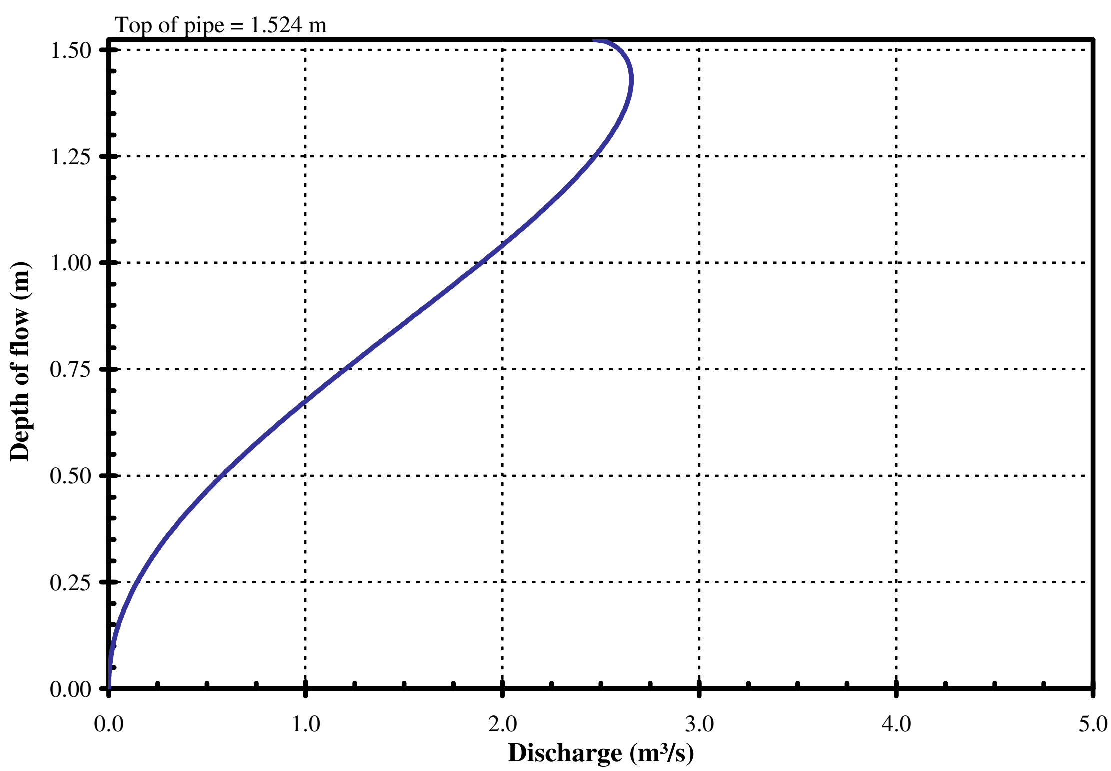


Figure 5.7 – Taylor Street catchment COS flow monitor stage - discharge curve.

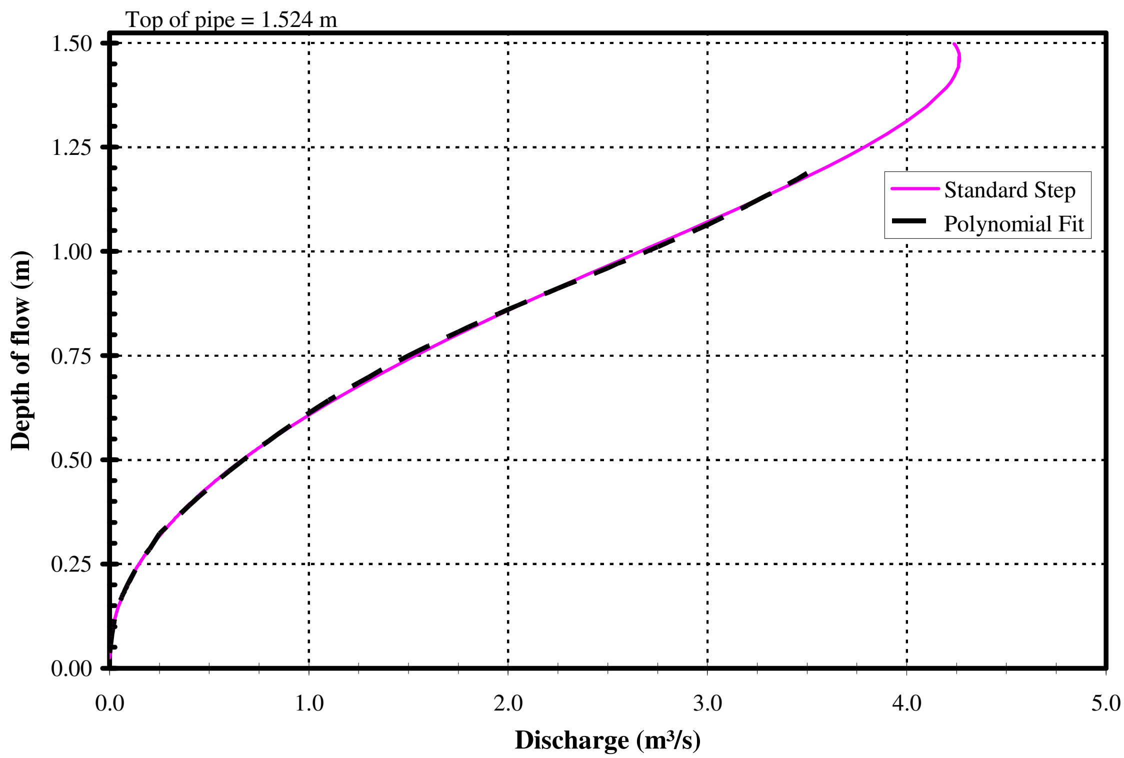


Figure 5.8 – Taylor Street catchment, U of S flow monitor stage-discharge curve.

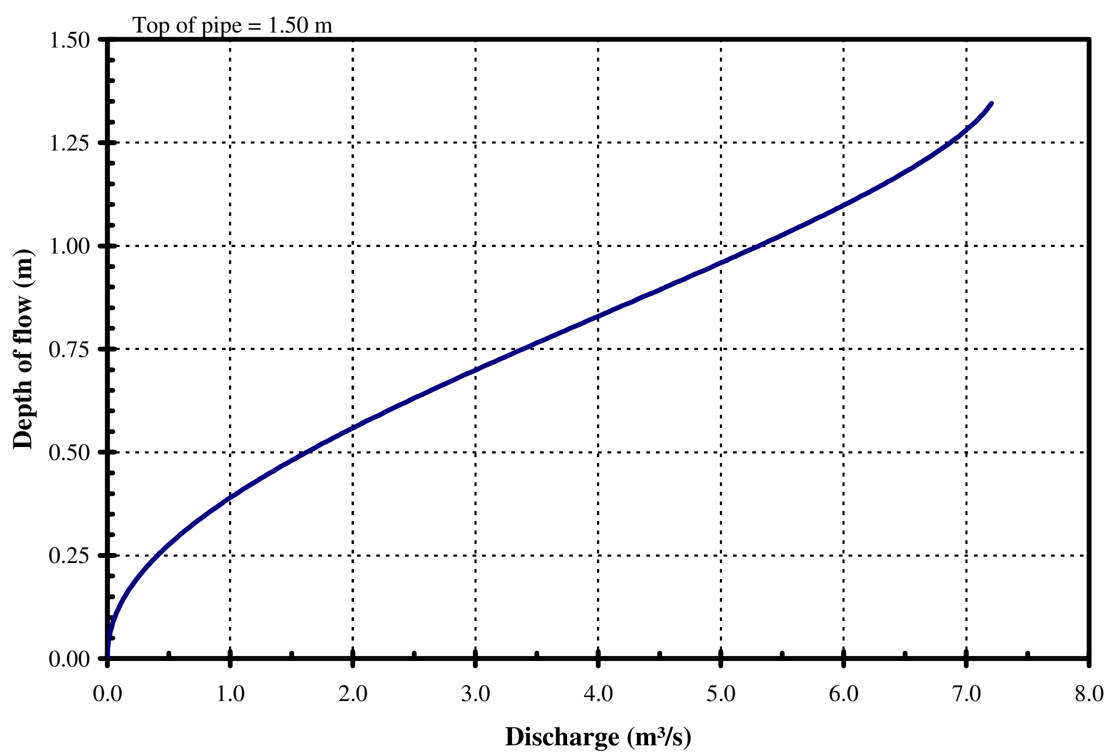


Figure 5.9 – Silverwood catchment stage-discharge curve.

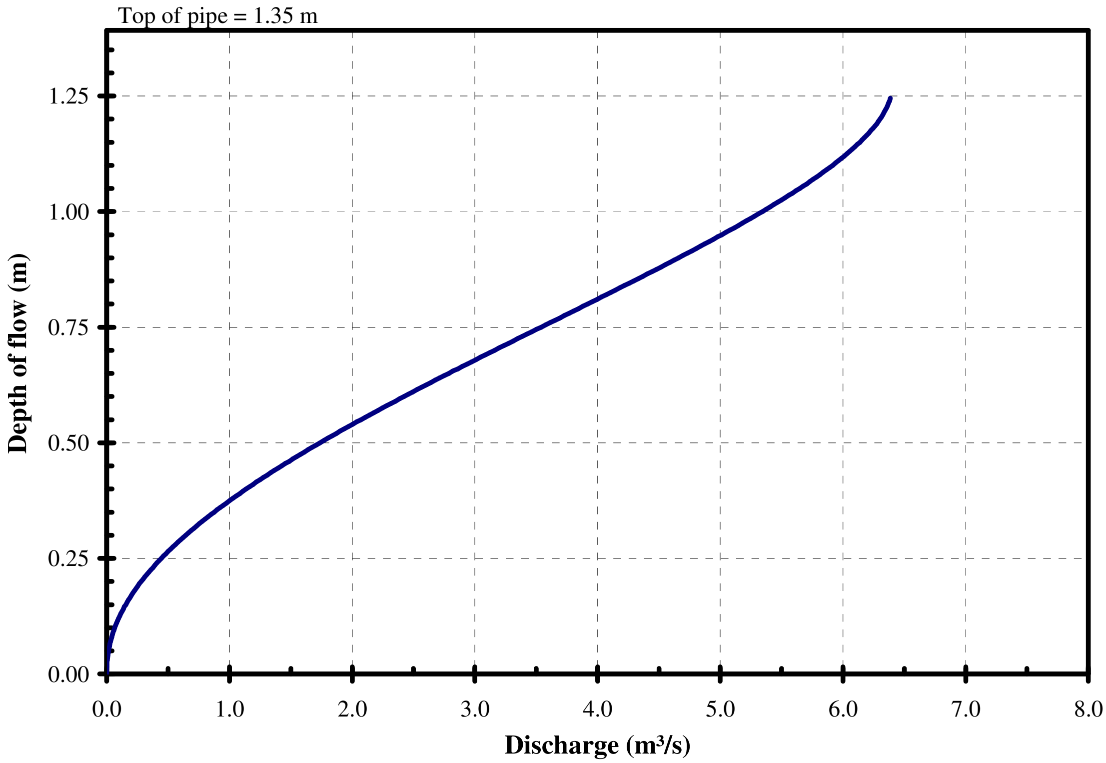


Figure 5.10 – Sturgeon Drive catchment stage-discharge curve.

5.4 WATER QUALITY PARAMETER CONCENTRATION RESULTS

5.4.1 Event Composite and Baseflow Sample Concentration

For most rainfall events that were sampled over the two years of the study, 27 water quality parameters were analyzed by the Provincial Lab. The ranges of the measured concentrations are summarized in Table 5.5. Events that did not meet the minimum rainfall requirements are not included in Table 5.5, however events for which flow data were unavailable are included. A measure of central tendency of the concentrations in Table 5.5 is not shown, as the most appropriate measure of central tendency for the composite samples is a weighted average concentration, or SMC, which is presented in Section 5.6.2. The numbers of samples represented are listed adjacent to the location.

Many of the parameters summarized in Table 5.5 exhibit large variations. For example, the total TSS results range over almost three orders of magnitude, from 3 to 1300 mg/L. In contrast, the range of total phosphorous (TP) was from 0.17 to 3.3 mg/L.

The pH results range from 6.6 to 8.3 and indicate that the pH tended to remain somewhat balanced at about pH 7 or neutral pH.

Table 5.5 – Ranges of rainfall-runoff event composite sample water quality parameter concentrations in the two study seasons (mg/L).

Parameters	Avenue B (15)[e]		Taylor (26)[e]		Silverwood (19)[e]		Sturgeon (6)[e]	
	Max	Min	Max	Min	Max	Min	Max	Min
TSS (fixed)	790	93	610	3	1100	12	400	2
TSS (volatile)	190	26	140	4	190	5	120	2
TSS (total)	970	120	740	8	1300	17	500	3
Conductivity	1000	130	1200	99	550	83	2200	440
pH	7.4	6.8	7.6	6.7	7.1	6.6	8.3	6.9
Turbidity	570	54	360	6.7	520	6.8	98	0.72
Bicarbonate	210	44	220	39	170	32	420	110
Total Alkalinity[a]	170	36	180	32	140	6	340	88
Calcium	75	13	120	10	51	9	260	46
Magnesium	38	3	57	2	14	2	130	14
Sodium	83	5	76	3	45	2	100	16
Hardness (Calc)[a]	340	45	540	33	190	31	1200	190
Chloride	78	4	100	4	40	2	76	8
Potassium	13	2	10	2	10	1	11	3
Sulphate	240	19	350	11	100	13	820	96
$NH_3 - N$	2.8	0.12	1.3	0.03	1.7	0.09	1.2	0.33
$NO_3 - N$	1.1	0.02	1.8	0.03	1.5	0.25	3.9	0.96
TKN	15	1.8	16	1.1	21	0.7	6.7	1.4
TP	3.3	0.56	1.6	0.27	2.7	0.24	0.88	0.17
OP	0.63	0.05	0.56	0.11	0.74	0.08	0.11	0.04
DOC[b]	68	7	53	6	98	8	66	13
BOD_5[c]	58	4.2	39	3.6	40	2.4	34	14
COD[c]	820	35	360	29	730	15	510	32
TDS (Calc)	740	71	870	67	430	53	1800	300
TC[d]	$>2.4x10^6$	$8.6x10^4$	$1.2x10^7$	$1.5x10^4$	$>2.4x10^6$	$8.6x10^4$	$1.1x10^6$	$7.5x10^3$
FC[d]	$2.3x10^4$	$4.0x10^1$	$9.5x10^4$	$9.0x10^1$	$2.3x10^4$	$4.0x10^1$	$3.0x10^3$	$4.0x10^1$

[a] mg/L as $CaCO_3$
[b] Dissolved Organic Carbon
[c] mg/L as O_2
[d] orgs/100 mL
[e] number of events represented

TC and FC also have large ranges. The TC range spans from 7,500 to 12,000,000 orgs/100 mL. FC ranges from 40 to 95,000 orgs/100 mL. It is notable that both TC's and FC's were detected in all of the rainfall-runoff event composite samples.

Table 5.6 presents a summary of the water quality parameter concentrations for the baseflow samples collected from all of the catchments between April 1 to October 31 of 2001 and 2002. The averages presented in the table were determined using the geometric mean. The number of baseflow samples in each catchment is shown in the Baseflow samples column of Table 5.1. Table 5.6 shows significantly smaller ranges for most parameters compared with Table 5.5. For example, the largest range of TSS in the baseflow samples is 2 to 26 mg/L, approximately one order of magnitude, for the Avenue B catchment, whereas from Table 5.5 the smallest range is 3 to 500 mg/L, approximately two orders of magnitude, for the Sturgeon Drive catchment.

TC results in the baseflow range from 90 orgs/100 mL to greater than 2,400,000 orgs/100 mL. Average TC concentrations in the baseflow range from 12,000 (Sturgeon) to 260,000 orgs/100 mL (Ave. B). The FC results ranged from less than the detection limit of 10 orgs/100 mL to 430,000 orgs/100 mL, with catchment averages ranging from 96 (Sturgeon) to 49,000 orgs/100 mL (Ave. B). FC were detected in all baseflow samples from the Taylor Street and Avenue B catchments, which indicates that there was a continuous source of fecal matter.

Table 5.6 – Summary of baseflow sample concentrations in the two study seasons.

Parameters	Taylor Street			Silverwood		
	Average	Max	Min	Average	Max	Min
TSS (fixed)	2.2	3	1	1.9	3	1
TSS (volatile)	1.5	2	1	1.0	1	1
TSS (total)	4.1	5	4	3.1	4	2
Conductivity	1800	2600	1100	2000	2700	1400
pH	8.0	8.1	7.7	7.8	8.2	6.7
Turbidity	1.9	3.3	1.1	1.2	2.0	0.62
Bicarbonate	330	480	190	310	390	270
Total Alkalinity[a]	270	390	160	260	320	220
Calcium	200	300	100	240	360	150
Magnesium	96	150	54	110	180	74
Sodium	90	130	57	78	110	55
Hardness (Calc)[a]	900	1400	480	1100	1600	670
Chloride	75	110	32	48	55	38
Potassium	8.6	10	6	11	14	9
Sulphate	640	1000	290	820	1300	490
NH_3 – N	0.20	0.74	0.06	0.07	0.14	0.03
NO_3 – N	4.0	6.7	1.5	10	16	6.3
TKN	1.6	5.1	0.8	1.4	2.1	0.9
TP	0.24	0.47	0.11	0.25	0.62	0.11
OP	0.19	0.42	0.07	0.23	0.54	0.1
DOC[b]	13	22	8	15	20	12
BOD_5 (mg/L as O_2)	1.0	1.3	0.7	0.4	0.5	0.2
COD (mg/L as O_2)	31	53	22	35	41	19
TDS (Calc)	1500	2200	820	1600	2400	1100
TC (orgs/100 mL)	6.7×10^4 [c]	2.3×10^5	8.2×10^3	1.3×10^3 [c]	3.1×10^4	2.3×10^3
FC (orgs/100 mL)	3.4×10^3 [c]	1.5×10^4	2.3×10^2	1.4×10^2 [c]	2.2×10^3	<30

[a] mg/L as $CaCO_3$
[b] Dissolved Organic Carbon
[c] median

Table 5.6 – Summary of baseflow sample concentrations in the two study seasons (continued).

Parameters	Avenue B			Sturgeon Drive		
	Average	Max	Min	Average	Max	Min
TSS (fixed)	3.9	18	1	1.4	3	1
TSS (volatile)	2.8	8	1	1.6	2	1
TSS (total)	7.6	26	3	3.0	5	2
Conductivity	2000	2400	1500	1800	2400	1200
pH	7.9	8.3	7.6	7.8	8.1	6.8
Turbidity	4.1	16	1.7	1.3	3.8	0.36
Bicarbonate	330	400	280	360	480	230
Total Alkalinity[a]	270	330	230	300	390	190
Calcium	160	220	110	220	310	130
Magnesium	110	150	66	100	160	56
Sodium	140	170	120	81	110	57
Hardness (Calc)[a]	840	1200	550	970	1400	550
Chloride	110	160	90	59	78	37
Potassium	12	14	11	9.6	12	7
Sulphate	670	950	420	710	1100	400
NH_3 – N	0.78	8.4	0.09	0.06	0.15	0.04
NO_3 – N	3.2	6.8	0.86	3.8	6.2	2.03
TKN	2.5	10.9	0.8	1.0	1.2	0.8
TP	0.38	1.5	0.14	0.15	0.24	0.1
OP	0.22	0.8	0.12	0.13	0.21	0.1
DOC[b]	14	30	9	15	21	12
BOD_5 (mg/L as O_2)	1.9	11	0.5	1.2	1.7	0.9
COD (mg/L as O_2)	39	88	22	41	61	29
TDS (Calc)	1500	2100	1100	1500	2200	920
TC (orgs/100 mL)	2.6×10^5 [c]	$>2.4 \times 10^6$	1.5×10^4	1.2×10^4 [c]	9.8×10^4	9.0×10^1
FC (orgs/100 mL)	4.9×10^4 [c]	4.3×10^5	4.3×10^3	9.6×10^1 [c]	4.3×10^3	<10

[a] mg/L as $CaCO_3$
[b] Dissolved Organic Carbon
[c] median

The TSS concentration was determined for each discrete sample. Table 5.7 summarizes the TSS concentration ranges for the discrete samples. The minimum TSS concentration in the discrete samples was 1 mg/L, while the maximum was nearly 6,000 mg/L. As expected, the discrete sample TSS concentrations show larger ranges than the event composite samples. Discrete TSS results were used in further analyses instead of the composite TSS results because the discrete results provide a better representation of the intra-event variability.

Table 5.7 – Range of TSS concentrations (mg/L) of discrete rainfall-runoff event samples.

Parameter	Avenue B		Taylor		Silverwood		Sturgeon	
	Max	Min	Max	Min	Max	Min	Max	Min
TSS (total)	2,300	100	6,000	1	3,700	2	980	5

The remainder of the discretely measured parameters, (e.g. conductivity, temperature, etc.) is shown in the field data reports contained on the data CD in Appendix L.

5.4.2 QA/QC Results and Analysis

Three types of QA/QC samples were used as part of the laboratory testing protocol: blank samples, spiked samples, and replicate (duplicate) samples. A total of 12 QA/QC samples were prepared during the two field seasons. The breakdown is two blank, two spiked, and eight duplicate samples. The specific laboratory analysis results and discussion thereof are provided in Appendix F.

Generally, the water quality QA/QC results provide an acceptable level of confidence in the water quality parameter concentration results. One set of duplicate samples (June 19, 2002) appear to have something wrong, because the results are somewhat different. A poor TKN result in one of the blank samples (July 9, 2002) appears to be an anomaly, as the remainder of the samples show good agreement with their respective comparisons. The same situation is true for chloride as well. The variations from the expected results in all other QA/QC samples are able to be explained. Utilizing the COS Water Treatment Plant lab for biological parameter analyses corrected the delay problems and provided good results. Marsalek (1991) states that the level of precision required of the water quality parameter analyses is dependent upon the usage of the data. In the case of stormwater, which has many other vagaries (Novotny, 1992), the precision need not be high.

5.5 EVENT LOAD DETERMINATION

The event load is the total mass of water quality parameter discharged during a rainfall-runoff event. An example of the data reduction process is illustrated in Figure 5.11 for a rainfall-runoff event taken from the Taylor Street catchment, which began the evening of July 17, 2002. The general process is described for discretely measured TSS and then extended to composite sample parameters.

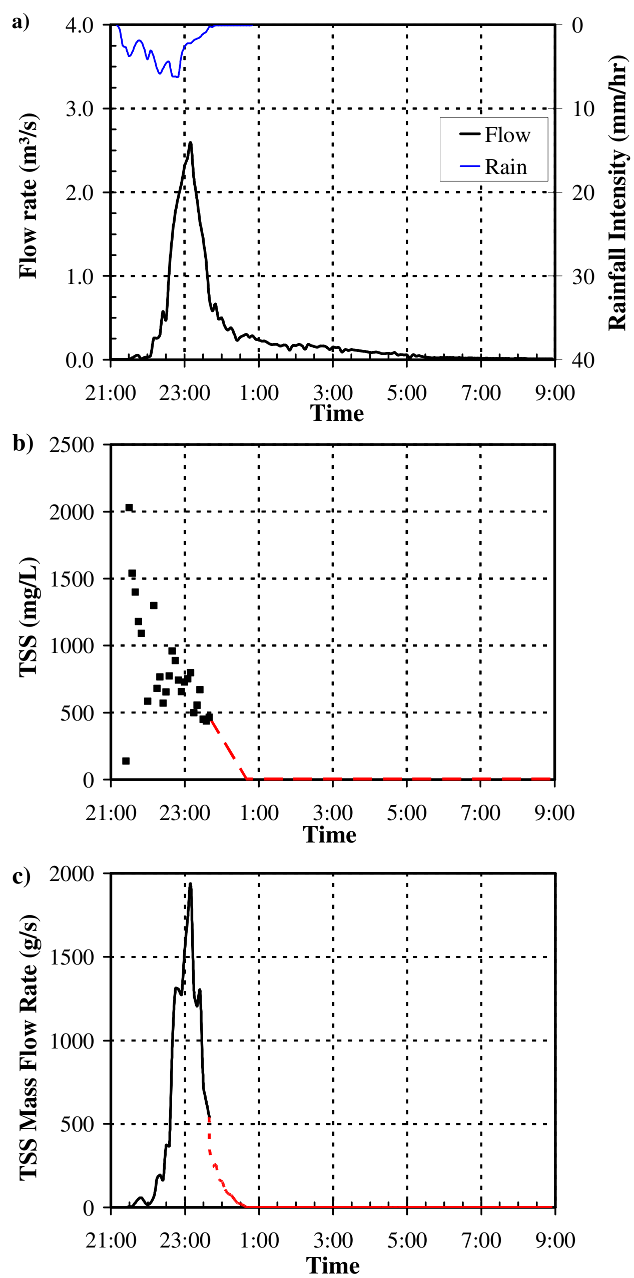


Figure 5.11 – Illustration of data reduction (Taylor Street catchment, beginning June 17, 2002): a) hyeto/hydrograph; b) pollutant concentration vs. time (pollutograph); and c) mass flow rate vs. time (loadograph).

The flow depth with time data are converted to volumetric flow rate (or discharge) with time using the appropriate stage discharge curve. The hydrograph for the example event is shown in Figure 5.11a. The total volume of runoff discharged during an event is the area under the curve (i.e. integration of the curve). A plot of the discrete TSS concentrations with time (pollutograph) is shown in Figure 5.11b. The combination of the hydrograph and the pollutograph yields a plot of mass flow rate with time (loadograph). The loadograph for the example event is shown in Figure 5.11c. The integration of the loadograph curve yields the total pollutant mass discharged during the rainfall-runoff event (event load). The event hyetograph, hydrograph, pollutograph (TSS only) and loadograph (TSS only) for each useable event are presented in Appendix G.

The example event, Figure 5.11, illustrates a common problem in which the entire event is not completely sampled. The lack of complete sampling is due to the limited number of samples (24 or 28) that could be drawn by the water quality sampler during an event. For the case shown, TSS concentrations during the declining leg of the hydrograph were estimated using an extrapolation based on the observed rate of decrease in the TSS concentrations at the end of the sampling interval. The extrapolation is shown in Figure 5.11 (and all of the event graphs) as the dashed line. The extrapolation was terminated at the average baseflow concentration. In the case of the Taylor Street catchment, the average baseflow TSS concentration is 4.1 mg/L. From that point on through the remainder of the event, the average baseflow concentration and the observed flow rates were used to calculate the event load.

The loads for all other parameters were determined based on the composite sample analysis. The composite sample concentrations (time weighted average concentrations) were applied to the volume discharged during the sampling window to calculate the total load. Extrapolation of the concentrations in the declining leg, as was done for TSS, was not made for composite sample parameters because the behaviour of the concentration profile in the declining leg was not known. When calculated in this manner, the loads determined using the composite sample concentrations are most likely underestimated because some load would invariably be discharged after the sampling was complete. However, as illustrated by the TSS results in Figure 5.11c, the majority of the pollutant load is discharged early in the event and generally within the sampling interval. Thus, a relatively small amount of pollutant load is missed in the declining leg. In the case of the example event, an estimated 94% of the TSS mass is represented by the discrete samples. The remaining six percent were estimated as shown by the dashed line in

Figures 5.11b and 5.11c. Approximately 64% of the event volume is represented in the example event by the water quality samples.

Table 5.8 presents a summary of the relative proportions of the useable events that were sampled, relative to both the volume discharged and the estimated total TSS load based on the discrete samples. Based on the event volume, the relative coverage ranged from 0.9 to 89.2% with averages from 31.6% (Ave. B) to 49.5% (Taylor). However, in the estimation of event loads, the relative amount of the estimated total mass that was sampled is more important. The relative TSS load sampled ranges from 15.5% to 99.5%, with averages from 61.8% to 81.6%. The relative coverage for each event can be inferred from the event graphs in Appendix G and is listed numerically in Appendix H. Both the event graphs and numerical representations indicate that some of the events were poorly sampled. The effect of a poorly covered event is to reduce the total event load, which in turn causes the characterization to be smaller. Ideally these events would be excluded from the data set, however they are used because of the small data set.

Table 5.8 – Average percentage of rainfall-runoff events sampled.

Catchment	% of volume sampled		% of total TSS load [a] sampled	
	Average	Range	Average	Range
Avenue B	31.6	12.9 - 58.4	61.8	15.5 - 93.1
Silverwood	42.0	0.9 - 89.2	65.0	17.3 - 99.5
Taylor	49.5	26.2 - 68.5	81.6	28.1 - 95.2

[a] estimated discrete TSS load

Several events required consideration other than that described above. The events fell into one of three categories. The first category is events for which the IETD was not used to define the end of the rainfall-runoff event. One event fell into this category. The event began June 29, 2002 at 18:32 in the Avenue B catchment. The combined hyeto/hydrograph is shown in Figure 5.12.

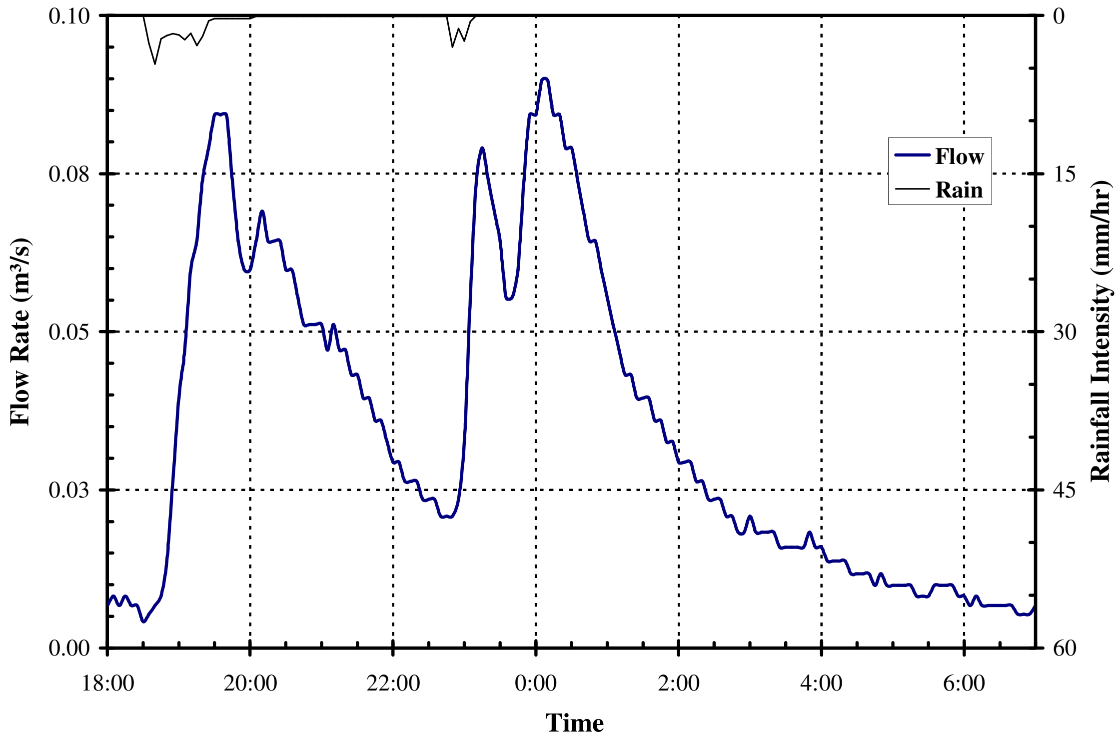


Figure 5.12 – Hyeto/hydrograph for Avenue B event beginning June 29, 2002.

In Figure 5.12, the hyetograph shows that the rainfall stopped shortly after 20:00 and then began again about 22:45, two hours forty-five minutes later. The water quality samples for this event were drawn between 18:52 and 20:24, and thus the second rainfall was not sampled for water quality. Due to the break in the rainfall and the lack of sampling in the second rainfall, this event was deemed to have been complete at 22:45 and the second rainfall event was disregarded. Some runoff, which would have been represented by the tail of hydrograph, is unaccounted for in the event volume.

The second category of events are those that had two sets of water quality samples collected from the same rainfall-runoff event. Three events fall into this category, with two events occurring in different catchments during the same rainfall event.

The first event began July 9, 2002 at 03:12 with samples collected in both the Avenue B and Taylor Street catchments. Figure 5.13 shows the event combined hydro/pollutograph for the Avenue B catchment. The full set of event graphs is shown in Figure G.25. The first set of water quality samples was collected during the leading leg of the hydrograph, while the second set was collected during the declining leg. The collection of the two sets of water quality samples leaves a period of nearly five hours without water quality data. The load from the intervening time was estimated using an average concentration of the last sample of the first set of samples and the first sample from the second set, yielding an average concentration of 307 mg/L. As shown in Figure 5.13, the concentration tends to somewhat follow the trend of the flow rate. The estimated mass of TSS represented by the first and second sample sets is 360 kg and 210 kg, respectively. The estimated mass of TSS during the intervening time is 3120 kg. The mass from the second set of samples includes an extrapolation of the concentrations from the last sample to the average baseflow concentration. The event TSS load is estimated to be 3700 kg.

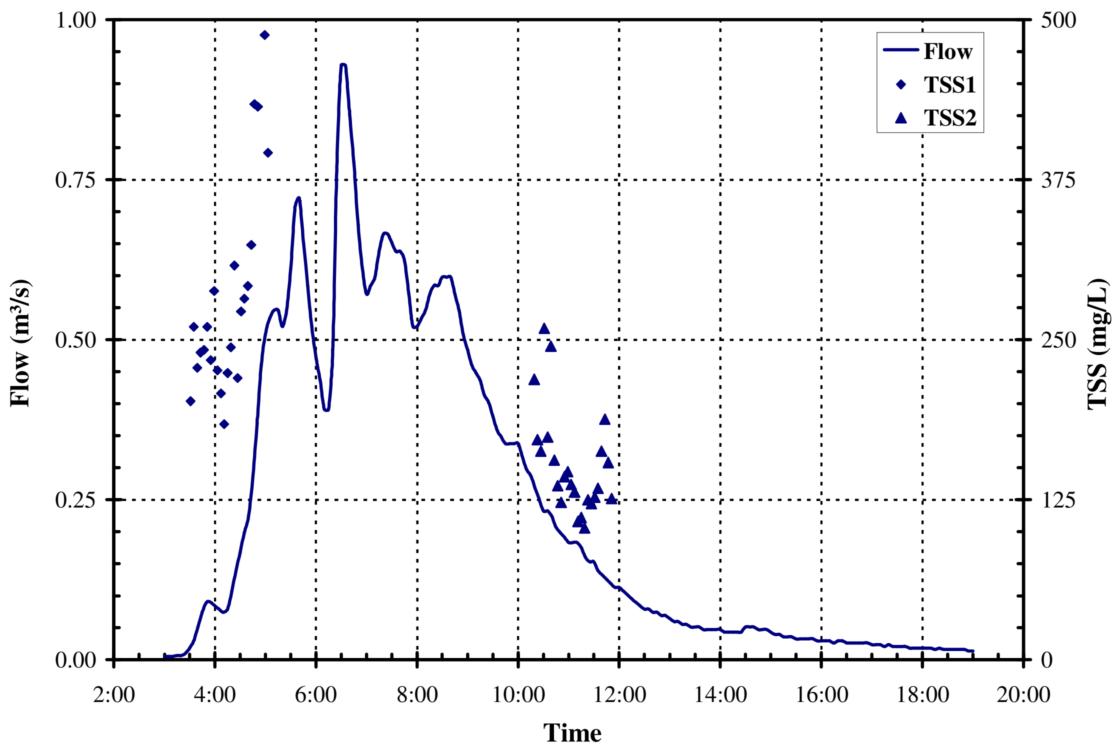


Figure 5.13 – Hydrograph and pollutograph for Avenue B event beginning July 9, 2002.

The Taylor Street catchment also had two sets of water quality samples collected during the July 9, 2002 rainfall event. The combined hydro/pollutograph is shown in Figure 5.14. The same analysis procedure was followed for this event as with the previous. The average concentration in the intervening time was 175 mg/L. The estimated mass of TSS represented by the first and second sample sets was 1030 kg and

320 kg, respectively. The estimated mass of TSS during the intervening time was 3700 kg. The TSS event load was estimated to be 5050 kg.

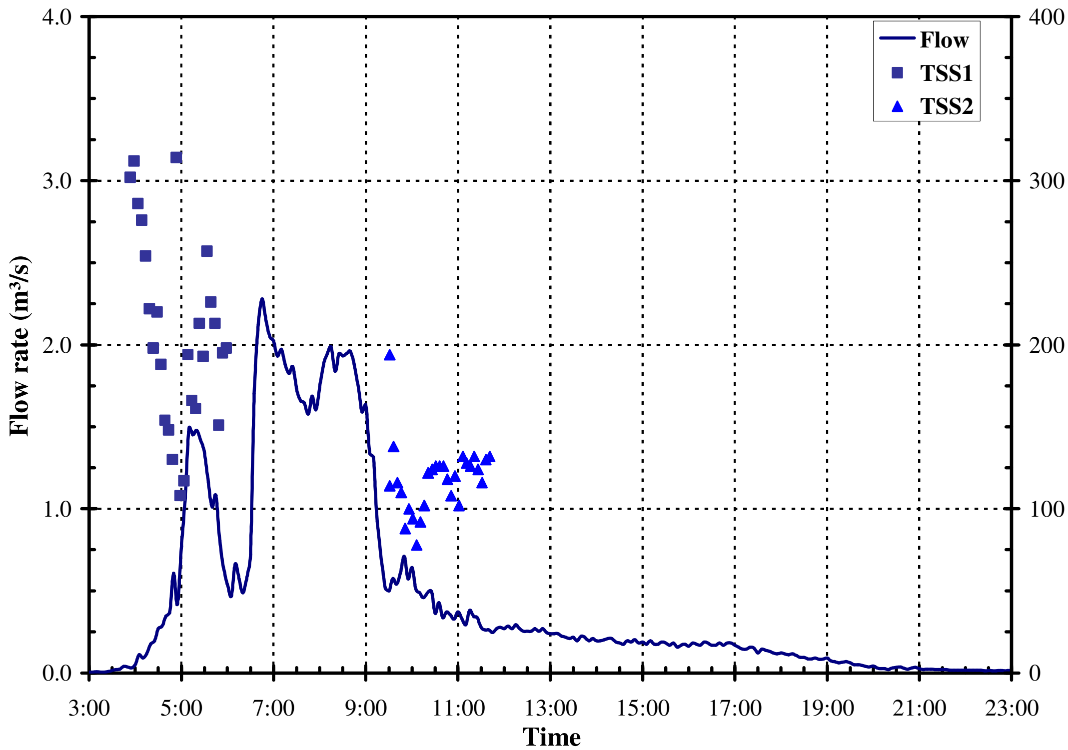


Figure 5.14 – Hydrograph and pollutograph for Taylor Street event beginning July 9, 2002.

The third event in this category is from the Avenue B catchment, which began June 10, 2002 at 19:47. The combined hydro/pollutograph is shown in Figure 5.15. The peculiar shape of the hydrograph was caused by fluctuations of the rainfall intensity (see Figure G.19). The average concentration between the first and second sets of samples is 406 mg/L. The scatter in the first set of samples is large and the scatter in the second set of samples is small. Generally in other events, the TSS concentration trend fell from the initial samples (e.g. Avenue B, June 20, 2002 shown in Figure G.20). A decision was taken to err on the side of smaller concentrations and consequently smaller loads and a concentration of 325 mg/L was used to calculate the mass load in the intervening time. This concentration represents the lower third point of the difference between the last sample of the first set and the first sample of the last set. The estimated mass of TSS represented by the first and second sample sets is 268 kg and 59 kg, respectively. The estimated mass of TSS during the intervening time is 1504 kg. The TSS event load is estimated to be 1830 kg.

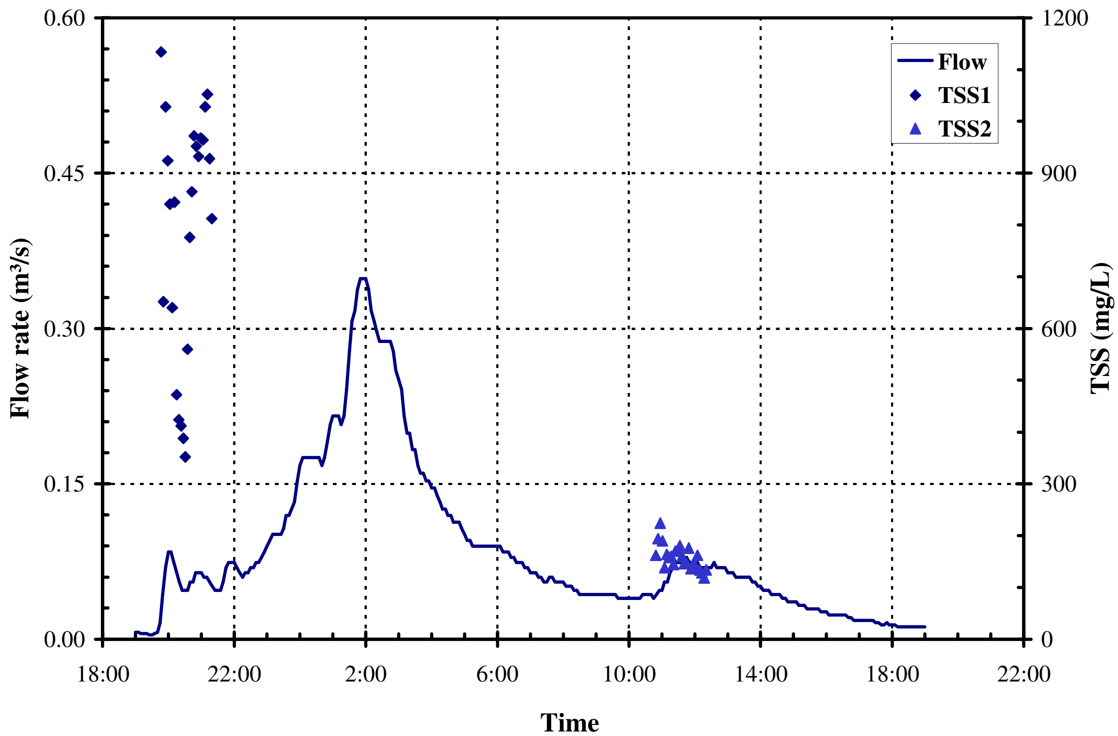


Figure 5.15 – Hydrograph and pollutograph for Avenue B event beginning June 10, 2002.

The event load for the three aforementioned events determined using the composite sample concentrations was calculated by allotting half of the flow volume in the intervening time to the first set of samples and half of the flow to the second set of samples.

The third category of events requiring special explanation is one event during which the flow metering equipment malfunctioned near the end of the event. The rainfall event began July 16, 2001 at approximately 09:20 in the Taylor Street catchment. The combined hydro/pollutograph are shown in Figure 5.16. The full set of event graphs are shown in Figure G.2. The equipment malfunctioned during the declining leg of the event. The flow data were truncated resulting in some volume and mass being not reported. Since the malfunction occurred in the declining leg of the hydrograph, the relative volume of flow and mass unaccounted for are assumed to be reasonably small, and the event was included for further analysis.

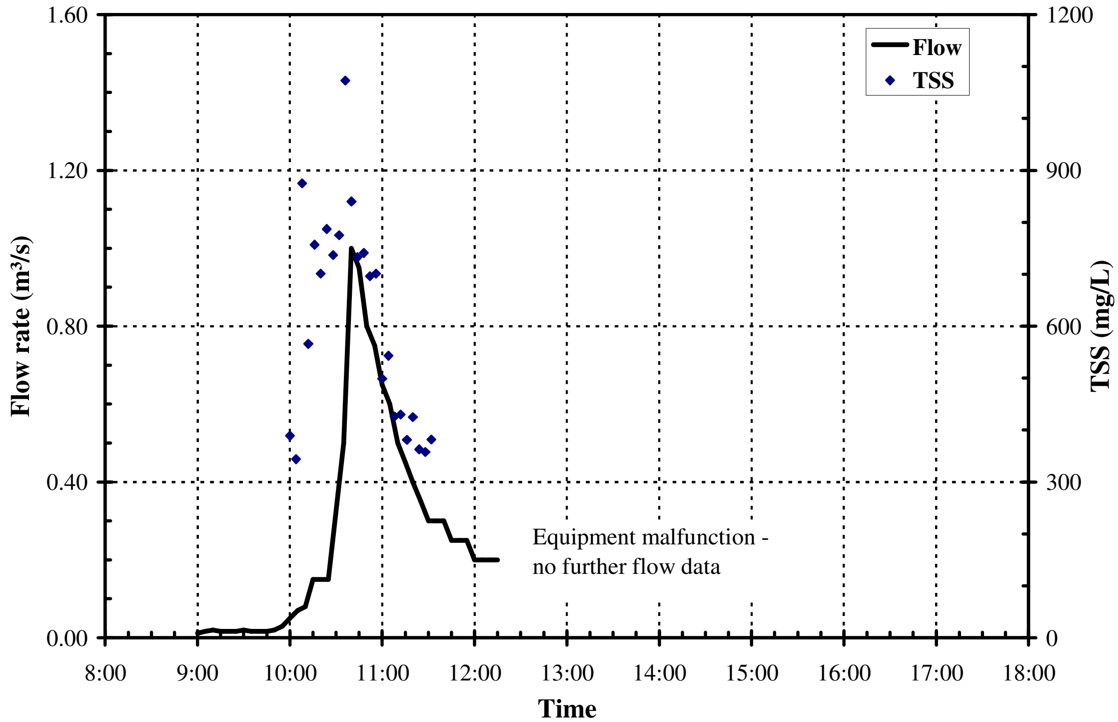


Figure 5.16 – Hydrograph and Pollutograph for Taylor Street event beginning July 16, 2001.

Table 5.9 presents the average, maximum, and minimum event loads from 2001 and 2002. The parameters reported are TSS load (discrete), TKN load, TP load, COD load and chloride load. The number of water quality parameters reported has been reduced to those parameters recommended by the US EPA (WEF/ASCE 1998), Alberta Environment (1999) and the National Water Research Institute (Marsalek and Schaefer, 2003) as parameters of interest in urban runoff. The list has also been shortened to those parameters for which, in this work, confident analysis can be made. For example, the same shipping delay concerns from the TC's and FC's apply to BOD_5, and so it has been eliminated from further analysis. COD is used instead to represent the oxygen demand. As shown in the table, the Taylor Street catchment had the highest average (arithmetic) load per event for each parameter. This is not unexpected because the Taylor Street catchment is the largest of the four studied. A summary of load results for the remaining parameters is shown in Appendix H.

The Sturgeon Drive catchment only has two useable TSS results, while the Avenue B, Taylor Street, and Silverwood catchments have nine, eight, and eight events, respectively. Due to the small number of usable events, the Sturgeon Drive catchment was eliminated from further load analysis.

Table 5.9 – Event load summary for selected parameters for useable events in 2001 and 2002.

Site		Discrete TSS (kg/event)	TKN (kg/event)	TP (kg/event)	COD (kg/event)	Chloride (kg/event)
Avenue B	Average	1,200	17	3.3	540	75
(9)*	Max	3,700	52	12	1,700	310
	Min	65	0.72	0.13	32	4.2
Taylor	Average	1,700	22	4.3	700	79
(8)*	Max	5,500	86	12	2,700	240
	Min	68	2.2	0.38	77	16
Silverwood	Average	960	3.8	1.1	240	4.7
(8)*	Max	5,300	19	5	1,200	14
	Min	21	0.05	0.02	6.2	0.34
Sturgeon	Average	390				
(2)*	Max	400				
	Min	380				

* - number of events represented

5.6 CATCHMENT WATER QUALITY CHARACTERIZATIONS

5.6.1 Introduction

Three characterizations of the catchment water quality were calculated using the event load data: the SMC, a set of regression equations, and the unit load. These characterizations are intended to provide a means to predict the urban runoff load based on the catchment land use and observed rainfall/runoff as well as to quantify the effect of pollution abatement efforts. The SMC and the regression equations are based directly upon the event load data from the previous section. The unit load, however, is an annual parameter and cannot be calculated from a few events scattered throughout two years of monitoring. Therefore, the SMC and regression characterizations are used to estimate total loadings from which seasonal unit loads can be determined. It should be noted that annual unit loads cannot be determined because the data only represent summer rainfall-runoff. It is suspected that winter/spring runoff will contribute significantly to the total annual loads. All three characterizations were used to estimate the summer rainfall-runoff loads.

The time period used for purposes of the following analyses is the window from April 1 to October 31 annually which, in this project, represents the window during which precipitation is assumed to produce runoff. Precipitation falling in the period outside of the window is assumed not to be represented by the data collected.

5.6.2 SMC Analysis

The calculation of mass loading using the SMC methodology requires the determination of the SMC for each water quality parameter and the volume of water discharged during a rainfall event. Although the storm sewer flow was monitored, the monitoring was not continuous and did not cover all of the April to October period. The rainfall data, however, did extend over the entire period. Therefore, in this study, a catchment runoff coefficient is used with the recorded rainfall depths to determine the volume of runoff used to estimate the loadings.

The determination of the SMC first requires the determination of the EMC's for each rainfall-runoff event. The EMC is determined by dividing the event load by the total volume of runoff discharged during the rainfall-runoff event.

The SMC is determined by taking the appropriate mean of the EMC's. The appropriate mean is dependent upon the distribution of the EMC's. For urban runoff work, the most commonly determined and assumed distribution is the log-normal distribution, which is fully defined, in log space, by the mean and standard deviation. The geometric mean is the log space mean converted to real space. Log-normally distributed data plot as a straight line on a log versus normal probability plot. Figure 5.17 shows the log-normal probability plot for TSS for each of the catchments.

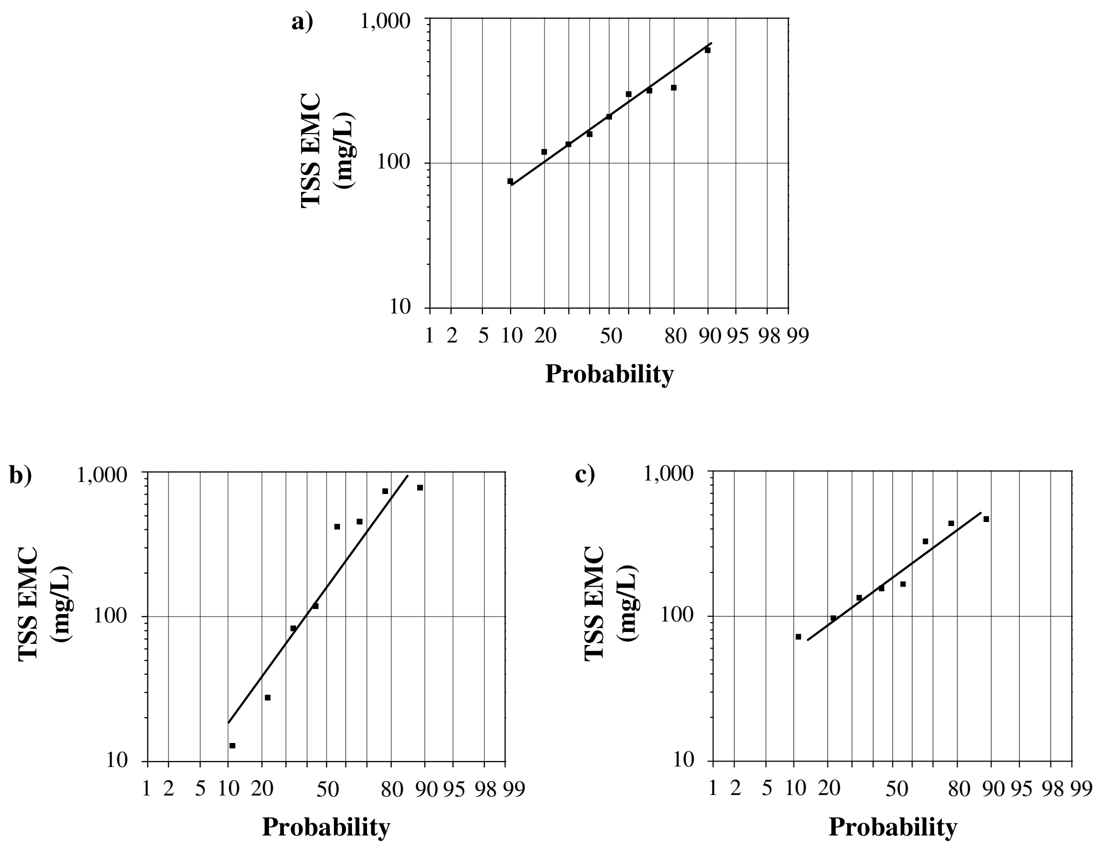


Figure 5.17 – TSS EMC log-normal probability plots for each catchment: a) Avenue B; b) Silverwood; c) Taylor Street.

As shown by the linear trend in Figure 5.17, the data are well represented by the log-normal distribution and thus the geometric mean. Table 5.10 presents the TSS SMC and a summary of the TSS EMC's for each of the catchments, as well as the range of one standard deviation (std. dev.) converted to real space (i.e. the log space mean plus/minus the log space standard deviation). Approximately 68% of all events should have concentrations within this range.

Table 5.10 – TSS SMC and EMC summary (mg/L).

	SMC	EMC Min.	EMC Max.	EMC one std. dev. range
Avenue B	210	75	600	110-400
Taylor	190	70	465	94-380
Silverwood	160	15	780	13-780

As shown in Table 5.10, the Silverwood catchment results exhibit the greatest variability as indicated by the range of EMC and one standard deviation range. The Avenue B and Taylor Street catchments have similar one std. dev. ranges and similar SMC's. The similar one standard deviation ranges indicate that the variability is similar.

Figure 5.18 shows the log normal probability plot for TKN for each of the catchments. The data, in all three plots, fits the linear trend well, except for one point from the Avenue B catchment. The event that produced the non-conforming point occurred August 15, 2002 and had a rainfall depth of 11.2 mm, which is in the upper third of recorded rainfall depths for this catchment. Further event parameters (e.g. rainfall depth, sampled volume, total volume, duration, etc.) are shown in Table D.1. Upon further examination of the event, the hydrograph (Figure G.27) was found to have two peaks, and the hyetograph indicated three periods of rainfall. The first period of rain did not produce runoff sufficient to trigger the automatic sampler. The water quality samples were drawn from the runoff resulting from the second period of rainfall. The runoff from the third period of rainfall was not sampled. Approximately 20% of the runoff volume was sampled for water quality. The TKN mass load carried in the non-sampled runoff from the third period of rainfall is unknown, but it was likely significant. Thus, the root problem is likely poor coverage of the event by the water quality samples. The net result of this problem is the reduction of the EMC, by dilution, from the true value. The different number of points between Figures 5.17 and 5.18 are caused by unreported results. Concentration results were not always reported for all composite sample water quality parameters, due to various factors such as test specific time requirements being exceeded and lab incidents which resulted in destruction or contamination of the sample.

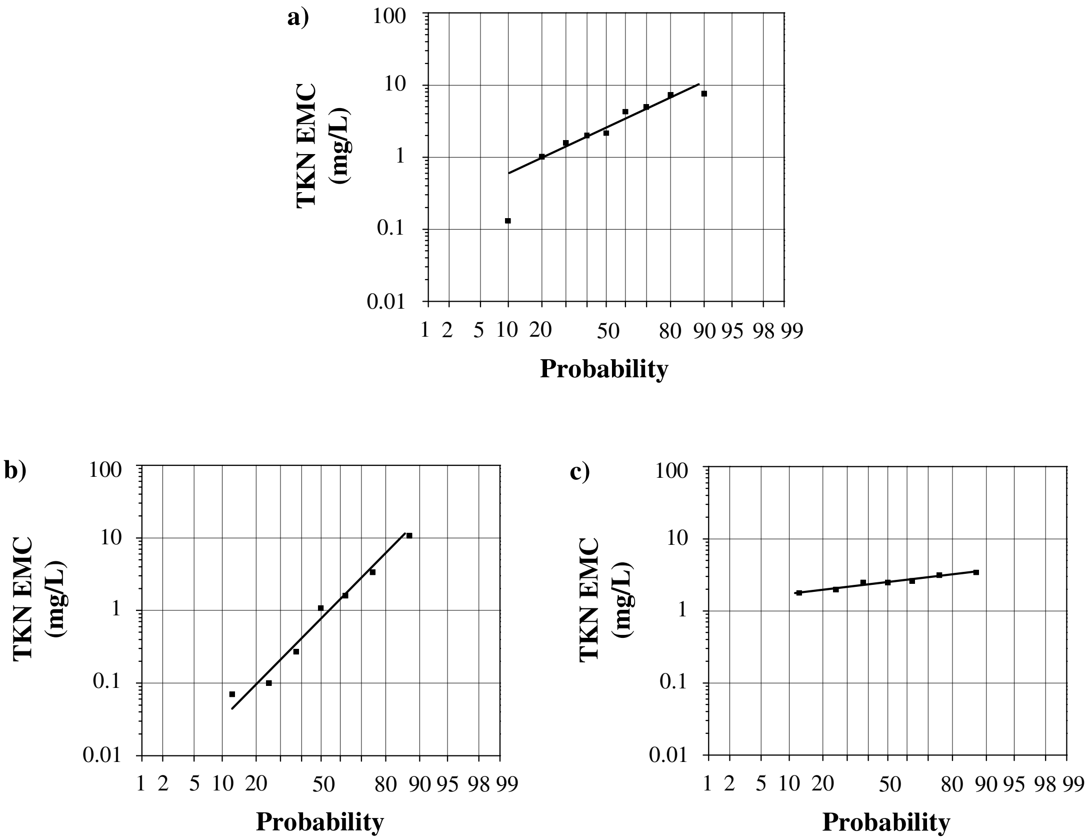


Figure 5.18 – TKN EMC log-normal probability plots for each catchment: a) Avenue B; b) Silverwood; c) Taylor Street.

Table 5.11 presents a summary of the TKN EMC's and SMC for each catchment. The Taylor Street catchment had the smallest range as well as the best visual fit to the line. The Silverwood catchment had the largest EMC range and the smallest SMC. As with TSS, the Silverwood catchment had the greatest variation.

Table 5.11 – TKN SMC and EMC summary (mg/L).

	SMC	EMC Min.	EMC Max.	EMC one std. dev. range
Avenue B	2.2	0.61	7.6	0.61-7.6
Taylor	2.5	1.8	3.4	2.0-3.1
Silverwood	0.73	0.07	11	0.1-10

Figure 5.19 shows the log-normal probability plot for TP EMC's for each catchment. Again, the Taylor Street data fit the linear trend well. The Avenue B data fit the linear trend well, except for one point. The non-conforming Avenue B point is from

the August 15, 2002 event and the same explanation as for TKN applies here. The Silverwood data have some scatter although they fit the trend of the line reasonably well. The Silverwood data show two points (0.01 and 0.02 mg/L) in the lower portion of the plot that appear to represent the slope of the line, but are displaced from the line. These two events (June 24, 2002 and June 29, 2002), similar to the Avenue B August 15, 2002 event, were poorly represented by the water quality samples. The hyetograph, hydrograph, pollutograph, and loadographs for these two events are shown in Figures G.15 and G.16, respectively. The figures illustrate the poor coverage of the event by the water quality samples. In the respective events, 4.8% and 0.9% of the runoff was sampled. As previously noted, the net effect of poor coverage of the event is reduction of the EMC from the true value because the load associated with the additional volume is not included in the EMC calculation. The linear trends shown in the figures support the log-normal assumption.

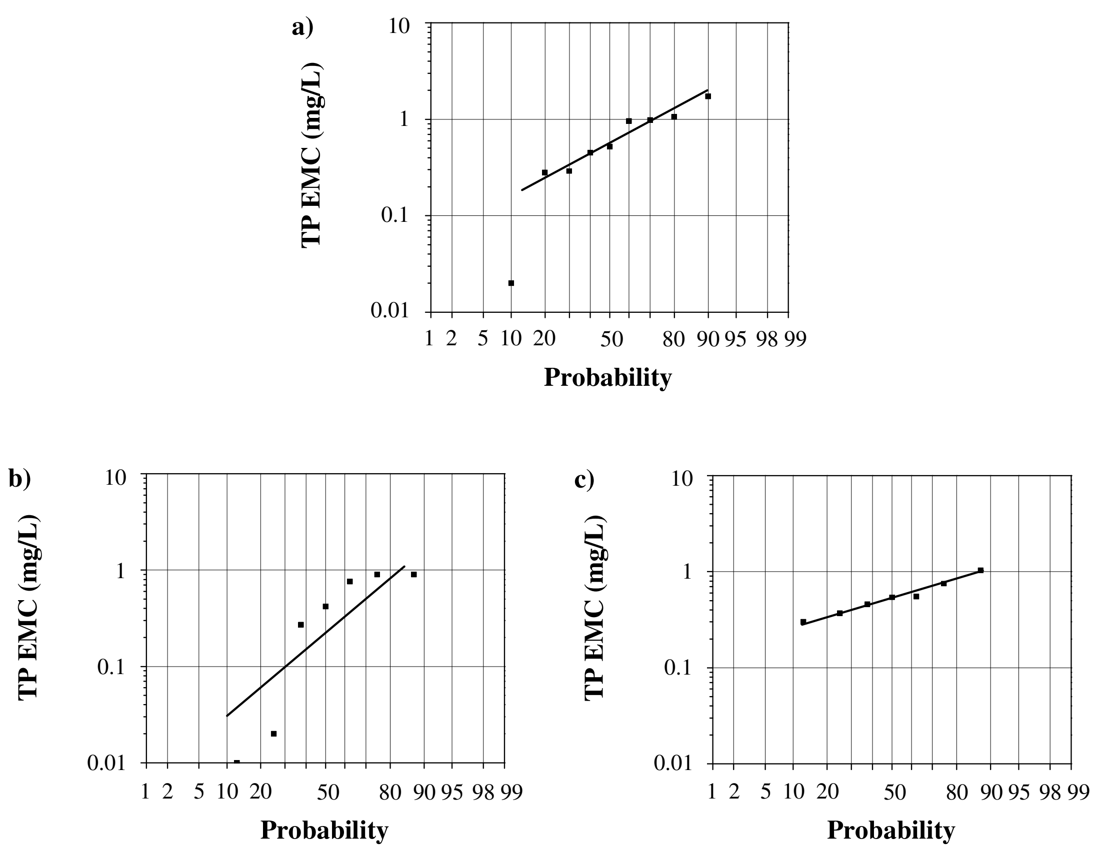


Figure 5.19 – TP EMC log-normal probability plots for each catchment: a) Avenue B; b) Silverwood; c) Taylor Street.

Table 5.12 presents a summary of the TP EMC's and SMC for each catchment. The Taylor Street catchment has the smallest range, smallest one standard deviation range and the largest SMC. The Avenue B catchment has the largest range of EMC's and one standard deviation. For this parameter, the largest range is for the Avenue B catchment.

Table 5.12 – TP SMC and EMC Summary (mg/L).

Catchment	SMC	EMC Min.	EMC Max.	EMC one std. dev. range
Avenue B	0.45	0.02	1.72	0.13-1.6
Taylor	0.53	0.3	1.0	0.35-0.80
Silverwood	0.21	0.01	0.90	0.033-1.3

Figure 5.20 shows the lognormal probability plots for COD EMC's for each catchment. Similar to the TP results, the same situation with one non-conforming event (August 15, 2002) in the Avenue B catchment and two events (June 24, 2001 and June 29, 2002) in the Silverwood catchment is observed in the COD results. The data show reasonably linear trends in both the Avenue B and Silverwood catchments, even with the non-conforming points. The Taylor Street data fit the linear trend well. The fit to the linear trends support the log-normal distribution assumption. Table 5.13 presents a summary of the COD EMC's and the COD SMC for each catchment. The Silverwood catchment COD EMC's have the largest range, largest one standard deviation range, and the smallest SMC. The Taylor Street catchment has the smallest range of COD EMC's, smallest standard deviation range of COD EMC's and the largest SMC.

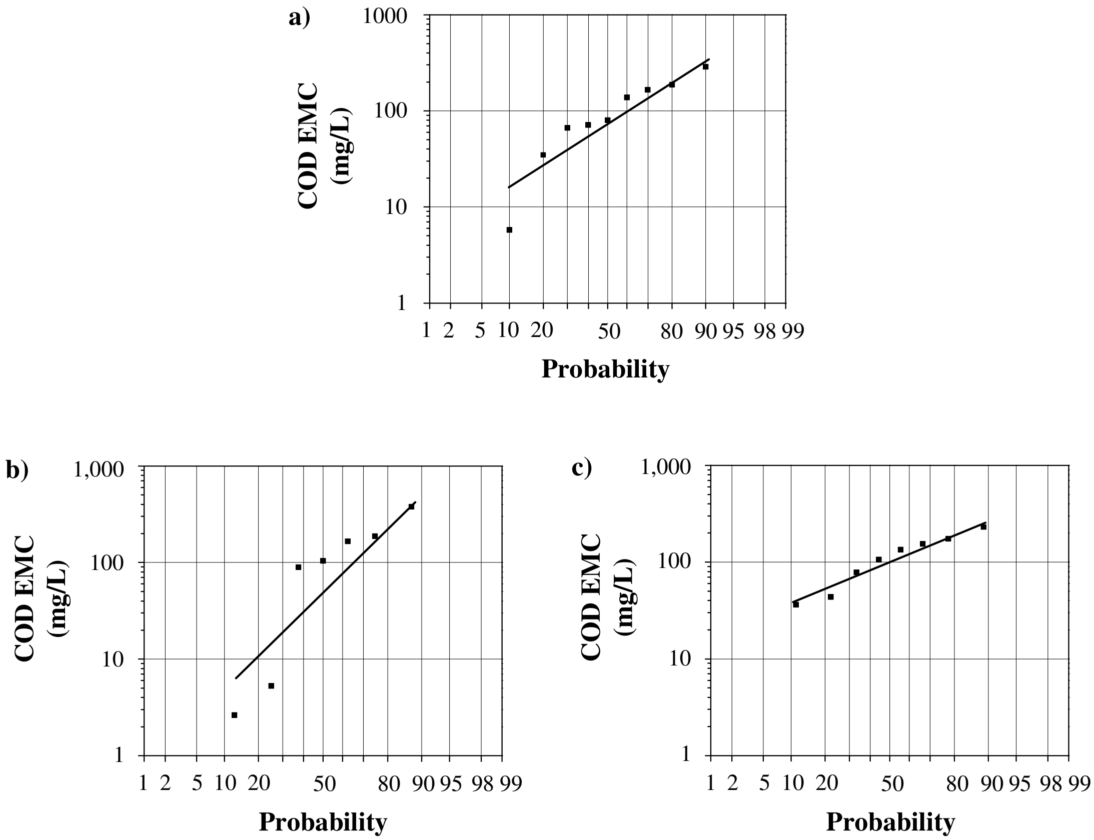


Figure 5.20 – COD EMC log-normal probability plots for each catchment: a) Avenue B; b) Silverwood; c) Taylor Street.

Table 5.13 – COD SMC and EMC summary (mg/L as O_2).

Catchment	SMC	EMC Min.	EMC Max.	EMC one std. dev. range
Avenue B	75	6	290	24-250
Taylor	100	52	195	52-200
Silverwood	55	2.6	380	8.0-370

Figure 5.21 presents the log-normal probability plot for chloride EMC's for each catchment. The data fit the linear trend reasonably in all of the plots. However, in each of the Avenue B and the Taylor Street catchments, there is a non-conforming point at the lower end. In the Avenue B plot, the previously discussed poorly covered event (August 15, 2002) is responsible for this point. The Taylor Street catchment shows one point at the lower end of the plot that does not fit the line. The event represented by this point occurred July 9, 2002, and was a large rainfall event. It had a rainfall depth of 31 mm,

which is one of the largest rainfall depths recorded during the two field seasons. This event had two sets of water quality samples collected from it. The first set of samples resulted in a composite concentration of 4 mg/L, while the second set of samples resulted in a concentration that was smaller than the detection limit of 2 mg/L. The result of less than the detection limit was treated like a null (zero) water quality concentration result. It is possible that the catchment had been washed clean of chloride resulting in a significantly lower overall concentration. The linear trends illustrated in the figures support the log-normal distribution assumption.

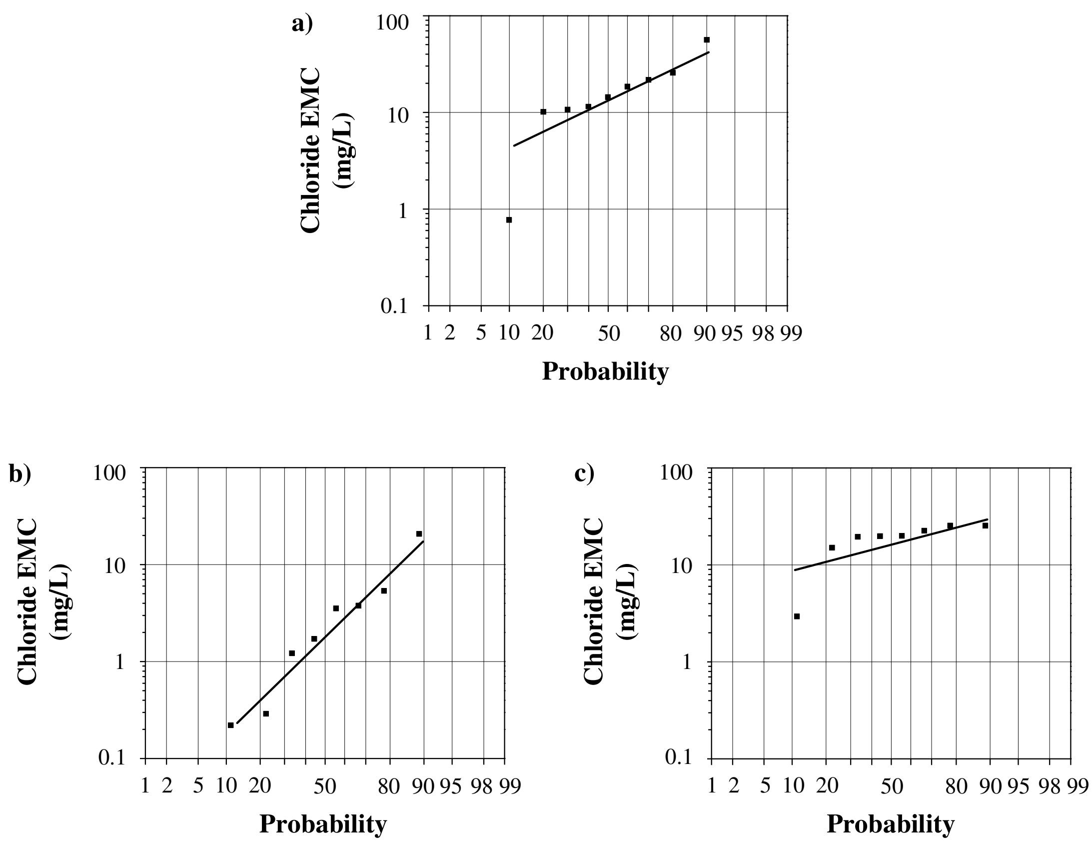


Figure 5.21 – Chloride EMC log-normal probability plots for each catchment: a) Avenue B; b) Silverwood; c) Taylor Street.

Error! Not a valid bookmark self-reference. presents a summary of the chloride EMC's and SMC for each catchment. The Avenue B catchment chloride EMC's have the largest range. The Silverwood catchment has the smallest one standard deviation range and the smallest SMC.

Table 5.14 – Chloride SMC and EMC summary (mg/L).

Catchment	SMC	EMC Min.	EMC Max.	EMC one std. dev. range
Avenue B	13	0.77	56	3.9-41
Taylor	15	2.9	25	8.0-33
Silverwood	1.9	0.22	21	0.43-8.8

The second requirement for calculation of mass load using the SMC method is the volume of water discharged. In this work, both the runoff volume and rainfall depth were measured. However, the monitoring for runoff volume was not continuous and many rainfall-runoff events were not recorded. Therefore, the volume of runoff was determined using a catchment runoff coefficient (C), which is the ratio of the runoff depth by the rainfall depth over the catchment. The runoff depth was found by dividing the total runoff volume discharged in an event by the contributing area. This analysis was performed for each catchment using all events for which both rainfall and runoff data were available. Figure 5.22 shows plots of runoff depth versus rainfall depth.

The line shown in each plot of Figure 5.22 was determined using a simple least squares linear regression. The runoff coefficient is the slope in each of the regression models. The runoff coefficients are 0.55, 0.18, and 0.29 for the Avenue B, Taylor Street, and Silverwood catchments, respectively. The regression produced good results with coefficients of determination (R^2) of 0.95, 0.97, and 0.80 for the Avenue B, Taylor Street, and Silverwood catchments, respectively. The regression equations logically indicate that some rainfall is required before runoff begins (i.e. the negative x-intercept). The runoff coefficient was then used to determine the total runoff volume of each event.

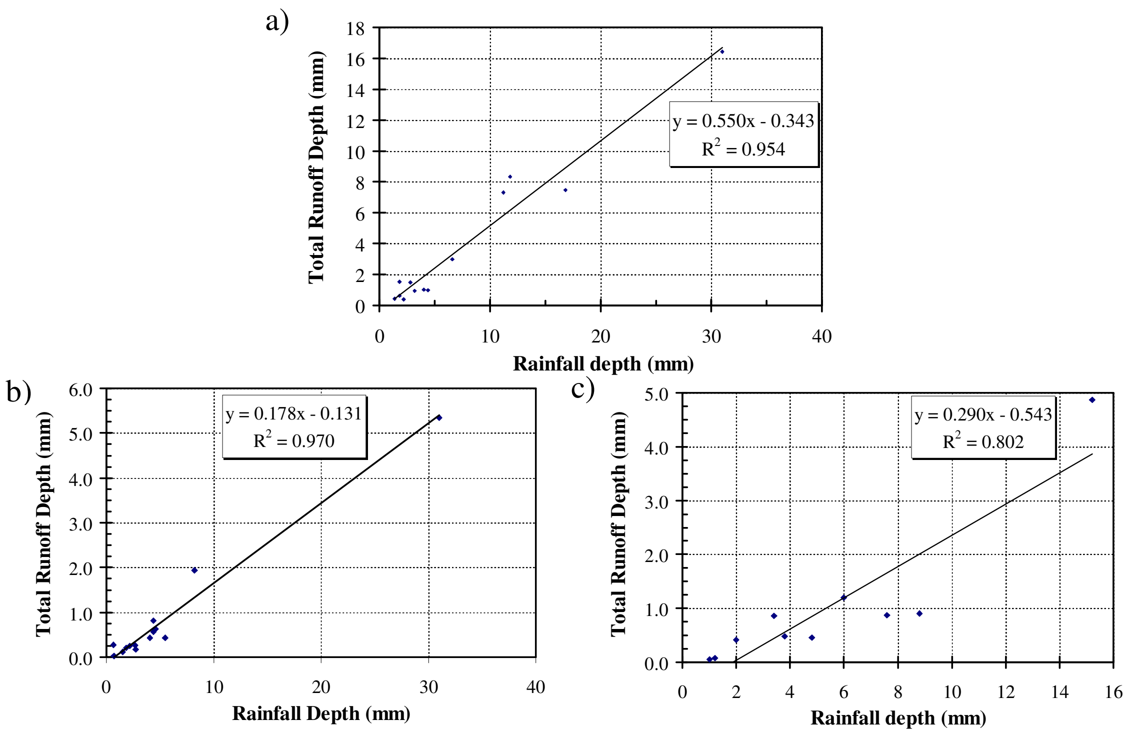


Figure 5.22– Runoff coefficient (C) determination for: a) Avenue B; b) Taylor Street; c) Silverwood.

Table 5.15 presents the estimated rainfall-runoff (period (April 1 to October 31 in 2001 and 2002) load for each study catchment made using the SMC methodology. The rainfall data used to calculate the results are shown in Appendix D. The Taylor Street catchment has the largest estimated load for each water quality parameter in each year. This is expected as the Taylor Street catchment is the largest catchment by about 2.5 and 8.2 times over the Silverwood and Avenue B catchments, respectively. The Avenue B catchment loads are larger than the Silverwood catchment loads even though the Avenue B catchment is 3.2 times smaller. The difference of the parameter loads in Table 5.15 between the two years of record is primarily because 2001 was the driest year on record in over one hundred years of record (World Meteorological Organization, 2002). From Table 5.4, 2001 had an average rainfall depth of 144.5 mm, which is below the average normal of 266 mm by 121.5 mm. In 2002, rainfall was slightly above the normal with an average rainfall depth of 292.4 mm.

Table 5.15 – Catchment load estimation using SMC method for the rainfall runoff period of 2001 and 2002.

Catchment	Rainfall/ runoff period	TSS load (kg)	TKN load (kg)	TP load (kg)	COD load (kg)	Chloride load (kg)
Ave B	2001	10,500	110	22	3,750	650
	2002	18,000	190	38	6,350	1,100
Taylor	2001	21,000	275	58	11,000	1,650
	2002	41,000	540	115	21,500	3,250
Silverwood	2001	8,700	40	11	3,000	105
	2002	12,500	55	17	4,350	150

Using the seasonal load estimates shown in Table 5.15, a unit load was estimated for each water quality parameter by dividing the estimated loads by the catchment area. The unit load shows different area-based load generation rates between the catchments. Table 5.16 shows the resulting unit loads in 2001 and 2002 and the average of the two values. These unit loads only represent the rainfall runoff period used in this study (April 1 to October 31).

Table 5.16 – Unit load (SMC method) in 2001 and 2002.

Catchment	Rainfall/ runoff period	TSS (kg/ha)	TKN (kg/ha)	TP (kg/ha)	COD (kg/ha)	Chloride (kg/ha)
Ave B	2001	140	1.5	0.30	50	8.7
	2002	240	2.5	0.51	85	15
	average	190	2.0	0.41	68	12
Taylor	2001	34	0.45	0.094	18	2.7
	2002	66	0.87	0.19	35	5.2
	average	50	0.66	0.14	26	4.0
Silverwood	2001	36	0.16	0.047	12	0.43
	2002	52	0.23	0.069	18	0.62
	average	44	0.19	0.058	15	0.52

The Avenue B catchment has the largest average unit load for each water quality parameter, while the Silverwood catchment has the smallest average unit load of all of the catchments. The average Avenue B unit load parameters are larger than the average Taylor Street unit load parameters by between 2.6 and 3.8 times. The average Taylor Street unit load parameters are larger than the average Silverwood catchment unit load parameters by 1.1 to 8.4 times. Differences between the years is likely due to the significantly below normal rainfall in 2001 and slightly above normal rainfall in 2002. The differences in the unit loads illustrate the differences between the catchments and the land use they represent.

5.6.3 Regression Analysis

Multiple variable regression analysis seeks to determine a relationship between several independent variables and one dependent variable by minimizing the total error between the data and the regression relationship. Regression analysis was undertaken to characterize the event loads for each water quality parameter in each catchment with several rainfall parameters as independent variables. The rainfall variables, their respective variable names, and data ranges are shown in Table 5.17. These rainfall variables are commonly used to represent rainfall events, except for the maximum five minute average intensity. The maximum five minute average intensity was included because it was felt that short-duration spikes obtained from tipping bucket rain gauge data may not properly represent the effect of rainfall intensity in terms of water quality response.

Table 5.17 – Rainfall variables and data ranges.

Parameter	Variable name	Units	Data range		
			Avenue B	Taylor	Silverwood
Rainfall event depth	*depth*	mm	1.0 - 30	0.6 – 31	1.0 - 15
Average intensity	*aveint*	mm/hr	0.4 – 4.7	0.29 – 39	0.6 - 31
Antecedent dry period	*ante*	days	1.0 – 5.1	1.3 – 8.2	0.70 - 29
Event duration	*dur*	min	25 – 106	7 – 610	7 - 250
Maximum instantaneous intensity	*max*	mm/hr	2.8 - 55	1.5 – 106	9.5 - 72
Maximum five minute average intensity	*5min*	mm/hr	2.7 - 23	1.5 – 20	4.2 - 33

In a U.S. nationwide study of urban runoff, USGS determined that the most suitable form of the predictor equation was a power type relation (Driver and Tasker, 1990). An example of the proposed model form is shown in [5.3] for TSS,

[5.3] $$TSS = a \cdot depth^{b} \cdot aveint^{c} \cdot ante^{d} \cdot dur^{e} \cdot max^{f} \cdot 5min^{g}$$

where TSS is the TSS event load (kg), a is a constant, and b through g are the exponents of the respective rainfall variable. For the models of the other four parameters, TKN, TP, COD and chloride (Cl^-) are substituted for the TSS term.

Driver and Tasker (1990) and LeBouthillier et al. (2000) used a logarithm transformation of the proposed model to yield a linear relation between the independent and dependant variables. The log transformation has a net effect of placing less emphasis on larger, potentially more uncertain, data points. USGS accepts this effective weighting, and found in other studies that the resultant parameter estimates are more accurate using the log transformation versus other regression methods based on the same model relation. LeBouthillier et al. (2000) used this type of analysis because linear regression allows for a greater suite of analysis tools to be used. In both cases, the resulting analysis was performed using statistical software, based on a forward stepwise linear regression methodology.

In the forward stepwise linear regression methodology, the most strongly correlated independent variable, based on individual R^2 values, is regressed first (Lebouthillier et al., 2000). Two statistical parameters, namely the F-statistic (evaluates the model's ability to describe the variability in the data) and the t-statistic (evaluates the parameter's difference from zero based upon the sample size and data variability), are evaluated based on the resultant model. The model is accepted if both parameters are significant. Then, the next most strongly correlated variable is added to the linear regression. If the addition of the next most strongly correlated variable results in F- or t-statistics that are not significant, or does not improve the overall model fit (R^2), that model is rejected and the previous model that meets the above conditions is accepted.

The statistical analysis package SPSS version 12.0 was used to perform the forward stepwise multivariate linear regression analysis on the log transformed data. The 10% level of significance was used in these analyses. SPSS automatically determines the F- and t- statistics and determines if they are significant. Only results with significant F- and t- statistics are reported by SPSS, therefore the F- and t- statistics are not reported herein.

The results of the regression analysis on the event loads, goodness-of-fit measures, and the degrees of freedom for each parameter and catchment are shown in Table 5.18. It should be noted that the size of the data set used is small for this type of analysis. This analysis is presented as an example of the methodology and should not be extrapolated beyond the variable ranges presented in Table 5.17.

Table 5.18 – Regression results summary and goodness-of-fit measures.

Parameter	Coefficient or exponent name							R^2	MPE[a]	DOF[b]
	constant	depth	aveint	ante	Dur	max	5min			
	a	b	c	d	e	f	g			
TSS_{AveB}	30.5	1.51	-	-	-	-	-	0.89	19%	7
TSS_{Taylor}	74.4	0.98	-	-	-	-	-	0.89	45%	5
$TSS_{Silverwood}$	0.012	-	3.69	-	2.2	-1.26	-	0.98	22%	4
TKN_{AveB}	1.89	2.09	-	-	-	-1.24	-	0.79	50%	6
TKN_{Taylor}	75	1.46	-	-	-0.745	-	-	0.97	23%	4
$TKN_{Silverwood}$	9.43×10^{-5}	-	2.52	-	1.4	-	-	0.87	54%	4
TP_{AveB}	0.532	2.11	-	-	-	-1.38	-	0.82	48%	6
TP_{Taylor}	483	1.46	-	-	-1.34	-	-	0.96	21%	4
$TP_{Silverwood}$	3.01×10^{-5}	-	2.52	-	1.36	-	-	0.88	52%	4
COD_{AveB}	63	2.01	-	-	-	-1.15	-	0.82	29%	6
COD_{Taylor}	104	0.944	-	-	-	-	-	0.75	33%	6
$COD_{Silverwood}$	17.2	-	0.897	-	-	-	-	0.47	94%	5
Cl^-_{AveB}	17.9	1.8	-	-	-	-1.22	-	0.84	32%	6
Cl^-_{Taylor}	28.5	0.552	-	-	-	-	-	0.55	27%	6
$Cl^-_{Silverwood}$	0.0095	-	1.7	-	0.79	-	-	0.94	72%	5

Refer to [5.3] for key to equation coefficient and exponents.
[a] – Median percent calibration error
[b] – Degrees of freedom

All three TSS models return reasonably good R² results. The manner in which R² is calculated can cause the value to be high, apparently showing very good model fit. R² is calculated using,

[5.4] $$R^2 = \frac{\left[\sum_{i=1}^{n}\left(O_i - \overline{O}\right)\left(P_i - \overline{P}\right)\right]^2}{\sum_{i=1}^{n}\left(O_i - \overline{O}\right)^2 \sum_{i=1}^{n}\left(P_i - \overline{P}\right)^2},$$

where P_i is the model prediction, $\overline{P}$ is the mean of the predictions, O_i is the observed value, and $\overline{O}$ is the mean of the observed values. When the range of observations is large (i.e. orders of magnitude), large relative errors about small value points are masked. For example, if the range is 1000, the datum point is three and the error about that point is one to six, the relative error about the point is about 200%, but compared to the overall range, the error is small and masked. Thus, in the situation of a large range of data, the R^2 value provides a good indication of overall goodness of fit. This work has a few points with large ranges, therefore R^2 was used to indicate overall goodness of fit. Further goodness of fit measures can be used along with R^2.

The median percent calibration error (MPE) is used as a goodness-of-fit measure to represent the relative average calibration error about each point. The median value was used as it is resistant to outliers and single large values that cause the arithmetic average to be skewed towards the larger value. The smallest TSS MPE is 19% for the Avenue B catchment and the largest MPE is 45% for the Taylor Street catchment.

When a model has only one predictor variable, a plot of the data and model fit can be used to provide visual evaluation of the result. Figure 5.23 shows the data and model fit for the Avenue B TSS model. In agreement with the goodness-of-fit parameters shown in Table 5.18, Figure 5.23 shows that the model fits the overall trend of the data better than the individual points, with the exception of one point.

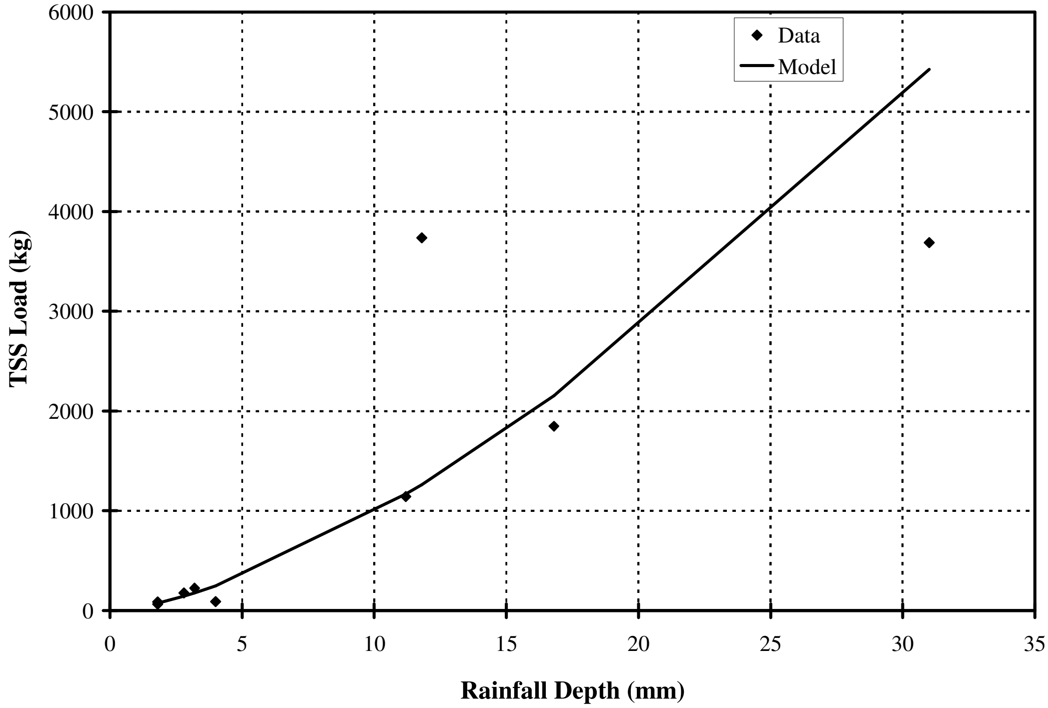


Figure 5.23 – Avenue B catchment TSS regression model fit.

In the TSS regression equations, four variables are found to be significant. The rainfall depth is significant in both the Avenue B and Taylor Street catchments. The average intensity, duration, and maximum intensity are significant in the Silverwood catchment.

The analysis of regression residuals provides an indication of problems with a regression model. The Avenue B TSS residual plots are shown in Figure 5.24. A residual plot should be randomly distributed about zero and have no distinct pattern. A distinct pattern or distribution that is not centered about zero indicates a problem with the regression. The Avenue B TSS residuals appear randomly distributed about zero with no distinct pattern. There are, however two large values that indicate greater error about those points. The plots are generally representative of all of the residual plots from all of the catchments indicating no obvious problems with the regressions. A complete set of residual plots are presented in Appendix K. It should be noted that, as with the regression models, the limited data set makes it difficult to fully assess whether the residual plots are randomly distributed or have a pattern.

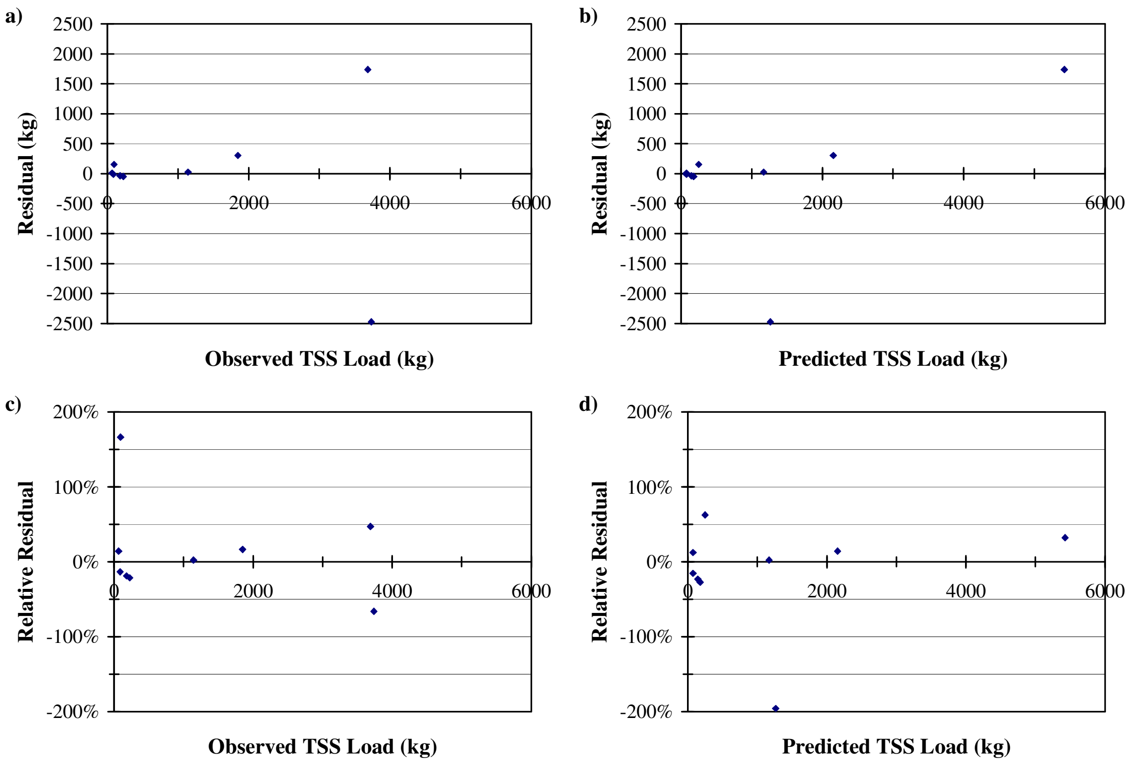


Figure 5.24 – Regression residual plots for TSS in the Avenue B catchment a) residual versus observed load, b) residual versus predicted load, and c) relative residual versus observed load, and d) relative residual versus predicted load.

The TKN models return R^2 values that range from acceptable (0.79) to very good (0.97), which indicate reasonable model fit. The MPE's range from 23% for the Taylor Street catchment to 54% for the Silverwood catchment. Again, for the Avenue B and Taylor Street models, the depth is returned as a significant variable. The Silverwood catchment model utilizes a small constant, as well as the average intensity and duration. For the TKN model different from the TSS model, the maximum intensity was not returned as a significant variable for the Silverwood catchment.

All three TP models, shown in Table 5.18, have acceptable R^2 values, which range from 0.82 to 0.96 for the Avenue B and Taylor Street catchments, respectively. The MPE's range from 21% for the Taylor Street catchment to 52% for the Silverwood catchment. All three models use the same predictor variables as the TKN models. The individual exponents in each TP model are also similar to the exponents in the TKN models except for the duration exponent in the Taylor Street catchment, which is almost double the value in the TKN model. The difference in the exponent may indicate a difference in the pollutant mobilization and washoff processes in this catchment between TKN and TP, such as particulate versus dissolved forms.

The COD models, shown in Table 5.18, have R^2 values that range from 0.82 to 0.47 for the Avenue B and Silverwood catchments, respectively. The Avenue B and Taylor Street R^2 results are reasonable, but the Silverwood R^2 results are poor. The MPE's are 29%, 33%, and 94% for the Avenue B, Taylor Street, and Silverwood catchments, respectively. The MPE's for the Avenue B and Taylor Street catchments are in line with the median errors of the other models. The MPE for the Silverwood catchment is large. Considered together, the goodness-of-fit parameters indicate that a trend may be evident in the Silverwood catchment and that the model describes the general trend of the data, but not very well. The model for the Silverwood catchment has only one significant variable. It is plotted in Figure 5.25 to permit visual examination of the fit. The figure shows one point that is substantially different from the rest of the data. Aside from the one non-conforming point, the model fits the trend in the data. The event that produced the non-conforming point (June 17, 2002) had the largest rainfall depth of the useable events in the Silverwood catchment, a large runoff volume, and a large load. Otherwise, the event was similar to most of the other events. This suggests that there may be other influences to the event loads.

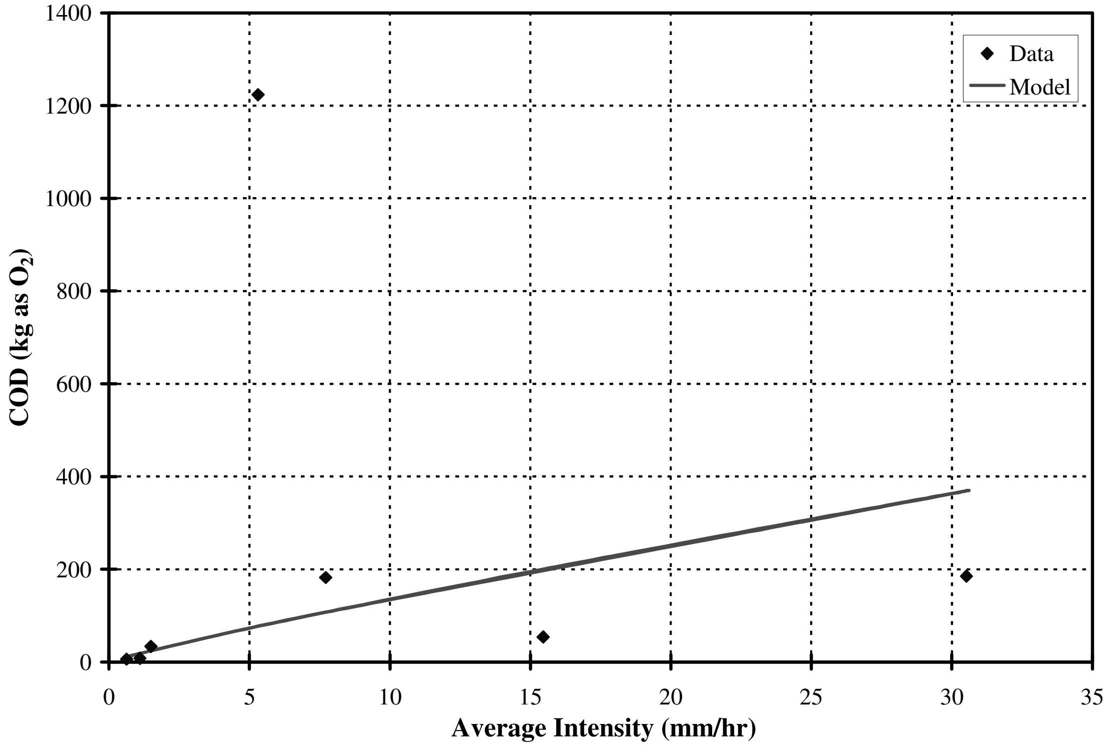


Figure 5.25 – Silverwood catchment COD model fit

The Tayor Street COD model has only one significant variable, the rainfall depth. The data and model are plotted in Figure 5.26, which shows good agreement between the model and the data at the lower end of the plot. The two non-conforming values come from events on June 17, 2002 and July 9, 2002. The events produced an estimated 2750 kg and 94 kg of COD, respectively. The June 17, 2002 event had the greatest instantaneous intensity (106 mm/hr) for events in this catchment while the July 9, 2002 event had the largest total event depth (31 mm) for the same set of events. The two extreme rainfall characteristics may have caused the poor fit to the regression model. It is also possible that an alternate model formulation may better represent this data, however the amount of data available is insufficient to assess a new model formulation.

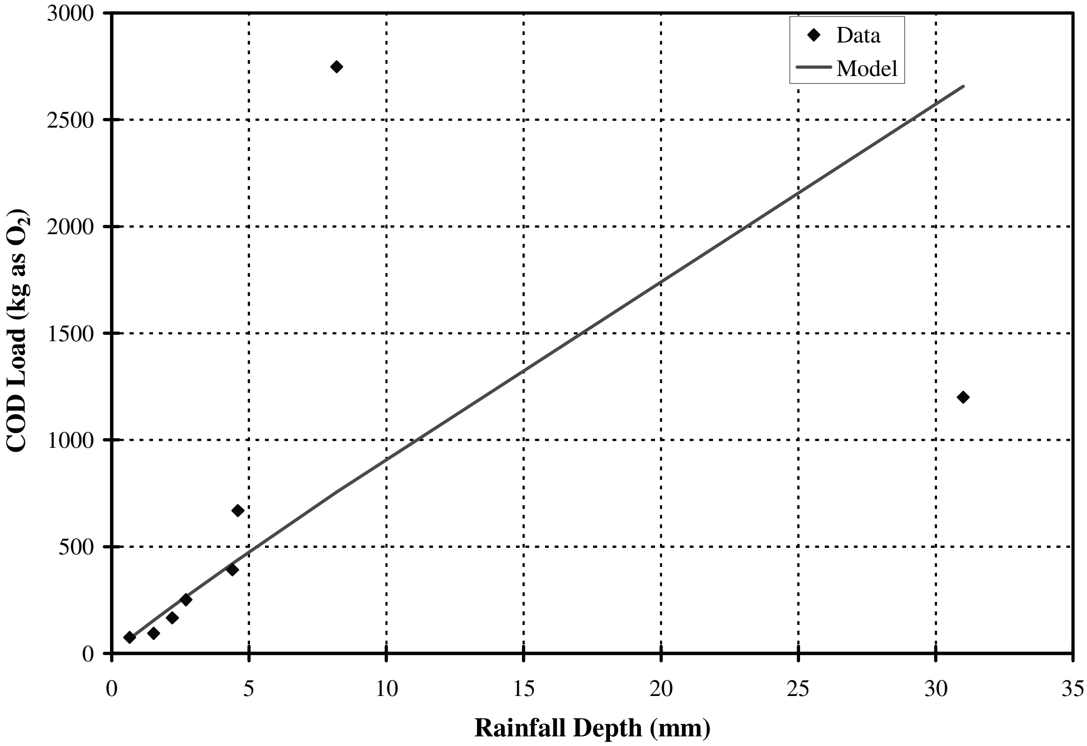


Figure 5.26 – Taylor Street catchment COD regression model fit.

The three chloride models, shown in Table 5.18, have R^2 values of 0.84, 0.55, and 0.92 for the Avenue B, Taylor Street and Silverwood catchments, respectively. The corresponding MPE's are 32%, 27%, and 72%, respectively. The Silverwood catchment chloride model has a poor MPE (72%), but good fit to the general trend based on the R^2 value (0.94). The Taylor Street catchment, conversely, yields a poor R^2 value (0.55) and better MPE (27%). The Taylor Street data and model fit are presented in Figure 5.27.

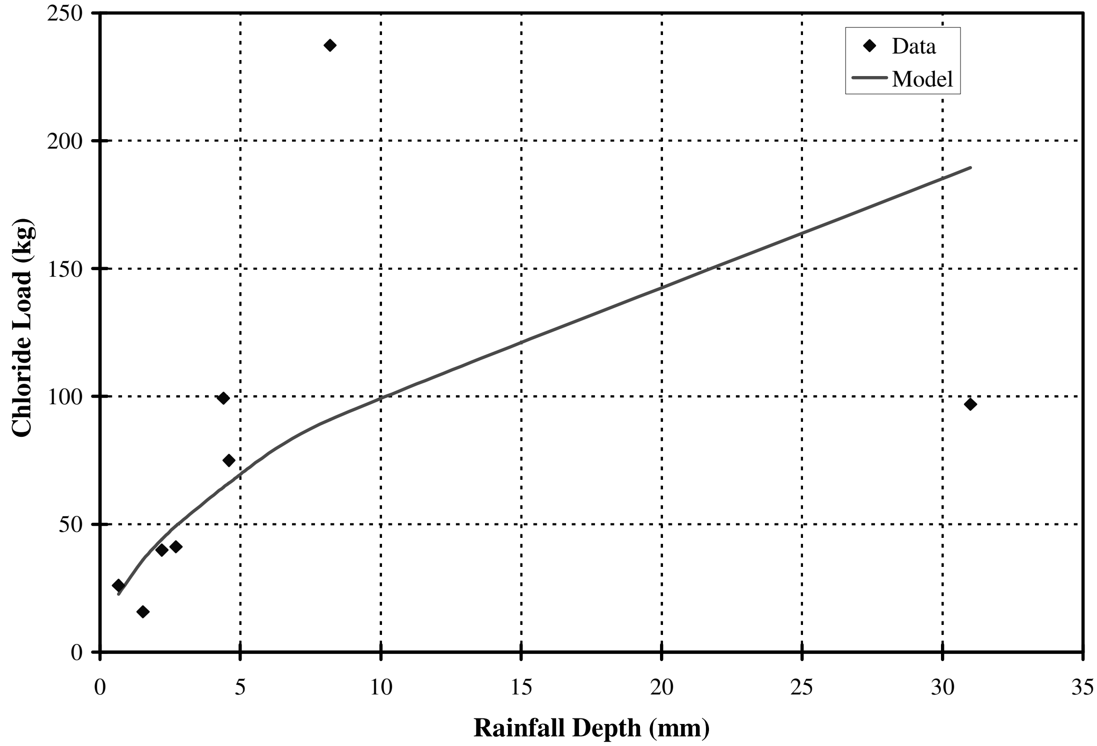


Figure 5.27 – Taylor Street catchment chloride regression model.

Figure 5.27 shows that the model fits the smaller values reasonably, but not the two larger values shown at the top and at the right of the figure. The two larger values come from events on June 17, 2002 and July 9, 2002. The events produced an estimated 237 kg and 98 kg of chloride, respectively. The June 17, 2002 event had the largest maximum instantaneous intensity (106 mm/hr) for events with water quality samples in this catchment, while the July 9, 2002 event had the largest total event depth (31 mm) for the same set of events. Both events are extreme results in the data set collected and cause the average trend to be skewed. Aside from the extreme values, the model represents the data reasonably. It is also possible that an alternate model formulation may better represent this data, however the amount of data available is insufficient to assess a new model formulation.

Similarity of predictor variables is observed in models for each of the Avenue B and Silverwood catchments. In the Avenue B catchment, the chloride, TKN, TP and COD models use the same predictor variables, however the exponents are different. The same is true for TKN and TP in the Silverwood catchment. Similar predictor variables within a particular catchment for different parameters may indicate that similar processes contribute to the load, but that the contribution is different.

In each catchment, the regression models were used along with the rainfall data to estimate the total catchment load over the rainfall-runoff period. Table 5.19 presents the estimated loads from each of the catchments, for the rainfall/runoff period, April 1 to October 31, in each of 2001 and 2002.

Table 5.19 – Catchment load estimation using the regression method for the rainfall-runoff period of 2001 and 2002.

Catchment	Rainfall/ runoff period	TSS load (kg)	TKN load (kg)	TP load (kg)	COD load (kg)	Chloride load (kg)
Ave B	2001	10,000	120	32	3,900	810
	2002	25,000	235	50	7,750	1,100
Taylor	2001	22,000	270	130	9,500	1,500
	2002	56,500	850	300	17,500	1,950
Silverwood	2001	15,000	30	8	1,100	52
	2002	32,500	45	13	1,550	76

The Taylor Street catchment has the largest estimated loads for all parameters, as would be expected given that it is significantly larger than the other catchments. The Silverwood catchment TSS loads are similar to but slightly larger than the Avenue B TSS loads. The Avenue B catchment TKN, TP, COD and chloride loads are larger than the Silverwood loads.

Using the SMC method, the Avenue B catchment loads were larger than the Silverwood catchment loads by at least 20%. Using the regression method, the same is true, except for TSS for which the trend is reversed. The loads estimated using the regression method are generally of the same order as the loads estimated using the SMC method. In each of the catchments and for each water quality parameter, the 2001 estimate is smaller than the 2002 estimate, which is as expected given the significant difference in rainfall.

The catchment loads estimated using the regression equations were also used to estimate the unit load parameter, which is presented in Table 5.20, for each of the water quality parameters. These unit loads only represent the rainfall runoff period used in this study (April 1 to October 31). For each of the water quality parameters, the Avenue B catchment has the largest average unit loads followed by the Taylor Street and Silverwood catchments, with the exception of TSS. The large unit loads in the Avenue B catchment are the result of large event loads and small catchment size. The average TSS unit load in the Taylor Street catchment is 64 kg/ha, which is smaller than the Silverwood average TSS unit load of 100 kg/ha. For the remaining parameters, the trend between the catchments reversed with the Taylor Street catchment having the larger unit load. The Taylor Street catchment TP and chloride unit loads are nearly an order of magnitude larger than that in the Silverwood catchment. The COD unit load in the Taylor Street catchment is four times larger than the Silverwood catchment.

Table 5.20 – Unit load parameter (regression method) for 2001 and 2002.

Catchment	Rainfall/ runoff period	TSS (kg/ha)	TKN (kg/ha)	TP (kg/ha)	COD (kg/ha)	Chloride (kg/ha)
Ave B	2001	140	1.6	0.42	52	11
	2002	340	3.1	0.67	100	15
	average	240	2.4	0.55	78	13
Taylor	2001	35	0.44	0.21	15	2.4
	2002	92	1.4	0.49	29	3.2
	average	64	0.91	0.35	22	2.8
Silverwood	2001	62	0.11	0.032	4.5	0.22
	2002	140	0.19	0.052	6.4	0.31
	average	100	0.15	0.042	5.5	0.27

With reference to Table 5.16, the Avenue B catchment unit load parameters estimated using the SMC method are similar but slightly smaller than the parameters estimated using the regression equations. In the Silverwood catchment, the average TSS unit load estimated using the SMC method (44 kg/ha) is about half of the unit load estimated using the regression equation (100 kg/ha). The difference may be due to the difference in the complexity of the estimation methods or the result of a poorly calibrated regression model. The remaining parameters determined using the regression equations are similar to but only slightly larger than the average parameters determined using the SMC method. In the Taylor Street catchment, the average unit load parameters estimated using the regression equations are similar to but larger than the

SMC estimated TSS, TKN, and TP. The COD and chloride unit load parameters are also similar, but the estimates using the SMC method are larger.

5.7 <u>CITY WIDE LOAD ESTIMATIONS</u>

Based on the characterizations developed in Section 5.6, city wide mass loads are estimated and presented herein. The estimated loads are based on the land use throughout COS which was apportioned into three categories: old residential (Taylor), new residential (Silverwood), and commercial/industrial (Avenue B). Industrial land uses were combined with commercial land uses because insufficient data were collected for industrial land uses alone (i.e. Sturgeon Drive catchment). The distribution of land uses is illustrated in Figure 5.28 along with the Thiessen polygons, which are used to distribute the rainfall from each of the rain gauges over the city.

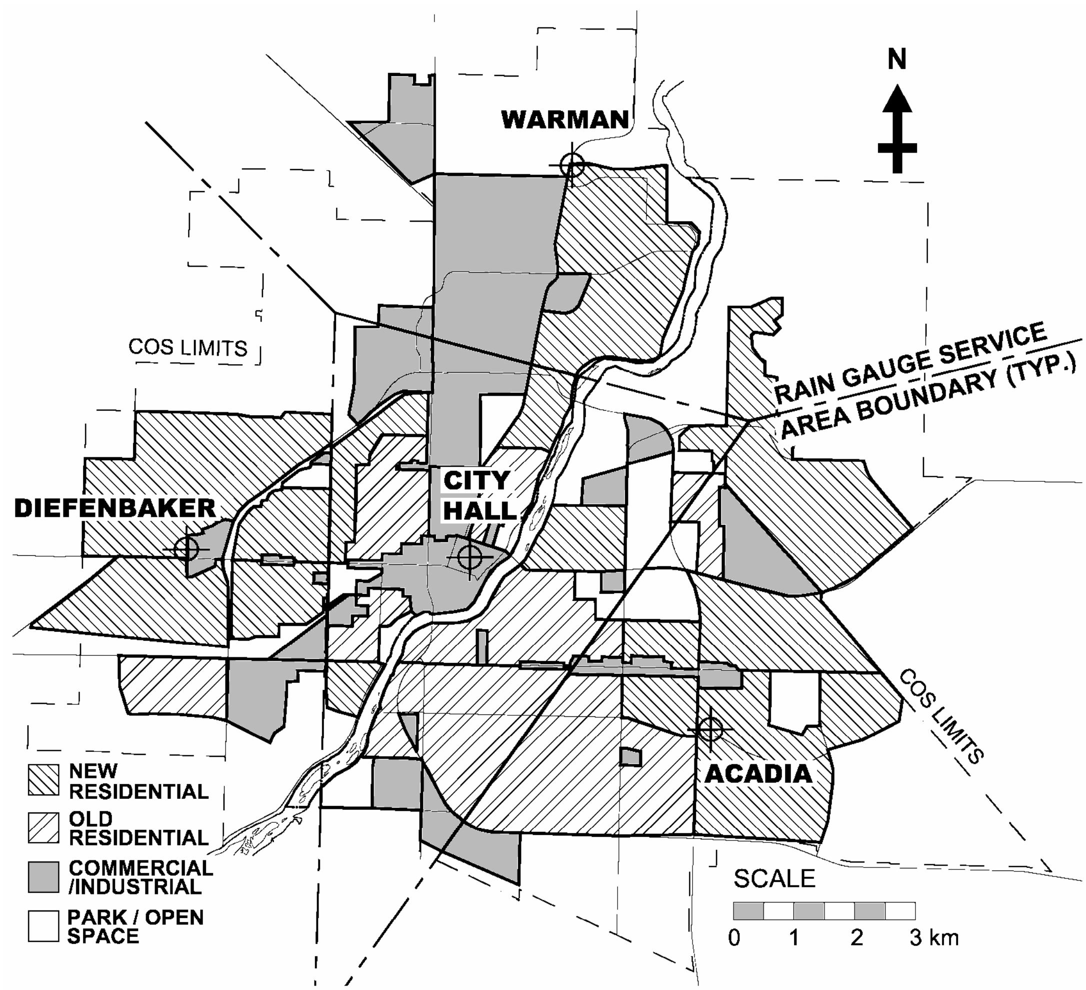


Figure 5.28 – Land use distribution within COS for load estimation.

Areas not hatched in Figure 5.28, but which within city limits, are open areas that, for this work, are assumed to not produce significant runoff, for example golf courses, large parks, and agricultural fields. The U of S campus is assumed to most closely resemble a new residential area. One notable exception is the John G. Diefenbaker International Airport in the northwest portion of the city. The data collected does not represent runoff from the airport, however airports are suspected to have significant urban runoff pollution loads associated with the large runoff volumes generated by impervious runways, taxiways and tarmacs (Richardson, 2006). Table 5.21 summarizes the area of each land use type within each rain gauge service area, which are used to predict the load. All three land use types are represented in each rain gauge service area, except for the Warman Road gauge, which does not include any older residential area.

Table 5.21 – Distribution of land use by area (ha) within each rain gauge service area.

Rainfall zone	Total	Older residential	Newer residential	Commercial /industrial	Open space (balance)
Acadia	4,375	786	1,616	310	1,661
Warman	4,215	0	763	789	2,664
City Hall	3,675	853	532	620	1,600
Diefenbaker	3,120	166	1,324	228	1,400
Total	15,385	1,806	4,235	1,947	7,326

Using the land distribution data, four estimates of city wide load were made: SMC method, regression equation method, unit load (SMC based) and unit load (regression based). The estimates are based upon the rainfall-runoff period (April 1 to October 31) in each of 2001 and 2002. The load estimate for each method is shown in Table 5.22 for each year along with the sum of the two years.

The loads estimated by the SMC method were found by multiplying the rainfall depth by the runoff coefficient and the area of the zone to estimate the runoff volume. The runoff volume was then multiplied by the SMC to yield the load. This method accounts for the spatial distribution of rainfall by using rain data from multiple rain gauges.

Table 5.22 – Summary of city wide load estimates for the rainfall-runoff period in 2001 and 2002.

Estimation method	Rainfall/ runoff period	TSS load (tonne)	TKN load (tonne)	TP load (tonne)	COD load (tonne)	Chloride load (tonne)
Unit load (SMC)	2001	470	4.2	0.93	180	23
	2002	780	7.2	1.6	300	40
	sum	1,250	11.4	2.5	480	62
SMC	2001	510	4.6	1.0	190	25
	2002	950	8.6	1.9	360	47
	sum	1,460	13	2.9	550	72
Unit load (regression)	2001	580	4.2	1.3	143	25
	2002	1,360	9.2	2.4	270	35
	sum	1,940	13.4	3.7	413	60
Regression	2001	480	5.1	2.2	160	28
	2002	1,570	12	3.3	320	43
	sum	2,050	17	5.5	480	72

The regression equations are based on specific catchments and thus the catchment area is implicit in the equation. The effect of area was accounted for by multiplying the estimate from the regression equations by the ratio of the area of interest to the area of the catchment for which the equation was developed. The method also accounts for the spatial distribution of rainfall by using data from multiple gauges.

The mass loads estimated using the unit loads are calculated solely on the basis of the unit load parameters, shown in Tables 5.15 and 5.19, and the catchment area. This method does not account for rainfall variation in any way.

In 2001, the TSS loads estimated using the SMC method are similar to those estimated using the regression method, but in 2002 the SMC method estimated 620 tonnes less than the regression equation. The total load for the two years is 590 tonnes less for the SMC method, however they are both within the same order of magnitude. The TKN, COD and chloride loads are similar for both methods. The SMC method produced TP loads that are smaller than the regression equation method, but the results are within the same order of magnitude.

The unit load (SMC) results are generally similar but slightly smaller than the loads estimated based on the SMC and distributed rainfall. The unit load (SMC) results are generally smaller than the loads estimated using the regression equation method except for COD, which has the same total over the two years. Compared with the unit load (SMC), the unit load (regression) estimates of TSS, TKN and TP are higher while the COD and Chloride loads are lower.

In general, the different estimation methods produce estimates of the COS load that are within the same order of magnitude. The estimations made using the regression equations produced the largest city wide loads, while the unit load (SMC method) produced the smallest city wide loads. The regression method is also the most complex estimation methodology as it takes into consideration several rainfall variables. The unit loads produced results smaller than the parameter that they are based upon. This is due to the simplification of the estimation and not accounting for rainfall variations.

5.8 TOXIC SUBSTANCES

Water quality parameters that fall into the category of toxic substances include heavy metals and pesticides. This section presents concentration results of both types of toxic substances. The parameters in this section include some of the parameters that are classified as "Priority Pollutants" by the U.S. EPA (EPA, 1982; WEF/ASCE, 1998). A select number of heavy metals is reported in this section with the full set of heavy metals analysis results listed in Appendix I.

The results presented in this section are end-of-pipe concentrations and make no attempt to account for dilution when the urban runoff water enters the river. The concentrations presented herein will be reduced by dilution many times over when the runoff mixes with the river water. The dilutions available in the river are dependent upon the flow rate in the river and the flow rate of the runoff. Detailed examination of the mixing of the urban runoff into the river water is beyond the scope of this work.

Two guideline concentrations are presented with the various results. The first guideline concentration is for the protection of drinking water supplies (i.e. human health). The second is for the protection of freshwater aquatic life. Generally, the latter is more restrictive for the parameters examined herein.

Three types of samples were collected: mid-event grab samples, baseflow samples, and rainfall event composite samples. The mid-event grab samples were collected at random times (i.e. flow rate unknown) during one rainfall event early in the project to provide an initial snapshot-in-time example of the concentrations. The baseflow samples give an indication of the concentration being discharged during dry weather while rainfall-runoff event composite samples provide an average concentration over the rainfall event.

Table 5.23 shows the reference guideline heavy metal concentrations from the Canadian Council of Ministers of the Environment (CCME, 2003) and the mid-event grab samples from June 6, 2001.

Table 5.23 – Heavy metal guideline concentrations and mid-event grab samples June 6, 2001 (mg/L).

Metal	Drinking Water[a]	Freshwater Aquatic[a]	Location		
			Sturgeon	Silverwood	Taylor
Mercury	0.001	0.0001	<0.00005[b]	<0.00005	<0.00005
Arsenic	0.025	0.05	0.004	<0.002	0.004
Cadmium	0.005	0.0002 - 0.008	0.001	<0.001	0.022
Chromium	0.05	0.002	0.055	0.027	0.046
Copper	1.0	0.002 - 0.004	0.04	0.008	0.028
Iron	0.3	0.3	14	4.1	17
Lead	0.010	0.001 - 0.007	0.064	0.006	0.062
Silver	0.01	0.0001	<0.001	<0.001	<0.001
Zinc	5.0	0.03	0.42	0.094	0.25

[a] – maximum guideline concentrations (from CCME, 2003)
[b] – The '<' symbol indicates that the parameter was not detected at the noted concentration (i.e. detection limit)

For the three samples shown in Table 5.23, mercury and arsenic were both found to be less than the respective guideline concentrations. Cadmium was found to range from <0.001 mg/L to 0.022 mg/L. The Taylor Street concentration of 0.022 mg/L is greater than both the drinking water and the freshwater aquatic life protection guidelines. Dilution of about four times is required to satisfy the drinking water guideline and about three times to satisfy the freshwater aquatic life protection guideline. It should be noted that even concentrations below the noted detection limit (0.001 mg/L) for cadmium would the lower freshwater aquatic guideline concentration.

The chromium results range from 0.027 mg/L to 0.055 mg/L. All results exceed the aquatic protection guideline while only the largest result exceeds the drinking water guidelines. Dilutions of 1.1 to 27 times are required to reduce the concentrations to fresh water aquatic guideline values. Copper concentrations range from 0.008 to 0.04 mg/L. These concentrations are well below the drinking water guideline, however the concentrations all exceed the aquatic protection guidelines. Dilutions of five to 10 times are required to meet the guidelines. Iron concentrations range from 4.1 to 17 mg/L. Dilutions of 13 to 57 times are required to meet the iron guideline of 0.3 mg/L. Lead concentrations range from 0.006 to 0.064 mg/L. The two upper values exceed both the drinking water guideline and the fresh water aquatic life protection guideline. The lower value lies on the upper bound of the freshwater aquatic protection guideline. Dilutions of approximately six times are required to meet the drinking water guidelines. Silver was not detected in the grab samples. Zinc was detected in concentrations from 0.094 to 0.42 mg/L. The zinc results exceed the aquatic protection guideline and require dilutions of three to 14 times to meet the guideline.

Table 5.24 presents the heavy metal analysis results from baseflow samples in 2001 and 2002. A total of six samples were obtained from the four study catchments.

Table 5.24 – Heavy metals analysis results – baseflow samples 2001 and 2002 (mg/L).

Metal	Sturgeon	Silverwood		Taylor		Ave B
	31/07/01	31/07/01	09/10/02	31/07/01	09/10/02	31/07/01
Mercury	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
Arsenic	0.0008	0.001	<0.002	0.0007	0.0032	0.0008
Cadmium	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Chromium	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Copper	0.005	0.003	0.002	0.004	<0.001	0.004
Iron	0.053	0.048	<0.001	0.11	0.085	0.33
Lead	<0.002	<0.002	<0.002	<0.002	<0.002	0.002
Silver	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Zinc	0.021	0.006	<0.005	0.017	0.011	0.053

Mercury was undetected and below guideline concentrations in all of the baseflow samples. Arsenic was detected in concentrations ranging from 0.0008 to 0.0032 mg/L, which are below both guideline values. Cadmium and chromium were not detected in the baseflow samples. Copper was found in five of six baseflow samples and ranged from 0.002 to 0.005 mg/L. The concentrations of copper are lower than the drinking water guideline concentrations, but are the same as or slightly greater than the freshwater aquatic life protection guideline. Dilutions of one to three times are required to meet the guideline concentration. Iron was detected in five of the six baseflow samples, ranging from 0.048 to 0.33 mg/L. The 0.33 mg/L result is slightly higher than the guideline concentration of 0.3 mg/L. Dilution of 1.1 times of more is required to meet the guideline concentration. Lead was detected in one of the six baseflow samples. The detected concentration is below the drinking water guideline and at the low end of the range of the freshwater aquatic guideline concentrations. A dilution of two times or more will cause the concentration to fall below the lower guideline. Silver was not detected in the baseflow samples. Zinc was detected in five of six baseflow samples. The range of concentration is 0.006 to 0.053 mg/L. All of the sample concentrations were below the guideline values, expect for one, which would require dilution of approximately two times to reduce it to less than the freshwater aquatic life guideline concentration.

The heavy metal analysis results from event composite samples in 2001 and 2002 are presented in Table 5.25. A total of six event composite samples were analyzed for heavy metals. As with all of the previous heavy metal samples, mercury was not detected in the event composite samples. Arsenic was detected in five of the six composite samples and is below both the drinking water and freshwater aquatic life protection guidelines. Cadmium was detected in one of five samples and one sample had no reported result. The concentration detected, 0.002 mg/L, is below the drinking water guidelines and in the middle of the range of guidelines for freshwater protection. Dilution greater than 10 times would be required reduce the detected concentration below the minimum guideline value. Chromium was detected in all composite samples, ranging in concentration from 0.004 to 0.065 mg/L, with one sample above the drinking water guideline (0.05 mg/L) and two samples above the freshwater guideline (0.002 mg/L). Dilutions of 1.5 to 33 times are required to reduce the detected chromium concentrations below the guideline values. Copper was detected in all composite samples. The concentration range is 0.078 to 0.19 mg/L, all of which are below the drinking water guideline. All of the samples the freshwater aquatic life protection guideline, requiring dilutions of 20 to 50 times to bring the concentrations below the guideline. Iron was detected in all six composite samples and all samples both the drinking water and freshwater aquatic guidelines. Dilutions of 2.7 to 10 times are required to reduce the concentrations below the guidelines. Lead was detected in all samples, ranging from 0.016 to 0.056 mg/L. Five of the six samples equal or exceed the drinking water guideline and all of the samples exceed the freshwater aquatic life protection guideline. As with all of the other heavy metal samples, silver was not detected in the composite samples. Zinc concentrations range from 0.068 to 0.47 mg/L. All of the samples are below the drinking water guideline, while all of the samples exceed the freshwater aquatic life protection guideline. Dilutions of three to 16 times are required to bring the concentrations below the aquatic guideline.

Table 5.25 – Heavy metal analysis – event composite samples in 2001 and 2002 (mg/L).

Metal	Sturgeon	Silverwood		Taylor		Ave B
	14/08/01	16/07/01	25/08/02	16/07/01	25/08/02	14/08/01
Mercury	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
Arsenic	0.0008	0.004	0.0028	0.0058	<0.002	0.0018
Cadmium	NR	<0.001	<0.001	<0.001	<0.001	0.002
Chromium	0.022	0.015	0.015	0.004	0.005	0.065
Copper	0.037	0.011	0.19	0.015	0.08	0.078
Iron	14	0.81	6.3	7.9	2.9	30
Lead	0.026	0.006	0.016	0.02	0.01	0.056
Silver	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Zinc	0.32	0.068	0.13	0.13	0.096	0.47

NR – no result

Table 5.26 presents pesticide guideline concentrations and results from mid-event grab samples collected June 6, 2001. Background information for each pesticide is provided in Appendix B. Only three samples were obtained. The third sample, collected from the Sturgeon Drive catchment, was destroyed in transit to the laboratory. All parameters in both samples, except for Mecoprop, are below detection limits. The concentrations of Mecoprop, 0.0008 and 0.0015 mg/L, do not exceed guideline concentrations.

Table 5.26 – Pesticide guideline concentrations and mid-event grab samples June 6, 2001 (mg/L).

Parameter	Drinking Water	Freshwater Aquatic	Location	
			Silverwood	Taylor
Trifluralin	0.045	0.00020	<0.00005	<0.00005
Triallate	None	0.00024	<0.00005	<0.00005
Diclofop-Methyl	0.009	0.0061	<0.00010	<0.00010
Mecoprop (MCPP)	none	0.004	0.0008	0.0015
2,4-D	0.1	0.004	<0.00010	<0.00010
2,4-DB	2	0.004	<0.00020	<0.00020
Dichlorprop	none	0.004	<0.00010	<0.00010
MCPA	none	0.0026	<0.00010	<0.00010
Dicamba	0.12	0.010	I/F	I/F
Bromoxynil	0.005	0.005	<0.00010	<0.00010

I/F – interference, no result

Table 5.27 presents the pesticide concentration results for the baseflow samples from the Sturgeon Drive and Silverwood catchments. Two samples from each catchment were collected and all samples were below the detection limits.

Table 5.27 – Pesticide baseflow sample concentration results for the Sturgeon Drive and Silverwood catchments in 2001 (mg/L).

Parameter	Sturgeon		Silverwood	
	31/07/01	28/08/01	31/07/01	28/08/01
Trifluralin	<0.00004	<0.00004	<0.00004	<0.00004
Triallate	<0.00004	<0.00004	<0.00004	<0.00004
Diclofop-Methyl	<0.00020	<0.00020	<0.00020	<0.00020
Mecoprop (MCPP)	<0.00004	<0.00004	<0.00004	<0.00004
2,4-D	<0.00010	<0.00010	<0.00010	<0.00010
2,4-DB	<0.00010	<0.00010	<0.00010	<0.00010
Dichlorprop	<0.00004	I/F	<0.00004	I/F
MCPA	<0.00004	<0.00004	<0.00004	<0.00004
Dicamba	<0.00010	<0.00010	<0.00010	<0.00010
Bromoxynil	<0.00010	<0.00010	<0.00010	<0.00010

I/F – interference, no result

Table 5.28 presents the pesticide concentration results for baseflow samples from the Taylor Street and Avenue B catchments. All of the parameters in all of the samples, except for Mecoprop in the Taylor Street catchment sample from August 28, 2001, were below the detection limit. The Mecoprop concentration is well below the fresh water aquatic guideline.

Table 5.28 – Pesticide baseflow sample concentration results for the Taylor Street and Avenue B catchments (mg/L).

Parameter	Taylor		Ave B	
	31/07/01	**28/08/01**	**31/07/01**	**28/08/01**
Trifluralin	<0.00004	<0.00004	<0.00004	<0.00004
Triallate	<0.00004	<0.00004	<0.00004	<0.00004
Diclofop-Methyl	<0.00020	<0.00020	<0.00020	<0.00020
Mecoprop (MCPP)	<0.00004	0.0017	<0.00004	<0.00004
2,4-D	<0.00010	<0.00010	<0.00010	<0.00010
2,4-DB	<0.00010	<0.00010	<0.00010	<0.00010
Dichlorprop	<0.00004	<0.00004	<0.00004	I/F
MCPA	<0.00004	<0.00004	<0.00004	<0.00004
Dicamba	<0.00010	<0.00010	<0.00010	<0.00010
Bromoxynil	<0.00010	<0.00010	<0.00010	<0.00010

I/F – interference, no result

Table 5.29 presents the concentration results from event composite samples from all three catchments sampled in 2001: Taylor Street, Silverwood, and Sturgeon Drive. The majority of the parameters are below the detection limit. Mecoprop and MCPA were detected in the Silverwood sample of August 8, 2001 at concentrations slightly higher than the freshwater aquatic life protection guideline. 2,4-D, dicamba, and bromoxynil were detected but the concentrations are below the drinking water MAC and freshwater aquatic life protection guidelines. Pesticides were not detected in composite samples from the Taylor Street catchment.

Table 5.29 – Pesticide event composite sample concentrations (mg/L).

Parameter	Sturgeon			Silverwood	Taylor
	15/06/2001 A	15/06/2001 B	17/06/01	08/08/01	23/06/01
Trifluralin	<0.00001	<0.00001	<0.00001	<0.00001	<0.00001
Triallate	<0.00004	<0.00004	<0.00004	<0.00004	<0.00004
Diclofop-Methyl	<0.00004	<0.00004	<0.00004	<0.00004	<0.00004
Mecoprop (MCPP)	<0.00004	<0.00004	<0.00004	0.0042	<0.00004
2,4-D	0.0013	0.0018	0.0014	0.0015	<0.00010
2,4-DB	<0.00020	<0.00020	<0.00020	<0.00020	<0.00020
Dichlorprop	<0.00010	<0.00010	<0.00010	<0.00010	<0.00010
MCPA	<0.00004	<0.00004	<0.00004	0.0049	<0.00004
Dicamba	I/F	I/F	I/F	0.0011	I/F
Bromoxynil	<0.00004	0.0004	0.0003	<0.00010	<0.00004

I/F – interference, no result

5.9 NOTABLE OCCURRENCES

During the field portion of the research project, there were some notable occurrences germane to the study and/or urban runoff within COS. This section contains descriptions of these occurrences followed by some general observations. These occurrences provide further insight into the urban runoff quality that is not necessarily reflected in the previous characterizations and descriptions.

Asphalt Contamination

On June 18, 2001, the automated sampler at the Sturgeon Drive catchment metering manhole collected urban rainfall-runoff samples clearly contaminated with asphalt oil. The contamination was identified by the strong odour from the samples, as well as by residues left on the intake strainer, suction tubing, pump tubing, and sampler bottles. Approximately the last third of the samples (17 to 24) were heavily contaminated, while samples 12 to 16 had an odour of asphalt.

The sampling sequence began at 21:54 on June 18, 2001 when the depth of flow in the manhole was 0.10 m, which corresponds to a volumetric flow rate of approximately 0.070 m^3/s. Samples 12-16 were collected between 22:37 and 22:57, while samples 17 to 24 were collected between 22:57 and 23:26. The event hydrograph and TSS pollutograph are shown in Figure 5.29.

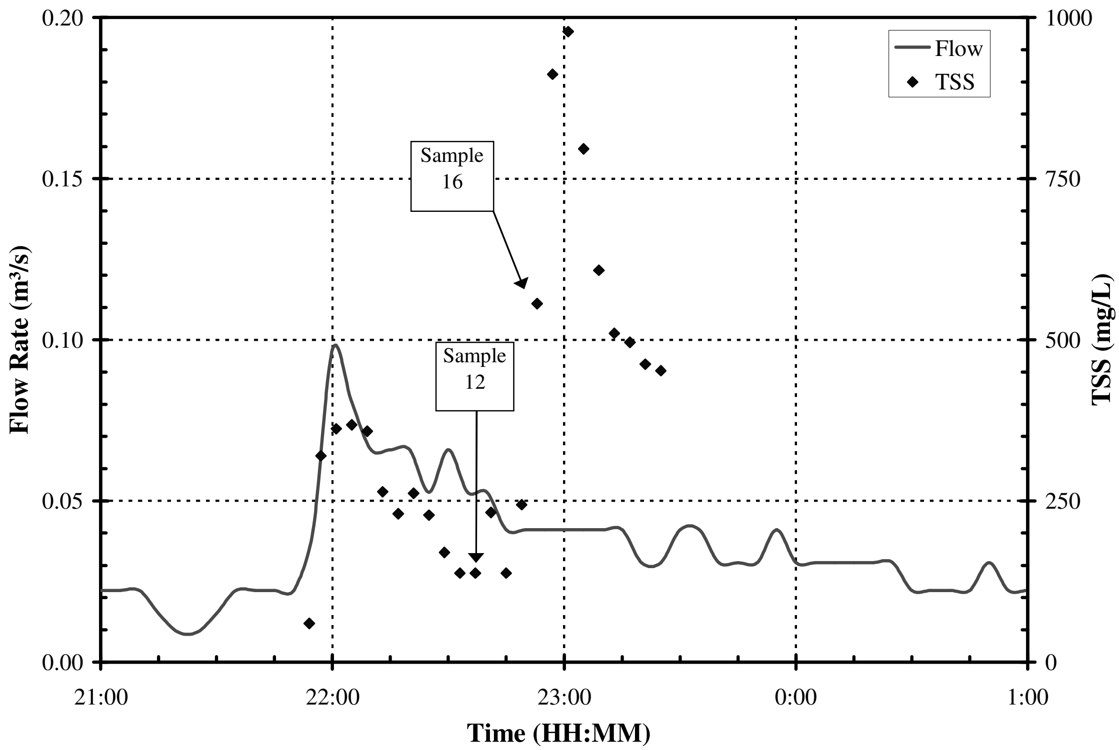


Figure 5.29 – Hydrograph and pollutograph for Sturgeon Drive event on June 18, 2001.

The rainfall event was relatively small in comparison to other recorded Sturgeon events. The peak flow rate was approximately 0.10 m³/s. The average rainfall intensity for this event was 5.8 mm/hr, the maximum intensity was 22.5 mm/hr, the rainfall depth was 1.8 mm, the duration of the rainfall was 19 minutes, and the antecedent dry period was 1.4 days. Between samples 16 and 24, 68 m³ of flow was recorded by the flow meter.

The first of the contaminated samples was collected on the declining leg of the hydrograph, more than 30 minutes after the peak in the flow rate. The asphalt odour was noted to have begun at sample 12 and the heavy contamination began at sample 16. All of the equipment that the contaminated samples came into contact with was irreparably fouled, except for the strainer which was able to be satisfactorily cleaned. Sample 12 had a TSS concentration of 138 mg/L. The peak TSS concentration prior to the asphalt contamination was 368 mg/L, while the peak during the contamination was 978 mg/L. The pollutograph clearly indicates an uncharacteristic secondary spike in the TSS concentration, which occurs in the same samples as the significant odour and obvious contamination. The samples were also significantly discoloured. Figure 5.30 is a photograph of the samples aligned from 1 to 24 (left to right). The samples are in clear glass turbidity bottles. The bottles show a slight change in colour in the first 10 samples and significant darkening in samples 16 to 24, similar to what is shown by the pollutograph.

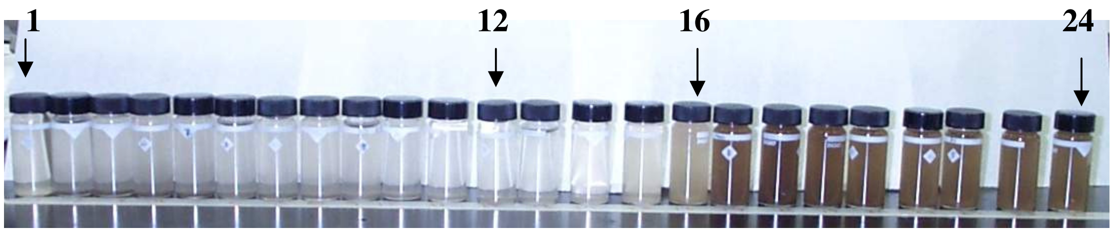


Figure 5.30 – Sturgeon Drive asphalt contaminated samples June 18, 2001, chronologically left to right.

After retrieval of the samples, an excursion through the catchment was taken and two paving projects were observed. The first paving project was in the same block as the metering site and would have produced contamination early in the sample set. The second paving project was on Lenore Drive between Pinehouse Drive and Warman Road. Based on review of pipe length to the project, this is relatively close to the monitoring location and the contamination would have been observed earlier in the sample set than sample 16. As such it was concluded that the two observed paving projects were not likely the source of the contamination.

Based on the extent of the fouling of the sampling equipment, the number of heavily contaminated samples and the recorded flow rate, it is likely that the volume of asphalt oil carried by the storm water is greater than what would be expected to be accidentally spilled during the course of a paving project. The incident was immediately reported to SE and COS.

<u>Varsol-like Odour</u>

On a few occasions in 2001, a varsol-like odour was noted at the Sturgeon Drive metering and sampling sites. The odour was strong enough to be detected by a person standing next to the open six metre deep manhole. The odour was similar to the tar remover used in the cleaning of asphalt equipment. In 2002, no occurrences of this odour were noted.

Abnormal Baseflow Colour

On July 12, 2001 during the installation of water quality sampling equipment at the Sturgeon Drive site, the baseflow was observed to have a yellow colour similar to a deep mustard colour. A sample was collected and negligible odour was noted. The sample was sent to the Provincial Lab for analysis, the results of which are shown in Table 5.30.

Table 5.30 – Water quality analysis results for baseflow sample collected July 12, 2001 from Sturgeon Drive catchment.

Parameter	Result (mg/L)	Parameter	Result (mg/L)
TSS (fixed)	5	Potassium (ICP)	10
TSS (volatile)	16	Sulphate	530
TSS (total)	20	NH3 - N	NR
Conductivity	1640	NO3 - N	0.93
pH	7.4	TKN	4.9
Turbidity	8.62	TP	0.37
Bicarbonate	312	OP	NR
Total Alkalinity	256	DOC	111
Calcium (ICP)	174	BOD_5	NR
Magnesium (ICP)	83	COD	359
Sodium (ICP)	82	TDS (Calc)	1268
Total Hardness (Calc)	776	TC (orgs/100 mL)	1.1E+06
Chloride	77	FC (orgs/100 mL)	2.3E+03

The majority of the parameters in Table 5.30 are similar to the average baseflow values (see Table 5.6) except for TSS, TKN, TP, DOC, and COD, which have values greater than the maximum baseflow results for those parameters. The results indicate that the source is organic in nature, composed of organic nitrogen and organic carbon, and whose chemical decomposition requires significant amounts of oxygen.

Chlorine

In August 2002, chlorine tests were performed on baseflow samples. The testing was initiated partly due to an overpowering odour of chlorine around the Avenue B outfall on August 2, 2002 during baseflow. A definite explanation of the source of the chlorine smell was not found, however watermain maintenance operations are suspected. After this occurrence, chlorine tests were performed on the baseflow samples using a portable DPD based colour comparator. The samples were tested for total chlorine. Table 5.31 summarizes the results.

Table 5.31 – Summary of chlorine tests on baseflow samples (mg/L Cl_2).

Parameter	Avenue B		Taylor Street		Silverwood	Sturgeon
Chlorine	13/08/02	09/10/02	13/08/02	09/10/02	09/10/02	09/10/02
Total	0.2	0	0.8	0.2	0	0

In both the Avenue B and Taylor Street catchments, chlorine was detected in the baseflow. The August 13, 2002 sample from the Avenue B catchment and the October 9, 2002 sample from the Taylor Street catchment both had small results. The Taylor Street sample collected August 13, 2002 had a total chlorine concentration of 0.8 mg/L. The detected concentration of total chlorine exceeds the minimum residual required for Saskatchewan drinking water distribution systems of 0.5 mg/L. COS maintains a concentration of approximately 1.5 mg/L total chlorine entering the distribution system. The fresh water aquatic habitat protection guideline concentration for chlorine species is 0.0005 mg/L (CCME, 2003). Based on the results in Table 5.31, it is likely that treated drinking water is entering the storm sewer system in somewhat significant quantities. Considering the results and the very low tolerance for chlorine in aquatic habitat, activities which discharge large quantities of treated drinking water to the storm sewer (i.e. at flow rates nearing half of the baseflow rate) are recommended to dechlorinate the water prior to discharge. This is especially true for watermain disinfection where very high concentrations of chlorine are used.

Hydrocarbons

On the free liquid surface of several samples, primarily in 2001 in the Sturgeon Drive and Silverwood catchments, ‘rainbows’ were observed, similar to those seen when hydrocarbons have been spilled on water. Specific laboratory analysis of hydrocarbons was not undertaken, however the presence of the rainbows suggests the presence of hydrocarbons.

Litter

Litter was a significant problem. Among other problems, it collects on sampling equipment, plugs suction strainers and causes the sample volumes to be erratic. It also collects on the safety grating at outfalls, creating an unsightly mess as shown in Figure 5.31, which was taken at the Avenue B site in 2002.

Figure 5.31 – Example of litter caught on grating at the Avenue B outfall in 2002.

The litter in Figure 5.31 was comprised of dead vegetation, plastic bags, plastic straps, ribbons, cigarette butts, cigarette package wrappers, paper, old electrical wires, prophylactics, and a decomposing rodent. This composition was not uncommon at any of the outfalls of the monitored catchments. On one occasion, a snow dump sign on a

3 m pole, as shown in Figure 5.32, was observed in the washout from the Taylor Street outfall.

Figure 5.32 – Snow dump sign in the Taylor Street outfall washout in 2001.

Litter caught on outfall gratings causes an increase in the head loss across the grating. The increased head loss causes greater force to be placed on the anchorage system of the grating. The increased head loss can also cause an increase in the upstream depth of flow, which in extreme circumstances may cause the storm sewer system to surcharge.

CHAPTER 6 DISCUSSION OF RESULTS

6.1 WATER QUALITY CHARACTERIZATIONS

6.1.1 SMC

SMC's for five water quality parameters were developed. The EMC's for each parameter plotted with a linear trend on log-normal probability plots, which indicates that the data are acceptably represented by the log-normal distribution. The appropriate measure of central tendency in the log-normal distribution is the geometric mean. Charbeneau and Barrett (1997) and the EPA (1982) also found urban runoff EMC data to be best represented by the log-normal distribution.

While the data are, in general, acceptably represented by the log-normal distribution, some of the log-normal plots had points that did not fit the trend. The primary cause of the poor fits is likely poor coverage of the rainfall-runoff event by water quality samples (i.e. the available sample containers ran out significantly before the event was complete). Ultimately, events such as these underestimate the actual load discharged and cause any characterization to be underestimated. In future work where sufficient amounts of data are available, events with poor sample coverage should be excluded from analysis because they underestimate the event load. In this study, these events are included because of the limited data available but with the understanding that the event loads and related characterizations are likely underestimated.

The SMC results, summarized in Table 6.1, show that the Avenue B and Taylor Street catchments are generally similar for all parameters, and the Silverwood catchment SMC's are generally lower than the other two study catchments. The variability of the water quality parameters appears to be greater than the differences between the catchments, as indicated by the one standard deviation range relative to the SMC's. T-tests were performed to determine if the SMC's are significantly different from each other at the 5% level of significance. The t-tests indicate that the differences between the SMC's are not significant, with respect to the variability in the data, except for chloride in the Silverwood catchment. The Silverwood chloride SMC is significantly different from the Avenue B and Taylor Street SMC's. The difference between the Taylor Street and Avenue B chloride SMC's is not significantly different. On this basis, it can be concluded that the SMC results, on the whole, do not show any significant difference between the land uses. Therefore, overall average SMC's are also shown in Table 6.1.

Table 6.1 – Summary of COS SMC's and one standard deviation ranges.

Parameter	Avenue B		Taylor Street		Silverwood		Overall
	SMC (mg/L)	One std dev range (mg/L)	SMC (mg/L)	One std dev range (mg/L)	SMC (mg/L)	One std dev range (mg/L)	average SMC (mg/L)
TSS	210	110-400	190	94-380	160	13-780	185
TKN	2.2	0.61-7.6	2.5	2.0-3.1	0.73	0.11-4.7	1.8
TP	0.45	0.13-1.6	0.53	0.35-0.80	0.21	0.033-1.3	0.40
COD	75	24-250	100	52-200	55	8.0-370	75
Cl^-	13	3.9-41	15	8.0-33	1.9	0.43-8.8	10

NURP (US EPA, 1982) also concluded that apparent differences in SMC's between land use types are negated by the variability of each parameter within the land use types. Therefore, determination of SMC's based on land use is not justified. This study supports that conclusion. The result does not, however, prevent the use of land use specific data in load calculations where such data have been specifically collected. Smullen et al. (1999) note that, even in light of the NURP conclusion, many urban runoff investigators continue to use land use specific SMC's.

With reference to the SMC's from other sources presented in Table 2.2, the COS SMC results in Table 6.1 are generally similar to those presented, with a tendency to be slightly larger. The NURP results are generally smaller than the COS SMC's. Exceptions are the Silverwood TKN, TP and COD SMC's, which are slightly smaller than the NURP SMC's.

SMC's for all urban sites from the update to the U.S. nationwide urban runoff quality database, reported by Smullen et al. (1999), generally showed reductions from the NURP results. The reported TSS and TP SMC's are smaller than those found in this study, while the TKN and COD SMC's are comparable.

The Vancouver SMC's for all urban sites, reported by Macdonald (2003), are smaller than the Avenue B and Taylor Street catchments SMC's and similar to the Silverwood SMC's. The large differences in the SMC's (approximately two to four times) is likely due to differences in the climates. Vancouver receives an average of 1274 mm of rainfall annually (Environment Canada, 2003), which is substantially greater than Saskatoon's average annual rainfall of 266 mm.

SMC's for all urban land uses from Wisconsin and Denmark, reported by Bannerman et al. (1996) and Ellis and Hvitved-Jacobsen (1996), respectively, are similar to the COS SMC's. The COS TSS SMC's are larger than the Denmark TSS SMC.

Choe et al. (2002) reported SMC's for specific land uses for the Republic of Korea (South Korea). The SMC's reported for South Korea are much larger than the COS SMC's. The Korean catchments studied do not receive any street sweeping. Choe et al. (2002) suspect that the lack of street sweeping is one reason that the SMC's are significantly larger than the majority of reported SMC's. Other potential explanations of the differences include differences in the build-up processes and regionally less stringent environmental discharge requirements.

Duncan (1997) reported TSS, TKN, and TP SMC's from pooled worldwide data as well as the log space standard deviation from which the real space one standard deviation range can be determined. The SMC's are similar to each other. The COS standard deviation ranges are similar to and slightly larger than those reported by Duncan (1997). The similar one standard deviation ranges indicate that a comparable amount of variability is seen in both the small sample size COS data and the large sample size used by Duncan (1997). This result suggests the sample variability reasonably represents the true variability.

In summary, the COS urban runoff SMC's are similar but slightly larger than those from other jurisdictions with the exception of South Korea. Greater similarity is observed in the data from Wisconsin, which has a climate that is more comparable to that in COS. This suggests that climate may be significant in the determination of characteristic parameters, such as the SMC. By comparison to sample variability from larger data pools, the variability of the COS SMC's appear to be representative of the true variability.

6.1.2 Regression Analysis

Multiple variable linear regression analyses were undertaken to characterize the water quality parameter mass loads discharged to the South Saskatchewan River. Fifteen equations were generated, one for each of five water quality parameters in each of the three catchments. The data set used for this analysis is limited. This analysis was pursued as a means of comparison with the SMC method and as a demonstration of the procedure. If used in other work, the resultant regression equations should be applied cautiously, respecting the ranges of data used in the calibration (Table 5.17).

The regression equations produced median percent calibration errors (MPE) ranging from 19 to 84%, with coefficients of determination (R^2) ranging from 0.47 to 0.98. MPE was used to represent the average error about each point. R^2 was used to indicate the overall model fit to the data. The majority of the models had MPE's of about 20% to 50% and R^2 greater than 0.75. The model fit parameters are generally reasonable. Based on R^2 being quite good compared to the MPE, the goodness-of-fit statistics indicate that the models better represent the overall trend of the data than individual points, which may be subject to large relative error. This finding suggests that the regression equations are more applicable to estimating longer term loads, such as annual or planning level estimates, where the analysis does not require as precise a result (i.e. order of magnitude type analysis).

LeBouthillier et al. (2000) had R^2 values ranging from 0.63 to 0.80, and percentage errors ranging from 3% to 88%. USGS had R^2 values ranging from 0.55 to 0.86 and average percentage errors from 68% to 334%. The coefficients of determination and median percentage error from this study indicate that slightly better fit to the data was achieved in this study.

Albeit somewhat simplistic, insight can be gained by examining the similarities in the regression equations. Within a given catchment, the predictor equations had a common variable. In the Avenue B and Taylor Street catchments, the variable common to all equations was the event rainfall depth, while in the Silverwood catchment, the common variable was the average intensity. For a given water quality parameter, the equations for each catchment are different; that is, different combinations of predictor variables were required. These two points (similarity within catchment, difference between parameters) indicate that the individual catchment characteristics have a greater role in the water quality parameter loads than parameter specific mechanisms. The catchments were specifically chosen to represent different land uses. Therefore, the regression equations indicate that land use is a significant factor in the pollutant load generation. This finding is contrary to the finding for the SMC analysis discussed in the previous section.

With respect to land use, the regression results appear to show a contrary conclusion to the SMC results. With the exception of one SMC, the SMC results show no significant difference in the SMC's between catchments, while the regression equations show difference between the catchments. The regression equations utilize rainfall variables as input, which relate to the volume of runoff that is generated from the catchment. No other accounting or normalization for the runoff volume is made in the regression equations. The difference between the two methods is that the regression equations implicitly include the runoff volume, but the SMC is normalized to the volume of runoff. For example, the relatively 'dirty' and impervious Avenue B catchment had the largest unit loads and relatively large runoff volumes. However, when the large loads were normalized by the large volumes, the SMC for the commercial catchment (Avenue B) was not significantly different from the residential catchment (Taylor). Therefore, the effect of land use is implicit in the volume of runoff.

The rainfall depth is the most frequently significant variable, being utilized in 10 of the 15 regression equations. The other variables that were found to be significant are average rainfall intensity, duration, and maximum instantaneous rainfall intensity. LeBouthillier et al. (2000) and USGS (Driver and Tasker, 1990) also found the rainfall depth to be the most frequent significant variable. LeBouthillier et al. (2000) also found average intensity and the maximum intensity as significant variables. In addition to the depth, USGS found area, duration, land use, mean annual rainfall, and mean annual nitrogen load in precipitation (TKN prediction only) as significant variables. In the COS study, the area, land use, and mean annual rainfall are constant. Area and land use were constant because only one example of each land use type was used. The mean annual rainfall is constant for the city. Thus in this study, all of the variables found to be significant in the prediction of the water quality parameter mass load were also found to be significant in other studies.

The regression results suggest that the largest loads will be discharged during periods when the rainfall depth is the greatest. In Saskatoon, the greatest rainfall depths generally occur during the months of June and July, thus an abatement strategy, such as an increased frequency of street sweeping prior to and during this period, will yield the greatest return in terms of pollution interception. Also, since the antecedent dry period is not significant in any of the models, the period when rainfall events occur most frequently will also yield the greatest amounts of material. In Saskatoon, this period is generally in June and July.

6.1.3 Unit Loads

The unit load is the mass of a water quality parameter divided by the area from which it is generated. This parameter is intended to show different area based mass load generation rates between the catchments. Table 6.2 presents a summary of the unit loads over the April 1 to October 31 period, based on the average loads of 2001 and 2002. Comparing the two methods used to generate the estimates, the results are generally similar, however some of the parameters show some difference.

Table 6.2 – Summary of estimated unit load parameters for April 1 to October 31.

Catchment	Method	TSS (kg/ha)	TKN (kg/ha)	TP (kg/ha)	COD (kg/ha)	Chloride (kg/ha)
Ave B	SMC	190	2.0	0.41	68	12
	Regression	240	2.4	0.55	76	13
Taylor	SMC	50	0.66	0.14	26	4.0
	Regression	64	0.91	0.35	22	3.0
Silverwood	SMC	44	0.19	0.06	15	0.5
	Regression	100	0.15	0.04	5.0	0.27

Between the catchments, most of the parameters show a difference of about five to 10 times. The TSS unit loads show the smallest difference between the catchments. The unit loads from the Avenue B catchment are the largest despite the catchment being the smallest. Aside from TSS, the Silverwood catchment unit loads are the smallest. The unit loads clearly show a difference between the land uses.

With reference to Table 2.3, the unit loads found in the literature are generally much larger than the estimated unit loads shown in Table 6.2. The presumed primary reason for the difference is the difference in the period covered. The literature values represent a full year, including winter and spring runoff, which will likely contain larger amounts of gravel, sand, and salt placed on the roads for driving safety. As a consequence of the difference in the period of coverage, the residential (Taylor Street and Silverwood) TSS unit loads are three to 10 times smaller than the literature values. The commercial TSS unit loads are five to 10 times smaller than the literature values. Similar and greater differences are seen in the other water quality parameter unit loads. The other two characterizations, SMC and regression, exhibited some similarity to their respective literature comparisons. If, like the other two characterizations, the actual annual unit loads for the COS catchments are similar to the literature values, the loads discharged over the winter and early spring period must be significantly greater than that discharged during the period studied. This conclusion indicates the need for further study to define the loads discharged during winter and spring runoff.

Further insight into the planning of pollution abatement strategies can be gained from the unit loads. Area based pollution abatement strategies, such as street sweeping, undertaken in commercial areas will yield the greatest amount of pollutant removed by sweeping the least amount of area. Further, concentrating any abatement undertaken during periods of frequent rain will provide the greatest pollutant interception for a given effort. Street sweeping has the added benefit of improving the aesthetics by keeping the streets clear of debris and dust.

6.2 CITY WIDE LOAD ESTIMATIONS

Estimates of the mass load discharged by COS urban runoff were made using the two different unit load parameters, the SMC method, and the regression equations. The methods, as listed, are progressively more complex and provide greater confidence in the prediction than the previous method. The unit load parameter is a basic parameter using only catchment area to estimate the load. The two unit load parameters are estimated based on the other two methods. The SMC method uses an average water quality parameter and the recorded rainfall depth to estimate the load and is expected to be more representative than the unit loads. The regression equations are developed based on the most significant predictor variables, as determined by statistical multivariable regression analysis. The regression equations are expected to be the most representative of the estimations because they attempt to incorporate more variables in the prediction process.

The loads are estimated based on the land use distribution (Figure 5.28) and rainfall distribution represented by data from each of four COS operated rain gauges (Figure 3.1). The estimations are made based on the rainfall record from April 1 to October 31 in 2001 and 2002. Due to the northern climate of Saskatoon, any precipitation falling outside of this interval generally falls as other than rain, is not represented by the data collected, and is not considered in this analysis.

The estimations of the COS urban runoff load for April 1 to October 31 (i.e. 7 months) in 2001 and 2002 are shown Table 6.3. Of the COS urban runoff load estimations, the two unit load parameters produce loads similar to but smaller than the method on which they are based (i.e. SMC or regression). This suggests that misrepresentations are introduced into the parameters when they are simplified to the unit load. The similarity of the mass loading estimated from unit load to the parameter upon which it was based indicates that it would produce reasonable planning level estimates. However, due to the similarities of the unit load results to the other two methods, only the SMC and regression method loads are discussed further.

The regression method produced the largest COS urban runoff loads. The SMC and regression methods produced loads that are all less than 2.2 times different from each other. Given the difference in the level of sophistication of each of the methods and the uncertainty related to the stage-discharge curves and water quality concentrations, the effort required for the more complex regression model does not appear to be justified. The SMC load estimation produces an acceptable result compared with the regression equation load estimation. Other researchers have also found that the SMC method produces results comparable to more complex methods in the prediction of longer term loads (Chandler, 1994; Charbeneau and Barrett, 1998).

Table 6.3 – Summary of estimated water quality parameter loads discharged by COS urban runoff, COS WWTP and Akzo Nobel Chemicals Ltd. – Saskatoon.

Source	Rainfall/ runoff period	TSS load (tonne)	TKN load (tonne)	TP load (tonne)	COD load (tonne)	Chloride load (tonne)
Unit load (SMC)	2001	470	4.2	0.93	180	23
	2002	780	7.2	1.6	300	40
	Sum	1300	11	2.5	480	62
Unit load (regression)	2001	580	4.2	1.3	140	25
	2002	1400	9.2	2.4	270	35
	Sum	2000	13	3.7	410	60
SMC	2001	500	4.6	1.0	190	25
	2002	950	8.6	1.9	360	47
	Sum	1500	13	2.9	550	72
Regression	2001	480	5.1	2.2	160	28
	2002	1600	12	3.3	320	43
	Sum	2100	17	5.5	480	72
COS WWTP[a]	2001	170		10	360	
	2002	230		12	390	
	Sum	400		22	750	
Akzo Nobel[a]	2001	11	1.3		73	
	2002	11	1.4		90	
	Sum	22	2.6		63	

[a] – results represent a complete year (i.e. 12 months)

Table 6.3 also presents data from two point source discharges within COS: COS Wastewater Treatment Plant (WWTP) and Akzo Nobel Chemicals Ltd. – Saskatoon (Akzo Nobel). The data for each point source represents a complete year (i.e 12 months) The WWTP loads were calculated using data from the COS Utility Services Department Annual Report, 2003 (COS, 2003). WWTP COD data were determined based on the BOD_5 data from the annual report and divided by the ratio of BOD_5 to COD as determined by parallel tests run in the U of S lab. The point source load calculations and information are shown in Appendix J.

Akzo Nobel provided average concentration and average flow data from which the annual loads were calculated. Akzo Nobel – Saskatoon produces fatty organic chemical compounds that are used in the potash industry and other manufacturing processes.

Table 6.3 shows that the TSS load discharged by COS urban runoff, estimated using the regression equations, is about five times larger than the COS WWTP loads. However, as rainfall depth was the most significant variable in the estimation of the urban runoff load, consideration must be given to the rainfall that occurred in each year. In 2001, the annual rainfall was the smallest in the over 100 years of records. In 2002, the annual rainfall was near normal. As such, comparisons based on the 2002 estimations are felt to be more representative. In 2002, the urban runoff TSS loads are nearly 10 times larger than the COS WWTP loads, which is significant. The larger TSS loads from urban runoff are cause for concern because the more toxic and bioaccumulative substances have a tendency to adsorb to TSS (Sansalone and Buchberger, 1997; Stahre and Urbonas, 1990; Characklis and Wiesner, 1997).

TKN data were not reported by the COS WWTP, thus comparison is not made.

The TP loads from the COS WWTP are four to eight times larger than the urban runoff loads. In 2002, the WWTP load was approximately five times larger than the COS urban runoff load.

The COD loads from the COS WWTP are slightly larger than the urban runoff load estimates. Considering the uncertainties of both estimates, both sources are considered as discharging comparable loads.

Using the average of the COS WWTP load estimates for 2001 and 2002 as a benchmark, the relative input to the South Saskatchewan River of COS urban runoff can be summarized as shown in Table 6.4. In short, the relative input of the urban runoff when compared with the COS WWTP is approximately 4.25 times for TSS, 0.17 times for TP and 0.82 times for COD.

Table 6.4 – Average annual load comparisons: WWTP and urban runoff.

Source	TSS load (tonnes/year)	TP load (tonnes/year)	COD load (tonnes/year)
COS WWTP[a]	200	11	375
COS urban runoff [b]	750 – 1050	1.45 - 2.25	240 – 275
Ratio	3.8-5.3	0.13 - 0.20	0.64 – 1.0

[a] – 12 months
[b] – 7 months

The Akzo Nobel loads are smaller than the COS urban runoff loads. The Akzo Nobel TSS load is about 100 times smaller than the COS urban runoff load. The Akzo Nobel TKN loads are nearly 10 times smaller than the urban runoff load. TP results were not reported by Akzo Nobel. The Akzo Nobel COD loads are between 2.5 and 5 times smaller than the COD loads from the COS urban runoff.

The loading comparisons show that COS urban runoff is likely the largest contributor of TSS and TKN to the South Saskatchewan River. COD is in the same range as the COS WWTP. The COS WWTP is the largest contributor of TP to the South Saskatchewan River. TP loading is of concern because TP is the most likely the growth limiting nutrient for aquatic plants in the fresh water of South Saskatchewan River (Chambers et el., 1997). It is also important to consider that the event loads used to generate the characterizations, and subsequently the city wide load estimations, are likely underestimated due to data limitations as explained in Chapter 5. Further the WWTP loads represent 12 months where as the urban runoff loads only represent seven months.

Urban runoff is a substantial contributor of pollutant load from the COS to the South Saskatchewan River. Further study is recommended to quantify the mass loads discharged in the spring and winter, which will permit an annual urban runoff load to be estimated and compared to point-source discharges. Assessment of the assimilative capacity of the South Saskatchewan River at Saskatoon is beyond the scope of this work. However, the assimilative capacity is necessary to assess the impact upon the receiving stream, set acceptable discharge loads for the various discharges, and subsequently determine appropriate controls. Further study to assess the assimilative capacity of the South Saskatchewan River at Saskatoon should be undertaken

6.3 BACTERIAL RESULTS: COLIFORMS

Total and fecal coliforms were excluded from the SMC and regression characterization analyses because of the concerns over the accuracy of the results and the limited number of usable results. Further, the effects of TC's and FC's are generally acute (short term) and not cumulative in nature (Ellis and Hvitved-Jacobsen, 1996), thus characterization using parameters that are better suited for prediction of cumulative (long term) loads provides limited additional insight. However, examination of the FC concentration results reveals a number of interesting items.

The general surface water quality guideline for the Province of Saskatchewan for FC's is 1,000 org/100 mg/L (SERM, 1997). The specific surface water quality guideline for contact recreation is 200 org/100 mL of FC. The South Saskatchewan River is used for contact recreation as well as a source of drinking water. The drinking water guideline is zero org/100 mL. The average baseflow FC concentration in each of the catchments (shown in Table 5.6) violates the contact recreation guideline, while all of the maximum concentrations violate general Surface Water Quality guidelines.

Urban runoff discharge from Saskatoon may impact upon the use of the river as a drinking water source and use of the river for contact recreation. The Taylor Street outfall is upstream of and on the opposite side of the river channel to the COS water treatment plant secondary raw water intake. It is unlikely that discharge from this outfall could reach the intake, however, other storm sewer outfalls, with presumably similar characteristics, are upstream from and on the same side of the river as the intake. The South Saskatchewan River at Saskatoon is used for water skiing, wading, boating, and other types of contact recreation. FC's in Saskatoon's urban runoff, from both baseflow and rainfall-runoff, are an issue relating to human safety in terms of both possible drinking water source water contamination and contact recreation. Makepeace et al. (1995) also generally conclude the same. In the period following rainfall-runoff discharges, contact recreation should be avoided in the vicinity of storm sewer outfalls.

Under baseflow conditions, the FC concentrations were greater than the detection limit in all samples at two of the four sites. These are the Avenue B and Taylor Street catchments; the oldest of the catchments monitored. This result suggests that a relatively continuous source of fecal matter was available during baseflow conditions to provide the FC's. Since little or no surface runoff to the storm sewers is observed during baseflow conditions, the FC's were not likely washed from the catchment surface into the storm sewer. Potential sources of the FC's may be leakage (exfiltration) from the sanitary sewer into the storm sewer and/or direct cross connections. The infrastructure in the Taylor Street catchment was constructed around 1950 and as early as 1912 in the Avenue B catchment. Since the time of construction of these two systems, construction practices, and materials have improved in quality. Both the Sturgeon Drive and Silverwood storm sewer mains were constructed around 1980 and both had baseflow samples with FC results below the detection limit. Munch and Keller (1983) found that the Grey Avenue storm sewer outfall was highly contaminated with FC. The source was later determined to be exfiltration from the sanitary sewer into the storm sewer, which was remedied by lining the sanitary sewer to prevent exfiltration. Cross connection between the systems was considered as a source, however, considering the nature of sanitary sewage (paper, rags, other floatables, etc.), under baseflow conditions visual inspection of the storm sewer flow can readily identify cross connections. Considering the above points, the results suggest that cross contamination from the sanitary sewer via exfiltration/infiltration is likely ocurring.

The event composite samples generally had higher concentrations of FC than did the baseflow samples. Considering the substantially larger dilution available during a rainfall-runoff event compared to baseflow conditions, a significant source of FC is likely to have been exposed to rainfall during the rainfall runoff events. Leakage from the sanitary sewer through the soil is not likely the source of the FC detected in the rainfall-runoff event samples, as the infiltration rate into the storm sewer would be relatively small with respect to the storm sewer flow and unable transport sufficient quantities of FC's to overcome the dilution. Cross connections between the two systems may be able to provide sufficient quantities of FC, however the connection would also need to be significant to overcome dilution in both sewers. A significant cross connection, in either catchment, is not suspected by COS's cross connection monitoring program. Thus, the likely source of the FC's in the event composite samples is likely animal fecal wastes on the catchment surface. This conclusion is consistent with

Novotny (1992) and Mills (1977) who found that animal waste on the surface was the likely source of fecal contamination during rainfall-runoff events.

6.4 TOXIC SUBSTANCES

Several urban runoff samples were analyzed for heavy metals and pesticides and both were found. The heavy metal results are of most concern.

Heavy metals were found to exceed aquatic life protection guideline concentrations in 14 of 15 samples and to exceed the drinking water guidelines in eight of 15 samples. The heavy metal results are shown in Tables 5.22 to 5.24. In baseflow samples, iron concentrations violate both guidelines, while copper, lead, and zinc concentrations violate the aquatic life protection guidelines. In event composite samples, iron and lead concentrations violate both guidelines, while cadmium, chromium, copper, and zinc concentrations violate the aquatic life protection guideline concentrations. The mid-event grab samples had concentrations of cadmium, chromium, iron, and lead that violate drinking water guidelines, while cadmium, chromium, copper, iron, lead, and zinc concentrations were also found to violate the aquatic guidelines. Dilutions of one to 50 times would be required to reduce heavy metal concentrations to below the respective guideline values.

The concentrations found in the mid-event grab samples were significantly greater than the concentrations found in the event composite samples (i.e. average concentration). This result suggests that an intra-event spike in concentration may exist.

The most concerning heavy metal results are for copper with respect to aquatic life protection with concentrations from less than one to 32.5 times greater than the freshwater aquatic guideline. Copper is very toxic to aquatic plants and animals (EXTOXNET, 2001) and the copper concentrations are the highest metal concentration relative to the guideline value. Urban runoff copper sources include tire wear, brake lining wear, combustion of lubrication oils, corrosion of building materials, wear engine components, and industrial emissions (Makepeace et al., 1995). In higher pH conditions, such as those observed in COS urban runoff, copper favours a solid form (Makepeace et al, 1995; CCREM, 1987). Therefore, an increased street sweeping program and/or TSS removal may help to reduce the load discharged.

The limited heavy metal results do not appear to show any trends with respect to the catchments and thus land use. This suggests that the source is diffuse within the city. Potential sources include air borne deposition and deposition from vehicles travelling on roadways.

Pesticides were detected in seven of 15 samples analyzed. Mecoprop, 2,4-D, dicamba, MCPA, and bromoxynil were identified. Mecoprop and 2,4-D were the most frequently detected and MCPA had the highest concentration relative to the freshwater guideline at nearly two times greater. Bromoxynil is the active ingredient in the agricultural herbicide Buctril or Buctril M, for post-emergent control of broadleaf weeds (EXTOXNET, 1996c). Mecoprop, 2,4-D, dicamba, and MCPA are selective, general use herbicides for pre- and post-emergent control of broadleaf weeds (EXTOXNET, 1995b; EXTOXNET, 1996a; EXTOXNET, 1996d; EXTOXNET, 1996e). These four pesticides are found in many common home and garden herbicide preparations. (See Appendix B for further information.) In 2000, the Pest Management Regulatory Agency (PMRA) of Health Canada initiated a priority review of these four common pesticides to re-evaluate the public risk (PMRA, 2000). Subsequent to this review, sales of mecoprop were voluntarily discontinued by the manufacturers due to a lack of supporting background data to indicate its safety (PMRA, 2004).

The largest number of pesticide detections occurred in the event composite samples. In all of the baseflow samples, one pesticide (mecoprop) was detected in one sample. The difference in the detections suggests that the main source of pesticides is washoff from the catchment surface, therefore sample collection should be focussed on rainfall-runoff events. The results also suggest that the total mass load of pesticide discharged could be significant, as the composite samples represent a generally substantial volume of runoff. The potentially significant mass load could result in a temporary spike in concentration in the South Saskatchewan River.

The available dilution in the South Saskatchewan River is dependent upon the flow rate in the river and the flow rate of the urban runoff into the river. A review of Figures 5.1 to 5.4 indicates an average peak flow rate of about 1 m^3/s in all of the study catchments. Data from the Saskatchewan Watershed Authority for the South

Saskatchewan River indicate a median summer flow rate in the range of 120-180 m^3/s (Saskatchewan Watershed Authority, 2005). The regulated minimum flow rate from Gardiner Dam is 42.5 m^3/s. Therefore, normally during the summer period a dilution ratio of 120-180 times is available when considering the entire width of the river. However, storm sewers discharge on one bank and immediate, complete mixing across the river width does not occur. The flow rate through the mixing zone of a storm sewer outfall is, among other factors, dependent upon the depth of the river at the outfall, the horizontal alignment of the river, the vertical cross-section of the river in the vicinity of the outfall, and the local roughness, which makes estimation of the flow rate through the mixing difficult.

A dilution of 50 times, as required to dilute some of the heavy metal concentrations, would require a flow rate of 50 m^3/s, or approximately one third of the river, through the mixing zone, which is unlikely. Therefore, within the mixing zones, potentially toxic concentrations may exist when significant concentrations are discharged from the storm sewers. The river concentration will be further exacerbated by intra-event concentration variability.

Most of the parameters examined do not require large dilutions to meet guideline concentrations and the South Saskatchewan River can reasonably provide the dilutions required to meet water quality guidelines in average conditions. Consequently, in other than the worst case or the mixing zone adjacent to the outfall, acute toxicity effects from urban runoff discharges do not appear to be a problem. Bioaccumulation (chronic effect) of toxic substances may still occur both in the vicinity of the outfall and mixing zone and further downstream, where the effects of all of the outfalls coalesce.

Generally, toxic substances have an affinity for TSS. Considering the apparently large TSS loads discharged, river sediments in the area of outfalls may contain significantly elevated concentrations of toxic substances. Control of the sediment load discharged to the river is recommended.

The number of samples of toxic substances in this study is insufficient to assess the frequency of occurrence of acute effects or the potential chronic effects (e.g. annual load) of the toxic substances. Further study is recommended to assess these items.

CHAPTER 7 SUMMARY, CONCLUSIONS AND RECOMMENDATIONS

7.1 SUMMARY

A multi-agency study of urban runoff in Saskatoon was conducted over the summers of 2001 and 2002. The project, initiated by Saskatchewan Environment, aimed to determine the water quality characteristics of urban runoff from the City of Saskatoon, as little had been done to date involving the water quality of urban runoff in Saskatchewan. The participating agencies were the City of Saskatoon, Saskatchewan Environment, University of Saskatchewan Department of Civil and Geological Engineering, and the Meewasin Valley Authority.

Urban runoff has been reported by many jurisdictions to be a source of pollutant input to receiving streams. The National Water Research Institute of Environment Canada has classified urban runoff as a threat to drinking water supplies and aquatic habitat in Canada (Environment Canada, 2001). The U.S. EPA, after more than 30 years of work, still consider urban runoff to be a 'stressor' on the waterways of the U.S. (Mays, 2001).

A field program was undertaken to collect representative data from four different types of land use: new residential, old residential, commercial, and industrial. The catchments chosen to represent these land uses were Silverwood, Taylor Street, Avenue B, and Sturgeon Drive, respectively. The water quality sampling and flow rate data collection portions of the field program were fraught with equipment issues. From 194 station events, water quality samples were collected from 73, and flow rate data were collected from 44. The marrying of the two types of data resulted in 27 rainfall events with useable data. Due to equipment issues, the Sturgeon Drive catchment had only two useable events (for discrete TSS only) and was excluded from the subsequent characterization and prediction exercises.

The water quality data covered a wide range of concentrations, often two or three orders of magnitude. The volumetric flow rates in the storm sewers varied from zero to 4 m^3/s. The paired water quality and flow rate data produced estimates of the total mass load of pollutant discharged during individual rainfall runoff events (event load).

The event loads were then used to characterize the water quality of the runoff. The characterizations are intended to provide a means of predicting urban runoff loads and measuring the effect of future pollution abatement (i.e. BMP's). The first characterization undertaken was the site mean concentration (SMC). The SMC is the geometric average of the flow weighted event mean concentrations (EMC). The second characterization comprises a set of multi-variable non-linear regression equations for predicting individual event loads. Each of the two characterizations were then used to determined the unit area mass load characterization, or unit load (kg/ha).

The characterizations were developed for five water quality parameters: TSS, TKN, TP, COD, and chloride. The water quality parameters were chosen based on their significance to the river environment, suggested parameters of other agencies, parameters that are frequently used in the regulation of surface water discharges, and parameters for which, in this work, confident analysis could be made.

A summary of the SMC's is presented in Table 6.1. The SMC's do not show any significant difference with land use when considering the large variations observed in the data. Therefore, it is concluded that land use is not a significant factor in the SMC's. Comparison to reported literature values showed that the COS SMC's were similar to and slightly larger than most. The range of variability of the COS SMC's compared well with reported variability, suggesting that the variability of the sample data is representative of the true variation. The COS SMC's were most similar to SMC's determined for urban sites in Wisconsin.

Multiple variable regressions were performed to determine 15 equations to predict the event loads of five water quality parameters in three catchments. The software package SPSS version 12.0 was used to perform the analyses. The resultant equations and goodness-of-fit measures are shown in Table 5.18. The goodness-of-fit measures are similar to those reported in the literature and indicate that the models are more applicable to longer term (i.e. annual) load predictions than to individual event load predictions. This is shown by the R^2 values generally being better than the median percentage calibration error (MPE). The most frequently significant variable is the event rainfall depth. The depth was found to be significant in 10 of the 15 regression equations. The literature sources also show the depth to be the most frequently significant variable. The other significant variables are the maximum intensity, duration, and average intensity. In each catchment, the models have similar predictor variables for each of the five parameters. In contrast, for a given parameter, the predictor variables are different for each catchment. These differences indicate that the effect of land use is important in the water quality parameter loads. The antecedent dry period was not found to be significant. This result, and the frequent significance of the rainfall depth, suggests that targeted street sweeping or similar management practices during rainy portions of the summer may be an effective pollution abatement strategy for TSS and related pollutants.

Unit loads were determined based on estimations of the catchment load during the April 1 to October 31 period using the SMC and regression characterizations. The unit loads determined are shown in Table 6.2. The unit load estimations are similar for the two methods (SMC and regression), and the regression based unit load is generally larger. The unit loads show a difference of between five and 10 times between the catchments and clearly show a difference between the land uses. The Avenue B catchment (smallest area, commercial land use) has the largest unit loads.

The unit loads estimated are much smaller than the literature reported values. This is in part because the literature reported unit loads are for a full year, while the unit loads in this study represent the seven month period from April to October. The difference is, therefore, the effect of the winter and early spring runoff. If, similar to the SMC's, the actual Saskatoon unit loads are comparable to the literature values, the load discharged during the winter and early spring must be significant. The winter and early spring mass loads remain unknown and should be examined in a future study.

The unit load results indicate that area based pollution abatement strategies undertaken in commercial areas, such as street sweeping, will yield the greatest amount of pollutant removed by sweeping the least amount of area. Further, concentrating efforts prior to and during periods of frequent rain will provide the greatest pollutant interception.

Based on the characterizations, four estimates of the mass load discharged from the city were made. The regression method is expected to produce the most accurate estimate and the unit loads are expected to produce the least accurate because of the differences in the level of sophistication of the estimation procedure. The unit load (SMC based) produced the smallest load estimates. While the regression method generally produced the largest load estimates, the SMC method load estimates were somewhat similar. Considering the difference in the complexity of the regression method versus the SMC method, the SMC method performed well in the load estimations and can be used instead of the regression method for long term load estimates.

In comparison to the two other discharges to the South Saskatchewan River examined herein, urban runoff is likely the largest source of TSS and TKN load. The COD mass load is similar to that of the COS WWTP. The COS WWTP, however, is still the largest source of TP. It should be noted that the COS WWTP employs advanced tertiary wastewater treatment and produces good quality effluent. The Akzo Nobel loads are generally two orders of magnitude smaller than the urban runoff loads.

Assessment of the assimilative capacity of the South Saskatchewan River is beyond the scope of this work. However, such an assessment is necessary to determine appropriate long term water quality controls. Further study to assess the assimilative capacity is recommended.

Fecal coliforms were found in both the rainfall-runoff and baseflow samples. During rainfall-runoff events, the FC source is likely animal excrement on the catchment surface. The Avenue B and Taylor Street sites appear to have a continuous supply of FC during baseflow conditions because FC's were detected in all baseflow samples at these sites. During baseflow conditions, the source is suspected to be exfiltration from sanitary sewer pipes, a condition which is known to have occurred previously in Saskatoon. The detected concentrations exceed both the contact recreation and drinking water guideline concentrations. Fecal contamination of outfalls and adjacent areas is a concern with respect to contact recreation and the potential use of the river as a drinking water source. In the periods following rainfall-runoff events, contact recreation should be avoided in the vicinity of storm sewer outfalls and backwater areas directly downstream.

The toxic substances analysis identified six potentially problematic heavy metals and detected five synthetic pesticides. Heavy metal concentrations were found to violate freshwater aquatic life protection guidelines in 14 of 15 samples and to exceed the drinking water guidelines in eight of 15 samples. The heavy metals that exceeded guideline values are cadmium, chromium, copper, iron, lead, and zinc. Copper is of most concern due to the relative concentration detected. Simply stated, heavy metals exist in the city's urban runoff in concentrations that may be toxic to aquatic life. Under average summer conditions and outside of the mixing zones adjacent to outfalls, however, the South Saskatchewan River can provide dilutions sufficient to reduce the concentrations below the guideline values, if the metals are in aqueous form. If the metals are sediment bound, harmful concentrations may develop in the sediment in the vicinity of the outfalls. Considering the affinity of heavy metals for sediments, control of sediment discharge should help to control heavy metal discharge. Sediment control measures should be implemented and monitored in Saskatoon.

Pesticides were detected in seven of 15 samples. The concentrations of the pesticides detected were below guideline concentrations in all samples except for one sample for two pesticides, namely mecoprop, and MCPA, with respect to the freshwater aquatic protection guideline. The pesticides detected are 2,4-D, bromoxynil, dicamba, mecoprop and MCPA. Mecoprop was the most frequently detected, while MCPA had the highest concentration relative to its guideline concentration at nearly two times greater.

The toxic substances results indicate the need for further investigation to determine the impact upon the receiving stream and to identify appropriate controls. Given the affinity of the toxic substances for TSS, control of TSS will likely help reduce the loads discharged.

7.2 CONCLUSIONS

Based on the work of this study the following conclusions are made:

- Urban runoff is a large source of material input to the South Saskatchewan River, and likely the largest source of total suspended solids (TSS) and total Kjeldahl nitrogen (TKN);
- Land use is important in the prediction of urban runoff load. While the SMC results do not show significant difference with land use, the regression and unit load methods showed an influence due to land use. Land use primarily affects the quantity of runoff generated from a catchment. Because water quality is integrally related to water quantity, collection of data and estimation of mass loads based on land use is justified;
- The SMC method can be used instead of more complex methods for estimating annual loading with land use specific runoff data;
- The SMC characterization parameters are larger for Saskatoon than for Vancouver and the United States national average. The Saskatoon SMC's are similar to those reported for Wisconsin;
- Based on the regression results, the most frequently significant variable in urban runoff load generation is the rainfall depth;
- Equipment malfunction was a major cause of poor overall rates of useable events (i.e. 27 useable events out of 73 events). Operation of a successful sampling program requires careful attention to and planning of details such as monitoring location, depth sensor placement, maximum elapsed time for specific water quality parameter analyses, and equipment selection. Automated sampling equipment with consistent, regular cleaning and servicing is necessary to facilitate appropriate sample collection;
- Toxic substances were detected in urban runoff with end-of-pipe concentrations that exceed guideline values. Under normal flow conditions, the South Saskatchewan River appears to be able to provide adequate dilution for most aqueous forms, however concentrations that are acutely toxic to some aquatic species may exist in the mixing zones adjacent to outfalls.

Some of the toxic substances have a preference for adsorbing to sediment which may accumulate in the area of outfalls. Further study is required to examine toxic substances and their potential impact upon the river; and

- Considering that the rainfall depth was the most frequently significant predictor variable and since the antecedent dry period was not found to be significant, the greatest amount of pollutant load is discharged during periods when rainfall events occur frequently and with the greatest rainfall depths.

7.3 RECOMMENDATIONS

Based on the work presented herein, the following recommendations are made:

- Saskatoon's urban runoff should continue to be monitored and sampled in an intensive manner to permit pollutant loads to be determined, their impact upon the South Saskatchewan River to be assessed, and appropriate controls to be determined. To aid in determination of appropriate controls and allocation of discharge loads, the assimilative capacity of the South Saskatchewan River at Saskatoon should be assessed;
- Winter and spring runoff sampling, with an emphasis on snow melt and early spring rainfall runoff, should be undertaken to estimate the loads discharged during this period when both salt and sand/gravel have been placed on the City's streets and the mass loads of pollutant are likely to be significant;
- The examination of toxic substances should be expanded to provide an indication of the loads discharged, which could then be used to assess the impact upon the river. The concentration of petroleum products in the runoff should also be examined;
- Considering the affinity of the more concerning pollutants for TSS (e.g. metals, pesticides, etc.) and the large TSS loads discharged, sediment controls should be implemented and integrally incorporated into new developments at all stages of development; and

- During periods of frequent rainfall, regular high efficiency (i.e. vacuum assisted) street sweeping in areas with large unit loads (i.e. commercial areas, such as the downtown core) should be considered and evaluated as a pollution abatement strategy.

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APPENDIX A

Saskatchewan General Surface Water Quality Objectives Summary

This appendix contains a summary of the Saskatchewan General Surface Water Quality Objectives.

Table A.1 – General Surface Water Quality Objectives (SERM, 1997).

Parameter	Objectives
Bacteria:	
Total Coliforms	Not to exceed a density of 5,000 per 100 mL in any sample.
Fecal Coliform	Not to exceed a density of 1,000 per 100 mL in any sample.
Biochemical Oxygen Demand (BOD_5)	Dependent on the assimilative capacity of the receiving water. The BOD must not exceed a limit which would create a dissolved oxygen content of less than 5 mg/L.
Colour (apparent)	Not to be increased by more than 30 colour units above ambient natural value.
Flow	Water quantities (flow and lake levels) should not be altered unreasonably so as to result in water quality impairment such that water uses are adversely affected.
Major Ions	The predominant cations of sodium, calcium, and magnesium, and anions of sulphate, chloride, and bicarbonate are too variable in the natural water quality state to attempt to define objectives. In general, however, levels of these major ions should not be suddenly increased or decreased in a receiving water so as to result in injury to aquatic life, or alter the sodium:cation ratio beyond the range of 0.30 to 0.75.
Nutrients	Nitrogen or phosphorus or other nutrient concentrations should not be altered from natural levels by discharges of effluents such that nuisance growths of algae or aquatic weeds result.
Oxygen (dissolved)	A minimum of 5 mg/L at any time.
Pesticides	To provide reasonably safe concentrations of these materials in receiving waters an application shall not exceed 1/100 of the 96 hr LC_{50}. Consideration must also be given to sublethal effects and to objectionable taste and odour generation, and to tainting of fish flesh.
pH	To be in the range of 6.5 to 8.5 pH units but not altered by more than 0.5 pH units from ambient values.
Radium 226	0.11 Bq/L
Suspended Solids	Not to be increased by more than 10 mg/L over existing background values for waters with levels of less than 100 mg/L, or not to be increased by more than 10 percent over existing values for waters with levels greater than 100 mg/L.
Sulphide	0.01 mg/L
Turbidity	Not to be increased by more than 25 turbidity units above ambient value.

APPENDIX B

Supplemental Water Quality Parameter Information

This appendix, which is essentially an extension of the literature review in Chapter 2, contains supplemental background water quality information. Contained herein are physical parameters (temperature and pH), further nitrogen background, expanded metals discussion, expanded organic chemical parameter discussion (including petroleum fractions and pesticides), and expanded micro-organism discussion.

B.1 PHYSICAL PARAMETERS

Temperature is not usually a suggested parameter of study (Alberta Environment Protection, 1999a; Smullen et al., 1999). It is, however, a parameter that needs consideration because of its effect on other parameters. Dissolved oxygen, pH, bacterial activity levels, and the solubility of hydrocarbons and metals are sensitive to changes in temperature (Snoeyink and Jenkins, 1980; Adams and Papa, 2000; Van Buren et al., 2000). For both dissolved oxygen and pH, temperature affects the solubility of gasses in solution. Generally, as the temperature increases, the solubility of gasses in solution decreases. In the literature, the temperature of urban runoff has been reported to range from 10°C to 30.5°C (Makepeace et al., 1995).

pH is a measure of the concentration of the hydrogen ion or the acidity of a solution. Rainfall in Wisconsin was generally found to have a pH of 5, while in the area of Milwaukee pH values of 3 to 6 were found (Novotny et al., 1985). Based on the Canadian fresh water aquatic life guideline of 6.5 to 9.0 (CCME, 2003), Makepeace et al. (1995) conclude that pH could be a problem in stormwater at the lower limits reported. pH affects the toxicity of other chemical parameters by causing a change in the dominant form to a potentially more toxic form. Arsenic and copper are examples of parameters whose toxicity is influenced by pH.

B.2 NITROGEN

Nitrogen has four predominant forms in water: Ammonia, nitrate, nitrite, and organic nitrogen (APHA et al., 1992; Snoeyink and Jenkins, 1980). Ammonia is discussed in Chapter 2. Nitrate is a nutrient important in plant growth. The Canadian drinking water guideline MAC for nitrate is 45 mg/L (CCME, 2003). Nitrate has aquatic acute toxicities ranging from 5 mg/L to 6000 mg/L and has been detected in stormwater in concentrations ranging from 0.01 mg/L to 12.0 mg/L (Makepeace et al., 1995).

Nitrite is not often found in significant quantities in natural waters because it's oxidation to nitrate is rapid (Peavy et al., 1985, CCREM, 1987). Based on their literature review, Makepeace et al. (1995) report urban runoff concentrations ranging from 0.02 to 1.49 mg/L. Nitrite has a reported acute aquatic toxicity in the range of 0.19

to 140.0 mg/L. The Canadian freshwater guideline is 0.060 mg/L (CCME, 2003). The Canadian and U.S. drinking water guidelines for nitrite are 3.2 mg/L as NO_2^- (CCME, 2003; EPA, 2003). Excess concentrations of nitrite in drinking water cause Blue Baby Syndrome (EPA, 2003).

B.3 METALS

While many metals have been detected in urban runoff (Makepeace et al., 1995; Bannerman et al., 1996; Munch and Keller, 1985; Marsalek and Schroeter, 1988), only those thought to have significance in urban runoff are examined here. Makepeace et al. (1995) found copper, lead, zinc, and cadmium were the most frequently studied metals, while lead, copper, zinc, cadmium, nickel, arsenic, and beryllium were found to be of "greatest concern" because the concentrations found exceeded guideline values by 10 times or more.

Arsenic is introduced into urban runoff from sources such as industrial emissions, fossil fuel combustion, smelting, erosion/dissolution of geologic formations, some laundry products, some pesticides (weed killers and defoliants) and chromated copper arsenate (CCA) (a preservative) treated wood (Makepeace et al., 1995; EPA, 2003; Rice et al., 2002). It was found in 52% of the NURP Priority Pollutant samples. In urban runoff, arsenic has been found at concentrations ranging from 0.001 to 0.21 mg/L. The Canadian drinking water guideline interim maximum acceptable concentration (IMAC) for arsenic is 0.025 mg/L (CCME, 2003), however aquatic guidelines for the various forms of arsenic range from 0.013 to 0.036 mg/L (marine chronic) and 0.360 mg/L (freshwater acute). As seen by the difference between the guideline concentrations, humans are more sensitive to arsenic than freshwater aquatic life. Arsenic is a carcinogen in humans (Rice et al., 2002). The dominant form of arsenic in water, which governs the toxicity, is governed by the redox potential and pH. Arsenic is primarily associated with suspended solids in runoff.

Rice et al. (2002) found that in an urban environment the predominant sources of arsenic were CCA treated lumber and roadway runoff. CCA treated lumber leached arsenic into surrounding soil and water. Further, they determined that a recreational urban lake retained 70% of the arsenic input, primarily in the lake sediments. In

Canada, the use of CCA treated lumber in non-industrial applications is being voluntarily phased out by the lumber supply industry (Health Canada, 2003c).

Beryllium was detected in 12% of the NURP Priority Pollutant samples and had an urban runoff concentration range reported in the literature of 0.001 to 0.049 mg/L (Makepeace et al., 1995). Canada does not have a drinking water guideline for beryllium (CCME, 2003). The only jurisdiction to implement such a guideline is the United States (U.S.) at 0.004 mg/L (Makepeace et al., 1995; EPA, 2003). The U.S. has also established a fresh water aquatic chronic exposure guideline of 0.0053 mg/L. Beryllium in urban runoff is associated with TSS. The toxicity of beryllium is higher in soft water. Combustion of fossil fuels is the major source of beryllium. Other anthropogenic sources include metal refining and discharge from electrical, aerospace, and defence industries (EPA, 2003).

Cadmium is associated with both drinking water and aquatic life concerns (Makepeace et al., 1995). It has been reported in stormwater to range from 0.00005 mg/L to 13.73 mg/L and was detected in 48% of the NURP Priority Pollutant samples. It has a Canadian drinking water guideline maximum acceptable concentration (MAC) of 0.005 mg/L and aquatic life guidelines of 0.000017 to 0.0093 mg/L (Makepeace et al., 1995; CCME, 2003). Its toxicity is affected by hardness, pH, water temperature, organic acids, and other metal content. Cadmium is associated with both the dissolved (primarily) and colloidal fractions. Cadmium bioaccumulates in aquatic plants and animals (CCREM, 1987). Its sources to urban runoff include brake linings, combustion of lubricating oils, tire wear, metal finishing, geologic formations, agricultural use of wastewater sludge, fertilizers, pesticides, corrosion of galvanized metal, paints, and waste batteries (Makepeace et al., 1995; EPA 2003).

Chromium was detected in 58% of the NURP Priority Pollutant Study samples (Makepeace et al., 1995). It is a concern for both drinking water and aquatic life, with guidelines of 0.05 mg/L and 0.001 to 0.056 mg/L (CCME, 2003), respectively. It has been found in urban runoff in concentrations from 0.001 to 2.30 mg/L (Makepeace et al., 1995). Chromium bioaccumulates, has a higher toxicity in soft water, and is primarily associated with TSS (Makepeace et al., 1995; CCREM, 1987). Sources of chromium to urban runoff include wear of engine parts, corrosion of some metals, geologic formations, paints, dyes, ceramics, paper, heating and cooling coils, fire suppression sprinklers, steel mill effluent, fertilizers, and pesticides (Makepeace et al., 1995; CCME, 2003; EPA, 2003).

Copper was detected in 91% of NURP Priority Pollutant samples and is the major aquatic toxic metal in urban runoff (Makepeace et al., 1995). Testament to its toxicity is the fact that it is used as an herbicide, insecticide, algaecide, and fungicide (EXTOXNET, 2001; Makepeace et al., 1995). Makepeace et al. (1995) report a range of urban runoff concentrations from the literature of 0.00006 to 1.41 mg/L. The Canadian drinking water AO for copper is 1.0 mg/L (CCME, 2003). The Canadian freshwater aquatic life guideline is 0.002 to 0.004 mg/L. The U.S. EPA's drinking water maximum concentration level goal (MCLG) is 1.3 mg/L (EPA, 2003). The U.S. EPA has a freshwater acute toxicity criterion of 0.018 mg/L and chronic toxicity criterion of 0.012 mg/L (Makepeace et al., 1995). Copper is most commonly found in the dissolved state; however, it is also found in colloidal suspension, which contributes to the TSS load (CCREM, 1987). The form of copper is primarily dependent upon the pH. In waters with a pH below 6.5, copper is generally dissolved. In waters with a pH above 6.5, copper is generally colloidal. Copper bioaccumulates quickly in plants and animals. Rice et al. (2002) found that in an urban lake in Virginia, 20% of the annual copper load entering the lake was retained and suggest that the copper load was primarily dissolved or remained in suspension.

The NURP SMC for copper was found to be 0.034 mg/L and shows correlation with vehicular traffic (Makepeace et al., 1995; EPA, 2003; Rice et al., 2002). Its sources include tire wear, brake linings, combustion of lubrication oils, corrosion of building materials, wear of moving engine components, industrial emissions, erosion of natural deposits, CCA treated lumber and corrosion of brass and copper piping.

Iron has been reported in urban runoff at concentrations ranging from 0.08 to 440 mg/L (Makepeace et al., 1995). Its sources include automobile corrosion, corrosion of other steel structures, burning of coal products, iron and steel manufacturing emissions, landfill leachate, and geologic formations. Aquatic guidelines vary from 0.3

mg/L (Canadian) to 1.0 mg/L (U.S. freshwater chronic), while the Canadian drinking water guideline AO is less than 0.3 mg/L. In aerobic water, iron is generally colloidal and contributes to TSS. In low pH waters, iron is generally dissolved.

Lead is suggested by researchers as the most important parameter in urban runoff (Makepeace et al., 1995). It was detected in 94% of the NURP Priority Pollutant study samples. In 74% of NURP samples, lead violated drinking water guidelines. It has been detected in concentrations in urban runoff ranging from 0.00057 to 26.00 mg/L (Makepeace et al., 1995). Since the introduction of unleaded gasoline in 1975 and the total ban of leaded gasoline for automobiles in 1990, environmental concentrations of lead have decreased (Health Canada, 2003a; Hall et al., 1999). The Canadian drinking water guideline MAC is 0.010 mg/L (CCME, 2003). Freshwater aquatic guidelines range from 0.001 to 0.007 mg/L (CCME, 2003). Lead is primarily associated with TSS, adsorbing to clays and forming sparingly soluble precipitates and organic complexes (CCREM, 1987). The solubility of lead increases with decreasing pH and alkalinity. At pH values less than five or six, lead tends to exist in the dissolved state. Lead is readily bioaccumulated by most aquatic species. Lead is also known to produce “additive and synergistic (toxic) effects” with copper and zinc (Chambers et al., 1997; Makepeace et al., 1995). Sources of lead include leaded gasoline (where still used), gasoline additives, tire wear, industrial emissions, old leaded paint, erosion of natural deposits, corrosion of leaded piping systems, and municipal wastewater effluent applied to agricultural fields (Makepeace et al., 1995; Health Canada, 2003a; EPA, 2003; CCREM, 1987).

Mercury concentrations have been found to range from 0.00005 to 0.067 mg/L in urban runoff (Makepeace et al., 1995). The Canadian drinking water guideline MAC for mercury is 0.001 mg/L (CCME, 2003) which was violated by some of the results examined by Makepeace et al. (1995). In humans, mercury is known to cause kidney damage (EPA, 2003). Mercury is highly toxic in the aquatic environment. Aquatic guidelines range from 0.000004 to 0.00026 mg/L (CCME, 2003). The smaller aquatic guidelines are violated by all samples reported by Makepeace et al. (1995). The toxicity of mercury is dependent upon the pH, organic content, and biological activity in the receiving water (CCREM, 1987). In natural waters, mercury tends to sorb to aquatic sediments, however anaerobic biological degradation and/or low pH can release mercury back to the water column. Mercury is rapidly bioaccumulated in aquatic organisms. Sources of mercury include coal combustion, dental amalgam, fluorescent lamps, electrical component waste and emissions from older chlor-alkalai industrial facilities (Makepeace et al, 1995; CCME, 2003; CCREM, 1987).

Nickel was detected in 43% of the NURP Priority Pollutant samples and is reported in the literature with urban runoff concentrations ranging from 0.001 to 49.0 mg/L (Makepeace et al., 1995). Canada and the U.S. have not set drinking water guidelines for nickel (CCME, 2003; EPA, 2003). Canadian aquatic guidelines range from 0.025 to 0.15 mg/L (CCME, 2003). In urban runoff, it is primarily associated with TSS and organic matter. The toxicity of nickel increases as the hardness of the water decreases (CCREM, 1987). At a pH value below 6.0, nickel tends not to absorb to sediments. Nickel can be re-mobilized from sediments by aerobic micro-organisms. Nickel is bioaccumulated in aquatic plants and animals. Sources of nickel are welded metal plating, wear of moving engine parts, electroplating, alloy manufacturing, and food production (Makepeace et al., 1995; CCREM 1987).

Silver is one of the most toxic metals to aquatic life (Makepeace et al., 1995; CCREM, 1987). It has been found in urban runoff in concentrations ranging from 0.0002 to 0.014 mg/L. The Canadian freshwater aquatic guideline for silver is 0.0001 mg/L (CCME, 2003), while the U.S. freshwater acute guideline is 0.0041 mg/L (Makepeace et al., 1995). Both guidelines are exceeded by the upper end of the aforementioned detected range. Animals are less tolerant of silver than plants. The aquatic toxicity of silver is inversely proportional to water hardness. Silver is found both in sediments and dissolved in the water column. At pH greater than 7.5 to 8.0, silver tends to precipitate out of solution (CCREM, 1987). Silver is also known to adsorb to oxidized manganese floc, oxidized iron floc, and clay minerals. CCREM (1987) notes that chloride levels above 35 mg/L may be useful for binding dissolved silver ions in a less toxic form. Sources of silver to urban runoff include dental, medical and electrical waste, coal combustion, effluents of the photography processing industry, and oil refining (Makepeace et al., 1995; CCREM, 1987).

Zinc in urban runoff was found to range from 0.0007 mg/L to 22.0 mg/L and was detected in 94% of NURP Priority Pollutant samples (Makepeace et al., 1995). Animals and plants easily bioaccumulate zinc. The U.S. freshwater, chronic toxicity guideline is 0.110 mg/L, which was violated by 77% of all NURP samples. The Canadian freshwater aquatic guideline is 0.030 mg/L (CCME, 1997). The Canadian AO for zinc in drinking water is 5.0 mg/L (CCME, 2003). The NURP SMC for zinc was 0.16 mg/L. In aquatic systems, zinc is found in the dissolved state, adsorbed to suspended sediments, with a preference for colloids, and precipitated with oxidized iron and manganese compounds. Zinc aquatic toxicity is affected by the pH and hardness of the water, and by additive and synergistic effects of other heavy metals (CCREM, 1987; Makepeace et al, 1995). Zinc can be re-mobilized to the water column from bottom sediments in anaerobic conditions. Sources of zinc to urban runoff include corrosion of building materials (e.g. roofs), corrosion of galvanized steel, combustion of lubricating oils, tire wear, brake pad wear and wood combustion (Makepeace et al., 1995; CCREM, 1987).

B.4 ORGANIC CHEMICAL PARAMETERS

Many organic chemical contaminants have been identified in numerous studies (Makepeace et al., 1995; Marsalek and Schroeter, 1988; Adams and Papa, 2000; Bannerman et al., 1996). NURP alone examined over 100 parameters, many of which have toxicity concerns (Makepeace et al., 1995). Among the organic parameters, many are associated with TSS. Organic chemical parameters are derived from pesticides, petroleum fractions, dry cleaning agents, industrial degreasers, dyes, preservatives and plastics manufacturing (Health Canada, 2003b; Makepeace et al., 1995; EPA, 2003). Unfortunately, most pesticides have not been studied in relation to stormwater (Makepeace et al., 1995). Makepeace et al. (1995) identified the following organic chemical parameters as “most critical stormwater contaminants”: polychlorinated biphenyls (PCB’s), polycyclic aromatic hydrocarbons (PAH’s), and tetrachloroethylene. The following organic chemicals are commonly used in Saskatchewan (Saskatchewan Health, 2001): 2,4-D; 2,4-DB; Bromoxynil; Dicamba; Diclofop-methyl; Dichlorprop; MCPA; Mecoprop; Triallate; and Trifluralin. For reference where toxicities are discussed in lieu of a guideline value, guideline values are typically set 100 times less than the toxic concentration.

PCB's are a family of 209 combinations of chlorinated biphenyls that adsorb easily to sediments and are only slightly soluble in water (Makepeace et al., 1995). The U.S. MCL for PCB is 0.0005 mg/L (EPA, 2003), while the Canadian aquatic maximum is 0.000001 mg/L (CCREM, 1987). Aquatic toxicity data show a chronic impact exposure range of 0.0002 to 0.015 mg/L. PCB manufacturing and importation was banned in North America in 1977, but existing PCB's were allowed to remain (Health Canada, 2003b). Chronic exposure to PCB's is suggested to increase the chances of humans contracting cancer. PCB's are known to bioaccumulate to high levels in animal tissue, especially fatty tissue (Makepeace et al., 1995; Health Canada, 2003b). PCB's are found in insulating oils of old electrical transformers, some lubricants, and landfill leachate (Makepeace et al., 1995). PCB's were formerly used in sealants and caulking, inks, and paint additives (Health Canada, 2003b).

Polycyclic aromatic hydrocarbons (PAH's) are found in urban runoff in concentrations ranging from 0.00000003 to 0.056 mg/L and have toxicities ranging from 0.016 to 1.7 mg/L (Makepeace et al., 1995). Many PAH's have been detected in stormwater, and as such total PAH is often used as the measurement parameter. High molecular weight PAH's are highly insoluble in water and tend to adsorb to sediments, whereas low molecular weight PAH's are also highly insoluble, but tend to volatilize to atmosphere. The only Canadian drinking water guideline MAC for PAH's is for benzo(a)pyrene, which is 0.00001 mg/L (CCME, 2003). PAH's may be supplied from incomplete combustion of organics such as gasoline, coal and refuse, from leaching of preservative from creosote treated wood (Makepeace et al., 1995), and leaching from the linings of water storage tanks and distribution pipelines (EPA, 2003). Estimates of total PAH loadings in urban runoff have found that 70% of the load is derived from roadway runoff (Marsalek and Schroeter, 1988).

Tetrachloroethylene, in urban runoff, is reported to have concentrations ranging from 0.0045 to 0.043 mg/L (Makepeace et al., 1995). The Canadian drinking water

MAC is 0.03 mg/L (CCME, 2003), while the aquatic guideline is 0.26 mg/L (CCREM, 1987). Tetrachloroethylene is a solvent used in dry cleaning and industrial degreasing.

Bis(2-ethylhexyl) phthalate (DEHP) is used in the manufacturing of PVC, in textile and paper mills, and is found in the leachate from landfills (Makepeace et al., 1995). It has been found in stormwater in the range of 0.007 to 0.039 mg/L. The U.S. drinking water guideline for DEHP is 0.006 mg/L. The freshwater aquatic guideline for Canada is 0.0006 mg/L. Both guidelines are exceeded by the range detected. Detected in 22% of the NURP Priority Pollutant samples, DEHP was the most frequently detected of all organic chemical organic parameters.

The remaining parameters in the aquatic health list of Table 2.2 - γ-BHC (also known as Lindane), chlordane, heptachlor, and heptachlor epoxide - are all insecticides whose detected ranges violated aquatic guidelines by more than 10 times as determined by Makepeace et al. (1995). Lindane is used as an insecticide on cattle, lumber, and in commercial gardens (EPA, 2003). Lindane sparingly sorbs to sediments, and appears to be relatively persistent in the environment. Chlordane is an insecticide used for termite control, and is banned in the U.S. (EPA, 2003). Chlordane has a strong affinity for sediments, bioaccumulates, and is persistent in the environment (Makepeace et al., 1995). Sedimentation is the most effective removal of chlordane. Heptachlor is an insecticide used to control termites and is banned in the U.S. Both heptachlor and heptachlor epoxide have an affinity for adsorption to clays, and bioaccumulate. Heptachlor and heptachlor epoxide are grouped together because heptachlor epoxide is the product of heptachlor decomposition (Makepeace et al., 1995; EPA, 2003).

The remaining ten organic chemical parameters listed in the previous paragraph are pesticides that are commonly used in Saskatchewan for which analysis was provided by the Saskatchewan Health Provincial Water Laboratory (Saskatchewan Health, 2001). The toxicity and usage classifications referenced below are based upon U.S. standards. In the U.S, 'general use pesticides' are available to the homeowner to use, while 'restricted use pesticides' must be purchased and used by those licensed to do so.

2,4-D is a general use pesticide that is used to control of broadleaf weeds (EXTOXNET, 1996a). It is a selective, systemic, post-emergence, chlorinated phenoxy herbicide used in some lawn and garden preparations. In humans, it is classified as slightly toxic by ingestion, and highly toxic in ocular exposure. The Canadian drinking water guideline IMAC is 0.1 mg/L (CCME, 2003). In aquatic organisms, acute toxicities range from 1.0 mg/L to 100 mg/L, with the majority of aquatic organisms showing little to no adverse effect (EXTOXNET, 1996a). 2,4-D is degraded readily by miroorganisms under oxygenated conditions with a half life ranging from 1 week to several weeks.

2,4 – DB is a general use pesticide that is used to control broad leaf weeds. It is classified as slightly toxic to humans (EXTOXNET, 1996b). It is a selective, systemic, post-emergence, phenoxy herbicide used in some lawn and garden preparations. It has an acute toxicity range of 2 to 18 mg/L in fish and is classified as slightly to moderately toxic in fish. A Canadian drinking water standard has not been set for this compound (CCME, 2003).

Bromoxynil is a restricted use pesticide, and is classified as moderately toxic to humans (EXTOXNET, 1996c). It is the active ingredient in the agricultural herbicide Buctril (also in Buctril M), which is used to control post-emergent broadleaf weeds (Bayer Crop Science, 2002). It is moderately to very highly toxic in fish, with acute toxicity ranging from 0.05 to 5.0 mg/L. The Canadian drinking water guideline IMAC is 0.005 mg/L (CCME, 2003).

Dicamba is a general use pesticide, which is used to control post-emergent broadleaf weeds (EXTOXNET, 1996d). It is a selective, pre- and post-emergence, benzoic acid herbicide used in many lawn and garden preparations. It is classified as slightly toxic to humans and has a low toxicity in aquatic organisms. It has mammalian acute toxicities ranging from 35 to 465 mg/L. Dicamba is not known to adsorb to soil particles and is primarily found in the dissolved state. The Canadian Drinking water guideline MAC is 0.12 mg/L (CCME, 2003).

Diclofop-methyl is a restricted use pesticide, which is used to selectively control post-emergent broadleaf weeds (EXTOXNET, 1995a). It has an acute toxicity range in aquatic organisms of 0.35 to 4.0 mg/L. The Canadian drinking water guideline MAC is 0.009 mg/L (CCME, 2003).

Dichlorprop (also known as 2,4 –DP) is a general use pesticide and is mildly toxic to humans (Information Ventures, 1995). 2,4-DP is highly toxic to fish and practically non-toxic to invertebrates. It has an acute toxicity range of 0.005 to 232 mg/L for aquatic organisms. It is not known to adsorb to sediments. A Canadian drinking water guideline has not been established (CCME, 2003).

MCPA is a general use pesticide and is slightly toxic to humans (EXTOXNET, 1996e). It is a selective, systemic post-emergence, phenoxy herbicide used in many lawn and garden preparations. MCPA is a relatively safe for most aquatic organisms, is slightly toxic for fish and practically non-toxic for invertebrates, with acute toxicity ranges from 117 mg/L to 232 mg/L. MCPA is rapidly biodegraded in both soil and water. A Canadian drinking water guideline has not been established (CCME, 2003).

Mecoprop is a general use pesticide and is slightly toxic to humans (EXTOXNET, 1995b). It is a selective, horomone-type, post-emergence, phenoxy herbicide used in many lawn and garden preparations. It is practically non-toxic to fish, with acute toxicities ranging from 100 mg/L and up. Mecoprop adsorbs to organic soil particles. A Canadian drinking water guideline has not been established (CCME, 2003).

Triallate is a selective, pre-emergence, general use pesticide and is classified as slightly toxic to humans (EXTOXNET, 1996f). It is the active ingredient in the agricultural herbicide Avadex BW and highly toxic to aquatic organisms, with acute toxicities ranging from 0.05 to 1.7 mg/L. It strongly adsorbs to soil particles, and is almost insoluble in water. The Canadian freshwater aquatic guideline concentration is 0.00024 mg/L (CCME, 2003).

Trifluralin is a selective, pre-emergence, general use pesticide and is classified as slightly toxic to humans (EXTOXNET, 1996g). It is found in many combination formulations. It is very highly toxic to aquatic organisms with acute toxicities ranging from 0.02 to 3.4 mg/L. It strongly sorbs to sediments and does not dissolve easily in water. The Canadian drinking water guideline IMAC is 0.045 mg/L (CCME, 2003).

In 2000, the Pest Management Regulatory Agency (PMRA) of Health Canada initiated a priority review of Mecoprop, 2,4-D, dicamba, and MCPA to re-evaluate the public risk (PMRA, 2000). Subsequent to this review, sales of mecoprop have been voluntarily discontinued by the manufacturers due to a lack of supporting background data (PMRA, 2004).

For more organic chemical parameters, the reader is directed to Makepeace et al., 1995.

B.5 MICRO -ORGANISMS

The use of indicator organisms to indicate microbiological quality is common (Peavy et al, 1985). As defined by Peavy et al. (1985), an indicator organism is one whose presence presumes that contamination has occurred and suggests the nature and extent of the contaminant(s). Two commonly used indicators are Total Coliforms (TC) and Fecal Coliforms (FC). FC are a subset of TC.

TC's include fecal and non-fecal coliform bacteria. In addition to fecal sources, TC are found in soil and decaying vegetation (Peavy et al., 1985). TC have been found in urban runoff in concentrations ranging from 7.0 to 1.8×10^7 CFU/100 mL (Makepeace et al., 1995). The Canadian drinking water guideline MAC for TC is zero CFU/100 mL (CCME, 2003). Saskatchewan's surface water guideline for TC is less than 5000 CFU/100 mL for non-contact recreation (SERM, 1997).

FC originate exclusively from the intestinal tracts of warm blooded animals (Peavy et al., 1985) and are comprised primarily (97%) of Escherichia coli (E. coli) (CCREM, 1987). FC are used to indicate fecal contamination. FC concentrations greater than 2000 CFU/100 mL have a 97.6% detection rate for Salmonella, another pathogen (Makepeace et al., 1995). Fecal coliforms have been found in concentrations in urban runoff ranging from 0.2 to 1.9×10^6 CFU/100 mL (Makepeace et al., 1995). The Canadian drinking water guideline MAC is zero CFU /100 mL (CCME, 2003). Saskatchewan has a surface water guideline of less than 200 CFU/100 mL for contact recreation and less than 1000 CFU/100 mL for non-contact recreation (SERM, 1997).

Other micro-organisms have been identified in urban runoff (Glasner and McKee, 2002; Makepeace et al., 1995). These include Fecal Streptococci (FS), Pseudomonas aeruginosa, salmonella, Staphylococcus aureus, viruses, protozoa (e.g. Giardia and Cryptosporidium) and fungi. It has been noted that because of the diversity of micro-organisms, of which some may be pathogenic, TC and FC may not accurately indicate the pathogenic risk and therefore caution is required (Makepeace et al., 1995; CCREM, 1987; Glasner and McKee, 2002). Makepeace et al. (1995) suggest the use of FS as an indicator for protozoan contamination. They further suggest that more research is required regarding micro-organism content and hazard in stormwater runoff because of a significant lack of available data. Glasner and McKee (2002) also note that viruses have been found to linger in shellfish after the common indicator E. coli had been flushed out. They further suggest that viruses are the principal water-borne pathogen.

APPENDIX C

Equipment Issues

This appendix contains further explanation of issues experienced with trigger mechanisms used in conjunction with the automated water sampling equipment and another time saving upgrade to the triggers. The intake strainer difficulties are also described herein.

C.1 TRIGGER PROBLEMS

External triggering mechanisms were required to activate all of the samplers except for the ISCO 6700. Two triggers were built for a previous project. The triggers had problems that caused them to trigger erroneously or to not trigger when they should. The triggers frequently triggered with no flow event and drew a set of partial samples. The erroneous triggerings required more frequent site visits, resulting in a significant loss of time. The difficulty originated from a misunderstanding of the operating principle of the trigger. The trigger was thought to be a simple open circuit, which was closed when water touched the contacts. Therefore, a lamp cord was used to provide little electrical resistance and physical durability. The trigger, however, is a capacitance loop which triggers at a certain level of capacitance. The lamp cord used has a high capacitance. The electronics technician in Engineering Shops suggested that a 300 ohm television antenna wire would provide a significantly lower capacitance than the lamp cord. The antenna wire solution was tested, worked well, and all of the triggers were subsequently outfitted with a new television antenna trigger wire.

Another problem with the triggers lay with the electronics. The original triggers were not built as rugged as the use ultimately required and the internal connecting wires often fatigued and broke. This problem was solved by having printed circuit boards made for the existing two triggers and two more were fabricated. This left, at full deployment, three triggers in service and one extra, which provided a desirable level of redundancy, especially given the previous difficulties.

To aid in efficiently checking if the samplers had triggered, an LED on a piece of wire was added to each trigger in 2002. When the sampler was triggered, the LED was illuminated. The wire and LED were taped to the side of the sampler such that it was visible from the surface. This allowed the status of the sampler to be checked by only removing the manhole cover, which saved a significant amount of time and effort by not needing to remove the sampler from the manhole at each visit to the site.

C.2 INTAKE STRAINERS

As described in Chapter 4, significant difficulties were experienced with the strainers supplied with the samplers. The supplied strainers appear to be intended for low velocity flow such as lakes or small streams and in the high velocity storm sewer were pushed to the surface of the sewer flow. This was described in previous projects as the strainer skiing on the surface of the flow. With the strainer on the surface of the flow, a significant amount of air is drawn by the pump, which causes erratic water sample volumes.

The difficulties were investigated using the flumes in the Hydrotechnical Laboratory at the U of S. The strainers were observed in the flume, along with various methods for preventing the skiing from occurring. The light strainers were pushed downstream. With the intake tubing originating at the sampler hung above the flow, the suction tubing pulled tight against the top of the sewer pipe and the strainer was pushed to the water surface by the force of the flow.

The strainer with the large mass on the bottom (middle of Figure 4.5) was from previous projects and did not work in the higher velocity flows, because the top portion of the strainer pivoted up about the weighted end. As the strainer pivoted, the flow pushed the strainer off to the side of the sewer out of the water. Based on the observations made in the Hydrotechnical laboratory, it was determined that the best solution was to have the significant mass on the top of the strainer rather than the bottom. New strainers (right most in Figure 4.5) were subsequently built in Engineering Shops. The new strainers are made from stainless steel to reduce the potential for metal contamination by the strainer. The new strainers functioned substantially better than the previous strainers and the previous problems were alleviated.

APPENDIX D

Rainfall Data by Gauge

This appendix contains a summary of all of the rainfall events which met the minimum rainfall requirements. The data is sorted by rain gauge.

Table D.1 – Warman Road rain gauge data summary 2001.

Start Date/Time (dd/mm/yy hh:mm)	Duration (min)	Antecedent Period (days)	Depth (mm)	Average Intensity (mm/hr)	Maximum Intensity (mm/hr)	Maximum five minute average intensity (mm/hr)
04/04/01 10:06	55	*	0.6	0.65	0.47	0.47
07/04/01 14:53	115	3.16	11.2	5.84	90.0	27.5
29/04/01 3:27	82	21.44	1.0	0.73	1.77	1.77
19/05/01 19:54	595	20.63	12.6	1.27	10.0	6.97
29/05/01 15:28	221	9.40	4.2	1.14	9.73	6.06
06/06/01 8:40	467	7.56	6.4	0.82	10.4	6.77
09/06/01 23:05	21	3.28	1.6	4.51	5.95	4.89
15/06/01 18:01	325	5.77	1.6	0.30	1.98	1.72
17/06/01 7:27	264	1.33	9.8	2.23	4.59	4.12
18/06/01 20:58	19	1.38	1.8	5.77	22.5	11.4
23/06/01 8:59	185	4.49	1.6	0.52	0.72	0.72
24/06/01 6:34	257	0.77	3.0	0.70	5.00	3.74
01/07/01 13:24	96	7.11	1.4	0.88	5.45	3.31
16/07/01 8:42	357	14.74	7.8	1.31	48.0	24.4
22/07/01 12:26	91	5.91	4.4	2.89	24.8	16.2
22/07/01 19:53	57	0.25	1.0	1.05	1.67	1.62
24/07/01 22:59	379	2.09	14.2	2.25	51.4	20.7
25/07/01 21:51	246	0.69	8.8	2.14	72.0	27.6
28/07/01 20:55	403	2.79	7.6	1.13	18.0	10.0
08/08/01 6:57	48	10.14	1.2	1.50	9.47	5.50
14/08/01 5:12	37	5.89	4.8	7.71	34.3	25.8
14/08/01 13:45	7	0.33	3.4	30.5	72.0	32.7
08/09/01 0:44	112	24.45	1.2	0.64	3.38	2.75
19/09/01 21:55	297	11.80	3.8	0.77	19.5	13.2
21/09/01 21:09	211	1.76	1.0	0.28	0.38	0.38

* - first event

Table D.2 – Warman Road rain gauge data summary 2002.

Start Date/Time (dd/mm/yy hh:mm)	Duration (min)	Antecedent Period (days)	Depth (mm)	Average Intensity (mm/hr)	Maximum Intensity (mm/hr)	Maximum five minute average intensity (mm/hr)
10/04/02 11:58	29	*	0.8	1.64	1.53	1.53
22/04/02 22:19	124	12.41	4.0	1.93	5.67	5.23
06/06/02 11:00	4	44.4	1.0	15.5	19.5	9.60
10/06/02 19:41	609	4.36	7.6	0.75	3.27	2.77
17/06/02 21:22	172	6.65	15.2	5.31	40.0	22.1
19/06/02 7:42	105	1.31	2.0	1.14	1.91	1.91
24/06/02 7:20	361	4.91	3.8	0.63	14.7	4.27
29/06/02 18:36	327	5.22	6.0	1.10	48.0	25.6
04/07/02 0:30	297	4.02	1.2	0.24	1.29	1.29
09/07/02 3:35	421	4.92	23.0	3.28	25.7	18.8
17/07/02 5:02	3	7.77	1.8	31.6	42.4	16.5
17/07/02 20:39	71	0.65	8.8	7.39	20.6	16.7
26/07/02 19:19	60	8.89	4.2	4.23	65.5	33.9
31/07/02 9:52	241	4.57	2.8	0.70	4.42	3.24
03/08/02 2:38	291	2.53	1.0	0.21	3.43	3.03
04/08/02 20:29	49	1.54	1.0	1.23	1.42	1.42
05/08/02 9:54	168	0.53	1.4	0.50	26.7	6.71
06/08/02 2:58	112	0.59	8.0	4.27	40.0	25.8
07/08/02 6:57	30	1.09	5.0	10.0	37.9	21.8
08/08/02 4:43	61	0.89	0.8	0.79	1.18	1.18
10/08/02 20:39	871	2.62	8.89	1.07	8.89	6.81
15/08/02 9:58	398	3.61	7.8	1.17	13.6	7.63
25/08/02 22:42	26	10.25	1.4	3.25	3.17	3.02
29/08/02 23:48	32	4.03	6.2	11.6	36.0	24.8
30/08/02 23:32	342	0.97	10.4	1.82	48.0	23.3
01/09/02 16:03	145	1.45	3.8	1.57	3.32	2.63

* - first event

Table D.3 – Acadia rain gauge data summary 2001.

Start Date/Time (dd/mm/yy hh:mm)	Duration (min)	Antecedent Period (days)	Depth (mm)	Average Intensity (mm/hr)	Maximum Intensity (mm/hr)	Maximum five minute average intensity (mm/hr)
19/05/01 19:05	951	38.24	17.4	1.10	10.3	8.51
29/05/01 15:12	235	9.18	5.0	1.27	17.1	9.53
06/06/01 8:23	391	7.55	10.2	1.57	60.0	30.7
09/06/01 17:29	373	3.11	2.4	0.39	4.83	4.08
15/06/01 17:39	359	5.75	2.8	0.47	2.12	2.09
17/06/01 7:31	371	1.33	5.6	0.91	3.81	3.63
23/06/01 4:39	456	5.62	1.4	0.18	0.57	0.57
24/06/01 7:05	247	0.78	4.4	1.07	5.63	5.16
25/06/01 6:19	548	0.80	0.8	0.09	0.27	0.27
01/07/01 14:33	87	5.96	2.0	1.38	2.59	2.10
16/07/01 9:35	427	14.73	4.6	0.65	4.50	3.84
22/07/01 12:50	169	5.84	3.4	1.20	3.85	3.69
24/07/01 22:58	527	2.30	4.6	0.52	1.91	1.89
25/07/01 21:54	347	0.59	1.6	0.28	0.70	0.70
28/07/01 21:18	717	2.73	2.2	0.18	0.51	0.51
08/08/01 8:33	328	9.97	0.6	0.11	0.09	0.09
14/08/01 14:34	228	6.02	1.0	0.26	0.28	0.28
08/09/01 2:39	580	24.34	1.0	0.10	0.10	0.10

Table D.4 – Acadia rain gauge data summary 2002.

Start Date/Time (dd/mm/yy hh:mm)	Duration (min)	Antecedent Period (days)	Depth (mm)	Average Intensity (mm/hr)	Maximum Intensity (mm/hr)	Maximum five minute average intensity (mm/hr)
06/06/02 1:10	36	36.5	0.6	1.01	0.73	0.73
10/06/02 18:58	1727	4.72	21.8	0.76	5.11	4.69
16/06/02 21:44	86	4.92	0.8	0.56	0.53	0.53
17/06/02 21:40	185	0.94	3.8	1.23	3.73	3.73
19/06/02 7:15	301	1.27	2.6	0.52	1.92	1.92
24/06/02 6:51	439	4.77	2.4	0.33	0.97	0.97
29/06/02 19:03	419	5.20	2.2	0.31	0.47	0.47
01/07/02 18:01	122	1.67	0.8	0.39	0.33	0.33
04/07/02 1:17	339	2.22	1.2	0.21	0.30	0.30
Equipment malfunction - use City Hall for remainder of year						

Table D.5 – City Hall rain gauge data summary 2001.

Start Date/Time (dd/mm/yy hh:mm)	Duration (min)	Antecedent Period (days)	Depth (mm)	Average Intensity (mm/hr)	Maximum Intensity (mm/hr)	Maximum five minute average intensity (mm/hr)
03/04/01 17:58	965	*	1.8	0.11	0.36	0.36
07/04/01 15:00	40	3.21	5.6	8.41	51.4	24.2
11/04/01 8:09	435	3.69	0.6	0.08	0.08	0.08
29/04/01 3:29	79	17.50	1.0	0.76	4.07	2.93
19/05/01 18:35	875	20.57	16.4	1.13	13.1	9.09
29/05/01 14:09	241	9.21	5.2	1.30	16.0	6.12
06/06/01 8:02	410	7.58	9.4	1.38	37.9	22.0
09/06/01 19:25	245	3.19	2.0	0.49	6.32	4.31
14/06/01 15:00	26	4.65	0.8	1.88	13.1	2.82
15/06/01 17:46	400	1.10	3.6	0.54	6.26	5.28
17/06/01 7:31	262	1.29	7.0	1.60	4.21	4.11
23/06/01 6:05	443	5.76	1.4	0.19	0.73	0.73
24/06/01 6:43	292	0.72	3.8	0.78	5.22	4.87
25/06/01 6:44	487	0.80	2.8	0.35	90.0	13.5
28/06/01 11:16	14	2.85	1.4	6.06	12.0	7.76
01/07/01 14:22	41	3.12	1.8	2.62	4.74	3.12
16/07/01 9:21	287	14.76	4.6	0.96	11.1	5.87
17/07/01 0:38	165	0.44	3.0	1.09	55.4	18.0
22/07/01 12:12	100	5.37	5.2	3.12	14.7	9.58
24/07/01 22:39	389	2.37	13.8	2.13	65.5	24.0
25/07/01 21:43	242	0.69	12.2	3.02	120	70.5
28/07/01 20:44	404	2.79	8.2	1.22	22.5	13.1
08/08/01 6:53	114	10.1	0.8	0.42	2.51	2.28
14/08/01 13:45	4	6.21	3.2	54.34	103	36.0
08/09/01 0:50	73	24.46	1.2	0.98	5.03	3.22
09/09/01 3:51	17	1.07	0.6	2.13	3.79	2.54
13/09/01 8:49	38	4.20	0.6	0.95	1.73	1.73
19/09/01 21:23	45	6.50	1.4	1.85	13.3	2.85
21/09/01 20:49	258	1.94	1.2	0.28	0.89	0.89
02/10/01 1:45	86	10.03	0.8	0.56	1.20	1.20

* - first event

Table D.6 – City Hall rain gauge data summary 2002.

Start Date/Time (dd/mm/yy hh:mm)	Duration (min)	Antecedent Period (days)	Depth (mm)	Average Intensity (mm/hr)	Maximum Intensity (mm/hr)	Maximum five minute average intensity (mm/hr)
22/04/02 22:33	131	*	6.8	3.13	9.47	8.02
30/04/02 11:50	294	7.46	0.8	0.16	0.62	0.62
06/06/02 0:48	45	36.34	0.8	1.07	1.93	1.93
10/06/02 19:05	1061	4.73	16.8	0.95	6.86	4.81
11/06/02 18:01	306	0.22	0.8	0.16	0.18	0.18
17/06/02 21:13	151	5.92	11.8	4.68	12.6	10.2
19/06/02 7:17	308	1.31	2.8	0.54	2.82	2.70
24/06/02 6:41	506	4.76	4.0	0.47	13.9	6.15
29/06/02 18:32	268	5.14	3.2	0.72	8.57	4.63
04/07/02 0:08	301	4.05	1.8	0.36	2.76	2.68
09/07/02 3:11	609	4.92	31.0	3.05	28.8	19.4
17/07/02 4:53	239	7.65	0.8	0.20	10.6	2.43
17/07/02 20:37	113	0.49	31.6	16.82	144	71.3
26/07/02 14:09	487	8.65	7.6	0.94	144	55.3
31/07/02 9:39	252	4.47	4.4	1.05	3.91	3.66
03/08/02 2:26	25	2.52	1.8	4.40	6.15	5.02
04/08/02 20:19	194	1.73	2.2	0.68	2.41	2.35
05/08/02 12:19	66	0.53	0.8	0.73	7.27	3.51
06/08/02 2:52	351	0.56	6.6	1.13	31.3	13.0
07/08/02 6:57	125	0.93	10.4	4.98	65.5	38.4
07/08/02 19:38	34	0.44	0.8	1.42	5.76	2.96
08/08/02 4:50	308	0.36	1.0	0.20	1.26	1.26
10/08/02 20:38	460	2.44	4.2	0.55	4.83	3.76
15/08/02 10:17	380	3.57	11.2	1.77	55.4	23.4
25/08/02 22:40	39	10.25	1.4	2.17	3.53	2.95
29/08/02 23:38	280	4.01	6.6	1.41	45.0	20.6
31/08/02 0:25	73	0.84	11.8	9.68	90.0	44.5
01/09/02 15:36	244	1.58	5.0	1.23	5.29	4.33

* - first event

Table D.7 – Diefenbaker rain gauge data summary 2001.

Start Date/Time (dd/mm/yy hh:mm)	Duration (min)	Antecedent Period (days)	Depth (mm)	Average Intensity (mm/hr)	Maximum Intensity (mm/hr)	Maximum five minute average intensity (mm/hr)
29/04/01 3:19	236	*	1.0	0.25	2.09	1.96
19/05/01 18:48	224	20.48	9.6	2.58	15.3	10.4
20/05/01 4:00	333	0.23	4.2	0.76	3.73	3.36
29/05/01 13:38	277	9.17	6.2	1.34	27.7	7.71
06/06/01 9:00	463	7.61	3.6	0.47	4.59	2.85
09/06/01 22:50	152	3.26	1.4	0.55	4.53	4.09
15/06/01 17:15	338	5.66	3.8	0.67	4.56	4.31
17/06/01 7:32	246	1.36	7.4	1.81	5.58	4.32
23/06/01 4:45	421	5.71	1.8	0.26	1.10	1.10
24/06/01 6:46	236	0.79	3.0	0.76	5.58	4.87
01/07/01 14:24	28	7.15	1.0	2.18	2.33	2.10
16/07/01 8:38	320	14.74	9.2	1.73	36.0	20.9
17/07/01 0:32	9	0.44	2.6	17.11	27.7	17.3
22/07/01 12:10	101	5.48	3.4	2.02	5.76	4.86
22/07/01 20:02	95	0.26	0.8	0.51	1.19	1.19
24/07/01 22:33	381	2.04	17.4	2.74	60.0	26.6
25/07/01 21:44	224	0.70	1.8	0.48	2.56	2.49
28/07/01 20:45	430	2.49	9.4	1.31	30.0	20.0
08/08/01 6:51	139	10.12	0.8	0.34	3.79	2.60
08/08/01 15:33	26	0.27	1.0	2.33	22.5	4.73
14/08/01 13:42	4	5.90	1.0	13.58	51.4	8.58
08/09/01 0:52	201	24.46	1.4	0.42	6.61	3.95
21/09/01 20:52	238	13.69	0.8	0.20	0.91	0.91
02/10/01 1:38	53	10.03	0.8	0.91	1.56	1.53

* – first event

Table D.8 – Diefenbaker rain gauge data summary 2002.

Start Date/Time (dd/mm/yy hh:mm)	Duration (min)	Antecedent Period (days)	Depth (mm)	Average Intensity (mm/hr)	Maximum Intensity (mm/hr)	Maximum five minute average intensity (mm/hr)
10/04/02 12:18	13	*	0.8	3.62	4.77	3.42
22/04/02 22:09	125	12.40	6.4	3.07	10.3	6.20
06/06/02 0:28	56	44.01	0.8	0.86	2.06	1.96
06/06/02 10:21	4	0.37	0.6	10.3	11.4	4.70
10/06/02 19:35	1592	4.38	14.4	0.54	5.67	4.40
16/06/02 21:09	21	4.96	0.8	2.28	2.83	2.23
17/06/02 21:06	153	0.98	14.0	5.49	20.6	15.0
19/06/02 7:19	136	1.32	2.6	1.15	2.6	2.57
24/06/02 7:04	695	4.90	6.4	0.55	42.4	17.4
29/06/02 18:38	270	5.00	3.0	0.67	13.1	7.55
04/07/02 0:13	305	4.05	2.6	0.51	7.42	4.82
09/07/02 3:12	623	4.91	26.2	2.52	16.4	9.75
17/07/02 20:34	72	8.29	21.0	17.5	90.0	59.7
21/07/02 10:02	321	3.51	2.0	0.37	11.6	3.05
26/07/02 19:00	214	5.15	7.2	2.02	120	40.4
31/07/02 9:40	304	4.46	3.8	0.75	4.14	2.97
03/08/02 2:24	22	2.49	1.2	3.30	4.56	4.24
04/08/02 20:08	286	1.72	1.8	0.38	2.02	1.85
05/08/02 12:16	28	0.47	1.8	3.88	16.7	9.12
06/08/02 2:56	136	0.59	8.4	3.69	80.0	43.4
Equipment malfunction - use City Hall for remainder of year						

* – first event

APPENDIX E

Hydraulic Analyses

This appendix contains supplemental information regarding the hydraulic analyses.

E.1 MANNING'S EQUATION AND GEOMETRIC RELATIONS

Hydraulic analyses are based on Manning's equation,

[E.1] $$Q = \frac{1}{n} A R^{2/3} S^{1/2}$$

where Q is the volumetric flow rate (m^3/s), n is Manning's roughness coefficient, A is the cross-sectional area of the flow (m^2), R is the hydraulic radius (m), and S is the slope of the channel (m/m). The area of the flow and the hydraulic radius are computed using the following geometric relations taken from Sturm (2001). The parameters used in the equations are defined in Figure B.1.

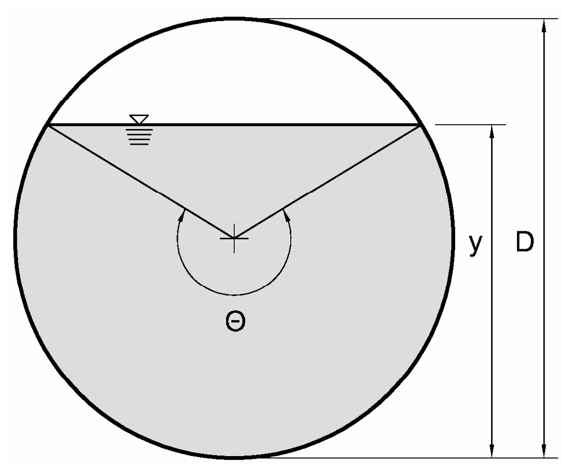

Figure B.1

Figure E.1 – Definition sketch for circular pipe geometric relations.

The area occupied by the flow is calculated,

[E.2] $$A = \frac{1}{8}\left(\Theta - \sin^{-1}\Theta\right) \cdot D^2$$

The angle, Θ, is calculated,

[E.3] $$\Theta = 2 \cdot \mathrm{ACOS}\left(1 - \frac{2y}{D}\right)$$

The hydraulic radius is calculated using, [E.4] $$R = \frac{1}{4}\cdot\left(1-\frac{\sin\Theta}{\Theta}\right)\cdot D$$

E.2 FIELD ROUGHNESS MEASUREMENT

The measurement technique is based upon the dilution of a brine solution that was pumped into the storm sewer flow at a constant, known rate. The conductivity was continuously measured downstream of the injection site using a data recording conductivity meter and probe. The test was run until a stable conductivity measurement was achieved. A site specific calibration curve for the volume of brine in the storm sewer flow was developed by collecting a sample of stormwater at the time of the test, adding metered amounts of the brine solution used in the field test, and measuring the conductivity of the mixed solution. Using the calibration curve, the volume ratio of brine in the stormwater was determined. The flow rate in the storm sewer was calculated by dividing the brine flow rate by the volume ratio. This test was performed under baseflow conditions, when relatively stable depths of flow (i.e relatively stable flow rate) were available. Perfectly stable conditions, in terms of both conductivity and depths of flow, were never achieved, and the stable values used in the calculations are averages over the period of reasonable stability.

Manning's equation can be rearranged to isolate the Mannings n roughness coefficient,

[E.5] $$n = \frac{1}{Q}AR^{2/3}S^{1/2}$$

The unknowns in the equation are the area occupied by the flow (A) and hydraulic radius (R), which are geometrically related to the depth of flow (y) and pipe diameter (D), and the pipe slope (S). The depth of flow was either measured manually at regular intervals or recorded by the flow monitor at the respective location. The pipe slope was determined by field survey for the change of elevation and the length of the pipe was taken from record drawings provided by the City of Saskatoon (COS).

The Avenue B test is presented in the following as an example of the analysis. Figure E.2 is the conductivity profile from the Avenue B tracer test. In this particular test the brine flow rate was increased three times to provide a conductivity profile that was somewhat greater (two to three times) than the background, which improved the stability of the reading. The portion of the curve used for the calculation is enlarged in Figure E.3. The value used in the calculation is an average conductivity reading over the stable period shown.

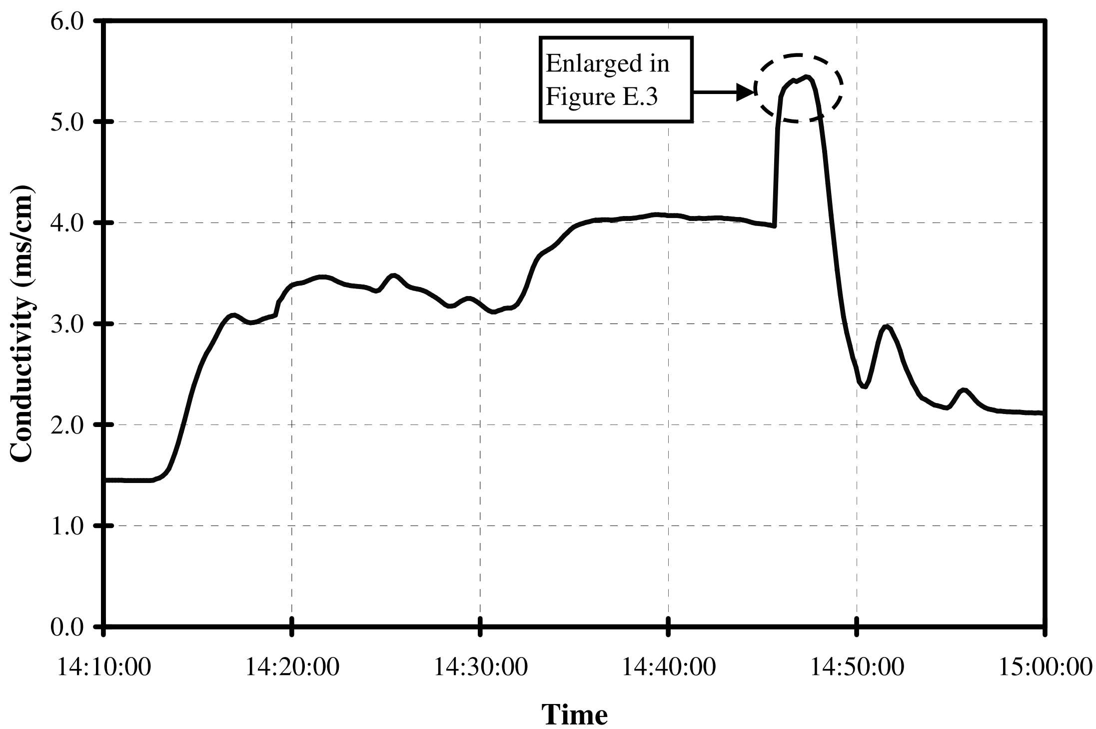

Figure E.2 – Avenue B tracer test conductivity profile.

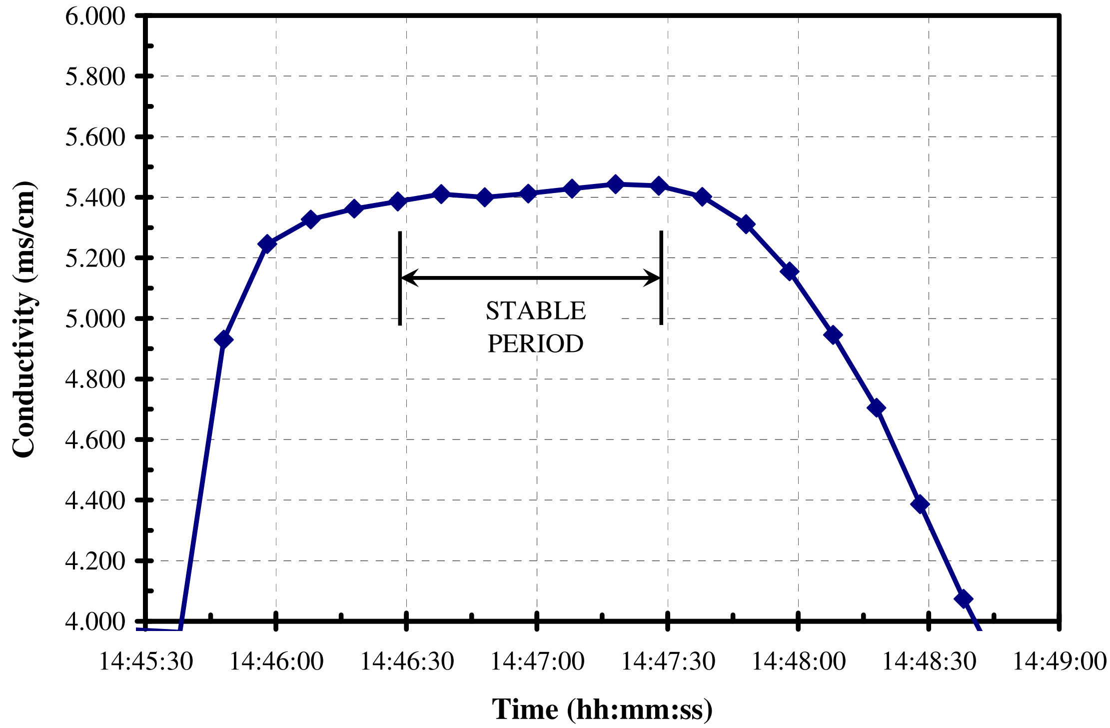

Figure E.3 – Avenue B tracer test conductivity profile enlargement.

Based on the conductivity observed in the stable period, the volumetric ratio of tracer in the storm sewer flow was determined using Figure E.4. The storm sewer flow rate was then calculated using

[E.6] $$Q_{\text{storm sewer}} = Q_{\text{brine}}\left(\frac{1}{\%_{\text{brine}}} - 1\right).$$

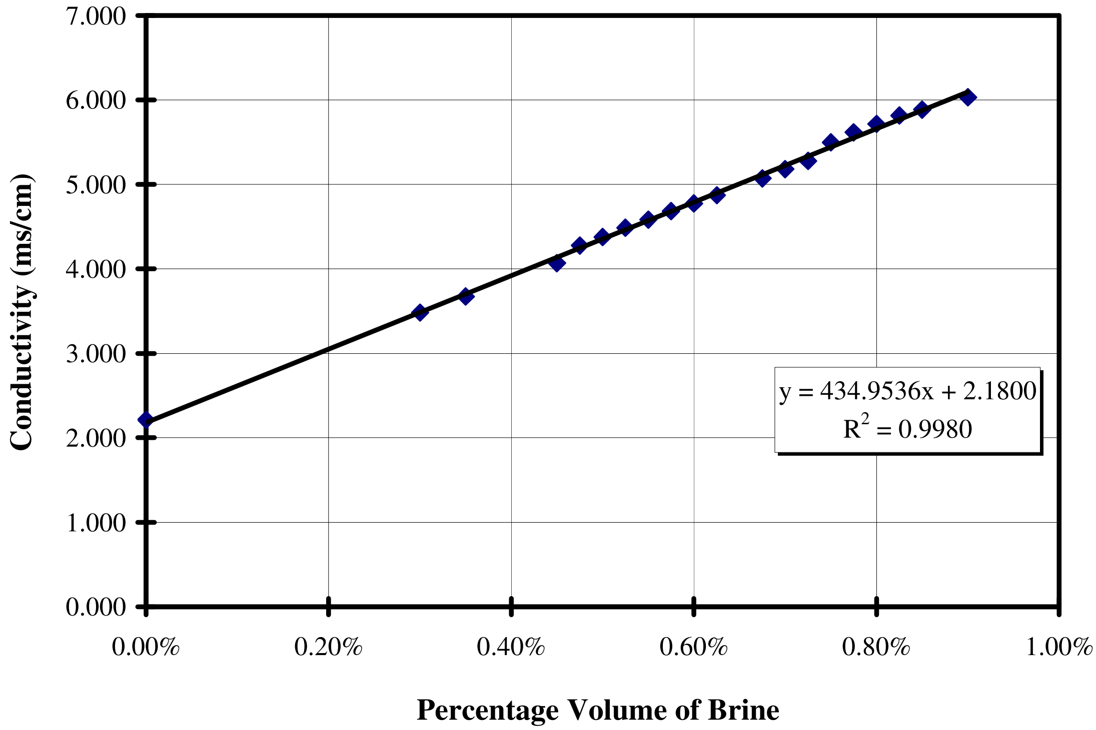

Figure E.4 – Calibration curve for brine in Avenue B storm sewer flow.

The average conductivity over the stable period, shown in Figure E.3, is 5.422 ms/cm. This correlates to a brine volume percentage of 0.745%. Brine was injected at a rate of 3.0 litres per minute, which was confirmed with a bucket and stopwatch. The flow rate in the storm sewer was therefore approximately 400 litres per minute or 6.71 litres per second. The average measured depth of flow was 0.050 m during the test. Therefore, the calculated Manning's roughness coefficient is 0.019. This roughness coefficient is consistent with in situ measurements made by Gerard et al. (1986) for in service concrete sanitary sewer pipes. This roughness coefficient is within the range of roughness for unfinished concrete with rough wood forms (Sturm, 2001).

Table E.1 is a summary of the pertinent data and results for all sites. Following the table, the same array of figures as for the Avenue B catchment is presented for the Taylor Street and Sturgeon Drive catchments. Due to the very shallow depths of flow in the Silverwood catchment, a tracer test was not possible.

Table E.1 – Summary of tracer test data and results.

Parameter	Avenue B	Taylor U of S*	Taylor COS	Sturgeon†
Pipe diameter (m)	1.372	1.524	1.524	1.35
Slope (%)	0.518	0.495	0.411	2.38
Depth of flow (m)	0.050	0.070	0.060	0.018
Q_{brine} (Lpm)	3.0	5.04	5.04	2.40
Measured Conductivity (mS/cm)	5.422	7.755	7.755	7.855
Percentage brine (%)	0.745	1.174	1.174	0.948
$Q_{storm\ sewer}$ (Lps)	6.71	7.15	7.15	4.22
Calculated Roughness	0.019	0.050	0.0249	0.007
Pipe Material	concrete	stainless steel	CSP	concrete

* Backwater effects; see discussion in Chapter 5.

† Result unreasonably small; results not used; see discussion in Chapter 5.

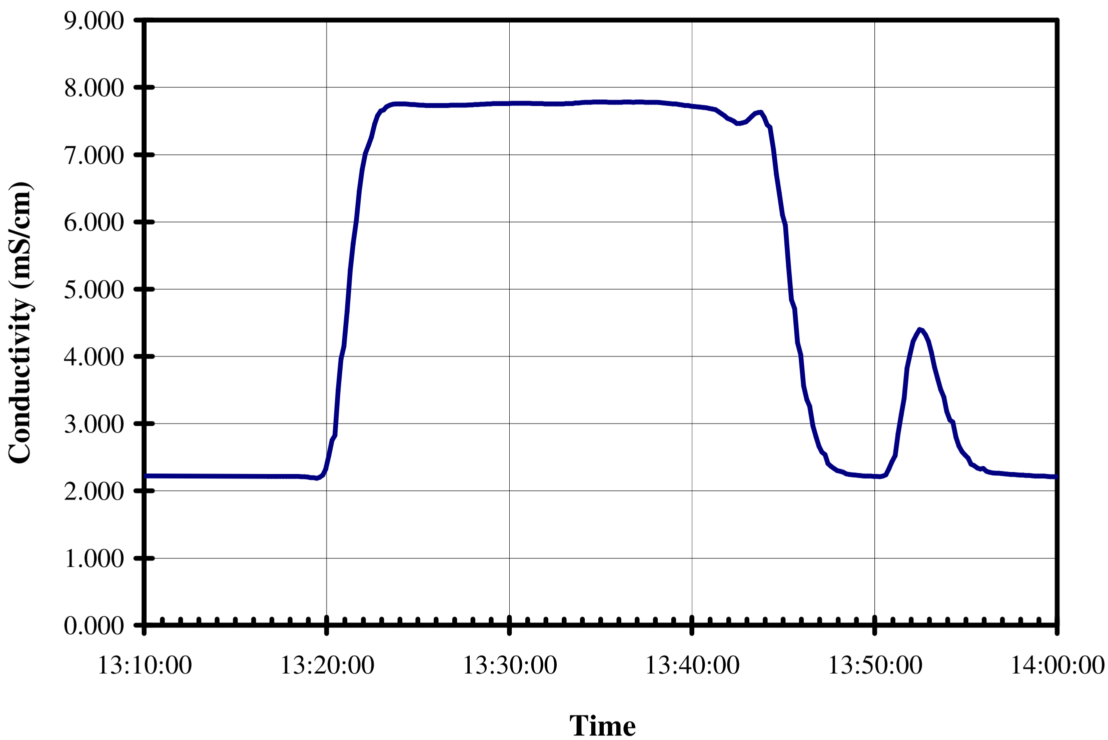

Figure E.5 – Taylor Street tracer test conductivity profile.

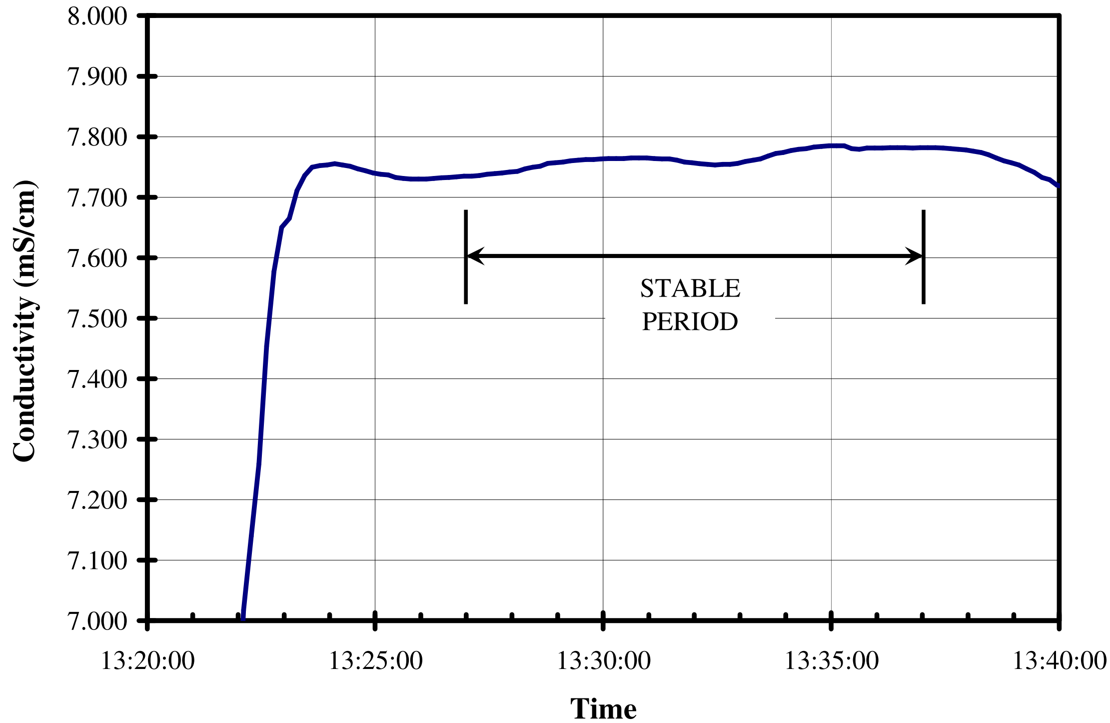

Figure E.6 – Taylor Street tracer test conductivity profile enlargement.

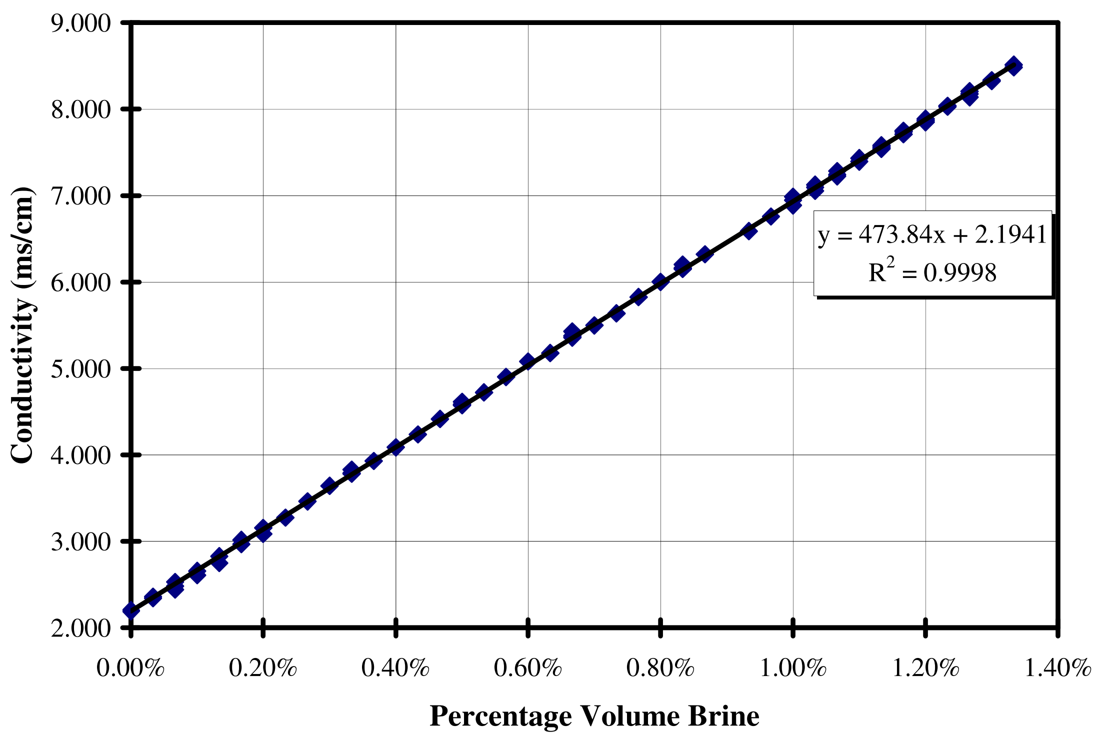

Figure E.7 – Calibration curve for brine in Taylor Street storm sewer flow.

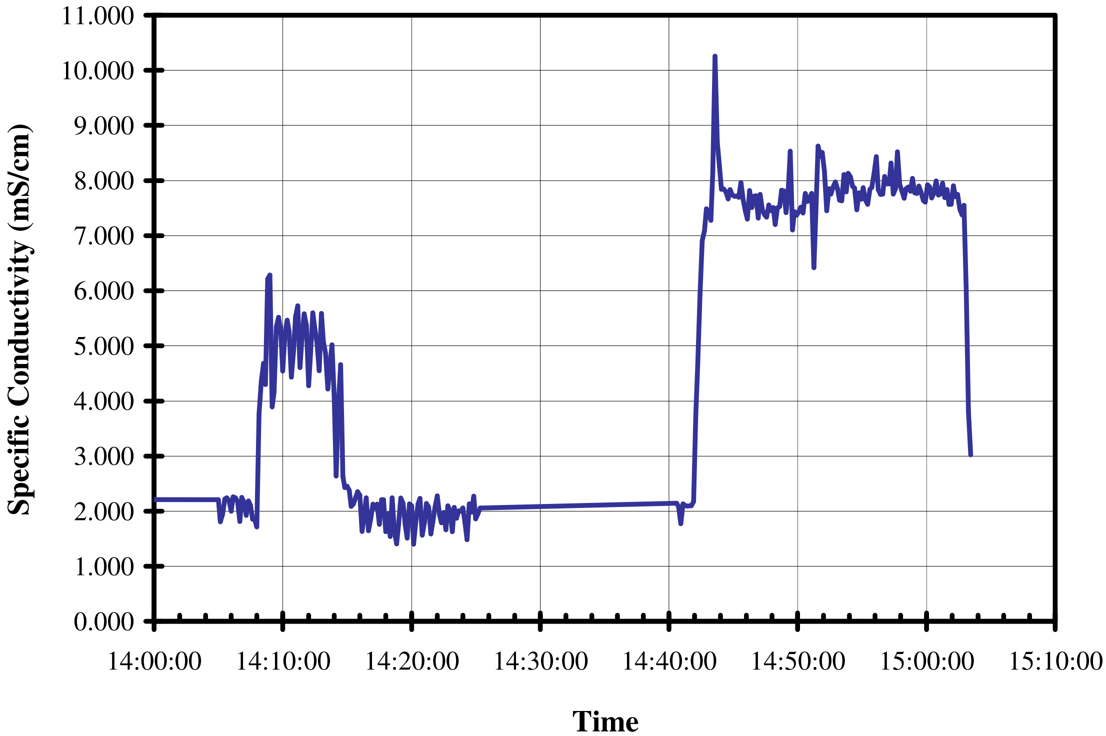

Figure E.8 – Sturgeon Drive tracer test conductivity profile.

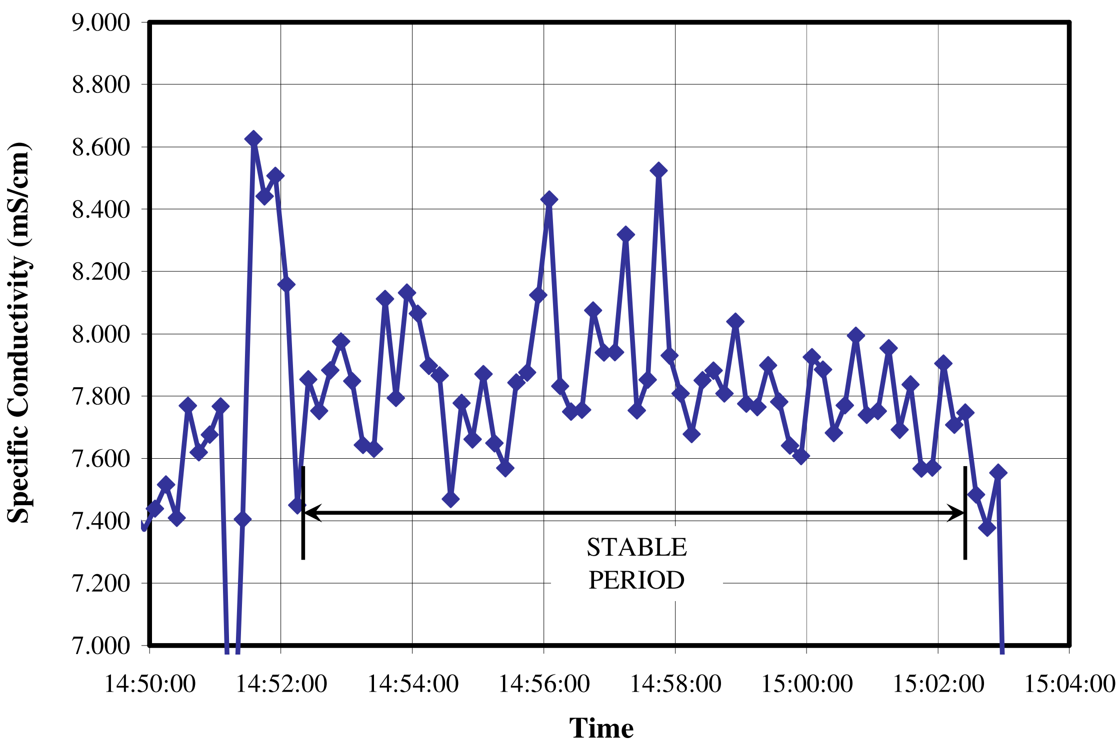

Figure E.9 – Sturgeon Drive tracer test conductivity profile enlargement.

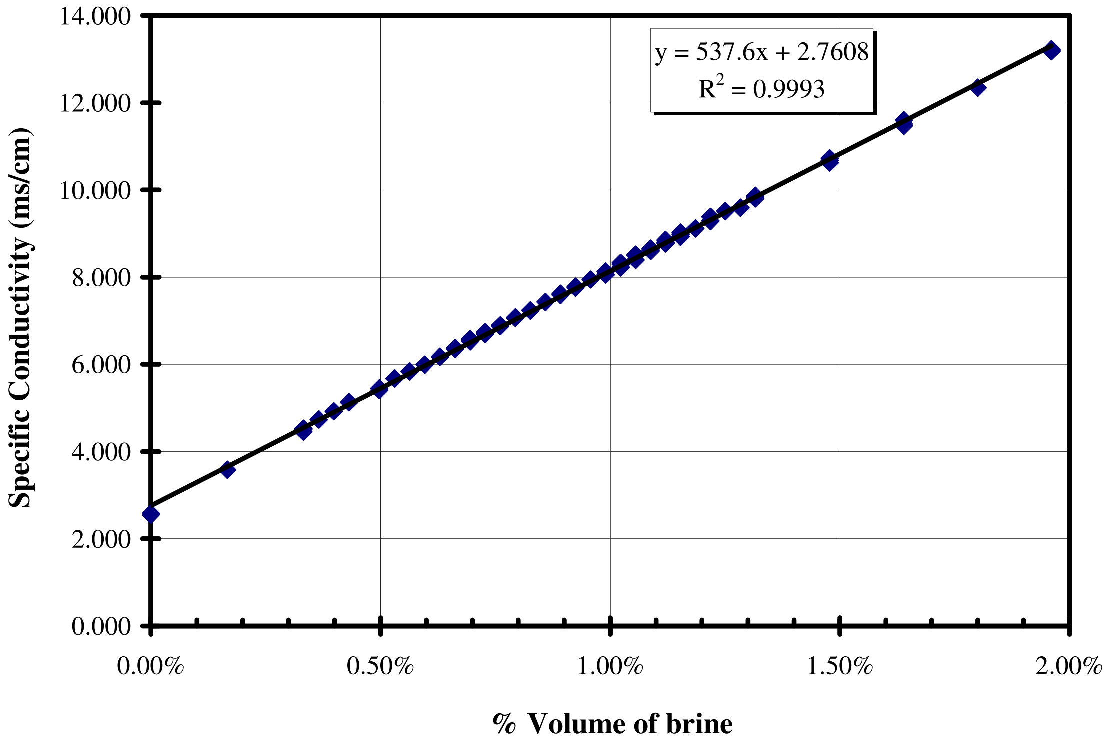

Figure E.10 – Calibration curve for brine in Sturgeon Drive storm sewer flow.

E.3 Location Specific Stage-Discharge Curve Development

E.3.1 General

Manning's equation was used to determine the rating curves. The slope used in Manning's equation to calculate the flow rate is dependent on the water surface profile which is determined by the flow regime (subcritical or supercritical) and the location of the change in slope. In general, if the flow is subcritical both entering and exiting the manhole, the slope of the downstream pipe is used for the calculation, while if the flow is supercritical, the slope of the upstream pipe is used in the calculation. The slope that is used in each case is also dependent on where the change in slope occurs. With the assistance of COS personnel, each location was visually inspected to determine the location of the change in slope. The location specific conditions are examined further in Sections E.3.2 to E.3.5.

E.3.2 Avenue B

The salt tracer technique worked well at the Avenue B site. The storm sewer flow rate calculated from the salt tracer test is subcritical. Therefore, the slope of the downstream pipe was used in Manning's equation to calculate a Manning's n value of 0.019. Both the upstream and downstream storm sewer pipes are concrete. The calculated roughness coefficient is consistent with in-situ roughness measurements made on in-service concrete sanitary sewer pipes by Gerard et al. (1986). The calculation of the roughness coefficient using Manning's equation assumes that the depth measurements were not taken on a transitional water surface profile (e.g. M1).

A schematic profile of the storm sewer pipes and water surface profile at the Avenue B metering location is shown in Figure E.11, with the water flow from left to right. During the visual inspection, the change of slope at the Avenue B metering location was observed to occur at the upstream side of the manhole as illustrated in Figure E.11. The dashed line shows the extension of the normal depth of flow in the upstream pipe. The flow regime during the salt tracer test was determined to be subcritical. The subcritical flow regime and change in slope yield an M1 (backwater) water surface profile that begins at the upstream side of the manhole and proceeds into the upstream pipe. Therefore, the depth observed in the manhole bottom was the same as that in the downstream pipe. As long as the flow regime remains subcritical, the upstream water surface profile will remain M1 and will not interfere with the depth observed in the manhole.

To determine the stage-discharge curve, Manning's equation was used to calculate the flow rate in the Avenue B trunk storm sewer at regular intervals of flow depth. At each interval of the flow depth, the flow was assumed to be subcritical and the assumption was checked after the flow rate was calculated. The flow regime was found to be subcritical for the entire pipe diameter. The stage-discharge curve is shown in Figure E.12.

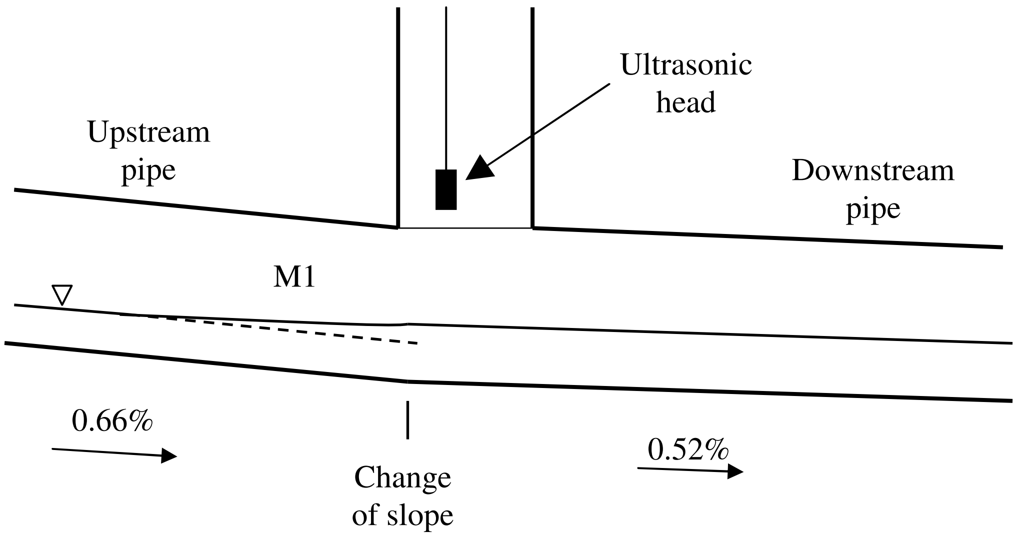

Figure E.11 – Schematic profile of the Avenue B metering location.

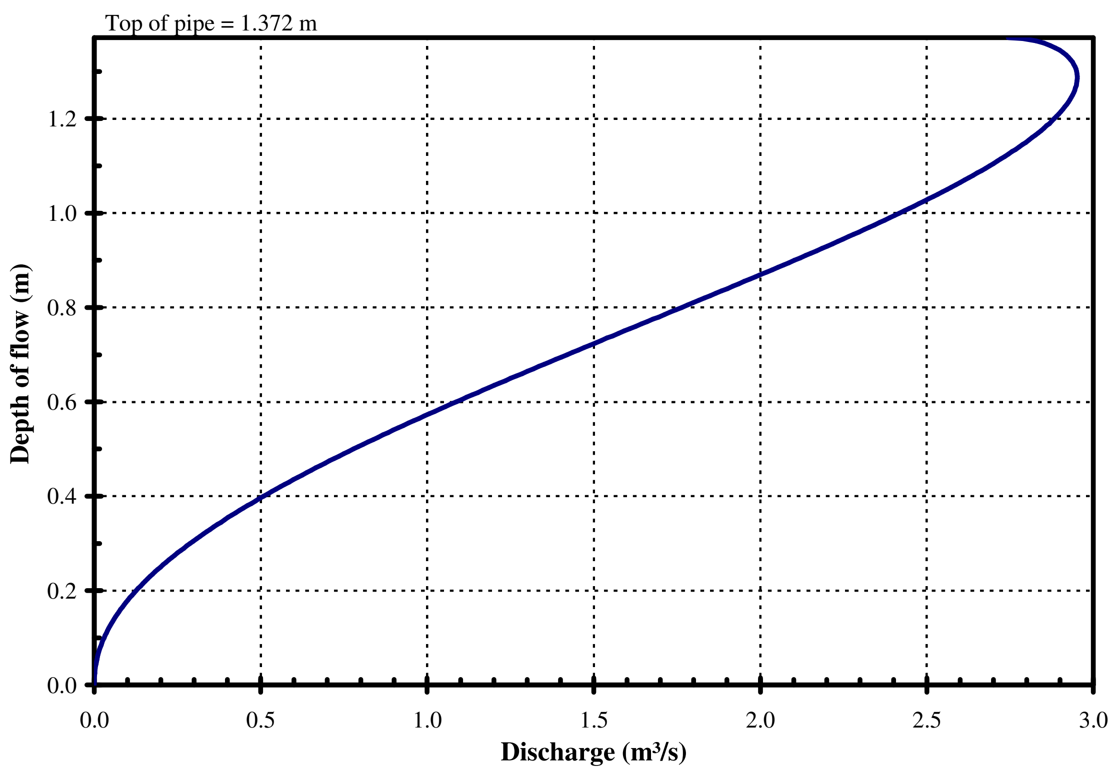

Figure E.12– Avenue B stage-discharge curve.

E.3.3 Taylor Street

In the Taylor Street catchment, two locations were used to monitor the flow rate, necessitating the development of two stage-discharge curves. The two monitoring sites are shown in Figure 3.18. The tracer test was performed, however, the resulting window of stable readings was too small. The tracer test was repeated and injection was continued until sufficient data were collected to allow stability to be established. The COS flow monitoring site is examined first, followed by the U of S flow monitoring site.

The storm sewer flow rate was calculated at the COS flow metering site and the flow was determined to be subcritical. Therefore, the downstream pipe controls the depth of flow. Based on the tracer test data, Manning's n was determined to be 0.025 for the COS flow monitoring site. The section of storm sewer pipe downstream of the COS metering location is the original corrugated metal pipe. The calculated Manning's n agrees well with the normal value listed for corrugated metal storm drains of 0.024, as reported by Sturm (2001). An in-service roughness that is larger than the normal value is not unexpected.

A schematic profile of the storm sewer pipes and water surface profile at the Taylor Street COS flow monitoring site is shown in Figure E.13, with the water flowing from left to right. The change in grade at the COS monitoring site was observed at the upstream side of the manhole. The location of the grade change at this site produces an M1 water surface profile upstream of the manhole. Therefore, the depth of flow measured in the manhole is unaffected by the transitional water surface profile.

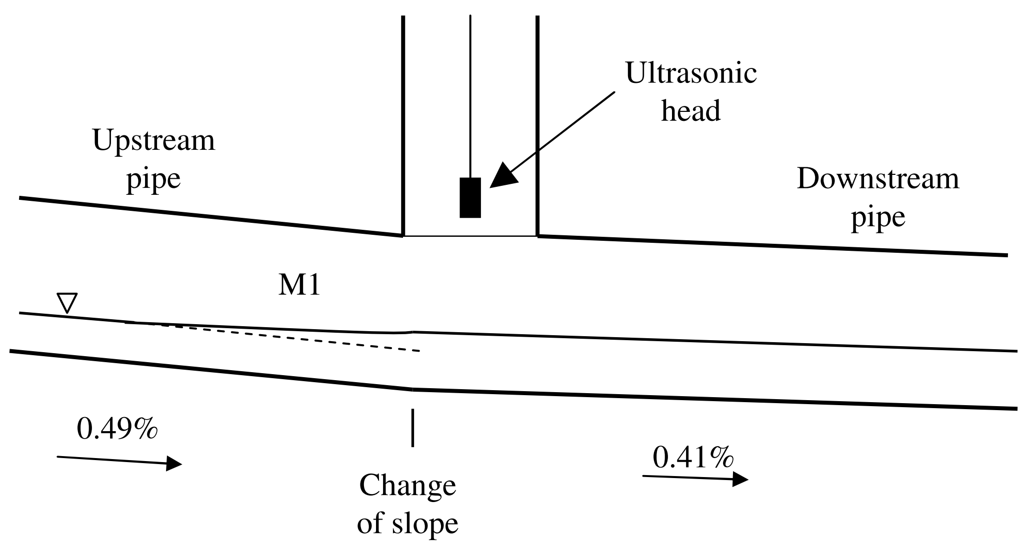

Figure E.13 – Schematic profile of the Taylor Street – COS flow metering location.

To determine the stage-discharge curve, Manning's equation was used to calculate the flow rate in the Taylor Street storm sewer at the COS flow metering site at regular intervals of flow depth. At each interval of the flow depth, the flow was assumed to be subcritical and the assumption was checked after the flow rate was calculated. The flow regime was found to be subcritical for the entire pipe diameter. The stage-discharge curve is shown in Figure E.14 for the Taylor Street COS flow monitoring site.

The layout of the U of S flow monitoring site created a situation that was more difficult for calculating a stage-discharge curve. Using the flow rate determined from the salt tracer test along with the depth measured at the U of S flow monitoring site during the test, the flow regime was found to be subcritical. A Manning's n of 0.051 was calculated using the downstream storm sewer pipe slope. COS has lined both the upstream and downstream pipes with stainless steel to reduce the flow resistance. The calculated Manning's n value is unreasonably high for stainless steel, which has a suggested maximum Manning's n of 0.014 (Sturm, 2000). Further visual inspection of the storm sewer pipe was undertaken. An apparent high point was noted downstream of the manhole, with the backwater area appearing to run into the pipe upstream of the manhole.

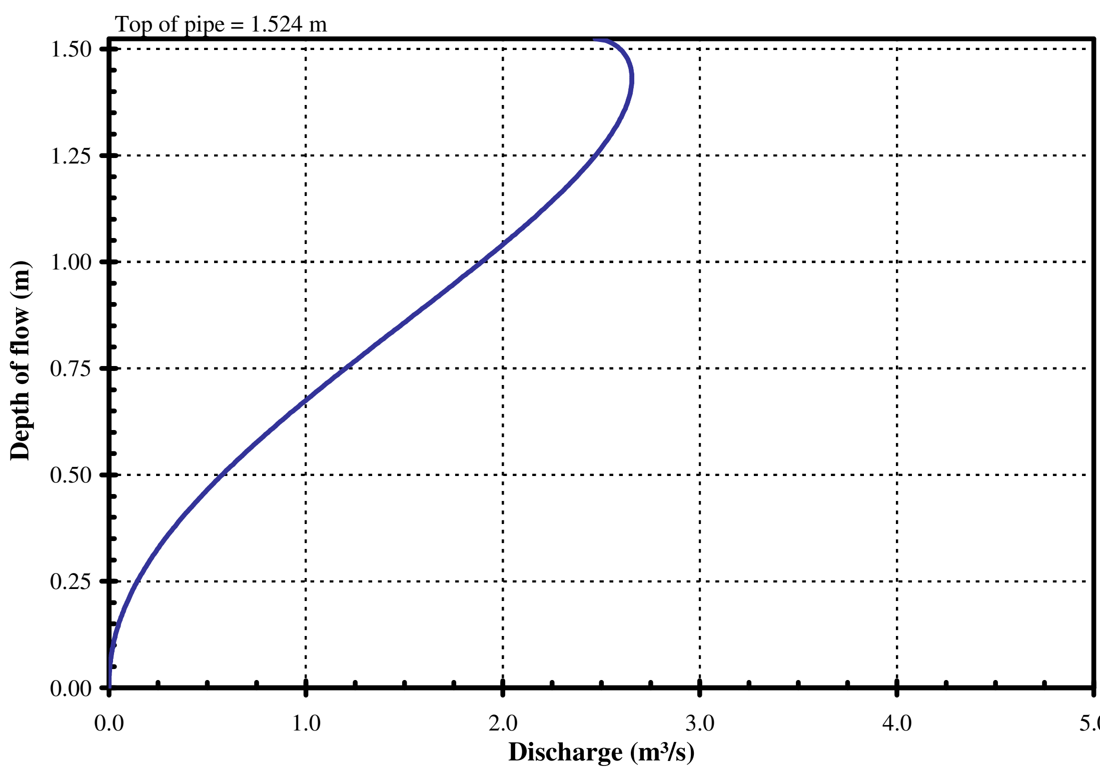

Figure E.14 – Taylor Street Catchment COS flow monitor stage - discharge curve.

An examination of the high point was undertaken to establish its location and elevation. The high point was identified by visual inspection by obstructing the flow in the sewer to better expose the high point. The high point appeared to be caused by the combination of the pipe heaving and a buckle in the stainless steel liner. The storm sewer profile is schematically shown in Figure E.15. With the flow nearly stopped, the water surface was level or nearly level. The water surface was therefore the same elevation in the manhole as it was at the high point. The elevation of the high point was found by measuring the depth of water at the high point and measuring the depth from the rim of the manhole (known elevation) to the surface of the water. The elevation of the high point was found to be the same as the elevation of the pipe invert in the manhole. The storm sewer invert profile was assumed to be approximately horizontal. The distance from the manhole to the high point was 7.5 m. Accounting for the reduced distance between the manholes, the average slope from the high point to the COS flow metering manhole was computed to be 0.54%. Based on this revised information, the water surface profile was re-evaluated.

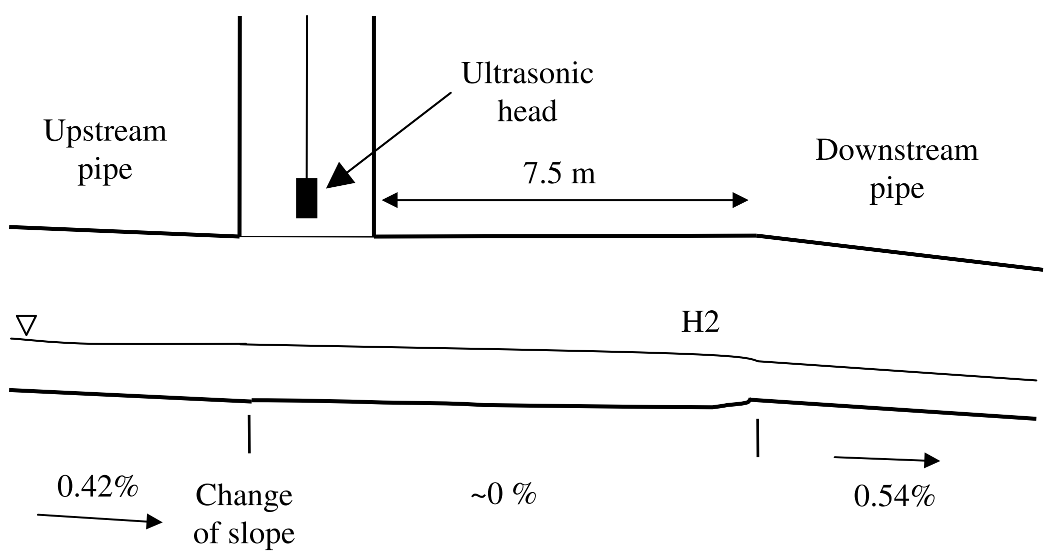

Figure E.15 – Schematic profile of the U of S flow monitoring site in the Taylor Street storm sewer trunk.

The water surface profile is classified as H2, based upon the subcritical flow regime determined in the tracer test and the approximately horizontal pipe invert. The H2 water surface profile influenced the depth measured using the ultrasonic head.

The standard step method was chosen to compute the profile because it allows the depth of flow determined at specified locations (e.g. below the ultrasonic head). The flow rate and depth of flow (downstream in subcritical conditions) are required as input to the standard step method calculations. The downstream control depth was determined using Manning's equation. The slope used in the calculation was the recalculated slope downstream of the high point. The stainless steel liner was lock banded and welded together as well as partially riveted to the original pipe. From Sturm (2000), the maximum roughness for lock banded and welded steel is 0.014 and was chosen for the analysis. The choice of a roughness that is slightly higher than normal roughness was based upon the riveting of the steel, wrinkles in the steel, and the bends in the horizontal alignment of the pipe.

The flow rates were chosen at regular intervals with increased frequency at the smaller values to provide increased resolution in the smaller, more frequently encountered depths. At each step, the flow regime was checked to ensure that the flow remained subcritical. The flow was subcritical for the entire diameter of the pipe. A polynomial line was fit to the data to allow interpolation of depths in between those evaluated. The data from the standard step calculations and the polynomial line fit are shown in Figure E.16. As evident in the figure, the polynomial line fit the standard step data almost perfectly. The polynomial line was fit only as high as the recorded depths of flow required.

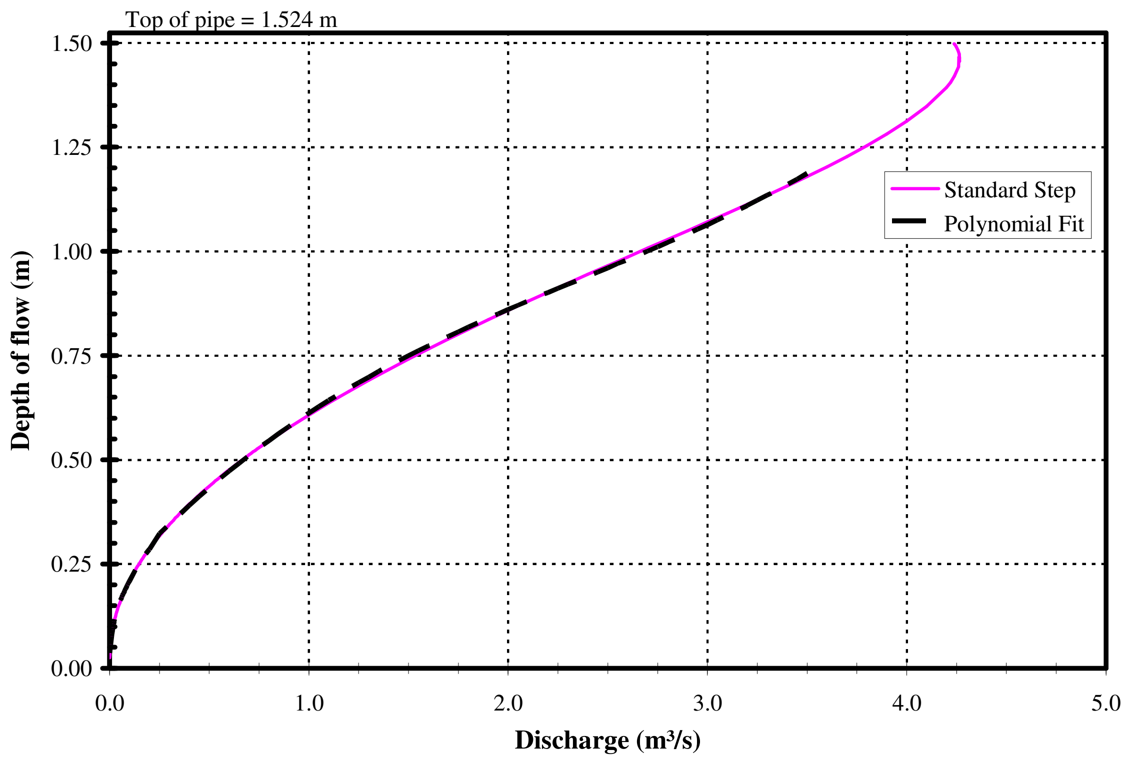

Figure E.16 – Taylor Street catchment, U of S flow monitor stage-discharge curve.

E.3.4 Silverwood

The Silverwood catchment had little or no baseflow. As such, a tracer test was not possible without artificial flow augmentation. Given the diameter of pipe (1.5 m), the volume of flow required to provide reasonable augmentation was not practical. A simple gravimetric method (bucket and stop watch) was also considered. However, the depth of flow was small, making it difficult to accurately measure. The rating curve for this location was prepared using Manning's equation and an assumed Manning's n of 0.019. This roughness was chosen as the pipe appears to be in similar condition (determined by visual inspection) to the Avenue B pipe. The choice of Manning's n is also in line with the findings of Gerard et al. (1986). A schematic profile of the Silverwood monitoring location is shown in Figure E.17.

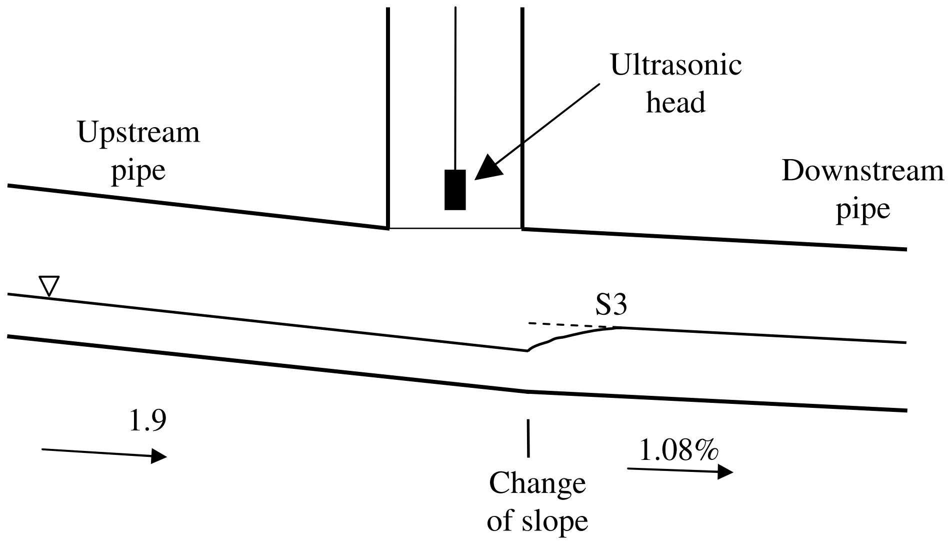

Figure E.17 – Schematic profile of the COS flow monitor location in the Silverwood storm sewer trunk.

As shown in Figure E.17, the change in grade at the Silverwood COS monitoring location occurs at the downstream side of the manhole. Preliminary calculations showed the flow to be supercritical for the majority of the diameter. When the flow is supercritical, an S3 water surface profile develops downstream of the manhole and does not affect the depth of flow in the manhole. The flow rate was then calculated for the entire diameter of the pipe at regular intervals of depth. The flow is supercritical for the majority of the rating curve, except for small depths (i.e. less than 0.010 m) and near the top of the pipe (i.e. greater than 1.35 m). At the small depth of flow, the change of flow regime has a negligible effect on the flow rates. At the large depths, the water begins to contact more of the top of the pipe, increasing the resistance to flow, and the system transitions from supercritical open channel flow to full pipe flow. Since the recorded flow depths did not exceed 0.84 m, examination of the transition from open channel to pipe flow was not undertaken. The resulting stage-discharge curve is shown in Figure E.18.

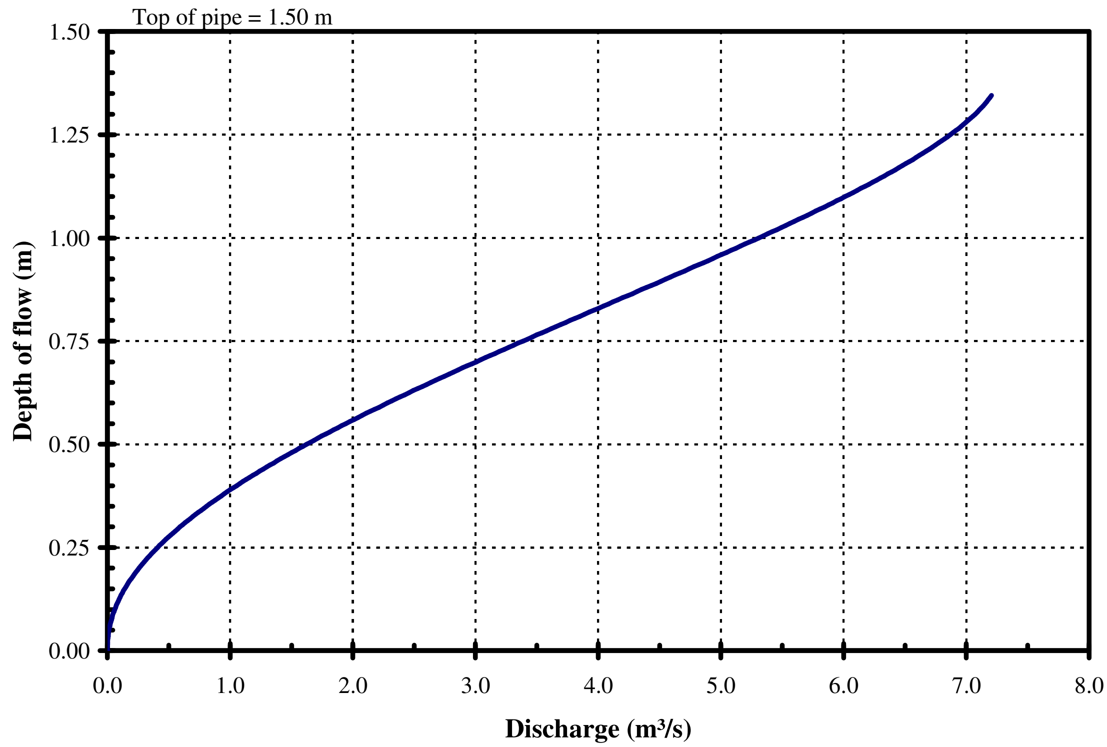

Figure E.18 – Silverwood catchment stage-discharge curve.

E.3.5 Sturgeon Drive

The Sturgeon Drive catchment monitoring site had a tracer test performed. From the flow rate determined in the tracer test, a Manning's n of 0.007 was determined, which is unreasonably small. From Sturm (2000), a glass conduit flowing partly full has a minimum roughness of 0.009, while Lucite has a minimum roughness of 0.008. By visual inspection, the storm sewer pipe was not smoother than glass or Lucite. The cause of this result is likely a combination of factors.

First, the flow depth during the test was small and any misreading of the depth would cause significant change in the calculated roughness. For example, an increase of only 2 mm in the measured depth of flow would cause the roughness to increase to 0.009. The accuracy of this measurement in this test is estimated to be ± 3 mm. The second factor was that, while every effort was made to maintain adequate submergence of the conductivity probe, it may have, from time to time, been inadequately submerged. The third factor was insufficient mixing of the brine solution with the storm sewer flow. Each of the above factors will cause the conductivity meter to report a smaller or fluctuating conductivity measurement. A smaller conductivity measurement causes the dilution of the brine to appear to be larger, which, for the same depth of flow, causes the flow rate to appear larger. The larger flow rate then causes the roughness to appear smaller. The Manning's n determined using the tracer test was disregarded because of the unreasonably small value.

By visual inspection, the pipe was judged to be cleaner, smoother and in slightly better condition than the Avenue B and Silverwood pipes. A Manning's n of 0.018 was chosen. The stage discharge curve was generated using Manning's equation. The flow was supercritical for all flow depths, except for when the flow would begin to contact the top of the pipe at the transition from open channel flow to pipe flow. The recorded flow depths at the Sturgeon Drive monitoring location did not exceed 0.40 m. Therefore, as with the Silverwood monitoring location, the recorded flow depth did not reach the transition and thus the transition was not examined further. The stage-discharge curve is shown in Figure E.19.

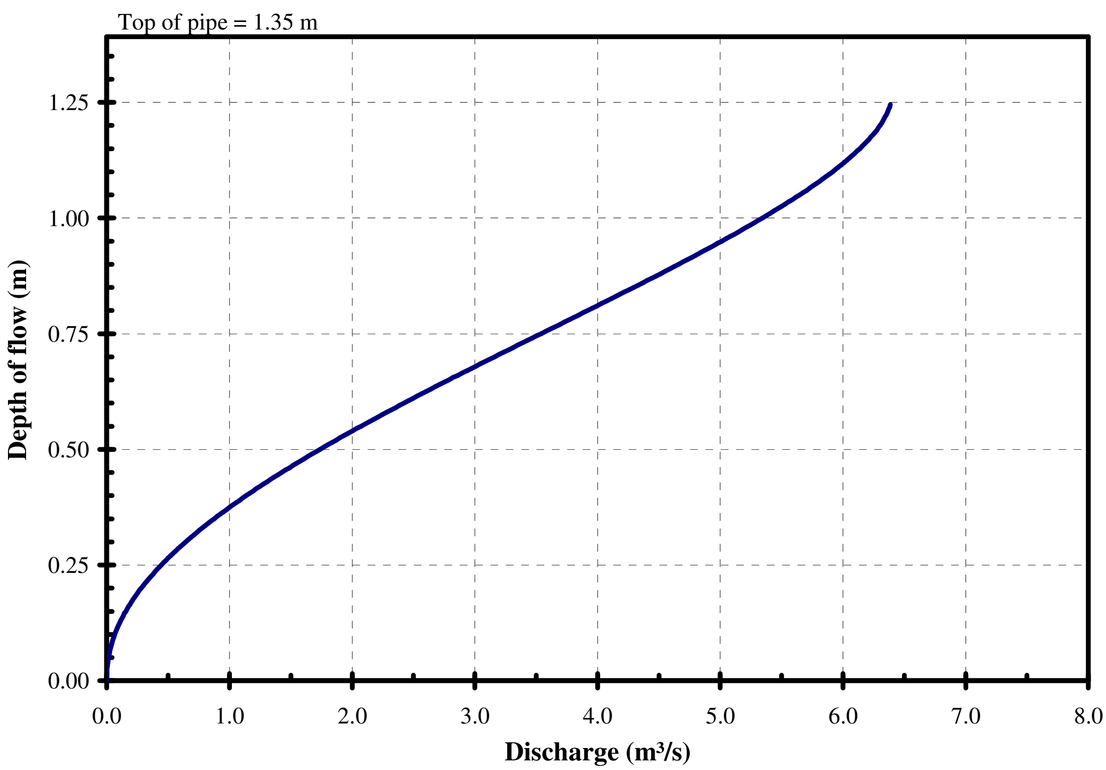

Figure E.19 – Sturgeon Drive catchment stage-discharge curve.

APPENDIX F

QA/QC Data and Analysis

This appendix contains QA/QC data and analysis.

F.1 QA/QC DATA AND ANALYSIS

Three types of QA/QC samples were used as part of the laboratory testing protocol: blank samples, spiked samples, and replicate (duplicate) samples. A total of 12 QA/QC samples were prepared. The breakdown is two blank, two spiked, and eight duplicate samples.

Blank samples were prepared using ultra pure water (UPW). One blank sample was submitted in each of 2001 and 2002. The sample submission bottles were double rinsed with regular water and single rinsed with UPW. The UPW used had a conductivity of 1-3 µs/cm. Table F.1 shows the results of the analysis of the blank samples. In the ideal situation, all results should be less than the detection limit. In these two samples, most parameters are below detection limits and are acceptable blank samples. The parameters that are not less than the detection limit are relatively small values and parameters that can reasonably be explained. In both samples, the conductivity is larger than when the sample was placed in the bottles. This is not unexpected, because the UPW immediately begins to dissolve atmospheric gasses into solution, thus introducing more ions into solution and increasing the conductivity. Also, both conductivity results, 5 and 14 µs/cm, are relatively small compared the results shown in Tables 5.5and 5.6. The conductivity results are acceptable.

Table F.1 – QA/QC blank samples water quality results (samples prepared using UPW).

Parameter	June 18, 2001	July 9, 2002
TSS (fixed) (mg/L)	<1	<1
TSS (volatile) (mg/L)	<1	<1
TSS (total) (mg/L)	<1	<1
Cond. (μs/cm)	5	14
pH	5.7	6.5
Turbidity (NTU)	0.02	0.65
Sulphate (mg/L)	<10	<10
Bicarbonate (mg/L)	5	5
T Alk (mg/L $CaCO_3$)	4	4
Sodium (mg/L)	<1	<1
Magnesium (mg/L)	<1	<1
Calcium (mg/L)	<1	<1
Hardness (Calc)[a]	0	0
Chloride (mg/L)	<2	<2
Potassium (mg/L)	<1	<1
BOD_5 (mg/L as O_2)	<0.1	0.2
DOC (mg/L)	1	2
NH_3 - N (mg/L)		0.03
NO_3^- - N (mg/L)	<0.02	0
TKN (mg/L)	<0.1	0.5
TP (mg/L)	<0.02	<0.02
OP (mg/L)	<0.02	<0.02
COD (mg/L)	<11	<11
TDS (mg/L) (calc)	5	5
TC (org/100 mL)	<1	<10
FC (org/100 mL)	<1	<10
Herbicide Scan Analysis		
Trifluralin	<0.01	
Triallate	<0.04	
Diclofop	<0.04	
Mecoprop (MCPP)	<0.04	
24-D	<0.10	
24-DB	<0.20	
Dichlorprop	<0.10	
MCPA	<0.04	
Dicamba	interference	
Bromoxynil	<0.04	

[a] mg/L as $CaCO_3$

The pH of UPW is very sensitive and not a good measure for QA/QC. Neutral pH, approximately 7, is expected when using pure water. However, as before, the UPW will dissolve gasses from the air, primarily carbon dioxide, which forms carbonic acid, causing the pH to be reduced. The UPW does contain any ability to buffer (counteract) the increase in acid, making it very sensitive with respect to pH. The pH result is acceptable.

Turbidity is an imprecise test (APHA et al., 1992; Metcalf and Eddy, 2003). At low values such as these, the test is sensitive to procedure, experimental setup, and other factors. Finger prints, condensation, scratches on testing vials, and apparatus lamp condition are all potential aggravating factors. The result in the 2001 sample is acceptable, while the result in the 2002 sample is passable, but does raise some concern. The UPW used was extremely clear and should have produced a much lower result.

Both the bicarbonate and total alkalinity concentration results are related to the dissolution of carbon dioxide from the atmosphere. The results are small and acceptable.

The BOD_5 result that is above detection is small and acceptable. The BOD_5 test is also somewhat imprecise and a result such as this is not unexpected.

The DOC results are small but not completely insignificant. There are a few potential sources of the DOC. One possibility is leaching from the plastic sample bottle or the bottle cap liner. Another possibility is contamination in handling of the sample in the testing process (APHA et al., 1992). Also the concentrations in the blank samples are small in comparison to the concentrations in Tables 5.5 and 5.6. While the result is not perfect, it is acceptable.

The ammonia result from the 2002 sample is small and acceptable. The TKN result from 2002 is cause for concern; it was relatively large compared with the range of results presented in Tables 5.5 and 5.6. Further QA/QC testing with regard to TKN was undertaken and is reported in subsequent paragraphs.

The TDS result is acceptable. As it is a calculated value, the value shows the of the bicarbonate concentration.

The Dicamba analysis resulted in an "interference" result. The Dicamba analysis was performed using a gas chromatograph. The "interference" result is returned when the peak of another compound overlaps or distorts the peak of the substance of interest.

Dicamba interference can be caused by small concentrations of plasticizers used in common flexible plastics (APHA et al., 1992). This contamination may have occurred because some of the conduits carrying the UPW are flexible plastics. Standard Methods (APHA et al., 1992) also notes that cross-contamination can easily occur from handling flexible plastics, then handling sampling/testing equipment.

Two spiked samples were carefully prepared and submitted to the Provincial Lab. Table F.2 shows the dates of the samples, the prepared concentrations, and results from the Provincial Laboratory.

Table F.2 – QA/QC spiked samples.

Sample date (mm/dd/yy)	Parameter	Prepared Concentration (mg/L)	Lab Result (mg/L)
08/06/02	DOC	50.0[a]	48
10/09/02	TKN	10.0	10
	OP	0.50	0.5

[a] - U of S Lab determination 52.5 mg/L

The results of the spiked samples are excellent. The first spiked sample, which contained 50.0 mg/L of DOC, yielded a Provincial Lab result of 48 mg/L and a U of S Lab analysis result of 52.5 mg/L. The two samples have acceptable results.

The second spiked sample contained 10.0 mg/L of TKN and 0.50 mg/L of OP. The results from the Provincial Lab match the prepared concentrations very well. The TKN result provides some confidence that the test results are representative and the previous poor TKN QA/QC result was likely an anomaly. The Provincial Lab OP concentration matches the prepared concentration.

Since duplicate samples are much easier to prepare than blank or spiked samples, duplicate samples were used more frequently. Three sets of duplicates were submitted in each sampling season. Table F.3 presents the duplicate samples submitted in 2001. The duplicate samples were created by thoroughly mixing a sufficient volume of each discrete sample in a large jug and pouring the thoroughly mixed composite sample into two separate, clean sample submission bottles. In both full sets of analyses, all of the duplicate results agree reasonably (i.e. within approximately 15% or less), except for turbidity in the first set, chloride in the second set, and the TC and FC results in both sets. The difficulties with turbidity have already been presented in the blank sample discussion. This result further supports the previous discussion. The chloride result in the August 14, 2001 samples is out of the ordinary. While several substances can interfere with chloride determination, duplicate samples have the same content and therefore should have similar results. The chloride results of August 14, 2001 do not have an obvious explanation. Possible causes are inadvertent sample switching at the laboratory, mislabelling of samples split for analysis, analysis error, mislabelling of the result or incorrect entry of the result into the reporting system.

The TC and FC results in the first set (June 28, 2001) are poor. The TC results are about an order of magnitude different. While some variability within the same order of magnitude in biological results can be expected, this result is too different. The major contributing factor is likely the delay between collection and analysis of the samples of seven days for the June 28, 2001 samples caused by delays in shipping. The maximum time between sample collection and analysis for TC and FC is 24 hours (APHA et. al, 1992). With such a large delay between sampling and analysis, this test should neither have been performed nor had results reported. The second set of duplicates also shows differences approaching an order of magnitude. The delay in shipping was only two days for this sample. The results are poor, but are at least still able to provide some indication of the order of magnitude of the TC's and FC's. The final duplicate sample (August 23, 2001) exhibits good agreement in the TC's and reasonable agreement in the FC's. The delay between sampling and analysis for these samples was less than one day. This sample was specifically collected and submitted to examine the TC and FC results and the results indicate that the delay due to shipping is a problem.

Table F.3 – 2001 QA/QC duplicate samples (mg/L).

Parameter	Sample date (mm/dd/yy)					
	06/28/01	06/28/01	08/14/01	08/14/01	08/23/01	08/23/01
TSS (fixed)	122	142	332	392		
TSS (volatile)	34	41	64	73		
TSS (total)	155	183	396	465		
Cond. (µs/cm)	1161	1159	435	437		
pH	7.6	7.6	7.2	7.3		
Turbidity (NTU)	109.6	82.3	22.5	21.7		
Sulphate	349	354	100	100		
Bicarbonate	217	217	107	110		
Total Alkalinity [a]	178	178	88	90		
Sodium	54	54	16	16		
Magnesium	57	56	17	17		
Calcium	122	122	46	47		
Hardness[a] (calc)	539	535	185	187		
Chloride	57	56	8	24		
Potassium	7	7	3	3		
BOD_5			18	20		
DOC	23	23	44	44		
NH_3 - N			1.09	1.08		
NO_3^- - N			0.96	1		
TKN			4.7	2.7		
TP			0.82	0.88		
OP			<0.02	<0.02		
COD	165	165	290	290		
TDS (calc)	863	866	297	317		
TC (org/100 mL)	1.5×10^4	2.4×10^5	1.2×10^5	1.1×10^6	2.0×10^2	2.1×10^2
FC (org/100 mL)	9.0×10^1	4.3×10^2	1.1×10^2	7.5×10^2	<30	1.5×10^2

[a] mg/L as $CaCO_3$

Table F.4 presents the results of the QA/QC duplicate samples for 2002. The first pair of samples (June 19, 2002) contains a several parameters that were significantly different from one another. TSS and turbidity are both somewhat methodologically dependent and can show difference between duplicate samples, however these differences are significant. BOD_5 is a biologically based test and can occasionally produce poor results because of various factors related to the biological seed used in the test. The differences in the other results are not easily explained. Kavelaars (1998) noted that the largest source of error in their analysis program was inadvertent sample switching in the laboratory. Perhaps this is the case here, however it is not possible to identify a specific cause. The number of anomalies suggests that something has happened to one of the June 19, 2002 samples. The TC and FC results are reasonable, as they are in the same order of magnitude and close in value. The analysis of the TC and FC samples was performed at the COS Water Treatment Plant Lab, while the other parameters were analyzed by the Provincial Lab.

Table F.4 – 2002 QA/QC duplicate samples (mg/L).

Parameter	**Date (mm/dd/yy)**					
	06/19/02	**06/19/02**	**07/16/02**	**07/16/02**	**08/07/02**	**08/07/02**
TSS (fixed)	256	148	2	2	287	262
TSS (volatile)	67	48	5	5	62	56
TSS (total)	324	197	8	8	348	318
Cond. (µs/cm)	533	617	1900	1890	130	141
pH	7	6.9	7.6	7.6	7.2	7.4
Turbidity (NTU)	151.9	81.2	3.72	3.93	108.5	96.6
Sulphate	113	156	655	668		29
Bicarbonate	113	120	339	339	44	46
Total Alkalinity [a]	92.8	98	278	278	36	38
Sodium	36	34	129	131	5	5
Magnesium	19	22	110	108	3	4
Calcium	39	59	150	149	13	18
Hardness[a] (calc)	176	238	828	817	45	61
Chloride	34	33	90	89	4	4
Potassium	6	6	12	12	2	2
BOD_5	21.9	8.3	15.6	18.3	5.6	5.4
DOC	21	23	18	19	7	7
NH_3 - N	0.73	0.29	8.4	8.58	0.39	0.36
NO_3^- - N	0.81	0.79	0.86	0.67	0.58	0.6
TKN	3.4	2.9	10.9	9.2	4.5	3.5
TP	0.84	0.84	1.53	1.52	0.8	0.87
OP			0.8	0.77	0.07	0.07
COD	213	100	44.4	44.4	76.2	89.9
TDS (calc)	360	130	1485	1496	71	108
TC (org/100 mL)	$4.9 x 10^6$	$6.5 x 10^6$	$4.5 x 10^5$	$4.6 x 10^5$	$6.6 x 10^5$	$4.8 x 10^5$
FC (org/100 mL)	$1.7 x 10^5$	$2.8 x 10^5$	$8.5 x 10^4$	$4.1 x 10^4$	$2.0 x 10^4$	$2.4 x 10^4$

[a] mg/L as $CaCO_3$

The July 16, 202 duplicate samples agree very well. The FC results are approximately half an order of magnitude different. While this result is not great, variation within an order of magnitude is not unexpected. These two samples provide an acceptable result.

The results of the August 7, 2002 sets of duplicate samples agree quite well, including TC's and FC's. These two samples provide an acceptable result.

A set of duplicate biological QA/QC samples were collected October 10, 2002 when baseflow samples were collected. The results are presented in Table F.5. Both the TC and FC results are in the same order of magnitude and are similar in value. These sample results are acceptable.

Table F.5 – Additional biological duplicates.

Parameter	Date (mm/dd/yy)	
	10/09/02	10/09/02
TC (org/ 100mL)	1.4 x10^4	2.3 x10^4
FC (org/ 100 mL)	2.2 x10^3	2.1 x10^3

Generally speaking, the water quality QA/QC analysis provides an acceptable level of confidence in the water quality parameter concentration results. One set of duplicate samples (June 19, 2002) appear to have something wrong, because the results are somewhat different. A poor TKN result in one of the blank samples (July 9, 2002) appears to be an anomaly, as the remainder of the samples show good agreement with their respective comparisons. The same situation is true for chloride as well. The variations from the expected results in all other QA/QC samples, are able to be explained. The shift of the analysis of the biological parameters to the COS Water Treatment Plant lab corrected the delay problems and provided good results. Marsalek (1991) states that the level of precision required of the water quality parameter analyses is dependent upon the usage of the data. In the case of stormwater, which has many other vagaries (Novotny, 1992), the precision need not be high.

APPENDIX G

Event Graphs

This appendix contains the hydro/hyeto, polluto- and loadographs for all of the useable events. The pollutograph and loadograph are plotted for the discrete TSS data only. Each set of figures contains a) hyteo/hydrograph (flow and rainfall), b) pollutograph (concentration), and c) loadograph (mass flow rate).

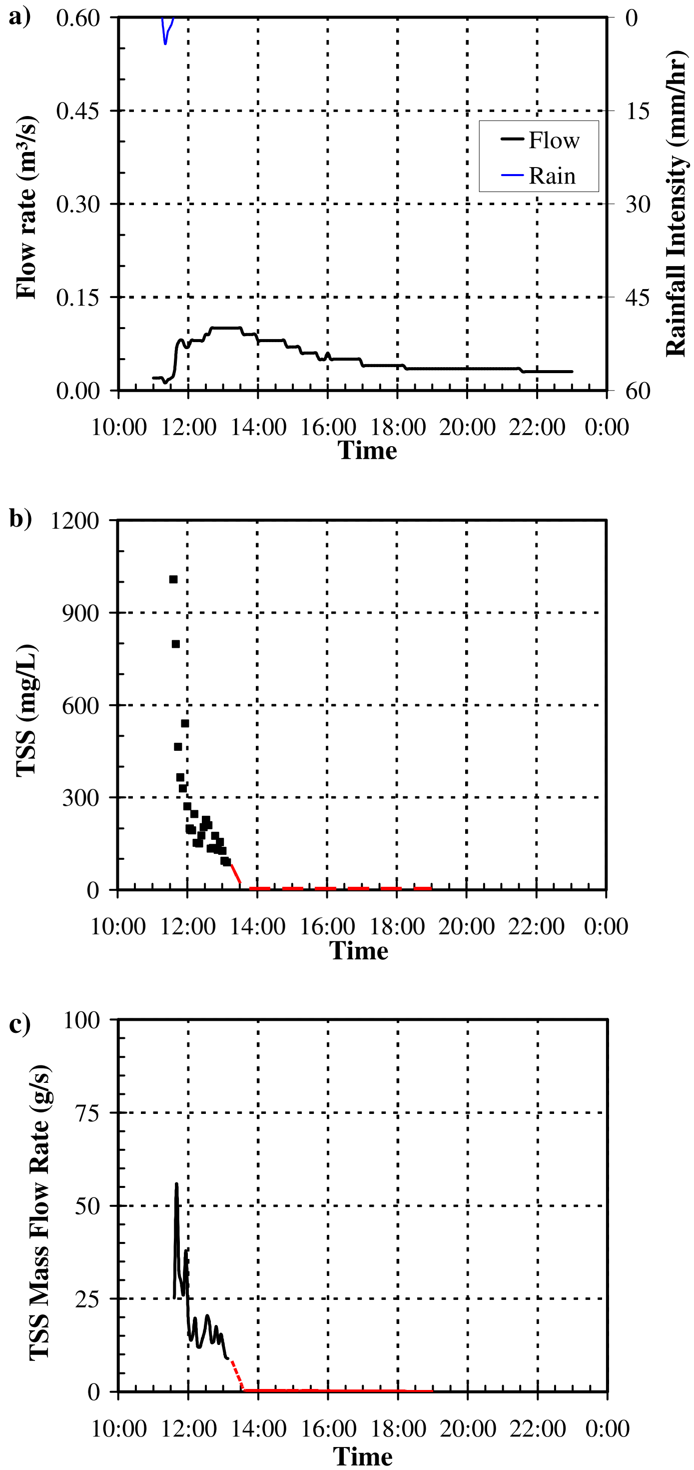

Figure G.1 – Taylor Street – June 28, 2001.

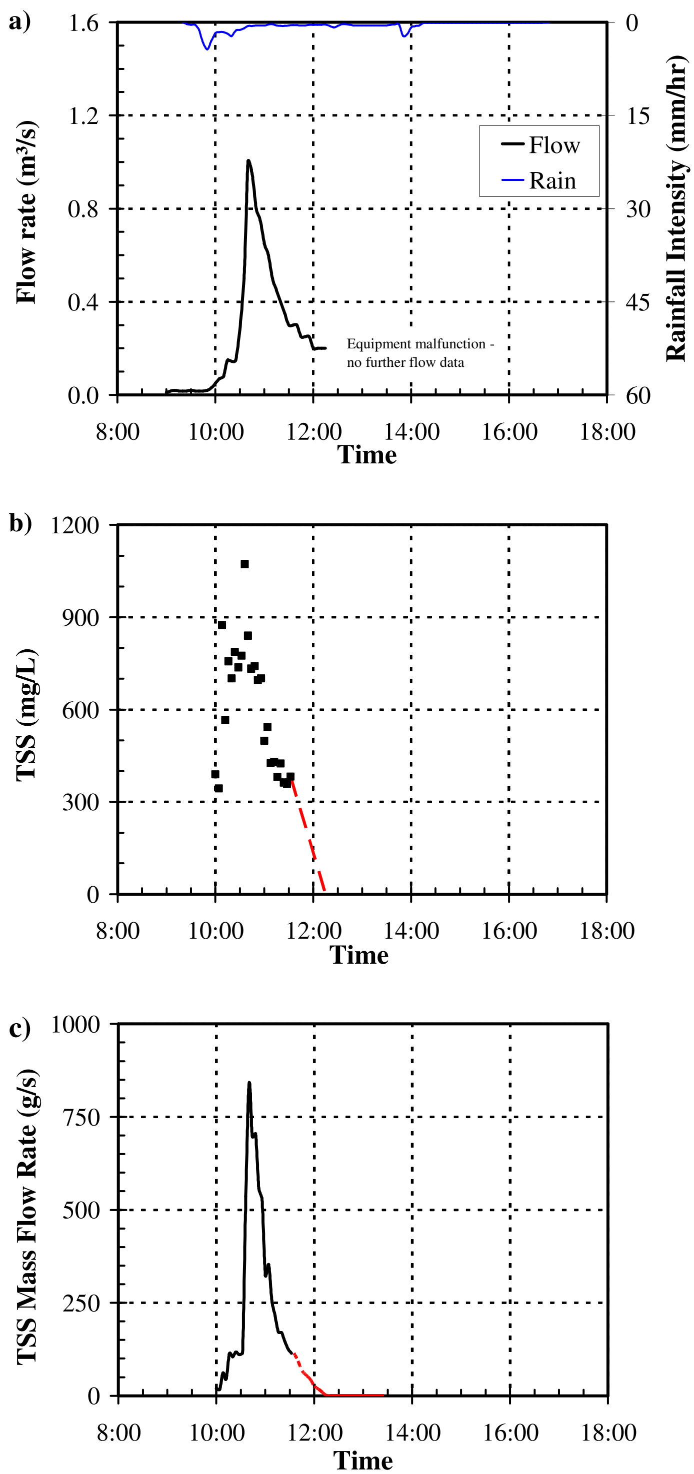

Figure G.2 – Taylor Street – July 16, 2001.

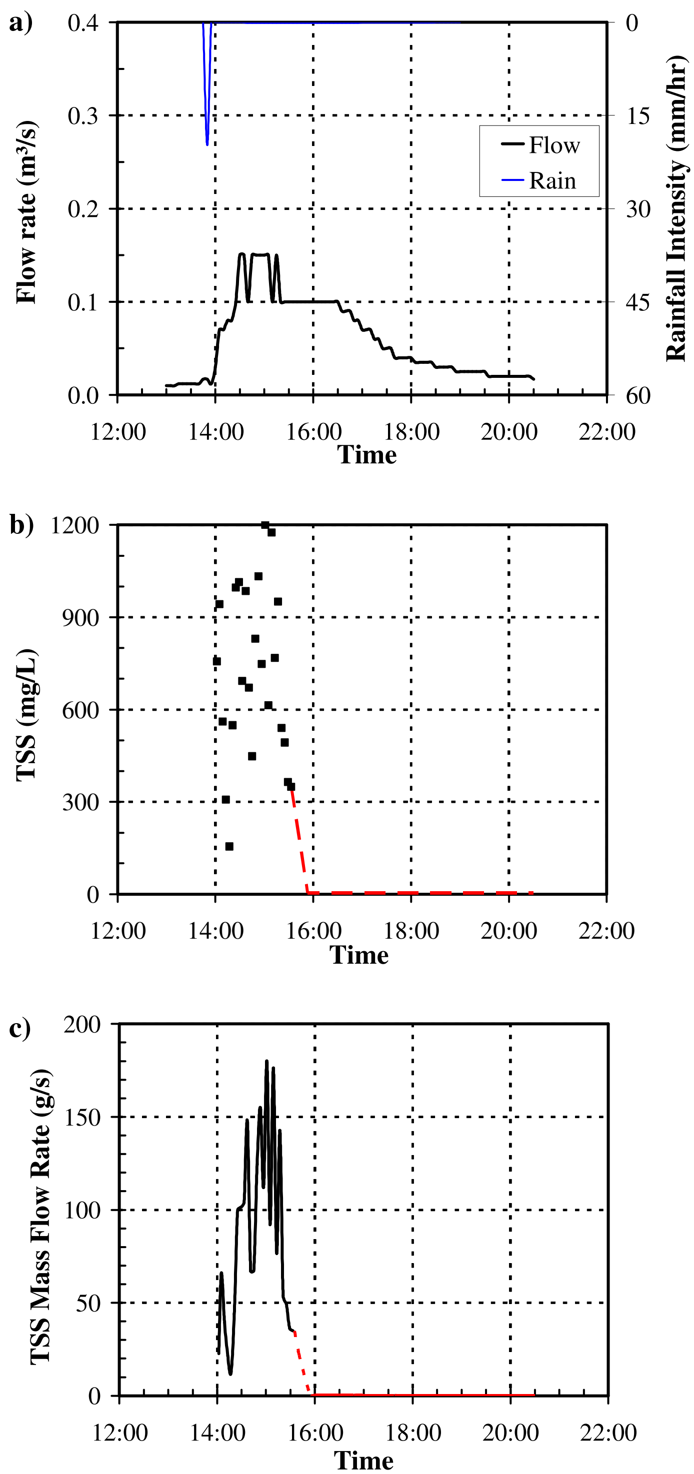

Figure G.3 – Taylor Street – August 14, 2001 – Event B.

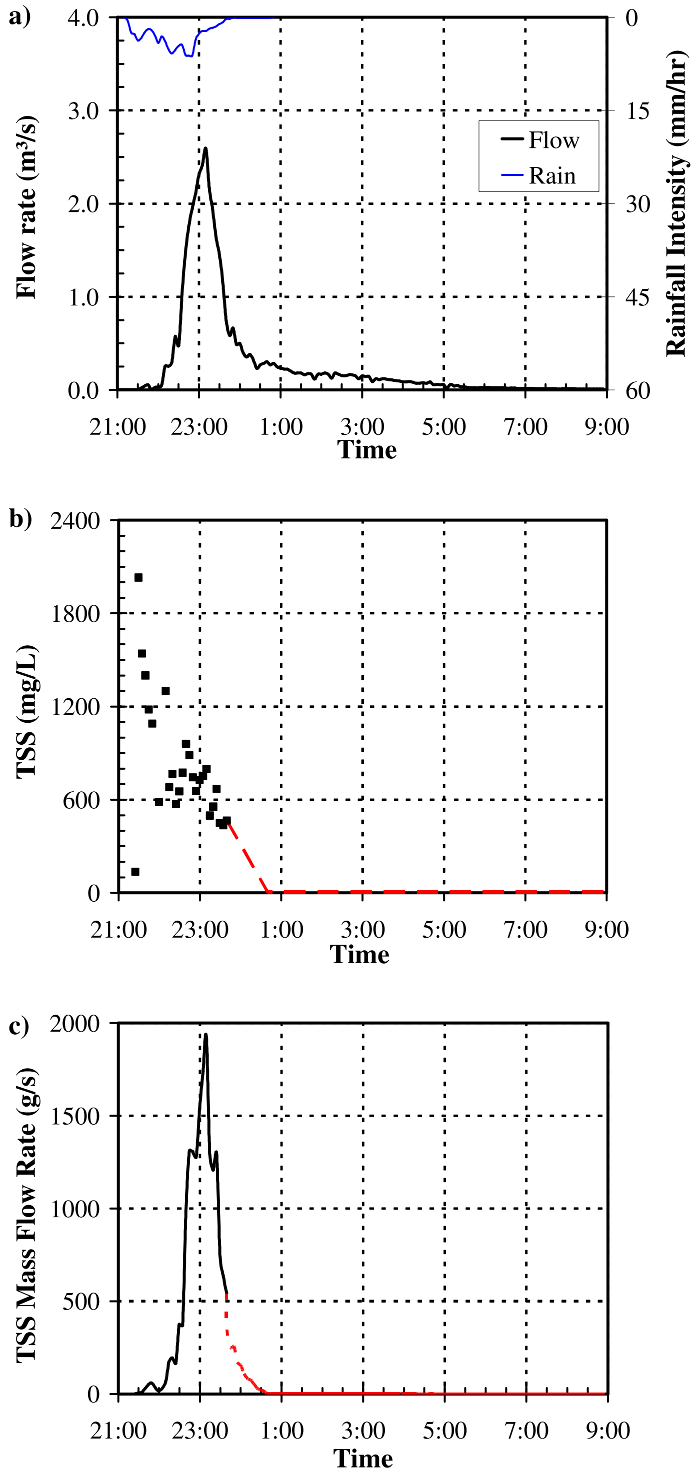

Figure G.4 – Taylor Street – June 17, 2002.

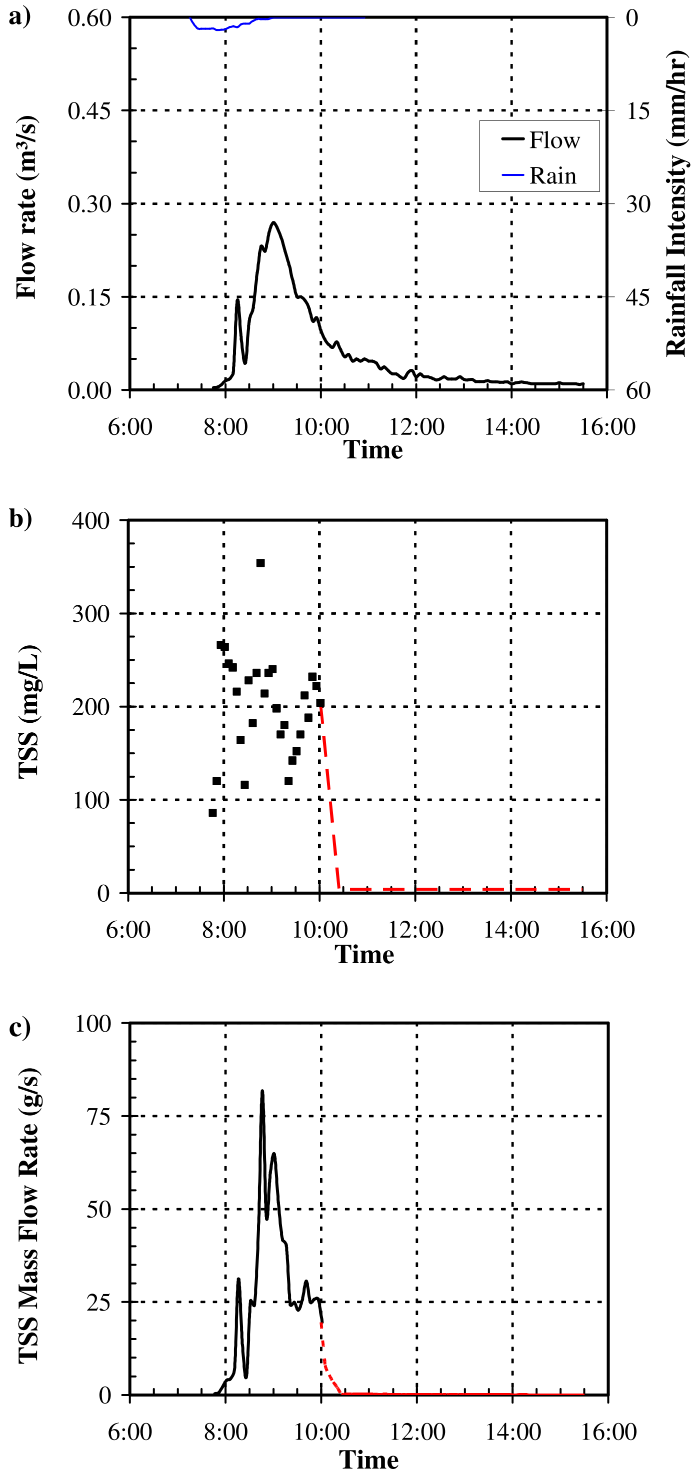

Figure G.5 – Taylor Street – June 19, 2002.

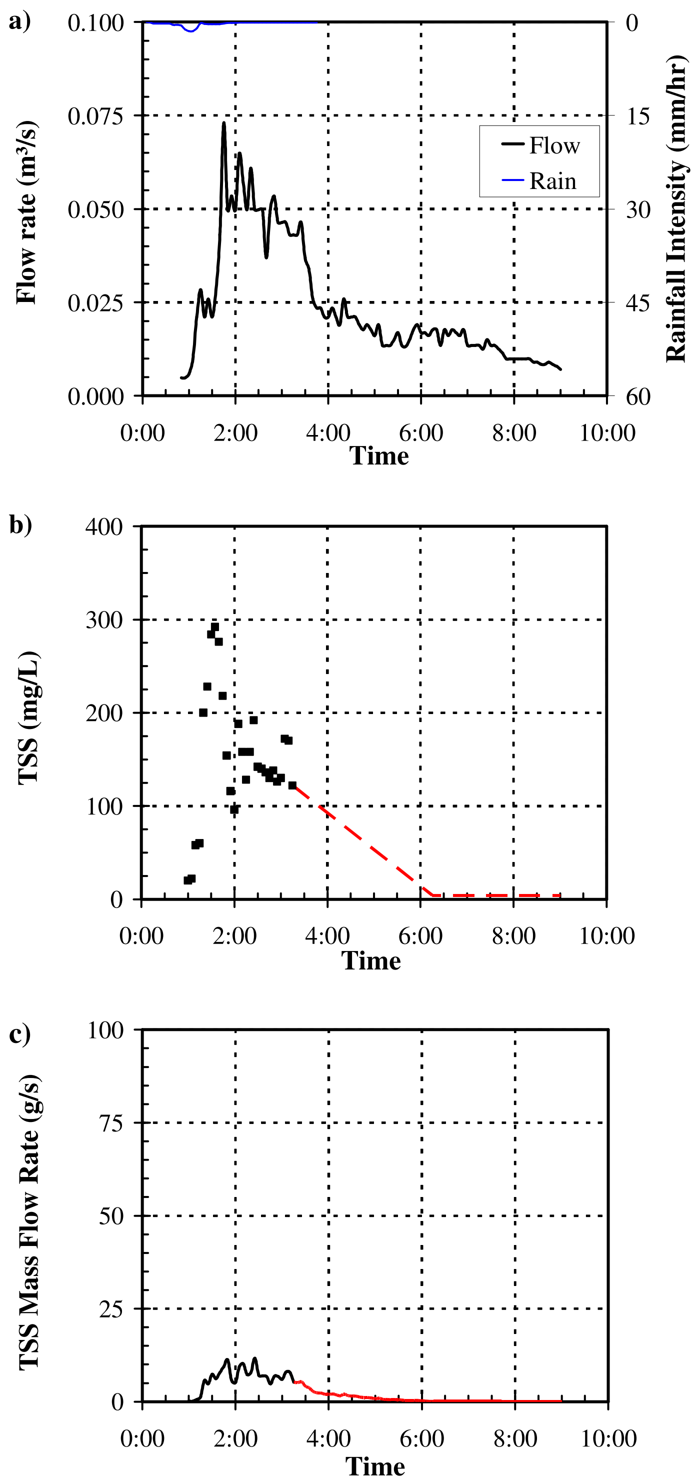

Figure G.6 – Taylor Street event beginning July 4, 2002.

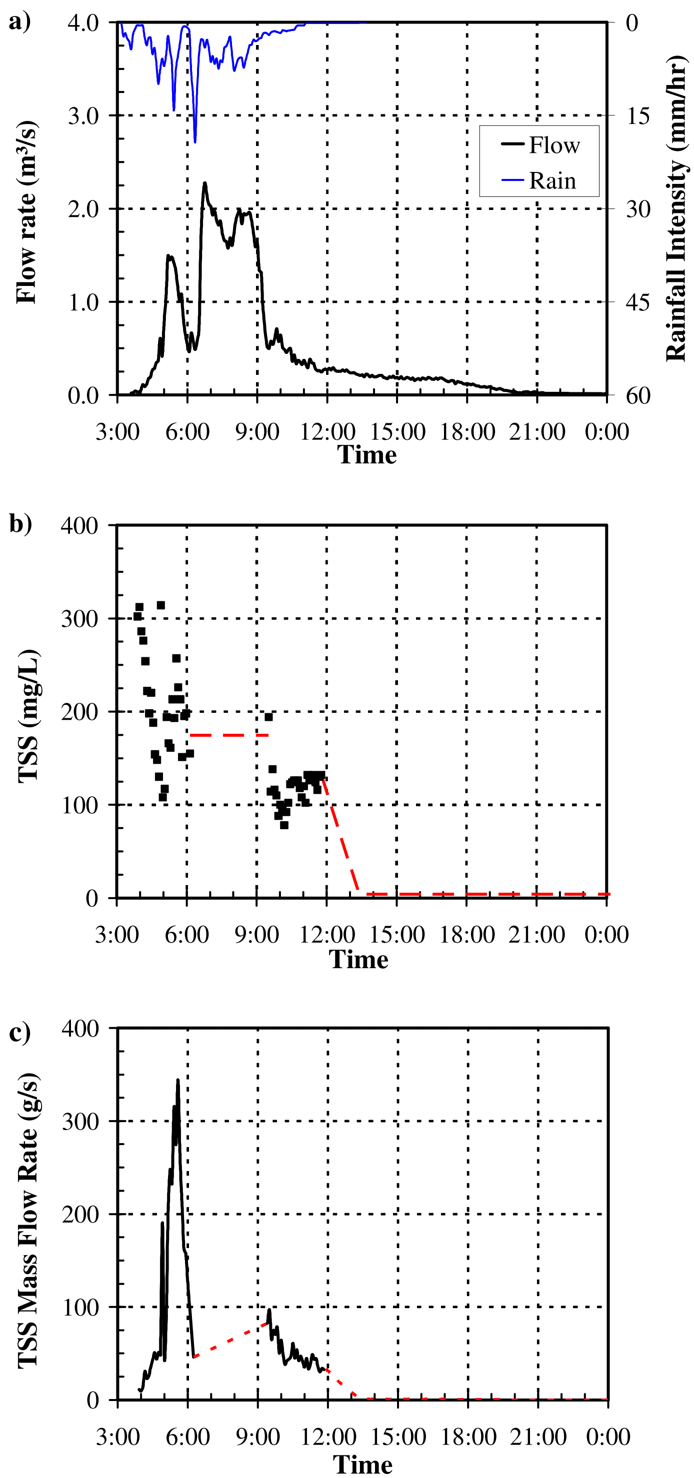

Figure G.7 – Taylor Street event beginning July 9, 2002.

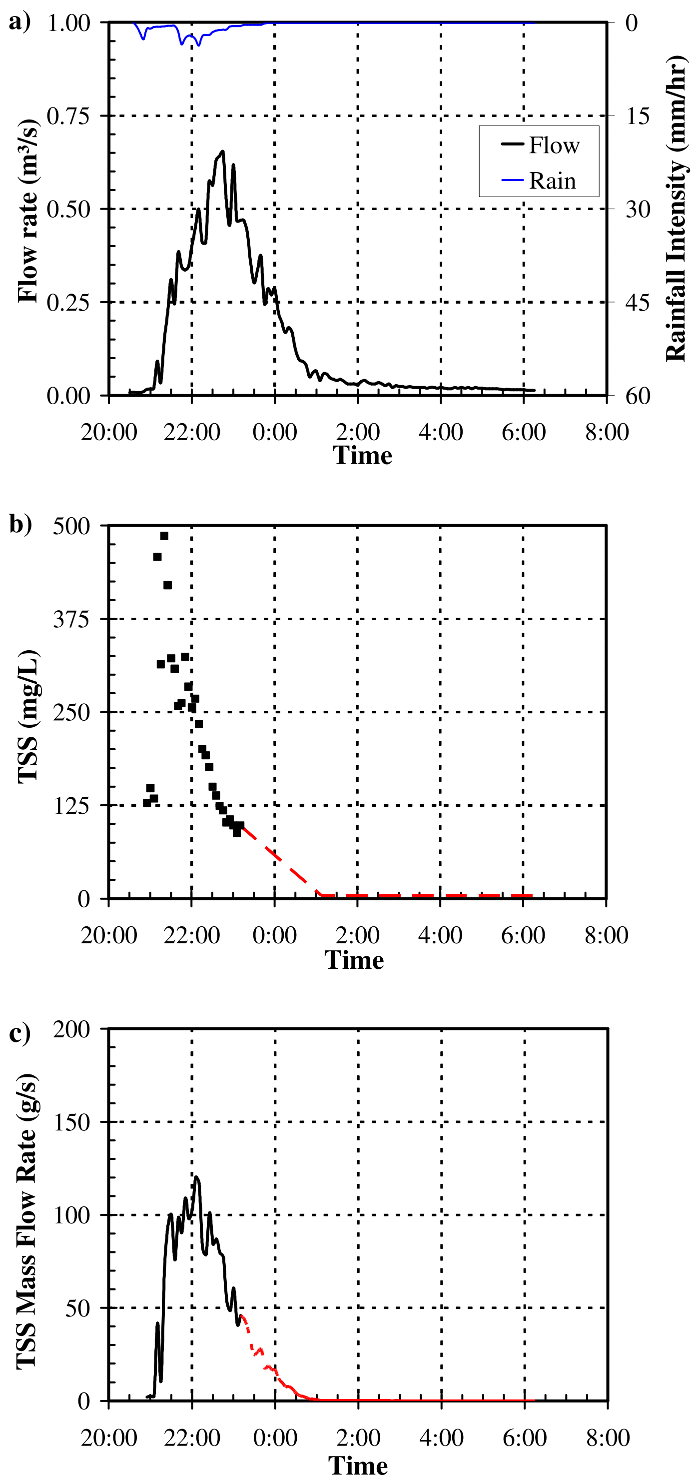

Figure G.8 – Taylor Street event beginning August 10, 2002.

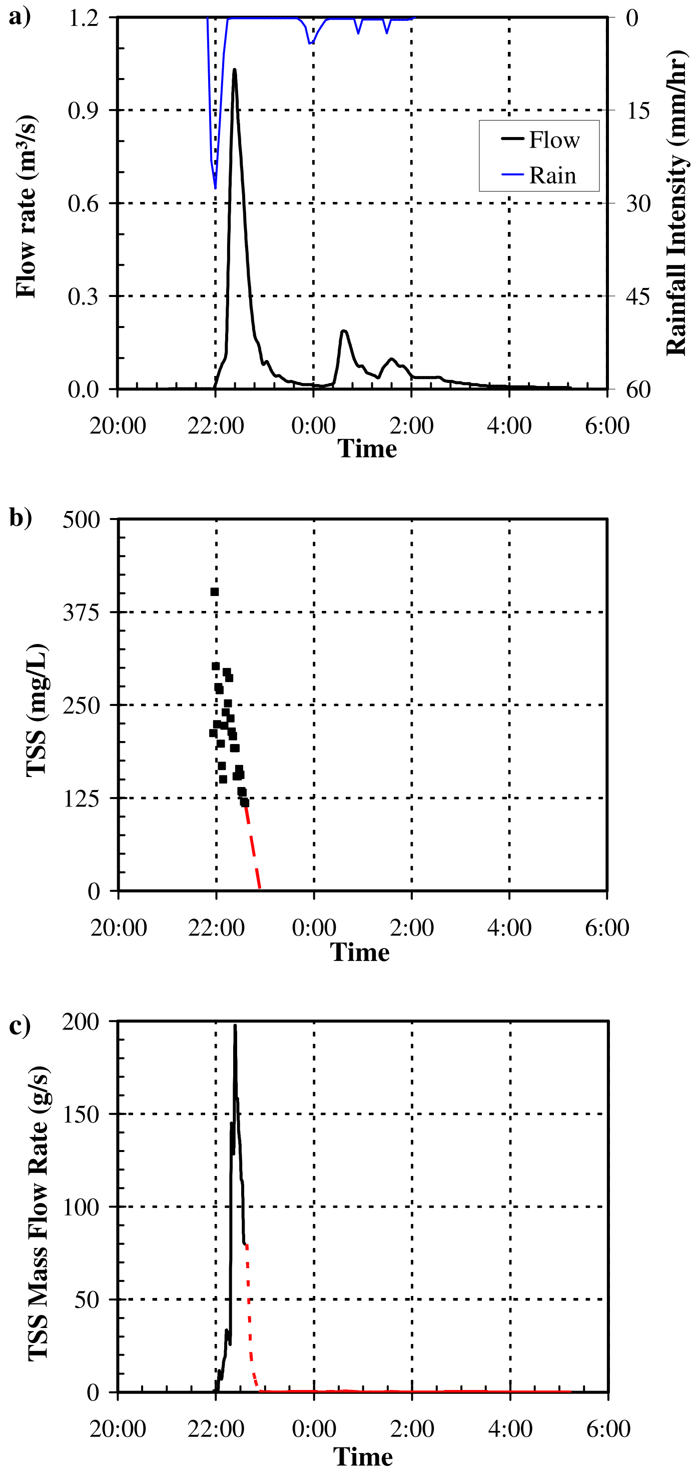

Figure G.9 – Silverwood July 25, 2001.

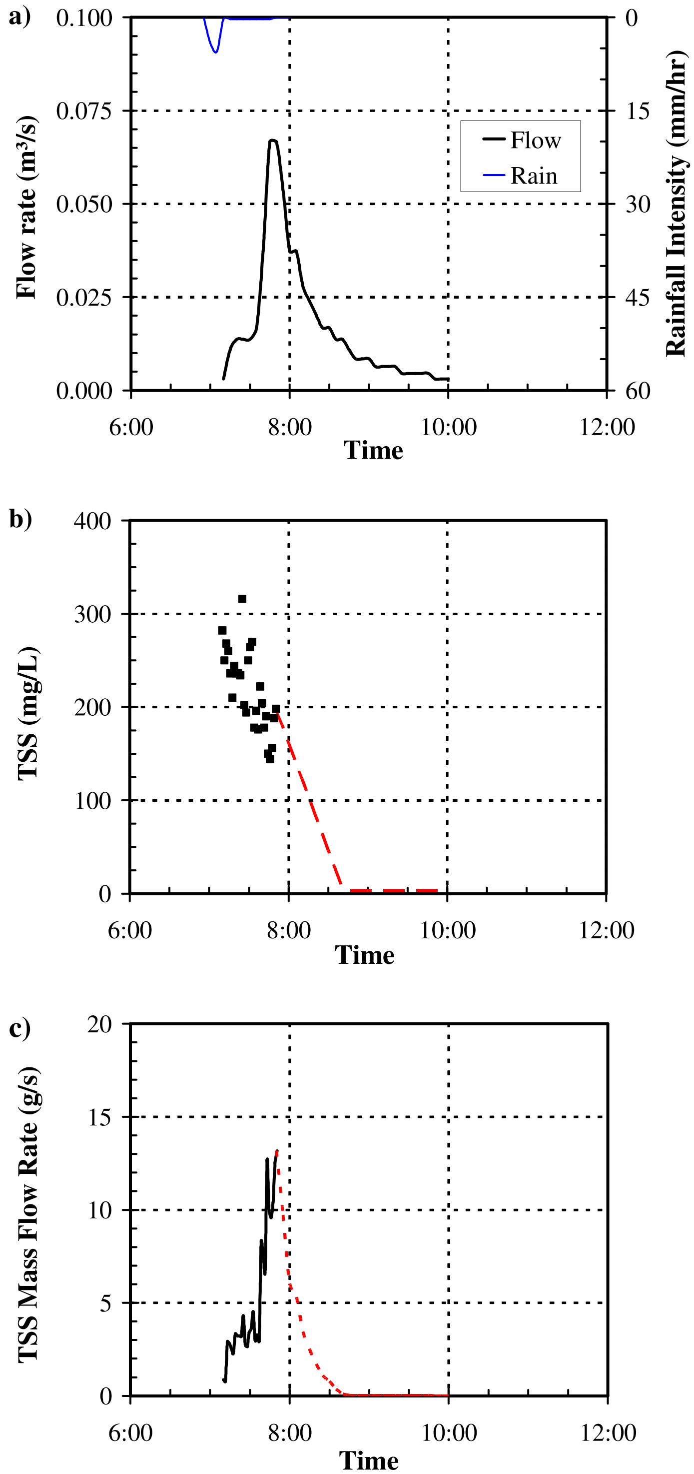

Figure G.10 – Silverwood August 8, 2001.

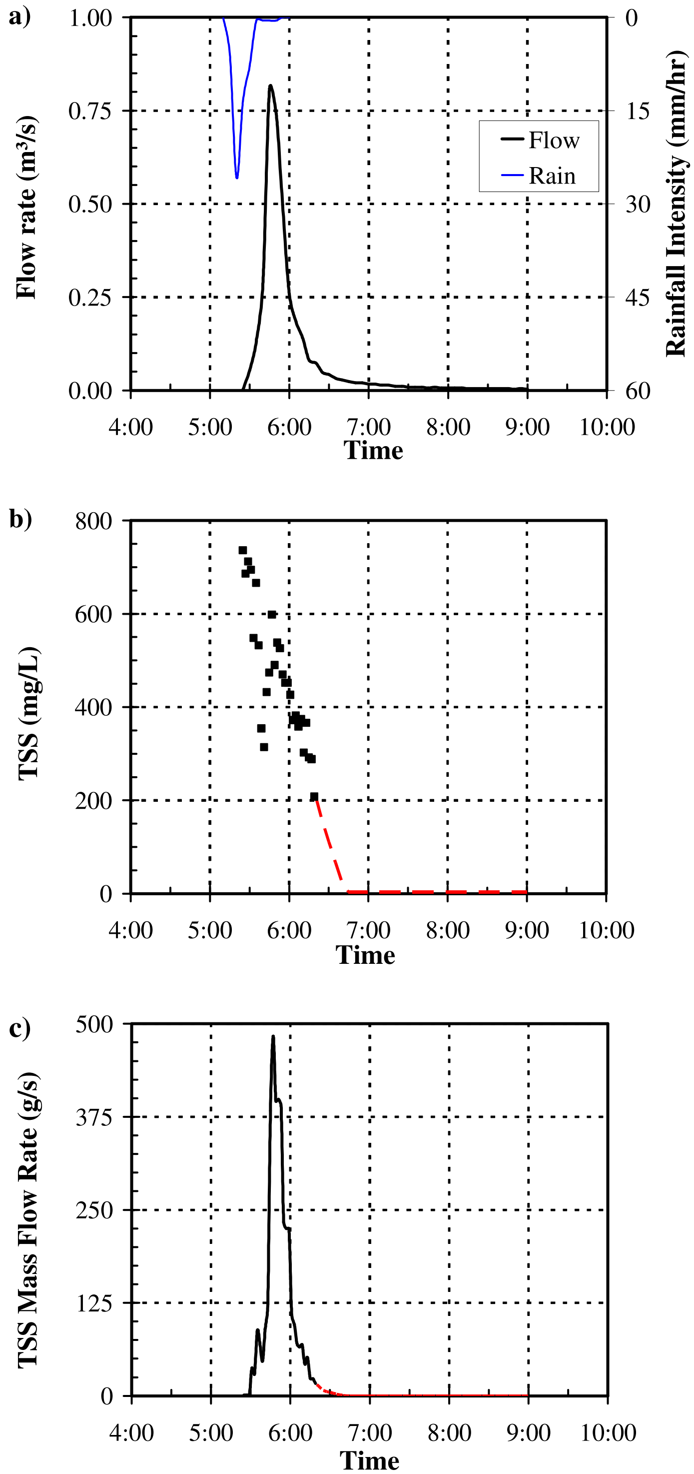

Figure G.11 – Silverwood August 14, 2001 – Event A.

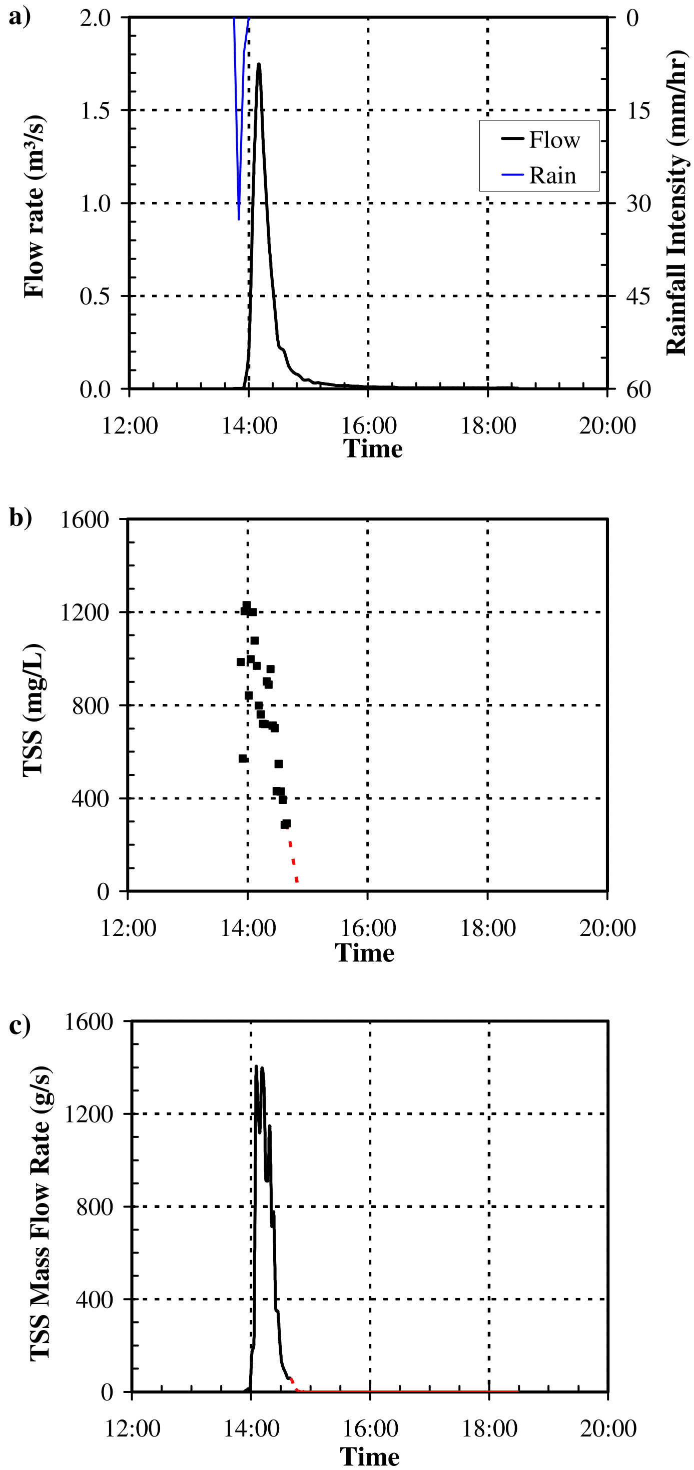

Figure G.12 – Silverwood August 14, 2001 – Event B.

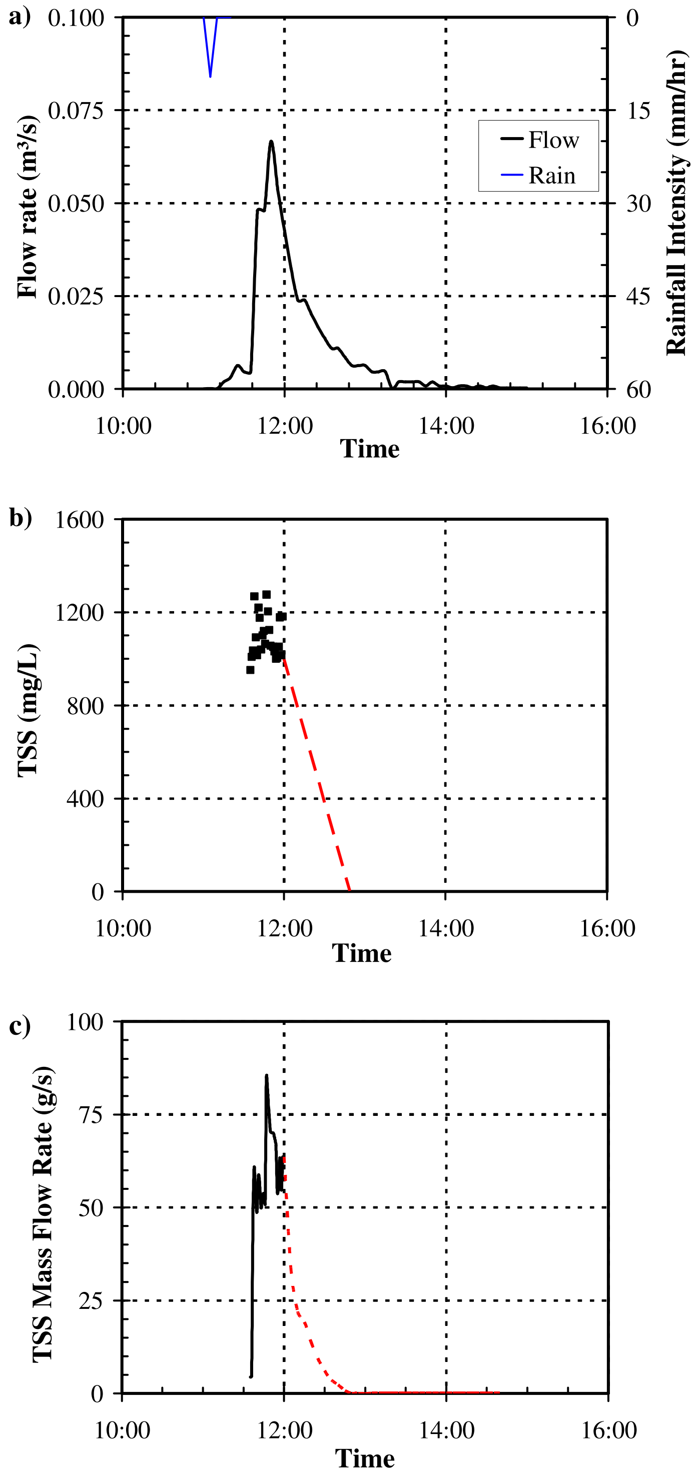

Figure G.13 – Silverwood June 6, 2002.

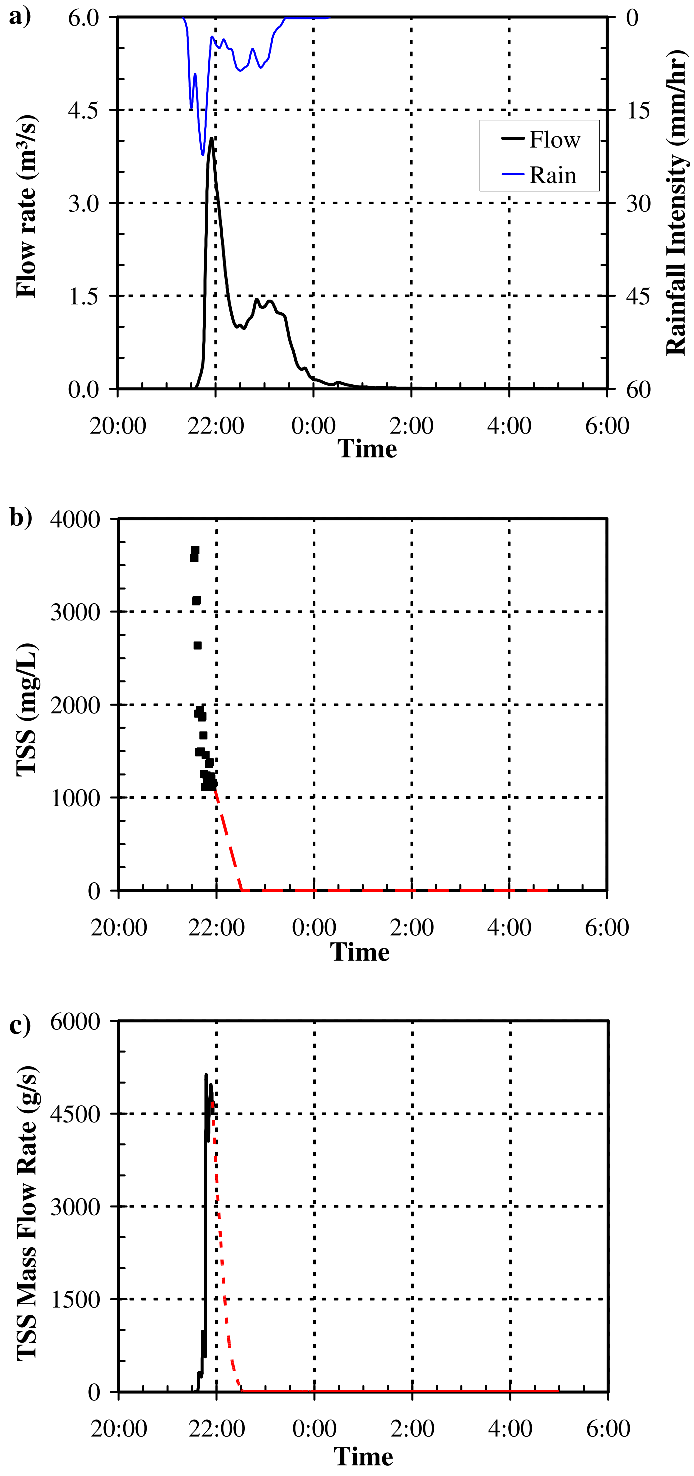

Figure G.14 – Silverwood June 17, 2002.

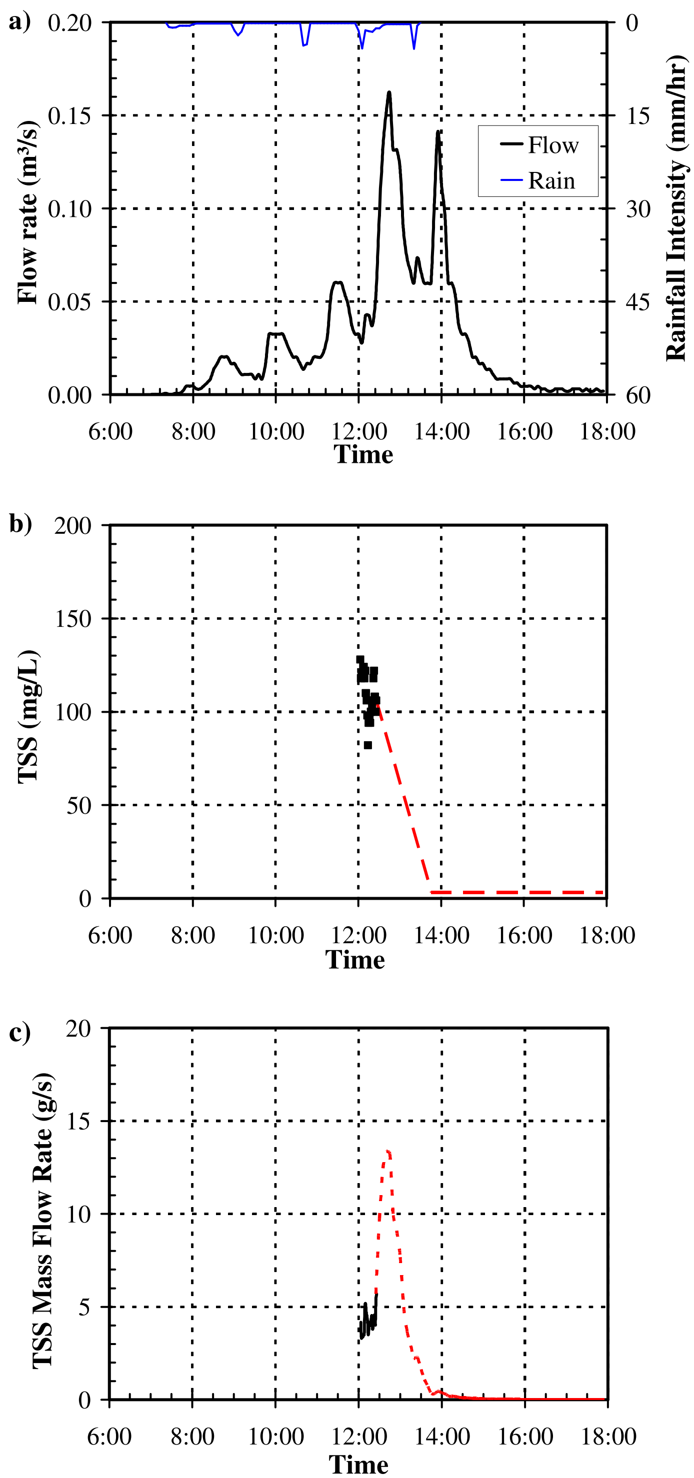

Figure G.15 – Silverwood June 24, 2002.

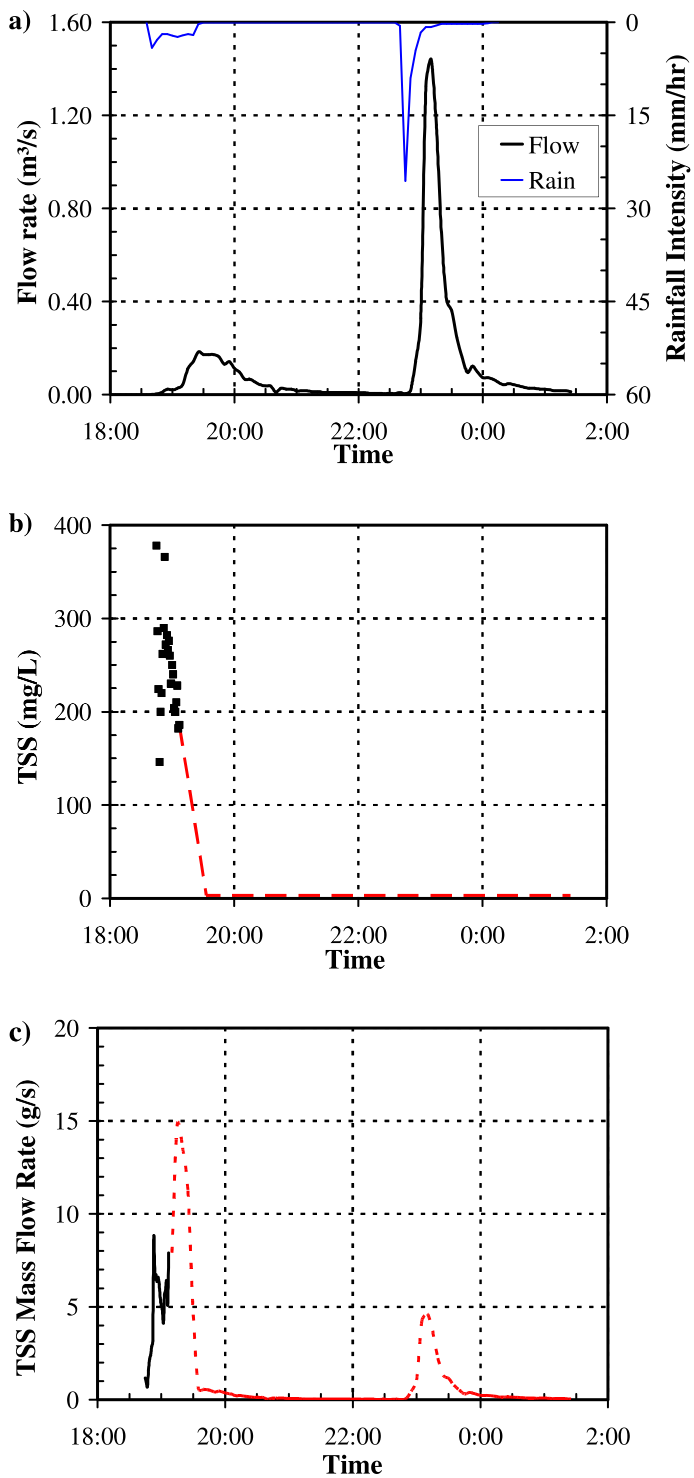

Figure G.16 – Silverwood June 29, 2002.

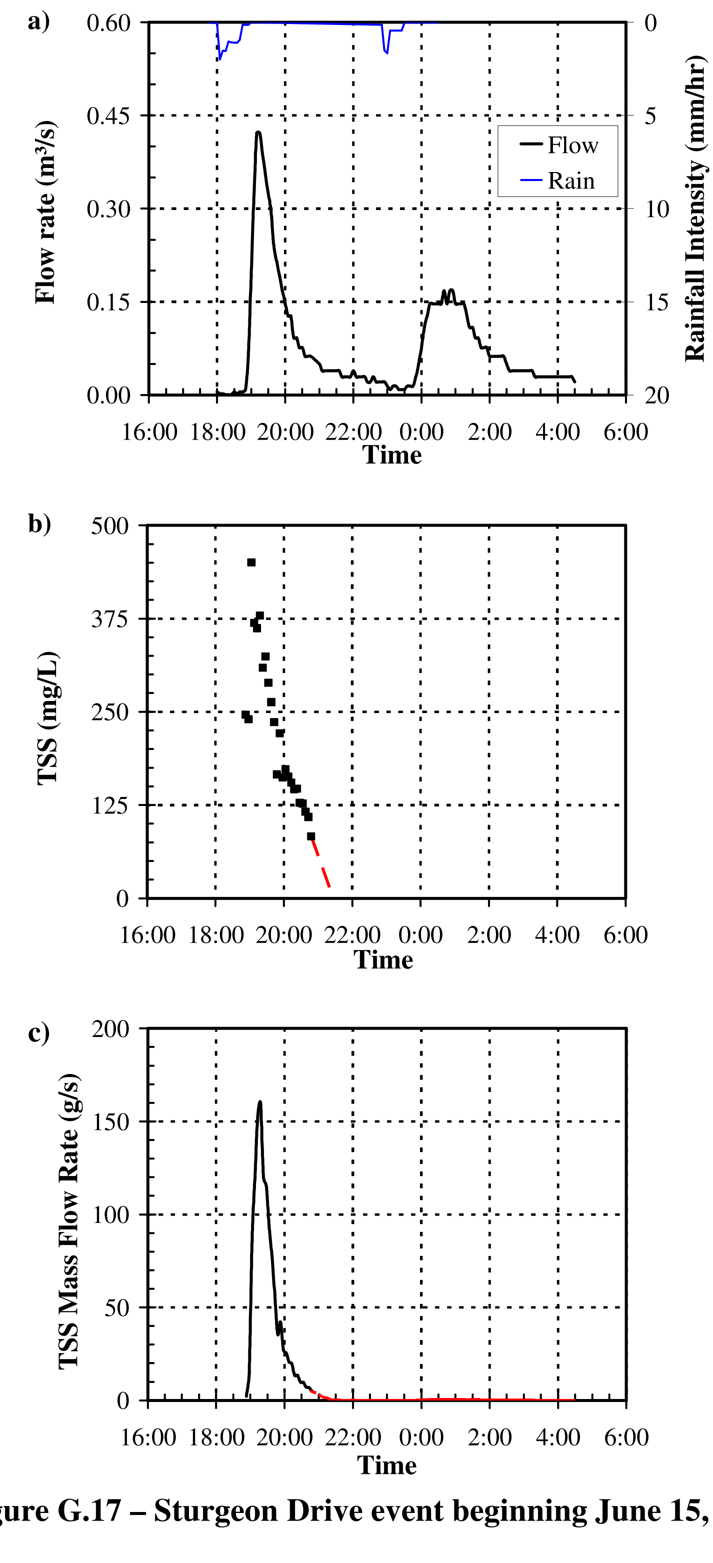

Figure G.17 – Sturgeon Drive event beginning June 15, 2001.

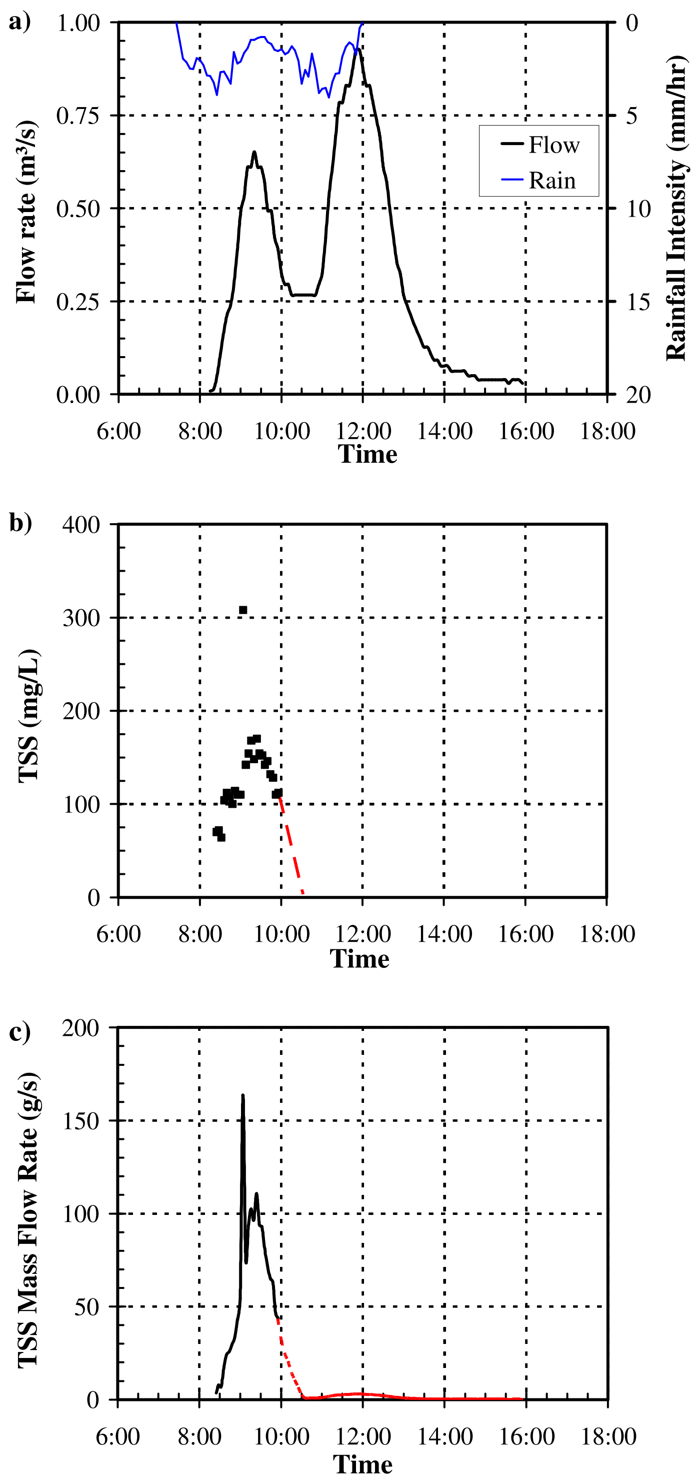

Figure G.18 – Sturgeon Drive event beginning June 17, 2001.

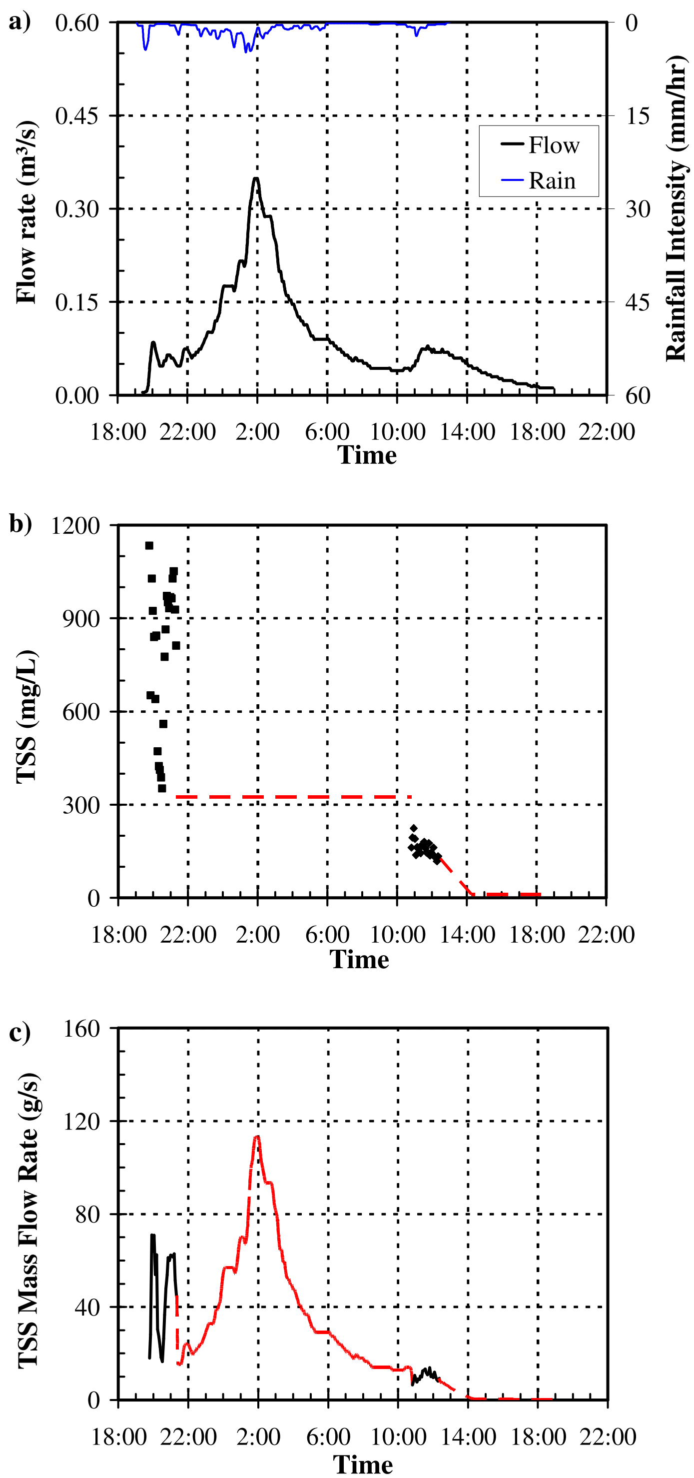

Figure G.19 – Avenue B event beginning June 10, 2002.

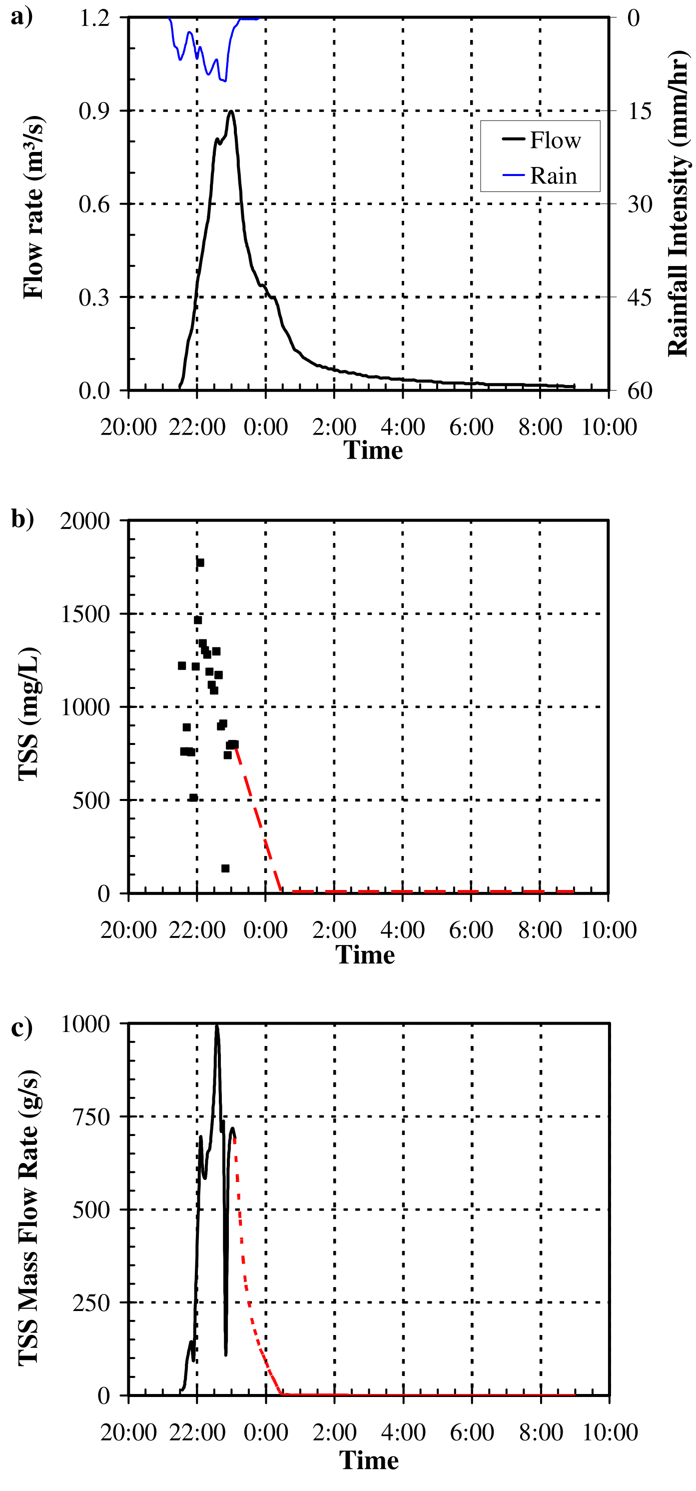

Figure G.20 – Avenue B event beginning June 17, 2002.

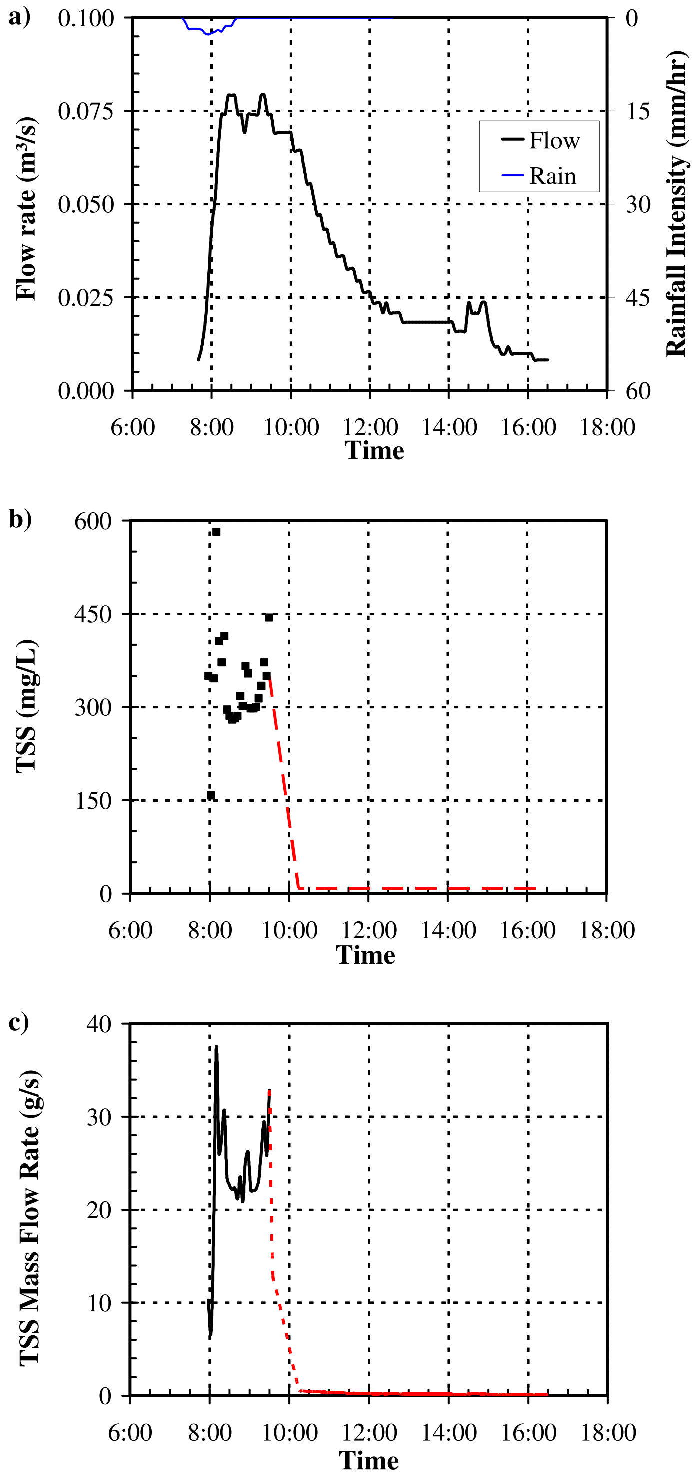

Figure G.21 – Avenue B event June 19, 2002.

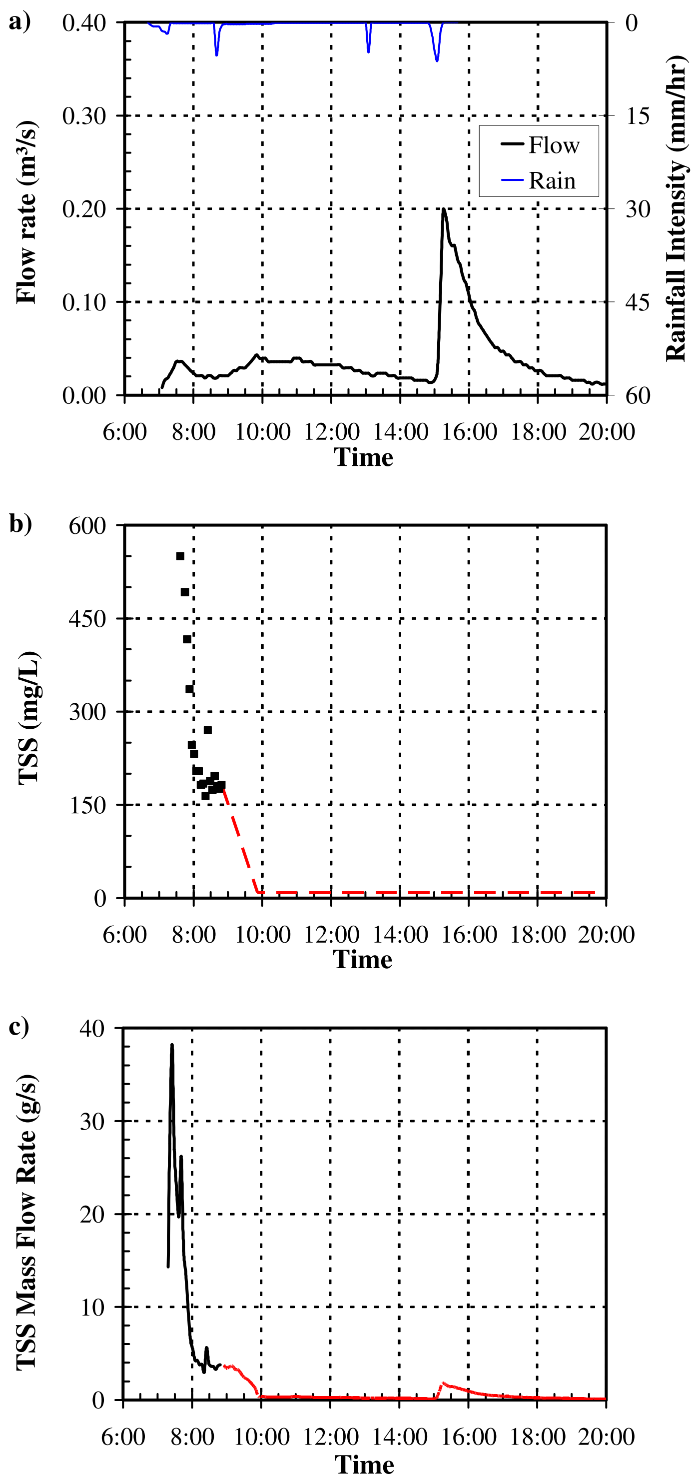

Figure G.22 – Avenue B event June 24, 2002.

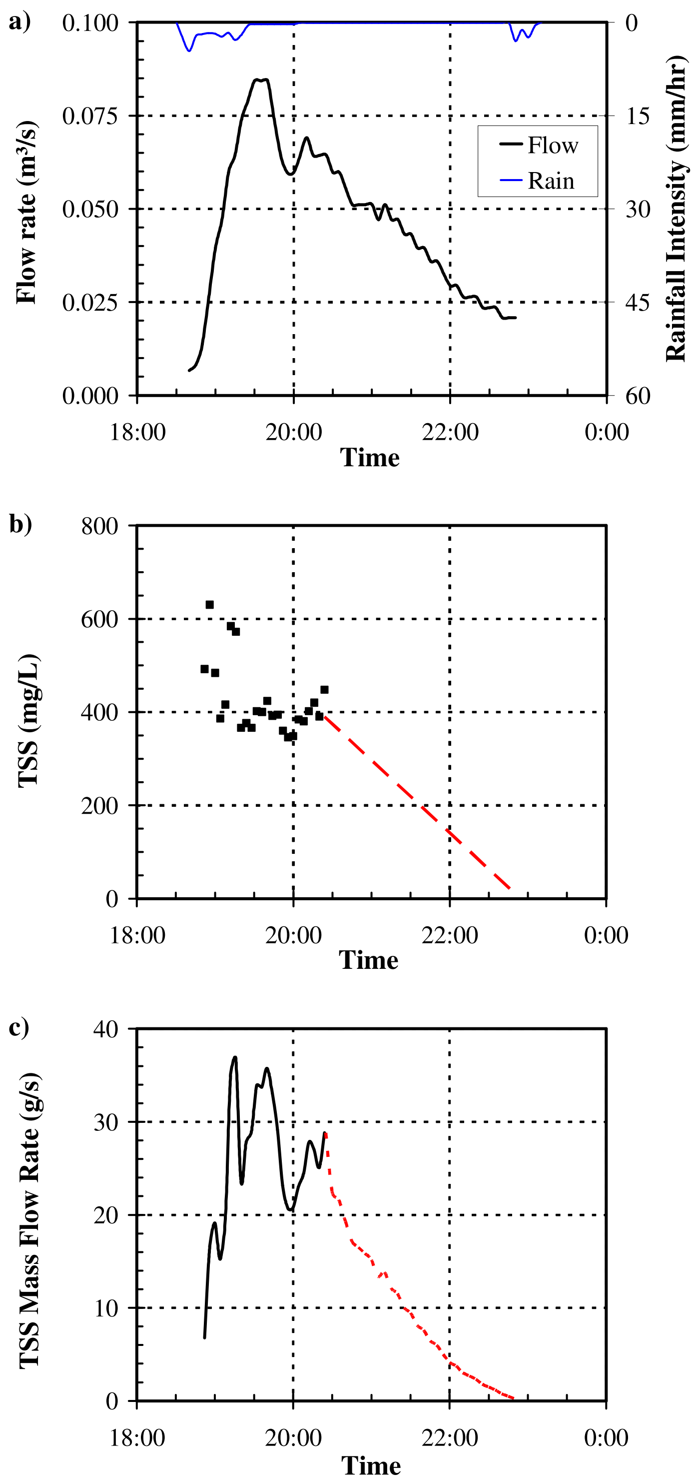

Figure G.23 – Avenue B event June 29, 2002.

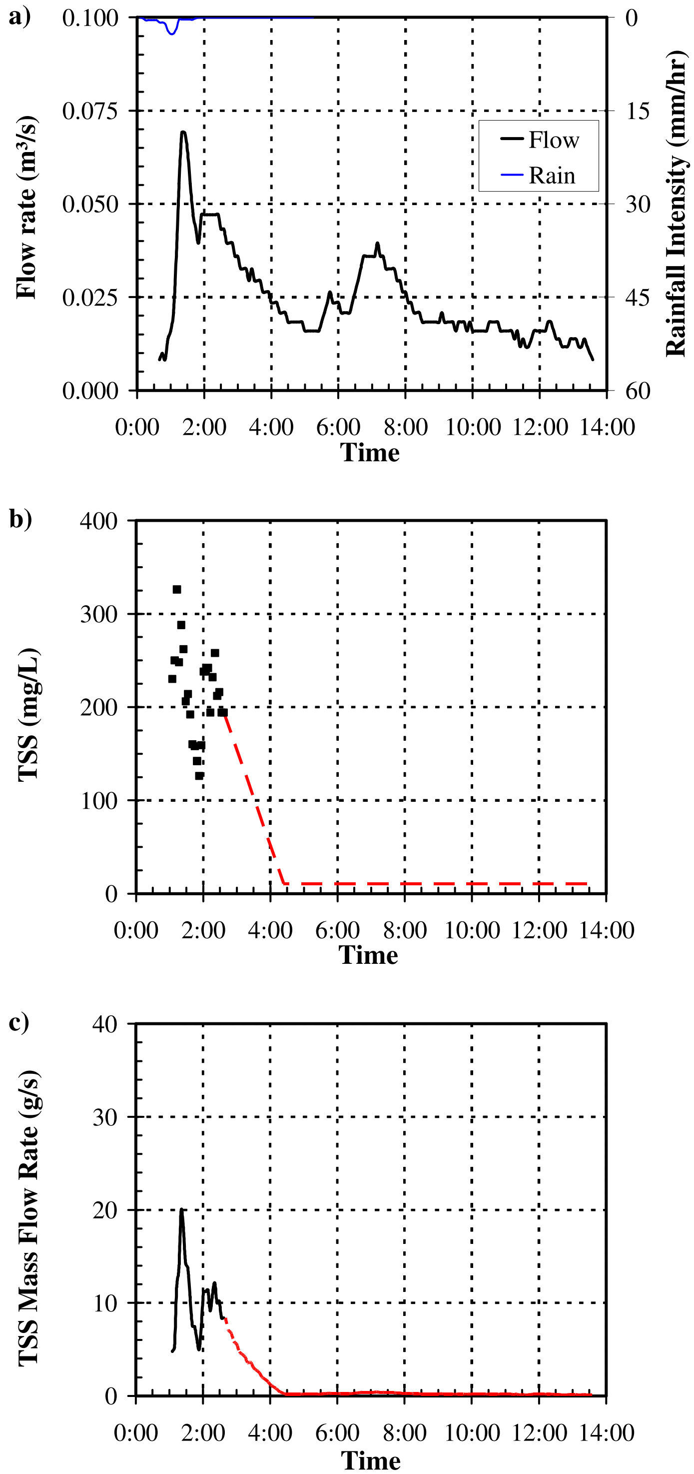

Figure G.24 – Avenue B event July 4, 2002.

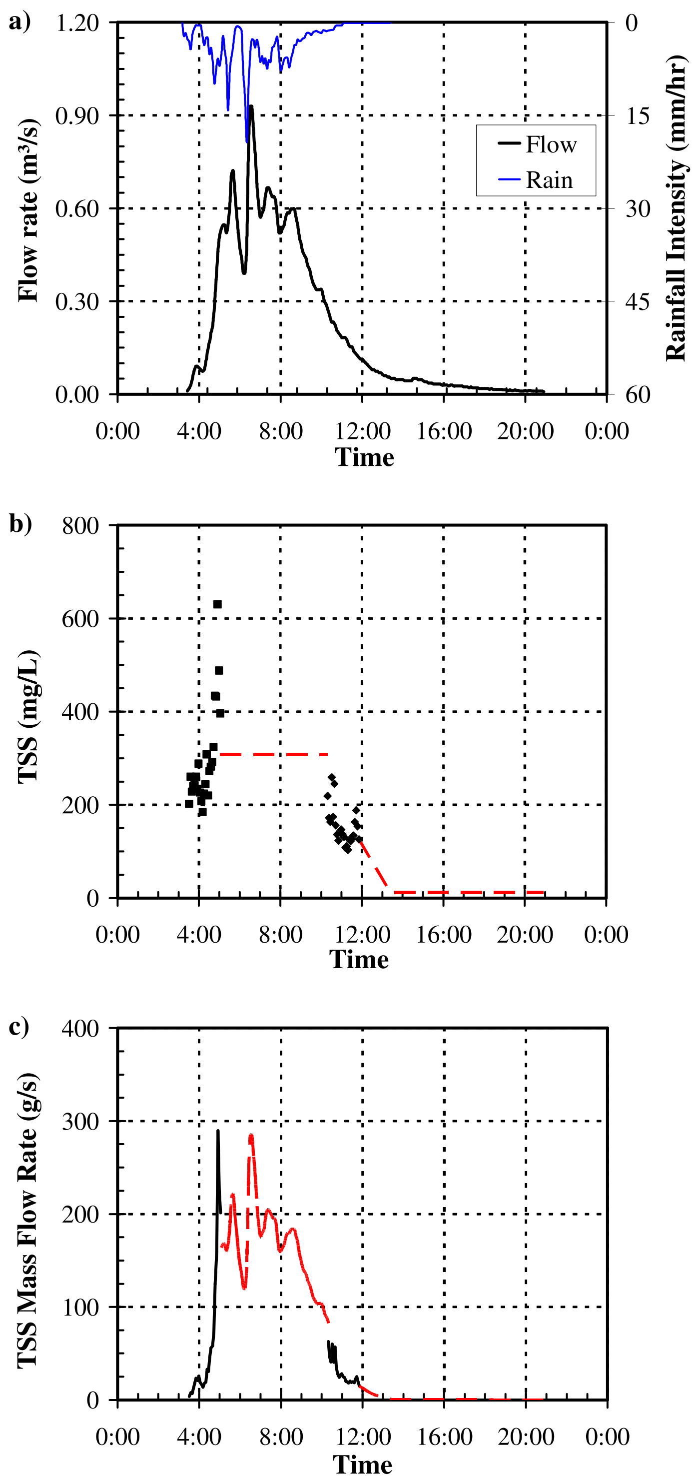

Figure G.25 – Avenue B event July 9, 2002.

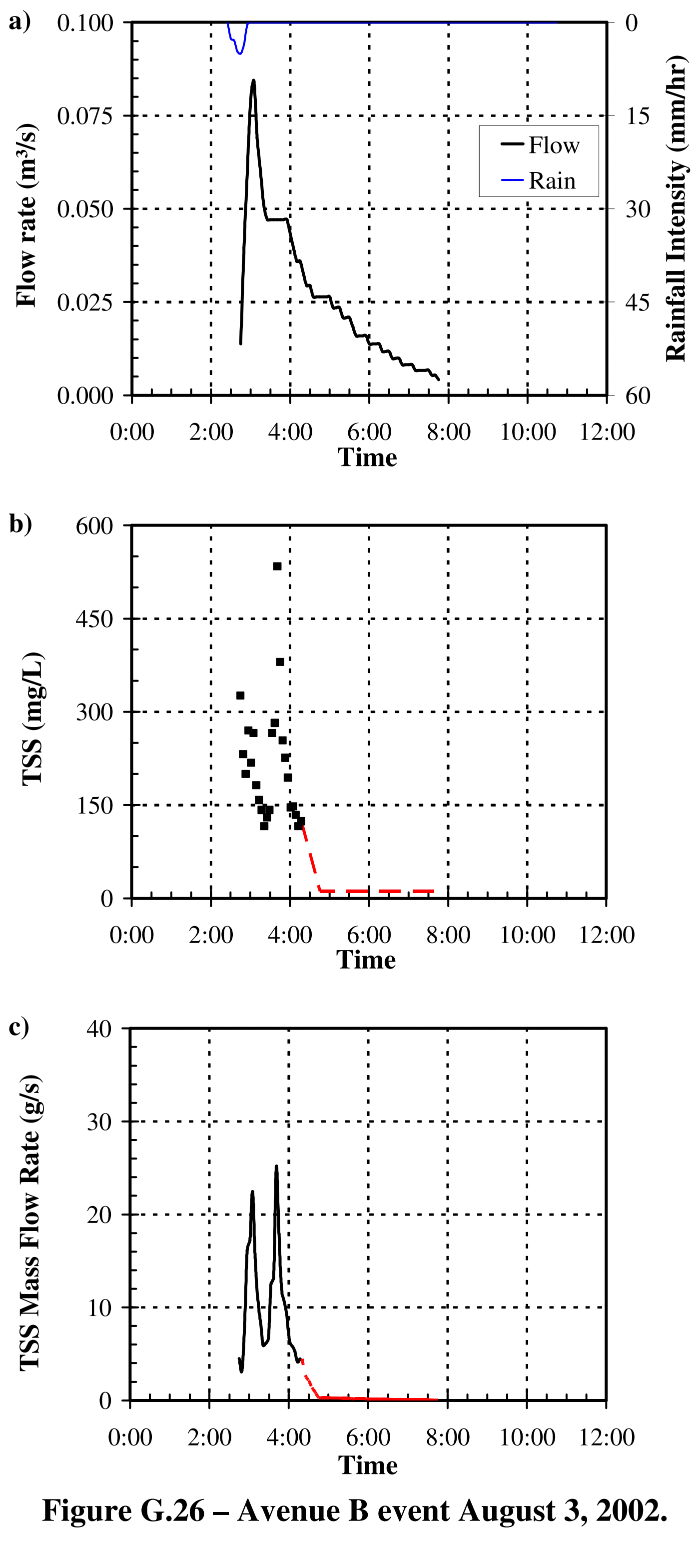

Figure G.26 – Avenue B event August 3, 2002.

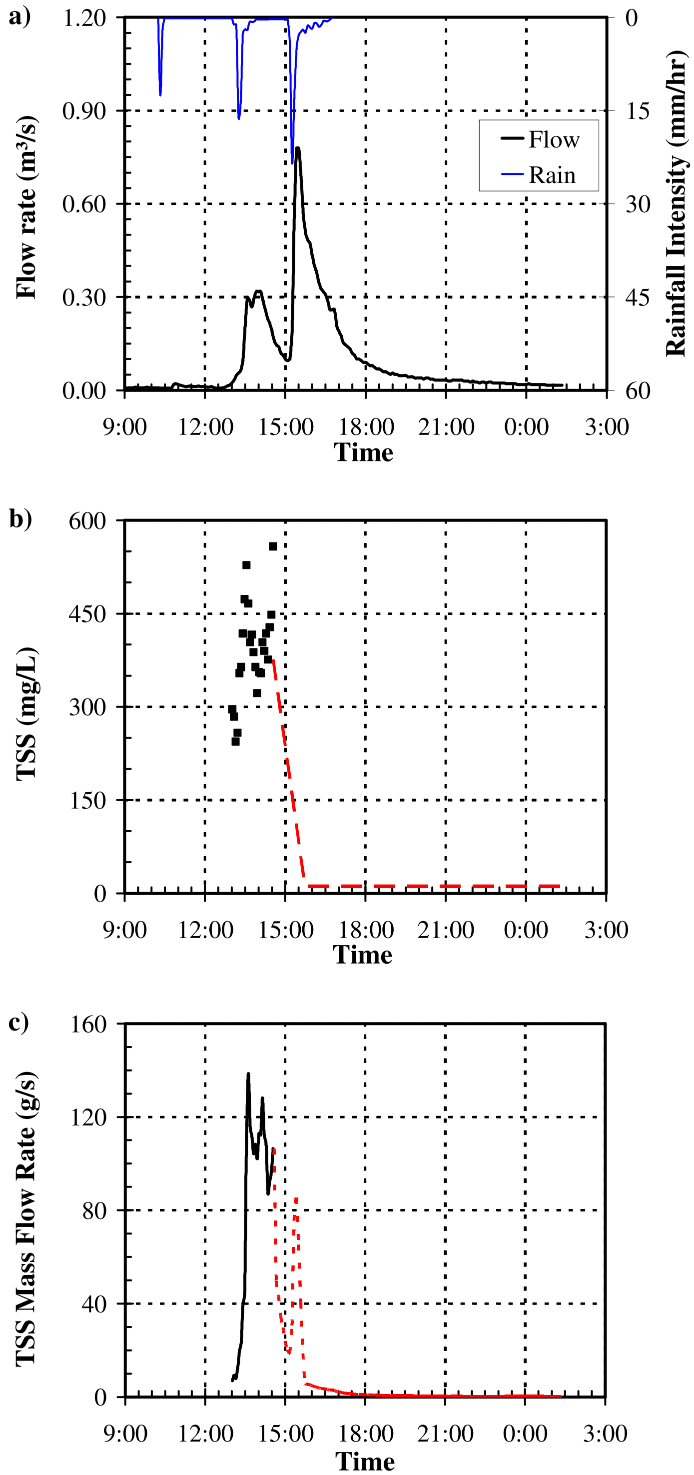

Figure G.27 – Avenue B event August 15, 2002.

APPENDIX H

Summary of Event Data

This appendix contains a summary of the event data including concentrations, loads, rainfall and runoff parameters for events that were used for load calculations. The rainfall data are processed to represent the specific catchment.

Table H.1 – Avenue B rainfall, runoff and relative proportion of event sampled data.

	10/06/2002	17/06/2002	19/06/2002	24/06/2002	29/06/2002	04/07/2002	09/07/2002	03/08/2002	15/08/2002
Total event volume (m³)	5,576	6,241	1,124	776	711	1,139	12,295	482	5471
Sampled volume (m³)	717	3,003	391	143	357	275	2,046	281	1,138
Max instantaneous intensity (mm/hr)	6.86	12.6	2.82	13.9	8.57	2.76	28.8	6.15	55.4
Rainfall depth (mm)	16.8	11.8	2.8	4.0	3.2	1.8	31.0	1.8	11.2
Duration (hh:mm)	17:41	2:31	5:08	8:25	4:28	5:01	10:09	0:24	6:20
Average intensity (mm/hr)	0.9	4.68	0.54	0.47	0.72	0.36	3.05	4.40	1.77
Max 5 min avg intensity (mm/hr)	4.8	10.21	2.70	6.16	4.62	2.69	19.34	5.02	23.34
Antecedant period (hrs)	93.7	23.55	31.55	52.57	123.42	97.12	118.04	60.57	85.72
Antecedant period (days)	3.9	1.0	1.3	2.2	5.1	4.0	4.9	2.5	3.6
Runoff C	0.444	0.707	0.537	0.259	0.297	0.846	0.530	0.846	0.653
Event Runoff depth (mm)	7.45	8.34	1.50	1.04	0.95	1.52	16.44	0.64	7.31
% of runoff sampled	12.9	48.1	34.7	18.4	50.3	24.2	16.6	58.4	20.8
% of total TSS mass sampled	17.7	78.7	79.1	74.6	70.1	78.0	15.5	93.1	49.3

Table H.2 – Taylor rainfall, runoff and relative proportion of event sampled data.

	28/06/2001	16/06/2001	14/08/2001 B	17/06/2002	19/06/2002	04/07/2002	09/07/2002	10/08/2002
Total event volume (m³)	1,753	3,845	1,569	11,965	1,629	701	33,268	5,013
Sampled volume (m³)	464	2,418	632	7,654	1,115	335	8,719	3,010
Max instantaneous intensity (mm/hr)	6.59	6.08	56.5	106.	1.55	1.52	28.8	4.83
Rainfall depth (mm)	0.66	4.6	2.2	8.2	2.7	1.50	31.0	4.4
Duration (hh:mm)	0:07	7:06	3:48	3:05	5:08	5:39	10:09	4:48
Average intensity (mm/hr)	3.3	0.8	30.0	39.2	0.5	0.3	3.1	0.9
Max 5 min avg intensity (mm/hr)	4.3	4.4	19.8	6.2	2.1	1.5	19.3	3.8
Antecedant period (hrs)	68.4	54.5	116.0	197.2	31.1	77.3	118.0	58.7
Antecedant period (days)	2.8	2.3	4.8	8.2	1.3	3.2	4.9	2.4
Runoff C	0.432	0.139	0.910	0.237	0.098	0.074	0.174	0.185
Event Runoff depth (mm)	0.28	0.62	0.25	1.94	0.26	0.11	5.40	0.81
% of runoff sampled	26.5	62.9	40.3	64.0	68.5	47.8	26.2	60.0
% of total TSS mass sampled	89.9	92.5	95.2	93.9	85.9	78.9	28.1	88.4

Table H.3 – Silverwood rainfall, runoff and relative proportion of event sampled data.

	25/07/2001	08/08/2001	14/08/2001 A	14/08/2001 B	06/06/2002	17/06/2002	24/06/2002	29/06/2002
Total event volume (m³)	2,179	179	1,099	2,076	142	11,745	1,165	2,880
Sampled volume (m³)	934	68	968	1,851	73	2,390	56	27
Max instantaneous intensity (mm/hr)	72.0	9.47	34.3	72.0	19.5	40.0	14.7	48.0
Rainfall depth (mm)	8.8	1.2	4.8	3.4	1.0	15.2	3.8	6.0
Duration (hh:mm)	4:06	0:48	0:37	0:06	0:03	2:51	6:00	5:26
Average intensity (mm/hr)	2.1	1.5	7.7	30.5	15.5	5.3	0.6	1.1
Max 5 min avg intensity (mm/hr)	27.6	5.5	25.8	32.7	9.6	22.1	4.3	25.6
Antecedant period (hrs)	16.6	243.3	141.4	7.9	691.3	23.8	24.8	125.3
Antecedant period (days)	0.7	10.1	5.9	0.3	28.8	1.0	1.0	5.2
Runoff C	0.103	0.062	0.095	0.253	0.059	0.320	0.127	0.199
Event Runoff depth (mm)	0.90	0.07	0.46	0.86	0.06	4.87	0.48	1.19
% of runoff sampled	42.9	37.7	88.1	89.2	51.7	20.3	4.8	0.9
% of total TSS mass sampled	89.4	62.2	98.7	99.5	76.9	56.8	18.9	17.3

Table H.4 – Sturgeon rainfall, runoff and relative proportion of event sampled data.

	15/06/2001	17/06/2001
Total event volume (m³)	2,955	9,103
Sampled volume (m³)	1,349	2,354
Max instantaneous intensity (mm/hr)	1.87	4.34
Rainfall depth (mm)	1.7	9.6
Duration (hh:mm)	6:40	4:23
Average intensity (mm/hr)	0.3	2.2
Max 5 min avg intensity (mm/hr)	2.0	4.0
Antecedant period (hrs)	132.4	32.0
Antecedant period (days)	5.5	1.3
Runoff C	0.412	0.229
Event Runoff depth (mm)	0.70	2.17
% of runoff sampled	45.7	25.
% of total TSS mass sampled	97.8	86.3

Table H.5 – Avenue B water quality parameter EMC's and load data.

	10-Jun-02		17-Jun-02		19-Jun-02		24-Jun-02		29-Jun-02	
	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)
Sulphate	85.1	475	20.2	126	54.2	60.9	44.2	34.3	40.7	57.3
Bicarbonate	103	576	57.7	360	41.7	46.9	39.2	30.4	52.2	73.4
Total Alkalinity (as $CaCO_3$)	84.6	472	47.2	294	34.0	38.3	32.1	24.9	42.9	60.4
Sodium	45.6	254	9.14	57.1	11.8	13.3	15.3	11.9	14.7	20.6
Magnesium	14.9	83	3.37	21.0	7.64	8.59	7.00	5.43	7.15	10.1
Calcium	36.0	201	15.9	99.1	20.5	23.0	13.8	10.7	17.2	24.1
Total Hardness (calc – as $CaCO_3$)	151	843	53.4	333	82.7	93.0	63.3	49.1	72.2	101.6
Chloride	56.2	313	10.1	63.1	11.5	12.9	14.4	11.1	15.4	21.6
Potassium	7.69	43	2.89	18.0	2.08	2.34	2.39	1.86	3.57	5.03
Biochemical Oxygen Demand (as O_2)	34.7	194	15.4	96.1	2.88	3.24	7.36	5.71	22.2	31.2
DOC	37.6	209	12.0	75.1	7.99	8.98	12.2	9.42	24.3	34.2
Preserved Ammonia (as N)	1.21	6.77	0.25	1.56	0.10	0.11	0.41	0.32		
Nitrate (as N)	0.52	2.90	0.03	0.21	0.27	0.31	0.0074	0.0057	0.00715	0.010
Total Kjeldahl Nitrogen (as N)	7.61	42	7.27	45.4	1.01	1.13	1.57	1.21	3.50	4.93
Total Phosphorous (as P)	1.72	10	0.98	6.10	0.29	0.33	0.28	0.22	0.75	1.06
Ortho Phosphorous (as P)			0.096	0.60			0.083	0.06	0.16	0.231
Chemical Oxygen Demand (as O_2)	288	1,610	187	1,170	34.7	39.1	71.4	55.4	118	166
Total Dissolved Solids	353	1,970	119	745	45.7	50.8	136	106	151	212
Discrete TSS	331	1,850	599	3,740	158	178	120.	92.7	224	316
Total Coliform (orgs/100ml)			7.47E+06	4.66E+09	1.70E+06	1.91E+08	3.51E+05	2.72E+07		
Fecal Coliform (orgs/100ml)			7.60E+04	4.75E+07	6.01E+04	6.76E+06	1.16E+04	9.00E+05		

Table H.5 con't – Avenue B water quality parameter concentration and load data.

	04-Jul-02		09-Jul-02		03-Aug-02		15-Aug-02	
	EMC (mg/L)	**Load (kg)**	**EMC (mg/L)**	**Load (kg)**	**EMC (mg/L)**	**Load (kg)**	**EMC (mg/L)**	**Load (kg)**
Sulphate	39.4	44.9	44.6	548	61.3	29.5	3.70	20.3
Bicarbonate	33.4	38.0	76.1	936	52.6	25.3	2.81	15.4
Total Alkalinity (as $CaCO_3$)	27.4	31.2	62.3	766	43.2	20.8	2.30	12.6
Sodium	10.6	12.1	16.0	197	18.7	9.00	0.86	4.71
Magnesium	6.29	7.2	8.01	98.5	10.5	5.07	0.57	3.14
Calcium	13.8	15.7	26	320	23.4	11.3	1.23	6.75
Total Hardness (calc – as $CaCO_3$)	60.2	68.6	97.6	1,200	102	49.0	5.45	29.8
Chloride	10.6	12.1	19	228	25.7	12.4	0.77	4.24
Potassium	1.93	2.2	4.50	55.4	4.67	2.25	0.11	0.63
Biochemical Oxygen Demand (as O_2)	10.6	12.1	19.6	240	6.66	3.21	0.36	1.98
DOC	14.0	16.0	30.1	370	13.4	6.47	0.60	3.30
Preserved Ammonia (as N)	0.68	0.78	0.95	11.7				
Nitrate (as N)	0.24	0.27	0.42	5.22	0.58	0.28	0.029	0.16
Total Kjeldahl Nitrogen (as N)	2.15	2.5	4.26	52.4	1.99	0.96	0.13	0.72
Total Phosphorous (as P)	0.52	0.59	0.96	11.8	0.45	0.22	0.023	0.13
Ortho Phosphorous (as P)	0.068	0.077	0.115	1.41	0.064	0.031	0.0014	0.0079
Chemical Oxygen Demand (as O_2)	79.8	91	137	1,690	66.6	32.1	5.77	31.6
Total Dissolved Solids	116	132	194	2,390	197	94.8	10.1	55.1
Discrete TSS	75.0	85	300	3,690	135	64.8	170	930
Total Coliform (orgs/100ml)	6.19E+04	7.05E+06	1.80E+06	2.21E+09				
Fecal Coliform (orgs/100ml)	2.64E+04	3.00E+06	2.07E+04	2.54E+07				

Table H.6 – Taylor water quality parameter concentration and load data.

	28-Jun-01		16-Jul-01		14-Aug-01		17-Jun-02		19-Jun-02	
	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)
Sulphate	92.3	162	82.4	317	137	214	76.1	911	77.4	126
Bicarbonate	57.4	101	73.6	283	89.5	140	88.9	1060	77.4	126
Total Alkalinity (as $CaCO_3$)	47.1	82.6	60.4	232	73.3	115	72.9	873	63.4	103
Sodium	14.3	25.0	15.1	58.0	23.8	37.3	18.6	222	24.6	40.1
Magnesium	15.1	26.4	13.8	53.2	22.6	35.4	12.2	145	13.0	21.2
Calcium	32.3	56.6	35.2	135	44.7	70.2	34.5	413	28.1	45.7
Total Hardness (calc – as $CaCO_3$)	143	250	145	556	205	321	136	1630	124	202
Chloride	15.1	26.4	19.5	75.0	25.4	39.8	19.8	237	25.3	41.3
Potassium	1.85	3.25	2.52	9.67	3.22	5.06	4.48	53.6	4.11	6.69
Biochemical Oxygen Demand (as O_2)			8.81	33.9	8.46	13.3			15.4	25.1
DOC	3.47	6.08	18.9	72.5	16.5	25.9	14.1	168	15.1	24.5
Preserved Ammonia (as N)			0.094	0.36	0.089	0.14	0.18	2.14	0.52	0.85
Nitrate (as N)			0.62	2.39	0.58	0.90	0.019	0.23	0.55	0.90
Total Kjeldahl Nitrogen (as N)			2.45	9.4	1.77	2.78	3.39	40.6	2.46	4.01
Total Phosphorous (as P)			0.46	1.77	0.55	0.86	1.02	12.2	0.75	1.22
Ortho Phosphorous (as P)			0.094	0.36			0.102	1.22		
Chemical Oxygen Demand (as O_2)	43.6	76.5	174	670	106	167	230	2750	155	252
Total Dissolved Solids	228	400	242	931	346	543	255	3050	250	407
Discrete TSS	71.9	126	434	1,670	325	511	464	5550	166	270
Total Coliform (orgs/100ml)	3.97E+03	6.96E+05	2.89E+05	1.11E+08	4.43E+05	6.96E+07	7.70E+06	9.21E+09	4.20E+06	6.84E+08
Fecal Coliform (orgs/100ml)	2.38E+01	4.17E+03	5.85E+03	2.25E+06	8.06E+02	1.26E+05	5.18E+04	6.20E+07	1.36E+04	2.21E+06

Table H.6 con't – Taylor water quality parameter concentration and load data.

	04-Jul-02		09-Jul-02		10-Aug-02	
	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)
Sulphate	133	93.1	11.1	368	84.1	421
Bicarbonate	88.0	61.6	37.3	1,240	66.0	331
Total Alkalinity (as $CaCO_3$)	72.2	50.6	30.4	1,010	54.0	271
Sodium	23.4	16.4	3.43	114	16.8	84.3
Magnesium	19.6	13.7	2.13	71.0	13.2	66.2
Calcium	46.4	32.5	10.2	340	32.4	163
Total Hardness (calc – as $CaCO_3$)	196	138	34.1	1,130	135	677
Chloride	22.5	15.7	2.94	97.8	19.8	99.3
Potassium	3.35	2.35	2.13	71.0	2.40	12.0
Biochemical Oxygen Demand (as O_2)	18.2	12.7	5.87	195	6.30	31.6
DOC	21.5	15.1	7.31	243	12.0	60.2
Preserved Ammonia (as N)	0.62	0.43	0.14	4.59		
Nitrate (as N)	0.79	0.56	0.17	5.57	0.74	3.70
Total Kjeldahl Nitrogen (as N)	3.11	2.17	2.59	86.3	1.98	9.93
Total Phosphorous (as P)	0.54	0.38	0.37	12.2	0.30	1.50
Ortho Phosphorous (as P)	0.10	0.070	0.097	3.21	0.066	0.33
Chemical Oxygen Demand (as O_2)	135	94.5	36.5	1210	78.1	391
Total Dissolved Solids	336	236	68.1	2270	235	1,180
Discrete TSS	96.7	67.8	155	5150	134	670
Total Coliform (orgs/100ml)	2.61E+05	1.83E+07	4.60E+05	1.53E+09		
Fecal Coliform (orgs/100ml)	4.54E+04	3.18E+06	1.57E+03	5.22E+06		

Table H.7 – Silverwood water quality parameter concentration and load data.

	25-Jun-01		08-Aug-01		14/08/2001 A		14/08/2001 B		06-Jun-02	
	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)
Sulphate	6.00	13.1	34.7	6.22	20.3	22.3			52.2	7.42
Bicarbonate	6.30	13.7	206	37.0	43.2	47.4	43.7	90.7	88.4	12.6
Total Alkalinity (as $CaCO_3$)	5.12	11.1	168	30.2	35.3	38.7	35.7	74.0	72.4	10.3
Sodium	1.29	2.80	6.04	1.08	4.41	4.84	1.78	3.70	23.3	3.31
Magnesium	0.86	1.87	5.28	0.95	2.64	2.9	2.67	5.55	7.24	1.03
Calcium	4.29	9.34	17.4	3.11	14.1	15.5	9.81	20.4	26.4	3.75
Total Hardness (calc – as $CaCO_3$)	14.1	30.8	65.3	11.7	45.8	50.3	35.7	74.0	95.6	13.6
Chloride	1.71	3.74	3.77	0.68	3.53	3.87	5.35	11.1	20.7	2.94
Potassium	0.43	0.93	2.26	0.41	2.64	2.90	1.78	3.70	5.17	0.735
Biochemical Oxygen Demand (as O_2)			11.7	2.09	13.2	14.5	4.01	8.33		
DOC			37.0	6.62	29.1	32.0	8.92	18.5	48.6	6.91
Preserved Ammonia (as N)			0.31	0.055	0.67	0.74	0.080	0.17	0.90	0.13
Nitrate (as N)			0.57	0.10	0.57	0.63	0.22	0.46	0.46	0.065
Total Kjeldahl Nitrogen (as N)			0.27	0.049	3.35	3.68	1.07	2.22	10.6	1.51
Total Phosphorous (as P)			0.27	0.049	0.76	0.83	0.90	1.87	0.90	0.13
Ortho Phosphorous (as P)					0.10	0.11			0.18	0.025
Chemical Oxygen Demand (as O_2)			187	33.5	166	182	89.2	185	379	53.8
Total Dissolved Solids	28.3	61.6	106	19.1	90.8	99.7	65.1	135	223	31.7
Discrete TSS	82.8	180	118	21.2	419	461	738	1,530	780	111
Total Coliform (orgs/100ml)			9.05E+07	1.62E+07	2.12E+07	2.32E+07	1.12E+07	2.32E+07	1.04E+08	1.48E+07
Fecal Coliform (orgs/100ml)			8.68E+05	1.55E+05	2.28E+06	2.51E+06	1.21E+06	2.51E+06	2.25E+05	3.20E+04

Table H.7 con't – Silverwood water quality parameter concentration and load data.

	17-Jun-02		24-Jun-02		29-Jun-02	
	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)
Sulphate	3.87	45.4	0.77	0.90	0.806	2.32
Bicarbonate	27.9	327	2.50	2.92	0.843	2.43
Total Alkalinity (as $CaCO_3$)	22.8	268	2.07	2.41	0.693	2.00
Sodium	1.63	19.1	0.39	0.45	0.178	0.51
Magnesium	0.81	9.56	0.14	0.17	0.103	0.30
Calcium	6.71	78.9	0.77	0.90	0.375	1.08
Total Hardness (calc – as $CaCO_3$)	20.1	237	2.50	2.92	1.36	3.91
Chloride	1.22	14.3	0.29	0.34	0.23	0.65
Potassium	0.61	7.17	0.14	0.17	0.066	0.19
Biochemical Oxygen Demand (as O_2)	3.91	45.9	0.419	0.488	0.234	0.675
DOC	3.66	43.0	1.11	1.29	0.506	1.46
Preserved Ammonia (as N)	0.118	1.39	0.0188	0.022		
Nitrate (as N)	0.0834	0.98	0.0385	0.045	0.0064	0.018
Total Kjeldahl Nitrogen (as N)	1.59	18.6	0.10	0.12	0.068	0.197
Total Phosphorous (as P)	0.415	4.87	0.0193	0.022	0.013	0.036
Ortho Phosphorous (as P)	0.022	0.263	0.0058	0.0067	0.0069	0.020
Chemical Oxygen Demand (as O_2)	104	1,220	5.29	6.17	2.64	7.61
Total Dissolved Solids	42.7	502	5.01	5.83	2.60	7.47
Discrete TSS	454	5,330	27.6	32.2	12.8	37.0
Total Coliform (orgs/100ml)	1.81E+04	2.13E+05	4.62E+05	5.38E+05		
Fecal Coliform (orgs/100ml)	6.33E+01	7.43E+02	5.45E+03	6.35E+03		

Table H.8 – Sturgeon water quality parameter concentration and load data.

	15-Jun-01		17-Jun-01	
	EMC (mg/L)	Load (kg)	EMC (mg/L)	Load (kg)
Sulphate				
Bicarbonate				
Total Alkalinity (as $CaCO_3$)				
Sodium				
Magnesium				
Calcium				
Total Hardness (calc – as $CaCO_3$)				
Chloride				
Potassium				
Biochemical Oxygen Demand (as O_2)				
DOC				
Preserved Ammonia (as N)				
Nitrate (as N)				
Total Kjeldahl Nitrogen (as N)				
Total Phosphorous (as P)				
Ortho Phosphorous (as P)				
Chemical Oxygen Demand (as O_2)				
Total Dissolved Solids				
Discrete TSS	128	379	44.2	402
Total Coliform (orgs/100ml)				
Fecal Coliform (orgs/100ml)				

APPENDIX I

Heavy Metal Concentration Results

This appendix contains a complete summary of the heavy metal analyses conducted.

Table I.1 – June 6, 2001 mid – event grab samples.

Metal	Units	Sturgeon Drive	Silverwood	Taylor Street
Mercury	mg/L	<0.00005	<0.00005	<0.00005
Aluminum	mg/L	14	2.3	10
Antimony	mg/L	0.001	<0.001	0.001
Arsenic	mg/L	0.004	<0.002	0.004
Barium	mg/L	0.21	0.035	0.097
Berrylium	mg/L	<0.001	<0.001	<0.001
Bismuth	mg/L	<0.001	<0.001	<0.001
Boron	mg/L	0.054	0.015	0.067
Cadmium	mg/L	0.001	<0.001	0.022
Calcium	mg/L	57	18	63
Chromium	mg/L	0.055	0.027	0.046
Cobalt	mg/L	0.007	0.001	0.004
Copper	mg/L	0.04	0.008	0.028
Iron	mg/L	14	4.1	17
Lead	mg/L	0.064	0.006	0.062
Magnesium	mg/L	16	4.1	23
Manganese	mg/L	0.56	0.12	0.69
Molybdenum	mg/L	<0.001	<0.001	<0.001
Nickel	mg/L	0.024	0.006	0.02
Phosphorus	mg/L	0.53	0.27	1.1
Potassium	mg/L	5.3	2.4	0.96
Seleium	mg/L	<0.005	<0.005	<0.005
Silicon, soluble	mg/L	5	1.8	3.8
Silver	mg/L	<0.001	<0.001	<0.001
Sodium	mg/L	25	7.4	21
Strontium	mg/L	0.13	0.036	0.18
Tin	mg/L	0.004	0	0.004
Titanium	mg/L	0.029	0.012	0.018
Vandium	mg/L	0.012	0.007	0.011
Zinc	mg/L	0.42	0.094	0.25
Zirconium	mg/L	0.004	0.001	0.003

Table I.2 – Heavy metal concentration results in baseflow samples.

Metal	Units	Sturgeon 31/07/2001	Silverwood 31/07/2001	Silverwood 09/10/2002	Taylor 31/07/2001	Taylor 09/10/2002	Avenue B 31/07/2001
Mercury	mg/L	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
Aluminum	mg/L	0.028	0.049	2.5	0.025	<0.005	0.13
Antimony	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Arsenic	mg/L	0.0008	0.001	<0.002	0.0007	0.0032	0.0008
Barium	mg/L	0.073	0.065	0.026	0.078	0.076	0.079
Berrylium	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Bismuth	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Boron	mg/L	0.15	0.19	0.29	0.28	0.11	0.44
Cadmium	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Calcium	mg/L	26	300	0	300	0	160
Chromium	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Cobalt	mg/L	<0.001	<0.001	<0.001	0.002	<0.001	<0.001
Copper	mg/L	0.005	0.003	0.002	0.004	<0.001	0.004
Iron	mg/L	0.053	0.048	<0.001	0.11	0.085	0.33
Lead	mg/L	<0.002	<0.002	<0.002	<0.002	<0.002	0.002
Magnesium	mg/L	120	120	0	140	0	98
Manganese	mg/L	0.03	0.002	<0.001	0.19	0.055	0.075
Molybdenum	mg/L	<0.001	<0.001	0.002	<0.001	<0.001	0.046
Nickel	mg/L	0.005	0.002	0.001	0.006	0.002	0.004
Phosphorus	mg/L	0.07	0.13	0.01	0.15	0.07	0.18
Potassium	mg/L	12	13	0	13	0	14
Seleium	mg/L	0.022	0.075	0.069	0.006	<0.005	0.006
Silicon, soluble	mg/L	7.7	6.5	8.6	8.8	3.3	5.6
Silver	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Sodium	mg/L	90	90	0	140	0	150
Strontium	mg/L	1.4	1.3	1.5	1.7	0.68	0.87
Tin	mg/L	<0.002	0.003	0.003	0.004	0.004	0.002
Titanium	mg/L	0.002	0.003	<0.001	0.003	<0.001	0.006
Vandium	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Zinc	mg/L	0.021	0.006	<0.005	0.017	0.011	0.053
Zirconium	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001

Table I.3 – Heavy metal concentration results for event composite samples.

Metal	Units	Sturgeon 14/08/2001	Silverwood 16/07/2001	Silverwood 25/08/2002	Taylor 16/07/2001	Taylor 25/08/2002	Avenue B 14/08/2001
Mercury	mg/L	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005	<0.00005
Aluminum	mg/L	10	1.9	5.5	7.2	2.5	32
Antimony	mg/L	0.001	<0.001	<0.001	<0.001	<0.001	0.001
Arsenic	mg/L	0.0008	0.004	0.0028	0.0058	<0.002	0.0018
Barium	mg/L	0.21	0.062	0.12	0.089	0.12	0.43
Berrylium	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	0.001
Bismuth	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Boron	mg/L	0.045	0.062	0.11	0.078	0.17	0.23
Cadmium	mg/L	0	<0.001	<0.001	<0.001	<0.001	0.002
Calcium	mg/L	69	25	0	67	0	98
Chromium	mg/L	0.022	0.015	0.015	0.004	0.005	0.065
Cobalt	mg/L	0.03	<0.001	0.003	0.002	0.002	0.007
Copper	mg/L	0.037	0.011	0.19	0.015	0.08	0.078
Iron	mg/L	14	0.81	6.3	7.9	2.9	30
Lead	mg/L	0.026	0.006	0.016	0.02	0.01	0.056
Magnesium	mg/L	27	6.5	0	26	0	45
Manganese	mg/L	0.45	0.1	0.27	0.48	0.3	0.73
Molybdenum	mg/L	<0.001	<0.001	0.004	<0.001	0.004	<0.001
Nickel	mg/L	0.022	0.008	0.015	0.009	0.01	0.05
Phosphorus	mg/L	0.39	0.3	0.87	0.41	0.54	1.5
Potassium	mg/L	4	4.2	0	7	0	8.4
Seleium	mg/L	0.002	<0.001	<0.005	0.002	<0.005	<0.001
Silicon, soluble	mg/L	2.9	1.9	2.5	2.9	5	6
Silver	mg/L	<0.001	<0.001	<0.001	<0.001	<0.001	<0.001
Sodium	mg/L	15	12	0	24	0	27
Strontium	mg/L	0.21	0.072	0.13	0.31	0.74	0.33
Tin	mg/L	<0.002	<0.002	<0.002	<0.002	0.004	<0.002
Titanium	mg/L	0.28	0.01	0.11	0.21	0.062	0.79
Vandium	mg/L	0.032	0.007	0.02	0.006	0.009	0.097
Zinc	mg/L	0.32	0.068	0.13	0.13	0.096	0.47
Zirconium	mg/L	0.013	<0.001	0.003	0.002	<0.001	0.034

APPENDIX J

Point Source Load Data

This appendix contains the data regarding the loads discharged from two point sources: COS WWTP and Akzol Nobel – Saskatoon Site. The discharge data contained herein was provided by the respective organizations. The parallel BOD_5 and COD tests were conducted at the U of S. In Table J.1, sample numbers 1 to 3 and 4 to 7 were collected concurrently.

Table J.1 – Parallel testing results for BOD_5 to COD ratio determination.

Sample number	BOD_5	Sample average BOD_5	COD	BOD_5 to COD ratio
1	7.55	7.83	12.92	0.606
1	8.45			
1	7.50			
2	7.85	8.90	11.08	0.803
2	9.65			
2	9.05			
2	9.05			
3	6.65	7.03	12	0.585
3	6.35			
3	6.95			
3	8.15			
4	8.20	7.90	12.68	0.623
4	7.60			
5	10.60	10.3	17.75	0.580
5	10.00			
6	8.20	6.80	16.06	0.423
6	7.60			
7	5.70	5.70	13.52	0.422
7	5.70			
Average				**0.58**

Table J.2 – COS WWTP Loads.

	2001	2002
Flow (x 10^6 m³)	28.02	28.69
BOD		
Mass removed (tonnes)	7705	8262
Process efficiency (%)	97%	97%
Mass discharged (tonnes)	238	256
COD discharged (BOD to COD ratio = 0.58) (tonnes)	411	441
TSS		
Mass removed (tonnes)	5351	5451
Process efficiency (%)	97%	96%
Mass discharged (tonnes)	165	227
Phosphorous		
Mass removed (tonnes)	164	154
Process efficiency (%)	94%	93%
Mass discharged (tonnes)	10	12

Table J.3 – Effluent data and calculated loads provided by Akzo Nobel.

Parameter	Units	2001	2002
Annual volume	(m^3)	47084	49189
Annual averages			
TKN (as N)	(mg/L)	27	29
BOD (as O_2)	(mg/L)	698	1173
TSS	(mg/L)	232	224
COD (as O_2)	(mg/L)	1559	1830
Annual Loads			
TKN (as N)	(tonne)	1.27	1.37
BOD (as O_2)	(tonne)	32.86	55.23
TSS	(tonne)	10.92	10.55
COD (as O_2)	(tonne)	73.4	90.0

APPENDIX K

Regression Residual Plots

This appendix contains the regression residual plots. Each set of figures represents one water quality parameter in one catchment and contains a) Residual versus observed load, b) Residual versus predicted load, and c) Relative residual versus observed load, and d) Relative residual versus predicted load.

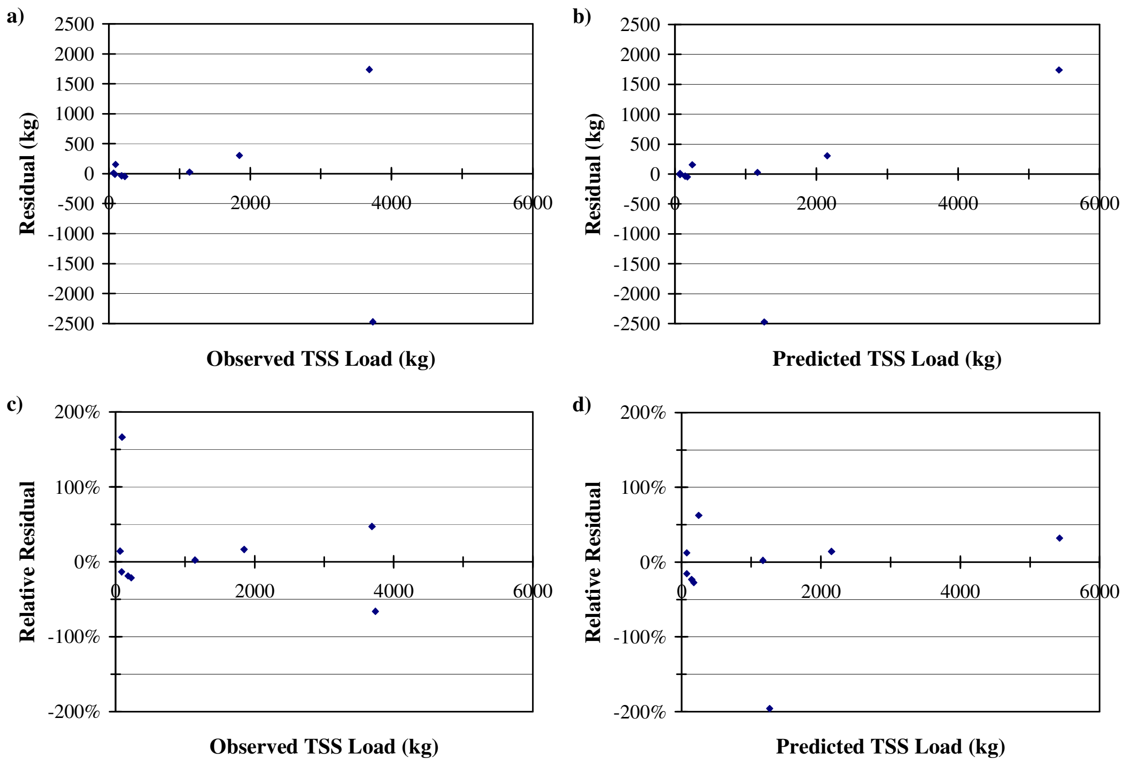

Figure K.1 – Avenue B TSS Residuals.

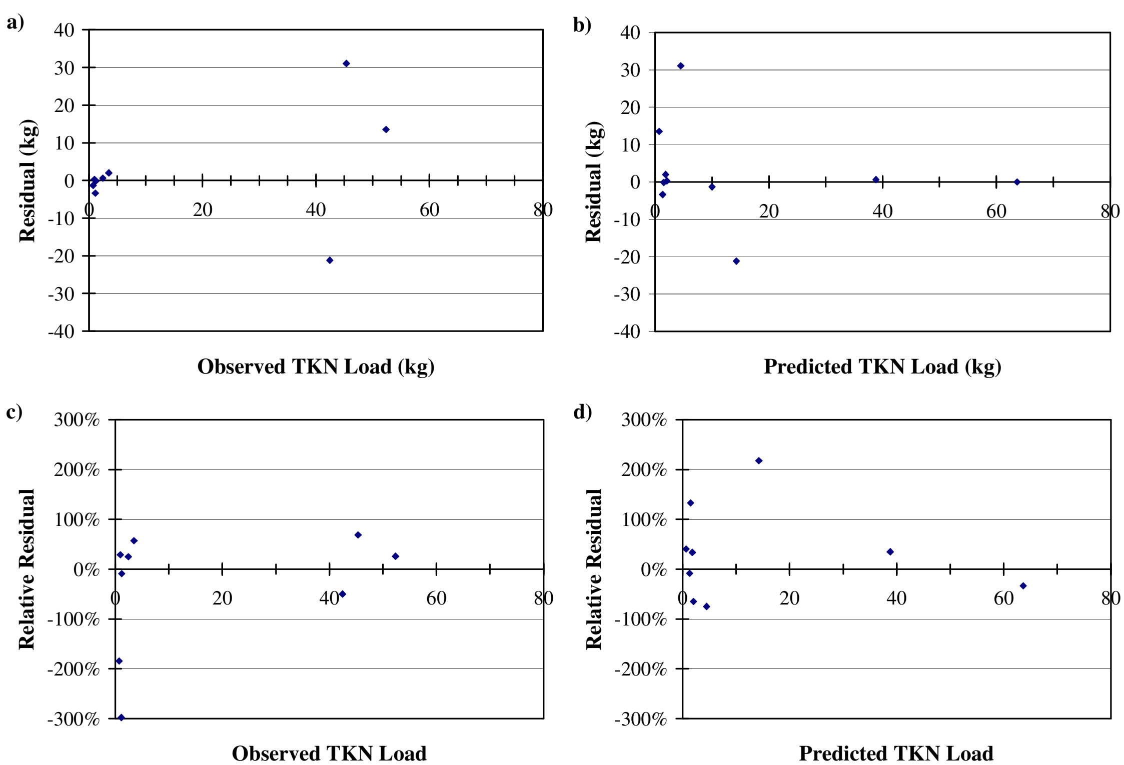

Figure K.2 – Avenue B TKN Residuals.

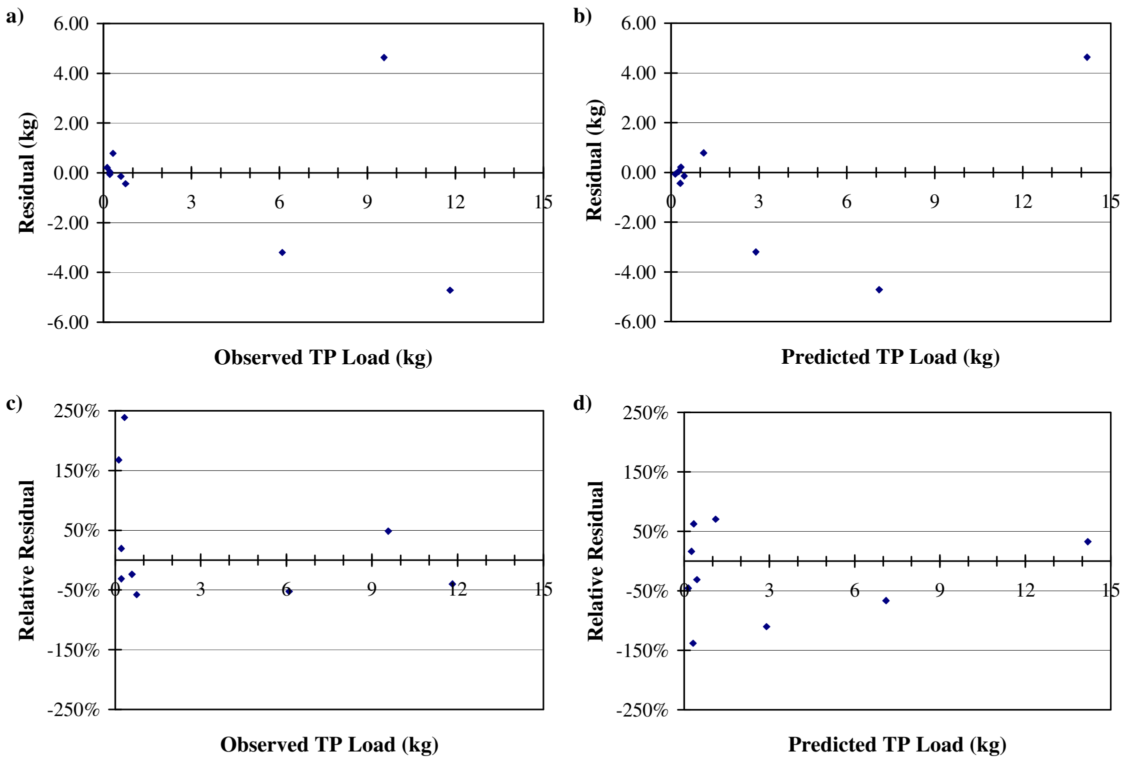

Figure K.3 – Avenue B TP Residuals.

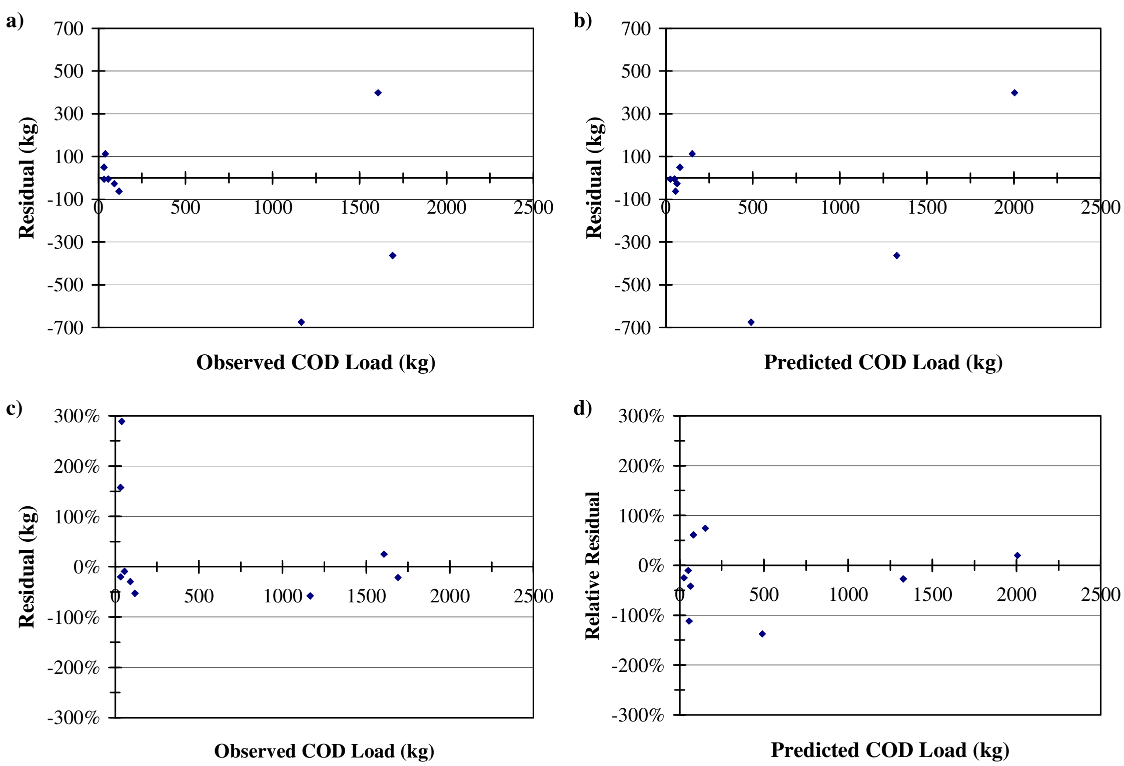

Figure K.4 – Avenue B COD Residuals.

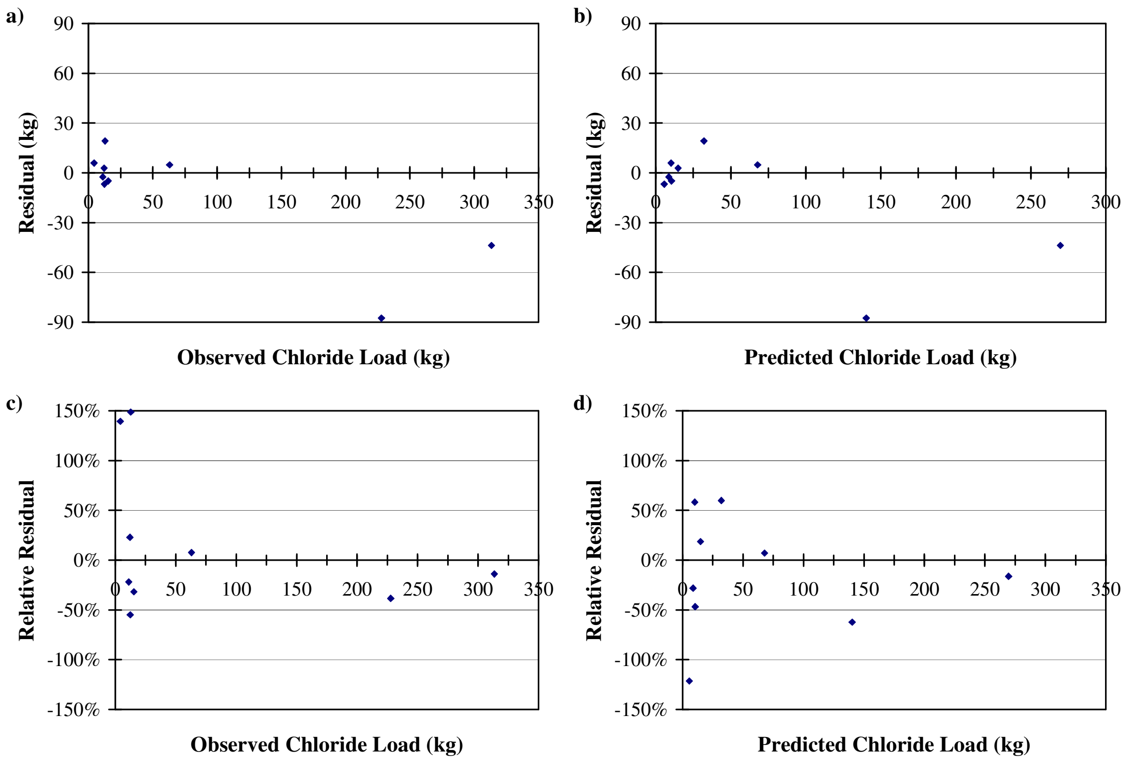

Figure K.5 – Avenue B Chloride Residuals.

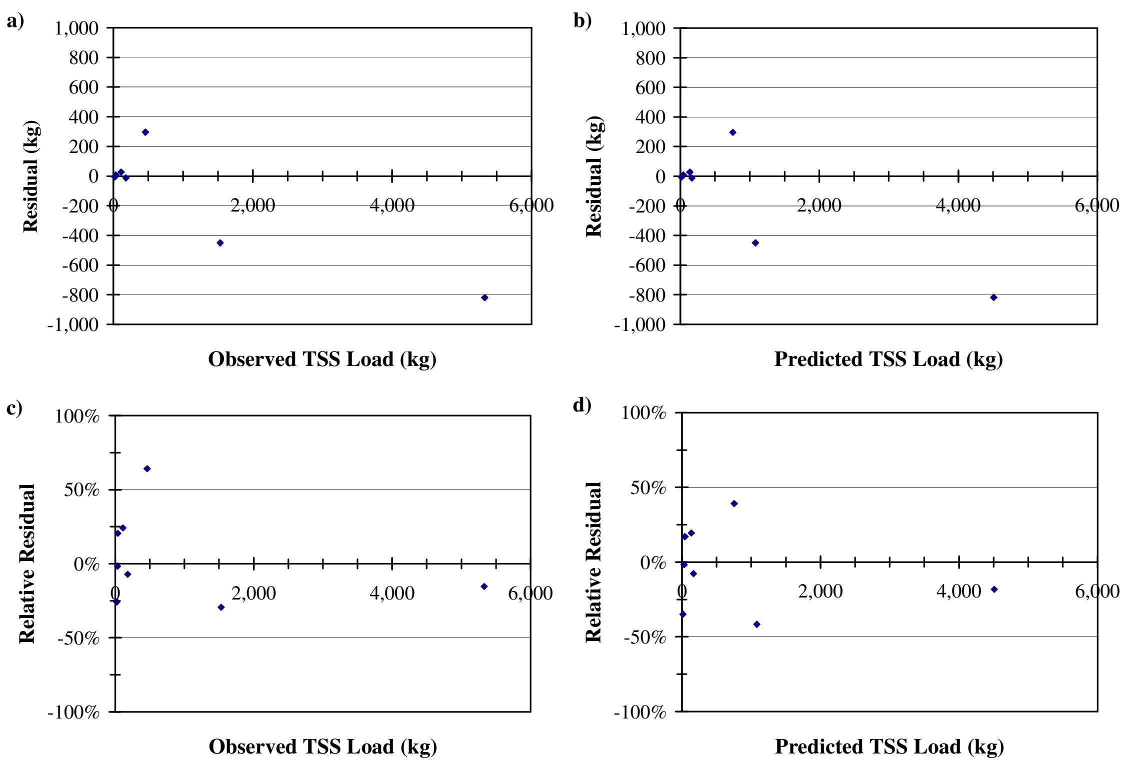

Figure K.6 – Silverwood TSS Residuals.

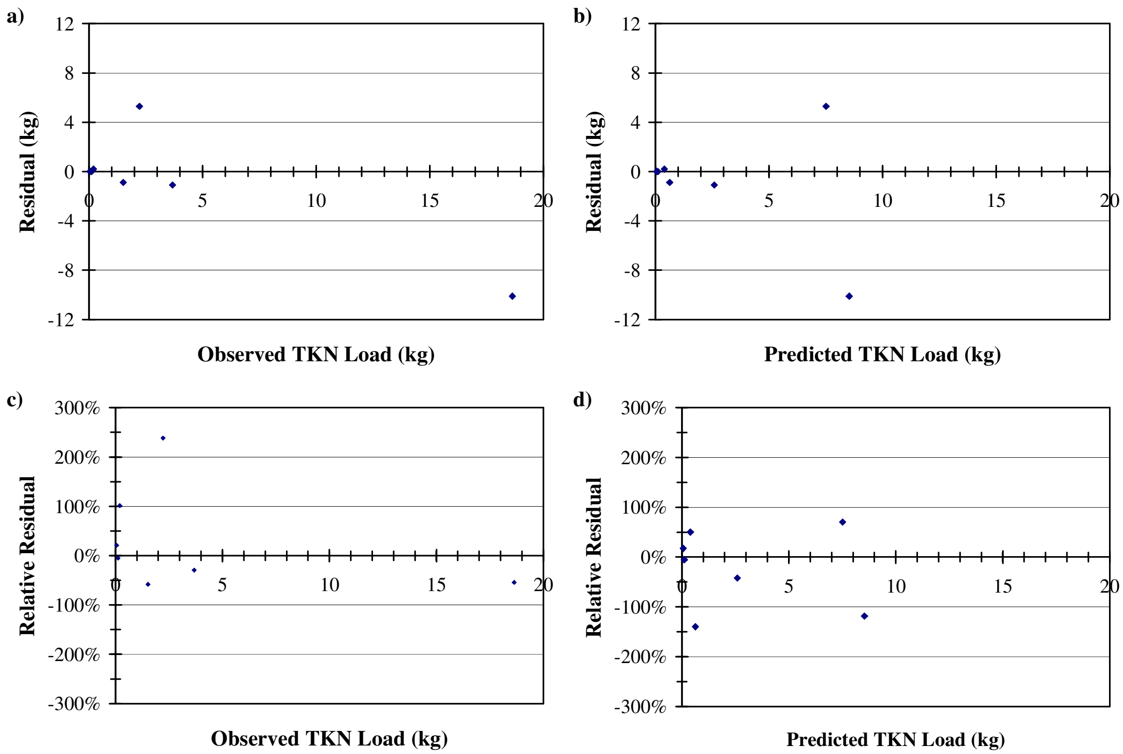

Figure K.7 – Silverwood TKN Residuals.

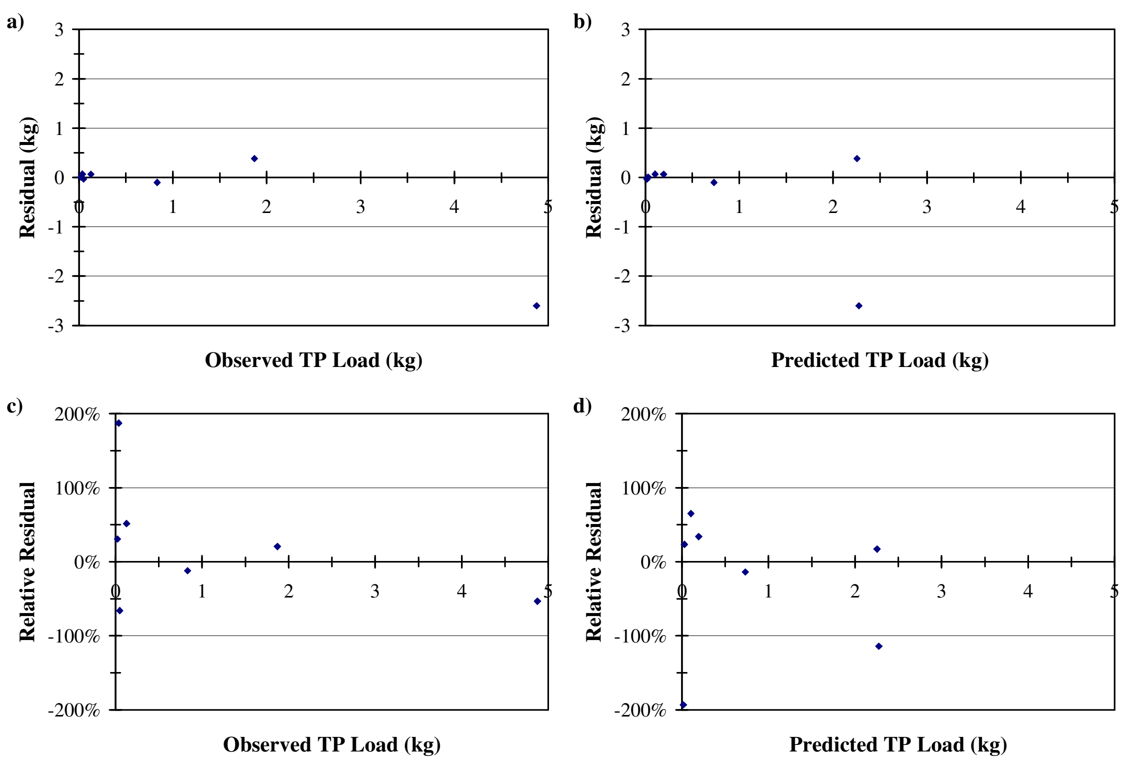

Figure K.8 – Silverwood TP Residuals.

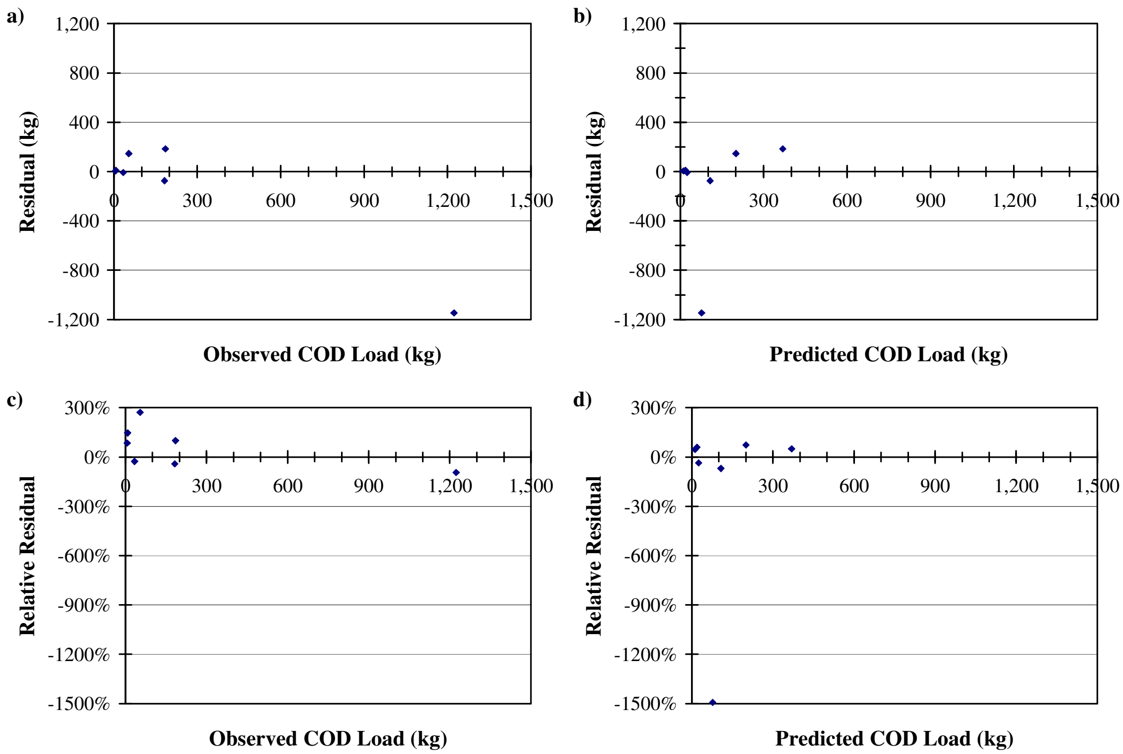

Figure K.9 – Silverwood COD Residuals.

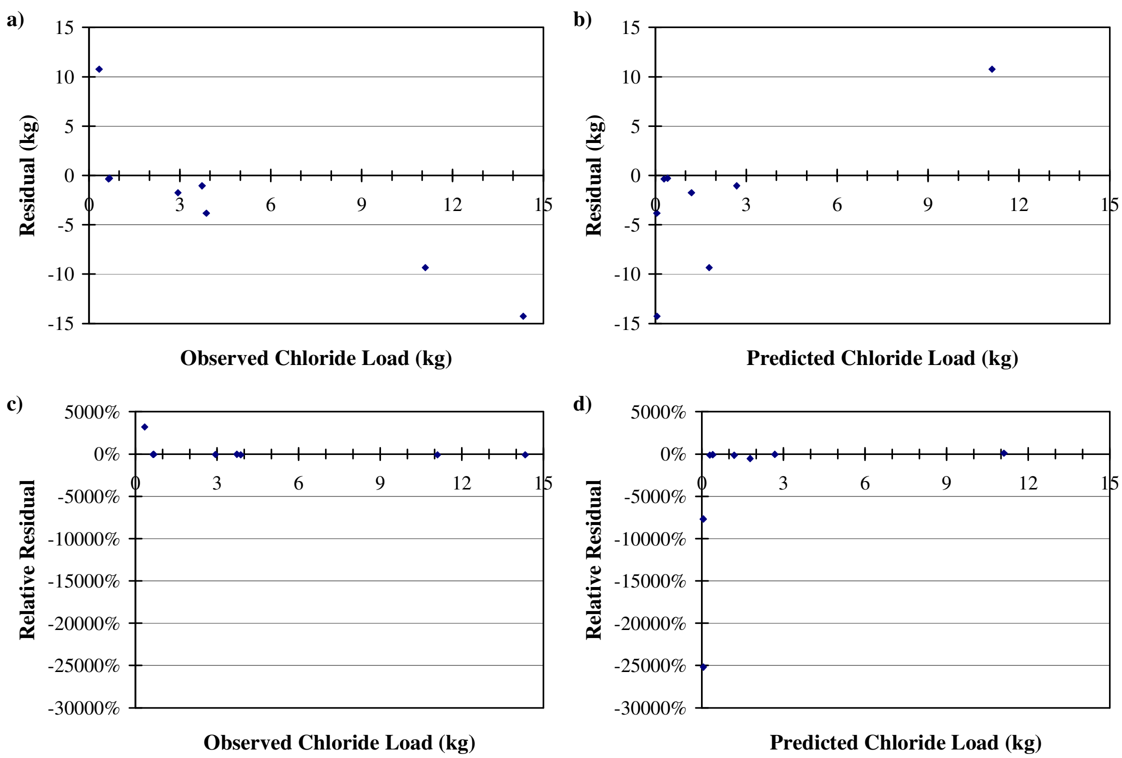

Figure K.10 – Silverwood Chloride Residuals.

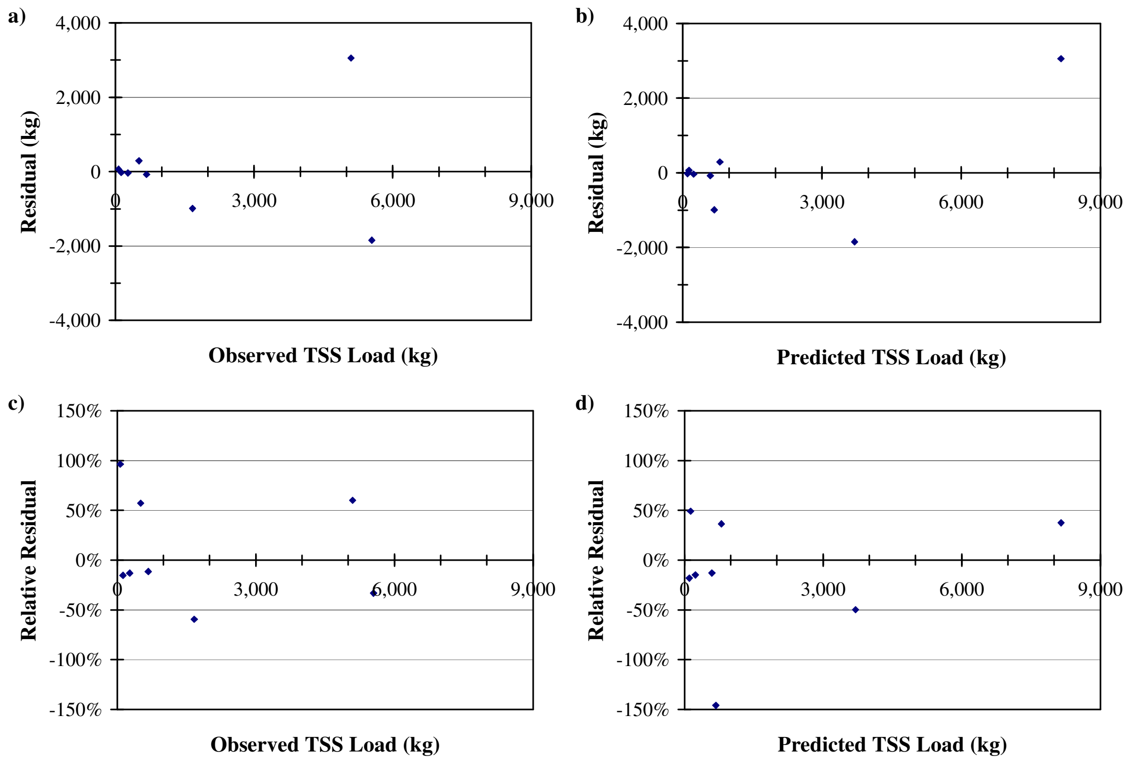

Figure K.11 – Taylor TSS Residuals.

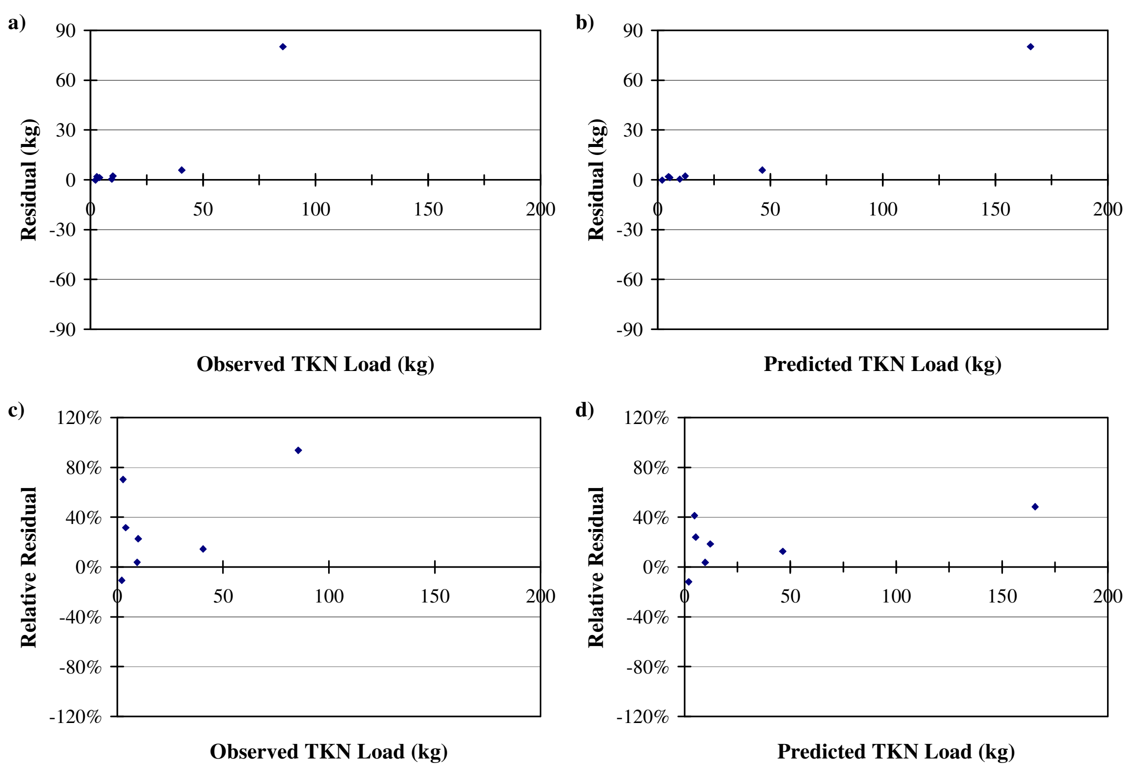

Figure K.12 – Taylor TKN Residuals.

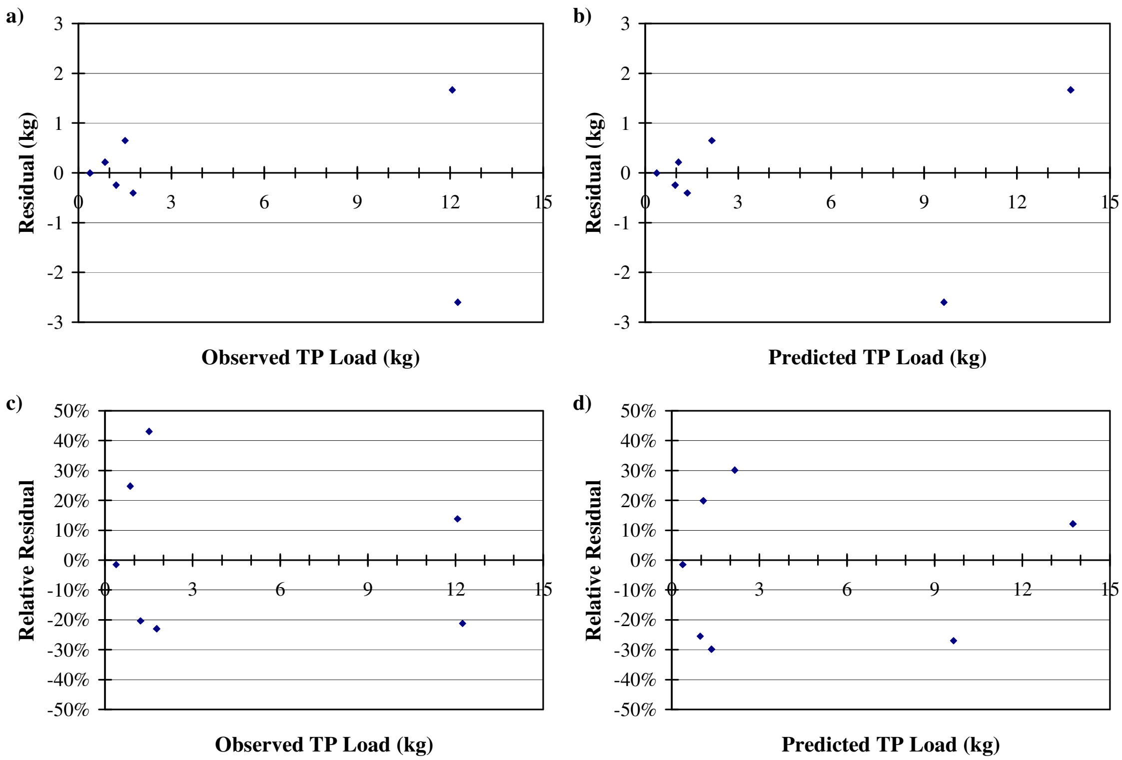

Figure K.13 – Taylor TP Residuals.

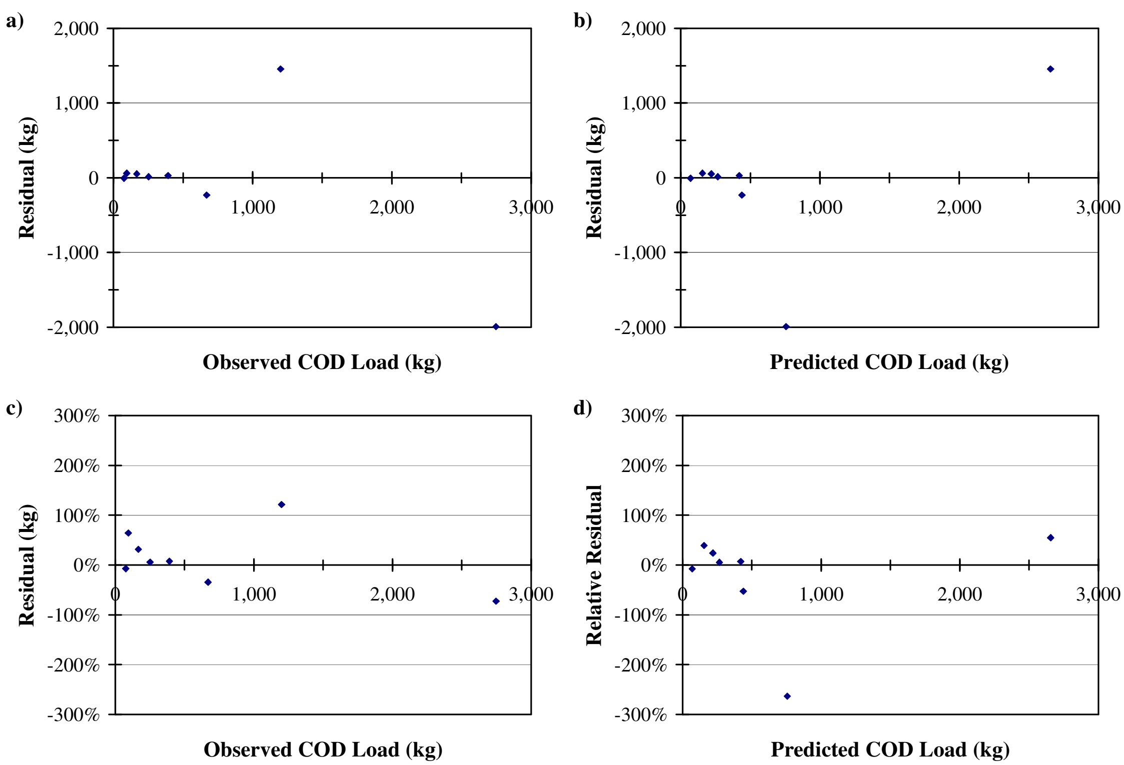

Figure K.14 – Taylor COD Residuals.

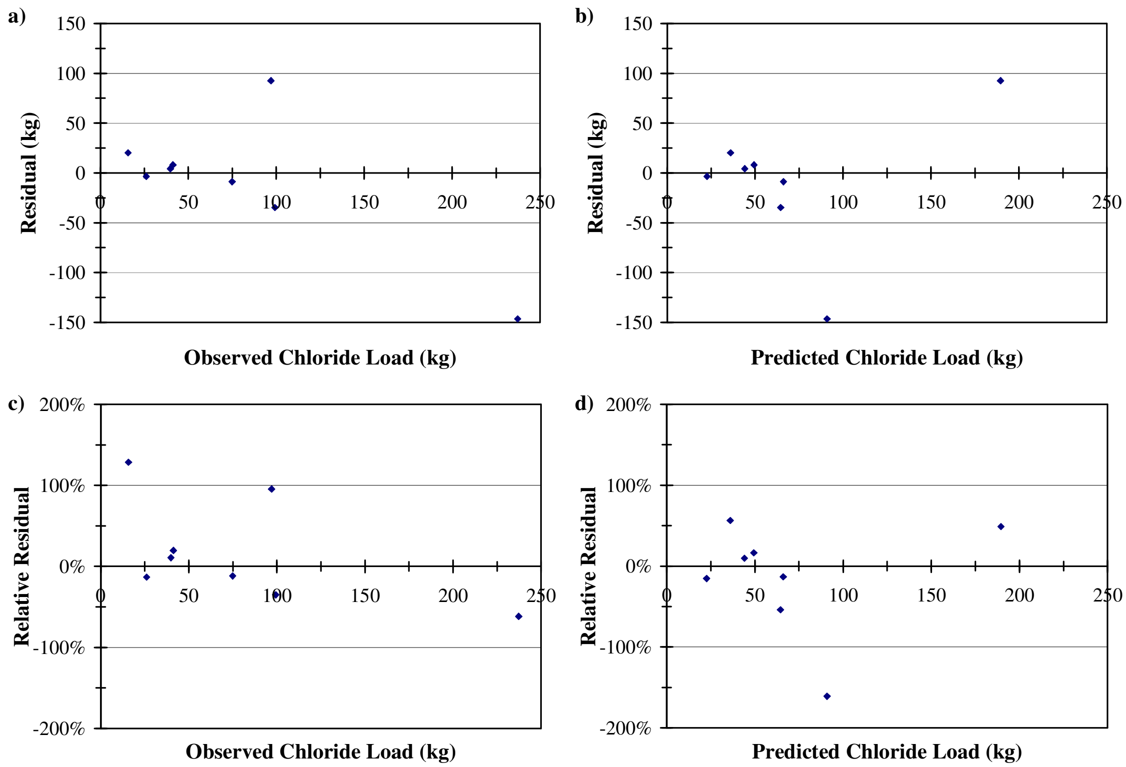

Figure K.15 – Taylor Chloride Residuals.

APPENDIX L

Flow Depth and Field Report Data

This appendix contains the flow depth data files and field retrieval reports for the useable events on the compact disc that is affixed to the back cover. The flow data and field reports are located in separate directories.

The flow data is in separate sub-directories directories for each flow meter. Each file within the directory is for one calendar year. The files are in comma delimited format.

The field retrieval reports for the useable events are in four PDF files – one for each catchment. A key for the sampler type noted on the report is also in the directory.