AN INVESTIGATION OF THE PERFORMANCE OF A CONSTRUCTED WETLAND IN TREATING URBAN STORMWATER

by
James Nagle Carleton

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APPROVED:

Thomas J. Grizzard, Co-Chairman

Adil N. Godrej, Co-Chairman

Harold E. Post

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ABSTRACT

This study investigated the pollutant removal performance of a constructed wetland treating stormwater runoff from a residential townhome complex in Manassas, Virginia. The facility was constructed by retrofitting a dry detention basin to retain a permanent shallow pool and included additional temporary storage to detain roughly the first half inch of rainfall for approximately 24 hours. Vegetation was allowed to establish itself through volunteer colonization, rather than active planting of selected wetland species.

Flow measurements showed substantially greater volume passing through the outlet than entering through the single gaged inlet. The extra volume (about 41% of the total) was attributed to ungaged overland flow which drained a wooded/grassy area adjacent to the site. Mass balance calculations employing the rational method with a runoff coefficient of 0.2 to estimate the flow from this area showed good agreement between long-term total estimated inflow and measured outflow. However, this method was not effective in accounting for the discrepancies between inlet and outlet volumes of individual storms.

Thirty-three runoff events were monitored between April, 1996 and May, 1997. Because of greater flow volumes passing through the outlet, constituent mass calculations which ignored the overland contribution generally exhibited higher loads exiting than entering the facility. With the results from a limited number of grab samples representing concentrations in overland input, estimated efficiencies improved substantially, showing overall net removal for most constituents. Less than one year after being retrofitted, the basin showed signs of beginning to develop a diverse wetland flora.

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TABLE OF CONTENTS

LIST OF FIGURES

FIGURE PAGE

LIST OF TABLES

I. Introduction

Background

As point sources have been brought under control in the years following passage of the Federal Clean Water Act of 1972, it has become increasingly apparent that nonpoint sources now play a dominant role in degrading the surface waters of the U.S. Recent estimates suggest that greater than 65 percent of total pollutant loads to inland U.S. surface waters result from nonpoint pollution sources (EPA, 1989). Nonpoint sources, by definition, are diffuse, and include inputs such as urban and agricultural stormwater runoff, which wash sediments and deposited pollutants from the landscape and ultimately into receiving waters. In some regions, stormwater runoff has been identified as contributing the bulk of heavy metal and sediment loadings to receiving waters, along with substantial amounts of nutrients and biochemical oxygen demand (BOD) (Livingston, 1985 in: Rushton and Dye).

Wetlands are transitional ecosystems that exist at the interface between aquatic and terrestrial systems. Because of their position in the landscape, they are frequently the default recipients of stormwater runoff, including that which drains agricultural lands and urban developments. Historically, wetlands have been viewed as areas with little economic value, best drained and converted to other uses. Until the mid-1970s, drainage and destruction of wetlands was common in the United States, and in some cases encouraged by government policies. It is estimated that of the original wetland acreage in the lower 48 states, more than half has been drained since the mid-1800s (Mitsch and Gosselink, 1993). However, in recent years wetlands have been recognized as fulfilling a number of critical ecological functions, including providing wildlife habitat and food chain support. Research also has shown that many wetland types can improve water quality through a combination of physical, chemical, and biological mechanisms (Stockdale, 1991). Today, wetlands in the U.S. are protected by many federal laws and regulations which were originally intended for other purposes, including land use and water quality regulation.

Natural wetlands in the U.S. have long been used as wastewater discharge sites (Kadlec, 1996). During the 1960s and 1970s water quality monitoring at some discharge sites promoted an awareness of the water purification potential of wetlands, which led in turn to deliberate efforts to exploit wetlands for water pollution treatment. The effectiveness of artificial ponds with wetland vegetation was studied as a method for treating municipal wastewater. The success of these investigations led to development of full-scale treatment wetlands, and research on wetland treatment for various types of wastewater (CH2M-HILL, 1996). To date, most research has focused on treatment of secondary wastewater and acid-mine drainage. Wetlands constructed for wastewater treatment are widespread in the U.S. today, and are becoming common in Europe, Australia, and parts of Asia and Africa (Magmedov *et al.*, 1996). Recently, interest also has grown in making use of wetland processes to treat urban and agricultural stormwater runoff.

Natural wetlands can remove or transform pollutants and undoubtedly play an important role in improving water quality and protecting downstream receiving waters (Olsen, 1993). However, wetlands also can be degraded by nonpoint source pollution. The construction of wetlands designed specifically as best management practices (BMPs) for treatment of stormwater runoff seems, therefore, an especially attractive alternative, because natural wetlands need not be impacted. Anticipated ancillary benefits to the constructed wetlands approach include the potential creation of valuable patches of wildlife habitat in landscapes that have been substantially denuded of their original wetlands.

A major drawback to using constructed wetlands is the cost and difficulty associated with planting a wetland community. Guidelines from the state of Maryland on constructing stormwater wetlands recommend planting five emergent species, two of which are chosen from a list of "primary species", and three of which are selected from a list of "secondary species" (MD DOE, 1987a). However, recent studies in the mid-atlantic area (Athanas and Stevenson, 1991; OWML, 1990; Shenot, 1993) have indicated that despite intensive planting efforts, constructed stormwater wetlands frequently become dominated by species that colonize the wetland voluntarily. In some cases volunteer colonization leads to monospecific stands of aggressive plants such as cattails, while in others, diverse communities of macrophytes develop. Either way, deliberate planting at the outset may have little effect on the eventual vegetative composition of a wetland site.

Objective

The primary purpose of the present study was to examine the pollutant removal performance of an urban wetland constructed in a former dry detention pond, and to investigate any seasonal trends in removal efficiencies. An additional objective was to examine the outcome of establishing a constructed stormwater wetland by allowing volunteer colonization of a shallow impoundment.

II. Literature Review

Properties of Urban Runoff

Recently, it has been recognized that the urbanization of watersheds can be deleterious to receiving waters, causing decreased infiltration of rainfall, increased runoff volume, increased flooding, streambank erosion, and pollutant export via runoff (Schueler, 1987). In the 1970s, researchers began documenting that nonpoint pollution sources contribute substantial contaminant loads to receiving waters, and that one great source of nonpoint pollution is runoff from urban and industrial areas (Whipple and Hunter, 1977). In 1978, the U.S. Environmental Protection Agency (EPA) established the Nationwide Urban Runoff Program (NURP) to gather basic data on the physical and chemical characteristics of urban runoff across the country (EPA, 1983). At each location studied, event mean concentrations (EMCs) of common stormwater constituents from numerous categories, including nutrients, metals, and suspended solids were measured and compared. Results showed that trace metals, especially Cu, Pb, and Zn, were the most prevalent priority pollutants found in urban runoff. Nutrients were found generally to be present as well, at loadings around an order of magnitude less than those typical of emissions from a publicly owned treatment works (POTW). Oxygen demanding substances were found to be present at concentrations similar to those in secondary treatment plant effluents, while total suspended solids (TSS) concentrations were high compared with treatment plant discharges. In general, EMC variability among events at most sites was greater than variability among sites, thus EMCs did not vary significantly among land use types. However, significant correlations were found between land use type and pollutant loads.

Best Management Practices

In response to the growing recognition of the role of urban runoff in the degradation of receiving waters, a series of management control options, commonly called best management practices were developed (Schueler, 1987). BMPs can be grouped into general categories of structural and nonstructural controls (Horner *et al.*, 1994). Nonstructural BMPs include practices such as street sweeping, and land use controls. Structural BMPs include storage practices such as detention ponds (wet and dry), and vegetative practices such as swales and grassed filter strips. Detention ponds originally were not designed for water quality control, but to control downstream flooding by temporarily detaining runoff before release to receiving streams. However research showed that detention ponds could improve water quality as well, by functioning as sedimentation basins (Rushton and Dye, 1993). The EPA (1983) concluded that detention basins are capable of providing "very effective removal of pollutants in urban runoff". In particular, detention ponds with a permanent pool of water ("wet ponds") showed the highest performance, particularly when adequately sized. Detention devices without a permanent pool ("dry ponds") were found to be largely ineffective at reducing pollutant loads. Subsequent studies have confirmed the ability of wet ponds to remove various categories of pollutants, including trace metals, in deposited sediments (Baker and Yousef, 1995; Mesuere and Fish, 1989; Wigington, *et al.* 1983). Long-term retention of nutrients and metals in deposited sediments has also been demonstrated to take place in both natural and constructed wetlands (Crites, *et al.* 1995; Schiffer 1989).

Mechanisms of Constituent Removal in Wetlands

Detention basins are designed to remove pollutants from stormwater singularly by suspended solids sedimentation. In contrast, wetlands enhance water quality using a combination of physical, biological, and chemical mechanisms (EPA, 1993). Physical mechanisms include filtration and sedimentation. Emergent wetland vegetation filters out large debris and slows the velocity of inflow, thereby enhancing sedimentation and reducing bottom scouring (Rhode Island Dept. of Environmental Management, 1989). Potential chemical processes in pollutant attenuation include sorption, precipitation, and evaporation. Biological processes include microbial transformations (i.e. biochemical oxidation, ammonification, nitrification, and denitrification) and vegetative uptake. The degree to which each mechanism operates is different for each pollutant, and also is dependent on site-specific variables such as local soil type and hydrology.

Phytoremediation

Seidel (1976) of the Max Planck Institut in Germany first used plants to attenuate environmental pollutants in the 1950s. Her research demonstrated a surprising capability of certain plant species for survival in and degradation of high strength phenolic wastes. More recent studies have used plants to clean soil and water, partially due to lower anticipated costs in comparison to conventional treatment methods (Black, 1995). Various studies have used plants effectively to remediate pollutants ranging from petroleum hydrocarbons (Schnoor *et al.*, 1995) to trace metals (Dushenkov, *et al.* 1995; Srivastav, *et al.*, 1994). Mechanisms believed to be responsible for the pollutant removal properties of plants include enhancement of oxidative microbial processes in the rhizosphere, and direct pollutant uptake. Plants have been identified which sorb metals and store them in their tissues (Black, 1995). Plants also have been studied for their ability to remove nutrients from water. For example, poplar trees planted in a riparian buffer strip were found to reduce concentrations of agriculturally-generated nitrate, as well as atrazine, in surficial groundwater (Schnoor *et al.*, 1995).

Use of natural or constructed wetlands to treat wastewaters is thus closely linked to the concept of phytoremediation. Phytoremediation studies typically focus on the pollutant removal capabilities of a single plant. In contrast, wetland studies examine whole system processes, including the soil component and hydrology, and diverse assemblages of plants.

Studies of Natural Wetland Processes

Investigators working in different regions of the country have demonstrated that many wetland types can act as sinks or transformers of nutrients and other water constituents, often according to seasonal variations (Mitsch *et al.*, 1989). The following is a brief description of some studies of natural wetlands and pertinent findings.

Midwestern U.S.

In an investigation of a Madison, Wisconsin freshwater marsh, Novitzky (1978) estimated that 80 percent of the sediment, 21 percent of the nitrogen and 7 percent of the phosphorus that entered annually was retained. Fetter *et al.* (1978) also examined a creek-fed Wisconsin marsh which also received upstream wastewater and industrial effluents, and urban and agricultural stormwater runoff. During passage through the marsh, concentrations of BOD, total phosphorus (TP), and TSS declined by 80, 13, and 29 percent, respectively. Johnston *et al.* (1984) studied a lakeside fringe wetland in northeast Wisconsin receiving upland dairy farm runoff. Despite high nitrogen and phosphorus loadings from the watershed, the lake itself continued to have high water quality, which was attributed to nutrient trapping in the wetland. The authors measured sediment deposition in the wetland, and concluded that soil mechanisms were more important than vegetative uptake in long-term nutrient and sediment retention. They also noted that, with the exception of denitrification, the ability of wetlands to trap sediments and nutrients could not continue indefinitely. Summarizing years of research, Lee *et al.* (1975), concluded that Wisconsin marshes acted as nutrient sinks during the growing season (summer), and sources at other times of the year. This seasonal pattern was believed to be of benefit in reducing summer algal blooms in downstream lakes.

Examining an Iowa prairie pothole marsh, Davis *et al.* (1981) found that the marsh removed 86 percent of the nitrate entering from agricultural drainage, although phosphorus and Kjeldahl nitrogen were not appreciably affected, and a net export of soluble organic carbon was reported. Richardson and Marshall (1986) conducted field and microcosm studies on fen peatland soils in Michigan, and found that most added phosphorus was removed from the water column within one hour by adsorption to fine sediments and uptake by microorganisms. Plant uptake accounted for over half of the phosphorus removed over one year; however seasonal dieback released substantial phosphorus back to the water column. They concluded that whereas microorganisms and sediments controlled initial uptake rates, soil adsorption and peat accumulation controlled “long-term phosphate sequestration”.

In an extensive study on nonpoint source pollution and wetland occurrence in 17 watersheds in greater Minneapolis-St. Paul, Minnesota, Oberts (1982) found that median annual loads of TSS, total Kjeldahl nitrogen (TKN), nitrate, and TP were inversely related to the areal percent of the watershed consisting of wetlands. This observation led him to conclude that preservation of urban wetlands as treatment systems was beneficial from a pollution management standpoint.

Eastern U.S.

In studying a tidal freshwater wetland near Philadelphia, Grant and Patrick (1970) found that concentrations of nutrients and BOD in sewage-contaminated water passing through the wetland decreased, while concentrations of dissolved oxygen increased. Simpson *et al.* (1983) found that freshwater tidal wetlands on the Delaware river retained trace metals (Cu, Pb, Ni) and reactive phosphorus, but usually exported TP. Nitrogen was retained early in the growing season, and exported late in the growing season. Soil Pb concentrations were high near storm drain outlets, suggesting that sedimentation was a major removal mechanism. Macrophyte litter also was found to retain substantial quantities of nutrients and trace metals following plant dieback, though the authors speculated that this was only a temporary storage compartment, given high rates of decomposition.

Craft *et al.* (1989) examined fluxes of nitrogen, phosphorus, and organic carbon between two recently transplanted brackish marshes, and adjacent estuarine waters in North Carolina. Their estimates indicated that the marshes removed ammonium and phosphate, and exported dissolved organic carbon. Peterjohn and Correll (1984) reported that a 50 m wide riparian forest wetland in Maryland removed an estimated 89% of the total nitrogen (TN), and 80% of the TP that entered from cropland runoff. In laboratory studies, Bartlett *et al.* (1979) found denitrification to be the predominant removal mechanism for nitrate experimentally added to muck soils from a Massachusetts wetland.

Wetlands For Treatment of Nonpoint Source Pollution

Most studies of NPS pollutant removal performance in wetlands have been conducted in a few geographic locations, especially in Florida and Minnesota (Strecker *et al.*, 1992). In addition to constructed marshes, studies have focused on natural wetland types ranging from cypress swamps to northern peatlands. Variables that differ between the wetlands studied include hydrologic conditions, vegetation, climate, source of runoff, and degree of pretreatment. In addition, various monitoring and performance calculation procedures were employed, making comparisons among studies difficult. The following discussion describes most of the reports available to date. Many studies were conducted by entities such as municipalities, and were not published in peer-reviewed journals, but rather were published as project reports. Key details and performance results are presented in Tables 1 through 5.

Case Studies — Natural Wetlands

The results of several studies suggest that wetland performance is generally a function of both inflow rate and retention time, which are in turn functions of storm intensity, runoff volume, and wetland size (area and volume). Inflow rate probably influences pollutant retention by affecting the degree of bottom scouring and resuspension of settled solids, and therefore the retention of solids and solids-associated pollutants. As with wet ponds, storm volume and wetland volume determine the fraction of a runoff event potentially captured, and therefore made available for treatment, especially during quiescent periods between events (Woodward-Clyde, 1986). The importance of proper sizing was recognized in early design guidelines published by the state of Maryland (MDE, 1987a), which recommended that the surface area of a constructed stormwater wetland be at least three percent of the contributing watershed area. Since that time, other authors have suggested that the area ratio may not be as important as the volume ratio (ratio of average runoff volume to storage volume) in determining performance (Strecker, 1992). More recent guidelines (Schueler, 1992) recommend both a minimum area ratio of two

Table 1. Literature studies on wetland treatment of runoff

Name	Location	Study	Wetland Type*	System	Drainage†	Calculation method
St. Joseph	MN	Stark and Brown 1987	N	wetland (bog)	U	Load summation, 1 yr.
			N	wetland (marsh)	U	Load summation, 1 yr.
Hidden Lake	FL	Harper et al. 1986	N	wetland	U	Load summation, 1 yr.
Lake Jackson	FL	Tuovila et al. 1987	C	wetland	U	Loads, 1 storm
Lake McCarrons	MN	Wotzka and Oberts 1988		detention pond	U	Load summation, 2 yrs.
			C	wetland	U	Load summation, 2 yrs.
				system	U	Load summation, 2 yrs.
Mays Chapel	MD	Stack 1989	C	wetland	U	Median EMC reduction
Silver Star Road, original	FL	Martin and Smoot 1986		detention pond	U	Regression of loads, 13 storms
			N	wetland	U	Regression of loads, 13 storms
				system	U	Regression of loads, 13 storms
Silver Star Road, modified	FL	Gain 1996		detention pond	U	Geo. mean EMC reduction, 22 storms
			N	wetland	U	Geo. mean EMC reduction, 22 storms
				system	U	Geo. mean EMC reduction, 22 storms
Franklin Farms	VA	OWML 1990	C	wetland	U	Load summation, estimated long-term
Island Lake	FL	Schiffer 1988	N	wetland	U	Median grab sample conc. reduction
B31	WA	Reinelt and Horner 1995	N	wetland	U	Load sum estimation, 2 yrs.
Swift Run	MI	Scherger and Davis 1983	N, impounded	wetland	U	Load summation, 5 storms
Tampa Office Pond	FL	Rushton and Dye 1993	C	wetland	U	Load summation, 2 yrs.
Palm Beach PGA	FL	Blackburn et al. 1986	C, N	wetland (1982)	U	Mean grab sample conc., 36/yr
				wetland (1985)	U	Mean grab sample conc., 36/yr
Tanner's lake	MN	Oberts et al. 1989	N, impounded	pond + wetland	U	Load summation, 1 yr.
Carver ravine	MN		C	wetland	U	Load summation, 1 yr.
Clear lake	MN	Barten 1987	C	wetland	U	Load sum estim., grab samples, 6 yrs
Wayzata	MN	Hickok 1977	N	wetland	U	Load sum estimation, 1 yr.
Lake Munson	FL	Maristany and Bartel 1989	N, impounded	wetland	U	Mean EMC reduction, 3 storms
Josephine	MN	Willenbring 1985	N, impounded	wetland	U	Mean grab sample (43) conc. reduction
E2	MN		N, impounded	wetland	U	Mean grab sample (64) conc. reduction
Jones lake	MN		N, impounded	wetland	U	Mean grab sample (11) conc. reduction
Franklin County	OH	Niswander and Mitsch 1995	C	wetland	U	Load sum estimation, annual
Queen Anne	MD	Athanas and Stevenson 1991	C	wetland	U	Load summation, 23 months

* C = constructed, N = natural

† U = urban, A = agricultural

Table 1. (cont.) Literature studies on wetland treatment of runoff

Name	Location	Study	Wetland Type*	System	Drainage†	Calculation method
Hidden River	FL	Carr and Rushton 1995	N, impounded	wetland	U	Load summation, 30 months
Shop Creek	CO	Urbonas 1994		detention pond	U	Load summation, 3 yrs.
			C	wetland	U	Load summation, 3 yrs.
				system	U	Load summation, 3 yrs.
Pacific Steel	NZ	Leersnyder 1993	C	pond + wetland	U	Load summation, 6 storms
Glenwood	WA	Koon 1995	C	wetland	U	Mean EMC reduction, 5 storms
Greenwood	FL	McCann and Olson 1994	C	wetland	U	Load summation, 6 months
DUST Marsh	CA	Meiorin 1986	C	wetland cell A	U, A	Load summation, 11 storms
	CA		C	wetland cell B	U, A	Load summation, 11 storms
	CA		C	wetland cell C	U, A	Load summation, 11 storms
	CA		C	system	U, A	Load summation, 11 storms
Des Plains	IL	Hey 1994; Mitsch et al. 1995	C	wetland cell EWA3	U, A	Load summation, 2 growing seasons
	IL		C	wetland cell EWA4	U, A	Load summation, 2 growing seasons
	IL		C	wetland cell EWA5	U, A	Load summation, 2 growing seasons
	IL		C	wetland cell EWA6	U, A	Load summation, 2 growing seasons
Ash slough	FL	Goldstein 1986	N, impounded	wetland	A	Load summation, 3 yr. annual avg.
Armstrong slough	FL		C	wetland	A	Load summation, 3 yr. annual avg.
Boney Marsh	FL	Moustafa, 1995	C	wetland	A	Load sum estimation, 1 yr.
Fish Lake	MN	Brown 1985	N, impounded	wetland	U	Load summation, 1 yr.
Lake Elmo	MN		N	wetland	A	Load summation, 1 yr.
Lake Riley	MN		N	wetland	A	Load summation, 1 yr.
Spring Lake	MN		C	wetland	U	Load summation, 1 yr.
Lake Apopka	FL	Reddy et al. 1986	C	reservoirs	A	Mean grab sample conc. reduction
			C	flooded fields	A	Mean grab sample conc. reduction
PC12	WA	Reinelt and Horner 1995	N	wetland	A	Load summation, 20 months
Reedy Creek	FL	German 1989	N	wetland	A	Load sum estimation, 1 yr.
Lower watkins	MN	Willenbring 1985	N	wetland	A	Flow-wt. mean grab sample concs. (21)
Kingston	MN		N	wetland	A	Flow-wt. mean grab sample concs. (17)
Cache River	AR	Dortch 1996	N	wetland	A	Load sum estimation, 3 yrs.
Spring Creek	ND	Downer and Myers 1995	C	wetland	A	Load sum estimation, 2 yrs
Monocacy	MD	Cronk 1996	C	wetland	A	Mean grab sample conc. reduction

* C = constructed, N = natural

† U = urban, A = agricultural

Table 2. Reported literature values on wetland treatment of runoff - nutrient removal

Name	Study	System	Pollutant Removal Efficiency (percent)									
			OP	TSP/SRP/DP	TP	PP	NH3	SKN	TKN	NO3/NO2	Org. N	TN
St. Joseph	Stark and Brown 1987	wetland (bog)			14				14			
		wetland (marsh)			18				22			
Hidden Lake	Harper et al. 1986	wetland	-109		7		62.2			80.2	-24	-1.6
Lake Jackson	Tuovila et al. 1987	wetland		52.7	90	78.3	37.4			69.8/75.3		75.9
Lake McCarrons	Wotzka and Oberts 1988	detention pond		57	79				78	62		76
		wetland		15	32				28	24		27
		system		48	77				78	64		76
Mays Chapel	Stack 1989	wetland		29	-7		22.1			28		
Silver Star Road, original	Martin and Smoot 1986	detention pond	57	70	33		60			-17	17	19
		wetland	2	0	17		54			40	23	21
		system	28	57	43		61			9	39	36
Silver Star Road, modified	Gain 1996	detention pond	26	35	30		17			24	20	16
		wetland	-67	-46	-55		40			-193	-34	-49
		system	-24	5	-9		50			-123	-7	-25
Franklin Farms	OWML 1990	wetland	23.6	14.5	14.9		-0.5	-9.8	4.4	59.8		
Island Lake	Schiffer 1988	wetland			87		96		41			
B31	Reinelt and Horner 1995	wetland			7.5							
Swift Run	Scherger and Davis 1983	wetland			49				20			
Tampa Office Pond	Rushton and Dye 1993	wetland	67		65		39			65	59	
Palm Beach PGA	Blackburn et al. 1986	wetland (1982)			33		12		7	18/44		
		wetland (1985)			62		17		16	33/71		
Tanner's lake	Oberts et al. 1989	pond + wetland	26	10	24				40	23		36
Carver ravine		wetland	27	21	24				14	18		15
Clear lake	Barten 1987	wetland	52	40	54		55		25			
Wayzata	Hickok 1877	wetland			78		-44					
Lake Munson	Maristany and Bartel 1989	wetland	-70.3		62.7		-39.4	-57.2	10.6	14.9		10.9
Josephine	Willenbring 1985	wetland			59				41			
E2		wetland			45				19			
Jones lake		wetland			9				19			
Franklin County	Niswander and Mitsch 1995	wetland			16							
Queen Anne	Athanas and Stevenson 1991	wetland	68.7	44.3	39.4	7.2	55.8			54.9/37.5	-5.4	22.8

Table 2 (cont.) Reported literature values on wetland treatment of runoff - nutrient removal

			Pollutant Removal Efficiency (percent)									
Name	Study	System	**OP**	**TSP/SRP/DP**	**TP**	**PP**	**NH3**	**SKN**	**TKN**	**NO3/NO2**	**Org. N**	**TN**
Hidden River	Carr and Rushton 1995	wetland	67		70		79		34	94	29	46
Shop Creek	Urbonas 1994	detention pond		38	14				-137	38/-115		-6
		wetland		3	36				65	21/-1		41
		system		36	44				18	51/-117		53
Pacific Steel	Leersnyder 1993	pond + wetland		75	79		-43			62		
Glenwood	Koon 1995	wetland		66	33		72			67		
Greenwood	McCann and Olson 1994	wetland	76.7		61.5		10.2		-10.3	-13.2/8.1		-11
DUST Marsh	Meiorin 1986	wetland cell A	53		17		-22		7	9		
		wetland cell B	19		-44		27		-32	5		
		wetland cell C	28		51		12		-17	8		
		system	56		48		10		-28	15		
Des Plains	Hey 1994;	wetland cell EWA3			79.1					80.6		
	Mitsch et al. 1995	wetland cell EWA4			70.8					49.8		
		wetland cell EWA5			87.4					82.3		
		wetland cell EWA6			>99.2					98.8		
Ash slough	Goldstein 1986	wetland	59.3		61.2							43.8
Armstrong slough		wetland	42.9		32.4							30.7
Boney Marsh	Moustafa 1995	wetland			71							26
Fish Lake	Brown 1985	wetland		28	37		0				36	-20
Lake Elmo		wetland		25	27		50				-36	38
Lake Riley		wetland		-30	-43		25				7	20
Spring Lake		wetland		-10	-7		-86				11	-14
Lake Apopka	Reddy et al. 1986	reservoirs	75.1		60.9		57.5		4.8	68.1		
		flooded fields	16.7		7.3		51.9		-7.6	64.2		
PC12	Reinelt and Horner 1995	wetland			82.4							
Reedy Creek	German 1989	wetland			33		88			62	16	36
Lower watkins	Willenbring 1985	wetland			9				2.5			
Kingston		wetland			38				12			
Cache River	Dortch 1996	wetland			3							21.4
Spring Creek	Downer and Myers 1995	wetland			39.6				-15.5	11		
Monocacy	Cronk 1996	wetland			54		30		60	90		

Table 3. Reported literature values on wetland treatment of runoff - misc. constituents removal

			Pollutant Removal Efficiency (percent)							
Name	Study	System	**Turb**	**TSS**	**VSS**	**ISS**	**TDS**	**COD**	**BOD5**	**TOC**
St. Joseph	Stark and Brown 1987	wetland (bog)		34						
		wetland (marsh)		44						
Hidden Lake	Harper et al. 1986	wetland		82.9					81.3	
Lake Jackson	Tuovila et al. 1987	wetland			94.1	96.3				
Lake McCarrons	Wotzka and Oberts 1988	detention pond		93	94			88		
		wetland		84	82			63		
		system		96	96			89		
Mays Chapel	Stack 1989	wetland								
Silver Star Road, original	Martin and Smoot 1986	detention pond		65	60		7	7		3
		wetland		66	60		38	18		18
		system		89	85		40	17		22
Silver Star Road, modified	Gain 1996	detention pond		43			-19			-30
		wetland		-170			-4			-1
		system		-24			-24			-31
Franklin Farms	OWML 1990	wetland		61.5						
Island Lake	Schiffer 1988	wetland								
B31	Reinelt and Horner 1995	wetland		13.6						
Swift Run	Scherger and Davis 1983	wetland	64			-3	-3			
Tampa Office Pond	Rushton and Dye 1993	wetland		55						
Palm Beach PGA	Blackburn et al. 1986	wetland (1982)	69	25					6	-10
		wetland (1985)	68	50					35	10
Tanner's lake	Oberts et al. 1989	pond + wetland		84	82			63		
Carver ravine		wetland		46						
Clear lake	Barten 1987	wetland		76						
Wayzata	Hickok 1877	wetland		94						
Lake Munson	Maristany and Bartel 1989	wetland	90.07	92.5			-5	28.18	42.16	
Josephine	Willenbring 1985	wetland		92						
E2		wetland		88						
Jones lake		wetland		56						
Franklin County	Niswander and Mitsch 1995	wetland								

Table 3. (cont.) Reported literature values on wetland treatment of runoff - misc. constituents removal

			Pollutant Removal Efficiency (percent)							
Name	Study	System	**Turb**	**TSS**	**VSS**	**ISS**	**TDS**	**COD**	**BOD5**	**TOC**
Hidden River	Carr and Rushton 1995	wetland		86						9
Shop Creek	Urbonas 1994	detention pond		50				-25		
		wetland		25				48		
		system		68				35		
Pacific Steel	Leersnyder 1993	pond + wetland		78				2		
Glenwood	Koon 1995	wetland								
Greenwood	McCann and Olson 1994	wetland		68.3			-100			
DUST Marsh	Meiorin 1986	wetland cell A		42			-9		-26	
		wetland cell B		24			-20		-22	
		wetland cell C		45			-50		-8	
		system		64			-49		-35	
Des Plains	Hey 1994;	wetland cell EWA3		90.4						
	Mitsch et al. 1995	wetland cell EWA4		85.1						
		wetland cell EWA5		95.9						
		wetland cell EWA6		99.6						
Ash slough	Goldstein 1986	wetland								
Armstrong slough		wetland								
Boney Marsh	Moustafa 1995	wetland								
Fish Lake	Brown 1985	wetland		95	78					
Lake Elmo		wetland		88	80					
Lake Riley		wetland		-20	20					
Spring Lake		wetland		-300	-20					
Lake Apopka	Reddy et al. 1986	reservoirs								
		flooded fields								
PC12	Reinelt and Horner 1995	wetland		56						
Reedy Creek	German 1989	wetland					4			
Lower watkins	Willenbring 1985	wetland		74						
Kingston		wetland		27						
Cache River	Dortch 1996	wetland				29.5				
Spring Creek	Downer and Myers 1995	wetland		77.7						

Table 4. Reported literature values on wetland treatment of runoff - metals removal, group 1

			Pollutant Removal Efficiency (percent)									
Name	Study	System	**TFe**	**SFe**	**TAl**	**SAl**	**TPb**	**SPb**	**TCr**	**SCr**	**TCd**	**SCd**
St. Joseph	Stark and Brown 1987	wetland (bog)										
		wetland (marsh)										
Hidden Lake	Harper et al. 1986	wetland	-90.1	-131	63.1	-25.4	54.8	56.1	72.5	74.5	70.7	79.4
Lake Jackson	Tuovila et al. 1987	wetland										
Lake McCarrons	Wotzka and Oberts 1988	detention pond					88					
		wetland					74					
		system					93					
Mays Chapel	Stack 1989	wetland										
Silver Star Road, original	Martin and Smoot 1986	detention pond					39	29				
		wetland					73	54				
		system					83	70				
Silver Star Road, modified	Gain 1996	detention pond	42		48		73					
		wetland	-41		-61		-187					
		system	18		16		23					
Franklin Farms	OWML 1990	wetland										
Island Lake	Schiffer 1988	wetland	19				98		40			
B31	Reinelt and Horner 1995	wetland										
Swift Run	Scherger and Davis 1983	wetland	62				83					
Tampa Office Pond	Rushton and Dye 1993	wetland	25									
Palm Beach PGA	Blackburn et al. 1986	wetland (1982)										
		wetland (1985)										
Tanner's lake	Oberts et al. 1989	pond + wetland					63					
Carver ravine		wetland					42					
Clear lake	Barten 1987	wetland										
Wayzata	Hickok 1877	wetland					94				67	
Lake Munson	Maristany and Bartel 1989	wetland					55.49		91.42			
Josephine	Willenbring 1985	wetland										
E2		wetland										
Jones lake		wetland										
Franklin County	Niswander and Mitsch 1995	wetland										
Queen Anne	Athanas and Stevenson 1991	wetland										

Table 4. (cont.) Reported literature values on wetland treatment of runoff - metals removal, group 1

			Pollutant Removal Efficiency (percent)									
Name	Study	System	**TFe**	**SFe**	**TAl**	**SAl**	**TPb**	**SPb**	**TCr**	**SCr**	**TCd**	**SCd**
Hidden River	Carr and Rushton 1995	wetland	5				83				88	
Shop Creek	Urbonas 1994	detention pond	70	73								
		wetland	32	-9								
		system	79	71								
Pacific Steel	Leersnyder 1993	pond + wetland					93					
Glenwood	Koon 1995	wetland					35					
Greenwood	McCann and Olson 1994	wetland					59.7				0	
DUST Marsh	Meiorin 1986	wetland cell A					30		40			
		wetland cell B					27		20			
		wetland cell C					83		53			
		system					88		68			
Des Plains	Hey 1994;	wetland cell EWA3										
	Mitsch et al. 1995	wetland cell EWA4										
		wetland cell EWA5										
		wetland cell EWA6										
Ash slough	Goldstein 1986	wetland										
Armstrong slough		wetland										
Boney Marsh	Moustafa 1995	wetland										
Fish Lake	Brown 1985	wetland										
Lake Elmo		wetland										
Lake Riley		wetland										
Spring Lake		wetland										
Lake Apopka	Reddy et al. 1986	reservoirs										
		flooded fields										
PC12	Reinelt and Horner 1995	wetland										
Reedy Creek	German 1989	wetland										
Lower watkins	Willenbring 1985	wetland										
Kingston		wetland										
Cache River	Dortch 1996	wetland										
Spring Creek	Downer and Myers 1995	wetland										

Table 5. Reported literature values on wetland treatment of runoff - metals removal, group 2

Name	Study	System	Pollutant Removal Efficiency (percent)								
			TCu	**SCu**	**TMn**	**SMn**	**TNi**	**SNi**	**TZn**	**DZn**	**THg**
St. Joseph	Stark and Brown 1987	wetland (bog)									
		wetland (marsh)									
Hidden Lake	Harper et al. 1986	wetland	39.9	28.6	7.7	-10	70	69.5	40.9	57.1	
Lake Jackson	Tuovila et al. 1987	wetland									
Lake McCarrons	Wotzka and Oberts 1988	detention pond									
		wetland									
		system									
Mays Chapel	Stack 1989	wetland									
Silver Star Road, original	Martin and Smoot 1986	detention pond							15	-17	
		wetland							56	75	
		system							70	65	
Silver Star Road, modified	Gain 1996	detention pond	42		24				52		
		wetland	-67		23				-14		
		system	3		41				45		
Franklin Farms	OWML 1990	wetland									
Island Lake	Schiffer 1988	wetland	88						67		
B31	Reinelt and Horner 1995	wetland							30.6		
Swift Run	Scherger and Davis 1983	wetland									
Tampa Office Pond	Rushton and Dye 1993	wetland							51		
Palm Beach PGA	Blackburn et al. 1986	wetland (1982)									
		wetland (1985)									
Tanner's lake	Oberts et al. 1989	pond + wetland									
Carver ravine		wetland									
Clear lake	Barten 1987	wetland									
Wayzata	Hickok 1877	wetland	80						82		29.25
Lake Munson	Maristany and Bartel 1989	wetland	-4.25				-17.13		59.2		
Josephine	Willenbring 1985	wetland									
E2		wetland									
Jones lake		wetland									
Franklin County	Niswander and Mitsch 1995	wetland									
Queen Anne	Athanas and Stevenson 1991	wetland									

Table 5. (cont.)Reported literature values on wetland treatment of runoff - metals removal, group 2

			Pollutant Removal Efficiency (percent)								
Name	Study	System	**TCu**	**SCu**	**TMn**	**SMn**	**TNi**	**SNi**	**TZn**	**DZn**	**THg**
Hidden River	Carr and Rushton 1995	wetland	79		2				84		
Shop Creek	Urbonas 1994	detention pond	18	25	26	50			45	37	
		wetland	-15	20	25	36			24	7	
		system	6	49	44	68			58	41	
Pacific Steel	Leersnyder 1993	pond + wetland	84						88		
Glenwood	Koon 1995	wetland	25								
Greenwood	McCann and Olson 1994	wetland	58.9						68.9		
DUST Marsh	Meiorin 1986	wetland cell A	5		-22		34		6		
		wetland cell B	-10		-1		-30		-22		
		wetland cell C	32		-86		12		51		
		system	31		-111		20		33		
Des Plains	Hey 1994;	wetland cell EWA3									
	Mitsch et al. 1995	wetland cell EWA4									
		wetland cell EWA5									
		wetland cell EWA6									
Ash slough	Goldstein 1986	wetland									
Armstrong slough		wetland									
Boney Marsh	Moustafa 1995	wetland									
Fish Lake	Brown 1985	wetland									
Lake Elmo		wetland									
Lake Riley		wetland									
Spring Lake		wetland									
Lake Apopka	Reddy et al. 1986	reservoirs									
		flooded fields									
PC12	Reinelt and Horner 1995	wetland							23.2		
Reedy Creek	German 1989	wetland									
Lower watkins	Willenbring 1985	wetland									
Kingston		wetland									
Cache River	Dortch 1996	wetland									
Spring Creek	Downer and Myers 1995	wetland									

percent (or one percent for wetlands with extended detention), and a treatment volume large enough to capture 90% of all storm events (a 1.25 inch storm in the Washington metropolitan area).

In a study of a natural wetland (Swift Run) receiving runoff from a mixed urban/agricultural watershed in Michigan, the wetland demonstrated good removal of turbidity (64%), TP (49%), TKN (20%), total Fe ("TFe", 62%), and total Pb ("TPb", 83%) (Scherger and Davis, 1982). Removal efficiencies were generally higher for events with longer retention times. Efficiencies for a single snowmelt event were lower, attributable in part to the large flow volume passing through the wetland in a relatively short period. In another study, a natural wetland (Wayzata) receiving urban runoff in Minnesota was reported to have retained an estimated 77% of the TP and 94% of the TSS entering during a one year period (Hickok *et al.*, 1977). The authors attributed the wetland's high performance in part to slow inlet water velocity, and physical entrapment of suspended sediments by the organic soils present.

Another study examined a natural herbaceous wetland (Hidden River) treating urban runoff in southwest Florida, after pretreatment by sedimentation basins at the two inlets (Carr and Rushton, 1995). The marsh was found to effectively reduce total influent loads of TSS (86%), nutrients (29-94%), and trace metals (>79% for TCd, TCu, TPb, TZn). Performance was better during the dry season than during the wet season, which was attributed in part to higher inflow concentrations during the dry season yielding better treatment efficiencies. It is possible that, like in the Swift Run and Wayzata studies, lower storm volumes and slower inflow velocities also contributed to the superior performance of the Hidden River wetland during the dry season.

Additional evidence of the importance of wetland size to pollutant removal performance was provided in a review of data from five natural wetlands associated with lake restoration projects in Minnesota (Willenbring, 1985). Three of the wetlands (Josephine, E2, Jones Lake) received primarily urban runoff. One (Kingston) received primarily agricultural runoff, and one (Lower Watkins) received high loadings from a creamery wastewater treatment plant upstream. Based on inlet and outlet grab samples, and instantaneous flow measurements, positive removals of TSS, TP and TKN were estimated for all five wetlands. The Josephine wetland showed the highest performance for the three constituents, which was attributed to its having the greatest wetland area to watershed area ratio (6%).

Some studies examined spatial distributions of constituents within wetlands, highlighting the importance of sedimentation in the removal of solids, TP, and various metals. In a study of a freshwater natural marsh (Island Lake) receiving urban runoff in central Florida, water samples were collected at the inlet, outlet, and various locations throughout the marsh on one day each of the wet and dry seasons of 1986 (Schiffer, 1989). Constituent concentrations (nutrients, metals, TOC, color) were found to decrease with distance from the inlet. Sediment samples also contained decreasing concentrations of phosphorus and metals (Cd, Cu, Fe, Pb, Zn) with increased distance from the inlet. The samples collected farthest from the inlet contained relatively high Cr concentrations in comparison to other constituents, suggesting that Cr might have traveled farther than the other metals before being removed from the water column. Similarly, Harper *et al.* (1986) investigated the fate of urban runoff-derived nutrients and trace metals, which had been draining into a hardwood wetland (Hidden Lake) near Orlando, Florida for about a decade at the time of the study. Surface water samples were taken during storm events and baseflow at the inlet. Soil core and water grab samples were also taken at 25 m intervals along the flow path of the runoff. Results showed that trace metals (Cd, Zn, Mn, Cu, Fe, Pb, Ni, Cr), especially particulate fractions, were effectively retained in the upper sediment layers.

Other studies provided evidence that processes other than sedimentation also can play important roles in the removal of various constituents. In the Hidden Lake study, no significant removal of TN was observed, and water column OP concentrations actually increased with travel distance through the wetland. The latter was hypothesized to be due to the release of sediment-bound phosphate, as oxidation-potential and pH decreased along the flow path. Another study involved two natural wetlands in Washington state, one receiving runoff from an urbanized watershed, and one draining a rural/forested watershed (Reinelt and Horner, 1995). In the case of the urban wetland (referred to as "B3I") an estimated net release of TSS and TP took place during baseflow conditions, and a net uptake of TSS and TP occurred during stormflow conditions. Overall removal was positive for both TSS and TP, though not high compared to most other BMPs (<15 %). The authors speculated that reducing conditions during baseflow might account for increased phosphorus in the water column, in turn leading to greater algal productivity, and greater TSS washout. In contrast, Zn removal (particulate and dissolved) was 31%, occurring primarily during storms. The non-urban wetland (referred to as "PC12") had greater removal performance than the urban wetland for TSS, TP, and Zn, which was attributed to lower loading rates in the non-urban wetland.

While most studies have involved wetlands with recent applications of runoff, a few focused on wetlands that had been receiving runoff and wastewater for decades. Stark and Brown (1987) evaluated a natural bog and marsh system (St. Joseph) in Minnesota which had received both secondarily treated wastewater and urban runoff for 24 years. The bog was estimated to retain 34% of annual inputs of suspended solids, and the marsh retained 44% of suspended solids. The bog retained both TP and TKN at 14%. The marsh retained 18% of TP, and 22% of TKN. The authors noted that the vegetation in both the bog and the marsh had changed greatly since wastewater discharge began, with invasions of cattails and a loss of original vegetation. Similarly, Maristany and Bartel (1989) studied a wetland system (Lake Munson) in Tallahassee, Florida, which had received wastewater effluent and stormwater for over 30 years. Based on limited sampling, results indicated that the lake (a former cypress swamp, impounded in 1950) was highly effective at retaining some constituents, including TSS (92.5%) and TP (62.7%). The authors concluded that removal rates were similar to those expected from a new wet detention pond of similar dimensions, despite the facility's age and lack of maintenance. However as a result of years of nutrient inputs, the lake was in an advanced state of eutrophication, with heavy algal blooms, and water hyacinth invasions neccessitating periodic herbicide applications for control.

The amount and type of hydrologic and sampling data collected varied greatly among the studies. In the absence of complete information, some studies used simplifying assumptions or modeling techniques to fill in critical data gaps. German (1989) examined the effect of a natural wetland (Reedy Creek) on nutrients in runoff from citrus groves and urban areas in Orlando, Florida. Using measured daily discharge, and linear regressions of concentration vs. flow rate to estimate daily concentrations, the author estimated annual load removals for ammonia and nitrate as being greater than 50%, and TN and TP removals as greater than 30%. Long-term dissolved solids loads were estimated to be essentially unchanged by passage through the wetland. Dortch (1996) reported on three years of mass flux measurements taken at the upstream and downstream boundaries of a bottomland riparian forest wetland (Cache River) in eastern Arkansas. A source of uncertainty in the calculations was the fact that 23% more water exited the system than entered it at the upstream gage. The additional input was attributed to ungaged flow from tributaries entering the wetland between the upstream and downstream gages. To account for this, the author assumed that constituent concentrations in these inputs were the same as the concentrations measured at the upstream gage. Results showed 20 to 30% removal of TN and inorganic suspended solids, and only 3% removal of TP.

Case Studies — Constructed Wetlands

<u>The Role of Vegetation in Constituent Removal</u>

Previous authors, after reviewing the literature, have concluded that constructed wetlands generally performed slightly better, and with less variability, than natural wetlands at removing various constituents (Strecker *et al.*, 1992). In contrast to the wide variety of natural wetlands which have been examined, constructed stormwater wetlands tended to be much more homogeneous in design, all being essentially shallow freshwater marshes with emergent vegetation. Several constructed wetland studies suggest that the vegetation itself may play an important role in constituent removal, but that selectively establishing particular plant species may be difficult.

One study examined a system of man-made wetlands within an impounded natural wetland (Palm Beach PGA) being used to treat runoff from a residential/golf course development in Palm Beach, Florida (Blackburn *et al.*, 1986). At the time of construction, the man-made wetlands were planted with six native emergent species, including spikerush (*Eleocharis cellulosa*) and arrowhead (*Sagittaria* spp.). Based on grab sample monitoring, the authors estimated greater than 50% removal of influent TSS, turbidity, TP, and nitrate. Estimated removals were greater in 1985 than in 1982, which was attributed to the man-made portions of the system being only "35% planted" in 1982, versus "completed" in 1985. However, the man-made portions were described as being plagued during the study by invasions of aggressive nuisance plant species including cattails, hydrilla, and melaleuca and brazilian pepper trees.

Another study evaluated a shallow, heavily vegetated wet detention pond (Tampa Office Pond) which received runoff, via a grass swale, from an office parking lot in Tampa, Florida (Rushton and Dye, 1993). Although the pond had been planted with cypress trees (*Taxodium distichum*), arrowhead (*Sagittaria latifolia*), pickerelweed (*Pontederia cordata*), and cordgrass (*Spartina bakeri*) when built in 1986, an invasion of cattails (*Typha* spp.) had reduced the areal coverage of the planted species at the time of the study (1989-1991). Treatment efficiencies were greater than 50% for all constituents except ammonia and total Fe (TFe), for which removals were 39% and 25%, respectively. The authors suggested that increasing the pond retention time would improve pollutant load removals. In a follow-up study, this idea was tested by reshaping the pond to increase treatment volume, thereby increasing the retention time from 2 days to 14 days (Rushton *et al.*, 1997). In the altered pond configuration, pollutant load removals improved by at least 20% for all monitored constituents. Recontouring of the pond resulted in loss of most of the desirable species, so the littoral zone was replanted with pickerelweed and arrowhead. Vegetation surveys showed that over the next two years, diversity increased from 3.67 to

6.70 species per square meter. The planting of pickerelweed was believed to have reduced the expansion of the nuisance species torpedo grass (*Panicum repens*). However, by the end of the study cattails had again begun to invade the pond, and were not believed to be reduced by competition with either planted vegetation, or more desirable volunteer species.

Athanas and Stevenson (1991) studied two constructed wetlands (Queen Anne and Prince Georges) treating urban stormwater in Maryland. At the Queen Anne site, approximately 4000 plants of three species (*Scirpus americanus*, *Sauruus cernus*, and *Sagittaria latifolia*) were established by planting. By contrast, to study "natural revegetation", no vegetation was planted in the Prince Georges wetland, which was constructed in a former dry detention basin. Despite some technical difficulties, water quality monitoring at the Queen Anne site showed good removal of TSS (65.1%), OP (68.7%), nitrate (54.9%), and ammonium (55.8%). Removals of organic nitrogen and organic phosphorus were low or negative, suggesting that the wetland served to transform inorganic nitrogen and phosphorus into organic forms. Despite early invasions by cattails (*Typha latifolia*) and spike rush (*Eleocharis quadrangulata*), the site developed a diverse assemblage of plant species (mostly volunteer) that has persisted (Stevenson, 1997). Water quality monitoring at the Prince Georges site was plagued by technical problems which made performance calculations impossible. By contrast with the vegetation at the Queen Anne site, the wetland became covered by a dense growth of cattails during the first year of the study.

Another study examined a brackish marsh (Demonstration of Urban Stormwater System, or "DUST") system treating urban/agricultural runoff in Fremont, California (Meiorin, 1989). The marsh consisted of three separate systems (A/B/C), each with a different design. System A was a long narrow pond with shallow edges containing rooted aquatic plants, especially cattails (*Typha latifolia*). System B was an overland flow system followed by a pond which was transected into four separate cells by shallow underwater sills vegetated with cattails. Plant colonization in systems A and B was described as being incomplete. System C consisted of shallow, meandering channels planted with "heavy cattail and alkali bulrush" vegetation. Systems A and B, with retention times of 5-48 hours, flowed into system C, which had a retention time of 1-12 days. Of the three systems, C performed most consistently at reducing organic phosphorus, and all monitored metals except Mn (20-88% for total forms of Cr, Cu, Pb, Ni, and Zn (TCr, TCu, etc.)), attributable to the longer retention time and heavier vegetation in system C. As a whole, the marsh (systems A/B/C together) effectively reduced TSS (64%), inorganic N (10-15%),TP (48%), OP (56%), and Pb (88%). Plant tissue analyses revealed that trace metals taken up were concentrated primarily in roots, in some cases to concentrations above the background in soils. Overall, cattails exhibited a greater ability to accumulate metals than did bulrushes.

Niswander and Mitsch (1995) examined phosphorus dynamics and vegetational trends in a constructed riparian wetland (Franklin County) receiving urban runoff in Ohio. Based on limited sampling and simulation modeling, the wetland retained 16% of TP inputs during a one year period. In the two years following establishment, the planted herbaceous vegetation increased in density at the site. Plant diversity also increased, with the addition of a number of volunteer species through natural colonization.

Downer and Myers (1995) investigated pollutant removals from agricultural runoff in a wetland (Spring Creek) newly constructed along a tributary to a flood control reservoir in North Dakota. The wetland, which was located within the flood control pool of the reservoir, was found to remove 77.7% of TSS. Total phosphorus and nitrate were also removed (at 39.6% and 11% respectively), although the wetland was found to export Kjeldahl nitrogen. During the monitoring period, aquatic vegetation was described as being sparse. The authors speculated that pollutant removals would improve as the wetland reached maturity and the vegetation continued to spread.

Barten (1987) reported on a study of an artificial wetland (Clear Lake) constructed within a natural wetland for the purpose of treating urban and agricultural runoff in Waseca, Minnesota. Over the six year duration of the study, the facility was estimated to have removed 54% of the influent TP, primarily through sedimentation. The TP removal efficiency was greater in the spring and fall than in the summer, despite longer retention times during summer. The author attributed this result to increased dissolved phosphorus release from decomposing vegetation during the warmer months. A retention time of three days was found to result in 93% removal of suspended solids; however beyond three days, increased retention time did not result in increased suspended solids removal.

As part of a project to evaluate the performance of stormwater detention basin retrofits, the City of Baltimore (1989) converted an existing dry detention pond into a wetland (Mays Chapel). Wetland flora was established by planting, and was reported to greatly increase the cost of the retrofit. During storms, the wetland exhibited moderate retention of all constituents monitored. However, overall retention was offset somewhat by the net baseflow export of all constituents except nitrate. The median estimated overall removal for storms and baseflow combined was positive but low (<30%) for all constituents except TP, which exhibited a net export of seven percent.

Cronk (1995) examined the efficacy of two wetlands constructed as BMPs to treat dairy wastewater and barnyard runoff prior to its entering the Monocacy River in Maryland. The wetlands were planted with cattails (*Typha latifolia*) and bulrushes (*Scirpus validus*). During a nine month period from August to April, load calculations based on grab samples showed good removal of BOD_5 (70%), TSS (90%), TP (54%), TKN (60%), ammonia (30%), and nitrate (90%). However, difficulties were encountered in maintaining live vegetation within the wetlands, which neccessitated some replanting. High influent constituent loadings (*e.g.* 7.3 $g/m^2/day$ BOD_5) probably contributed to poor vegetation survival. The author speculated that enhanced solids pretreatment upstream might improve the functioning of the wetlands.

In a companion to the present study, a constructed urban marsh (Franklin Farms) was established in a former dry detention basin located adjacent to an elementary school in Chantilly, Virginia (OWML, 1990). The site retrofit included regrading and removal of existing cattail stands, followed by establishment of a 1.5-foot weir at the pond outlet, and the planting of over 3,000 plugs of native emergent plants within the facility. Like the

Crestwood wetland (the site in the present study), the outlet weir was designed to detain additional vertical (extended detention) storage above the permanent pool. While constituent removals for the entire set of 23 “synoptic” storms were disappointing (-8.2 to 15% for nutrients), performance was found to be substantially better for a subset of storms which had volumes less than the capacity of the marsh (>59% for nutrients). Over the course of two growing seasons, the total area covered by vegetation expanded substantially, including proliferations of spike rush (*Eleocharis obtusa*) and cattails, attributable to rhizomes left over from the cattail stand that had been removed.

Other Factors Affecting Constituent Removal

In addition to plant uptake, several authors identified soil, hydrologic, and other factors as having substantial impacts on pollutant removals within constructed wetlands. Reddy *et al.* (1982), in a study near Lake Apopka, Florida, examined nitrogen and phosphorus removal from agricultural drainage by shallow reservoirs planted with water hyacinths, elodea, and cattails, and flooded fields planted with cattails. A system of three reservoirs in series, each containing one of the three plant types, was found to exhibit better nitrate, ammonium, and OP removals (68.1%, 57.5%, and 75.1%, respectively) than a single large reservoir (54.4%, 41.8%, and 62.5%, respectively). A control reservoir without plants also showed good nitrate and ammonium removals (55.4% and 33.5% respectively), and somewhat less OP removal (20.9%). Similarly, a series of three flooded fields gave better nitrate and ammonium removals (64.2% and 51.9% respectively) than a single large flooded field (50.7% and 43.5% respectively). As with the reservoirs, a control flooded field without plants also gave good nitrate and ammonium removals (48.4% and 38.7% respectively). Removals of TKN were poor (<10%) for all reservoir systems, and negative for all flooded fields. Inorganic nitrogen removals were attributed by the authors to a combination of plant uptake, nitrification, denitrification and volatilization. Phosphorus removals were attributed to plant and algal uptake, and adsorption and precipitation reactions.

Moustafa *et al.* (1995) examined TP and TN removal from agricultural runoff by a constructed wetland (Boney Marsh) in Florida. Mass removal efficiencies were higher for TP (71%) than for TN (26%). Constituent retentions were found to be linearly related to inflow loading rates, and the authors attributed the relatively low TN retentions to low input loads.

Tuovila *et al.* (1987) examined the efficacy of an urban stormwater treatment system (Lake Jackson) in Tallahassee, Florida which consisted of a constructed marsh preceded by both a detention pond and an underdrain sand filter. Based on results from a single storm (monitoring period of 13 days), the system showed good removals for TN (75.9%), TP (90%), and suspended solids (96%). Treatment volume and the amount of flow bypassing the facility were identified as crucial to the facility's performance. The authors noted that periodic scraping of the filter and dredging of the detention pond would be needed to maintain the effectiveness of the system.

In one of the most ambitious constructed wetland projects to date, investigators examined nutrient and suspended solids removals from four restored riparian marshes built along the Des Plains River in northeastern Illinois (Hey *et al.* 1995; Mitsch *et al.*1995; Mitsch and Cronk 1995; Phipps and Crumpton 1994). The river drains a watershed of mixed agricultural and urban land. To allow studies of the effect of various hydrologic loading rates, pumps were used to deliver controlled amounts of river water at two different flow rates ("low-flow" and "high-flow") to the four wetlands. At both loading rates, the wetlands removed substantial percentages of nitrate, TN and TP. Organic nitrogen was exported at the higher loading rates, but was unchanged at lower loading rates. Isopleths of water column TP and soluble reactive phosphorus showed relatively homogeneous concentrations in the two lower-flow wetlands, suggesting that most of the phosphorus was removed near the inlet. By contrast, concentrations in the two higher-flow wetlands exhibited a decreasing gradient from inlet to outlet, suggesting phosphorus was removed from the water column throughout the wetland.

Case Studies—Natural and Constructed Wetland Comparisons

Two studies that examined natural and constructed wetlands concurrently were identified. Although of limited scope, both studies suggested that constructed wetlands, when adequately sized, may provide pollutant removals comparable to those of natural wetlands.

Goldstein (1986) studied a 20 acre natural wetland (Ash Slough) and a 30 acre constructed wetland (Armstrong Slough) in southern Florida. Both received runoff from watersheds involved in cattle production. In general, the wetlands exhibited similar performances, both showing greater than 50% removal of inorganic nitrogen, and 20 to 30% removal of OP and TP. Both were least effective at TN removal. Over the three year study period, both wetlands showed a trend of decreasing phosphorus uptake. It was unclear whether this decrease was representative of a long-term trend. It also was not known whether significant nutrient losses might be the result of infiltration to groundwater.

Brown (1985) compared three natural wetlands (Fish Lake, Lake Elmo, Lake Riley) and a constructed wetland (Spring Lake) receiving runoff from primarily agricultural land in Minnesota. While the natural wetlands reduced most nutrient and sediment concentrations to some degree, the constructed wetland was ineffective (export of most constituents), which was attributed to its being vastly undersized relative to the volumes of runoff it received.

Case Studies—Wetlands Combined with Detention Ponds

Oberts *et al.* (1989) studied a natural wetland (Tanner's Lake), a wetland preceded by a detention basin (McCarron's), and a wetland/detention system (Carver Ravine), all receiving urban runoff in Minnesota. Although drought conditions during the study limited the number of sampling events, all facilities reduced pollutant loads by 7-96%. The most effective system was McCarron's, and the detention basin in that system had higher removal percentages than the wetland for monitored pollutants (TSS, TP, TN, TPb). Lower efficiencies in the wetland were attributed to pre-treatment by the detention basin.

Martin and Smoot (1986) examined the effectiveness of a detention pond and natural wetland in series (Silver Star Road) treating highway runoff near Orlando, Florida. The system as a whole substantially reduced loads of suspended solids (89%), metals (>40% for Zn, Pb), and nutrients (9-61%). While the detention pond reduced suspended constituent loads, the wetland was effective at reducing both suspended and most dissolved constituents. In a later report, Gain (1996) evaluated the same system after installation of a flow barrier to lengthen the flow path and increase the retention time within both the pond and the wetland. Surprisingly, the overall effect of the flowpath modifications was a reduction in performance for all monitored constituents except TZn and ammonia. During periods of rapid inflow, more efficient flushing of formerly quiescent areas within the system may have caused the reduced performance.

McCann and Olson (1994) studied a constructed wetland system (Greenwood) with a "sediment control" pretreatment basin, treating urban runoff in Orlando, Florida. While the performance of the detention basin was "sporadic", over an eight month period the wetland was effective at removing estimated total loads of phosphorus (61.5%), suspended solids (68.3%), and some trace metals (>58% for TCu, TPb, TZn; 0% for Cd). The performance at removing nitrogen was poor, including a net export of TKN, TN, and nitrate. The latter may have been due in part to large contributions of nitrate from groundwater believed to be discharging into the wetland. The reason for poor performance of the detention basin may have been related to sediment accumulation and the need for periodic cleanout.

Urbonas *et al.* (1994) reported on a three year study of a detention pond and constructed wetland (Shop Creek) treating urban stormwater in Aurora, Colorado. Dry weather base flows were found to have a substantial impact on overall load removal calculations, improving the apparent performance for some constituents (nitrate, dissolved phosphorus), while decreasing the performance for others (TKN, chemical oxygen demand (COD), TSS, Cu, Mn). The wetland appeared to "even out" low or negative constituent removals from the detention pond, resulting in net removal of all constituents (6-79%) except nitrite, which was exported.

General Observations

As demonstrated in Tables 1 through 5, constituent removals varied substantially among case studies. This undoubtedly is related to the differing site-specific characteristics among studies. Given the variability, drawing definitive conclusions is difficult. Nevertheless some general observations can be made (Strecker, 1992). Suspended solids (TSS) appeared to most consistently show high removal efficiencies, followed by Pb, Cr, and TP. Total metal removals generally exceeded dissolved metal removals, though not always by much. Pretreatment by a detention pond seemed to improve overall system performance for most constituents, confirming the importance of sedimentation as a removal mechanism for solids-associated pollutants. In many cases nutrient removals appeared to vary as a function of season, emphasizing the importance of vegetative uptake in the removal of dissolved constituents.

III. Methods and Materials

Site Description

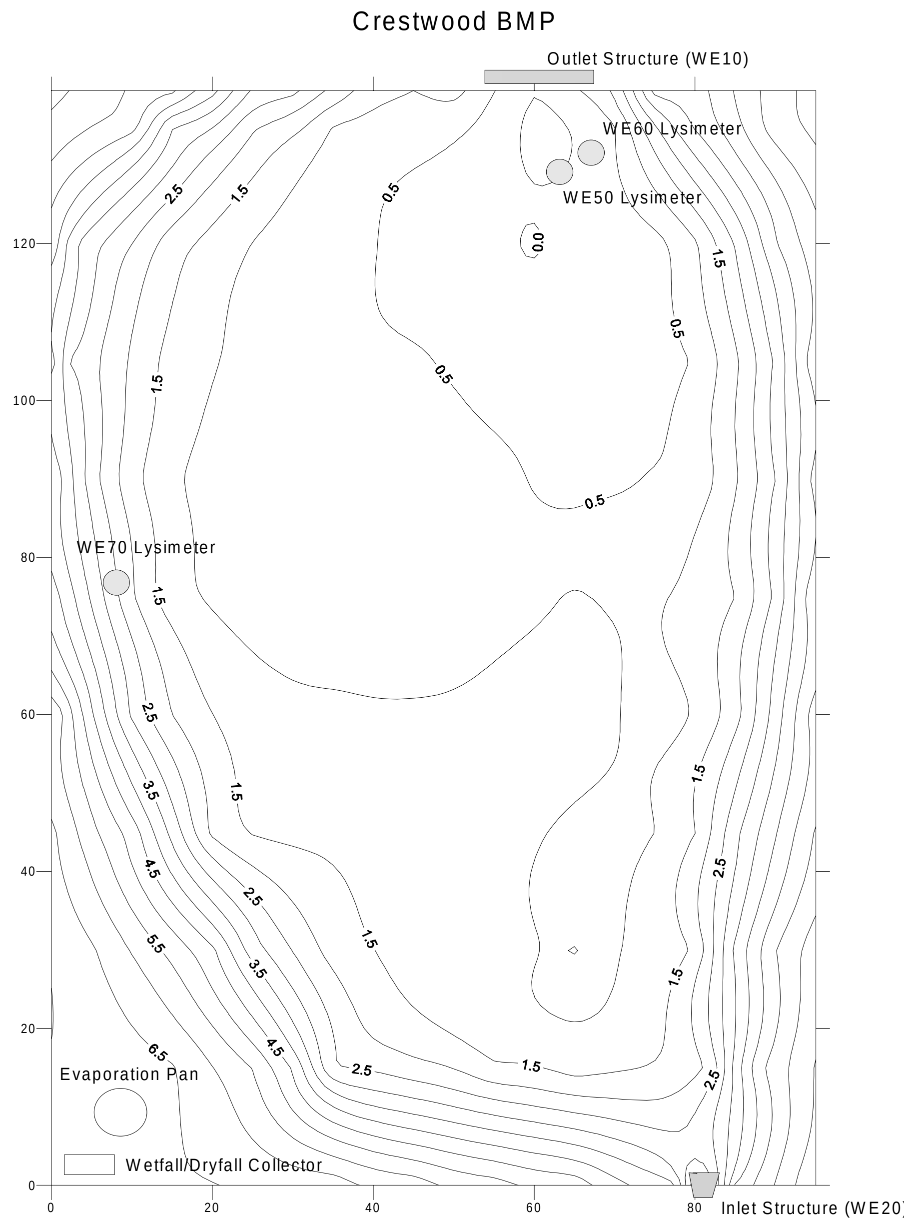
The dry detention basin/constructed wetland investigated in this study was located in the Tarawood townhome complex near route 234 in Manassas, Virginia. Prior to being modified by OWML staff, the detention basin contained numerous volunteer plant species, including various wetland plants growing in the wet zone along a well-defined flow path channel. The basin was equipped with a single storm sewer inlet which transported runoff from an estimated 3.2 acre drainage area within the townhouse complex. An additional 4.03 acres of wooded and grassed land is immediately upland of the pond on one side, and serves to separate the complex from a working stone quarry nearby. Although details of the initial construction of the detention pond were not available to OWML, the facility is thought to be underlain by an impermeable clay or compressed earth layer.

Site Retrofit

Before its conversion to a wetland, the detention basin had been completely surveyed. A contour plot of the site is shown in Figure 1. Figures 2 and 3 show elevation-surface area and elevation-storage volume plots, respectively. In December of 1995, the detention basin was modified with the addition of a 1.5-foot weir (referenced to the invert of the original outlet structure) at the outlet. A 1.25-inch diameter orifice was installed in the weir in order to impound a permanent pool of water at a depth of 0.5 feet at the riser. Placement of the orifice was designed to drain the pool from the top of the weir to the orifice invert in 24 hours.

Station Equipment

Automatic flow gaging/sampling stations were located at the inlet (WE20) and outlet (WE10) of the facility, inside fiberglass utility sheds. Inlet and outlet flows passed through standard 18 inch diameter Palmer-Bowlus flumes (outlet diameter 15 inches) in order to enable flow measurements. At each station, a submerged Keller-PSI model 2005



X and Y coordinates are relative coordinates (in feet)

Vertical Datum: Outlet Structure invert.

Inlet Structure Invert Elevation = 1.52 ft
Outlet Structure:
Permanent Pool Elevation = 0.5 ft
Max Pool Elevation at weir top = 1.5 ft
Overflow Elevation = 2.43 ft.

Figure 1. Contour map of Crestwood marsh BMP

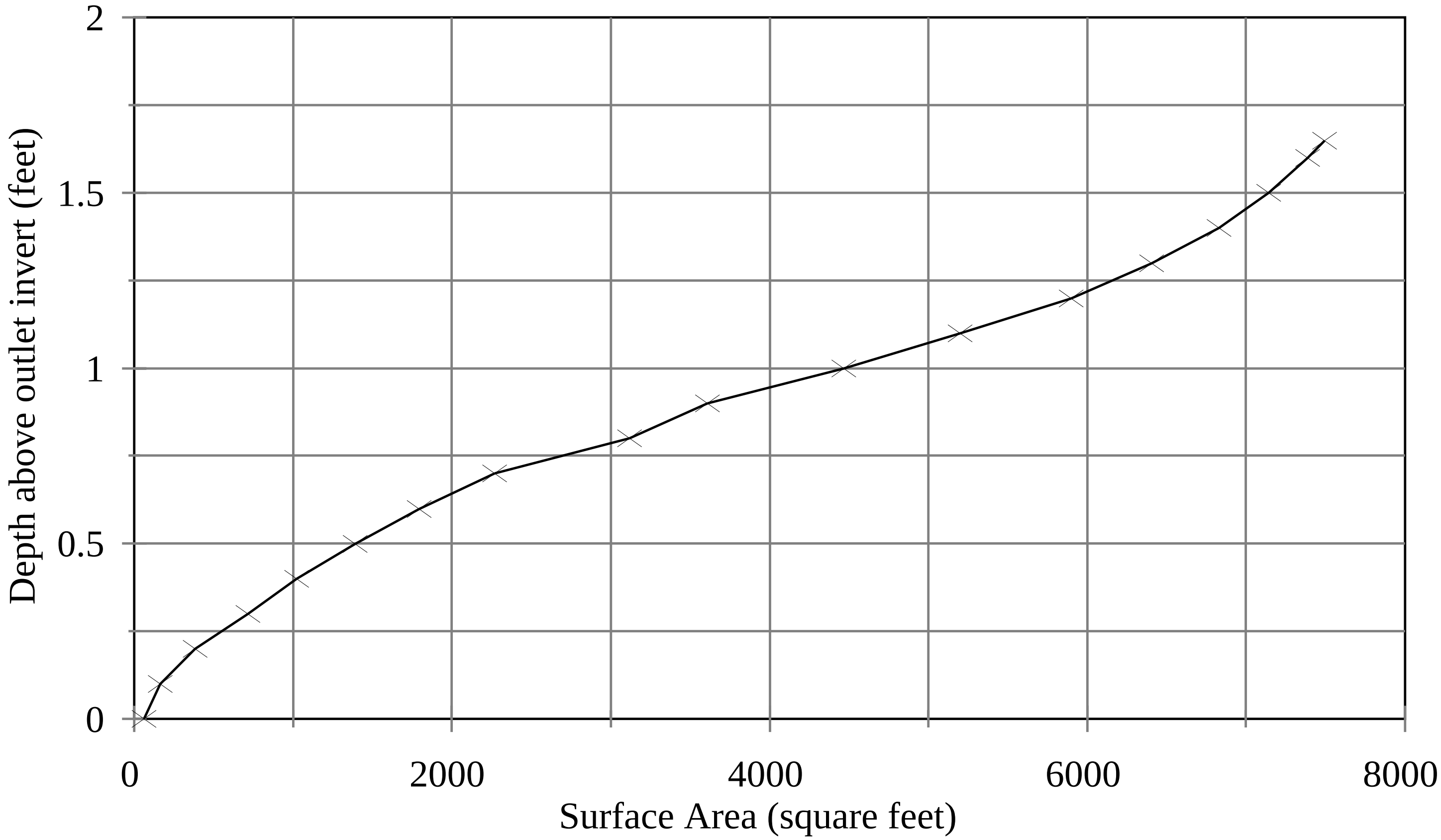


Figure 2. Crestwood wetland BMP - depth vs. area

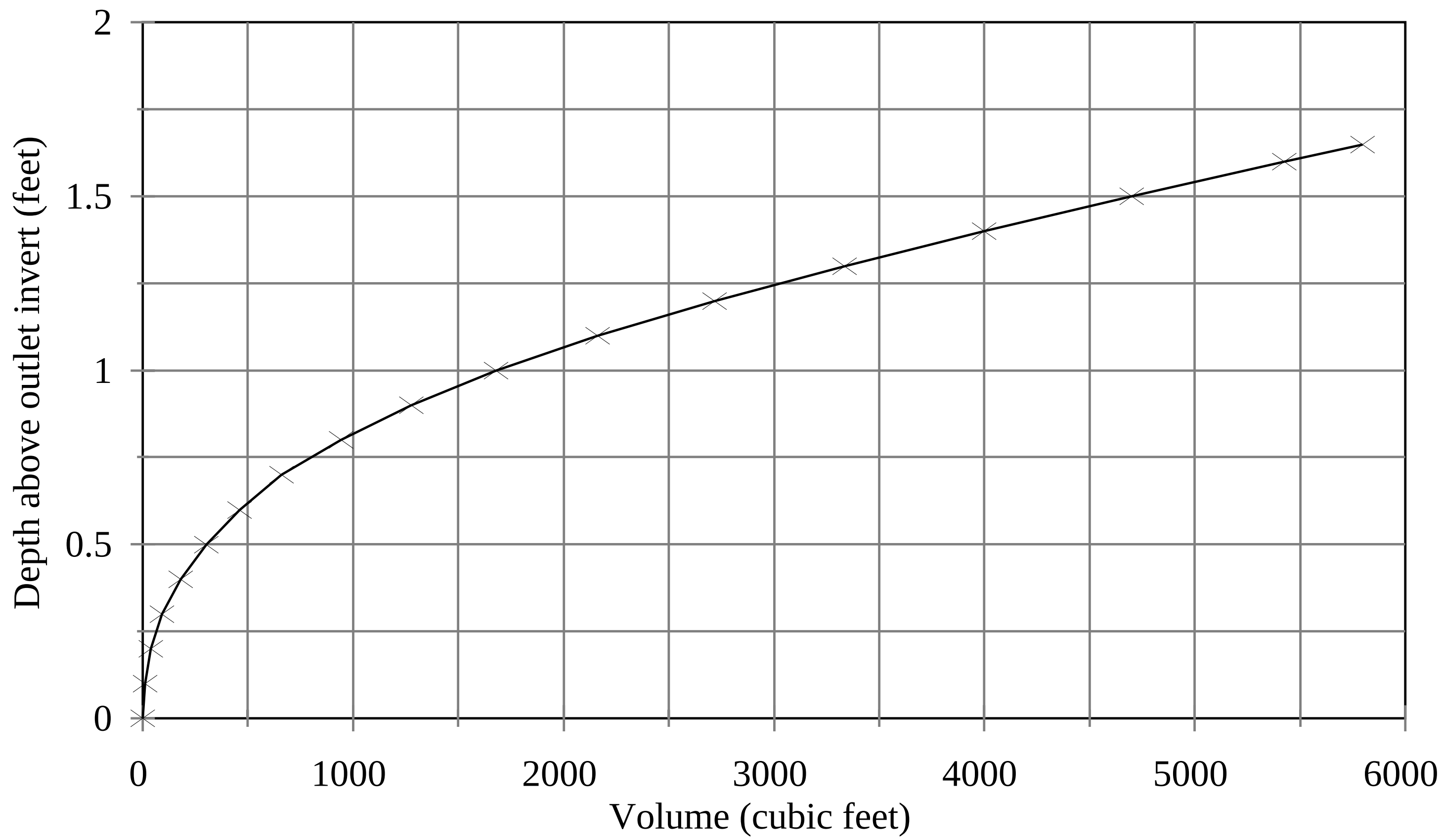


Figure 3. Crestwood wetland BMP - depth vs. volume

pressure transducer inside the flume fed signals to a computer through a Datamite JS+D analog-to-digital (A/D) converter. The computer converted the signal to a stage reading, then calculated the associated discharge from the stage, and a rating curve stored in memory.

Hydrologic Monitoring and Sample Collection

An increase in stage triggered the computer to enter a mode wherein it calculated incremental volume passing through the flume. At set volume intervals, the computer sent a signal via the A/D converter to a peristaltic sampling pump, which withdrew a sample of fixed volume from the water in the pipe. All samples from a given event were composited. In this way, flow-weighted composite samples were collected at the inlet and outlet during storm events. Chemical analyses of these samples yielded estimated EMCs for each storm at inlet and outlet.

Baseflow monitoring was performed by OWML personnel, who collected grab samples from the pool near the outlet, initially on a weekly basis. After several months of monitoring the baseflow data showed no apparent temporal concentration trends, and the frequency of sampling was reduced to approximately every two weeks.

Overland Runoff Grab Samples

Periodically during and following rainfall events, a substantial amount of water was observed to be flowing overland into the facility from the grassed/wooded area adjacent to the pond. A portion of this flowed through a 6-inch diameter polyvinyl chloride (PVC) drainage pipe located outside the perimeter fence around the site. The pipe drained a small depression adjacent to a dirt access road, which was located between the facility and the wooded area. After flow calculations began to suggest that a substantial hydrologic input came from this area, attempts were made to collect samples in order to characterize the chemical nature of the overland input. On three occasions, during runoff events, OWML personnel collected grab samples directly from the PVC drainage pipe (WE15).

Subsequent attempts to establish automated sampling of this source were unsuccessful, therefore a major portion of the hydrologic input (the overland flow) was not well characterized chemically. An unfortunate consequence of this data gap is uncertainty introduced into long-term constituent load removal estimates.

Soil Pore Water Monitoring

Three soil water lysimeters were installed within the facility to allow infiltrating and percolating pore water to be sampled. The lysimeters were constructed from five foot lengths of 2.5-inch diameter PVC pipe (OWML, 1990). Lysimeter installation entailed excavating a three foot hole into the soil and inserting the lysimeter. The hole was then backfilled, and tightly packed with soil to prevent downward leakage of surface water. Two lysimeters (WE50, WE60) were installed within the area of the permanent pool near the outlet of the pond, at depths of 0.5 and 1.0 foot below the bottom of the pool (reference datum 0 ft. above the original outlet invert), respectively. The third lysimeter (WE70) was installed above the permanent pool level in one of the banks of the basin (reference datum 2.5 ft. above the original outlet invert), to a depth of 1.5 feet below the ground level. Lysimeter samples were collected concurrently with baseflow samples, unless a storm was in progress. Approximately 24 hours prior to sample collection, the lysimeters were charged by placing a vacuum of approximately 18 mm Hg on them using a hand-held pump. Samples were retreived by ejecting them under positive pressure using a hand held pump.

Precipitation Measurements

A tipping-bucket rain gage (Weather Measure Corp., model P501) was situated on top of the inlet instrumentation shed at the site. At 0.01 inch increments of rainfall, the bucket would become full and tip, thereby triggering a contact closure signal which was relayed to a solid state totalizing rainfall recorder mounted in the equipment shelter. The recorder accumulated a count of the bucket tips, and recorded incremental rainfall as a function of time.

Wetfall/Dryfall Sampling

A wet/dry precipitation collector (Aerochemetrics, model 301), located on-site within the perimeter fence, and 1.5 meters above the ground, was used to collect atmospheric deposition samples. The collector included separate vessels for wetfall (WE40) and dryfall (WE30) samples. During dry weather periods, the dryfall vessel was exposed to the atmosphere, while a lid covered the wetfall vessel. The beginning of a rainfall event would trigger a signal to move the lid over the dryfall vessel, thus exposing the wetfall vessel to the atmosphere. Wetfall samples were collected after the rainfall event was complete. Dryfall vessels were collected after approximately two weeks of exposure. Sample buckets were transported to and from the field with the lids on and taped down.

In the laboratory, wetfall volumes in the collection bucket were measured and recorded, along with the pH of the sample. Samples were mixed as needed before being transferred to sample bottles for analysis. Dryfall samples were prepared by the removal of gross contamination such as bird droppings or leaves before adding 200 mL of reagent grade water to the bucket. After rubbing down the sides of the bucket with a rubber policeman, the samples were transferred to 250 mL volumetric flasks and diluted to the mark with Milli-Q (Millipore Corporation) water.

Field Measurements

Dissolved oxygen, temperature, pH, total alkalinity, and conductivity in samples were measured in the field by OWML personnel. Conductivity and pH values were subsequently corrected for extant temperature to the equivalent value at 25° C.

Sample Handling and Analysis

Between uses in the field, sample bottles were cleaned by the following procedure:

1. Phosphorus-free detergent hand-wash with hot tap water
2. 15 minute soak in 50% HCl
3. Rinse 3X with RO water
4. Rinse 2X with Milli-Q water
5. Air dry

Samples were transported in iced coolers. Upon receipt at the laboratory, each sample was given a unique identification number for unambiguous in-house identification.

Nutrient analyses were performed by OWML staff using a Bran and Luebbe TrAAcs 800 system, as described in Technicon Methods 780-86T through 787-86T, and modified in the *Quality Assurance Plan for the Occoquan Watershed Monitoring Laboratory* part III. Trace metals were analyzed using a Perkin Elmer model 5100 Atomic Absorption Spectrophotometer as described in either EPA Method 200.9 or Standard Method 3111, and modified in the *Quality Assurance Plan for the Occoquan Watershed Monitoring Laboratory* part III. Metals, TSS, COD, turbidity, total petroleum hydrocarbons (TPH), pH, and total hardness were analyzed according to published APHA and EPA methods (APHA, 1992; USEPA, 1979).

Some constituents, notably TPH and both extractable and soluble forms of Al, were highly left censored, that is, many measurements were below the analytical detection limit (LOD). Measurements falling below the LOD were substituted with one half the LOD. When simultaneous measurements of related constituents gave occasional incongruous results (*e.g.* OP>TP, SKN>TKN, etc.), the higher value was substituted for the more inclusive parameter (*e.g.* OP value substituted for TP value).

Pollutant Removal Performance Calculations

System efficiency was estimated using seven different calculation methods. These are described as follows:

- The average EMC reduction (AVE) is based on the mean % EMC reduction over all monitored storms.

$$AVE = Average\ of\ All: \left[\frac{\sum(EMC_i - EMC_o)}{EMC_i} \right] \times 100$$

- The median EMC reduction (MED) is based on the median % EMC reduction over all monitored storms.

$$MED = Median\ of\ All: \left[\frac{\sum(EMC_i - EMC_o)}{EMC_i} \right] \times 100$$

- The average load reduction (AOL) is based on the mean % load reduction over allmonitored storms.

$$AOL = Average\ of\ All: \left[\frac{\sum(Load_i - Load_o)}{Load_i} \right] \times 100$$

- The median load reduction (MOL) is based on the median % load reduction over all monitored storms.

$$MOL = Median\ of\ All: \left[\frac{\sum(Load_i - Load_o)}{Load_i} \right] \times 100$$

• The summation of loads (SOL) is based on the difference between the sum of the inlet and outlet storm loads over all monitored storms. The summation of loads for subset A (SOL_A) is based on the difference between the sum of the inlet and outlet storm loads over the subset A storms (#1,5,6,8,13) only.

$$SOL = \left[1 - \frac{\sum(V_o EMC_o)}{\sum(V_i EMC_i)} \right] \text{x } 100$$

• The long term efficiency (LTE) is based on the difference between the estimated total input (inlet + overland + wetfall + dryfall) and estimated total output (storms + baseflow) loads, over all monitored storms.

$$LTE = \left[1 - \frac{[\sum(V_o EMC_o)] + V_B FWA_B}{[\sum(V_i EMC_i)] + V_R MC_R + WF + DF} \right] \text{x } 100$$

where: V_B = total baseflow volume,

FWA_B = flow-weighted average baseflow concentration,

V_R = estimated total overland flow volume,

MC_R = median overland grab sample concentration,

WF = direct wetfall loading to pool,

DF = direct dryfall loading to pool

Statistical Analysis

Seasonal trends in load removal efficiencies, as measured by paired inlet and outlet storm loads only (overland loads not included), were examined using Statistical Analysis System 6.02 (SAS, Cary, N.C.) software. Because analyses for metals were not performed for all storms, there was insufficient data to examine seasonal trends in metal removals (*e.g.* only one event analyzed for some seasons), therefore statistical analysis was performed only for nutrients, COD, TSS, and turbidity. Data was analyzed using analysis of variance (ANOVA). Because the ANOVA test is not designed to handle data sets containing both positive and negative values, removal percentages were subject to an additive transformation (a large positive number was added to each removal percentage) to convert all values to positive. For each constituent, removals were then grouped according to season (spring, summer, fall, winter, spring2). Constituents which met the required assumptions of normality and homogeneity of variance (OX_N, TN) were analyzed using ANOVA. Constituents which did not meet these assumptions (OP, total soluble phosphorus (TSP), NH_3, SKN, TKN, COD, TSS, TURB) were subject to logarithmic transformation, and retested for normality and homogeneity. Constituents which still failed the tests for homogeneity and/or normality (all constituents), were then rank transformed and retested. Rank-transformed constituents which met the homogeneity and normality assumptions (all except TSP) were then subjected to ANOVA. Constituents for which the ANOVA test identified significant seasonal differences were subject to pairwise t-tests with a Bonferroni adjustment to control for the cumulative experimentwise error rate (α =0.05), in order to identify which seasons differed significantly from each other. Because the TSP data could not be normalized, a separate analysis was done using the nonparametric Wilcoxon rank sum test, which does not require the data to be normally distributed.

IV. Results

Flow Balance

Monitored storms were found to exhibit substantial discrepancies between inlet and outlet flow volumes. Figure 4 displays a bar chart comparison between the inlet and outlet flow volumes for each storm. Figure 5 displays the same information in x-y graph form. Linear regression of storm outlet volume *vs.* inlet volume resulted in an r^2 value of 0.94 and a slope of 1.39, suggesting that the outlet storm volume was typically about 39% greater than the inlet volume. The excess outlet volume was found to be significantly related ($p<0.05$) to the inlet volume (Figure 6), however the percent excess volume over inlet volume was not (Figure 7). In order to obtain a satisfactory flow balance, subsequent calculations used the rational method with a runoff coefficient of 0.2 to estimate the overland flow volume. This approach resulted in only 3% difference between estimated long-term total inflow and outflow (Table 6). This method was less effective at accounting for the volume discrepancies of individual storms (Figure 8).

Atmospheric Contributions

Wetfall/dryfall monitoring was performed throughout the study, making it possible to evaluate areal mass depositions, and to compare them to other sources of constituent loadings to the wetland. Calculated annual wetfall and dryfall deposition rates (lb/acre/yr) are presented in Table 7, along with rates from similar studies previously conducted in the greater Washington, D.C. area and the southeast. In general, atmospheric loadings of nutrients and Zn at the Crestwood wetland were comparable to these earlier studies. However, loadings of Cu and Cd were 1 to 2 orders of magnitude lower. Wetfall loadings of Al appeared to be about an order of magnitude greater than those at Franklin Farms, while dryfall loadings were about an order of magnitude lower. However, these results for Al should be interpreted cautiously, because the data were highly censored, *i.e.* most of the results fell below the analytical detection limit (Figures A.26, A.27).

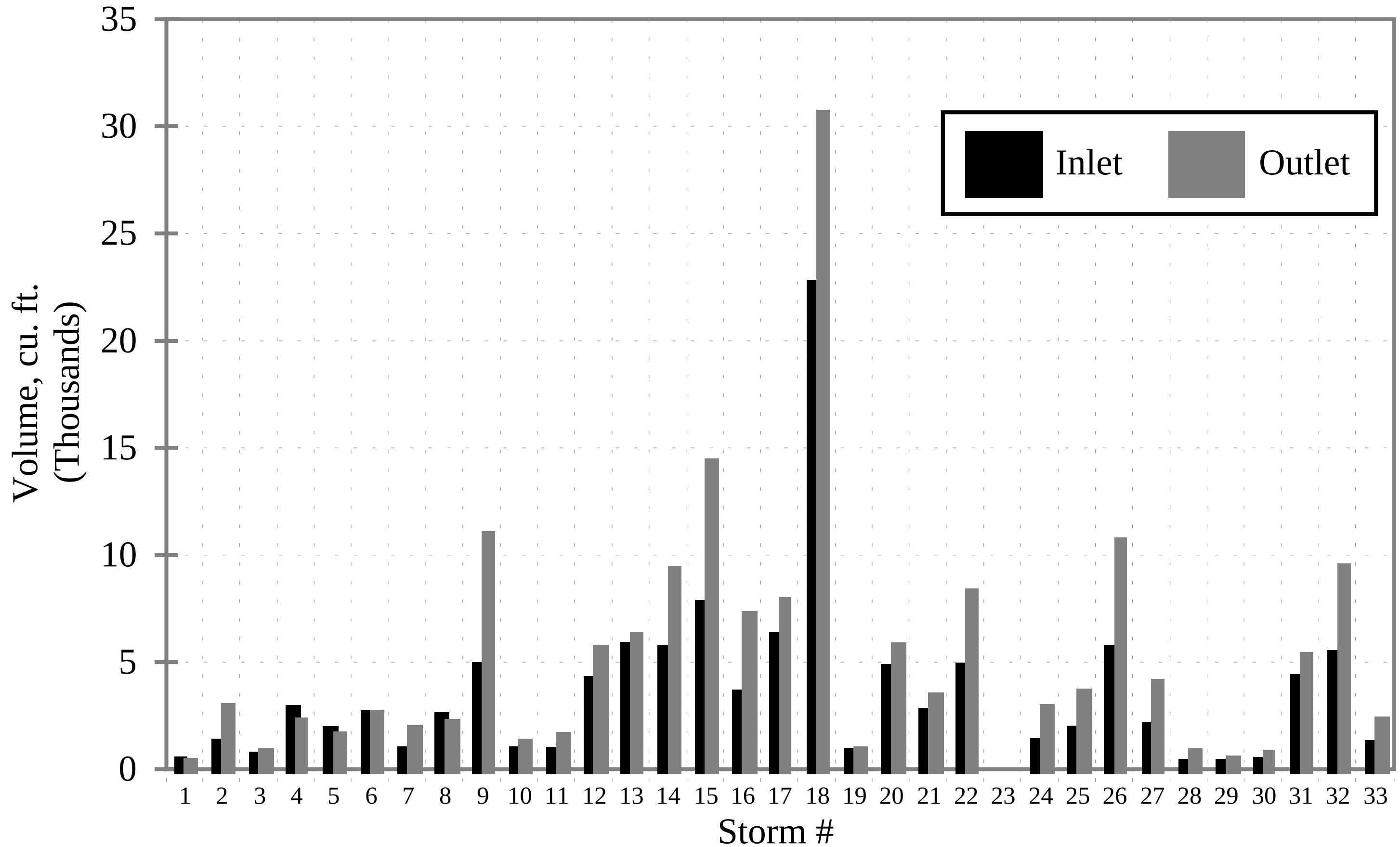


Figure 4. Storm volume comparison, inlet and outlet

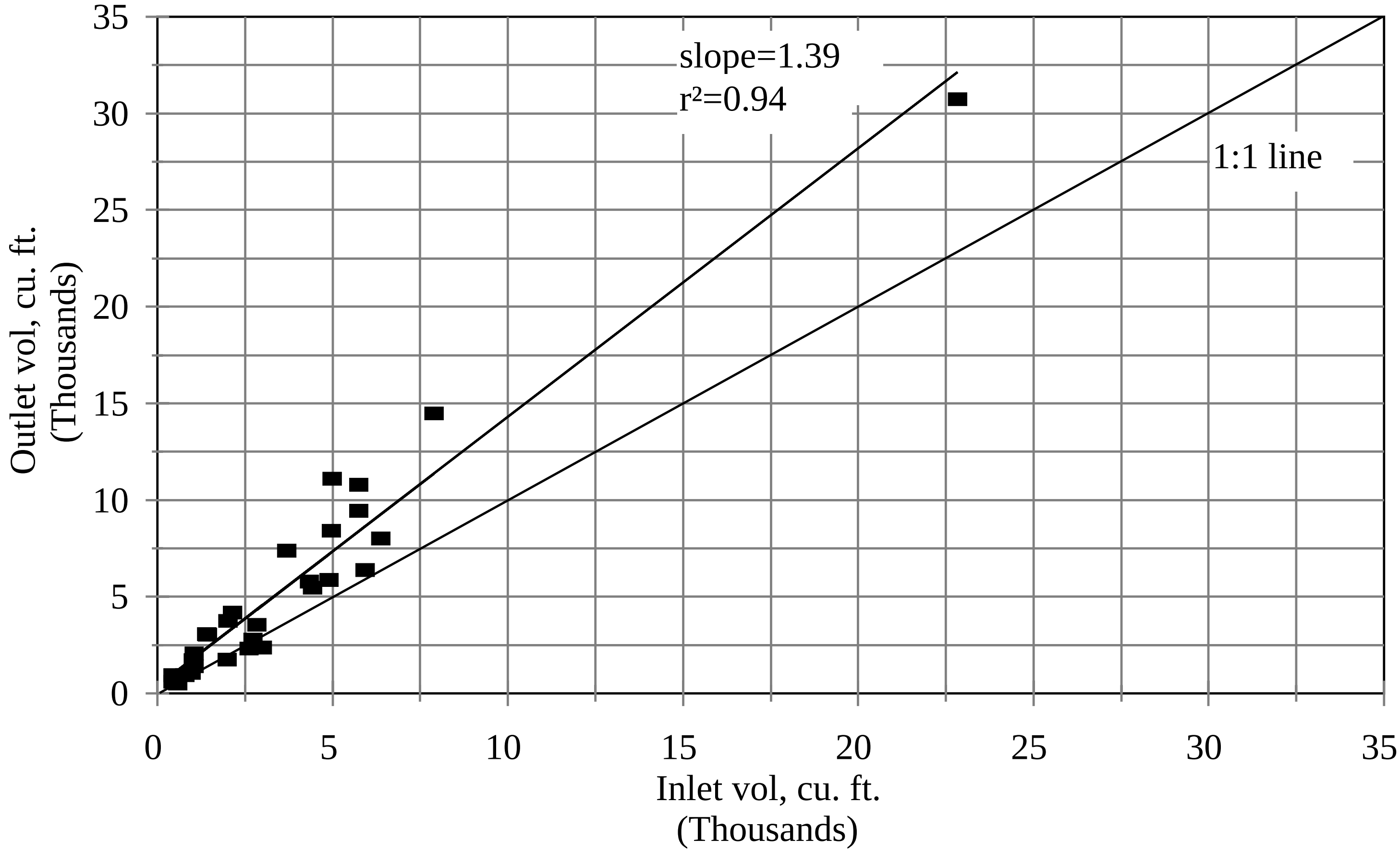


Figure 5. Storm volumes: inlet vs. outlet, and least squares fit

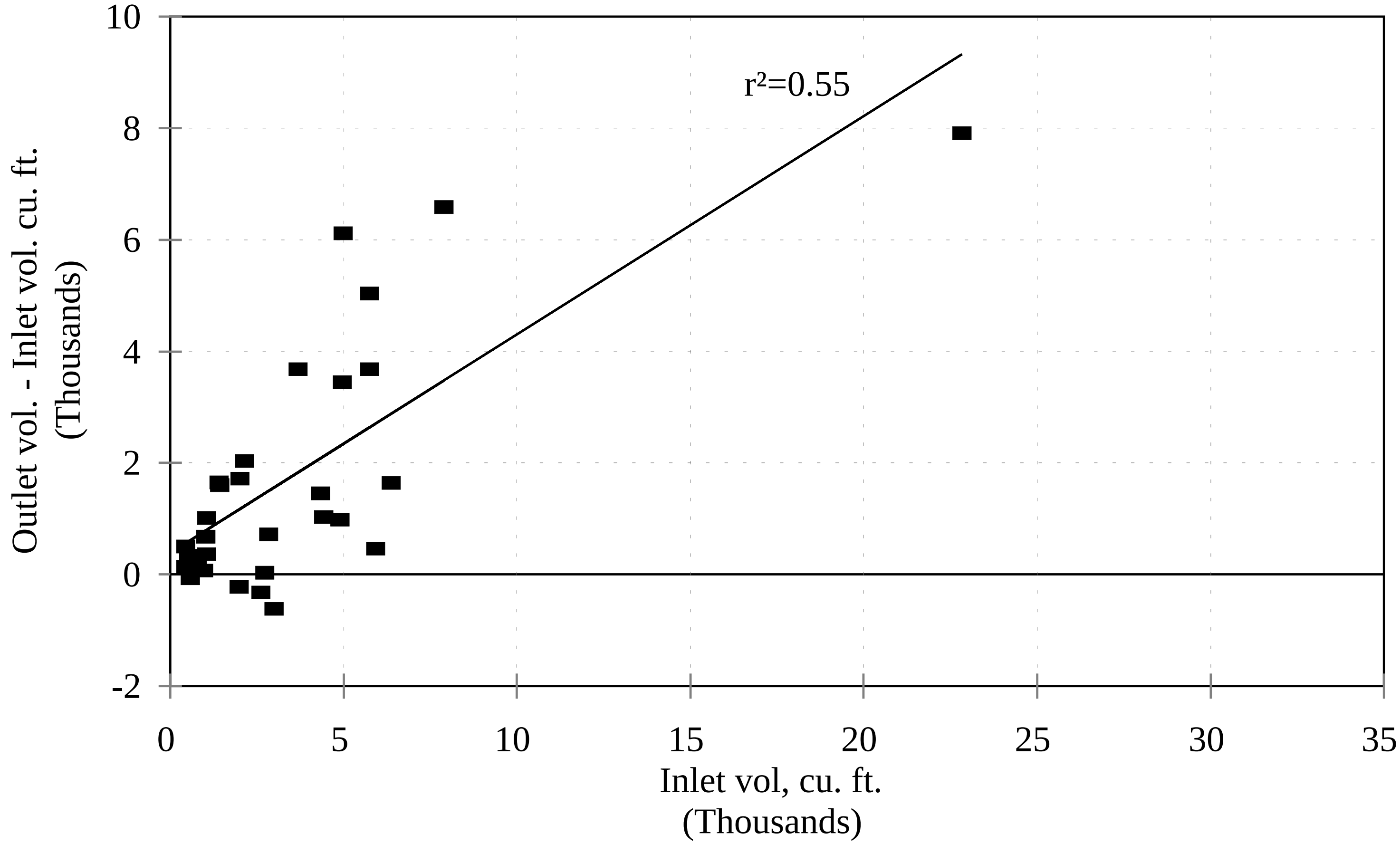


Figure 6. Storm volume difference vs. inlet volume, and least squares fit

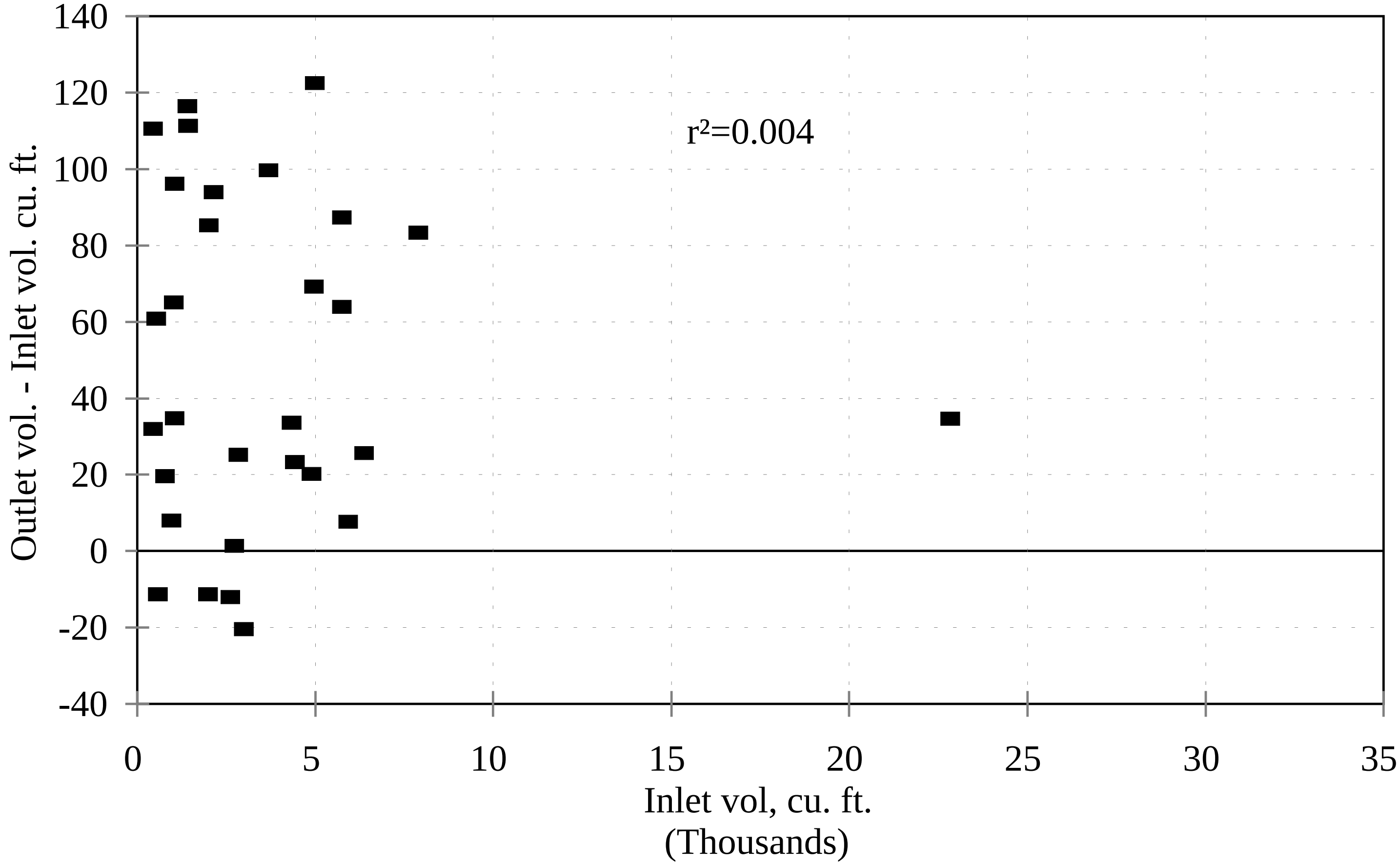


Figure 7. Percent storm volume difference vs. inlet volume

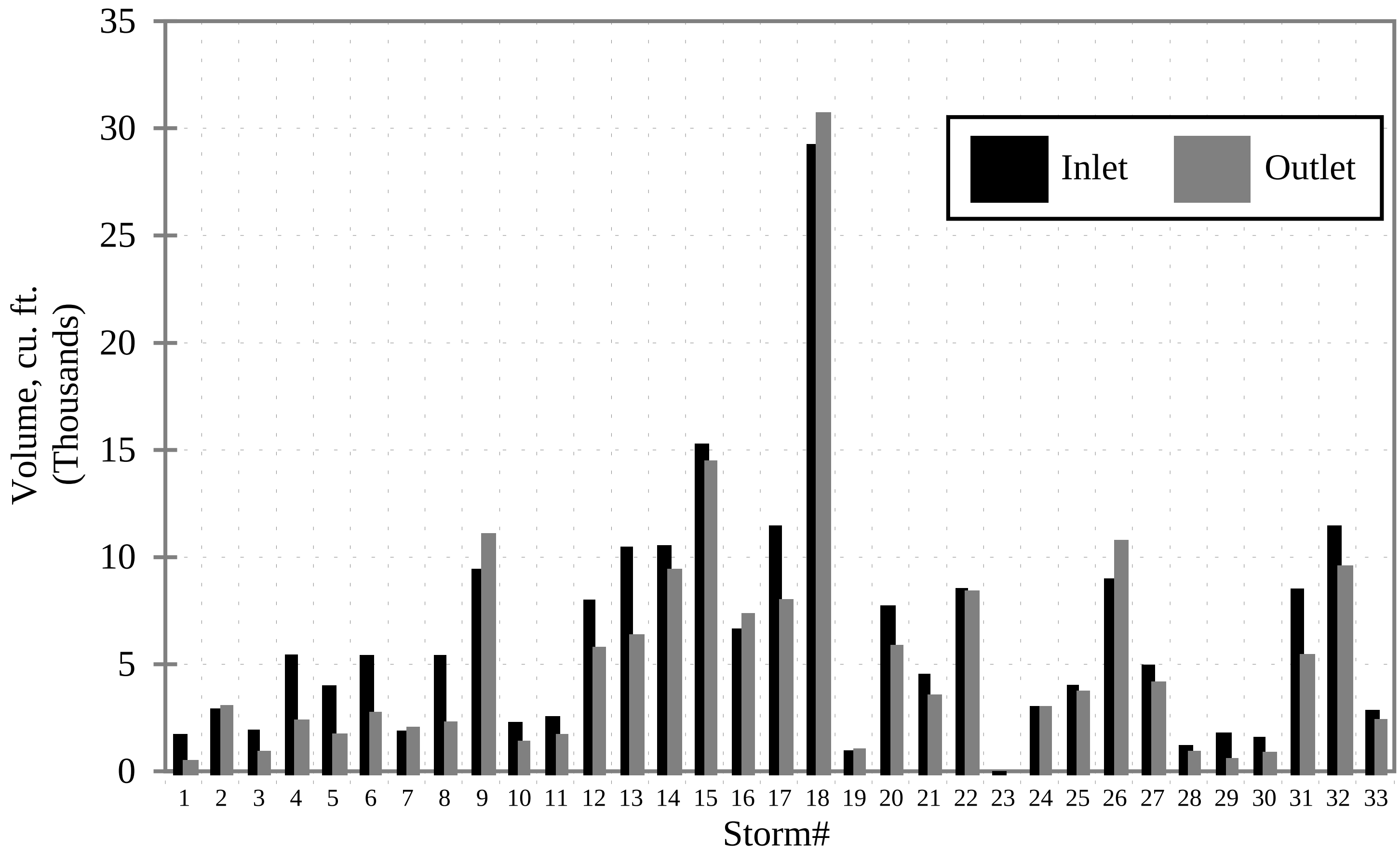


Figure 8. Inlet (including estimated overland) and outlet storm volumes for monitored storms

Table 6. Crestwood flow balance

4/28/96-12/28/96, Including all storms, except during periods with flow data gaps:

Inputs		Outputs		% difference
Gaged inlet	151710	Gaged outlet, total	271020	
Estimated overland	116820	Evapotranspiration	3410	
Direct precipitation	9010			
Total	**277540**	**Total**	**274430**	**-1.1**

4/28/96-12/28/96, Including only sampled storms:

Inputs		Outputs		% difference
Gaged inlet	91910	Gaged outlet, storms	131420	
Estimated overland	64690	Gaged outlet, baseflow	30120	
Direct precipitation	9010	Evapotranspiration	3410	
Total	**165610**		**164950**	**-0.4**

4/28/96-6/1/97, Including only sampled storms:

Inputs		Outputs		% difference
Gaged inlet	116110	Gaged outlet, storms	173190	
Estimated overland	89010	Gaged outlet, baseflow	32040	
Direct precipitation	12530	Evapotranspiration	5800	
Total	**217650**		**211030**	**-3.0**

Table 7. Estimated Annual Atmospheric Deposition Rates (lb/Ac/yr): Wetfall + Dryfall

Nutrients:

Location	Loading type	OX_N	NH_3_N	TKN	TN	OP	TSP	TP
SE U.S. (TN)*	Total	6.74	2.25					
Suburban D.C. **	Total	5.70	1.07	7.3	12.9	0.27		0.51
Franklin Farms	wetfall	3.34	2.93	3.35	5.93	0.11	0.14	0.18
	dryfall	0.91	0.52	1.94	3.09	0.07	0.09	0.34
	Total	4.25	3.45	5.29	9.02	0.18	0.23	0.52
Crestwood (current study)	wetfall	4.94	3.60	4.15	9.19	0.09		0.13
4/29/96-4/29/97	dryfall	0.73	0.42	1.78	2.55	0.07		0.21
	Total	5.67	4.02	5.93	11.74	0.16		0.34

Trace Metals:

Location		TCu	TPb	TZn	TCd	TAl
		lb/A/yr	lb/A/yr	lb/A/yr	lb/A/yr	
SE U.S. (TN)*	Total					
Suburban D.C. **	Total	0.21	0.44	1.35	0.09	
Franklin Farms	wetfall	0.12		0.48	0.07	1.24
	dryfall	0.73		1.51	0.21	70.8
	Total	0.85		1.99	0.28	72
Crestwood (current study)	wetfall	0.02	0.02	0.6	0.01	11.69
4/29/96-4/29/97	dryfall	0.01	0.01	0.62	0.002	5.08
	Total	0.03	0.03	1.22	0.01	16.77

*** Lindberg et al. (1986)**
**** MD Dept. of Envt. (1987b)**

Consistent with the Franklin Farms study, wetfall nitrogen loadings were substantially higher than dryfall loadings, by approximately an order of magnitude, while wetfall and dryfall phosphorus depositions were more similar. Wetfall and dryfall Zn loadings were also similar. For all other metals, observed wetfall loadings were about 2 to 5 times higher than dryfall loadings. Time series plots of wetfall and dryfall areal loadings are presented in Appendix A.

Characterization of Runoff

For a lognormally distributed variable, the population median is equal to the geometric mean (Burmaster, 1997). Because pollutant concentrations in runoff have been found to be lognormally distributed (EPA, 1983), the sample median is typically used to describe the central tendency of runoff concentration data. Accordingly, Table 8 presents median inlet (WE20) EMCs, wetfall sample (WE40), and ungaged area (WE15) grab sample concentrations of the various consituents measured. Median urban runoff concentrations from other studies in the Washington, D.C. metropolitan area are presented as well, for comparative purposes. The table shows that for nutrients, TSS, COD, and all metals except Pb, the WE20 samples at the Crestwood site were similar to urban runoff concentrations measured in the other studies, though generally somewhat lower.

While ammonia and nitrate concentrations in the forested drainage (WE15) were lower, TKN, TP, TCu, and TPb were substantially higher than in the townhouse runoff (WE20). Only three ungaged samples were collected, as grab samples, whereas flow-weighted composites were collected at WE20, thereby substantially improving the precision of measurements at WE20 relative to WE15. Table 9 presents the three WE15 sample results, along with results from other local studies on constituent concentrations in overland drainage from forested areas. The Crestwood concentrations were consistent with those from the other studies for most constituents. The WE15 TKN and TP results, while higher than median concentrations in the townhouse drainage, were within the range of values reported in the other forested runoff studies. Table 9 shows that TPb

Table 8. Urban Stormwater Runoff - Median Concentration Comparisons

Nutrients and Physical Parameters:											
	OP	TSP	TP	NH _N	SKN	TKN	OX_N	TN	TS§	TURB	COD
Site	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	NTU	mg/L
Crestwood, WE15	0.06	0.09	0.17	0.03	0.64	1.22	0.10	1.31	32	80	47.1
Crestwood, WE20	0.04	0.05	0.14	0.13	0.37	0.81	0.56	1.40	37	19	45.5
Crestwood, WE40	0.005		0.005	0.31	0.50	0.35	0.34	0.74			
Franklin Farms, UM40	0.17	0.16	0.35			1.44	1.22	2.66			
NE Washington DC, HL30*	0.06		0.37	0.11		2.24	0.46		43	13	53.2
Townhouse/Garden Apts., DC area 208†	0.08		0.43	0.12		2.08	0.51	2.67	106.3		88.3
Townhouse/Garden Apts., DC area NURP†	0.14		0.35	0.34		1.62	0.70	2.40	32.8		43.8
NURP urban site overall‡		0.12	0.33			1.50	0.68	2.18††	100		65

Trace Metals:										
	EAl	SAl	ECd	SCd	ECu	SCu	EPb	SPb	EZn	SZn
Site	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l	µg/l
Crestwood, WE15	2100	1000	0.5	0.5	12	6.3	4.3	1.5	39	24
Crestwood, WE20	1000	1000	1.20	0.5	7.60	2.80	1.50	1.50	61	33.0
Crestwood, WE40	1000		0.5		1		1.5		23	
Franklin Farms, UM40										
NE Washington DC, HL30*			1		18.1		14.8		212	
Townhouse/Garden Apts., DC area 208†			5		22		334		148	
Townhouse/Garden Apts., DC area NURP†					29		151		74	
NURP urban site overall‡					34		144		160	

* Rabanal *et al.*, 1995

† Washington Metropolitan Area Urban Runoff Demonstration Project - NVPDC, 1983

‡ Results of the NURP Vol. I - Final Report

†† Inferred value: sum of median TKN and median OX_N

Table 9. Concentrations in "forested" runoff

	Nutrients and Physical Parameters:									
	OP	TSP	TP	NH _N	SKN	TKN	OX_N	TN	TSS§	TURB
Site	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	mg/L	NTU
Norman Forest*	0.004		0.17	0.08		1.48	0.02	1.50	290	
Norman Forest mean†	0.01	0.12	0.36	0.10		2.82	0.02	2.84	621	
Norman Forest flow-wt. mean†	0.02	0.15	0.65	0.14		4.33	0.03	4.36		
NURP forest values‡	0.02		0.15	0.07		0.61	0.17	0.78		
Crestwood: WE15-12/31/96	0.05	0.05	0.11	0.02	0.33	0.87	0.10	0.97	32	80
Crestwood: WE15-3/3/97	0.06	0.12	0.23	0.04	0.95	1.56	0.09	1.65	70	120
Crestwood: WE15-4/28/97	0.08	0.15	0.23	0.03	1.3	1.52	0.02	1.54	27	80

	Trace Metals:									
	TAl	SAl	TCd	SCd	TCu	SCu	TPb	SPb	TZn	Szn
Site	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L	µg/L
Norman Forest*			1				16		72	
Norman Forest mean†							100		108	83
Norman Forest flow-wt. mean†										
NURP forest values‡										
Crestwood: WE15-12/31/96										
Crestwood: WE15-3/3/97	2100	1000	0.5	0.5	12	6.3	4.3	1.5	39	24
Crestwood: WE15-4/28/97										

*As cited in: Land Use/Runoff Quality Relationships in the Washington Metro Area (1978)

† Numbers provided by Harry Post, 1996

‡ Numbers cited by Schueler, 1987

concentrations in the WE15 samples, while higher than in the WE20 samples, were lower than concentrations reported in the other forested runoff studies. Figures 9 and 10 show that for most events, total hardness and conductivity were higher in outlet than in inlet samples.

Lysimeter Samples

Appendix B presents time series plots of the results from lysimeter monitoring. Median values of all constituents for each of the three lysimeters are presented in Table 10. Concentrations of OP and TP in each sample were similar, with both frequently falling below detection limits. Ammonia and TKN concentrations were generally highest in the WE50 samples, followed by WE60 and WE70, respectively. In comparison with other nitrogen species, nitrate concentrations in the three lysimeters

Table 10. Lysimeter Median Values

Constituent	WE50	WE60	WE70
OP, mg/L	0.005	0.005	0.01
TSP, mg/L	0.01	0.005	0.015
TP, mg/L	0.01	0.005	0.01
NH_3_N, mg/L	0.1	0.04	0.005
SKN, mg/L	0.75	0.22	0.03
TKN, mg/L	0.61	0.3	0.02
OX_N, mg/L	0.12	0.1	0.085
TN, mg/L	0.73	0.43	0.11
SCd, µg/L	0.5	0.5	0.5
SCu, µg/L	1	1	1
SPb, µg/L	1.5	1.5	1.5
SZn, µg/L	20	17	7.5
SAl, µg/L	1000	1000	1000

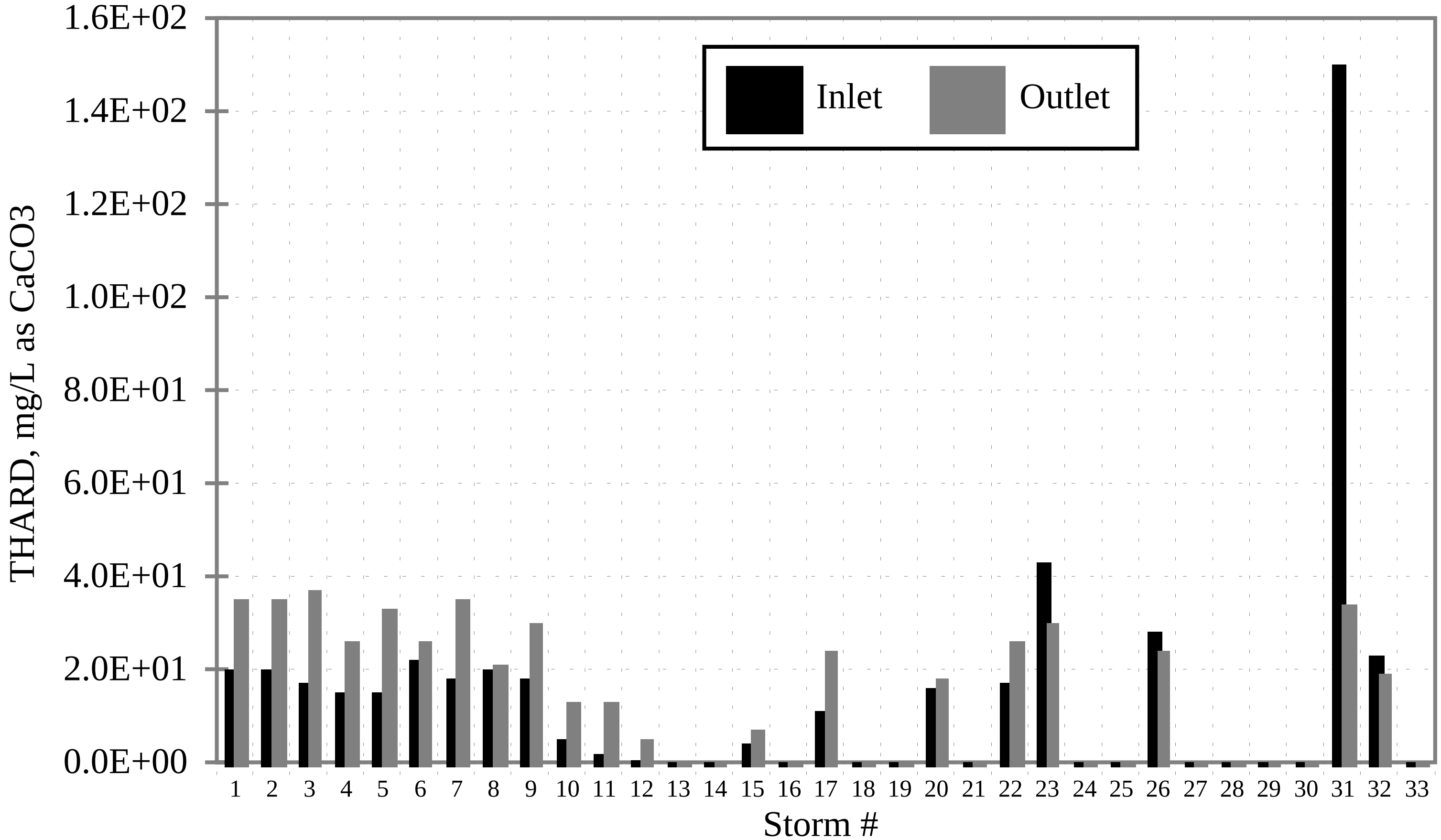


Figure 9. Inlet-outlet EMC comparison: total hardness

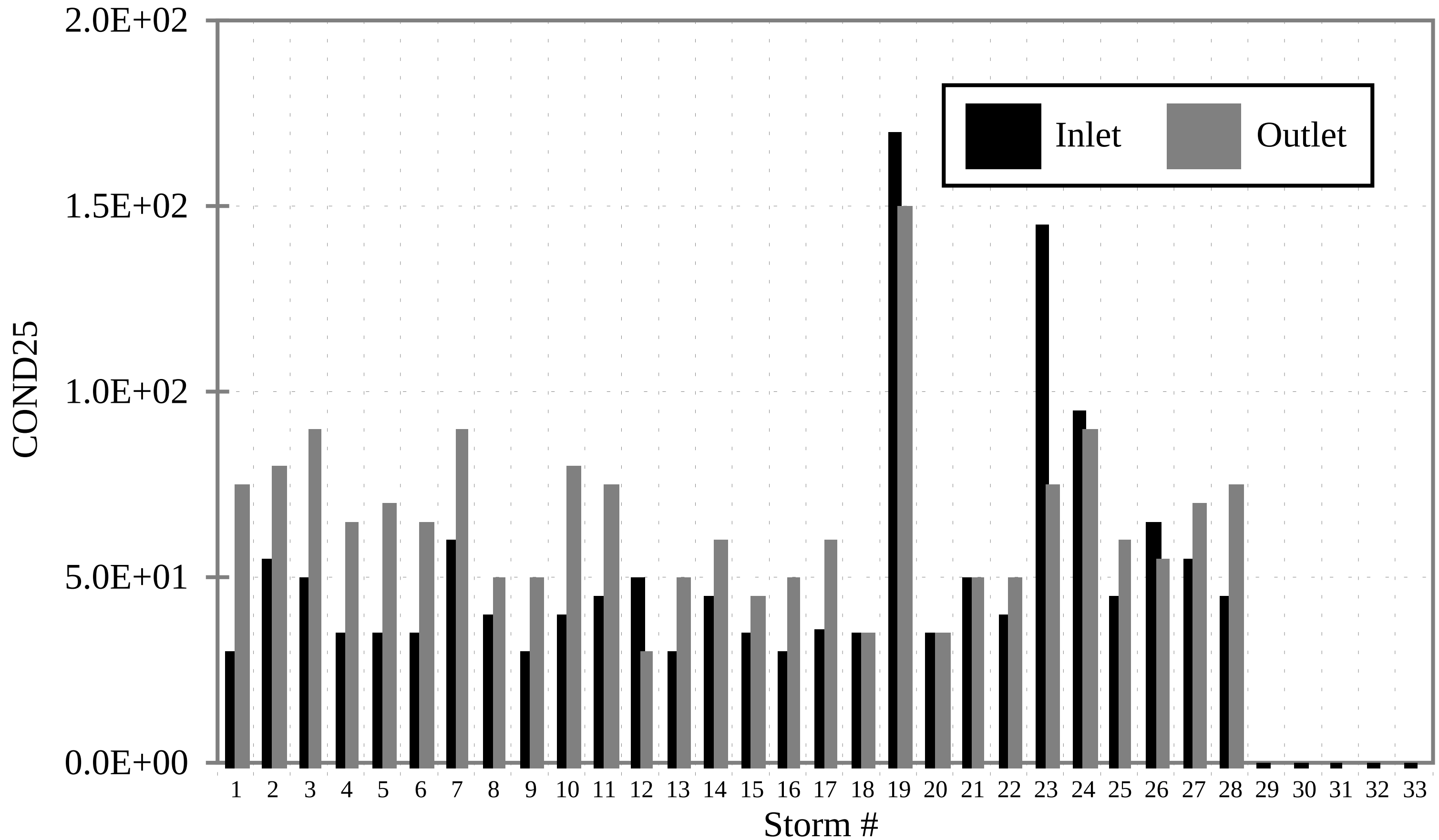


Figure 10. Inlet-outlet EMC comparison: conductivity at 25° C

appear to follow each other very closely. The Al, Cd, and Pb results fell mostly below detection limits. By contrast, concentrations of Zn and Cu were observed more often at detectable levels.

Outlet Baseflow Monitoring

The Crestwood wetland received only intermittent hydrologic inputs following rainfall events, because of the relatively small drainage area. There was no inlet baseflow, nor any perennial outlet baseflow, because all hydrologic outputs also were ultimately the result of intermittent storms. Nevertheless, because of the sometimes long outlet hydrographs, a fraction of the outflow typically occurred after the monitoring equipment had ceased collecting samples. For purposes of analysis this was considered to be outlet baseflow. For the mass balance and removal efficiency calculations, it was important to be able to account for the hydrologic and pollutant loads exported during these time periods. For this reason, outlet grab samples were regularly collected, as previously described, to characterize this portion of the total outflow.

Time series comparison plots of baseflow and outlet storm EMCs are presented in Appendix C. The baseflow sample concentrations appeared to follow the storm EMCs, though at slightly lower concentrations. In the case of nitrate (Figure C.7) however, the storm EMCs were consistently several fold higher than the baseflow concentrations.

Plots of baseflow concentrations as a function of the instantaneous outlet flow at sampling time, are also presented in Appendix C. The plots show an inverse relationship between concentration and flow rate for most constituents. This relationship between flow and concentration was accounted for in the long-term mass removal calculations by calculating the flow-weighted mean baseflow concentration for each constituent, together with the total baseflow volume, thereby estimating the total baseflow export.

Pollutant Removal Performance

Pollutant removal performance was calculated using the seven methods described in the previous chapter (Table 11). The two methods which focused exclusively on inlet and outlet concentrations (AVE and MED) showed positive removals for most constituents, with the exceptions of SKN, TKN, turbidity, and COD. However, this apparent improvement in water quality between the inlet and outlet was offset by greater outlet storm volumes, resulting in much lower load removal efficiencies (AOL, MOL, and SOL calculations). In fact, according to these calculations, most constituents exhibited a net export from the facility.

For another calculation, a subset of storms was identified for which inlet and outlet volumes differed by less than 12%, herein referred to as "subset A" (Table D.1). The difference between the total summed inlet and outlet volumes for these five storms was less than one percent. The summed load removal efficiencies calculated for this subset (SOL_A) were substantially better than the efficiencies calculated across all storms, and for some constituents were better than efficiencies based only on EMCs, which are shown in Table 11.

All of the calculation methods discussed thus far focused exclusively on constituent concentrations or loads passing through the gaged inlet and outlet, and entering the wetland through direct wetfall and dryfall. However, as previously mentioned, flow balance calculations suggested that as much as 41% of the total volume entering the wetland was attributable to ungaged overland flow from the wooded/grassed area adjacent to the facility. Thus, in order to conduct an accurate pollutant mass balance, it was neccessary to account in some way for the pollutant loading contributed by this extra hydrologic input. Calculations which ignored this factor tended to show poor or negative removals, as is shown in Table 11. Therefore, a seventh method for calculating removal performance was derived which used the median concentration for each constituent in the WE15 grab samples to represent all ungaged overland runoff entering the wetland, the

Table 11. Crestwood Wetland Removal Efficiencies, 4/96-5/97

Constituent	LTE	SOL	SOLa	AVE	MED	AOL	MOL
OP	35.8	-8.1	31.0	23.9	35.4	-6.4	-11.9
TSP	48.1	-10.1	55.0	16.7	20.0	-22.8	-11.9
TP	45.9	-1.1	71.1	14.8	33.3	-15.5	-0.3
NH_3_N	54.7	52.9	68.9	44.1	68.8	24.9	62.7
SKN	19.8	-90.8	-25.1	-47.4	-26.6	-128.1	-64.4
TKN	25.5	-55.0	7.3	-27.2	-3.1	-85.6	-50.0
OX_N	39.4	34.7	65.1	54.5	61.7	39.4	40.8
TN	21.7	-18.3	27.2	13.4	21.9	-23.8	-24.9
TURB	56.4	-83.3	43.0	-121.7	-16.1	-195.6	-63.9
TSS	57.9	37.2	75.5	28.9	57.9	2.2	49.6
COD	21.9	11.1	-62.8	-126.2	-21.0	-174.6	-65.3
TPH	-110.1	-90.2	21.4	-1.2	0.0	-53.2	-33.6
TAl	68.4	-12.8	59.6	10.5	0.0	-25.7	-27.6
SAl	45.5	-24.7	37.1	2.8	0.0	-31.6	-27.6
TCd	30.8	4.9	90.7	-190.0	50.0	-92.1	41.5
SCd	28.0	-15.2	82.4	-99.6	0.0	-53.2	-27.6
TCu	65.5	1.8	19.3	5.0	0.0	-40.2	1.0
SCu	47.7	-98.8	-23.2	-60.4	-22.2	-131.8	-68.7
TPb	74.7	36.4	67.4	25.8	0.0	1.1	10.6
SPb	33.2	-33.3	16.8	2.8	0.0	-30.8	-26.6
TZn	29.2	-8.5	5.4	8.9	23.4	-14.0	-17.8
SZn	35.5	-20.9	45.7	10.5	11.1	-15.1	14.3

LTE = long-term efficiency
SOL = sum of loads, all storms
SOLa = sum of loads, subset A storms (#1,5,6,8,13)
AVE = mean EMC reduction
MED = median EMC reduction
AOL = mean load reduction
MOL = median load reduction

estimated long-term efficiency (LTE). This method also was the only one which accounted for baseflow mass exports from the wetland, and resulted in estimated net retention for all constituents except TPH. For many constituents, this method gave the highest performance of the seven calculation methods employed.

A comparison of contributions to input and output loadings in the LTE calculations demonstrates that for many constituents, the estimated overland load was a major fraction of the total input, in some cases exceeding even the gaged inlet load, as shown in Table 12, and Figures 11 through 17. For these constituents (TSP, TP, SKN, TKN, TAl, SAl, TCu, SCu, TPb, SPb, and SZn) inclusion of the overland input makes the difference between apparent net retention and net export, but greatly increases the uncertainty of the result.

Nitrogen

Ammonia and nitrate displayed net retention by all the calculation methods used. Ammonia removals in particular were among the highest for any constituent, by all calculation methods. However, while ammonia removals were consistently high, Kjeldahl nitrogen, especially soluble Kjeldahl nitrogen (SKN) displayed low or negative removals via most methods. Total nitrogen and COD removals were also low as calculated by the LTE estimate, and low or negative by the other methods. The percentage of TN in the form of TKN generally increased between inlet and outlet samples (median values were 58.8% and 79.8% at the inlet and outlet, respectively) (Figure 18).

Phosphorus

All three forms of phosphorus (OP, TSP, TP) displayed the same general trend. AVE and MED efficiencies were positive, though less than 40%, while AOL, MOL, and SOL efficiencies were all negative. However, subset A and LTE removal efficiencies were all positive. The percentage of TP in the form of TSP varied widely, but was generally

Table 12. Breakdown of total input and output loads, 4/96-5/97

	Inputs, g					Outputs, g		
	Gaged inlet	Wetfall	Dryfall	Overland	Total	Storms	Baseflow	Total
OP	174.8	1.4	0.9	151.2	328.3	190.4	20.4	210.8
TSP	223.6	1.4	0.9	302.4	528.3	247.7	26.3	274.0
TP	492.8	2.7	2.5	579.7	1077.6	501.1	81.7	582.8
NH3_N	458.0	50.1	3.9	75.6	587.6	239.2	26.7	265.9
SKN	1376.5			2394.4	3770.9	2627.0	395.4	3022.4
TKN	2479.5	61.4	19.6	3831.0	6391.5	3937.8	826.6	4764.4
OX_N	1689.0	70.5	7.1	226.8	1993.5	1149.4	58.1	1207.5
TN	4168.5	131.9	26.7	3301.7	7628.9	5087.2	884.8	5971.9
TURB	60230.5			201633.1	261863.5	102129.8	9962.5	112092.4
TSS	125365.8			80653.2	206019.1	78673.4	8102.8	86776.2
COD	600702.8			118711.5	719414.3	533991.4	27531.5	561522.9
TPH	1816.5				1816.5	3305.5	345.2	3650.6
TAl	1873.0	69.6	49.0	5292.9	7284.4	2191.3	110.0	2301.2
SAl	1687.3	69.6		2520.4	4277.3	2191.3	140.0	2331.2
TCd	2.9	0.043	0.0	1.3	4.2	2.8	0.1	2.9
SCd	1.7	0.043		1.3	3.0	2.1	0.1	2.2
TCu	14.8	0.167	0.1	30.2	45.4	14.7	0.9	15.6
SCu	4.7	0.167		15.9	20.8	9.7	1.1	10.9
TPb	6.1	0.161	0.1	10.8	17.1	4.0	0.4	4.3
SPb	3.0	0.161		3.8	7.0	4.2	0.4	4.6
TZn	110.3	5.1	5.5	60.5	181.4	125.1	3.4	128.5
SZn	56.4	5.1		60.5	122.0	74.4	4.3	78.7

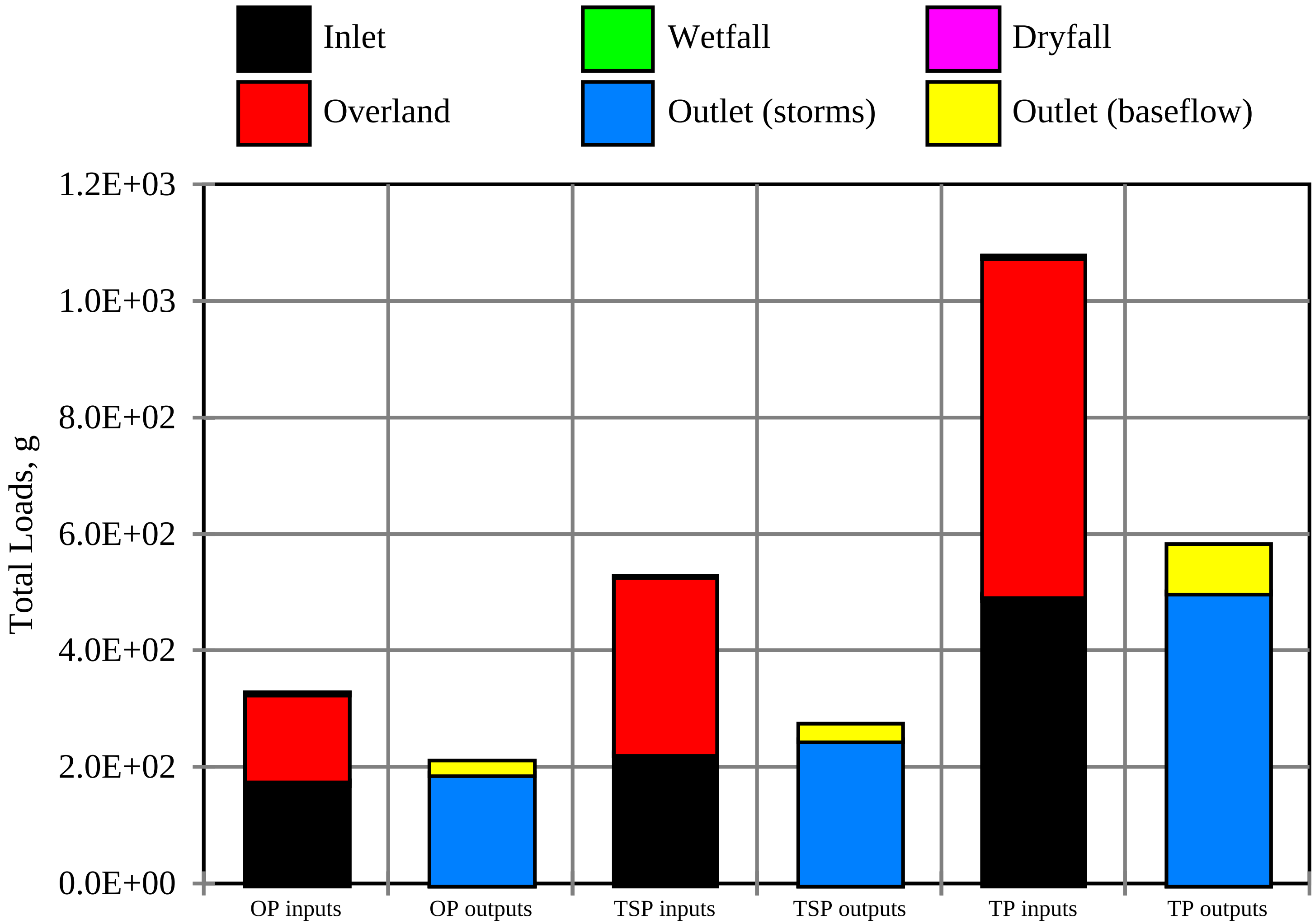


Figure 11. Phosphorus total mass inputs and outputs for all monitored storms

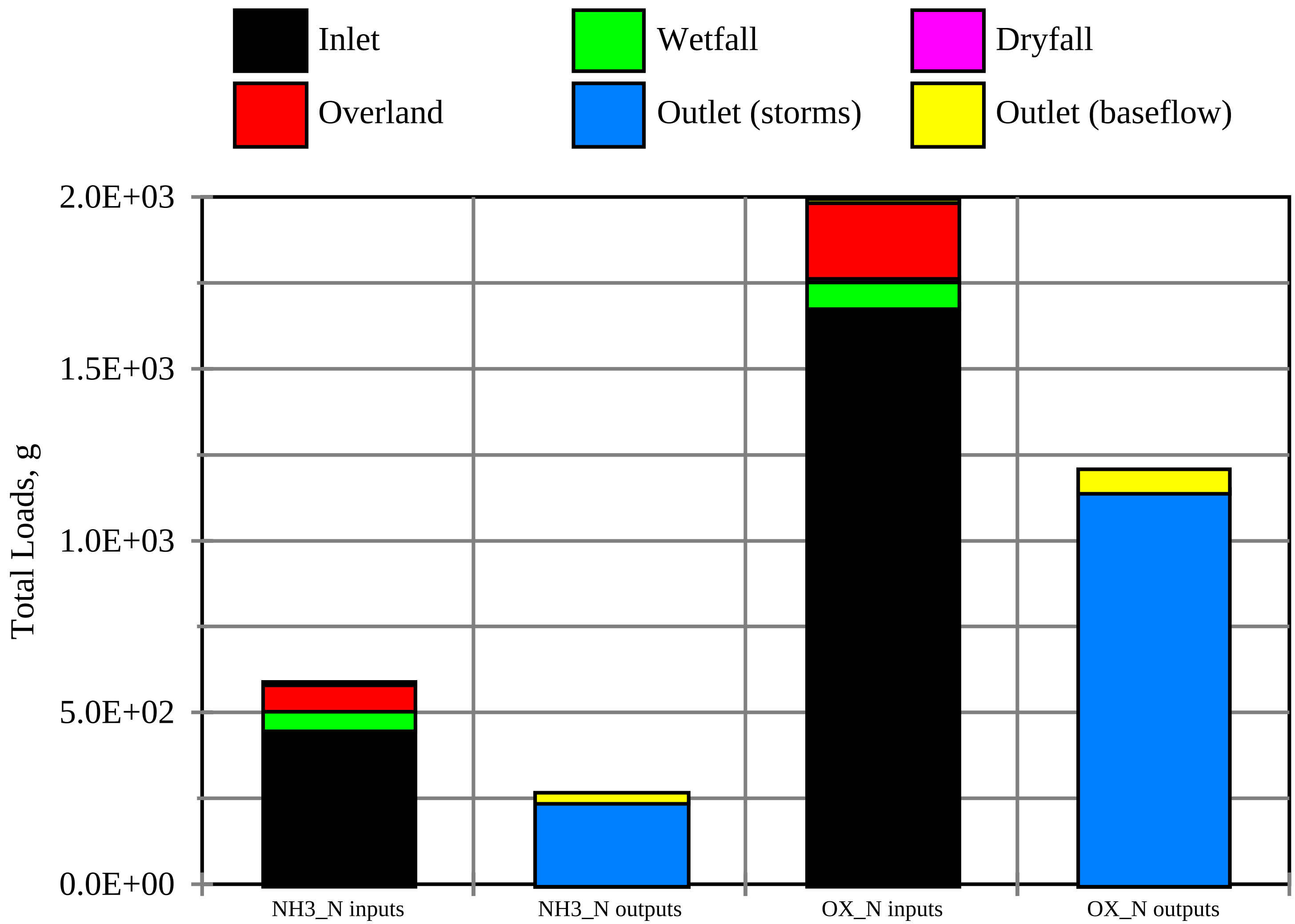


Figure 12. Ammonia and oxidized nitrogen: total mass inputs and outputs for all monitored storms

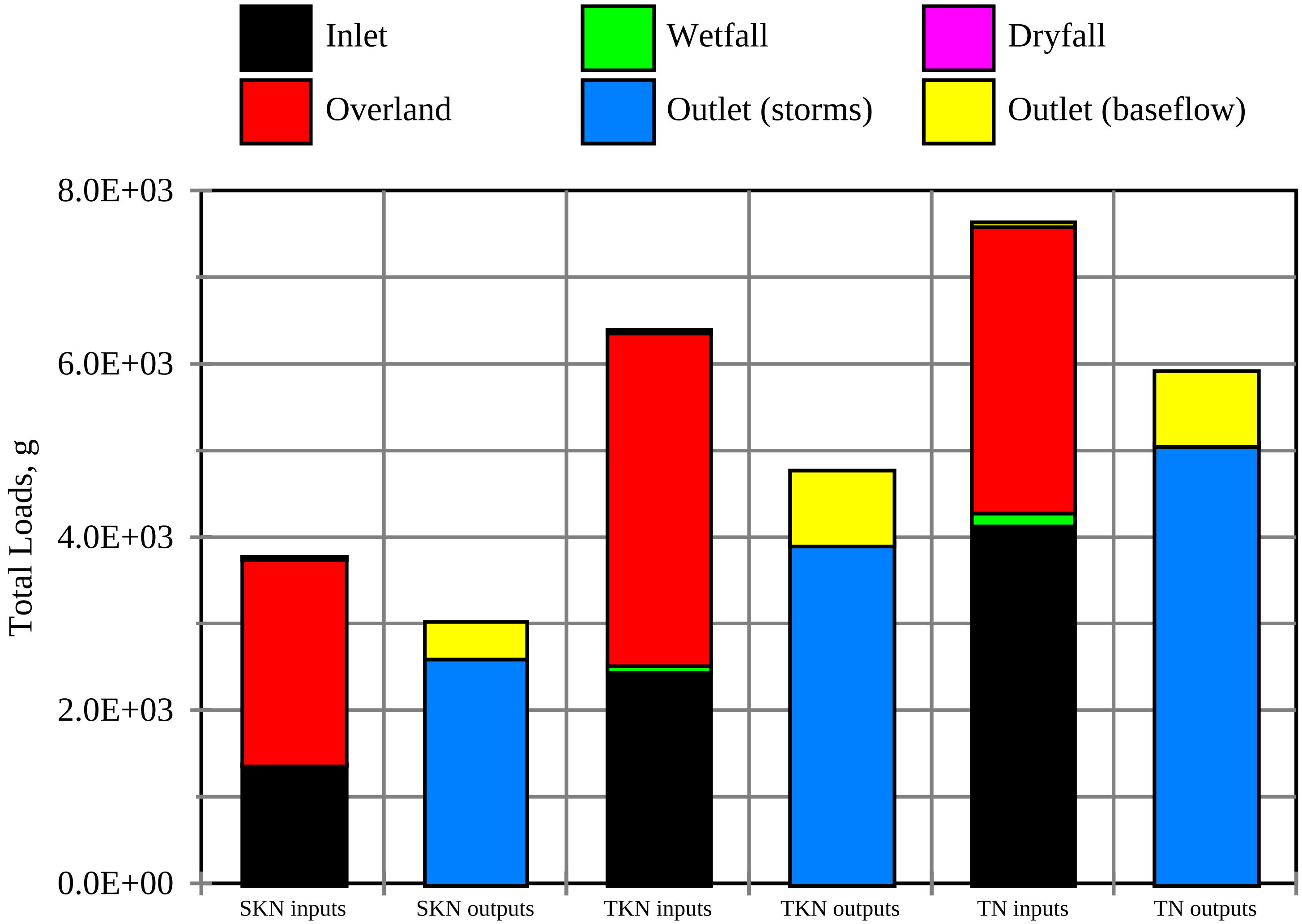


Figure 13. Kjeldahl and total nitrogen: total mass inputs and outputs for all monitored storms

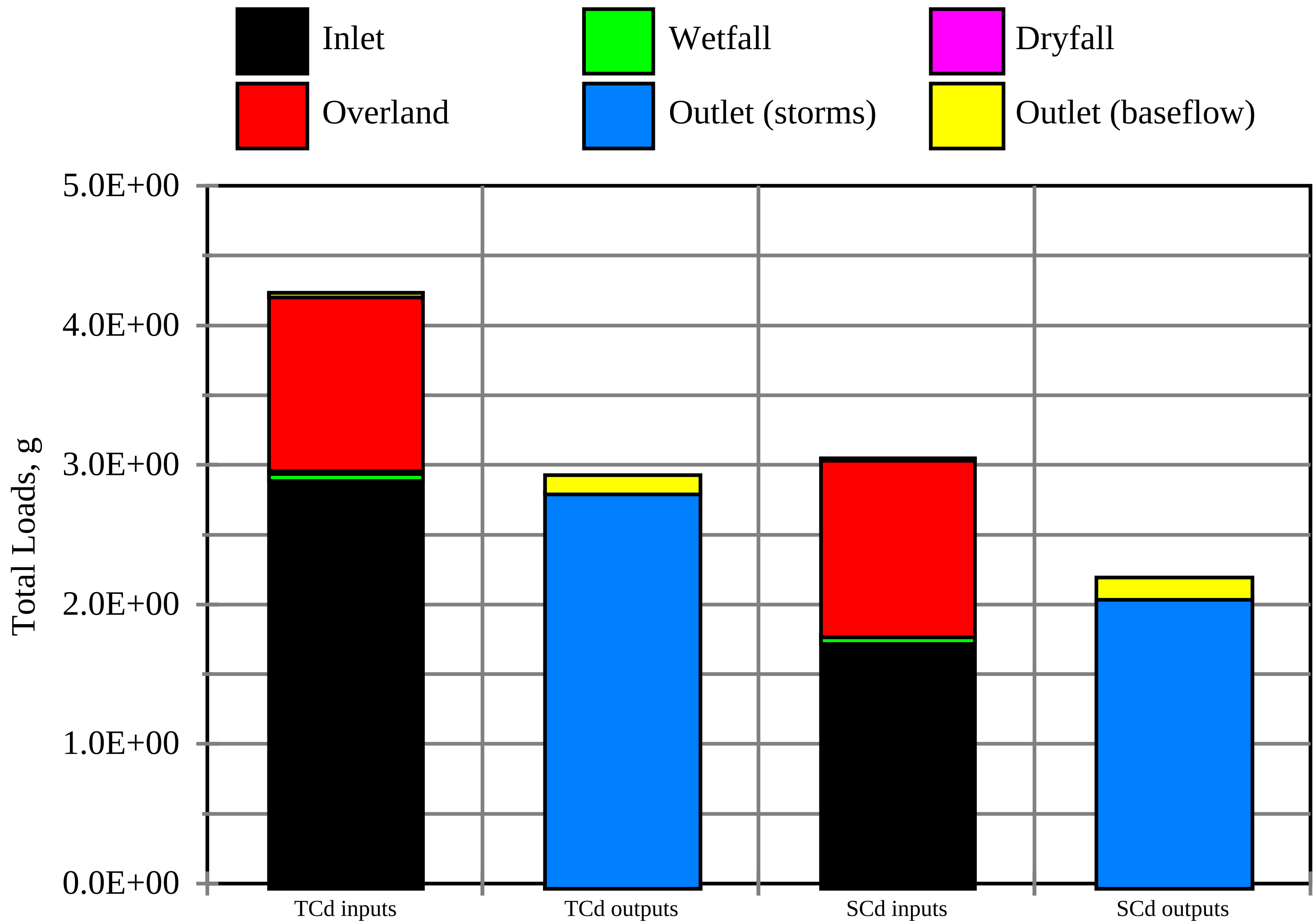


Figure 14. TCd and SCd: total mass inputs and outputs for all monitored storms

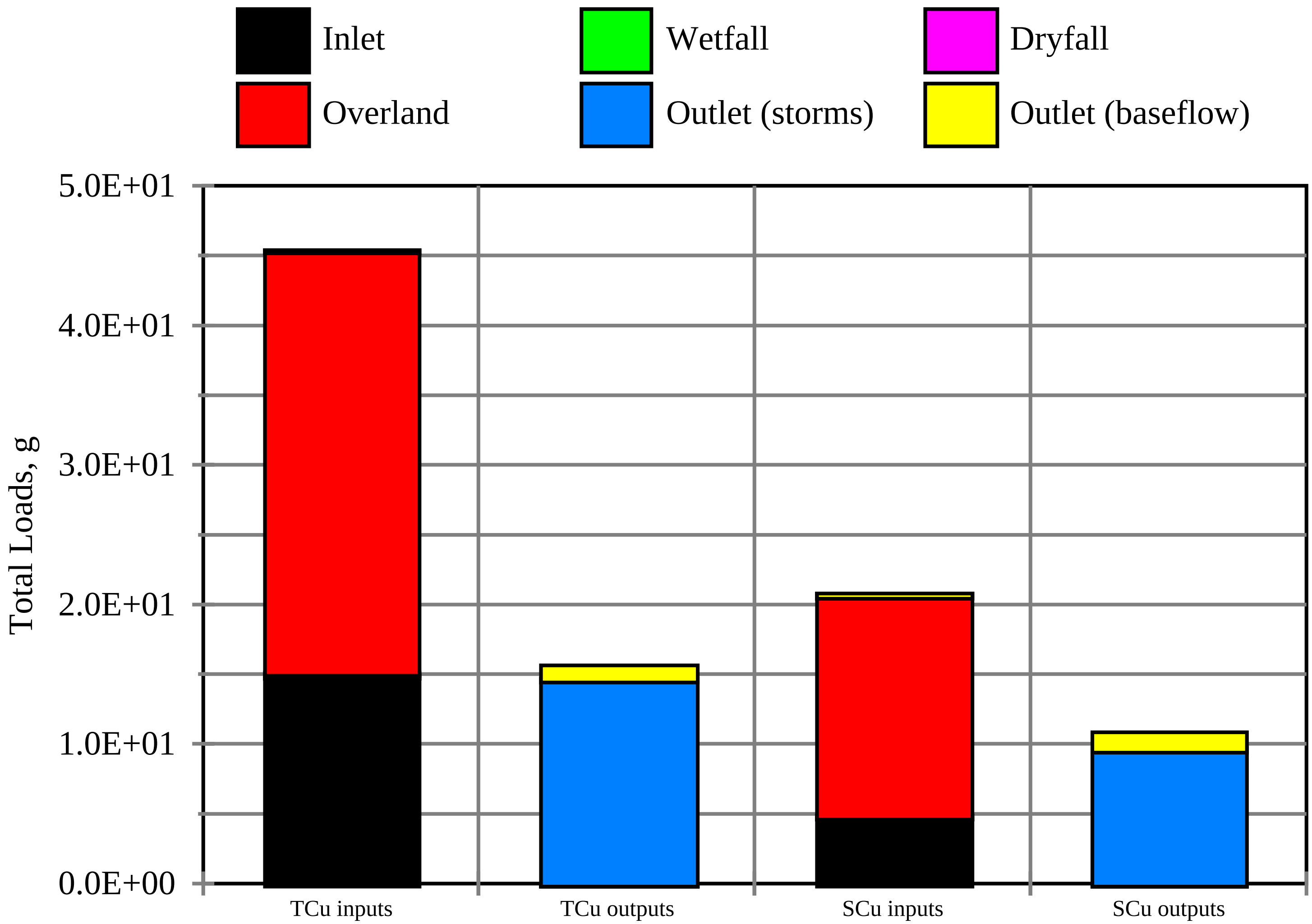


Figure 15. TCu and SCu: total mass inputs and outputs for all monitored storms

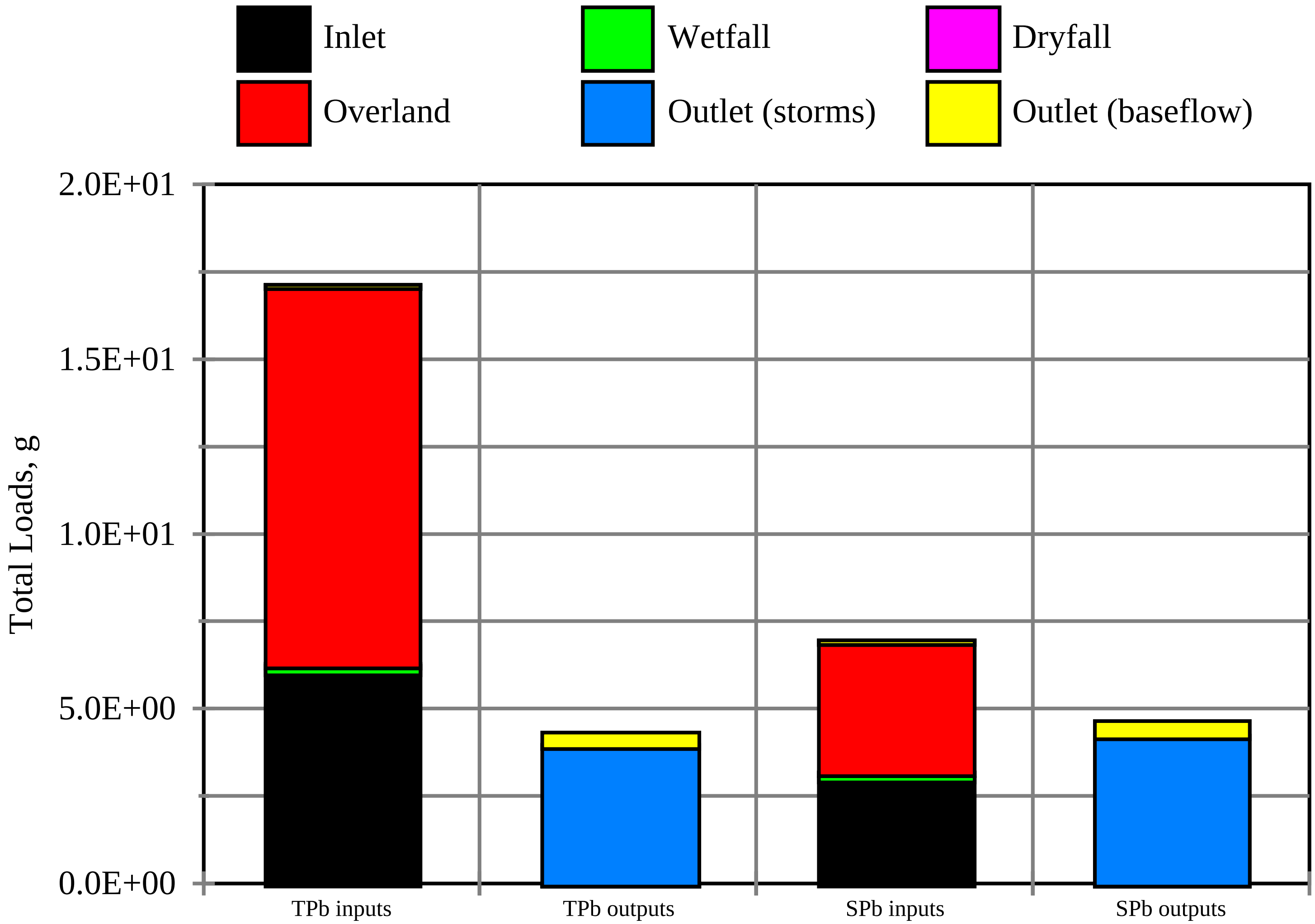


Figure 16. TPb and SPb: total mass inputs and outputs for all monitored storms

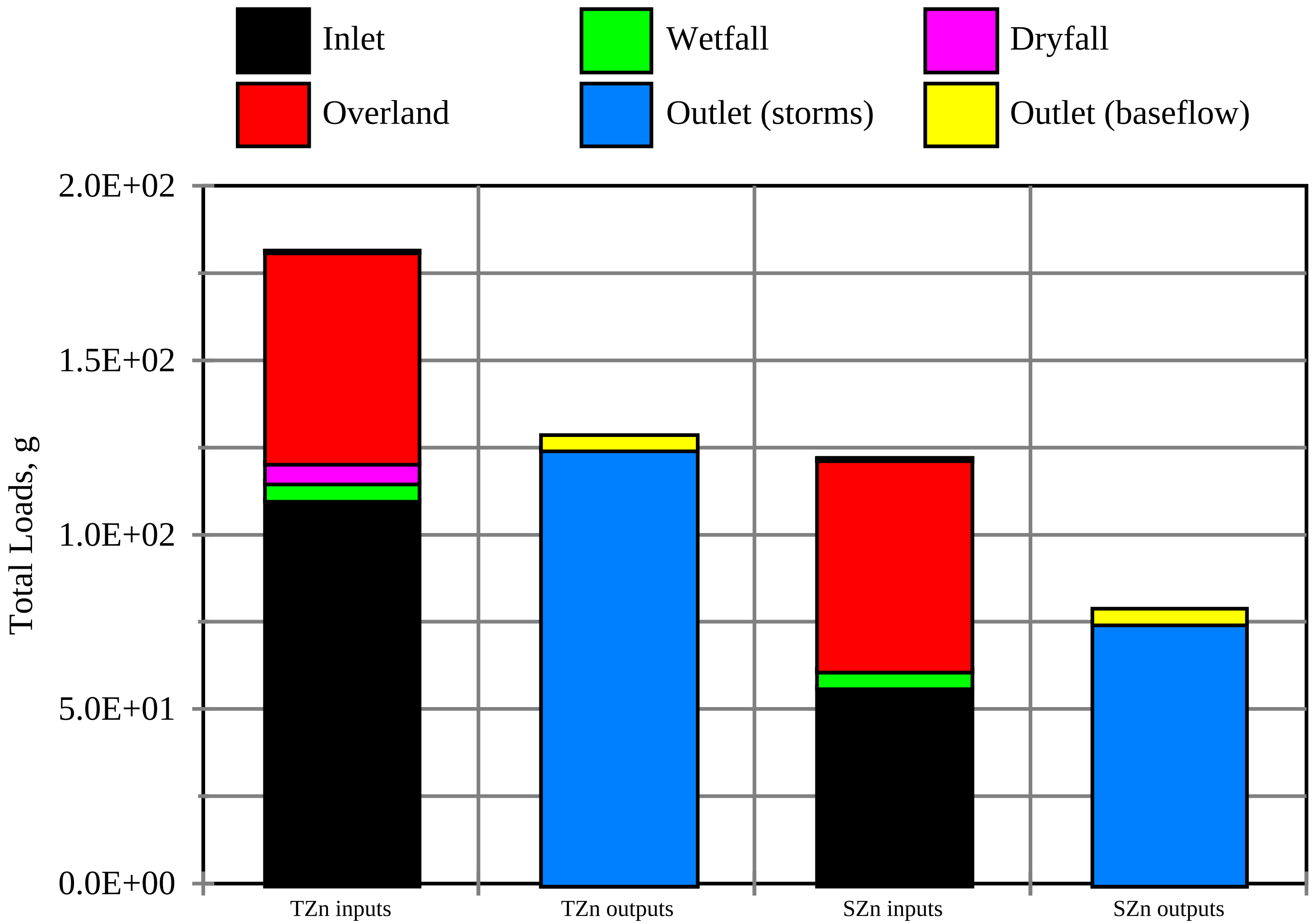


Figure 17. TZn and SZn: total mass inputs and outputs for all monitored storms

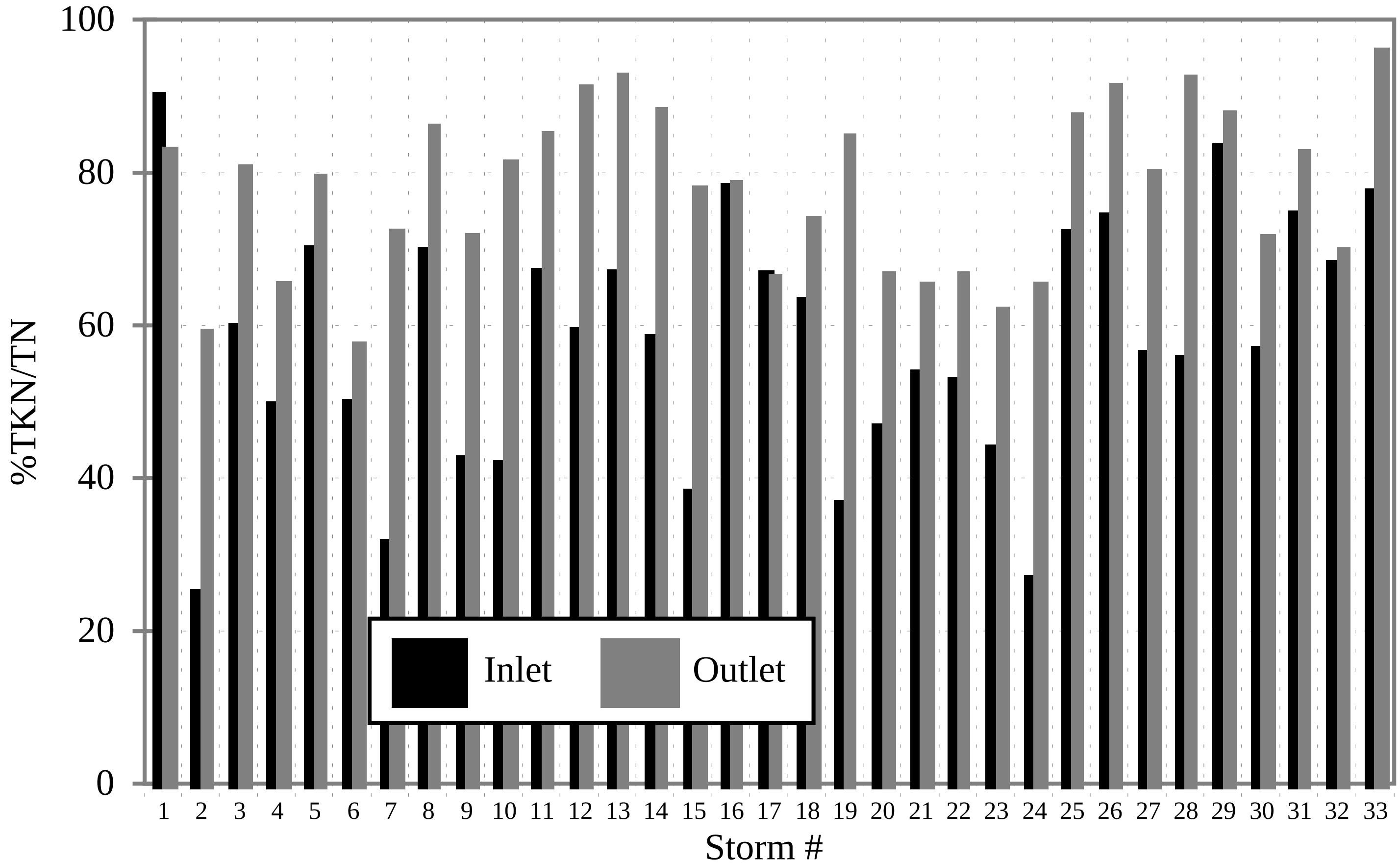


Figure 18. Percent TKN of TN in inlet and outlet EMC samples

greater in the outlet than in the inlet samples (median values were 33.3% and 45.3% at the inlet and outlet, respectively) (Figure 19). For subset A, TP removal was 71.1%, compared to 31% for OP and 55% for TSP.

TSS and Turbidity

TSS removal was positive by every estimation method. This was despite the fact that the overland flow concentrations were very similar to those in the inlet EMC samples (Table 8). By contrast, turbidity removal estimates were negative by all methods except subset A and LTE "loads".

Trace Metals

Of the seven calculation methods used, the LTE method was the only one which resulted in positive removal efficiencies for both forms (extractable and soluble) of all metals. For subset A, metal removals were all positive except for soluble copper. LTE performance estimates for all metals ranged from 28 to 75% removal.

Vegetation Analysis

On September 13, 1996, approximately nine months after the site retrofit was implemented, a comprehensive inventory of plant species was conducted at the Crestwood site (Fleming, 1996). Three well-defined zones were identified within the facility: a low, wet area with standing water; a slightly higher surrounding area with saturated soil; and a relatively dry area on the upper slopes of the basin. In general, plant diversity at the site was noted to be high, including some native species not commonly found in the local area (*e.g. Eleocharis englemanni*, *Liatris squarrosa*) (Table 13). The pond showed indications of recent inundation, with several upland species found still growing within the two

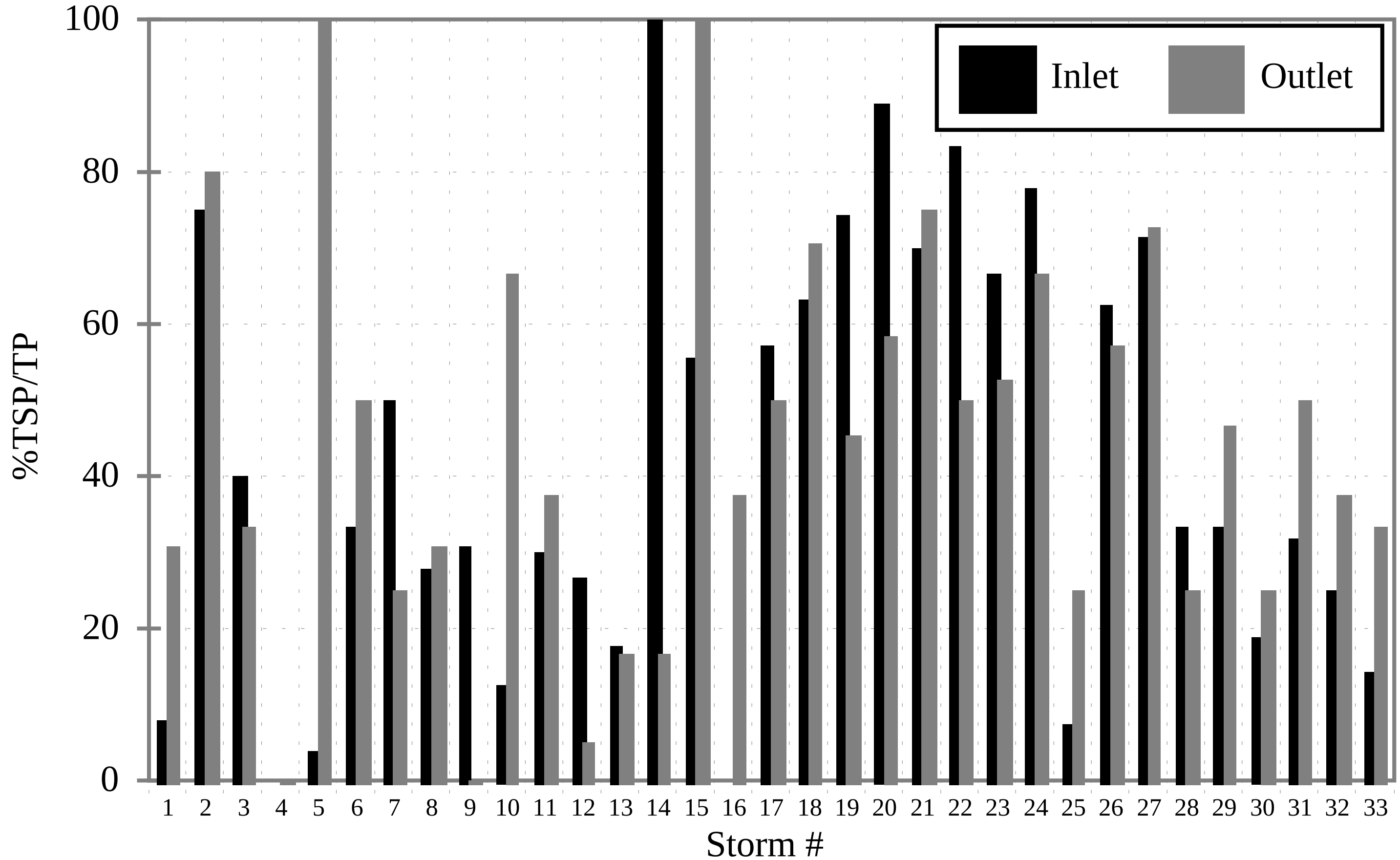


Figure 19. Percent TSP of TP in inlet and outlet EMC samples

wetter zones. Nevertheless, the wet zones appeared to have already begun developing a characteristic wetland flora, with species such as *Carex crinata*, *Juncus effusus*, *Typha latifolia*, and *Rumex verticillata*.

Table 13. Vascular Plants Identified at the Crestwood Wetland on 9/13/96 (Fleming, 1996)

WETLAND (LOWEST AND WETTEST AREA)

Scientific Name	Common Name	Wetland Status*
Apocynum cannibum	Indian hemp	FACU
Aster dumosus	Bushy aster	FAC
Aster lateriflorus	Calico aster	FACW-
Bidens polylepsis	Tickseed-sunflower	FACW
Carex crinita	Drooping sedge	OBL
Cyperus strigosus	Straw-colored nutsedge	FACW
Echinochloa crusgalli	Barnyard grass	FACU
Eleocharis englemanni	Englemans spikerush	FACW+
Eupatorium hyssopifolium	Hyssop-leaved boneset	
Juncus acuminatus	Taper-tip rush	OBL
Juncus effusus	Soft rush	FACW+
Juncus marginalis	Grass-leaved rush	FACW
Juncus tenuis	Path rush	FAC-
Lespedeza striata	Japanese clover	FACU
Panicum virgatum	Switchgrass	FAC
Polygonum pensylvanicum	Pennsylvania smartweed	FACW
Prunella vulgaris	Healall	FACU+
Rumex verticillatus	Swamp dock	OBL
Schizachyrium scoparium	Little bluestem	FACU-
Solidago canadensis	Tall goldenrod	FACU
Typha latifolia	Broad-leaved cattail	OBL

Table 13 (cont.) Vascular Plants Identified at the Crestwood Wetland on 9/13/96 (Fleming, 1996)

MIDLAND AREA (1-3 FEET ABOVE WETLAND)

Scientific Name	Common Name	Wetland Status*
Agalinis purpurea	Purple gerardia	FACW-
Agropyron repens	Quackgrass	FACU-
Allium vineale	Wild garlic	FACU-
Ambrosia artemisiifolia	Common ragweed	FACU
Aster dumosus	Bushy aster	FAC
Aster lateriflorus	Calico aster	FACW-
Bidens polylepsis	Tickseed-sunflower	FACW
Cuphea petiolata (vicosissima)	Clammy cuphea	FAC-
Cyperus echinatus (ovularis)	Globose nutsedge	FACU
Cyperus strigosus	Straw-colored nutsedge	FACW
Eragrostis spectabilis	Purple lovegrass	
Eupatorium hyssopifolium	Hyssop-leaved boneset	
Juncus effusus	Soft rush	FACW+
Juniperus virginiana	Eastern red-cedar	FACU
Lespedeza cuneata	Chinese bushclover	FACU
Lespedeza striata	Japanese clover	FACU
Lonicera japonica	Japanese honeysuckle	FAC-
Phleum pratense	Timothy grass	FACU
Poa pratense	Kentucky bluegrass	FACU
Pyrus calleryana	Bradford pear	
Rubus argutus	Blackberry	FACU
Schizachyrium scoparium	Little bluestem	FACU-
Setaria glauca	Yellow foxtail grass	FAC
Solidago graminifolia	Grass-leaved goldenrod	

Table 13 (cont.) Vascular Plants Identified at the Crestwood Wetland on 9/13/96 (Fleming, 1996)

UPLAND (HIGHEST, DRIEST AREA)

Scientific Name	Common Name	Wetland Status*
Acalypha rhomboidea	Three-seeded mercury	FACW-
Agalinis purpurea	Purple gerardia	FACW-
Agrimonia parviflora	Small-flowered agrimony	FAC
Asclepias syriaca	Common milkweed	
Aster lateriflorus	Calico aster	FACW-
Aster pilosus	White heath aster	
Aster vimineus	Small white aster	FAC
Cichorium intybus	Chickory	
Cirsium discolor	Pasture thistle	
Convolvulus sepium	Hedge bindweed	
Dactylis glomerata	Orchard grass	FACU
Daucus carota	Queen Anne's lace	
Dianthus armeria	Deptford pink	
Eleagnus umbelleta	Autumn olive	
Eragrostis spectabilis	Purple lovegrass	
Erigeron strigosus	Lesser daisy fleabane	FACU+
Eupatorium hyssopifolium	Hyssop-leaved boneset	
Fraxinus americana	White ash	FACU
Juniperus virginiana	Eastern red-cedar	FACU
Lespedeza cuneata	Chinese bushclover	FACU
Lespedeza stipulacea	Korean bushclover	FACU
Liatris squarrosa	Scaly blazing-star	
Lonicera japonica	Japanese honeysuckle	FAC-
Morus alba (saplings)	White mulberry	
Nyssa sylvatica (sapling)	Tupelo	FAC
Pinus virginicus	Virginia pine	
Prunella vulgaris	Healall	FACU+
Pyrus calleryana	Bradford pear	
Rhus copallinum	Shining sumac	NI

Table 13 (cont.) Vascular Plants Identified at the Crestwood Wetland on 9/13/96 (Fleming, 1996)

UPLAND (HIGHEST, DRIEST AREA) (cont.)

Rhus glabra	Smooth sumac	
Rudbeckia fulgida	Orange coneflower	FAC
Schizachyrium scoparium	Little bluestem	FACU-
Solanum carolinense	Horse-nettle	
Solidago canadensis	Tall goldenrod	FACU
Solidago gigantea	Smooth goldenrod	FACW
Solidago nemoralis	Gray goldenrod	
Sorghastrum nutans	Indian grass	
Toxicodendron radicans	Poison ivy	FAC
Tridens flavus	Purpletop grass	FACU
Trifolium pratense	Red clover	FACU-
Vernonia noveboracensis	New York ironweed	FACW+

* U.S. Fish and Wildlife Service classification system:
OBL = obligate wetland
FACW = facultative wetland
FAC = facultative
UPL = upland
NI = no indicator
NO = no occurrence
(+) = toward higher frequency of occurrence within a category
(-) = toward lower frequency of occurrence within a category

V. Discussion

Characterization of Runoff and Atmospheric Depositions

Compared to other studies of urban runoff in the Washington metropolitan area, constituent concentrations in townhouse drainage at the Crestwood site appear to be similar for some constituents (ammonia, nitrate), slightly lower for others (OP, TSP, TP, TKN, TN, TSS, COD, ECu, ECd), and substantially lower for a few (EPb, EZn) (Table 8). Thus in general, the Crestwood drainage seems to be relatively “clean” for urban runoff in the national capital area. In part, this may reflect variations between previously studied sites in terms of proximity to local sources of atmospheric pollutants, and differences in land uses within the watersheds.

The substantially lower Pb concentrations in urban runoff at Crestwood compared to earlier studies probably reflects the reduction in leaded gasoline usage that has occurred since those studies were conducted. This trend is also evident in the atmospheric data, which shows much lower areal deposition rates for Pb compared with earlier studies, as illustrated in Table 7. Areal loadings of Cu and Cd were also lower at Crestwood (by one to two orders of magnitude), while loadings of nutrients and Zn were comparable to results from earlier studies.

Comparison between concentrations in rainfall (WE40), and both townhome runoff (WE20) and “forested” runoff (WE15) suggests that both sub-watersheds (townhomes and forested area) are net sources for some constituents (OP, TP, TKN, TN, ECu, EZn), and net sinks for others (ammonia). That the forested runoff contained higher concentrations of ECu and EPb than the townhouse drainage is surprising. However, given that this reflects only a single WE15 sample which was analyzed for metals, this result must be considered highly uncertain, and deserving of further investigation.

Lysimeter Samples

An interesting trend was seen in the data for Spring 1997, where ammonia concentrations in WE50 climbed substantially, while those in WE60 and WE70 did not.

This may have resulted from the development of anaerobic conditions in the shallow sediments, which would suppress nitrification and lead to ammonia build-up. In contrast, the similarity in nitrate concentrations among the three lysimeters reflects the general mobility of anions within the soil matrix, and suggests that nitrate is able to readily diffuse throughout the wetland sediments.

Outlet Baseflow Samples

The fact that baseflow concentrations of nitrate were consistently lower than storm (outlet) concentrations, unlike most constituents, suggests that nitrate removal primarily occurred during the quiescent periods between storms. This would be consistent with either denitrification or plant uptake being the primary removal mechanism for this constituent. In contrast, solids-associated constituents may be mostly removed by settling during storm events. Subsequently developing anaerobic conditions in newly deposited sediments would result in re-release of some of these pollutants to the water column, which for some soluble constituents might account for the observed inverse relationship between outlet flow rate and baseflow concentrations. In the case of suspended solids and insoluble constituents, the inverse relationship between concentration and flow rate may reflect greater algal densities occurring in the water column during quiescent conditions.

Pollutant Removal Performance

Ammonia, nitrate, and TSS showed net removal by all calculation methods employed, and were the only constituents which did so. Total nitrogen and Kjeldahl forms of nitrogen did not display this trend. This suggests that nitrification/denitrification was not solely responsible for the loss of ammonia and nitrate, and that microbial/vegetative uptake might play a significant role in the fate of nitrogen within the wetland. The fact that the percentage of TN in the form of TKN was usually greater in outlet (WE10) than in inlet (WE20) EMC samples suggests that inorganic forms of nitrogen (nitrate, ammonia) were transformed into exportable organic nitrogen within the wetland, via uptake into algae and/or incorporation into macrophyte tissues. Uptake may have been followed by re-release into the water column in soluble organic form as cellular exudate, or export from the wetland in particulate organic form, *i.e.* algal cells. The presence of algae in the wetland was noted by OWML staff, who periodically observed the presence of an algal scum on the surface of the water near the facility outlet. On one occasion, a build-up of this material had to be manually removed from the outlet weir, because it was impeding flow through the outlet orifice. Export of algae from the facility could account for substantial Kjeldahl nitrogen and COD export, and could account for the relatively poor removals of these constituents.

As previously mentioned, conductivity and total hardness concentrations were generally higher in outlet than in inlet samples (Figures 9, 10). This may reflect the influence of soils in the wetland, or of overland runoff from the ungaged areas on the chemical make-up of the water column in the wetland, however these constituents were not measured in the WE15 samples.

The high TSS removal performance at the Crestwood wetland is generally consistent with the results of other researchers, as shown in Table 3. The apparent increase in turbidity "loads" which occurred coincident with the decrease in TSS loads might be the result of biotic processes releasing dissolved organic matter into the water column. Similarly, the fact that the percentage of TP as TSP was usually greater in outlet (WE10) than in inlet (WE20) EMC samples reflects a relatively greater loss of insoluble than soluble phosphorus during passage through the wetland. This is consistent with sedimentation as a removal mechanism for particulate-associated phosphorus. However, with the exception of the subset A calculation, TP load removals were not substantially better than OP or TSP removals, suggesting that other processes besides sedimentation played a role in phosphorus removal. The greater relative removal of TP compared with TSP for the subset A storms is consistent with the fact that these storms were of relatively small volume, and may reflect the relative lack of sediment scouring and resuspension during these events. At the Franklin Farms site, performance for a number of constituents was found to be substantially better for the subset of storms whose total runoff volumes were less than the capacity of the wetland.

Efficiencies for extractable (*i.e.* total) forms of all metals, with the exception of TZn, were slightly higher than for the soluble forms, suggesting that retention of metals took place within the wetland primarily via sedimentation of particulate-associated forms. This is consistent with the results of previous researchers, who have documented the build-up of metals within the sediments of detention basins and wetlands (Baker and Yousef, 1995; Crites, *et al.*, 1995; Mesuere and Fish, 1989; Schiffer, 1989; Wigington, *et al.*, 1983).

In terms of long-term performance, the Crestwood wetland generally fell within the range of percent removal estimates reported in other selected urban stormwater wetland studies, as shown in Table 14. With the exception of oxidized nitrogen and suspended solids, the Crestwood facility appeared to perform better than the Franklin Farms marsh, despite having a lower area ratio. The better performance at Crestwood may be attributable in part to the greater volume ratio at Crestwood compared with Franklin Farms. This would be consistent with the observation by Schueler (1992) that extended detention storage, in the event that a wetland appears to be undersized with respect to area, is a desirable design feature from a pollutant removal standpoint. Not all the wetland studies examined in this report included sufficient information to allow area or volume ratios to be calculated, therefore it is not possible to compare all the studies on the basis of these parameters. However, the bulk of the available information in the literature suggests that wetland size and layout play a critical role in determining pollutant removal performance.

Estimated long-term pollutant removals at the Crestwood wetland were encouragingly high, and may be expected to improve as wetland vegetation spreads throughout the site. Results from the literature suggest performance would improve further with the addition of a detention pond or sediment forebay to pretreat runoff prior to its entering the wetland. Design guidance documents typically recommend the inclusion of such a feature, which

Table 14. Treatment Efficiencies: Estimated Long-Term % Removals for Selected Urban Stormwater Wetlands and Wet Ponds

Study	VR*	AR†	OP	TSP	TP	NH _N	SKN	TKN	OX_N	TN	TSS	TCu	TPb	TZn
Constructed wetlands														
Crestwood (VA)	0.81	0.004	35.8	48.1	45.9	54.7	19.8	25.5	39.4	22.4	57.9	65.5	74.7	29.2
DUST marsh cell A (CA)		0.002	53		17	-22		7	9		42	5	30	6
Mays Chapel (MD)		0.006		29	-7	22.1			28					
Franklin Farms (VA)	0.19	0.008	23.6	14.5	14.9	-0.5	-9.8	4.4	59.8		61.5			
Clear Lake wetland (MN)		0.049		40	54	55		25			76			
Natural wetlands														
Wayzata wetland (MN)		0.099			78	-44					94	80	94	82
St. Joseph marsh (MN)		0.095			18			22			44			
Hidden Lake wetland (FL)		0.045	-109		7	62.2			80.2	-1.6	82.9	39.9	54.8	40.9
Lake McCarrons wetland (MN)	4.9	0.010		15	32			28	24	27	84		74	
Lake McCarrons wetland + wet pond				48	77			78	64	76	96		93	
Silver Star Road wetland (FL)	2.2	0.018	2	0	17	54			40	21	66		73	56
Silver Star Road wetland + wet pond			28	57	43	61			9	36	89		83	70
Wet ponds														
Westleigh wet pond (MD)‡		0.024		50.9	69.8		48.6	45.8	71.1		86.8			
Burke wet pond (MD)‡		0.035		56.2	59.2		39.8	37.1	83.6		36.8			

*** Volume Ratio: (pool vol/avg. storm vol.)**

† Area Ratio: (pool area/drainage area)

‡ As reported in MDE (1987b)

would extend the lifetime of the wetland, allowing periodic cleanout of accumulated sediments without disruption of the vegetation.

Temporal Trends

Ammonia removal was significantly better (α=0.05, Bonferroni pairwise t-test) in spring, 1996 than in both fall 1996, and spring 1997. OP removal was also better in spring, 1996 than in spring, 1997. Oxidized nitrogen removal was significantly better in summer, 1996 than in fall, 1996. Relatively greater removal in spring and summer compared with fall is consistent with vegetative uptake as an important pathway for nitrogen removal, which would be expected to occur to a greater extent during the growing season. However, the relatively better removals in spring, 1996 as compared with spring, 1997 were unexpected, and may reflect the high degree of uncertainty that results from not having sufficient information on the ungaged overland inputs in the calculations. No significant seasonal differences were observed for any other constituents.

Total alkalinity in outlet baseflow samples generally decreased in the latter part of the study, while pH generally increased. The reason for this seemingly incongruous behavior is not known, but may have been due to precipitation of $CaCO_3$ as photosynthesis raised the pH of the water column.

VI. Conclusions

From the results of the research described in this thesis, the following conclusions were derived:

1. Ungaged overland flow from the wooded/grassy area constituted about 41% of the long term total hydrologic input to the wetland. Assuming a runoff coefficient of 0.2 for this area resulted in a good match (~3% difference) between estimated long-term total hydrologic inputs and outputs. However, this method proved ineffective at explaining the inlet/outlet flow imbalances for individual storms.
2. Most constituent EMCs were lower in outlet than in inlet samples. However, constituent outlet loads were generally greater than inlet loads, due to the larger outlet flow volumes.
3. Estimated long-term efficiency for the facility, which included overland runoff contributions, was positive for most constituents. Results fell within the range of values reported in other runoff wetland studies. Consistent with most of the literature, long-term removals for total suspended solids, total phosphorus, and lead were among the best for all constituents monitored.
4. The estimated overland input load was comparable to the inlet load for many constituents, though fairly uncertain, because it was inferred based on indirect measurements. Performance calculations which ignored this input showed generally poor or negative removals, illustrating the importance of the overland inputs in the apparent performance of the wetland.
5. Ammonia removal was significantly better (α=0.05, Bonferroni pairwise t-test) in spring, 1996 than in both fall, 1996, and spring, 1997. Orthophosphate removal was also better in spring, 1996 than in spring 1997. Oxidized nitrogen removal was significantly better in summer 1996 than in fall 1996. No significant seasonal differences were observed for any other constituent.
6. Despite the short time period since the conversion of the facility to a wetland, there are signs of the development of a diverse wetland flora.

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APPENDIX A
Wetfall and Dryfall Areal Loading Plots; Wetfall and Runoff Concentrations

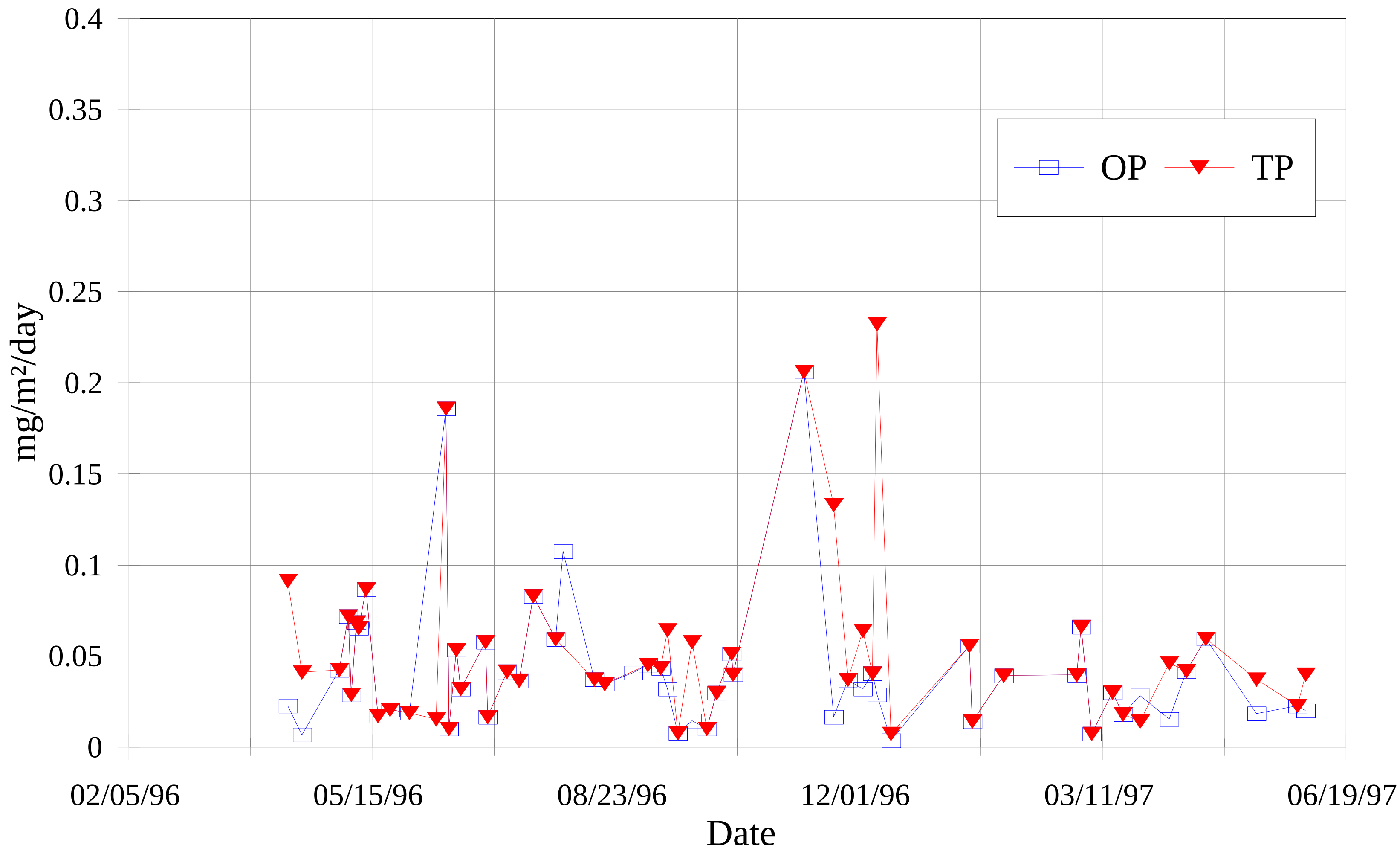

Figure A.1 Wetfall OP and TP areal loads, mg/m²/day

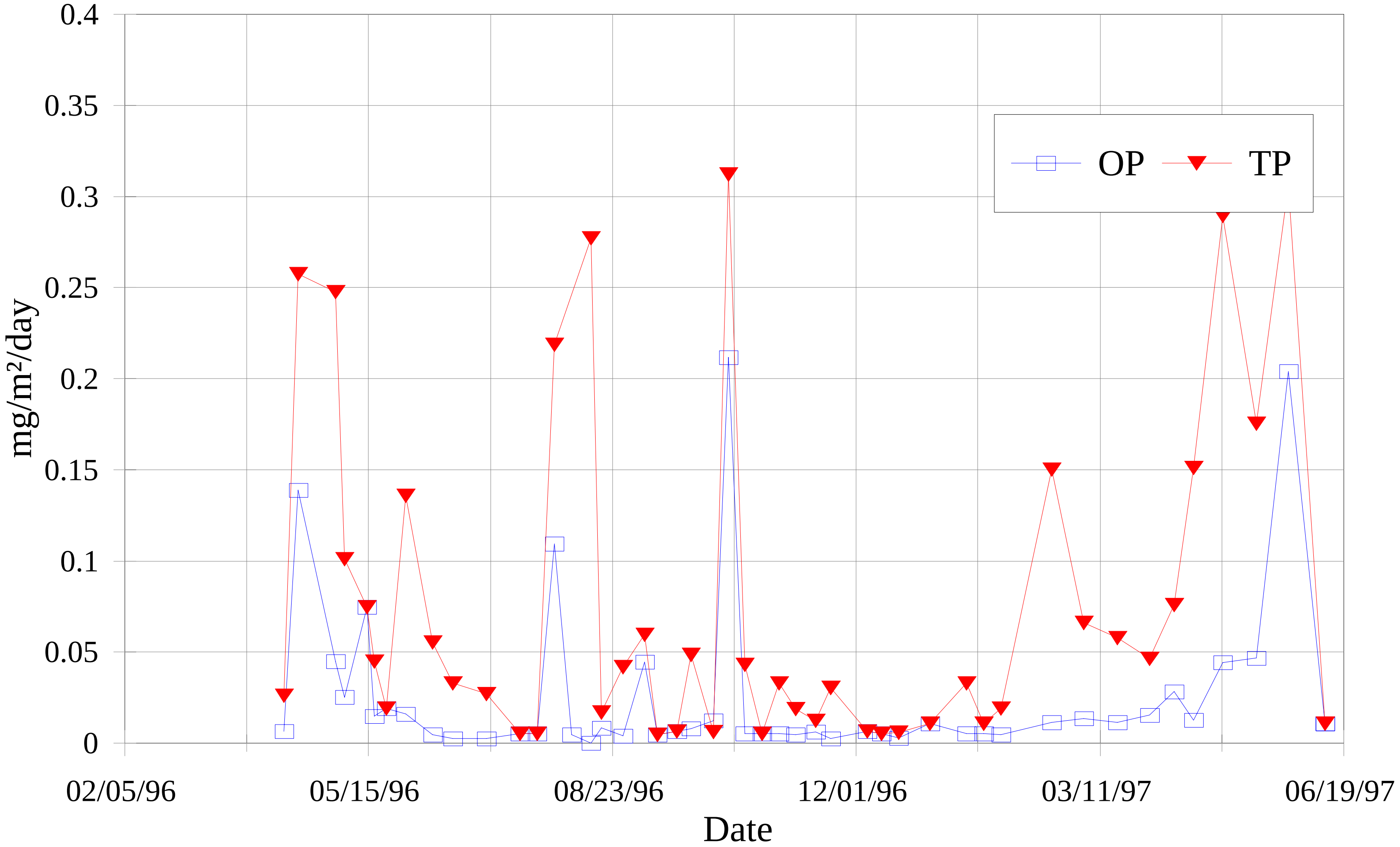

Figure A.2 Dryfall OP and TP areal loads, mg/m²/day

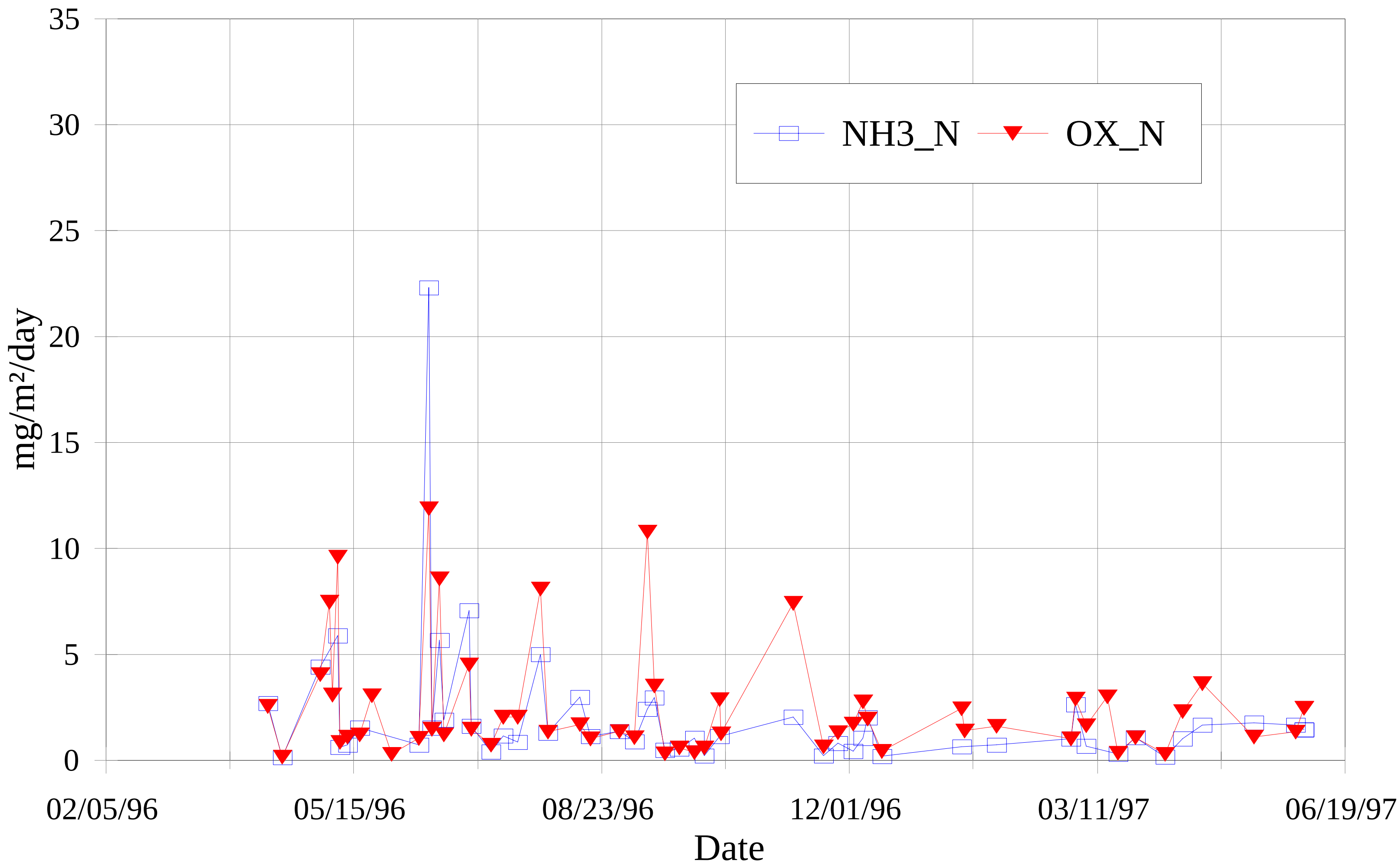

Figure A.3 Wetfall NH3_N and OX_N areal loads, mg/m²/day

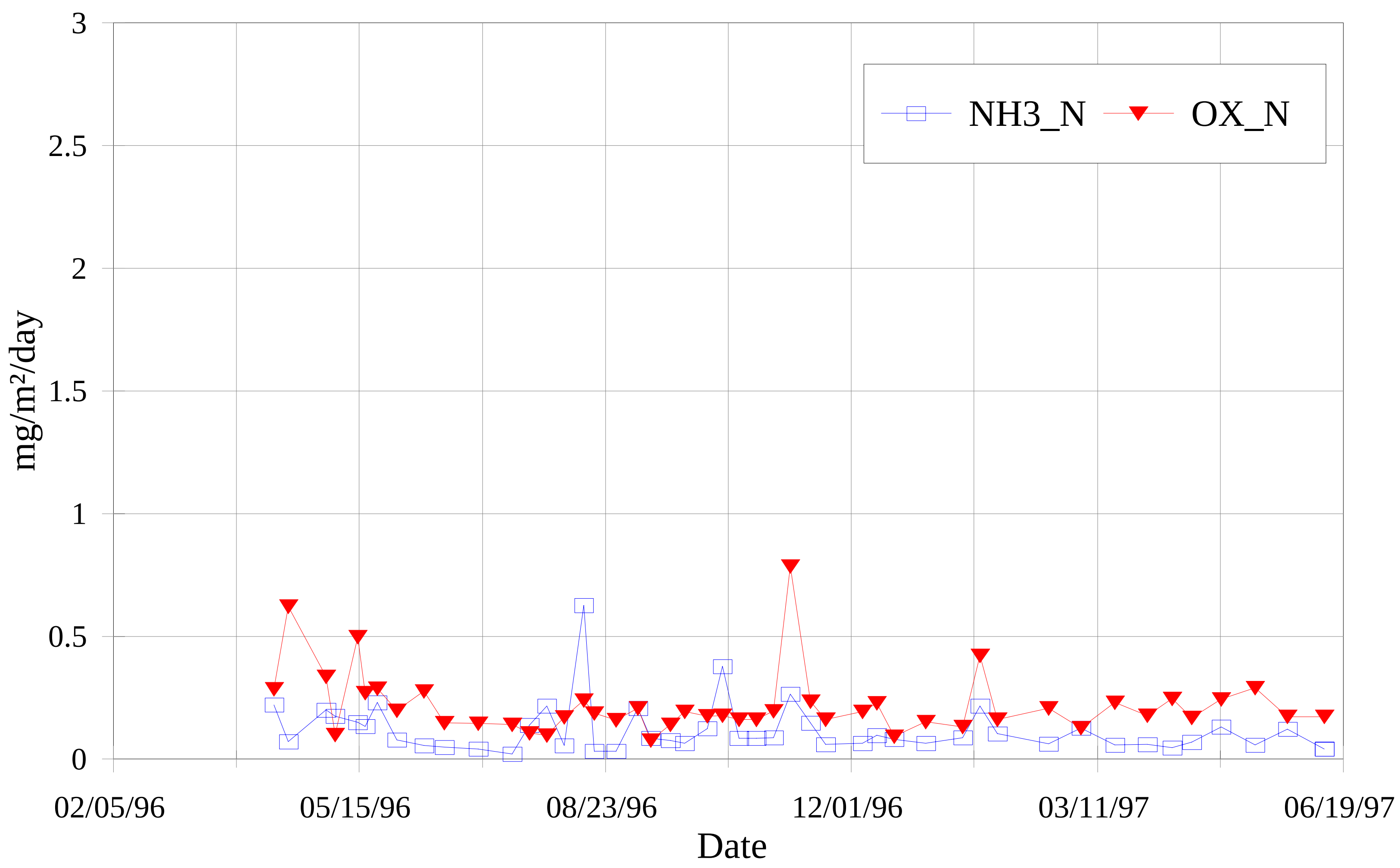

Figure A.4 Dryfall NH3_N and OX_N areal loads, mg/m²/day

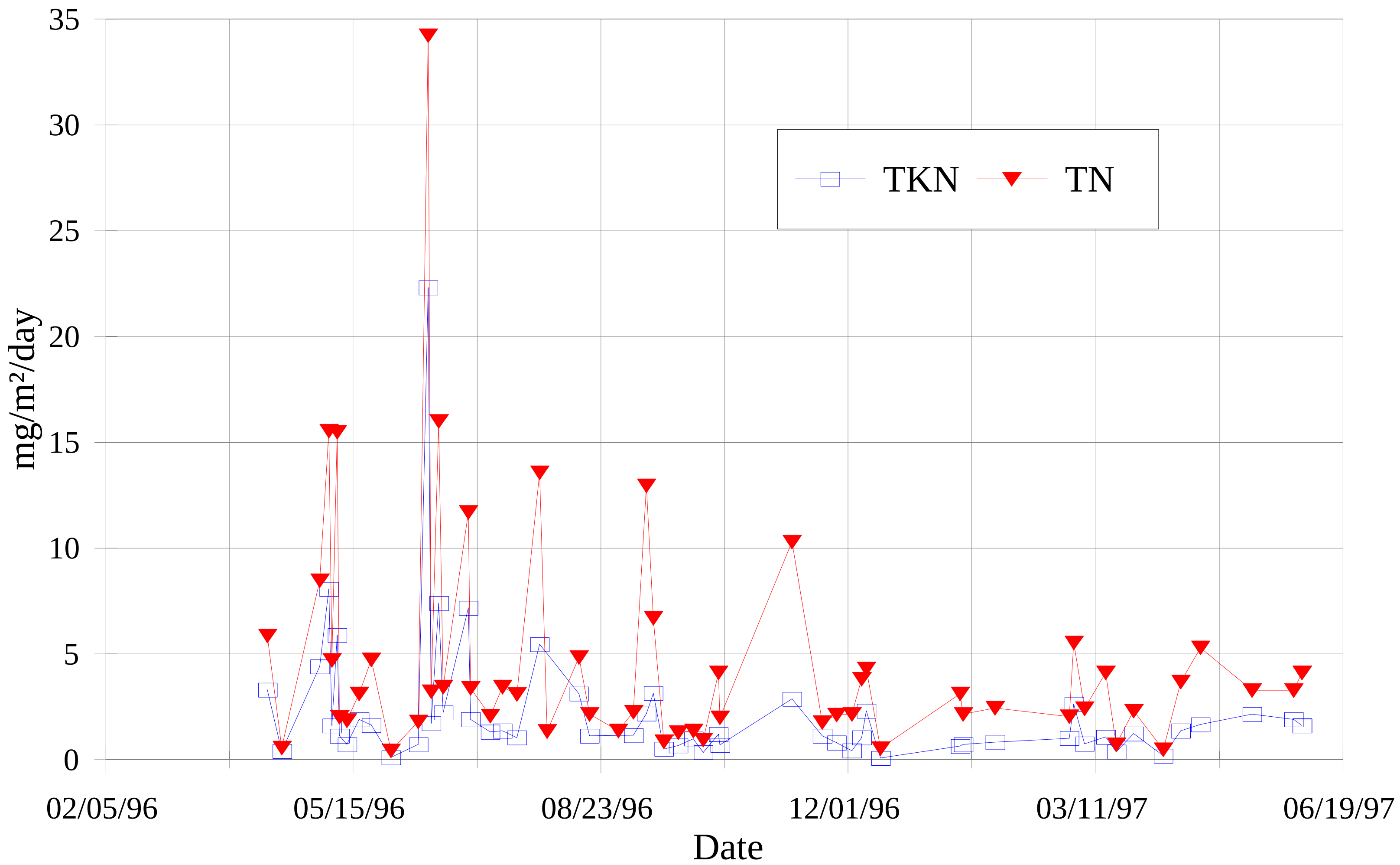

Figure A.5 Wetfall TKN and TN areal loads, mg/m²/day

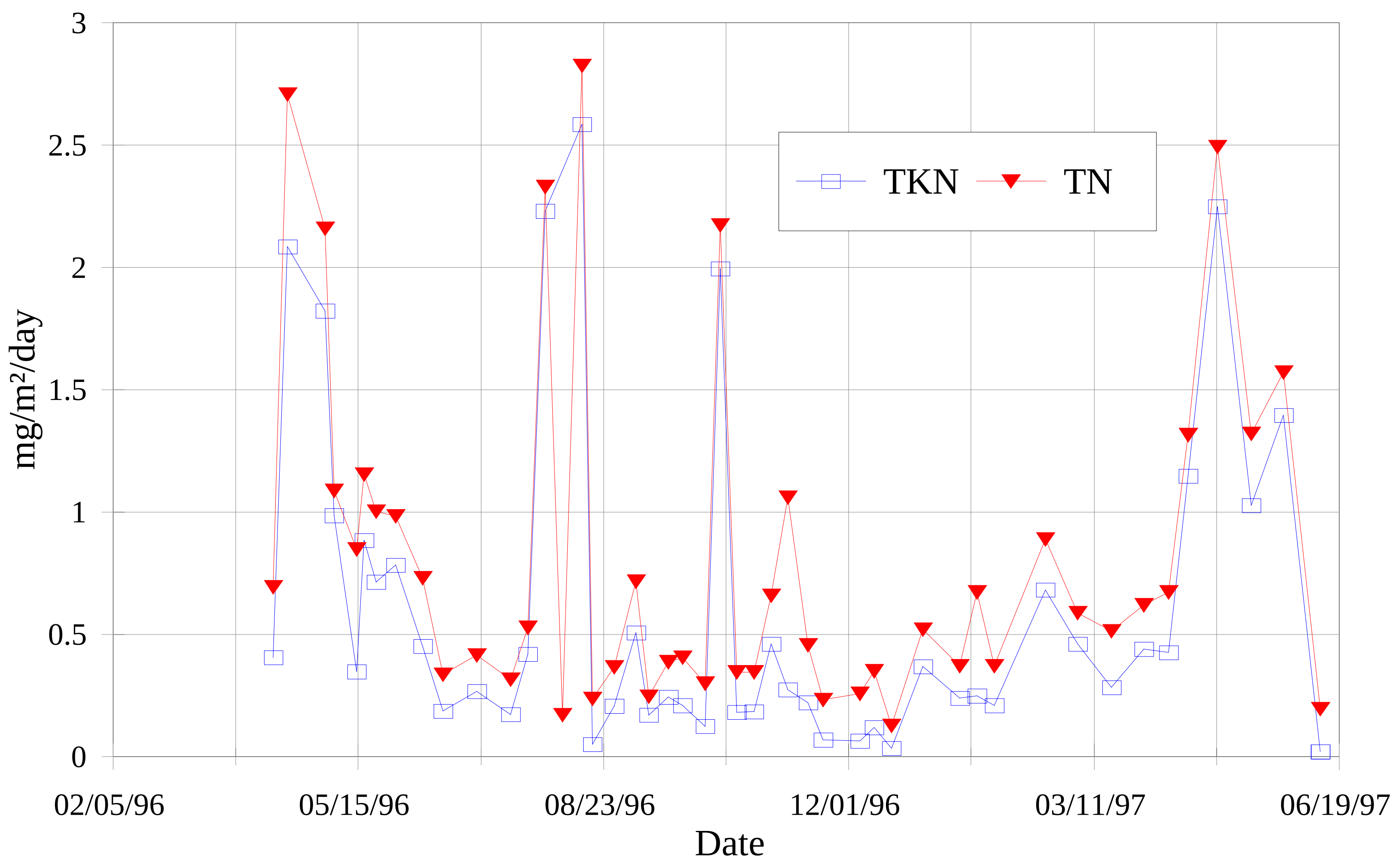

Figure A.6 Dryfall TKN and TN areal loads, mg/m²/day

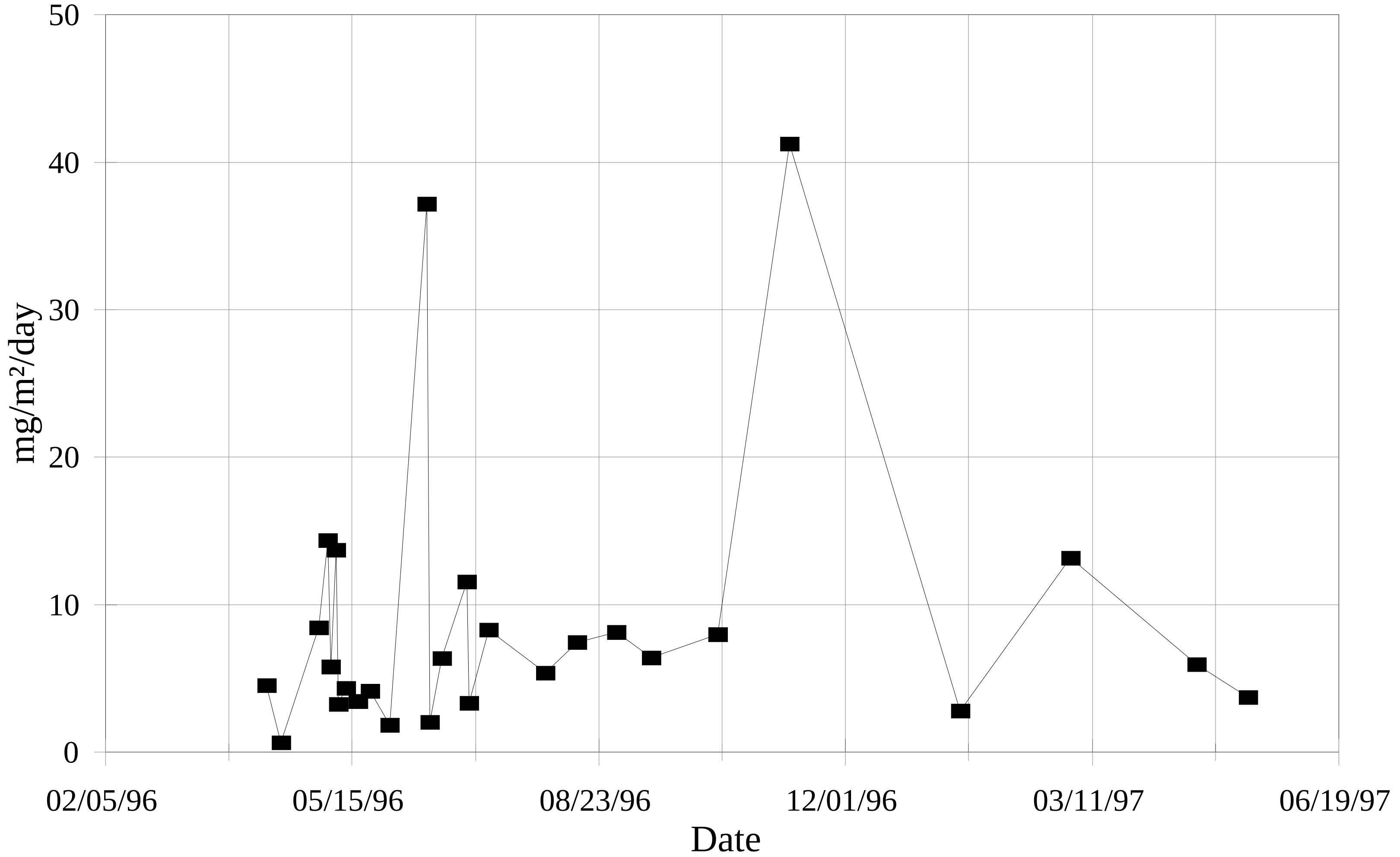

Figure A.7 Wetfall TAl areal loads, mg/m²/day

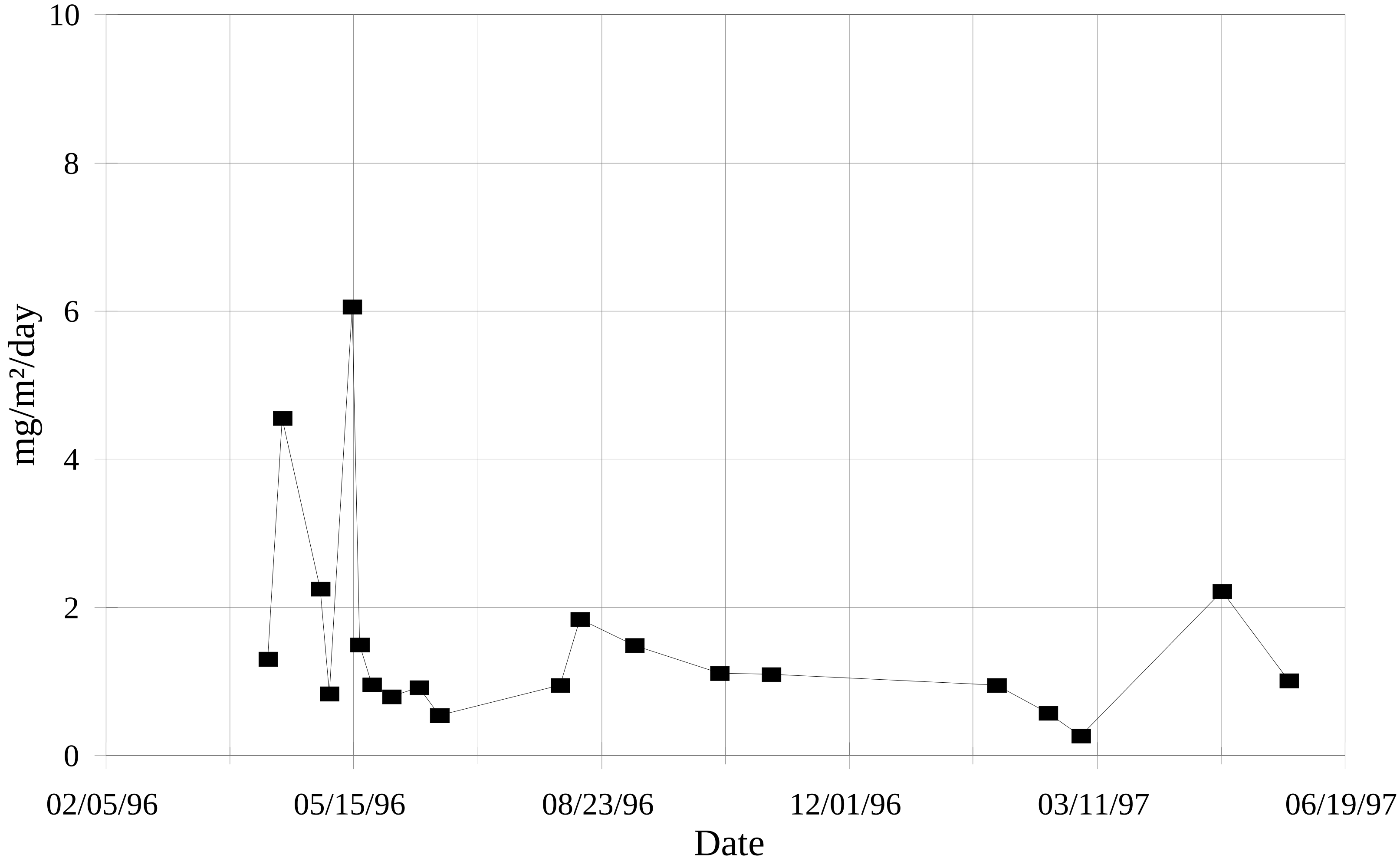

Figure A.8 Dryfall TAl areal loads, mg/m²/day

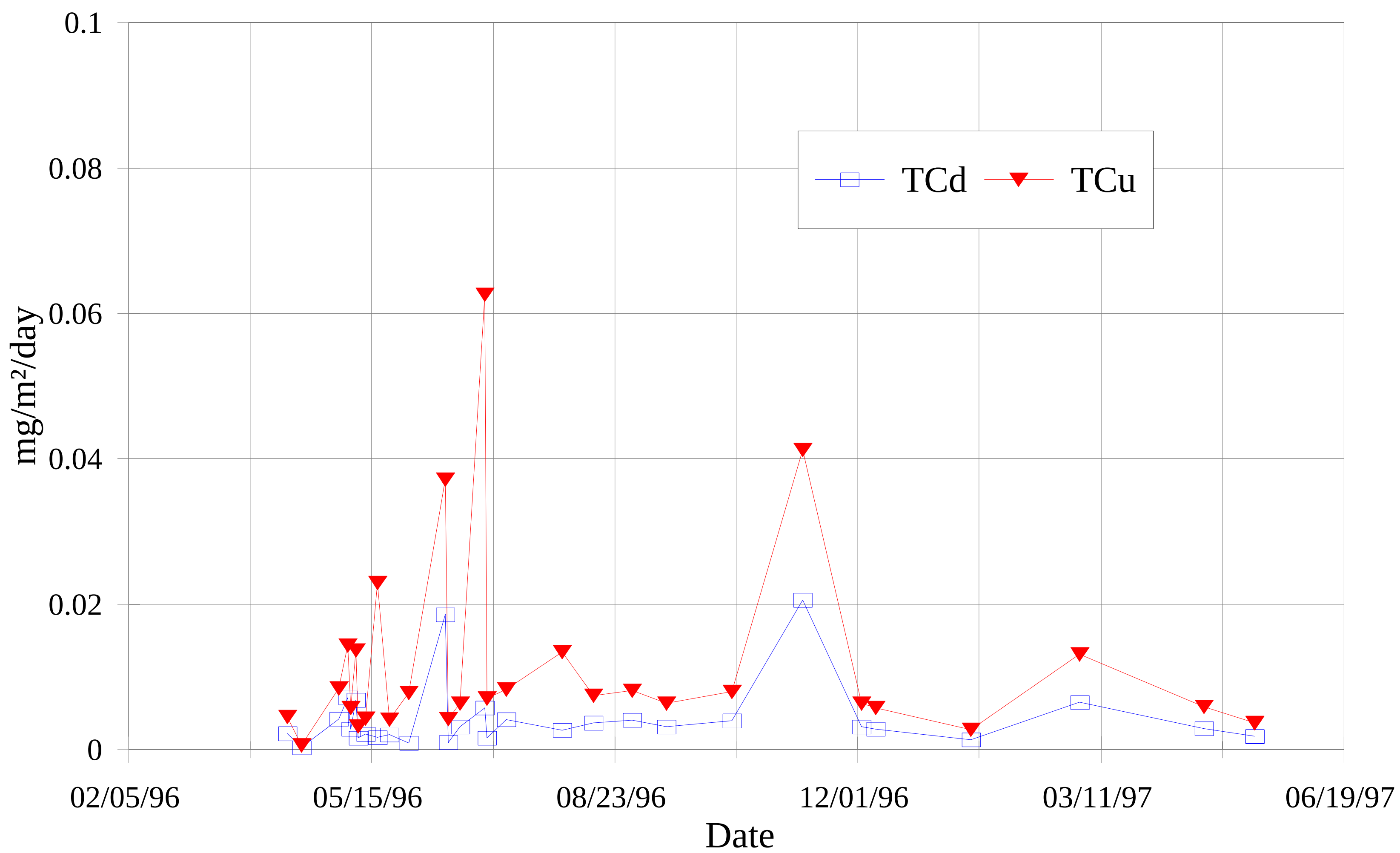

Figure A.9 Wetfall TCd and TCu areal loads, mg/m²/day

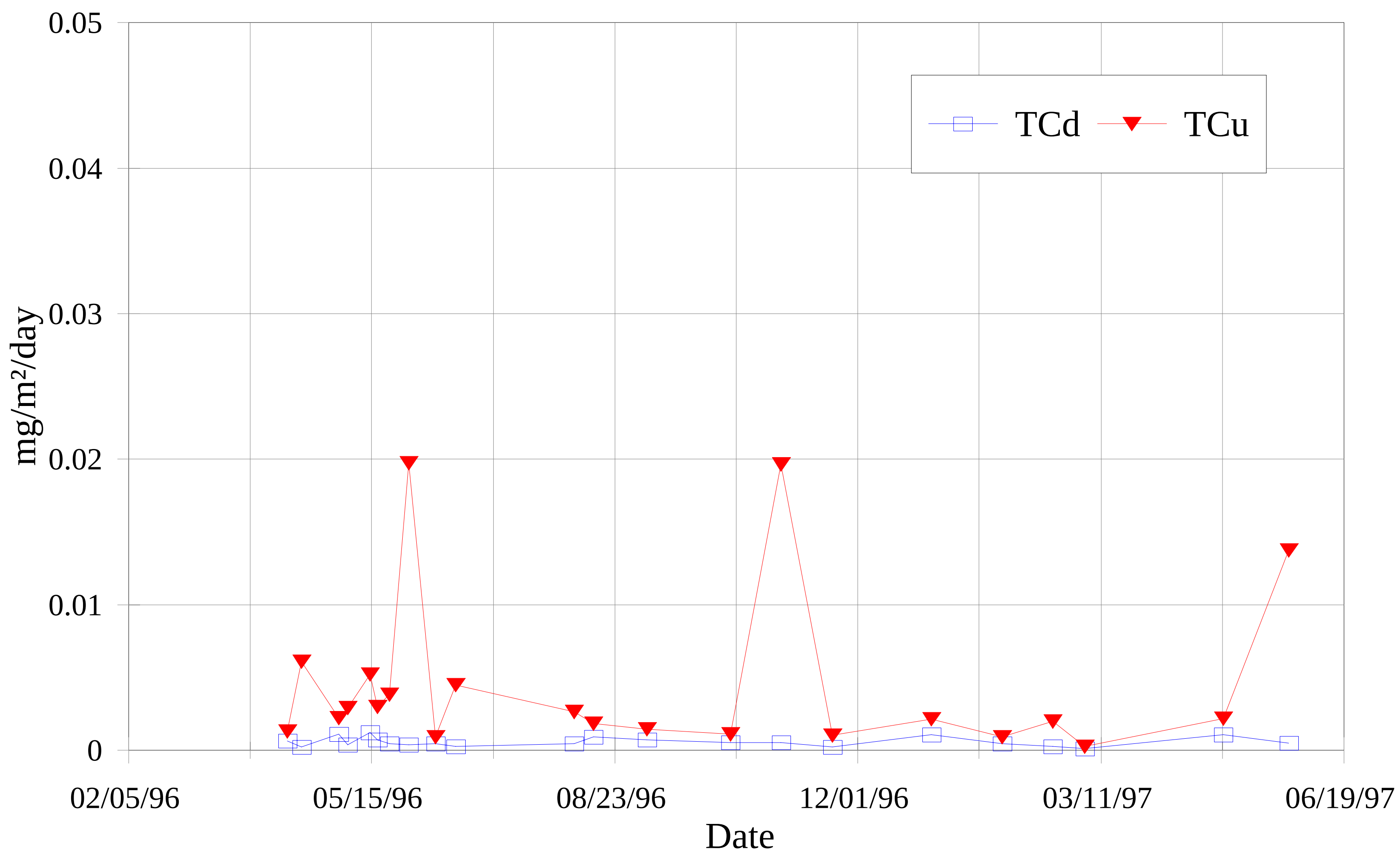

Figure A.10 Dryfall TCd and TCu areal loads, mg/m²/day

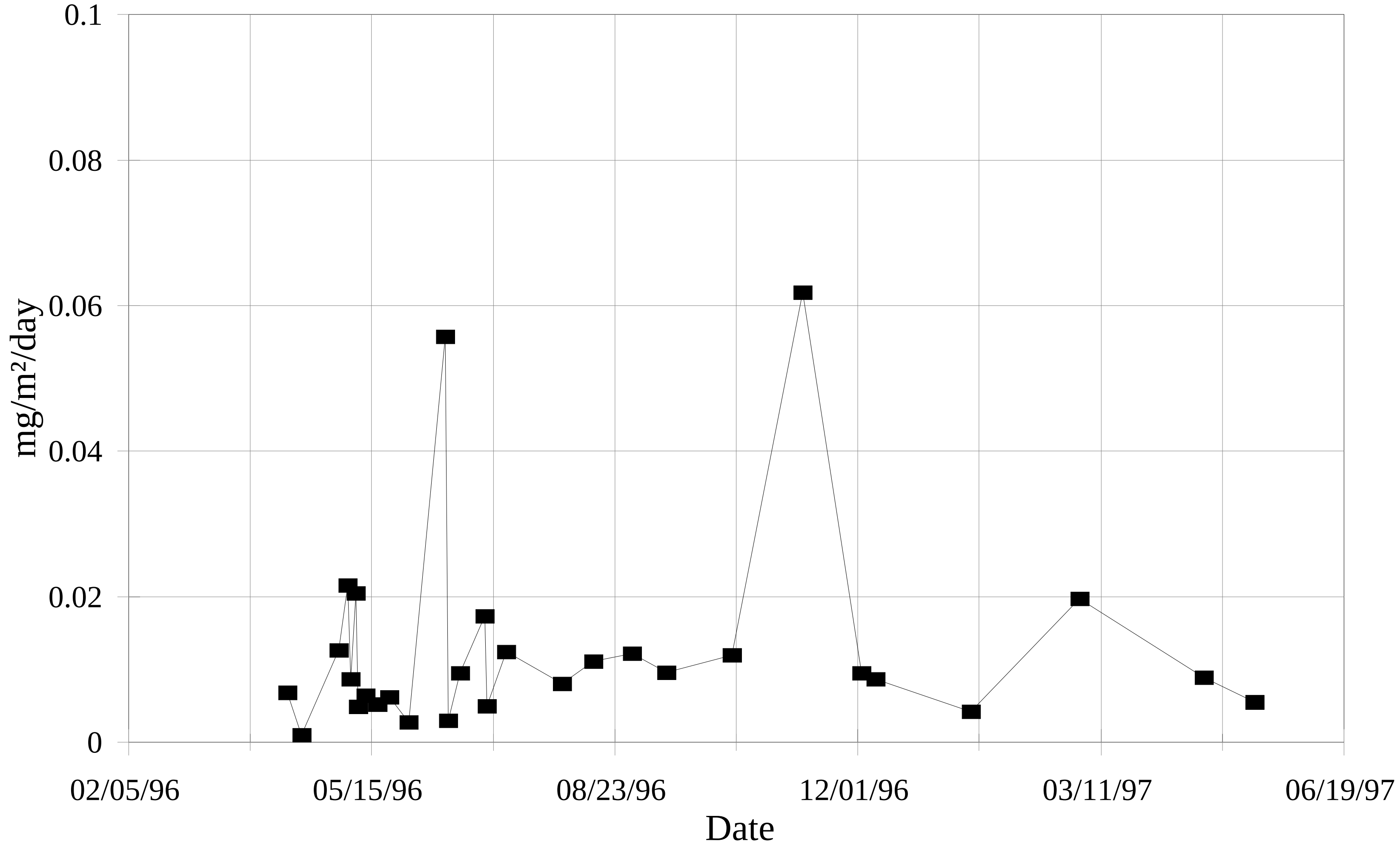

Figure A.11 Wetfall TPb areal loads, mg/m²/day

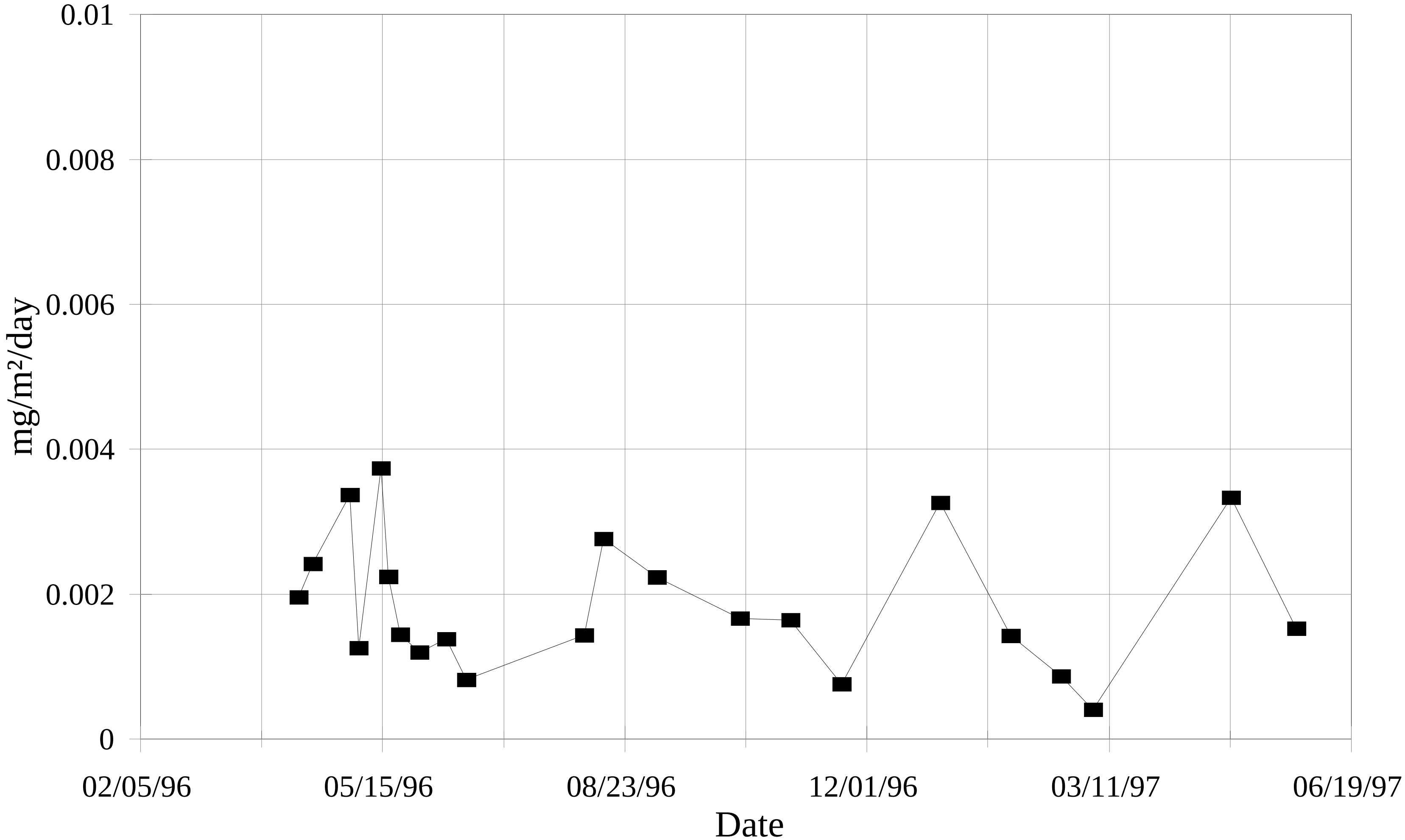

Figure A.12 Dryfall TPb areal loads, mg/m²/day

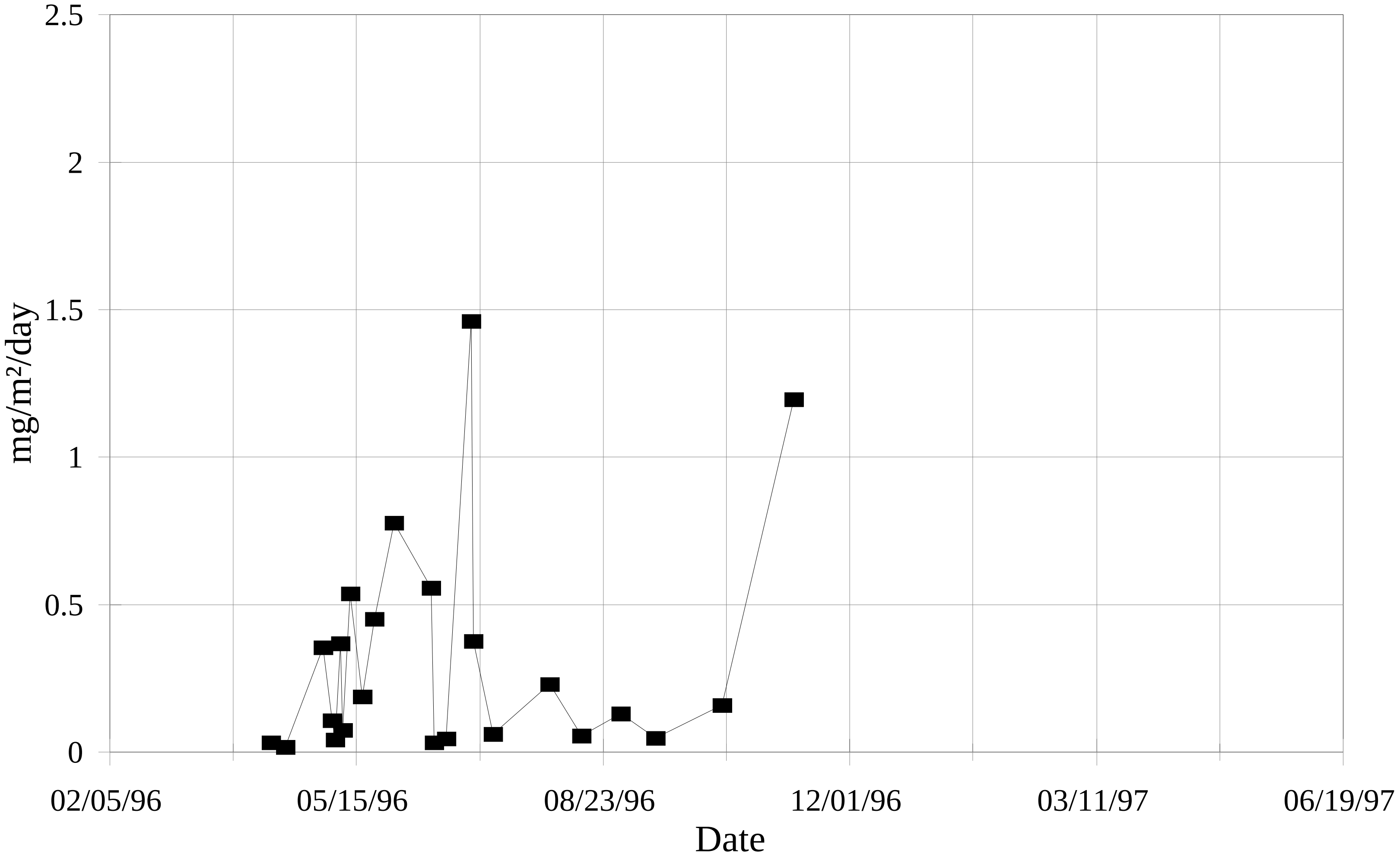

Figure A.13 Wetfall TZn areal loads, mg/m²/day

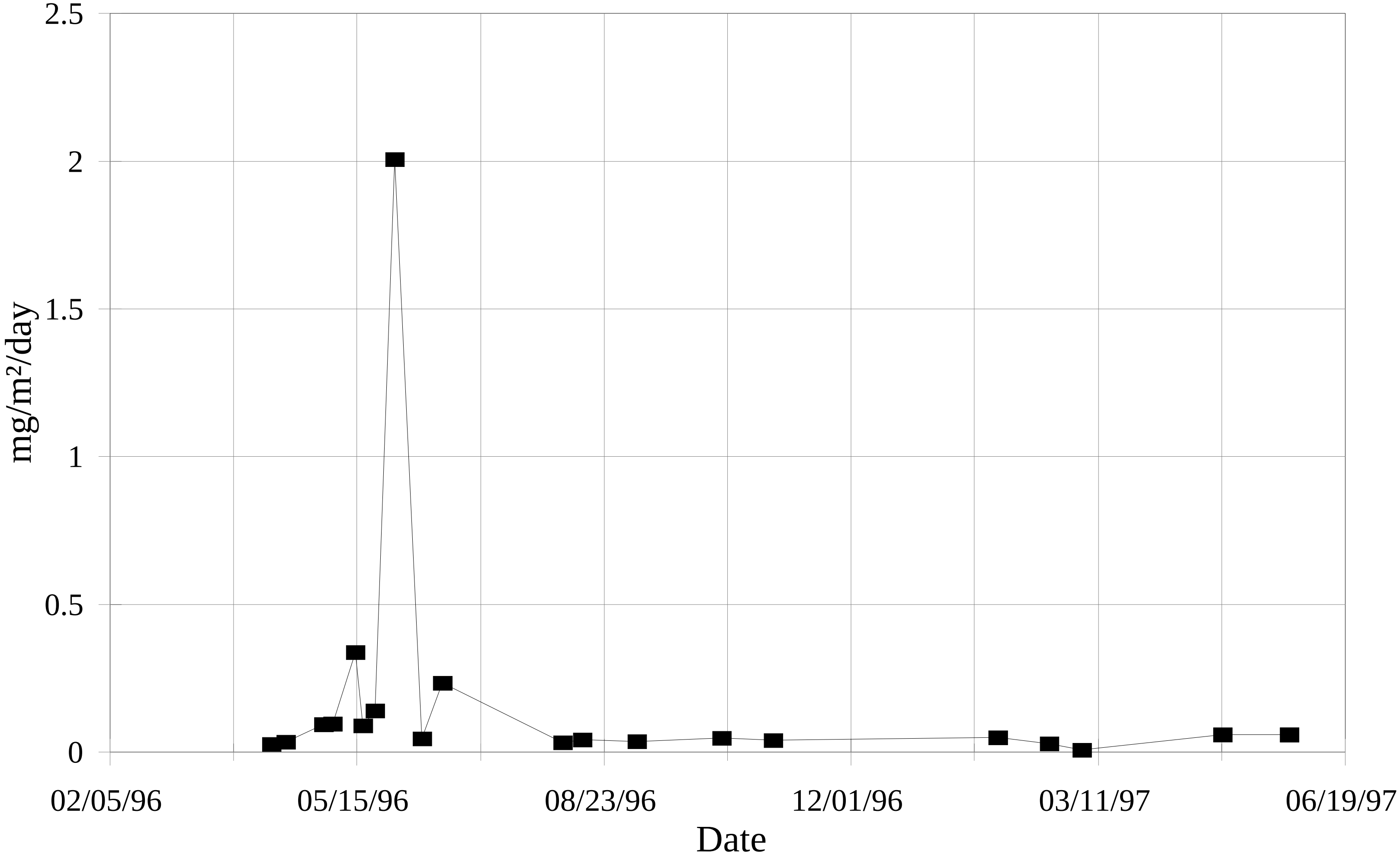

Figure A.14 Dryfall TZn areal loads, mg/m²/day

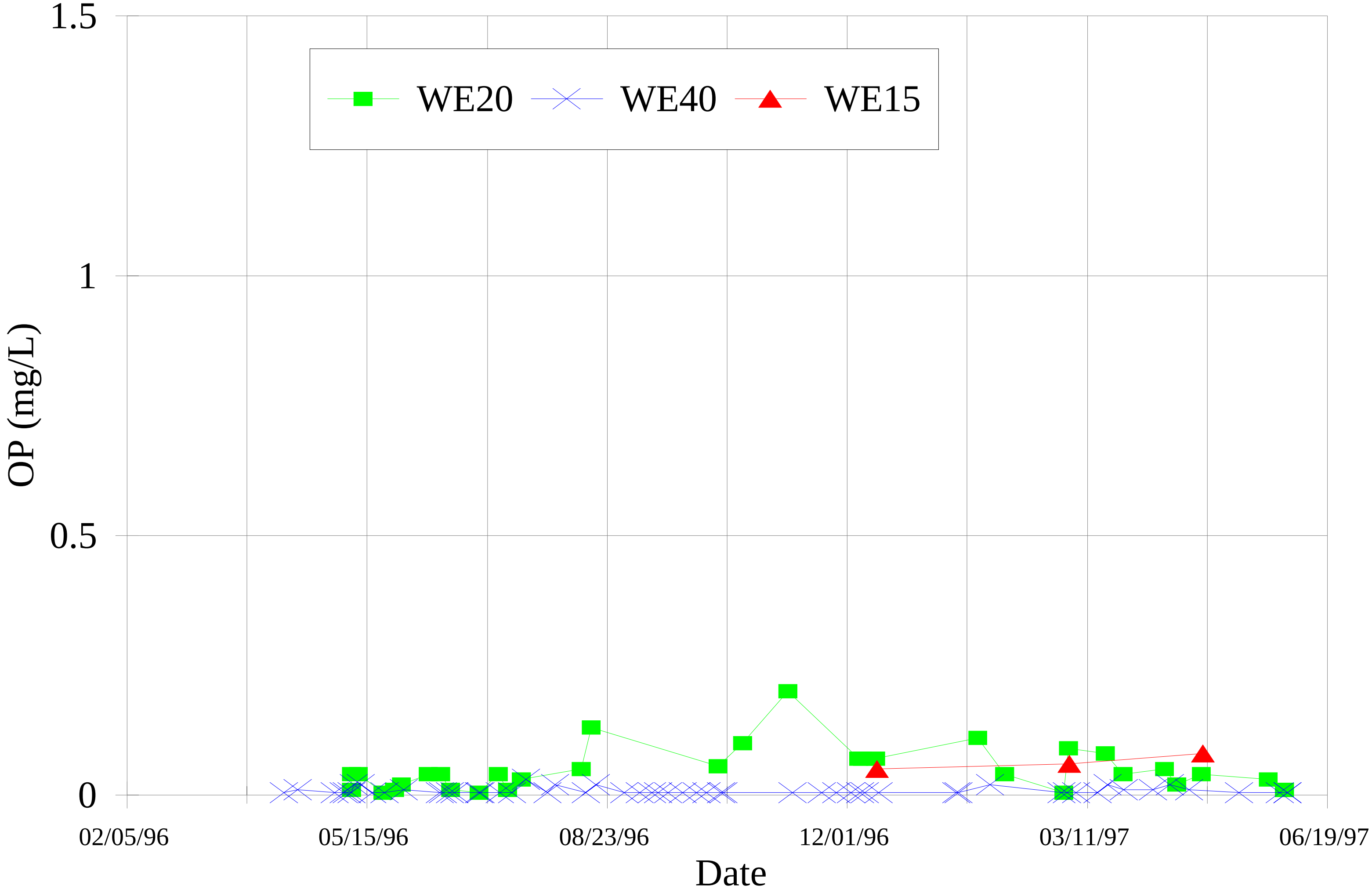

Figure A.15 Inlet EMC, overland grab, and wetfall sample concentrations: OP, mg/L

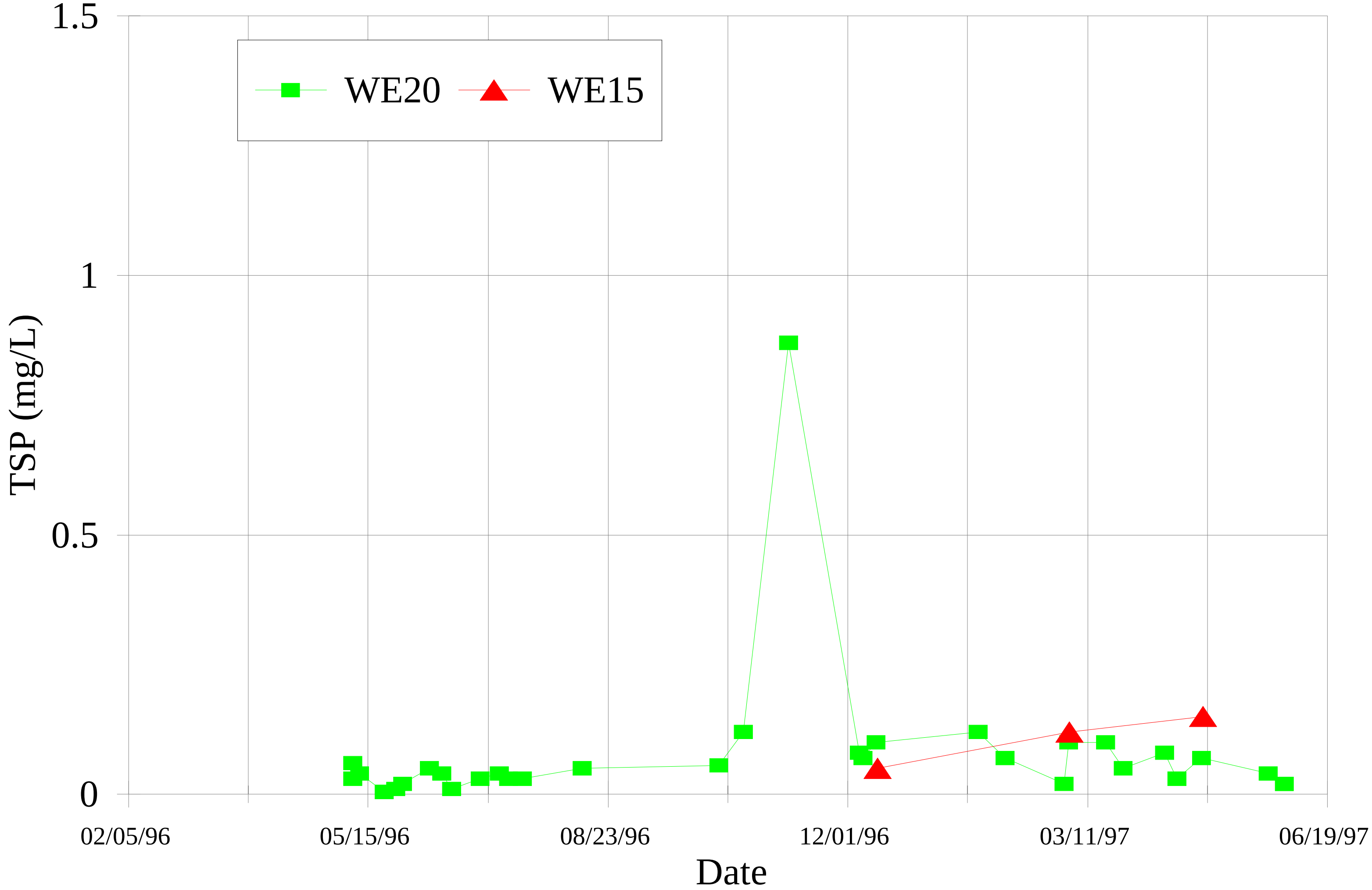

Figure A.16 Inlet EMC, overland grab, and wetfall sample concentrations: TSP, mg/L

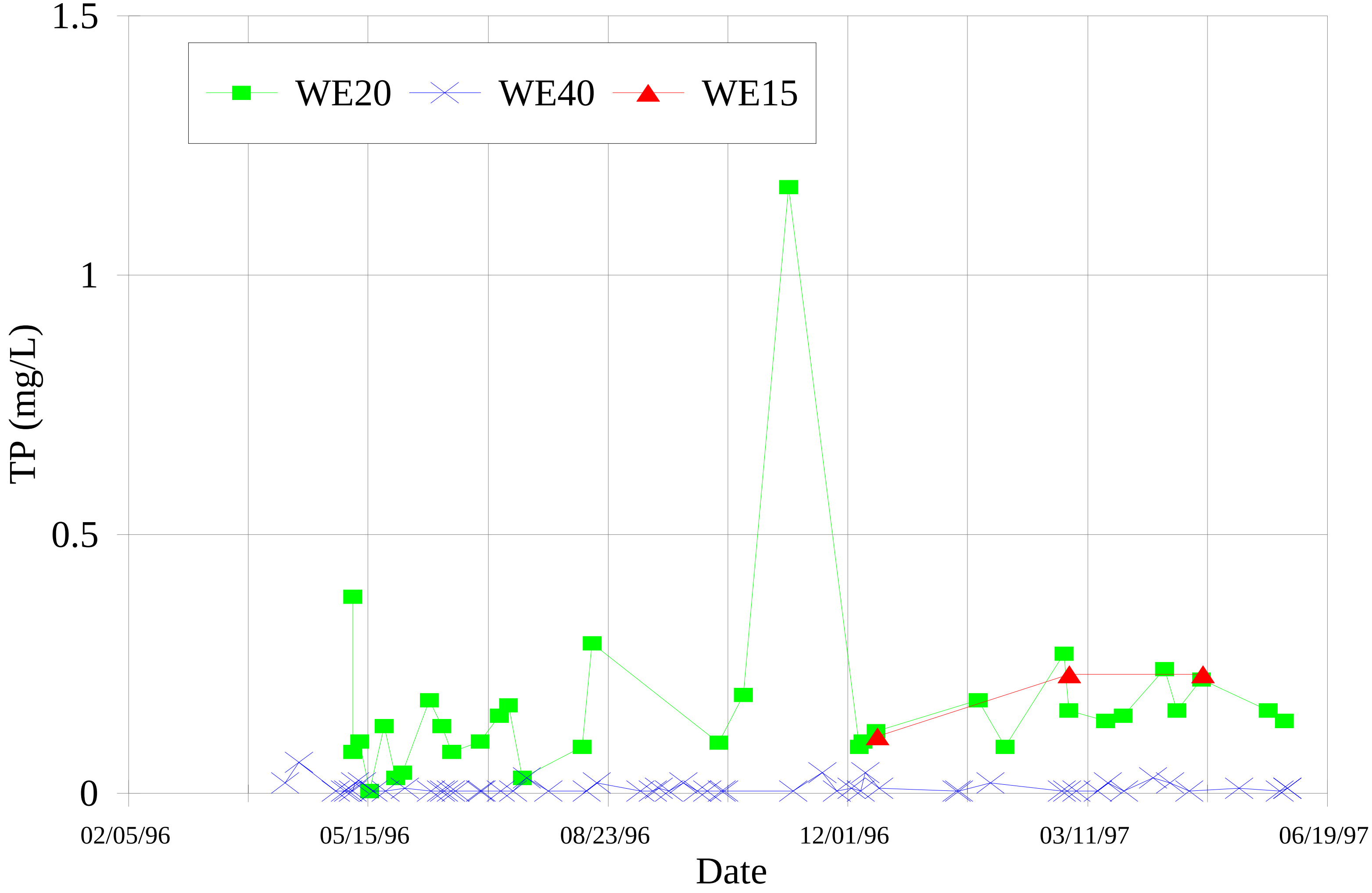

Figure A.17 Inlet EMC, overland grab, and wetfall sample concentrations: TP, mg/L

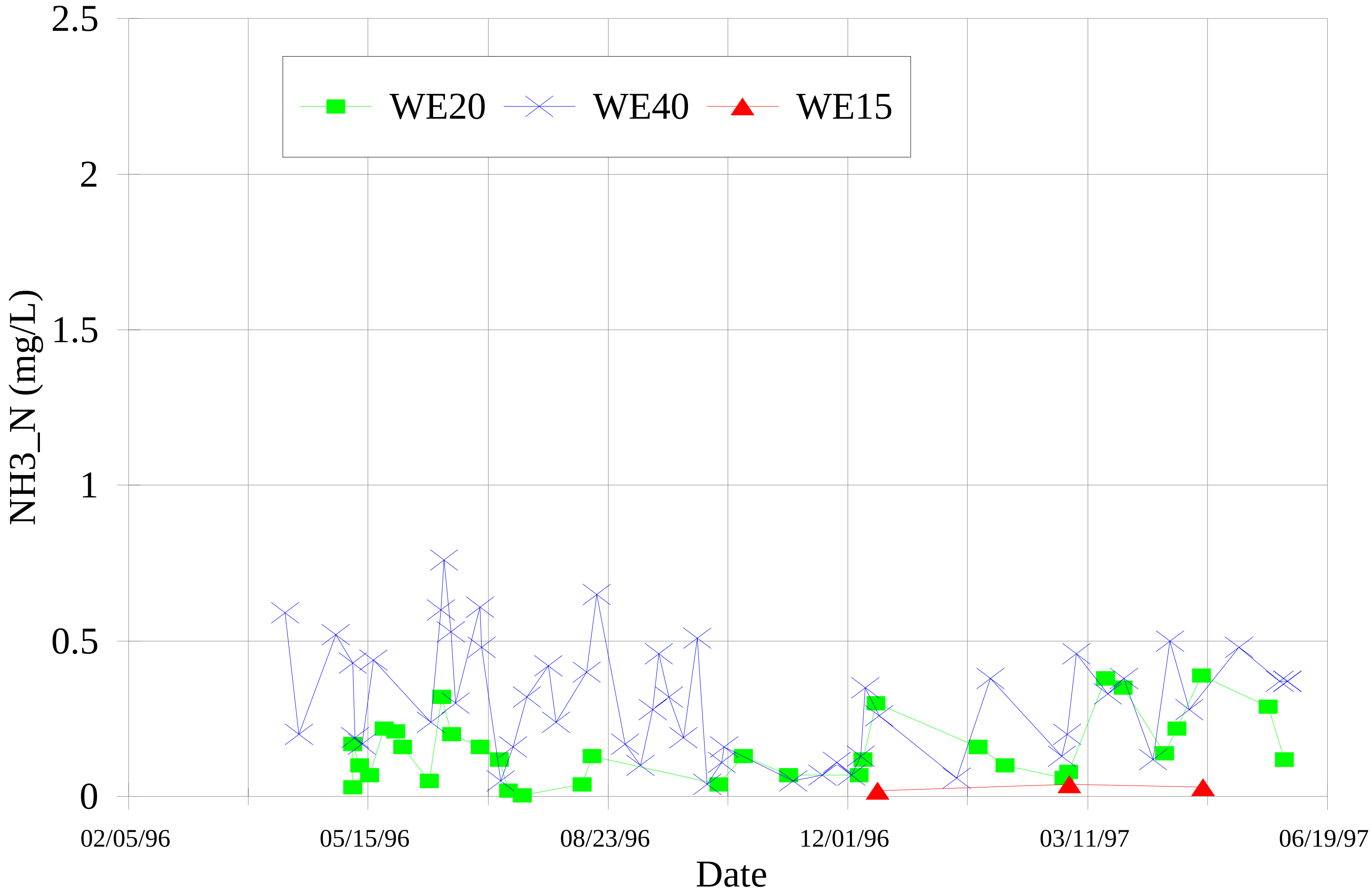

Figure A.18 Inlet EMC, overland grab, and wetfall sample concentrations: NH3_N, mg/L

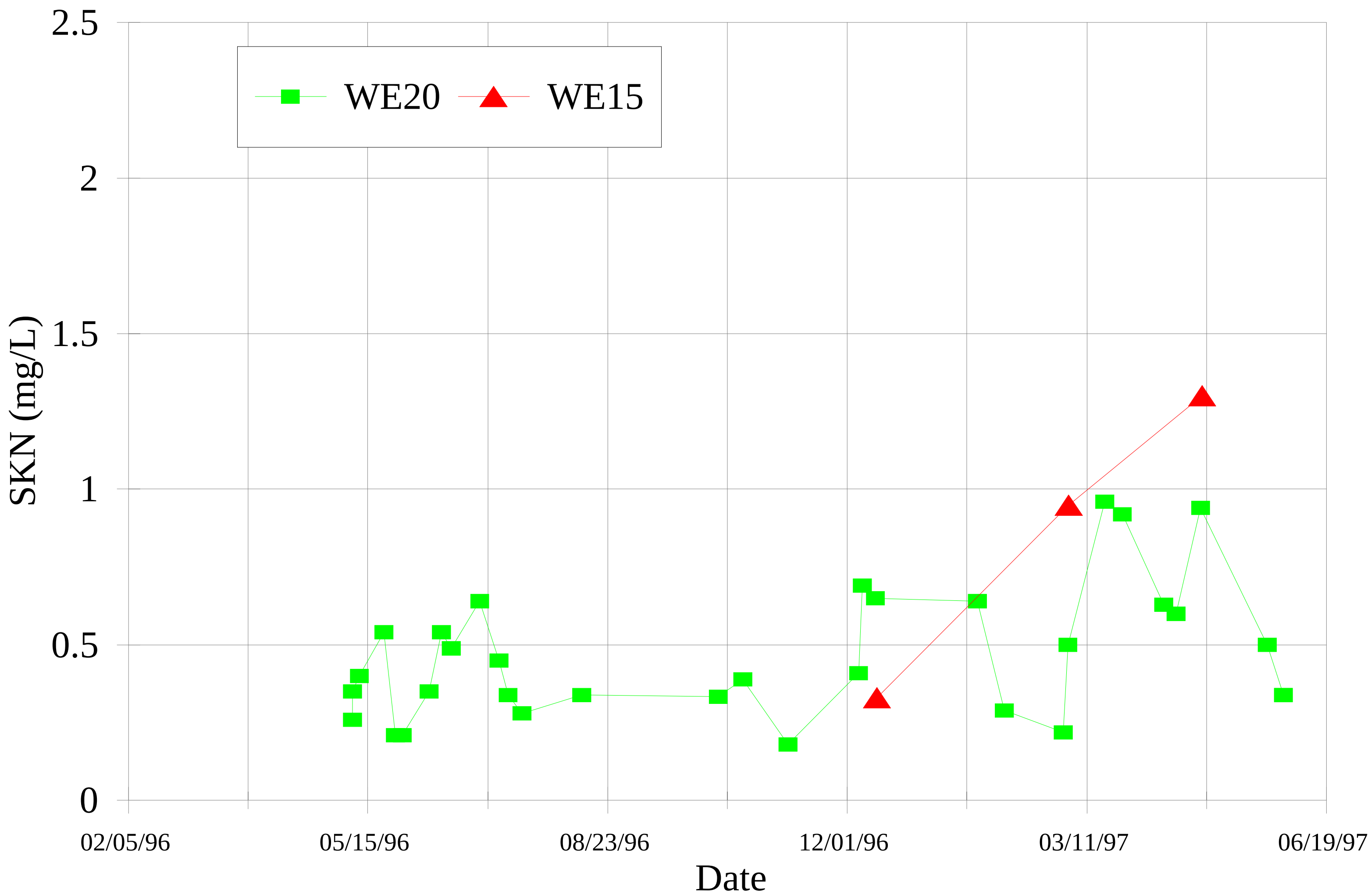

Figure A.19 Inlet EMC, overland grab, and wetfall sample concentrations: SKN, mg/L

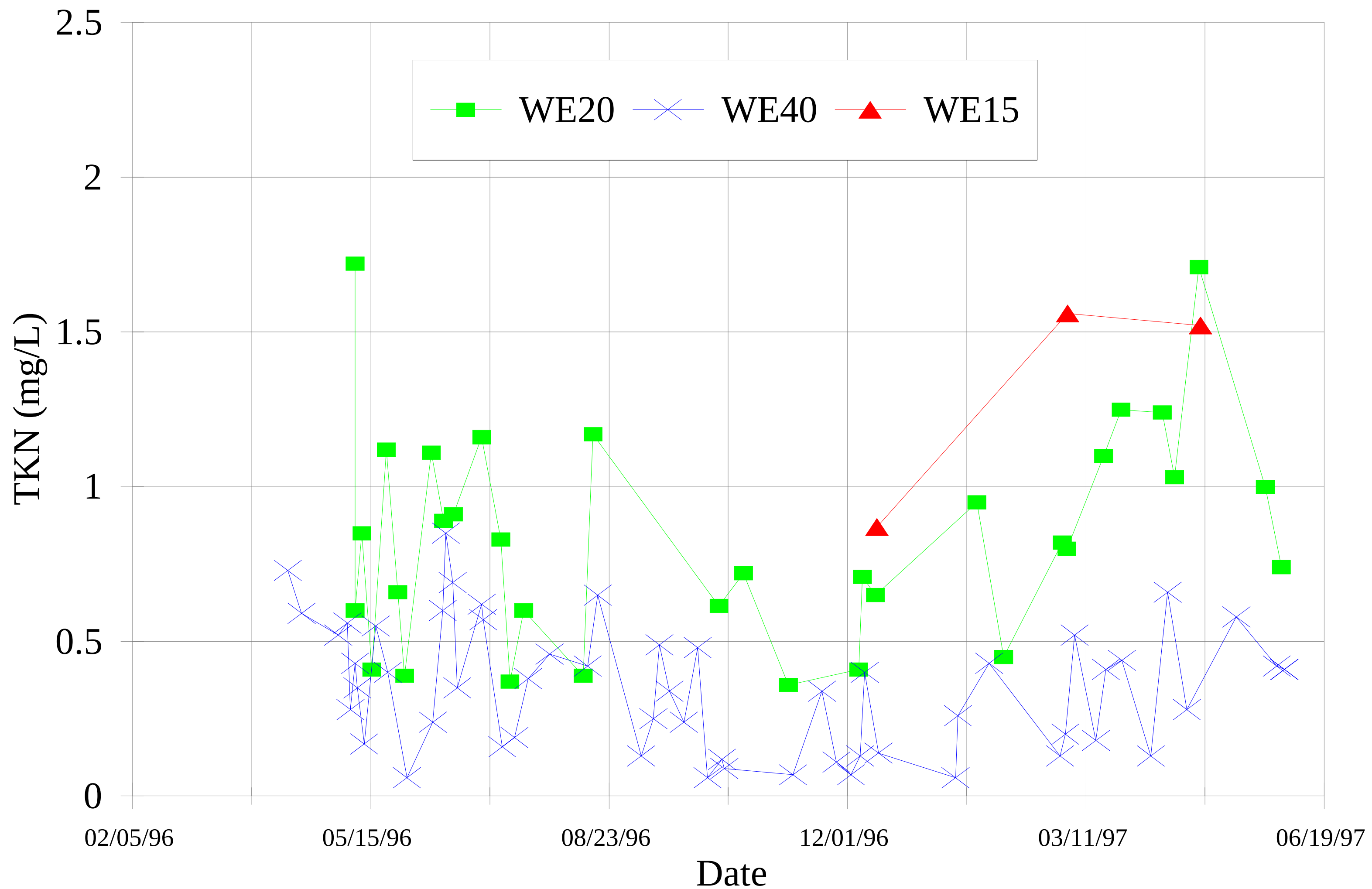

Figure A.20 Inlet EMC, overland grab, and wetfall sample concentrations: TKN, mg/L

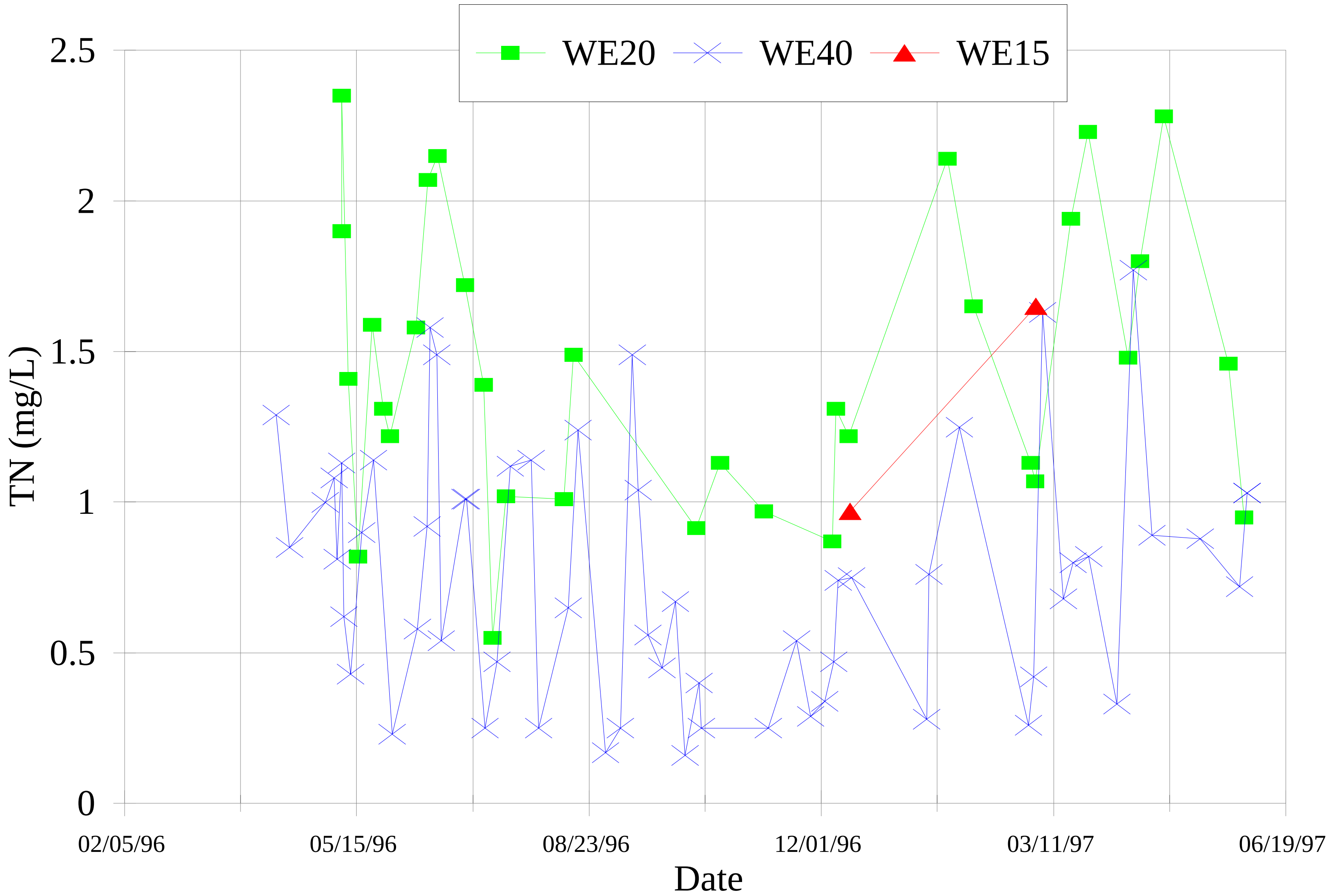

Figure A.21 Inlet EMC, overland grab, and wetfall sample concentrations: TN, mg/L

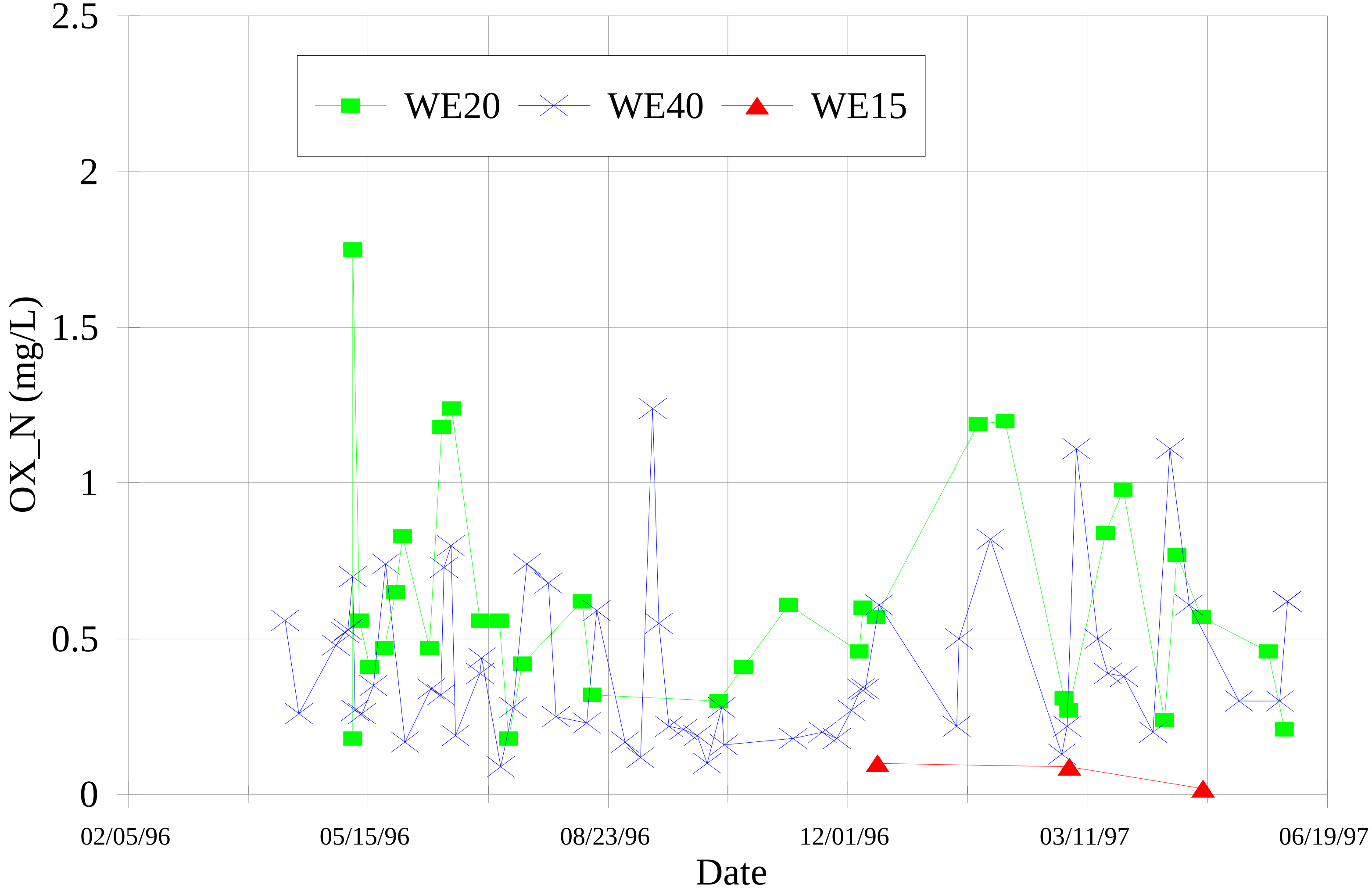

Figure A.22 Inlet EMC, overland grab, and wetfall sample concentrations: OX_N, mg/L

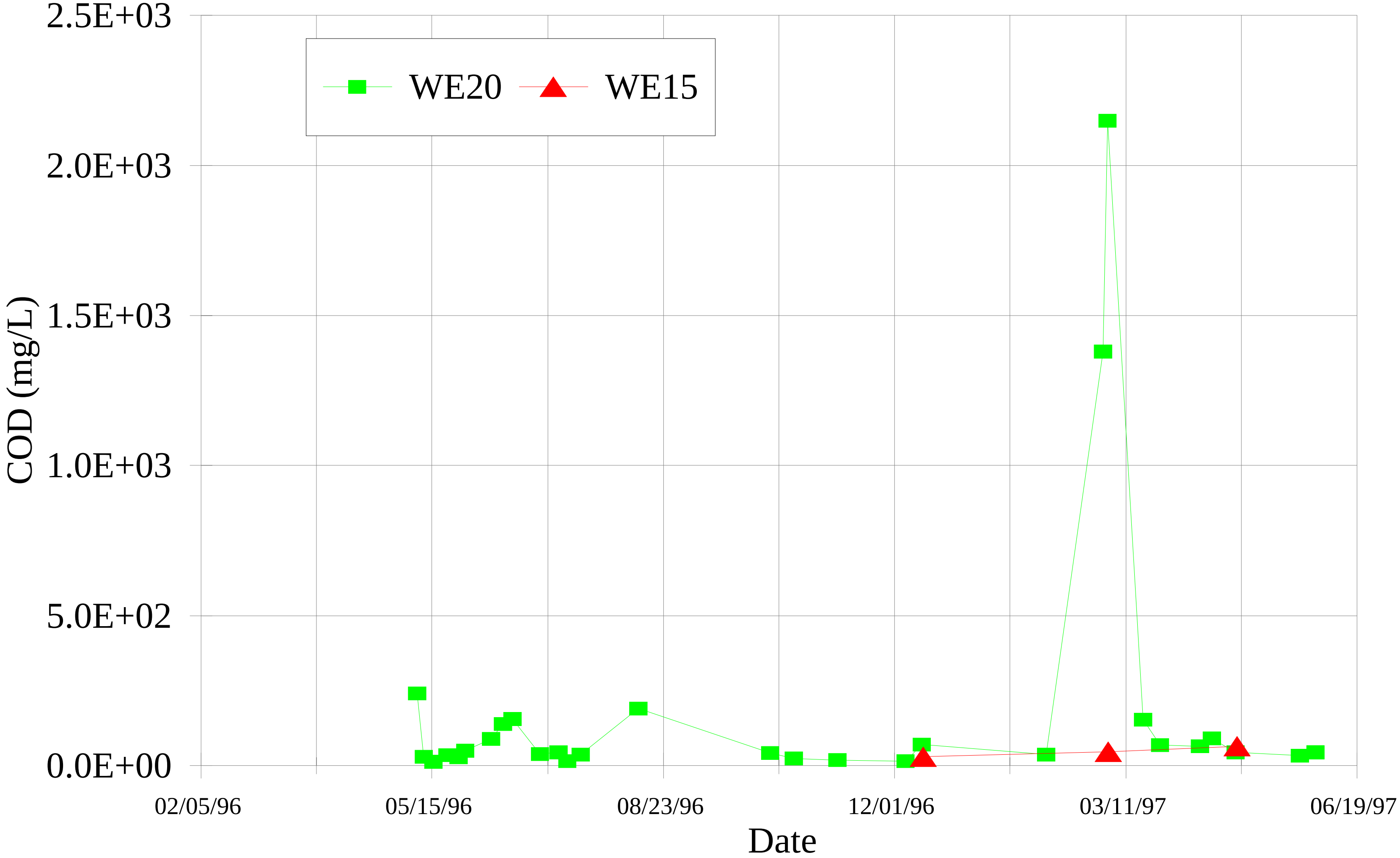

Figure A.23 Inlet EMC, overland grab, and wetfall sample concentrations: COD, mg/L

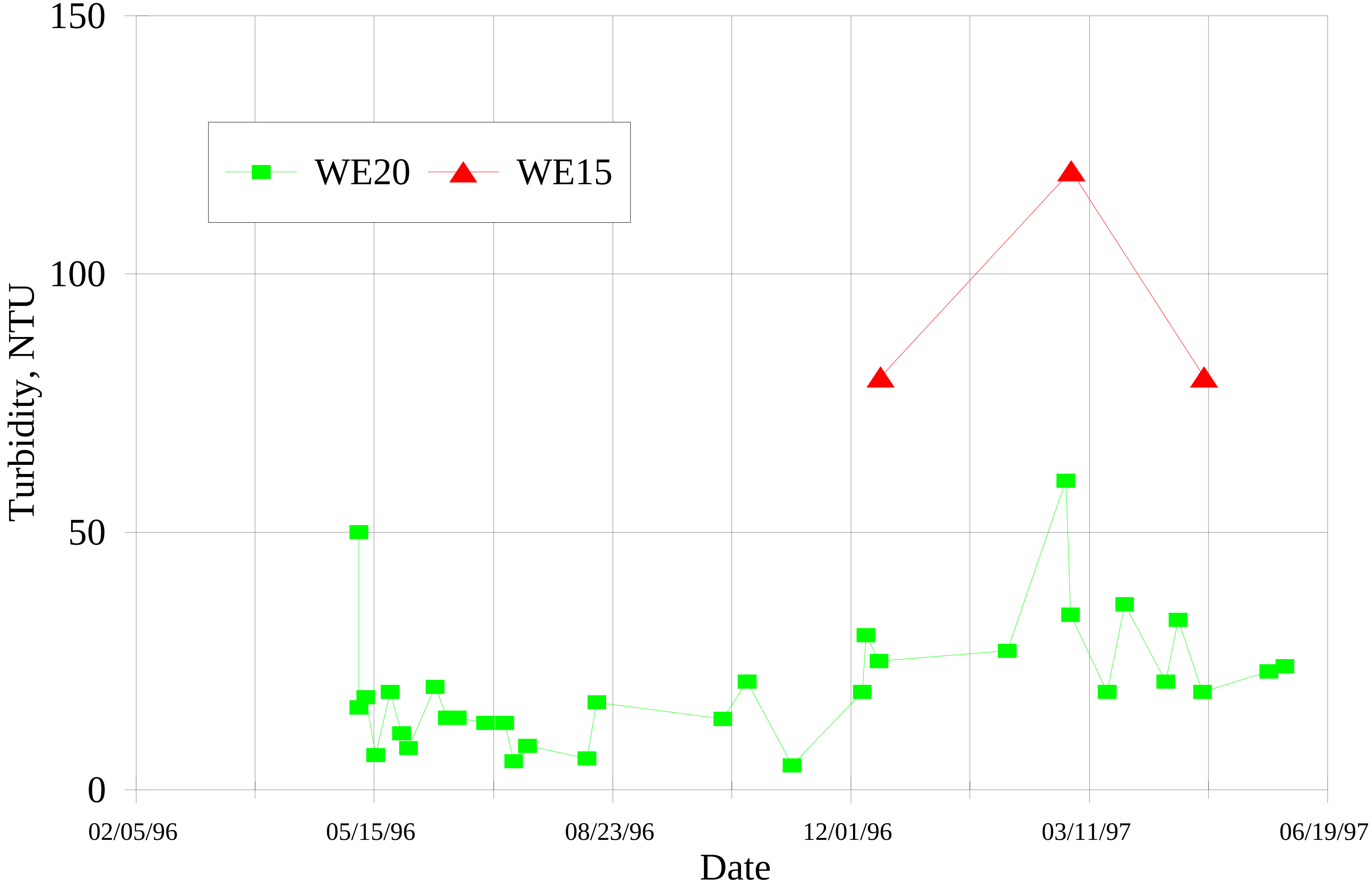

Figure A.24 Inlet EMC, overland grab, and wetfall sample concentrations: Turbidity,NTU

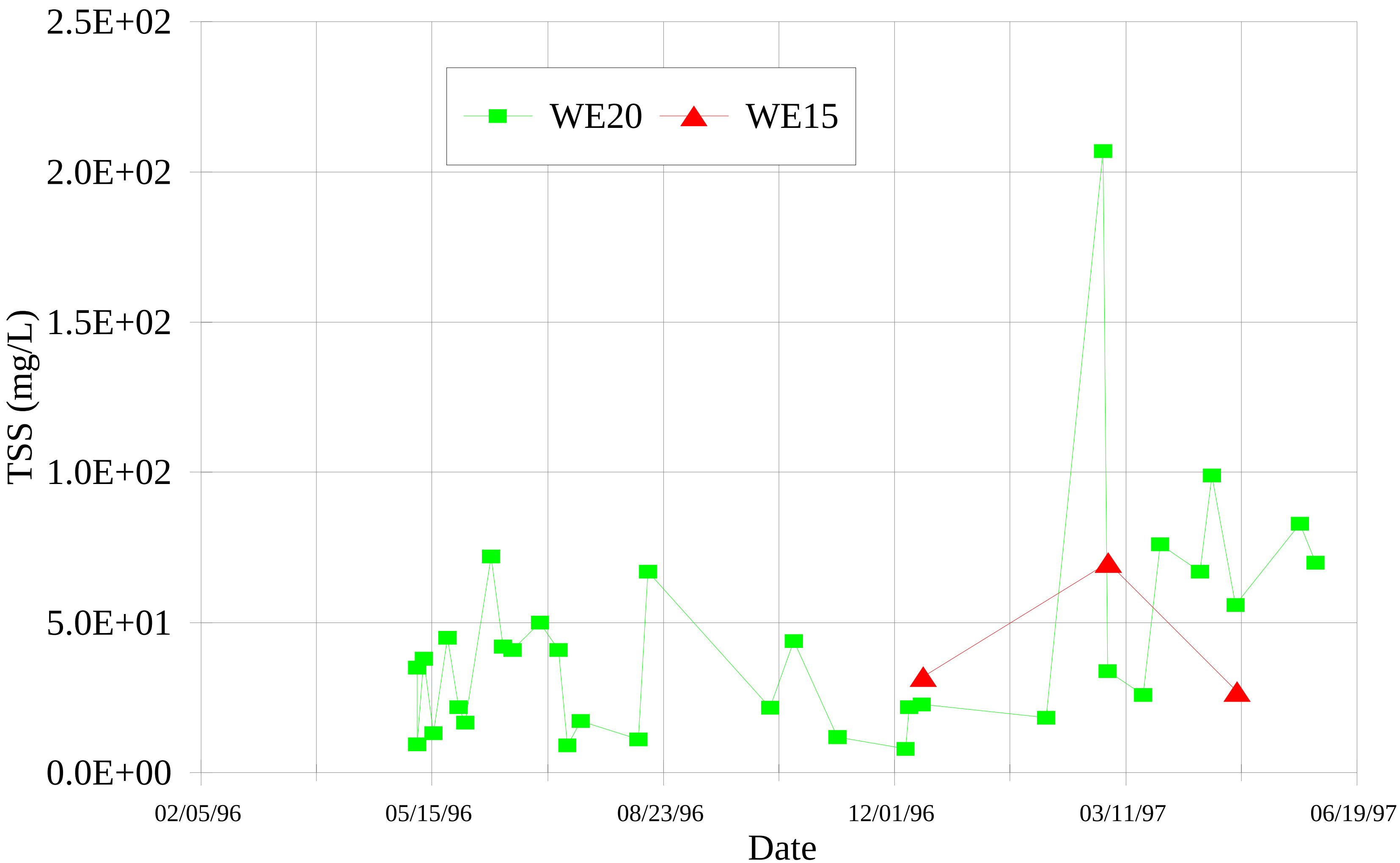

Figure A.25 Inlet EMC and overland grab sample concentrations: TSS, mg/L

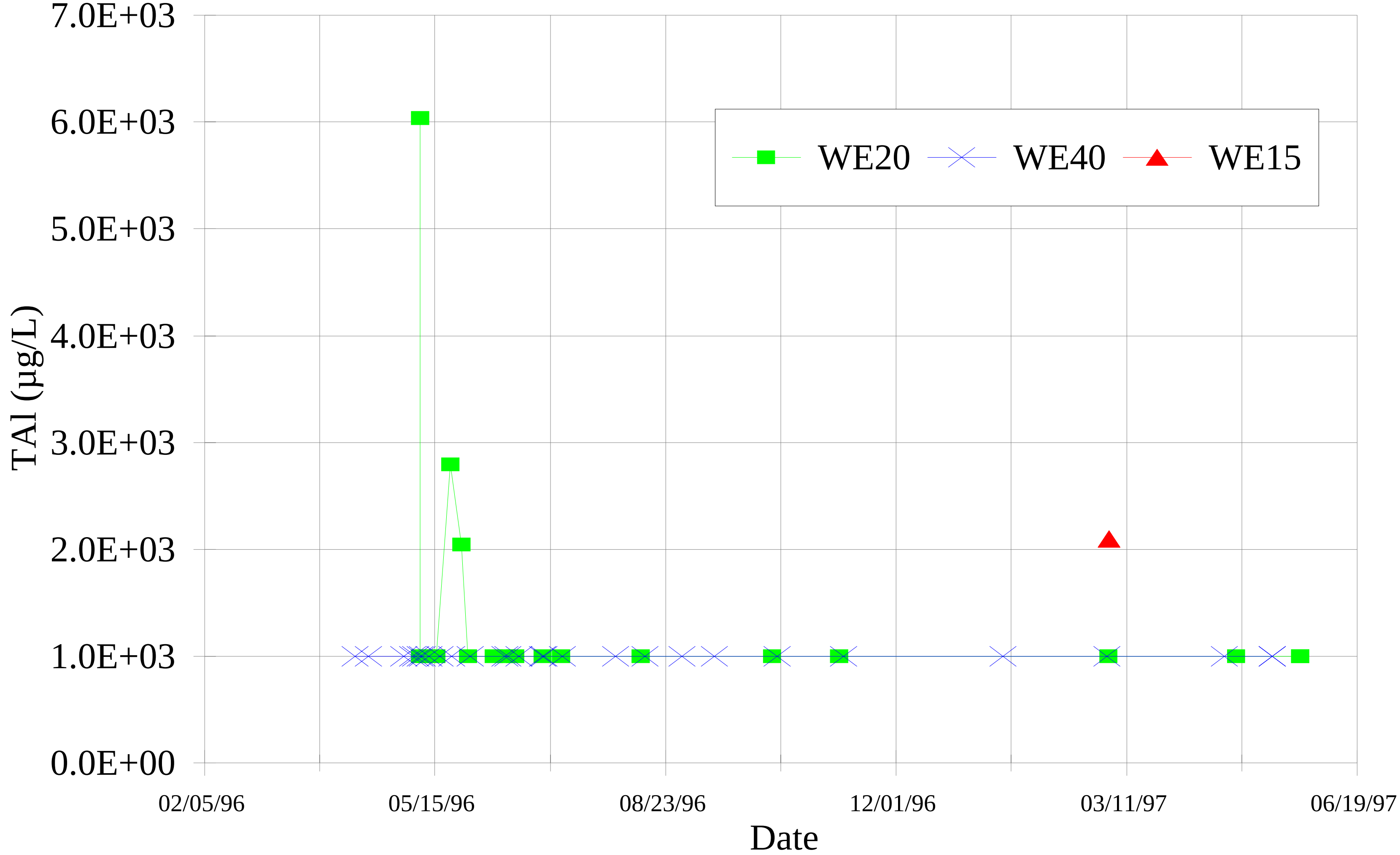

Figure A.26 Inlet EMC, overland grab, and wetfall sample concentrations: TAl, µg/L

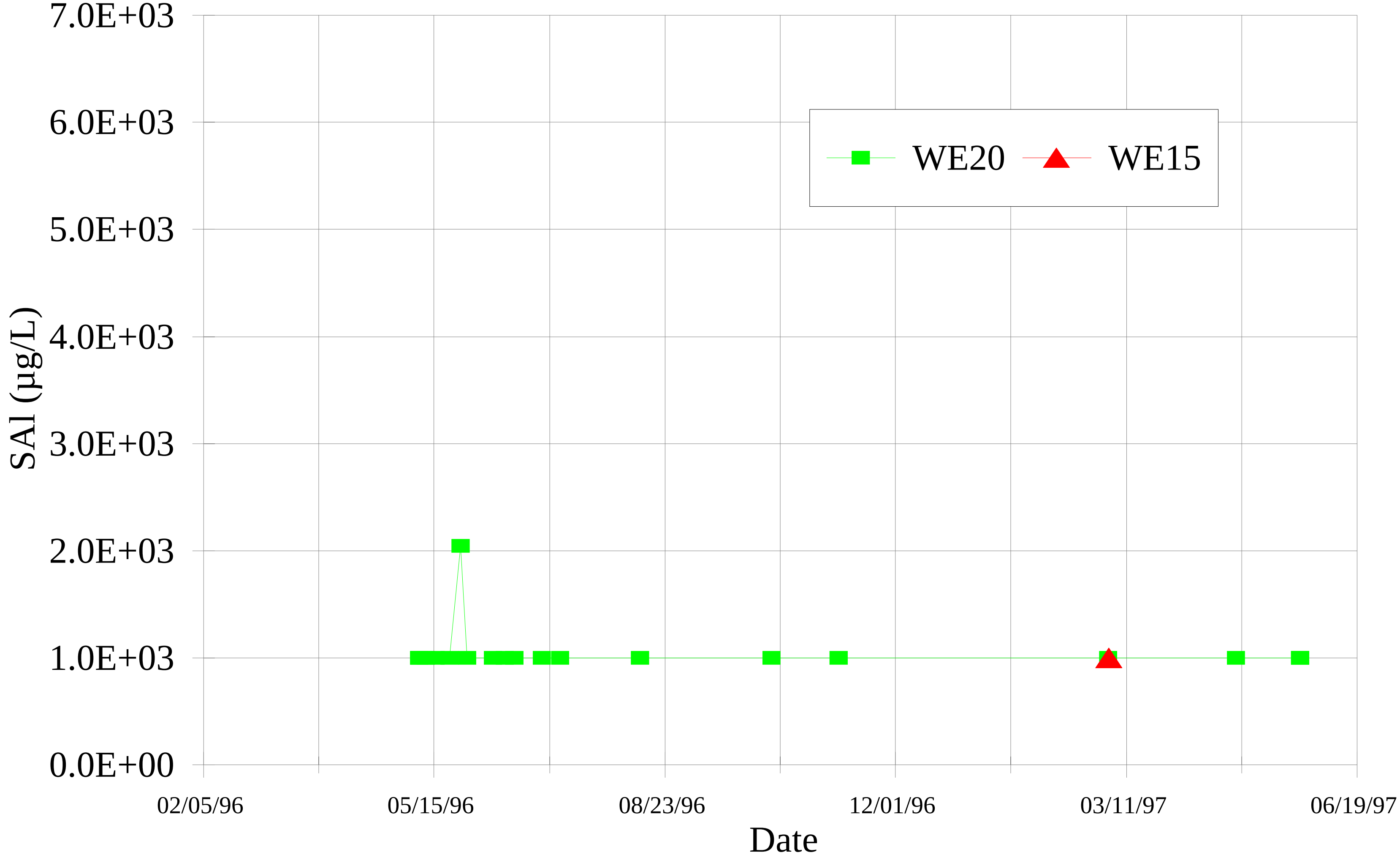

Figure A.27 Inlet EMC, overland grab, and wetfall sample concentrations: SAl, µg/L

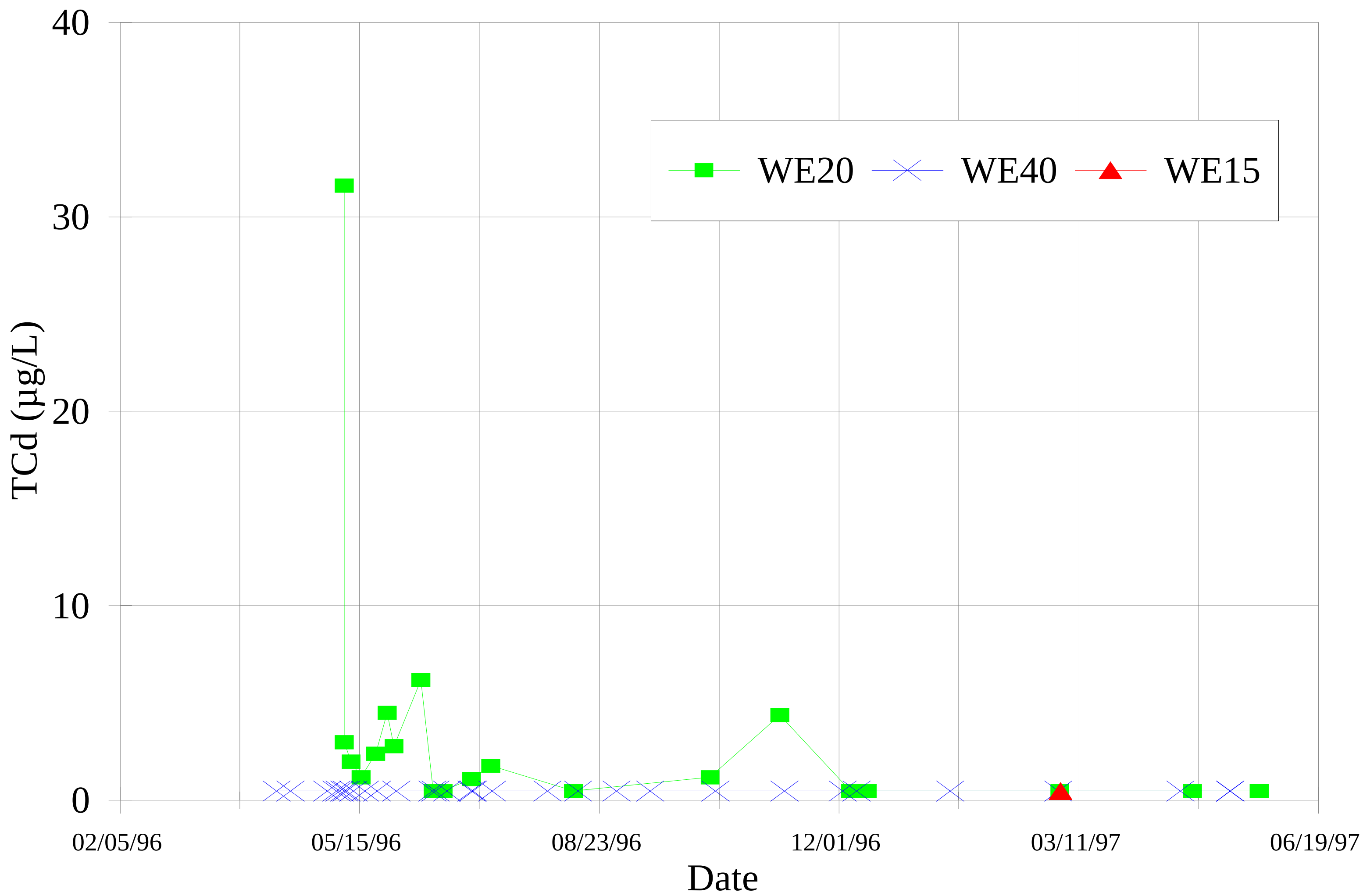

Figure A.28 Inlet EMC, overland grab, and wetfall sample concentrations: TCd, µg/L

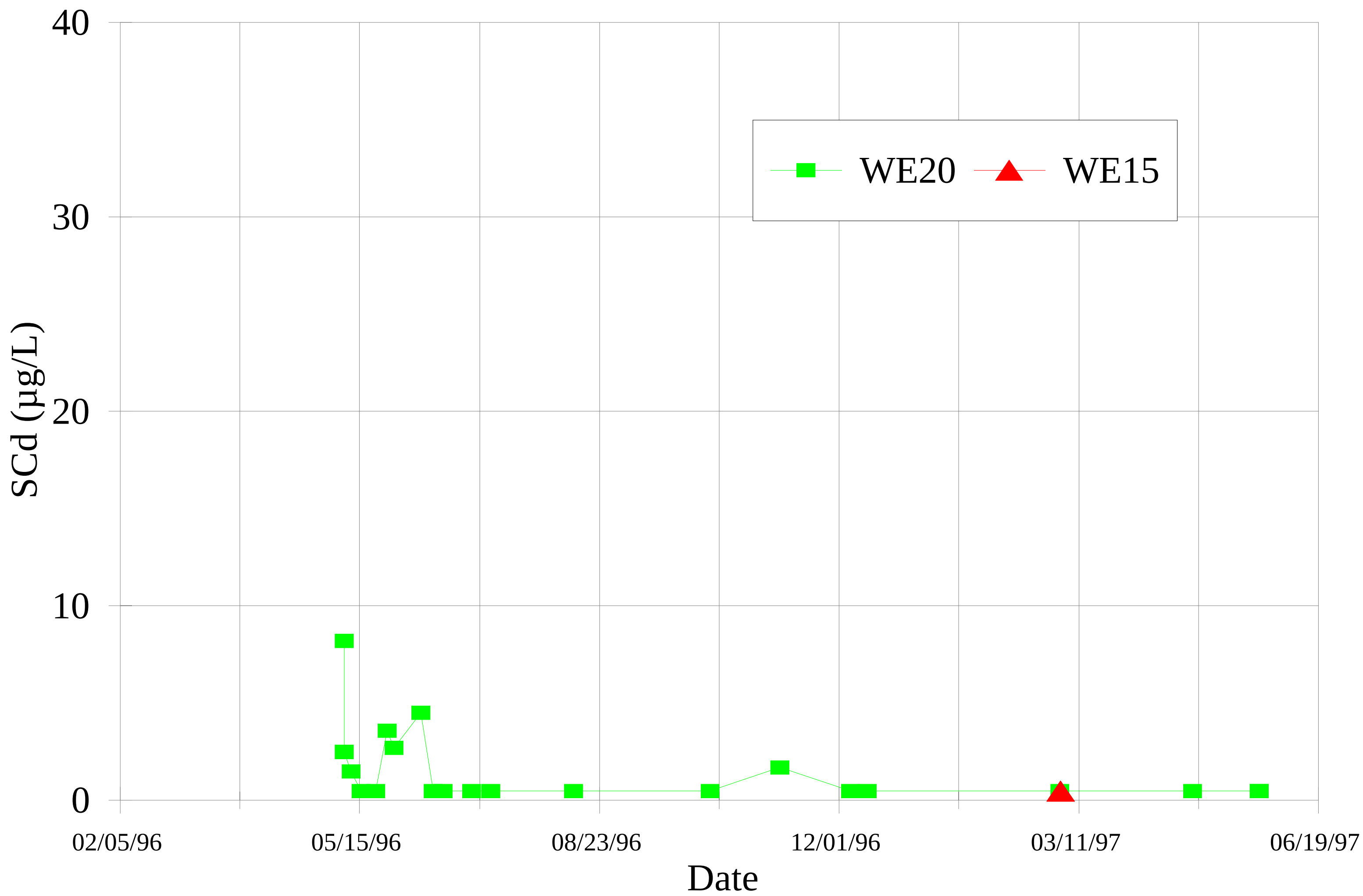

Figure A.29 Inlet EMC, overland grab, and wetfall sample concentrations: SCd, µg/L

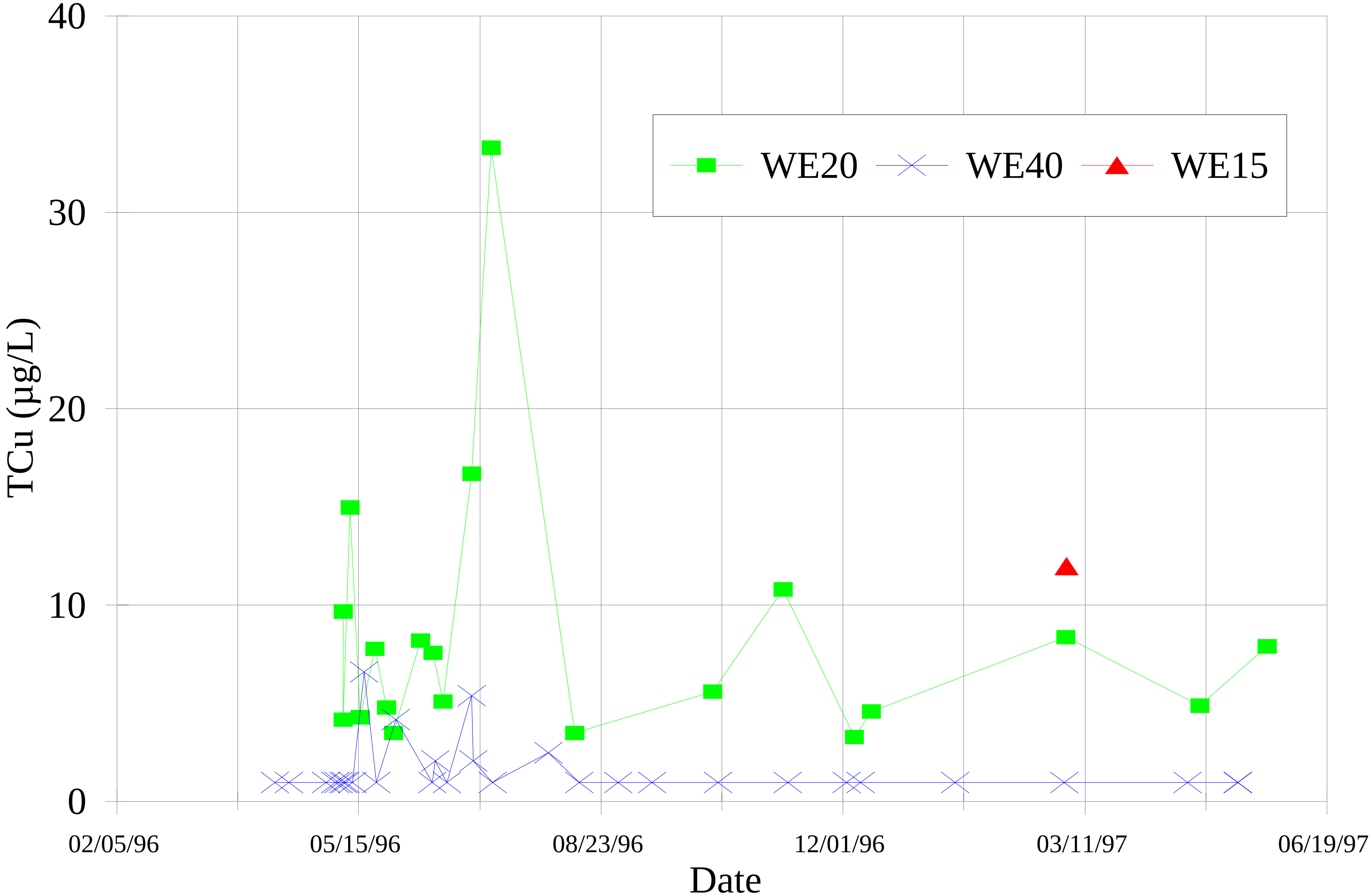

Figure A.30 Inlet EMC, overland grab, and wetfall sample concentrations: TCu, µg/L

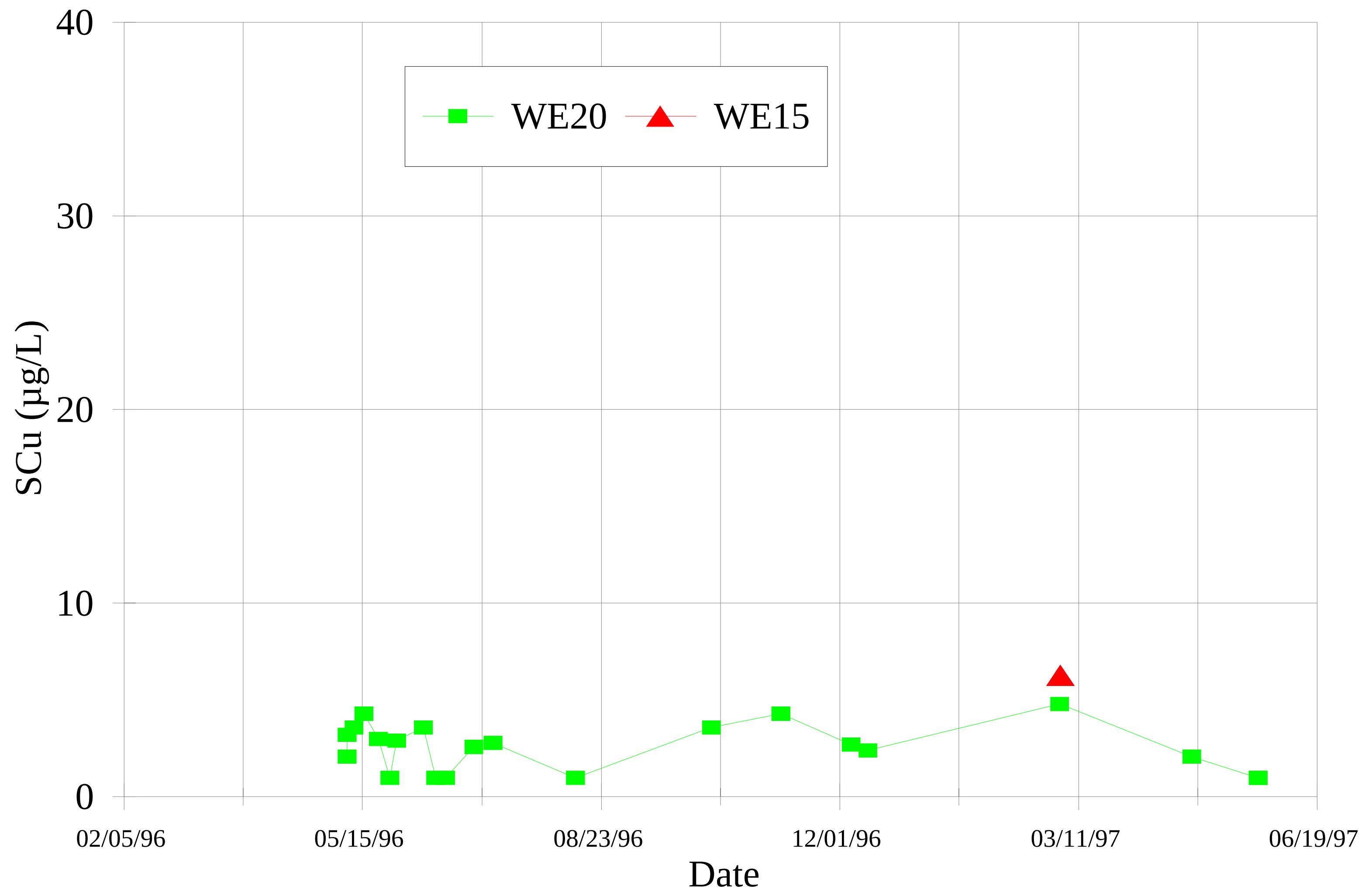

Figure A.31 Inlet EMC, overland grab, and wetfall sample concentrations: SCu, µg/L

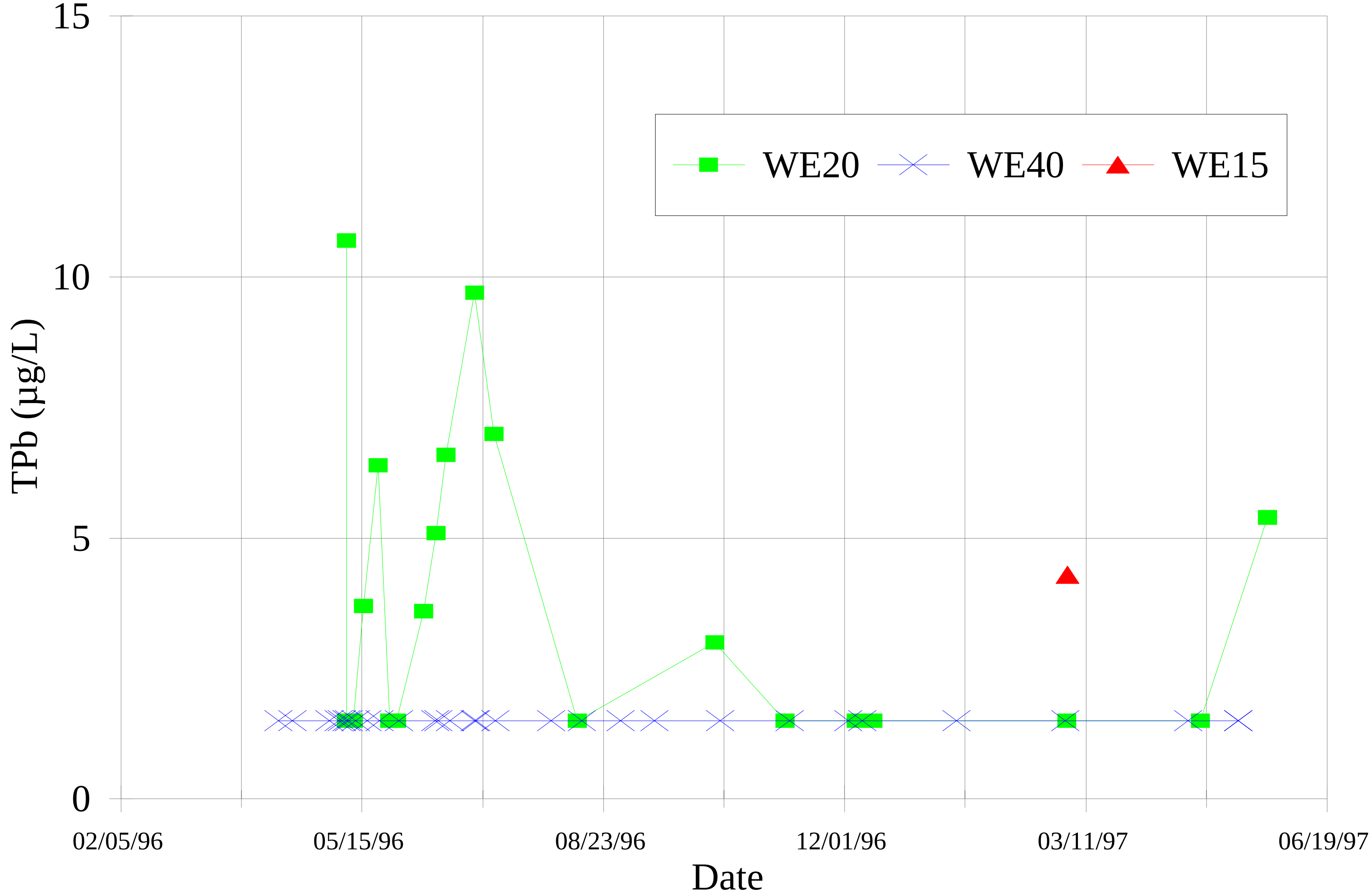

Figure A.32 Inlet EMC, overland grab, and wetfall sample concentrations: TPb, µg/L

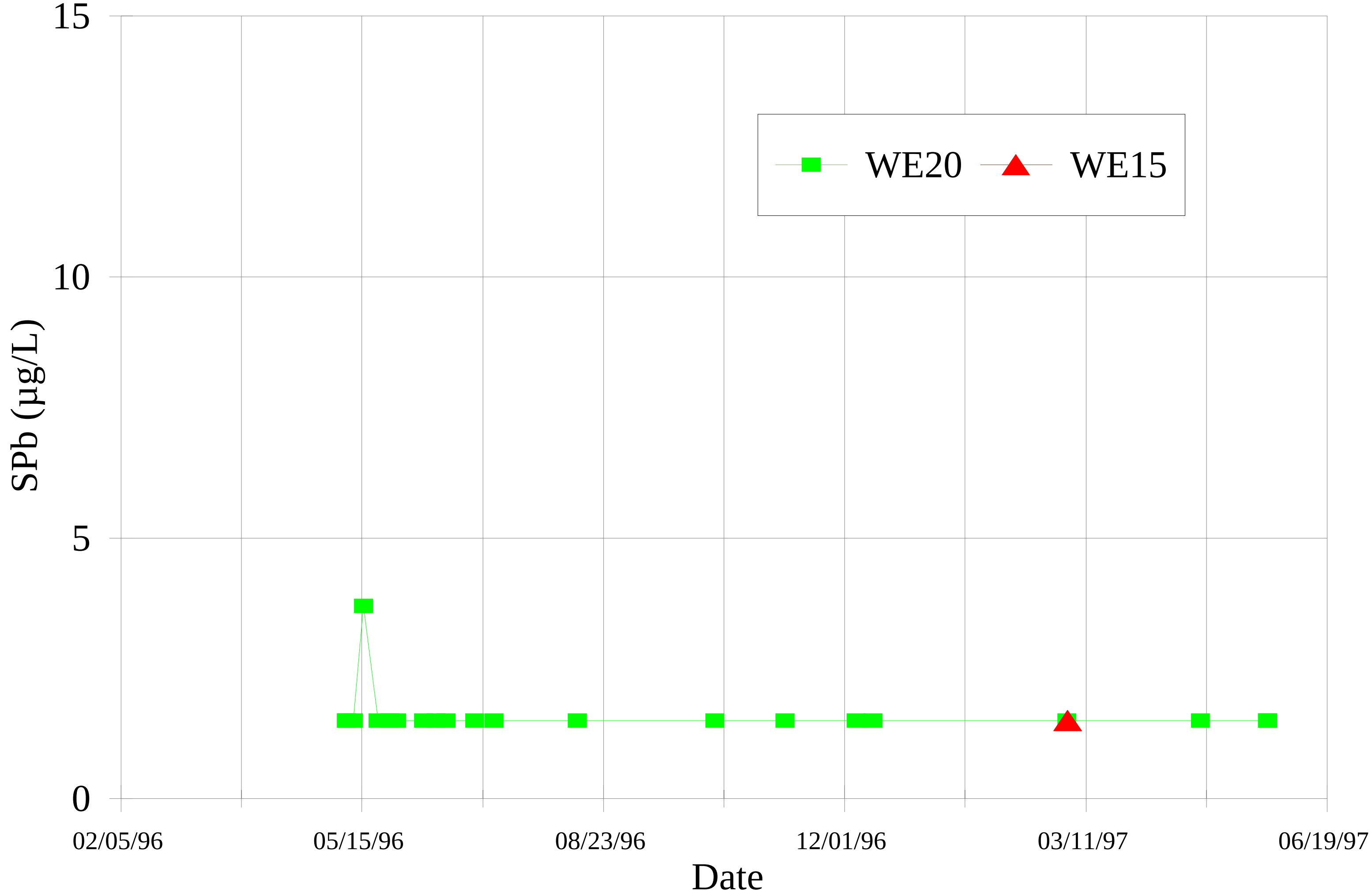

Figure A.33 Inlet EMC, overland grab, and wetfall sample concentrations: SPb, µg/L

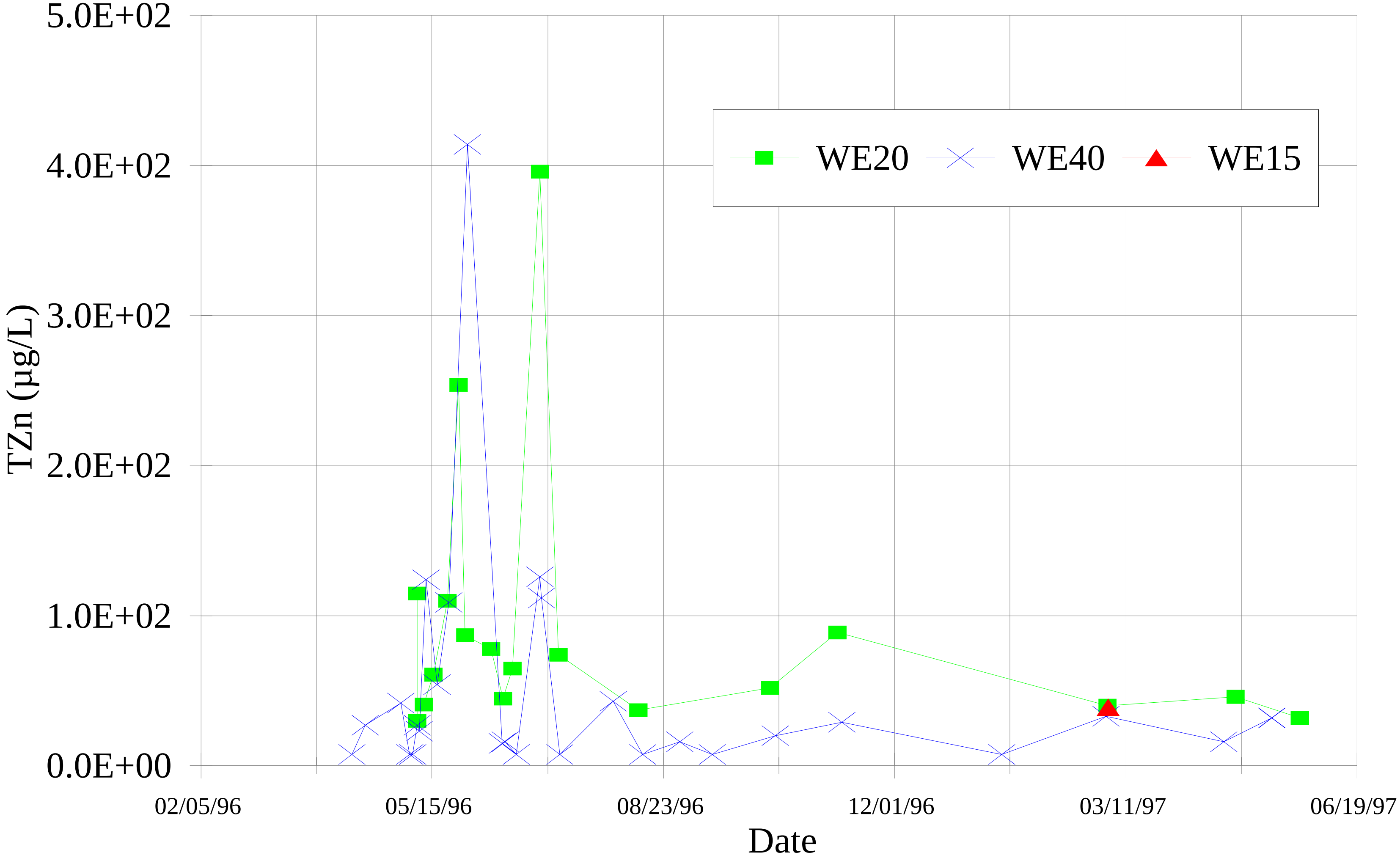

Figure A.34 Inlet EMC, overland grab, and wetfall sample concentrations: TZn, µg/L

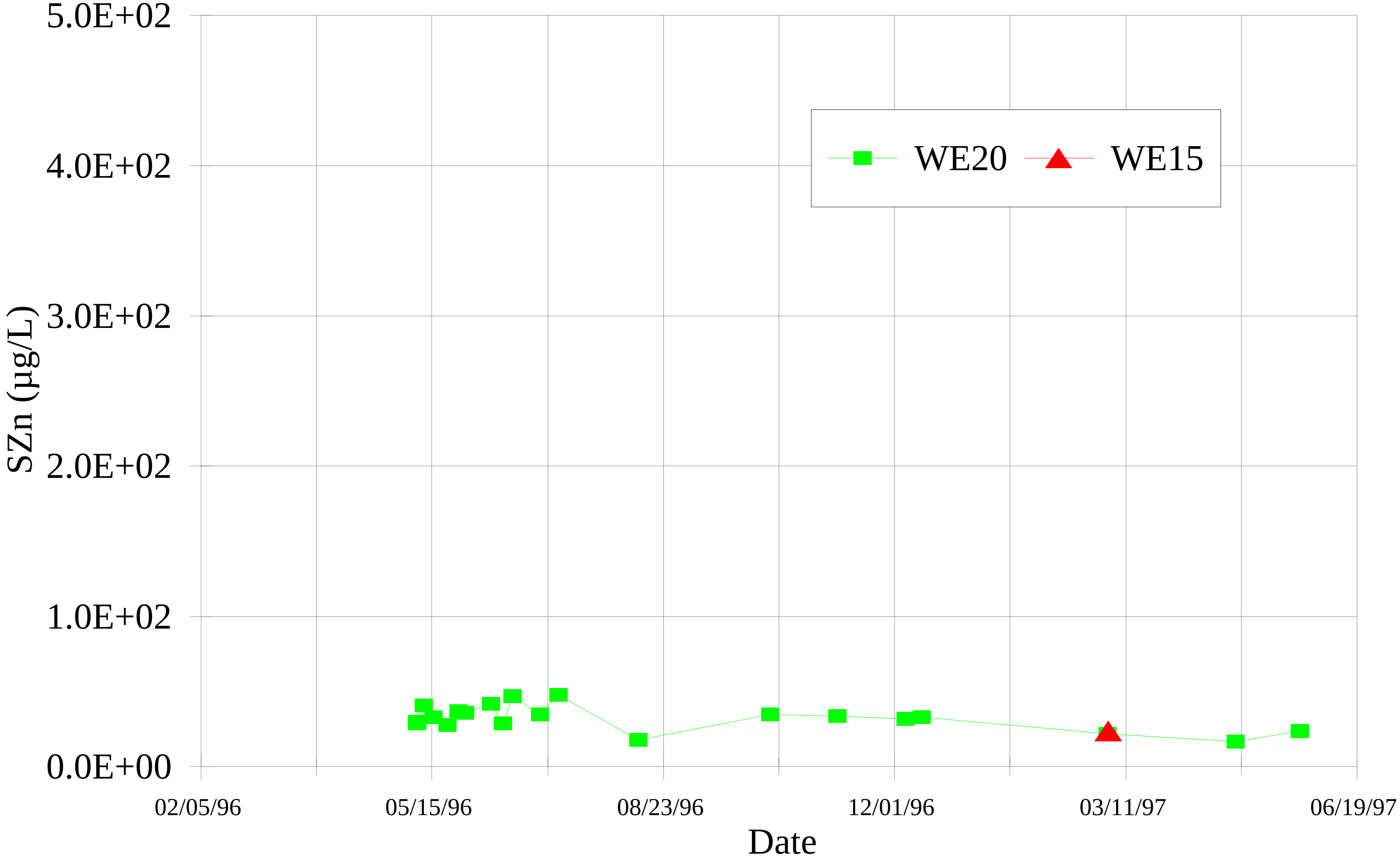

Figure A.35 Inlet EMC, overland grab, and wetfall sample concentrations: SZn, µg/L

APPENDIX B
Lysimeter Time Course Plots

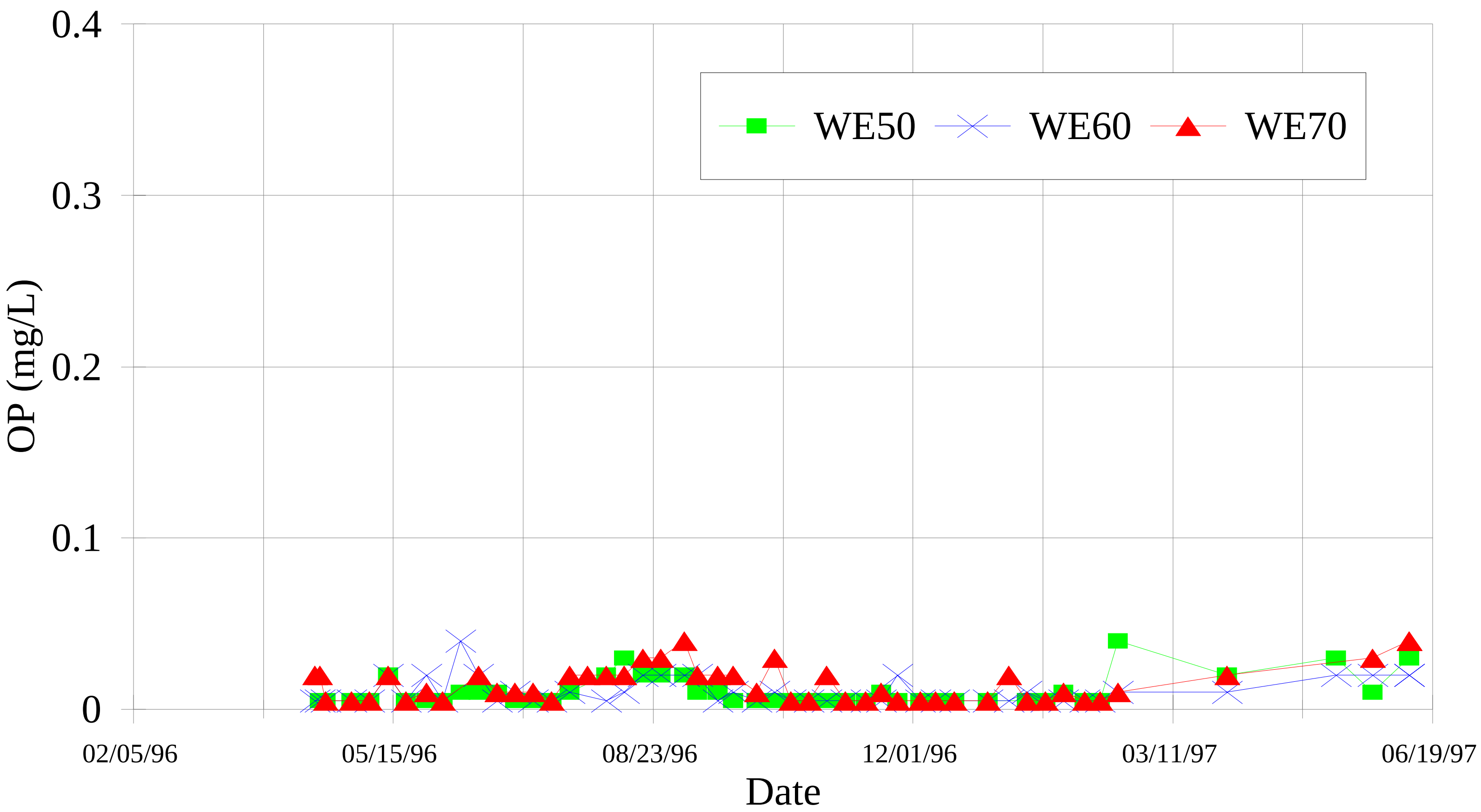

Figure B.1 Lysimeter OP data, mg/L

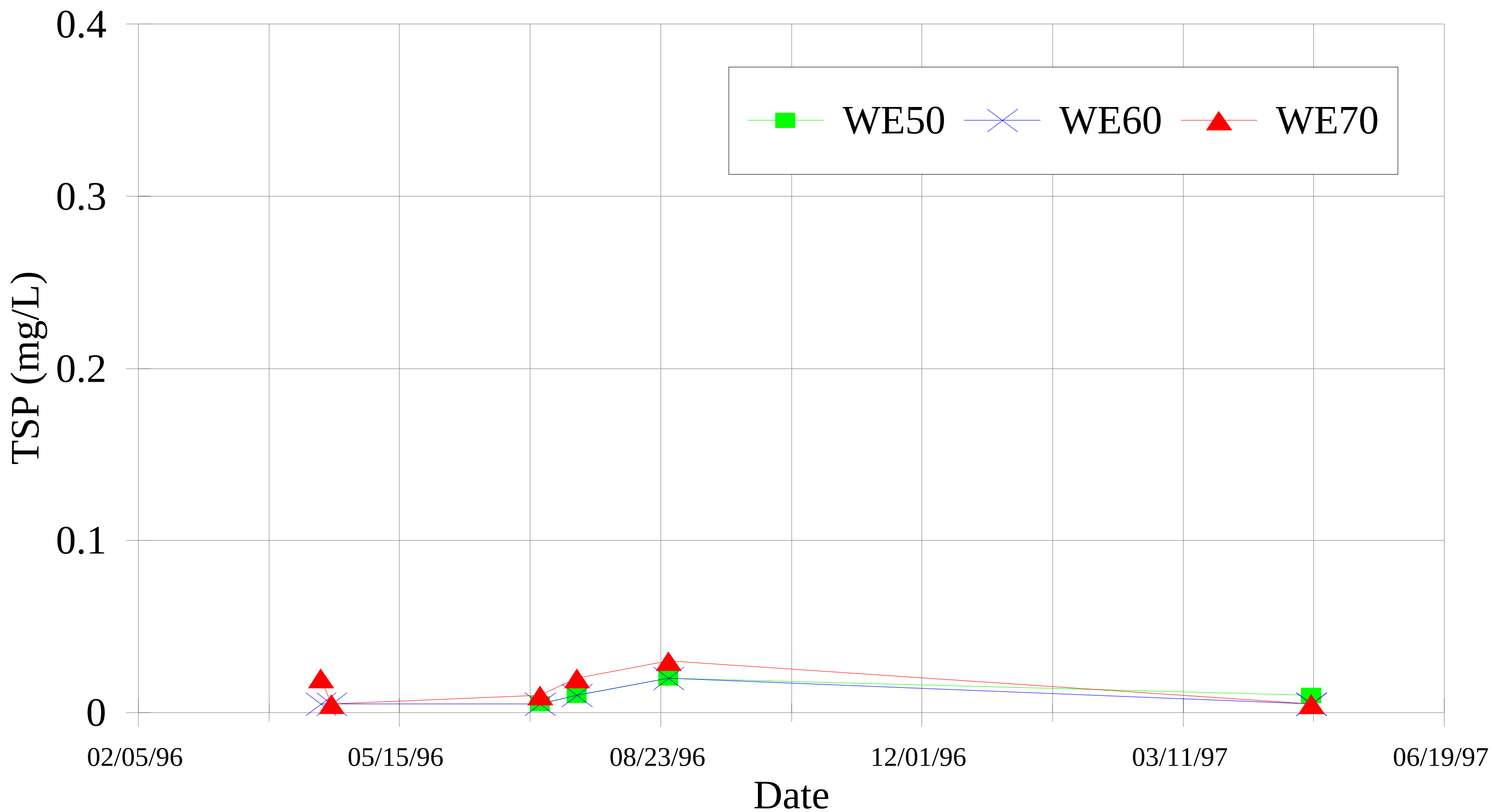

Figure B.2 Lysimeter TSP data, mg/L

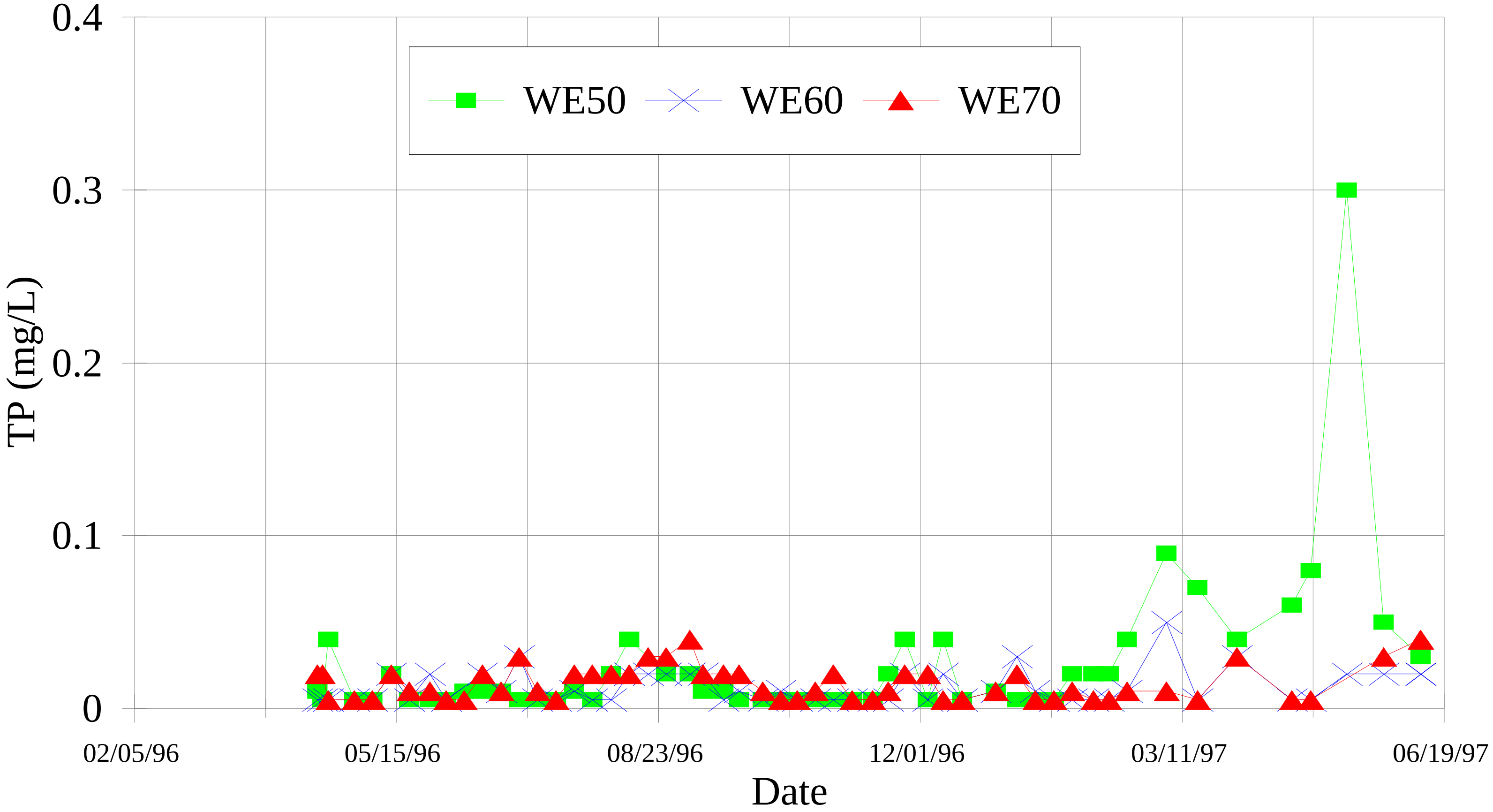

Figure B.3 Lysimeter TP data, mg/L

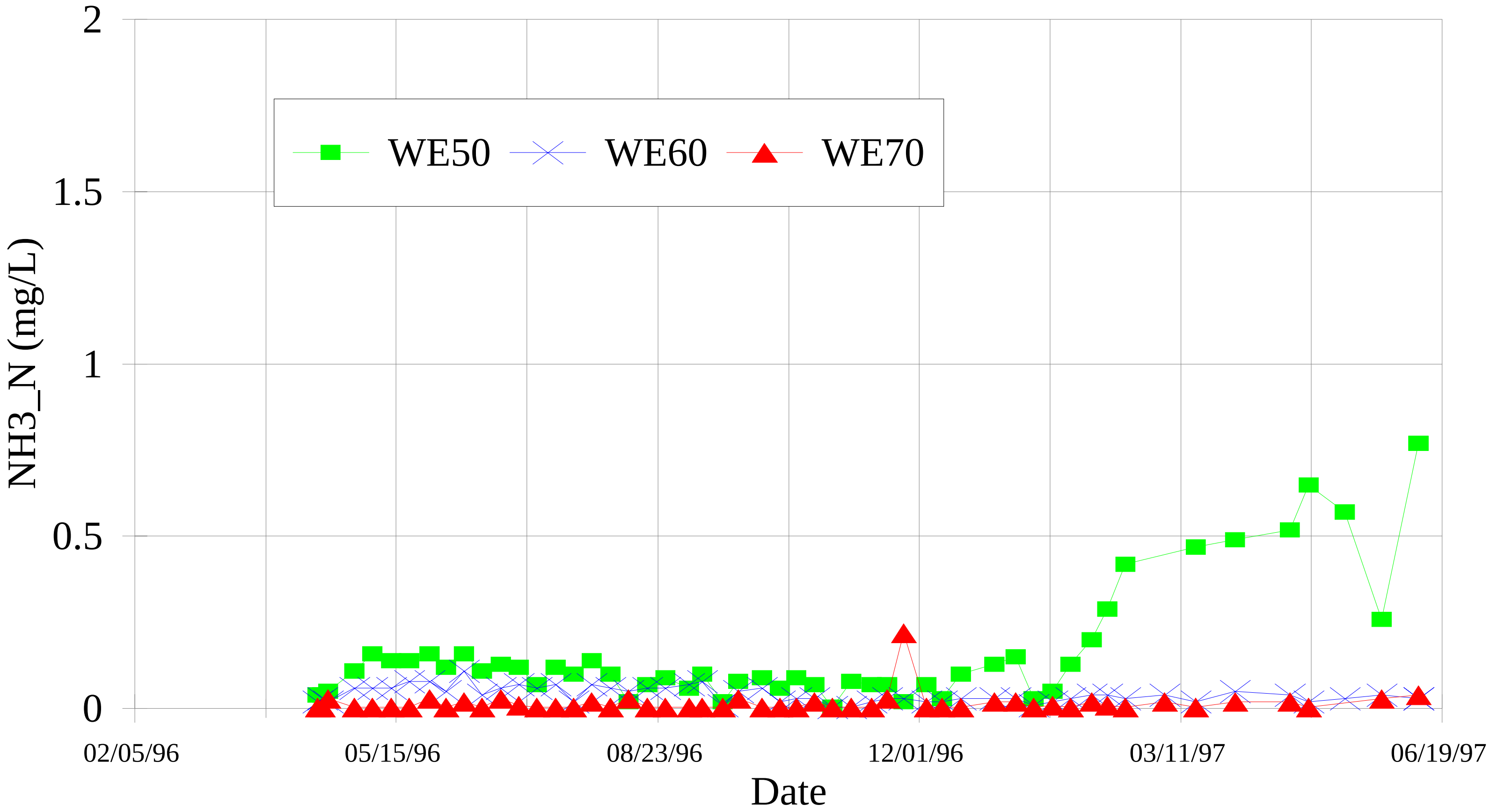

Figure B.4 Lysimeter NH3_N data, mg/L

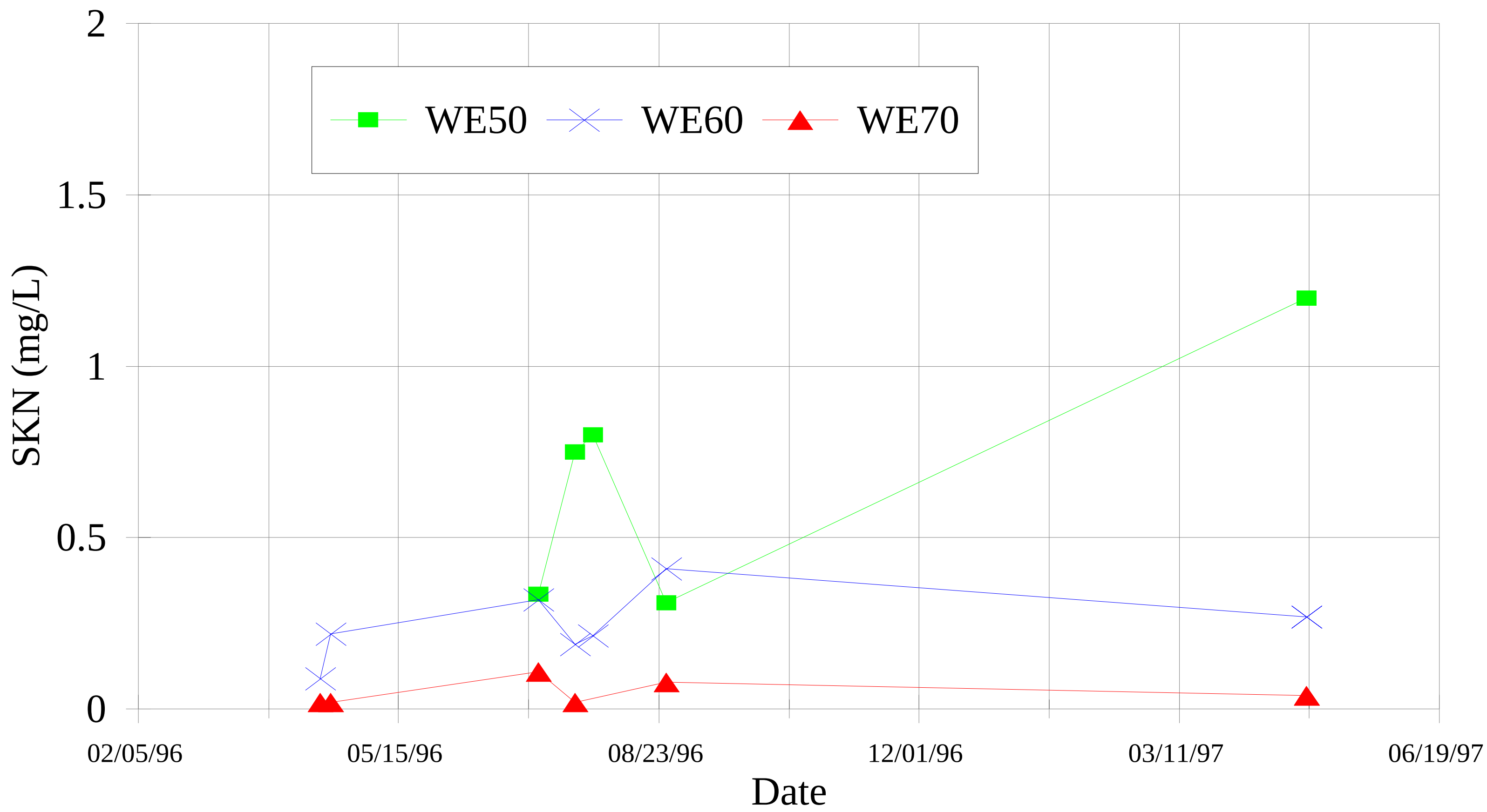

Figure B.5 Lysimeter SKN data, mg/L

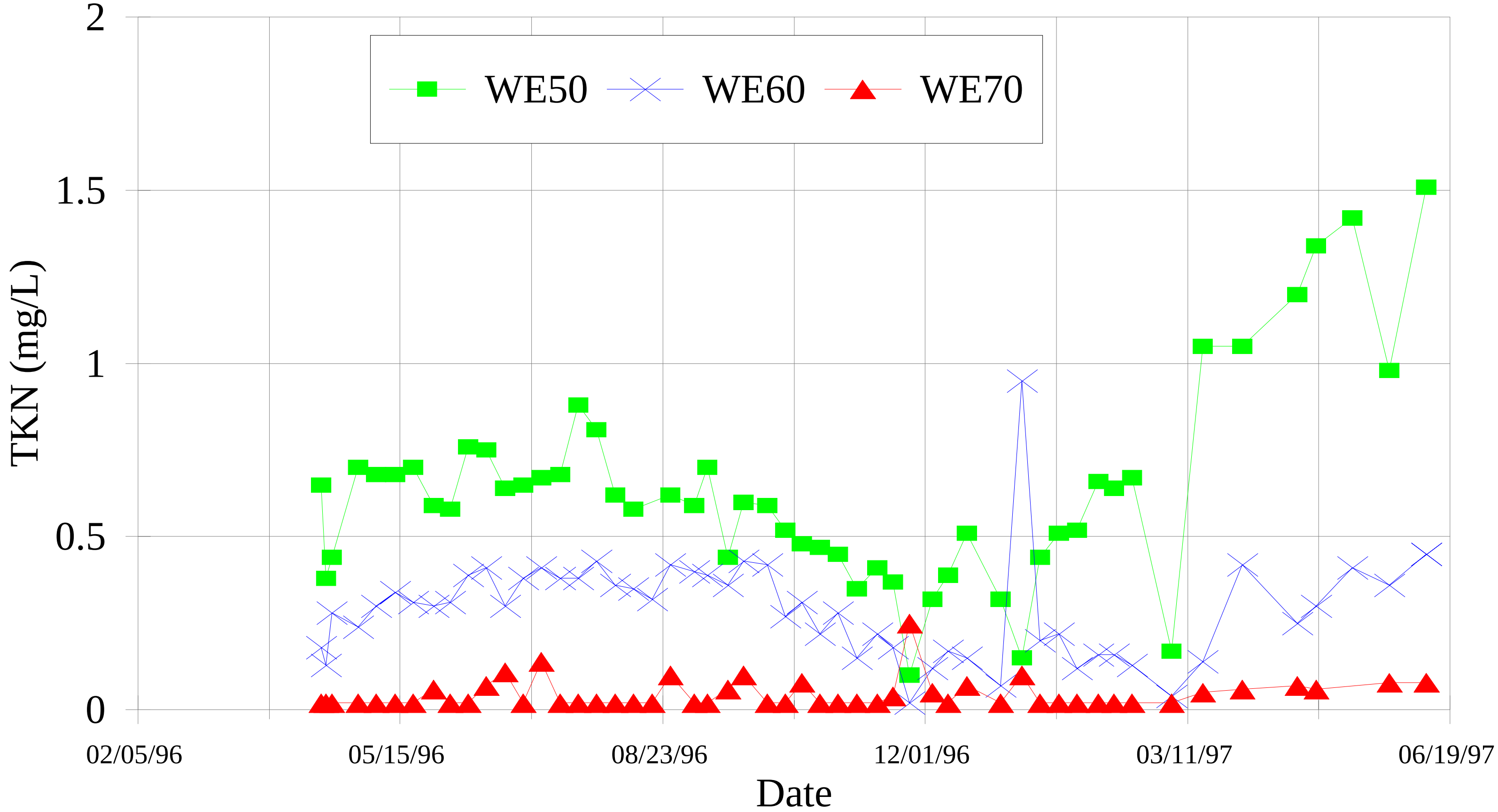

Figure B.6 Lysimeter TKN data, mg/L

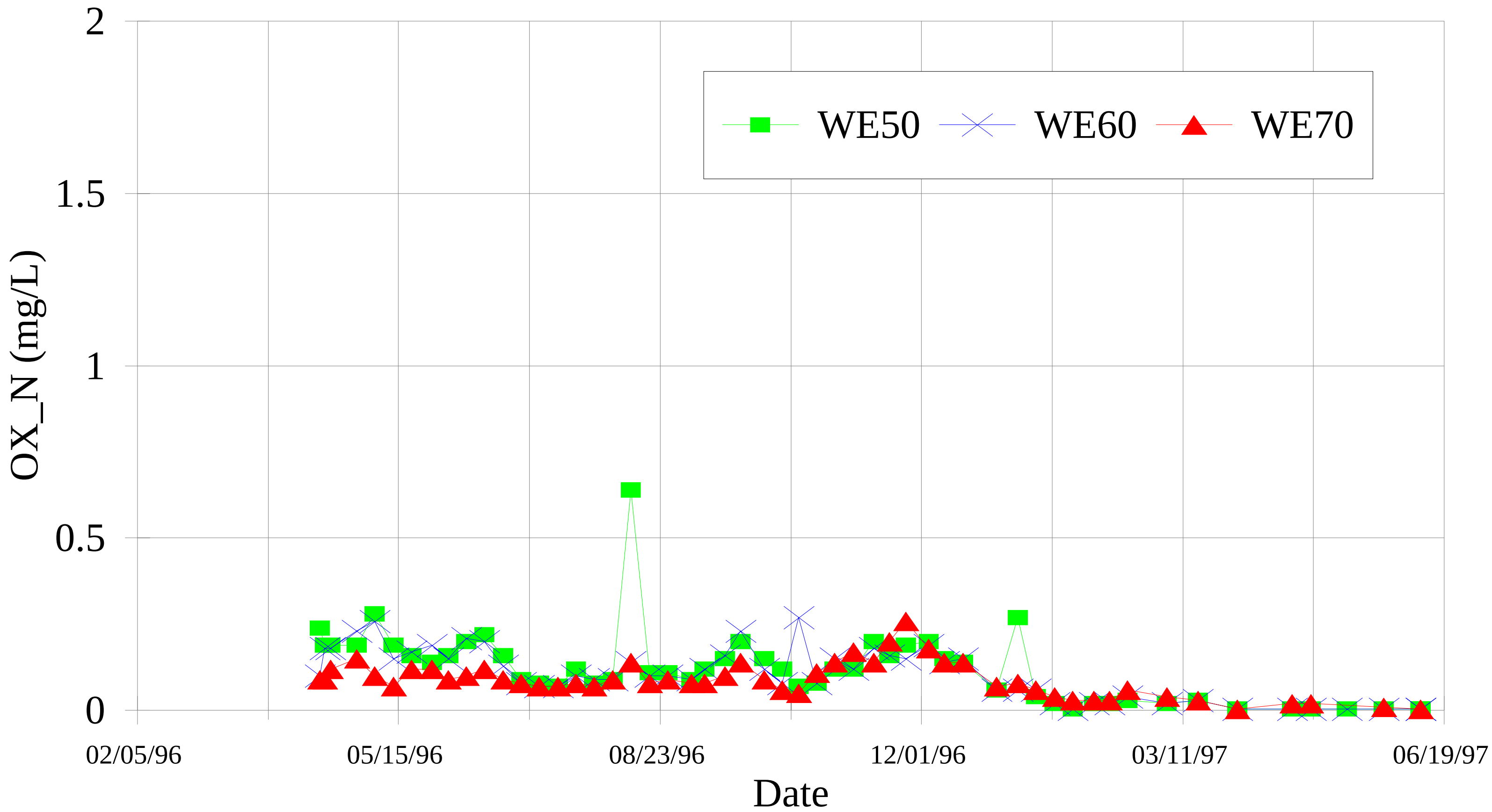

Figure B.7 Lysimeter OX_N data, mg/L

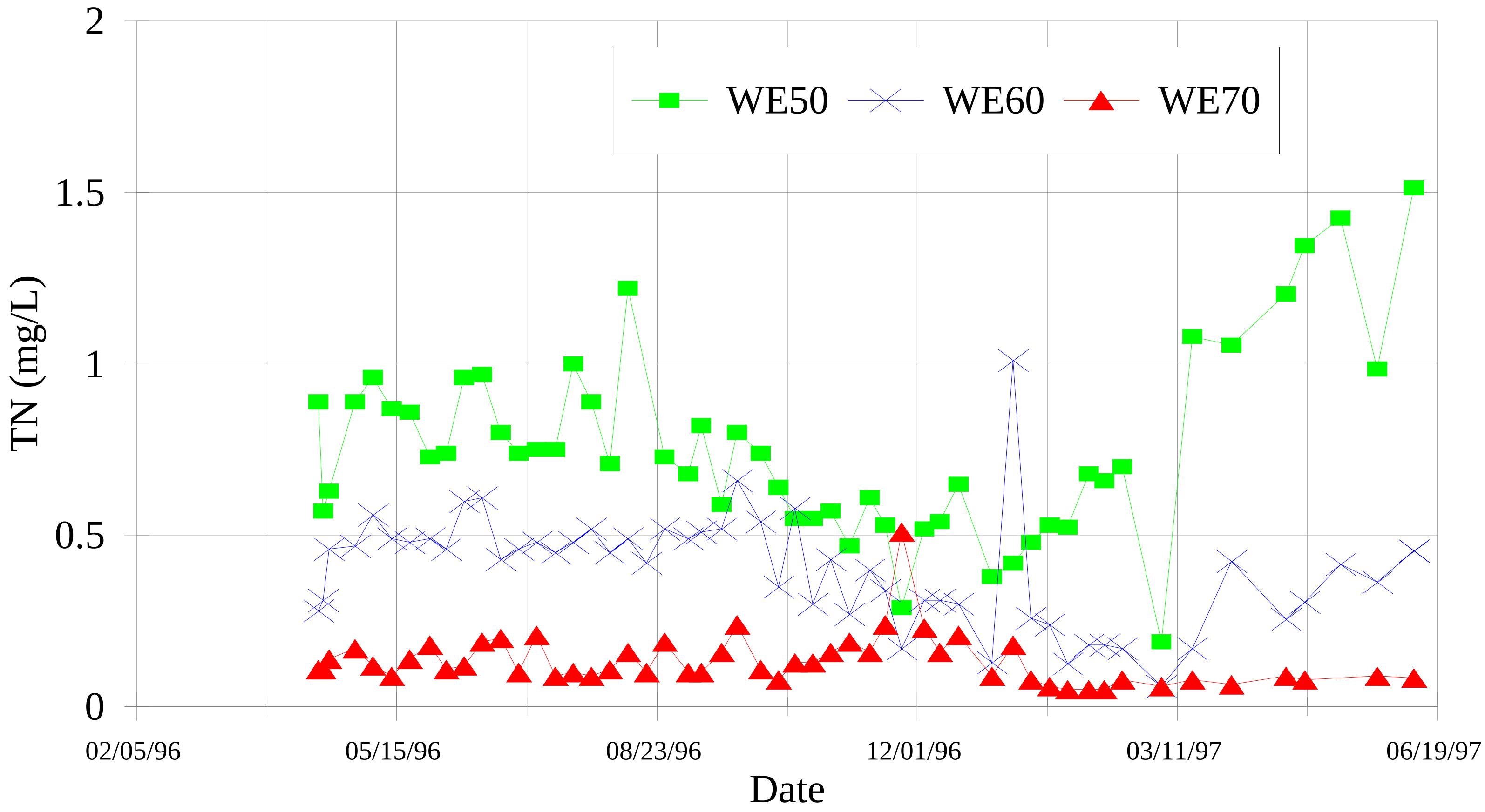

Figure B.8 Lysimeter TN data, mg/L

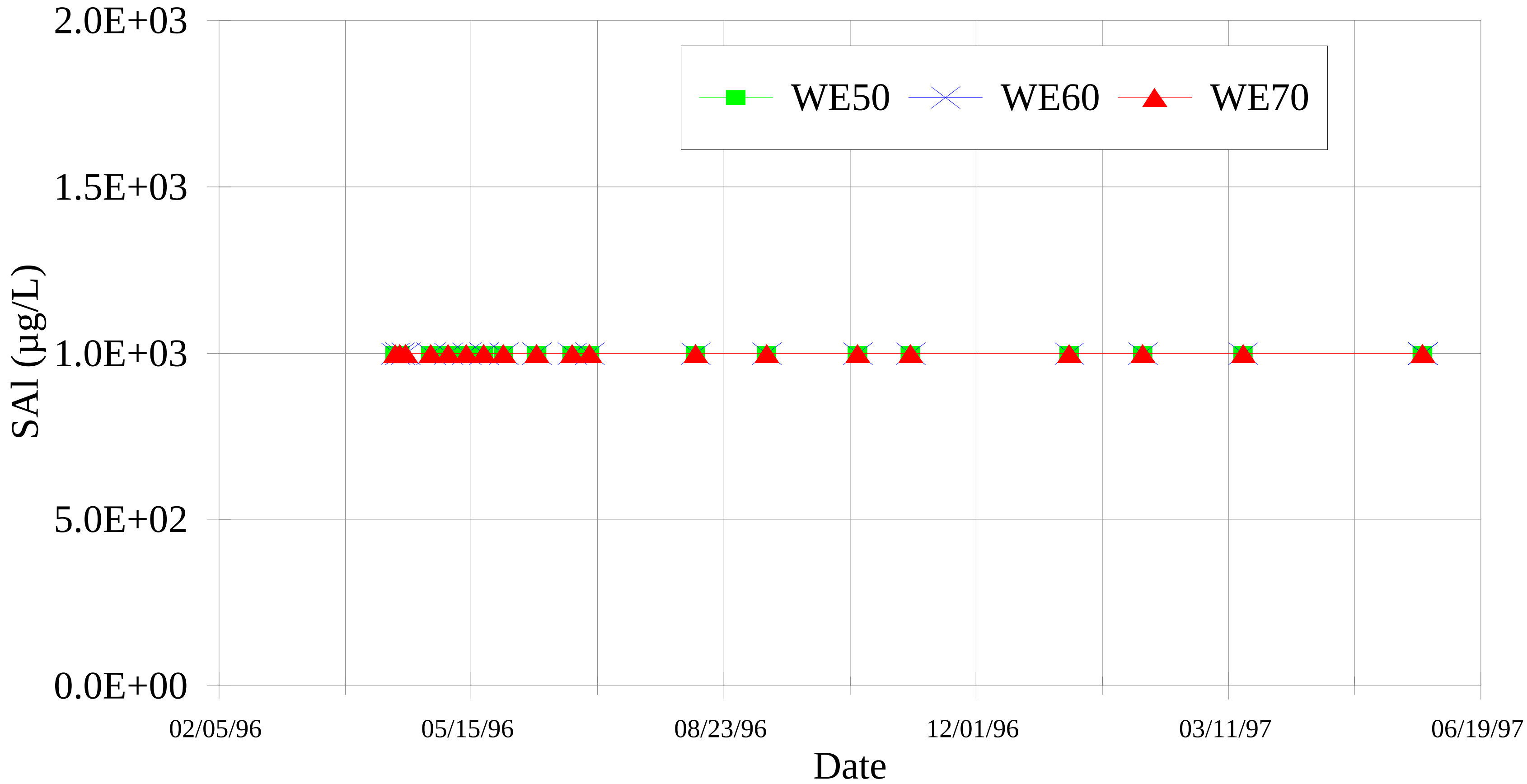

Figure B.9 Lysimeter SAl data, µg/L

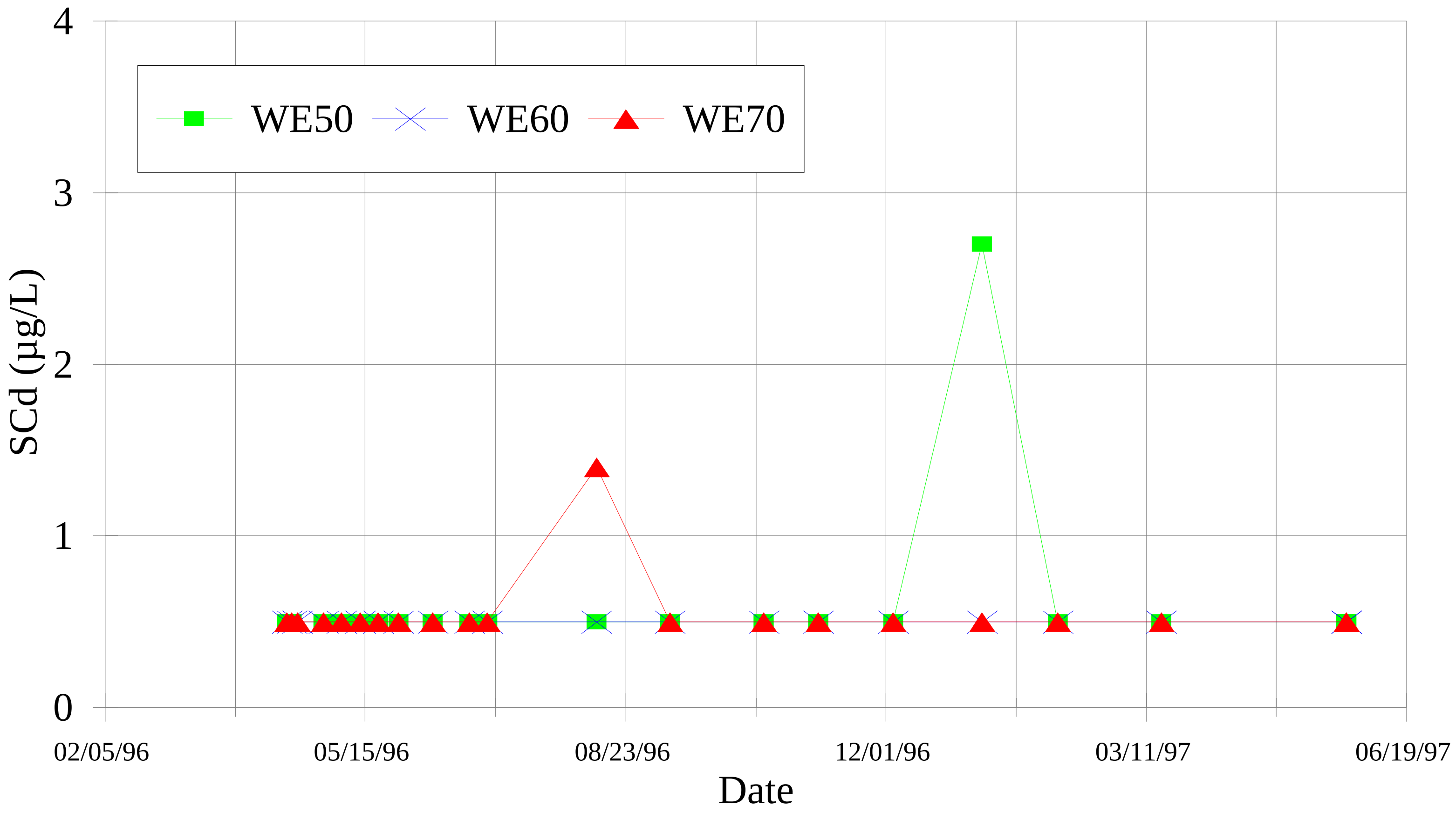

Figure B.10 Lysimeter SCd data, µg/L

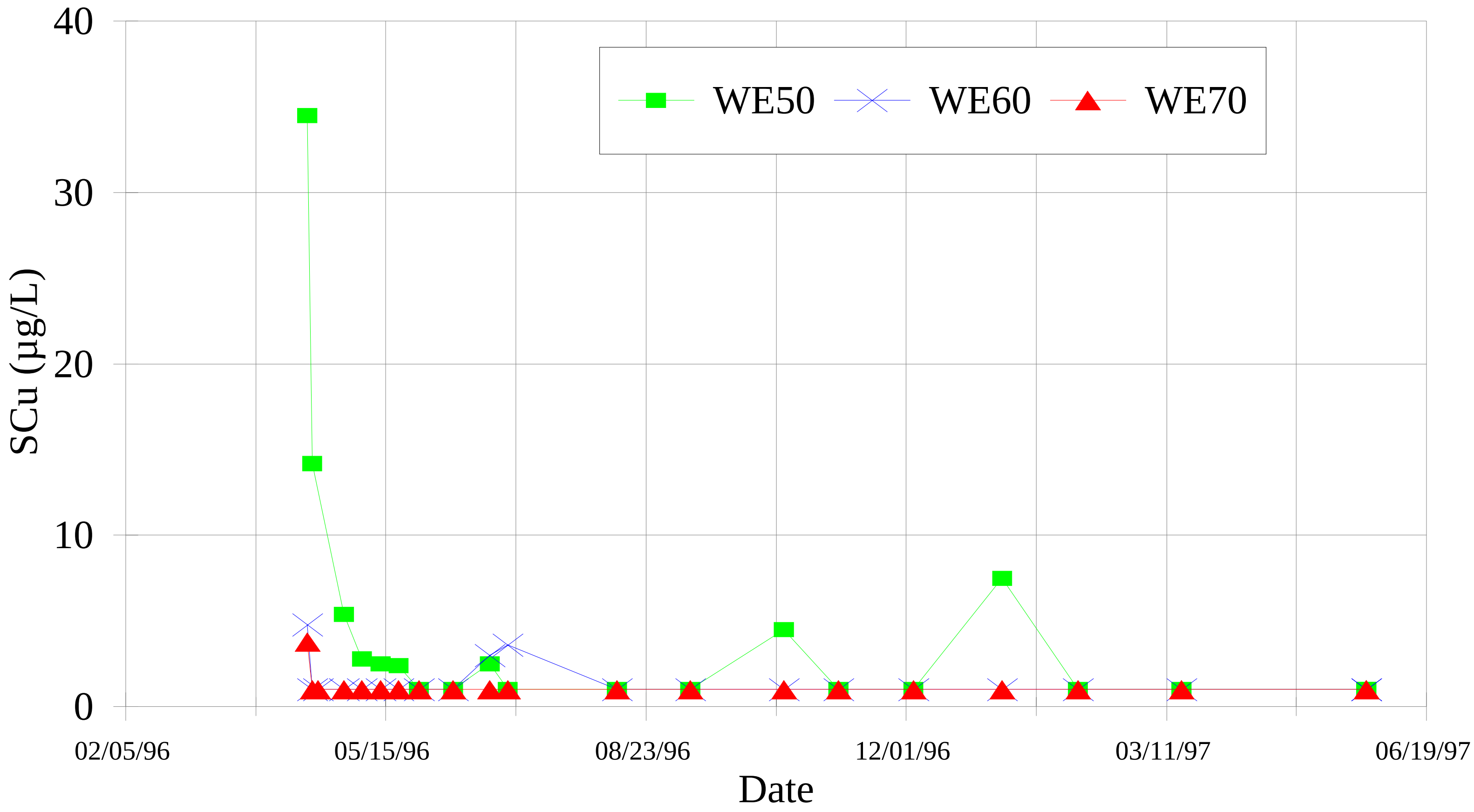

Figure B.11 Lysimeter SCu data, µg/L

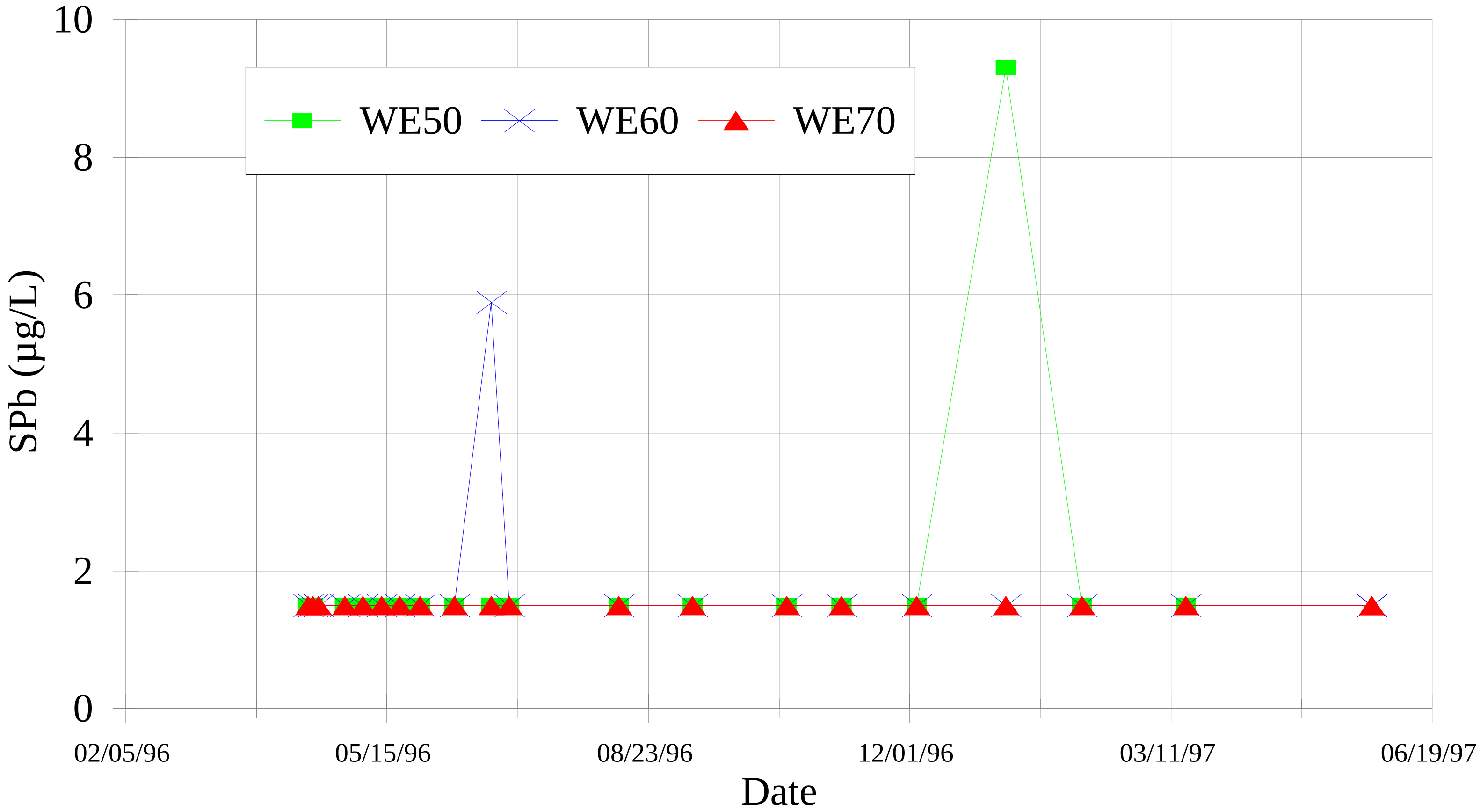

Figure B.12 Lysimeter SPb data, µg/L

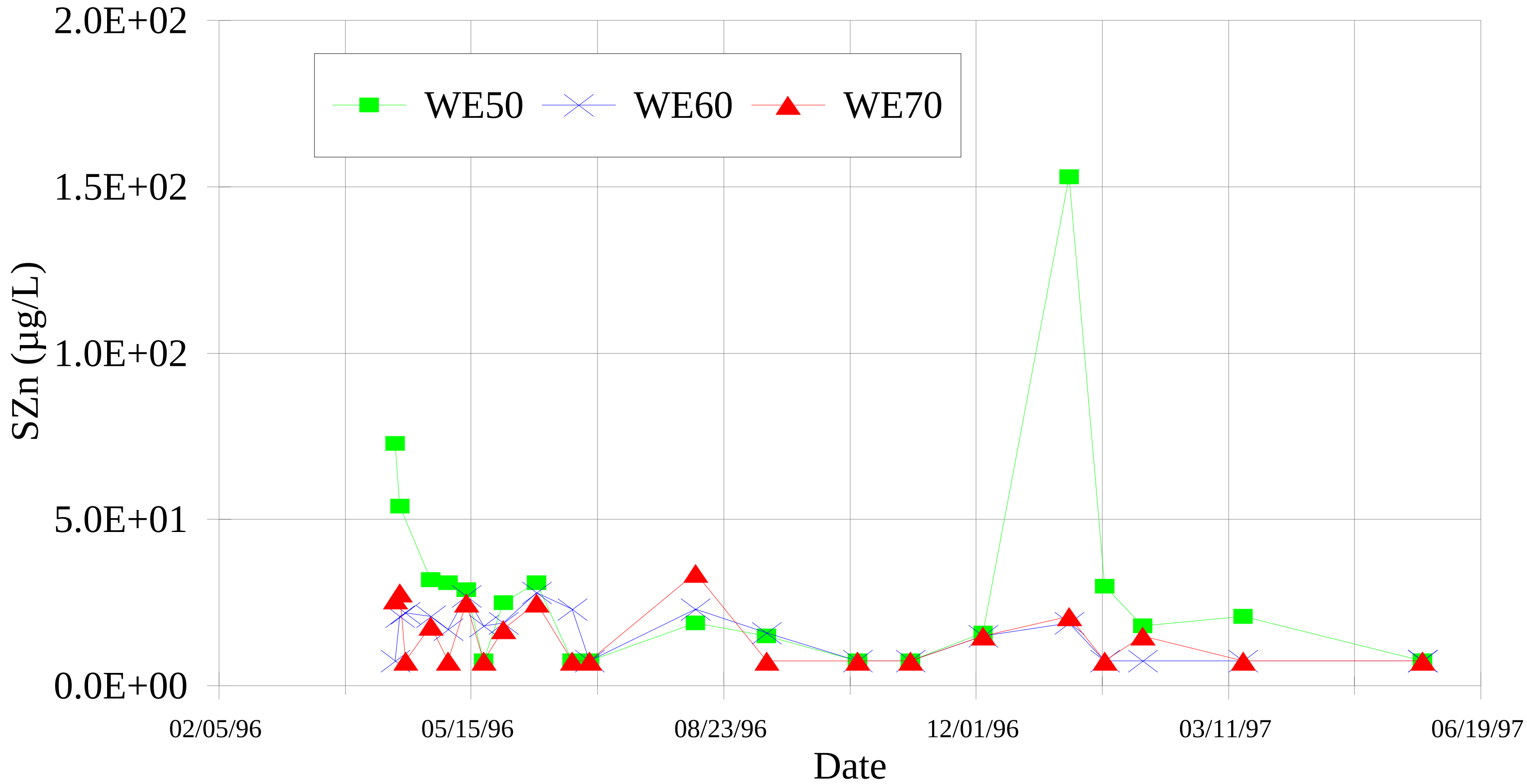

Figure B.13 Lysimeter SZn data, µg/L

APPENDIX C
Outlet Baseflow Concentrations vs. Storm EMCs and vs. Flow Rates

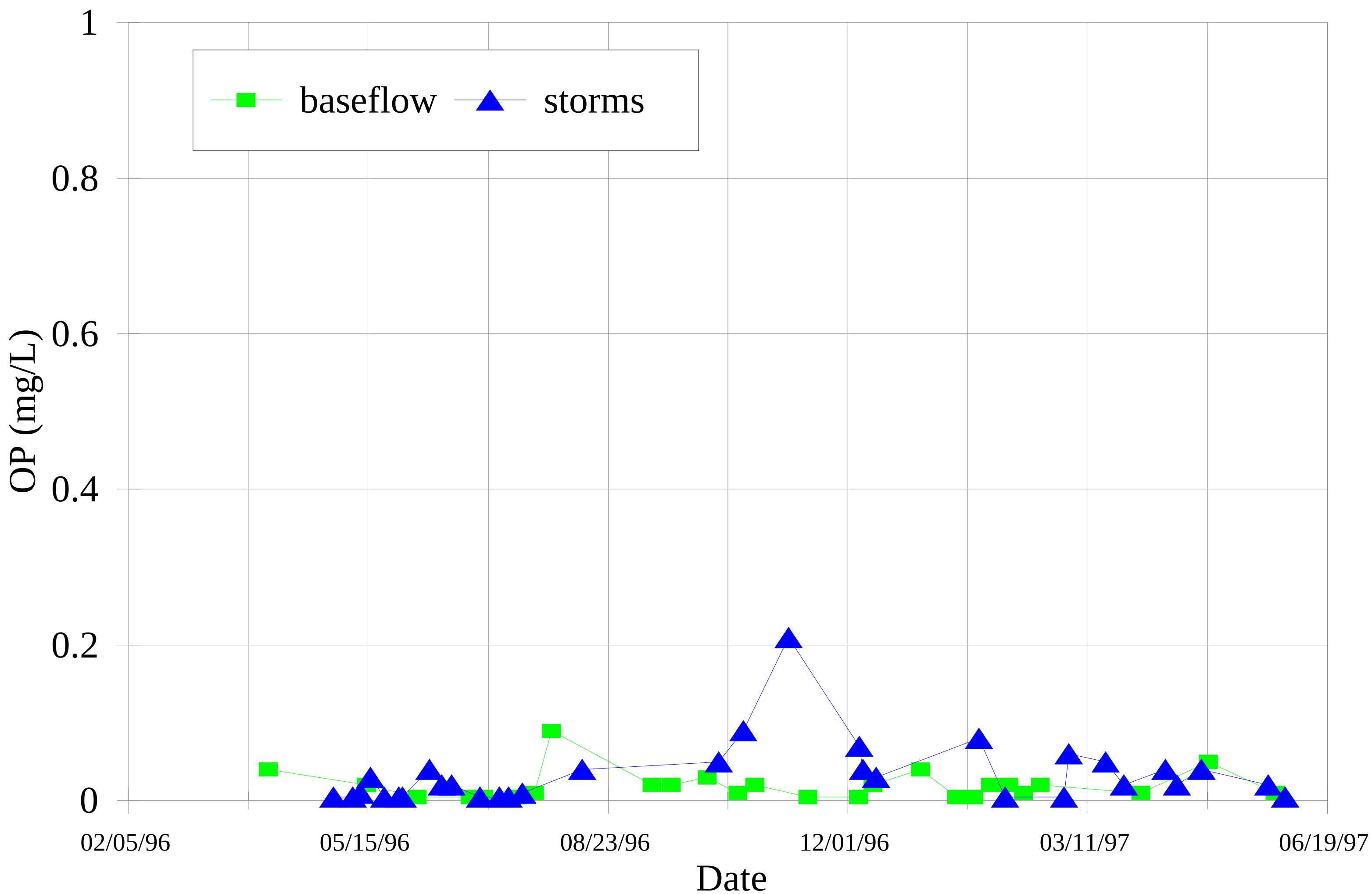

Figure C.1 Outlet storm EMCs and baseflow concentrations: OP, mg/L

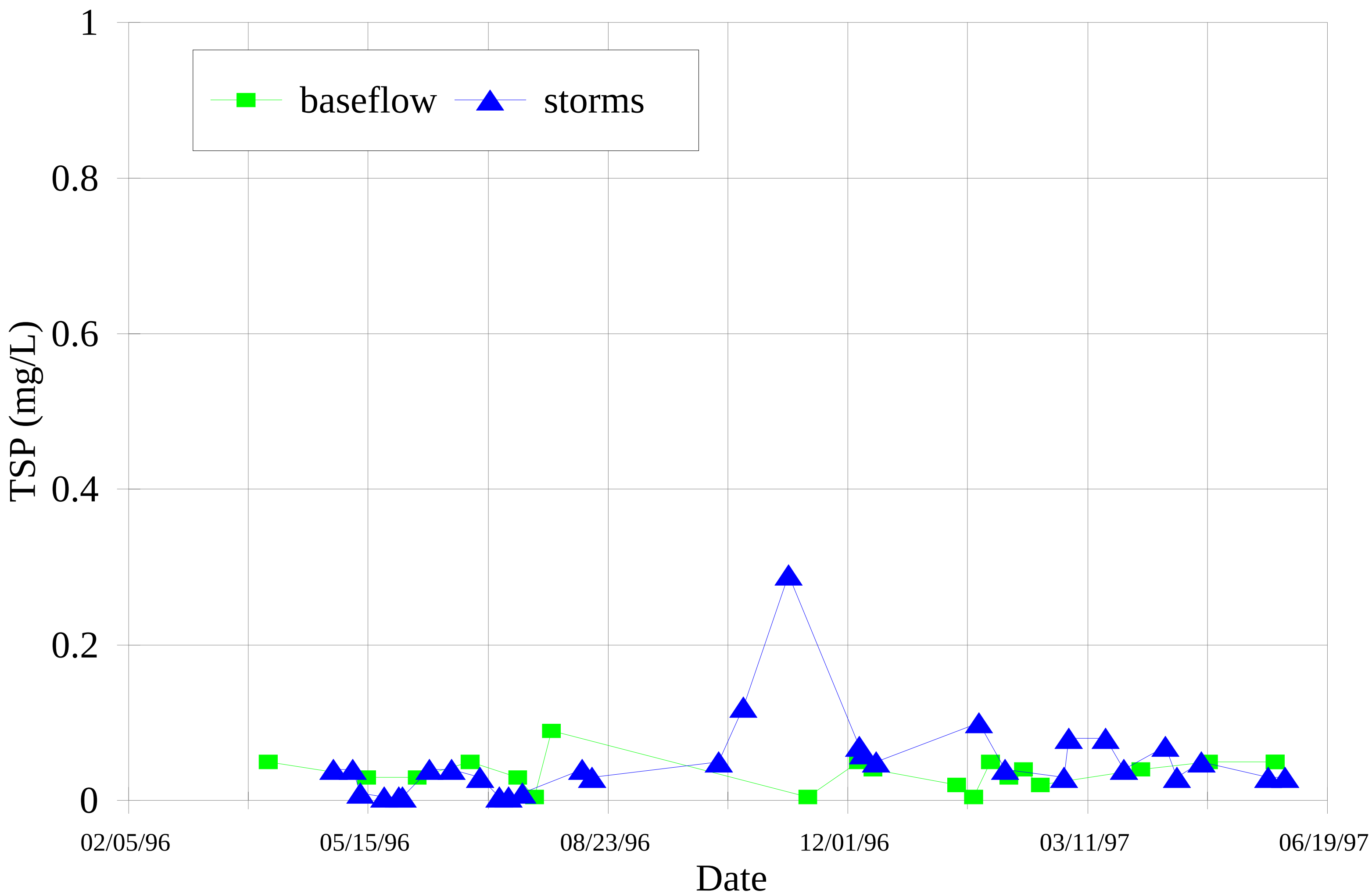

Figure C.2 Outlet storm EMCs and baseflow concentrations: TSP, mg/L

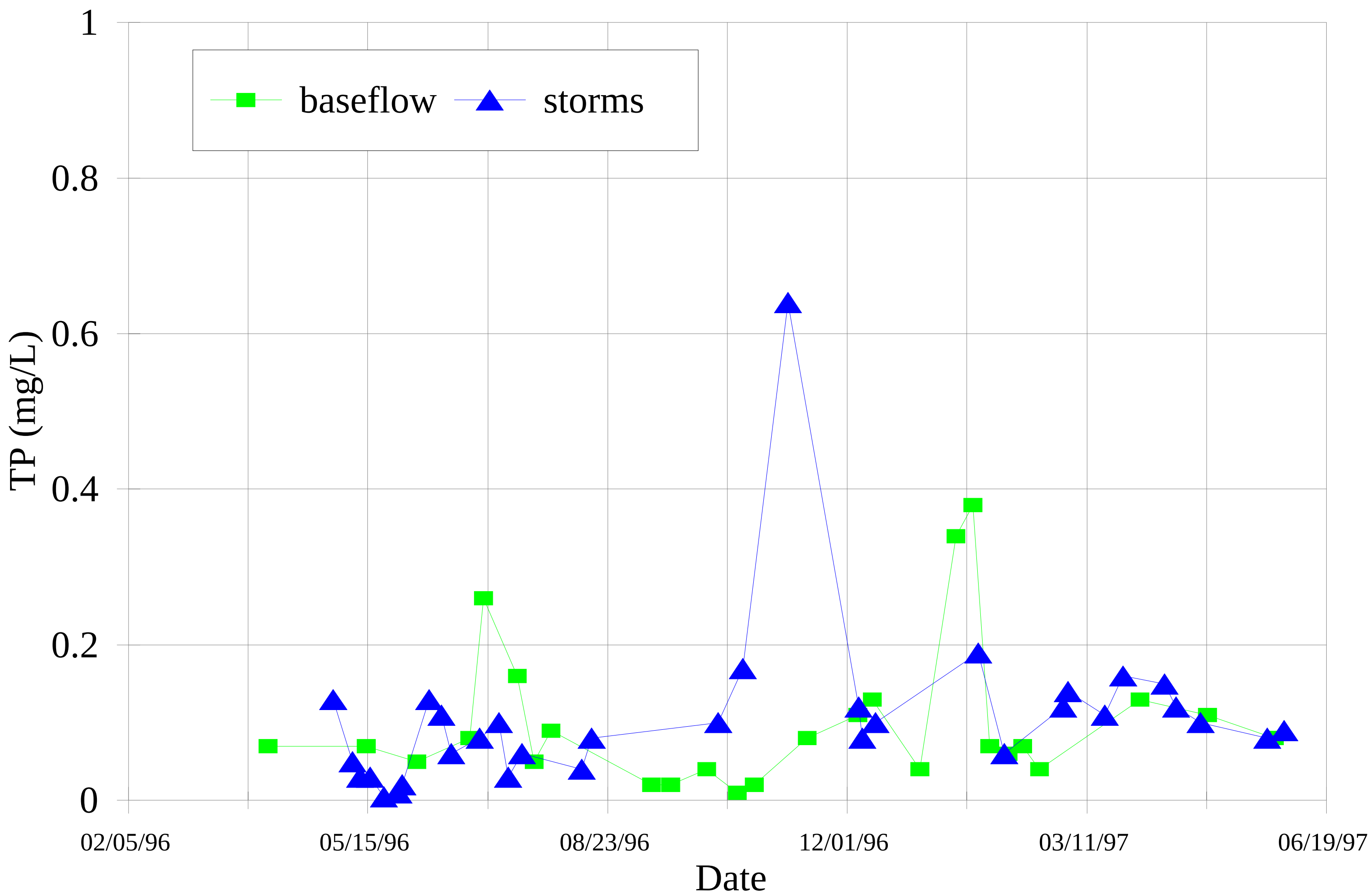

Figure C.3 Outlet storm EMCs and baseflow concentrations: TP, mg/L

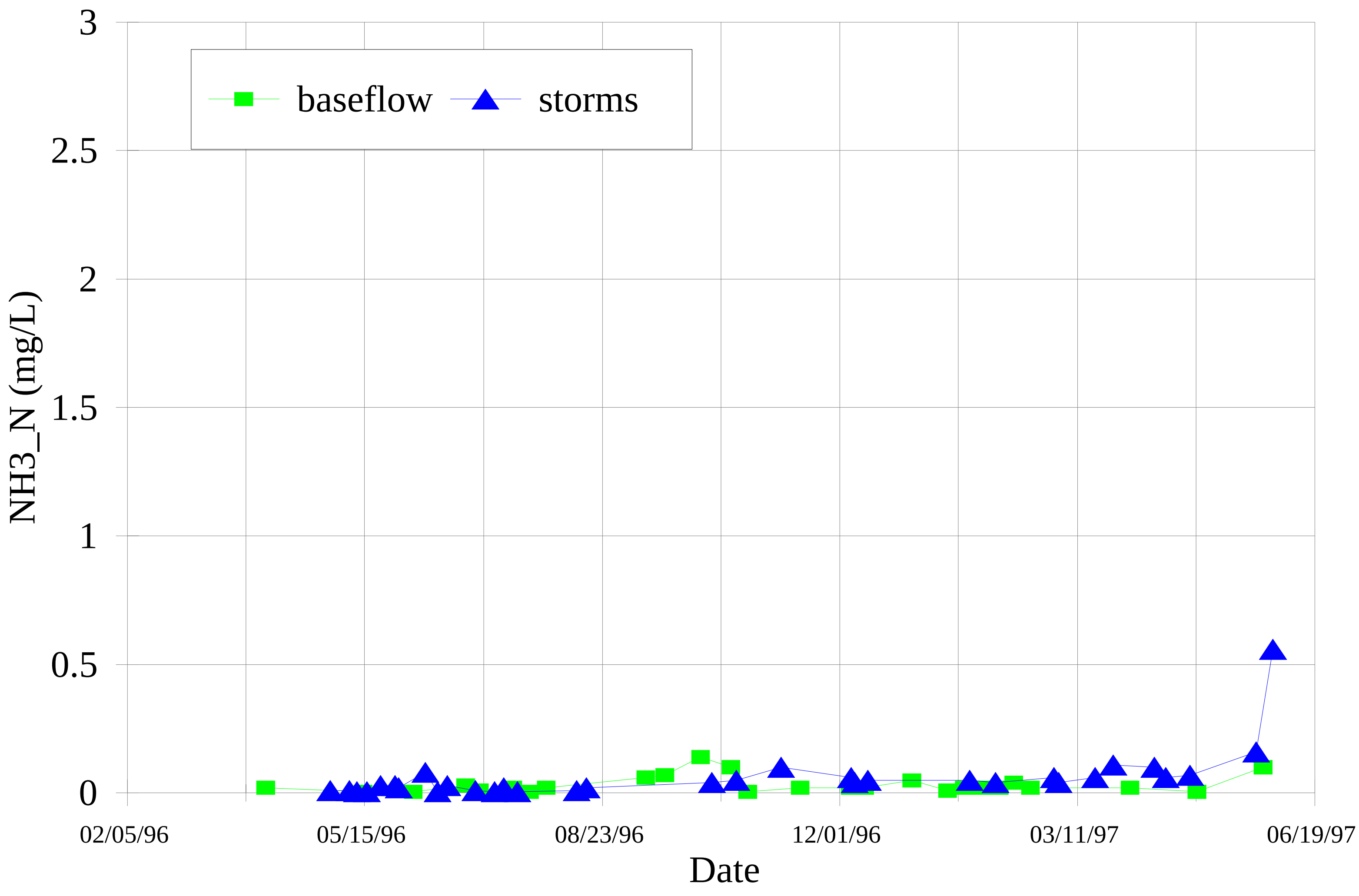

Figure C.4 Outlet storm EMCs and baseflow concentrations: NH3_N, mg/L

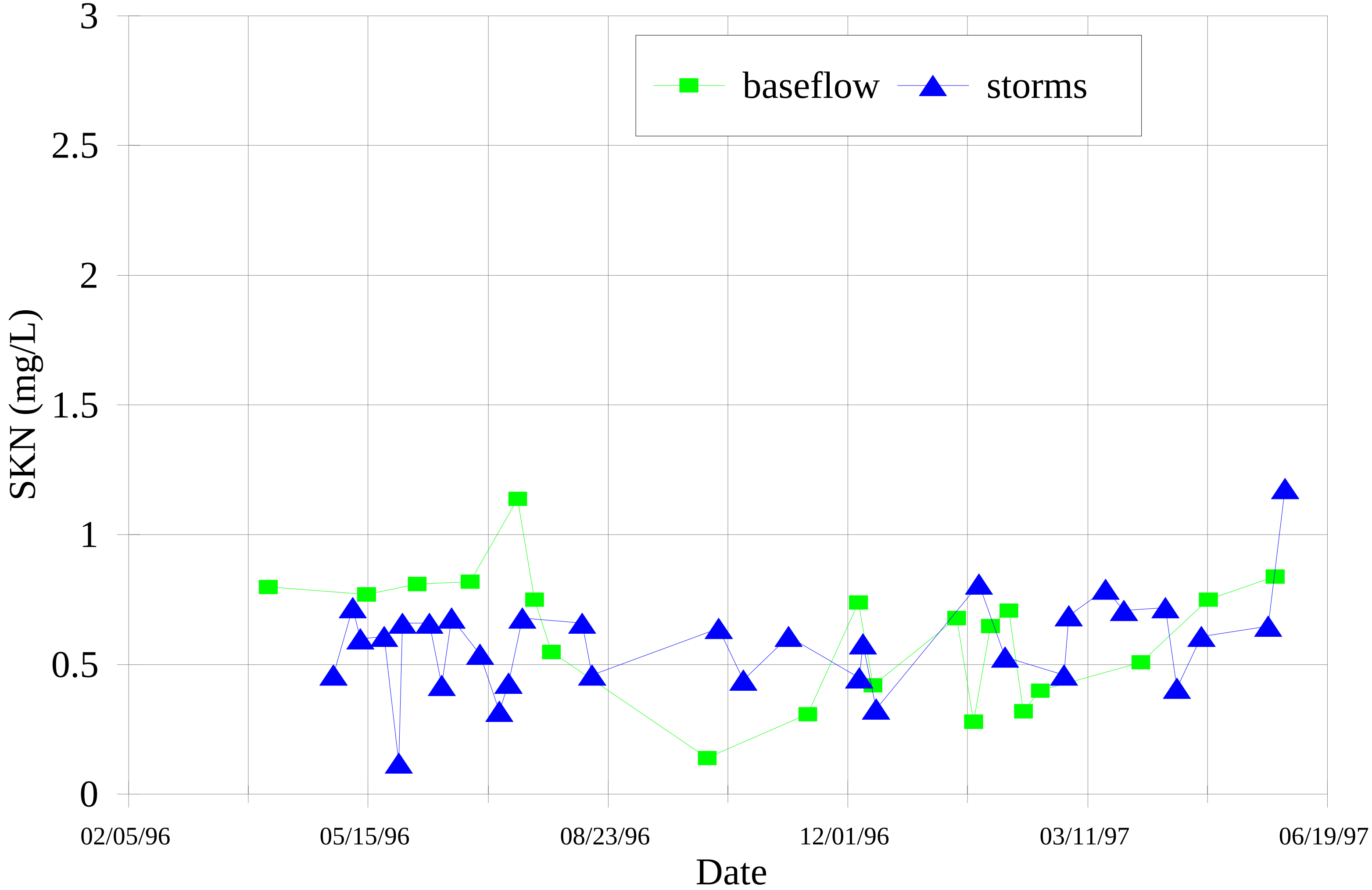

Figure C.5 Outlet storm EMCs and baseflow concentrations: SKN, mg/L

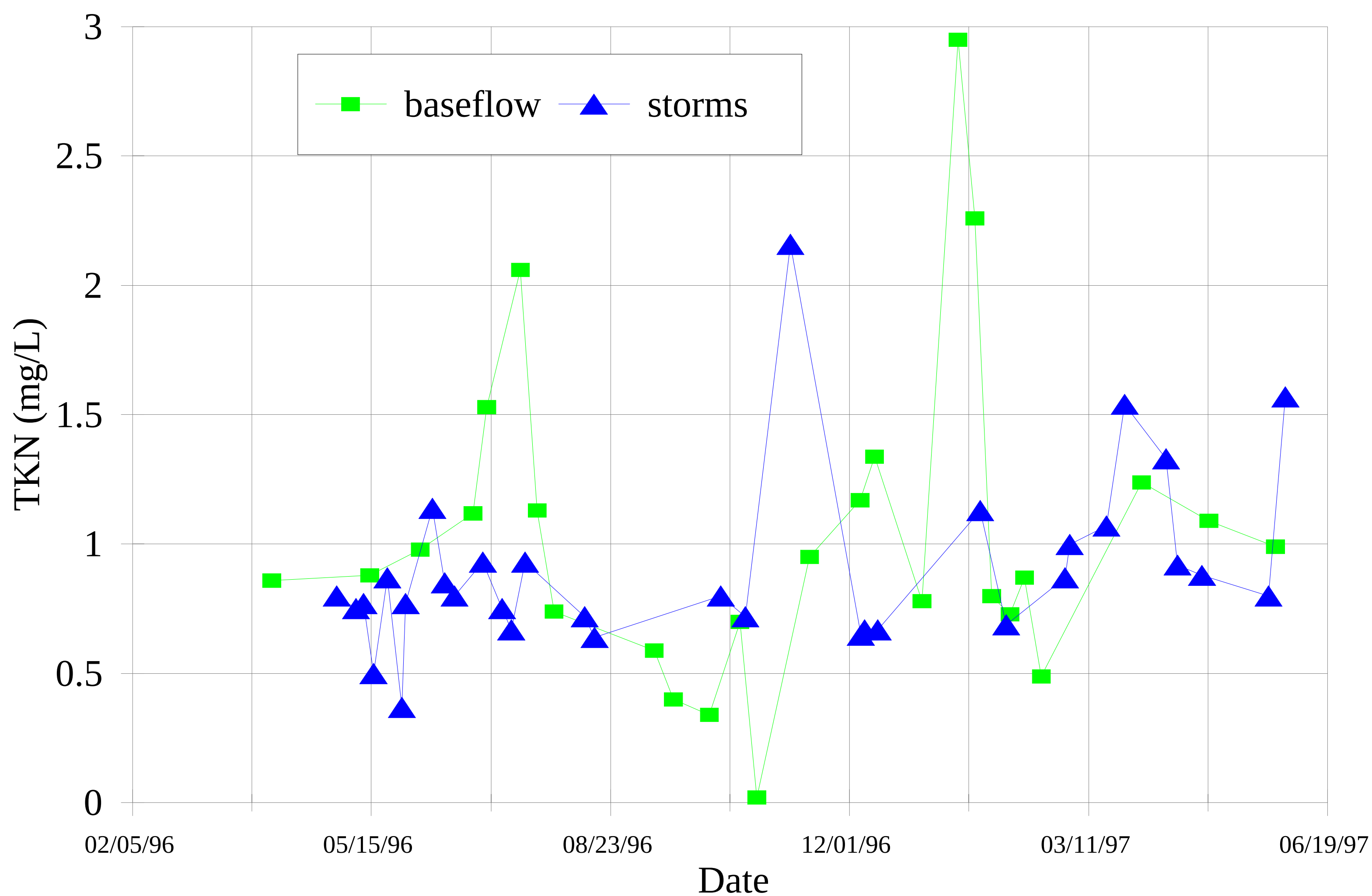

Figure C.6 Outlet storm EMCs and baseflow concentrations: TKN, mg/L

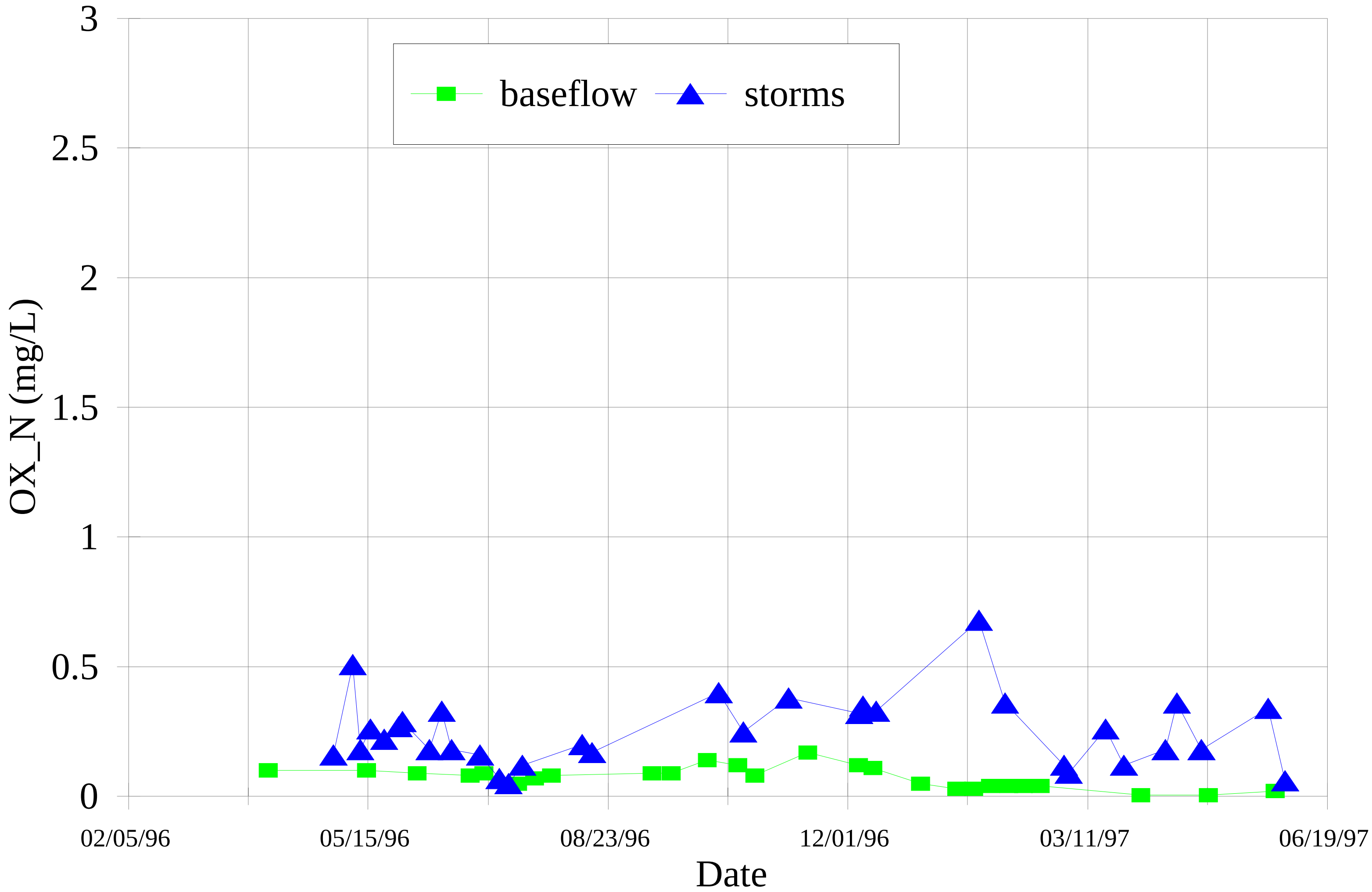

Figure C.7 Outlet storm EMCs and baseflow concentrations: OX_N, mg/L

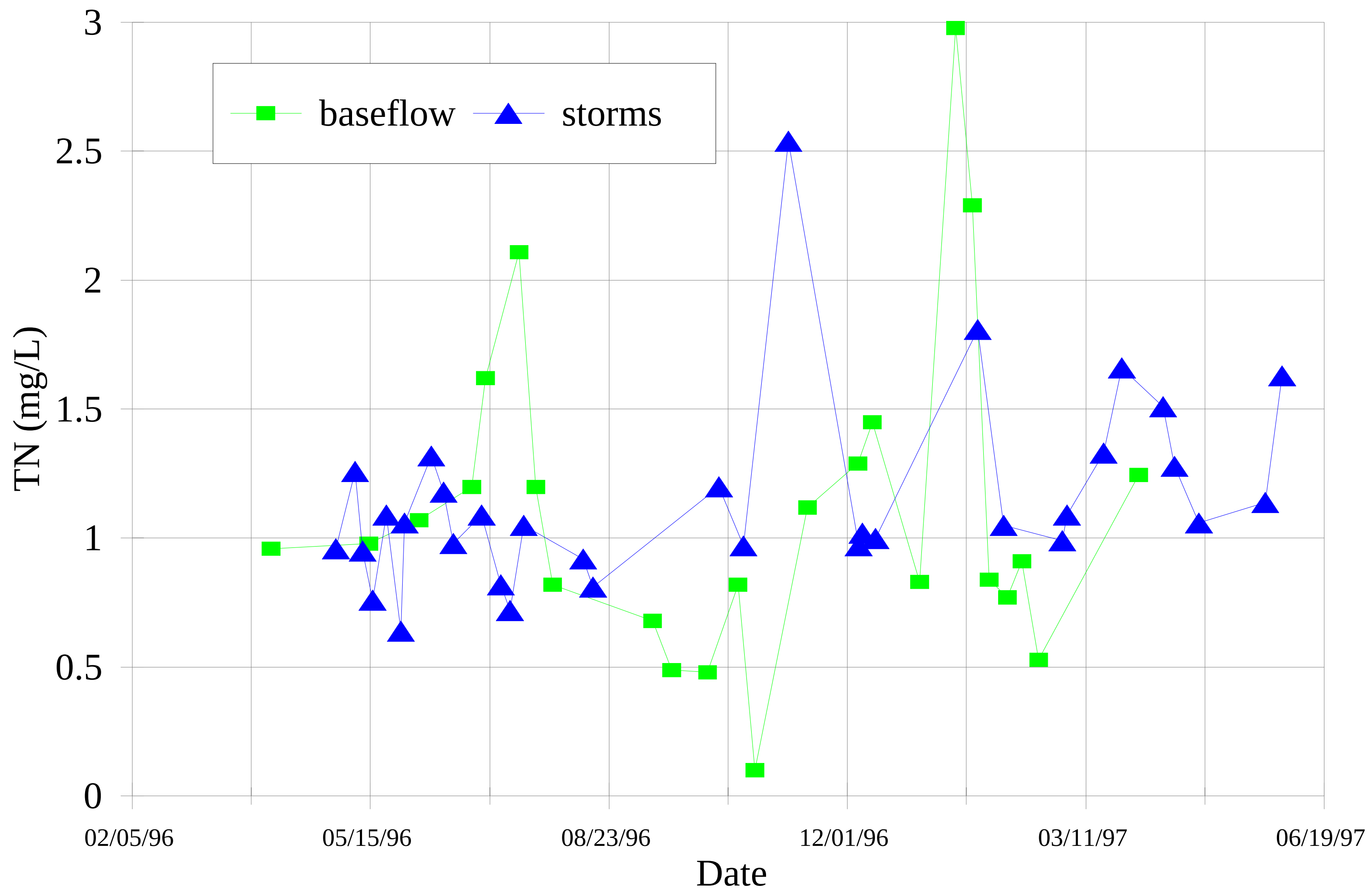

Figure C.8 Outlet storm EMCs and baseflow concentrations: TN, mg/L

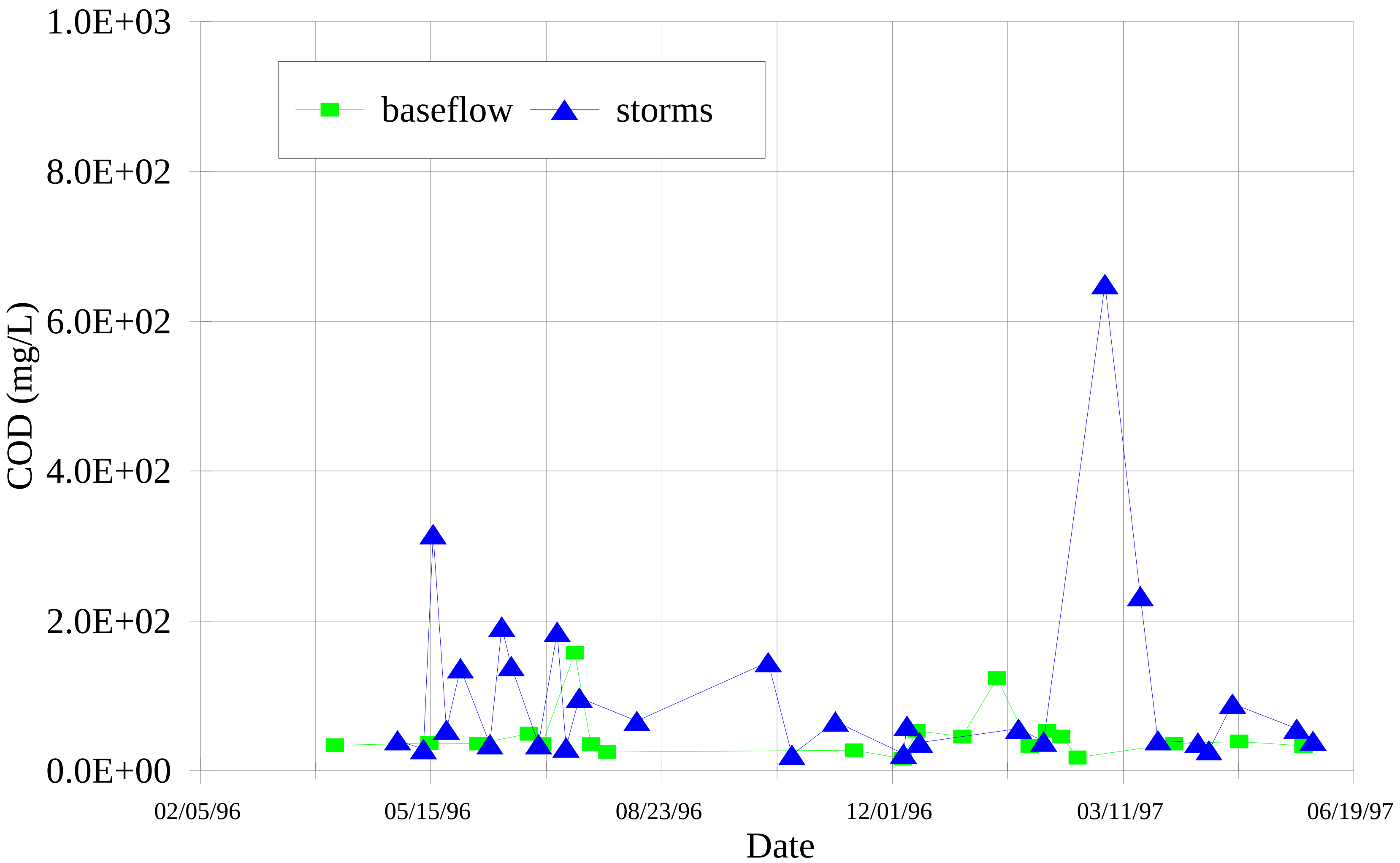

Figure C.9 Outlet storm EMCs and baseflow concentrations: COD, mg/L

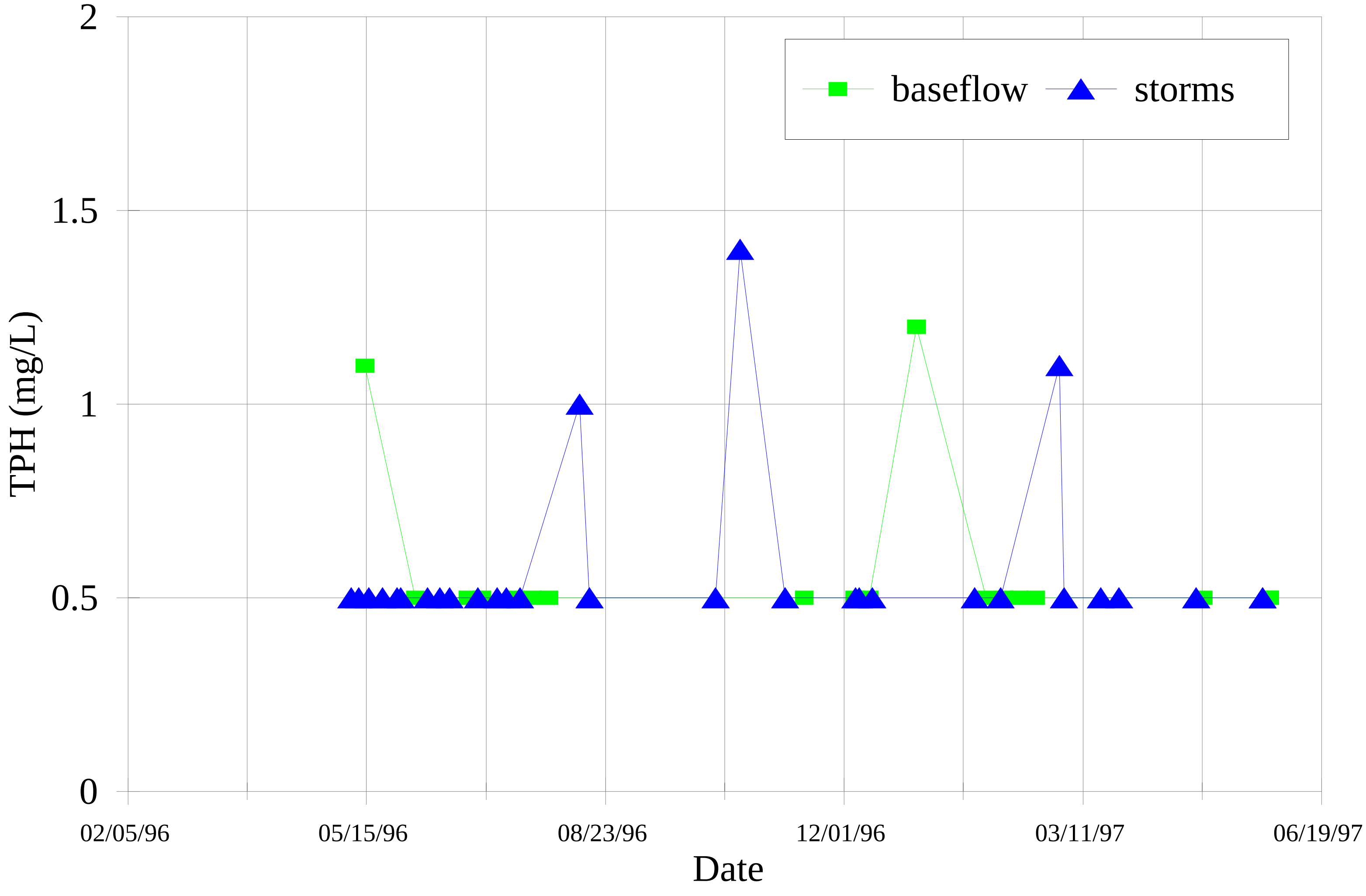

Figure C.10 Outlet storm EMCs and baseflow concentrations: TPH, mg/L

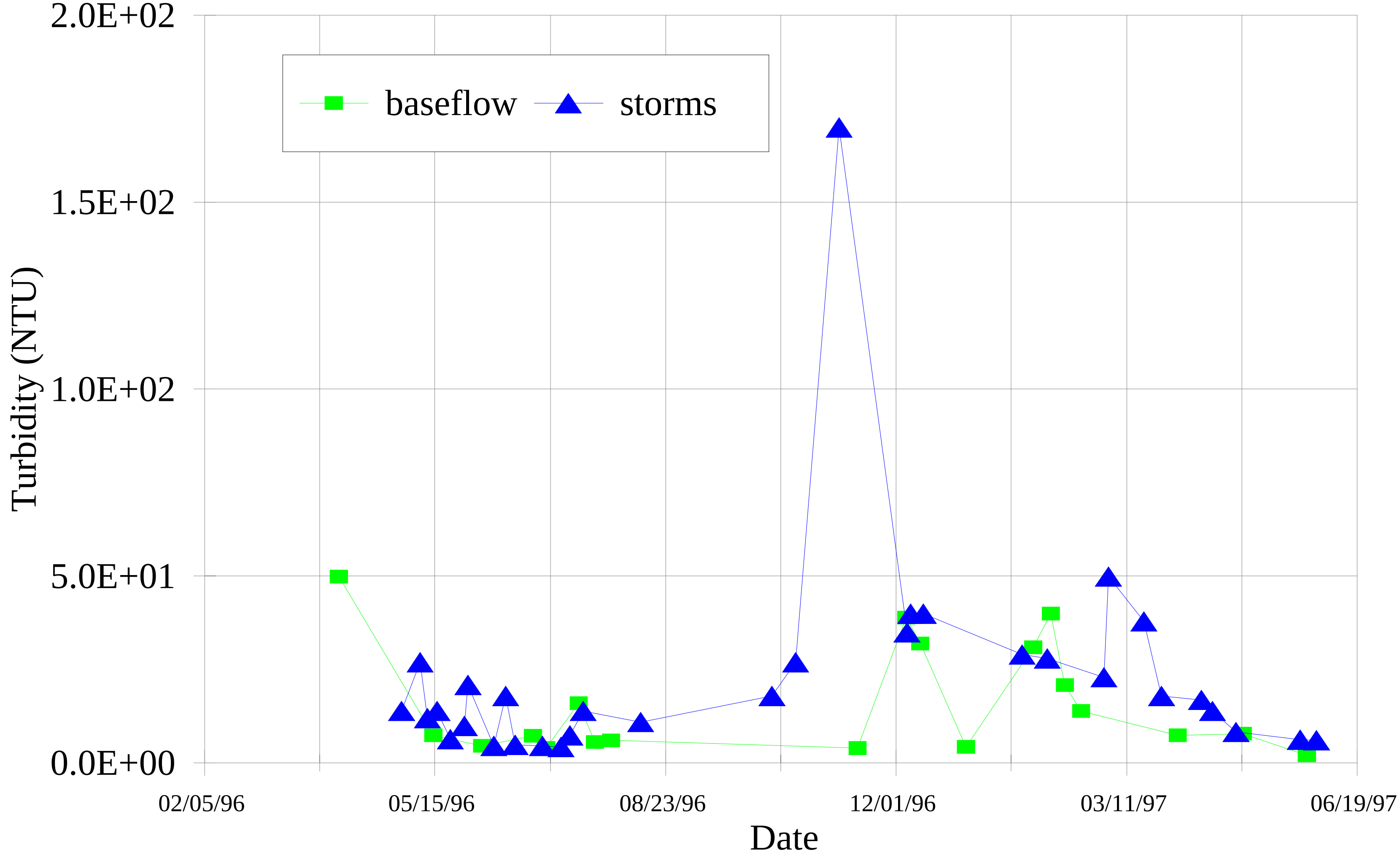

Figure C.11 Outlet storm EMCs and baseflow concentrations: Turbidity, NTU

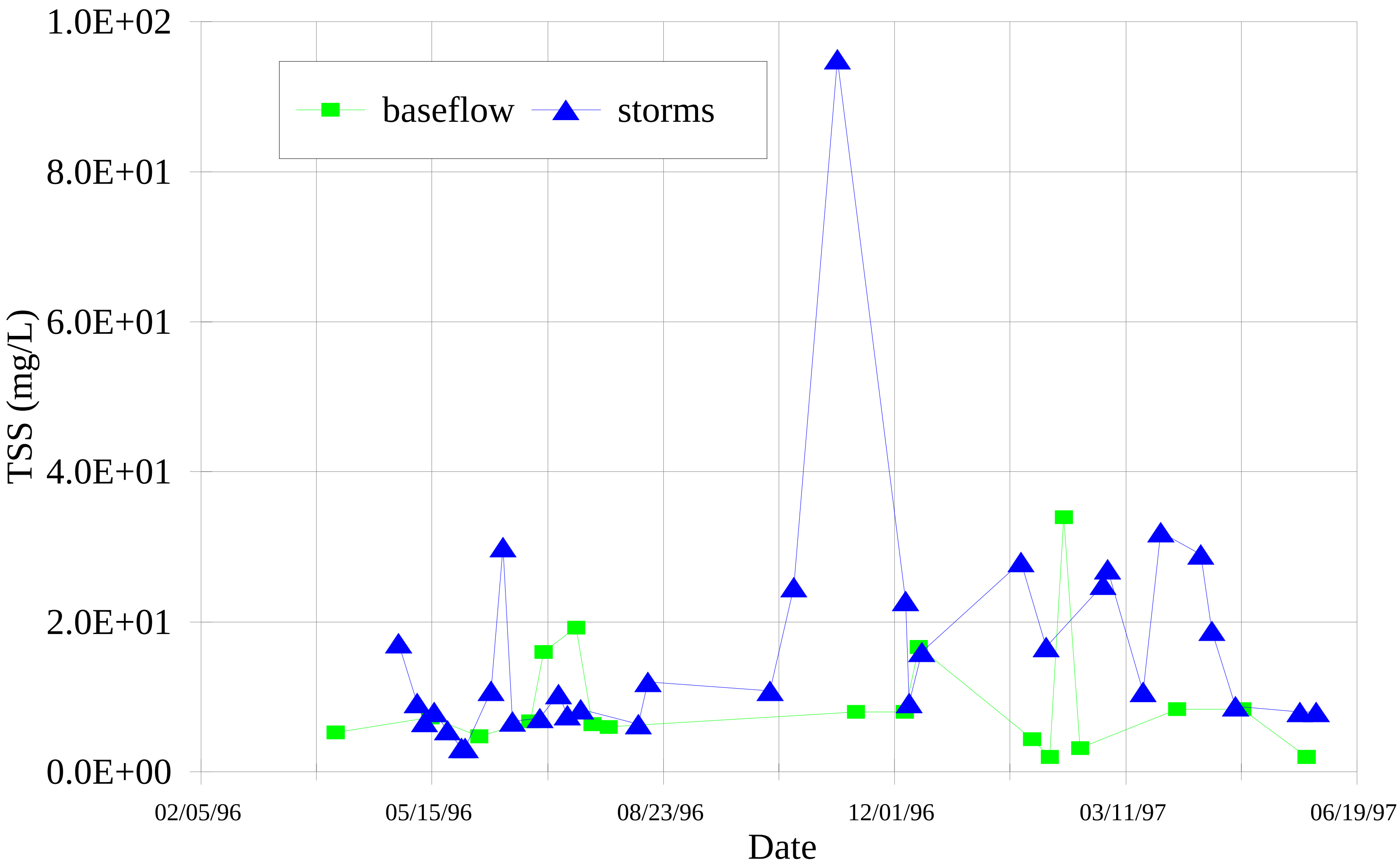

Figure C.12 Outlet storm EMCs and baseflow concentrations: TSS, mg/L

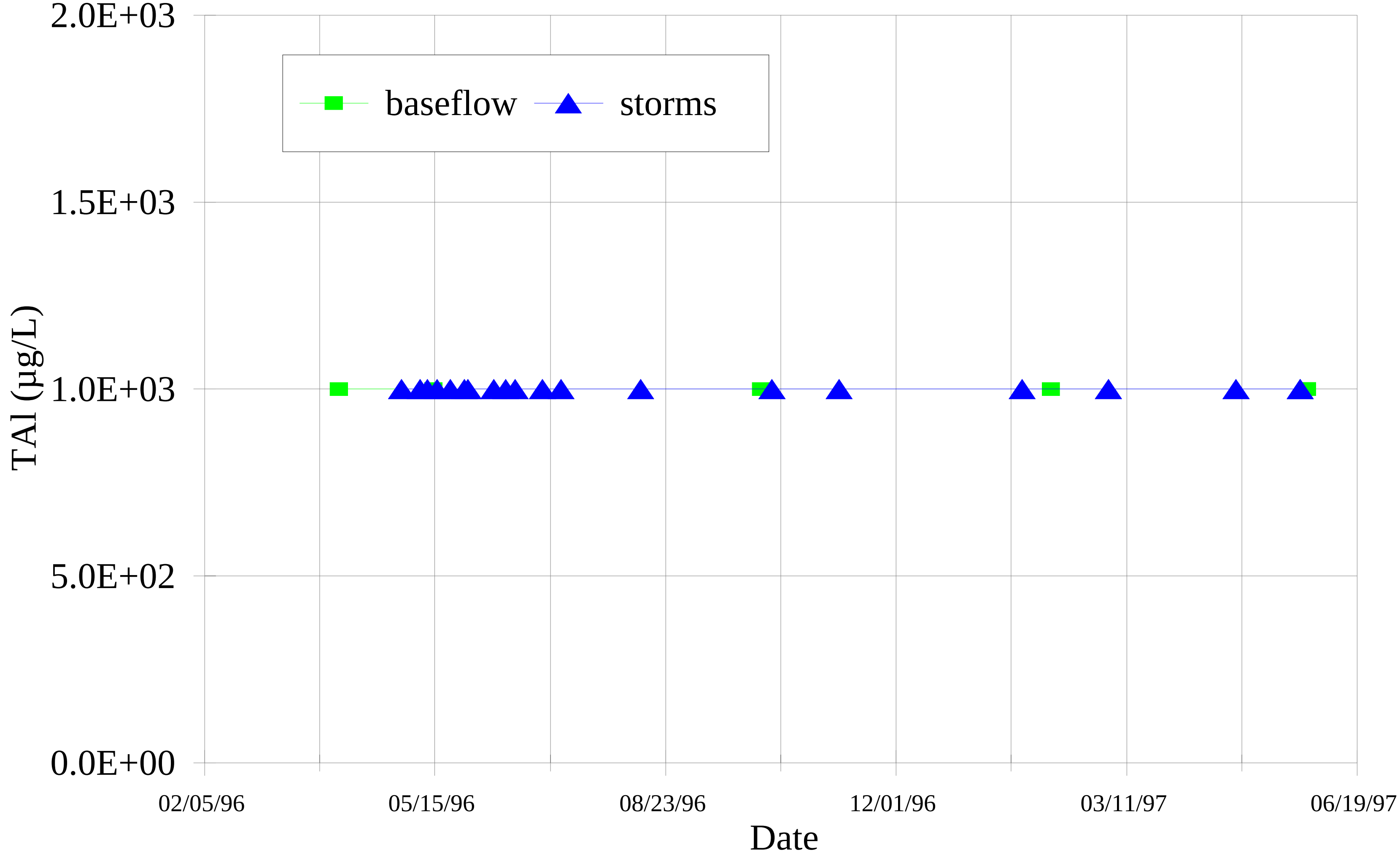

Figure C.13 Outlet storm EMCs and baseflow concentrations: TAl, µg/L

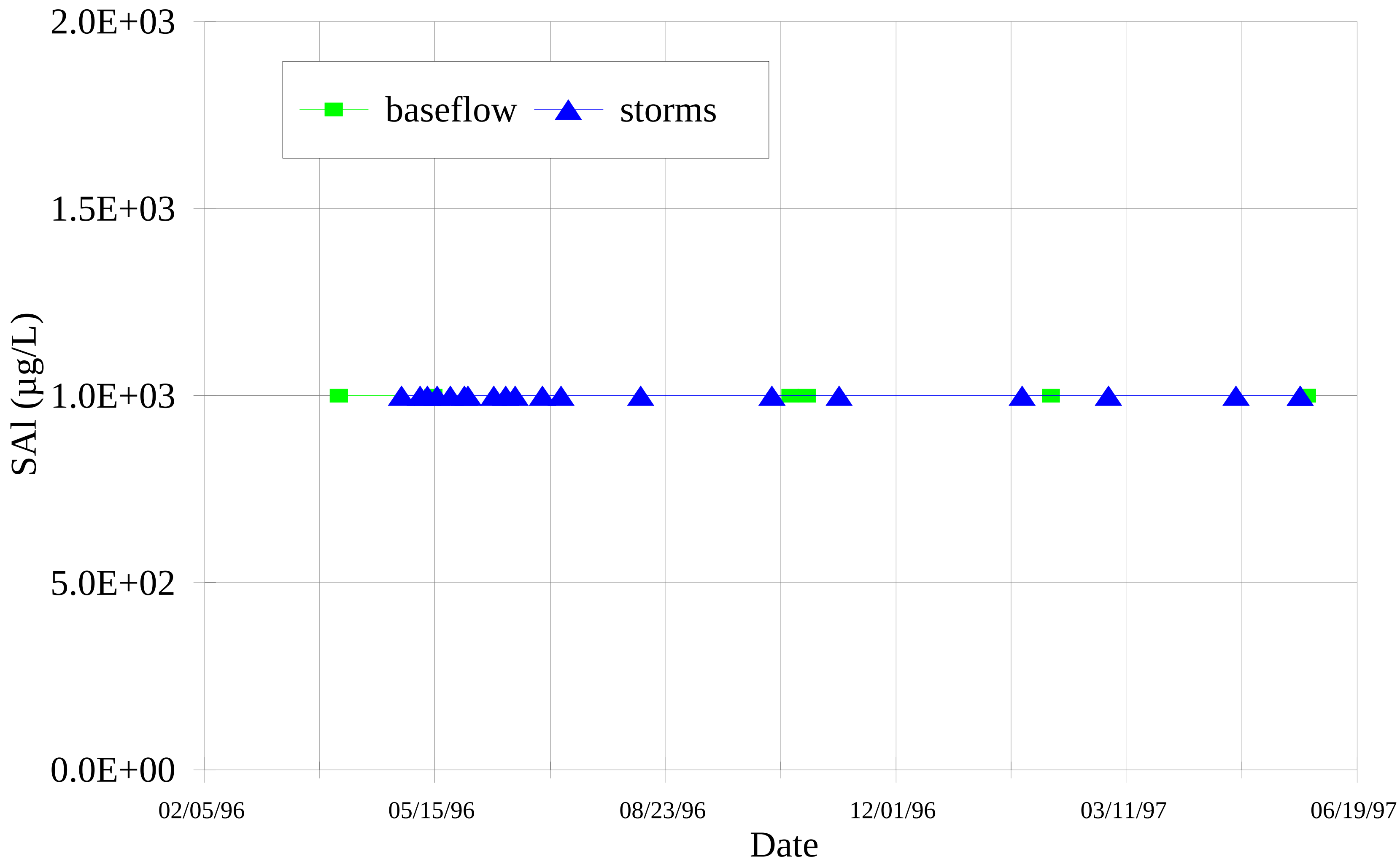

Figure C.14 Outlet storm EMCs and baseflow concentrations: SAl, µg/L

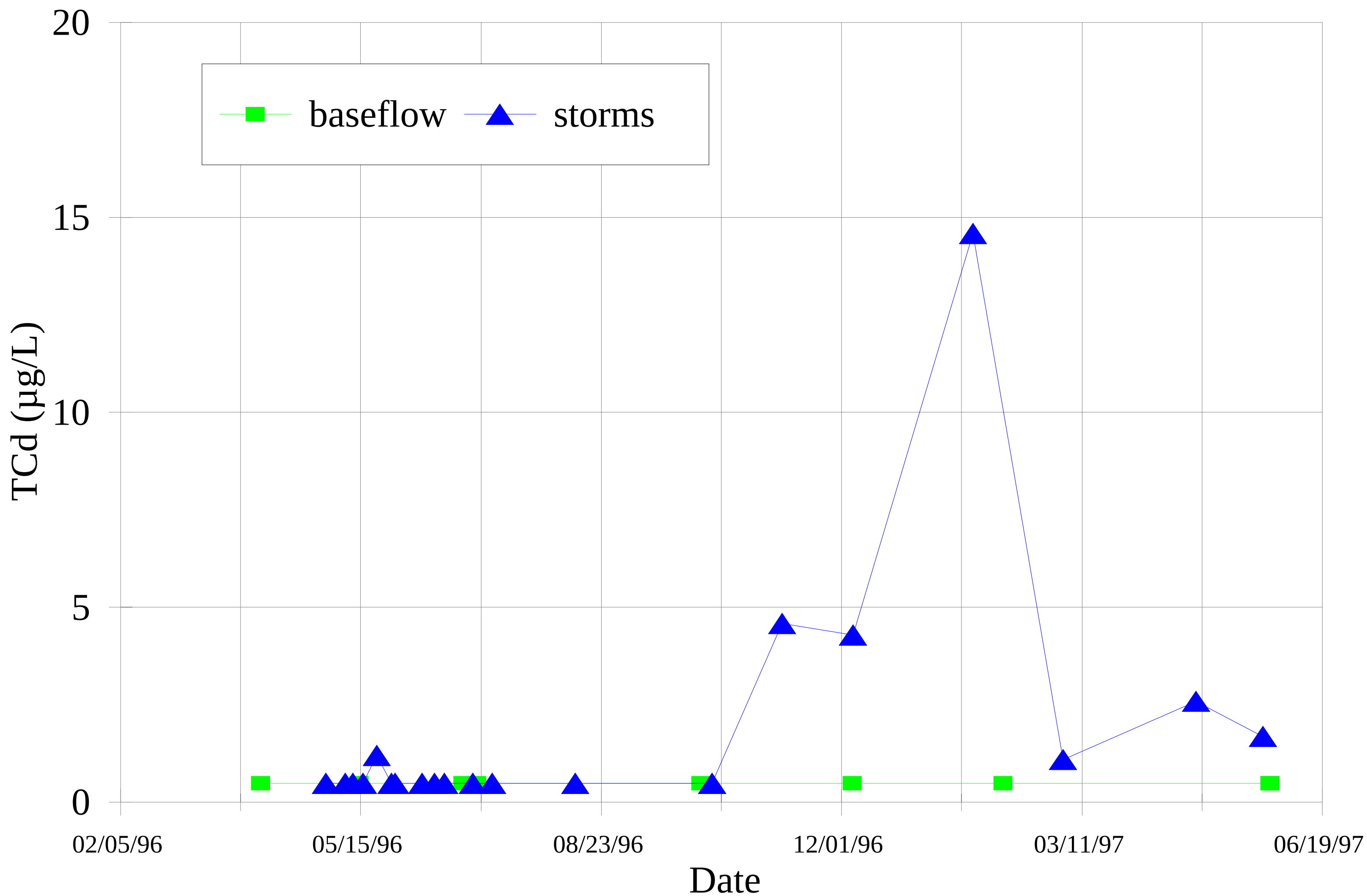

Figure C.15 Outlet storm EMCs and baseflow concentrations: TCd, µg/L

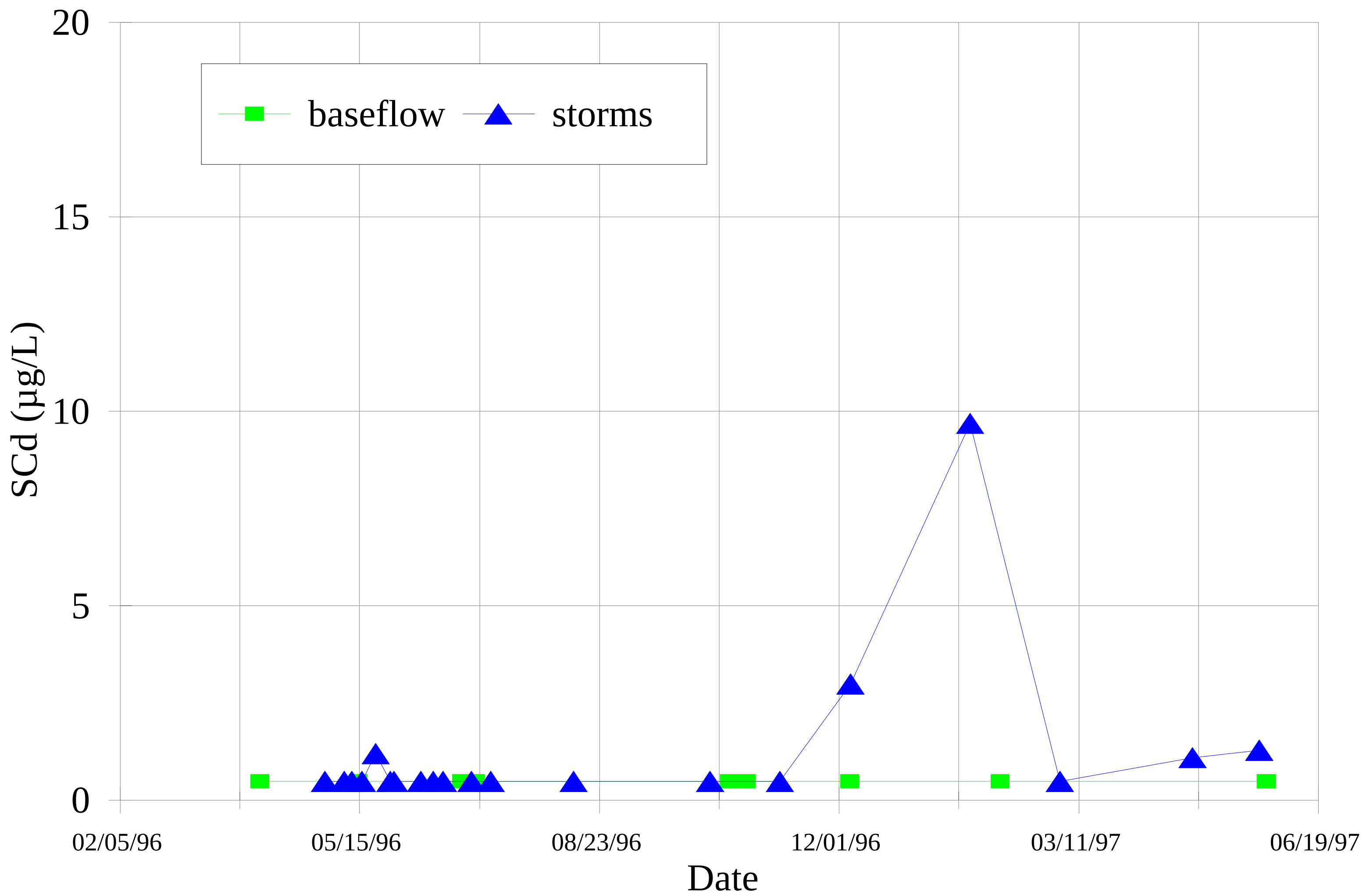

Figure C.16 Outlet storm EMCs and baseflow concentrations: SCd, µg/L

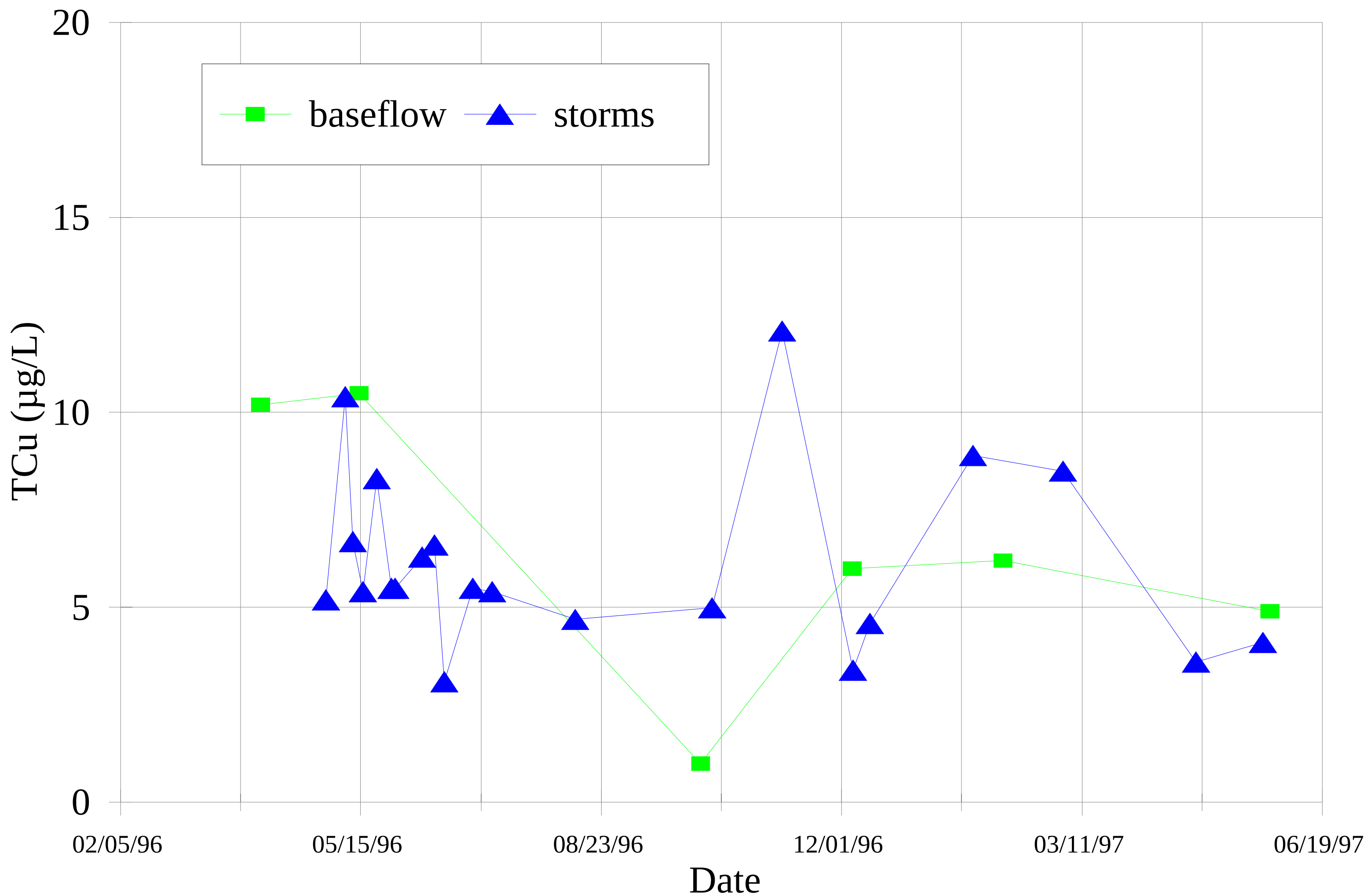

Figure C.17 Outlet storm EMCs and baseflow concentrations: TCu, µg/L

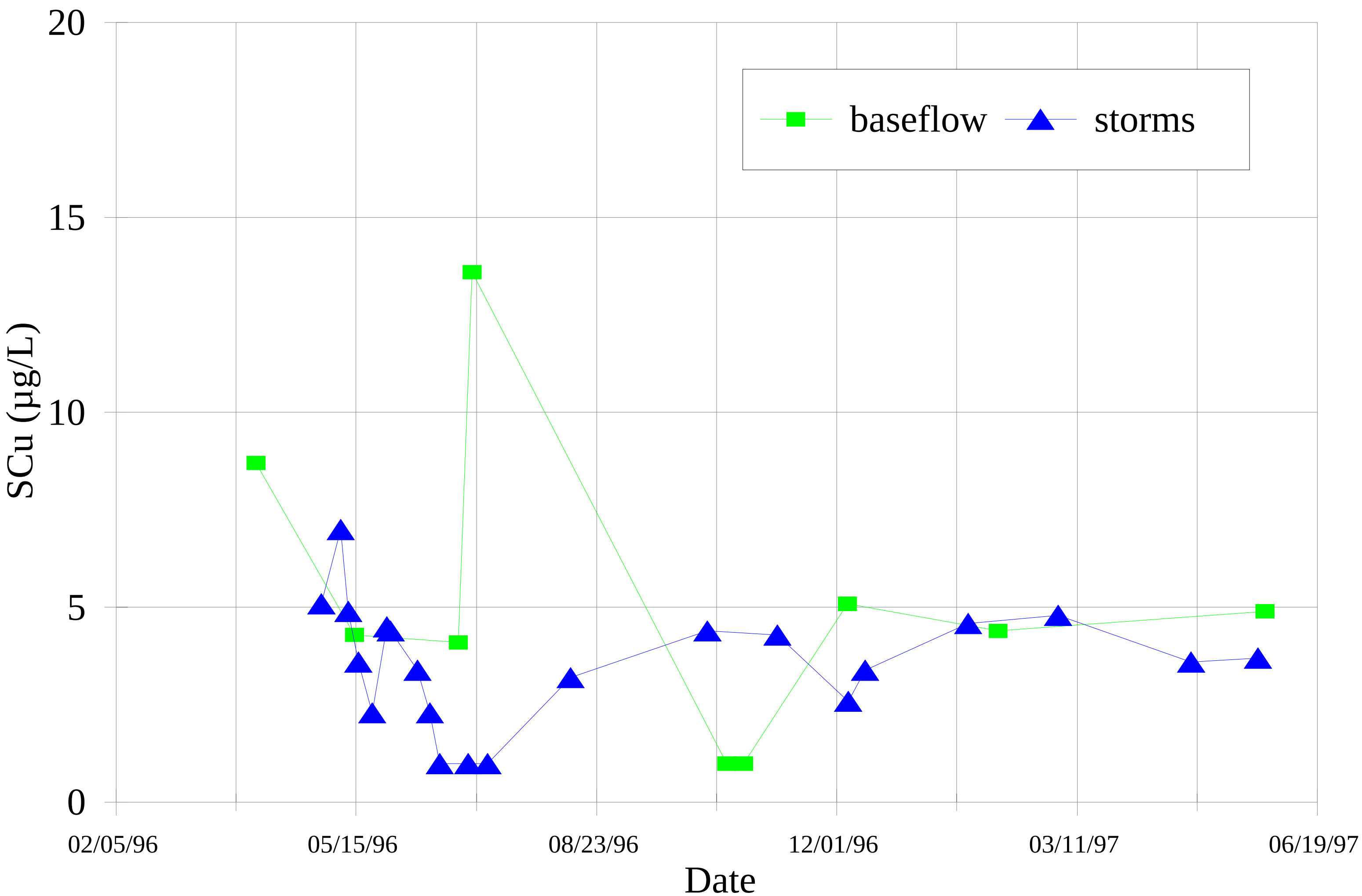

Figure C.18 Outlet storm EMCs and baseflow concentrations: SCu, µg/L

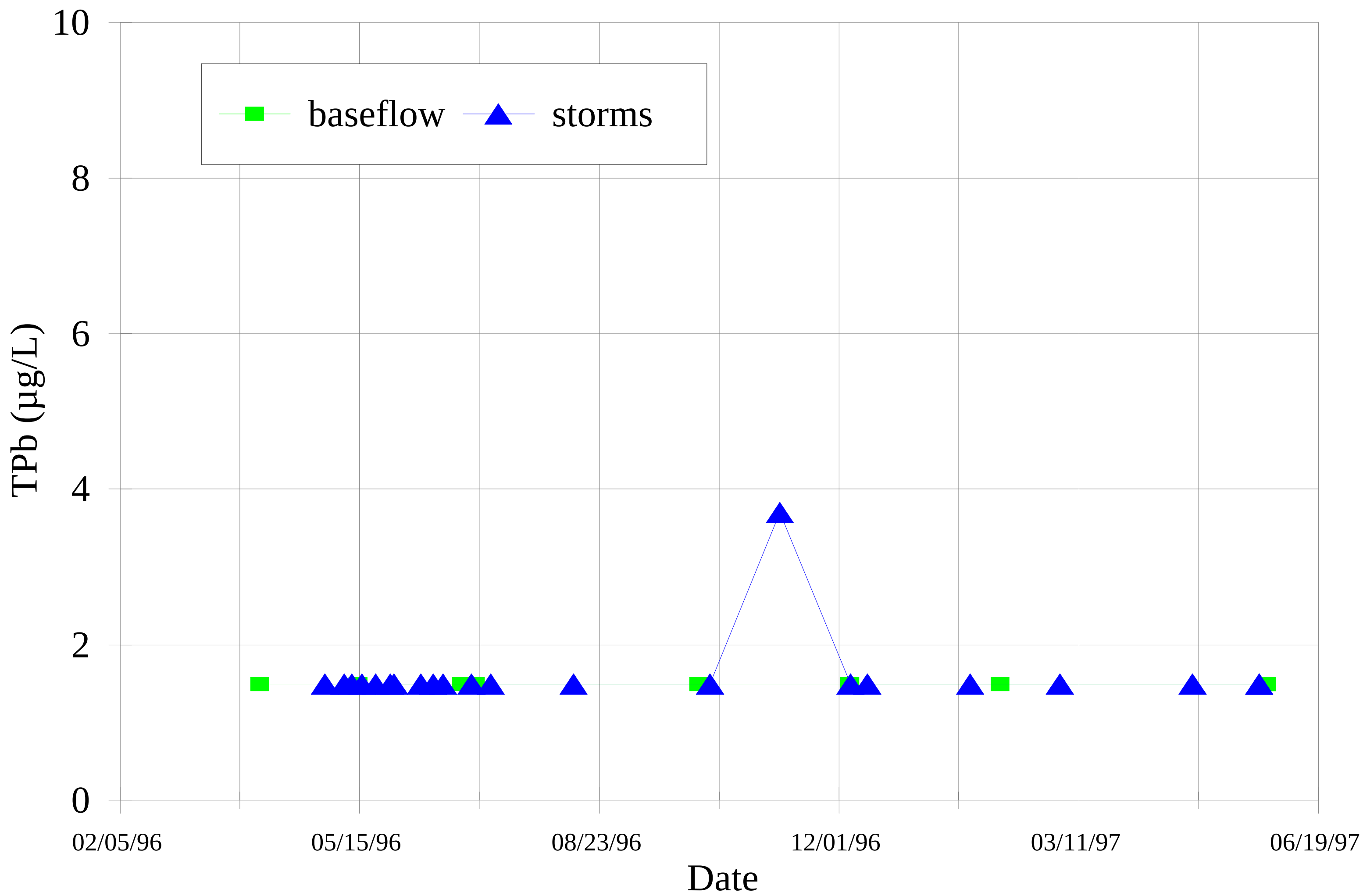

Figure C.19 Outlet storm EMCs and baseflow concentrations: TPb, µg/L

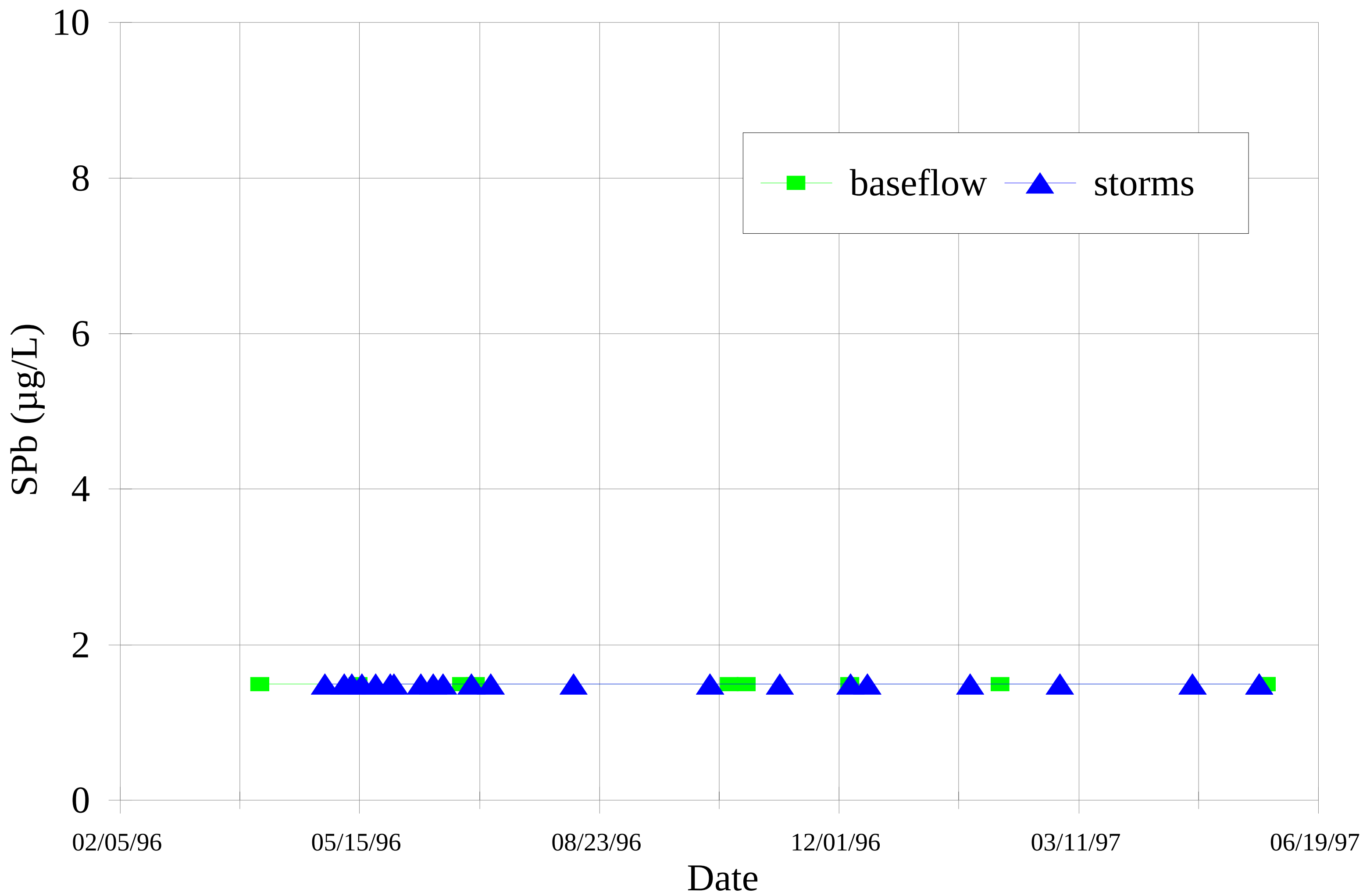

Figure C.20 Outlet storm EMCs and baseflow concentrations: SPb, µg/L

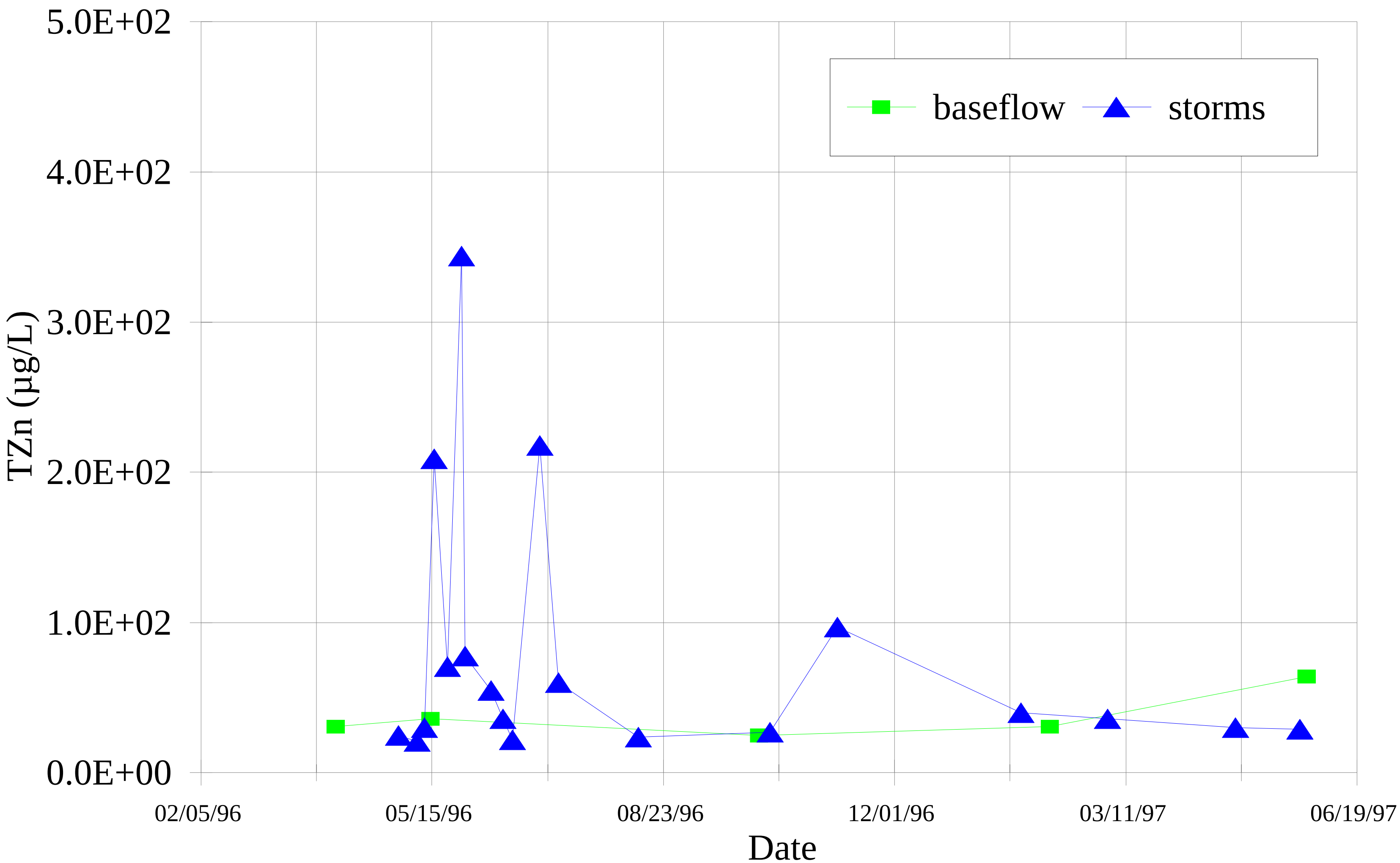

Figure C.21 Outlet storm EMCs and baseflow concentrations: TZn, µg/L

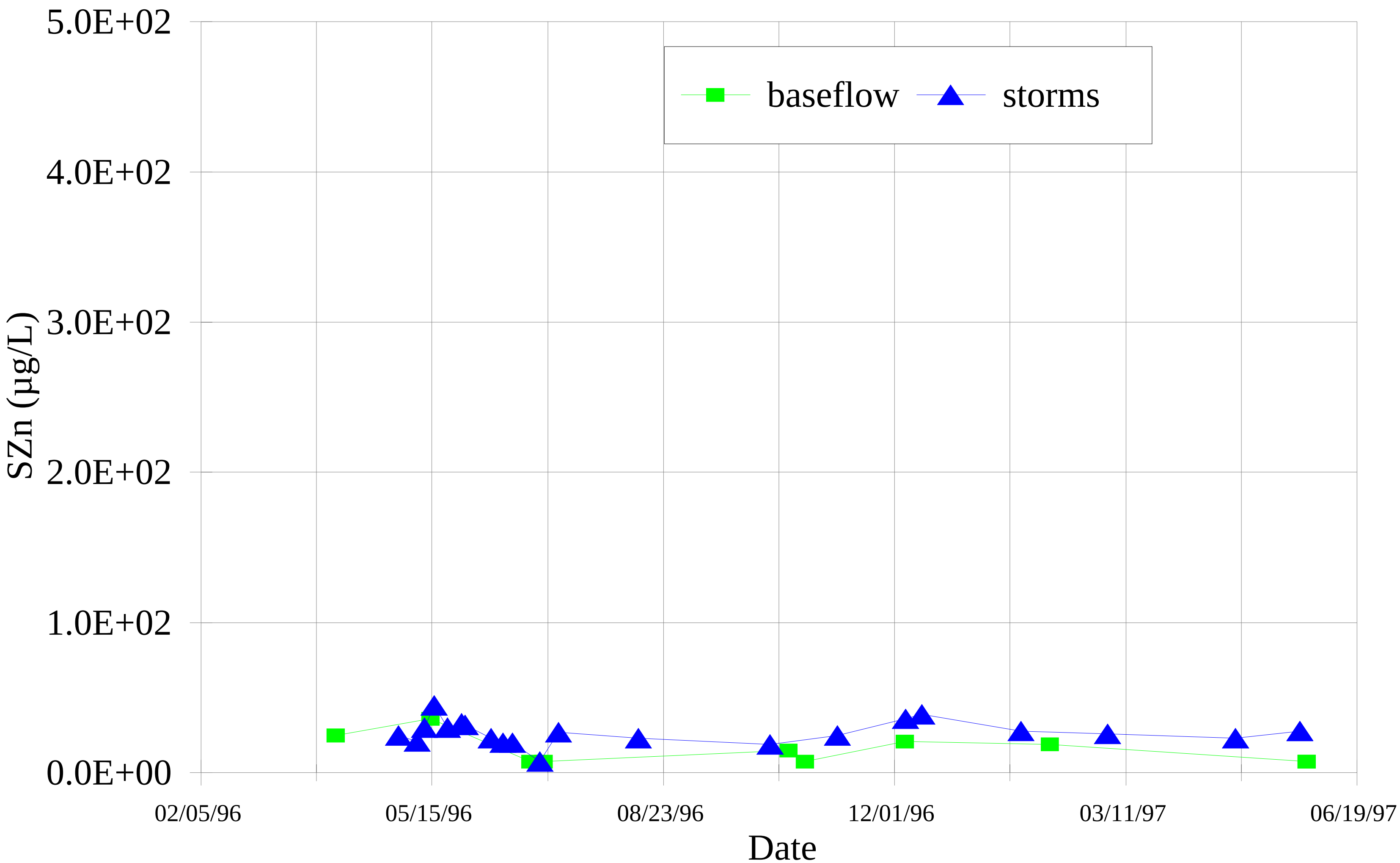

Figure C.22 Outlet storm EMCs and baseflow concentrations: SZn, µg/L

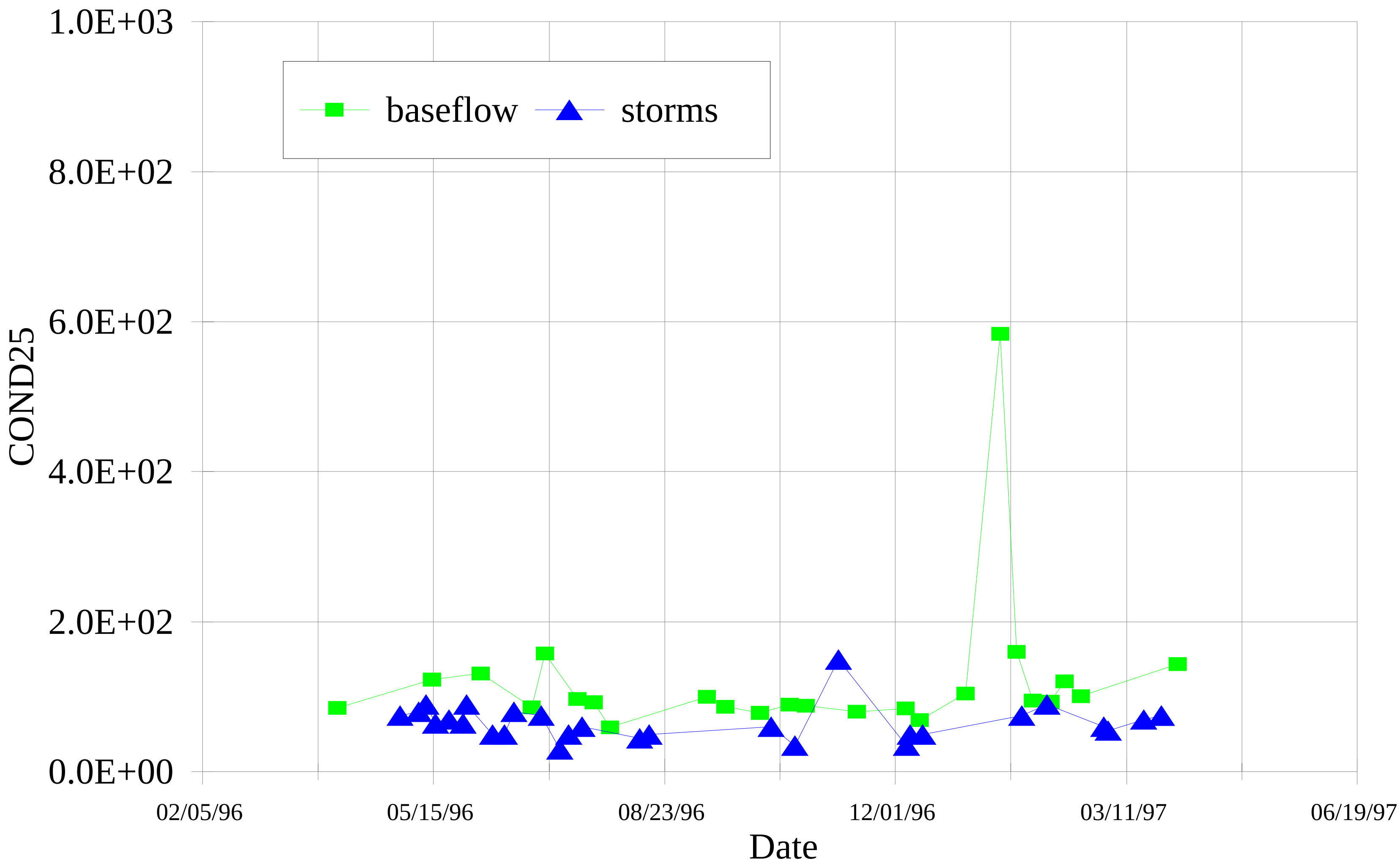

Figure C.23 Outlet storm EMCs and baseflow concentrations: conductivity at 25 deg C

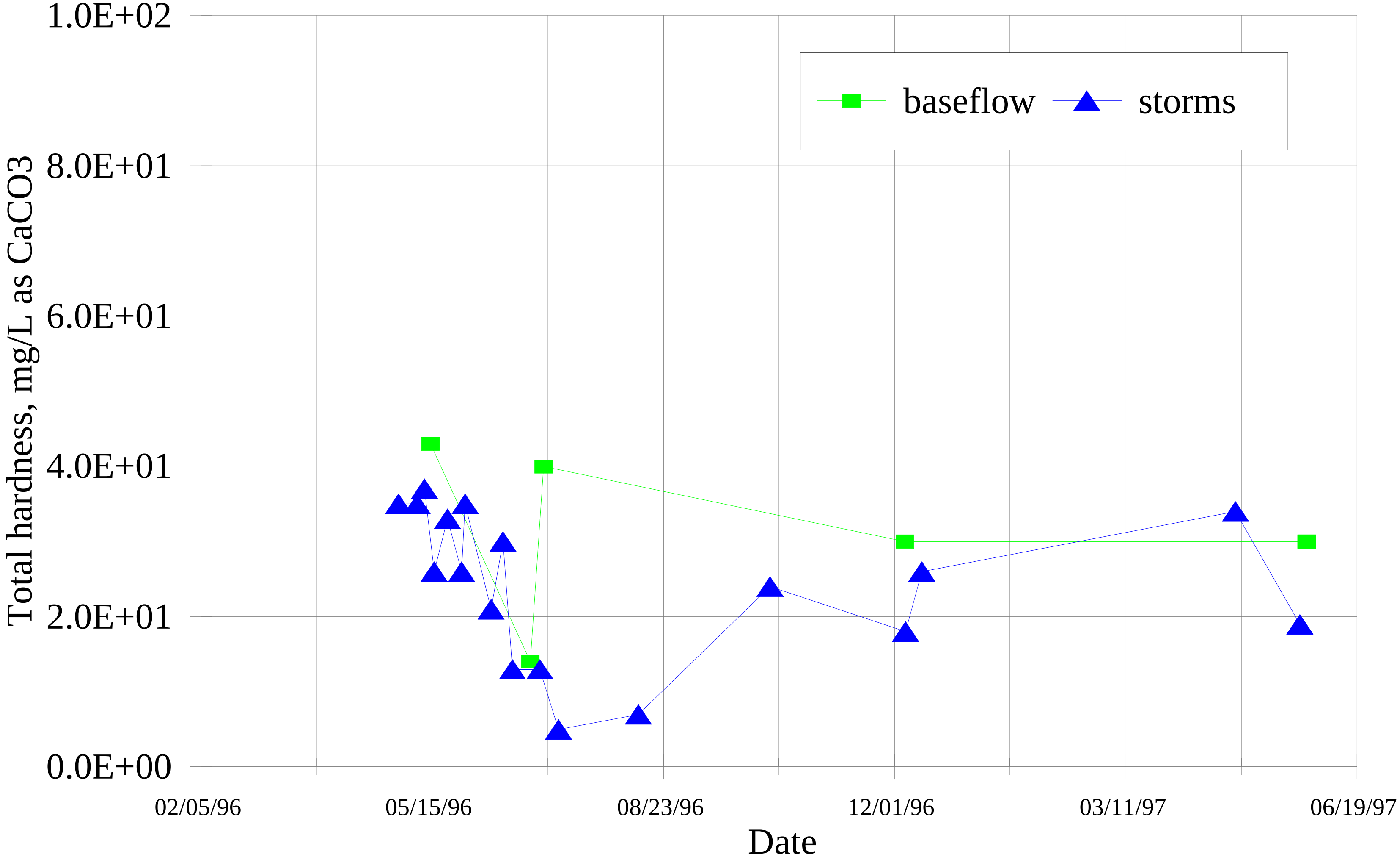

Figure C.24 Outlet storm EMCs and baseflow concentrations: Total hardness, mg/L as CaCO3

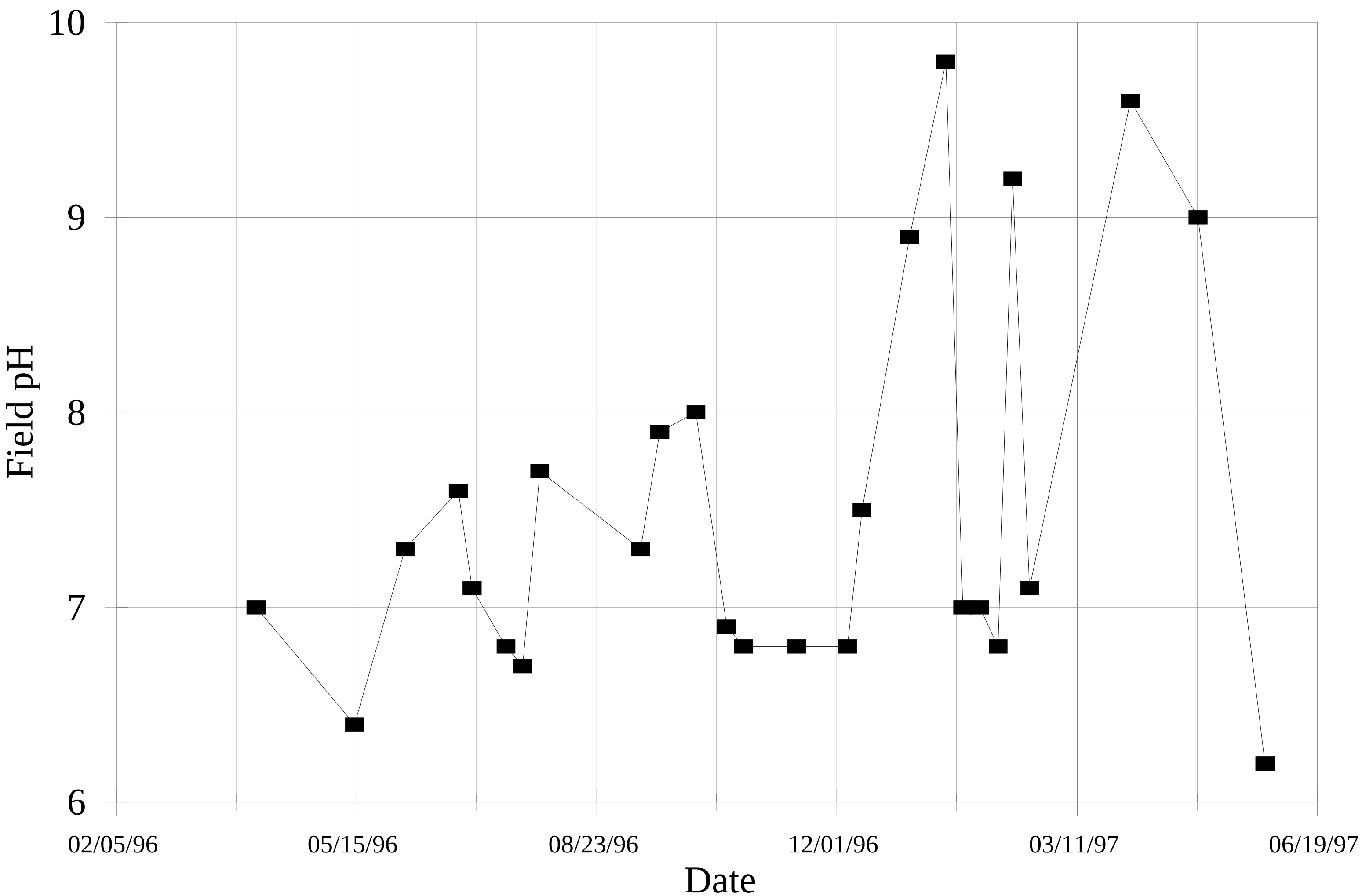

Figure C.25 Baseflow concentrations: field pH

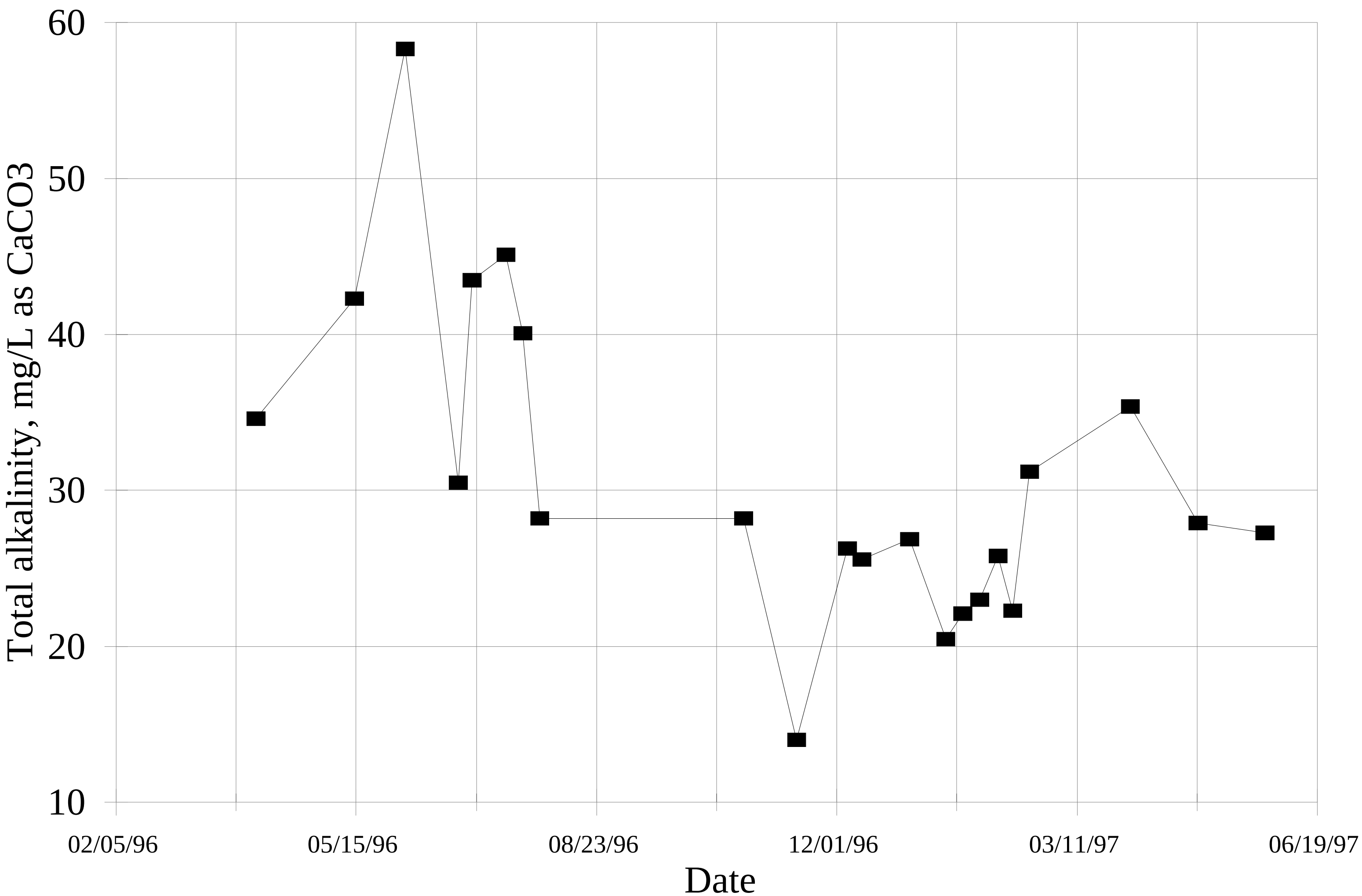

Figure C.26 Baseflow concentrations: Total alkalinity, mg/L as CaCO3

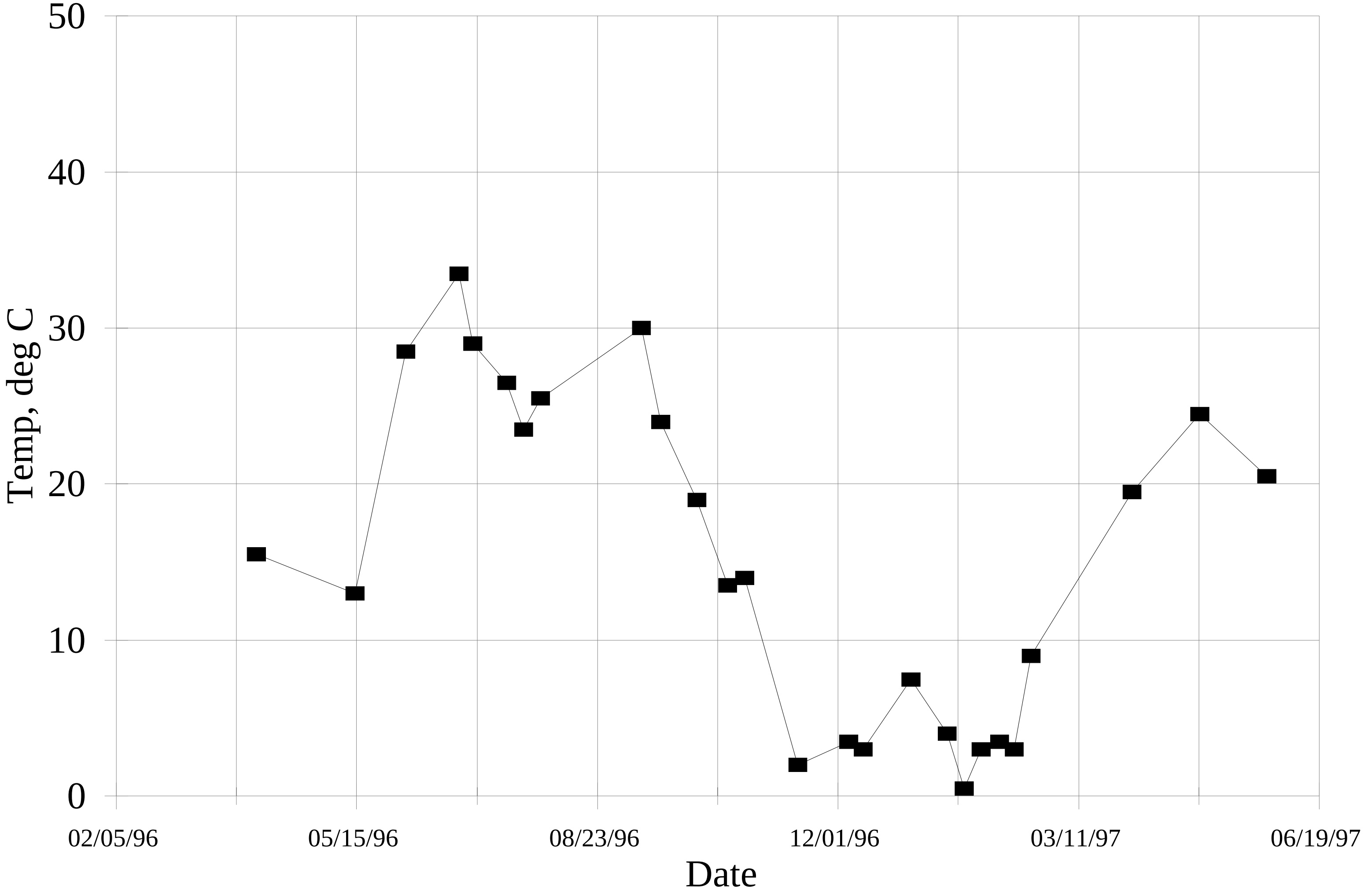

Figure C.27 Baseflow sample temperatures, deg C

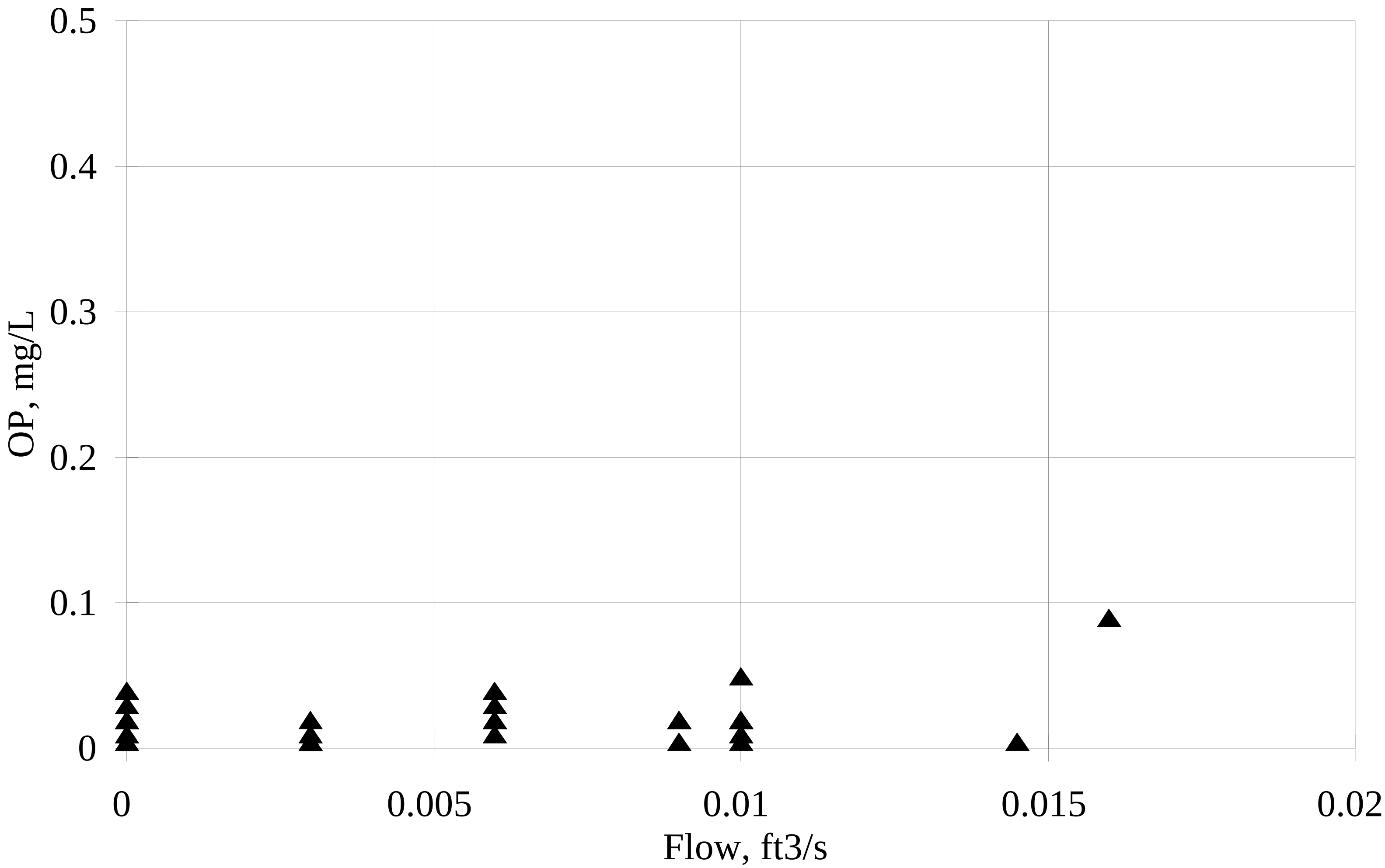

Figure C.28 Baseflow OP conc. vs. flow rate

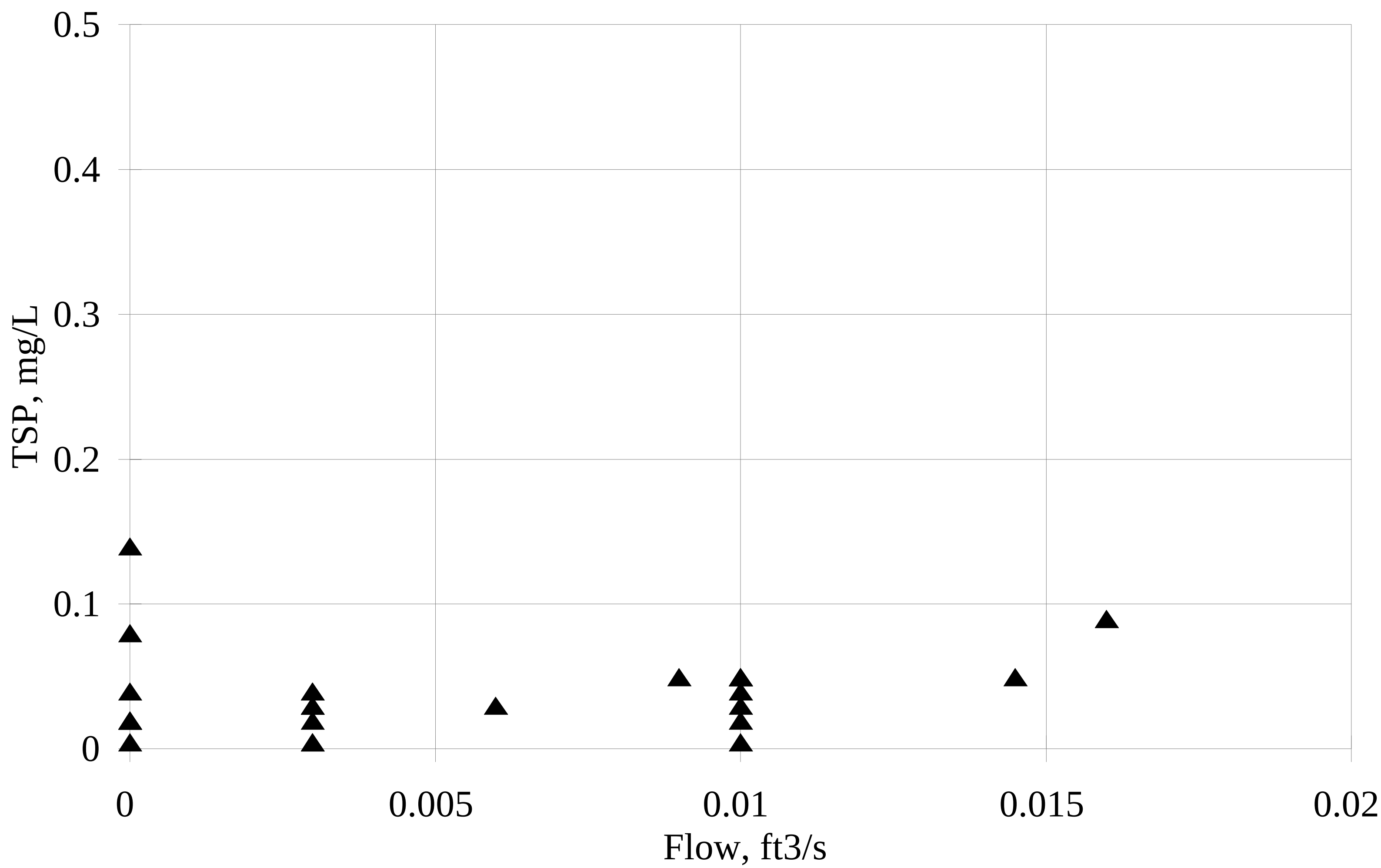

Figure C.29 Baseflow TSP conc. vs. flow rate

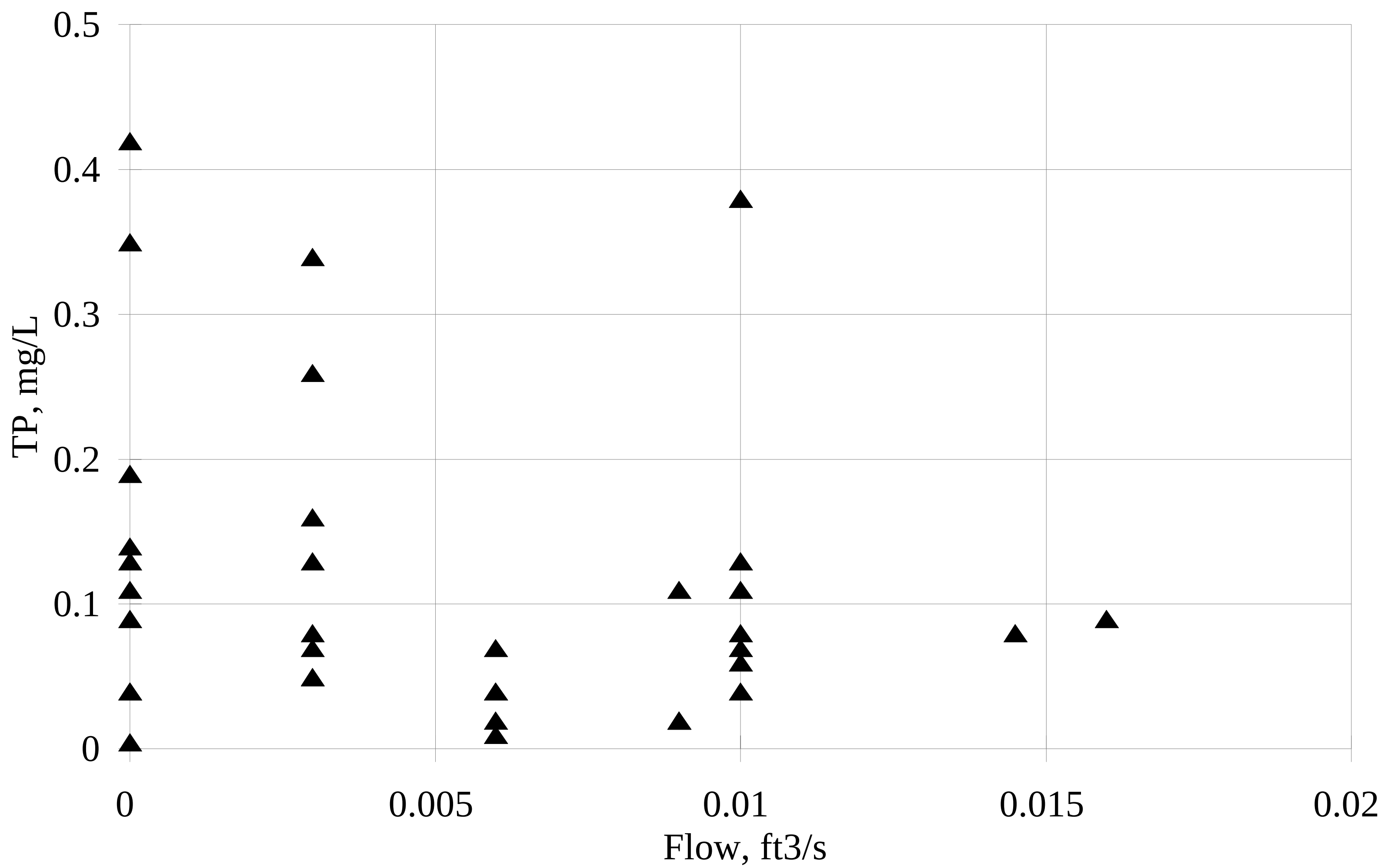

Figure C.30 Baseflow TP conc. vs. flow rate

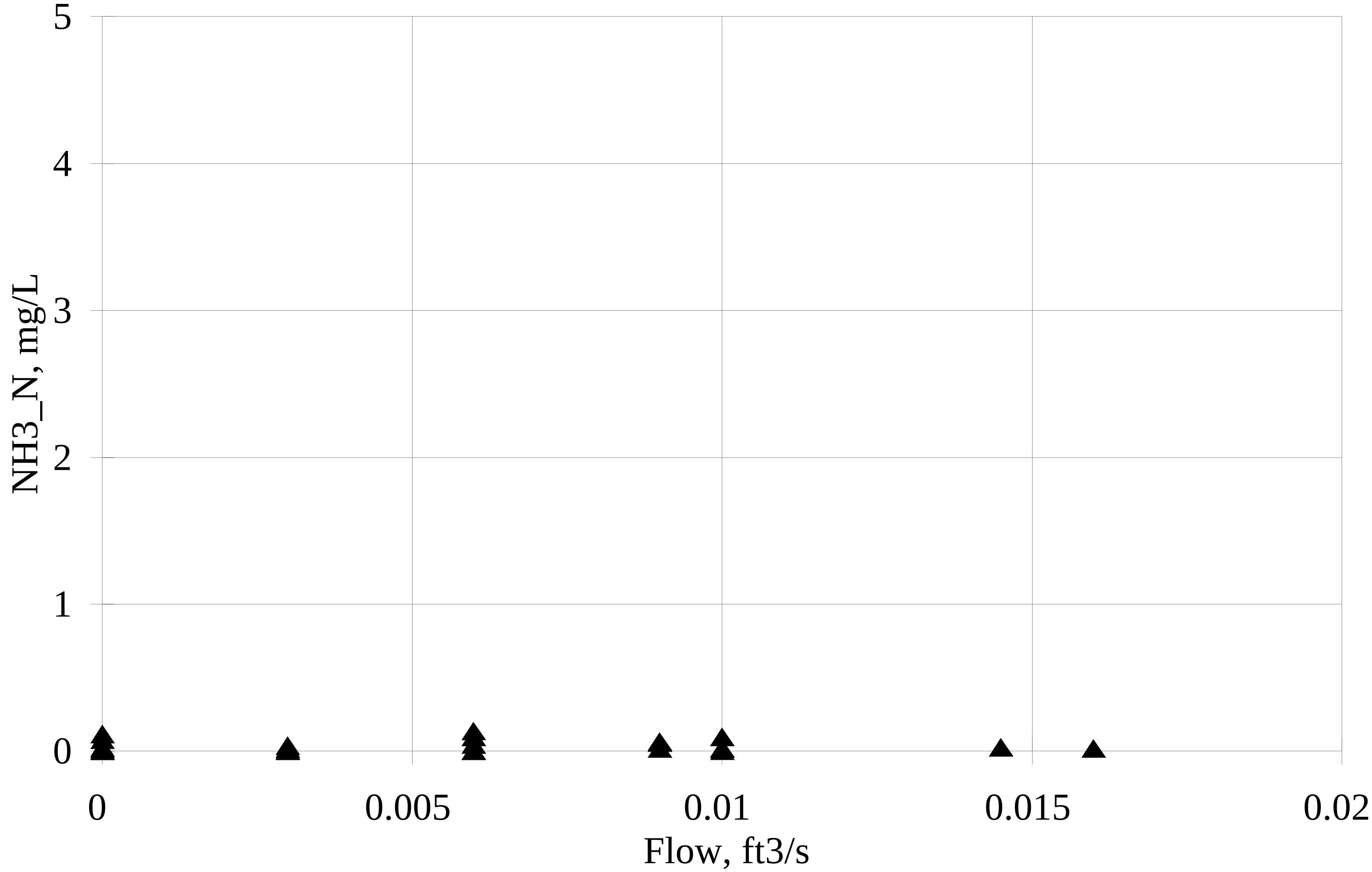

Figure C.31 Baseflow NH3_N conc. vs. flow rate

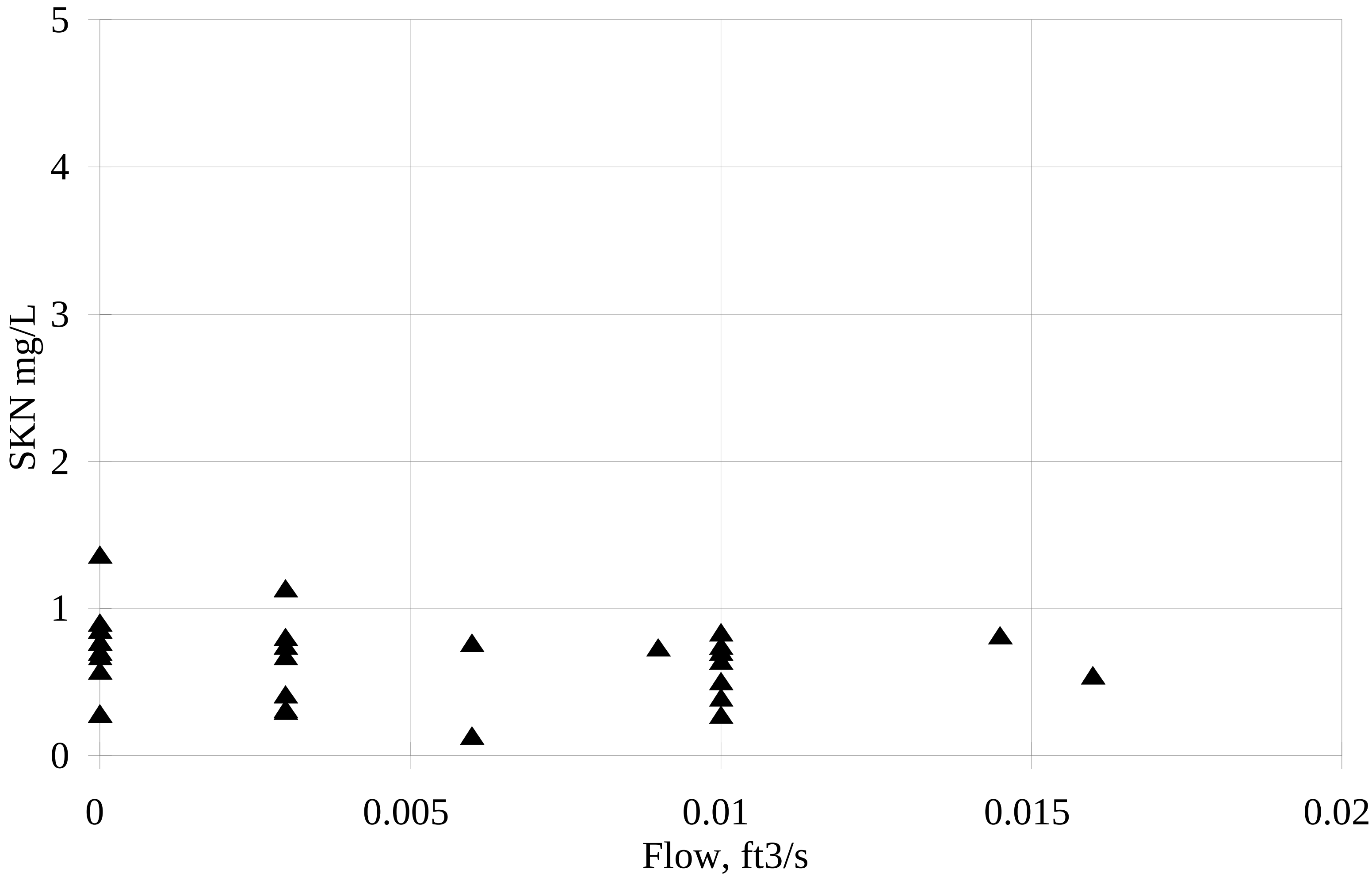

Figure C.32 Baseflow SKN conc. vs. flow rate

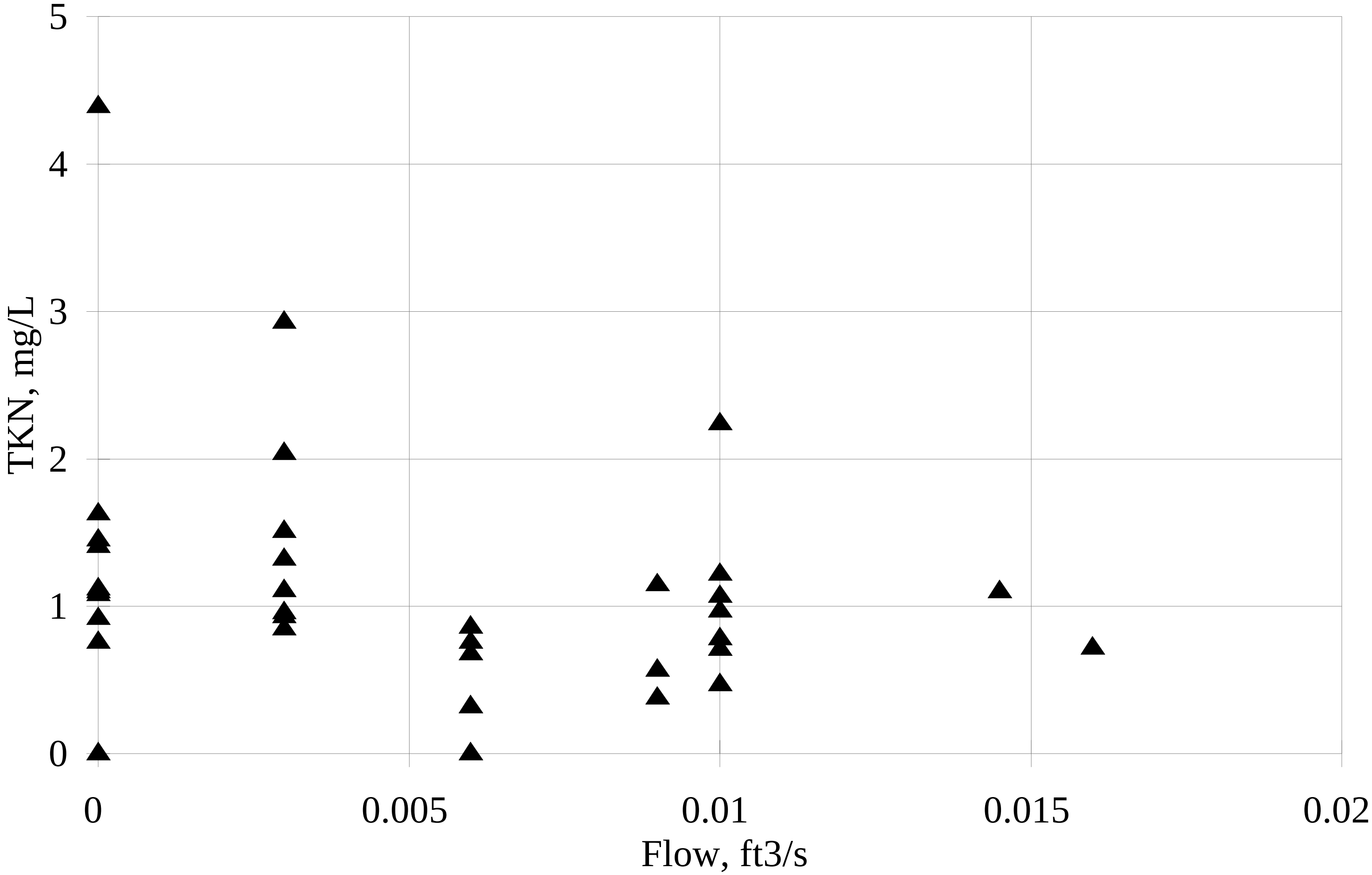

Figure C.33 Baseflow TKN conc. vs. flow rate

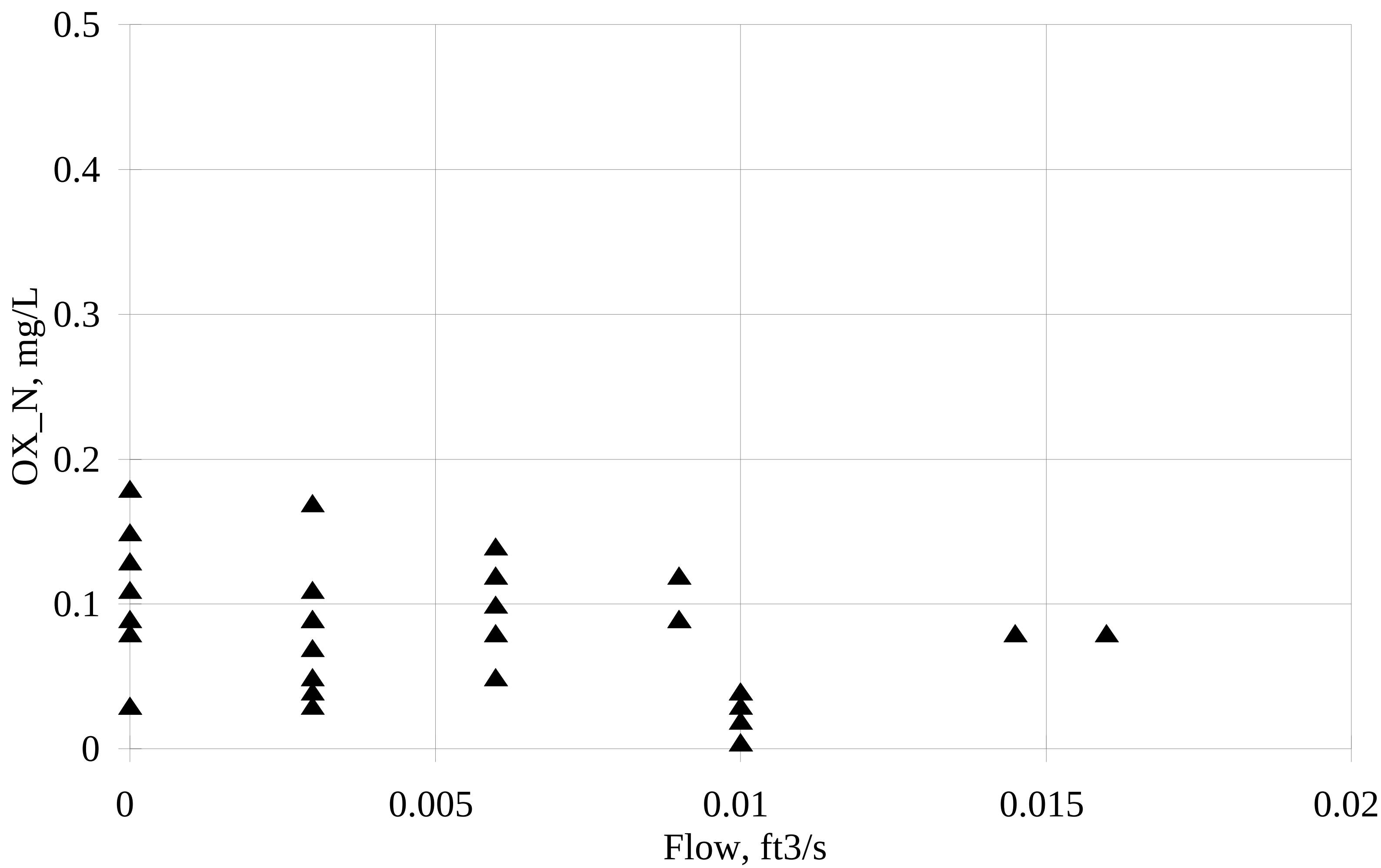

Figure C.34 Baseflow OX_N conc. vs. flow rate

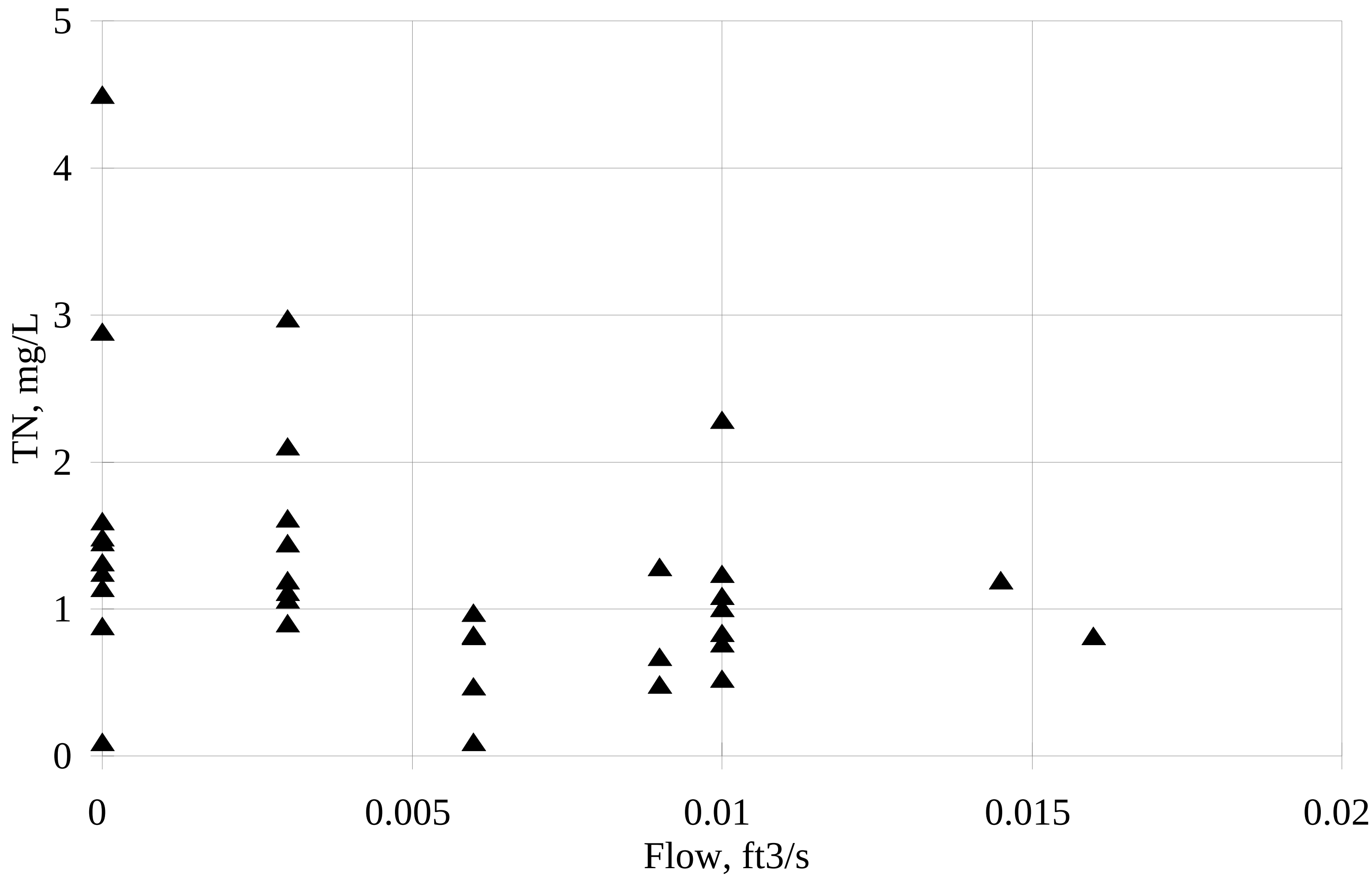

Figure C.35 Baseflow TN conc. vs. flow rate

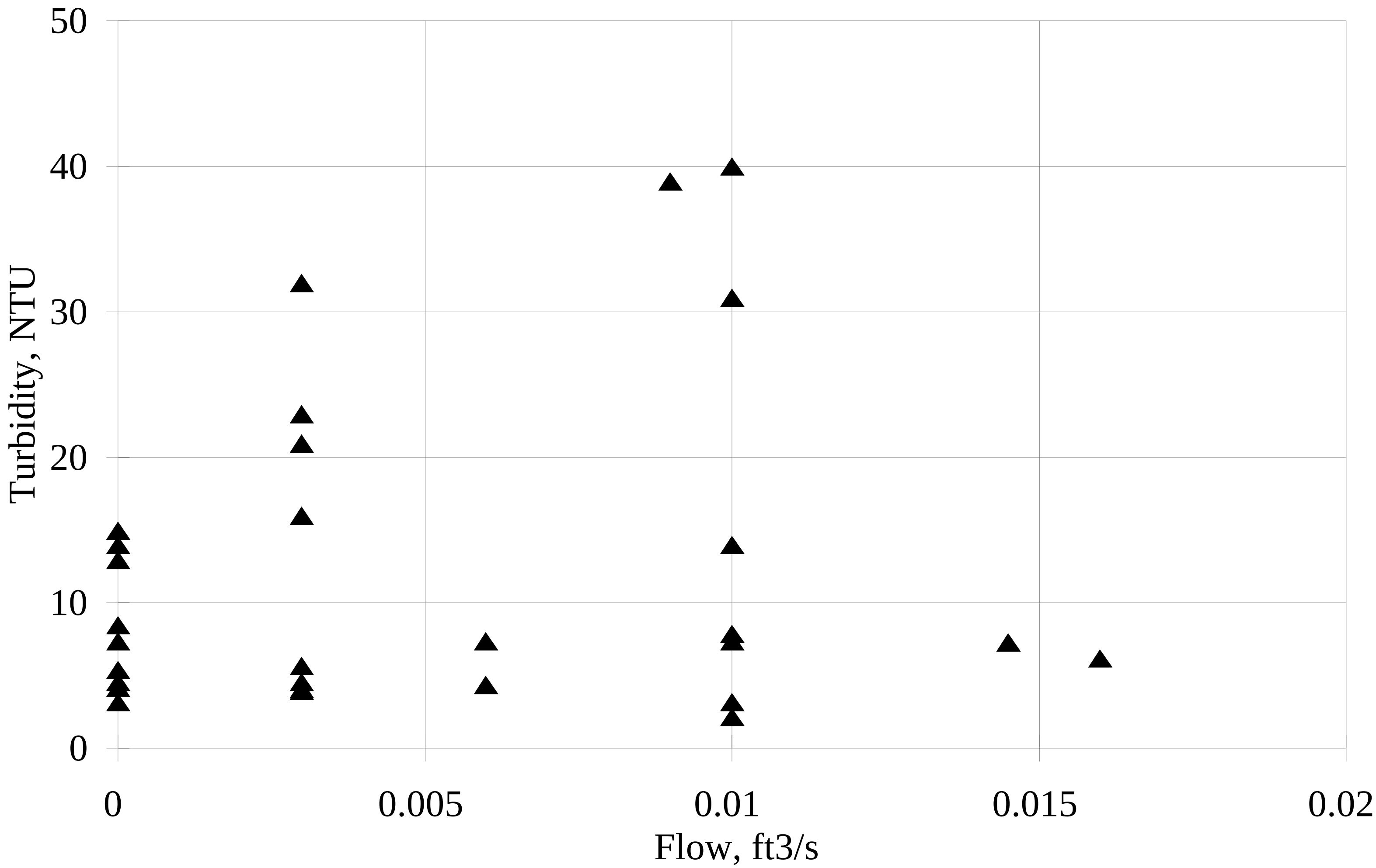

Figure C.36 Baseflow turbidity vs. flow rate

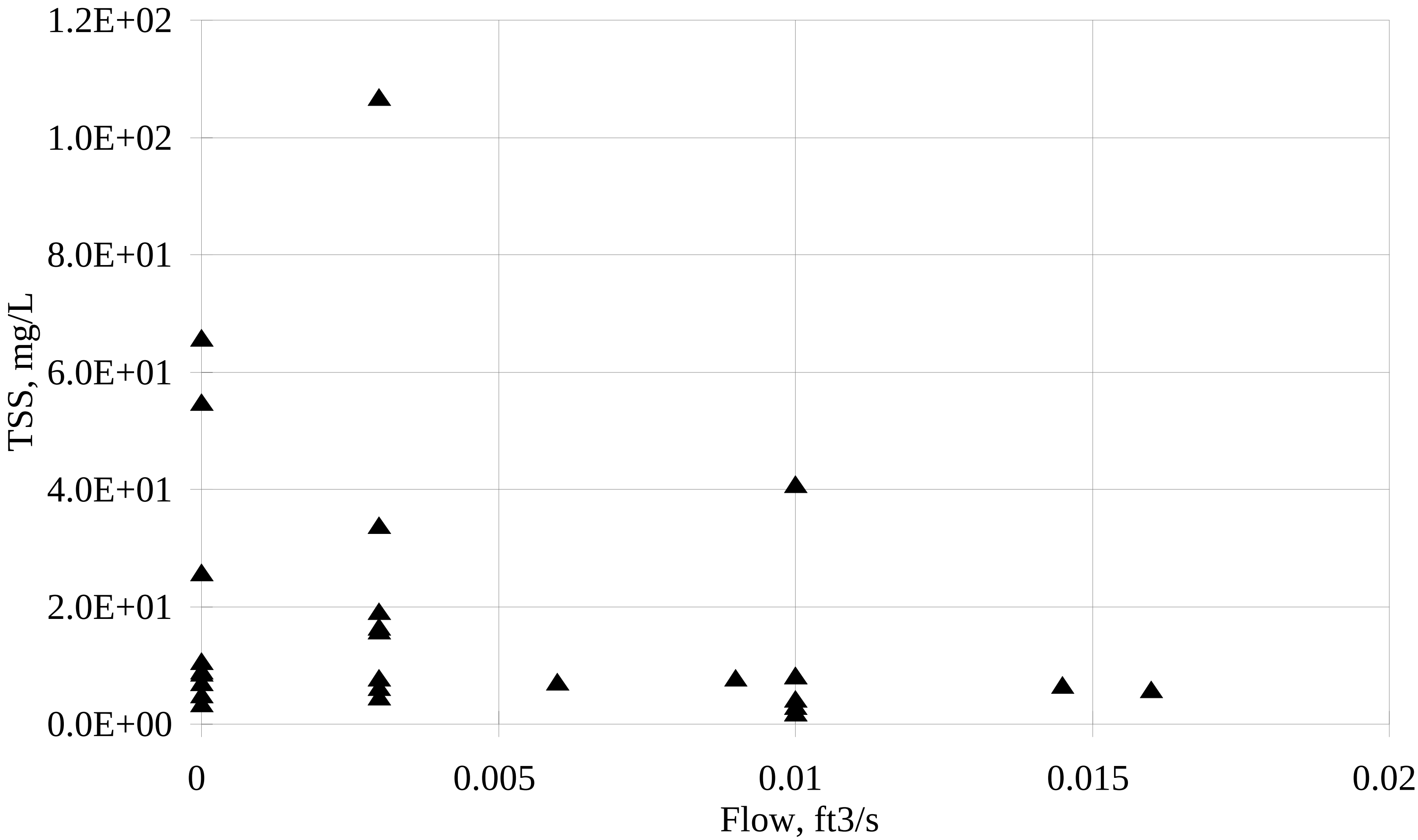

Figure C.37 Baseflow TSS conc. vs. flow rate

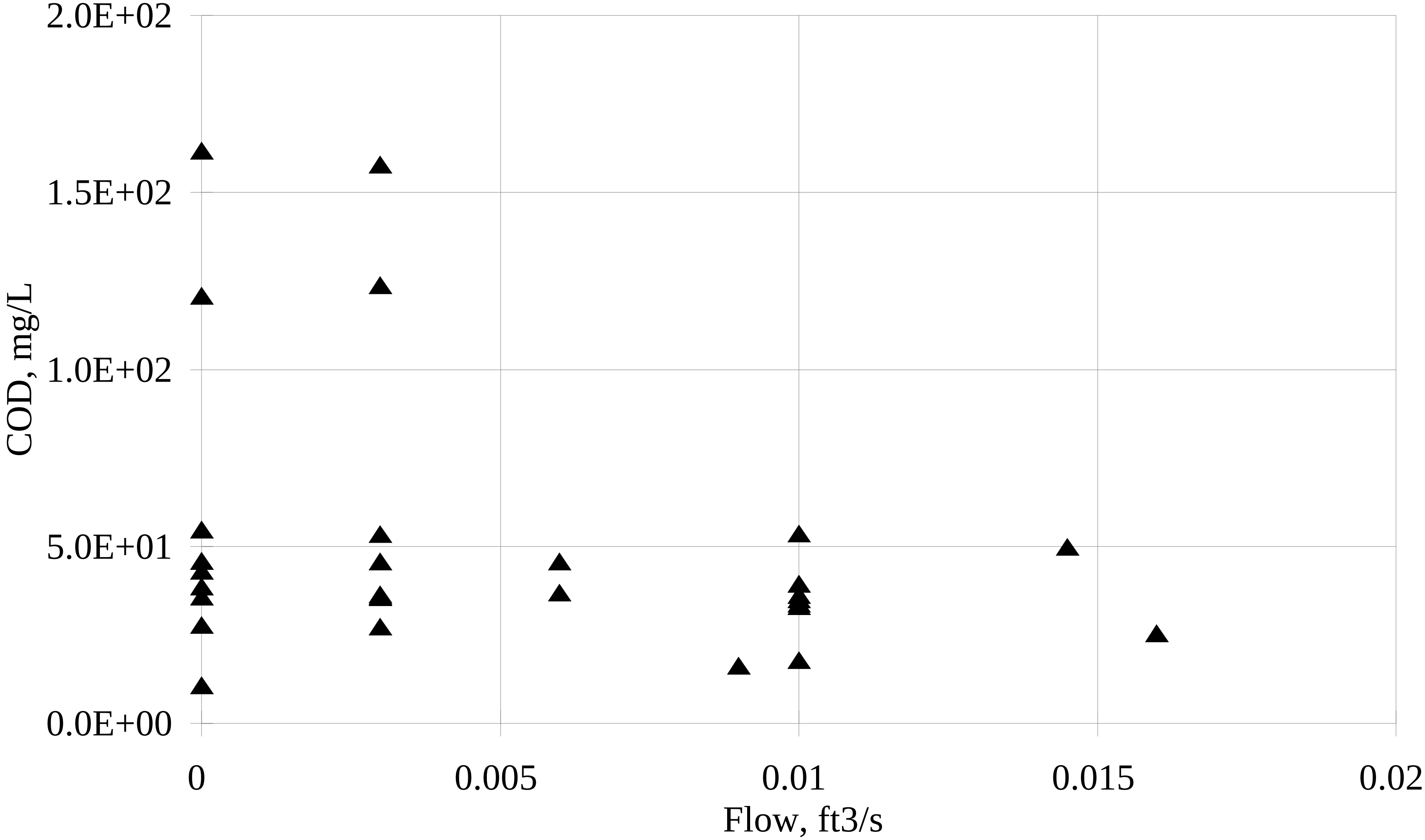

Figure C.38 Baseflow COD conc. vs. flow rate

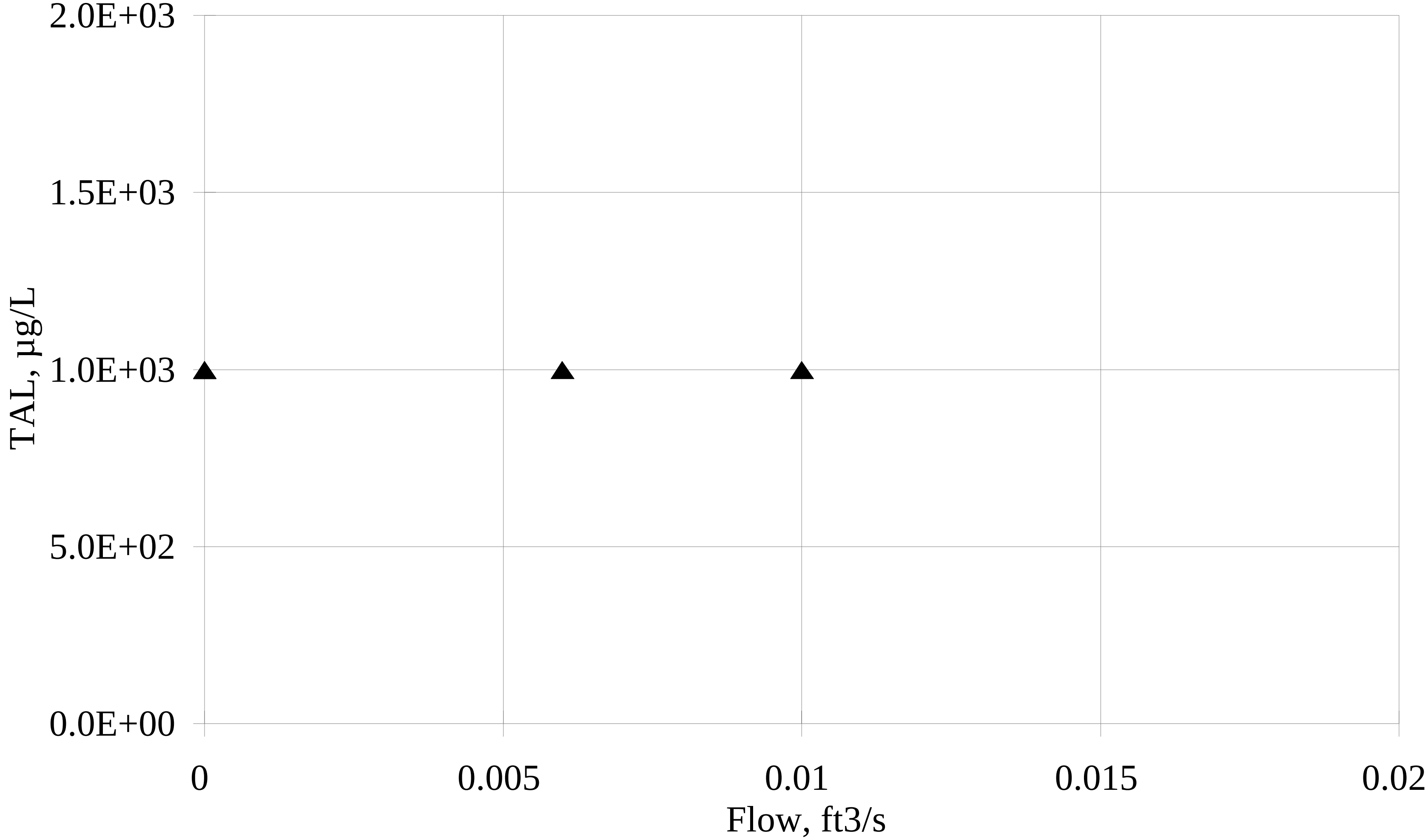

Figure C.39 Baseflow TAl conc. vs. flow rate

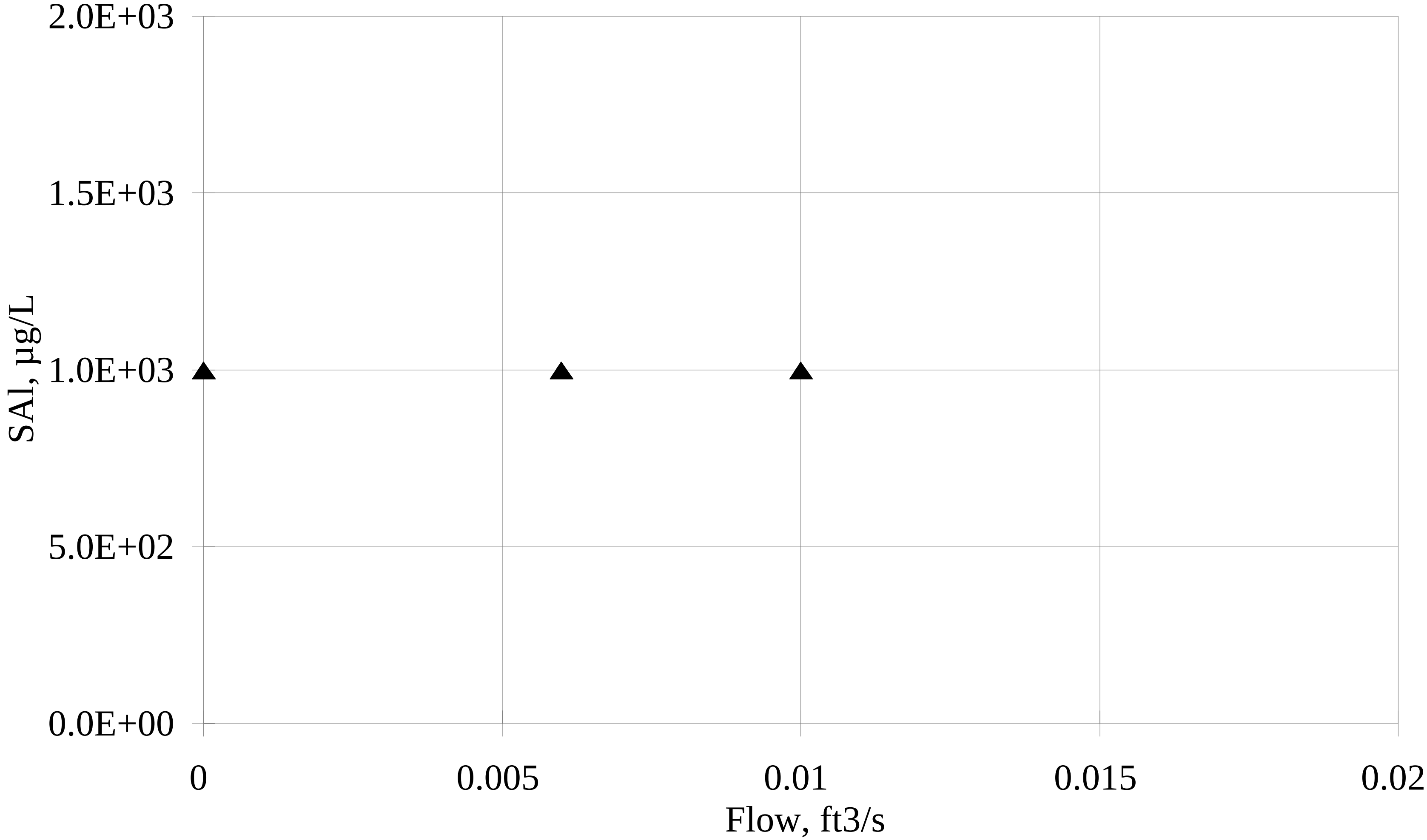

Figure C.40 Baseflow SAl conc. vs. flow rate

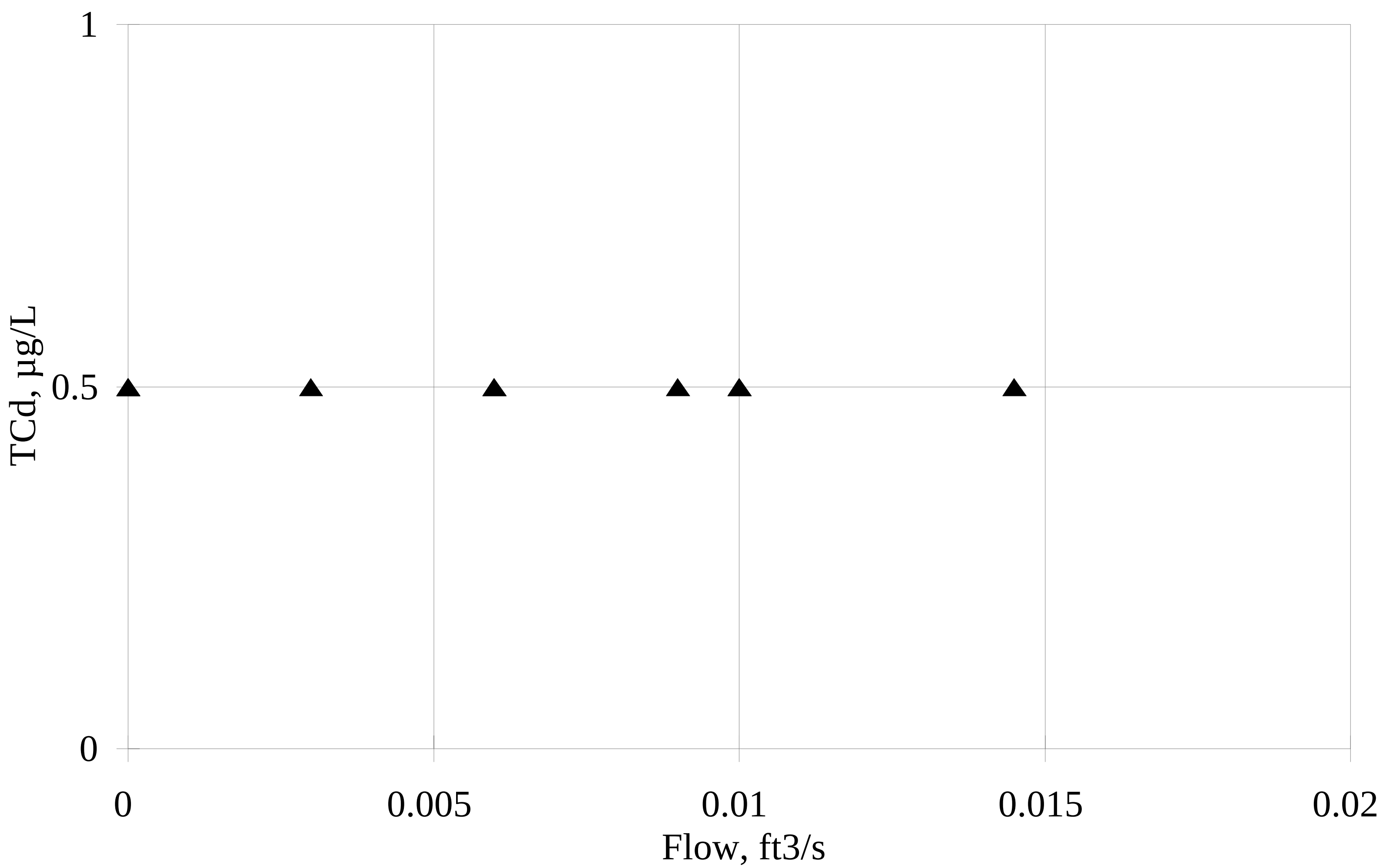

Figure C.41 Baseflow TCd conc. vs. flow rate

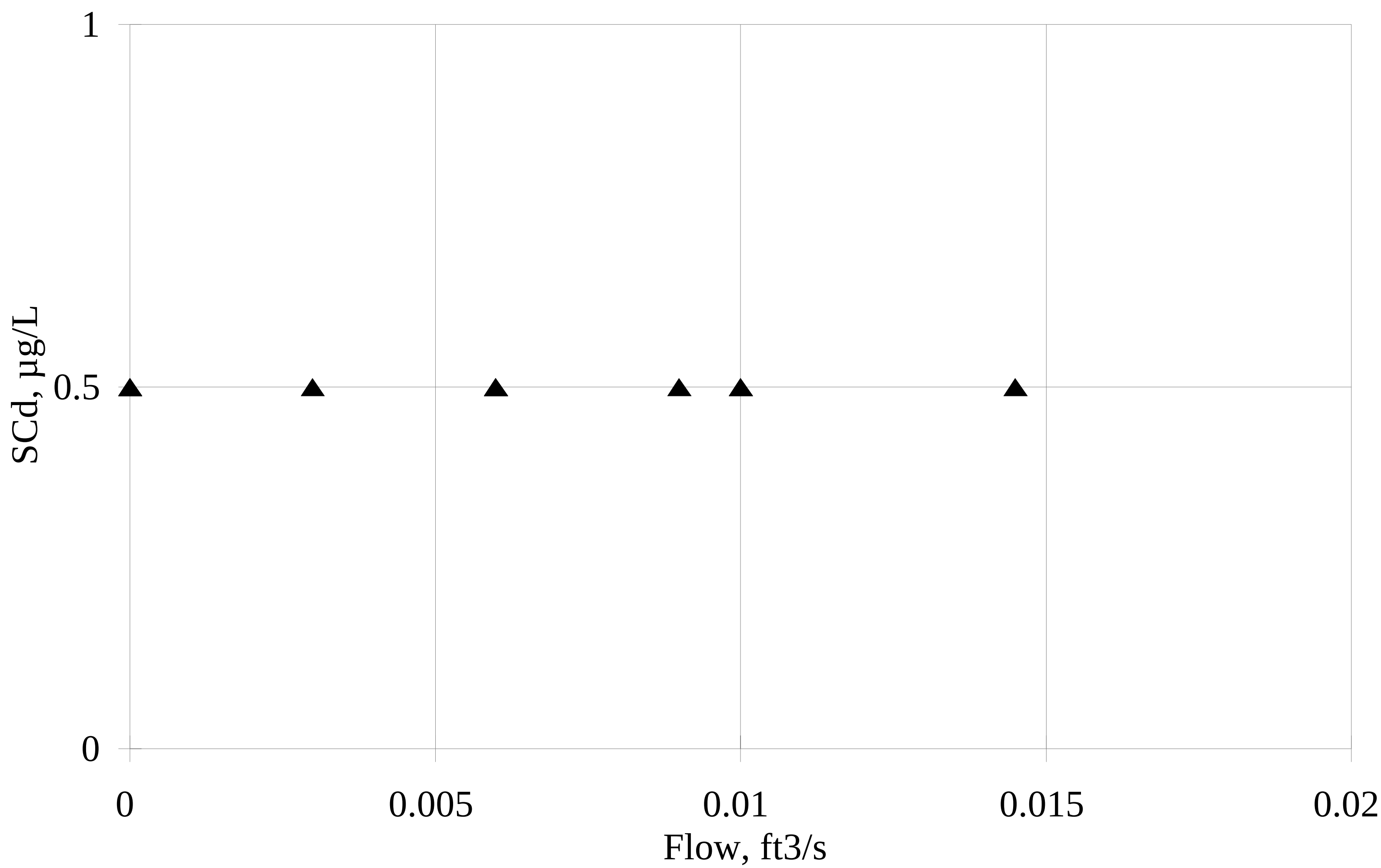

Figure C.42 Baseflow SCd conc. vs. flow rate

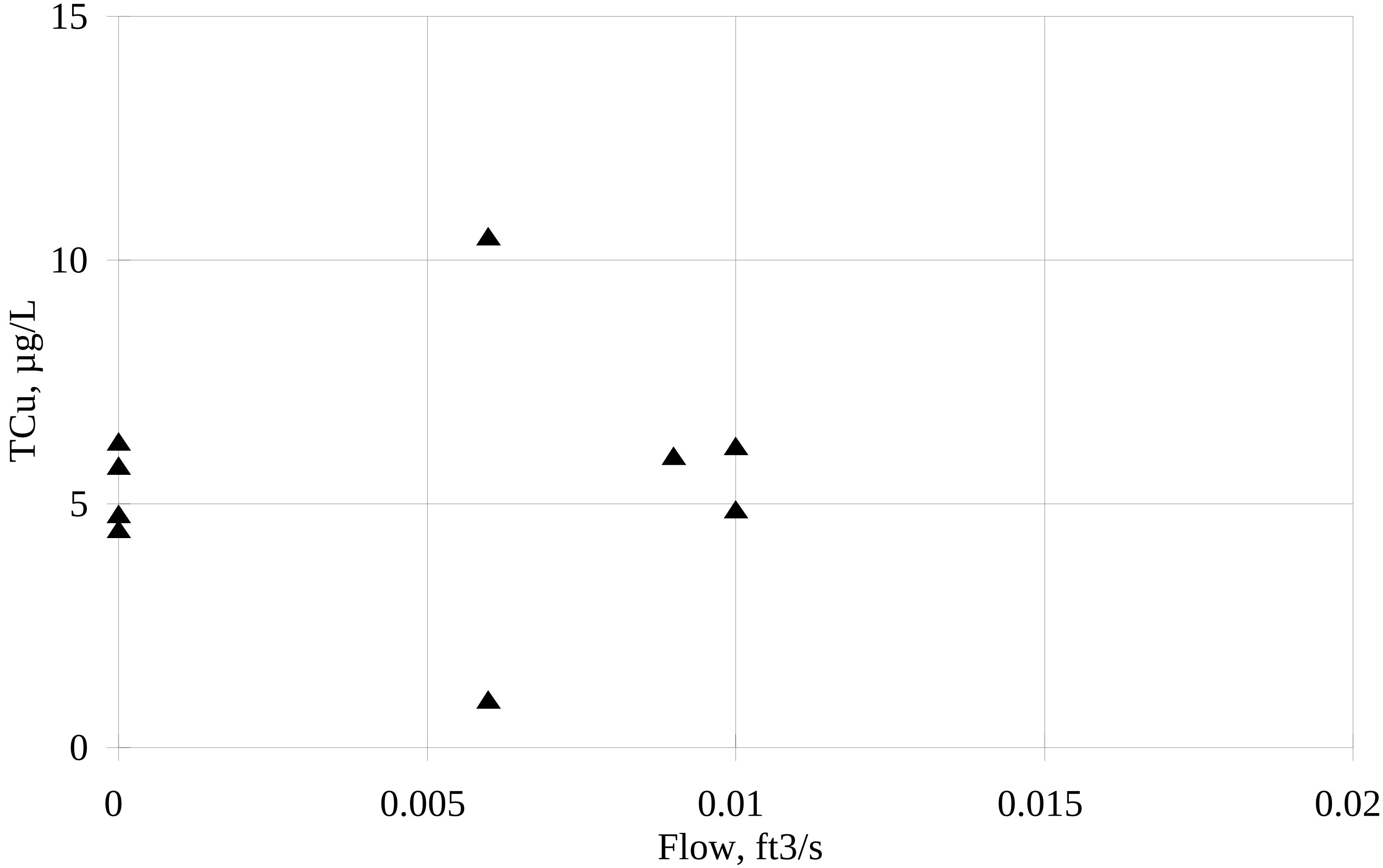

Figure C.43 Baseflow TCu conc. vs. flow rate

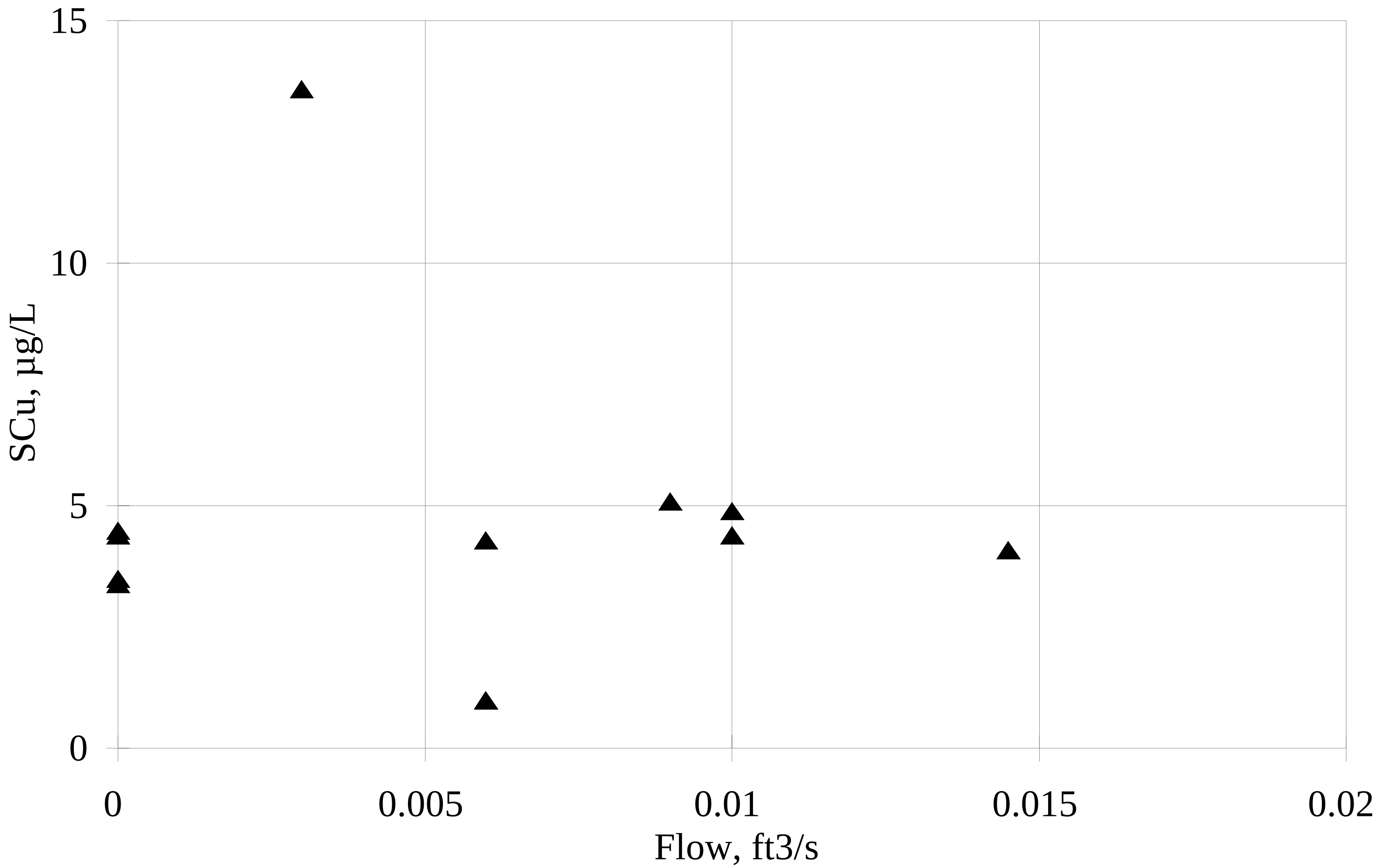

Figure C.44 Baseflow SCu conc. vs. flow rate

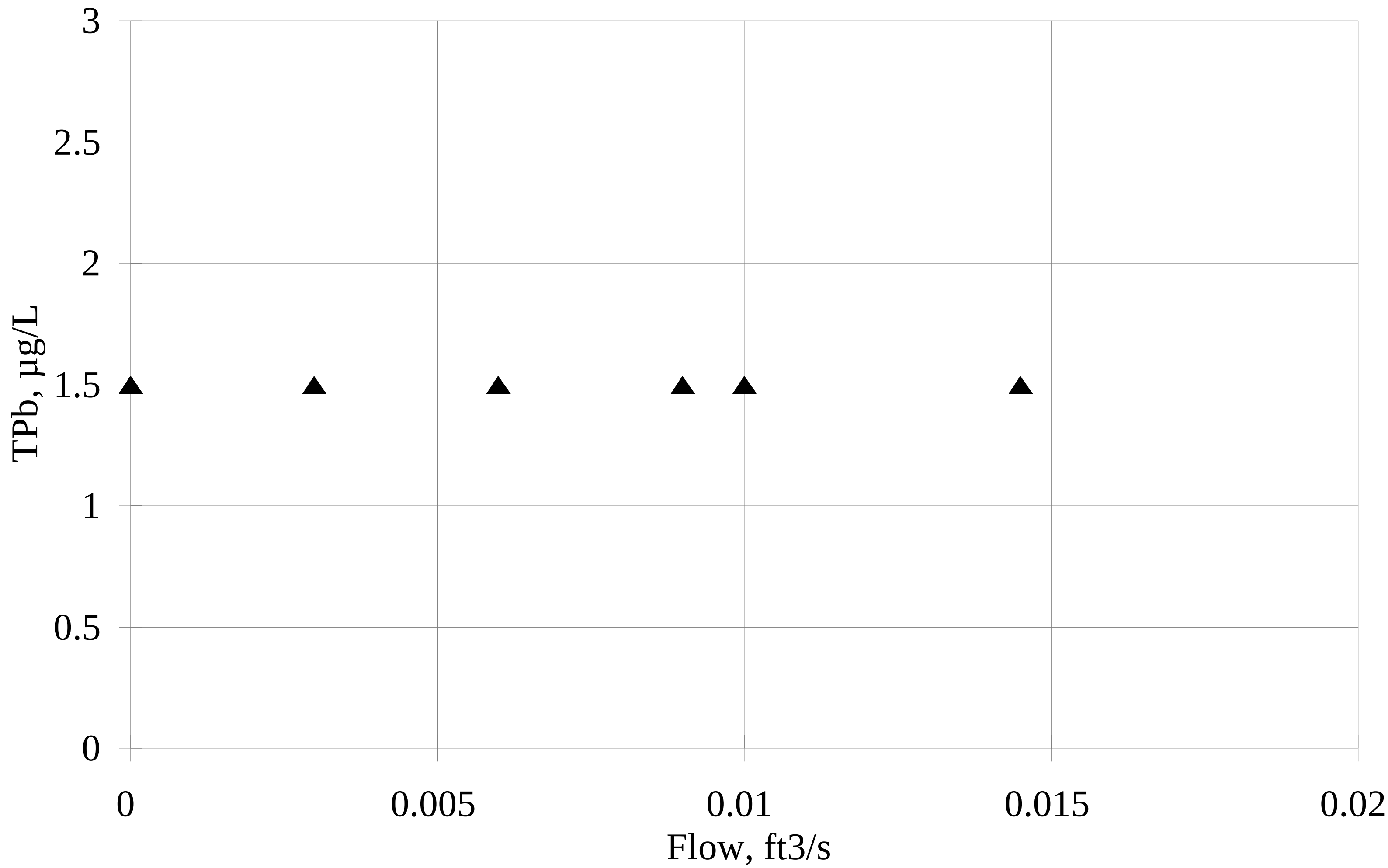

Figure C.45 Baseflow TPb conc. vs. flow rate

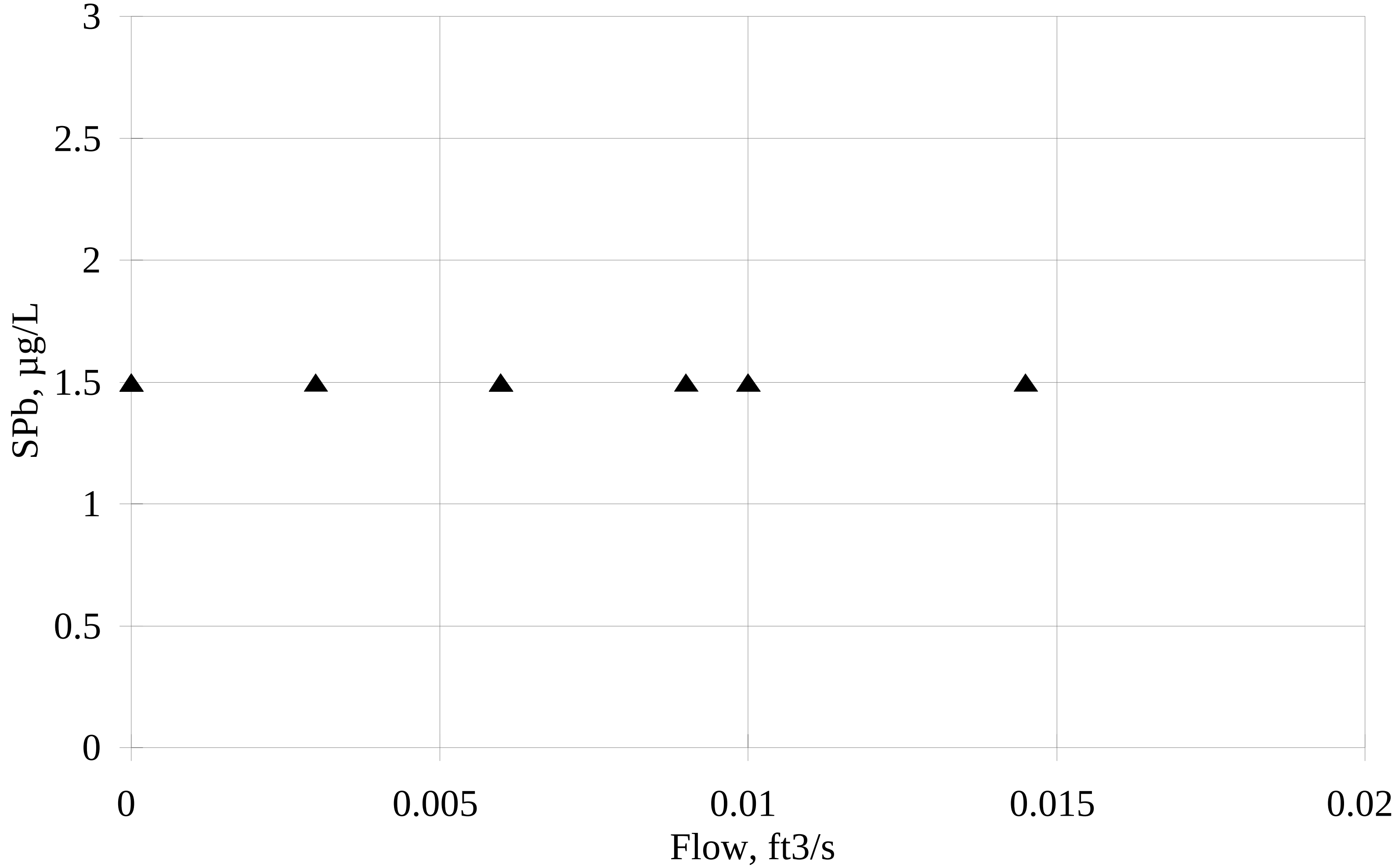

Figure C.46 Baseflow SPb conc. vs. flow rate

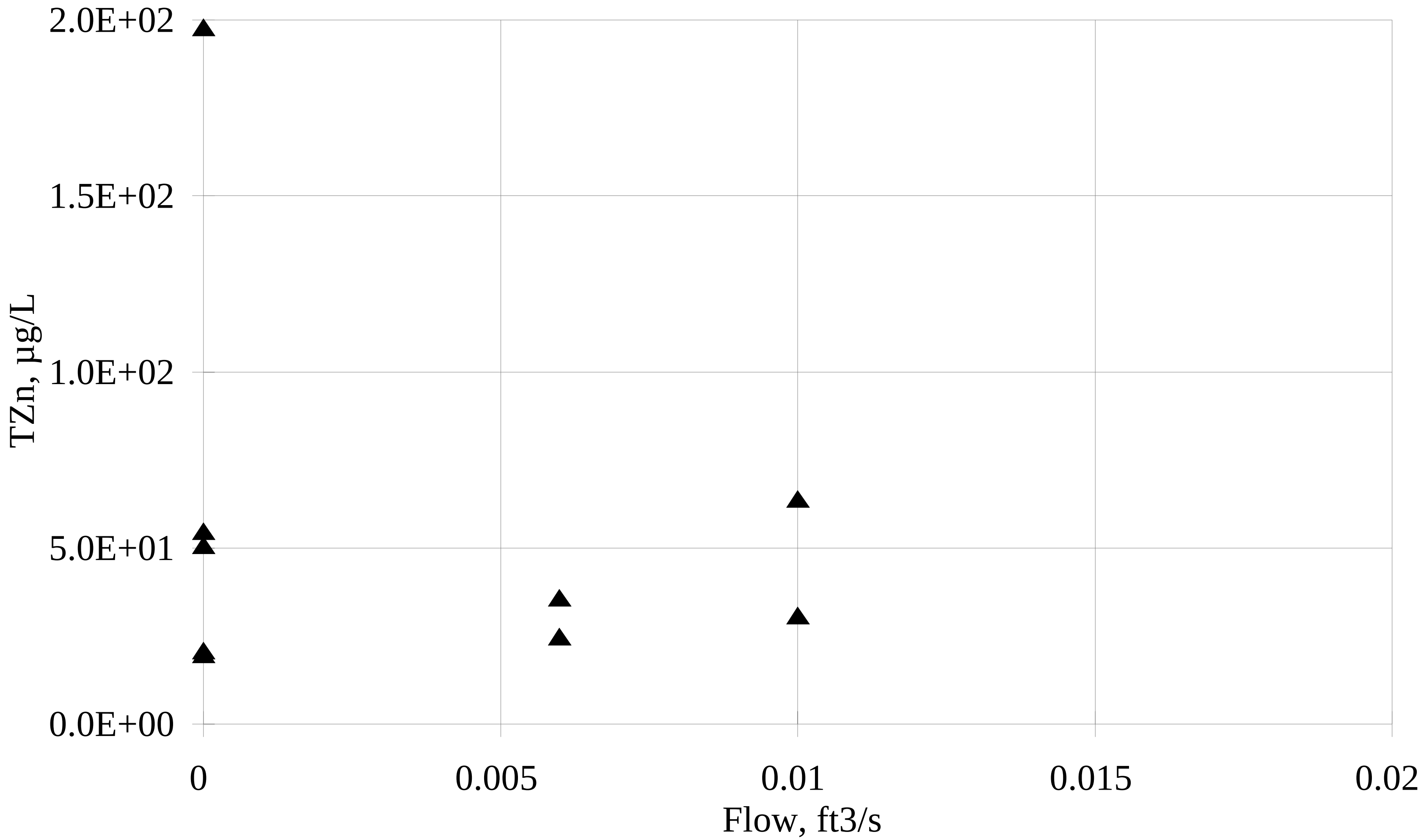

Figure C.47 Baseflow TZn conc. vs. flow rate

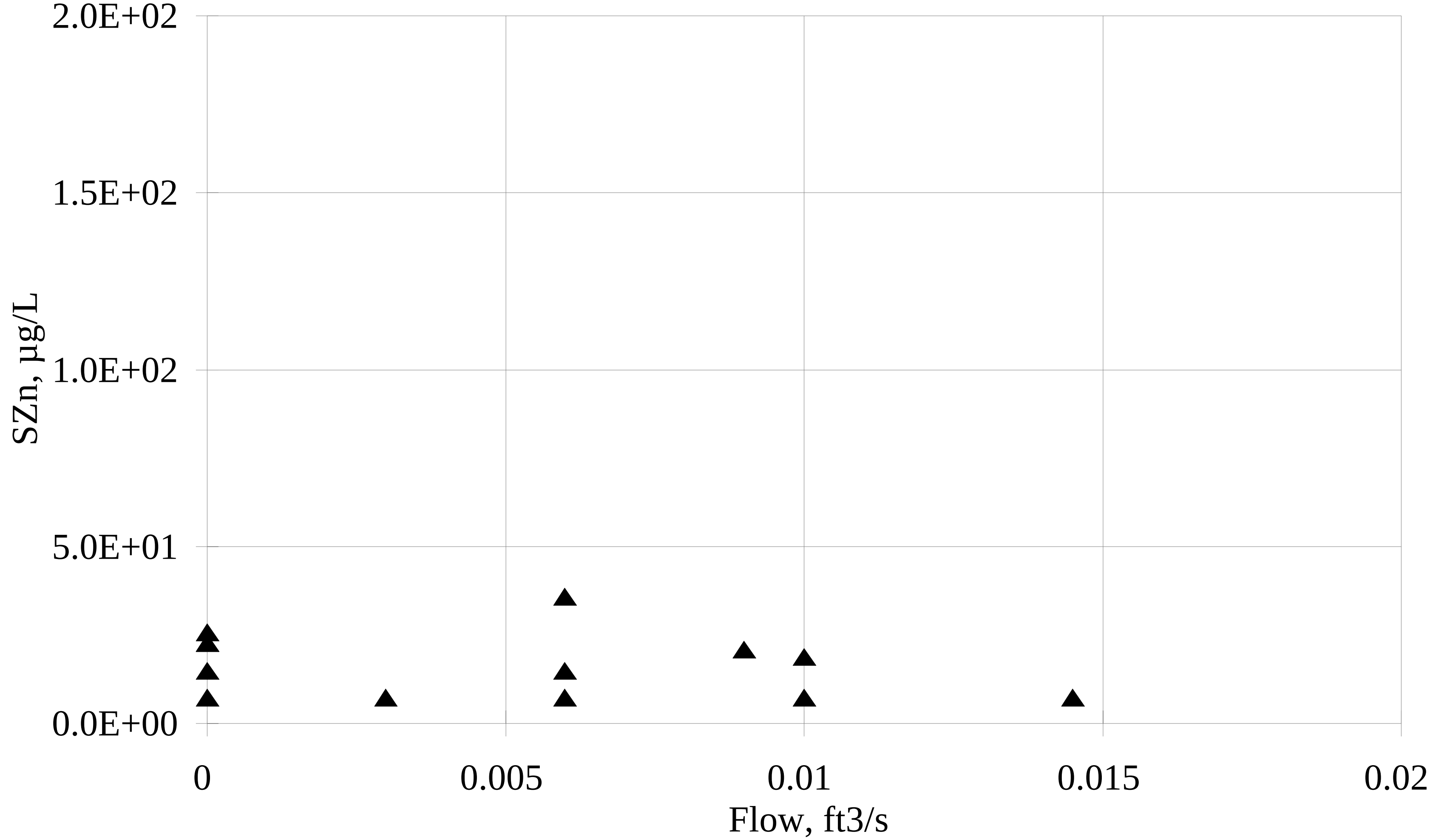

Figure C.48 Baseflow SZn conc. vs. flow rate

APPENDIX D
Paired Inflow and Outflow EMCs and Loads

Table D.1 Storm number/date key

Storm#	Start Date
1*	04/30/96
2	05/08/96
3	05/11/96
4	05/15/96
5*	05/21/96
6*	05/27/96
7	05/29/96
8*	06/09/96
9	06/14/96
10	06/19/96
11	06/30/96
12	07/08/96
13*	07/12/96
14	07/18/96
15	08/12/96
16	08/16/96
17	10/08/96
18	10/18/96
19	11/06/96
20	12/05/96
21	12/07/96
22	12/13/96
23	01/24/97
24	02/04/97
25	03/01/97
26	03/03/97
27	03/18/97
28	03/26/97
29	04/12/97
30	04/17/97
31	04/27/97
32	05/25/97
33	06/01/97

* Subset A storms

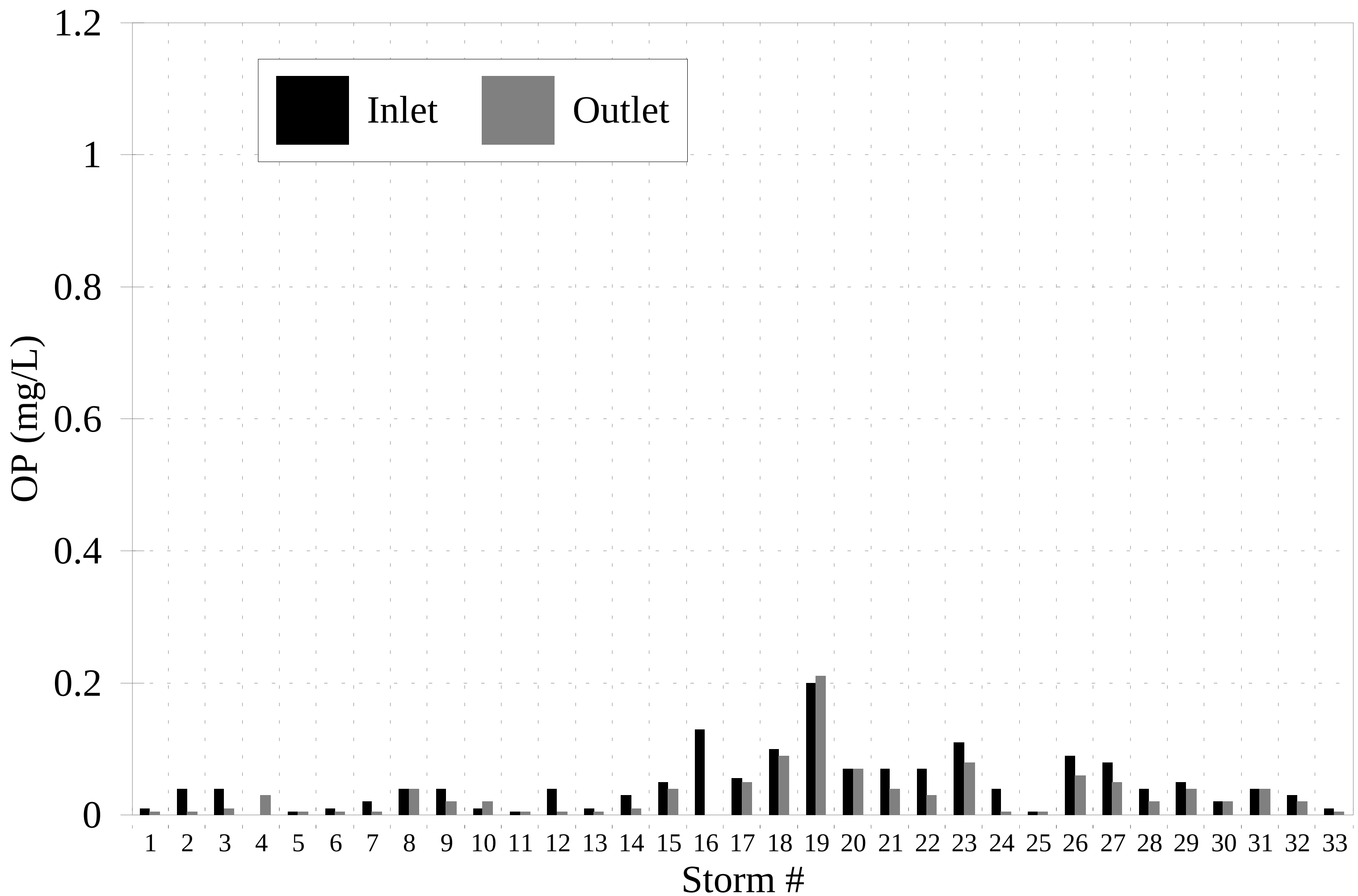

Figure D.1 Inlet-outlet EMC comparison: OP, mg/L

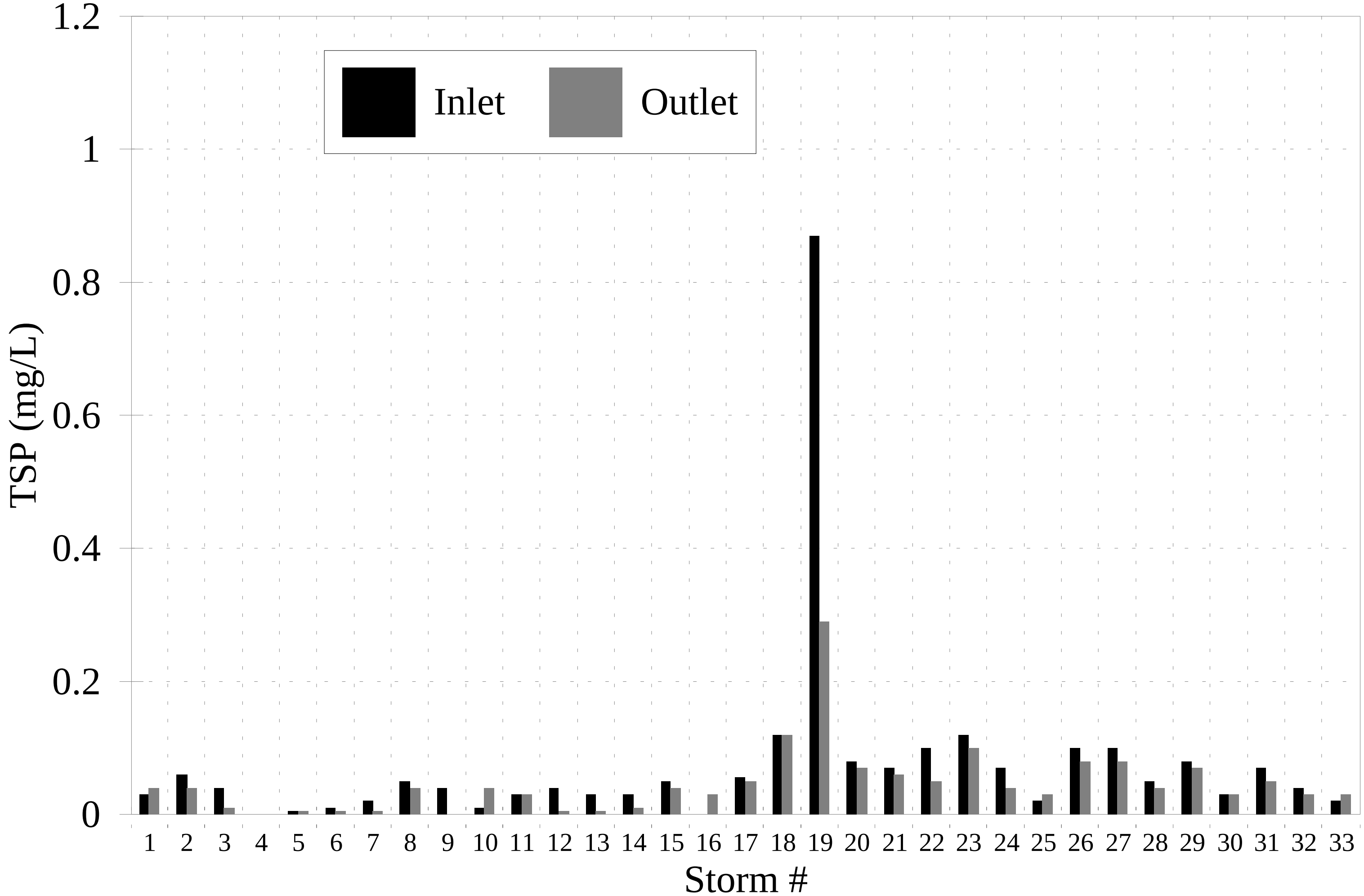

Figure D.2 Inlet-outlet EMC comparison: TSP, mg/L

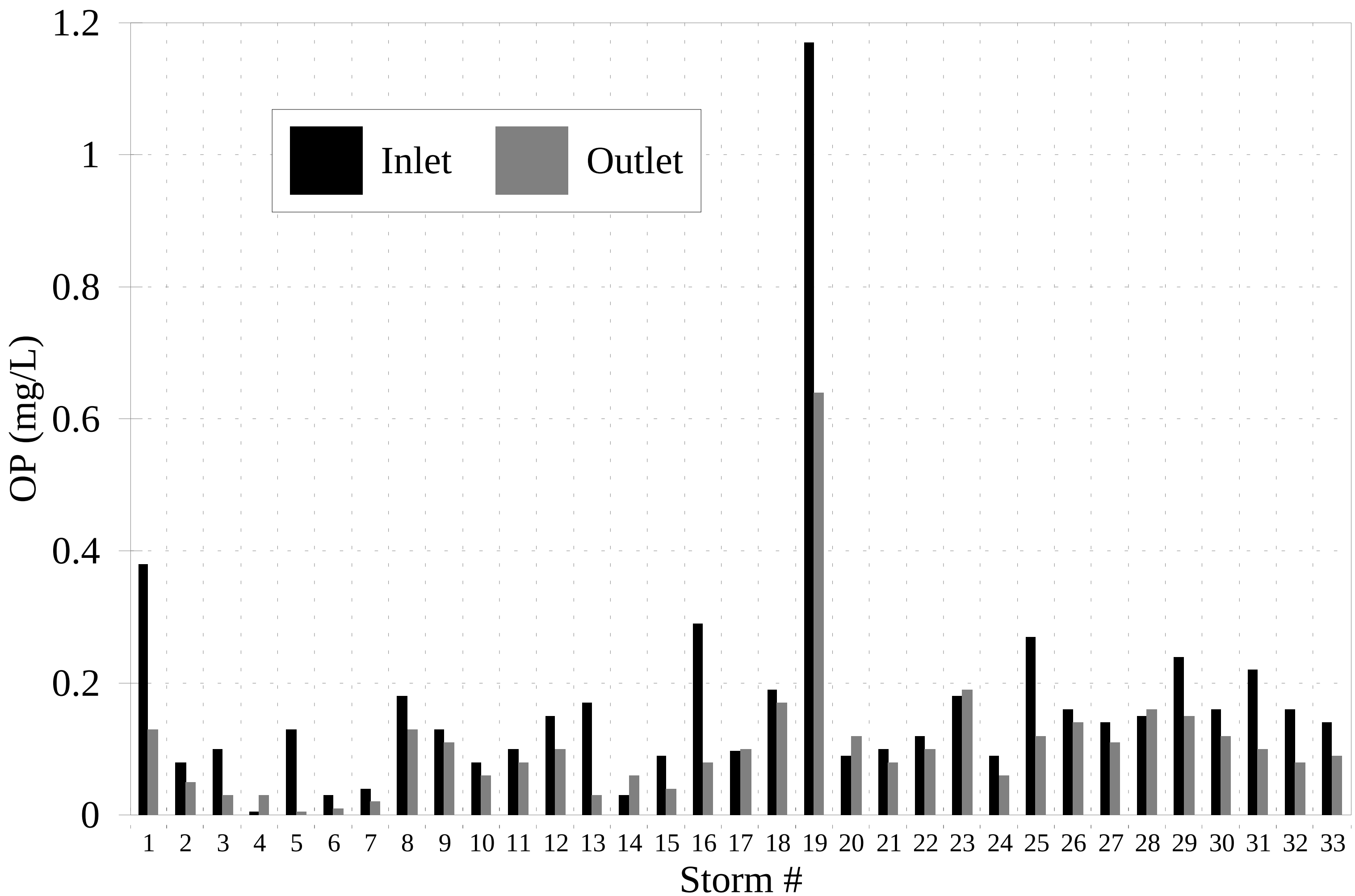

Figure D.3 Inlet-outlet EMC comparison: TP, mg/L

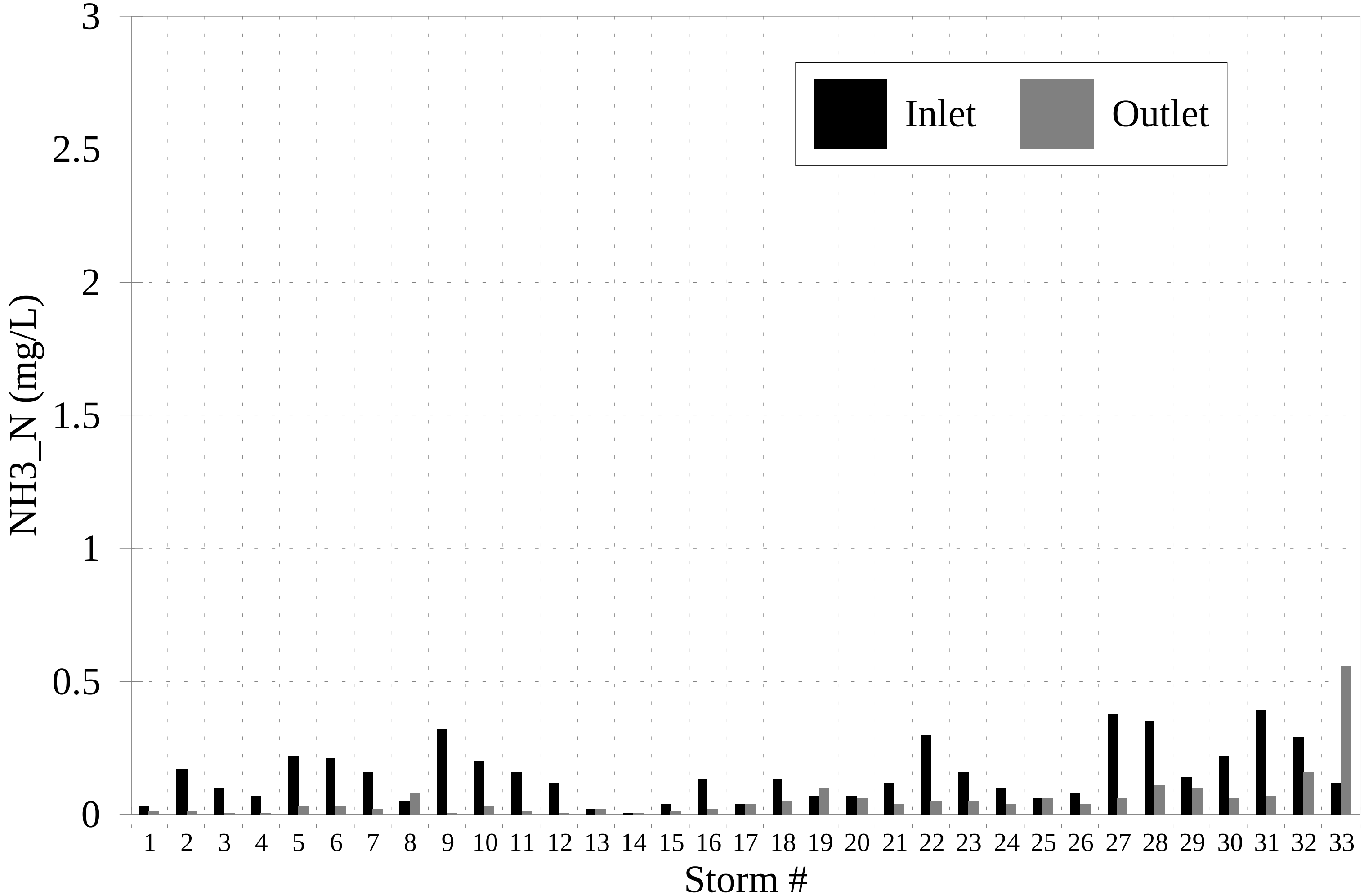

Figure D.4 Inlet-outlet EMC comparison: NH3_N, mg/L

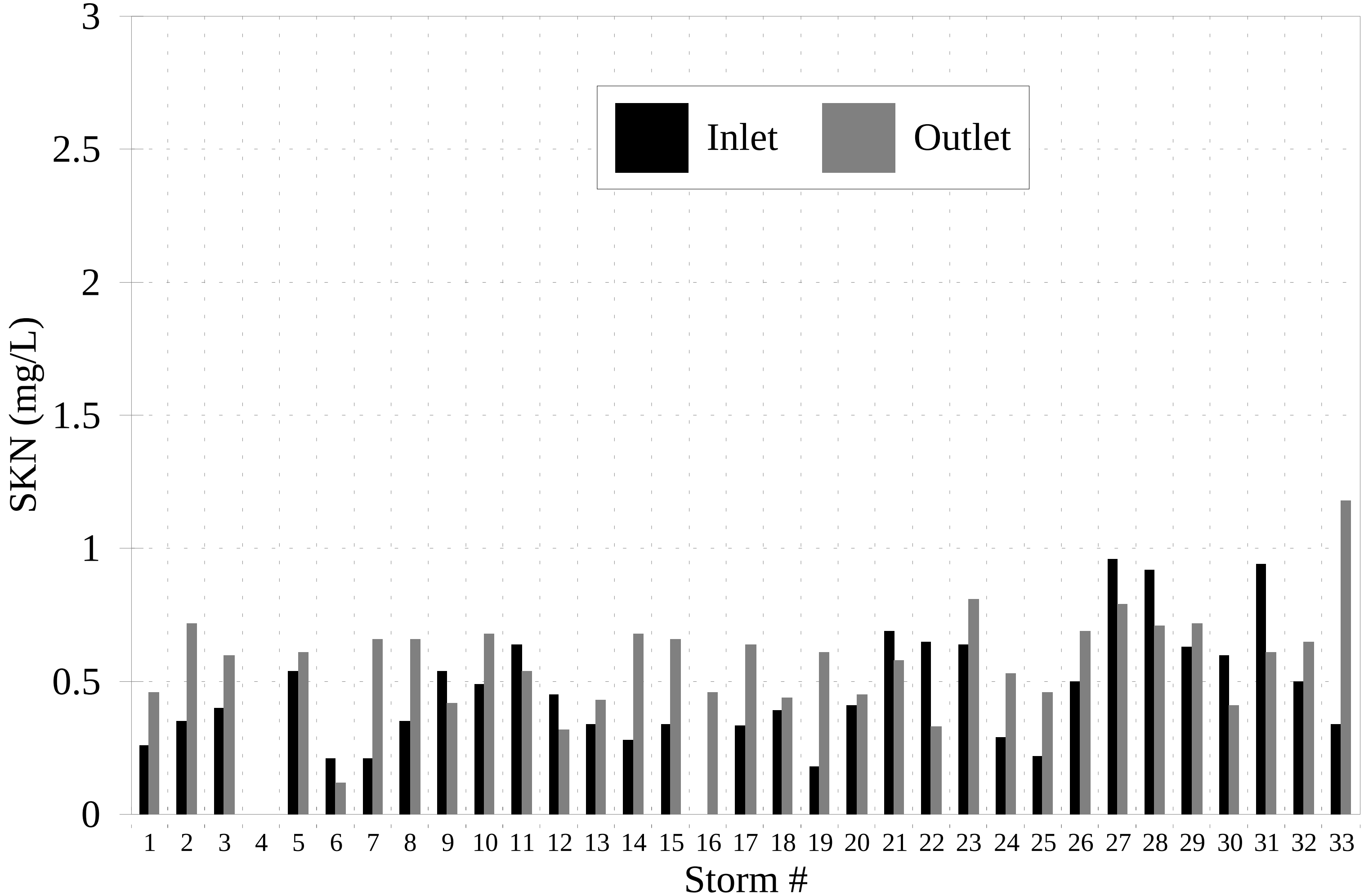

Figure D.5 Inlet-outlet EMC comparison: SKN, mg/L

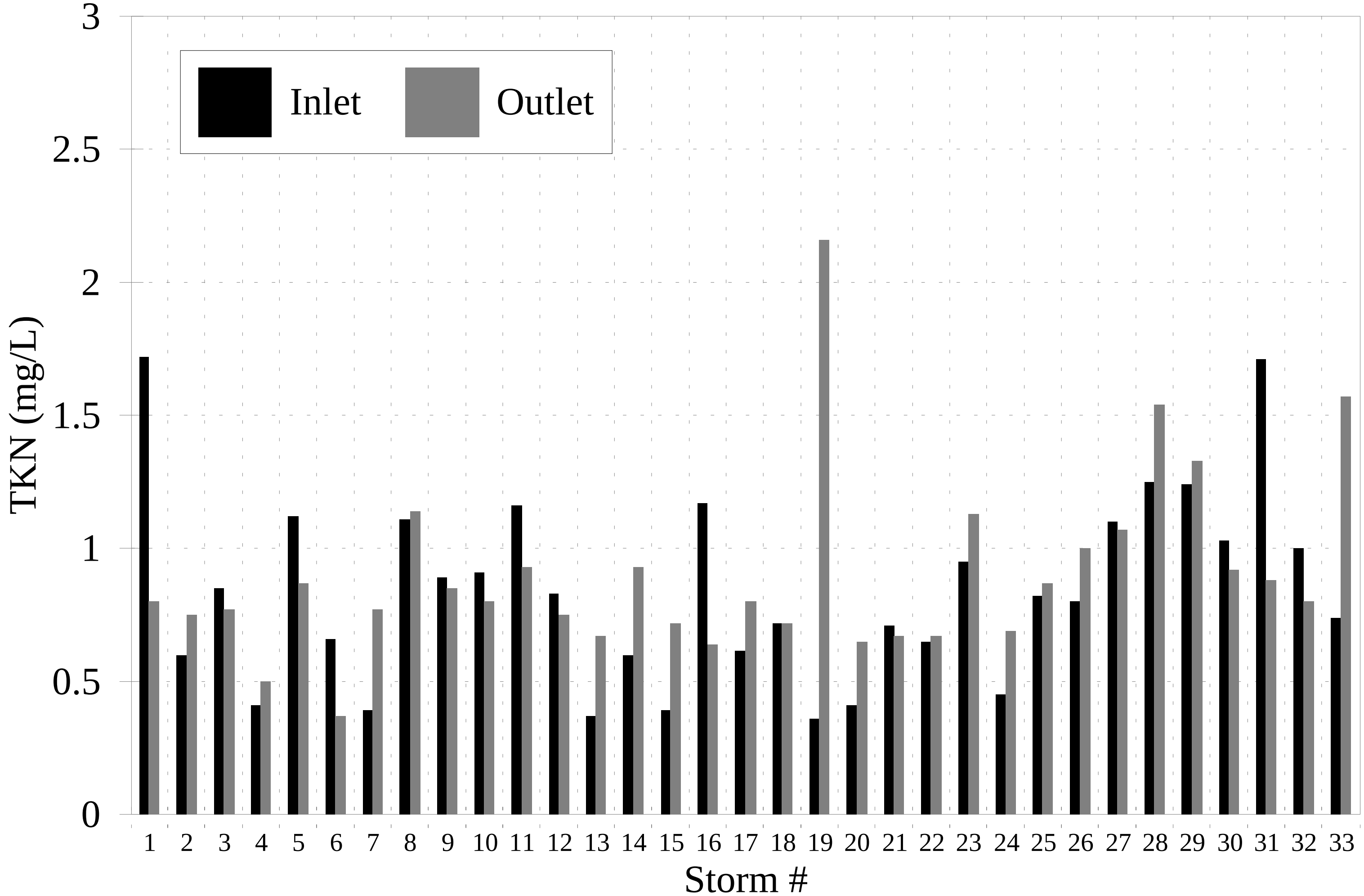

Figure D.6 Inlet-outlet EMC comparison: TKN, mg/L

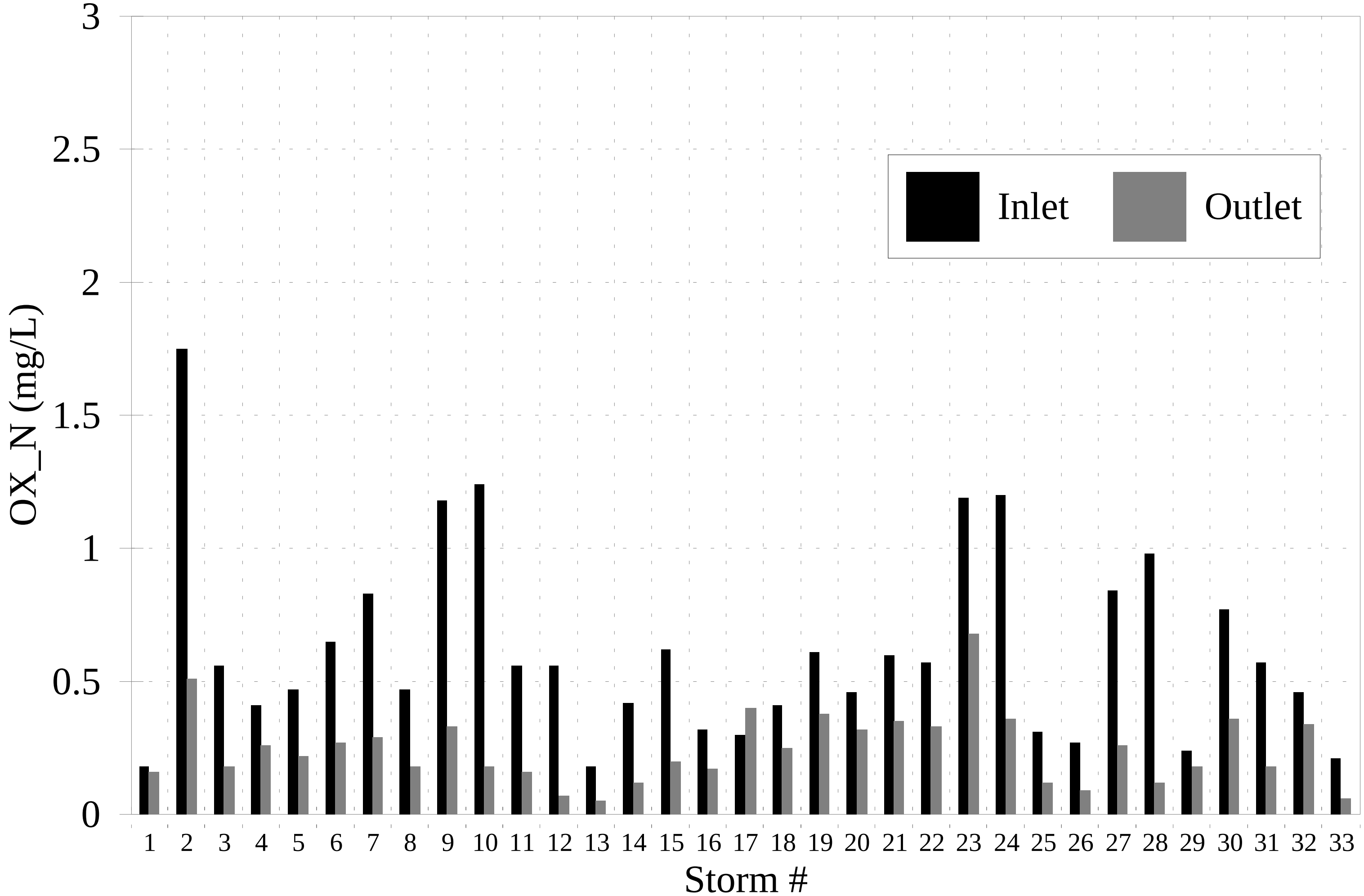

Figure D.7 Inlet-outlet EMC comparison: OX_N, mg/L

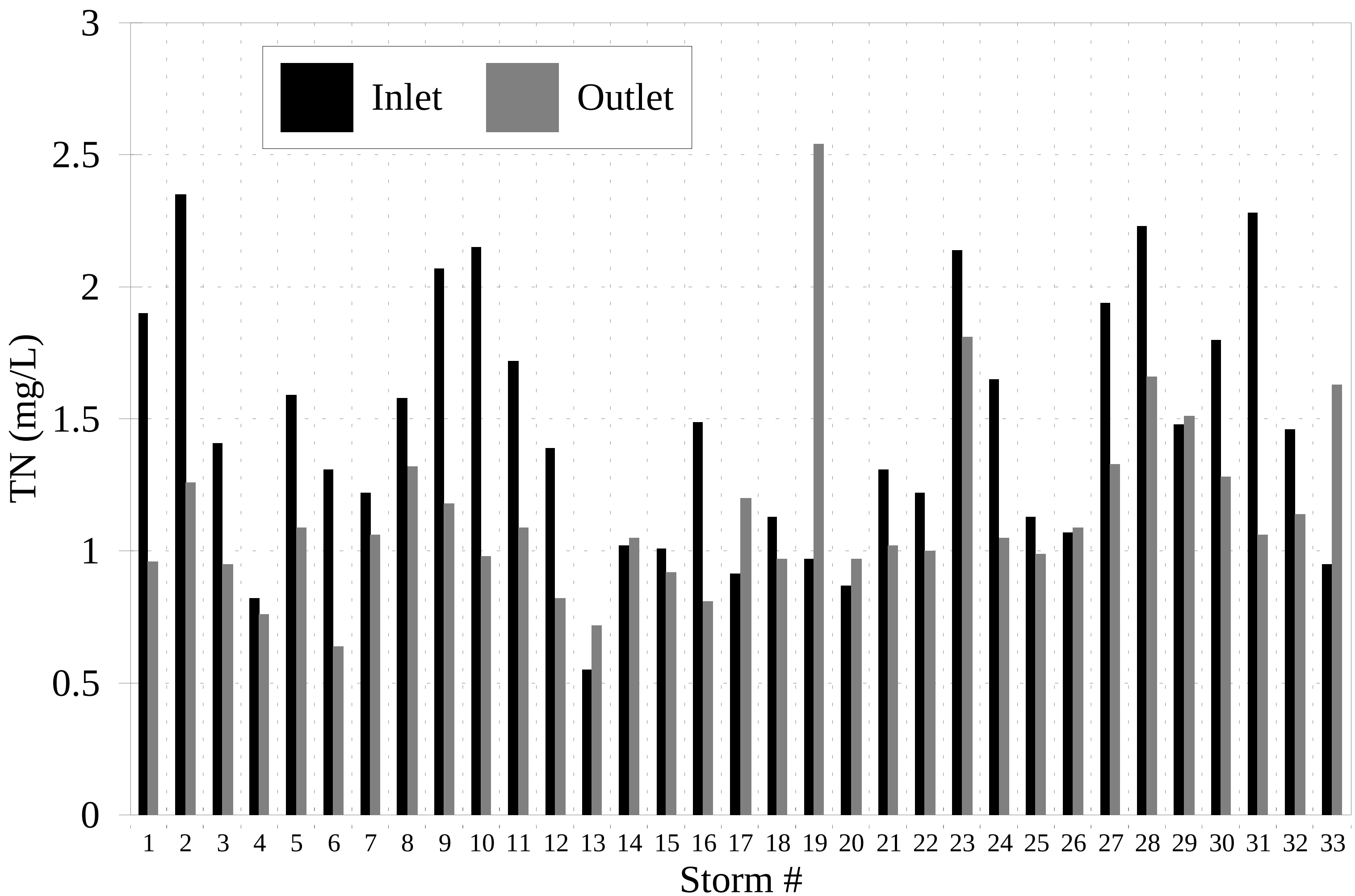

Figure D.8 Inlet-outlet EMC comparison: TN, mg/L

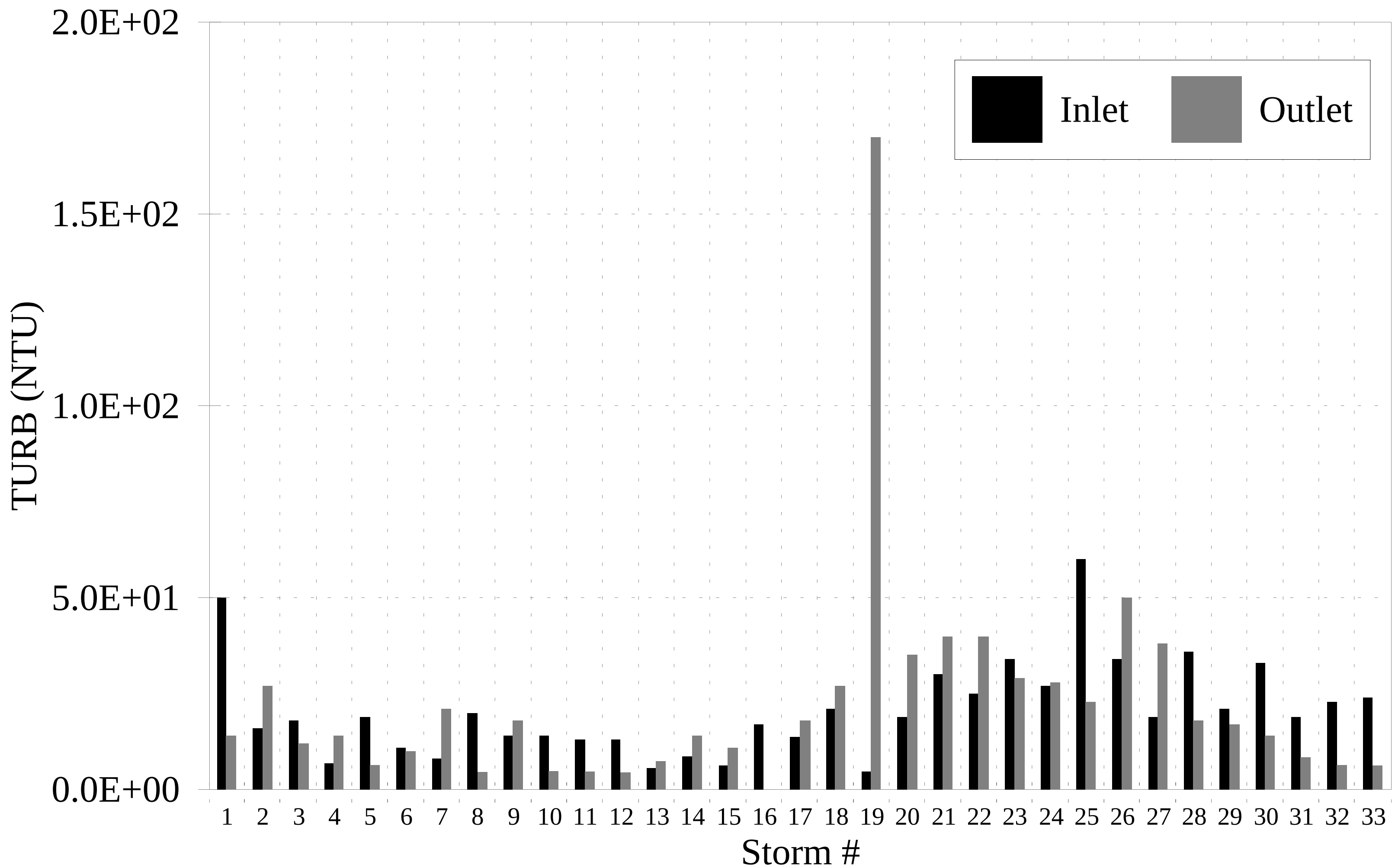

Figure D.9 Inlet-outlet EMC comparison: turbidity, NTU

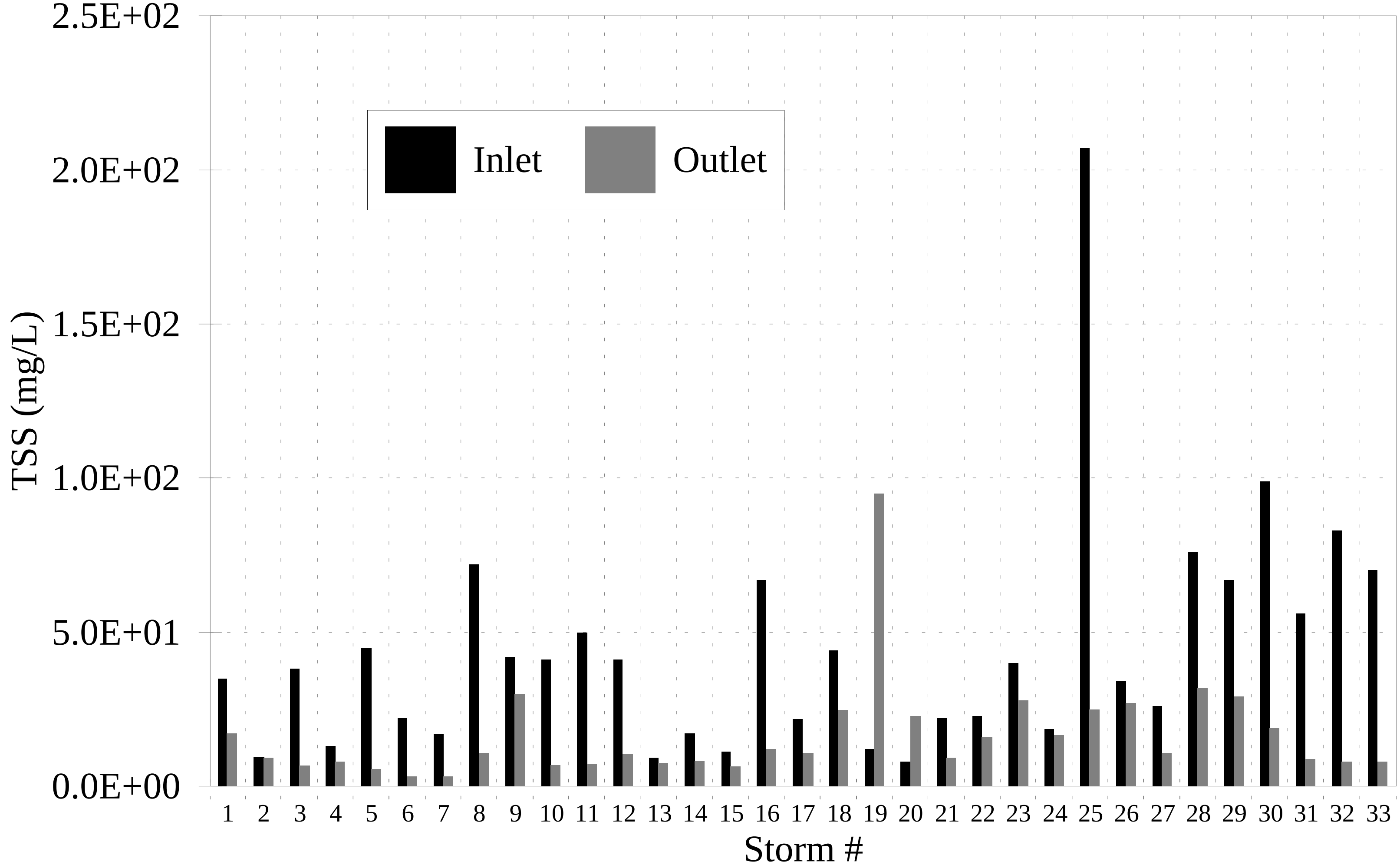

Figure D.10 Inlet-outlet EMC comparison: TSS, mg/L

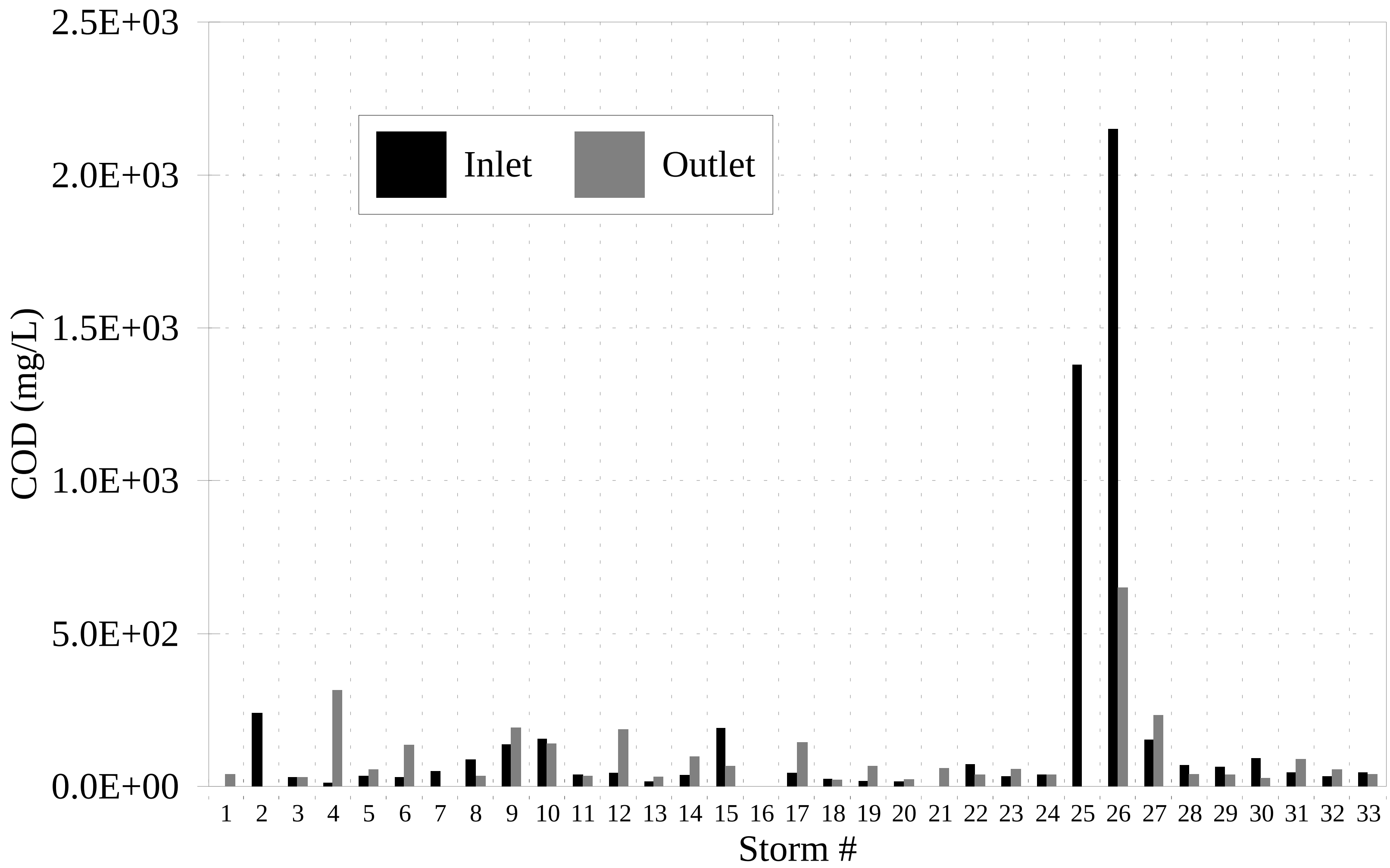

Figure D.11 Inlet-outlet EMC comparison: COD, mg/L

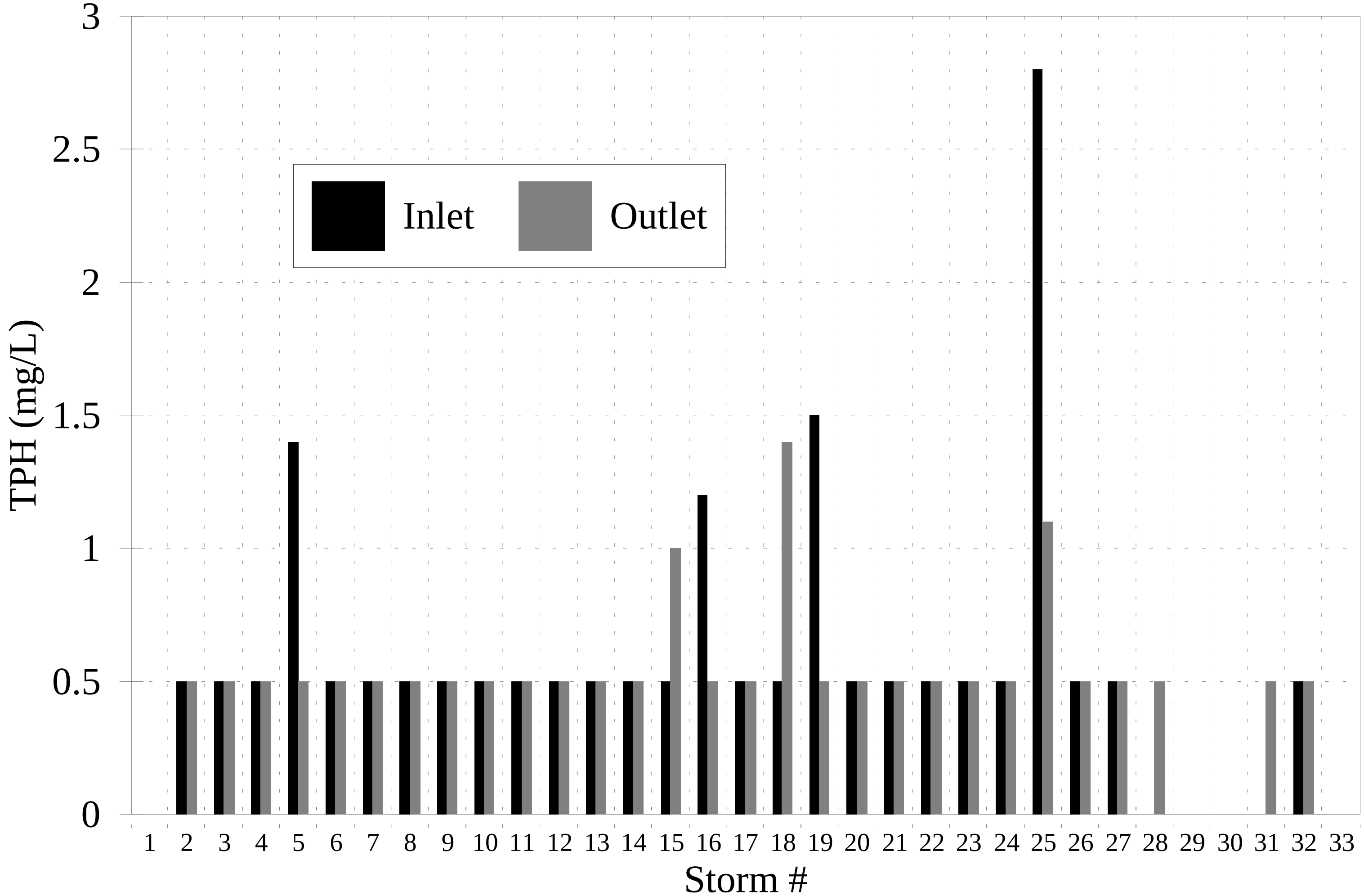

Figure D.12 Inlet-outlet EMC comparison: TPH, mg/L

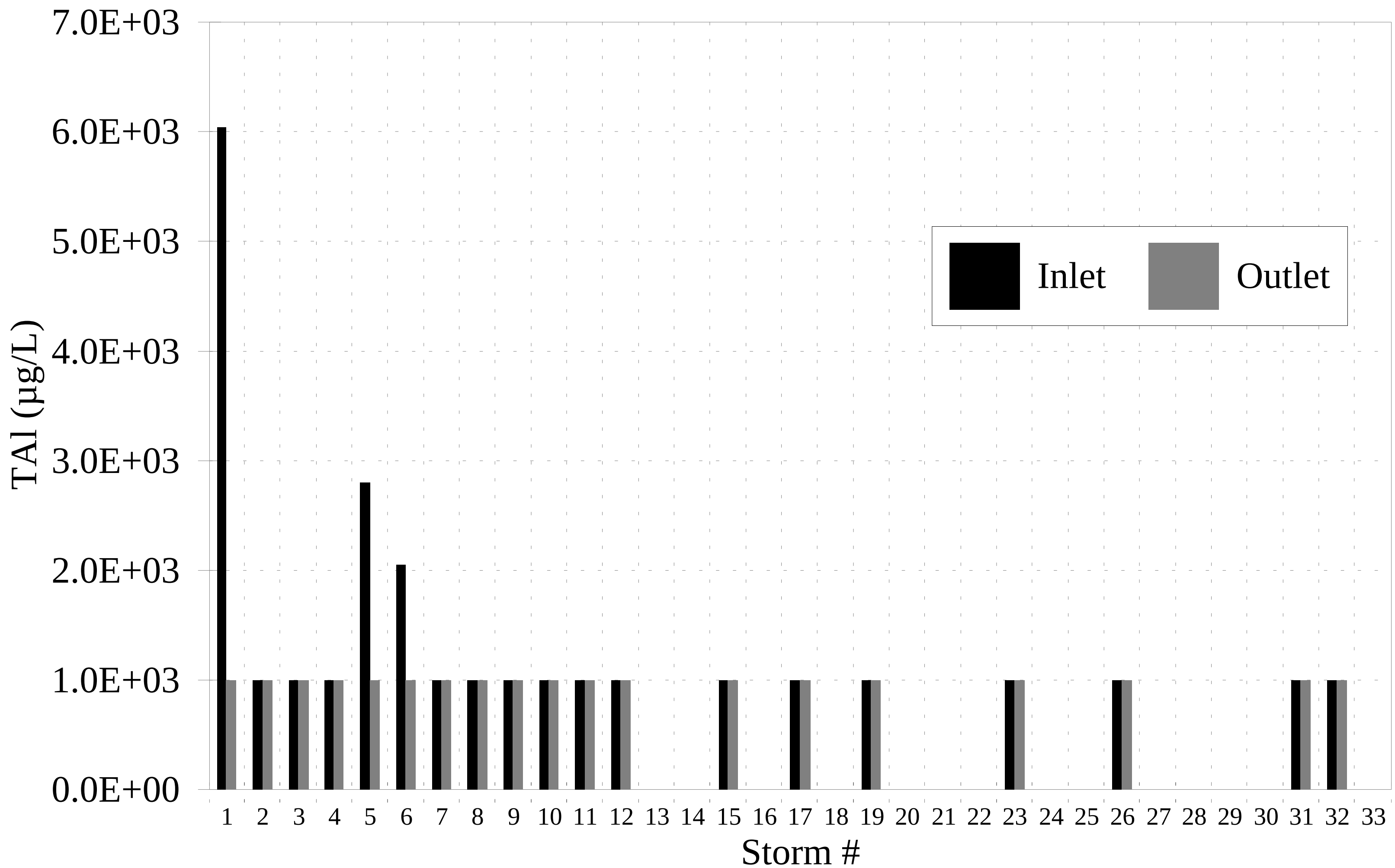

Figure D.13 Inlet-outlet EMC comparison: TAl, µg/L

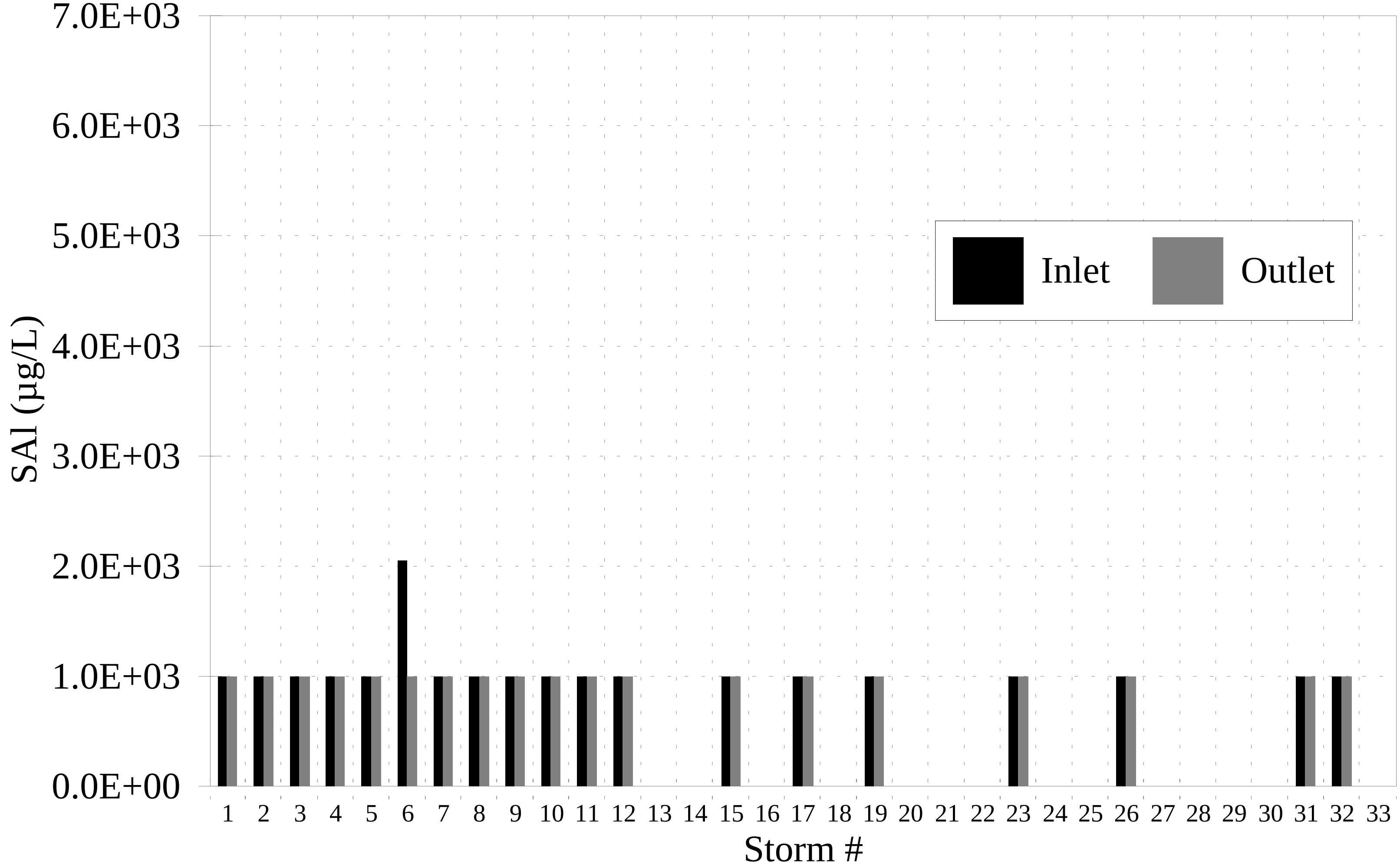

Figure D.14 Inlet-outlet EMC comparison: SAl, µg/L

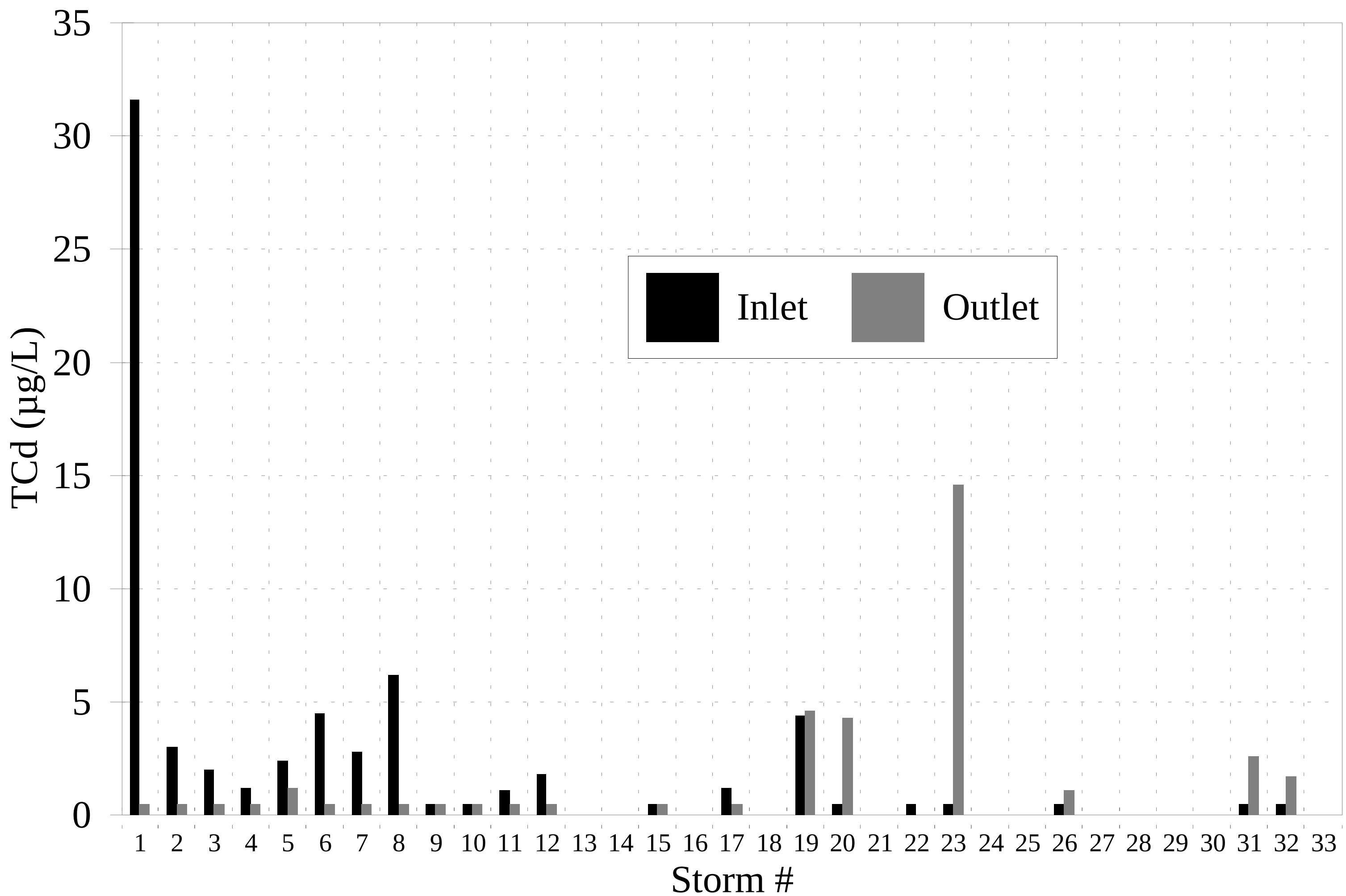

Figure D.15 Inlet-outlet EMC comparison: TCd, µg/L

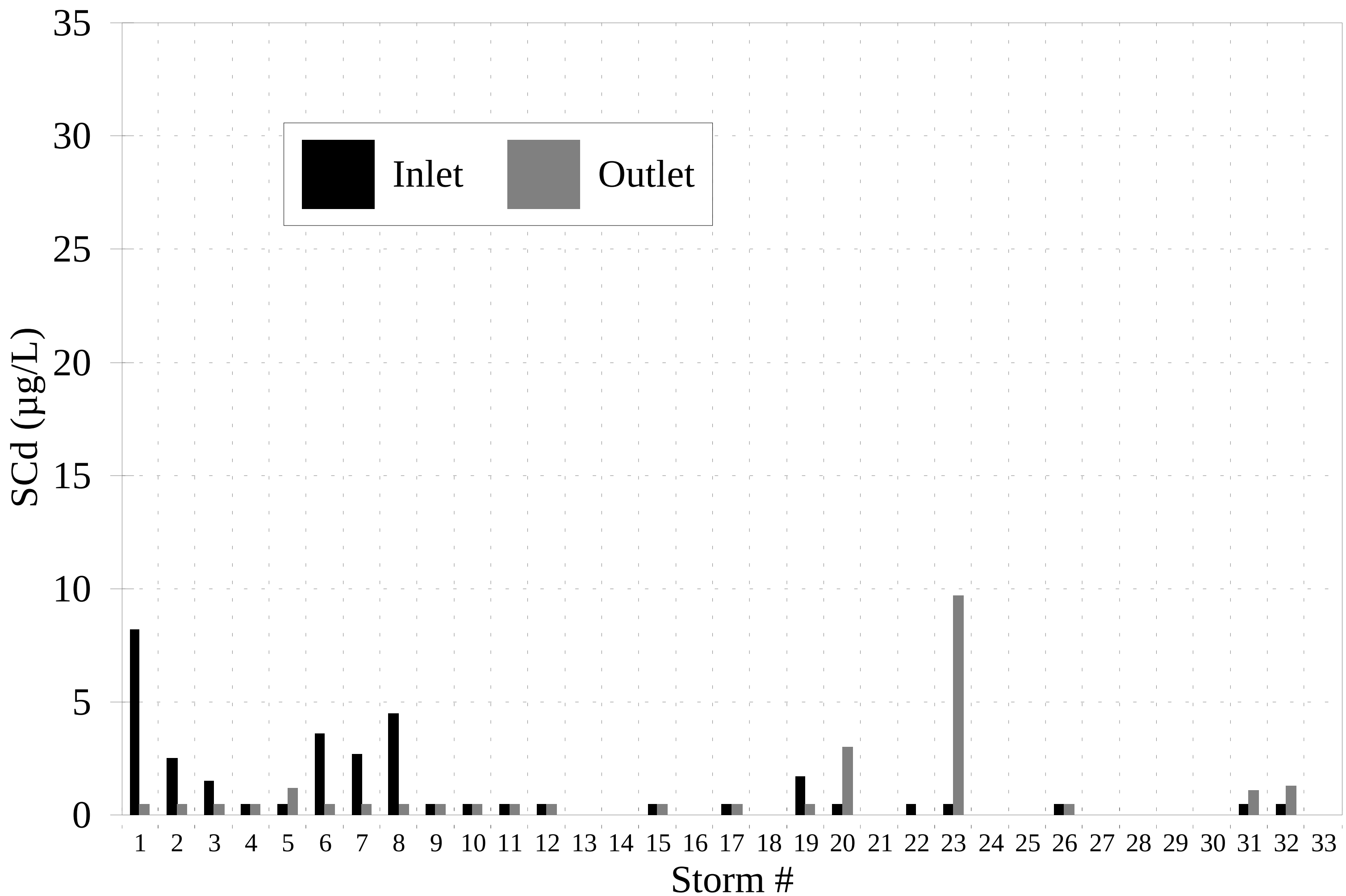

Figure D.16 Inlet-outlet EMC comparison: SCd, µg/L

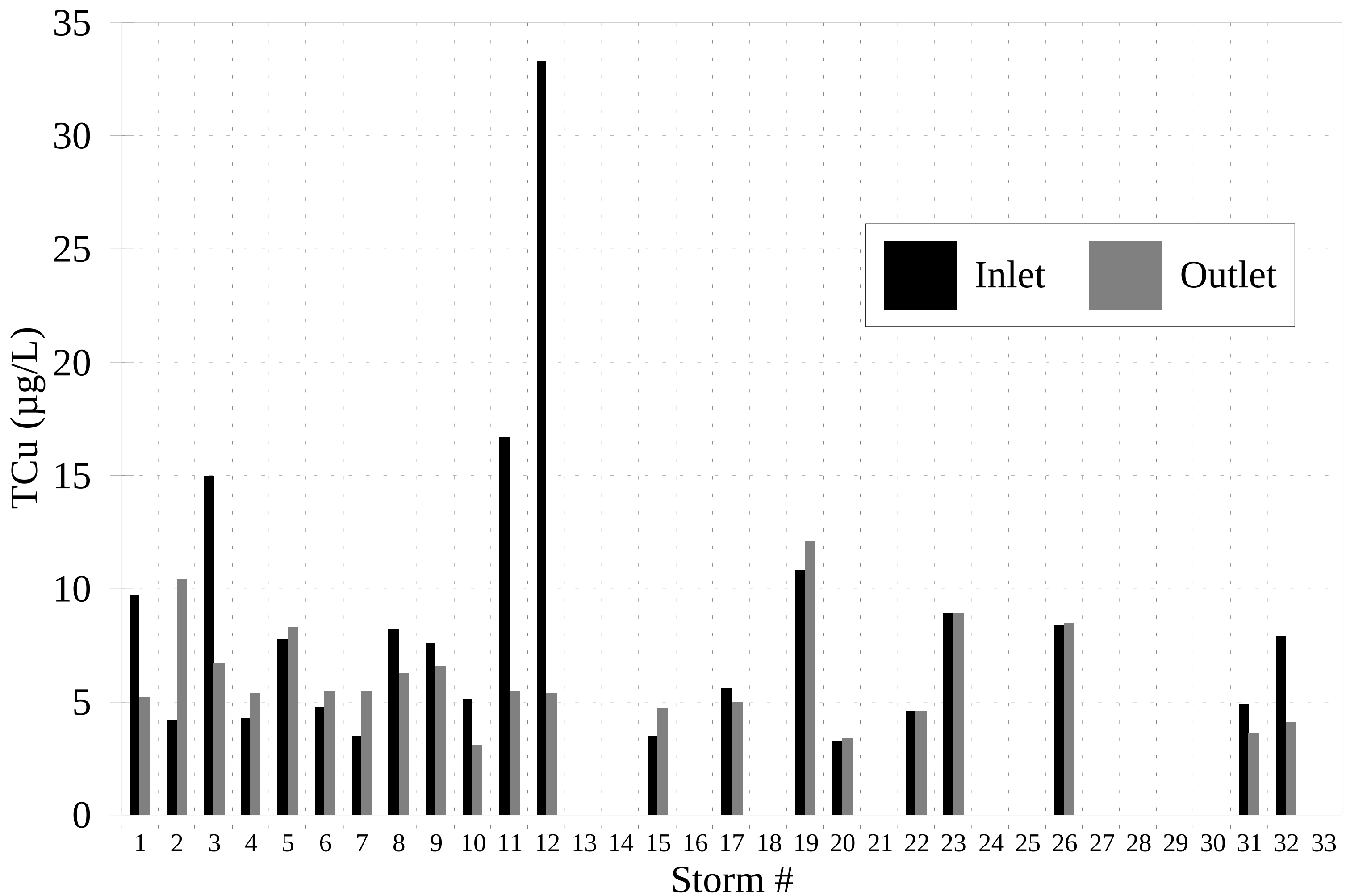

Figure D.17 Inlet-outlet EMC comparison: TCu, µg/L

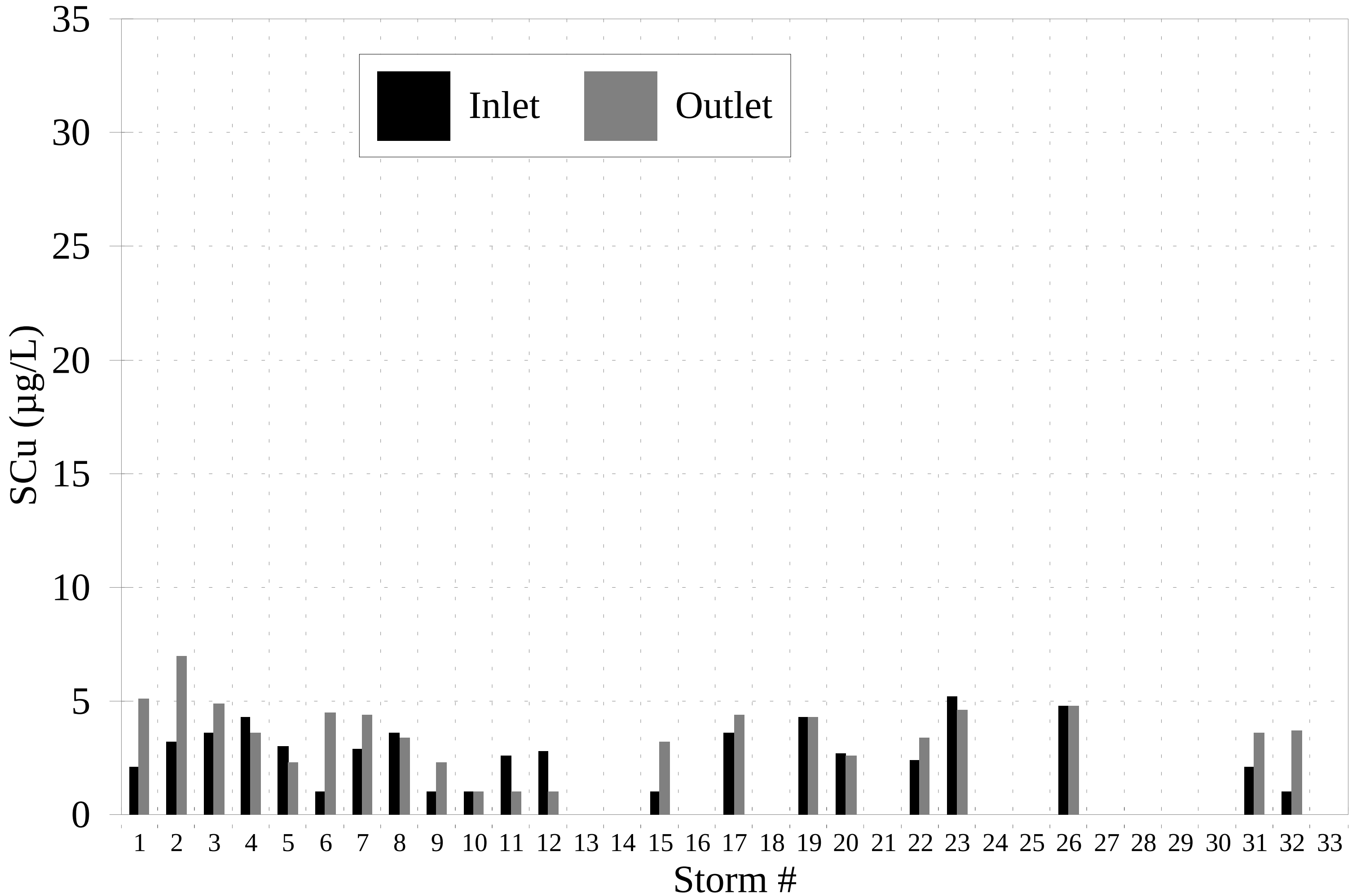

Figure D.18 Inlet-outlet EMC comparison: SCu, µg/L

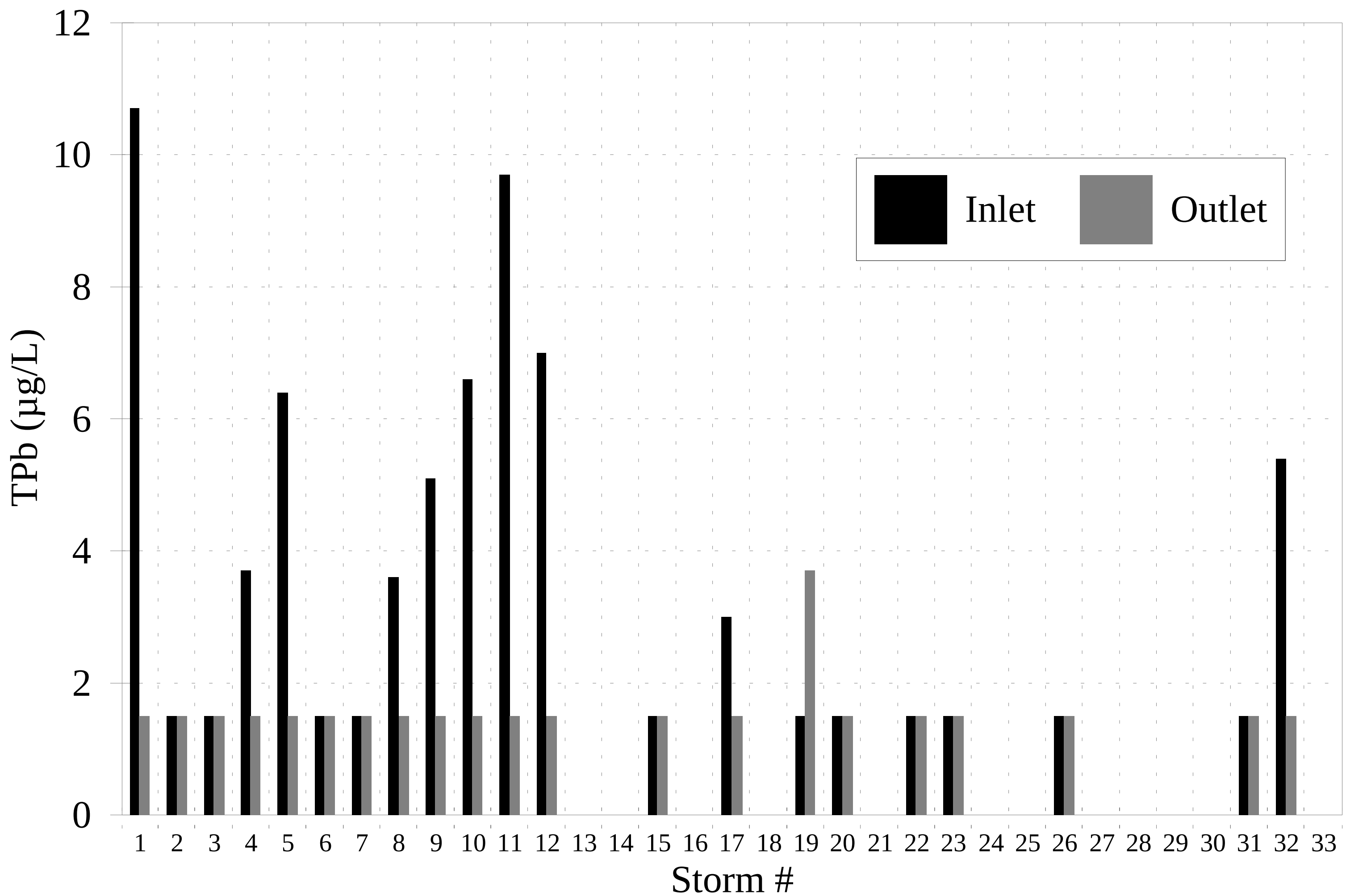

Figure D.19 Inlet-outlet EMC comparison: TPb, µg/L

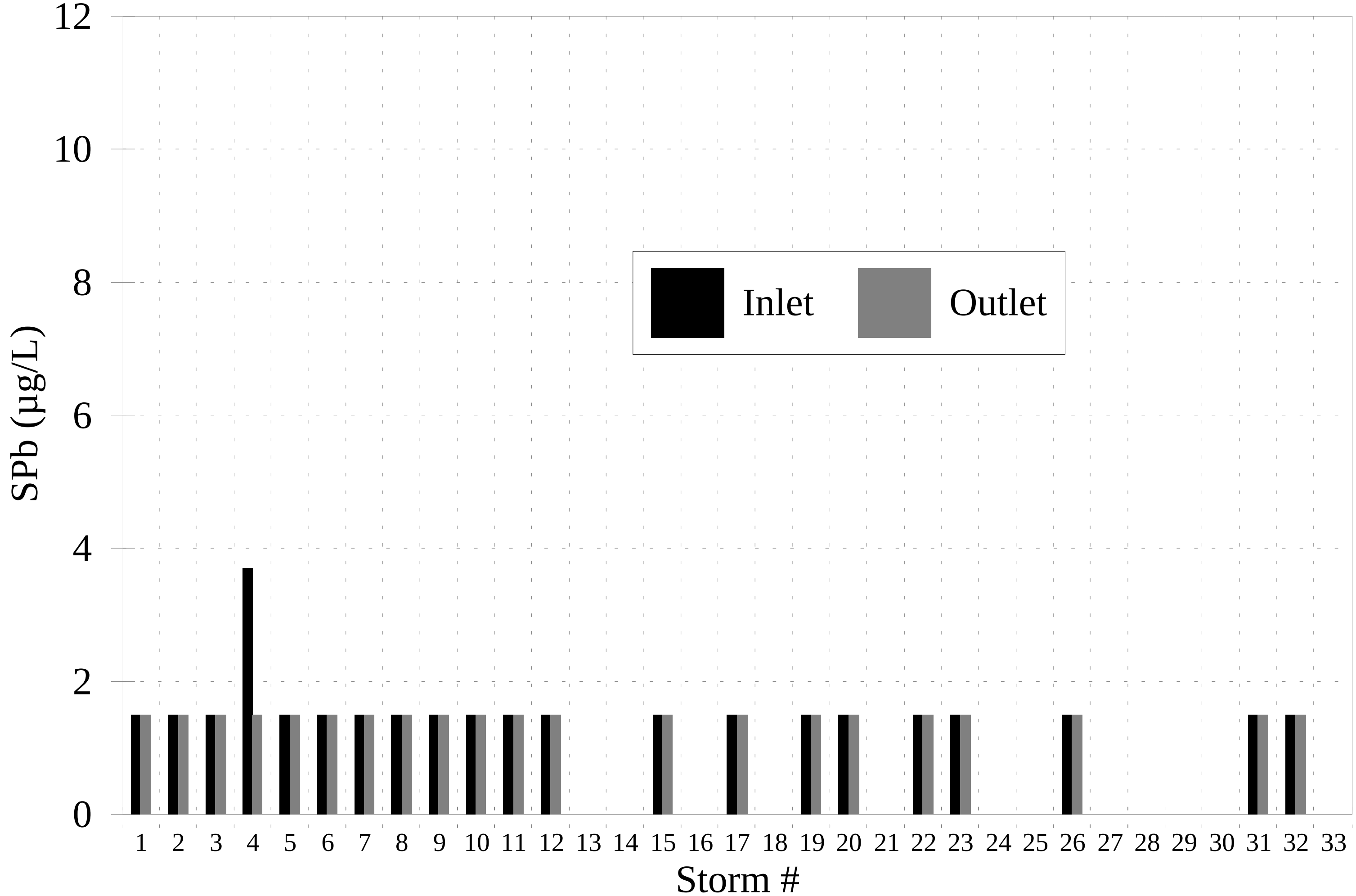

Figure D.20 Inlet-outlet EMC comparison: SPb, µg/L

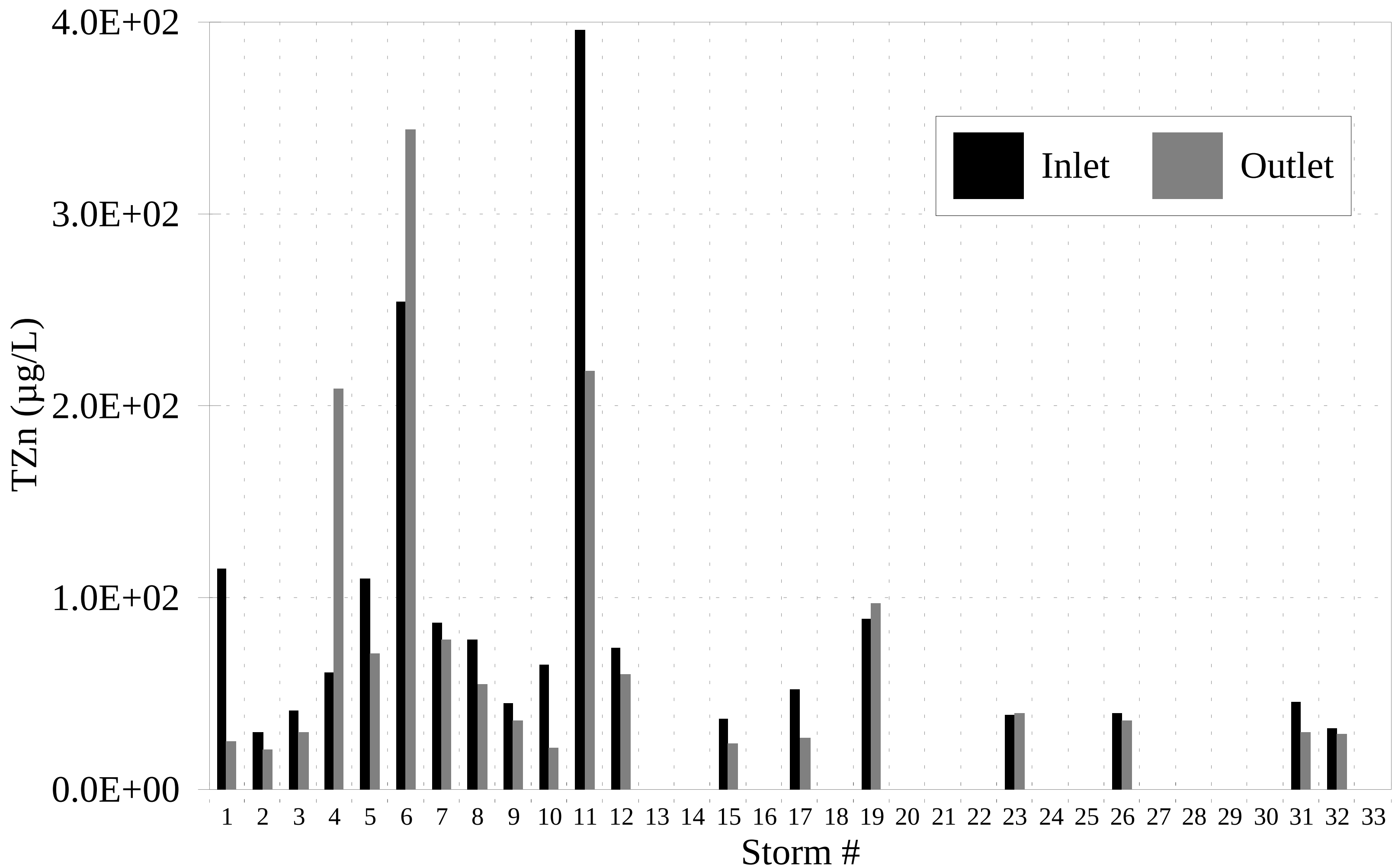

Figure D.21 Inlet-outlet EMC comparison: TZn, µg/L

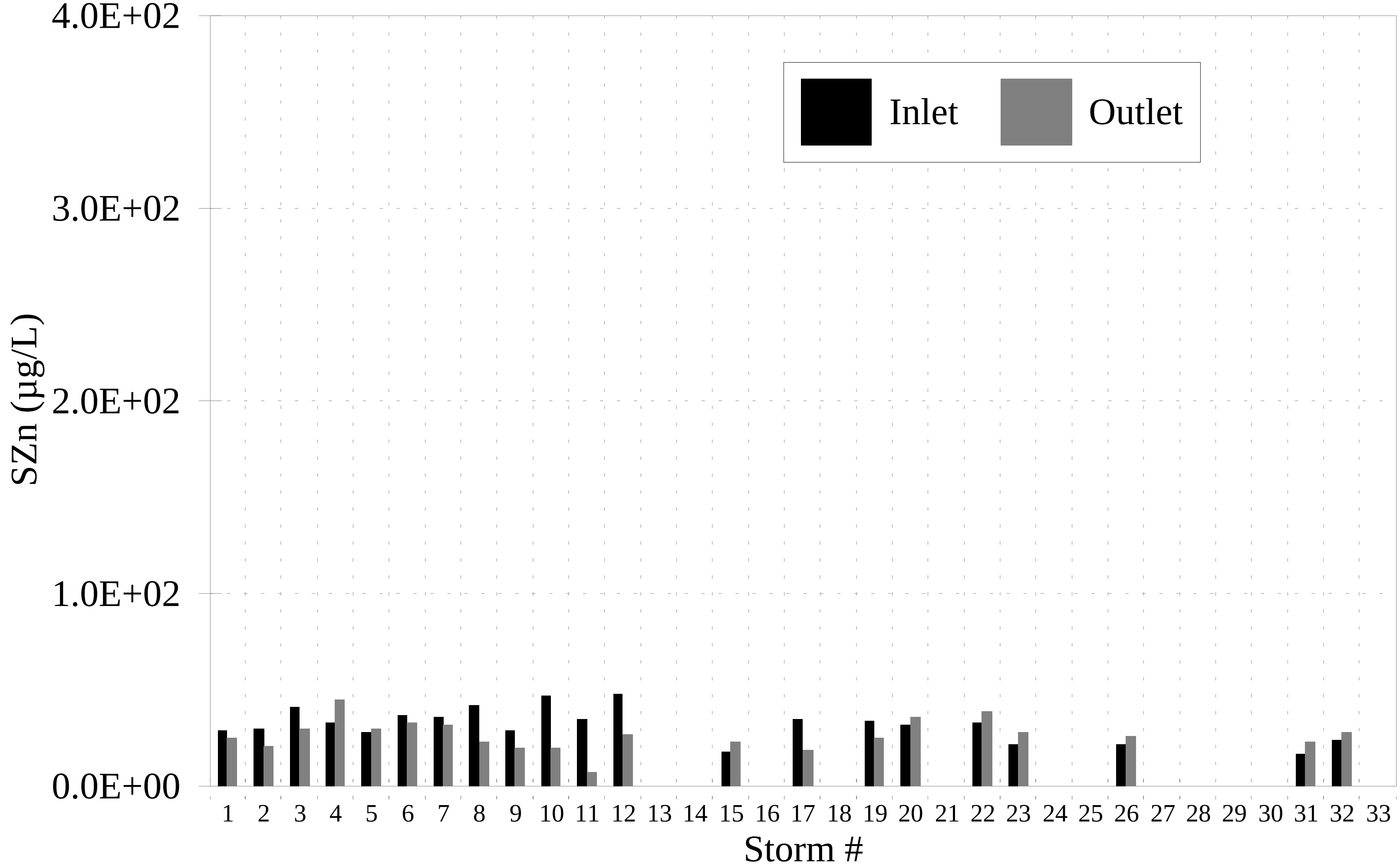

Figure D.22 Inlet-outlet EMC comparison: SZn, µg/L

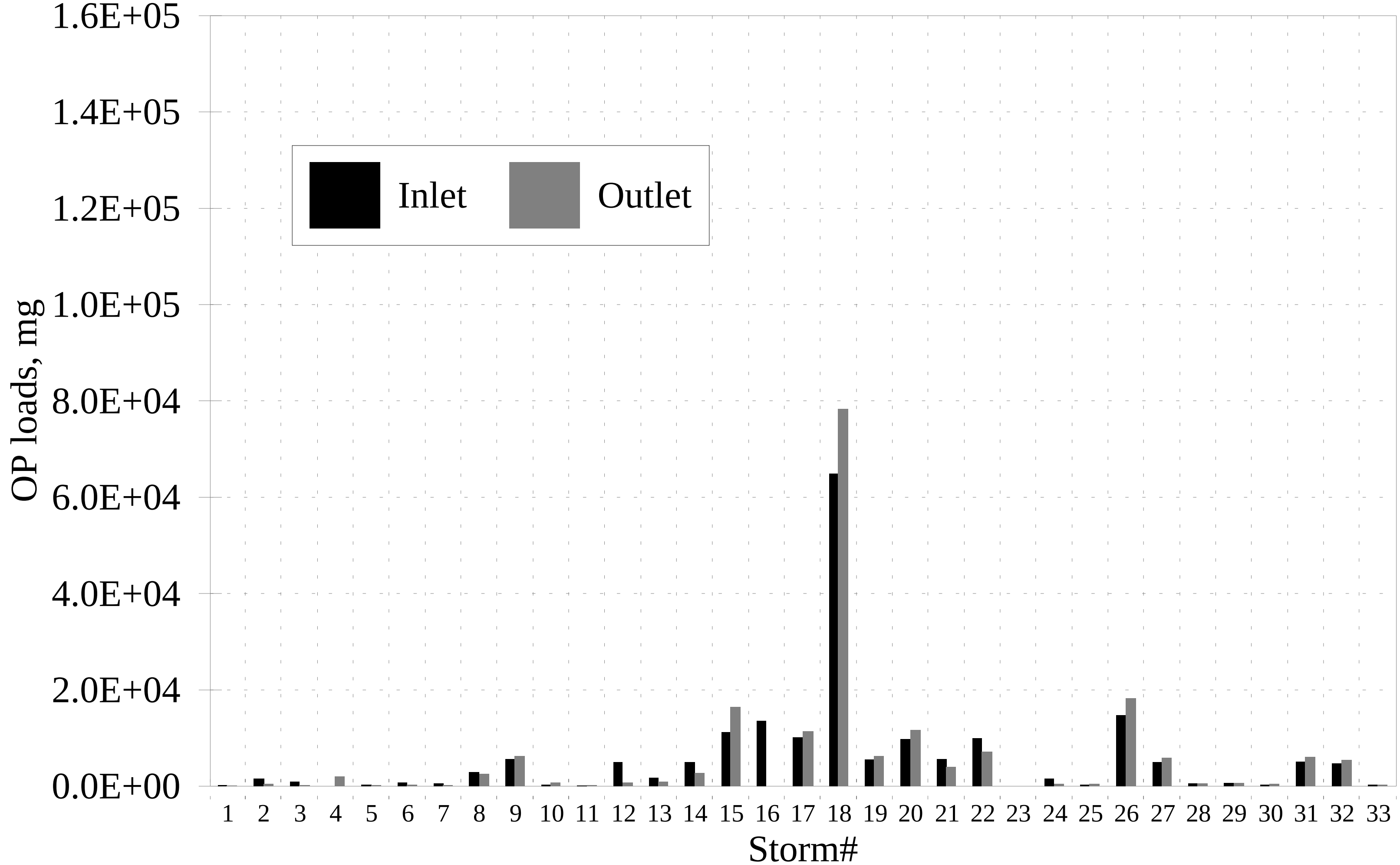

Figure D.23 Inlet-outlet load comparison: OP, mg

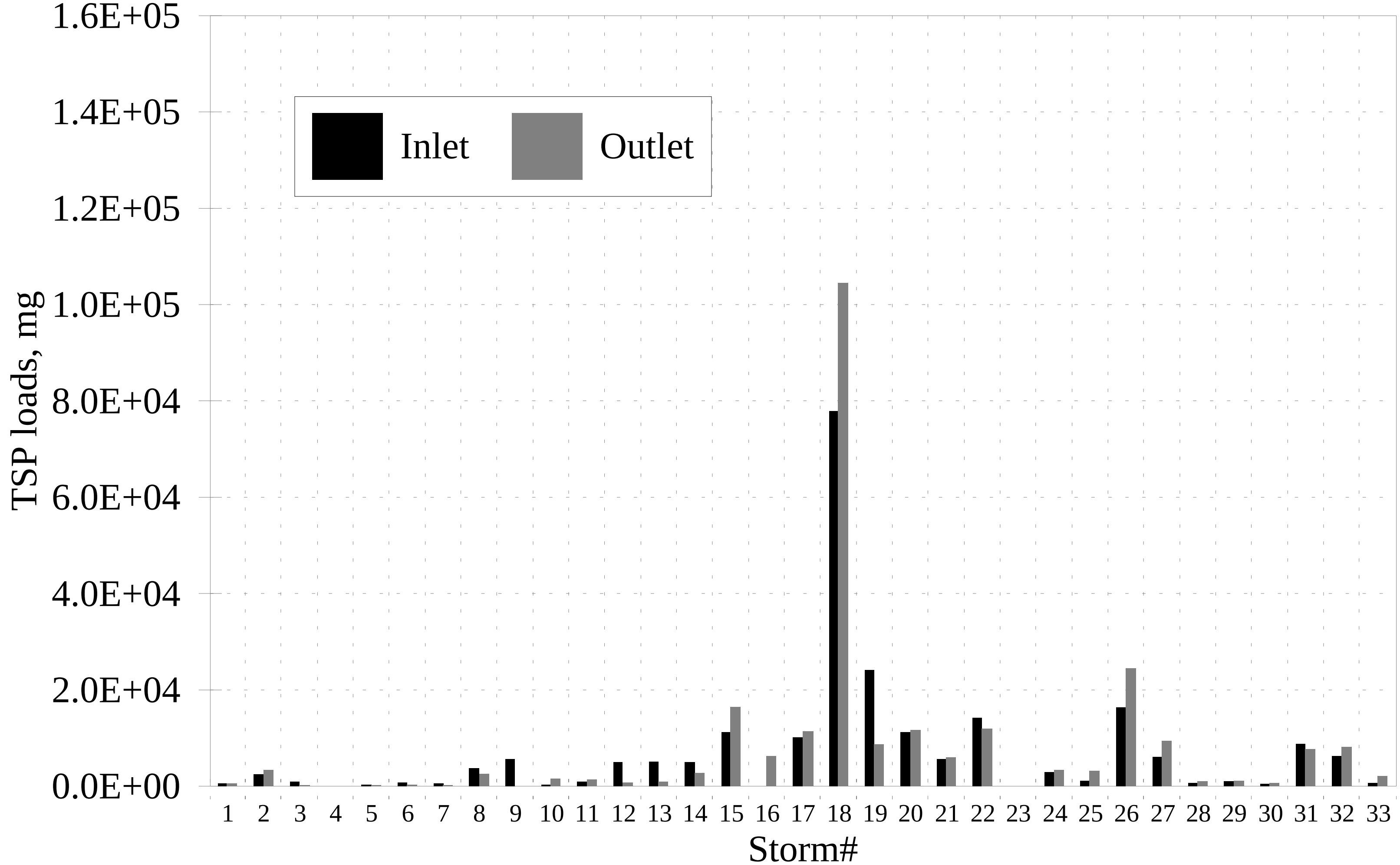

Figure D.24 Inlet-outlet load comparison: TSP, mg

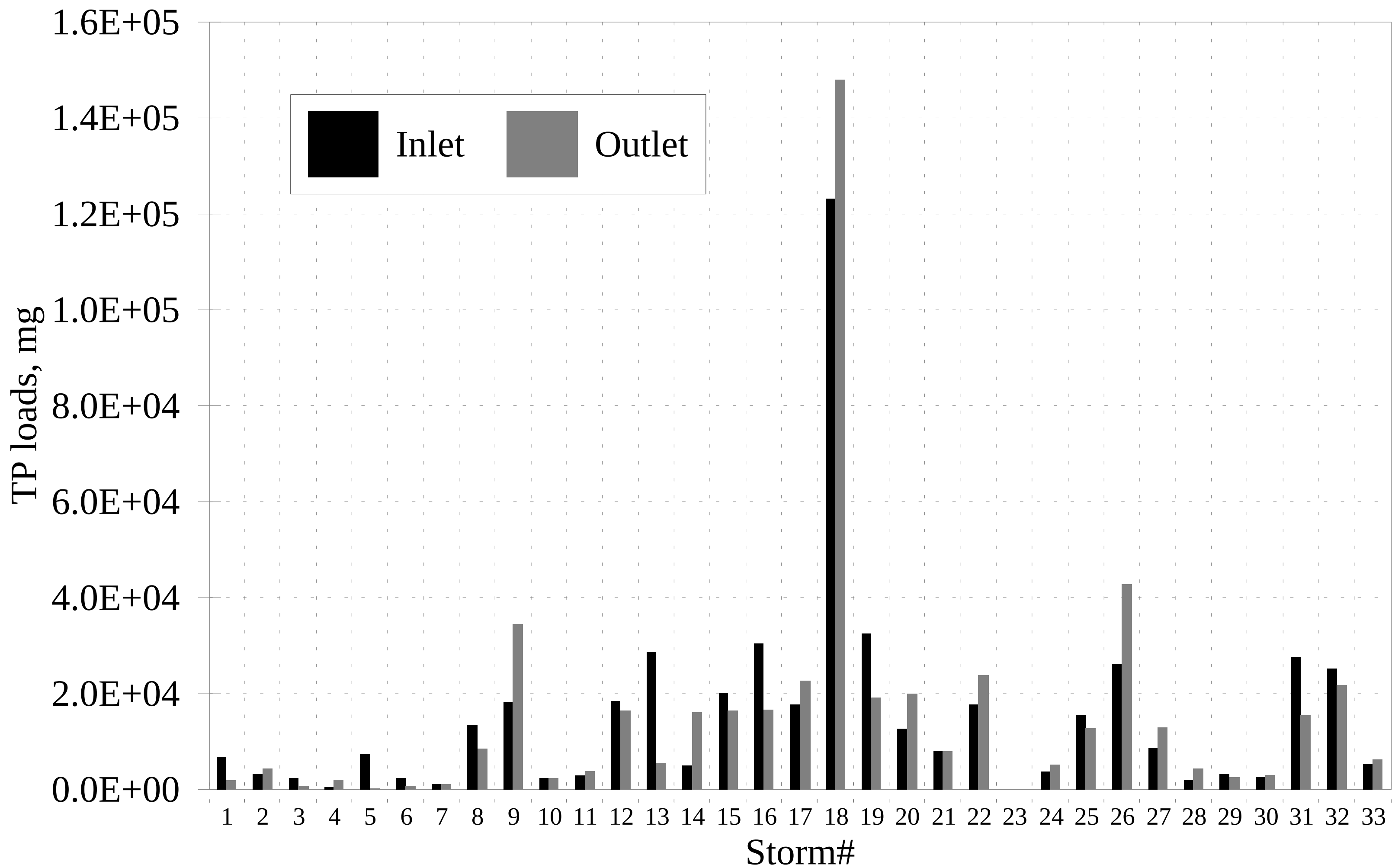

Figure D.25 Inlet-outlet load comparison: TP, mg

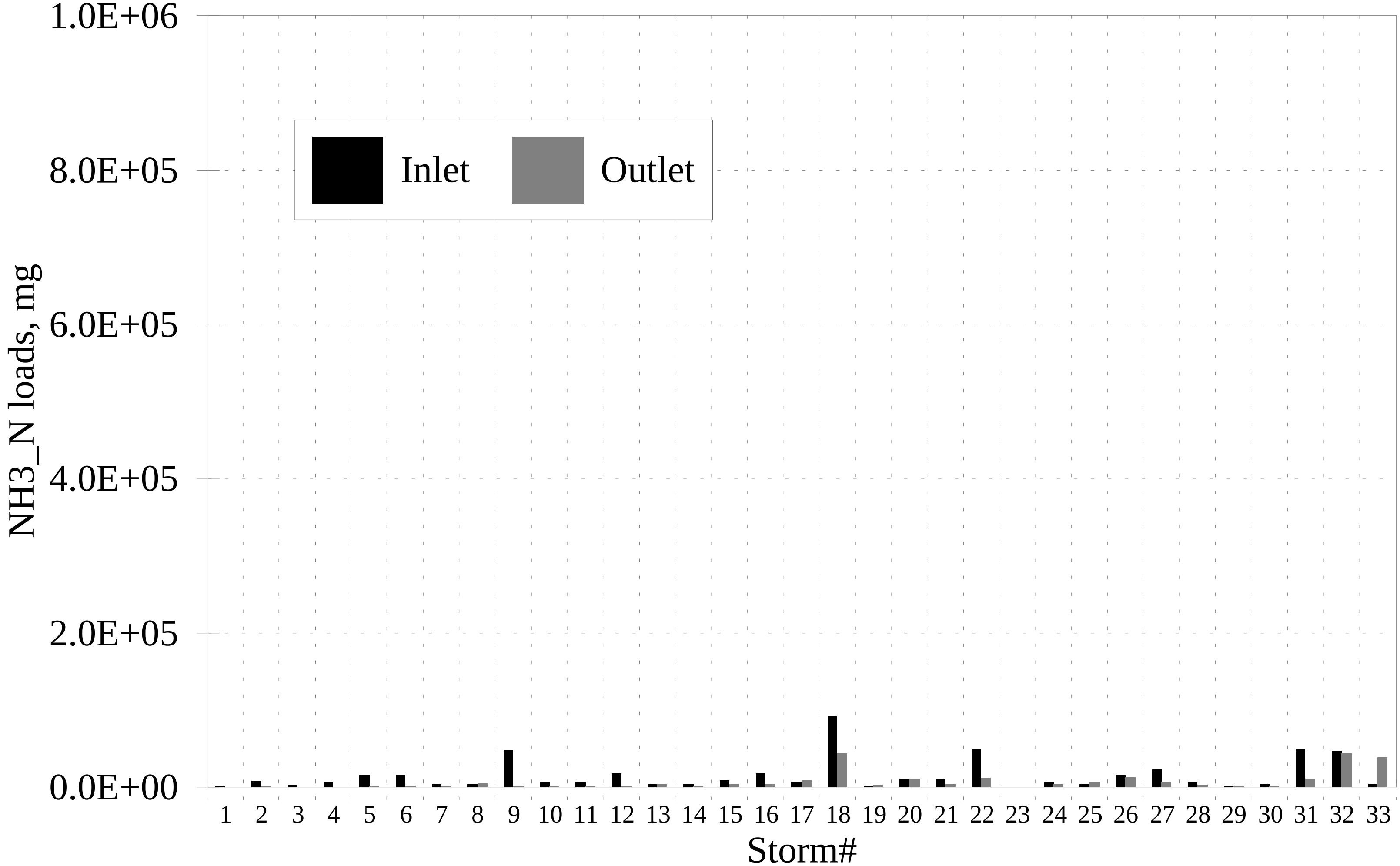

Figure D.26 Inlet-outlet load comparison: NH3_N, mg

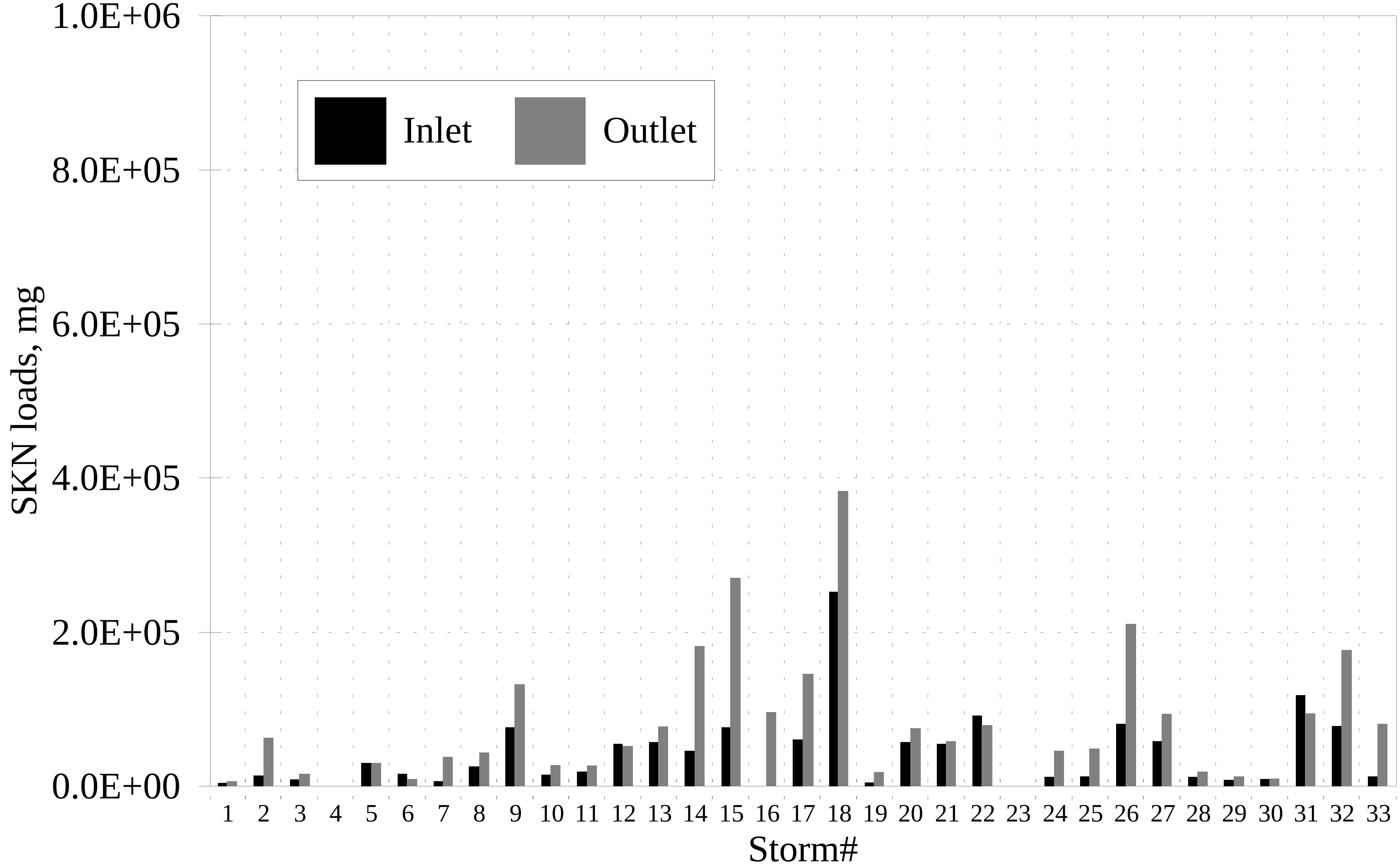

Figure D.27 Inlet-outlet load comparison: SKN, mg

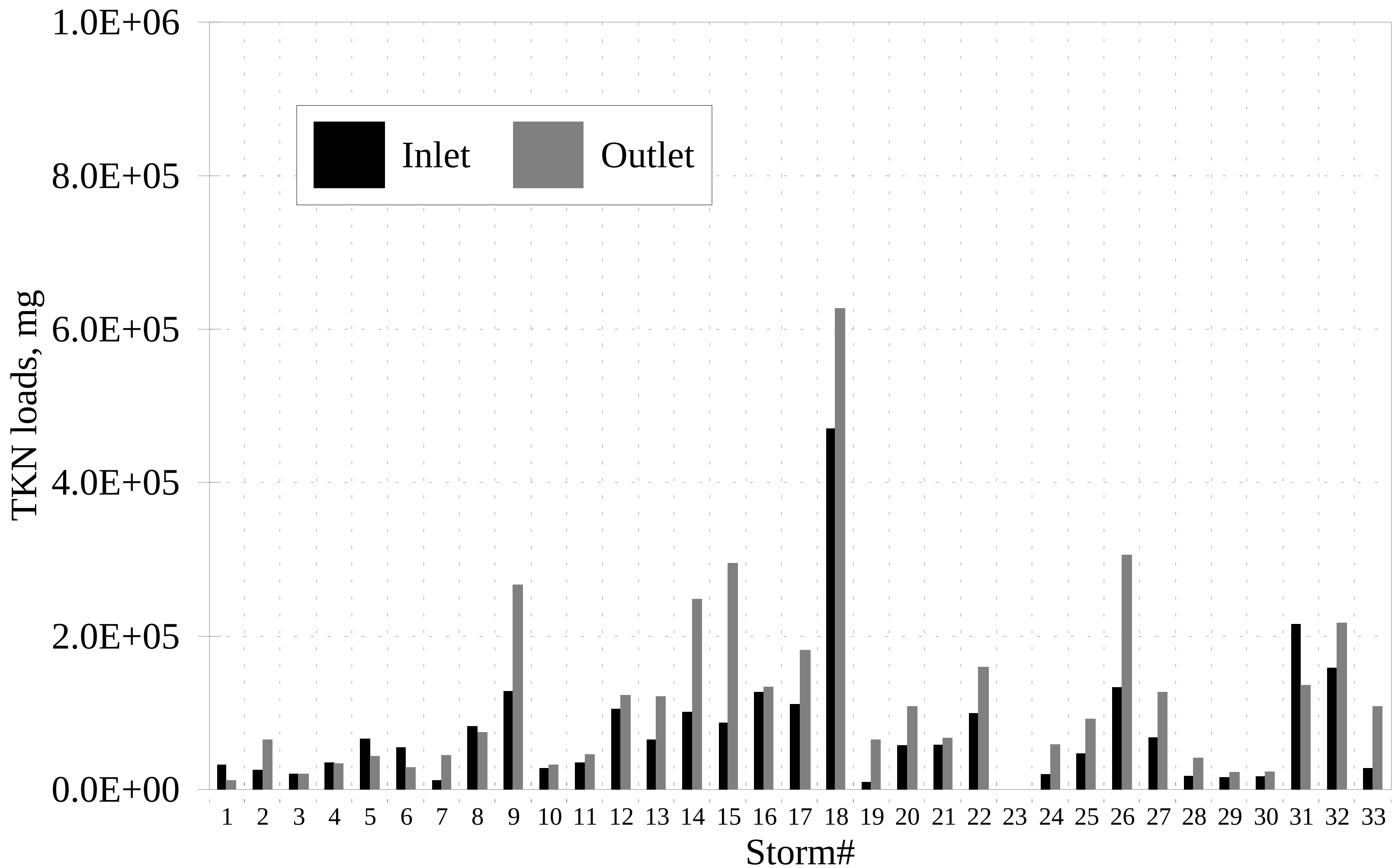

Figure D.28 Inlet-outlet load comparison: TKN, mg

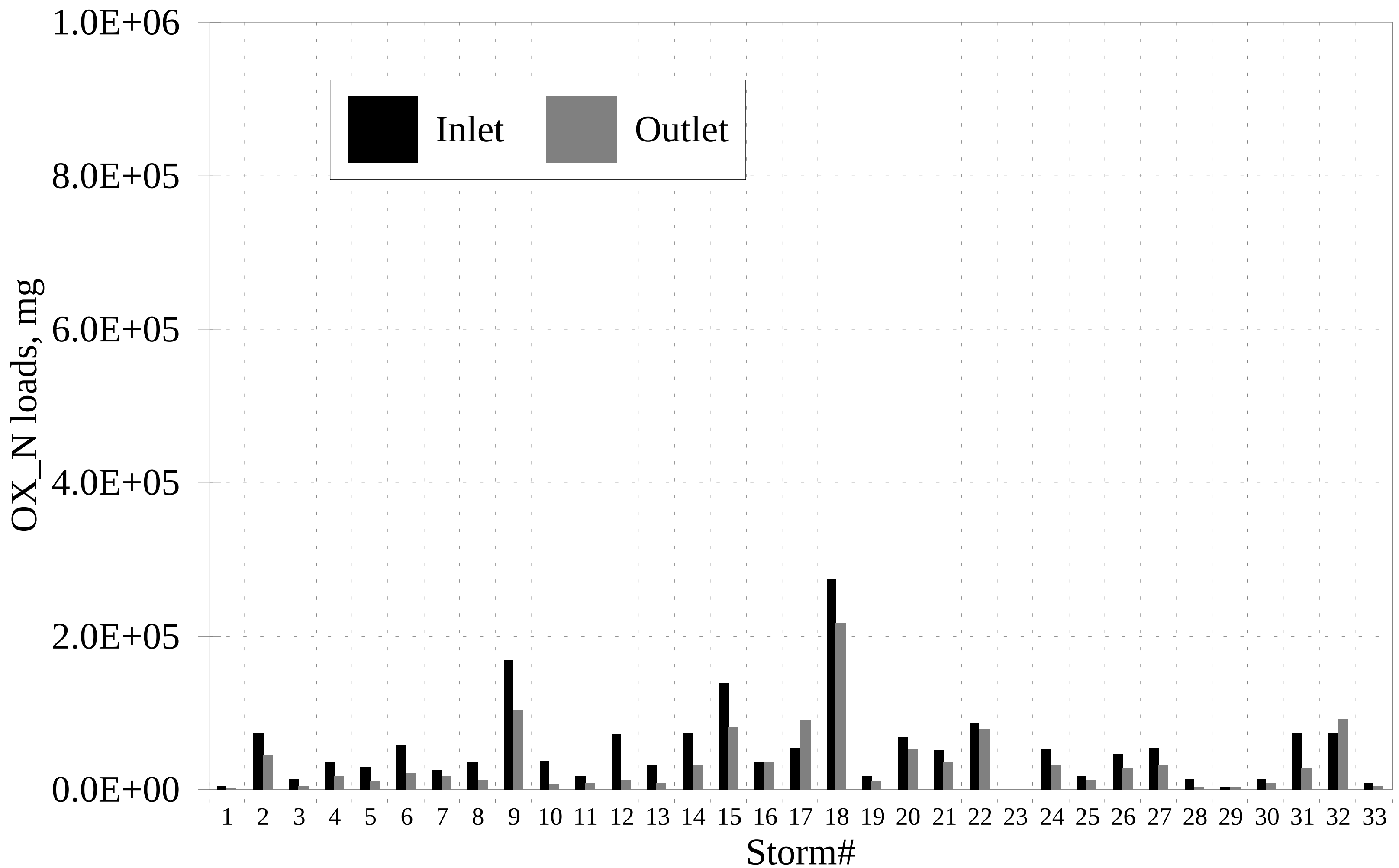

Figure D.29 Inlet-outlet load comparison: OX_N, mg

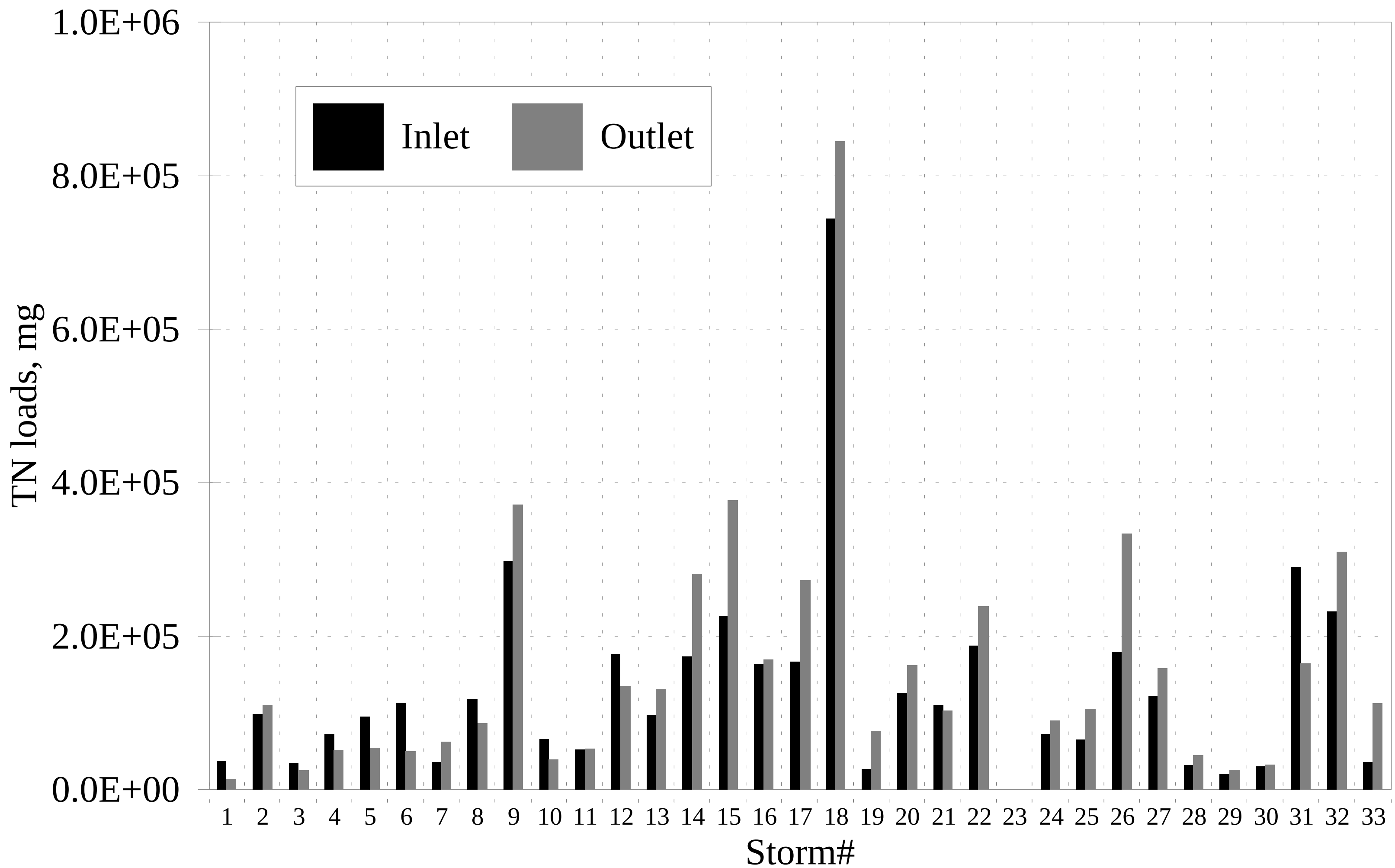

Figure D.30 Inlet-outlet load comparison: TN, mg

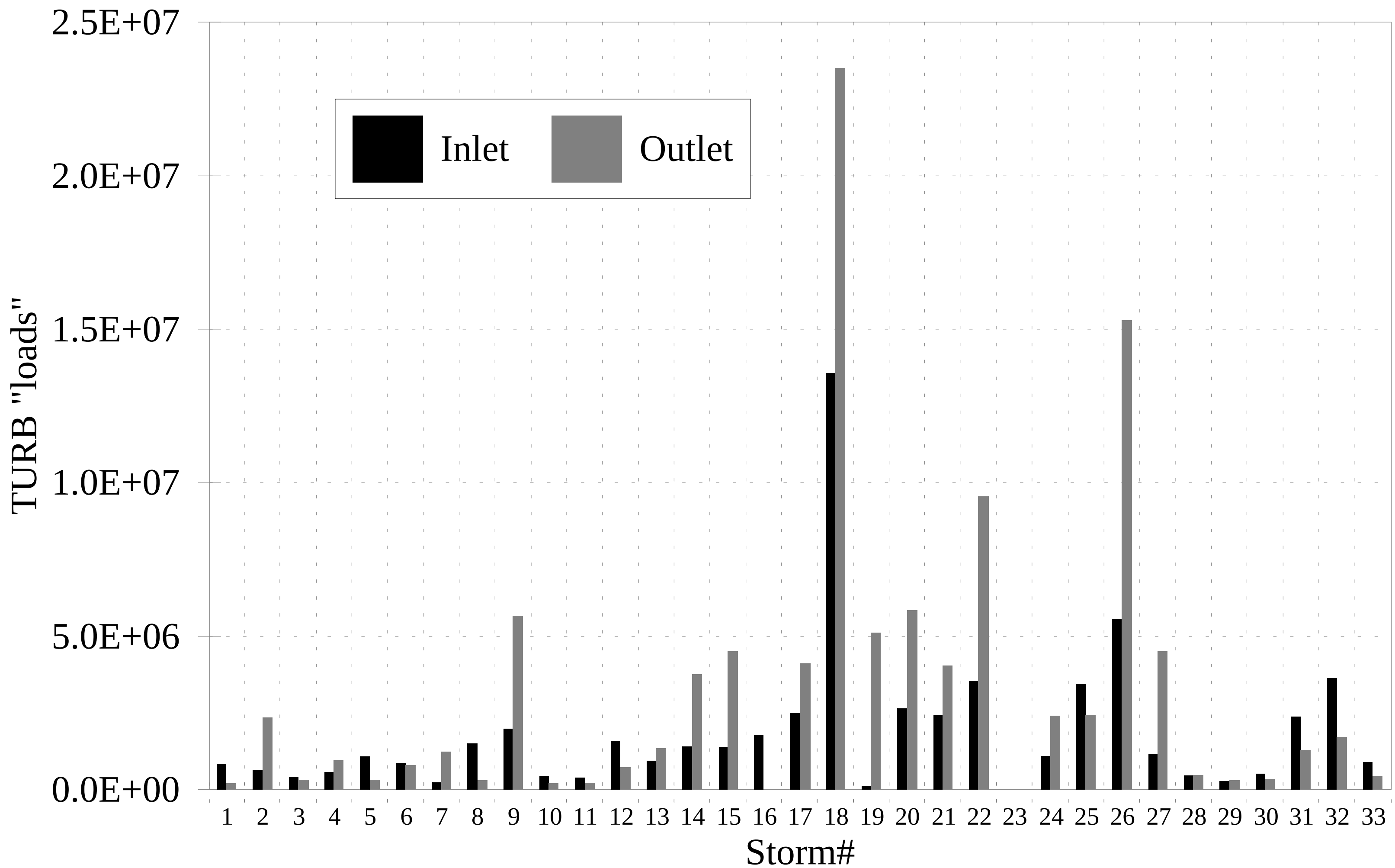

Figure D.31 Inlet-outlet "load" comparison: turbidity

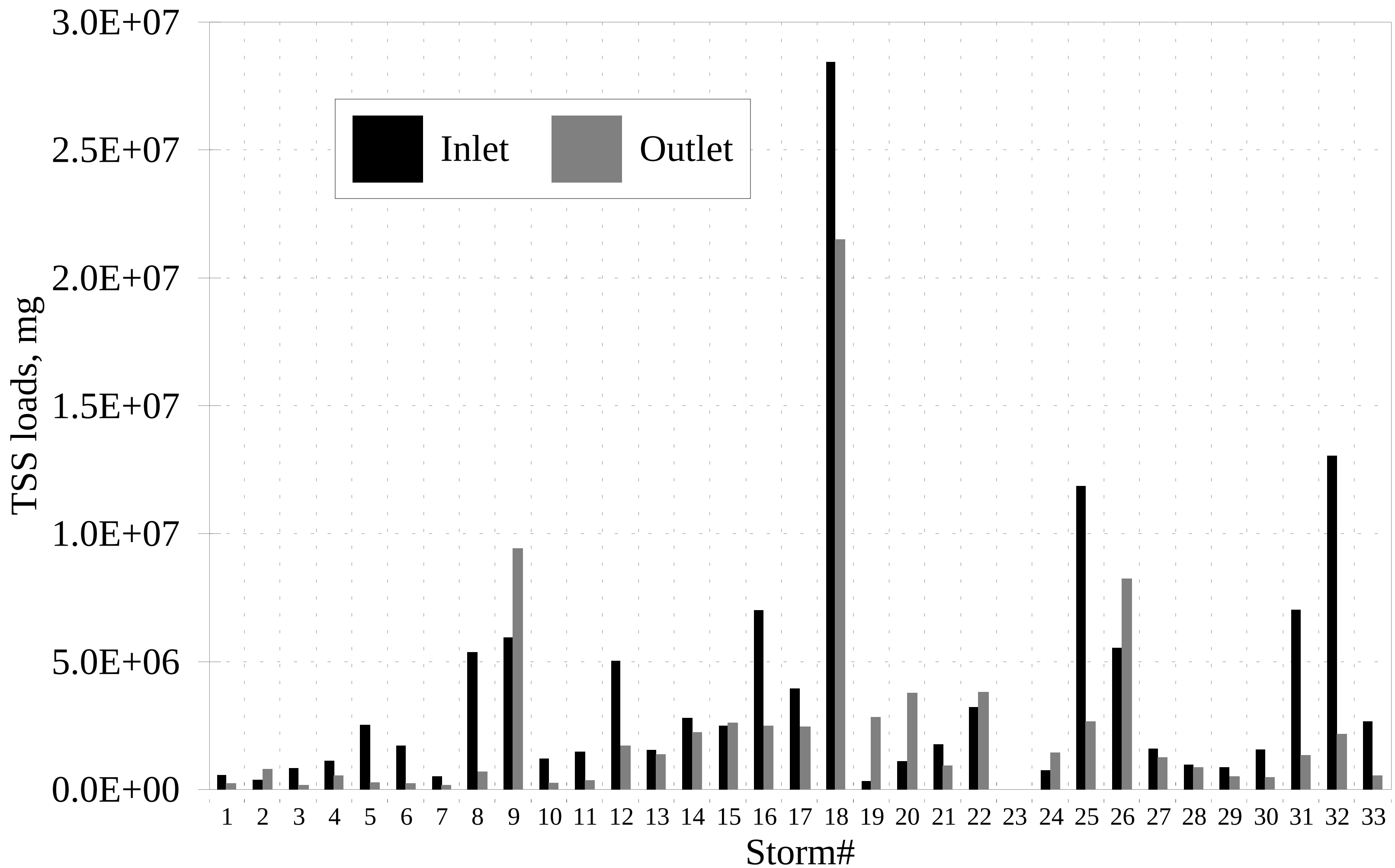

Figure D.32 Inlet-outlet load comparison: TSS, mg

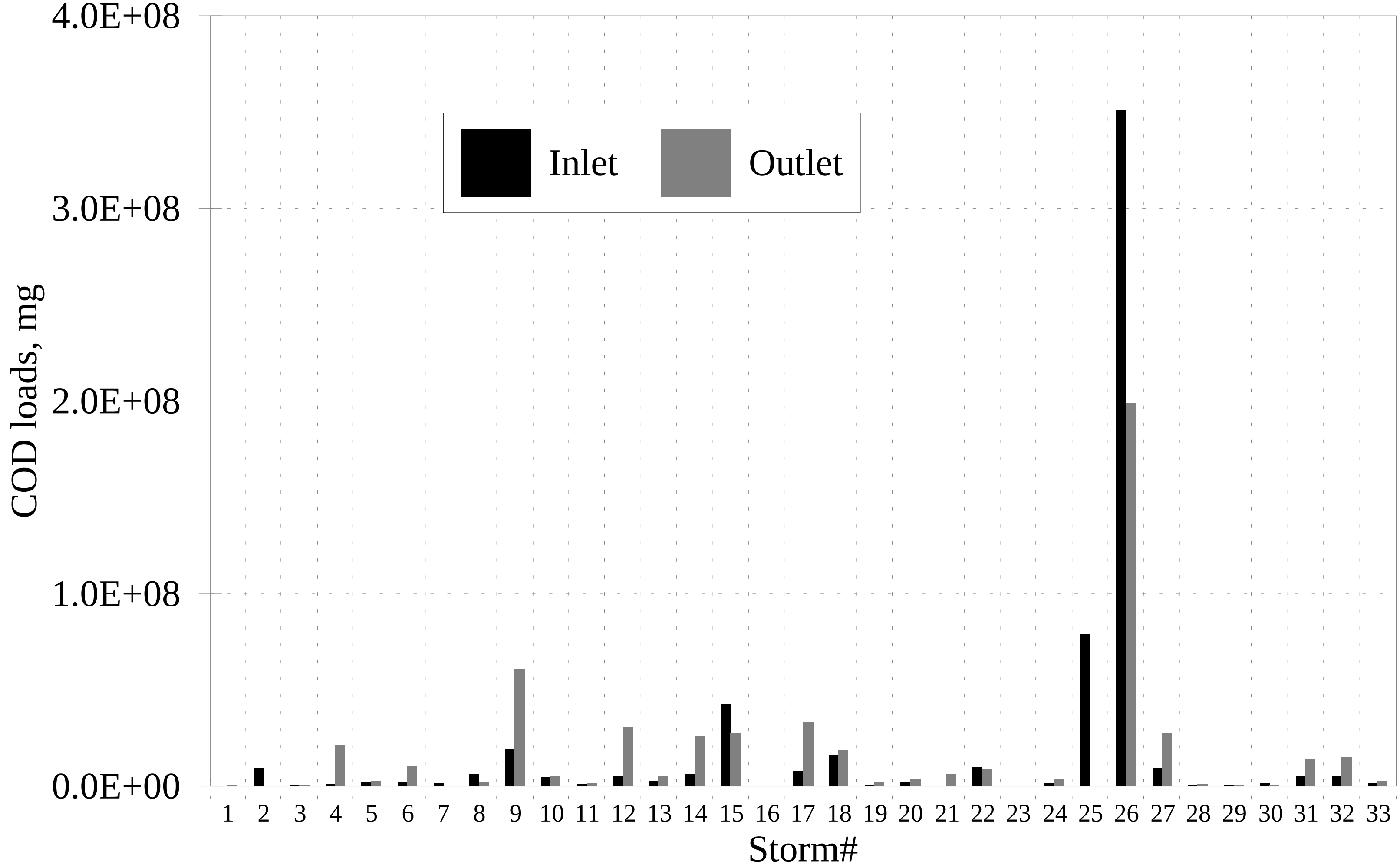

Figure D.33 Inlet-outlet load comparison: COD, mg

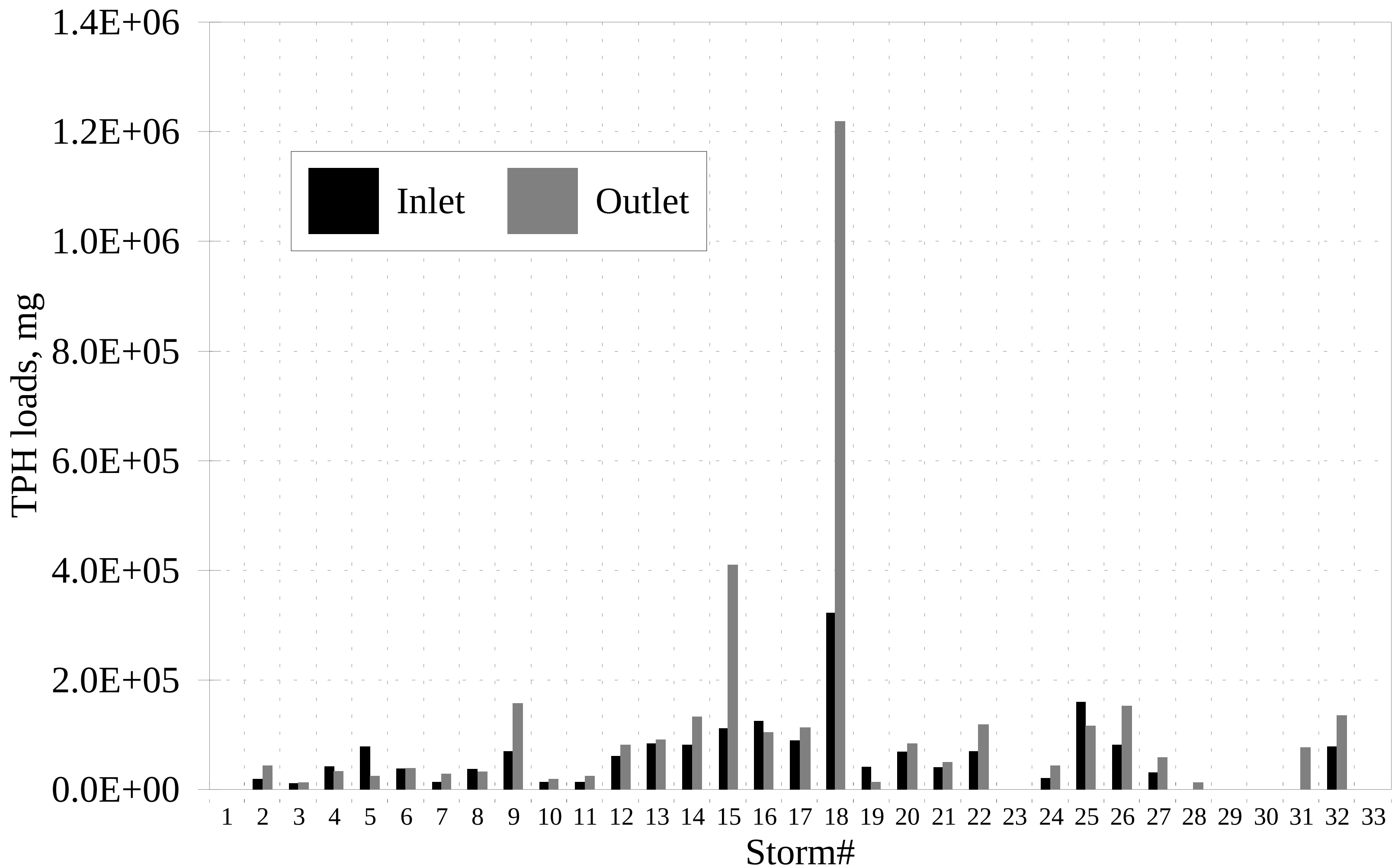

Figure D.34 Inlet-outlet load comparison: TPH, mg

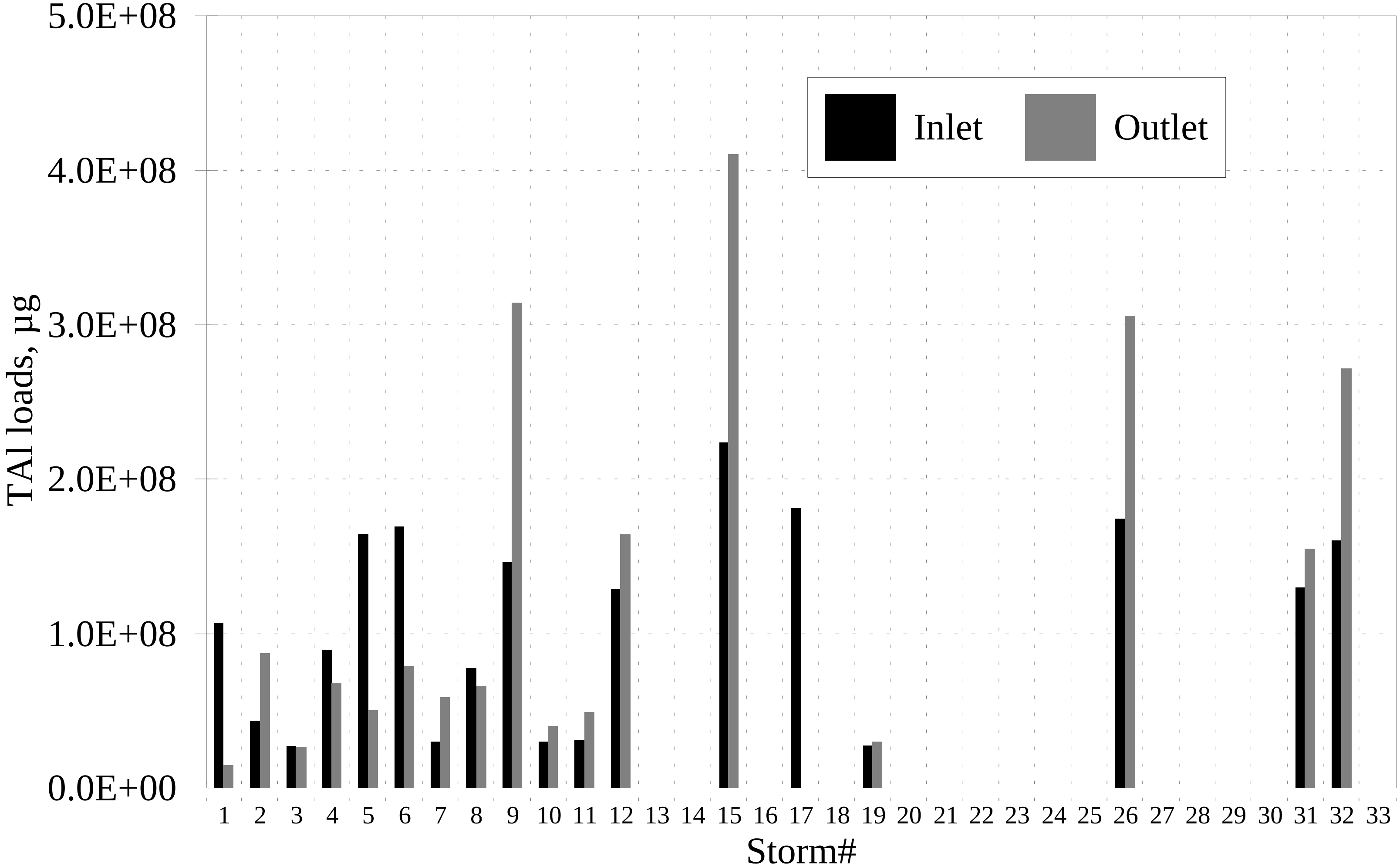

Figure D.35 Inlet-outlet load comparison: TAl, µg

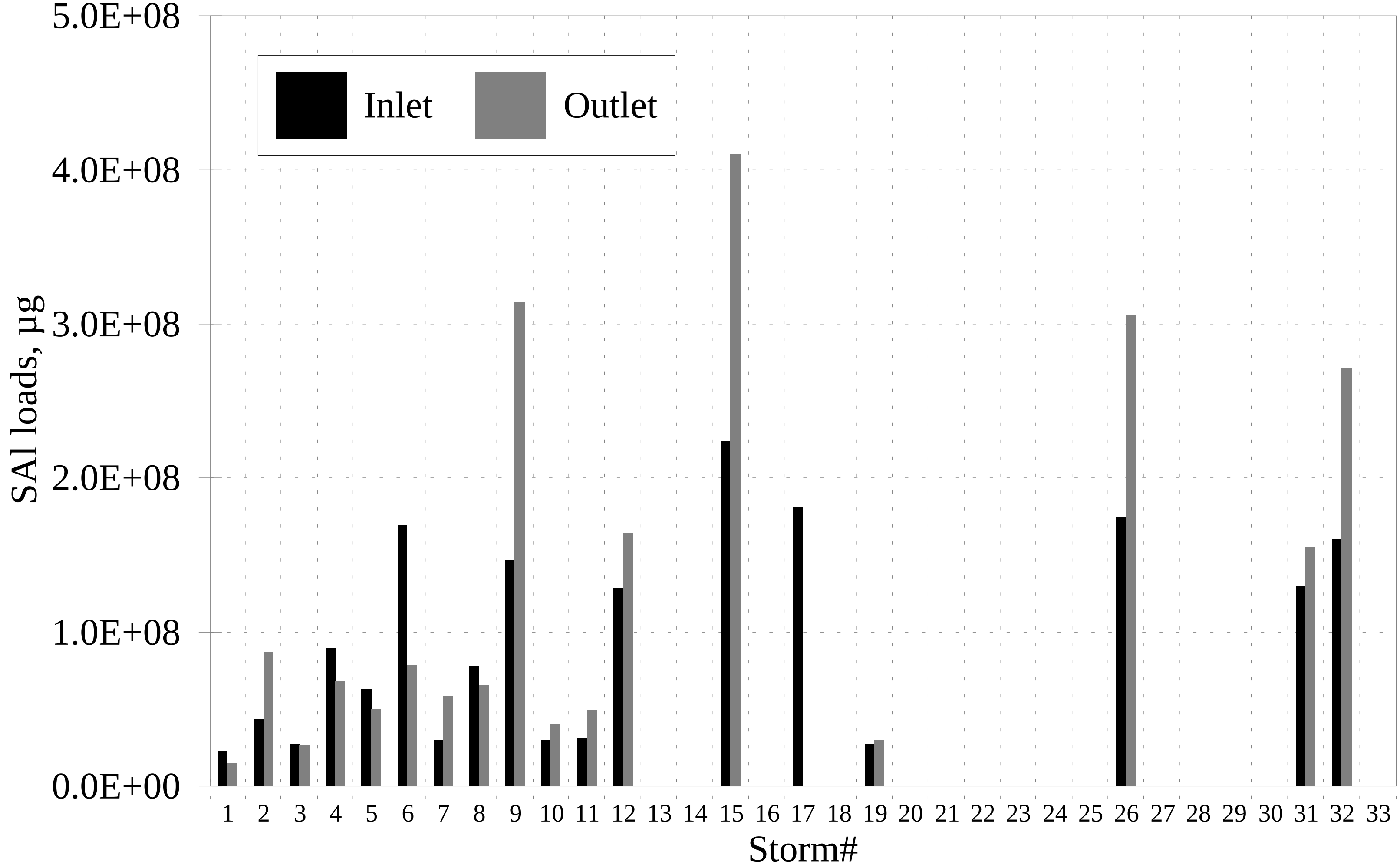

Figure D.36 Inlet-outlet load comparison: SAl, µg

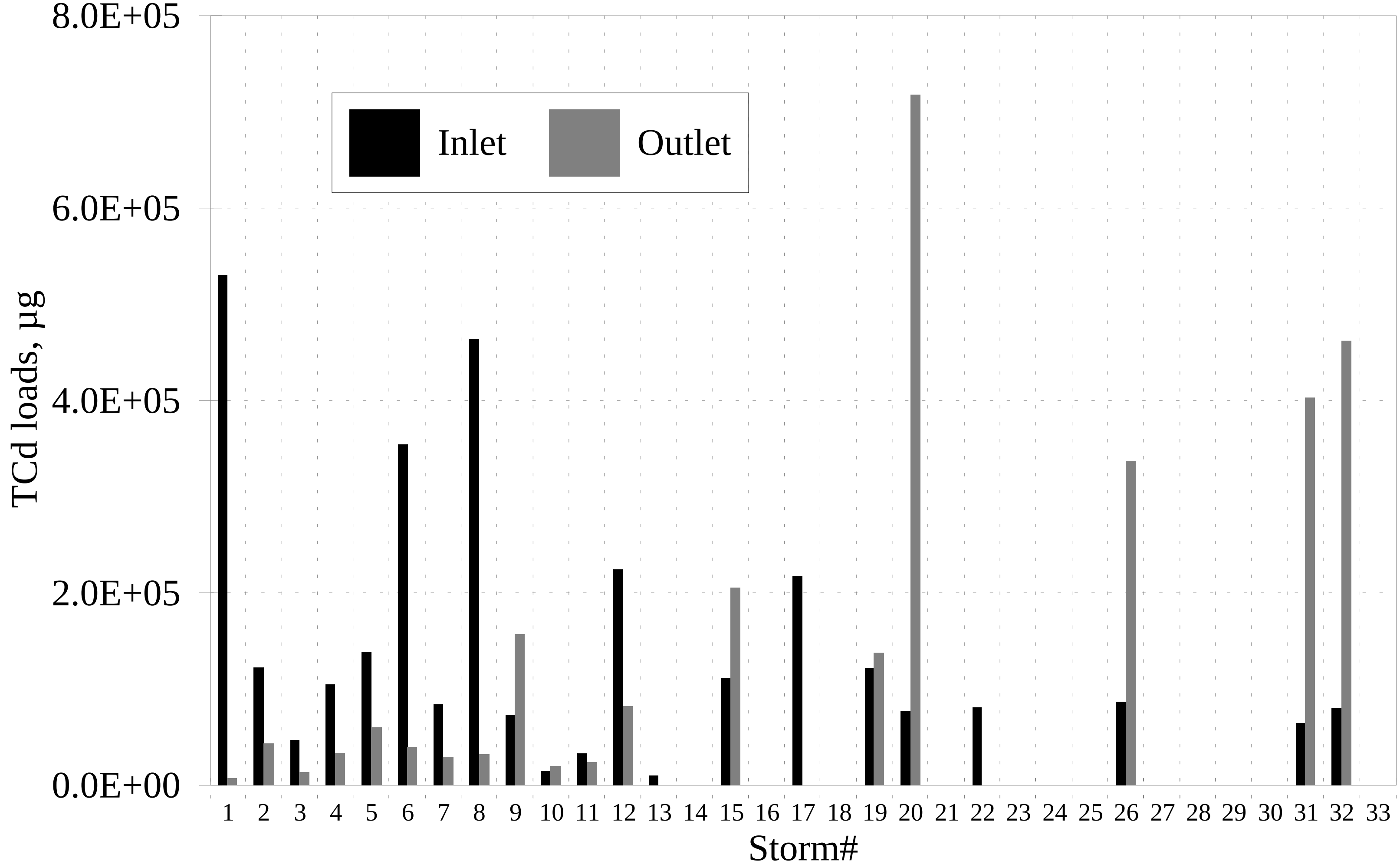

Figure D.37 Inlet-outlet load comparison: TCd, µg

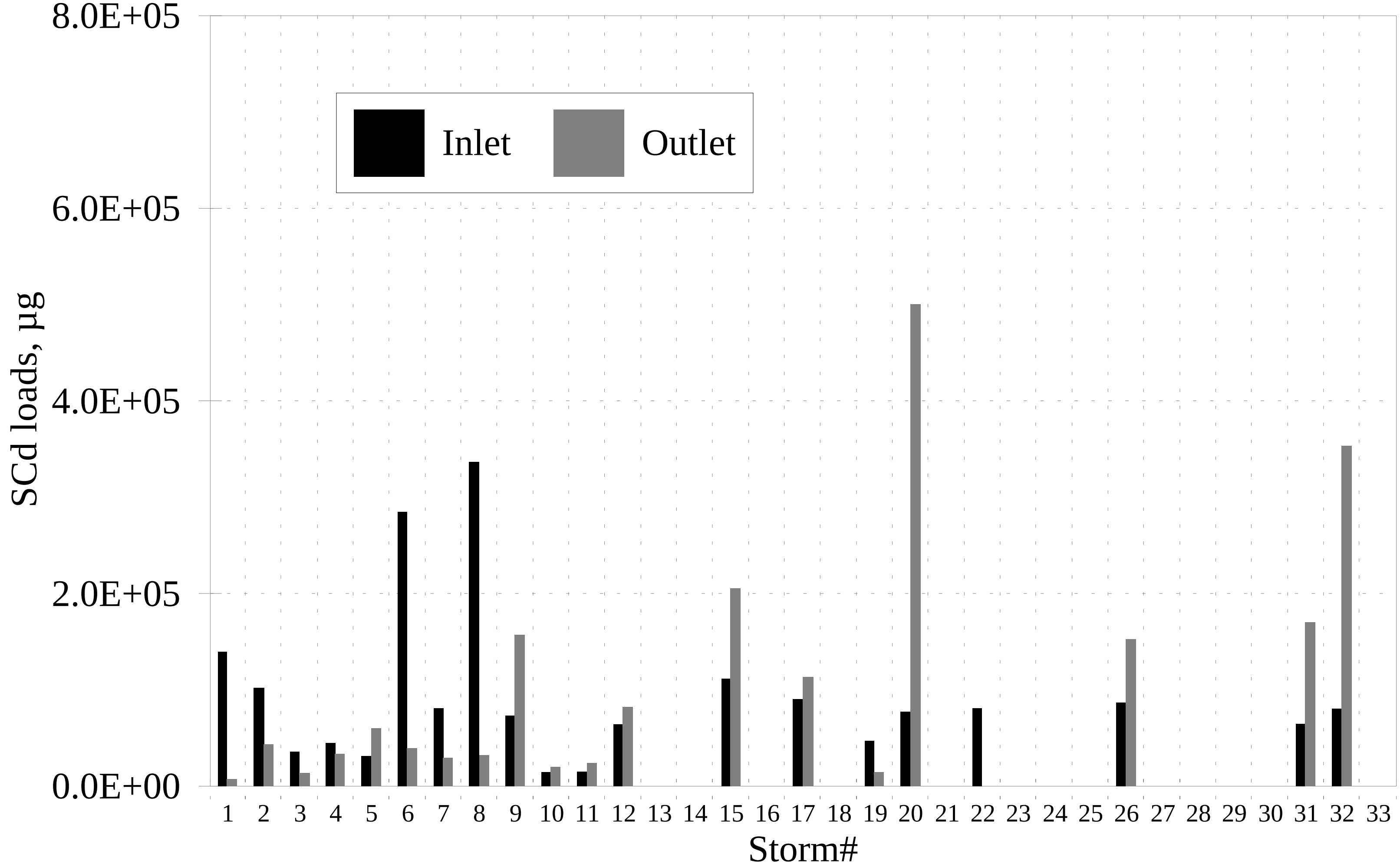

Figure D.38 Inlet-outlet load comparison: SCd, µg

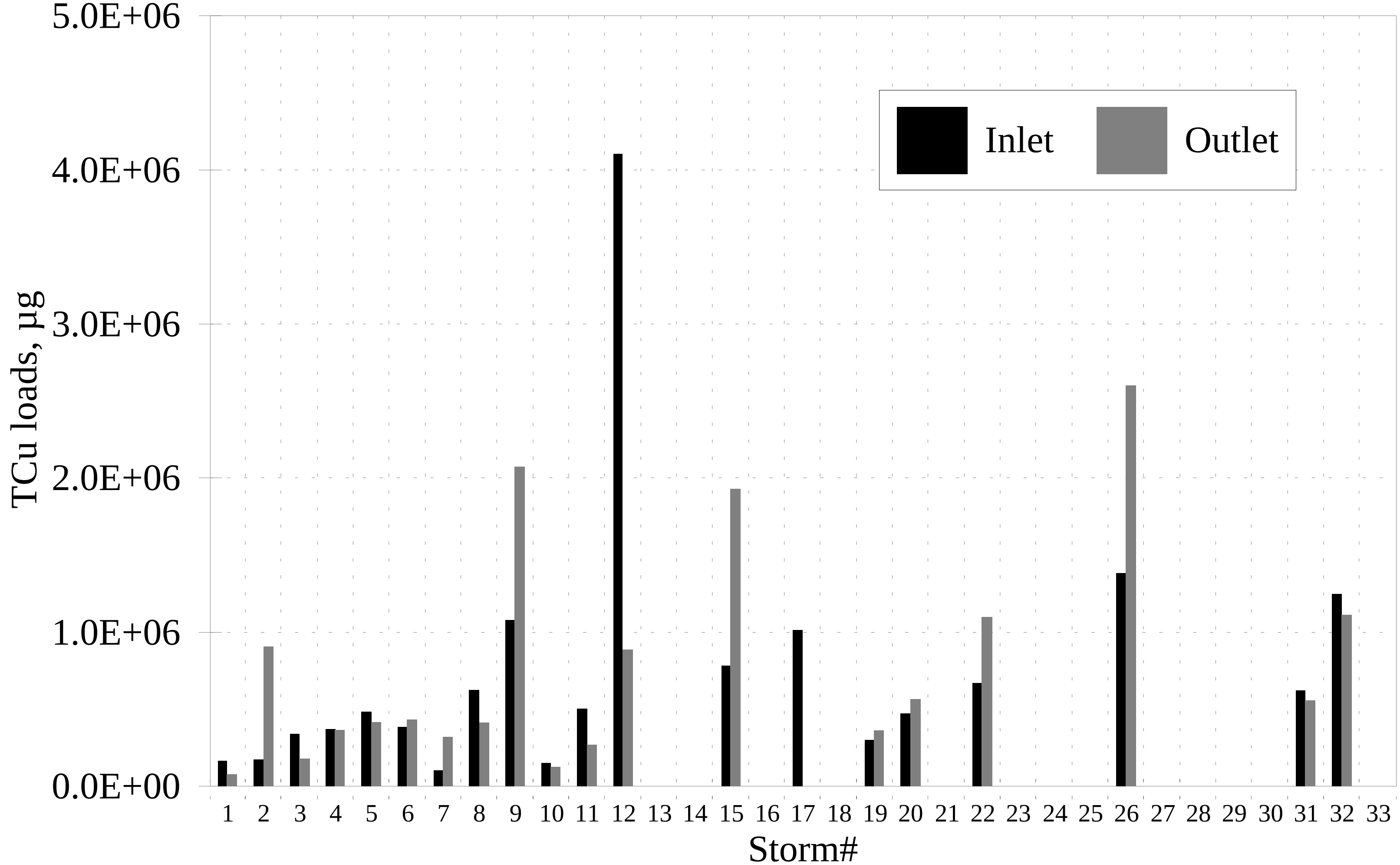

Figure D.39 Inlet-outlet load comparison: TCu, µg

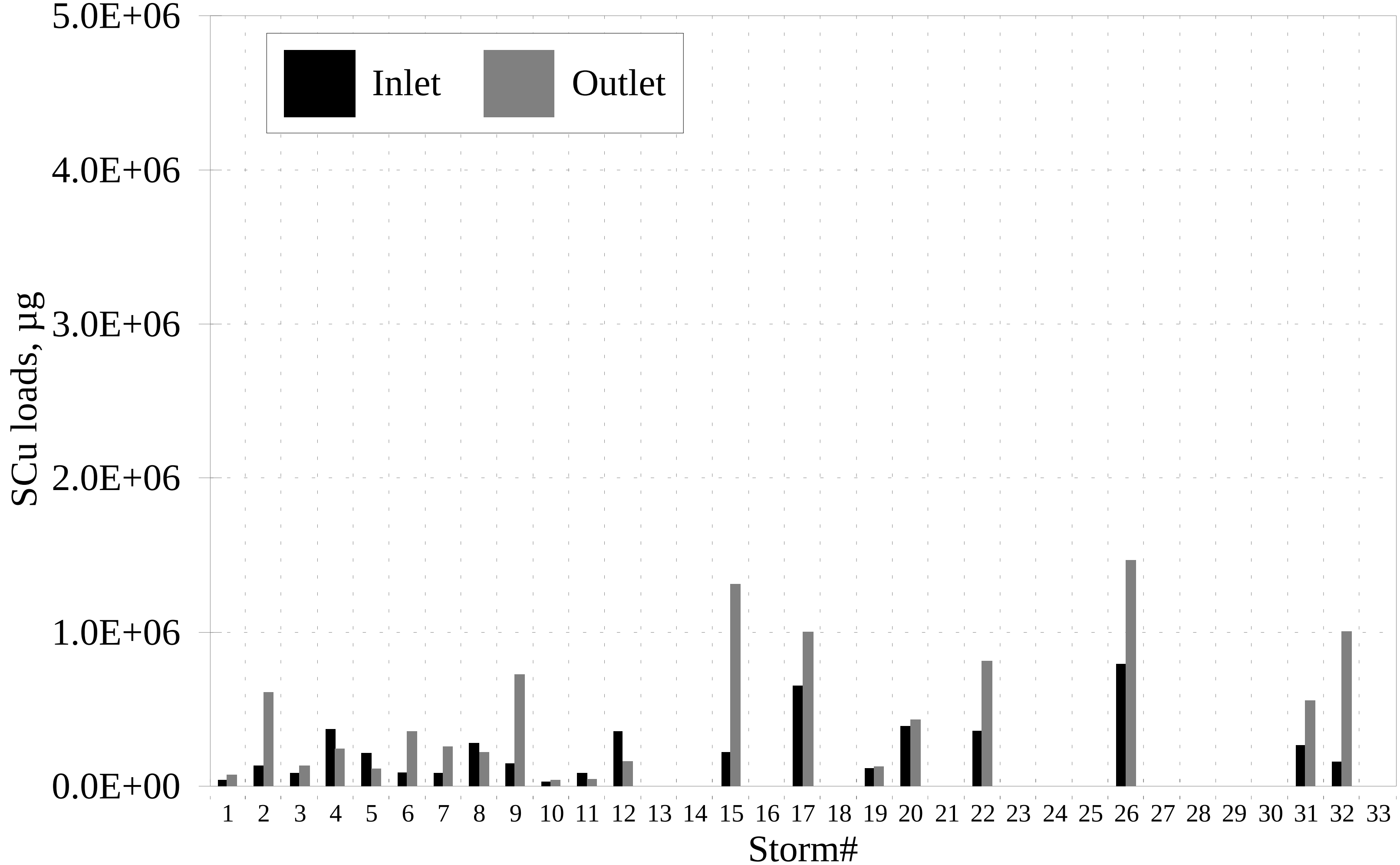

Figure D.40 Inlet-outlet load comparison: SCu, µg

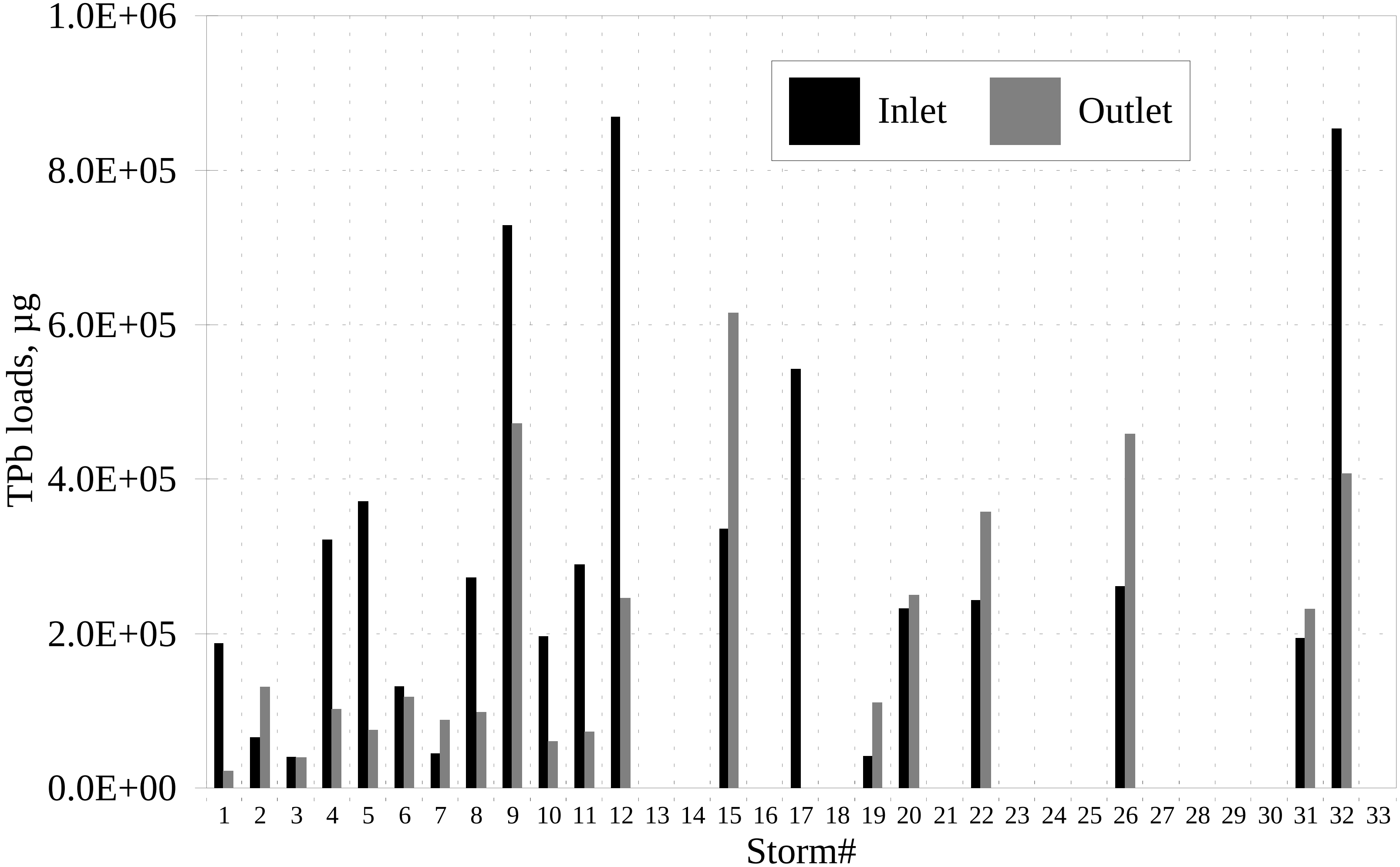

Figure D.41 Inlet-outlet load comparison: TPb, µg

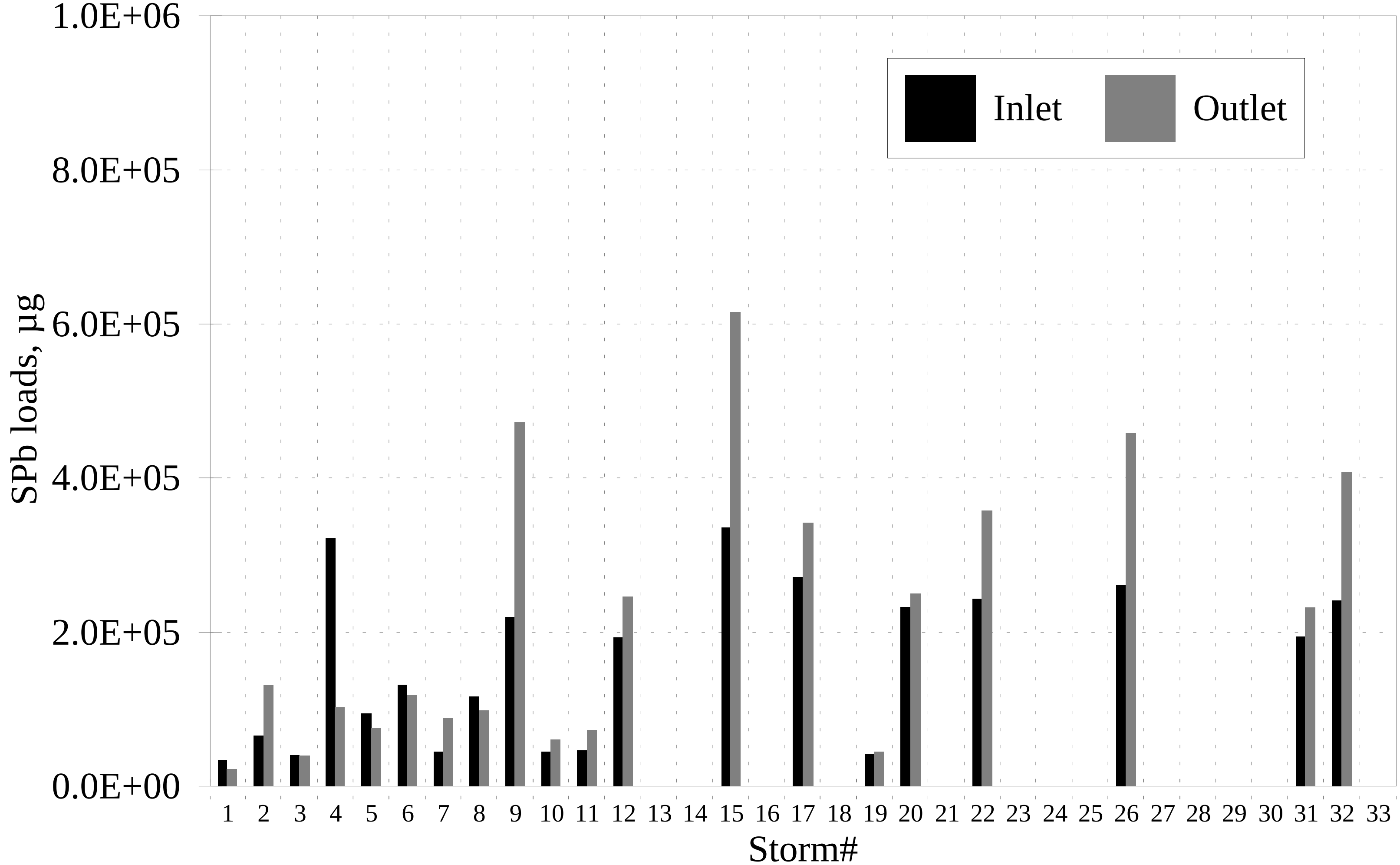

Figure D.42 Inlet-outlet load comparison: SPb, µg

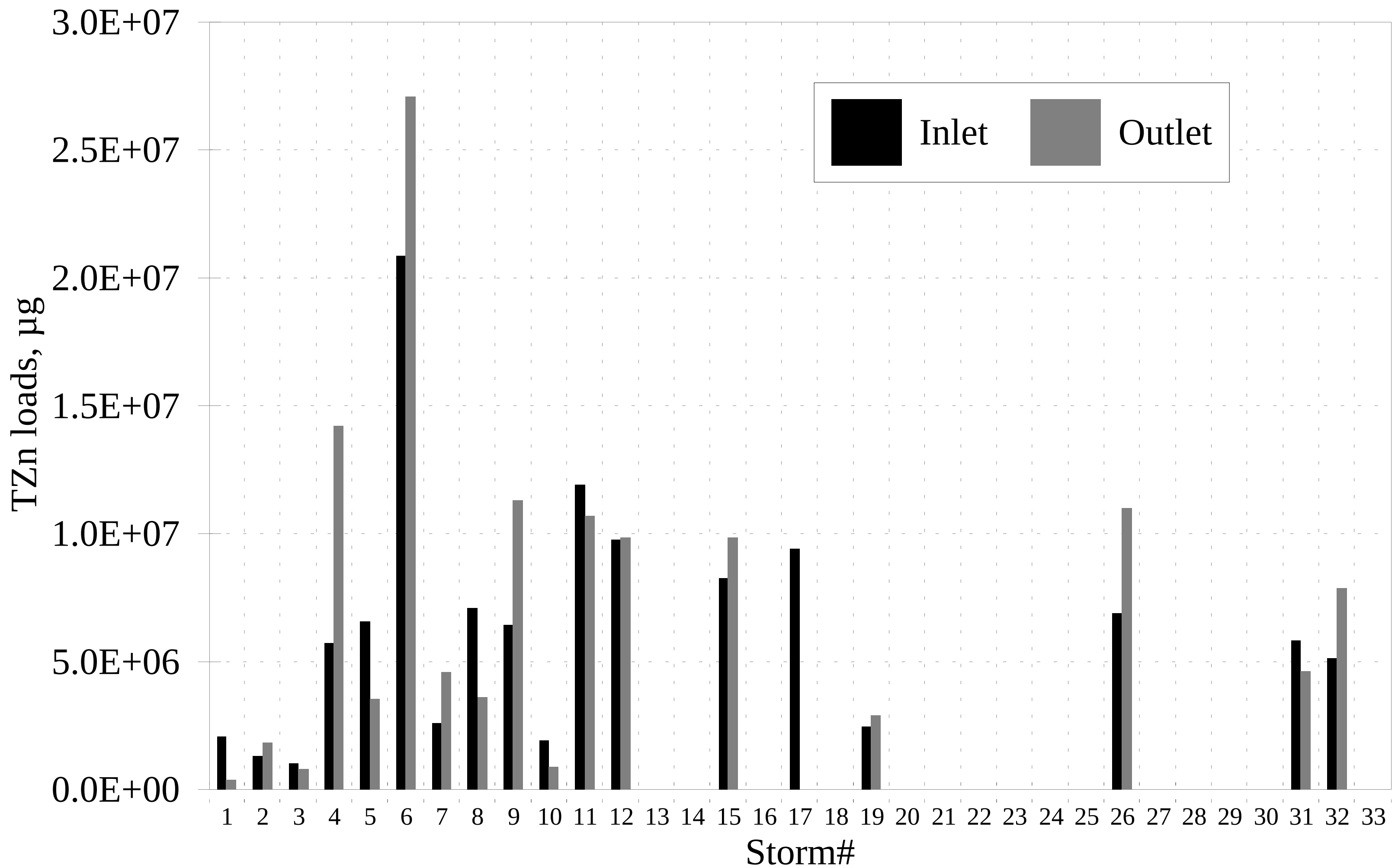

Figure D.43 Inlet-outlet load comparison: TZn, µg

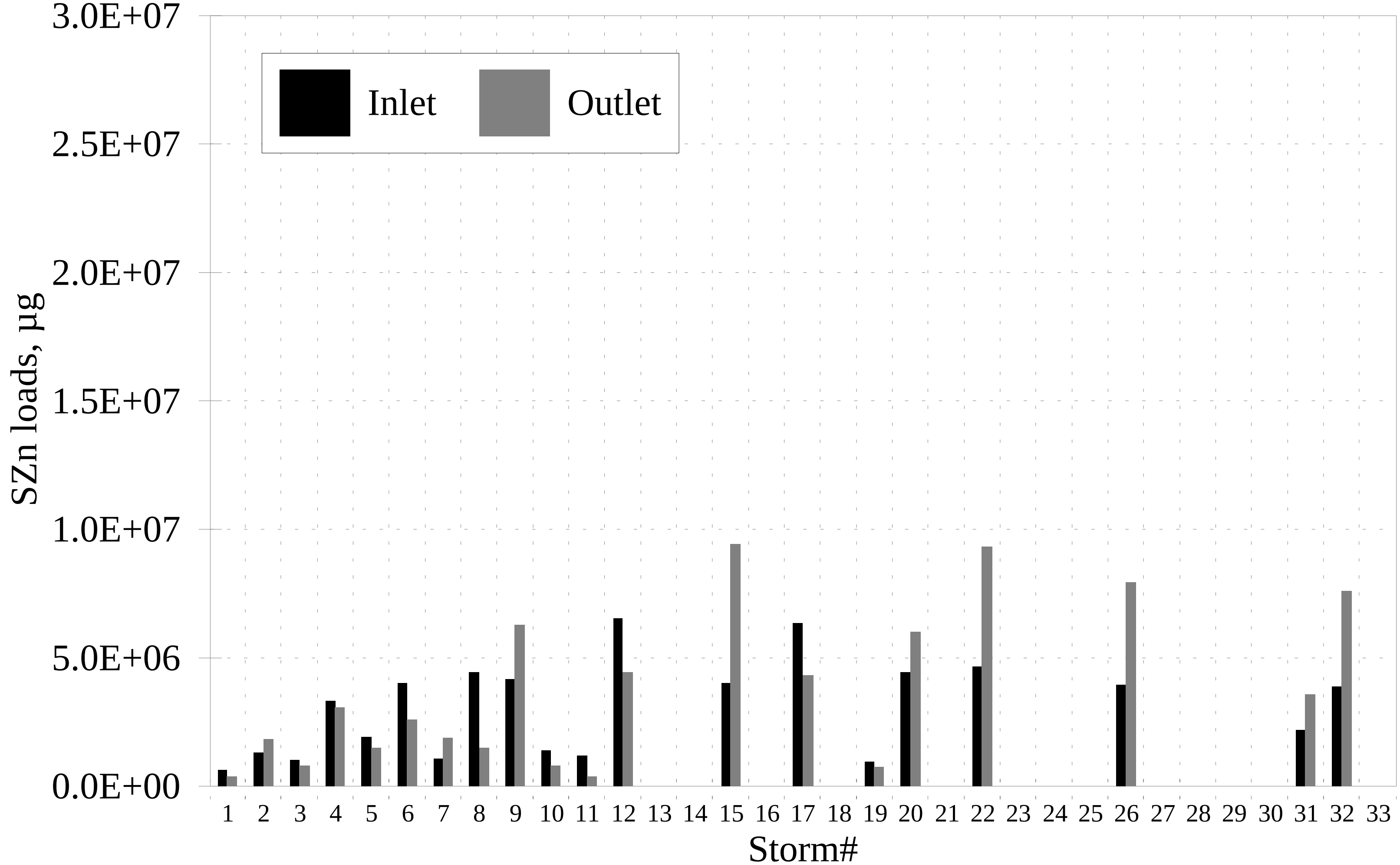

Figure D.44 Inlet-outlet load comparison: SZn, µg

VITA

James N. Carleton was born on November 9, 1959, in Madison, Wisconsin. After having lived in Missouri, New Jersey, Illinois, New Hampshire, and North Carolina, Jim attended the University of North Carolina at Chapel Hill, where he received a B.S. degree in chemistry in 1982.

Since graduating, the author has worked in various fields, including blood coagulation research, GC/MS analysis of environmental samples, and research related to Sudden Infant Death Syndrome. The author is currently employed by the Environmental Protection Agency, where he works as a chemist in the Environmental Fate and Effects Division of the Office of Pesticide Programs.