SURFACE WATER IMPROVEMENT AND MANAGEMENT (SWIM) PROGRAM

TECHNICAL REPORT

Removal of microbial indicators from stormwater using sand filtration, wet detention, and alum treatment best management practices

Raymond C. Kurz, Ph.D.

Southwest Florida Water Management District
1998

Protecting Your Water Resources

*The mission of the **Southwest Florida Water Management District (SWFWMD)** is to manage the water and water-related resources within its boundaries. Central to the mission is maintaining the balance between the water needs of current and future users while protecting and maintaining the natural systems that provide the District with its existing and future water supply.*

The Governing Board of the District assumes its responsibilities as authorized in Chapter 373 and other chapters of the Florida Statutes by directing a wide-range of programs, initiatives, and actions. These programs include, but are not limited to, flood control, regulatory programs, water conservation, education, and supportive data collection and analysis efforts.

*The **Surface Water Improvement and Management (SWIM) Program** plans and coordinates the preservation and restoration of threatened waterbodies of regional or statewide significance. It was formed in 1987 by the Florida Legislature in response to increasing degradation of Florida's water resources. SWIM is administered by the Florida Department of Environmental Protection (FDEP) through each of the State's five water management districts.*

Within the Southwest Florida Water Management District, ten waterbodies have been identified for restoration, management, or protection. These include: Tampa Bay, Sarasota Bay, Crystal River/Kings Bay, Lake Panasoffkee, Charlotte Harbor, Lake Tarpon, Lake Thonotosassa, Rainbow River, Banana Lake, and the Winter Haven Chain of Lakes.

The primary goals of the SWIM Program include improving or maintaining water quality, restoring or protecting natural systems, and providing watershed management assistance to local governments. Specifically, the SWIM Program of the Southwest Florida Water Management District has implemented more than 50 projects to improve water quality and natural ecosystems within its ten priority waterbodies. These activities include habitat restoration, stormwater treatment, diagnostic water quality studies, and lake restoration.

Cover

Top left photo: Stormwater outfall at Little Bayou in St. Petersburg, Florida.
Top right photo: An above-ground sand filter treatment facility (small building in center of photo) in Madeira Beach, Florida.
Bottom left photo: Aerial view of two experimental stormwater treatment ponds at the Southwest Florida Water Management District's Tampa Service Office.
Bottom right photo: Site of an alum injection stormwater treatment system in Pinellas Park, Florida and one of two sites used for stormwater collection for alum jar testing.

REMOVAL OF MICROBIAL INDICATORS FROM STORMWATER USING SAND FILTRATION, WET DETENTION, AND ALUM TREATMENT BEST MANAGEMENT PRACTICES

Raymond C. Kurz, Ph.D.

Surface Water Improvement and Management (SWIM) Section
Southwest Florida Water Management District
7601 U.S. Hwy. 301 N.
Tampa, Florida 33637
U.S.A.

(813) 985-7481
(813) 987-6747 FAX
ray.kurz@swfwmd.state.fl.us

October
1998

Southwest Florida Water Management District
Technical Report

TABLE OF CONTENTS

LIST OF TABLES iii

LIST OF FIGURES v

EXECUTIVE SUMMARY x

CHAPTER ONE - INTRODUCTION 1
Best Management Practices for the Treatment of Stormwater 5
Sand Filtration 6
Wet Detention 7
Alum Coagulation 8
Indicator Microorganisms Used as Surrogates for Microbial Pathogens 9
Stormwater Best Management Practices for Removing Microorganisms 12

CHAPTER TWO - METHODS AND MATERIALS 15
Site Locations and Site-Specific Sampling Protocol 15
Sand Filtration 15
Wet Detention 21
Alum Coagulation 29
Laboratory Analyses and Physicochemical Parameters 32
Data Analysis 35

CHAPTER THREE - RESULTS 40
Individual Stormwater Treatment Systems 40
Sand Filter 40
Microbial Indicators 40
Bacteria Speciation 49
Physicochemical Parameters 50
Metal Toxicity 51
Comparisons with Water Quality Standards 52
Wet Detention 54
Physicochemical Parameters 54
Microbial Indicators 57
Bacteria Speciation 74
Decay Rates 75
In-Pond and Sediment Concentrations 75
Monitoring Well Samples 78
Water Budget 80
Comparisons with Water Quality Standards 80
Alum Coagulation 81

Physicochemical Parameters 81
Microbial Indicators 84
Decay Rates 94
Bacteria Speciation 99
Comparison with Water Quality Standards 99
Relationships Between Parameters Using Data From All Three Sites 101
MS2 as an Indicator for Total and Fecal Coliforms 101
Relationships with Turbidity 101
Comparisons Between Exceedences of Surface Water Quality Standards 104
Comparisons Between Sand Filtration, Wet Detention, and Alum Coagulation 108
Treatment Train Reductions 108
Potential Reductions in Health Risks 110

CHAPTER FOUR - DISCUSSION 117
Individual Stormwater Treatment Systems 117
Sand Filter 117
Ability to Meet State Water Quality Standards 125
Wet Detention 125
Ability to Meet State Water Quality Standards 135
Alum Coagulation 136
Ability to Meet Water Quality Standards 140
Comparisons between Sand Filtration, Wet Detention, and Alum Treatment . . 140
Recommendations 144
CONCLUSIONS 148

LITERATURE CITED 149

APPENDICES 161
Appendix 1. Raw data for microbial indicators from the three stormwater treatment systems. 162
Appendix 2. Raw data for physicochemical parameters from the three stormwater treatment systems. 165
Appendix 3. Raw data for heavy metal samples from the sand filter trials. 168

LIST OF TABLES

Table 1. Patients discharged from hospitals by category of foodborne/waterborne disease in the U.S., 1987-1990. 5

Table 2. State of Florida Surface Water Quality Standards (Chapter 62-302) for microbial indicators and turbidity. 39

Table 3. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the sand filter treatment system challenge. 41

Table 4. List of coliform bacteria identified in inflow and ten outflow samples of the sand filter. 49

Table 5. Mean heavy metal concentration, removal efficiency, and correlation coefficient (between metal concentrations and optical densities of *E. coli* toxicity assays) from the sand filter challenge from Trials 2 and 3. 51

Table 6. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the 5-day shallow wet detention pond challenge. 58

Table 7. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the 5-day deep wet detention pond challenge 58

Table 8. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the 14-day shallow wet detention pond challenge. 59

Table 9. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the 14-day deep wet detention pond challenge. 59

Table 10. List of coliform bacteria identified in inflow and five outflow samples of the wet detention pond (5-day trial). 74

Table 11. Concentrations of indicator and physical parameters from grab samples in the deep zones of each of the wet detention ponds during the 5-day trial. 78

Table 12. Removal efficiencies based on differences between concentrations of indicator and physical parameters in control versus alum treated stormwater samples from Lowry Park at a dose of 10 mg/L alum. 86

Table 13. Removal efficiencies based on differences between concentrations of indicator and physical parameters in control versus alum treated stormwater samples from Pinellas Park at a dose of approximately 600 mg/L alum. 86

Table 14. List of coliform bacteria identified in control (CON) and replicate alum-treated stormwater samples taken from Lowry Park. No bacteria were present after alum treatment in the high dose trial. 100

Table 15. Comparisons of exceedences of state surface water quality standards between sand filtration, wet detention, and alum treatment of stormwater. 106

Table 16. Treatment train reduction calculations. 111

Table 17. Potential reductions in risk of disease using various stormwater treatment technologies. 114

LIST OF FIGURES

Fig. 1. Location of the stormwater treatment systems and collection sites in the Tampa Bay area. 16

Fig. 2. Schematic of the sand filter system in Madeira Beach, Florida. 17

Fig. 3. Dimensions of the wet detention ponds at the SWFWMD Tampa Service Office. 22

Fig. 4. Cross-section diagrams of wet detention ponds. 23

Fig. 5. Schematic of the wet detention ponds at the SWFWMD Tampa Service Office 24

Fig. 6. Changes in water levels during the 5-day and 14-day storm simulation trials at the SWFWMD wet detention ponds in Tampa, Florida. 26

Fig. 7. Monthly trends in total and fecal coliform bacteria concentrations in Hamilton Creek (Lowry Park) prior to the collection of stormwater for jar testing in February 1998 31

Fig. 8. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, and turbidity during Trial 1 (unsaturated) sand filter challenge. 42

Fig. 9. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, and turbidity during Trial 2 (saturated) sand filter challenge. 43

Fig. 10. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, and turbidity during Trial 3 (saturated) sand filter challenge. 44

Fig. 11. Trends in total and fecal coliform bacteria concentrations over time using sand filtration. 45

Fig. 12. Trends in concentrations of MS2 coliphage and fluorescent beads over time using sand filtration. 47

Fig. 13. Trends in turbidity and total suspended solids concentrations over time using sand filtration. 48

Fig. 14. Trends in temperature, dissolved oxygen, and pH over time using sand filtration 50

Fig. 15. Trends in heavy metal concentrations before (To) and after (1-10) sand filtration and trends in Zn concentrations and toxicity (inverse of optical density of a solution containing *E. coli* cultures) before (To) and after (1-10) sand filtration. 53

Fig. 16. Trends in temperature, pH, and conductivity over time for the shallow and deep wet detention ponds during the 5-day trials. 55

Fig. 17. Trends in temperature, pH, and conductivity over time for the shallow and deep wet detention ponds during the 14-day trials. 56

Fig. 18. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the 5-day seeded challenge of the shallow wet detention pond. 60

Fig. 19. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the 5-day seeded challenge of the deep wet detention pond. 61

Fig. 20. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the 14-day seeded challenge of the shallow wet detention pond. 62

Fig. 21. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the 14-day seeded challenge of the deep wet detention pond. 63

Fig. 22. Relationship between log-transformed total coliform bacteria concentrations and turbidity and total suspended solids in outflow samples from the wet detention ponds. 65

Fig. 23. Trends in total and fecal coliform bacteria concentrations over time during the 5-day wet detention pond trials. 66

Fig. 24. Trends in total and fecal coliform bacteria concentrations over time during the 14-day wet detention pond trials. 67

Fig. 25. Trends in MS2 and bead concentrations over time during the 5-day wet detention pond trials. 69

Fig. 26. Trends in MS2 and bead concentrations over time during the 14-day wet detention pond trials. 71

Fig. 27. Trends in turbidity and total suspended solids concentrations over time during the 5-day wet detention pond trials. 72

Fig. 28. Trends in turbidity and total suspended solids concentrations over time during the 14-day wet detention pond trials. 73

Fig. 29. Decay rates for total and fecal coliforms and MS2 during 5-day shallow and deep wet detention pond trials. 76

Fig. 30. Decay rates for total and fecal coliforms and MS2 during 14-day shallow and deep wet detention pond trials. 77

Fig. 31. Comparisons between inflow, in pond, and outflow concentrations for total coliforms, fecal coliforms, MS2, beads, turbidity, and total suspended solids in shallow and deep wet detention ponds during the 5-day trial. 79

Fig. 32. Trends in pH and conductivity from Pinellas Park stormwater after dosing with alum at 600 mg/L. 82

Fig. 33. Trends in pH and conductivity from Lowry Park stormwater after dosing with alum at 10 mg/L. 83

Fig. 34. Trends in heavy metal concentrations over time in alum treated stormwater using a dose of approximately 600 mg/L. 85

Fig. 35. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the Lowry Park stormwater seeded challenge using alum coagulation treatment 87

Fig. 36. Trends in the removal of turbidity and total suspended solids from Lowry Park stormwater using an alum dose of 10 mg/L. 89

Fig. 37. Trends in the removal of total and fecal coliform bacteria from Lowry Park stormwater using an alum dose of 10 mg/L. 90

Fig. 38. Trends in the removal of MS2 and 3 μm fluorescent beads from Lowry Park stormwater using an alum dose of 10 mg/L. 91

Fig. 39. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the Pinellas Park stormwater seeded challenge using alum coagulation treatment 92

Fig. 40. Trends in the removal of total and fecal coliform bacteria from Pinellas Park stormwater using an alum dose of 600 mg/L. 93

Fig. 41. Trends in the removal of MS2, and 3 μm fluorescent beads from Pinellas Park stormwater using an alum dose of 600 mg/L. 95

Fig. 42. Trends in the removal of turbidity and total suspended solids from Pinellas Park stormwater using an alum dose of 600 mg/L. 96

Fig. 43. Decay rates for total and fecal coliforms and MS2 using control (no alum) sample data during low dose and high dose alum trials. 97

Fig. 44. Decay rates for total and fecal coliforms and MS2 during low dose and high dose alum treatments. 98

Fig. 45. Relationships between log-transformed MS2 concentrations and log-transformed total and fecal coliform concentrations from inflow samples from the three stormwater BMPs. 102

Fig. 46. Regression of total coliform and fecal coliform values with turbidity for all inflow samples from the three stormwater BMP sites. 103

Fig. 47. Histogram showing frequencies of fecal:total coliform ratios for all data combined from the sand filtration, wet detention, and alum treatment BMP's. Box plot comparing fecal:total coliform ratios for inflow and outflow samples taken from sand filtration, alum treatment, and wet detention ponds. 105

Fig. 48. Resulting dendrogram from cluster analysis of removal efficiency comparisons of microbial indicators for the seven different stormwater treatment trials. 107

Fig. 49. Removal efficiencies of three stormwater treatment methods around Tampa Bay, Florida. 109

Fig. 50. Schematic of a treatment train or series of stormwater BMPs that could provide optimal microorganism removal. 112

Fig. 51. Schematic of a wet pond treatment design that could provide greater microorganism removal than current commonly-used designs. 113

EXECUTIVE SUMMARY

Water resources throughout the U.S. have suffered decades of pollution and overuse. As a result, the restoration and improvement of water quality in aquatic ecosystems are important issues in states like Florida where various human activities vie for limited water resources. Technological advances in wastewater treatment have improved dramatically in the last decade which has led to higher quality effluent from many domestic wastewater treatment facilities (Smith and Smith, 1982). Despite this significant point-source reduction in pollutant loading to surface waters, other sources still contribute excess pollution. Previous research has determined that a large proportion of pollutants originate from non-point sources such as urban and agricultural stormwater runoff (U.S. EPA, 1983). In fact, the U.S. EPA in 1984 determined that nearly 90% of fecal coliform pollution to surface waters originates from stormwater runoff.

In Tampa Bay, for example, several tributaries which receive agricultural, industrial, and urban runoff exhibit consistent, elevated total and fecal coliform bacteria concentrations which often exceed state standards for shellfish harvesting and recreational exposure (Hillsborough County Environmental Protection Commission, 1996). Based on state water quality standards, 45% of these tributaries did not meet their intended use for recreation and the propagation and maintenance of a healthy, well-balanced population of fish and wildlife (FDEP, 1996).

A number of microbial pathogens including bacteria (*Escherichia coli*, *Pseudomonas aeruginosa, Salmonella* sp., *Shigella, Staphylococcus aureus, Vibrio*), viruses (poliovirus, Echovirus, Coxsackie virus, hepatitis A), and protozoa (*Giardia*, *Cryptosporidium*) can be present in stormwater and have been implicated in waterborne disease outbreaks (Olivieri *et al.*, 1977; Hayes *et al.*, 1987; Bemiss *et al.*, 1989; Koenig *et al.*, 1991; Enriquez *et al.*, 1992; Kebabjian, 1994). A massive *Cryptosporidium* outbreak in Milwaukee in 1992 that affected over 400,000 individuals and a smaller outbreak of hepatitis A in Florida (Vonstille *et al.*, 1993) were caused by contaminated stormwater runoff. In urbanized areas, contaminated stormwater can impact recreational beaches in both marine and freshwater environments and can cause a number of bathing-related illnesses including eye, ear, nose, and upper respiratory ailments, skin irritation, and gastrointestinal infections (Herwaldt *et al.*, 1991; Levesque *et al.*, 1993).

Several best management practices are used throughout the U.S. for stormwater treatment, however, little research has been performed to evaluate their effectiveness for the removal of microorganisms. In this study, indicators and surrogates of microbial pathogens (total and fecal coliform bacteria, MS2 coliphage, and a 3 μm fluorescent bead representing the pathogenic protozoa, *Cryptosporidium parvum*) were used to challenge sand filtration, wet detention, and alum coagulation treatment systems using simulated storm events.

Significant ($p \leq 0.05$) reductions in total and fecal coliform bacteria, MS2, and bead concentrations were observed between inflow and outflow samples for each of the three stormwater treatment systems. On a few occasions, however, greater concentrations

of total coliform bacteria, turbidity, and total suspended solids were found in outflow samples than at the inflow. Using flow-weighted sampling techniques, estimates of load reductions for microbial indicators were determined for each treatment system. Removal efficiencies with the sand filter ranged from 59.4 to 99.5% while wet detention reductions ranged from -284.5 to 99.5% and alum treatment ranged from -3233.3 to 99.9995%. Removal efficiencies for beads were consistently greater than 90% while MS2 coliphage removal was consistently greater than 80% for all three treatment systems. Removal efficiencies for total and fecal coliform bacteria varied widely with total coliform removal values consistently less than 70% while fecal coliform values ranged from 65 to 100%.

Overall, alum coagulation (dose = 10 mg/L) provided greatest removal efficiencies for total and fecal coliform bacteria, MS2 coliphage, and turbidity under semi-controlled conditions using jar tests. Removal efficiencies using sand filtration were generally high for turbidity, MS2, and beads but not for total or fecal coliforms. Wet detention using the current regulatory standard of a 5-day bleed-down period provided consistently high removal efficiencies for fecal coliform bacteria, MS2 and beads and had the greatest TSS removal of the three treatment systems. Recommendations for optimizing current stormwater treatment systems for the removal of microorganisms are addressed and include the use of a multiple treatment (treatment train) approach.

CHAPTER ONE

INTRODUCTION

Water resources throughout the U.S. have suffered decades of pollution and overuse. As a result, the restoration and improvement of water quality in aquatic ecosystems are important issues in states like Florida where various human activities vie for limited water resources. The passage of the Clean Water Act and the Safe Drinking Water Act in the early 1970s and recent federal legislation (National Pollutant Discharge Elimination System - NPDES) regulating pollutant discharges to surface waters have resulted in significant improvements in water quality in certain areas of the U.S., including Tampa Bay, Florida. Historically, municipal wastewater treatment plants were one of the leading sources of surface water pollution and often discharged a number of contaminants including heavy metals, excess nitrogen and phosphorus loads, suspended solids, and microbial pathogens to rivers, lakes, and coastal waters.

Technological advances in wastewater treatment have improved dramatically in the last decade which has led to higher quality effluent from many domestic wastewater treatment facilities (Smith and Smith, 1982). Despite this significant point-source reduction in pollutant loading to surface waters, other sources still contribute excess

pollution. Previous research has determined that a large proportion of pollutants originate from non-point sources such as urban and agricultural stormwater runoff (U.S. EPA, 1983). A follow-up report by the U.S. EPA in 1984 determined that nearly 90% of fecal coliform pollution to surface waters originates from stormwater. In Tampa Bay, for example, several tributaries which receive agricultural, industrial, and urban runoff exhibit consistent, elevated total and fecal coliform bacteria concentrations which often exceed state standards for shellfish harvesting and recreational exposure (Hillsborough County Environmental Protection Commission, 1996). Based on state water quality standards, 45% of these tributaries did not meet their intended use for recreation and the propagation and maintenance of a healthy, well-balanced population of fish and wildlife (FDEP, 1996).

A number of microbial pathogens including bacteria (*Escherichia coli*, *Pseudomonas aeruginosa, Salmonella* sp., *Shigella, Staphylococcus aureus, Vibrio*), viruses (poliovirus, Echovirus, Coxsackie virus, hepatitis A), and protozoa (*Giardia*, *Cryptosporidium*) can be present in stormwater and have been implicated in waterborne disease outbreaks (Olivieri *et al.*, 1977; Hayes *et al.*, 1987; Bemiss *et al.*, 1989; Koenig *et al.*, 1991; Enriquez *et al.*, 1992; Kebabjian, 1994). A massive *Cryptosporidium* outbreak in Milwaukee in 1992 that affected over 400,000 individuals and a smaller outbreak of hepatitis A in Florida (Vonstille *et al.*, 1993) were caused by contaminated stormwater runoff. In urbanized areas, contaminated stormwater can impact recreational beaches in both marine and freshwater environments and can cause a number of bathing-related illnesses including eye, ear, nose, and upper respiratory ailments, skin irritation, and

gastrointestinal infections (Herwaldt *et al.*, 1991; Levesque *et al.*, 1993). Historical reports of illnesses among bathers and surfers in Santa Monica Bay, California led to the design of a large-scale epidemiological study to assess the impacts of contaminated stormwater effluent on recreational beaches. Haile *et al.* (1996) found elevated concentrations of bacterial indicators at several beaches and determined that bathers who swam within 25 m of a major storm drain outfall had a significantly greater risk of upper respiratory and gastrointestinal infection than bathers swimming more than 400 m from the same outfall.

Microbial pathogens in stormwater runoff can originate from both human and natural sources. Examples of human activities which contribute to microbial pathogen loading include agriculture (cattle pastures, feedlots, dairies), urban/residential development (septic tanks with inadequate separation distances or which leach into overlying flood waters, poor pet waste management, leaky or cross connected sewer lines), and industry (disposal of inadequately treated septage and biosolids). Natural sources include birds (Levesque *et al.*, 1993), mammals (Sherer *et al.*, 1992), and indigenous soil bacteria. In either case, microbes can enter rivers and lakes and contaminate potable water supplies (Rose *et al.*, 1988). Closures of shellfish harvesting areas in many coastal regions of Florida are correlated with excessive rainfall events since pathogenic microorganisms like bacteria, viruses, and protozoa can be mobilized and transported in overland runoff to coastal estuaries (Hesselman *et al.*, 1992). Oysters and other harvestable shellfish are known to concentrate pathogenic microorganisms,

including *Cryptosporidium*, since these filter feeding organisms routinely concentrate particulate matter during feeding and respiratory activity (Fayer *et al.*, 1997).

The economic impacts of waterborne diseases can be significant. Recent estimates of the annual cost of foodborne/waterborne disease outbreaks in the U.S. range from $500 million to over $2 billion per year (Steahr and Roberts, 1993) (Table 1). In 1996, more than 2,500 beach closings and advisories were posted throughout the coastal U.S. which can have an enormous economic impact on revenues from tourism (U.S. EPA, 1998). In Florida, approximately 55% of the beach closings between 1994 and 1996 were attributed to stormwater runoff (FDEP, 1996). Concurrently, the annual risk of contracting a waterborne infection is estimated to range between 1:100 to 1:10,000 which clearly demonstrates the need to investigate potential sources and methods of removal of microbial pathogens from surface waters from both a human health and economic perspective (Table 1).

Microbial contamination of stormwater is often a short-term impact on receiving waterbodies since many pathogenic microorganisms are vulnerable to die-off or inactivation in the environment (Hvitved-Jacobsen, 1986). However, the impact can be significant if the risk of human exposure is elevated, such as in cases where stormwater outfalls are adjacent to recreational beaches or shellfish harvesting areas. A number of surveys to identify the potential sources and relationships of microbial indicators and pathogens to environmental factors have been performed in urbanized coastal areas in Florida (Charlotte Harbor [Lipp *et al.*, in prep.], Sarasota Bay [Rose and Lipp, 1997], Biscayne Bay [McCorquodale and Burney, 1993]), Massachusetts (Buzzard's Bay

[Heufelder 1988]), and California (Santa Monica Bay [Gold *et al.*, 1990; Haile *et al.*, 1996]) and have developed extremely useful regional databases. The next logical step in the process to remediate watersheds which generate contaminated runoff is an assessment of current stormwater treatment technologies for the removal of disease-causing organisms.

Table 1. Patients discharged from hospitals by category of foodborne/waterborne disease in the U.S., 1987-1990 (Steahr and Roberts, 1993).

Agent	**Average annual number of patients discharged with a diagnosed disease**	**Average annual hospital costs**
Salmonella	15,408	$79,623,000
Shigella	5,344	$16,964,000
Protozoa	6,124	$34,014,000
Hepatitis A virus	12,403	$76,119,000
Ill defined	31,431	$141,878,000
Unspecified gastroenteritis	530,689	$1,971,039,000
Total:	601,399	$2,319,637,000

Best Management Practices for the Treatment of Stormwater

Stormwater management involves two general methods for reducing pollutants to the aquatic environment. These methods include preventive (nonstructural) and control (structural) measures. Preventive measures include source reduction practices, land use management practices, animal waste collection, curb elimination, debris removal,

exposure reduction, landscaping and lawn maintenance controls, minimization of pollutants, parking lot and street cleaning, streambank stabilization, creation of vegetated buffer zones, sanitary waste management, and education programs (Scheuler, 1987; Urbonas, 1994). Structural control methods include dry detention basins, infiltration or exfiltration devices, chemical coagulation and sedimentation, ozone disinfection (Greene, 1992), oil and grease trap devices, porous pavement, sand filters (Shaver, 1992), filter strips, grassed swales, wet detention ponds, and constructed (Kehoe, 1993; Rushton and Dye, 1993) and natural wetlands (Carr, 1993; Kehoe *et al.*, 1994). The focus of this study involves the use of common (wet detention, sand filtration) and less common (alum treatment) structural control measures used in Florida which are described in greater detail below.

Sand Filtration

Sand filtration is a commonly-used technique for removing contaminants in both the water and wastewater treatment industries (Ellis, 1984). During the last decade, the use of sand filters has also become an accepted treatment technique for stormwater, particularly in situations where high property values reduce the cost-effectiveness of other best management practices (BMPs), such as wet detention ponds, which require large surface areas for construction (Shaver, 1992; Urbonas, 1994).

Typical sand filter systems are constructed either in underground trenches or in above-ground, pre-cast concrete boxes. Treatment occurs as pollutants and suspended

particles are adsorbed or trapped by smaller sand particles. Generally, the removal of sediments and trace metals is greater than the removal of soluble pollutants like nitrogen and phosphorus since the filter functions primarily by mechanical straining. The use of sand filters has several advantages including aesthetics (can be installed underground and out of public view) and the ability to provide consistent pollutant removal when properly maintained.

Wet Detention

Wet detention is one of the most common stormwater treatment methods in Florida and other states in the U.S. A typical wet detention system involves routing stormwater from an adjacent contributing basin to a pond which has been either excavated to depths below the seasonal high groundwater table or where a confining layer of soil or clay holds a permanent pool of water. The detention pond slows the flow of stormwater, reduces downstream flooding, and enhances the removal of many pollutants through several mechanisms including gravitational settling, biological uptake by plants and microorganisms, and chemical and photochemical degradation. Treated stormwater is then slowly discharged at an outlet structure after being detained within the pond over a number of hours or days. Many wet detention ponds are planted with aquatic vegetation to aid in the physical and biological removal of pollutants and often serve as aesthetic enhancements in urban settings where they attract various waterfowl and wildlife. The removal of pathogens can occur through sedimentation, filtration, natural die-off, and UV

degradation while heavy metals can be removed by adsorption and complexation with organic matter or, for particulate-associated metals, through gravitational settling.

Alum Coagulation

Alum has been used as a coagulant for the treatment of drinking water and wastewater in the U.S. for several decades. More recently, alum (in the form of aluminum sulfate or $Al_2(SO_4)_3 \cdot 18H_2O$) has been used in the treatment of stormwater (Harper, 1990, Price and Yonge, 1998) and lake systems (Bulson *et al.*, 1984) to remove a number of pollutants. When used under optimal conditions and dosing rates, alum can form non-toxic precipitates of $Al(PO_4)$ and $Al(OH_3)$ which can attract and bind with suspended solids, heavy metals, nutrients (nitrogen and phosphorus) and microorganisms (bacteria, viruses, protozoa). As larger particles are created with alum, these materials become heavier and settle out as flocculent material.

When colloidal matter such as virus and bacteria particles are treated with metal salts such as aluminum sulfate, the metal salts act as primary coagulants. Positively-charged metal ions bond with negatively-charged colloidal particles which results in charge neutralization. The particles then repel each other less strongly and tend to coagulate or bind into larger particles. Decreases in pH to about 3 to 5 often occur as a result of adding metal salts and may result in more effective treatment. When pH is subsequently adjusted to between 8 and 11 the soluble metal salts and other soluble metals in the solution form insoluble hydroxide particles that are large enough to settle. These hydroxide particles

coprecipitate other contaminants in the solution, including oil particles and other colloids. Addition of a coagulant such as alum after pH adjustment and formation of hydroxide particles usually results in the rapid growth of large flocculant material which sweeps smaller particles out of solution and quickly settles. Bench scale jar testing is commonly used to determine the most effective combination and dose of chemicals, optimum pH, and type of coagulant for maximum pollutant removal.

Indicator Microorganisms Used as Surrogates for Microbial Pathogens

A number of microbial indicators have been used as both tracers and surrogates of microbial pathogens in both surface water and groundwater studies (Harvey, 1997). Since the use of disease-causing organisms in environmental studies can pose significant health risks, indicators including various bacteria, coliphage, and conservative tracers are often used in their place to evaluate water treatment systems. Three groups of indicators were used in this study including total and fecal coliform bacteria, MS2 coliphage, and a fluorescent bead which represented *Cryptosporidium* oocysts.

Total and fecal coliforms, widely-used indicators of fecal pollution, represented the bacterial community. The total coliform group includes both the fecal coliform group as well as a number of other bacterial species, some of which are commonly found in soils (e.g., *Enterobacter cloacae*). Fecal coliforms are non-spore forming, facultatively anaerobic bacteria which are common flora of the human gut. When found in high

concentrations, fecal coliform bacteria can indicate the presence of disease-causing pathogens.

The fecal coliform standard for recreational waters was established in 1968 with a maximum concentration of 200 colony forming units (cfu) per 100 ml of sample water expressed as the geometric mean of at least 5 consecutive samples. Concentrations of fecal coliform bacteria in runoff in the Pacific Northwest have been found to range between approximately 7.1×10^7 and 3.6×10^{10} cfu/100 ml (Horner *et al.*, 1994). In the Tampa Bay area, total and fecal coliform concentrations can often exceed 10^4cfu/100 ml in a number of tributaries which receive polluted runoff (Hillsborough County Environmental Protection Commission, 1998). Other estimates indicate levels of coliform bacteria in stormwater runoff to be two to five times greater than in secondarily-treated wastewater (Bastian, 1986).

Both state and federal regulatory agencies normally require monitoring for coliform bacteria (indicators of human or animal fecal contamination) in local surface water supplies, wastewater treatment effluent, and NPDES discharges. However, the establishment of maximum contaminant levels (MCL's) has been extremely controversial since the coliform bacteria indicator is often incapable of predicting the presence of waterborne pathogens in surface waters (Rose *et al.*, 1988). In many cases, the abundance and distribution of human pathogens including representative species of bacteria, protozoa, and viruses are poorly correlated with the presence of coliform bacteria since their sources and ability to survive in the environment differ. Testing for all potential pathogens would be economically unfeasible, however, and so methods for

using other indicator organisms (e.g., coliphage, *Clostridium perfringens*) have been suggested (Payment and Franco, 1993).

Coliphages are bacteriophages (viruses) which infect and replicate in coliform bacteria. Previous studies have shown coliphage to be correlated with the presence of total and fecal coliforms (Wentsel *et al.*, 1982) and, as a result, has been used as a tracer in drinking water treatment efficiency (Payment and Franco, 1993), groundwater recharge (Powelson and Gerba, 1994), and marine pollution (Lucena *et al.*, 1994) studies. Due to its similarity in size (25 nm diameter virion) and biochemical characteristics (icosohedral, single stranded RNA), MS2 coliphage was used as a surrogate for pathogenic viruses and served as a model for viral transport and removal. The infective dose of many viruses including rotavirus can be as low as 10 to 100 infectious viral particles (U.S. Food and Drug Administration, 1992).

Fluorescent beads (microspheres) were used as a surrogate for oocysts of, *Cryptosporidium parvum*, a pathogenic protozoa which has been implicated in numerous waterborne disease outbreaks throughout the world (Hayes *et al.*, 1989; Joseph *et al.*, 1991; Richardson *et al.*, 1991). Previous research by Li *et al.* (1997) found significant correlations between latex microspheres and *Cryptosporidium* oocyst concentrations in microbial removal experiments validating the use of these beads as a surrogate for protozoa.

Infection by *Cryptosporidium* can cause severe gastrointestinal disorders (including diarrhea and nausea) and even death in the elderly and immunocompromised. Intestinal cryptosporidiosis is self-limiting in most healthy individuals, with watery

diarrhea lasting 2 to 4 days, however, in some outbreaks at day-care facilities, diarrhea has lasted 1 to 4 weeks (U.S. Food and Drug Administration, 1992). Currently, there is no known effective drug for the treatment of cryptosporidiosis. The oocyst phase of *C. parvum* is highly resistant to environmental conditions and is relatively unaffected by chlorine treatment in drinking water disinfection processes. DuPont *et al.* (1995) reported that the infectious doses (ID_{50}) for *C. parvum* in healthy individuals is approximately 132 oocysts, however, doses as low as 10 oocysts are capable of causing illness. A number of hosts have been identified for this parasite including several mammalian and avian species (Gomez *et al.*, 1992; Kaminjolo *et al.*, 1993; Smith *et al.*, 1993).

Stormwater Best Management Practices for Removing Microorganisms

Research evaluating pollutant removal efficiencies for various stormwater treatment systems and best management practices (BMPs) have focused primarily on physical and chemical contaminants such as total suspended solids, nutrients, and metals (Urbonas, 1994). Relatively little research has been performed to investigate the efficiencies of BMPs for the removal of microbial pathogens (Horner *et al.*, 1994; O'Shea and Field, 1992). These organisms are known to be present in stormwater (Qureshi and Dutka; 1979) and can pose serious health risks to certain high-risk groups including elderly and immunocompromised individuals. Transport of bacteria and viruses has been shown to occur over long distances in a variety of environments (Lucena *et al.*, 1994; McFeters *et al.*, 1993) while the potential for exposure to microbial pathogens in

stormwater can occur through several mechanisms. Direct exposure can occur through accidental ingestion of contaminated runoff during flooding conditions or while bathing at recreational beaches that receive stormwater discharges. Indirect exposure can result from ingestion of contaminated foods (vegetables, shellfish) or water supplies (surface and groundwater) which have been exposed to or recharged with contaminated runoff.

To date, little information exists as to the effectiveness of current regulatory criteria for stormwater treatment systems in the removal of human microbial pathogens. This information will become more critical as several alternative sources of drinking water are developed in Florida including the diversion and storage of stormwater runoff and treated wastewater to recharge depleted aquifers, rivers, and lakes (Bishop, 1992; SWFWMD, 1995). In Florida, state regulations (Chapter 17-40, Florida Administrative Code [F.A.C.]) recommend that stormwater treatment systems achieve an annual average of 80 percent pollutant load reduction. This standard is based primarily on the removal of heavy metals and nutrients (nitrogen and phosphorous) and does not specifically address microbial pathogens. Although standards for bacterial indicators (total and fecal coliforms) exist for surface waters, there are no maximum contaminant levels for a wide range of specific waterborne pathogens including other species of bacteria (*Clostridium*, *E. coli*, *Salmonella*, *Klebsiella*), viruses (hepatitis a, Coxsackie, rotavirus), and protozoa (*Cryptosporidium*, *Giardia*) that can cause human disease.

This study was conducted to determine the removal efficiencies for bacteria, viruses, and protozoa using three different stormwater treatment technologies used in Florida and other parts of the U.S.: an above-ground sand filter, a wet detention pond, and

alum coagulation. Indicator organisms (total and fecal coliforms, coliphage, and fluorescent beads representing *Cryptosporidium* oocysts) were used as surrogates for the broad spectrum of human pathogens which may be present in urban stormwater. Removal efficiencies were calculated based on comparisons of total inflow and outflow loads of seeded microbial indicators. Effluent concentrations for total and fecal coliform bacteria and turbidity were compared with the State of Florida's Surface Water Quality Standards (Chapter 62-302) to determine the extent to which each BMP could treat stormwater to meet current regulatory goals.

CHAPTER TWO

METHODS AND MATERIALS

Site Locations and Site-Specific Sampling Protocol

Sand Filtration

The sand filter stormwater treatment facility used in this study was constructed in 1991 to treat runoff from a 2.73 ha (6.75 ac) light commercial/urban drainage basin in the coastal community of Madeira Beach, Florida (Fig. 1). Prior to the construction of this facility (141st Ave. Pump Station Stormwater Filter), stormwater runoff from the contributing basin received no treatment and discharged to a state-designated aquatic preserve (Boca Ciega Bay). After construction, stormwater was diverted to a 51,566 L (13,642 gal) holding tank below a small building which houses two high speed hydraulic pumps (Fig. 2). The holding tank fills during single, large rain events or over a period of small, successive rain events. When the water level in the holding tank reaches a height of approximately 1.5 m, a floating switch is triggered and the collected water is pumped vertically, approximately 3 m, into one of three, rectangular sand filter chambers.

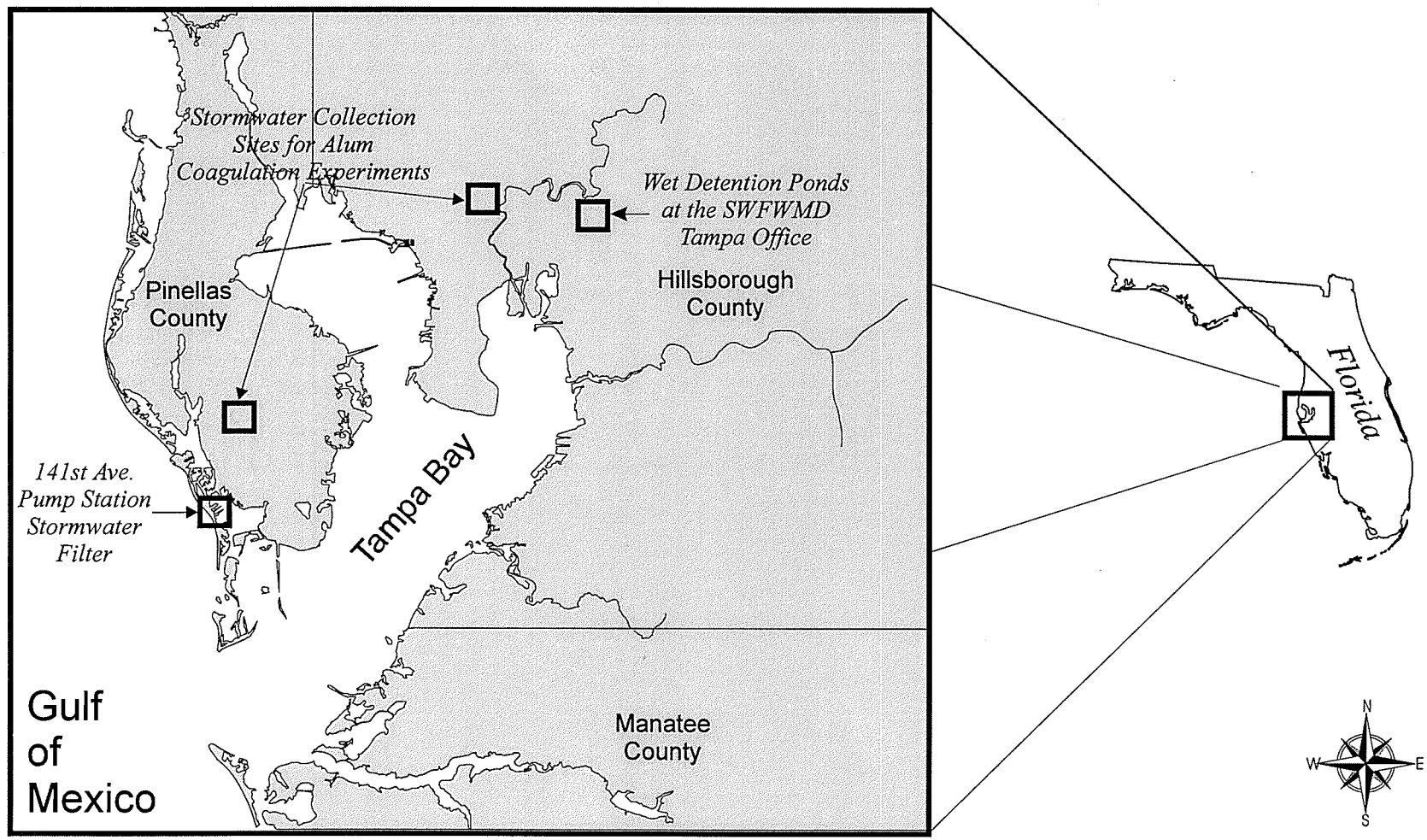


Fig. 1. Location of the stormwater treatment systems and collection sites in the Tampa Bay area.

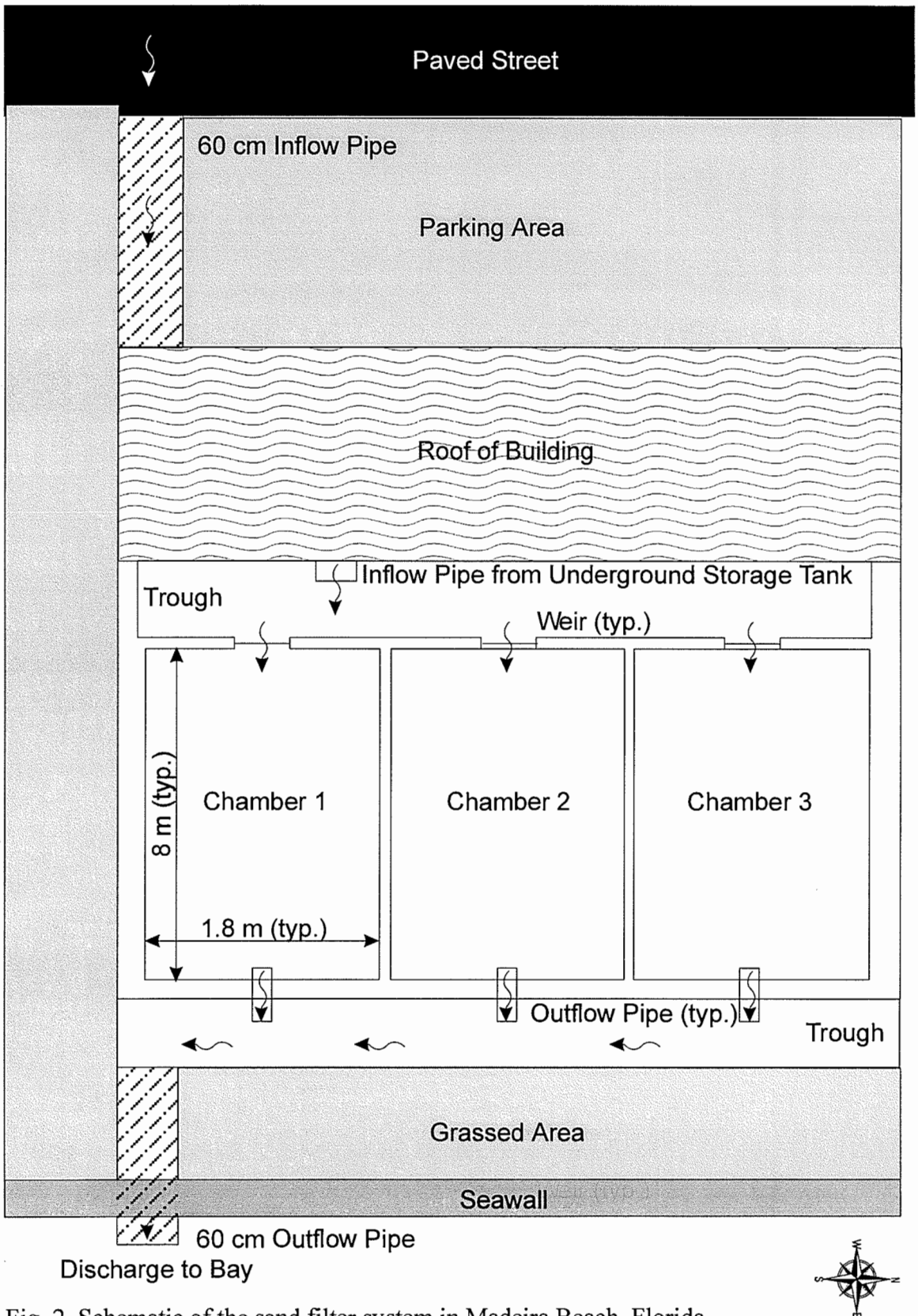


Fig. 2. Schematic of the sand filter system in Madeira Beach, Florida.

Treatment occurs as the stormwater percolates through a filter composed of approximately 1 m of clean creek gravel and sand. The design specifications called for a sand media with an effective size (D_{20}) of approximately 0.42 mm and vertical permeability rate (K) of 39.6 m/day. Field measurements determined that the actual permeability rate ranged between 90.0 to 120.0 m/day. After traveling through a gravel underdrain and perforated drainage pipe, the treated stormwater is discharged to an adjacent residential canal which is tidally connected to Boca Ciega Bay.

Three seeded trials were performed between September 1995 and November 1996. During each sampling event, approximately 5,160 L (1,365 gal.) of residual stormwater which had accumulated in the holding tank were pumped onto one of the three sand filters. Adjustable weirs were installed so that stormwater could be diverted to any or all three of the filters. Prior to the first trial (Trial 1), the northernmost filter bed had been isolated from stormwater inflows and was challenged in an unsaturated condition. The filter was then left open and challenged several months later (Trial 2) in a saturated condition (a few hours after being used to treat unseeded stormwater). The middle bed was later challenged once (Trial 3) in a saturated condition.

Titers of MS2 coliphage (approximately 5×10^{11} virions) and 3.0 μm fluorescent beads (1×10^{10} beads) were simultaneously mixed with raw stormwater to reach final inflow concentrations of 9.69×10^{5}/ml and 1.94×10^{3}/ml, respectively. A one-log greater titer of MS2 was used during Trial 2. The concentrations of beads and viruses used for seeding experiments were adjusted to ensure that adequate numbers of each surrogate could be recovered for analysis using a relatively small outflow sample volume. Total

and fecal coliforms were not seeded since background concentrations were sufficiently elevated for influent-effluent comparisons.

Total coliforms, fecal coliforms, MS2 coliphage, fluorescent beads, and turbidity were all measured in the inflow (three replicate grab samples) and then at ten evenly-spaced intervals during the drawdown period in the outflow. Total suspended solids (TSS) were only measured during Trial 3. Total inflow loads were calculated based on the total volume pumped into each chamber times the average concentration of each parameter. Outflow loads were calculated by summing each of the ten outflow sample concentrations times the corresponding volume discharged between sampling events. Temperature, pH, and conductivity were measured in the holding tank and also during outflow sampling.

A number of factors influence the survival of bacteria in water including predation, dessication, lack of nutrients, suboptimal pH and temperature, and metal toxicity. Since a variety of heavy metals are found at elevated concentrations in stormwater, the effect of metal toxicity may play a significant role in regulating bacterial populations. Toxicity effects were assessed using an *E. coli* model which has been used in a number of laboratory procedures to assess metal toxicity in environmental samples (Kong *et al.*, 1995; Jung *et al.*, 1996).

The MetPad© test kit (Group 206 Technologies, Inc.) was used during Trials 2 and 3 to determine the toxic effects of elevated heavy metal concentrations on *E. coli*. Metal toxicity for bacteria was measured using an enzyme inhibition test developed by Bitton *et al.* (1992, 1994). The test measures the activity of the enzyme ß-galactosidase

as it hydrolzyes lactose to glucose and galactose and has been shown to be sensitive specifically to heavy metals (Jung et al., 1996). The assay was performed by mixing freeze-dried *E. coli* with a moderately hard water diluent to produce a bacterial suspension. A 0.1 ml aliquot of bacterial suspension was then mixed with 0.9 ml of stormwater in a sterile glass test tube and incubated for 1 hour at 35°C. A 0.2 ml aliquot of the suspension was then dispensed into a well of a microplate followed by 0.1 ml of an enzyme substrate. The microplate was then incubated at 35°C until a purple color developed in the sample (approximately 15 minutes). The intensity of color development indicated enzyme (ß-galactosidase) activity and was inversely proportional to the sample toxicity.

Toxicity was expressed as the degree of inhibition of enzyme activity measured by absorbance values (i.e., a decrease in optical density) with the negative control representing 0% inhibition. Absorbance was measured using a microplate reader (Biotek© Instruments) at 490 nm. Five dilutions (full strength to 10^{-4}) were assayed for the inflow (a composite of the three inflow samples) and the outflow (a composite of the ten outflow samples). Duplicate assays were performed using undiluted aliquots from each of the ten outflow samples. Triplicate positive and negative controls and blanks were also performed for each trial. Separate samples were also taken for metals analysis from the raw stormwater and the ten consecutive outflow samples and were analyzed by atomic absorption (APHA, 1992) for Zn, Cd, Cu, Cr, Ni, and Pb expressed as µg/L (ppb). Correlation coefficients were calculated between optical densities and individual metal concentrations to determine which metal had the most toxic effect on *E. coli.*

Wet Detention

The Southwest Florida Water Management District (SWFWMD) Tampa Service Office Experimental Stormwater Treatment Ponds were constructed in the summer of 1990 for a project to evaluate pollutant removal efficiencies of chemical constituents using conventional wet detention methods (Cunningham, 1993). Two 0.06 ha (0.15 ac) ponds with depths of 1 m (3.3 ft) and 2.75 m (9.0 ft) were constructed to meet Chapter 40D-4, Basis of Review, Florida Administrative Code guidelines to compare the effect of pond depth on stormwater treatment efficiency. Both ponds had surface areas of approximately 511 m^2 (5,500 ft^2) (Fig. 3), and treatment volumes (defined as the storage volume up to 18 inches above the invert of the bleed-down device) of approximately 252,000 L (65,500 gal) (Fig. 4). This volume represented the same amount of runoff produced from a 2.5 cm (1.0 in) storm over a 0.97 ha (2.39 ac) impervious watershed. Both ponds had a shallow, littoral shelf comprising approximately 50% of their total surface area which was colonized by submerged vegetation including maidencane (*Panicum hemitomon*) (Fig. 5). However, the extent of vegetated bottom in the deep pond was limited to only the shallow littoral shelf (approximately 50% coverage) and the shallow pond had approximately 95% of its benthos colonized by aquatic plants.

Both ponds were constructed by excavating soils to create depressional areas which were then surrounded by an approximately 1 m high containment berm (Fig. 5). The normal pool volumes were 362,800 L (94,328 gal) and 443,046 L (115,192 gal), respectively, for the shallow and deep ponds. The underlying soils in the area were

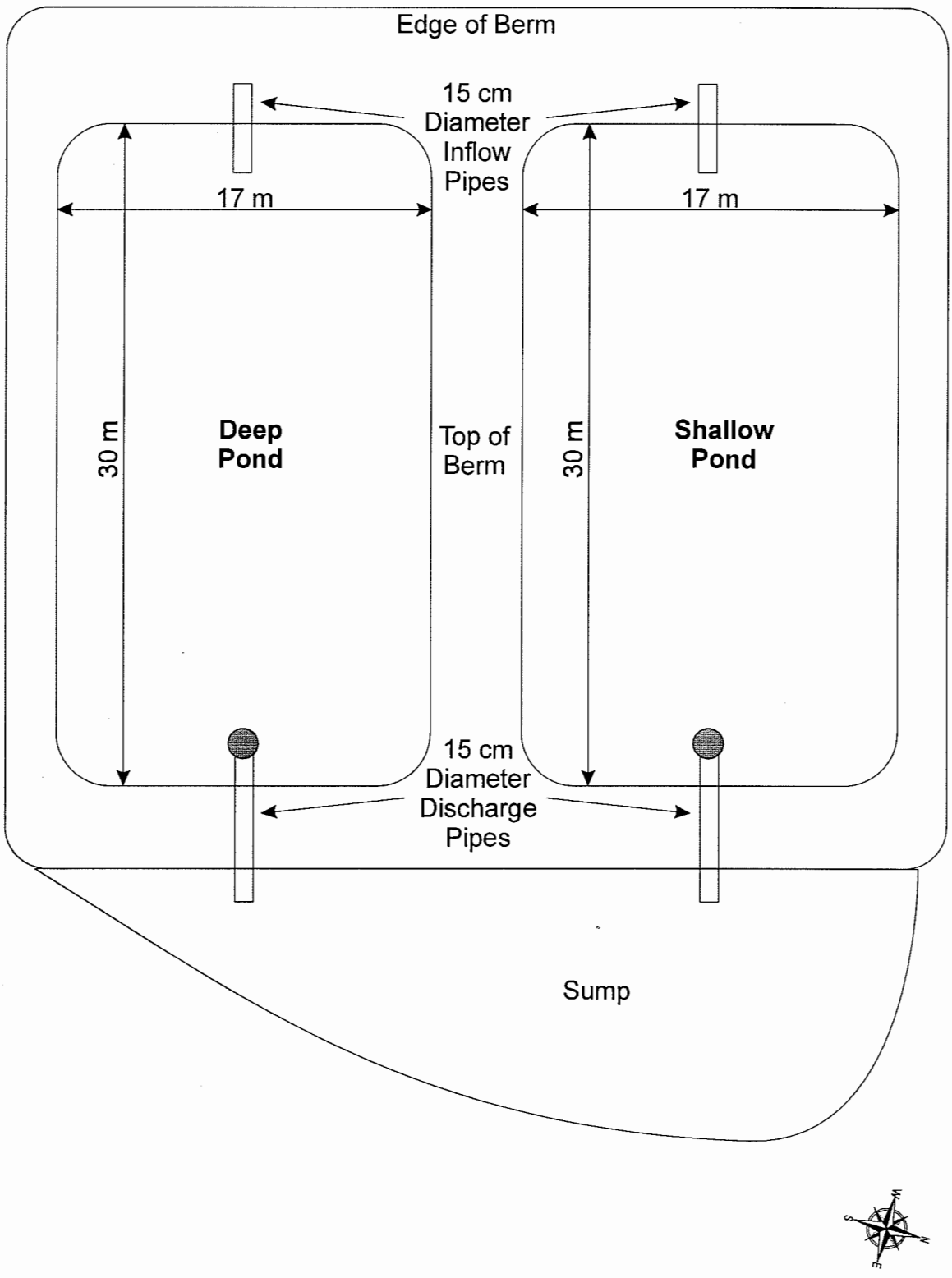


Fig. 3. Dimensions of the wet detention ponds at the SWFWMD Tampa Service Office.

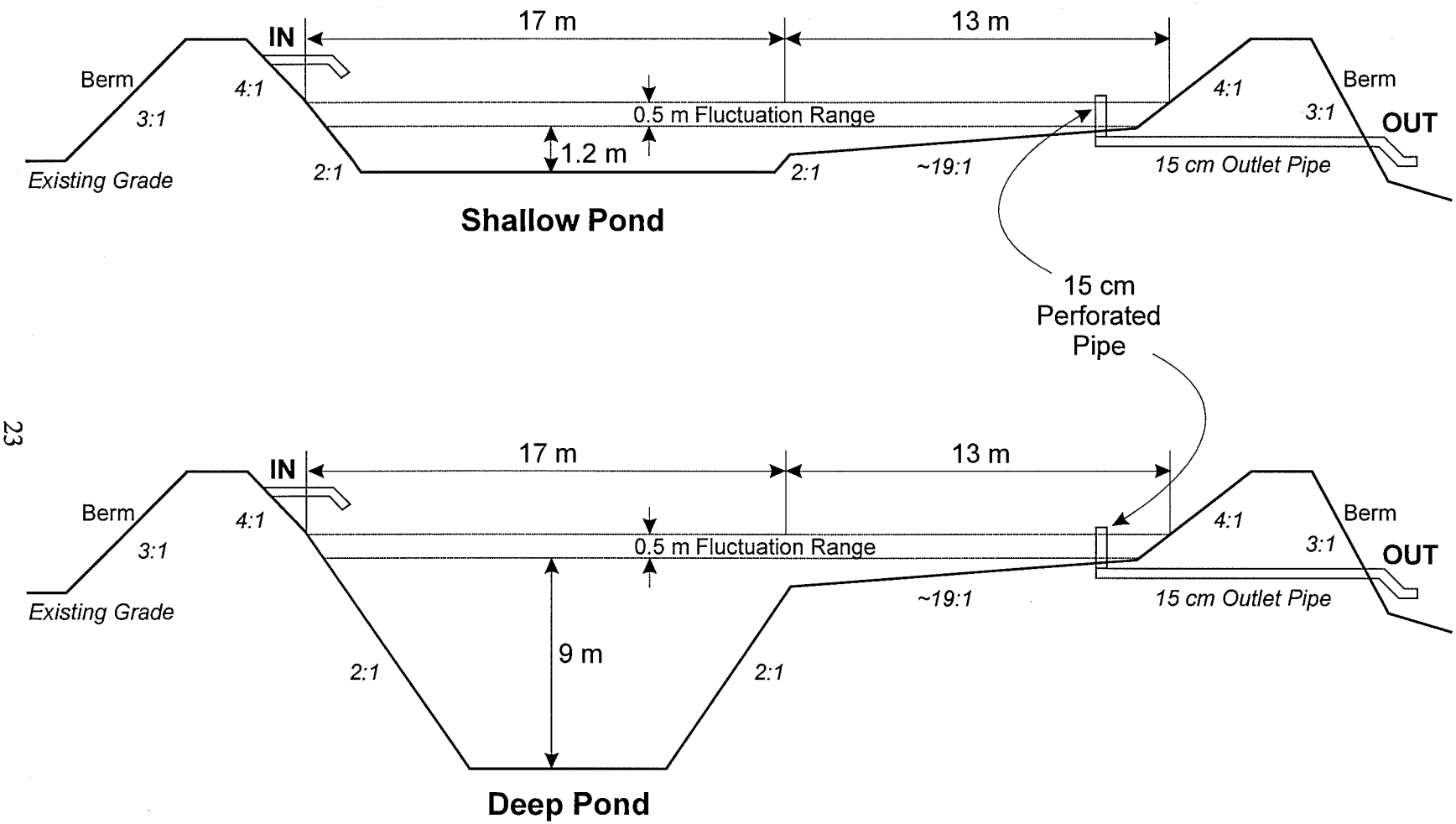


Fig. 4. Cross-section diagrams of wet detention ponds.

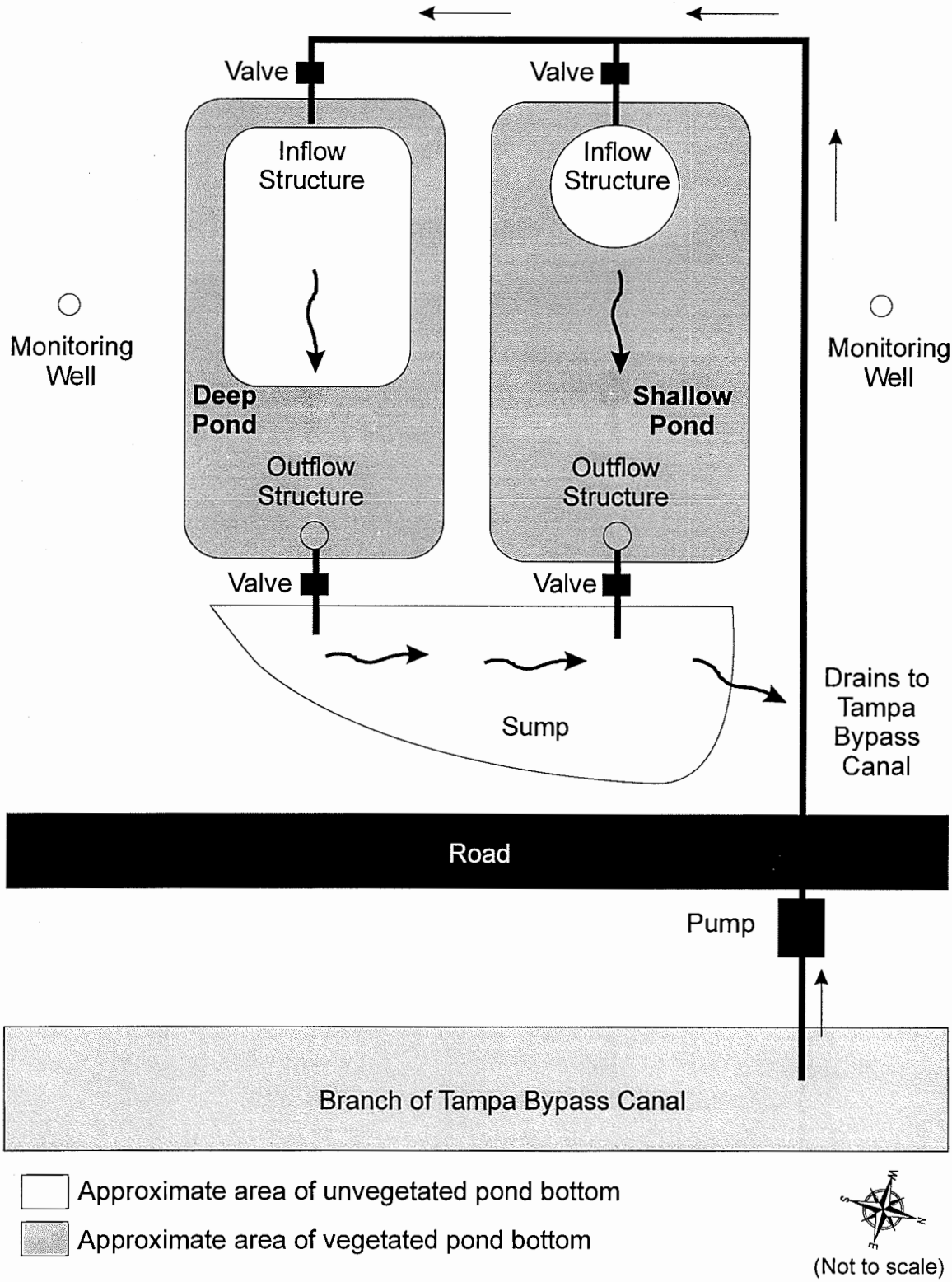


Fig. 5. Schematic of the wet detention ponds at the SWFWMD Tampa Service Office.

described by Cunningham (1993) and included mixed sand and clay overburden from the surface to two feet followed by gray sandy clay from two to seven feet and stiff green clay from seven to eleven feet. These soil types are relatively impervious and were expected to limit any subsurface movement of water between the two adjacent ponds and to the locally shallow aquifer system.

In order to simulate various storm events, water was pumped from the Tampa Bypass Canal into the western end of each pond through a series of 10 cm diameter underground PVC pipes. This section of the canal received industrial and light commercial runoff and was also adjacent to heavily forested upland and wetland communities harboring various birds, reptiles, and mammals. Impeller type flowmeters (Water Specialties Corp.) were used to measure flow rates and the total volume of water pumped into each pond. Flow to the ponds ranged between 400 to 1200 L/min and was controlled using Fast Co. gate valves. Vertical 15 cm diameter PVC pipes were used to drain the ponds at the outfall. The pipes traveled down approximately one foot below the pond bottom and then horizontally through the surrounding berm to an adjustable valve controlling the outflow rate to a vegetated outlet sump.

Two separate trials were performed representing different treatment or bleed down periods. In the first trial, a 14-day residence time was achieved by adjusting the outflow valve so that the initial outflow rate was limited to approximately 13 L/min. At this bleed down rate, approximately 50% of each pond's volume was discharged after 7 days (Fig. 6). In the second trial, a 5-day residence time was simulated by increasing the discharge flow rate to 36 L/min. Approximately 50% of each pond's volume was discharged after

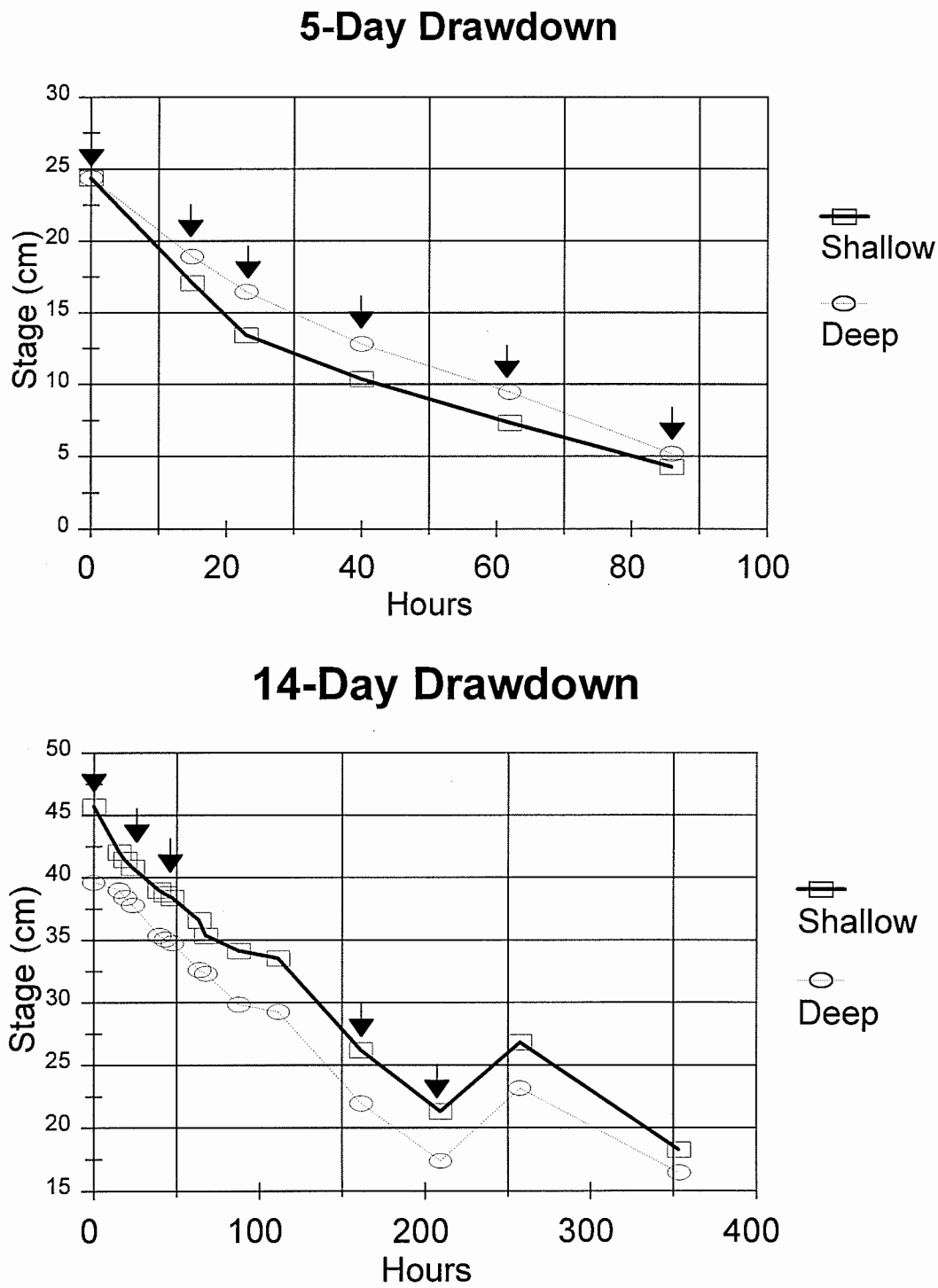


Fig. 6. Changes in water levels during the 5-day (top) and 14-day (bottom) storm simulation trials at the SWFWMD wet detention ponds in Tampa, Florida. Arrows represent sample collection times.

2.5 days. Outflow rates were measured by the number of liters which filled a graduated 16 L capacity bucket in one minute.

The first simulation was performed for the 14-day treatment or bleed-down period and 212,597 L (56,150 gal) and 216,293 L (57,126 gal) of stormwater were pumped into the shallow and deep ponds, respectively. This represented a 2.5 cm (1.0 in) storm event. MS2 coliphage and 3.0μm fluorescent beads were seeded into each pond at loads of approximately 1.5×10^{12} virions and 4×10^{10} beads, respectively. Due to the extended detention time, three samples were collected and composited during each 24 hour sampling period for the first 5 days. Single samples were then collected on subsequent daily sampling events. Samples were stored on ice until the final sample of each composite group was collected.

During the second simulated storm event, a 5-day treatment period was carried out based on an approximately 0.5 inch storm event. A total of 107,415 L (28,370 gal) and 129,807 L (34,284 gal) were pumped into the shallow and deep ponds, respectively. Each pond was seeded with approximately 1.2×10^{13} MS2 virions and 4×10^{10} beads. Grab samples were collected at five, evenly-spaced intervals following the drawdown period (Fig. 6). Discrete (uncomposited) samples were collected at the outfall since a shorter treatment period was being measured. Total and fecal coliforms were not seeded during either storm event simulation since pilot sampling in the source waters (canal) indicated concentrations were sufficiently elevated for influent-effluent comparisons.

Physicochemical parameters including temperature, pH, turbidity, TSS, and conductivity were measured in each pond prior to pumping and also concurrently with

each grab sample. As water was pumped into each pond, unseeded grab samples were taken in the canal to determine background coliphage concentrations. Concentrated suspensions of fluorescent beads and MS2 coliphage were mixed with approximately 30 L of water taken from the canal and then seeded with the inflow water entering each pond using a peristaltic pump and sterile polyethylene tubing at a rate of approximately 180 ml/min.

Additional grab samples were taken in the deepest zones of the ponds before water began to discharge from the outlet structure to determine initial concentrations of total and fecal coliforms and coliphage during the 14-day treatment trial. Grab samples were also taken at approximately 0.5 m depth at the deep end of each pond prior to seeding during the 5-day treatment trial and analyzed for total and fecal coliforms, coliphage, and fluorescent beads to assess the extent of sediment resuspension as water was pumped into each pond. Sediment samples were taken and analyzed for total and fecal coliforms at both the deep end and the shallow zone near the outfall structure to determine sediment concentrations.

To confirm that seeded surface waters were not leaching from the ponds, groundwater samples were taken from monitoring wells located approximately 10 m laterally from each pond before and then 7 days after the ponds were filled during the first seeded trial (Fig. 5). Based on estimates of hydraulic conductivity for the surrounding soils (4.2 m/day, D. Dewitt, pers. comm.), 7 days was considered an adequate amount of time to detect virus and bead tracer movement from the ponds. The presence of either

MS2 coliphage or fluorescent beads in these wells would have indicated subsurface flow (and water loss) from the ponds to the adjacent surficial aquifer.

Alum Coagulation

Stormwater was collected from drainage ditches which collected runoff from two heavily urbanized watersheds to evaluate microorganism removal using alum coagulation. Bench-scale jar tests were employed during two separate trials which included a relatively high dose of alum (600 mg/L) versus a lower dose typically employed for stormwater treatment (10 mg/L).

The first collection site was in Pinellas Park, Florida, upstream of an existing in-line alum treatment system consisting of an alum injection system and downstream settling pond. The point of collection was located at a channel which drains an approximately 33 ha (83 ac) residential/light commercial watershed. Downstream of the collection point, the existing treatment system used alum to coagulate and remove pollutants from stormwater using a flow-weighted dosing system. An average dose of 10 mg/L concentration of aluminum sulfate ($Al_2(SO_4)_3 \cdot 18H_2O$) was determined by previous jar tests to be optimal for pollutant removal at the site (Environmental Research and Design, Inc., 1995), however, no evaluation of microorganism removal was performed. After flowing through a series of culverts, settling ponds, open ditches, and a small lake, stormwater from this basin eventually discharges to middle Tampa Bay.

Stormwater from this site was challenged in triplicate in the summer of 1997 after an approximately 5 cm rainfall event. Approximately 16 L of stormwater was pumped into each of four (4) 20 L capacity plastic containers from a collection point upstream of a settling pond and existing alum injection system. Three of the four containers were dosed with 160 ml of industrial-grade liquid alum to simulate a high dose treatment of approximately 600 mg/L concentration. The fourth container was used as a control to measure the effects of natural die-off and settling of the microbial indicators. Loads of approximately 3.40×10^{12} MS2 virions and 9.80×10^{6} fluorescent beads were added to each container.

A second trial was performed using water sampled from a large creek (Hamilton Creek) draining an 184 ha (460 ac) urbanized watershed in downtown Tampa near the Lowry Park Zoo. The site was adjacent to a number of residential homes using septic tank systems. Total and fecal coliform concentrations in the creek were consistently elevated above Class III water quality standards (200 cfu/100 ml) prior to sampling (Fig. 7). A large (12.0 cm) storm event occurred within 24 hours of sample collection. During this trial, a lower dose of alum was added at a concentration of 10 mg/L expressed as Al_2O_3. The correct dose of alum was obtained by mixing a solution containing 10 ml of concentrated liquid aluminum sulfate ($Al_2(SO_4)_3 \cdot 18H_2O$) with 1 L of deionized water. From this diluted alum solution, 14 ml was taken and mixed in each of three containers containing 14 L of stormwater. Approximately 1.36×10^{9} fluorescent beads and 2.4×10^{12} MS2 virions were then added to each container. Total and fecal coliforms were not

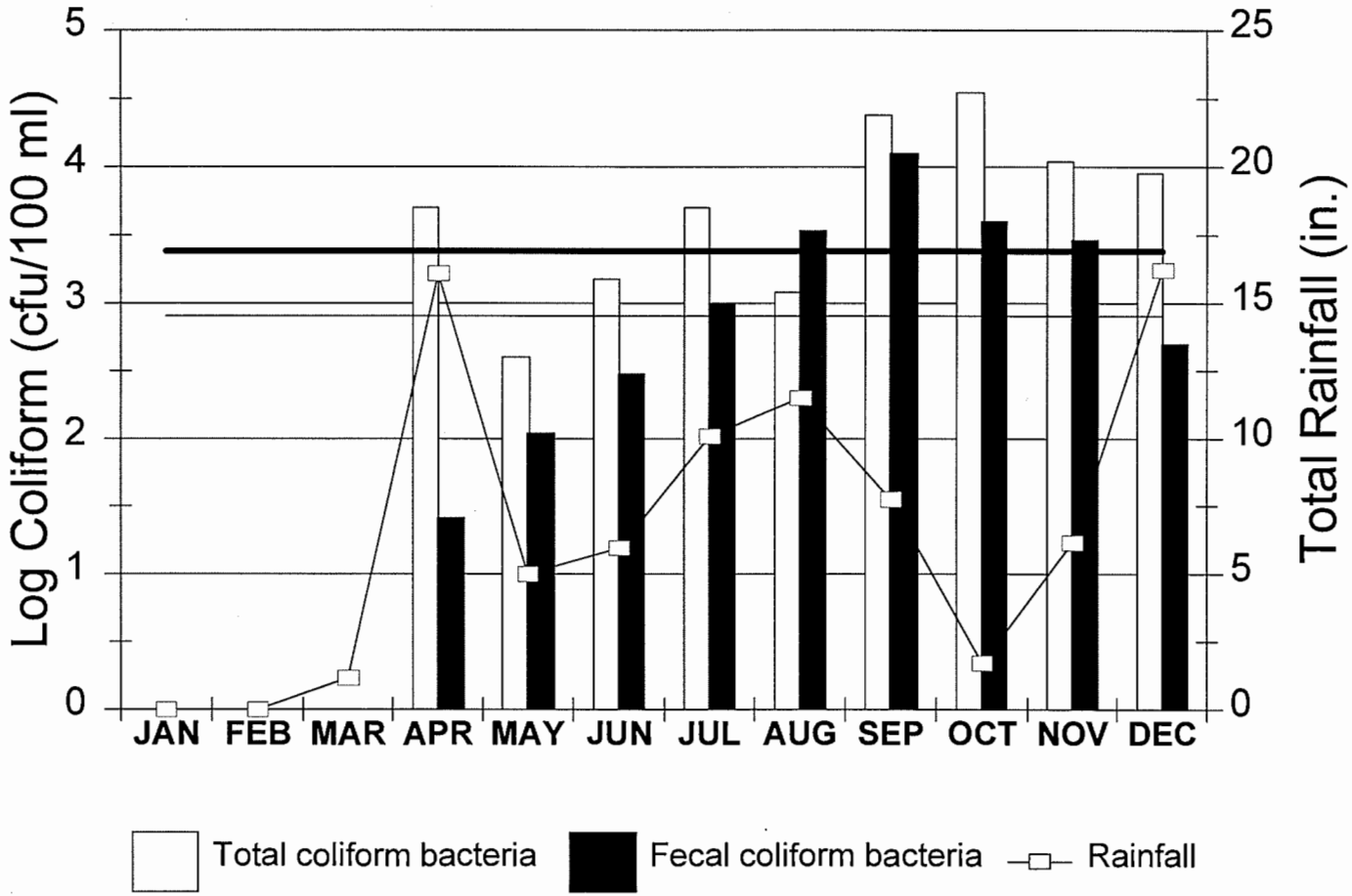


Fig. 7. Monthly trends in total and fecal coliform bacteria concentrations in Hamilton Creek (Lowry Park) prior to the collection of stormwater for jar testing in February 1998. Thick and thin horizontal lines represent Class III daily maximum concentrations for total coliforms and fecal coliforms, respectively.

seeded during either trial since pilot sampling in each of the two source waters indicated ambient concentrations were sufficiently elevated for jar test comparisons.

During both trials, samples were collected within the top 10-15 cm of each container using sterilized tygon tubing and a peristaltic pump. The first set of samples was collected prior to seeding with MS2 and beads to determine background concentrations of MS2. High titers of MS2 and beads were then stirred into all four containers and then a second set of samples (designated as T_0) was taken to determine actual seeded concentrations. This was followed by the addition of alum to each of three of the four containers. Each container was mixed thoroughly for approximately 30 seconds to distribute the alum throughout the water column. Subsequent samples from all four containers were collected 24 and 48 hours after the addition of alum. Load values were adjusted after each sampling event by subtracting the load associated with the 2 L of water that was taken from each container during the previous collection. The floc layer that settled during coagulation was also sampled by decanting each container until approximately 2 L of water remained. The remaining water and floc material were then mixed for approximately 30 seconds and then collected for analyses.

Laboratory Analyses and Physicochemical Parameters

Temperature (°C), pH (s.u.), and conductivity (µS/cm) were measured *in-situ* using a Hydrolab® Surveyor III. Metals, turbidity, and TSS were collected in clean plastic bottles and analyzed at the SWFWMD lab in Brooksville according to Standard Methods

for the Examination of Water and Wastewater (APHA, 1995). Total and fecal coliform samples were collected in sterile 500 ml Nalgene® bottles and analyzed within 6 hours using the membrane filtration method (APHA, 1995). Several serial dilutions were filtered to ensure that a valid colony count could be expressed numerically. Too numerous to count (TNTC) results were not acceptable since removal efficiencies could not be calculated using a non-numerical value. Confirmation of total and fecal coliform colonies were made using the Enterotube® multitest system (Roche Bioscience), an accepted methodology by the Florida Department of Health (M. Rials, pers. comm.). MS2 coliphage and fluorescent bead samples were collected in sterile 50 ml polyethylene tubes and analyzed at the Department of Marine Science lab on the University of South Florida campus in St. Petersburg. MS2 samples were analyzed within 24 hours or were stabilized with tryptic soy broth (TSB) during a few occasions when holding times exceeded 24 hours. All samples were stored on ice prior to analysis.

MS2 coliphage (ATCC 15597-B1) and its host bacterium (*E. coli*, ATCC 15597) were obtained from the American Type Culture Collection (ATCC) and stored at 4°C prior to each experiment. High titers of MS2 were propagated using host bacterial cultures and purified by filtration through 0.45 μm pore size disposable filters. MS2 samples were assayed in duplicate over several serial dilutions using the plaque-forming technique (APHA, 1992). A 1.0 ml sample of stormwater was mixed with 0.5 ml of a 3-hour culture of host bacterium (*E. coli*) and 3 ml of melted tryptic soy agar (TSA). This mixture was poured and spread onto 100 mm diameter petri dishes containing solid TSA which were then allowed to cool and solidify. Plates were then incubated for 24 hours at

37°C. Coliphage infect and multiply within their host bacteria which results in the lysis of the host cells and a release of phage particles which infect adjacent cells. As the infected coliforms are lysed, visible clear areas (or plaques) are formed in the lawn of confluent bacterial growth. By counting the number of plaques on each plate and multiplying by the reciprocal of the dilution factor, the number of plaques per ml of sample was obtained.

Fluorescent latex beads (Fluoresbrite® beads, Polysciences, Inc.) were used as surrogates to model the transport and fate of *Cryptosporidium* spp. The methods for enumerating fluorescent beads were similar to the methods used by Paul *et al.* (1995) in their investigation of on-site sewage disposal systems in the Florida Keys. The beads used in this study were similar in size ($3.0 \pm 0.1\mu m$ in diameter) and density to *C. parvum*, were relatively inert in aqueous solutions, and have been used as tracers in both environmental contamination assessments (Harvey, 1989; Paul *et al.*, 1995; Dr. Joan Rose, University of South Florida, pers. comm.) and cytometry studies.

Fluorescent beads were enumerated by filtering 10-50 ml of either full strength or diluted stormwater through a 10 mm diameter Sartorius® filter with a 1.5 μm pore size. Filters were soaked in 1X PBS and then placed in stainless steel filter holders. Each volume of sample was injected slowly through the filter using a 10 cc syringe. The filter was then removed using flame-sterilized forceps and placed on a glass slide prepared with approximately 0.1 ml of 2% DABCO-glycerol mounting medium. Negative control membranes were mounted first, followed by stormwater samples, and then positive control membranes.

Duplicate filters were counted at a magnification of 20x with an Olympus BH-2 epifluorescence microscope equipped with blue light excitation. Only those fluorescent beads which met the size tolerance for the *Cryptosporidium* oocyst surrogate (3μm) under 40x magnification were counted. Bead concentrations per 1.0 ml were then calculated by dividing the average number of beads counted from duplicate filters by the volume of sample water filtered.

Data Analysis

Geometric means were calculated for all inflow and outflow values for microbial indicators. Arithmetic means were calculated for all physical parameters. Log removal values were calculated based on log concentration differences between inflow and outflow samples. Removal efficiencies were calculated using mass balance equations for each of the four indicators and TSS. For turbidity, removal efficiency was calculated by the difference between mean inflow and outflow concentrations, dividing by the inflow concentration, and multiplying by 100%. Loading values were known for fluorescent beads since none of the treatment systems had been exposed to this tracer prior to the study. For MS2, background samples were collected from the source water prior to seeding to determine ambient coliphage concentrations. The geometric mean of these values was then multiplied by the total volume of water entering the treatment system to estimate an ambient loading value which was then added to the known seed load to calculate a total inflow load:

$$load_{in} = ([\bar{x}_{in}] \times volume_{total}) + seed$$

where x = sample replicate

Bacteria and total suspended solid loads were based on ambient concentrations. To estimate loads leaving each system, outflow concentrations were multiplied by the corresponding volume of water discharged between each sample collection interval and summed to estimate total outflow loads ($load_{out}$) as follows:

$$load_{out} = \sum_{1}^{x} [outflow]_x \times volume_x$$

where x = sample collection interval

A total of ten evenly-spaced outflow samples were collected at the sand filter. Five samples were taken during the drawdown period for the wet detention ponds, and two post-treatment samples were taken during the alum jar testing. Inflow and outflow loads were then used in the following equation to determine removal efficiencies:

$$\%removal = \frac{load_{out} - load_{in}}{load_{in}}$$

For the alum jar tests, removal efficiencies were based on concentration differences between the control and alum treated samples at a given sampling time. The $load_{in}$ value was equal to the concentration of a parameter in a given container (jar) at a given time of

sampling (24 or 48 hours) after the addition of alum. The value for $load_{out}$ was equal to the concentration of a parameter in the control jar.

Decay rates can be used to predict the concentration of a given parameter at a specific point in time and have often been used in mechanistic models simulating pollutant transport and fate (Canale *et al.*, 1993). Decay rates were determined for total and fecal coliforms and MS2 using data from both the wet detention pond and alum jar test experiments using the following equation:

$$C_t = C_0 \times 10^{-kt}$$

where C_t = concentration in cfu/100 ml or pfu/ml at t hour at the outflow

C_0 = concentration in cfu/100 ml or pfu/ml at time zero at the inflow

t = time in hours

k = die-off coefficient (larger k values represent faster die-off rates)

Since the decay rate, k, was the unknown variable, natural logs of the fractional decreases in concentrations were plotted over time (t). The slope of the best fit linear regression line to these points was used to estimate k.

Whenever possible, parametric statistics (ANOVA) were used to compare concentrations between inflow and outflow samples for each of the microbiological indicators. Due to wider than expected variations in bacterial and coliphage concentrations, non-parametric analyses (Kruskal-Wallis Test) were used in cases where the assumption of homogeneity of variance could not be met even after log

transformation of the data. Post-hoc comparisons were made using either the Kruskal Wallis Z test or Fisher's LSD test, depending on whether non-parametric or parametric analyses were used, respectively. Correlation analyses were performed using simple linear regression. All comparisons were considered significant at the 95% confidence level and were analyzed using NCSS® software.

Effluent concentrations were also compared with Florida's Surface Water Quality Standards (Chapter 62-302, F.A.C.) to determine if individual treatment methods could meet regulatory standards (Table 2). Results of these comparisons are presented as the percent of samples which exceeded the water quality standard. Currently, no standards exist for coliphage or protozoa.

Table 2. State of Florida Surface Water Quality Standards (Chapter 62-302) for microbial indicators and turbidity.

Parameter	Class I - Potable Water Supply	Class II - Shellfish Propagation or Harvesting	Class III (Marine)- Recreation, Propagation and Maintenance of a Healthy, Well-Balanced Population of Fish and Wildlife
Total Coliform Bacteria (cfu/100 ml)	≤ 1,000 as a monthly average, nor exceed 1,000 in more than 20% of samples examined during any one month, nor exceed 2,400 at any one time.	Median most probably number value shall not exceed 70, and not more than 10% of samples shall exceed 230.	≤ 1,000 as a monthly average, nor exceed 1,000 in more than 20% of samples examined during any one month, nor exceed 2,400 at any one time. Monthly averages shall be expressed as geometric means based on a minimum of 10 samples taken over a 30 day period.
Fecal Coliform Bacteria (cfu/100 ml)	Counts shall not exceed a monthly average of 200 nor exceed 400 in 10% of the samples, nor exceed 800 on any one day. Monthly averages shall be expressed as geometric means based on a minimum of 5 samples taken over a 30 day period.	Counts shall not exceed a median most probable number value of 14 with not more than 10% of samples exceeding 43 nor exceed 800 on any one day.	Counts shall not exceed a monthly average of 200 nor exceed 400 in 10% of the samples, nor exceed 800 on any one day. Monthly averages shall be expressed as geometric means based on a minimum of 10 samples taken over a 30 day period.
MS2 Coliphage	NONE	NONE	NONE
Cryptosporidium parvum	NONE	NONE	NONE
Turbidity (nephelometric units or NTU)	≤ 29 above natural background conditions	≤ 29 above natural background conditions	≤ 29 above natural background conditions

CHAPTER THREE

RESULTS

Individual Stormwater Treatment Systems

Sand Filter

Microbial Indicators

The mean inflow and outflow concentrations, removal efficiencies, log removal values (based on differences in mean log inflow and log outflow concentrations), and statistical significance of comparisons between log-transformed inflow and outflow concentrations for the sand filter treatment system are presented in Table 3. Removal efficiencies for the microbial indicators ranged from 59.4% to 99.5% and were greatest for the *Cryptosporidium* surrogate (fluorescent bead), followed by MS2 coliphage, fecal coliforms, and total coliforms. Differences between removal efficiencies for the four indicators were not significant ($p > 0.05$). For each trial, concentrations of the four

Table 3. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the sand filter treatment system challenge. Data from all three trials were used for comparisons.

Parameter	Inflow	Outflow	Mean Log Removal	Load Removal Efficiency
Turbidity (NTU)	15.70	2.76*	-	82.4%
TSS (mg/L)**	19.27	5.63*	-	71.0%
Total coliforms (cfu/100 ml)	2.44×10^4	4.24×10^3	0.88	59.4%
Fecal coliforms (cfu/100 ml)	1.19×10^4	1.19×10^3*	1.01	65.4%
MS2 coliphage (pfu/ml)	2.10×10^5	2.00×10^3*	2.02	87.7%
3 µm beads *(Cryptosporidium* surrogate) (beads/ml)	1.94×10^5	5.22×10^1*	3.57	99.5%

*statistically significant difference at the 95% confidence level between log transformed inflow and outflow concentrations. ** based on a single trial with multiple replicates.

microbial indicators and turbidity were all significantly ($p \leq 0.05$) lower in the outflow than in the inflow (Figs. 8 - 10) except for total coliform comparisons in Trial 3 (Fig. 10). Log-transformed concentrations of total coliforms ($r^2 = 70.1\%$), fecal coliforms ($r^2 = 80.3\%$), MS2 ($r^2 = 42.4\%$), and fluorescent beads ($r^2 = 42.6\%$) were all positively correlated ($p \leq 0.05$) with turbidity.

Trends in total coliform concentrations were similar during Trials 1 and 2. Elevated values in the inflow (T_0) generally (except for a few samples at the start and end of the treatment period) decreased below the Class III one-day maximum value of 2,400 cfu/100 ml in the outflow (Fig. 11). During Trial 3, total coliform concentrations decreased only slightly after filtration and remained elevated above the Class III maximum during the entire treatment period. Fecal coliform bacteria trends were nearly identical to total coliform values with the exception of fewer values exceeding the Class III maximum.

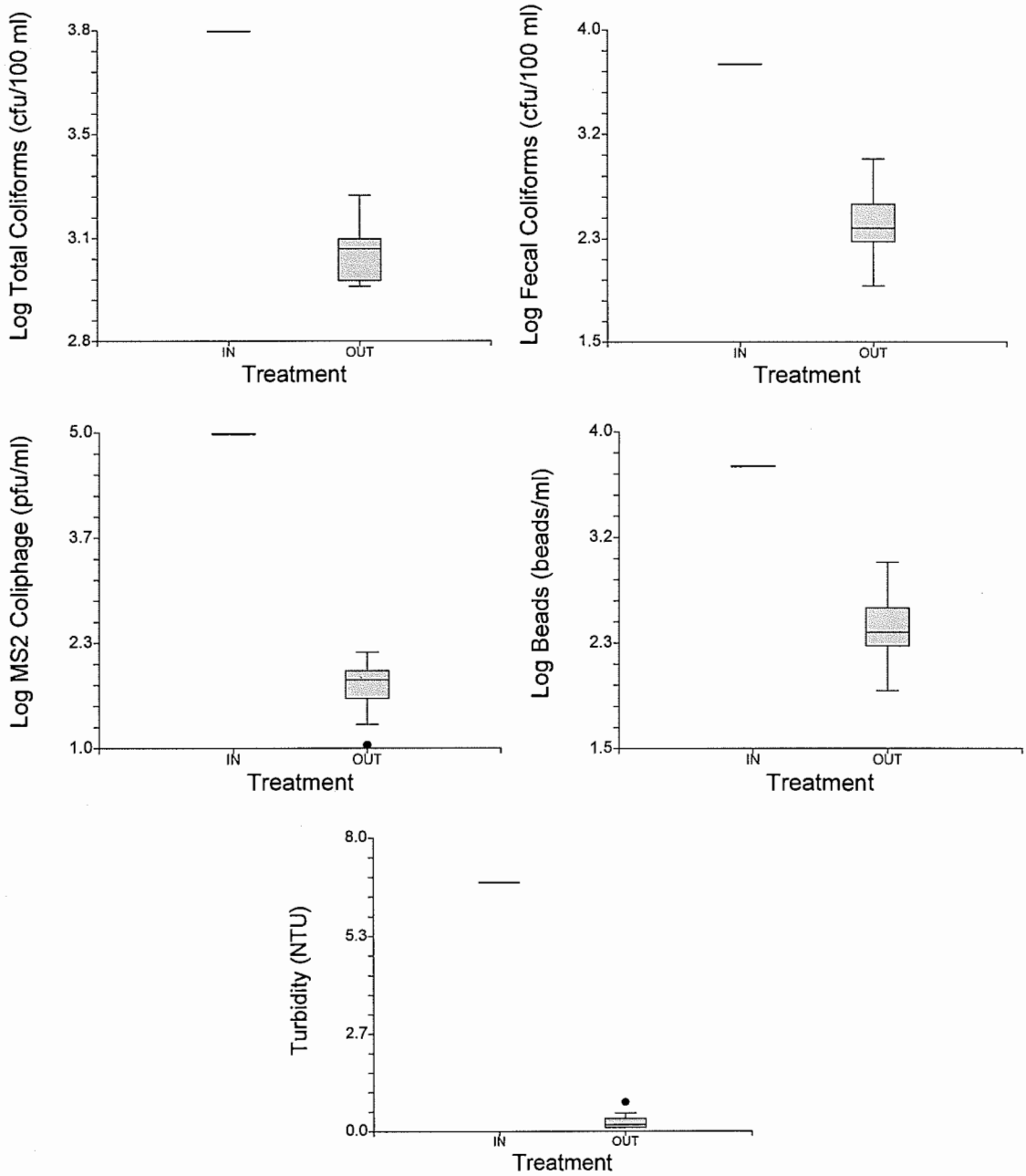


Fig. 8. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, and turbidity during Trial 1 (unsaturated) sand filter challenge. All comparisons were significantly different ($p \leq 0.05$).

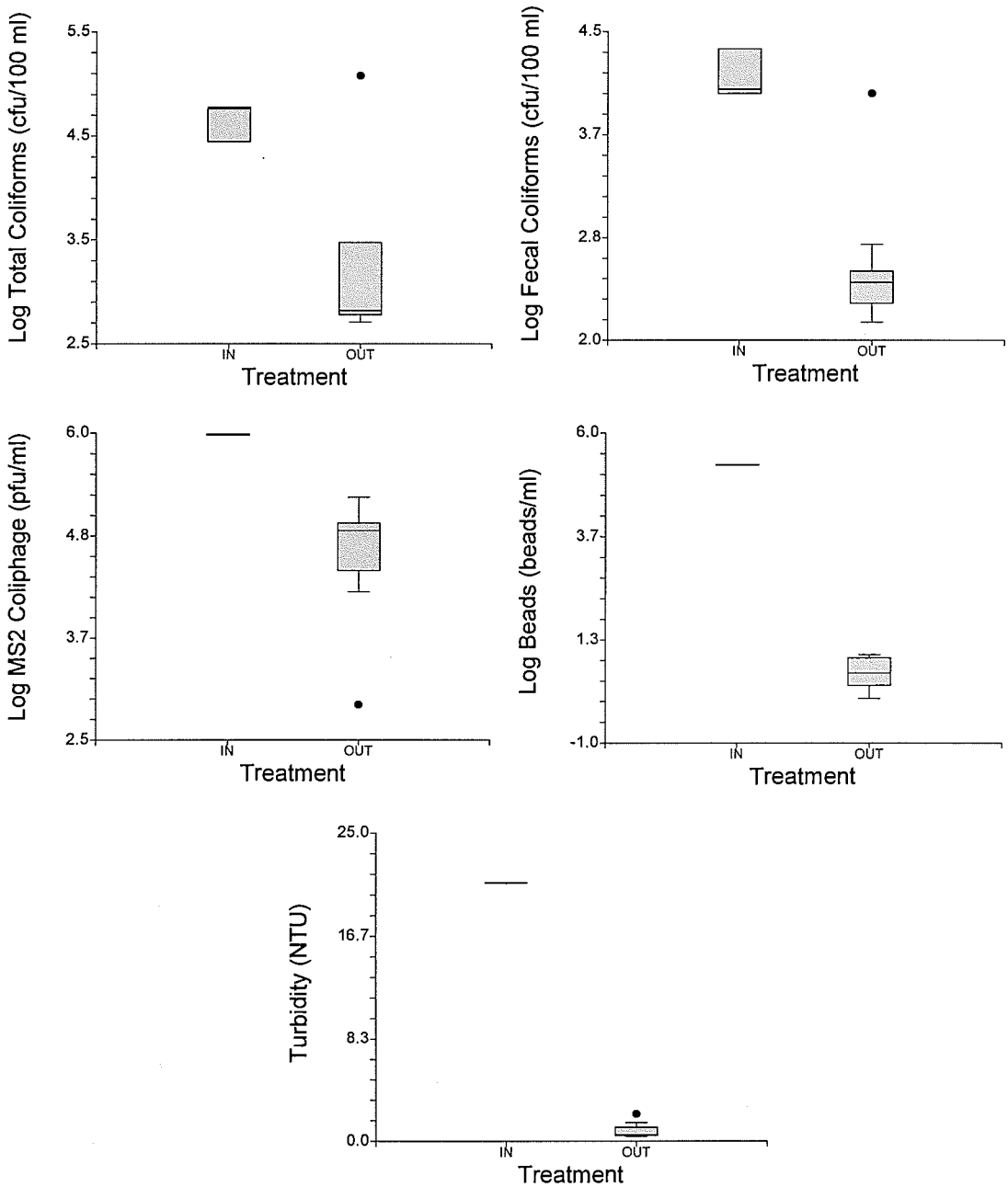


Fig. 9. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, and turbidity during Trial 2 (saturated) sand filter challenge. All comparisons were significantly different ($p \leq 0.05$).

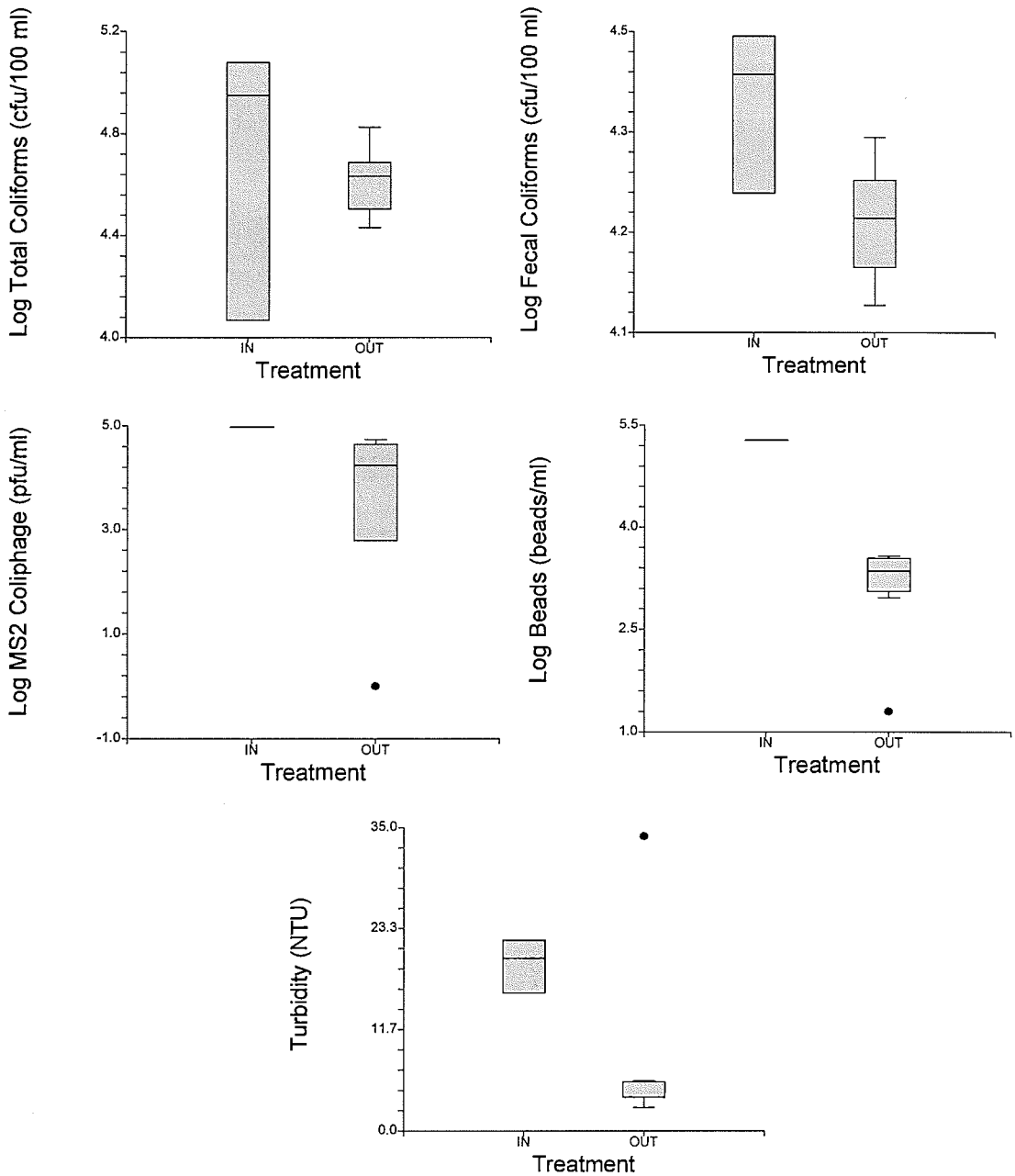
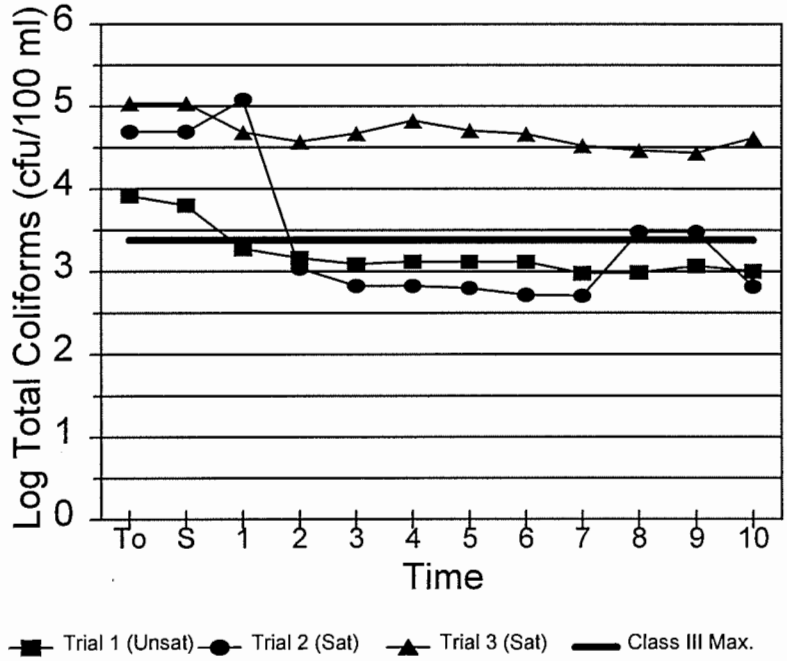


Fig. 10. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, and turbidity during Trial 3 (saturated) sand filter challenge.



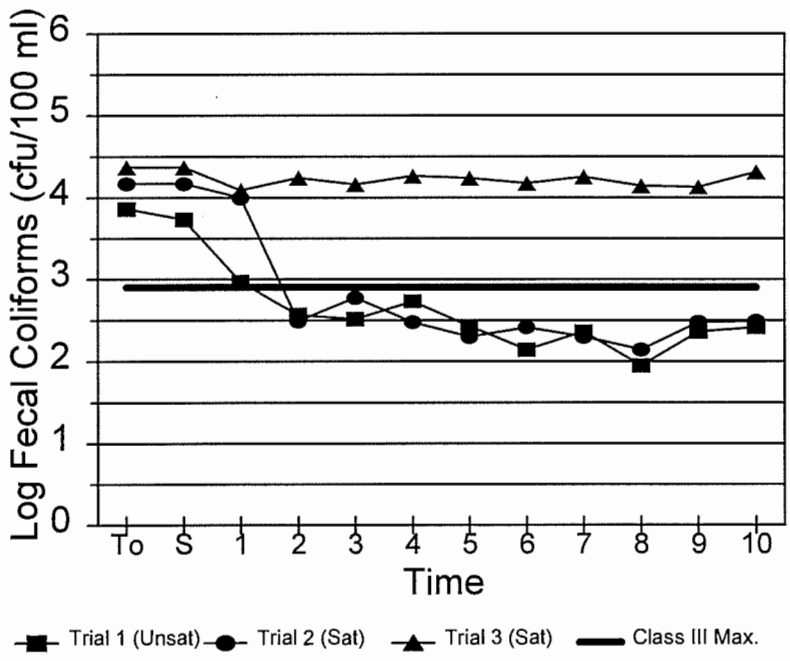
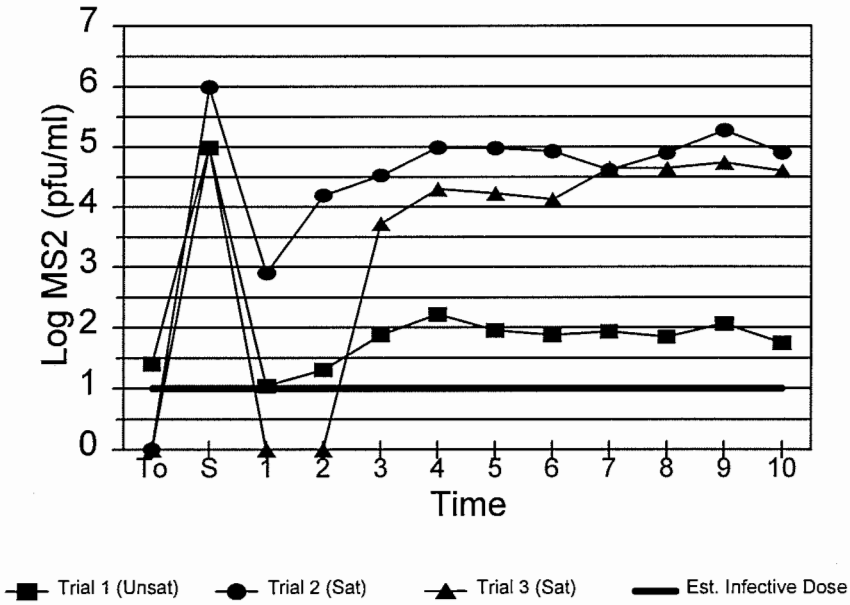


Fig. 11. Trends in total and fecal coliform bacteria concentrations over time using sand filtration. To = before seeding, S = after seeding of microorganisms but prior to filtration. Time (1-10) represents ten consecutive, equally-spaced, outflow samples after filtration.

Concentrations of fluorescent beads and MS2 coliphage dropped sharply between the seeded inflow sample and first outflow sample (Time = 1) (Fig. 12). This was due to the fact that the first few liters of effluent is composed of either water that had not yet been seeded or of existing pore-water that remained in the sand media from the most recent filter run. As the seeded portion of the water column travels through the filter, a few beads and viral particles that are not attenuated by the sand penetrate through the filter. This is reflected by a rise in the outflow concentrations (T = 2 through 10) for both of these indicators.

MS2 values rose more rapidly during saturated filter conditions than during unsaturated conditions while beads rose more rapidly during Trial 3 (120 m/day permeability rate) than during Trials 1 and 2 (90 m/day permeability rate). Horizontal lines indicating an estimated infectious dose for enteroviruses and a known ID_{50} value for *Cryptosporidium* were shown on plots for MS2 and beads since no water quality standard exists for viral or protozoan indicators. These infectious dose values were assumed to be extremely conservative (skewed toward lower doses) given the wide range of potential pathogenic viruses and protozoa in the environment. For beads, outflow samples were typically below the infectious dose except during Trial 3. For viruses, the infectious dose was exceeded in nearly every outflow sample during all three trials.

Turbidity and TSS values were elevated in all inflow samples and were reduced significantly during treatment (Fig. 13). Trends in turbidity indicated relatively rapid removal except in Trial 3, where a spike in turbidity and TSS occurred in the first outflow sample. Turbidity values in outflow samples during Trial 3 were greater than in Trials 1 and 2 despite having similar inflow concentrations.



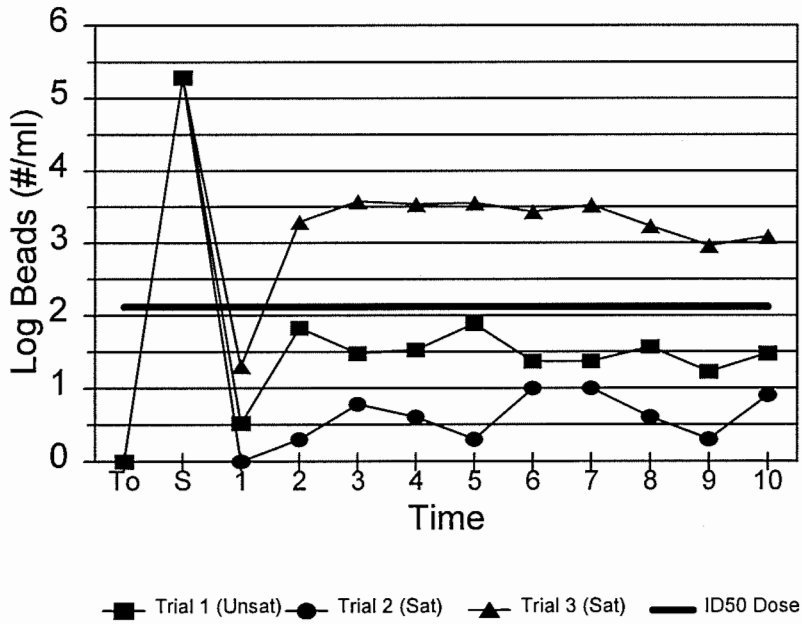


Fig. 12. Trends in concentrations of MS2 coliphage and fluorescent beads over time using sand filtration. To = before seeding, S = after seeding of microorganisms but prior to filtration. Time (1-10) represents ten consecutive, equally-spaced, outflow samples after filtration.

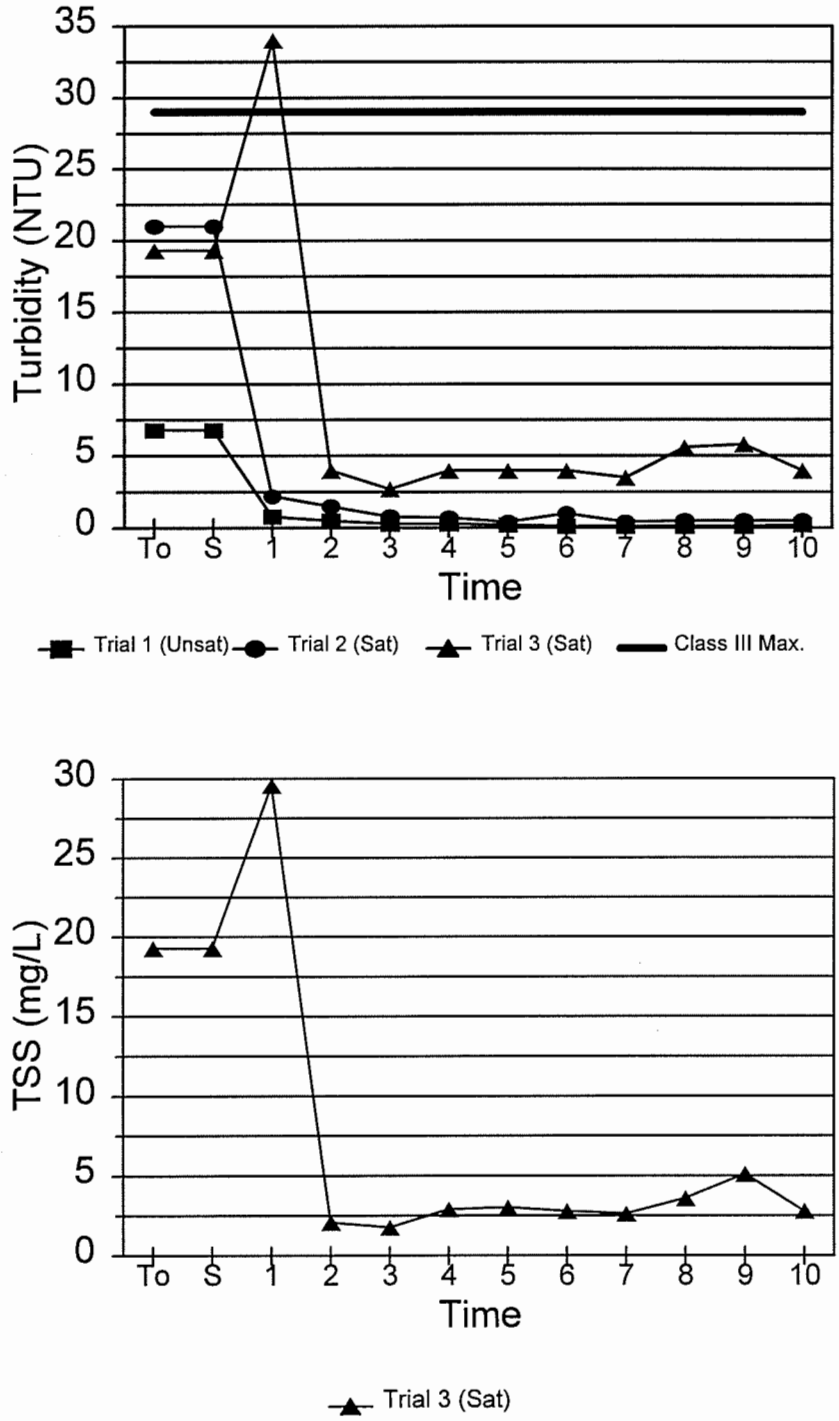


Fig. 13. Trends in turbidity and total suspended solids concentrations over time using sand filtration. To = before seeding, S = after seeding of microorganisms but prior to filtration. Time (1-10) represents ten consecutive, equally-spaced, outflow samples after filtration.

Bacteria Speciation

A number of gram-negative bacteria were identified in both the inflow and outflow samples taken from the sand filter including several which are capable of causing human disease (*E. coli*, *Klebsiella pneumoniae*, and *Salmonella enteritidis*) (Table 4). None of the various bacterial species appeared to be removed differentially since most were present in both the inflow and outflow samples. *Klebsiella pneumoniae* was the most ubiquitous species and was found in both inflow and nine of ten outflow samples.

Table 4. List of coliform bacteria identified in inflow and ten outflow samples of the sand filter. *X* denotes presence in sample.

		TIME									
Species	**IN**	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Enterobacter aerogenes	*X*		*X*		*X*	*X*	*X*		*X*		
Enterobacter agglomerans						*X*					
Enterobacter cloacae	*X*	*X*	*X*	*X*				*X*			
Enterobacter gergoviae	*X*										
Enterobacter sakazakii	*X*	*X*	*X*		*X*	*X*	*X*		*X*		
Escherichia coli		*X*		*X*		*X*	*X*				
Klebsiella ozaenae	*X*				*X*						
Klebsiella pneumoniae	*X*	*X*	*X*	*X*		*X*	*X*	*X*	*X*	*X*	*X*
Salmonella enteritidis											*X*
Serratia liquefaciens	*X*						*X*				
Serratia marcescens											
Serratia rubidea											
Citrobacter freundii											
Arizona sp.											

Physicochemical Parameters

Temperature values did not change significantly between inflow and outflow samples ($p > 0.05$) and ranged between 22 to 24°C (Fig. 14). For pH, values ranged from 6 to 7.57. Mean pH for inflow samples was 7.6 and 7.2 for outflow samples. Conductivity increased significantly ($p \leq 0.05$) from a mean of 314 µS/cm at the inflow to 783 µS/cm at the outflow.

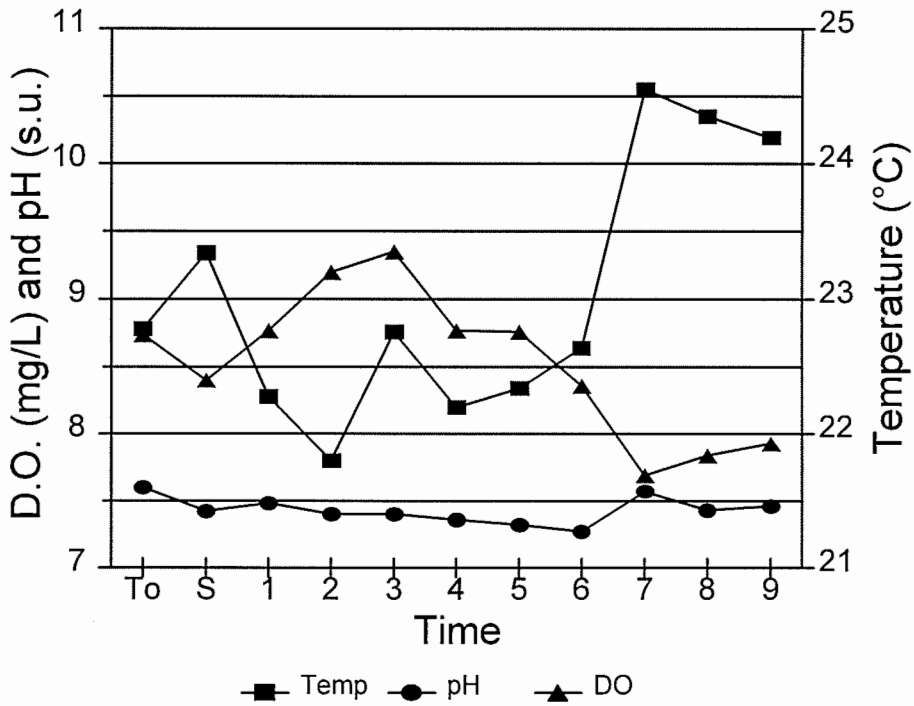


Fig. 14. Trends in temperature, dissolved oxygen, and pH over time using sand filtration. To = before seeding, S = after seeding of microorganisms but prior to filtration. Time (1-10) represents ten consecutive, equally-spaced, outflow samples after filtration.

Metal Toxicity

Metal concentrations ranged from 4.2 to115.3 ppb in inflow samples and below detection limits to 32.0 ppb in outflow samples. Mean concentrations for all metals analyzed in Trials 2 and 3 are presented in Table 5. Trends in heavy metal concentrations over time (Fig. 15) show that metals are found in greater concentrations in untreated stormwater and are removed rapidly as the water passes through the filter. Concentrations of Zn were greater than any of the other metals analyzed in the untreated stormwater. Of the six metals analyzed, Pb had the greatest mean load reductions (94.5%) followed by Cd (91.5%), Zn (82.5%), Ni (57.2%), Cu (49.1%), and Cr (9.3%).

Table 5. Mean heavy metal concentration, removal efficiency, and correlation coefficient (between metal concentrations and optical densities of *E. coli* toxicity assays) from the sand filter challenge from Trials 2 and 3.

Parameter	**Inflow**	**Outflow**	**Removal Efficiency**	**Correlation Coefficient**
Zn	108.17	2.37	82.3%	0.51
Pb	12.52	0.69	94.5%	0.39
Cd	2.37	0.20	91.5%	0.34
Ni	5.68	2.43	57.2%	0.37
Cu	13.40	6.81	49.1%	0.63
Cr	7.40	6.71	9.3%	0.48

Metal toxicity was greatest in inflow samples which coincided with peak concentrations of heavy metals. Toxicity values declined over time in the outflow samples and generally followed the declining trend in metal concentrations. Although

correlation coefficients were greatest between Cu and optical density, trends in Zn concentrations paralleled toxicity measurements more closely (Fig. 15) and probably had a greater effect on bacterial toxicity than any of the other metals due to its greater concentration in inflow and outflow samples.

Comparisons with Water Quality Standards

Surface water quality standards (≤ 29 NTU above background conditions) for turbidity in Class III waters were exceeded in only a single outflow grab sample but were never exceeded in any other inflow or outflow sample (Fig. 13). Total and fecal coliform bacteria concentrations exceeded the Class III maximum value at the inflow (raw stormwater) during every trial. Outflow concentrations for total coliform bacteria exceeded the Class III (recreational waters:< 2,400 cfu/100 ml) one day maximum value in 43% of all outflow samples. Outflow concentrations for fecal coliform bacteria exceeded the Class III (< 800 cfu/100 ml) one day maximum value in 40% of all outflow samples (Fig. 11). When analyzed by sand saturation conditions, total coliform concentrations exceeded Class III standards in 65% of outflow samples using a saturated sand filter and 0% using an unsaturated filter. Fecal coliform concentrations were exceeded in 55% of outflow samples during saturated filter conditions and 10% during unsaturated conditions. Of the six parameters, only turbidity, MS2 and the *Cryptosporidium* surrogate were reduced sufficiently to meet the State of Florida's 80% reduction goals.

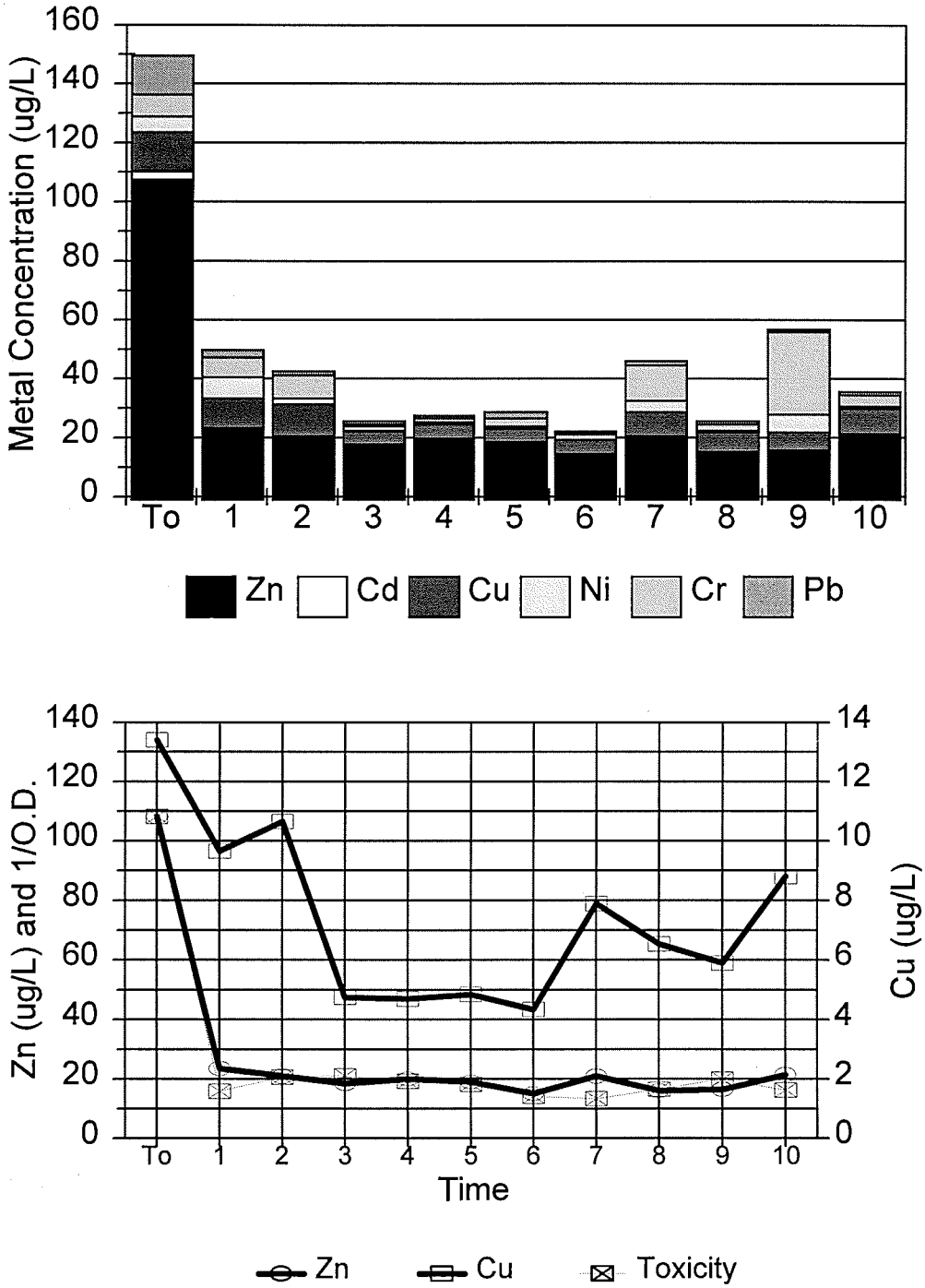


Fig. 15. Trends in heavy metal concentrations before (To) and after (1-10) sand filtration (top). Trends in Zn concentrations and toxicity (inverse of optical density of a solution containing *E. coli* cultures) before (To) and after (1-10) sand filtration (bottom).

Wet Detention

Physicochemical Parameters

Comparisons of temperature, pH, and conductivity values between shallow and deep ponds were not significantly different ($p > 0.05$) during either the 5-day or 14-day trial, and so data were grouped by trial for inflow versus outflow comparisons. Temperature decreased significantly ($p \leq 0.05$) from 21.5 to 16.5°C during the course of treatment for the 5-day trial (Fig. 16). Changes in pH were not significant ($p > 0.05$) and ranged between 7.4 and 7.9. Conductivity values decreased significantly ($p \leq 0.05$) from a mean of 550µS/cm at the inflow to 455µS/cm at the outflow.

For the 14-day trial, temperature did not change significantly ($p > 0.05$) between the inflow (22.8°C) and outflow (21.7°C). Values for pH ranged between 7.4 and 7.9 and decreased significantly ($p \leq 0.05$) from a mean of 7.5 at the inflow to 7.0 at the outflow (Fig. 17). Conductivity did not change significantly between the inflow and the outflow ($p > 0.05$). Differences in discharge rates between the shallow and deep ponds were not significantly different ($p > 0.05$) during either the 5-day or 14-day simulations. Water level differences also did not appear to vary between the shallow and deep ponds during the discharge periods (Fig. 6).

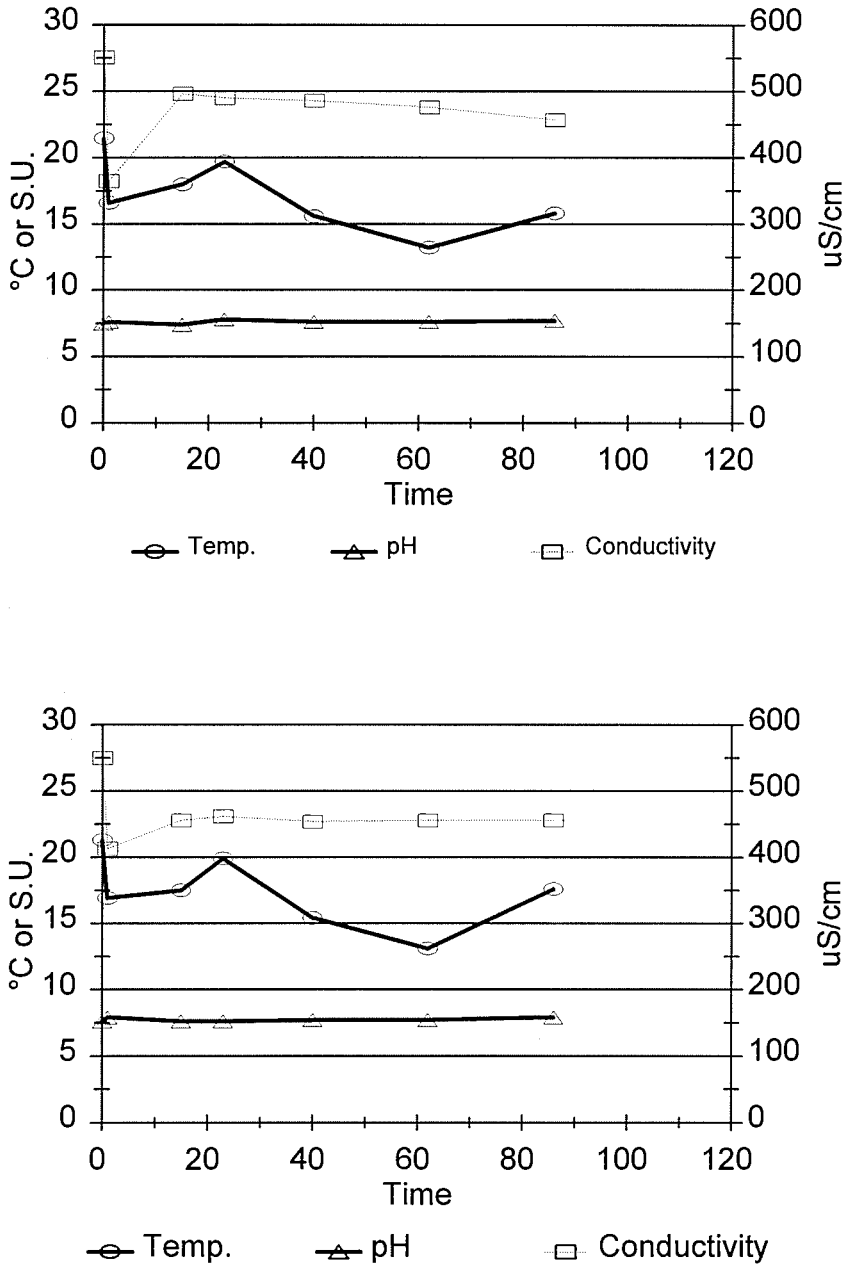


Fig. 16. Trends in temperature, pH, and conductivity over time for the shallow (top) and deep (bottom) wet detention ponds during the 5-day trials.

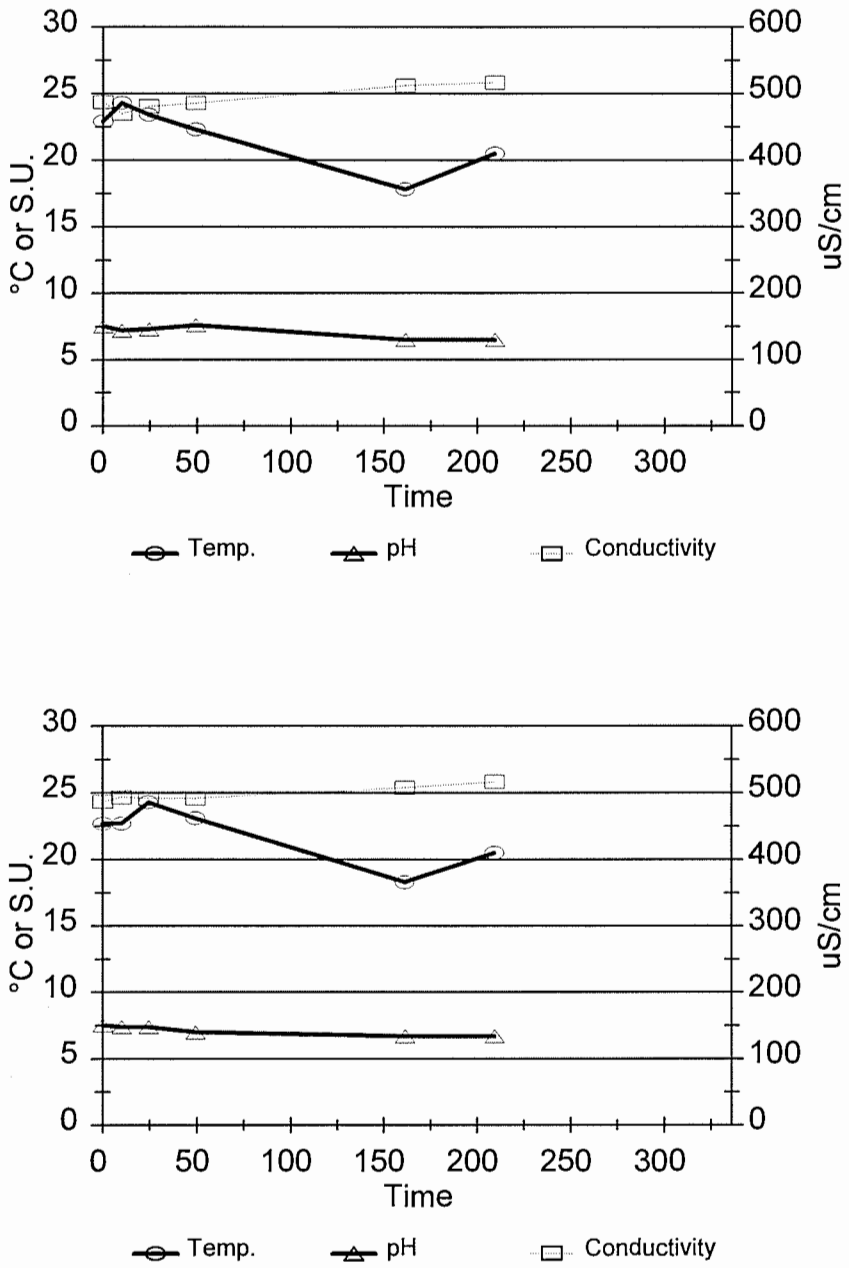


Fig. 17. Trends in temperature, pH, and conductivity over time for the shallow (top) and deep (bottom) wet detention ponds during the 14-day trials.

Microbial Indicators

The mean inflow and outflow concentrations, removal efficiencies, log removal values, and statistical significance of comparisons between inflow and outflow concentrations for the wet detention ponds are presented in Tables 6 through 9. Removal efficiencies for the four indicators ranged from -284.5% to 99.5% and were typically greater for fluorescent beads followed by MS2, fecal coliforms, and total coliforms.

Differences between inflow and outflow concentrations were significantly different for turbidity, TSS, total and fecal coliform bacteria, MS2, and beads during the 5-day shallow trial (Figs. 18 through 21). Concentrations of turbidity, TSS, and total coliforms were significantly greater in the outflow than the inflow during the 5-day deep trial, however, fecal coliforms, MS2, and beads were all significantly reduced. During the 14-day shallow trial, only turbidity, TSS, and bead concentrations were significantly lower in the outflow compared to the inflow.

During the 14-day deep trial, only TSS, MS2, and bead concentrations were significantly lower in the outflow compared to the inflow. Turbidity removal ranged from -281.2% to 37.4% and TSS ranged from -81.4% to 99.8%. Differences in removal efficiencies between the four microbial indicators for all four trials combined were significant ($p \leq 0.05$). Removal efficiencies for total and fecal coliform bacteria were both significantly less than fluorescent beads and total coliform bacteria removal was significantly ($p \leq 0.05$) less than MS2.

Table 6. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the 5-day shallow wet detention pond challenge.

Parameter	Inflow	Outflow	Mean Log Removal	Load Removal Efficiency
Turbidity (NTU)	1.23	0.86	-	30.3%
Total Suspended Solids (mg/L)	1.42	0.28*	-	99.8%
Total coliforms (cfu/100 ml)	1.14×10^3	$2.41 \times 10^{2*}$	0.67	64.0%
Fecal coliforms (cfu/100 ml)	2.29×10^2	$5.48 \times 10^{0*}$	1.62	98.2%
MS2 coliphage (pfu/ml)	9.25×10^4	$1.13 \times 10^{3*}$	1.91	93.9%
3 μm beads *(Cryptosporidium* surrogate) (beads/ml)	3.72×10^2	1.23×10^{0}*	2.48	99.5%

*statistically significant difference at the 95% confidence level between log-transformed inflow and outflow concentrations.

Table 7. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the 5-day deep wet detention pond challenge.

Parameter	Inflow	Outflow	Mean Log Removal	Load Removal Efficiency
Turbidity (NTU)	1.13	4.32*	-	-281.2%
Total Suspended Solids (mg/L)	1.67	4.21*	-	-81.4%
Total coliforms (cfu/100 ml)	6.80×10^2	3.03×10^{3}*	-0.65	-284.5%
Fecal coliforms (cfu/100 ml)	1.59×10^2	2.42×10^{0}*	1.82	88.5%
MS2 coliphage (pfu/ml)	9.24×10^4	6.94×10^{2}*	2.12	98.6%
3 μm beads *(Cryptosporidium* surrogate) (beads/ml)	3.08×10^2	2.61×10^{0}*	2.07	99.0%

*statistically significant difference at the 95% confidence level between log-transformed inflow and outflow concentrations.

Table 8. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the 14-day shallow wet detention pond challenge.

Parameter	Inflow	Outflow	Mean Log Removal	Load Removal Efficiency
Turbidity (NTU)	3.80	2.38*	-	37.4%
Total Suspended Solids (mg/L)	3.56	0.96*	-	72.2%
Total coliforms (cfu/100 ml)	4.34×10^3	4.82×10^2	0.96	4.2%
Fecal coliforms (cfu/100 ml)	2.08×10^3	4.44×10^1	1.67	76.4%
MS2 coliphage (pfu/ml)	7.07×10^3	6.96×10^2	1.01	88.9%
3 µm beads *(Cryptosporidium* surrogate) (beads/ml)	1.88×10^2	2.33×10^0*	1.91	99.1%

*statistically significant difference at the 95% confidence level between log-transformed inflow and outflow concentrations.

Table 9. Mean concentrations, log removal (based on concentrations), and removal efficiencies (based on loads) for indicator and physical parameters from the 14-day deep wet detention pond challenge.

Parameter	Inflow	Outflow	Mean Log Removal	Load Removal Efficiency
Turbidity (NTU)	3.83	4.12	-	-7.5%
Total Suspended Solids (mg/L)	3.40	2.22*	-	73.3%
Total coliforms (cfu/100 ml)	3.51×10^3	3.53×10^3	-0.003	37.9%
Fecal coliforms (cfu/100 ml)	1.57×10^3	1.53×10^2	1.01	69.2%
MS2 coliphage (pfu/ml)	6.95×10^3	1.90×10^2*	1.56	94.7%
3 µm beads *(Cryptosporidium* surrogate) (beads/ml)	1.85×10^2	2.11×10^0*	2.33	99.5%

*statistically significant difference at the 95% confidence level between log transformed inflow and outflow concentrations.

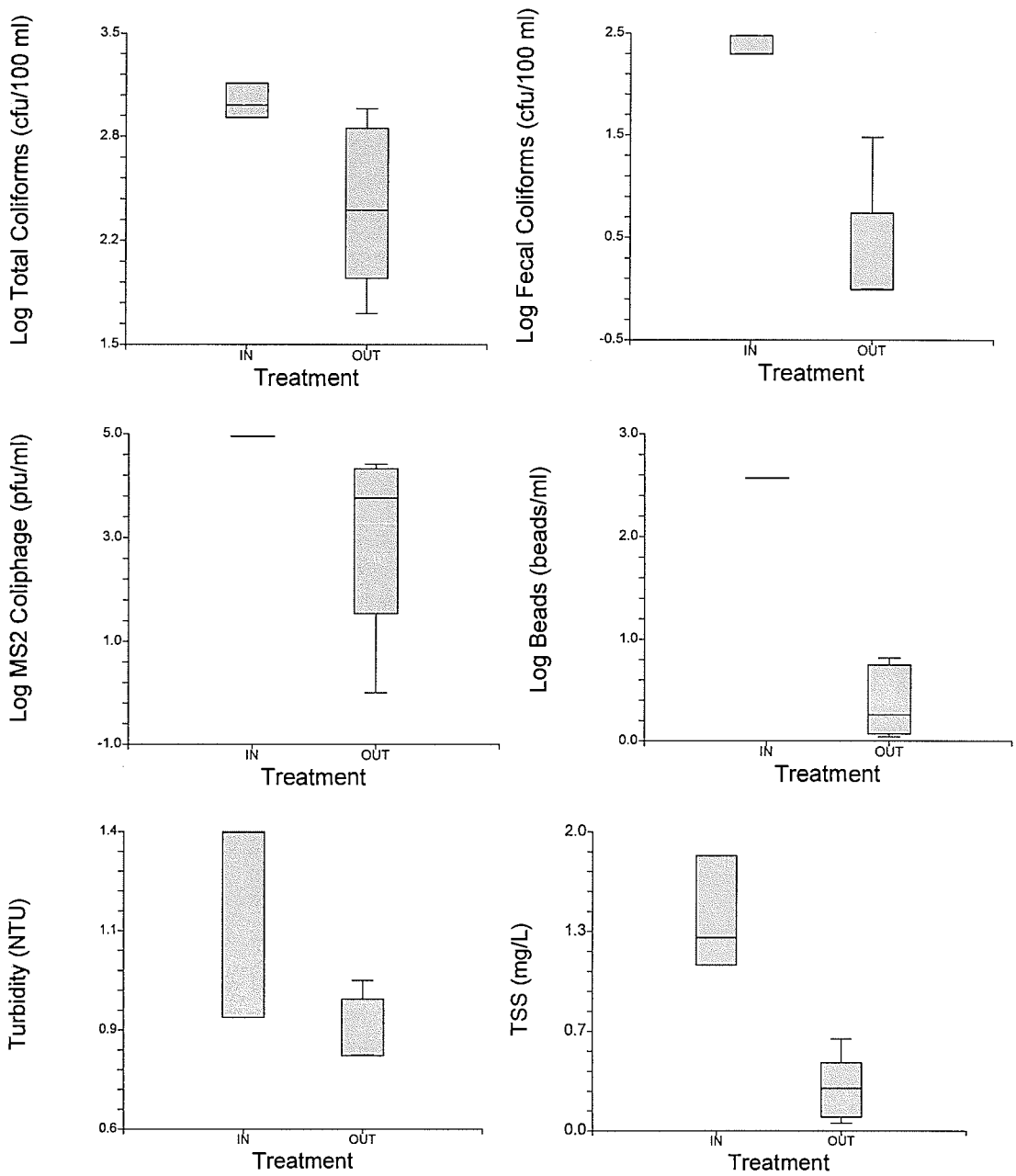


Fig. 18. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the 5-day seeded challenge of the shallow wet detention pond. All comparisons were significantly different ($p \leq 0.05$).

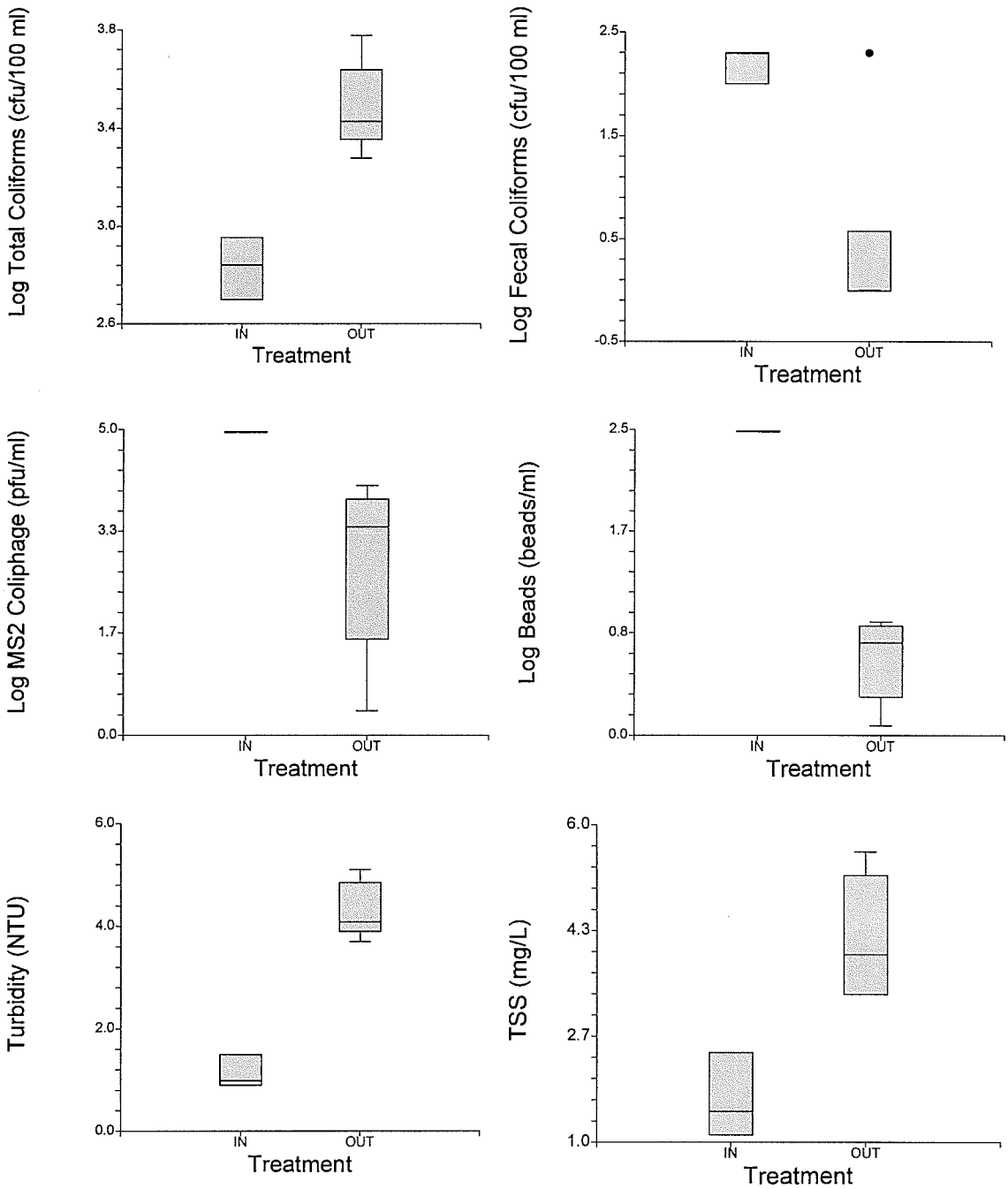


Fig. 19. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the 5-day seeded challenge of the deep wet detention pond. All comparisons were significantly different ($p \leq 0.05$).

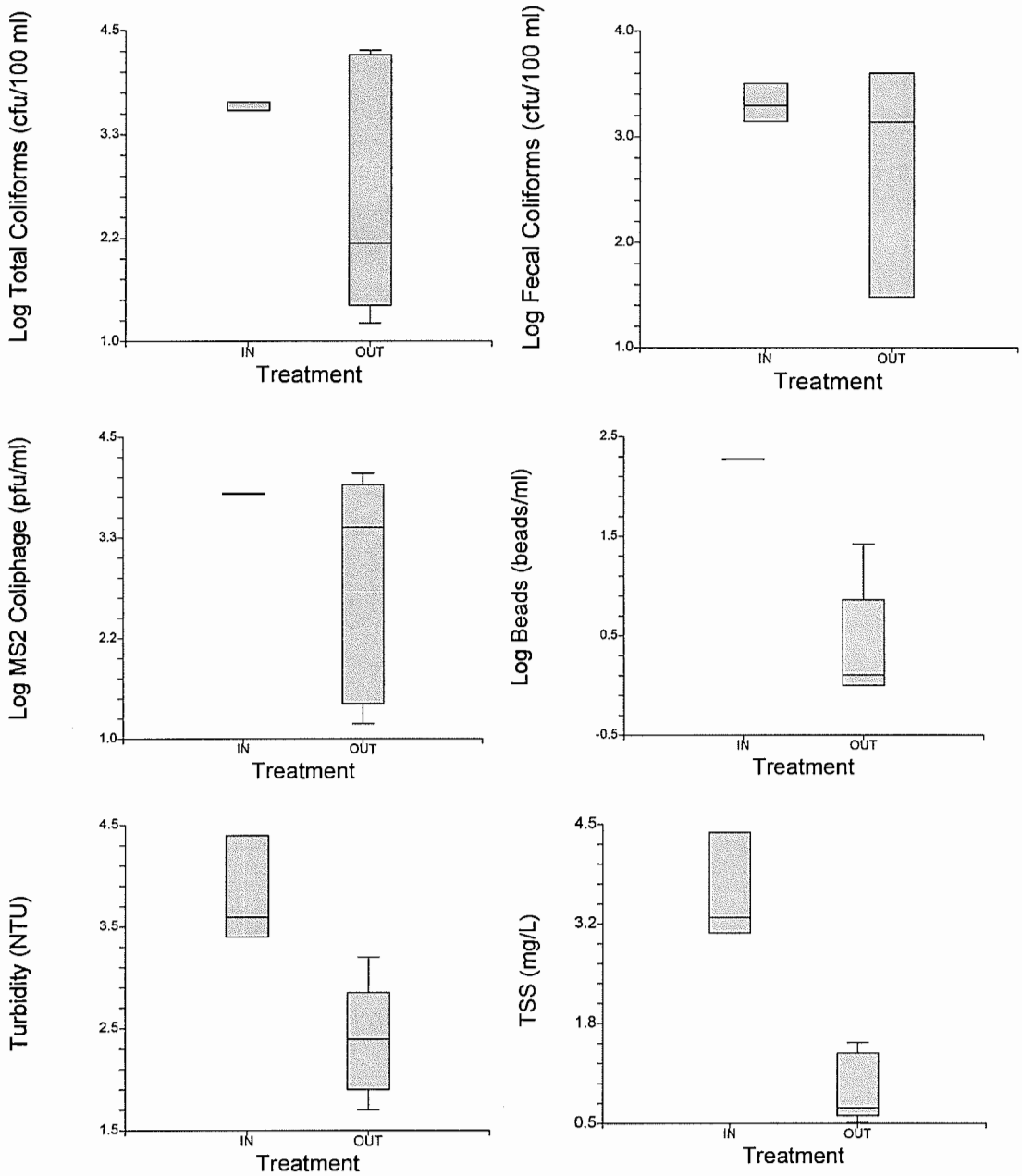


Fig. 20. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the 14-day seeded challenge of the shallow wet detention pond.

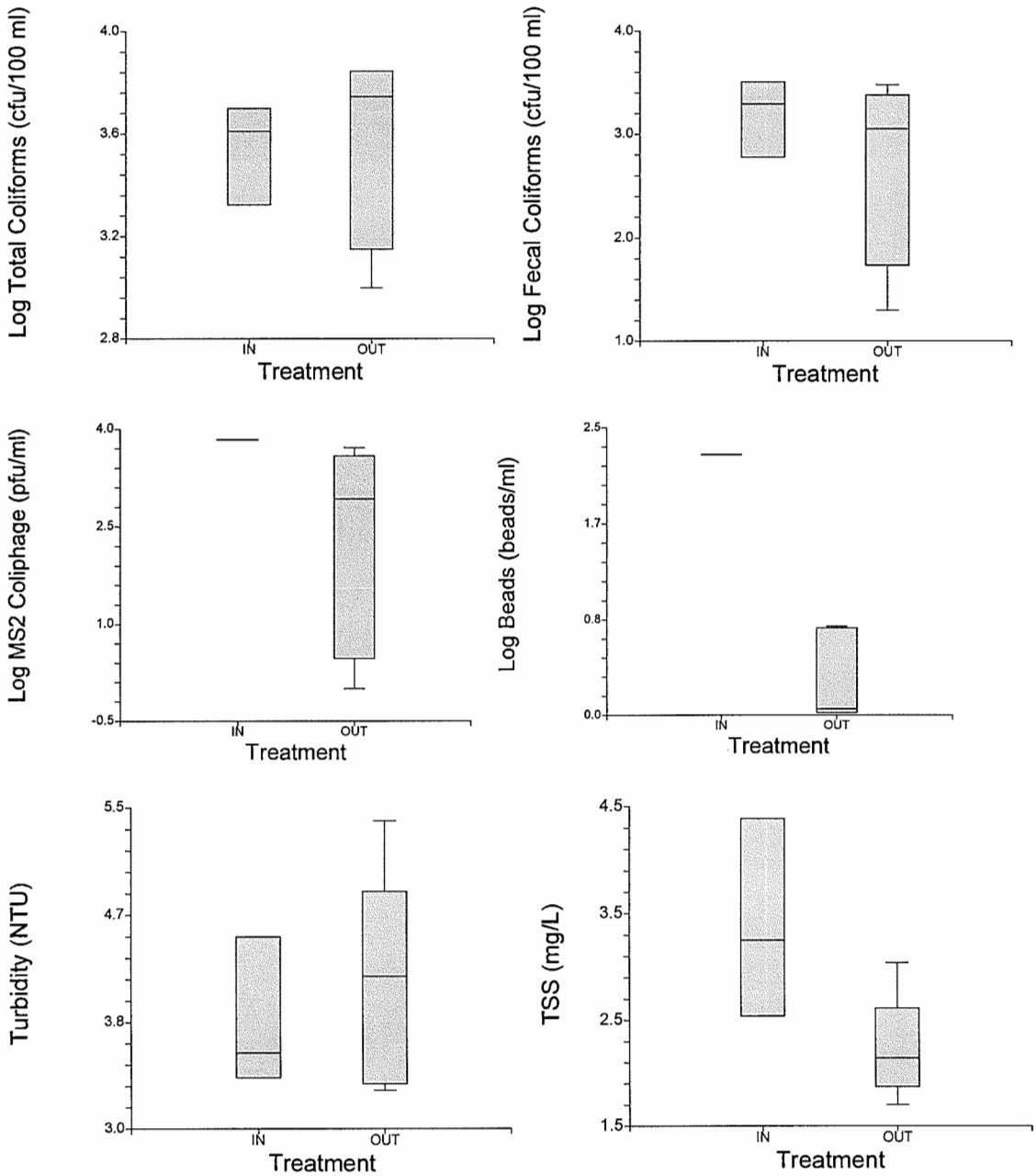


Fig. 21. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the 14-day seeded challenge of the deep wet detention pond.

Total coliform removal was greatest during the 5-day shallow pond trial followed by the 14-day deep, 14-day shallow, and 5-day deep trials. Log-transformed total coliform concentrations were positively ($p \leq 0.05$, $r^2 = 0.21$) correlated with turbidity and TSS ($r^2 = 0.30$) (Fig. 22) in both inflow and outflow samples. Fecal coliform removal was greatest during the 5-day shallow trial followed by the 5-day deep, 14-day shallow, and 14-day deep trials. Log-transformed fecal coliform concentrations were not correlated ($p \leq 0.05$) with either turbidity or TSS.

During the 5-day trial, concentrations for both total and fecal coliforms followed similar trends and peaked after the simulated storm event (pumping) at approximately 15 hours (Fig. 23). Both declined after 20 hours but total coliforms remained above the expected maximum concentration for the duration of the experiment. Total coliform values were consistently below the Class III maximum concentration in the shallow pond but were consistently greater than or equal to the Class III standard in the deep pond. Fecal coliform values were below the Class III maximum concentration in both the shallow and deep ponds after pumping began but did rise above the expected maximum concentration after approximately 20 hours and then declined below this value rapidly during the remainder of the experiment.

During the 14-day trial, total and fecal coliform concentrations also followed similar trends and peaked after approximately 15 hours (Fig. 24). In the shallow pond, both total and fecal coliforms declined below the Class III maximum concentration at approximately 25 hours but then rose above the standard after 50 hours. Both parameters then declined below the standard for the duration of the experiment. In the deep pond,

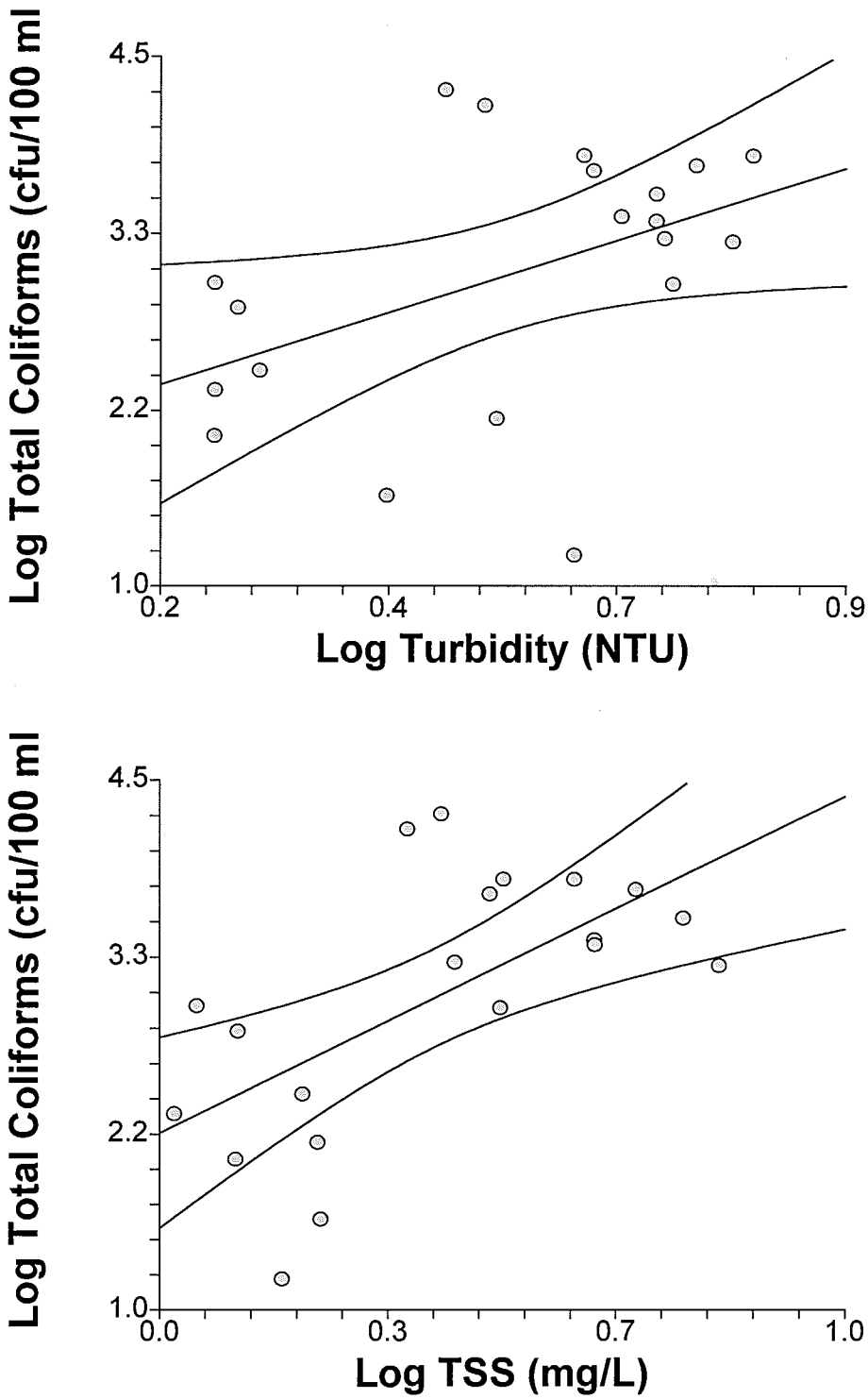


Fig. 22. Relationship between log-transformed total coliform bacteria concentrations and log-transformed turbidity (top) and total suspended solids (bottom) concentrations in outflow samples from the wet detention ponds.

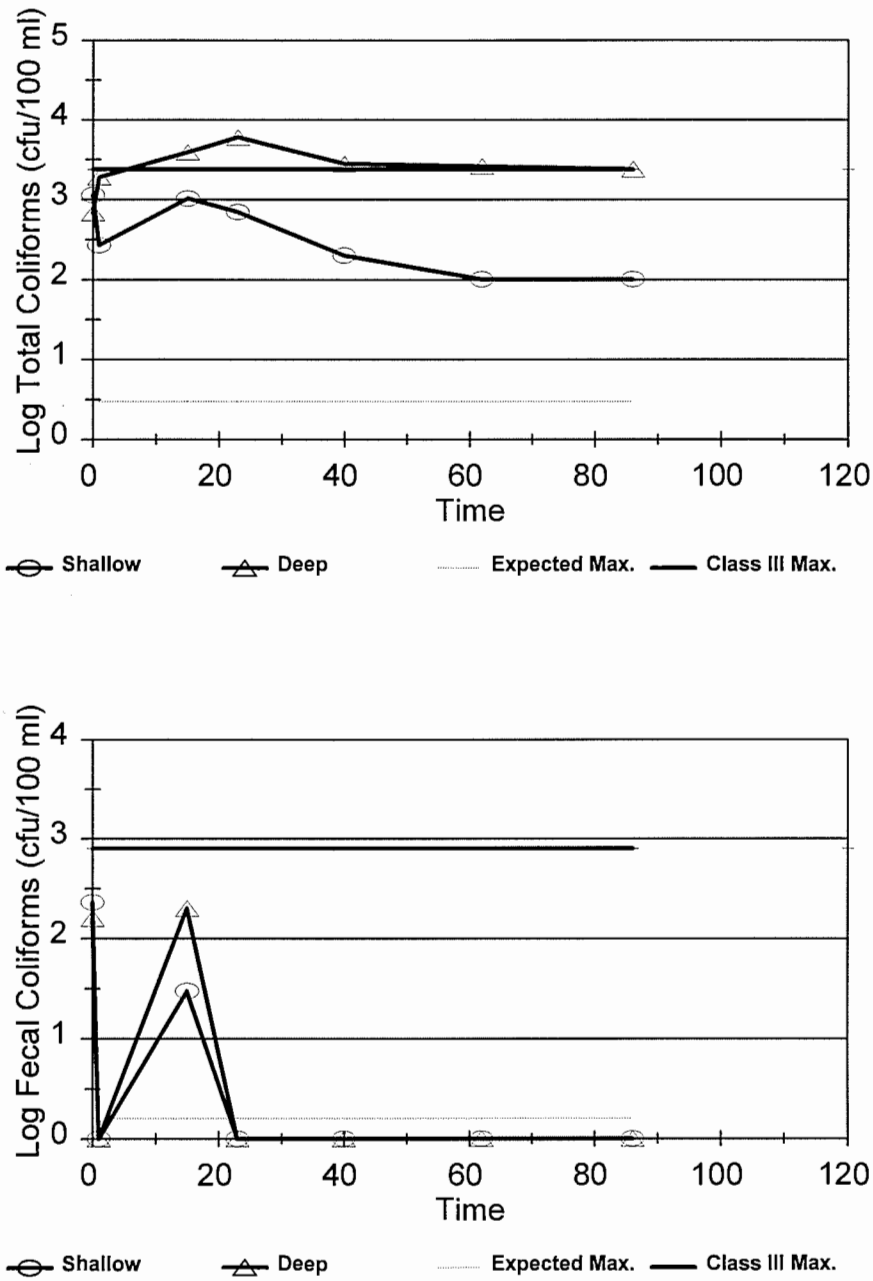


Fig. 23. Trends in total (top) and fecal (bottom) coliform bacteria concentrations over time during the 5-day wet detention pond trials. Expected Max. = maximum expected concentration assuming complete mixing of inflow load with existing pond volume. Class III Max. = one-day maximum allowable concentration under State of Florida Surface Water Quality Standards.

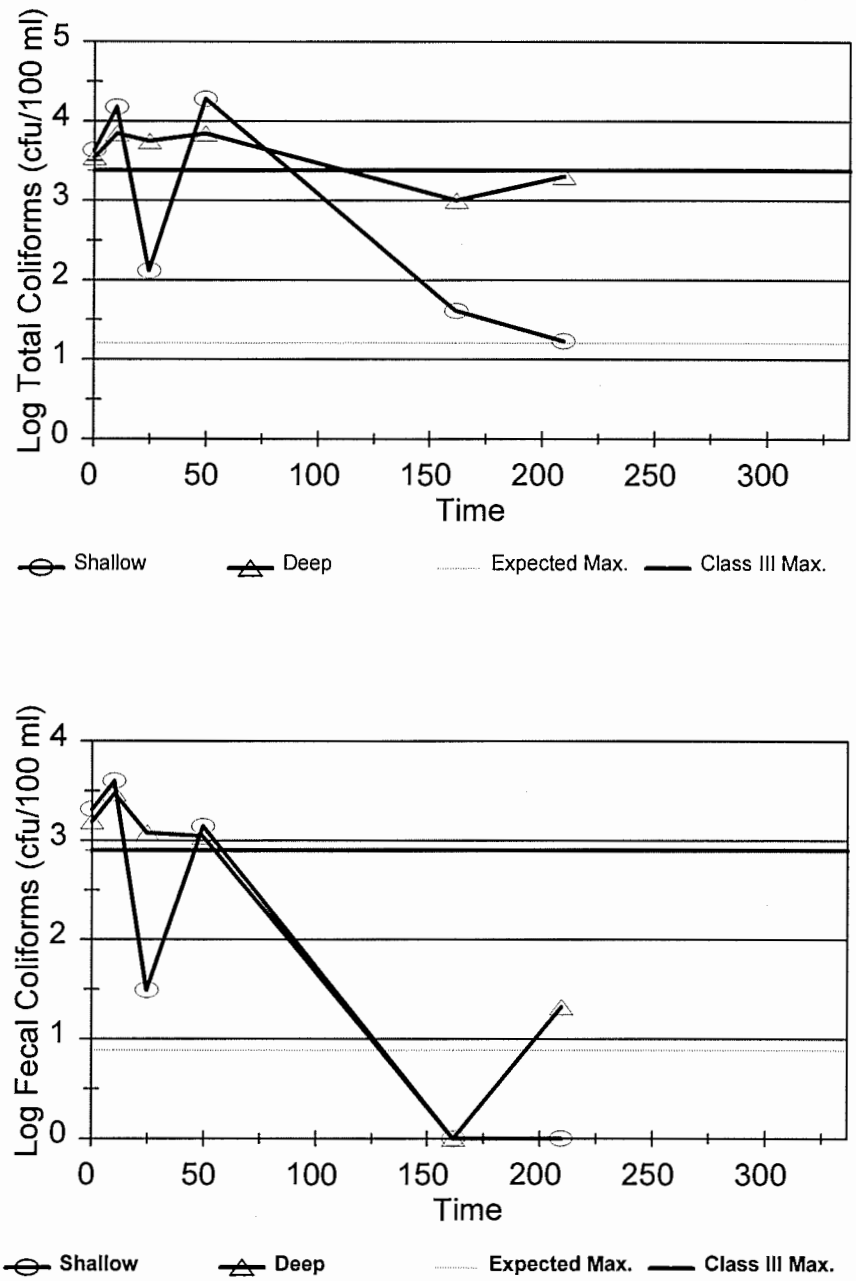


Fig. 24. Trends in total (top) and fecal (bottom) coliform bacteria concentrations over time during the 14-day wet detention pond trials. Expected Max. = maximum expected concentration assuming complete mixing of inflow load with existing pond volume. Class III Max. = one-day maximum allowable concentration under State of Florida Surface Water Quality Standards.

both total and fecal coliforms were elevated above the Class III maximum concentration until approximately 50 hours and then declined below the standard in the remaining samples.

MS2 removal was greatest during the 5-day deep trial followed by the 14-day deep, 5-day shallow, and 14-day shallow trials. Log-transformed MS2 concentrations never exceeded the expected maximum concentration given complete pond mixing during the 5-day trial (Fig. 25). MS2 concentrations were low during the first sample collection period since the inflow had not yet dispersed/traveled far enough to reach the outflow structure but did rise considerably after approximately 15 hours. Concentrations then declined at a near constant rate as a result of removal/inactivation. During the 14-day trial, MS2 concentrations were greater than the expected maximum concentration until approximately 25 hours but then declined below this value for the remainder of the experiment at a relatively constant rate in both the shallow and deep ponds (Fig. 26). Log-transformed MS2 concentrations were not correlated ($p \leq 0.05$) with either turbidity or TSS.

Bead removal was nearly identical among all four trials with slightly greater removal efficiency values during the 5-day shallow and 14-day deep trials, followed by the 14-day shallow, and 5-day deep trials. During the 5-day trial, bead concentrations never rose above the expected maximum concentration in either the shallow or deep ponds and declined after a brief peak at approximately 15 hours (Fig. 25). During the 14-day trial, bead concentrations followed similar trends in both the shallow and deep ponds and dropped sharply after the initial storm simulation to values much less than the

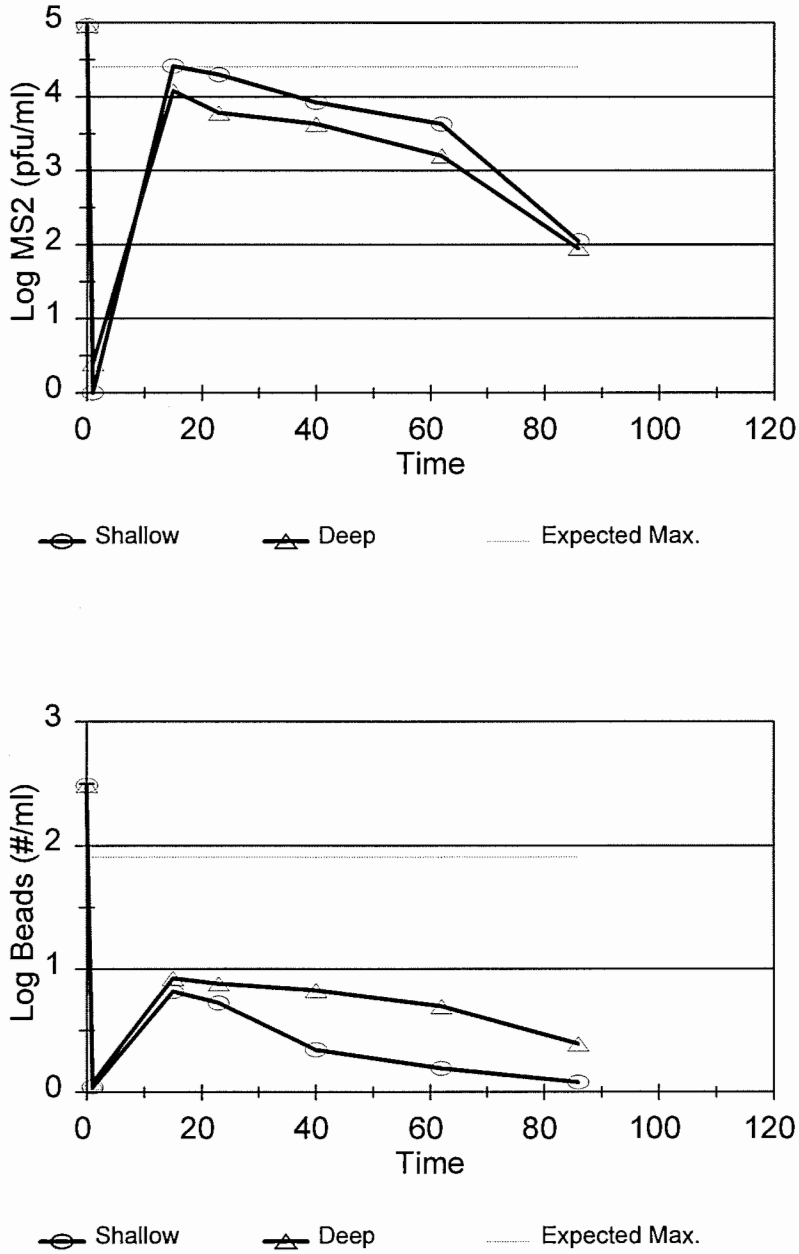


Fig. 25. Trends in MS2 (top) and bead (bottom) concentrations over time during the 5-day wet detention pond trials. Expected Max. = maximum expected concentration assuming complete mixing of inflow load with existing pond volume.

maximum expected concentration (Fig. 26). Bead concentrations then remained relatively low during the remainder of the experiment at the outflow for both ponds. Log-transformed bead concentrations were not significantly ($p > 0.05$) correlated with either turbidity or TSS.

Greatest turbidity removal occurred during the 14-day shallow pond trial followed by the 5-day shallow, 14-day deep, and 5-day deep trials. In both the 5-day and 14-day trials, turbidity was significantly ($p \leq 0.05$) greater in deep pond versus shallow pond outflow samples despite having similar inflow turbidity values (Figs. 27 and 28). Trends in turbidity were also dissimilar between shallow and deep pond outflow samples. Shallow pond turbidity values were generally flat during the 5-day trial while values declined rapidly during the 14-day trial and remained low until approximately 200 hours when a slight rise occurred. In the deep pond, turbidity rose rapidly during the simulated storm and remained high during both the 5-day and 14-day trials. This phenomenon resulted in relatively poor (negative) removal efficiency values.

TSS removal was greatest during the 5-day shallow pond trial followed by the 14-day deep, 14-day shallow, and 5-day deep trials. TSS was positively correlated ($p \leq 0.05$, $r^2 = 0.58$) with turbidity in both inflow and outflow samples. As a result, trends in TSS were highly similar to turbidity (Figs. 27 and 28) for both the 5-day and 14-day trials. During the 14-day trial, TSS concentrations dropped rapidly after the first 20 hours and remained relatively low during the remainder of the experiment. TSS also had negative removal efficiencies during the 5-day deep trial indicating a greater load was exported from the pond than had entered.

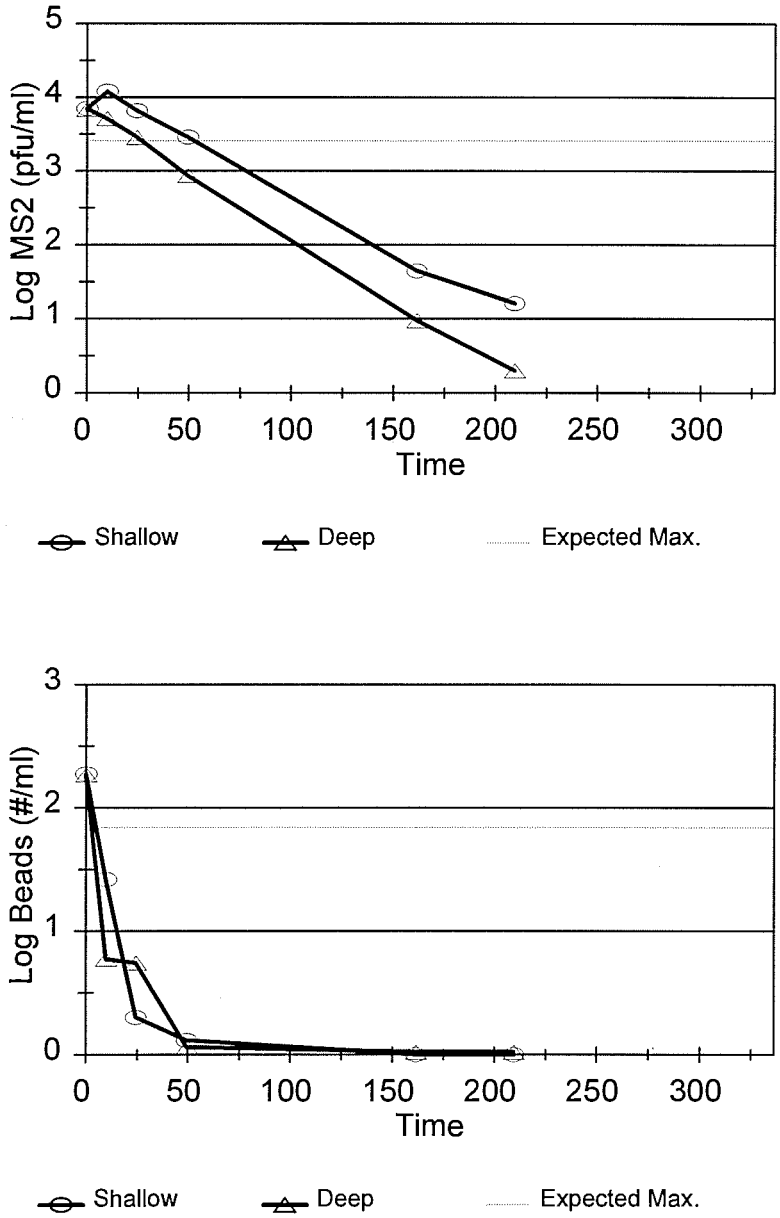


Fig. 26. Trends in MS2 (top) and bead (bottom) concentrations over time during the 14-day wet detention pond trials. Expected Max. = maximum expected concentration assuming complete mixing of inflow load with existing pond volume.

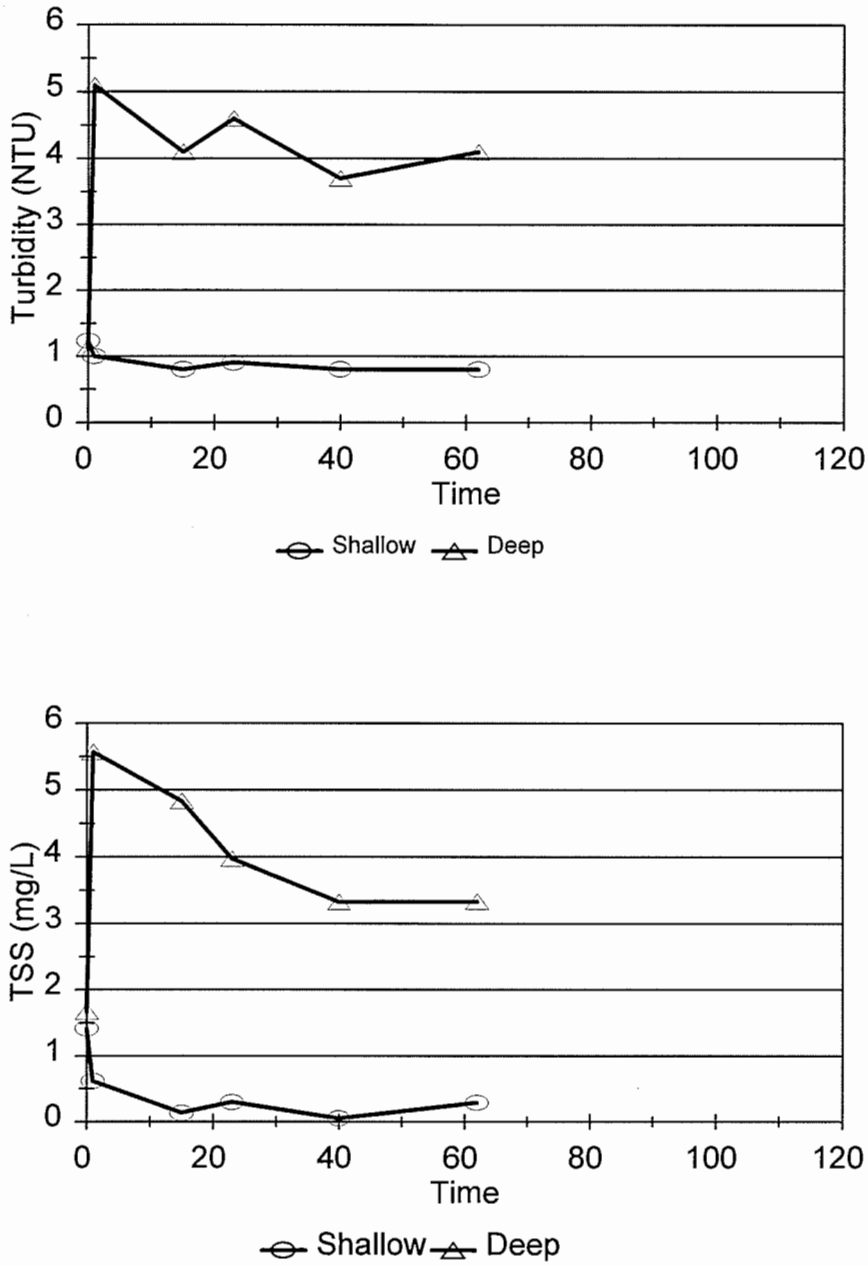


Fig. 27. Trends in turbidity (top) and total suspended solids (bottom) concentrations over time during the 5-day wet detention pond trials.

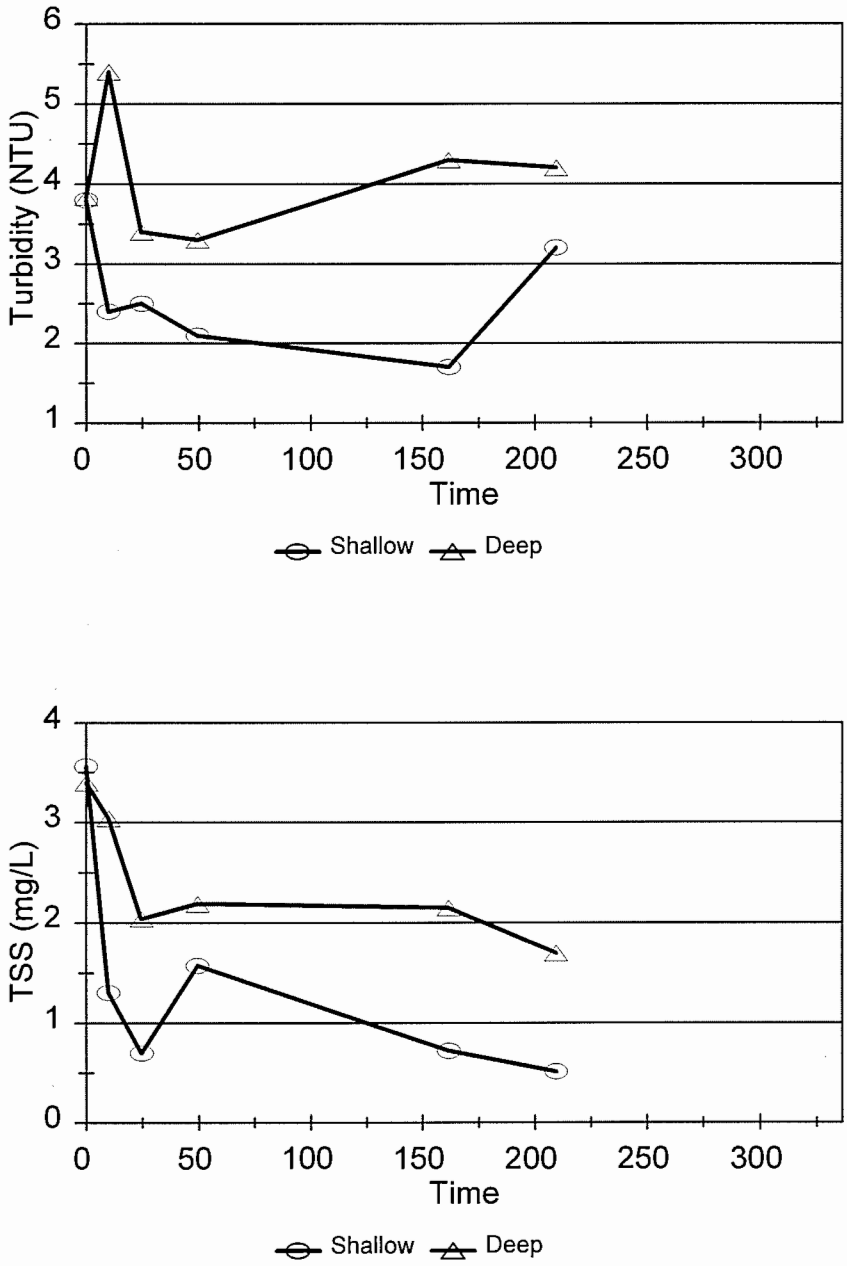


Fig. 28. Trends in turbidity (top) and total suspended solids (bottom) concentrations over time during the 14-day wet detention pond trials.

Bacteria Speciation

A number of gram-negative bacteria were also identified in both the inflow and outflow samples taken from the wet detention ponds including several which are capable of causing human disease (*E. coli*, *Klebsiella pneumoniae*, *Citrobacter freundii*, and *Salmonella enteritidis*) (Table 10). None of the various bacterial species appeared to be removed differentially since most were present in both the inflow and outflow samples. *Klebsiella pnuemoniae* was relatively persistent in the wet detention ponds during the 5-day trial and was found in both the inflow and in five of the five outflow samples.

Table 10. List of coliform bacteria identified in inflow and five outflow samples of the wet detention pond (5-day trial). *X* denotes presence in sample.

		TIME (days)				
Species	**IN**	**1**	**2**	**3**	**4**	**5**
Enterobacter aerogenes						
Enterobacter agglomerans						
Enterobacter cloacae						
Enterobacter gergoviae						
Enterobacter sakazakii						
Escherichia coli	*X*		*X*			
Klebsiella ozaenae				*X*		
Klebsiella pneumoniae	*X*	*X*	*X*	*X*	*X*	*X*
Salmonella enteritidis	*X*					
Serratia liquefaciens			*X*			
Serratia marcescens						
Serratia rubidea						
Citrobacter freundii		*X*	*X*	*X*	*X*	*X*
Arizona sp.						

Decay Rates

Decay rates were greatest for fecal coliforms followed by MS2 and total coliforms in both deep and shallow ponds during the 5-day trial (Fig. 29). The decay rate for total coliform bacteria was actually negative (implying a growth or increase in numbers) in the deep pond during the 5-day trial. During the 14-day trial, fecal coliform bacteria decay rates were greatest followed by MS2 and total coliform bacteria in the shallow pond while MS2 decay rates were greatest followed by fecal coliform and total coliform decay rates in the deep pond (Fig. 30). Decay rates were similar between MS2 and fecal coliforms in both ponds during the 14-day trial, however, the total coliform decay rate was much lower in the deep pond compared to the shallow pond.

In-Pond and Sediment Concentrations

Results from samples taken within each pond during the simulated storm event for the 5-day trial are presented in Table 11. After the pump had started to fill each pond, turbidity was approximately 2 times greater in the shallow pond and nearly 7 times greater in the deep pond than at the inflow (Fig. 31). TSS were also elevated within the pond during the simulated storm event in both the shallow and deep pond. However, removal of both turbidity and TSS still occurred despite the resuspension of sediments. This is indicated by a decrease in concentration between the in-pond and outflow concentrations (Fig. 31). Total coliform concentrations were greater within the deep

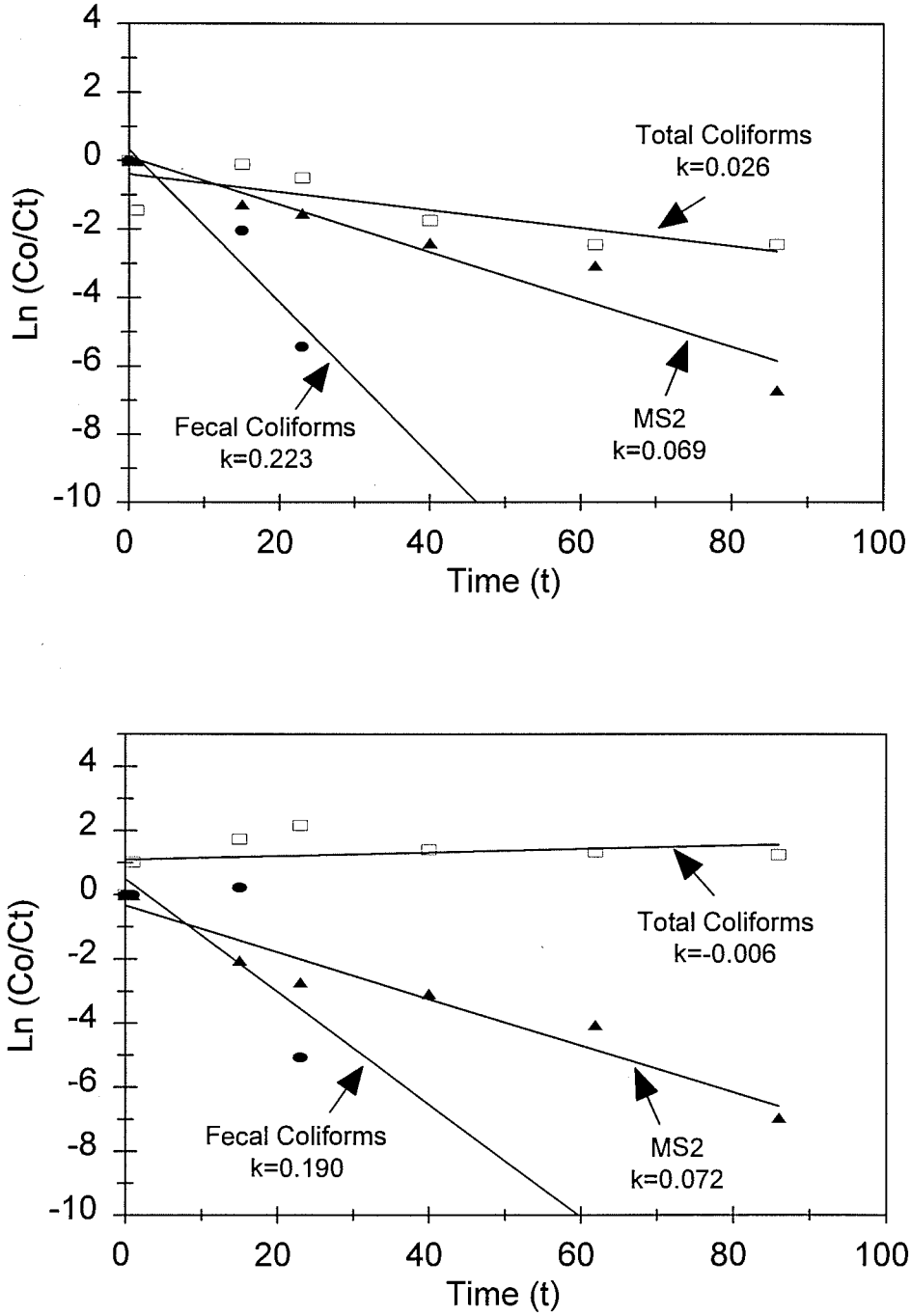


Fig. 29. Decay rates for total and fecal coliforms and MS2 during 5-day shallow (top) and deep (bottom) wet detention pond trials.

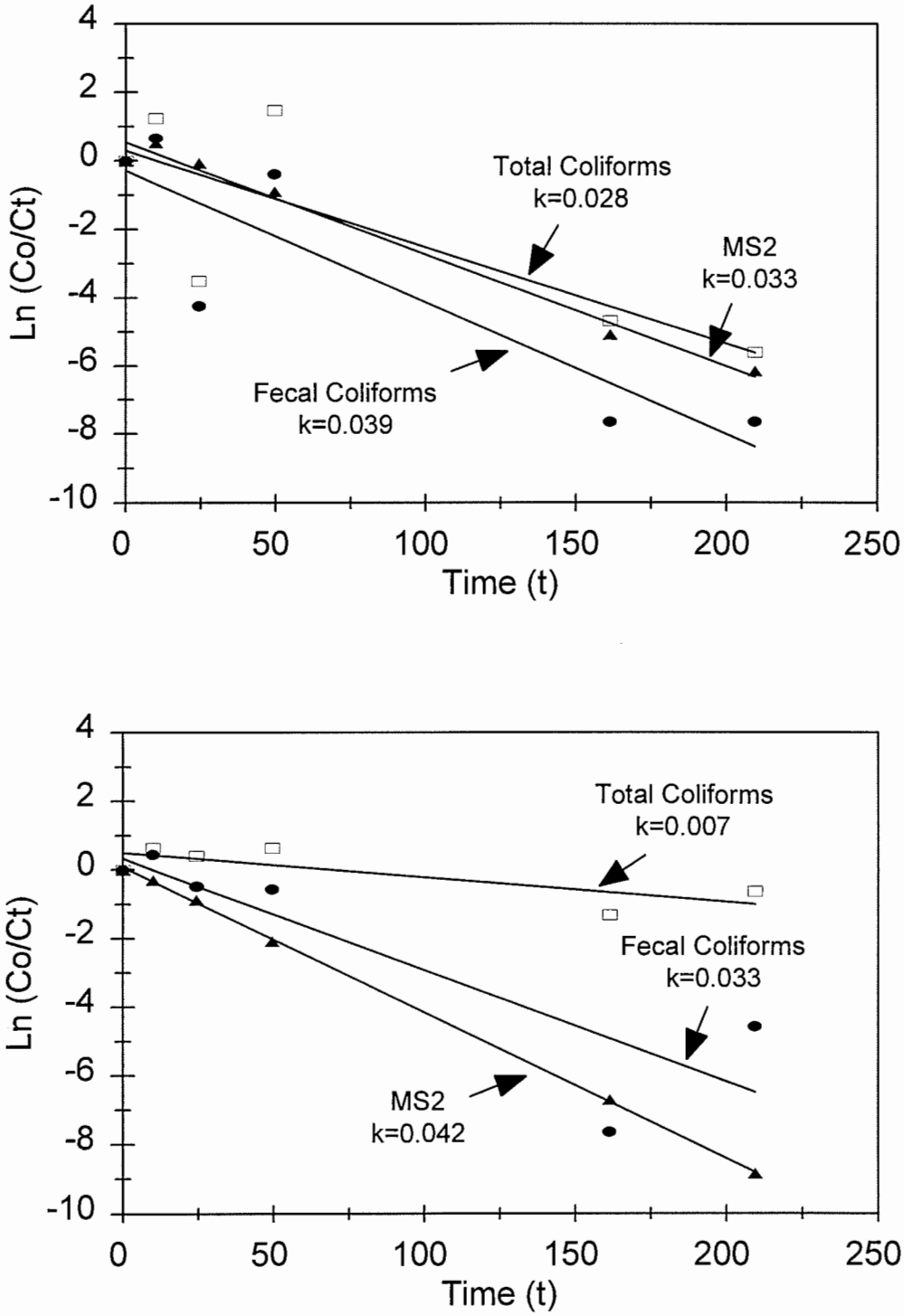


Fig. 30. Decay rates for total and fecal coliforms and MS2 during 14-day shallow (top) and deep (bottom) wet detention pond trials.

Table 11. Concentrations of indicator and physical parameters from grab samples in the deep zones of each of the wet detention ponds during the 5-day trial. Samples were taken during simulated storm flow.

Parameter	Shallow Pond	Deep Pond
Turbidity (NTU)	2.3	6.7
Total Suspended Solids (mg/L)	12.7	8.0
Total coliforms (cfu/100 ml)	4.00×10^{2}	1.9×10^{3}
Fecal coliforms (cfu/100 ml)	1.00×10^{2}	0
MS2 coliphage (pfu/ml)	3.30×10^{1}	2.00×10^{-1}
3 μm beads *(Cryptosporidium* surrogate) (beads/ml)	0	0

pond than the shallow pond and were greater than the inflow concentrations to the deep pond. Fecal coliform concentrations within the ponds were not elevated above inflow concentrations for either the shallow or deep pond.

Total coliform bacteria concentrations in sediment samples taken near the outfall structure in both the deep and shallow ponds were both $>1.6 \times 10^{4}$ cfu/g. Fecal coliform concentrations in sediments at the same locations were 3.0×10^{3} and 1.6×10^{4} cfu/g, respectively.

Monitoring Well Samples

Samples taken 7 days after the two ponds were seeded were negative for both MS2 and fluorescent beads in both monitoring wells. Fecal coliform bacteria were not present in either the initial or 7 day post-seed samples for either well, however, total

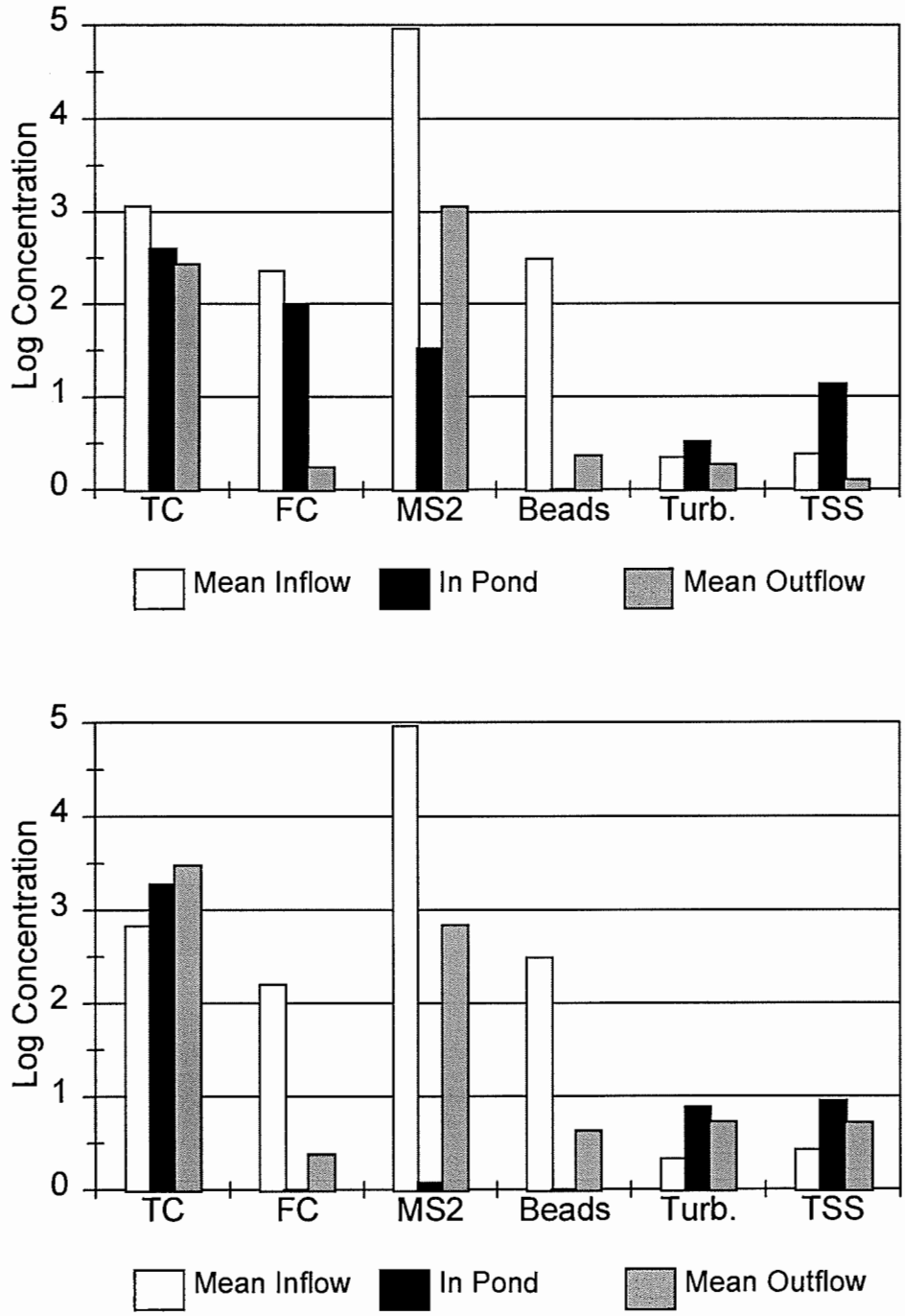


Fig. 31. Comparisons between inflow, in pond, and outflow concentrations for total coliforms (TC), fecal coliforms (FC), MS2, beads, turbidity (Turb.), and total suspended solids (TSS) in shallow (top) and deep (bottom) wet detention ponds during the 5-day trial.

coliform bacteria concentrations increased from 0 to 2300 cfu/100 ml during this period in the well adjacent to the deep pond.

Water Budget

By subtracting the inflow volume from the outflow volume and dividing by the total inflow volume, the percent of water discharged was calculated to evaluate each pond's water budget. During the 14-day simulation, approximately 55% and 54% of the inflow volume was discharged from the shallow and deep pond, respectively. Much of the remaining percentage was probably lost to evapotranspiration. During the 5-day simulation, approximately 86% and 84% of the inflow volume was discharged for the shallow and deep pond, respectively.

Comparisons with Water Quality Standards

Surface water quality standards (≤ 29 NTU above background conditions) for turbidity for Class III waters were never exceeded in either mean inflow or outflow concentrations for any of the wet detention pond trials. Total coliform bacteria concentrations exceeded Class III maximum values in 33% of inflow samples from the four pond trials. Total coliform bacteria concentrations exceeded the Class III maximum value in 83% and 60% of outflow samples from the 5-day and 14-day deep pond trials, respectively, and in 40% of outflow samples from the 14-day shallow pond trial. Fecal

coliform bacteria concentrations exceeded the Class III maximum value in 42% of inflow samples from all four pond trials. Fecal coliform bacteria concentrations exceeded the Class III maximum value in 40% of outflow samples during the 14-day shallow pond trial and in 60% of outflow samples for the 14-day deep pond trial but did not exceed the one day maximum value in outflow samples for either of the 5-day trials.

Of the six parameters, TSS, fecal coliforms, MS2, and beads were reduced sufficiently to meet the 80% reduction goals during the 5-day shallow pond trial. Fecal coliforms, MS2, and beads were reduced sufficiently to meet the 80% reduction goals during the 5-day deep pond trial, however, turbidity, TSS, and total coliform bacteria were not. MS2 and fluorescent beads were the only parameters reduced sufficiently to meet the 80% reduction goals during both the 14-day shallow and deep pond trials.

Alum Coagulation

Physicochemical Parameters

The parameters which exhibited the greatest changes during the high dose alum treatment were pH and conductivity. During the high dose trial, pH dropped from near neutral (7.0) to values <4.0 within the first 24 hours and remained below 4.0 during the remainder of the experiment (Fig. 32). In the low dose trial (Fig. 33), a slight increase in

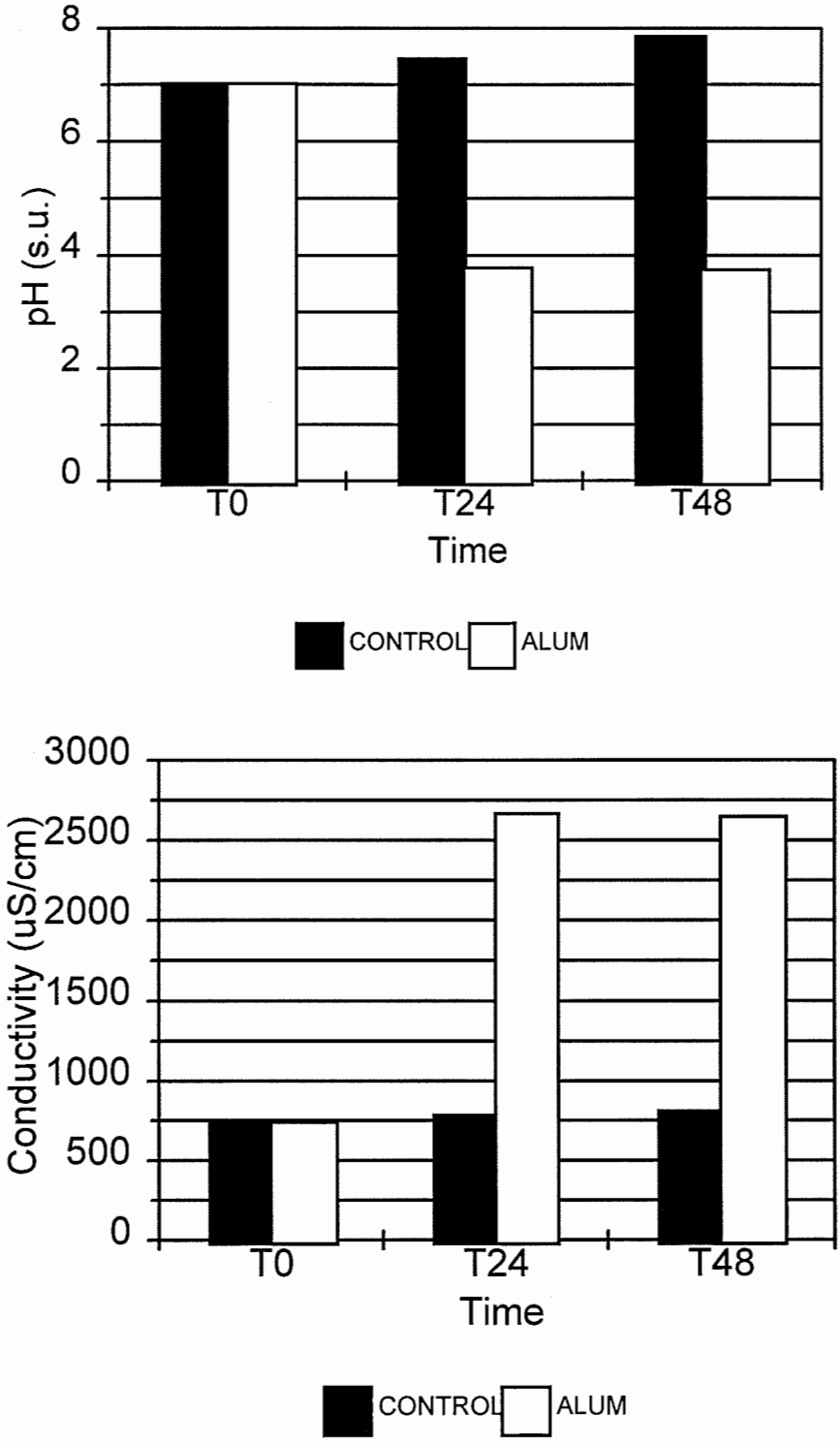
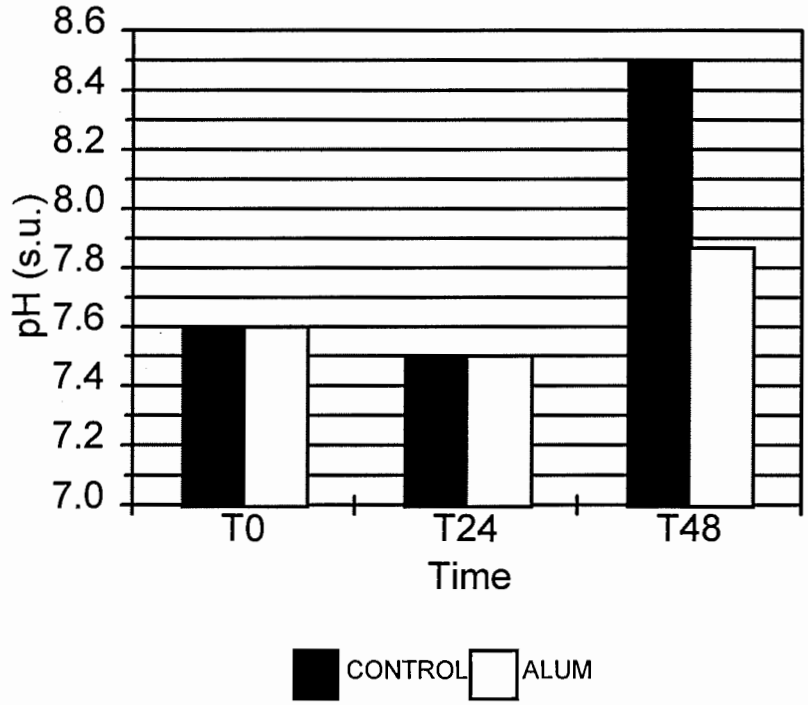


Fig. 32. Trends in pH (top) and conductivity (bottom) from Pinellas Park stormwater after dosing with alum at 600 mg/L.



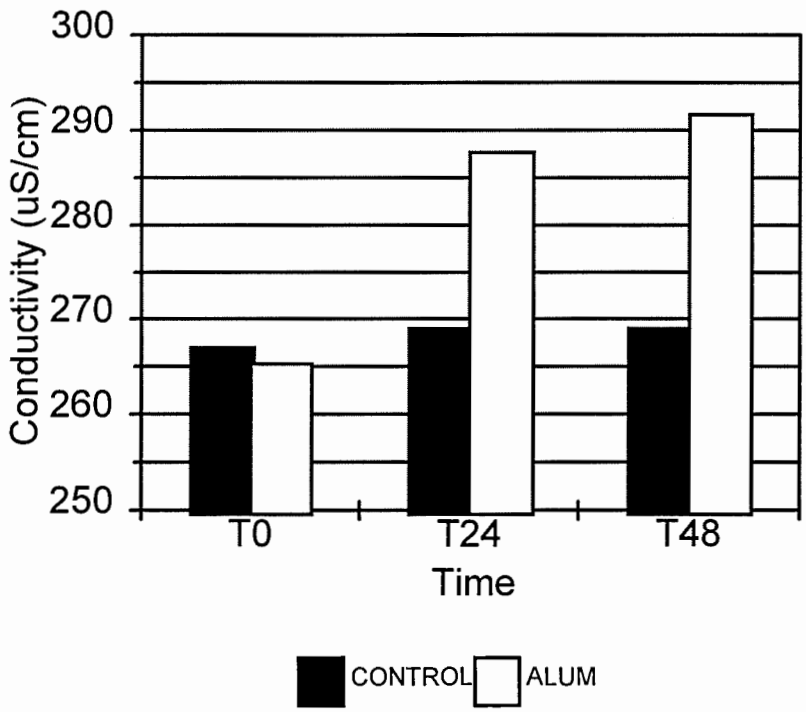


Fig. 33. Trends in pH (top) and conductivity (bottom) from Lowry Park stormwater after dosing with alum at 10 mg/L.

pH occurred from approximately 7.6 to 7.9 after 48 hours. Conductivity increased during both trials, however, the rise was much more pronounced during the high dose trial as a result of much higher concentrations of heavy metals and salts being added to the stormwater (Fig. 34). Temperature was relatively stable during the high dose trial and averaged about 25°C. During the low dose trial, a cold front passed through the Tampa Bay area just prior to sample collection. Temperatures dropped approximately 5°C within the first 24 hours of the trial and remained at approximately 13°C during the remainder of the experiment.

Microbial Indicators

Removal efficiencies and log removal values for comparisons between alum and control samples are presented in Tables 12 and 13. Greatest reductions in the concentrations of total and fecal coliforms and turbidity occurred within 24 hours after the addition of alum in both the high and low dose trials with removal efficiencies often exceeding 97% for most microbial indicators.

In the low dose (10 mg/L) jar tests, greater than 3-log reductions were observed for total and fecal coliforms and MS2 within the first 24 hours. After 48 hours, removal efficiencies (differences between the control and alum treated sample concentrations) for most parameters except TSS and fluorescent beads had declined (Table 12, Fig. 35).

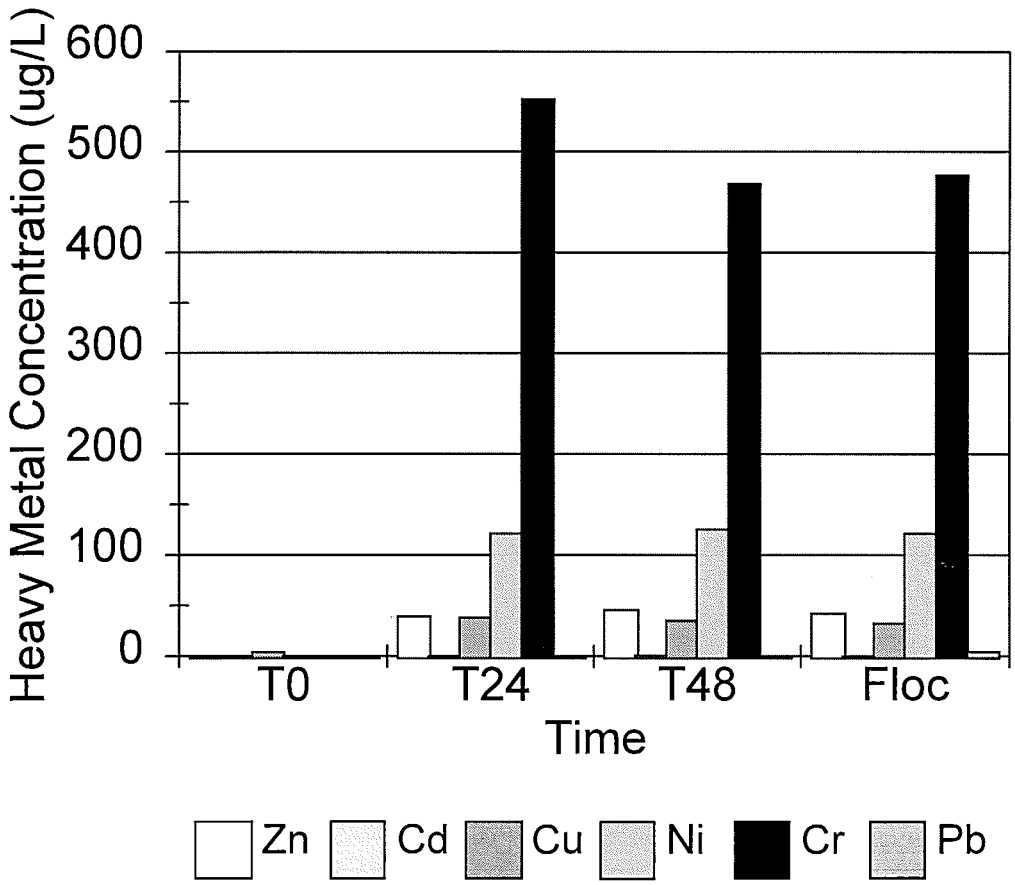


Fig. 34. Trends in heavy metal concentrations over time in alum treated stormwater using a dose of approximately 600 mg/L.

Table 12. Removal efficiencies based on differences between concentrations of indicator and physical parameters in control versus alum treated stormwater samples. Stormwater was taken from Lowry Park at a dose of 10 mg/L alum.

Parameter	**$Time_0$**	**$Time_{24}$**	**$Time_{48}$**	**Log Removal After 48 Hours**
Turbidity (NTU)	0%	88.1%	79.6%	-
Total Suspended Solids (mg/L)	0%	74.1%	84.4%	-
Total coliforms (cfu/100 ml)	0%	99.9%	98.5%	1.8
Fecal coliforms (cfu/100 ml)	0%	99.9%	99.6%	2.4
MS2 coliphage (pfu/ml)	0%	99.9995%	98.0%	1.7
3 μm beads *(Cryptosporidium* surrogate) (beads/ml)	0%	96.4%	98.2%	1.8

Table 13. Removal efficiencies based on differences between concentrations of indicator and physical parameters in control versus alum treated stormwater samples. Stormwater was taken from Pinellas Park at a dose of approximately 600 mg/L alum.

Parameter	**$Time_0$**	**$Time_{24}$**	**$Time_{48}$**	**Log Removal After 48 hours**
Turbidity (NTU)	0%	50.0%	7.6%	-
Total Suspended Solids (mg/L)	0%	-59.3%	-26.9%	-
Total coliforms (cfu/100 ml)	0%	33.3%	-3233.3%	-
Fecal coliforms (cfu/100 ml)	0%	100%	100%	>2.0
MS2 coliphage (pfu/ml)	0%	99.996%	99.998%	4.9
3 μm beads *(Cryptosporidium* surrogate) (beads/ml)	0%	81.2%	90.8%	1.0

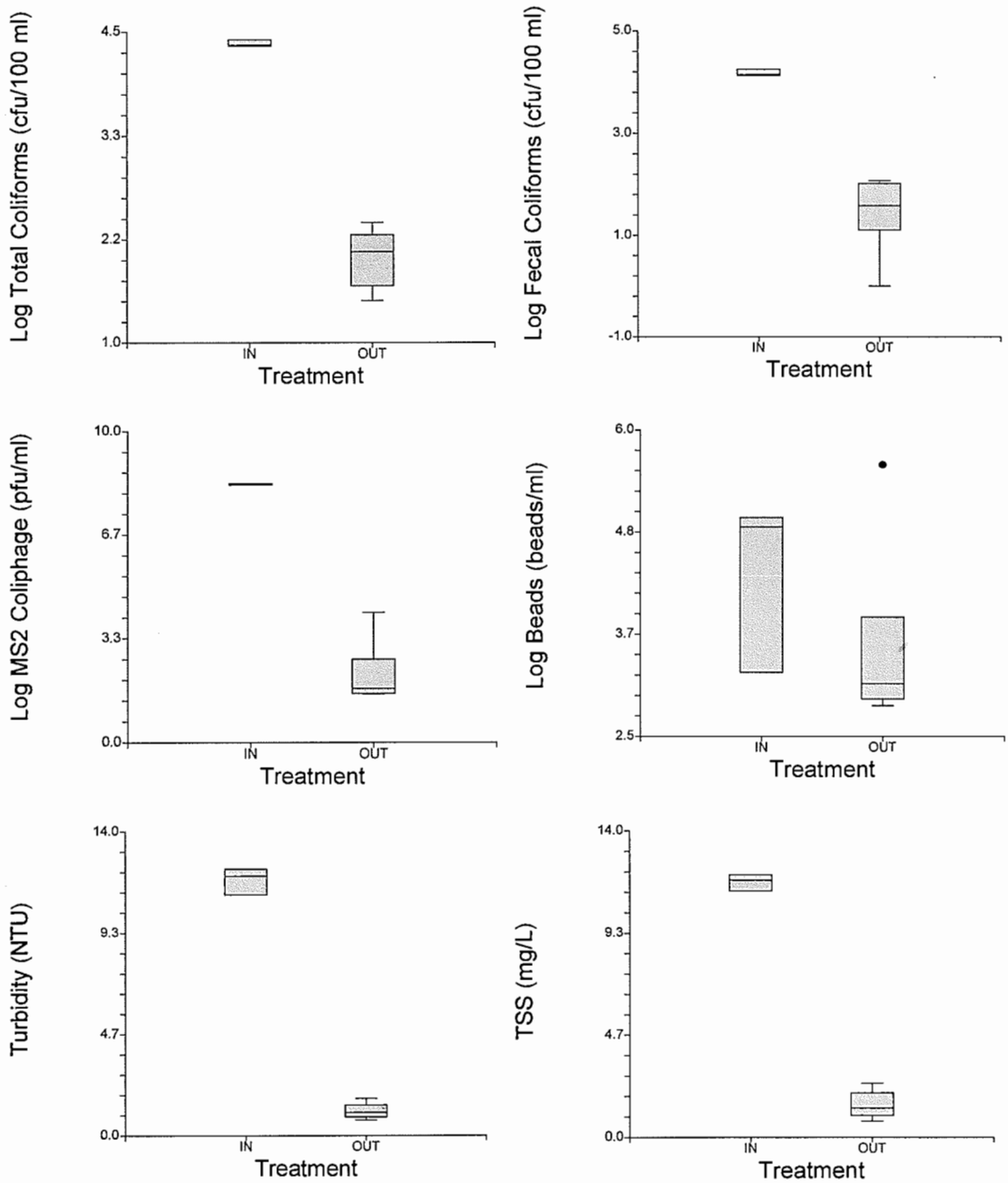


Fig. 35. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the Lowry Park stormwater seeded challenge using alum coagulation treatment.

During the low dose trial, turbidity and TSS concentrations were found at greater concentrations in the floc layer than in initial (T_0) water column concentrations prior to the addition of alum (Fig. 36). Concentrations of total and fecal coliforms and beads in the floc layer were within 1-log unit of T_0 seeded concentrations (Fig. 37). Greatest declines in MS2 concentrations occurred between T_0 and T_{24} and then remained relatively low, even in the floc layer (Fig. 38). Total and fecal coliform concentrations were significantly greater ($p \leq 0.05$) in the floc layer than in the water column 48 hours after the addition of alum. Bead concentrations were significantly greater in the floc than at T_0 or after 24 hours but not after 48 hours (Fig. 39). Log-transformed total coliform concentrations were positively correlated with TSS and log-transformed bead concentrations were positively correlated with turbidity.

During the high dose (600 mg/ L) jar tests, greatest removal efficiencies occurred within 24 hours for turbidity and total and fecal coliforms while removal efficiencies for TSS, MS2, and beads were greater after 48 hours (Table, 13, Fig. 40). Negative TSS and total coliform removal efficiencies were observed after 48 hours. Microscopic examination of undiluted T_0 and T_{24} samples revealed floc materialsin both control and alum treated samples. The appearance of alum floc may be a result of either the resuspension of floc material during sampling, a thicker than expected floc layer which extended into the sample collection area of the jar, or contamination of the source water from the full-scale alum treatment system located downstream (which may have unintentionally back-flushed alum upstream to the sample collection point for the jar

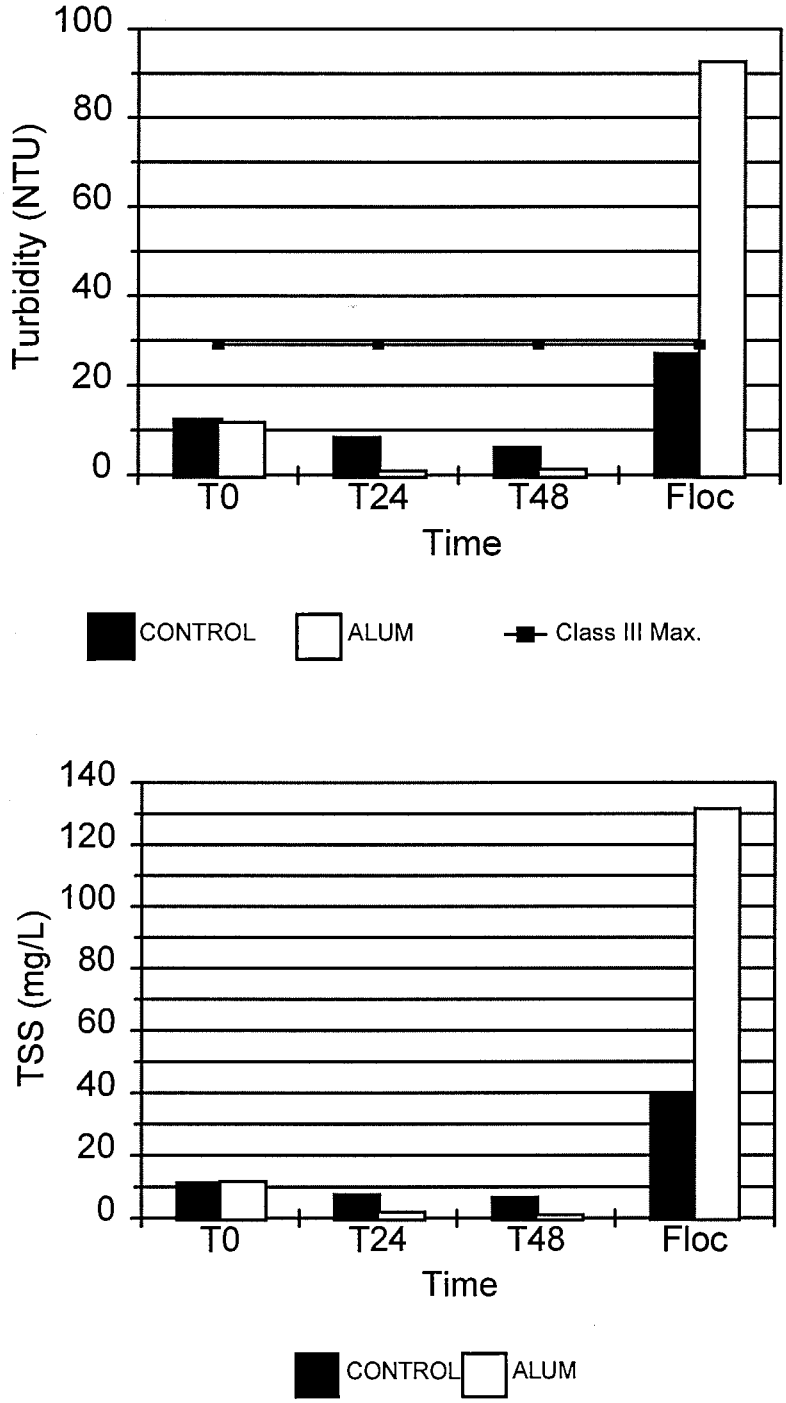
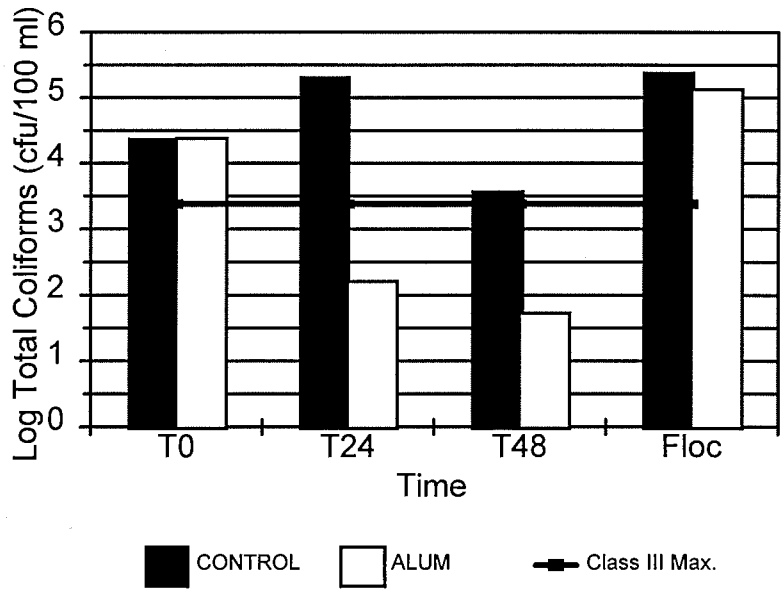


Fig. 36. Trends in the removal of turbidity (top) and total suspended solids (bottom) from Lowry Park stormwater using an alum dose of 10 mg/L.



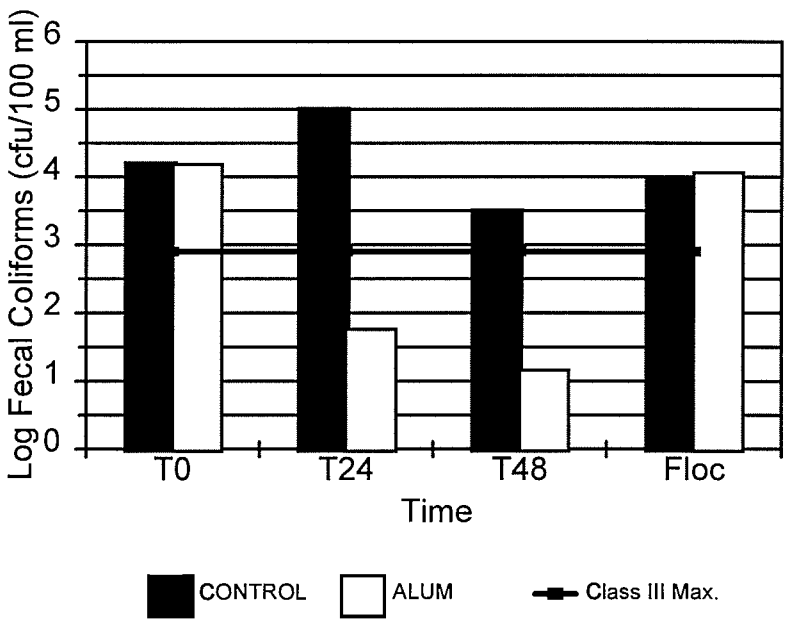


Fig. 37. Trends in the removal of total (top) and fecal (bottom) coliform bacteria from Lowry Park stormwater using an alum dose of 10 mg/L.

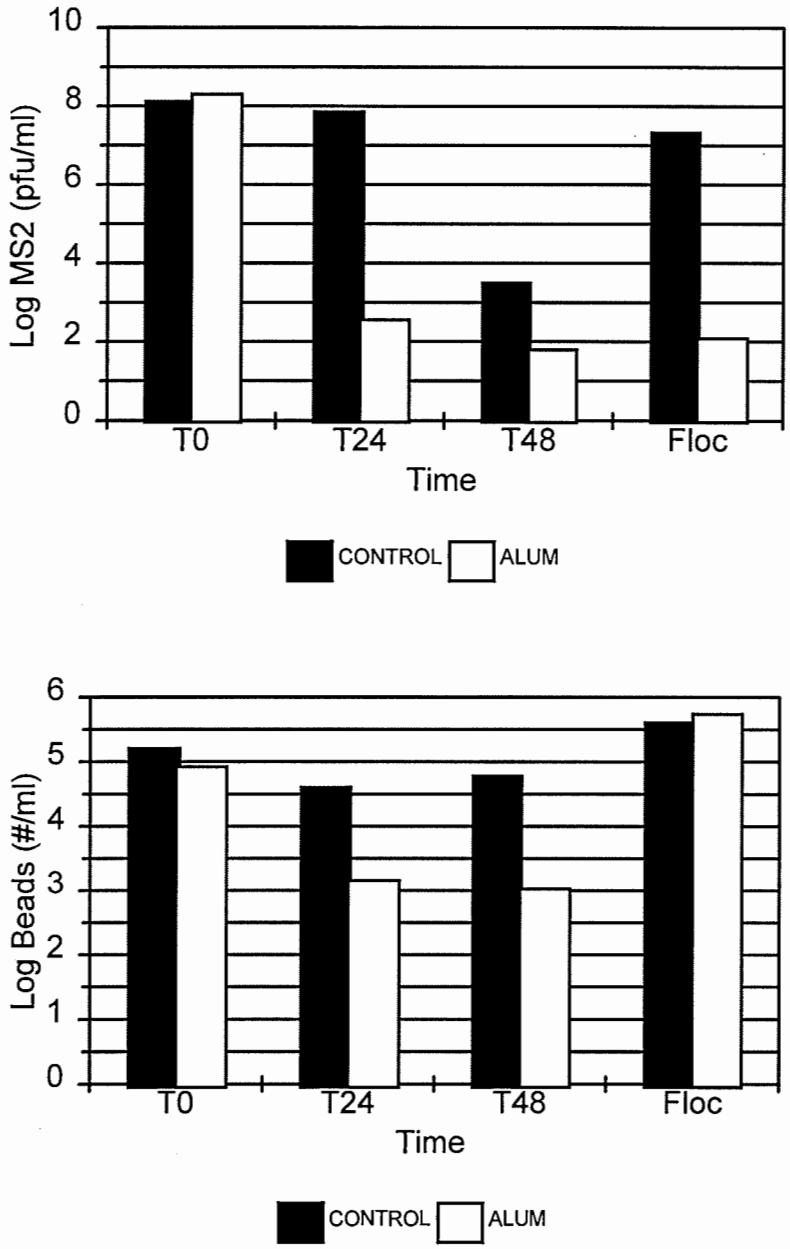


Fig. 38. Trends in the removal of MS2 (top), and 3 μm fluorescent beads (bottom) from Lowry Park stormwater using an alum dose of 10 mg/L.

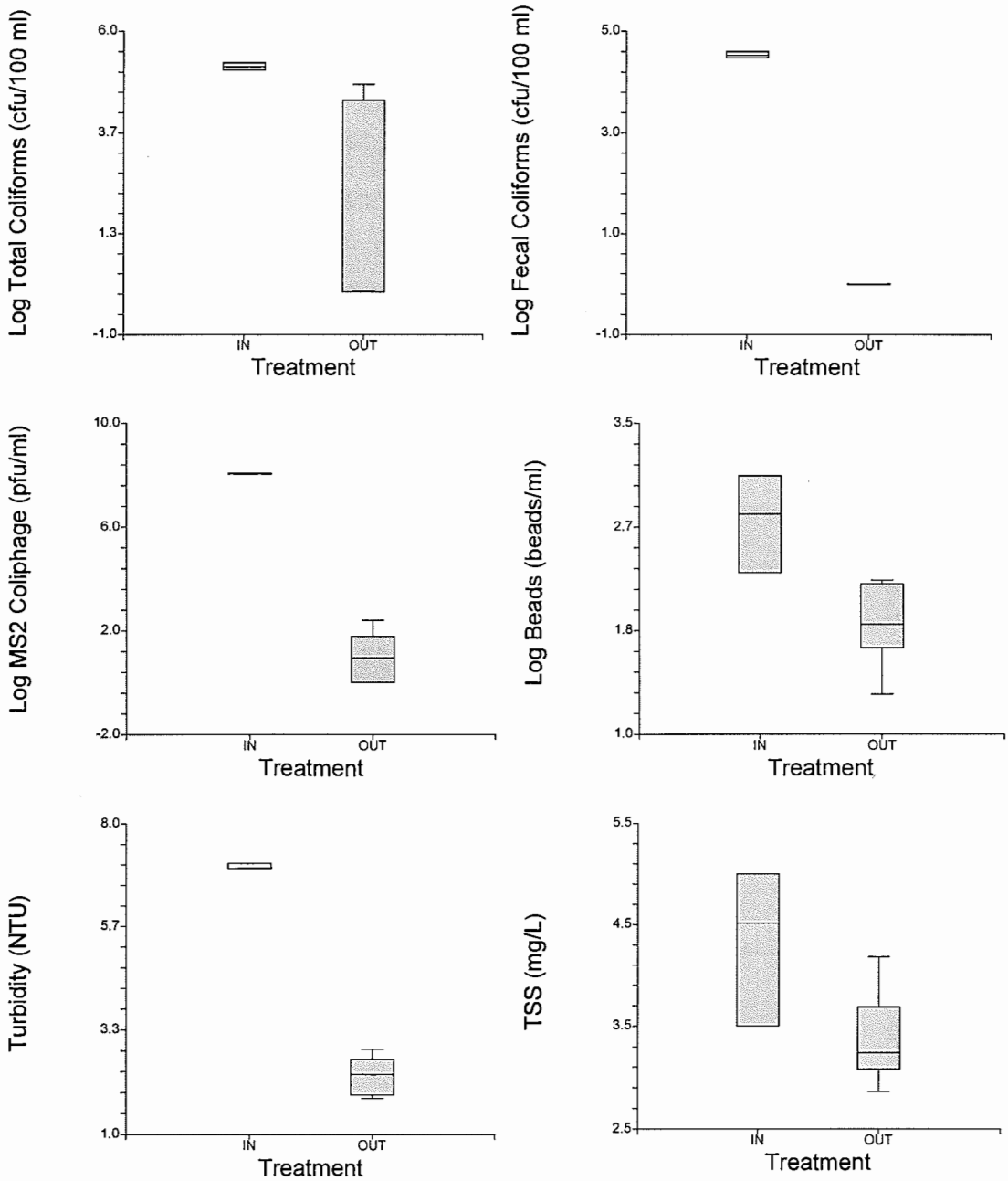


Fig. 39. Box plots comparing inflow and outflow concentrations of total coliforms, fecal coliforms, MS2 coliphage, 3 μm fluorescent beads, turbidity, and total suspended solids during the Pinellas Park stormwater seeded challenge using alum coagulation treatment.

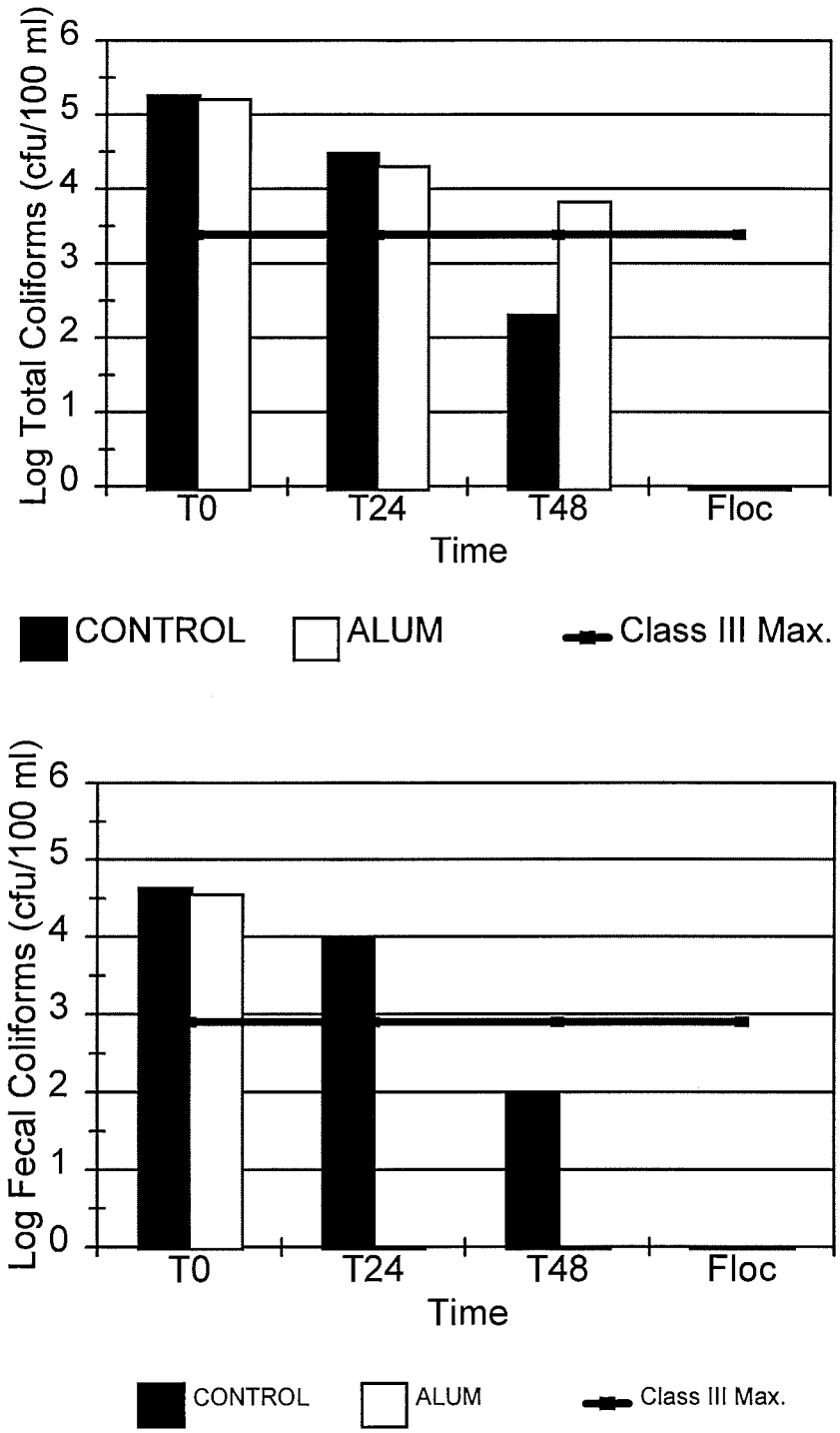


Fig. 40. Trends in the removal of total (top) and fecal (bottom) coliform bacteria from Pinellas Park stormwater using an alum dose of 600 mg/L.

tests). This phenomenon was also confirmed by elevated Al concentrations during the trial as well as greater than expected conductivity values.

The presence of alum at the collection point may reduce the validity of comparisons between control and alum treated samples, however, it does help explain the greater than expected declines for all of the microbial indicators, turbidity, and TSS in the control samples (Figs. 40 through 42). MS2, fecal coliforms, and beads were positively correlated with turbidity as a result of elevated die-off/inactivation rates and a significant decline in turbidity.

Decay Rates

Decay rates were greatest for MS2 followed by total coliforms and fecal coliforms in low dose control samples (Fig. 43). In high dose control samples, total coliform bacteria decay rates were greatest followed by fecal coliforms and MS2. Obvious differences occurred between low dose and high dose decay rates for both total and fecal coliforms. In the low dose control samples, total and fecal coliform decay rates were generally less than 0.038/hr, however, during the high dose trial, decay values were greater than 0.102/hr, probably as a result of alum contamination.

MS2 decay rates were greater than both fecal and total coliform decay values during both the low and high dose trials which was also reflected in greater MS2 removal efficiency values (Fig. 44). Total and fecal coliform decay rates were similar in the low dose trial but fecal coliform decay values were much greater than total coliform values

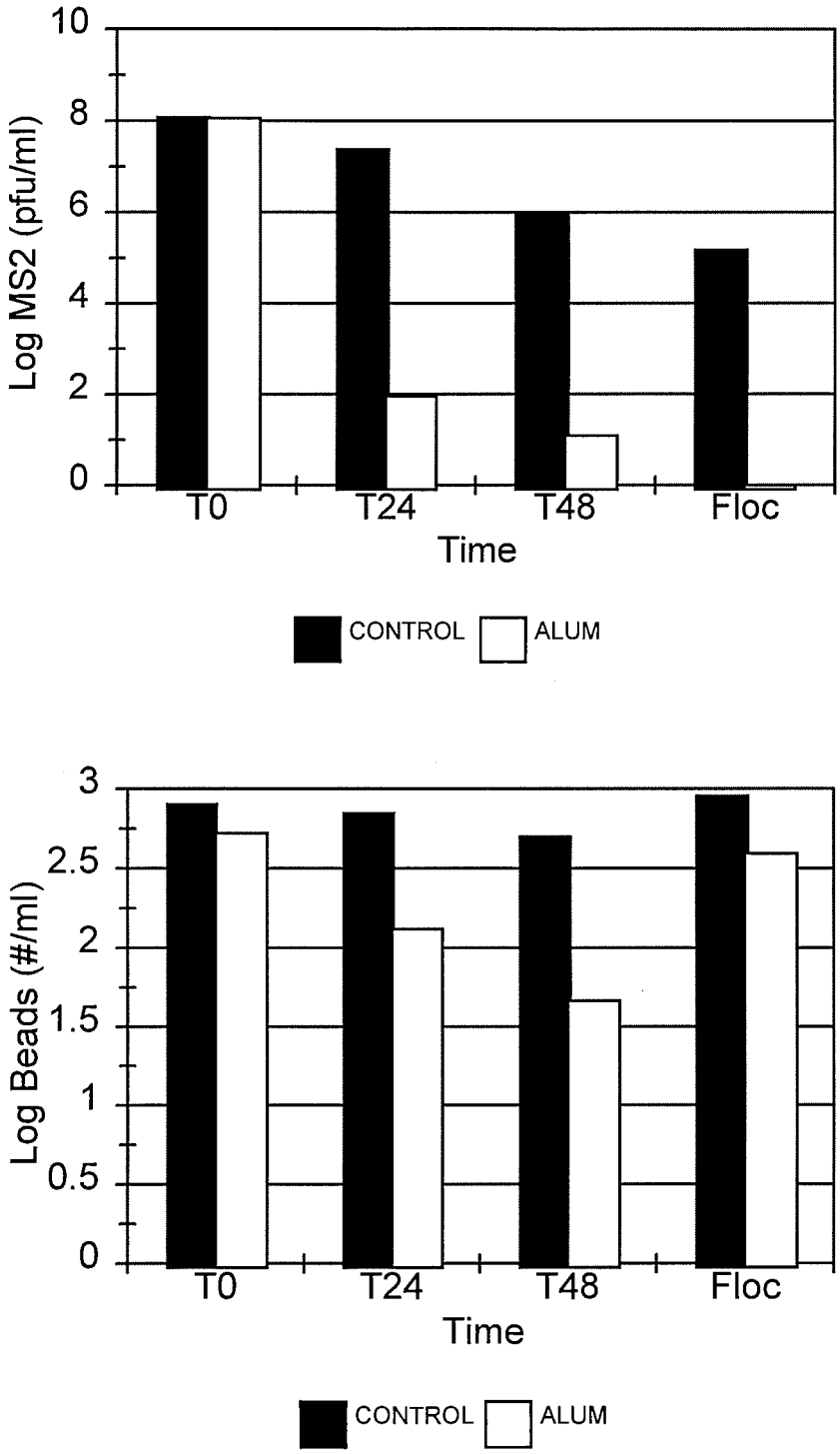


Fig. 41. Trends in the removal of MS2 (top) and 3 μm fluorescent beads (bottom) from Pinellas Park stormwater using an alum dose of 600 mg/L.

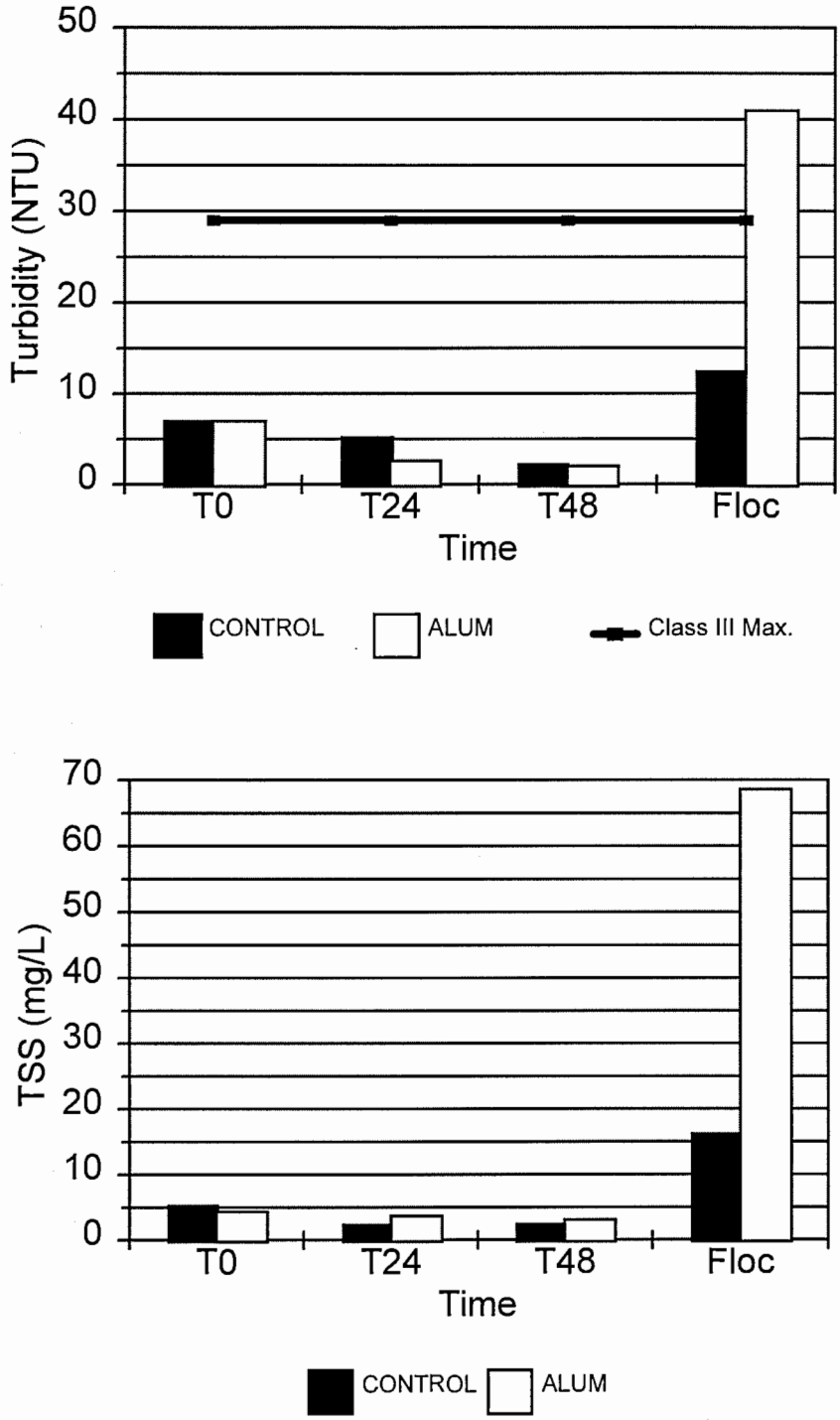


Fig. 42. Trends in the removal of turbidity (top) and total suspended solids (bottom) from Pinellas Park stormwater using an alum dose of 600 mg/L.

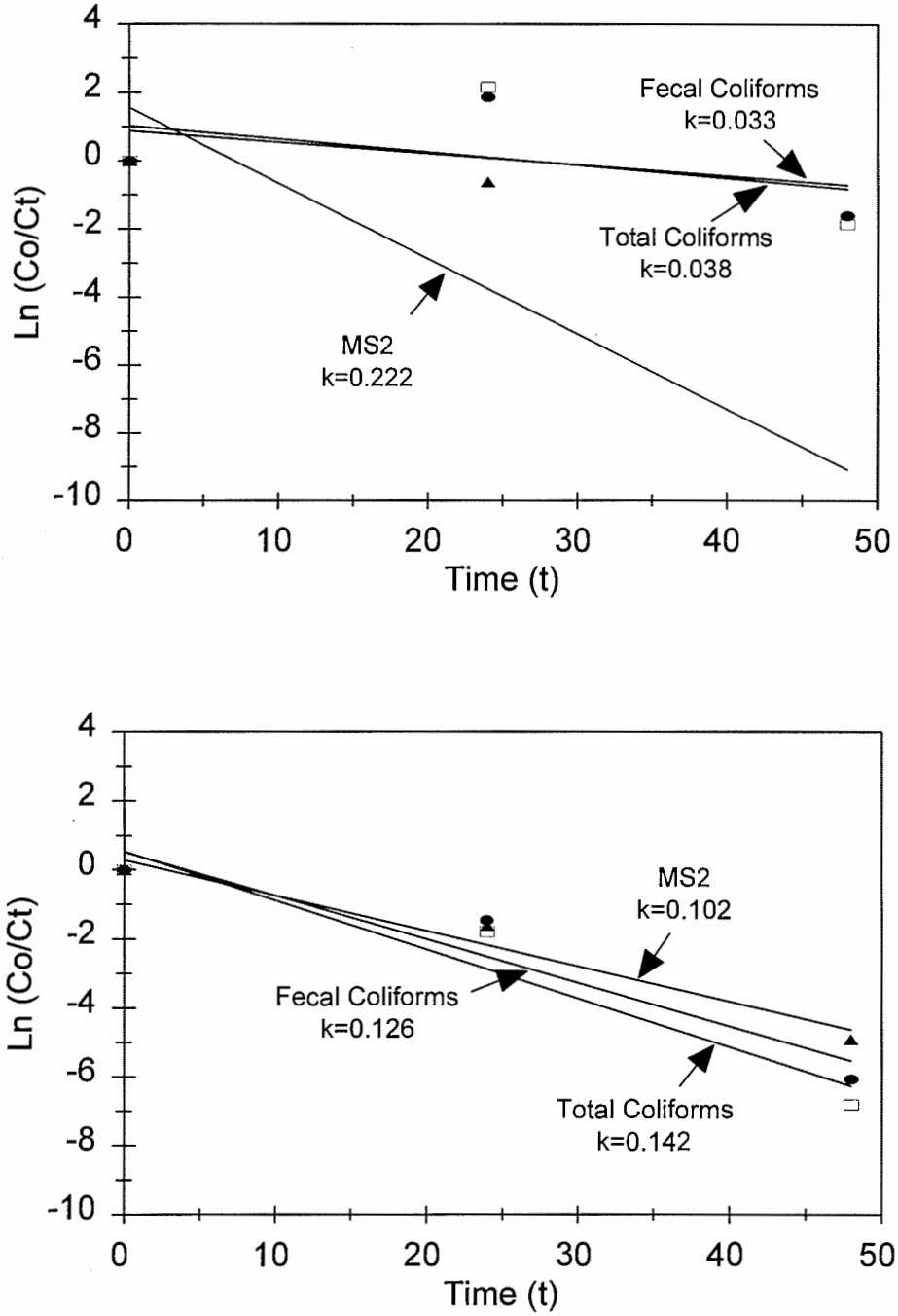


Fig. 43. Decay rates for total and fecal coliforms and MS2 using control (no alum) sample data during low dose (top) and high dose (bottom) alum trials.

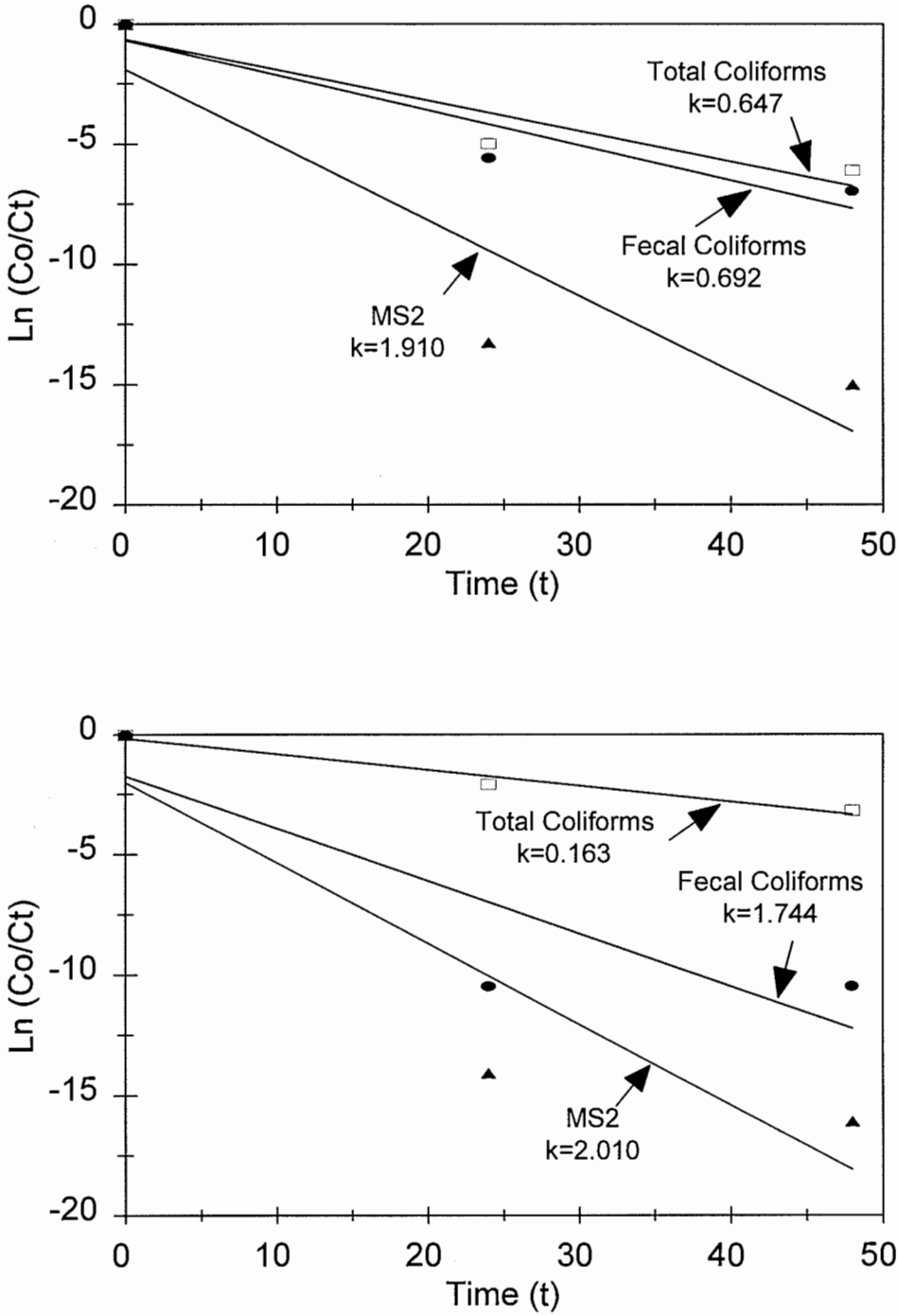


Fig. 44. Decay rates for total and fecal coliforms and MS2 during low dose (top) and high dose (bottom) alum treatments.

during the high dose trial. This pattern may suggest an increased susceptibility of various species within the fecal coliform bacteria group to metal toxicity.

Bacteria Speciation

A number of gram-negative bacteria were identified during each of the two trials and included several capable of causing human disease (*E. coli*, *Klebsiella pneumoniae*, and *Salmonella enteritidis*) (Table 14). None of the various bacterial species appeared to be removed differentially as a result of alum treatment. The complete lack of any fecal coliform bacteria in either the water column or floc layer after the addition of alum in the high dose trial suggests that this group of bacteria may be more susceptible to metal toxicity than other species from the total coliform group which were still present after 24 and 48 hours (Fig. 40).

Comparison with Water Quality Standards

Surface water quality standards for turbidity (≤ 29 NTU above background conditions) for Class III waters were never exceeded in any of the initial (T_0) nor subsequent samples after 24 and 48 hours. However, discharge of the concentrated floc would violate the Class III standards for turbidity. Both total and fecal coliform bacteria concentrations exceeded the Class III one day maximum value in all (100%) T_0 samples during both the high and low alum dose trials. Total coliform bacteria concentrations did

not exceed the Class III maximum value in any of the 48-hour (T_{48}) low dose alum samples but did exceed the one day maximum value for 50% of all T_{24} and T_{48} control samples and 33% of all T_{24} and T_{48} high dose alum samples. Fecal coliform bacteria concentrations did not exceed the Class III maximum value in any of the 48-hour (T_{48}) high or low dose alum samples but the one day

Table 14. List of coliform bacteria identified in control (CON) and replicate alum-treated stormwater samples taken from Lowry Park. No bacteria were present after alum treatment in the high dose trial. *X* denotes presence in sample.

Species	**CONTROL**	**CONTROL FLOC**	**STORMWATER w/ALUM**	**ALUM FLOC**
Enterobacter aerogenes	*X*			*X*
Enterobacter agglomerans				
Enterobacter cloacae				
Enterobacter gergoviae				
Enterobacter sakazakii				
Escherichia coli	*X*	*X*	*X*	*X*
Klebsiella ozaenae	*X*	*X*	*X*	
Klebsiella pneumoniae	*X*	*X*		*X*
Salmonella enteritidis				*X*
Serratia liquefaciens	*X*	*X*		*X*
Serratia marcescens	*X*			
Serratia rubidea			*X*	
Citrobacter freundii	*X*			
Arizona sp.	*X*			*X*

maximum value was exceeded in 50% of all control samples. Total and fecal coliform concentrations in floc samples from both alum treated and control tests from the low dose trial would have exceeded Class III standards if discharged to a protected waterbody. TSS, total and fecal coliforms, MS2, and beads were reduced sufficiently to meet the 80% reduction goals during the 10 mg/L trial. Total and fecal coliforms, MS2, and beads were reduced sufficiently during the 600 mg/L trial to meet the 80% reduction goal, however, TSS was not.

Relationships Between Parameters Using Data From All Three Sites

MS2 as an Indicator for Total and Fecal Coliforms

Background concentrations of MS2 (prior to seeding) were analyzed using linear regression to determine potential relationships with total and fecal coliform bacteria concentrations. Log-transformed total ($r^2 = 0.67$) and fecal ($r^2 = 0.61$) coliform concentrations were positively correlated ($p \leq 0.05$) with log-transformed MS2 concentrations (Fig. 45).

Relationships with Turbidity

Inflow concentrations of total and fecal coliforms for all data combined were both positively correlated ($p \leq 0.05$) with turbidity (Fig. 46). The r^2 value for total coliforms

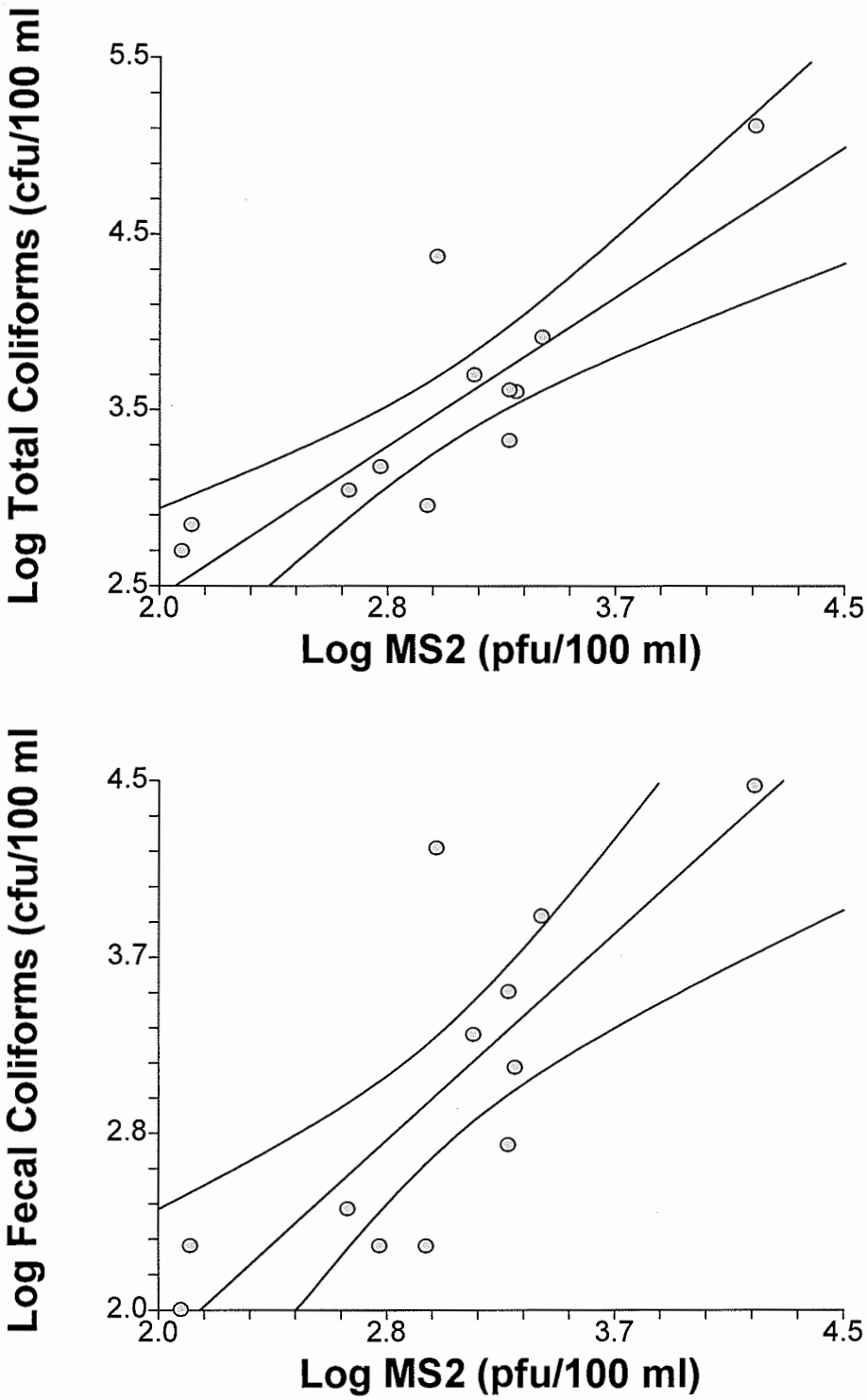


Fig. 45. Relationships between log-transformed MS2 concentrations and log-transformed total (top) and fecal (bottom) coliform concentrations from inflow samples from the three stormwater BMPs.

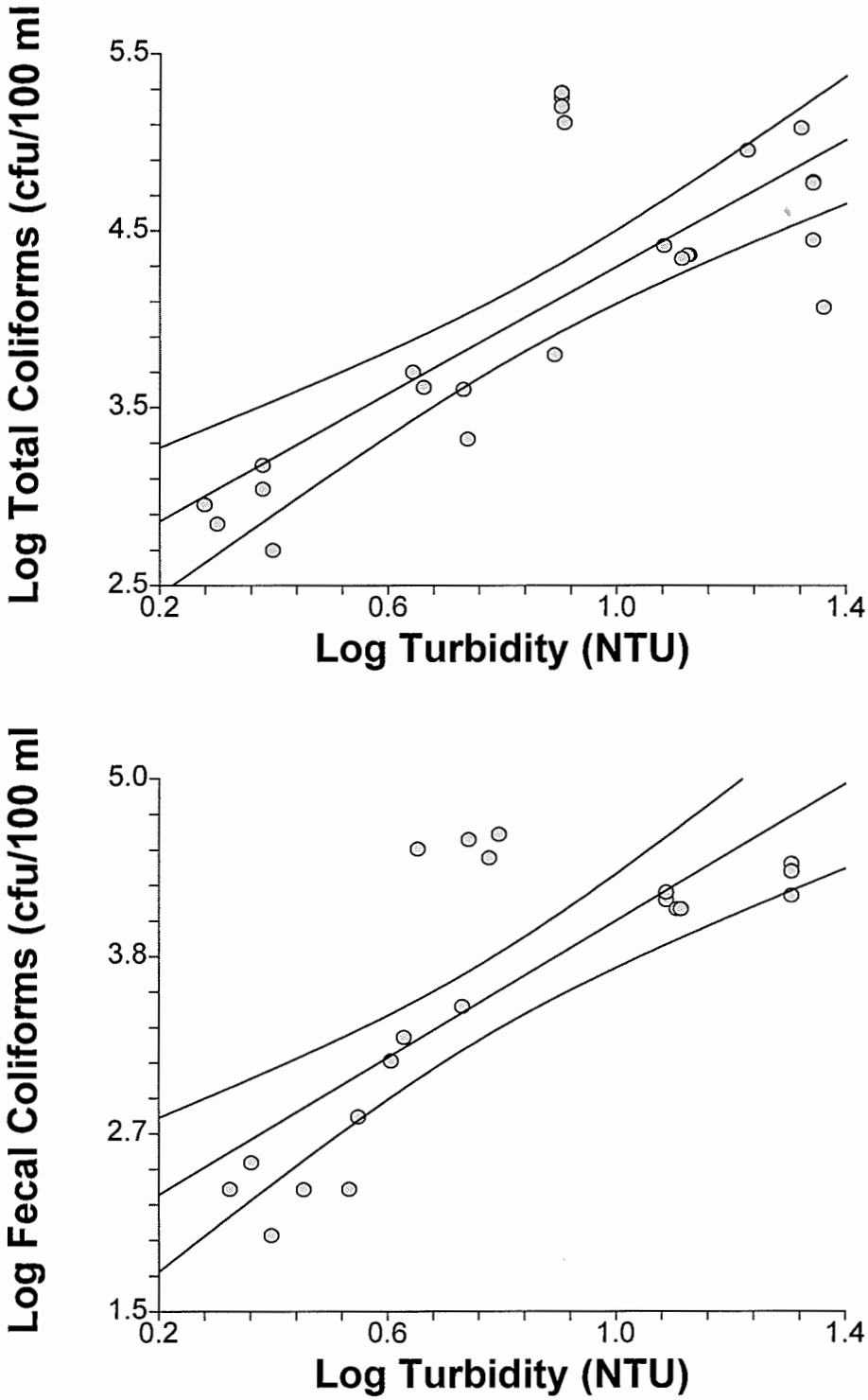


Fig. 46. Regression of total coliform (top) and fecal coliform (bottom) values with turbidity for all inflow samples from the three stormwater BMP sites.

was 0.78 compared to an r^2 value of 0.70 for fecal coliforms. Since fecal coliform bacteria comprise a subset of the total coliform group, a lower r^2 value for fecal coliforms would be expected given that fecal coliform concentrations are consistently lower than total coliform concentrations. This assumption was true for this study, since the mean fecal:total coliform ratio ($\bar{x} = 0.85 \pm .11$) was less than 1.0 and (Fig. 47).

Outflow concentrations of total and fecal coliforms, MS2, and beads were all positively correlated with turbidity. The significant, positive correlation between these indicators suggests that adsorption to particulate matter in the water column may be facilitating both their removal (through sedimentation) and transport out of the treatment pond. The significant differences between inflow and outflow ratios of fecal:total coliform bacteria may also indicate greater susceptibility and die-off of fecal coliform bacteria than the more tolerant soil-associated bacteria (Fig. 47).

Comparisons Between Exceedences of Surface Water Quality Standards

The percentages of total and fecal coliform samples exceeding Class III water quality standards were lowest and most similar between the low and high dose alum treatments, the 5-day shallow wet detention pond, and the unsaturated sand filter trial (Table 17). The greatest percentages of total and fecal coliform concentrations exceeding this standard occurred during both 14-day wet detention pond trials and saturated sand filtration trials. The 5-day deep wet detention pond had the greatest percentage of exceedence for total coliform concentrations.

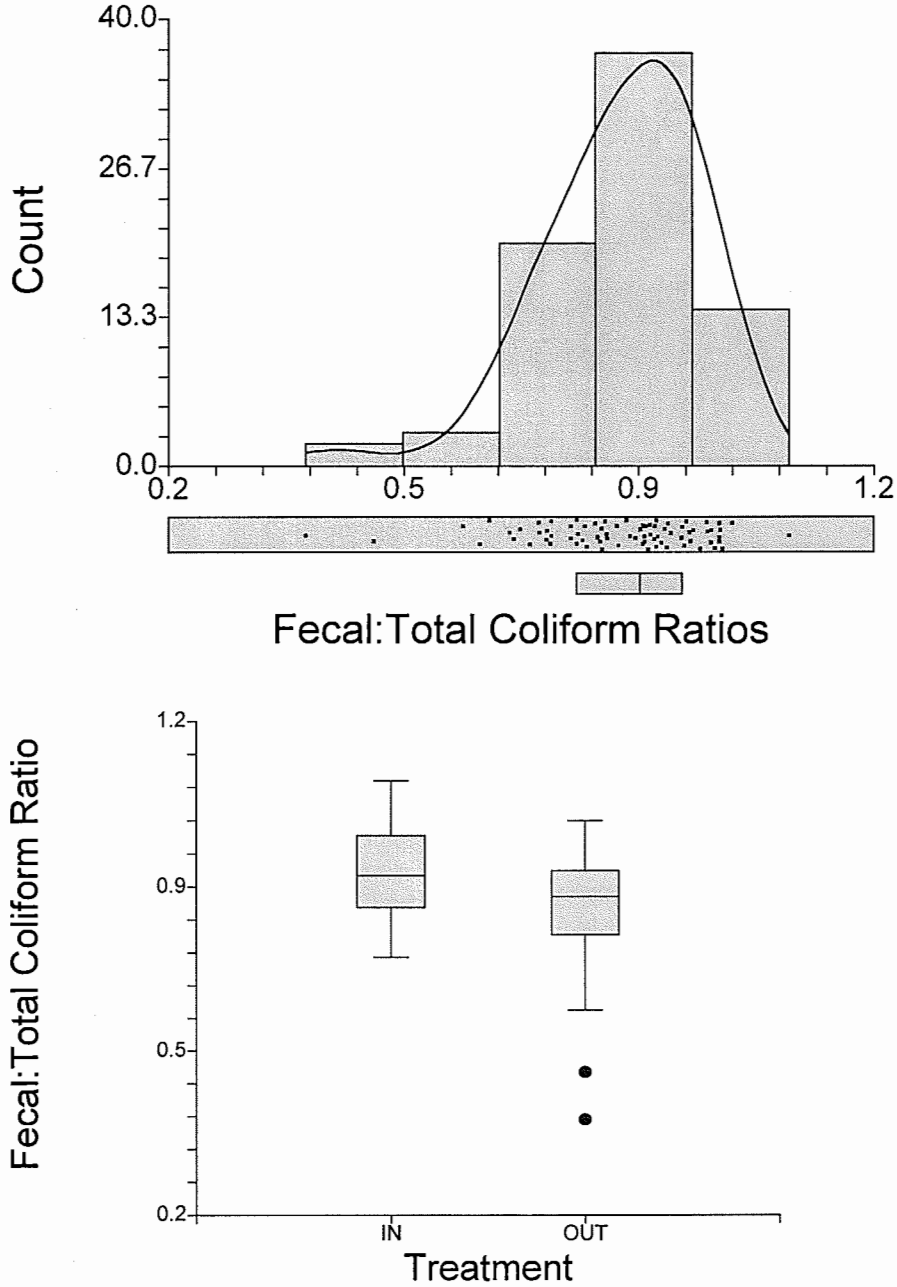


Fig. 47. (Top) Histogram showing frequencies of fecal:total coliform ratios for all data combined from the sand filtration, wet detention, and alum treatment BMP's. Dot plot below x-axis indicates spread and scatter of data. Box plot below dot plot indicates median and upper and lower quartiles of fecal:total coliform values. (Bottom) Box plot comparing fecal:total coliform ratios for inflow and outflow samples taken from sand filtration, alum treatment, and wet detention ponds. Inflow (IN) ratios were significantly greater ($p < 0.05$) than outflow ratios.

Table 15. Comparisons of exceedences of state surface water quality standards between sand filtration, wet detention, and alum treatment of stormwater.

	% of Outflow Samples Exceeding Class III Standards	
Treatment	**Total Coliforms**	**Fecal Coliforms**
Sand Filtration		
Unsaturated	0%	10%
Saturated	65%	55%
Wet Detention		
5-Day Shallow	0%	0%
5-Day Deep	83%	0%
14-Day Shallow	40%	40%
14-Day Deep	60%	60%
Alum Treatment		
High Dose (600 mg/L)	33%	0%
Low Dose (10 mg/L)	0%	0%

These comparisons were confirmed further by including MS2 and bead concentrations and using nearest-neighbor cluster analysis (NCSS®, 1997). The stormwater treatment BMPs which had the most similar removal efficiency values for all four indicators were the high and low dose alum treatments (Fig. 48). The next most similar treatment trials were the 14-day shallow and deep wet detention ponds, followed by the sand filtration and 5-day shallow wet detention pond systems. The 5-day deep wet detention pond had the greatest dissimilarity to any of the other treatment trials which was probably due to an extreme negative total coliform removal efficiency value.

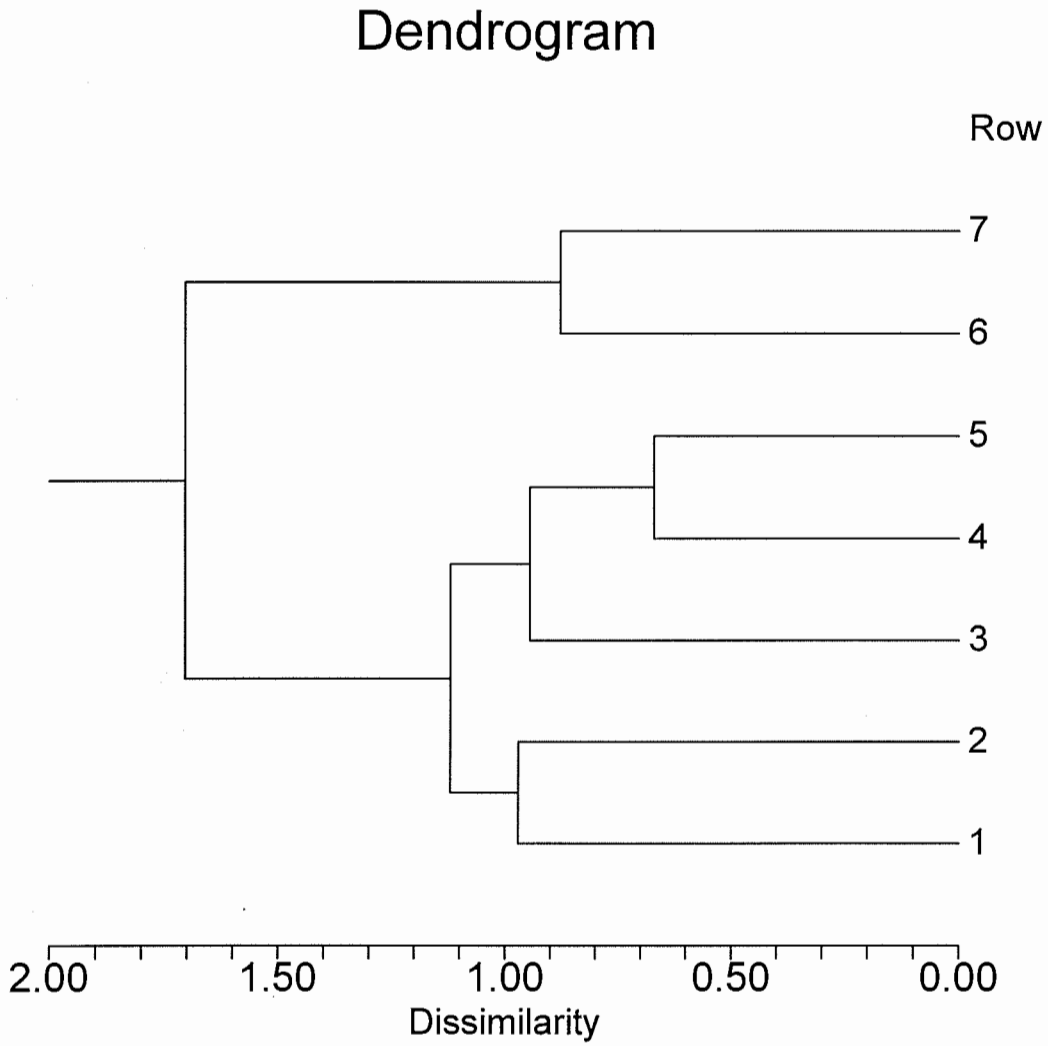


Fig. 48. Resulting dendrogram from cluster analysis of removal efficiency comparisons of microbial indicators for the seven different stormwater treatment trials (three trials from the sand filtration experiment were combined). 1=Sand, 2=Wet Detention (5-day, shallow pond), 3=Wet Detention (5-day, deep pond), 4=Wet Detention (14-day, shallow pond), 5=Wet Detention (14-day, deep pond), 6=Alum (high dose, 1000 mg/L), 7=Alum (low dose, 10 mg/L).

Comparisons between Sand Filtration, Wet Detention, and Alum Coagulation

Low dose alum coagulation treatment resulted in the greatest overall removal efficiency values for total and fecal coliforms and turbidity (Fig. 49). MS2 removal was greatest using alum treatment, but was also typically greater than 80% for all other BMPs. Removal efficiencies for beads were greater than 90% for all three treatment systems. The greatest bead removal (99.5%) was identical for sand filtration, 5-day shallow, and 14-day deep wet detention pond treatments. Greatest turbidity removal was achieved using the sand filter followed closely by alum treatment (low dose). Total suspended solids removal was greatest during the 5-day shallow pond treatment followed by alum treatment (low dose).

Treatment Train Reductions

The use of a multiple treatment system in which several different BMPs are joined in series may offer greater reductions for a broader collection of parameters than any single BMP. Since no single BMP evaluated during this study had consistently greater removals of all the parameters, this type of approach would be more effective, especially since the removal of bacteria was relatively poor using both sand filtration and wet detention. However, as discussed more thoroughly below, removal calculations for treatments in series are not always additive, since the easiest contaminants to remove are typically taken out during the first treatment phase.

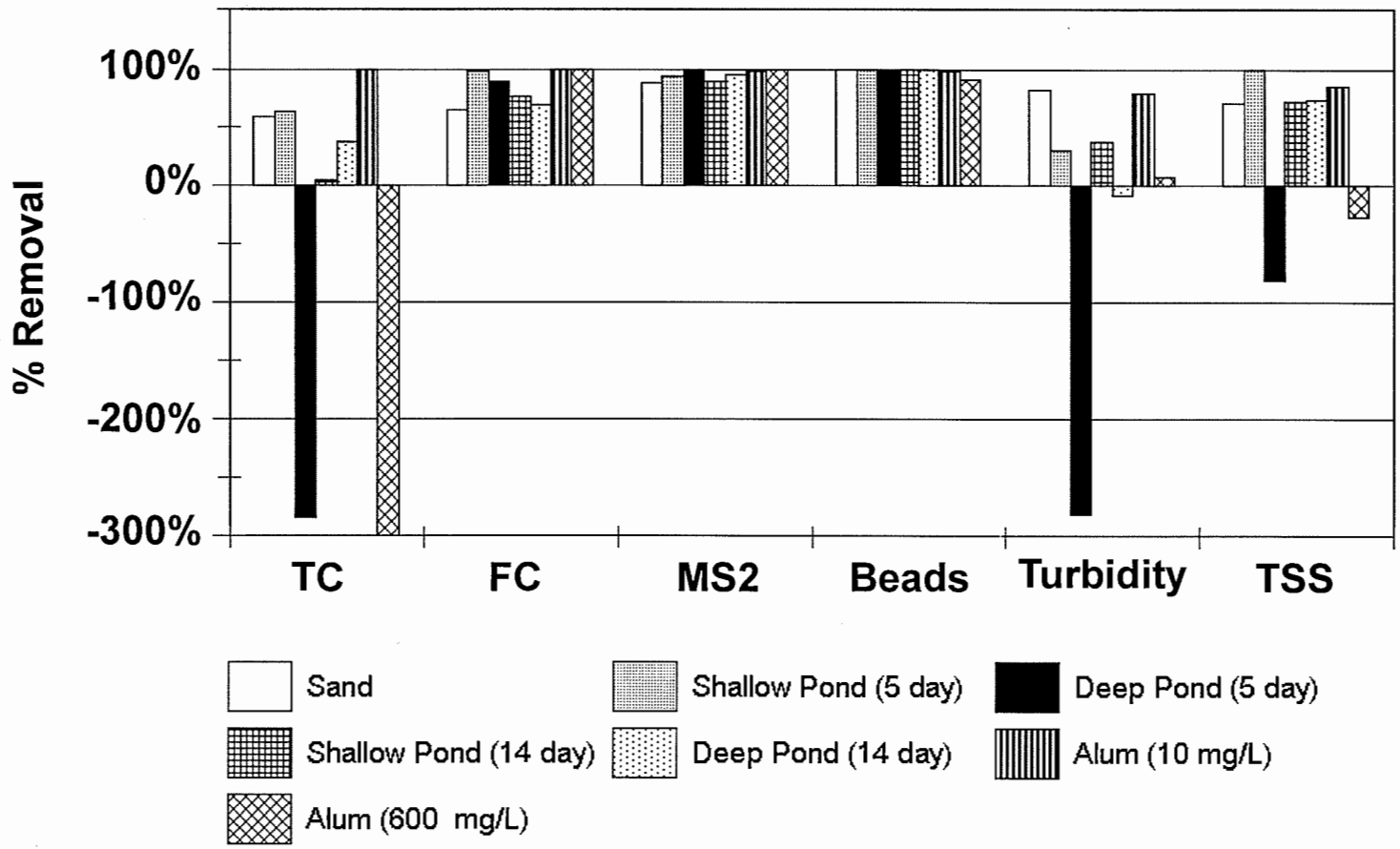


Fig. 49. Removal efficiencies of three stormwater treatment methods around Tampa Bay, Florida.

Using a series of treatments which include alum coagulation, wet detention, and sand filtration, greater than 90% reductions may be realized for all the parameters measured during this study (Table 18, Fig. 50). This includes using a penalty factor of 1.5 which reduces the efficiency of removal between consecutive treatment systems. In fact, MS2 may be removed by as much as 8-log units and both total and fecal coliform bacteria by 3 and 5-log units, respectively.

In cases where sand filters or alum coagulation systems are not logistically or financially feasible, a modified wet pond design may offer greater microbial pathogen removal than the commonly-used open pond design (Fig. 51). By incorporating a baffling system often used in primary settling basins in wastewater and drinking water treatment plant designs, greater detention/travel times can be achieved which can result in greater sedimentation and removal of microorganisms and suspended solids.

Potential Reductions in Health Risks

The removal of pathogenic microorganisms using any of the three stormwater BMPs in this study may result in a potential reduction in health risks from contaminated stormwater (Table 19). For the purposes of this study, a reduction in health risk was determined if the inflow concentration of a particular group of pathogenic microorganisms is reduced substantially to pose little or no health threat to a person exposed (via direct ingestion) to waters discharged at the outflow. The calculations assume that water from the outfall of a stormwater treatment system is not diluted and the

Table 16. Treatment train reduction calculations.

Method of Treatment	Turbidity (NTU)	TSS (mg/L)	Reduction (1 - % removal) Total Coliforms (cfu/100 ml)	Fecal Coliforms (cfu/100 ml)	MS2 (pfu/100 ml)	Beads (beads/ml)
Alum (10 mg/L)	0.107	0.412	0.002	0.001	2.000E-07	0.009
Shallow Pond (5-day detention)	0.697	0.002	0.360	0.018	0.061	0.005
Sand Filter	0.176	0.290	0.406	0.346	0.123	0.005
Initial Concentration	**50**	**25**	**1,000,000**	**1,000,000**	**1,000**	**5**
after Alum	8.0	15.50	3000.0	1500.0	0.00030	0.068
after Shallow Pond	8.4	0.05	1620.0	40.5	0.00003	0.001
after Sand Filtration	2.2	0.02	986.6	21.0	0.00001	0.000
Overall Percent Reduction	**95.6%**	**99.92%**	**99.90%**	**99.998%**	**99.9999996%**	**99.99992%**
Overall Log Reduction			**3.0**	**4.7**	**8.3**	**6.1**
Penalty Factor:	*1.5*					

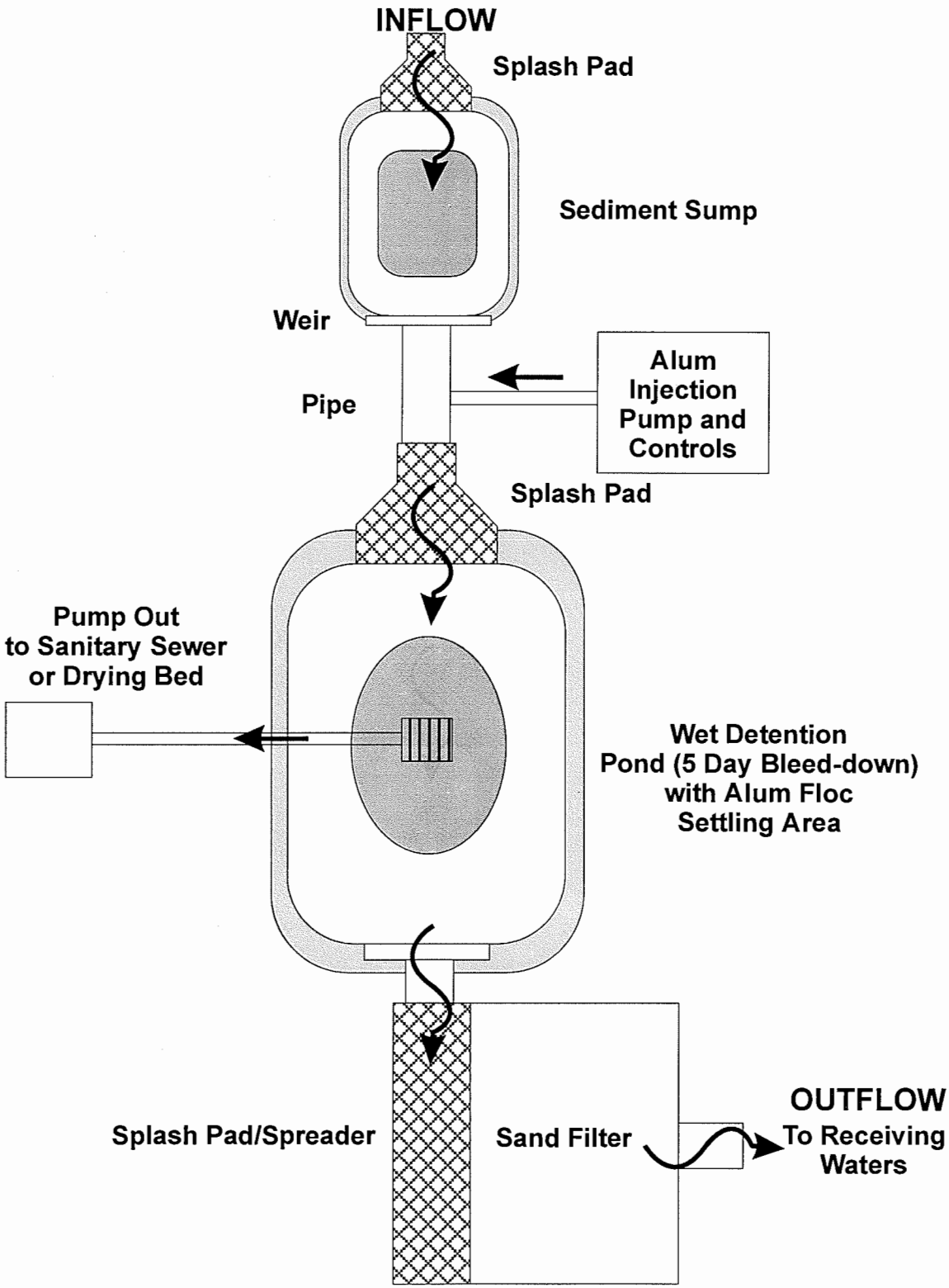


Fig. 50. Schematic of a treatment train or series of stormwater BMPs that could provide optimal microorganism removal.

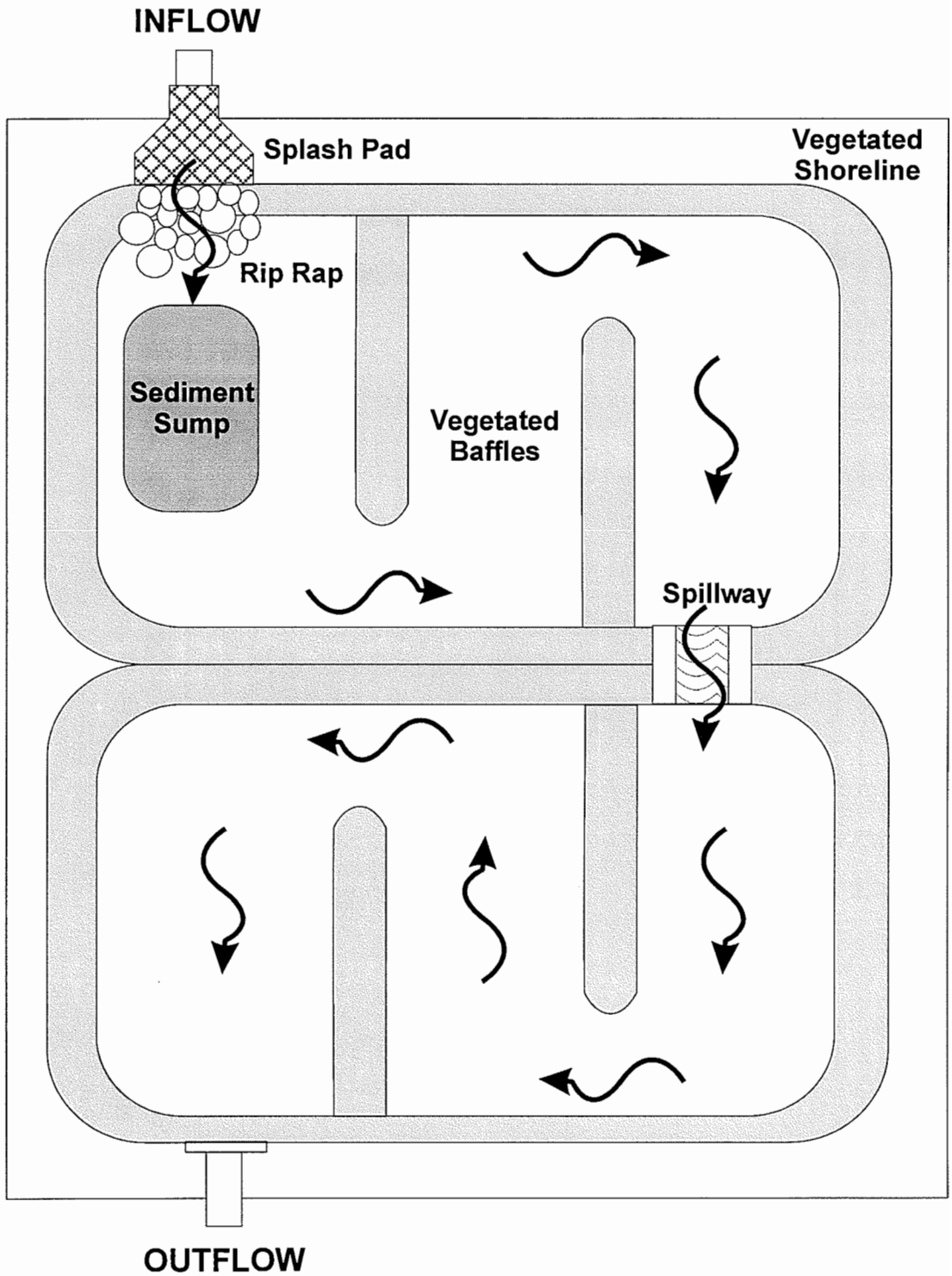


Fig. 51. Schematic of a wet pond treatment design that could provide greater microorganism removal than current commonly-used designs.

Table 17. Potential reductions in risk of disease using various stormwater treatment technologies.

Treatment	Pathogen	Level of Contam-ination	Initial Concentration (no./ml)	Removal Efficiency	Reduced Value	Infective Dose	Ingested Dose**	Risk Difference	Risk Reduction
Alum	Bacteria*	High	1.0×10^{4}	99.9%	1.0×10^{3}	1.0×10^{1}	1.0×10^{3}	9.9×10^{2}	N
		Medium	1.0×10^{3}	99.9%	1.0×10^{1}	1.0×10^{1}	1.0×10^{2}	9.0×10^{1}	N
		Low	1.0×10^{2}	99.9%	1.0×10^{-1}	1.0×10^{1}	1.0×10^{1}	8.9×10^{-15}	Y
	Enterovirus*	High	1.0×10^{4}	99.99998%	2.0×10^{-3}	1.0×10^{1}	2.0×10^{-1}	-9.8×10^{0}	Y
		Medium	1.0×10^{3}	99.99998%	2.0×10^{-4}	1.0×10^{1}	2.0×10^{-1}	-1.0×10^{0}	Y
		Low	1.0×10^{2}	99.99998%	2.0×10^{-5}	1.0×10^{1}	2.0×10^{-1}	-1.0×10^{1}	Y
	Protozoa	High	2.0×10^{4}	99.1%	1.8×10^{2}	1.32×10^{2}	1.8×10^{4}	1.8×10^{4}	N
		Medium	2.0×10^{3}	99.1%	1.8×10^{1}	1.32×10^{2}	1.8×10^{3}	1.7×10^{3}	N
		Low	2.0×10^{2}	99.1%	1.8×10^{0}	1.32×10^{2}	1.8×10^{2}	4.8×10^{1}	N
Wet Detention	Bacteria*	High	1.0×10^{4}	98.2%	1.8×10^{2}	1.0×10^{1}	1.8×10^{4}	1.8×10^{4}	N
		Medium	1.0×10^{3}	98.2%	1.8×10^{1}	1.0×10^{1}	1.8×10^{3}	1.8×10^{3}	N
		Low	1.0×10^{2}	98.2%	1.8×10^{0}	1.0×10^{1}	1.8×10^{2}	1.7×10^{1}	N
	Enterovirus*	High	1.0×10^{4}	93.9%	6.1×10^{2}	1.0×10^{1}	6.1×10^{4}	6.1×10^{4}	N
		Medium	1.0×10^{3}	93.9%	6.1×10^{1}	1.0×10^{1}	6.1×10^{3}	6.1×10^{3}	N
		Low	1.0×10^{2}	93.9%	6.1×10^{0}	1.0×10^{1}	6.1×10^{2}	6.0×10^{2}	N

Table 17. (Continued).

	Protozoa	High	2.0×10^4	99.5%	1.0×10^2	1.32×10^2	1.0×10^4	9.9×10^3	N
		Medium	2.0×10^3	99.5%	1.0×10^1	1.32×10^2	1.0×10^3	8.7×10^2	N
		Low	2.0×10^2	99.5%	1.0×10^0	1.32×10^2	1.0×10^2	-3.2×10^1	Y
Sand Filtration	Bacteria*	High	1.0×10^4	65.4%	3.5×10^5	1.0×10^1	3.5×10^5	3.5×10^5	N
		Medium	1.0×10^3	65.4%	3.5×10^3	1.0×10^1	3.5×10^4	3.5×10^4	N
		Low	1.0×10^2	65.4%	3.5×10^1	1.0×10^1	3.5×10^3	3.5×10^3	N
	Enterovirus*	High	1.0×10^4	87.7%	1.2×10^3	1.0×10^1	1.2×10^5	1.2×10^5	N
		Medium	1.0×10^3	87.7%	1.2×10^2	1.0×10^1	1.2×10^4	1.2×10^4	N
		Low	1.0×10^2	87.7%	1.2×10^1	1.0×10^1	1.2×10^3	1.2×10^3	N
	Protozoa	High	2.0×10^4	99.5%	1.0×10^2	1.32×10^2	1.0×10^4	9.9×10^3	N
		Medium	2.0×10^3	99.5%	1.0×10^1	1.32×10^2	1.0×10^3	8.7×10^2	N
		Low	2.0×10^2	99.5%	1.0×10^0	1.32×10^2	1.0×10^2	-3.2×10^1	Y

* Estimate of High concentration from O'Shea and Field (1992)

** Ingested dose assumes a person ingests an average of 100 ml of undiluted stormwater.

person ingests an average of 100 ml of water. An estimate of ingested dose was calculated by multiplying the outflow concentration by 100 ml and the difference between this value and the infective dose determined either a positive (Y) or negative (N) reduction in risk.

For bacteria, a conservative estimate of 10 vegetative cells was used as the infective dose with inflow concentrations ranging from 1.0×10^2 to 1.0×10^4 cfu/100 ml. Positive reductions in risk only occurred during low inflow concentrations using alum treatment (at a dose of 10 mg/L). For enteroviruses, a range of concentrations from 1.0×10^2 to 1.0×10^4 pfu/ml at the inflow was used with an infective dose of 10 virions. For protozoa (specifically *Cryptosporidium*), a range of concentrations from 2.0×10^2 to 2.0×10^4 oocysts/ml was used with an infective dose of 132 oocysts. Positive reductions in risk for enteroviruses were only observed using alum treatment (all levels of contamination). For protozoa, reductions in risk were only observed using wet detention and sand filtration at low levels of contamination.

CHAPTER FOUR

DISCUSSION

Total and fecal coliform densities found in the source waters used in the seeded trials for this study were similar to other surveys in the U.S. (Schillinger and Gannon, 1985; Ellis, 1988; Edwards *et al.*, 1997; Moorhead *et al.*, 1998). High titers of MS2 phage and *Cryptosporidium* oocysts at the magnitude used in this study are not typically found in source waters except in extremely contaminated agricultural (pasture) runoff or during sewage overflow events (Lijklema *et al.*, 1987).

Individual Stormwater Treatment Systems

Sand Filter

Temperature increased slightly during the course of sand filtration treatment, however, this increase may have been due to the exposure of the cooler raw water from the holding tank to the warmer ambient conditions in the open filter chamber. The significant difference in pH between inflow and outflow samples may have been due to

lower pH conditions in the soils within the sand filter. This may have been caused by chemical reactions in the soil resulting from microbial activity (metabolism) and the breakdown of organic material. Conductivity was significantly greater in the outflow samples probably as a result of fine silts and metals being flushed from the filter during the initial filtration period.

Microparticle reduction in sand filters typically occurs through two principal mechanisms - straining (for particles larger than the interstitial spaces between sand grains), and adsorption (for smaller, colloidal particles). Other, secondary mechanisms include flocculation and sedimentation of particles between and in the filter medium matrix. Factors that might affect adsorption include ambient pH, cation exchange capacity, percent clays, and ionic strength (McConnell *et al.*, 1984). The results of the three seeded trials demonstrated that microorganisms and microparticles can be reduced by the above-ground sand filter system. The ability of the treatment facility to remove total and fecal coliform bacteria to levels meeting state water quality standards was also demonstrated; however, the basis for this determination was made using the average discharge concentrations from the entire volume of raw stormwater pumped onto the filter. If only the first few liters of filtrate had been used to assess the performance of the facility, total and fecal coliform concentrations would have exceeded the one day maximum values for both shellfish harvesting and recreational waters in two of three trials for total coliforms and in all three trials for fecal coliforms.

In comparison to MS2 and bead removal, the sand filtration system performed relatively poorly in removing bacteria. Horner *et al.* (1994) reported even poorer removal

efficiencies of fecal coliform bacteria for sand filters in Texas which ranged from 36 to 37%. There are several explanations for this phenomenon. Both total and fecal coliform bacteria are known to exist in soils (Hunter and MacDonald, 1991) and can often survive in an aqueous environment for extended periods of time when associated with sediments (Lijklema *et al.*, 1987). Soil material from lawn runoff appeared to have created a rich organic filter skin on the surface of all three filter beds. Sediments and other organic materials that form this dark filter skin or *schmutzdecke* (German for "dirt layer") create a particularly hospitable environment for soil-associated bacteria. The *schmutzdecke* is a commonly used term for the stable, biologically active, top layer of a slow sand filter bed and can be highly effective in straining particulate matter. In this study, however, the filter skin is exposed to harsh environmental conditions and is subjected to extreme fluctuations in temperature, water saturation, and oxygen levels; factors which may not affect indigenous coliform bacteria populations but may be detrimental to the survival of predatory microorganisms and algae which are known to be important in the removal of contaminants in slow sand filters.

Although bacteria will attach to various types of silt, clay, and sand particles (Schillinger and Gannon, 1985; Harvey *et al.*, 1989), this attachment has been shown to be reversible (Kinoshita *et al.*, 1993) or even inhibited by differences in growth media (Schillinger and Gannon, 1985). Unlike a slow sand filter which has a relatively constant head pressure and excellent bacterial removal capabilities (Ellis, 1984), the treatment system tested in this study is best described as a rapid sand filter and experiences sudden, large pulses in head pressure as raw stormwater is pumped onto the filter beds. The

greatest head pressure occurs during the first few seconds after the sand filter chamber has been filled which could flush out silts, clays, and soil-associated bacteria. Ellis (1984) suggested that high treatment rates can carry silt deep into a sand filter bed. This forceful movement of water through the filter may explain the dramatic spike in turbidity observed in the first outflow sample in Trial 3. Concurrently, this phenomenon might have caused bacteria growing in the filter media to detach from the sand resulting in the elevated coliform concentrations observed in the first few effluent grab samples (Fig. 11).

An unquantifiable but probably small proportion of bacteria removal may have occurred as a result of heavy metal toxicity. Toxicity bioassays performed during Trials 2 and 3 strongly suggest that metals, specifically Zn and Cu, are correlated with the inhibition of bacterial (*E. coli*) growth in both inflow and outflow samples. Applications of toxicity bioassays have been employed in several other studies, primarily to trace or identify sources of contaminated sediments or water (Bitton et al., 1994; de Vevey et al., 1993; Liu and Dutka, 1984; Van Hattum et al., 1993). Few studies, however, have involved direct sampling of storm events for toxicity. In an evaluation of toxicity screening tests for stormwater runoff in the City of Fort Worth, Waller et al. (1994) used several test organisms including *Ceriodaphnia dubia*, *Pimephales promelas*, and bacteria (Microtox test). Of the heavy metals, zinc was the only analyte measured and was found in highest concentrations from runoff originating from commercial land uses followed by industrial and residential. Zinc concentrations were elevated to the point of causing acute toxicity in both *C. dubia* and the Microtox test.

Differences in bacteria and fluorescent bead removal efficiencies between the three separate trials may have been due to differences between the two filter chambers. The same filter (chamber 1) was used in both Trials 1 and 2 and had a relatively slow permeability rate (90.0 m/day) compared to chamber 2 (approximately 120.0 m/day) which was challenged in Trial 3. In Trials 1 and 2, both total and fecal coliform bacteria concentrations in the outflow dropped by a factor of 1 to 2 log units and were below the Class III maximum limits in 85% and 90% of all outflow samples, respectively. Fluorescent bead concentrations dropped by approximately 4 log units. In Trial 3, neither coliform bacteria nor fluorescent bead concentrations were reduced appreciably and all ten outflow samples exceeded Class III maximum limits for both total and fecal coliforms. Also, differences in fecal coliform concentrations were not statistically significant between inflow and outflow values for Trial 3. The differences in outflow concentrations between Trials 1 and 2 versus Trial 3 may have been due to differences in permeability rates between chambers.

Specifically, chamber 2 may have had a thinner filter skin, larger grain sizes, larger microchannels, or a combination of the three which facilitated the transport of bacteria and fluorescent beads through the filter and which may have also caused spikes in turbidity and TSS values in the first outflow sample in Trial 3 (Fig. 13). Viessman and Hammer (1993) described the aging of a sand filter which results in the accumulation of deposits in the upper layer of the filter bed. This results in a reduction of pore area and an increase in the velocity of water passing through the remaining voids. Floc and other particles are carried deeper into the filter bed until breakthrough occurs which results in

the creation of large flow channels which are not capable of trapping small suspended material. Anecdotal reports by the operator of the sand filter suggest that this particular chamber was used more frequently than the northern chamber used in Trials 1 and 2. This continuous (over)use may have resulted in particle breakthrough and, consequently, poor removal efficiencies for all four microbial indicators.

Another explanation for the lack of bacterial removal may be due to the fact that the effective size of the sand media for the filter beds was 0.42 mm and created interstitial pore spaces which were probably too large to retain bacteria cells. Schillinger and Gannon (1985) found that the majority of bacteria taken from stormwater samples were associated with particulate matter that could be retained by pore sizes <52 μm. The estimated pore size for the filter medium was approximately 56 μm, much larger than most bacteria (typically <15 μm in length). The fluorescent bead (*Cryptosporidium* surrogate) was also much smaller than this pore size at 3 μm and the MS2 virion has a diameter of approximately 25 nm (Powelson *et al.*, 1993). Thus most of the attenuation of bacteria, viruses, and fluorescent beads probably did not occur as a result of straining but through either direct adsorption to the surfaces of the filter medium or indirectly as a result of adsorption to large (>56 μm) particulate matter that were then strained by the sandbed.

Obvious differences occurred between the three different trials for MS2 coliphage. In Trial 1, the filter medium was in an unsaturated condition and produced a load removal of approximately 99.999%. In Trials 2 and 3, the filter media had recently been saturated with water from antecedent storms and had much lower load removal efficiencies (87.6%

and 75.4%, respectively) than in Trial 1 (Fig. 12). Powelson and Gerba (1994) evaluated the effects of saturated and unsaturated soils on the removal of MS2 and PRD1 bacteriophages and found similar results, i.e., unsaturated soils had virus removal coefficients that were nearly three times greater than saturated soils. Saturated soils may have fewer available binding sites, poorer electrostatic attraction characteristics, and/or a greater capacity for colloidal transport than unsaturated soils, all of which may reduce the ability for viruses to attach to soil particle surfaces.

Soil pH has also been found to influence virus adsorption since both soil and virus particles are negatively charged under neutral (pH = 7) conditions. Goyal and Gerba (1979) found that soils having a saturated pH of less than 5.0 tended to adsorb viruses, including MS2, more readily than at higher pH conditions. The pH_{iep} or isoelectric point (pH at which the net charge of a particle is zero) is 3.9 for MS2 (Overby *et al.*, 1966). In this study, pH values ranged between 6 and 7.6 which presented less than optimal conditions for MS2 adsorption since virus particles would have still remained oppositely charged to the filter medium during the filtration process (Dowd *et al.*, 1998). As a result, a consistent, low level concentration of MS2 was observed in each outfall sample. Nicosia (1998) found similar results for PRD1 in which low levels of phage were detected over several weeks after the application of seeded wastewater to approximately 0.75 m of fine sand.

Removal of MS2 was similar to a report by Powelson *et al.* (1990) who observed a 95% reduction in MS2 concentrations after flow through an unsaturated, 105 cm long soil column. McConnell *et al.* (1984) used a slow rate sand filtration column and found

that reovirus could be removed by more than 4 log concentration units under certain water quality, flow rate, and sand bed construction conditions. Goyal and Gerba (1994) also found that human enteric viruses tend to adsorb more readily to soils than MS2 which supports the use of this coliphage as a conservative tracer in soil filtration experiments.

The *Cryptosporidium* surrogate (fluorescent beads) had an average log removal of 3.6 which was greater than previous studies which used inactivated oocysts as tracers. Riesenberg *et al.* (1995) reported a 2 log reduction in *Giardia* cysts using precast concrete slow sand filters along hillsides in California. Nieminski and Ongerth (1995) reported a 2.97 average log removal rate for *Cryptosporidium* oocysts using a 1 m deep sand/anthracite filter bed.

The greater removal efficiency for fluorescent beads in this study is surprising since earlier studies used slow sand filtration which tends to have greater contaminant removal properties than rapid sand filtration. Also, the beads used in this study were assumed to be relatively inert (uncharged) in aqueous solutions and were slightly smaller in size (3.0 ± 0.1 μm in diameter) than *Cryptosporidium parvum* (approximately 5.0 μm in diameter) to simulate the naturally occurring effects of oocyst deformation and folding when transported through a porous medium. When examined microscopically, the fluorescent beads were typically found in clumps and associated with larger aggregates of organic material which suggests that the beads may exhibit an electrostatic attraction to other particles in the water column. As a result, the high removal efficiency of this surrogate may have been enhanced by its attachment to large (>56 μm) suspended solids in the raw stormwater which were strained by the filter during treatment.

Ability to Meet State Water Quality Standards

Water quality standards for total coliform bacteria were exceeded more often during the saturated filter trials. During the unsaturated trials, removal of both total and fecal coliform bacteria was greater. Similar to the virus removal phenomenon, bacteria may experience greater removal when binding sites on the surface of the sand particles are unoccupied by water molecules. Regardless, the sand filter performed relatively poorly in bacteria removal.

Pretreatment of stormwater prior to filtration could help reduce bacterial loads which are not attenuated by the sand filter. Even though the discharge enters marine waters, disinfection or die-off may not be a major factor since a number of pathogenic microorganisms are resistant to elevated salinity (Fleisher *et al.*, 1986).

Wet Detention

Percentages of water discharged during the 5-day trial were similar to those reported by Cunningham (1993) for the same ponds and detention time. Approximately 84-86% of the inflow volume was discharged at the outflow leaving 14-16% of the total volume lost to evapotranspiration and probably a minor percentage lost to the surficial aquifer. Water loss was much greater during the 14-day trial with approximately 45%-46% of the inflow volume unaccounted for in the outflow despite a storm which contributed approximately 6.4 cm of rainfall 11 days (250 hours) after the initial storm

simulation (Fig. 6). Since the detention time within the ponds was much longer in the 14-day trial, a greater period of time was available for both evaporation to the atmosphere and transpiration via the abundant emergent vegetation present in each pond. This evapotranspiration phenomenon can be significant for wetland systems (Parkhurst *et al.*, 1998; Souch *et al.*, 1998).

The significant decline in temperature during the 5-day trial was a result of a cold front which moved through the region approximately 60 hours after the start of the experiment. Temperature did not appear to affect any of the four microbial indicators. The significant decline in fecal coliform bacteria occurred prior to the drop in temperature and the rates of decline for total coliforms, MS2, and beads did not appear to change appreciably after the cold front (Figs. 23 and 25). The significant decrease in conductivity was not a result of a removal phenomenon but was caused by the dilution of inflow water of greater conductivity mixing with the existing pool of water in the pond which had much lower conductivity. Once the stormwater entered the pond, conductivity values rose to an equilibrium level which appeared to be approximately the mean of the inflow value and the ambient value in the pond prior to pumping.

During the 14-day trial, changes in temperature were not significant since no major weather fronts occurred during the storm simulation. Although conductivity did not change significantly between inflow and outflow samples, a slight rise occurred which continued throughout the experiment. Concurrently, a significant decline in pH occurred possibly as a result of biological activity (breakdown of organic material) during the extended detention time within each pond. The decline in pH may have been enough

to cause the release of free ions into the water column which may have resulted in the rise in conductivity.

Lower turbidity and TSS values in outflow samples for the shallow pond may have been due to the absence of a thick muck and fine silt layer in the deeper zone of the shallow pond than the deep pond. Sediments collected using a ponar sampler in the shallow pond were typically sandy and submerged aquatic vegetation covered the entire bottom of the shallow pond. The deep pond had a thick muck layer and virtually no submerged vegetation coverage in the deep zone below the inflow pipe. The amount of vegetation that did occur was restricted to shallower depths of about 1 m or less which represented approximately 50% of the pond bottom area.

In experiments evaluating the importance of vegetation on sedimentation and retention in stream channels, Thornton *et al.* (1997) determined that vegetation length and cross sectional area were important variables for sediment entrapment. The amount of cross sectional area for vegetation in the shallow pond was nearly twice that of the deep pond which may have enhanced total suspended solid and microorganism entrapment. This was more evident during the shorter 5-day detention time trials than during the 14-day trials since gravitational settling and ultraviolet light inactivation were probably the more dominant inactivation mechanisms during the longer detention trials.

Although heavy metal samples were not collected during the wet pond trials, Cunningham (1993) did report low levels of metals in both ponds during similar treatments. However, Cu and Fe were the only two metals which were found consistently in outflow samples and concentrations of these metals were typically lower than those

found during toxicity testing at the sand filter site. Metal toxicity probably did not play a significant role in the removal/disinfection of bacteria from the water column in either of the ponds during the 5-day and 14-day trials.

Harper (1995) reported mean removal efficiencies of TSS from an extensive literature search of wet detention systems in Florida. Except for the 5-day deep pond trial, the removal efficiencies calculated in this study were within the range of other Florida wet detention ponds (55-94% removal). However, previous work by Cunningham (1993) at the SWFWMD ponds suggested that greater removal of suspended solids occurs in the deep pond than the shallow pond with the explanation that sediments appeared to be re-entrained in the shallow pond during pumping.

Cunningham (1993) also reported occasional negative suspended solid removal for both ponds which was also observed in this study. Depending on the frequency of storm events, the shallow pond may either become scoured by repetitive, high inflow velocities or laden with fine sediments if inflows are less frequent or have slower velocities. If scoured, less internal resuspension could occur since only larger, heavier soil materials would remain in the shallow pond resulting in greater removal efficiencies than the deep pond. Based on the qualitative comparison of the sediments between the two ponds, the shallow pond did appear to have a greater sand content (larger grain size material) and lower organic much layer than the deep pond.

One of the most important factors that may affect removal efficiency of microorganisms in wet detention systems is adsorption and sediment resuspension. Schillinger and Gannon (1985) suggest that fimbriation or the formation of external pili

on the surfaces of bacterial cells influences their adsorption to suspended particles. The extended survival of bacteria and viruses in sediments has been shown by a number of authors and suggests that contaminated sediments can pose serious public health risks if resuspended or transported to recreational waters (Smith *et al.*, 1978; Goyal and Gerba, 1979; Bulson *et al.*, 1984; Schillinger and Gannon, 1985; Goyal et al., 1984). Evidence of resuspension was indicated by greater turbidity values from samples taken in the deeper zones of each pond near the inlet structure during the simulated storm event (pumping).

For total coliforms, sediment resuspension may increase the ability of bacteria to be released into the water column and transported to the outlet structure. The significant, positive correlation between total coliforms and turbidity in outflow samples (Fig. 22) suggests a strong relationship between soil-associated bacteria and particulate matter in the water column which facilitates both removal (through sedimentation) and transport out of the treatment pond under high flow conditions. The spike in fecal coliform concentrations which occurred approximately 15 hours after the simulated storm event in the 5-day deep pond trial was probably a result of resuspension of bacteria attached to sediments (Fig. 23).

Total coliform concentrations in the deep pond were also elevated above the maximum expected outflow concentration (given complete mixing of the inflow with the existing pond volume) despite having a nearly 100,000 L greater normal pool volume to dilute the incoming bacterial load. The high concentrations and loads leaving the deep pond strongly suggest that an internal source of bacteria was extremely important in

contributing a significant load to the system that was later partially discharged at the outflow. Jones and Langan (1996) also reported negative removal efficiency values for both fecal coliforms and *E. coli* in wet detention ponds in New Hampshire and suggest that bacterial regrowth, especially during warm summer months, can result in the export of bacterial loads to downstream waters.

Spikes in fecal coliform concentrations during the 14-day trial may have also been due to wildlife activity in and around the pond. During nearly every sample collection period, several birds (anhinga and osprey) were seen perched near each of the ponds or on top of the outfall pipe at the western end of each of the pond. A small alligator was also observed in both ponds on various occasions during the 14-day but not the 5-day trial. The disturbance of sediments caused by the alligator's swimming activity may have also contributed to elevated total and fecal coliform concentrations at the outfall resulting in non-significant differences between inflow and outflow concentrations. Although, the extent of this phenomenon could not be quantified, elevated concentrations of total and fecal coliform bacteria found in sediment samples support this hypothesis.

Interestingly, log removal values for the wet detention ponds used in this study were similar to the range of values determined by Fernández *et al.* (1992) for raw wastewater stabilization ponds in Spain. Using similar sized ponds and detention times, but higher inflow concentrations, the authors observed log reductions ranging from 0.2 to 1.1 for both total and fecal coliforms . In this study, log reduction values for total and fecal coliforms ranged from -0.58 to 1.73. These reductions may be the result of a number of factors including protozoan bactivory (Sibille *et al.*, 1998) which is known to

occur in drinking water distribution systems, and ultraviolet light inactivation which has been shown to significantly diminish bacterial populations over time (Fujioka *et al.*, 1981; Davies-Colley *et al.*, 1994).

A similar evaluation of water quality improvements was recently performed by Dames & Moore (1998) for a 4.5 ha (11 ac) lake (Jungle Lake) in St. Petersburg, Florida which was reconfigured to improve pollutant removal efficiency. Prior to 1994, the lake functioned as a flood attenuation pond with a short detention time (<2 hours) and relatively poor removal of a number of stormwater contaminants. In fact, using data from the Dames & Moore report, greater loads of total coliforms and fecal coliforms were discharging at the outfall than what had entered the lake during certain storm events. Removal efficiency values were -323.1 and -129.0% for total coliforms and fecal coliforms, respectively. Once the pond was reconfigured to increase detention time (>30 hours), removal efficiencies for total coliforms improved and ranged from -142.0 to 99.6% with a mean of 42.6% while fecal coliforms ranged from -120 to 94% with a mean of 25.1%.

Removal efficiencies for total coliforms (–284 to 64.5%) and fecal coliforms (88.5 to 98.2%) in the 5-day trial were within the same order of magnitude as the range of values for the stormwater pond in St. Petersburg and illustrate the difficulty in assessing pollutant removal efficiencies in open water systems. Negative removal values suggest that pollutant loads are being generated within and then exported from a wet detention pond. This can occur for several reasons including inputs from the feces of indigenous wildlife (e.g, birds, small mammals) or from self-sustaining bacterial populations in the

pond sediments. Tate (1978) described the ability of *E. coli* to catabolize organic materials in soil. This, in turn, could support elevated concentrations of fecal coliform bacteria. A number of studies have reported extended survival of bacteria in sediments (Sherer *et al.*, 1992; Gerba and McLeod, 1976). Sherer (1992) found that both fecal coliform and fecal streptococci bacteria could survive in sediments for periods of several weeks to months versus only a few days in the water column.

Lijklema *et al.* (1987) reported decay rates of total and fecal coliform bacteria in a 1 m deep wet detention pond in the Netherlands as ranging from 0.029 to 0.042/hr for total coliform bacteria and from 0.029 to 0.075/hr for fecal coliform bacteria. These values were similar to the decay rates for total coliforms (0.026 to 0.028/hour), however, the range for fecal coliforms (0.039 to 0.223/hour) calculated for the 5-day and 14-day shallow wet detention pond trials in this study was much wider.

In their development of loss kinetics for fecal coliform bacteria, Auer and Niehaus (1993) found significant, positive correlations between fecal coliform death (decay) rate coefficients and light irradiance in a lake in New York. Death rates ranged between 0.021 and 0.190/hour for *in situ* lake measurements which brackets most of the decay rate values for fecal coliforms found during all four wet detention pond trials. They also found decreasing trends in irradiance values with increasing depth as a result of light attenuation in the water column. Bacteria may have experienced greater decay rates in the shallow pond because of the shallow depth and the ability for a greater percentage of ultraviolet light to penetrate to the bottom of the pond. This might also explain the apparent internal bacterial loading phenomenon and the typically greater concentrations

of total coliform bacteria at the outflow of the deep ponds. If light is not penetrating to the deep, silty bottom of the deep pond, fewer bacteria would experience die-off than in the shallow pond.

For MS2 and the *Cryptosporidium* surrogate, the interaction with resuspended sediments may have the reverse effect to that of total and fecal coliform bacteria. Sobsey *et al.* (1980, 1995) described the adsorption of several viruses, including MS2, in suspensions of soils in wastewater and found greater virus adsorption with clay and muck soil types. Clay soils composed much of the bottom and sideslopes of both ponds while the deep pond had a muck layer in the deep zone near the inlet pipe. The presence of these two soil types may have been responsible for the more than 90% removal of MS2 which typically occurred at both ponds. As sediment and organic particles are resuspended during pumping (or a storm event), the smaller MS2 and bead particles may be adsorbing to larger soil particles and then settling out before reaching the outfall. This same theory is believed to be responsible for the removal/straining of soil-adsorbed MS2 and beads during sand filtration. Plots of observed MS2 concentrations during the 5-day trial were generally less than the expected concentrations at the outfall (based on the assumption that complete mixing of the inflow with the existing pond volume occurs) and indicate that coliphage are being removed or inactivated at a greater rate than explained by dilution alone (Fig. 25).

The MS2 decay rates for the 5-day and 14-day shallow wet detention pond trials (0.012 and 0.033/hour, respectively) were similar to values reported by Gersberg *et al.* (1987) for artificial wetlands used to treat wastewater. In the artificial wetlands, decay

rates for MS2 ranged from 0.012 to 0.052/hour. Gersberg *et al.* (1987) found lower decay rates in stagnant ponds during winter while higher rates were found in flowing ponds and stagnant ponds in summer. The artificial wetlands were only about one-half the size of the ponds used in this study, however, the season (effects of temperature), percentage of aquatic vegetation, and total pond depth were nearly identical to the shallow wet detention pond trials. Overall log-removal values were less for the wet detention ponds (1.01 to 1.91 log-units) compared to a mean 3-log-unit removal of MS2 in the artificial wetlands studied by Gersberg *et al.*

Wet detention ponds attract a number of bird species including the great blue heron (*Ardea herodias*) which has also been shown to be a source of waterborne giardiasis (Georgi *et al.*, 1986). Levesque *et al.* (1993) found significant correlations between bird abundances and bacterial concentrations along bathing beaches in Canada and suggest that birds can contribute significant loads of microbial pathogens to freshwater lakes. Graczyk *et al.* (1998) recently determined that *Giardia* spp and infectious *Cryptosporidium parvum* oocysts can be disseminated by the Canada geese. Based on these earlier reports, the potential for contamination of wet detention ponds by endemic and migratory waterfowl could be significant and should be considered when siting and determining the level of public access to these ponds. Fattal *et al.* (1992) also determined that fish (tilapia) exposed to polluted water could be contaminated by both bacteria and viruses within 24 hours of inoculation. If wet detention ponds are heavily used by both birds and harvestable fish species, the risk of infection from ingesting

contaminated fish could be significant if wet detention ponds are used as a source of recreational fishing.

The inflow pipe to a stormwater pond should discharge to a shallow, hard-bottom or densely vegetated buffer area before flowing to a deeper storage/treatment area. By slowing and dissipating energy from the incoming flow resuspension of sediments and organic materials which can harbor microorganisms should be reduced considerably and will allow treatment by sunlight inactivation, predation, and die-off. Since there is a potential risk of contamination of wet detention ponds by various wildlife species, property managers should restrict the level of access (no bathing) and use (no fishing) of stormwater ponds to help reduce the risk of exposure and infection to nearby human populations. Avoidance of groundwater contamination should also be a concern since microbial pathogens may be able to migrate from stormwater treatment ponds if pond bottoms are excavated to a depth which would allow exchange with the surficial or intermediate aquifer.

Ability to Meet State Water Quality Standards

Water quality standards for total coliform bacteria were exceeded more often during the 14-day trials than the 5-day trials. Prior to the 14-day trials, heavy rains may have resulted in bacteria-contaminated runoff to the Tampa Bypass Canal which was the source water used in these experiments. As a result, greater inflow concentrations and loading of bacteria occurred during the 14-day trials which, when coupled with internal

loading from resuspended sediments could not be removed sufficiently to meet state water quality standards.

Alum Coagulation

Alum appeared to be highly effective in removing turbidity, MS2, beads, and fecal coliform bacteria at both high and low doses as compared to simple gravitational settling. At the 10 mg/L dose, alum was more effective at removing turbidity, TSS, beads, and total coliforms than at the 600 mg/L dose. This may have been due to a thicker than expected floc layer in the high dose trial that may have contaminated samples with floc-bound bacteria and viruses drawn from the upper portions of the water column. Another cause may have been related to the extreme change in pH that accompanied the addition of the high dose of alum which, in turn, may have resulted in the dissociation and resuspension of flocculent organic material back into the water column.

Low doses of alum did not appear to be acutely toxic to bacteria since viable colonies were present in the water column and floc layer after 48 hours. The presence of bacteria even after alum treatment may be related to cell surface properties. Zita and Hermansson (1997) showed that increased cell surface hydrophobicity in *E. coli* strains from wastewater was positively correlated ($p \leq 0.05$) with adhesion to sludge floc. As a result, the bacterial isolates that were detected in the water column after alum treatment in this study may have had much lower hydrophobicity values than the isolates in the floc material. George *et al.* (1989) found no observable toxic effects of alum sludge on

bacteria (*Photobacterium phosphoreum*, using Microtox assays), protozoa (*Tetrahymena pyriformis*), fathead minnows (*Pimephales*), nor *Ceriodaphnia*. However, high doses of alum and other metals are toxic to bacteria and viruses as indicated by non-detectable concentrations of total and fecal coliform bacteria and MS2 in the floc layer of the 600 mg/L jar tests (Figs. 40 and 41).

The apparent toxic effects of alum treatment may not have been directly attributed to the alum. Commercial grade alum used in stormwater and wastewater treatment often contains a number of other metal ions that are not removed during the alum production process. Concentrations of zinc, chromium, copper, and nickel were all elevated in the water column and floc material after the addition of alum during the 600 mg/L jar test (Fig. 34). Again, this was probably due to the extreme drop in pH which can cause metals to dissociate from organic floc. These metals are all potentially toxic to certain species of bacteria, including *E. coli*, and probably caused the complete inactivation of total and fecal coliforms and MS2 in during the high dose trial.

A study by Carr (*in prep.*) estimated TSS removal to range from -370% to 95% with a mean of 21% for the full scale alum treatment system in Pinellas Park. The removal efficiencies calculated for the same dose in this study (74.1 to 84.4%) were within this range but would probably only occur during optimal, quiescent conditions in the field. The wide range of removal efficiencies calculated by Carr represented several storm events of varying intensity. Larger storms may have bypassed the treatment system or caused the resuspension of TSS in the settling pond which would have resulted in negative removal efficiency values.

Removal rates for bacteria in this study were similar to results from an alum treatment of a 288 ha lake in Washington state. Bulson *et al.* (1984) reported a 90% removal of fecal coliforms from the water column approximately 72 h after alum addition at a dose of approximately 10 mg/L. Data from this study and laboratory studies by Bulson *et al.* (1984) suggest bacteria can be removed rapidly and with high efficiencies in jar tests with removals ranging up to 99.9% after 48 hours. Due to the controlled nature of jar tests, a greater removal efficiency would be expected since little or no resuspension of floc material occurs over time and little chance of contamination by waterfowl or other fecal sources is possible.

Concentrations of total and fecal coliform bacteria in the floc layer were within one order of magnitude of the initial concentrations in the untreated stormwater and MS2 was found at lower concentrations in the floc than in the untreated stormwater. Much greater concentrations would be expected in the floc if these microbial populations acted as a conservative mass like TSS. Since bacteria and viruses are subject to die-off and inactivation, respectively, losses from metal toxicity, low pH, ultraviolet light, predation, or other environmental factors probably resulted in the complete absence of total coliforms, fecal coliforms, and MS2 in the floc layer of the high dose containers and lower than expected concentrations in the low dose containers. Bulson *et al.* (1984) found a similar die-off response (at a rate of approximately 200 cfu/100 ml per day) of fecal coliform bacteria in the floc layer of an alum treated lake.

Removal or die-off of MS2 was also generally high in both alum treated and control samples. Thompson *et al.* (1998) suggest that the air-water interface (AWI) can have a significant effect on the inactivation of MS2 by causing the denaturing or reconfiguration of capsid proteins. This increased inactivation response can be enhanced by the storage of viruses in polypropylene containers which was the same material used for jar tests and sample collection containers in this study. The greater than 90% reduction in MS2 for both control and alum treated stormwater suggests that other factors such as the interaction of virus particles with the AWI and sunlight inactivation may be significant when combined with adhesion to particulate floc matter and gravitational settling. The use of other viral surrogates may be warranted since MS2 has been shown to be particularly susceptible to low pH conditions.

Despite the observed die-off responses in both total and fecal coliform bacteria and MS2, considerable microbial loads can accumulate and remain viable in the floc layer. This contaminated floc has the potential to be resuspended by wave energy or storm generated flows in a flocculation basin or alum treated waterbody. If pathogens were removed from the water column and deposited in the benthos, the risk of infection would be much greater if a person were exposed to this highly concentrated floc. Careful management of the timing and dose of alum injection and the removal of alum floc is necessary to minimize public health risks associated with exposure to this potentially contaminated waste material.

Ability to Meet Water Quality Standards

In general, concentrations of nearly every parameter were below Class III exceedence standards during both the low dose and high dose trials. Alum treatment provided greater than 90% reduction of microbial indicators under controlled conditions at a dose of 10 mg/L, however, this estimate is probably over-conservative given the variability of flows and other factors such as wind driven resuspension that could occur in alum floc settling ponds. Alum floc would probably not meet Class III standards if discharged immediately to a surface waterbody due to elevated levels of TSS, turbidity, and viable bacteria.

Comparisons between Sand Filtration, Wet Detention, and Alum Treatment

Each of the three stormwater treatment systems evaluated in this study were capable of reducing microbial pollution and each had specific attributes that would make them more advantageous than the other for specific applications or site constraints. Sediment resuspension, adhesion, and natural die-off were common factors important for the removal of microorganisms, however, there were obvious differences in the removal efficiencies of specific indicators between the three BMPs (Fig. 49).

Alum coagulation (at a dose of 10 mg/L) exhibited some of the greatest removal of turbidity, total coliforms, fecal coliforms, and MS2 and would require the least amount of space to treat large watersheds. Sand filtration also requires a small area for construction and removal efficiencies were generally high for turbidity, MS2, and beads but not for

total coliforms or fecal coliforms. Wet detention ponds are probably the easiest to construct and maintain but can also be the most expensive treatment technology, especially where land costs are high, since they require the greatest surface area to treatment area ratio. However, the current regulatory standard for shallow wet detention ponds with a 5-day bleed-down period had the greatest TSS removal of the BMPs and greater than 90% fecal coliform bacteria, MS2 and bead removal. Removal efficiencies for beads were the same between sand filtration, 5-day shallow pond wet detention, and 14-day deep pond wet detention. Overall, bead removal was high among all treatment systems.

Comparisons of fecal:total coliform ratios between inflow and outflow samples also suggests that fecal coliforms are either removed or may experience greater die-off rates than the total coliform group as a whole. The mean fecal:total coliform ratio for inflow samples was significantly greater ($p \leq 0.05$) than outflow samples. This would suggest that fecal coliform bacteria may be more susceptible to die-off than non-enteric coliform bacteria since their normal growth environment has a relatively narrow level of tolerance. Exposure to adverse environmental conditions such as extreme fluctuations in temperature, nutrient availability, pH, and ultraviolet light may be causing differential die-off rates for the coliform group selecting for bacteria typically associated with soils.

The consistent presence of pathogenic strains of bacteria in both inflow and outflow samples from all of three sites evaluated in this study further stresses the importance of stormwater treatment to reduce potential public health risks. *Citrobacter freundii*, *Klebsiella pneumoniae*, *E. coli*, and *Enterobacter agglomerans* were all found in samples

from both the wet detention and alum coagulation trials and have been isolated in infected wounds caused by fecally-contaminated seawater (Kueh, *et al.*, 1992). *Salmonella enteritidis* can be isolated from egg shells and poultry carcasses and can cause abdominal cramps, vomiting and diarrhea 12 to 72 hours after exposure to contaminated food or water. The source of this bacteria which was found in the alum coagulation and sand filtration trials may have originated from leachate from garbage cans near homes or large trash bins used by local restaurants. Alum coagulation appeared to have the greatest effect on reducing the number of species present in the water column (from eight to three), however, several pathogenic strains (*E. coli, K. pneumoniae, S. enteritidis*) were still viable in the resultant alum floc.

The three stormwater treatment systems evaluated in this study are all commonly used technologies in the water and wastewater treatment industries (Viessman and Hammer, 1993). Stabilization ponds are often used during tertiary treatment of wastewater and have retention times between 10 to 15 days and are generally shallow (less than 2 m deep) to allow maximum sunlight penetration and mixing. Alum is one of the most widely used coagulants for water and wastewater treatment and is generally applied in doses of 5 to 50 mg/L in water treatment facilities. Sand filtration is a common method for removing colloidal impurities in water processing and tertiary wastewater treatment. Alum coagulation is often used as a pretreatment step prior to sand filtration to reduce turbidity and TSS.

Other methods which are commonly used in the drinking water and wastewater treatment industries have been suggested for the removal of microbial pathogens from

stormwater including high-rate chlorine disinfection, ozonation, and uv light irradiation (O'Shea and Field, 1992). The primary drawback in implementing large-scale stormwater treatment facilities has generally been associated with high infrastructure costs and difficulties in treating unpredictable volumes and flows of stormwater runoff . Unlike wastewater collection systems, storm flows are often unpredictable in duration and intensity. This characteristic makes the ability to plan and size the appropriate level of infrastructure (pipes, holding tanks, chemical storage, etc.) extremely difficult. Even if a facility were constructed, the performance of the plant would probably be inefficient for extreme storm events such as hurricanes or floods which can cause widespread microbial contamination of surface waters.

Greene (1992) evaluated the use of ozone disinfection for the treatment of urban runoff in Santa Monica, California. A pilot plant was constructed to disinfect dry-weather flows (flows originating from groundwater seepage into storm drains) within a heavily urbanized watershed. Despite increases in total organic carbon and TSS concentrations which often reduced the efficiency of ozonation, total and fecal coliforms, enterococci, and the vaccine strain of poliovirus were reduced over a range of 3 to 5 log units using ozone at doses of 10-20 mg/L. The removal of microbial indicators was successful enough that the effluent generated by the ozone treatment system was of sufficient quality to be used for landscape irrigation.

The use of high-rate chlorine disinfection for the reduction of coliform bacteria in stormwater overflows was evaluated by Haas *et al.* (1990) at a pilot plant in California. The problematic phenomenon of high intensity, short duration flows during a storm event

was incorporated into the design of the disinfection process. Since high flows result in turbulent mixing, the chlorine disinfectant used in the process was well-mixed with the waste stream, thereby reducing contact times and chlorine residuals. The disinfection process was deemed feasible since the observed total coliform concentrations in the effluent met the design criteria for the plant. This type of treatment is only cost-effective if large volumes of stormwater are treated and the runoff is contaminated to the point of being a consistent health risk to downstream users.

Recommendations

In general, mass balance calculations provided more accurate and conservative measures of removal efficiencies than calculations based on changes in concentration. This was more evident in comparisons of total and fecal coliform bacteria than with MS2 or bead concentrations. For example, the calculated removal efficiencies based on load values for total and fecal coliforms in the 14-day shallow pond trial were 4.2% and 76.4%, respectively. Removal efficiencies calculated using mean inflow and outflow concentrations for total and fecal coliforms would have produced values of 89.1% and 97.9%, respectively.

The reason for this phenomenon can be attributed to two factors: highly skewed coliform concentrations and differences in the volume of stormwater discharged between sample collection times. Calculations based on loads take into account the actual bacterial load for a given volume of water discharged during a discrete sample collection

interval. Calculations based on concentrations do not account for varying outflow rates and discharge volumes, and this can significantly influence removal efficiency calculations. Whenever possible, future evaluations of treatment systems for microbial removal should be assessed using mass balance calculations.

Since it is apparent that the resuspension of sediments can reduce the effectiveness of wet detention ponds, reducing flow rates at the inflow to the sump of a wet detention pond can be critical to achieving target load reductions and maintaining sanitary water quality at the discharge. The use of a splash pad or rip-rap at the outfall to dissipate the flows at the inflow can reduce erosion and sediment resuspension which should increase removal efficiencies for bacteria. For alum coagulation systems, the control and disposal of alum floc will also be an important issue since this and other research has shown that both bacteria and viruses can remain viable in the resulting floc layer. Separate floc collection sumps and pump out facilities can help reduce the potential for resuspension and transport of floc to receiving waters. Factors which affect the effectiveness of sand filters include the use of proper filter media (optimal grain size and chamber volumes), monitoring (checking hydraulic permeability rates, pollutant removal efficiencies), and maintenance (replacing clogged filter material). The significant correlation between several of the microbial indicators and turbidity may be useful in estimating the potential effectiveness of stormwater treatment systems during both the post-construction phase (monitoring and maintenance) and in watershed planning (selection of optimal bmp's for water quality remediation).

Each of the three treatment systems has unique advantages which are dependent upon the various watershed characteristics of the basin being treated. To take advantage of all of the attributes of each of these treatment systems, a treatment train or combination of treatment methods could be employed to reduce microbial loading. However, the effectiveness of a series of BMPs is not additive (Horner *et al.*, 1994), since the first treatment system will sequester the fraction of contaminant easiest to remove which makes subsequent reductions more difficult. Horner (1992) suggests that a penalty or performance reduction coefficient be incorporated into calculations which estimate the performance of a series of treatment systems. Unfortunately, little research has been performed to establish this penalty coefficient based on actual field data.

A treatment train or combination of treatment methodologies which could provide an optimal microbial removal efficiency might be configured using an alum injection system coupled with a wet detention pond and floc settling sump, and a sand filter (Fig. 50). Based on the removal efficiency results from this study and using conservative inflow concentration estimates, one could expect a 95.6% reduction in turbidity, 99.9% reduction in TSS, 99.9% (3.0 log) removal of total coliforms, 99.998% (4.7 log) removal of fecal coliforms, 99.9999995% (8.3 log) removal of MS2 and 99.99992% (6.1 log) removal of *Cryptosporidium* oocysts (assuming the fluorescent beads used in this study acted as a conservative tracer). These estimates reflect a fairly conservative penalty factor of 1.5 as discussed previously. However, in actual field conditions, these reductions will be highly variable depending on scale (the ratio of watershed area or volume of stormwater treated versus alum, pond, and sand filter capacities), quality of construction

and maintenance for each of the three treatment components, and the potential for stochastic events such as hurricanes or floods that could overload or bypass the entire treatment train.

Despite the findings of this study, it is obvious that more research is necessary to determine the extent of microbial contamination of stormwater in specific watersheds or basins of concern. The sources and fate of pathogens are highly variable due to differences in a number of factors including local geological formations (soil types), average seasonal temperatures, land uses (agricultural vs. commercial), building practices (septic tanks vs. central sewers), streamflow patterns, and natural attenuation. As a result, a number of waterbodies both in Florida and the U.S. are considered unfit for recreational or shellfishing without any direct knowledge as to the cause of microbial contamination. The development of new methods or a combination of existing monitoring tools are needed to diagnose sources of pathogenic strains. Remedial measures to protect public health and increase the function and accessibility of valuable water resources can then be implemented prudently and efficiently.

In addition, more data will be needed to determine the efficiencies of stormwater BMPs under a variety of field conditions and scales. For example, small wet detention ponds may function differently than larger ponds with greater surface areas, longer detention times, and more diverse aquatic vegetation. The use of a smaller effective grain size in a sand filter may be more effective at retaining bacteria, viruses, and protozoa but may also reduce the volume of water that can be treated for larger rainfall events (which would result in an emergency bypass of the system and the discharge of untreated

stormwater). Alum treatment may be more cost-efficient for treating larger watersheds than small basins due to the high initial startup costs for equipment. Also, additional data will be necessary to determine whether a treatment train approach would, in fact, be more effective than a single BMP that was redesigned specifically for optimizing the removal of microorganisms (i.e., determining the penalty factor between BMPs in series).

CONCLUSIONS

The results of this study are intended to assist in the decision-making process for the use of stormwater BMPs to reduce microbial pathogen loads to significant waterbodies including rivers, lakes, and estuaries. The preservation and protection of watersheds and the improvement of water quality in shellfish harvesting areas to protect human health are identified as major initiatives under the current federal Clean Water Act (U.S. EPA, 1998). Improving the quality of stormwater runoff that discharges to fragile aquatic ecosystems is also an important issue in Florida for a number of competing industries (drinking water suppliers, commercial shellfishing, tourism) which rely on clean water from both surface and groundwater sources. Implementing remedial actions to treat stormwater contaminated by microbial pathogens will be an important and necessary step in managing these limited water resources.

ACKNOWLEDGMENTS

I thank Dr. Betty Rushton (SWFWMD) and Dr. Joan Rose (USF) for inspiring this research and Mr. Mike Holtkamp, P.E. (SWFWMD), for his guidance and assistance with a number of calculations used in developing removal efficiency values. Chris Tomlinson, Rob Fenwick, Gloria Ake, Rebecca Hastings, and Forest Turbiville provided invaluable assistance in the field and Bonnie Gering (SWFWMD), Kelley Reynolds (USF), Lara Nicosia (USF), and Chuck Gibson (USF) provided additional help in the lab.

Special thanks to Dr. Daniel Lim, Dr. Stuart Brooks, Dr. My Lien Dao, Dr. Joan Rose, and Dr. Ted VanVleet for their critical review of this report. This project was funded through the Surface Water Improvement and Management (SWIM) Program and the Pinellas-Anclote, Alafia River, Hillsborough, Northwest Hillsborough, Manasota, and Peace River Basin Boards of the Southwest Florida Water Management District.

LITERATURE CITED

American Public Health Association. 1995. Standard methods for the examination of water and wastewater, 19th ed. American Public Health Association, Washington, D. C.

Auer, M. T. and S. L. Niehaus. 1993. Modeling fecal coliform bacteria - I. Field and laboratory determination of loss kinetics. Water Research 27(4): 693-701.

Bastian, R.K. 1986. Potential impacts on receiving waters. In: B. Urbonas and L. A. Roesner, eds. Urban Runoff Quality - Impact and Quality Enhancement Technology. Proceedings of an Engineering Foundation Conference. American Society of Civil Engineers, New York.

Bemiss, J. A., M. M. Logan, J. D. Sample, and G. P. Richards. 1989. A method for the enumeration of poliovirus in selected molluscan shellfish. Journal of Virological Methods 26: 209-218

Bishop, R. 1992. Integrated water resource plan. Florida Water Resources Journal (March): 34-39.

Bitton, G., B. Koopman, O. Agami 1992. MetPAD: a bioassay for rapid assessment of heavy metal toxicity in wastewater. Water Environment Research 64(6): 834-836.

Bitton, G., K. Jung, and B. Koopman. 1994. Evaluation of a microplate assay specific for heavy metal toxicity. Archives of Environmental Contamination and Toxicology 27: 25-28.

Bulson, P. C., D. L. Johnstone, H. L. Gibbons, and W. H. Funk. 1984. Removal and inactivation of bacteria during alum treatment of a lake. Applied and Environmental Microbiology 48(2): 425-430.

Canale, R. P., M. T. Auer, E. M. Owens, T. M. Heidtke, and S. W. Effler. 1993. Modeling fecal coliform bacteria - II. Model development and application. Water Research 27(4): 703-714.

Carr, D. 1993. A native herbaceous marsh used for stormwater treatment. Proceedings of the 3rd Biennial Stormwater Research Conference, Tampa, Florida. Ed. by C. Dye and B. Rushton. Southwest Florida Water Management District, Brooksville, Florida.

Cunningham, J. D. 1993. Comparative water quality data of a deep and a shallow wet detention pond. Master's Thesis. University of Central Florida.

Dames & Moore. 1998. Jungle Lake stormwater Rehabilitation and Lake Enhancement: Water Quality Monitoring Program Report. Technical report prepared for the Southwest Florida Water Management District.

Davies-Colley, R.J., R. G. Bell, A. M. Donnison. 1994. Sunlight inactivation of enterococci and fecal coliforms in sewage effluent diluted in seawater. Applied and Environmental Microbiology 60(6): 2049-2058.

de Vevey, E., G. Bitton, D. Rossel, L. D. Ramos, L. M. Guererro, and J. Tarradellas. 1993. Concentration and bioavailability of heavy metals in sediments in Lake Yojoa (Honduras). Bulletin of Environmental Contamination and Toxicology: 253-259.

Dowd, S. E., S. D. Pillai, S. Wang, and M. Y. Corapcioglu. 1998. Delineating the specific influence of virus isoelectric point and size on virus adsorption and transport through sandy soils. Applied and Environmental Microbiology 64(2): 405-410.

DuPont, H. L., C. L. Chappell, C. R. Sterling, P. C. Okhuysen, J. B. Rose, W. Jakubowski. 1995. The infectivity of *Cryptosporidium parvum* in healthy volunteers. New England Journal of Medicine 332:855–889.

Eastaugh, J. and S. Shepherd. 1989. Infectious and toxic syndromes from fish and shellfish consumption. Archives of Internal Medicine 149: 1735-1740.

Edwards, D. R., M. S. Coyne, P. F. Vendrell, T. C. Daniel, P. A. Moore, Jr., and J. F. Murdoch. 1997. Fecal coliform and streptococcus concentrations in runoff from grazed pastures in northwest Arkansas. Journal of the American Water Resources Association 33(2): 413-422.

Ellis, J. B. 1988. The development of environmental criteria for urban detention pond design. In: Design of Urban Runoff Quality Controls. Proceedings of an Engineering Foundation Conference on Current Practice and Design Criteria for Urban Water Quality Control. Edited by: L. A. Roesner, B. Urbonas, and M. B. Sonnen. American Society of Civil Engineers, New York, New York.

Ellis, K. V. 1984. Slow sand filtration. CRC Critical Reviews in Environmental Control 15(4): 315-354.

Enriquez, R., Frosner, V. Hochstein-Mintzel, S. Riedemann, and G. Reinhardt. 1992. Accumulation and persistence of hepatitis A virus in mussels. Journal of Medical Virology 37: 174-179.

Environmental Research and Design, Inc. 1995. Pinellas Park Alum Injection Stormwater Treatment Facility. Southwest Florida Water Management District MSSW Permit.

Fayer, R., C. A. Farley, E. J. Lewis, J. M. Trout, and T. K. Graczyk. 1997. Potential role of the eastern oyster, *Crassostrea virginica*, in the epidemiology of *Cryptosporidium parvum*. Applied and Environmental Microbiology 63(5): 2086-2088.

Fernández, A., C. Tejedor, and A. Chordi. 1992. Effect of different factors on the die-off of fecal bacteria in a stabilization pond purification plant. Water Research 26(8): 1093-1098.

Fleisher, J. M., D. Kay, R. L. Salmon, F. Jones, and M.D. Wyer. 1986. Marine waters contaminated with domestic sewage: nonenteric illnesses associated with bather exposure in the United Kingdom. American Journal of Public Health 86(9):1228-1234.

Florida Administrative Code, Chapter 40D-4, Basis of Review, Southwest Florida Water Management District, 1988.

Florida Department of Environmental Protection. 1996. Section 305B Report: Water Quality Assessment for the State of Florida.

Fujioka, R. S., H. H. Hashimoto, E. B. Siwak, and R. H. F. Young. 1981. Effect of sunlight on survival of indicator bacteria in seawater. Applied and Environmental Microbiology 41: 690-696.

George, D. B., S. G. Berk, and V. D. Adams. 1989. Toxicity of alum sludge discharge on the aquatic environment. Proceedings of the Residual Management Conference. American Water Works Association/Water Pollution Control Federation.

Georgi, M. E., M. S. Carlisle, and L. E. Smiley. 1986. Giardiasis in a great blue heron (*Ardea herodias*) in New York state: another potential source of waterborne giardiasis. American Journal of Epidemiology 123: 916-917.

Gersberg, R. M., R. Brenner, S. R. Lyon, and B. V. Elkins 1987 Survival of bacteria and viruses in municipal wastewaters applied to artificial wetlands. In: Aquatic Plants for Water Treatment and Resource Recovery. eds. K. R. Reddy and W. H. Smith. Magnolia Publishing, Inc. Orlando, Florida pp. 237-245

Gold, M., M. Bartlett, J. Dorsey, and C. McGee. 1990. An assessment of inputs of fecal indicator organisms and human enteric viruses from two Santa Monica storm drains: Technical Report. Santa Monica Bay Restoration Project.

Gomez, M. S., M. Gracenea, P. Gaslbez, C. Feliu, C. Ensenat, and R. Hidalgo. 1992. Detection of oocysts of *Cryptosporidium* in several species of monkeys and one prosimian species at the Barcelona Zoo. Parasitology Research 78 (7): 619-620.

Goyal, S. M., and C. P. Gerba. 1979. Comparative adsorption of human enteroviruses, simian rotavirus, and selected bacteriophages to soils. Applied and Environmental Microbiology 38(2): 241-247.

Goyal, S. M., W. N. Adams, M. L. O'Malley, and D. W. Lear. 1984. Human pathogenic viruses in sewage sludge disposal sites in the middle Atlantic region. Applied and Environmental Microbiology 48(4): 758-763.

Graczyk, T. K., R. Fayer, J. M. Trout, E. J. Lewis, C. A. Farley, I. Sulaiman, and A. A. Lal. 1998. *Giardia* sp. cysts and infectious *Cryptosporidium parvum* oocysts in the feces of migratory Canada geese (*Branta canadensis*). Applied and Environmental Microbiology 64(7): 2736-2738.

Greene, G. E. 1992. Ozone disinfection and treatment of urban storm drain dry-weather flows: A pilot treatment plant demonstration project on the Kenter Canyon storm drain system in Santa Monica. Santa Monica Bay Restoration Project Technical Report. Monterey Park, California.

Haas, C. N., K. Longley, and T. Selfridge. 1990. High-rate coliform disinfection of stormwater overflow. J. Water Pollut. Control Fed. 62(3): 282-287.

Haile, R. W., J. Alamillo, K. Barrett, R. Cressey, J. Dermond, C. Ervin, A. Glasser, N. Harawa, P. Harmon, J. Harper, C. McGee, R. C. Millikan, M. Nides, and J. S. Witte. 1996. An epidemiological study of possible adverse health effects of swimming in Santa Monica Bay. Final Report. Santa Monica Bay Restoration Project. 211 pp.

Harper, H. H. 1990. Long-term evaluation of the use of alum for treatment of stormwater runoff. Proceedings of the First Annual Southeastern Lakes Management Conference. Ed. by C. E. Watkins, H. McGinnis, and K. J. Hatcher. North American Lake Management Society, Alachua, Florida.

Harper, H. H. 1995. Pollutant removal efficiencies for typical stormwater management systems in Florida. Proceedings of the 4th Biennial Stormwater Research Conference, Tampa, Florida. Ed. by C. Dye and B. Rushton. Southwest Florida Water Management District.

Harvey, R. W. 1989. Transport of microspheres through a sandy aquifer. Environmental Science and Technology 23: 51-56.

Harvey, R. W. 1997. Microorganisms as tracers in groundwater injection and recovery experiments: a review. Federation of European Microbiological Societies Microbiology Reviews 20: 461-472.

Hayes, E. B., T. D. Matte, T. R. O'Brien, T. W. McKinley, G. S. Losdon, J. B. Rose, B. L. P. Ungar, D. M. Word, P. F. Pinsky, M. L. Cummings, M. A. Wilson, E. G. Long, E. S. Hurwitz, and D. G. Juranek. 1989. Large community outbreak of Cryptosporidiosis due to contamination of a filtered public water supply. New England Journal of Medicine 320:1372-1376.

Herwaldt, B. L., G. F. Craun, S. L. Stokes, and D. D. Juranek. 1991. Waterborne-disease outbreaks, 1989-1990. Morbidity and Mortality Weekly Reports Surveillance Series 40 (SS-3): 1-21.

Hesselman, D. H., J. H. Seagle, and R. L. Thompson. 1992. Comprehensive shellfish harvesting area survey of Sarasota Bay, Manatee and Sarasota Counties, Florida. Florida Department of Natural Resources Technical Report.

Heufelder, G. R. 1988. Bacteriological monitoring in Buttermilk Bay. Technical Report: EPA 503/4-88-001, Office of Marine and Estuarine Protection, Washington, DC.

Hillsborough County Environmental Protection Commission. 1991. Surface water quality, Hillsborough County, 1990-1991. Technical Report. Hillsborough County Environmental Protection Commission.

Horner, R. R. 1992. Water Quality Analysis for Covington Master Drainage Plan. Rep. R. W. Beck and Associates and King County Surface Water Management Division, Seattle, Washington.

Horner, R. R., J. J. Skupien, E. H. Livingston, and H. E. Shaver. 1994. Fundamentals of Urban Runoff Management: Technical and Institutional Issues. Terrene Institute, Washington, DC. 302 pp.

Hunter, C. and A. McDonald. 1991. Seasonal Changes in the Sanitary Bacterial Quality of Water Draining a Small Upland Catchment in the Yorkshire Dales. Water Resources, 25(4) : 447-453.

Hvitved-Jacobsen, T. 1986. Conventional pollutant impacts on receiving water, Proceedings of the NATO Workshop on Urban Runoff Pollution, Springer-Verlag, Heidelberg, pp. 345-378.

Jones, S. and R. Langan. 1996. Assessment of the effectiveness of permanent stormwater control measures. Technical Report for the New Hampshire Coastal Program, Office of State Planning.

Joseph, C., G. Hamilton, M. O'Connor, S. Nicholas, R. Marshall, R. Stanwell-Smith, R. Sims, E. Ndawula, D. Casemore, P. Gallagher, and P. Harnett. 1991. Cryptosporidiosis in the Isle of Thanet; an outbreak associated with local drinking water. Epidemiology and Infection 107: 509-519.

Jung, K., G. Bitton, and B. Koopman. 1996. Selective assay for heavy metal toxicity using a fluorogenic substrate. Environmental Toxicology and Chemistry 15(5): 711-714.

Kaminjolo, J. S., A. A. Adesiyun, R. Lorengard, and W. Kitsonpiggott. 1993. Prevalence of *Cryptosporidium* oocysts in livestock in Trinidand-and-Tobego. Veterinary Parasitology 45(3-4): 209-213.

Kebabjian, R. 1994. Monitoring the effects of urban runoff on recreational waters. Journal of Environmental Health 56(9): 15-18.

Kehoe, M., C. W. Dye and B. T. Rushton.1993. A survey of the water-quality of wetlands-treatment stormwater ponds. Technical Report, Southwest Florida Water Management District, Brooksville, Florida.

Kehoe, M.1994. Water-quality survey of twenty-four stormwater wet-detention ponds. Technical Report, Southwest Florida Water Management District, Brooksville, Florida.

Kinoshita, T., R. C. Bales, M. T. Yahya, and C. P. Gerba. 1993. Bacteria transport in a porous medium: retention of *Bacillus* and *Pseudomonas* on silica surfaces. Water Research 27(8): 1295-1301.

Koenig, K. L., J. Meuller, and T. Rose. 1991. *Vibrio vulnificus* hazard on the half shell. Western Journal of Medicine. 155(4): 400-403.

Kong, I., G. Bitton, B. Koopman, and K. Jung. 1995. Heavy metal toxicity testing in environmental samples. Reviews of Environmental Contamination and Toxicology 142: 119-147.

Kueh, C. S. W., P. Kutarski, and M. Brunton.1992. Contaminated marine wounds - the risk of acquiring acute bacterial infection from marine recreational beaches. Journal of Applied Bacteriology 73: 412-420.

Levesque, B., P. Brousseau, P. Simard, E. Dewailly, M. Meisels, D. Ramsay, and J. Joly. 1993. Impact of the ring-billed gull (*Larus delawarensis*) in the microbiological quality of recreational water. Applied and Environmental Microbiology 59(4): 1228-1230.

Li, S. Y., J. A. Goodrich, J. H. Owens, G. E. Willeke, F. W. Schaefer, and R. M. Clark. 1997. Reliability of surrogates for determining *Cryptosporidium* removal. Journal of the American Water Works Association 89(5): 90-99.

Lijklema, L., B. Habekotte, C. Hooijmans, R. H. Aalderink, and A. H. Havelaar. 1987. Survival of indicator organisms in a detention pond receiving combined sewer overflow. Water Science Technology 19(Rio): 547-555.

Liu, D. and B. J. Dutka. 1984. Toxicity screening procedures using bacterial systems, Vol. 1. Marcel Dekker, Inc., New York, 476 pp.

Lucena, F., J. Lasobras, D. McIntosh, M. Forcadell, and J. Jofre. 1994. Effect of distance from the polluting focus on relative concentrations of *Bacteroides fragilis* phages and coliphages in mussels. Applied and Environmental Microbiology 60(7): 2272-2277.

McConnell, L. K., R. C. Sims, and B. B. Barnett. 1984. Reovirus removal and inactivation by slow-rate sand filtration. Applied and Environmental Microbiology 48(4): 818-825.

McCorquodale, D. S., and C. M. Burney. 1993. Biscayne Bay sewage pollution indicators. Final Report to the Southwest Florida Water Management District. West Palm Beach, Florida.

McFeters, G. A., J. P. Barry, and J. P. Howington. 1993. Distribution of enteric bacteria in Antarctic seawater surrounding a sewage outfall. Water Research 27(4): 645-650.

Moore, J. A. 1991. Surface transport of microorganisms by water. Biotechnology: 41-53.

Moorhead, D. L., Davis, W. S., Wolf, C. F. 1998. Coliform densities in urban waters of west Texas. Journal of Environmental Health 60(7): 14-18.

Nicosia, L. A. 1998. Assessment of Florida on-site disposal system regulations for the removal of viruses. Master's Thesis, University of South Florida.

Nieminski, E. C., and J. E. Ongerth. 1995. Removing *Giardia* and *Cryptosporidium* by conventional treatment and direct filtration. Journal of the American Water Works Association (November): 96-106.

Olivieri, V. P., C. W. Kruse, K. Kawata, and J. E. Smith. 1977. Microorganisms in Urban Stormwater. U. S. EPA Report No. EPA-600/2-77-087.

O'Shea, M. L., and R. Field. 1992. An evaluation of bacterial standards and disinfection practices used for the assessment and treatment of stormwater. Advances in Applied Microbiology 37: 21-40.

Overby, L. R., G. H. Barlow, R. H. Doi, M. Jacob, and S. Spiegelman. 1966. Comparison of two serologically distinct ribonucleic acid bacteriophages. Journal of Bacteriology 91: 422-448.

Parkhurst, R. S., T. C. Winter, D. O. Rosenberry, and A. M. Sturrock. 1998. Evaporation from a small prairie wetland in the Cottonwood Lake area, North Dakota - an energy-budget study. Wetlands 18(2): 272-287.

Paul, J. H., J. B. Rose, J. Brown, E. A. Shinn, S. Miller, and S. R. Farrah. 1995. Viral tracer studies indicate contamination of marine waters by sewage disposal practices in Key Largo, FL. Applied and Environmental Microbiology 61(6): 2230-2234.

Paul, J. H., J. B. Rose, S. C. Jiang, P. London, X. Xhou, and C. Kellogg. 1997. Coliphage and indigenous phage in Mamala Bay, Oahu, Hawaii. Applied and Environmental Microbiology 63(1): 133-138.

Payment, P. and E. Franco. 1993. *Clostridium perfringens* and somatic coliphages as indicators of the efficiency of drinking water treatment for viruses and protozoa cysts. Applied and Environmental Microbiology 59(8): 2418-2424.

Powelson, D. K., and C. P. Gerba. 1994. Virus removal from sewage effluents during saturated and unsaturated flow through soil columns. Water Resources 28(10): 2175-2181.

Powelson, D. K., J. R. Simpson, and C. P. Gerba. 1990. Virus transport and survival in saturated and unsaturated flow through soil columns. Journal of Environmental Quality 19: 396-401.

Price, F. A and D. R. Yonge. 1998. Enhancing contaminant removal in stormwater detention basins by coagulation. Transportation Research Record of the National Research Council.

Qureshi, A. A. and B. J. Dutka. 1979. Microbiological studies on the quality of urban stormwater runoff in southern Ontario, Canada. Water Research 13: 977-985.

Richardson, A. J., R. A. Grankenberg, A. C. Buck, J. B. Selkon, J. S. Colbourne, J. W. Parsons, and R. T. Mayon-White. 1991. An outbreak of waterborne cryptosporidiosis in Swindon and Oxfordshire. Epidemiology and Infection 10: 485-495.

Rose, J.B. H. Darbin, and C. P. Gerba. 1988. Correlations of the protozoa, *Cryptosporidium* and *Giardia*, with water quality variables in a watershed. Water Science Technology 20(11/12): 271-276.

Rose, J. B., and E. K. Lipp. 1997. A study on the presence of human viruses in surface waters of Sarasota County. Technical Report, Southwest Florida Water Management District, Brooksville, Florida.

Risenberg, F., B. B. Walters, A. Steele, and R. A. Ryder. 1995. Slow sand filters for a small water system. Journal of the American Water Works Association (November): 48-56.

Rushton, B. T. and C. W. Dye. 1993. An in-depth analysis of a wet detention stormwater system. Technical Report, Southwest Florida Water Management District, Brooksville, Florida.

Scheuler, T. R. 1987. Controlling Urban Runoff: A Practical Manual for Planning and Designing Urban BMPs. Publication No. 87703. Metropolitan Washington Council of Governments. 275 pp.

Schillinger, J. E. and J. J. Gannon. 1985. Bacterial adsorption and suspended particles in urban stormwater. Journal of the Water Pollution Control Federation 57(5): 384-389.

Shaver, E. 1992. Sand Filter Design for Water Quality Treatment. In: Stormwater Management: Urban Runoff Management Workshop. Book 2. U. S. Environmental Protection Agency, Washington, DC.

Sherer, B. M., J. R. Miner, J. A. Moore, and J. C. Buckhouse. 1992. Indicator bacterial survival in stream sediments. Journal of Environmental Quality 21: 591-595.

Sibille, I., T. Sime-Ngando, L. Mathieu, and J. C. Block. 1998. Protozoan bactivory and Escherichia coli survival in drinking water distribution systems. Applied and Environmental Microbiology 64(1): 197-202.

Smith, E. M., C. P. Gerba, and J. L. Melnick. 1978. Role of sediment in the persistence of enteroviruses in the estuarine environment. Applied and Environmental Microbiology Apr: 685-689.

Smith, H. V., J. Brown, J. C. Coulson, G. P. Morris, and R. W. A. Girdwood. 1993. Occurrence of oocysts of *Cryptosporidium* sp. in *Larus* spp. gulls. Epidemiology and Infection 110(1): 135-143.

Smith, H. V., W. J. Patterson, R. Hardie, L. A. Greene, C. Benton, W. Tulloch, R. A. Gilmour, R. W. A. Girdwood, J. C. M. Sharp, and G. I. Forbes. 1989. An outbreak of waterborne cryptosporidiosis caused by post-treatment contamination. Epidemiology and Infection 103: 703-715.

Smith, W. E., and W. L. Smith. 1982. The effects of advanced wastewater treatment on the coliform concentration in Hillsborough Bay. Proceedings of the Tampa Bay Area Scientific Information Symposium, Tampa, Florida. Ed. by S. Treat, J. L. Simon, R. R. Lewis, and R. L. Whitman. Sea Grant Report No. 65, Bellwether Press.

Souch, C., C. S. B. Grimmond, and C. P. Wolfe. 1998. Evapotranspiration rates from wetlands with different disturbance histories: Indiana Dunes National Lakeshore. Wetlands 18(2): 216-229.

Steahr, T. E. and T. Roberts. 1993. Microbial foodborne disease: hospitilizations, medical costs and potential demand for safer food. Regional Research Project NE-165, Private Strategies, Public Policy and Food Marketing Performance, Working Papers.

Sobsey, M. D., C. H. Dean, M. E. Knuckles, and R. A. Wagner. 1980. Interactions and survival of enteric viruses in soil materials. Applied and Environmental Microbiology 40(1): 92-101.

Sobsey, M. D., R. M. Hall, and R. L. Hazard. 1995. Comparative reductions of hepatitis a virus, enteroviruses and coliphage MS2 in miniature soil columns. Wat. Sci. Tech. 31(5-6): 203-209.

Southwest Florida Water Management District. 1995. District Water Management Plan. Brooksville, Florida.

Steahr, T. E. and T. Roberts. 1993. Microbial foodborne disease: hospitilizations, medical costs and potential demand for safer food. Regional Research Project NE-165, Private Strategies, Public Policy and Food Marketing Performance, Working Papers.

Tate, R. L., III. 1978. Cultural and environment factors affecting the longevity of *Escherichia coli* in Histosols. Applied and Environmental Microbiology 35: 925-929.

Thompson, S. S., M. Flury, M. V. Yates, and W. A. Jury. 1998. Role of the air-water-solid interface in bacteriophage sorption experiments. Applied and Environmental Microbiology 64(1): 304-309.

Thornton, C. I., S. R. Abt, and W. P. Clary. 1997. Vegetation influence on small stream siltation. Journal of the American Water Resources Association 33(6): 1279-1288.

USDC, Strategic Assessment Branch, NOS, NOAA. 1991. The 1990 National Shellfish Register of Classified Estuarine Waters. Rockville, MD.

U.S. Environmental Protection Agency. 1976. Proceedings of a workshop on microorganisms in urban stormwater. EPA-600/2-76-244, Environmental Protection Technology Series.

U.S. Environmental Protection Agency. 1984. Nonpoint Source Pollution in the U.S. Report to Congress, EPA 440/9-84-001, Office of Water Programs, Washington, D.C.

U.S. Environmental Protection Agency. 1998. Clean Water Action Plan: Restoring and Protecting America's Waters. Report for the Vice President, Washington, D.C.

U.S. Food and Drug Administration. 1992. Foodborne Microorganisms and Natural Toxins. Center for Food Safety and Applied Nutrition.

Urbonas, B. 1994. Assessment of stormwater BMPs and their technology. Water Science and Technology 29(1-2): 347-353.

Van Hattum, B., G. Korthals, N. M. Van Straalen, H. A. J. Govers, and E. N. G. Joosse. 1993. Accumulation patterns of trace metals in freshwater isopods in sediment bioassays-influence of substrate characteristics, temperature and pH. Water Research 27(4): 669-684.

Veenhuis, J., Parish, J., and M. Jennings. 1989. Monitoring and Design of Stormwater Control Basins, Design of Urban Runoff Quality Controls, American Society of Civil Engineers, New York, New York, pp. 224-238.

Viessman, Jr., W. and M. J. Hammer. 1993. Water Supply and Pollution Control, Fifth Edition. Harper Collins College Publisher, New York, New York.

Vonstille, W. T., W. T. Stille III, and R. C. Sharer. 1993. Hepatitis A epidemics from utility sewage in Ocoee, Florida. Archives of Environmental Health 48(2): 120-124.

Waller, W. T., M. F. Acevedo, E. L. Morgan, K. L. Dickson, J. H. Kennedy, L. P. Ammann,H. J. Allen, and P. R. Keating. 1994. Biological and chemical testing in stormwater. Stormwater NPDES Related Monitoring Needs, Proceedings of the Engineering Foundation Conference, Mount Crested Butte, Colorado, pp. 177-193.

Wentsel, R. S., P. E. O'Neill, and J. F. Kitchens. 1982. Evaluation of coliphage detection as a rapid indicator of water quality. Applied and Environmental Microbiology 43(2): 430-434.

Zita, A., and M. Hermansson. 1997. Effects of bacterial cell surface structures and hydrophobicity on attachment to activated sludge flocs. Applied and Environmental Microbiology 63(3): 1168-1170.

APPENDICES

Appendix 1. Raw data for microbial indicators from the three stormwater treatment systems.

Treatment	Date	Replicate	Time	MS2	Load MS2	TC	Load TC	FC	Load FC	Bead	Load Bead
				pfu/ml	*pfu*	*cfu/100ml*	*cfu*	*cfu/100ml*	*cfu*	*beads/ml*	*beads*
sand filter	6 SEP 95	unsaturated	IN1	9.69E+04	5.00E+11	6.28E+03	3.24E+08	5.36E+03	2.77E+08	1.94E+05	1.00E+10
sand filter	6 SEP 95	unsaturated	IN2	9.69E+04	5.00E+11	6.28E+03	3.24E+08	5.36E+03	2.77E+08	1.94E+05	1.00E+10
sand filter	6 SEP 95	unsaturated	IN3	9.69E+04	5.00E+11	6.28E+03	3.24E+08	5.36E+03	2.77E+08	1.94E+05	1.00E+10
sand filter	6 SEP 95	unsaturated	T1	1.10E+01	5.68E+04	1.87E+03	9.65E+06	9.30E+02	4.80E+06	3.33E+00	1.72E+04
sand filter	6 SEP 95	unsaturated	T2	2.00E+01	1.03E+05	1.45E+03	7.48E+06	3.70E+02	1.91E+06	6.67E+01	3.44E+05
sand filter	6 SEP 95	unsaturated	T3	7.50E+01	3.87E+05	1.23E+03	6.35E+06	3.30E+02	1.70E+06	3.00E+01	1.55E+05
sand filter	6 SEP 95	unsaturated	T4	1.65E+02	8.51E+05	1.32E+03	6.81E+06	5.40E+02	2.79E+06	3.33E+01	1.72E+05
sand filter	6 SEP 95	unsaturated	T5	9.00E+01	4.64E+05	1.30E+03	6.71E+06	2.70E+02	1.39E+06	7.67E+01	3.96E+05
sand filter	6 SEP 95	unsaturated	T6	7.50E+01	3.87E+05	1.30E+03	6.71E+06	1.40E+02	7.22E+05	2.33E+01	1.20E+05
sand filter	6 SEP 95	unsaturated	T7	8.50E+01	4.39E+05	9.50E+02	4.90E+06	2.30E+02	1.19E+06	2.33E+01	1.20E+05
sand filter	6 SEP 95	unsaturated	T8	7.00E+01	3.61E+05	9.70E+02	5.01E+06	9.00E+01	4.64E+05	3.67E+01	1.89E+05
sand filter	6 SEP 95	unsaturated	T9	1.15E+02	5.93E+05	1.16E+03	5.99E+06	2.30E+02	1.19E+06	1.67E+01	8.60E+04
sand filter	6 SEP 95	unsaturated	T10	5.50E+01	2.84E+05	1.00E+03	5.16E+06	2.60E+02	1.34E+06	3.00E+01	1.55E+05
sand filter	10 SEP 96	saturated	IN1	9.69E+05	3.00E+12	6.00E+04	2.53E+09	2.30E+04	7.57E+08	1.94E+05	1.00E+10
sand filter	10 SEP 96	saturated	IN2	9.69E+05	3.00E+12	2.80E+04	2.53E+09	1.10E+04	7.57E+08	1.94E+05	1.00E+10
sand filter	10 SEP 96	saturated	IN3	9.69E+05	3.00E+12	5.90E+04	2.53E+09	1.00E+04	7.57E+08	1.94E+05	1.00E+10
sand filter	10 SEP 96	saturated	T1	8.00E+02	4.13E+08	1.20E+05	6.19E+08	1.00E+04	5.16E+07	0.00E+00	0.00E+00
sand filter	10 SEP 96	saturated	T2	1.55E+04	8.00E+09	1.10E+03	5.68E+06	3.10E+02	1.60E+06	2.00E+00	1.03E+06
sand filter	10 SEP 96	saturated	T3	3.30E+04	1.70E+10	6.70E+02	3.46E+06	6.00E+02	3.10E+06	6.00E+00	3.10E+04
sand filter	10 SEP 96	saturated	T4	9.70E+04	5.01E+10	6.70E+02	3.46E+06	3.00E+02	1.55E+06	4.00E+00	2.06E+04
sand filter	10 SEP 96	saturated	T5	9.50E+04	4.90E+10	6.30E+02	3.25E+06	2.00E+02	1.03E+06	2.00E+00	1.03E+04
sand filter	10 SEP 96	saturated	T6	8.40E+04	4.33E+10	5.20E+02	2.68E+06	2.60E+02	1.34E+06	1.00E+01	5.16E+04
sand filter	10 SEP 96	saturated	T7	4.08E+04	2.11E+10	5.10E+02	2.63E+06	2.00E+02	1.03E+06	1.00E+01	5.16E+04
sand filter	10 SEP 96	saturated	T8	7.90E+04	4.08E+10	3.00E+03	1.55E+07	1.40E+02	7.22E+05	4.00E+00	2.06E+04
sand filter	10 SEP 96	saturated	T9	1.87E+05	9.65E+10	3.00E+03	1.55E+07	3.00E+02	1.55E+06	2.00E+00	1.03E+04
sand filter	10 SEP 96	saturated	T10	8.00E+04	4.13E+10	6.60E+02	3.41E+06	3.10E+02	1.60E+06	8.00E+00	4.13E+04
sand filter	19 NOV 96	saturated	IN1	9.69E+04	5.00E+11	9.00E+04	5.62E+09	1.72E+04	1.20E+09	1.94E+05	1.00E+10
sand filter	19 NOV 96	saturated	IN2	9.69E+04	5.00E+11	1.20E+05	5.62E+09	2.78E+04	1.20E+09	1.94E+05	1.00E+10
sand filter	19 NOV 96	saturated	IN3	9.69E+04	5.00E+11	1.17E+04	5.62E+09	2.48E+04	1.20E+09	1.94E+05	1.00E+10
sand filter	19 NOV 96	saturated	T1	0.00E+00	0.00E+00	4.80E+04	2.48E+08	1.22E+04	6.30E+07	2.00E+01	1.03E+05
sand filter	19 NOV 96	saturated	T2	0.00E+00	0.00E+00	3.70E+04	1.91E+08	1.74E+04	8.98E+07	1.95E+03	1.01E+07
sand filter	19 NOV 96	saturated	T3	5.22E+03	2.69E+09	4.70E+04	2.43E+08	1.44E+04	7.43E+07	3.75E+03	1.93E+07
sand filter	19 NOV 96	saturated	T4	1.99E+04	1.03E+10	6.70E+04	3.46E+08	1.82E+04	9.39E+07	3.44E+03	1.78E+07
sand filter	19 NOV 96	saturated	T5	1.66E+04	8.57E+09	5.10E+04	2.63E+08	1.72E+04	8.88E+07	3.60E+03	1.86E+07
sand filter	19 NOV 96	saturated	T6	1.33E+04	6.88E+09	4.60E+04	2.37E+08	1.48E+04	7.64E+07	2.72E+03	1.40E+07
sand filter	19 NOV 96	saturated	T7	4.44E+04	2.29E+10	3.30E+04	1.70E+08	1.78E+04	9.18E+07	3.35E+03	1.73E+07
sand filter	19 NOV 96	saturated	T8	4.43E+04	2.29E+10	2.90E+04	1.50E+08	1.38E+04	7.12E+07	1.71E+03	8.82E+06
sand filter	19 NOV 96	saturated	T9	5.48E+04	2.83E+10	2.70E+04	1.39E+08	1.34E+04	6.91E+07	9.10E+02	4.70E+06
sand filter	19 NOV 96	saturated	T10	3.98E+04	2.05E+10	4.10E+04	2.12E+08	2.04E+04	1.05E+08	1.23E+03	6.35E+06
wet detention	26 JAN 98	5-Day Shallow	IN1	9.25E+04	1.20E+13	9.00E+02	1.46E+11	2.00E+02	3.65E+10	3.72E+02	4.00E+10
wet detention	26 JAN 98	5-Day Shallow	IN2	9.25E+04	1.20E+13	1.50E+03	1.46E+11	2.00E+02	3.65E+10	3.72E+02	4.00E+10
wet detention	26 JAN 98	5-Day Shallow	IN3	9.25E+04	1.20E+13	1.10E+03	1.46E+11	3.00E+02	3.65E+10	3.72E+02	4.00E+10
wet detention	26 JAN 98	5-Day Shallow	OUT1	0.00E+00	0.00E+00	2.70E+02	8.38E+07	0.00E+00	0.00E+00	1.00E-01	3.11E+06
wet detention	27 JAN 98	5-Day Shallow	OUT2	2.60E+04	8.07E+11	1.03E+03	1.56E+08	3.00E+01	4.54E+06	5.55E+00	8.39E+07
wet detention	27 JAN 98	5-Day Shallow	OUT3	2.00E+04	3.02E+11	7.00E+02	1.45E+08	0.00E+00	0.00E+00	4.30E+00	8.88E+07

Appendix 1 (Continued).

Treatment	Date	Replicate	Time	MS2	Load MS2	TC	Load TC	FC	Load FC	Bead	Load Bead
wet detention	28 JAN 98	5-Day Shallow	OUT4	8.40E+03	1.74E+11	2.00E+02	2.57E+07	0.00E+00	0.00E+00	1.20E+00	1.54E+07
wet detention	29 JAN 98	5-Day Shallow	OUT5	4.30E+03	5.53E+10	1.00E+02	1.19E+07	0.00E+00	0.00E+00	5.50E-01	6.53E+06
wet detention	30 JAN 98	5-Day Shallow	OUT6	1.13E+02	1.34E+09	5.00E+01	9.72E+06	0.00E+00	0.00E+00	2.00E-01	3.89E+06
wet detention	26 JAN 98	5-Day Deep	IN1	9.24E+04	1.20E+13	5.00E+02	8.43E+11	1.00E+02	2.06E+08	3.08E+02	4.00E+10
wet detention	26 JAN 98	5-Day Deep	IN2	9.24E+04	1.20E+13	7.00E+02	8.43E+11	2.00E+02	2.06E+08	3.08E+02	4.00E+10
wet detention	26 JAN 98	5-Day Deep	IN3	9.24E+04	1.20E+13	9.00E+02	8.43E+11	2.00E+02	2.06E+08	3.08E+02	4.00E+10
wet detention	26 JAN 98	5-Day Deep	OUT1	2.50E+00	7.38E+07	1.90E+03	5.61E+08	0.00E+00	0.00E+00	2.00E-01	5.90E+06
wet detention	27 JAN 98	5-Day Deep	OUT2	1.20E+04	1.43E+11	3.90E+03	4.63E+08	2.00E+02	2.38E+07	7.40E+00	8.79E+07
wet detention	27 JAN 98	5-Day Deep	OUT3	6.10E+03	1.12E+11	6.00E+03	1.10E+09	0.00E+00	0.00E+00	6.60E+00	1.21E+08
wet detention	28 JAN 98	5-Day Deep	OUT4	4.30E+03	6.81E+10	2.80E+03	4.44E+08	0.00E+00	0.00E+00	5.70E+00	9.03E+07
wet detention	29 JAN 98	5-Day Deep	OUT5	1.60E+03	2.22E+10	2.60E+03	3.60E+08	0.00E+00	0.00E+00	3.95E+00	5.47E+07
wet detention	30 JAN 98	5-Day Deep	OUT6	8.90E+01	1.73E+09	2.40E+03	4.67E+08	0.00E+00	0.00E+00	1.45E+00	2.82E+07
wet detention	3 NOV 97	14-Day Shallow	IN1	7.07E+03	1.50E+12	5.00E+03	9.24E+09	2.00E+03	4.42E+09	1.88E+02	4.00E+10
wet detention	3 NOV 97	14-Day Shallow	IN2	7.07E+03	1.50E+12	4.00E+03	9.24E+09	1.40E+03	4.42E+09	1.88E+02	4.00E+10
wet detention	3 NOV 97	14-Day Shallow	IN3	7.07E+03	1.50E+12	4.10E+03	9.24E+09	3.20E+03	4.42E+09	1.88E+02	4.00E+10
wet detention	3 NOV 97	14-Day Shallow	OUT1	1.20E+04	1.69E+11	1.50E+04	2.12E+09	4.00E+03	5.64E+08	2.54E+01	3.58E+08
wet detention	4 NOV 97	14-Day Shallow	OUT2	6.60E+03	6.82E+10	1.30E+02	1.34E+07	3.00E+01	3.10E+06	1.00E+00	1.03E+07
wet detention	5 NOV 97	14-Day Shallow	OUT3	2.90E+03	2.86E+10	1.90E+04	1.88E+09	1.40E+03	1.38E+08	3.00E-01	2.96E+06
wet detention	10 NOV 97	14-Day Shallow	OUT4	4.35E+01	1.22E+09	4.00E+01	1.12E+07	0.00E+00	0.00E+00	0.00E+00	0.00E+00
wet detention	12 NOV 97	14-Day Shallow	OUT5	1.50E+01	2.38E+08	1.60E+01	2.53E+06	0.00E+00	0.00E+00	0.00E+00	0.00E+00
wet detention	3 NOV 97	14-Day Deep	IN1	6.95E+03	1.50E+12	5.00E+03	8.07E+09	2.00E+03	4.18E+09	1.85E+02	4.00E+10
wet detention	3 NOV 97	14-Day Deep	IN2	6.95E+03	1.50E+12	4.10E+03	8.07E+09	3.20E+03	4.18E+09	1.85E+02	4.00E+10
wet detention	3 NOV 97	14-Day Deep	IN3	6.95E+03	1.50E+12	2.10E+03	8.07E+09	6.00E+02	4.18E+09	1.85E+02	4.00E+10
wet detention	3 NOV 97	14-Day Deep	OUT1	5.20E+03	9.04E+10	7.00E+03	1.22E+09	3.00E+03	3.90E+05	4.95E+00	8.61E+07
wet detention	4 NOV 97	14-Day Deep	OUT2	2.90E+03	3.82E+10	5.60E+03	7.37E+08	1.20E+03	1.44E+05	4.55E+00	5.99E+07
wet detention	5 NOV 97	14-Day Deep	OUT3	8.70E+02	9.00E+09	7.00E+03	7.24E+08	1.10E+03	8.80E+04	1.15E+00	1.19E+07
wet detention	10 NOV 97	14-Day Deep	OUT4	8.50E+00	2.30E+08	1.00E+03	2.70E+08	0.00E+00	0.00E+00	5.00E-02	1.35E+06
wet detention	12 NOV 97	14-Day Deep	OUT5	1.00E+00	1.37E+07	2.00E+03	2.74E+08	2.00E+01	8.00E+02	5.00E-02	6.84E+05
alum 600 mg/L	9 JUN 97	control	FLOC	1.50E+05	3.00E+08	0.00E+00	0.00E+00	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	7 JUN 97	control	0	1.20E+08	1.68E+12	1.80E+05	2.52E+09	4.30E+04	6.02E+08	n.d.	n.d.
alum 600 mg/L	8 JUN 97	control	24	2.40E+07	2.88E+11	3.00E+04	3.60E+08	1.00E+04	1.20E+08	n.d.	n.d.
alum 600 mg/L	9 JUN 97	control	48	8.90E+05	8.90E+09	2.00E+02	2.00E+06	1.00E+02	1.00E+06	n.d.	n.d.
alum 600 mg/L	9 JUN 97	rep 1	FLOC	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	7 JUN 97	rep 1	0	1.20E+08	1.68E+12	1.30E+05	1.82E+09	3.00E+04	4.20E+08	n.d.	n.d.
alum 600 mg/L	8 JUN 97	rep 1	24	4.50E+00	5.40E+04	0.00E+00	0.00E+00	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	9 JUN 97	rep 1	48	3.70E+01	3.70E+05	0.00E+00	0.00E+00	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	9 JUN 97	rep 2	FLOC	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	7 JUN 97	rep 2	0	1.10E+08	1.54E+12	1.60E+05	2.24E+09	4.00E+04	5.60E+08	n.d.	n.d.
alum 600 mg/L	8 JUN 97	rep 2	24	2.50E+02	3.00E+06	6.00E+04	8.40E+08	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	9 JUN 97	rep 2	48	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	9 JUN 97	rep 3	FLOC	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	7 JUN 97	rep 3	0	1.20E+08	1.68E+12	1.90E+05	2.66E+09	3.50E+04	4.90E+08	n.d.	n.d.
alum 600 mg/L	8 JUN 97	rep 3	24	1.90E+01	2.28E+05	0.00E+00	0.00E+00	0.00E+00	0.00E+00	n.d.	n.d.
alum 600 mg/L	9 JUN 97	rep 3	48	0.00E+00	0.00E+00	2.00E+04	2.00E+08	0.00E+00	0.00E+00	n.d.	n.d.
alum 10 mg/L	5 FEB 98	control	FLOC	2.10E+07	4.20E+10	2.38E+05	4.76E+08	9.80E+03	1.96E+07	4.00E+05	8.00E+08
alum 10 mg/L	3 FEB 98	control	0	1.30E+08	1.82E+12	2.30E+04	3.22E+08	1.60E+04	2.24E+08	1.60E+05	2.24E+09
alum 10 mg/L	4 FEB 98	control	24	7.10E+07	8.52E+11	2.02E+05	2.42E+09	1.03E+05	1.24E+09	4.00E+04	4.80E+08

Appendix 1 (Continued).

Treatment	Date	Replicate	Time	MS2	Load MS2	TC	Load TC	FC	Load FC	Bead	Load Bead
alum 10 mg/L	5 FEB 98	control	48	3.10E+03	3.10E+07	3.64E+03	3.64E+07	3.24E+03	3.24E+07	6.00E+04	6.00E+08
alum 10 mg/L	5 FEB 98	rep 1	FLOC	3.80E+01	7.60E+04	1.64E+05	3.28E+08	6.10E+03	1.22E+07	6.40E+05	1.28E+09
alum 10 mg/L	3 FEB 98	rep 1	0	2.00E+08	2.80E+12	2.30E+04	3.22E+08	1.80E+04	2.52E+08	1.00E+05	1.40E+09
alum 10 mg/L	4 FEB 98	rep 1	24	3.70E+01	4.44E+05	1.50E+02	1.80E+06	4.00E+01	4.80E+05	1.90E+03	2.28E+07
alum 10 mg/L	5 FEB 98	rep 1	48	3.60E+01	3.60E+05	3.00E+01	3.00E+05	0.00E+00	0.00E+00	1.30E+03	1.30E+07
alum 10 mg/L	5 FEB 98	rep 2	FLOC	1.10E+02	2.20E+05	1.00E+05	2.00E+08	2.60E+03	5.20E+06	6.30E+05	1.26E+09
alum 10 mg/L	3 FEB 98	rep 2	0	1.90E+08	2.66E+12	2.20E+04	3.08E+08	1.40E+04	1.96E+08	8.00E+04	1.12E+09
alum 10 mg/L	4 FEB 98	rep 2	24	7.80E+01	9.36E+05	1.20E+02	1.44E+06	4.00E+01	4.80E+05	9.00E+02	1.08E+07
alum 10 mg/L	5 FEB 98	rep 2	48	4.20E+01	4.20E+05	1.00E+02	1.00E+06	1.00E+02	1.00E+06	1.30E+03	1.30E+07
alum 10 mg/L	5 FEB 98	rep 3	FLOC	4.20E+02	8.40E+05	1.46E+05	2.92E+08	9.60E+04	1.92E+08	7.00E+04	9.80E+08
alum 10 mg/L	3 FEB 98	rep 3	0	2.10E+08	2.94E+12	2.60E+04	3.64E+08	1.40E+04	1.96E+08	1.70E+03	2.04E+07
alum 10 mg/L	4 FEB 98	rep 3	24	1.50E+04	1.80E+08	2.30E+02	2.76E+06	1.20E+02	1.44E+06	7.00E+02	7.00E+06
alum 10 mg/L	5 FEB 98	rep 3	48	1.50E+02	1.50E+06	5.00E+01	5.00E+05	3.00E+01	3.00E+05	4.00E+05	8.00E+08

Appendix 2. Raw data for physicochemical parameters from the three stormwater treatment systems (n.d. = no data collected).

Treatment	Date	Replicate	Time	Turbidity *NTU*	TSS *mg/L*	Load TSS *mg*	Temp. *°C*	pH *s.u.*	Cond. *µS/cm*
sand filter	6 SEP 95	unsaturated	IN1	6.8	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	IN2	6.8	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	IN3	6.8	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T1	0.8	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T2	0.5	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T3	0.3	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T4	0.3	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T5	0.2	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T6	0.1	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T7	0.1	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T8	0.1	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T9	0.1	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T10	0.2	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	10 SEP 96	saturated	IN1	21.0	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	10 SEP 96	saturated	IN2	21.0	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	10 SEP 96	saturated	IN3	21.0	n.d.	n.d.	28.43	7.51	1174
sand filter	10 SEP 96	saturated	T1	2.2	n.d.	n.d.	27.72	6	902
sand filter	10 SEP 96	saturated	T2	1.5	n.d.	n.d.	27.7	7.19	971
sand filter	10 SEP 96	saturated	T3	0.8	n.d.	n.d.	27.66	7.11	1214
sand filter	10 SEP 96	saturated	T4	0.7	n.d.	n.d.	27.6	7.08	1368
sand filter	10 SEP 96	saturated	T5	0.4	n.d.	n.d.	27.6	7.07	1411
sand filter	10 SEP 96	saturated	T6	1.0	n.d.	n.d.	27.61	7.06	1407
sand filter	10 SEP 96	saturated	T7	0.4	n.d.	n.d.	27.66	7.07	1399
sand filter	10 SEP 96	saturated	T8	0.5	n.d.	n.d.	27.72	7.07	1382
sand filter	10 SEP 96	saturated	T9	0.5	n.d.	n.d.	27.76	7.08	1377
sand filter	10 SEP 96	saturated	T10	0.5	n.d.	n.d.	27.81	7.08	1365
sand filter	19 NOV 96	saturated	IN1	15.9	19.27	n.d.	22.78	7.6	28
sand filter	19 NOV 96	saturated	IN2	20.0	19.27	n.d.	22.78	7.6	28
sand filter	19 NOV 96	saturated	IN3	22.0	19.27	n.d.	22.78	7.6	28
sand filter	19 NOV 96	saturated	T1	34.0	29.57	n.d.	23.34	7.42	206
sand filter	19 NOV 96	saturated	T2	4.0	2.08	n.d.	22.28	7.48	89
sand filter	19 NOV 96	saturated	T3	2.7	1.78	n.d.	21.8	7.4	75
sand filter	19 NOV 96	saturated	T4	4.0	2.90	n.d.	22.76	7.4	47
sand filter	19 NOV 96	saturated	T5	4.0	3.03	n.d.	22.2	7.36	40
sand filter	19 NOV 96	saturated	T6	4.0	2.78	n.d.	22.34	7.32	34
sand filter	19 NOV 96	saturated	T7	3.5	2.63	n.d.	22.64	7.27	31
sand filter	19 NOV 96	saturated	T8	5.6	3.60	n.d.	24.55	7.57	n.d.
sand filter	19 NOV 96	saturated	T9	5.8	5.12	n.d.	24.35	7.43	n.d.
sand filter	19 NOV 96	saturated	T10	4.0	2.78	n.d.	24.19	7.46	n.d.
wet detention	26 JAN 98	5-Day Shallow	IN1	0.9	1.11	1.52E+08	21.5	7.6	550
wet detention	26 JAN 98	5-Day Shallow	IN2	1.4	1.84	1.52E+08	21.5	7.5	551
wet detention	26 JAN 98	5-Day Shallow	IN3	1.4	1.30	1.52E+08	21.4	7.4	552
wet detention	26 JAN 98	5-Day Shallow	OUT1	1.0	0.62	1.91E+05	16.6	7.6	365
wet detention	27 JAN 98	5-Day Shallow	OUT2	0.8	0.13	2.00E+04	18	7.4	496
wet detention	27 JAN 98	5-Day Shallow	OUT3	0.9	0.30	6.18E+04	19.7	7.8	490
wet detention	28 JAN 98	5-Day Shallow	OUT4	0.8	0.05	6.56E+03	15.6	7.6	486

Appendix 2 (Continued).

Treatment	Date	Replicate	Time	Turbidity	TSS	Load TSS	Temp.	pH	Cond.
wet detention	29 JAN 98	5-Day Shallow	OUT5	0.8	0.29	3.45E+04	13.2	7.6	476
wet detention	30 JAN 98	5-Day Shallow	OUT6				15.8	7.7	457
wet detention	26 JAN 98	5-Day Deep	IN1	1.5	1.50	2.17E+08	21.2	7.7	550
wet detention	26 JAN 98	5-Day Deep	IN2	1.0	2.41	2.17E+08	21.2	7.5	550
wet detention	26 JAN 98	5-Day Deep	IN3	0.9	1.11	2.17E+08	21.5	7.6	550
wet detention	26 JAN 98	5-Day Deep	OUT1	5.1	5.57	1.64E+08	16.9	7.9	413
wet detention	27 JAN 98	5-Day Deep	OUT2	4.1	4.83	5.74E+07	17.5	7.6	456
wet detention	27 JAN 98	5-Day Deep	OUT3	4.6	3.97	7.29E+07	19.9	7.6	462
wet detention	28 JAN 98	5-Day Deep	OUT4	3.7	3.32	5.27E+07	15.4	7.7	454
wet detention	29 JAN 98	5-Day Deep	OUT5	4.1	3.33	4.62E+07	13.1	7.7	456
wet detention	30 JAN 98	5-Day Deep	OUT6				17.6	7.9	456
wet detention	3 NOV 97	14-Day Shallow	IN1	3.4	3.26	7.58E+08	22.1	7.8	489
wet detention	3 NOV 97	14-Day Shallow	IN2	4.4	3.04	7.58E+08	23.3	7.4	486
wet detention	3 NOV 97	14-Day Shallow	IN3	3.6	4.39	7.58E+08	23.3	7.4	487
wet detention	3 NOV 97	14-Day Shallow	OUT1	2.4	1.30	3.21E+07	24.3	7.2	470
wet detention	4 NOV 97	14-Day Shallow	OUT2	2.5	0.70	2.56E+07	23.4	7.3	481
wet detention	5 NOV 97	14-Day Shallow	OUT3	2.1	1.58	7.54E+07	22.3	7.6	486
wet detention	10 NOV 97	14-Day Shallow	OUT4	1.7	0.72	2.69E+07	17.8	6.5	512
wet detention	12 NOV 97	14-Day Shallow	OUT5	3.2	0.51	5.07E+07	20.5	6.5	517
wet detention	3 NOV 97	14-Day Deep	IN1	3.4	3.26	7.35E+08	22.1	7.8	489
wet detention	3 NOV 97	14-Day Deep	IN2	3.6	4.39	7.35E+08	23.3	7.4	487
wet detention	3 NOV 97	14-Day Deep	IN3	4.5	2.54	7.35E+08	22.7	7.4	484
wet detention	3 NOV 97	14-Day Deep	OUT1	5.4	3.04	5.25E+07	22.7	7.4	494
wet detention	4 NOV 97	14-Day Deep	OUT2	3.4	2.04	2.59E+07	24.3	7.4	492
wet detention	5 NOV 97	14-Day Deep	OUT3	3.3	2.19	7.97E+07	23.1	7	492
wet detention	10 NOV 97	14-Day Deep	OUT4	4.3	2.15	2.94E+07	18.3	6.7	508
wet detention	12 NOV 97	14-Day Deep	OUT5	4.2	1.70	8.57E+06	20.5	6.7	517
alum 600 mg/L	9 JUN 97	control	FLOC	12.4	16.15	3.23E+04	n.d.	n.d.	n.d.
alum 600 mg/L	7 JUN 97	control	0	7.0	5.24	7.34E+04	25.17	7.02	741
alum 600 mg/L	8 JUN 97	control	24	5.2	2.30	2.76E+04	25.37	7.46	787
alum 600 mg/L	9 JUN 97	control	48	2.2	2.42	2.42E+04	27.84	7.86	810
alum 600 mg/L	9 JUN 97	rep 1	FLOC	34.0	67.17	1.34E+05	n.d.	n.d.	n.d.
alum 600 mg/L	7 JUN 97	rep 1	0	7.1	5.00	7.00E+04	25.17	7.02	741
alum 600 mg/L	8 JUN 97	rep 1	24	2.9	3.29	3.95E+04	24.94	3.78	2683
alum 600 mg/L	9 JUN 97	rep 1	48	2.4	3.20	3.20E+04	27.32	3.75	2626
alum 600 mg/L	9 JUN 97	rep 2	FLOC	37.0	87.80	1.76E+05	n.d.	n.d.	n.d.
alum 600 mg/L	7 JUN 97	rep 2	0	7.0	4.52	6.33E+04	25.17	7.02	741
alum 600 mg/L	8 JUN 97	rep 2	24	2.3	3.52	4.22E+04	24.74	3.78	2653
alum 600 mg/L	9 JUN 97	rep 2	48	1.9	2.86	2.86E+04	27.79	3.75	2662
alum 600 mg/L	9 JUN 97	rep 3	FLOC	52.0	50.71	1.01E+05	n.d.	n.d.	n.d.
alum 600 mg/L	7 JUN 97	rep 3	0	7.0	3.50	4.90E+04	25.17	7.02	741
alum 600 mg/L	8 JUN 97	rep 3	24	2.6	4.18	5.02E+04	24.78	3.77	2663
alum 600 mg/L	9 JUN 97	rep 3	48	1.8	3.15	3.15E+04	28.2	3.74	2658
alum 10 mg/L	5 FEB 98	control	FLOC	27.0	39.73	n.d.	n.d.	n.d.	n.d.
alum 10 mg/L	3 FEB 98	control	0	12.4	11.26	3.74E+06	20.2	7.6	267
alum 10 mg/L	4 FEB 98	control	24	8.4	7.53	3.23E+06	13.8	7.5	269
alum 10 mg/L	5 FEB 98	control	48	6.2	6.50	2.69E+06	12.1	8.5	269

Appendix 2 (Continued).

Treatment	Date	Replicate	Time	Turbidity	TSS	Load TSS	Temp.	pH	Cond.
alum 10 mg/L	5 FEB 98	rep 1	FLOC	94.0	226.50	n.d.	n.d.	n.d.	n.d.
alum 10 mg/L	3 FEB 98	rep 1	0	12.3	11.26	3.72E+06	19.9	7.6	266
alum 10 mg/L	4 FEB 98	rep 1	24	0.7	1.52	3.48E+06	14.1	7.5	290
alum 10 mg/L	5 FEB 98	rep 1	48	1.2	0.74	2.88E+06	12.6	7.8	288
alum 10 mg/L	5 FEB 98	rep 2	FLOC	93.0	8.80	n.d.	n.d.	n.d.	n.d.
alum 10 mg/L	3 FEB 98	rep 2	0	12.0	11.78	3.70E+06	19.8	7.6	264
alum 10 mg/L	4 FEB 98	rep 2	24	1.0	1.88	3.46E+06	14.3	7.5	288
alum 10 mg/L	5 FEB 98	rep 2	48	0.9	1.08	2.98E+06	13.6	7.9	298
alum 10 mg/L	5 FEB 98	rep 3	FLOC	91.0	159.33	n.d.	n.d.	n.d.	n.d.
alum 10 mg/L	3 FEB 98	rep 3	0	11.1	12.00	3.72E+06	19.9	7.6	266
alum 10 mg/L	4 FEB 98	rep 3	24	1.3	2.46	3.42E+06	14.8	7.5	285
alum 10 mg/L	5 FEB 98	rep 3	48	1.7	1.23	2.89E+06	15.2	7.9	289

Appendix 3. Raw data for heavy metal samples from the sand filter trials.

Treatment	Date	Replicate	Time	Zn *ppb*	Cd *ppb*	Cu *ppb*	Ni *ppb*	Cr *ppb*	Pb *ppb*
sand filter	6 SEP 95	unsaturated	IN1	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	IN2	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	IN3	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T1	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T2	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T3	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T4	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T5	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T6	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T7	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T8	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T9	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	6 SEP 95	unsaturated	T10	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	10 SEP 96	saturated	IN1	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	10 SEP 96	saturated	IN2	n.d.	n.d.	n.d.	n.d.	n.d.	n.d.
sand filter	10 SEP 96	saturated	IN3	101.00	0.50	9.90	3.00	1.90	7.30
sand filter	10 SEP 96	saturated	T1	15.00	0.10	3.90	1.60	0.90	1.50
sand filter	10 SEP 96	saturated	T2	19.00	0.20	7.00	2.20	0.90	0.40
sand filter	10 SEP 96	saturated	T3	24.00	0.10	3.50	1.90	0.50	0.60
sand filter	10 SEP 96	saturated	T4	27.00	0.10	4.10	0.70	0.00	0.30
sand filter	10 SEP 96	saturated	T5	23.00	0.10	4.10	1.20	1.60	2.80
sand filter	10 SEP 96	saturated	T6	17.00	0.10	3.80	1.00	0.80	1.30
sand filter	10 SEP 96	saturated	T7	24.00	0.20	3.00	2.20	0.10	1.40
sand filter	10 SEP 96	saturated	T8	19.00	0.10	3.40	1.20	0.00	0.00
sand filter	10 SEP 96	saturated	T9	16.00	0.00	3.90	1.70	1.00	1.10
sand filter	10 SEP 96	saturated	T10	20.00	0.10	4.80	0.30	0.00	0.00
sand filter	19 NOV 96	saturated	IN1	115.33	4.23	16.90	8.37	12.90	17.73
sand filter	19 NOV 96	saturated	IN2	115.33	4.23	16.90	8.37	12.90	17.73
sand filter	19 NOV 96	saturated	IN3	115.33	4.23	16.90	8.37	12.90	17.73
sand filter	19 NOV 96	saturated	T1	32.00	0.70	15.40	13.10	12.90	2.40
sand filter	19 NOV 96	saturated	T2	23.00	0.40	14.30	2.10	15.10	0.40
sand filter	19 NOV 96	saturated	T3	13.00	0.20	6.00	0.00	1.20	0.20
sand filter	19 NOV 96	saturated	T4	13.00	0.20	5.30	1.10	3.40	0.00
sand filter	19 NOV 96	saturated	T5	15.00	0.20	5.60	0.00	3.50	0.20
sand filter	19 NOV 96	saturated	T6	13.00	0.30	4.90	0.00	2.10	0.00
sand filter	19 NOV 96	saturated	T7	18.00	0.30	12.80	6.20	23.90	0.00
sand filter	19 NOV 96	saturated	T8	13.00	0.00	9.70	0.60	3.70	0.50
sand filter	19 NOV 96	saturated	T9	17.00	0.30	7.90	9.90	55.00	0.00
sand filter	19 NOV 96	saturated	T10	23.00	0.30	12.80	1.60	7.60	0.60