Assessing the Performance of Two Stormwater Management Ponds in Waterloo, Ontario

by

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A thesis
presented to the University of Waterloo
in fulfillment of the
thesis requirement for the degree of
Master of Science
in
Geography

Waterloo, Ontario, Canada, 2010

AUTHOR'S DECLARATION

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Stormwater (SW) runoff in urban areas represents a major pathway for pollutant transfer to receiving waters. Best management practices (BMP) were introduced in the 1970s to help mitigate the negative effects of SW. In the 1990s, Stormwater management (SWM) ponds were established as a BMP to help increase the water quality of SW effluent. Many SWM ponds do not provide sufficient water quality treatment. Information on the internal processes influencing the reduction of total phosphorus (TP), soluble reactive phosphorus (SRP) and total suspended solid (TSS) concentrations in SWM ponds with different designs is lacking. Knowledge of the processes affecting TP, SRP and TSS retention can help improve the design of SWM ponds to enhance their treatment performance.

The purpose of this thesis is to provide an assessment of the internal chemical processes that affect the trap efficiency (TE) and spatial and temporal variability of TP, SRP and TSS concentrations at two structurally different SWM ponds (Pond 45; conventional and Pond 33; hybrid extended detention) in Waterloo, ON. Water samples were collected at the inflow and outflow at the two SWM ponds during six storm events and 30 baseflow periods. A mass balance approach was used to quantify the TE of TP, SRP and TSS concentrations at each pond. Pond 33 had a TE of 24.3%, 26.7% and 66.8% for baseflow and stormflow samples of TP, SRP and TSS. Pond 45 performed much better with TE of 93.8%, 94.2% and 98% for TP, SRP and TSS concentrations. Pond 33 was a source of TP, SRP and TSS for 3, 4 and 2 storm events sampled during the field season, respectively. Pond 45 was a sink for all parameters on all storm events samples.

The spatial and temporal variability of TP, SRP and TSS concentrations were examined to improve knowledge of external factors and internal processes that influence the TE of SWM ponds. The effects of storm magnitude, seasonality and vegetation growth and senescence on effluent water quality were investigated. Additionally, the role of sediment on P cycling in the ponds was evaluated by determining grain size distribution, porewater SRP concentrations, sediment geochemistry and mineralogy, and the sediment P buffering capacity. Vegetation senescence, anoxic conditions, porewater SRP concentrations, sediment characteristics and buffering capacity influenced the poor TE at Pond 33. Pond 45 had more favourable water column conditions, i.e. higher dissolved oxygen concentrations, therefore allowed greater amounts of P to adsorb onto sediment. Design and maintenance considerations are described to help improve the performance at Pond 33. Continual water quality monitoring of SW effluent will identify changes in quality and mitigation measures can be implemented to increase a SWM ponds performance.

Acknowledgements

I would like to start by thanking my advisor Dr. Mike Stone for providing me with support and guidance during this study. Whenever I felt discouraged or defeated, having a discussion with you always revived my excitement for my study.

I would also like to thank my examining committee, Dr. Jiri Marsalek, Dr. Stephen Murphy and Dr. Merrin Macrae for taking the time to provide comments and have an insightful conversation about this study with me.

This research would not have been completed without the support of Denise McGoldrick and John Burke at the City of Waterloo. Thank you for providing me with the background information, access to the ponds, installation and use of the sampling equipment. I would also like to thank the Canadian Water Network for providing funding for this study under the Innovative Stormwater Management Study.

A huge thank you goes to Derrick Hambly and Scott Easton who helped me with field and lab work. I could not have done this without your help. I would also like to acknowledge all the wonderful people I met while at Waterloo. Thank you for all the advice, encouragement and laughter over the last couple of years.

I would like to thank my family for their unwavering support during my studies at Waterloo. I am truly blessed to have such a fantastic family. Lastly, I would like to thank Brett for his love and support. You might know more about phosphorus cycling than you know what to do with but I thank you for always being there for me.

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Chapter 1: Introduction

1.1. Problem Statement

Stormwater runoff in urban areas represents a major pathway for pollutant transfer to downstream ecosystems (Mallin et al., 2002). The combined impacts of increased flows, erosion and pollutant concentrations in urban stormwater have degraded receiving waters in Ontario thus reducing the use of water resources for downstream environments while incurring significant remediation costs to society (Brabec, 2009; Paul and Meyer, 2001). To mitigate some of the adverse effects associated with urban development, a wide range of structural and vegetative Best Management Practices have been implemented in Ontario through the subwatershed and site management planning process (OMOE, 2003; TRCA, 2009). This planning process is used to implement low impact development scenarios that have stormwater management (SWM) practices designed to address stormwater quality, quantity and erosion concerns as an assumed part of the development form. The implementation procedure determines the management options to be used and the level of control (lot level, conveyance level, end of pipe) and then to test the performance of options on the key physical and biological systems of the watershed (OMEE, 1994). In addition to structural and vegetative best management practices, non-structural practices such as policies, awareness and public education can help improve stormwater quality (Marsalek and Chocat, 2002). This can be accomplished by altering the development communities business practices, as well as the public's perception of SWM and actions negatively affecting runoff wate quality (e.g. stopping excessive fertilizer use).

There is increasing emphasis and reliance on the performance of SWM controls to meet targets for environmental protection that are set through subwatershed strategies and their use in planning for future land use and resource management in Ontario. It is becoming increasingly important to predict the pollutant removal efficiency of various SWM practices and thereby ensure that environmental targets are met. Given the increasing reliance on these SWM practices by regulatory agencies (municipalities, Conservation Authorities, provincial agencies) to meet the environmental protection and enhancement targets, it is essential to develop a long-term database for use in measuring pollutant removal efficiency of a range of SWM practices.

SWM ponds are one of many end of pipe management practices commonly used in urban areas to improve water quality and reduce flooding. Initially introduced in the 1960s, SWM ponds were constructed to control flooding and erosion. However, due to insufficient storage volumes, accumulation and consequently resuspension of sediment and release of pollutants, early designs of SWM ponds were unable to significantly reduce the impacts of urban runoff on water quality because they were designed primarily for quantity control (Jones and Jones, 1984). More recently, improved pond designs (e.g. hybrid extended detention pond) have been proposed in the Stormwater Management Planning and Design Manual (OMOE, 2003) to meet water quality, flooding and erosion concerns. While these facilities have the potential to remove pollutants, field research focusing on treatment performance is lacking in Ontario. One of the main reasons for this discrepancy is that most of the initial experimental design and implementation research on SWM ponds was conducted either in the United States or Europe (Schueler, 1987) where environmental conditions can vary significantly from that of Ontario. The percentage of pollutant retained in a pond is governed by the treatment facility used and the fraction of runoff volume effectively treated (OMOE, 2003). Thus differences between observed and expected SWM pond performance are caused by variability in climate, pond design and pollutant characteristics (Van Buren et al., 1997). This is particularly significant for managing phosphorus (P) in urban streams because despite the promotion and adoption of SWM ponds in Ontario, **relatively little is known about the delivery, storage, export and internal cycling of sediment associated pollutants such as phosphorus in SWM ponds**. Such data on treatment effectiveness (performance) of SWM ponds are required to ensure targets are being met and to make recommendations for system adjustment if targets are not being met.

1.2. Literature Review

1.2.1. Effect of Urbanization on Water Quality

Urbanization can have a negative impact on the natural environment by altering the hydrologic cycle (Leopold, 1968) and decreasing the quality and quantity of both surface waters (Rast and Thornton, 1996; Booth and Jackson, 1997) and groundwater (Bradford and Gharabaghi, 2004). The percentage of impervious surface coverage (ISC) increases as urban areas expand (Arnold and Gibbons, 1996) which reduces infiltration capacity (Schueler, 1987; Stephens et al., 2002; Xiao et al., 2007) and increases the volume and velocity of runoff (OMOE, 2003; Bradford and Gharabaghi, 2004). Of particular interest is the measure of effective impervious area, which is defined as the area directly connected to a water course (Booth and Jackson, 1997).

Generally, in a natural environment the percentage of runoff is typically less than infiltration, approximately 10% and 50%, respectively (Arnold and Gibbons, 1996; Stephens et al., 2002). However, a residential area with 35-50% ISC and an industrial area with 75-100% ISC can increase runoff to approximately 30% and 55%, respectively

(Arnold and Gibbons, 1996). A decrease in infiltration of 35% and 15% (Arnold and Gibbons, 1996) can occur in residential and industrial areas (Stephens et al., 2002). Storm hydrographs in urban areas have a shorter lag time and a higher peak flow compared to pre-urbanization hydrographs. Untreated stormwater (SW) contains a range of particulate and dissolved materials (i.e. sediment, heavy metals, fecal matter and nutrients) which can be transported into receiving waters (Mallin et al., 2002).

A significant amount of research has focused on the impact of an increasing supply and transfer of sediment and associated P to water bodies from storm sewer systems (c.f. Schueler, 1987; Rast and Thornton, 1996). Stormwater can increase the sediment load of a river by incising stream banks (Stephens et al., 2002; OMOE, 2003). Embeddedness can occur (OMOE, 2003) which can increase turbidity, decrease light penetration, smother benthic invertebrates, and irritate fish gills (Schueler, 1987; Stephens et al., 2002). Because fine-grained sediment (<63 µm) has a higher surface area and cation exchange capacity compared to larger sand and gravel materials, it has a greater potential to carry pollutants into/within a water body (Stone and Mudroch, 1989; Jin et al., 2005).

In 1996, non-point source pollutants from human activity, such as inorganic and organic fertilizers, phosphate soaps and animal waste contributed an estimated 12,000 tonnes of P to Canadian waters (Chambers et al., 2001). Industrial wastewater and municipal sewage has been estimated to supply 5,600 tonnes P and 2,000 tonnes P to Canadian waters, respectively. The majority of Canadian inland waters are P-limited (Chambers et al., 2001) and an excess of P can cause eutrophication in lakes and reservoirs resulting in toxic algal blooms, foul odours and decreased aesthetic appeal for humans (Schueler, 1987). Decaying algae decrease dissolved oxygen (DO) concentrations in the hypolimnion which can cause massive fish kills due to asphyxiation (Schueler, 1987).

1.2.2. Use of Stormwater Management Ponds

In the 1970s, best management practices (BMPs) to help mitigate the effects of urban SW, including eutrophication, were introduced (Stephens et al., 2002; Bradford and Gharabaghi, 2004). Detention ponds were initially designed to control flooding downstream (Stephens et al., 2002; Bradford and Gharabaghi, 2004). However, in the early 1990s, degraded water quality became an issue and pond designs were enhanced to mitigate water quality concerns (Stephens et al., 2002; TRCA, 2009) such as wet detention ponds and constructed wetlands. Currently, watershed-based approaches, including low impact development, have been introduced (Stephens et al., 2002) as an attempt to keep as much rainwater on site as possible instead of conveying it to another location, thus reducing the number of end of pipe SWM facilities (TRCA, 2009).

To help mitigate the effects of urban stormwater on aquatic systems the government of Ontario published a document titled “Stormwater Management Planning and Design Manual” in 1994. The manual was revised in 2003 to provide new information on the design of SW BMPs. The manual promotes a treatment train approach, which includes the use of several SWM controls implemented at the lot level, conveyance and end-of-pipe (OMOE, 2003). End-of-pipe controls help to control flooding, erosion and increase water quality (OMOE, 2003). There are six end-of-pipe controls, of which wet ponds and constructed wetlands have the greatest potential for water quantity and erosion control and water quality improvement (OMOE, 2003).

Wet ponds and constructed wetlands decrease flow velocities and peak flow by retaining runoff and releasing it at slower rates into the receiving water body (OMOE, 2003). Wet ponds have a permanent pool of water at the pond inflow (Borden et al., 1998, Bavor et al., 2001) and fringe emergent vegetation for enhanced sedimentation and nutrient removal (Persson et al., 1999). Constructed wetlands are shallower than wet ponds and have a higher density of emergent vegetation (Persson et al., 1999). Emergent vegetation incorporated into a constructed wetland provides additional treatment of SW through biological uptake (Wong et al., 1998; Kröger et al., 2007). Schueler (1987) stated that wet ponds can increase property values, provide recreation and landscape amenities and create habitat for wildlife within an urban environment. However, wet ponds can also be a safety hazard, cause degradation of upstream or downstream aquatic habitats, cause occasional odor, algae and debris problems (Schueler, 1987) and potentially have a negative effect on wildlife because of contaminant accumulation in the pond (Bishop et al., 2000). Hybrid extended detention ponds, which include both a wet pond and constructed wetland are designed to enhance SW quality treatment. Research measuring the performance and cycling of P in hybrid extended detention ponds has been given little attention within peer-reviewed SW literature especially in Southern Ontario. Wet ponds and constructed wetlands are designed to meet a certain level of protection (enhanced, normal or basic) depending on the characteristics of receiving streams and their sensitivity to pollutant loading (OMOE, 2003). The terms enhanced, normal or basic protection are defined as meeting specified volumetric criteria to ensure 80%, 70% or 60% TSS removal, respectively (OMOE, 2003). The Provincial Water Quality Objectives limit for phosphorus (P) is 30 µg P L^{-1} in surface water because excessive plant and algal growth can occur at concentrations greater than this limit (OMEE, 1994). The Canadian Council of Ministers of the Environment (2002) has stated that an increase of 25 mg L^{-1} from background surface water concentrations is unfavourable to stream health. SWM ponds are not required to meet these thresholds. However, evaluating inflow and outflow concentrations will indicate whether effluent water quality is meeting the 2002 CCME and 1994 PWQO guidelines.

1.2.3. Design of Wet Ponds and Constructed Wetlands

Detention ponds are designed to optimize pollutant treatment by ensuring the inflow is well distributed throughout the pond and that the total pond volume is utilized (Persson et al, 1999). Shape, vegetation type, distribution and density as well as inlet/outlet location can affect the hydraulic characteristics of a wetland and wet pond (OMOE, 2003; Jenkins and Greenway, 2005). Hydraulic efficiency is defined as how well a system distributes incoming runoff (Wong and Somes, 1995; Holland et al., 2004) and is achieved when incoming runoff disperses into the pond and then travels as plug flow to the outlet. Plug flow is defined as a parcel of water that has uniform inflow velocity and travels through the pond at the same rate (Persson, 2000; Holland et al., 2004; Jenkins and Greenway, 2005).Water flowing through SWM ponds will travel through eddies and recirculate within the pond causing preferential flow paths and dead zones (Wong et al., 1999; Persson, 2000) and therefore, not achieve plug flow.

The detention time of a wet pond or constructed wetland is a measure of how long influent runoff is retained in the facility (Persson, 2000) and this determines how well runoff is treated (Wong et al., 1999). Detention time must ensure adequate settling of SS (Bavor et al., 2001) but must also make sure incoming runoff does not become stagnant and allow the growth of algae or stratification of bottom waters leading to anoxic conditions (Burge and Breen, 2006). A minimum detention time of 24 hours has been suggested by the OMOE (2003) to provide adequate time for SS settling.

Design factors can affect plug flow and detention time such as length:width (L:W) ratio, baffles, location of inlet(s) and outlet and vegetation characteristics. A high L:W ratio increases the mean detention time for ponds of equal volume and decreases short circuiting within a pond because influent runoff has a longer flow path to travel to the outlet (Matthews et al., 1997; Persson, 2000, Gharabaghi et al., 2006). Multiple inlets, the addition of an island and subsurface berms help increase the detention time (Persson, 2000). Baffles are used to increase the flow path and are commonly used to retrofit existing ponds that are not removing pollutants effectively (German et al., 2005). Persson (2000) considers L:W ratio of a wet pond the most important design variable, however, Jenkins and Greenway (2005) stated that vegetation characteristics have the greatest influence on hydraulic efficiency.

Changes in vegetation species, density and location cause changes to hydraulic characteristics, such as plug flow (Jenkins and Greenway, 2005). Detention time is increased in a constructed wetland or pond with vegetation because of increased drag between flowing water and surrounding vegetation (Reddy et al., 1999; Jenkins and Greenway, 2005). Vegetation around the edge of a pond causes differences in drag from the edge to the middle of the pond, and this can increase mixing and dispersion therefore, decreasing plug flow (Jenkins and Greenway, 2005). Vegetation spanning the width of a pond allows the same amount of drag on the entire body of water therefore, increasing hydraulic efficiency (Jenkins and Greenway, 2005).

Pond design, including optimum L:W ratios, inlet(s)/outlet location, baffles and vegetation, can increase hydraulic efficiency which allows water quality treatment processes to occur (Persson et al., 1999; Wong et al., 1999).

Transformations of P are governed by a range of chemical, physical and biological processes (Figure 1). Changes in pH, temperature and redox conditions can alter the rates and magnitudes of these processes. Each process helps control the amount of P adsorbed to sediment and dissolved in the water column. Soluble reactive phosphorus (SRP) is the dissolved fraction of total P (TP) and can pass through a 0.45 µm filter (Broberg and Persson, 1988). Particulate phosphorus (PP) is P that does not pass

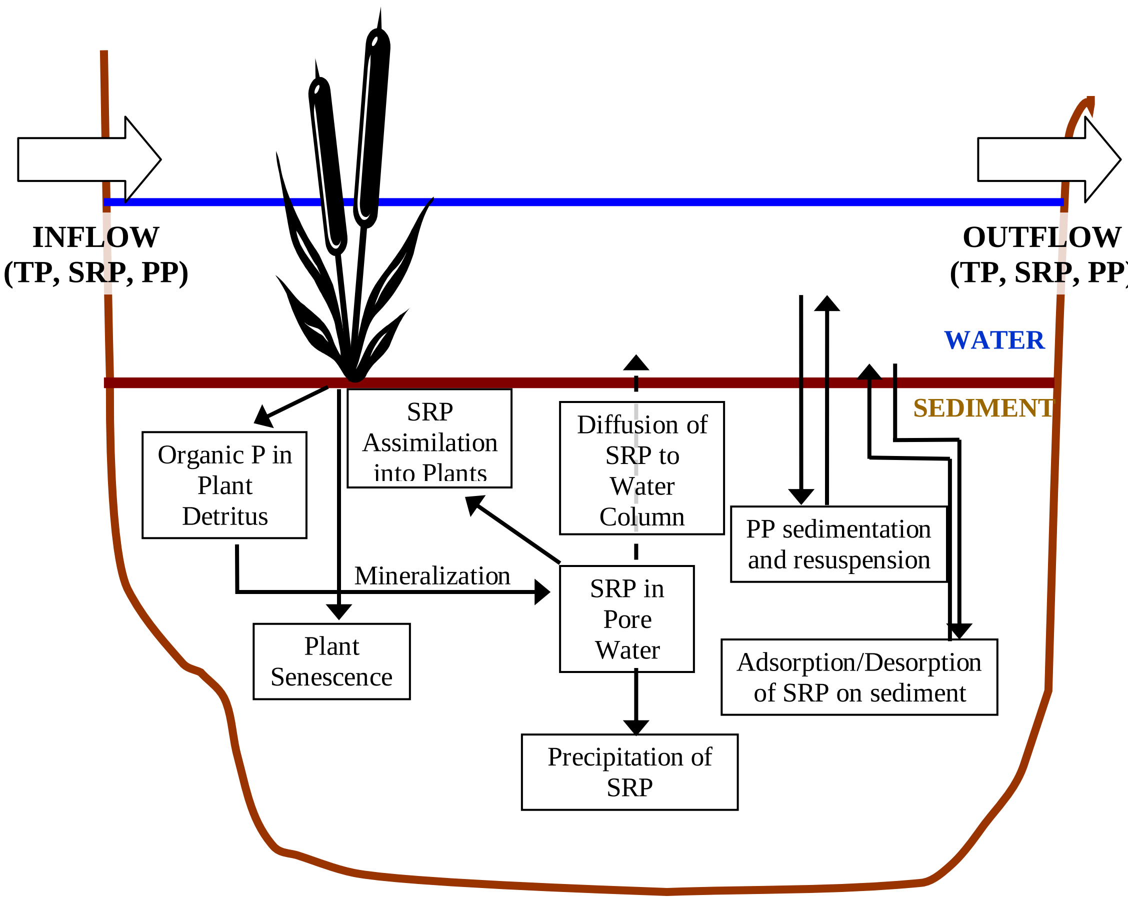


Figure 1. Conceptual diagram of P cycling in a SWM pond. Chemical, physical and biological processes can affect the solubility of SRP.

through the 0.45 µm filter and is adsorbed to sediment surfaces or organic matter (OM) (Boström et al., 1988; Broberg and Persson, 1988).

1.2.4. Biological, Physical and Chemical Processes in Aquatic Systems

1.2.4.1. Wetland Vegetation Growth and Senescence

In temperate latitudes, wetland plants have a lifecycle of four to six months for growth, senescence, and decomposition (Kadlec and Reddy, 2001). Assimilation and storage of P in plants will vary by season, latitude, species characteristics (e.g. growth rate and maximum biomass), litter decomposition rates, P leaching from plant detrital tissue and the amount of P translocated from above and below ground biomass (Cronk and Fennessy, 2001). Assimilation of P by wetland vegetation has been well documented in the literature. For example, floating plants, such as the water hyacinth (*Eichornia crassipes*) can assimilate 350-1125 kg P ha^{-1} yr^{-1} (Reddy and DeBusk, 1987), while, emergent plants (e.g. cattails (*Typha Latifolia*)) sequester 30-150 kg P ha^{-1} yr^{-1} (Gumbricht, 1993; Brix, 1994). In temperate environments, *Typha sp.* reaches its peak growth around mid-August then begins to senesce between September and November (Bayly and O'Neill, 1972; Bonneville et al., 2008).

Senescence occurs when plant tissues age and degrade causing nutrients to be translocated from aboveground biomass to belowground biomass (Bayly and O'Neill, 1972; Della Mea et al., 2007). Some P is translocated to plant rhizomes during senescence (Cronk and Fennessy, 2001) to ensure enough P is available for growth during the following spring (Reddy et al., 1999). In a laboratory study, Weng et al. (2006) found an increase in P concentrations in *Typha* sp. plant biomass above the root/rhizome system during the early stages of growth and after approximately 50-63 days, the upper plant biomass had a lower concentration of P compared to the lower plant biomass. The plant will eventually move from a dead standing stalk to litterfall, which will decompose over time (Kadlec, 1997).

Plants such as *Typha*, *Carex*, *Phalaris* and *Phragmites* have approximately a one year shoot life span (Bernard, 1999). This could have a negative effect on P removal from a SWM pond because of the short turnover rate from living to dead stalk. There have been varying estimates on the percentage of P released from plants during decomposition of fallen stalks. Nichols (1983) conducted a review of literature on natural wetlands, including *Typha sp*. and *Phragmites communis* Trin, and reported that 35-75% of a plants internal P is released. In contrast, Kadlec (2005) reported a release rate of 80-90%. If macrophytic plants such as *Typha sp.* store 20% of P permanently and assimilate 0.5-20 g P m^{-2} yr^{-1} (Kadlec, 1997), only 0.1-4 g P m^{-2} yr^{-1} is permanently removed from the system (Kadlec, 2005). Through decomposition processes, 0.4-16 g P m^{-2} yr^{-1} could become bioavailable from macrophytic plants. Decomposition of leaf litter can result in reducing conditions and potentially increase concentrations of SRP, ferrous iron (Fe^{2+}), total iron and total manganese in sediment porewater (Moore et al., 1994). Some detritus can return P in refractory residual form (Kadlec, 2005). The effect of vegetation growth and senescence on P removal for a SWM pond requires further study in temperate environments because of their short life cycles and potential increase of P to sediment porewater upon senescence and decomposition. An improved understanding of the total amount of P removed from the water column by *Typha Latifolia* plants is needed to optimize pond design for P removal.

1.2.4.2. Microbial Cycling of Phosphorus

Bacteria, algae (Brix, 1997) and periphyton (Reddy et al., 1999) are commonly found in wetlands and can influence P cycling through photosynthesis/respiration (Carlton and Wetzel, 1988), assimilation (Gächter and Meyer, 1993) and mineralization (Boström et al., 1988). Mineralization occurs when OM is transformed through microbial processes into inorganic matter and the degree of mineralization depends on the degradability of OM (Boström et al., 1988). The turnover rate of P between microbes and the water column is in the order of minutes to hours because microbes have a relatively short lifespan and will therefore release P back to the water column once the organisms decay (Vymazal, 2007).

Photosynthesis by periphyton can change O_2 concentrations and pH of both the water column and sediment-water interface and these changes can affect chemical or biological reactions that control P uptake and release (Carlton and Wetzel, 1988). Carlton and Wetzel (1988) studied the mobilization of P during light and dark periods with periphyton-sediment samples from Lawrence Lake in southwest Lower Michigan. They found that during light periods, periphyton were photosynthesizing, releasing O_2 and enhancing the immobilization of P. During dark periods however, respiration occurred and reducing conditions prevailed. These conditions can cause the reduction of ferric iron (Fe^{3+}) to Fe^{2+} and result in the mobilization of P (Gächter and Meyer, 1993).

At high concentrations of P in the water column microorganisms can assimilate greater amounts of P and release it when P is limiting in the environment (Reddy et al., 1999). When microbes die, decomposition of their tissues can transform organic P to SRP or particulate refractory organic P (Gächter and Meyer, 1993). SRP is considered to be the most bioavailable form of P to microbes and vegetation (Boströ m et al., 1988; Reddy et al., 1999). Particulate refractory organic P can be in microbial tissues, or adsorbed onto sediment surfaces which are not easily decomposed and are usually stored in sediments longer than any other P complex (Gächter and Meyer, 1993).

Microbes are important for P cycling in wetlands because they can attach to plants and contribute to biofilm growth (Reddy et al., 1999). Microbes have the ability to mobilize P through photosynthesis, respiration and decomposition. Depending on the life cycle and diurnal period, microbes can be either a source or sink for P within a wetland (Vymazal, 2007). However, net benefit to microbial cycling in SWM ponds is unknown and further investigation into this is needed.

1.2.4.3. Sedimentation and Resuspension

Ontario Ministry of the Environment (2003) promotes sedimentation as the primary treatment process for SW because many pollutants are sediment-bound (Vase and Chiew, 2004). Fine sediment (<63 µm) can transport a larger pollutant load than coarser sediment (Hesse, 1973; Stone and Mudroch, 1989; Krishnappan and Marsalek, 2002; Vaze and Chiew, 2004) because these particles have a higher surface area and cation exchange capacity.

According to Stokes Law, sediment with a higher density or larger mean diameter will settle out of suspension faster than smaller particles. Flocculation of fine sediment (<63 µm) is controlled by sediment geochemistry, particle size/density, water column turbulence (De Boer and Stone, 1999), pH, SS concentration and organic carbon content (Droppo and Ongley, 1994). Floc formation will increase sedimentation because of the relative increase in particle mass (Stephan et al., 2005). However, Krishnappan et al. (1999) developed a relationship between floc size, density and settling velocity from sediment collected in a SWM pond in Kingston, ON. They found that particles between 5 and 15 µm in diameter have a greater fall velocity than smaller particles (<5 µm) and larger less dense flocs. These larger flocs can be formed by microbiological processes which bind particles together with extra cellular polymeric substances (Krishnappan et al., 1999). A bed shear stress (BSS) of 0.056 N m^{-2} will have low deposition rates because there is a low incidence of fine particulate matter colliding and flocculating with other particles, however, a high BSS around 0.213 N m^{-2} will cause flocculated particles to break apart (Krishnappan and Marsalek, 2002).

Properly designed SWM ponds can decrease SS concentrations in SW. However, large portions of SS that settle out of suspension are >63 µm and the majority of fine grained particles (<63 µm) often remain in suspension (Greb and Bannerman, 1997). Incorporating vegetation into a SWM pond can reduce flow velocities due to friction between the water column, sediment and vegetation (Reddy et al., 1999) therefore increasing sedimentation of fine grained material (Oberts and Osgood, 1991; Stephan et al., 2005).

Waves and bioturbation can resuspend particles (Boström, 1988; Gumbricht, 1993; Fennessey et al., 1994; Granéli, 1999; Reddy et al., 1999; Søndergaard et al., 2003) if the particle settling velocity is exceeded by the vertical flow velocity (Bagnold, 1966). The critical shear stresses required to erode and deposit coarse grained material are equal, however, fine grained particles do not have equal critical shear stresses because of their cohesive nature (Krishnappan, 2007). Based on flume experiments, consolidated sediment from the Kingston pond has a critical shear stress of 0.21 N m^{-2} for erosion (Krishnappan and Marsalek, 2002). Providing quiescent waters to promote sedimentation of fine grained sediment (<63 µm) could also aid in the consolidation process at the sediment-water interface which may decrease erosion of settled material.

1.2.4.4. Chemical Cycling of Phosphorus

1.2.4.4.1. Phosphorus Speciation

Studies examining chemical cycling of P in aquatic systems have predominantly focused on aquatic environments (c.f. Boström et al., 1988, Broberg and Persson, 1988; Holtan et al., 1988; Pettersson and Istvanovics, 1988; Cooke et al., 1993; Stone and English, 1993; White and Stone, 1996; Hongve, 1997; Jin et al., 2005; Spears et al., 2007). There is a lack of scientific research investigating P cycling in SWM ponds. It is important to understand P cycling in SWM ponds because they can receive higher loads of P and could have different sediment P mobility potentials compared to natural aquatic systems (Peng et al., 2007). Phosphorus adsorption and desorption from sediment is influenced by PP speciation, sediment geochemistry (Stone and English, 1993) and water quality parameters including pH, DO and temperature at the sediment water interface (Boström et al., 1988).

Particulate phosphorus can be separated into three operationally defined chemical fractions using the modified Psenner sequential extraction scheme: non-apatite inorganic P (NAIP), apatite inorganic P (AP) and organic P (OP) (Pettersson and Istvanovics, 1988). Non-apatite inorganic phosphorus consists of P extracts from 1.0M NH_4Cl, 0.1M $NaHCO_3 \bullet Na_2S_2O_4$ and 1.0M NaOH. This PP form is a combination of loosely adsorbed P, reductant SRP and reactive P adsorbed to metal oxides such as Fe and Al (Psenner et al., 1988; Stone and English, 1993). Changes in pH and redox at the sediment-water interface can release NAIP to the water column (Boström, 1988). Phosphorus from 0.5M

HCl and 1.0M NaOH extractions at 85°C yields P forms in the AIP and OP chemical fractions, respectively (Psenner et al., 1988; Stone and English, 1993). Apatite inorganic phosphorus encompasses carbonate bound P, P desorbed after oxide dissolution and apatite P (Stone and English, 1993). Organic phosphorus is composed of refractory organic P (Stone and English, 1993). Large fluctuations in redox and pH have little effect on the solubility of AIP (Stumm and Morgan, 1996) and OP which are less bioavailable than NAIP. Quantifying PP forms and spatial distribution of PP in lake sediment will provide an understanding of the potential internal loading of P to the water column.

1.2.4.4.2. Phosphate Adsorption and Desorption

Sorption reactions (adsorption, surface precipitation, coprecipitation and diffusion) influence the release of P from particulate matter (Scheidegger and Sparks, 1996). Adsorption refers to the surface attraction of one substance or phase to another (Hillel, 1998). If large quantities of P sorb onto the surface of a particle, the formation of a surface precipitate could occur (Scheidegger and Sparks, 1996). Accordingly, a coprecipitate forms when dissolved ions from a particle and water bind together (Scheidegger and Sparks, 1996). Chemical reactions (e.g. metal complexation reactions and redox reactions) remove P from solution and promote sorption to the surface of a particle. The force holding the phosphate ion to the solid surface can be a weak electrostatic outer-sphere complex, van der Waals forces or a stronger inner sphere complex which requires a ligand exchange, hydrogen bonding or covalent bonding (Scheidegger and Sparks, 1996; Hillel, 1998). Phosphorus can be released from Fe, Al and Ca complexes when the solubility product is exceeded which is dependent on pH and redox values (Stumm and Morgan, 1996).

Research has focused on P adsorption with Fe^{3+} and Al^{3+} because of their attraction to P and relative abundance in lake sediment (Lijklema, 1980; Hupfer et al., 1995; Hongve, 1997). $Fe(OH)_3$ strongly binds to P (Cooke et al., 1993) forming $FePO_{4(s)}$ and OH^- (Fox, 1993). In well oxygenated water bodies, a zone of $Fe(OH)_3$ can form at the sediment-water interface which will increase P adsorption (Cooke et al., 1993). Changes in pH and/or redox can reduce Fe^{3+} to Fe^{2+} and release P to the water column (Lijklema, 1980; Lake et al., 2007; Peng et al., 2007). Fe-bound P increases in concentration at neutral pH values in aquatic systems, whereas, SRP is desorbed from Fe^{3+} at pH <6 therefore increasing SRP concentrations in the water column (Peng et al., 2007). The oxidized/reduced boundary for Fe and P is 200 mV at a pH of 7 (Wetzel, 1983; Boström et al., 1988) and larger reducing conditions will result in more SRP desorption from Fe^{3+} (Peng et al., 2007).

Insoluble aluminum hydroxide ($Al(OH)_3$) is present at neutral pH in aquatic systems and it is a colloidal, amorphous floc, which strongly binds to P ions (Cooke et al., 1993) resulting in $AlPO_{4(s)}$ and OH^-. Low redox values do not affect the solubility of Al-bound P (Lake et al., 2007), however, increasing pH will increase its solubility (Cooke et al., 1993). Rydin and Welch (1998) treated Fe rich surface sediment from two eutrophic lakes with a pH of 6 and anoxic conditions in Sweden with $Al_2(SO_4)_3$ and found that sediment treated with $Al_2(SO_4)_3$ released less P to the water column compared to untreated Fe rich sediment. Increasing Al^{3+} concentrations in a eutrophic lake with anoxic conditions can help stabilize P release to the water column (Rydin and Welch, 1998).

Photosynthesis can produce calcium hydroxide and calcium carbonate ($CaCO_3$) in lake waters (Boström et al., 1988; Cooke et al., 1993; House et al., 1995) which have a high affinity for P. At neutral to basic pH levels, Ca^{3+} and P coprecipitate out of solution (Boström et al., 1988; Stumm and Morgan, 1996) and form hydroxyapatite (Cooke et al., 1993; Stumm and Morgan, 1996). Phosphorus is absorbed into the crystal lattice of $CaCO_3$ (House et al., 1995) rendering it unavailable for desorption unless soil porewater or the water column become acidic and the solubility of hydroxyapatite and calcite increases (Cooke et al., 1993). Apatite is not considered bioavailable because minor fluctuations in pH and redox do not affect its solubility (Stumm and Morgan, 1996). It is essential to understand sediment geochemistry in a SWM pond because changes in pH, temperature, redox and DO can affect the solubility of Fe, Al or Ca and P. Stormwater managers can use sediment geochemistry results and water quality parameters to decide the potential for sedimentary P release to a SWM pond.

The presence of OM can increase or decrease SRP adsorption on inorganic particles (Stuanes, 1982). Phosphorus adsorption sites can be obstructed by organic anions and the level of hindrance is dependent on how strong Fe/Al-bound P complexes are compared to Fe/Al-bound organic complexes (Stuanes, 1982). This will effectively reduce the number of sorption sites for P therefore decreasing its removal from the water column. Phosphorus can also be released from OM during mineralization (Boström et al., 1988; Gächter and Meyer, 1993). However, during the mineralization process refractory organic P is also formed and is considered to be a permanently removed from the water column (Gächter and Meyer, 1993; Søndergaard et al., 2003)

1.2.4.4.3. Phosphorus Cycling at the Sediment Water Interface

Phosphorus cycling is controlled by sediment-water interactions in shallow water bodies (van Raaphorst and Kloosterhuis, 1994). Water column concentrations of P are usually highly dependent on sediment P reserves, which can have P concentrations 100 times higher than in the overlying water column (Søndergaard et al., 2003). It is important to understand how P is exchanged at the sediment-water interface and how the pool of porewater P changes seasonally in shallow lakes because sedimentary P can be a major supply to water column P, even after reductions of external P sources (Søndergaard et al., 2003).

In well oxygenated waters, a microzone of oxygenated sediment at the sediment-water interface can occur and this allows Fe, Al and P to adsorb onto the surface of sediment (Wetzel, 1983). Iron, P, and Al can be released from sediment surfaces and diffuse upward through a concentration gradient to the sediment-water interface where Fe-bound P and Al-bound P precipitate because of the aerobic microzone (Lijklema and Hieltjes, 1982; Holtan et al., 1988). Ca-bound P concentrations are not affected by decreasing redox conditions and therefore, are more stable in the sediment profile (Stumm and Morgan, 1996). When sorption sites on sediment are saturated, P can diffuse across the microzone and enter the water column (Søndergaard et al., 2003). During the summer, water temperatures rise and the oxygenated microzone can be reduced to anoxic conditions where SRP will diffuse through sediment to the water column (Lijklema, 1980). Bioturbation and ebullition can increase the rate and magnitude of SRP entering the water column (Boström et al., 1988; Jansson et al., 1988; Reddy et al., 1999) by disturbing the sediment-water interface. Quantifying the spatial distribution and the amount of SRP in sediment porewater will provide an indication of the potential increase of SRP in the water column to be expected if reducing conditions dominate the sediment-water interface or if sediment is resuspended and the microzone of oxidized sediment is disturbed.

1.2.4.4.4. Sediment Buffering Capacity

Froelich (1988) introduced the concept of the phosphate buffering mechanism (PBM) which describes the capacity of sediment to buffer SRP in the water column. This process results in almost constant concentrations of SRP. The amount of SRP adsorbed or desorbed from sediment depends upon environmental conditions such as pH, redox and sediment geochemistry (Froelich, 1988; McGechan and Lewis, 2002). The PBM incorporates a two step process for P adsorption or release (Froelich, 1988). The first step (fast reaction) allows $H_2PO_4^-$ to be adsorbed onto a charged sediment surface, while the second step (slow reaction) incorporates P into the crystal lattice of sediment through a diffusion gradient that was established after the first step (Barrow, 1983; Froelich, 1988, McGechan and Lewis, 2002, Wang et al., 2006). Fast and slow reactions can occur for desorption as well (Froelich, 1988), however, the rate and magnitude of reversibility depends on the amount of SRP adsorbed to the sediment surface and integrated into the crystal lattice (Barrow, 1983). Desorption of SRP is very slow if the majority of P has diffused into the crystal lattice compared to being adsorbed to the sediment surface (Barrow, 1983).

Batch experiments using a range of SRP concentrations in a known volume of water are used to quantify the change in SRP from adsorption and desorption processes between sediment and the water sample (Froelich, 1988; Stone and Mudroch, 1989; Schulz and Herzog, 2004; Wang et al., 2006). An isotherm is produced which relates the amount of P adsorbed to the surface of particles compared to the initial concentration of SRP in the aqueous solution (Giles, 1970). At high and low concentrations of P, SRP will be adsorbed and released from sediment, respectively (Froelich, 1988). In aquatic systems there is a tendency to reach equilibrium between SRP and PP (Froelich, 1988). At true equilibrium or steady state, called the equilibrium phosphate concentration (EPC), sediment is no longer able to adsorb or desorb SRP.

Several studies have reported values of EPC (Table 1) in freshwater and marine environments. The data show that EPC values range from 0.015-579.15 mg L^{-1} (Table 1). High SRP concentrations in the water column can result in higher EPC values (Wang et al., 1996; Haggard et al., 2004). The buffering capacity of P for surface sediment from Spring and Sager/Flint Creeks are lower than Mud and Osage Creeks in Arkansas, USA because they have higher EPC values. Haggard et al. (2004) stated that the SRP concentration in the downstream wastewater effluent was higher in Spring and Sager/Flint Creeks. Reducing SRP concentrations in the water column could cause a shift in EPC values and release SRP currently adsorbed to surface sediment (Haggard et al., 2004). Liu et al. (2009) studied the sorption characteristics in a multipond system in the Liuchahe watershed in Chaohu Lake, China. They found that sediment with a high EPC was a result of high P loadings to the pond. EPC values should be combined with sediment geochemistry, grain size, pH and redox data to have a complete understanding of sediment buffering in surface sediment.

Table 1. Summary of EPC values found in the literature.

Author	Location	Type of Sediment / Grain Size	EPC (mg L^{-1})
Stone et al., 1995	Big Creek and Big Otter Creek, Ontario	Freshwater sediment; 42 - 63 µm 31 - 42 µm 19 - 31 µm 12 - 19 µm 8 - 12 µm <8 µm	**0.015 – 0.080[1]**
McDowell & Sharpley 2001	Pennsylvania, USA	River bed sediment	**0.043**
		River bank sediment	**0.02**
McDowell & Sharpley 2003[2]	Pasture & forest sediment with addition of poultry or dairy manure, Pennsylvania, USA	Forest with poultry	**0.02**
		Forest with dairy	**0.022**
		Pasture with poultry	**0.027**
		Pasture with dairy	**0.024**
Haggard et al. 2004[3]	Mud Creek	Sediment downstream of a WWTP	**0.11**
	Osage Creek		**0.13**
	Spring Creek		**5.29**
	Sager/Flint Creek, Arkansas, USA		**0.88**
Schulz & Herzog 2004[4]	The lower River Spree, Northern Germany	Eutrophic river sediment	**W, 8°C = 0.09**
			W, 22°C = 0.072
			S, 8°C = 0.028
			S, 22°C = 0.480

Table 1 cont. Summary of EPC values found in the literature.

Wang et al. 2006	East Taihu Lake	360 – 480 mesh	**48.2**
		240 – 360 mesh	**41.3**
		60 – 240 mesh	**34.2**
		18 – 60 mesh	**30.1**
	Yue Lake	360 – 480 mesh	**634.1**
		240 – 360 mesh	**588.5**
		60 – 240 mesh	**562.7**
		18 – 60 mesh	**531.3**
	Wuli Lake, China	360 – 480 mesh	**214.8**
		240 – 360 mesh	**194.4**
		60 – 240 mesh	**212.3**
		18 – 60 mesh	**87.9**
Liu et al. 2009	Chaohu Lake, China	Hill Pond	**0.007 ± 0.002**
		Non-irrigation Pond	**0.008 ± 0.003**
		Rice Pond	**0.010 ± 0.003**
		River Pond	**0.011 ± 0.005**
		Village Pond	**0.023 ± 0.005**

[1]Range of EPC values for Big Creek and Big Otter Creek, Ontario for 6 sediment size classes
[2]Average values from 3 forest sites and 4 pasture sites
[3]Values reported are an average from 3 sampling locations downstream of a wastewater treatment plan (WWTP) within each creek before treatment of Alum
[4]W=winter and S=summer

1.2.5. Stormwater Management Pond Trap Efficiency (TE)

The TE of a SWM pond is estimated using a mass balance approach, where concentrations of TP, SRP and TSS and discharge are measured at the inlet(s) and outlet of each pond. The mass retained is indicative of how well the pond can treat the pollutant(s) of concern. A positive value indicates the pond is acting as a sink, however, a negative values indicates the pond is acting as a source for the pollutant of concern (Davis and McCuen, 2005). Table 2 summarizes the TE for TP, SRP and/or TSS from 11 studies previously reported in the literature. These studies show that the range of TP, SRP and TSS at the inlet are 51-1750 µg L^{-1}, 45-1170 µg L^{-1} and 3.6-212 mg L^{-1}, respectively. The large range between inlet concentrations is related to the pollutant source and pond design. For example, Yeh and Wu (2009) reported the inflow for a hybrid pond in Taiwan was sewage treated by secondary biological processes, however, Van Buren et al. (1997) measured inflow concentrations of TP, SRP and TSS from a commercial parking lot and online creek. Ponds treating sewage have much higher concentrations of TP, SRP, and TSS compared to those treating runoff from commercial parking lots.

In the literature, TE ranged from -35 to 77%, -77 to 87% and -37 to 96% for TP, SRP and TSS, respectively. These data show that some detention facilities perform well, while, others act as a source of TP, SRP and TSS. The hybrid pond studied by Oberts and Osgood (1991) has a high TE for TSS and TP, however, this is not the case for SRP which has a TE of 48%. This could partly result from vegetation senescence which causes a release of SRP to the water column. Gharabaghi et al. (2006) reported over 90% TSS reduction for the Greensborough and Ballymore ponds in Ontario, however, these ponds were designed to treat runoff from construction sites and the outflow concentrations

Table 2. Summary of TSS, TP and SRP inlet and outlet concentrations and corresponding trap efficiencies.

Author	Location	Pond Name	Facility Type[1]	Parameter	Concentration[2] Inlet	Concentration[2] Outlet	Trap Efficiency[3] (%)
Gharabaghi et al., 2006	Markham, ON	Greensborough	WP	TSS	nd[4]	nd	**>90**
		Ballymore	WP	TSS	nd	nd	**>90**
Van Buren et al. 1997	Kingston, ON	-	WP/DP	TSS	**212**	**73**	**65**
				TP	**216**	**84**	**61**
				SRP	**89**	**37**	**58**
Comings et al., 2000	Washington	Pond A	WP	TSS	**22.8**	**8.9**	**61**
				TP	**95**	**77**	**19**
				SRP	**15**	**14**	**7**
		Pond C	WP	TSS	**16.2**	**2.9**	**82**
				TP	**87**	**45**	**48**
				SRP	**26**	**10**	**62**
Oberts and Osgood 1991	Minnesota	McCarrons Wetland	HP	TSS	nd	nd	**96**
				TP	nd	nd	**77**
				SRP	nd	nd	**48**
Borden et al. 1998	North Carolina	Piedmont Pond	WP	TSS	**61**	**49**	**20**
				TP	**160**	**100**	**38**
				SRP	**33**	**27**	**18**
		Davis Pond	WP	TSS	**97**	**39**	**60**
				TP	**360**	**210**	**42**
				SRP	**220**	**100**	**55**

[1]CW = constructed wetland; WP = wet pond; HP = hybrid pond and DP = dry pond

[2]Concentration Units: TSS = mgL^{-1}; TP/SRP - μgL^{-1}

[3]Trap efficiency = $1-(C_{out}/C_{in}) * 100\%$

[4]nd = data not available

Table 2 cont. Summary of TSS, TP and SRP inlet and outlet concentrations and corresponding trap efficiencies.

Author	Location	Pond Name	Facility Type[1]	Parameter	Concentration[2] Inlet	Concentration[2] Outlet	Trap Efficiency[3] (%)
Mallin et al. 2002	North Carolina	Ann McCrary	WP	TSS	**10.5**	**3.7**	**65**
				TP	**61**	**47**	**23**
				SRP	**28**	**26**	**7**
		Silver Stream	WP	TSS	**4.3**	**5.9**	**-37**
				TP	**141**	**60**	**57**
				SRP	**100**	**23**	**77**
		Echo Farms Country Club	WP	TSS	**3.6**	**4.4**	**-22**
				TP	**51**	**69**	**-35**
				SRP	**25**	**44**	**-76**
Line et al. 2008	North Carolina	CMS	CW	TSS	nd	nd	**64**
				TP	nd	nd	**43**
		UNC	CW	TSS	nd	nd	**83**
				TP	nd	nd	**52**
Braskerud 2002	Norway	A	CW	TP	**170**	**100**	**41**
		C	CW	TP	**250**	**170**	**32**
		F	CW	TP	**220**	**170**	**21**
		G-1	CW	TP	**430**	**270**	**37**
		G-2	CW	TP	**430**	**240**	**44**
Istvanovics and Somlyódy 1999	Hungary	Upper Kis-Balaton (1991-1997)	WP	TSS	**44**	**29**	**34**
				TP	**290**	**224**	**23**
				SRP	**118**	**15**	**87**
Yeh and Wu 2009	Taiwan	-	HP	TSS	**12.42**	**1.65**	**87**
				TP	**1750**	**1170**	**33**
				SRP	**1070**	**960**	**10**
Birch et al. 2004	Australia	Mean of RA-RF	CW	TSS	nd	nd	**-4**
				TP	nd	nd	**12**

of TSS were still high. SWM ponds receiving high concentrations of TP, SRP, and TSS might have a high TE but the effluent concentration may still exceed PWQO (1994) and CCME (2002) criteria.

Assessing the TE of SWM ponds using a mass balance does not provide knowledge of mechanisms governing performance. For the purpose of this study, TE will describe only the mass balance calculation and performance will refer to how well a SWM pond reduces pollutant concentrations through physical, chemical and biological processes. Understanding the rates and magnitudes of the physical, chemical and biological processes that influence the TE of a pond is necessary to optimize its treatment capability. There is a lack of field research on P cycling in SWM ponds of varying designs in Southern Ontario, for example a wet pond versus a hybrid extended detention pond. Extrapolating P cycling studies from freshwater and marine environments might not be applicable because of higher loading of P in SWM ponds (Peng et al., 2007). Additional research is required on the performance and P cycling of conventional and hybrid extended detention ponds in Southern Ontario so they can be designed to address the internal physical, chemical and biological processes and increase effluent water quality.

1.3. Research Objectives

The specific objectives of this study are to:

1. Examine spatial and temporal variation of P (TP and SRP), total suspended solids (TSS) and grain size (GS) distribution at the inlet and outlet of two SWM ponds (conventional – Pond 45 and hybrid extended detention – Pond 33) in Waterloo, Ontario during baseflow (BF) and stormflow (SF) events.

2. Evaluate the role of sediment on P cycling within Pond 33 and Pond 45 by determining the vertical profile of SRP in pond sediments and the P buffering characteristics of pond sediments in relation to sediment geochemistry, mineralogy and grain size.
3. Examine the trap efficiency of P and sediment for Pond 33 and Pond 45.
4. Recommend design and maintenance improvements to enhance water quality of the SWM pond effluent.

The null hypotheses of this study are:

H_{O1}: No statistical difference in stormflow TP, SRP and TSS concentrations within and between Pond 33 and Pond 45

H_{O2}: No statistical difference between baseflow TP, SRP and TSS concentrations between Pond 33 and Pond 45.

Chapter 2: Methodology

2.1. Experimental Design

A mass balance approach was used to quantify the TE of two SWM pond designs (conventional – Pond 45 and hybrid extended detention – Pond 33) in Waterloo, Ontario as well as to determine the distribution, transport and fate of P in water and sediment compartments of each facility. Water quality parameters (pH, conductivity, DO and temperature) were measured during BF conditions from June 2, 2008 to October 31, 2008. Baseflow samples were collected twice a week from pond inlets and outlets, unless a storm event occurred during the week of sampling. A total of 30 BF samples were collected. Stormflow samples were also collected from June 28, 2008 to October 25, 2008. The sampling period followed peak growth and senescence of *Typha sp.* in Pond 33 in August (Bonneville et al., 2008) and September to November (Bayly and O'Neill, 1972) respectively. A total of ten storm events were sampled and six storm events will be discussed in detail herein. Treatment efficiency was calculated using the following equation

$$\text{TE} = \frac{Cin - Cout}{Cin} \text{ x } 100\% \qquad (1)$$

(Davis and McCuen, 2005). Where $C_{in/out}$ is the concentration entering and leaving a pond, respectively.

Internal cycling of P is determined through the use of porewater peepers (PWP), GS analysis, sediment geochemistry and sediment buffering capacity experiments. PWP were installed on August 13 and 14, 2008 and removed October 6, 7 and 14, 2008. Sediment samples were collected on October 21, 2008 and stored in a refrigerator at 4°C until required for analysis.

2.2. Study Site Description

Pond 33 and Pond 45 are located in the City of Waterloo. A map of the City of Waterloo indicating the relative locations of Pond 33 and Pond 45 to the University of Waterloo and Laurel Creek Reservoir is illustrated in Figure 2. Pond 33 and Pond 45 are located on the east and west sides of the City of Waterloo, respectively. Brief descriptions of the two ponds are found below.

Figure 2. Aerial map indicating the proximity of Pond 33 and Pond 45 to the University of Waterloo and Laurel Creek Reservoir.

2.2.1. Pond 33 – Eastbridge Subdivision

Pond 33 is located in the Eastbridge Subdivision which is part of the Genstar Development on the east side of Waterloo, Ontario (43°31'31.33" N, 80°29'57.04" W) (Figure 3). The nearest major intersection is located at University Avenue East and New Bedford Drive. Cumming Cockburn Limited (CCL) (1996; 2005) provided the design details discussed herein. The pond was built in 1995 and residential construction was completed in 2005. Pond 33 was designed to control erosion and water quantity, while increasing runoff water quality from the 79.5 ha residential community. The drainage area has an impervious level of 35%, with all runoff leading to Pond 33.

Pond 33 was designed as a hybrid extended detention pond, with two main cells, the sediment forebay and the extended detention cell, respectively. The sediment forebay has been designed to treat the first 4 $m^3 s^{-1}$, and flow over this threshold will be routed via a flow splitter located at the 2400 mm concrete inflow pipe to the overflow channel which bypasses the sediment forebay. The permanent pond depth in the forebay is 0.95 m, with a length and bottom width of 50 m and 17 m, respectively. Using the MOEE estimate of annual sediment loading (0.6 $m^3 ha^{-1}$), it is estimated that the total annual sediment loading to the pond will be 47.7 m^3 (CCL, 2005). Pond 33 has been designed with reference to the MOEE normal protection level of 70%

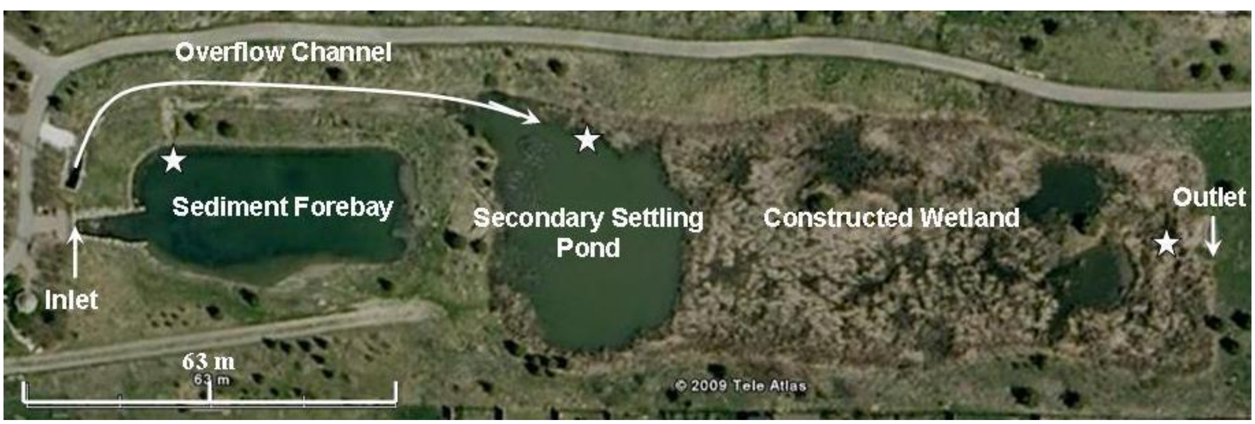


Figure 3. Aerial photograph of Pond 33. Note: The stars represent the approximate locations of PWP installation.

SS removal. The forebay is designed to store a minimum of 33.4 $m^3 yr^{-1}$. Dredging of the pond is required when 500 m^3 of sediment has accumulated in the forebay. Sediment removal in the forebay is recommended every 15 years.

The secondary cell can be divided into two compartments, the secondary settling pond and the extended detention wetland which has been heavily planted with *Typha Latifolia*. The permanent pool is 1685 m^3 and the extended detention portion increases the volume to 4242 m^3. The pond has a drawdown time of approximately 23.1 hours. Two sewers are connected to the 250 mm outlet; a 1500 mm pipe drains the treated water to the natural channel of Colonial Creek and the second 1800 mm sewer drains directly to the Grand River when outlet flows are in excess of 3.8 $m^3 s^{-1}$. Riprap has been installed to dissipate energy from the outlet discharge. Table 3 details the water quality objectives which Pond 33 has been designed to achieve.

Table 3. Water quality targets at the outlet of Pond 33.

Water Quality Parameter	**Water Quality Target**
Total Phosphorus	30 µg L^{-1} or lower
Suspended Solids	25 mg L^{-1} or lower
Dissolved Oxygen	6 mg L^{-1} or higher
Temperature	26°C or lower (June 1 - Aug 1) 29°C or lower (Aug 2 - Oct 31)

Source: CCL (1996)

2.2.2. Pond 45 – Columbia Forest Subdivision

Built in 1996, Pond 45 was designed to treat runoff from the Columbia Forest Subdivision in Waterloo, Ontario (43°27’48.22” N, 80°35’06.42” W) (Figure 4). The nearest major intersection is Erbsville Road and Columbia Forest Drive. The pond receives runoff from a 29.66 ha residential area with approximately 50% impervious surface cover. The first 20 mm of rainfall on all rooftops in the drainage area infiltrates into lot level soakaway pits to help maintain preurbanization infiltration rates. During the summer the first 20 mm of runoff is diverted into two infiltration galleries located in two parks to maximize the amount of infiltration within the residential area. The remaining runoff drains into Pond 45. The pond was designed with two

Figure 4. Aerial photograph of Pond 45. Note: The stars represent the approximate locations of PWP installation.

inlets located on the east and west side of the pond (Figure 5). Inlet 1 is a 1000 mm diameter pipe and discharges to an exposed pilot channel then to the pond. Inlet 2 is a submerged 900 mm pipe and is at a lower elevation than Inlet 1, and the permanent water level is at approximately half the diameter of the pipe. Outflow is controlled by a 150 mm diameter orifice which creates a detention time of 48 hours when the pond is at the 100 year storm event elevation.

Planning and Engineering Initiatives Ltd (2006) provided design details and drawings for Pond 45 described below. Two sediment forebays are located at each inlet; both are approximately 55 m long. Annual sediment loading to Pond 45 is approximately 17.79 m^3. The pond was designed to have a removal efficiency of 70% (Normal Protection) therefore 12.46 m^3 of sediment will be stored in the pond each year. The forebays require dredging once the depth is around 0.5 m, and which is approximately every 40 years.

The permanent pool area is 0.3 m deep with a storage volume of 6,925 m^3. There is also an extended detention area which adds another 1 m to the pond depth; totaling 12,450 m^3 of pond volume. The pond has been designed to decrease SS concentrations to below 25 mg L^{-1}. However, other water quality parameters, such as TP, DO and temperature, were not identified in the operation and maintenance manuals.

2.3. Meteorological Data

Meteorological data was obtained from the University of Waterloo Weather Station located at the University of Waterloo's North Campus (43°28'25.6" N, 80°33'27.5" W). The station is approximately 6.0 km and 2.5 km from Pond 33 and Pond 45, respectively. Rainfall was measured with an accuracy of ± 0.1 mm by a tipping bucket rain gauge (Texas Electronics Model TE525). Ambient air temperature with accuracy of ± 0.4°C was measured using a Vaisala Model HMP35C.

2.4. Automated and Manual Sampling Procedures

Baseflow samples were collected twice a week, however if a rain event occurred a day before or during a field day, sampling was not conducted. A total of 30 BF samples were collected from the inlet(s) and outlet of each pond. Grab samples were collected at the inlets and outlets of Pond 33 and Pond 45 for TSS, TP, SRP, and GS. Conductivity, pH, DO and temperature were measured in situ.

Stormflow samples were collected with ISCO 6712 Automatic Samplers. Each sampler is equipped with 24 1L acid-washed and triple rinsed bottles. To increase the number of samples collected during a storm event each bottle was used as an individual sample. Discharge at each inlet and outlet was measured with an ISCO 4150 Area Velocity Flow Logger. Flowlink 5 software was used to program the sampling frequency for each location. Flowlink 5 was unavailable until mid-August, therefore the samplers were initially time paced and triggered base on a pre-determined start time. Each sampler collected four samples in the first hour, six samples within the next three hours, and one sample an hour after that for a total of 24 samples within an 18 hour period. After the arrival of Flowlink, the samplers were programmed to trigger sampling once the flow measured by each ISCO 4150 Area Velocity Flow Logger exceeded a specified threshold. The Rational Method was used to calculate the peak discharge for each inlet

$$Q=c\ i\ A \tag{2}$$

where, c is the runoff coefficient of 0.50 from Dunne and Leopold (1978), i is rainfall intensity and A is the drainage area (Dunne and Leopold, 1978). Ten percent of the Rational Method value for each inlet was used as the threshold to trigger the ISCO 6712 to begin sampling. Flow pacing was used to collect samples at the three inlets which allow a greater proportion of sampling when the flow rate is high, and a lower proportion of samples when the flow rate decreases (Van Buren et al., 1997). The magnitude of a storm event dictated the flow pacing. For example, a storm event of 5 mm would collect a sample every 15 m^3, while a storm event of 30 mm would collect a sample every 100 m^3. This ensured that samples were collected throughout the entire storm event. The sampling program at each outlet followed the same time pacing as mentioned above. Samples were collected once the sampling program was completed and transferred to the Sediment and Water Quality Lab at the University of Waterloo for processing. Each sample was analyzed for TP, SRP, TSS, GS, pH, conductivity and temperature using methods described below.

2.5. Water Quality Parameters

2.5.1. pH, Conductivity, Dissolved Oxygen, Temperature

Conductivity and temperature were measured with an Orion 105A+ Conductivity Meter. An Orion 250A pH meter and YSI 55-12 dissolved oxygen (DO) meter were also used. All meters were calibrated weekly according to standard methods. The membrane and saturated potassium chloride solution for the DO meter was calibrated each time the meter was used for baseflow sampling.

2.5.2. Phosphorus Analysis

Water samples were transferred from a 1L sampling bottle to 120 mL acid-washed triple rinsed glass bottles and preserved with 1 mL of 20% H_2SO_4 within two hours of sample collection for TP analysis. Soluble reactive phosphorus samples were filtered with a 0.45 µm Whatman Schleicher and Schuell (Whatman) filter and luerlock syringe into a 20 mL scintillation vial within two hours of sample collection and were subsequently stored in a refrigerator at approximately 4°C. Storage times for TP and SRP are approximately 3-4 months and 2-3 weeks, respectively (Environment Canada, 1979).

The potassium persulphate digestion method was used to prepare samples for TP analysis (Environment Canada, 1979). Twenty mL aliquots were measured into 50 mL Erlenmeyer flasks with 0.5 mL of saturated potassium persulphate and boiled down to 2 to 5 mL. Each sample was diluted back up to 20 mL using de-ionized water and filtered with a 0.45 µm Whatman filter. A Technicon Auto-analyzer II single-channel colourimeter was used to analyze TP and SRP. The stannous chloride-ammonium molybdate method (Environment Canada, 1979) was used to measure TP and SRP concentrations ≤ 200 µg L^{-1} and ≤ 100 µg L^{-1}, respectively. Concentrations higher than the above values required dilution. Method detection limits for TP and SRP are 0.5 μg L^{-1} and 1.75 μg L^{-1}, respectively.

Vertical profiles of SRP in sediment porewater were determined using PWP. Fourteen PWPs were acid washed and triple rinsed, filled with deionized water. A 64 μm membrane and filter cloth was inserted in each PWP to restrict the flow of larger particles. PWP were stored in a new tupperware container filled with deionized water until installation. Three PWP were placed vertically in each compartment of Pond 33: sediment forebay, secondary settling pond and wetland (Figure 3). Five PWPs were installed in Pond 45; three near Inlet 2 and two between Inlet 1 and Inlet 2 (Figure 4). After eight weeks the PWPs were equilibrated with pond water and sediment porewater and were removed. A luerlock syringe and needle was used to extract a composite sample of 5 mL corresponding to 2.5 cm of the sediment profile. The sample was immediately filtered with a 0.45 μm Whatman filter into a scintillation vile and stored on ice until arrival at the laboratory. Samples were analyzed for SRP using a Technicon Auto-Analyzer.

2.5.3. Suspended Sediment Analysis

Total suspended solid concentrations were determined gravimetrically (Standard Methods, 2008). Grain size distribution of suspended solids was determined from using subsamples from each ISCO bottle. Each bottle was carefully inverted 3 to 4 times to resuspend sediment and a sample was poured into a known volumetric column and filtered. Larger particles will settle quickly and likely be underestimated using this method. The methods of Deboer and Stone (1999) were used to collect SS samples for subsequent image analysis and particle characterization. In this procedure, SS were deposited on a 0.45 μm Whatman filter. To distinguish particles from the background filter paper three drops of low viscosity microscope immersion oil is applied to render the filter paper semi-transparent. An inverted microscope

(Zeiss Axiovert S100) fitted with a Sony XC-75 CDD video camera and Northern Eclipse Image analysis software was used to measure particle size and shape characteristics. Selected SF events were used to establish a representative sample of GS variability based on storm intensity. Samples from the inlet(s) and outlet of Pond 33 and Pond 45 were used to measure particle size distributions. The data are expressed as D_{50} and D_{90} GS distributions and photomicrographs.

Five bottom sediment samples were collected with a Ponar sediment sampler at the locations near the PWP installations. The grain size distributions of two composite sediment samples from Pond 33 and Pond 45 were determined with a hydrometer (A.S.T.M., 1964). For each sample approximately 50 g of air dried sediment was covered with 125 mL of 4% sodium hexametaphosphate for 16-18 hours to ensure any ionic charge between particles was neutralized. Each sample was diluted with deionized water up to 1 L in a glass cylinder and was inverted 30 times within 1 minute. Hydrometer readings were taken at regular intervals and the test was completed after approximately 4 days.

2.6. Sediment Geochemistry and Mineralogy

Four cone and quartered bottom sediment samples from Pond 33 and Pond 45 were submitted to Activation Laboratories Ltd. in Ancaster, Ontario for geochemical and mineralogical analysis using X-ray fluorescence (XRF) and X-ray diffraction (XRD), respectively. Sediment samples were analyzed on a Panalytical Axios Advanced XRF and Phillips X'Pert PW3040-PRD diffractometer equipped with copper X-ray source for XRF and XRD analysis, respectively. Detection limits for XRF and XRD analysis were generally 0.01 wt% and 0.5 to 5% weight, respectively. One sediment sample was submitted from each of the design compartments of Pond 33. Pond 45 is essentially one compartment; therefore, one composite sediment sample from the two PWP sampling locations was submitted. The Certificate of Analyses for these methods are listed in Appendix A.

2.7. Sediment Buffering Capacity Experiments

Sediment buffering capacity experiments described by Stone and Mudroch (1989) were conducted. Triplicate sediment samples of ≤63 µm and >63 µm sediment size fractions from Pond 33 and Pond 45 were weighed individually into 50 mL centrifuge tubes. Twenty five mL aliquots of 0, 25, 50, 100, 200, 400 and 800 µg P L^{-1} were added to 50 mL centrifuge tubes with screw caps. Competitor ions were neutralized with the addition of 0.5 ml of 0.5M $CaCl_2$. The centrifuge tubes were placed on a shaker at room temperature for 18 hours, filtered with a 0.45 µm Whatman filter and analyzed for SRP on a Technicon Auto-analyzer.

2.8. Pond Trap Efficiency

Pond TE was determined using Equation 1. A positive value indicates the pond is acting as a sink, whereas a negative value suggests the pond is a source of the constituent of concern (Davis and McCuen, 2005).

2.9. Quality Assurance / Quality Control

Triplicate samples of TP and SRP from each sampling location were collected once a month during BF periods to assess the variability of grab samples. Three field blanks were also analyzed monthly to assess the contribution of P from field contamination. Triplicate samples of TP, SRP, TSS and GS were collected during a 3.4 mm storm event on November 25, 2008. Samples were collected at the outlet of Pond 45 only because the flow loggers at the inlets of Pond 45 and inlet and outlet of Pond 33 did not trigger. Laboratory blanks from ISCO 1L sample bottles, 20 mL syringes and 20 mL syringe with filter attached were analyzed for TP on a Technicon Auto-analyzer.

2.10. Statistical Analysis

Statistical analyses were conducted on TP, SRP and TSS data to determine if there were significant differences between the inflow and outflow concentrations for individual storm events and the entire BF period for Pond 33 and Pond 45. Significant differences between the inlets and outlets of Pond 33 and Pond 45 were assessed using a one sample Kolmogorov-Smirnov test to assess normality within the data. The Kolmogorov-Smirnov test indicated the samples were not normally distributed for this study. Most of the SF samples were not paired because the samplers were programmed to sample based on flow, therefore, the non-parametric Mann-Whitney U statistical test was used to test for significant differences.

Chapter 3: Results

3.1. Meteorological and Hydrological Results

A record amount of precipitation fell during the summer of 2008 which was the fifth wettest summer on record since 1915 (Weather Summary, 2008). Monthly precipitation collected from the University of Waterloo Weather Station from 1998 to 2008 is shown in Table 4. Monthly precipitation for 2008 is compared to the 30 year average from the Waterloo Wellington Airport data set (1971-2000) in Table 5. The data show that July and September received above average precipitation, while the months of August and October received average and below average amounts of precipitation, respectively. The average monthly high and low temperatures for the study period were comparable to the 30 year monthly high and low average temperature. For example, the average high for July, 2008 was 25.7ºC while the long term average was 25.9ºC. Temporal variation of precipitation and temperature values recorded at the University of Waterloo Weather Station during the field season is illustrated in Figure 5.

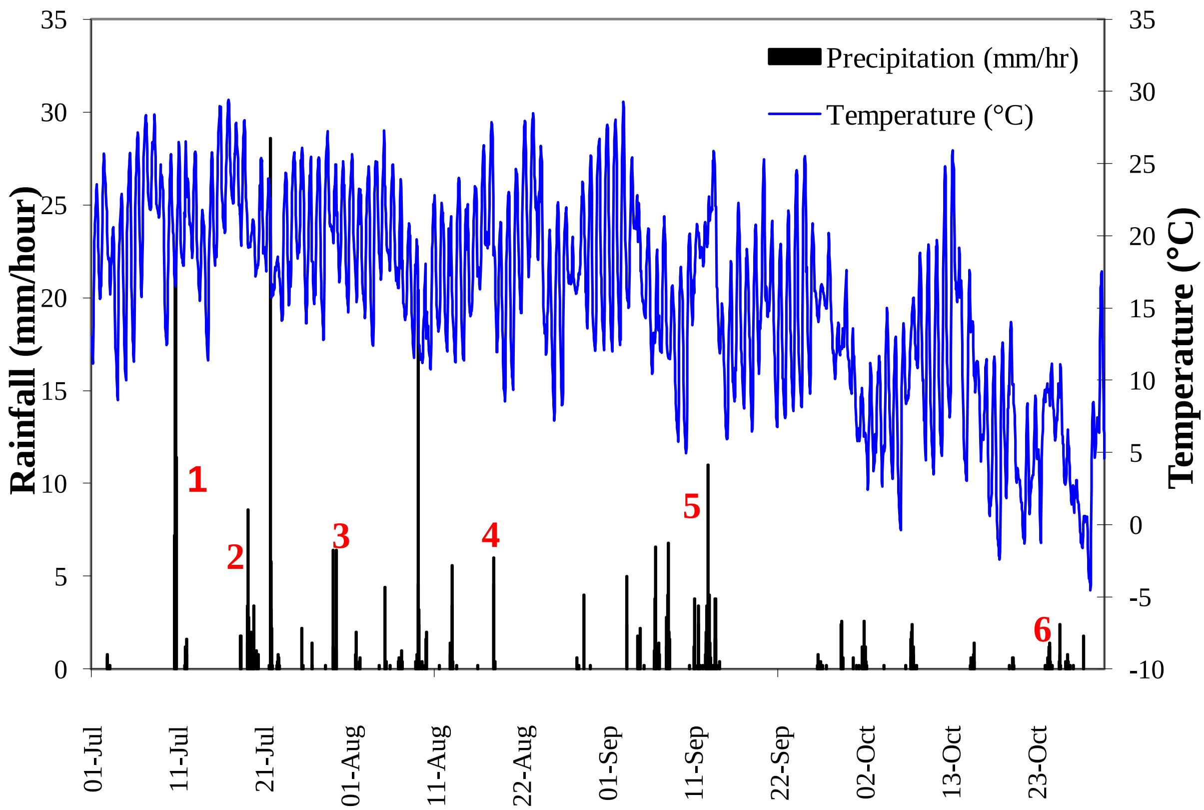


Figure 5. Precipitation and temperature variability during the 2008 field season. Storm events monitored are listed 1 through 6.

Table 4. Monthly precipitation values in millimeters recorded at the University of Waterloo Weather Station.

Year	January	February	March	April	May	June	July	August	September	October	November	December
1998	-	-	-	17.3	15.4	43.8	15.5	28.9	21.6	7.8	20.9	-
1999	-	-	-	21.4	23.2	58.7	29.2	24.9	38.9	32.3	28.7	-
2000	-	8.40	16.9	34.8	58.9	81.3	56.5	10.8	37.5	9.10	27.6	-
2001	-	-	-	19.5	52.9	22.3	5.10	15.6	37.5	60.9	24.4	-
2002	-	-	-	57.3	99.3	76.6	94.4	24.0	67.5	47.1	66.7	41.0
2003	46.9	68.8	65.8	73.6	122.7	40.0	52.7	39.0	84.9	84.7	84.0	66.1
2004	85.4	42.9	109.9	56.0	145	51.2	108.6	56.4	27.4	75.0	78.0	56.4
2005	73.1	79.0	29.1	75.2	24.1	43.0	111.8	92.3	81.6	37.4	127.9	43.0
2006	90.3	96.9	64.0	71.0	113.4	32.8	152.2	52.4	117.2	131.4	68.6	86.4
2007	65.1	27.4	56.7	70.7	59.1	26.6	50.8	62.6	36.8	47.6	76.2	57.2
2008	83.4	34.6	16.0	53.8	59.8	106.4	181	79.2	116.2	45.0	103.6	99.6

Table 5. Monthly (2008) and 30 year average precipitation values for four months.

Month	UW* (mm)	WWA** (mm)
July	181.0	92.9
August	79.2	87.0
September	116.2	87.5
October	45.0	67.1

* University of Waterloo Weather Station

** Waterloo Wellington Airport Weather Station

A total of six storm events were monitored (Figure 5). The return period, amount of precipitation and storm intensity of the six storm events sampled are listed in Table 6. Storm intensity was determined by dividing the rainfall depth by the storm duration. Subsequently, return periods were calculated from the local Intensity Duration Frequency curves supplied by Environment Canada. The event sampled on Julian Day (JD) 193 had the highest storm intensity of 13.6 mm hr^{-1} and a return period of 10 years. The other storm events are of lower intensity and have return periods <2 years. Small, frequent storms are of importance to stormwater management because the majority of rainfall is from these types of storms (Nehrke and Roesner, 2004).

Table 6. Rainfall depth, stormflow intensity and return period for six stormflow events captured during the field season.

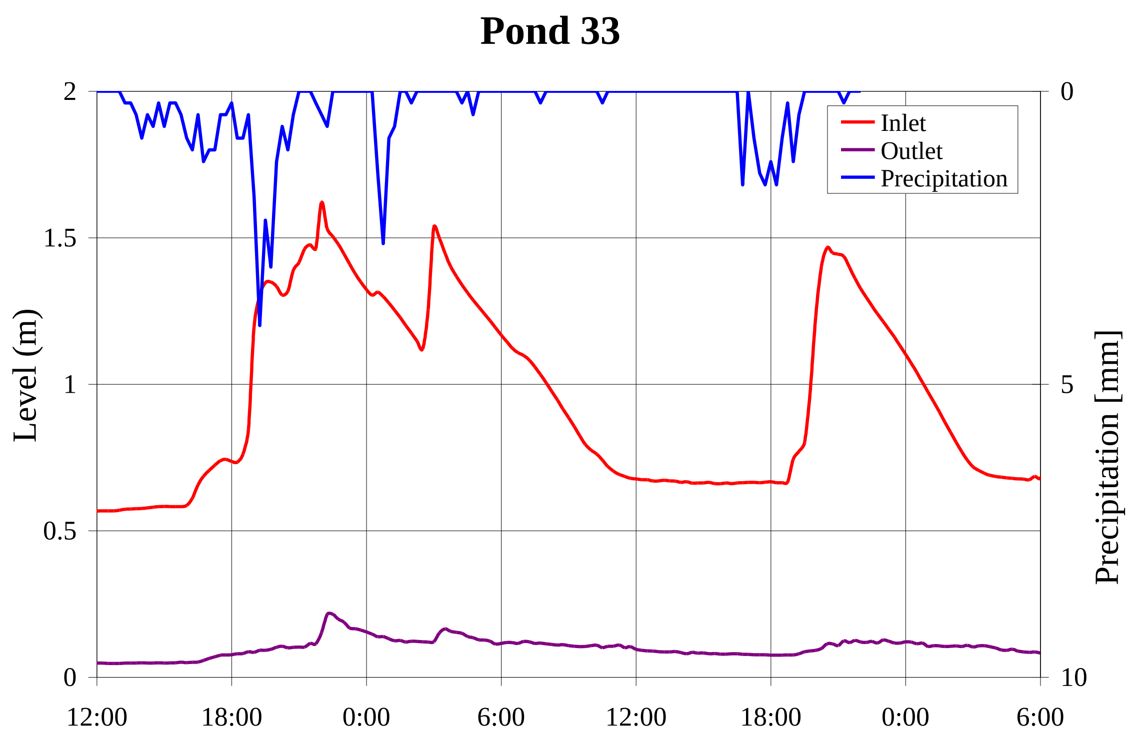
Julian Day	Rainfall Depth (mm)	Intensity (mm hr^{-1})	Return Period
193	78.0	13.6	10
201	9.4	4.1	<2
212	16.0	7.0	<2
231	11.0	8.5	<2
257	28.0	3.6	<2
299	6.4	1.3	<2

Water level, velocity and discharge were measured every fifteen minutes at each inlet and outlet of Pond 33 and Pond 45 with an ISCO 4150 Area Velocity Flow Logger. Discharge measurements are only available for the months of August, September and October at the outlet of Pond 45 and the months of September and October for the outlet of Pond 33 because of a delay receiving Flowlink 5 software and periodic equipment malfunctions. However, water level data is available for the majority of events and will be used as a surrogate to discharge values. Changes in inlet and outlet water levels at Pond 33 and Pond 45 during a storm event on JD 257 are illustrated in Figure 6. The storm event had an intensity of 3.6 mm hr^{-1} and was approximately 13 hours in duration.

The water level at the inlet of Pond 33 increased approximately 0.5 m over a three hour period. The soakaway pits, infiltration basins and smaller drainage area (37% of Pond 33) caused the water level response at Pond 45 to be less pronounced. Water levels at the outlet of Pond 33 gradually increased to the hydrograph peak and gradually declined after the storm event subsided. Pond 45 had more of a sudden increase in water level at the outlet after the first peak in precipitation; however, it steadily declined with the end of the storm event. Changes in water level at Pond 33 and Pond 45 for the other five storm events can be found in Appendix B.

3.2. Inflow, Outflow and Porewater Phosphorus Concentrations

Mean TP inflow concentrations ± standard deviation for individual storm events for Pond 33 and Pond 45 are presented in Table 7. They ranged from 38.9 ± 23.8 µg L^{-1} to 163.7 ± 81.0 µg L^{-1} on JD 212 and JD 193 and 26.7 ± 11.7 µg L^{-1} to 201.6 ± 144.5 µg L^{-1} on JD 201 and 299, respectively (Table 7). Mean TP outflow concentrations for individual storms are not as variable compared to the inlet(s) at Pond 33 and Pond 45. Mean stormflow TP outflow concentrations ± standard deviation ranged from 88.6 ± 47.2 µg L^{-1} to 107.1 ± 22 µg L^{-1} on JD 201 and JD 299 for Pond 33 and 7.9 ± 4.9 µg L^{-1} to 27.1 ± 9.0 µg L^{-1} JD 257 and JD 193 for Pond 45. Mean ± standard error around the mean TP concentrations for all storm events combined was 94.3 ± 7.7



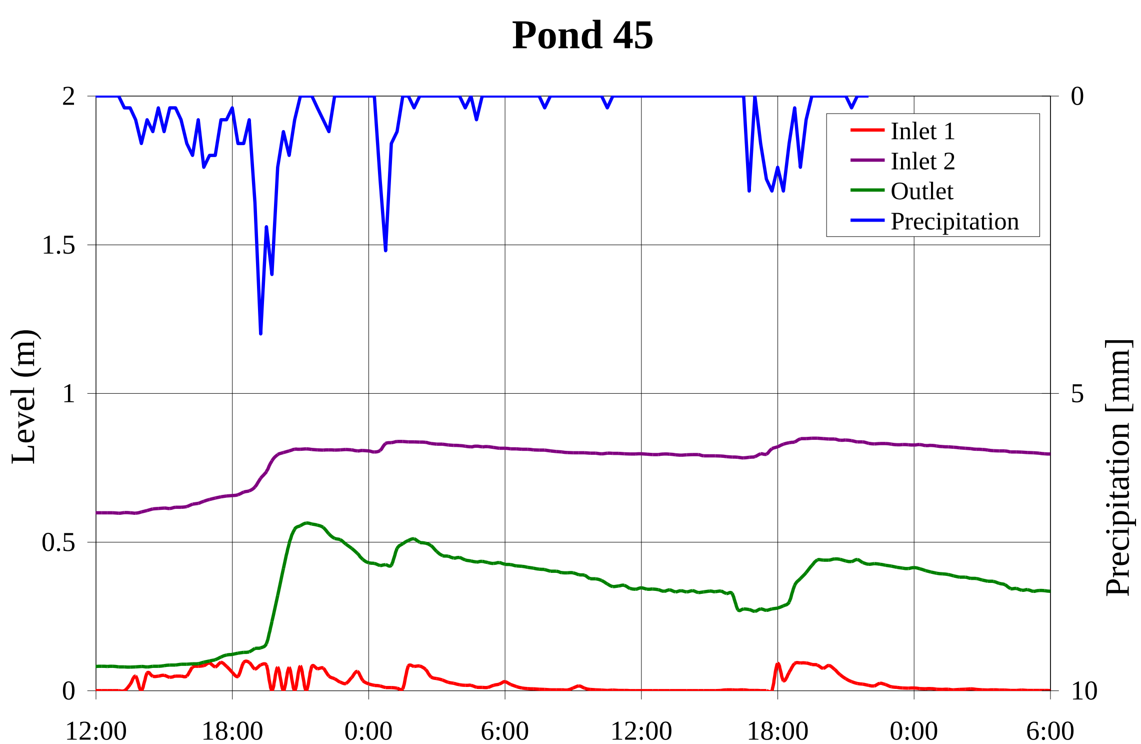


Figure 6. Change in water level at the inlet(s) and outlet in response to a rain event beginning on JD 257 for two stormwater ponds.

Table 7. Mean inflow and outflow TP and SRP concentrations (± standard deviation) for six storm events in two stormwater management ponds.

Parameter	**Julian Day**	**Pond 33**				**Pond 45**					
		Inlet		**Outlet**		**Inlet 1**		**Inlet 2**		**Outlet**	
		n	**Mean ± Std. Dev**	**n**	**Mean ± Std. Dev**	**n**	**Mean ± Std. Dev**	**n**	**Mean ± Std. Dev**	**n**	**Mean ± Std. Dev**
TP (µg L^{-1})	193	11	163.7 ± 81.0	24	99.9 ± 8.6	11	140.0 ± 75.5	11	116.8 ± 49.7	42	27.1 ± 9.0
	201	13	49.8 ± 17.2	13	88.6 ± 47.2	7	34.7 ± 8.9	13	26.7 ± 11.7	13	14.7 ± 4.0
	212	12	38.9 ± 23.8	24	91.9 ± 39.2	8	97.6 ± 79.0	12	79.2 ± 47.6	24	17.4 ± 3.6
	231	6	151.3 ± 34.5	24	98.1 ± 21.8	-	ND	24	71.5 ± 42.8	24	8.2 ± 2.1
	257	7	61.8 ± 6.3	24	97.1 ± 22.3	20	122.3 ± 72.0	20	91.8 ± 41.5	24	7.9 ± 4.9
	299	20	112.2 ± 46.9	13	107.1 ± 22.0	16	201.6 ± 144.5	24	61.7 ± 38.5	24	12.7 ± 13.0
SRP (µg L^{-1})	193	11	69.1 ± 42.3	24	62.8 ± 3.7	11	83.2 ± 57.2	11	59.1 ± 36.3	42	9.9 ± 6.5
	201	13	23.2 ± 11.5	13	30.1 ± 28.6	7	5.0 ± 6.1	13	3.2 ± 8.0	13	0.9 ± 0.6
	212	12	19.0 ± 9.9	24	40.7 ± 30.7	8	21.7 ± 47.3	12	20.5 ± 28.0	24	0.9 ± 0.5
	231	6	53.8 ± 31.2	24	30.2 ± 15.3	-	ND	24	29.5 ± 25.4	24	0.9 ± 0.01
	257	7	28.6 ± 4.8	24	47.4 ± 17.4	20	59.3 ± 37.6	20	42.7 ± 19.0	24	0.9 ± 0.3
	299	20	34.2 ± 18.4	13	39.5 ± 8.8	16	70.7 ± 49.9	24	31.3 ± 15.9	24	2.4 ± 2.7

ND = No Data

μg L^{-1} and 97.1 ± 2.5 μg L^{-1} at the inlet and outlet of Pond 33, respectively. Pond 45 had mean TP concentrations ± standard error at Inlet 1, Inlet 2 and the outlet of 132.8 ± 13.4 μg L^{-1}, 73.2 ± 4.5 μg L^{-1}, and 16.1 ± 0.7 μg L^{-1}, respectively. Inlet 1 at Pond 45 always had a higher mean TP concentration than inlet 2 and almost always had a larger standard deviation from the mean. The highest TP concentration sampled from the inlet of Pond 33 was 312 μg L^{-1} during a storm event on JD 193 and 547 μg L^{-1} from Inlet 1 in Pond 45 on JD 299. Mean TP baseflow concentrations ± standard error of the mean was 101.5 ± 33.9 μg L^{-1} and 131.3 ± 12.2 μg L^{-1} at the inlet and outlet of Pond 33, while Pond 45 had concentrations of 60.48 ± 6.1 μg L^{-1}, 24.0 ± 4.0 μg L^{-1} and 10.1 ± 1.3 μg L^{-1} at Inlet 1, Inlet 2 and outlet, respectively.

The distribution of TP and SRP at the inlet(s) and outlet for the six storm events sampled at Pond 33 and Pond 45 are presented in Figure 7. Total phosphorus concentration variability is large at the inlets of each pond as well as the outlet of Pond 33. Mean inlet concentrations at both ponds do not show a definite trend throughout the field season. Inlet TP and SRP concentrations at Pond 33 ranged from 9.5 μg L^{-1} to 903.3 μg L^{-1} and 0.9 μg L^{-1} to 936.9 μg L^{-1}, respectively. Minimum and maximum TP and SRP concentrations at Inlet 1 in Pond 45 were 0.9 μg L^{-1} to 720.5 μg L^{-1} and 0.9 μg L^{-1} to 442.0 μg L^{-1}, respectively. Total phosphorus and SRP concentrations at Inlet 2 in Pond 45 ranged from 0.3 μg L^{-1} to 188.3 μg L^{-1} and 0.9 μg L^{-1} to 111.2 μg L^{-1}, respectively. Pond 45 outlet TP concentrations exhibited the least amount of variability from all locations sampled. Outflow concentrations at Pond 45 were below the PWQO criteria of 30 μg L^{-1} 93% of the time. However, outlet TP concentrations at Pond 33 were above the PWQO guideline 99% of the time.

The temporal variation in TP concentrations at the inlet(s) and outlet of Pond 33 and Pond 45 in relation to three differing storm events are shown in Figure 8 and Figure 9. Three storms were chosen to contrast small (<10 mm), medium (10 mm to 30 mm) and large (>30 mm) storm events. For the purpose of this study, storm events on JD 299, 212 and 193 are classified as small, medium and large storm events, respectively. Antecedent dry days (ADD) for JD's 193, 212 and 299 are 1, 3 and 4 days, respectively. Outflow TP concentrations do not respond to a change in storm intensity compared to their corresponding inlet(s) (Figure 8 and Figure 9) and

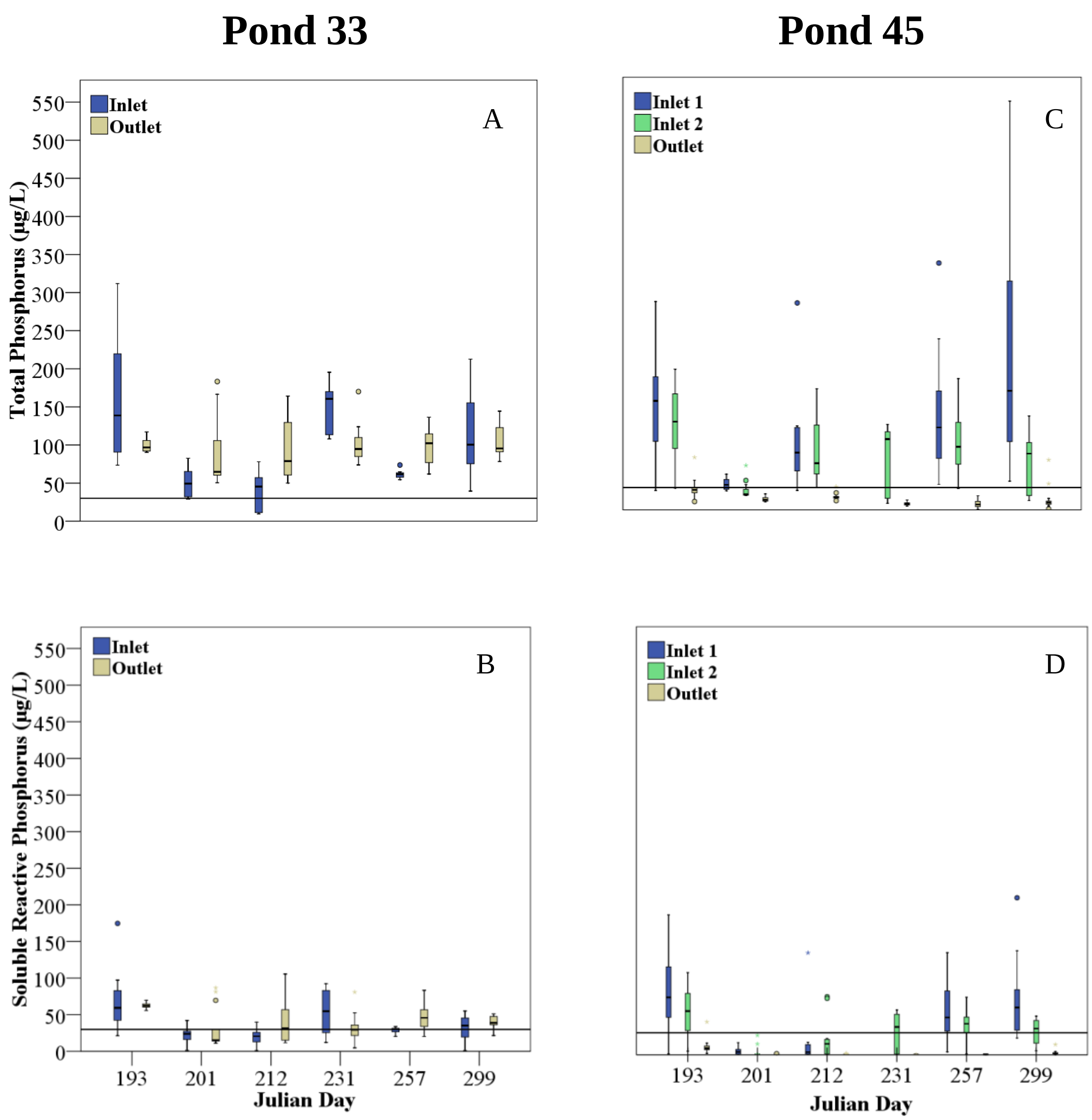


Figure 7. Distribution of TP and SRP at the inflow(s) and outflow of two stormwater management ponds. The solid line indicates the PWQO of 30 µg L^{-1} for TP. (*= extreme value; °=outlier)

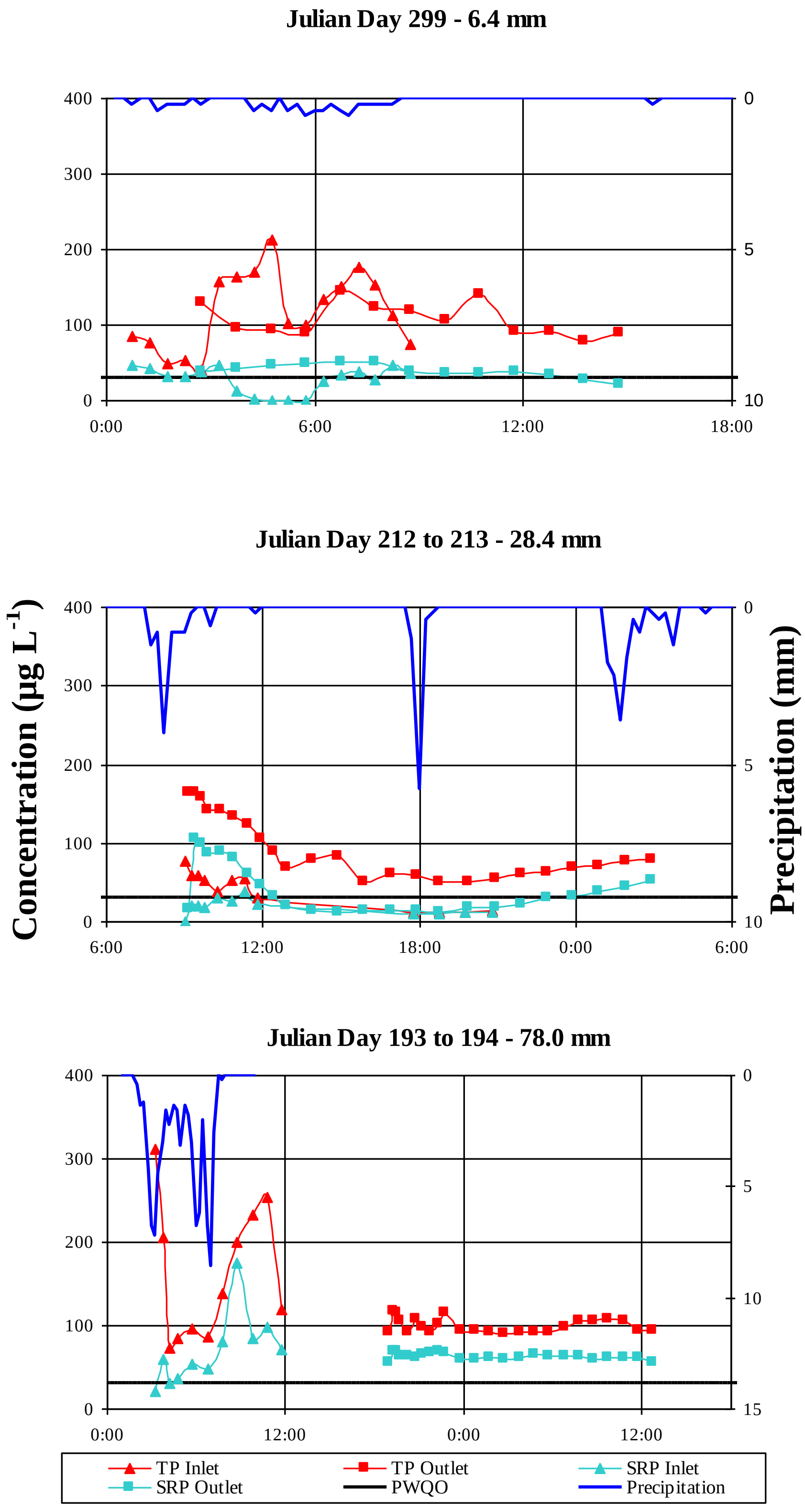


Figure 8. Temporal variability of TP and SRP for storm events of varying magnitudes at the inflow and outflow of Pond 33. PWQO: Provincial Water Quality Objectives.

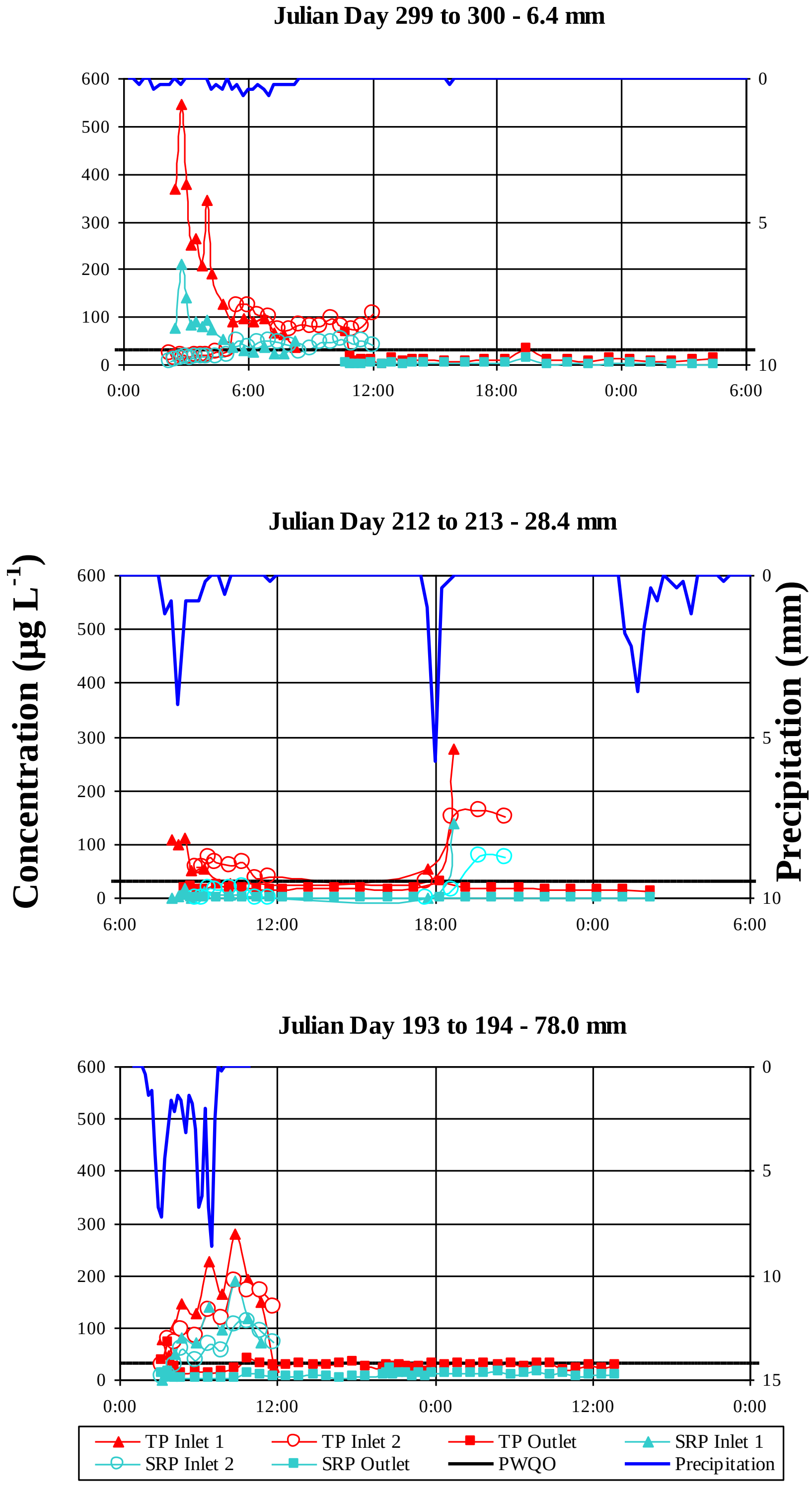


Figure 9. Temporal variability of TP and SRP for storm events of varying magnitudes at the inflow and outflow of Pond 45. PWQO: Provincial Water Quality Objectives.

TP is released at an almost constant rate for Pond 33 and Pond 45. Inlet 1 and Inlet 2 in Pond 45 increase with an increase in storm intensity during a storm event on JD 193 and JD 212. The inlet at Pond 33 shows the same response to increased storm intensity as Pond 45.

A single storm event captured for each month of July, August, September and October was chosen to represent the temporal variability of TP concentrations during the entire field season at the inlet(s) and outlet of Pond 33 and Pond 45 (Appendix B). The temporal variation of TP concentrations at the inlet and outlet of Pond 33 showed little variation for storm events sampled on JDs 201, 231, 257 and 299 with ADD of 2, 2, 1 and 4 days, respectively. Inlet concentrations ranged from a minimum of 28.8 μg L^{-1} on JD 201 to a maximum of 212.7 μg L^{-1} on JD 299, however, outlet TP concentrations ranged from 50.4 μg L^{-1} to 183.3 μg L^{-1} on JD 201. Inlet TP concentrations for Pond 45 exhibited greater variability compared to Pond 33. Peak inlet TP concentrations progressively increased throughout the field season from 47.7 μg L^{-1} (JD 201) to 547.1 μg L^{-1} (JD 299) and from 59.4 μg L^{-1} (JD 201) to 125.7 μg L^{-1} (JD 299) for Inlet 1 and Inlet 2, respectively. However, outlet TP concentrations remained relatively constant around 15 μg L^{-1}. Mean TP concentrations at the inlet(s) and outlet of Pond 33 and Pond 45 during BF and SF events throughout the field season are shown in Figure 10. Mean TP concentrations were usually higher during storm events and outlet TP concentrations were higher than inlet concentrations at Pond 33 even during BF periods.

Mean SRP concentrations ± standard deviation for individual storm events for Pond 33 ranged from 19.0 ± 9.9 μg L^{-1} to 69.1 ± 42.3 μg L^{-1} on JD 212 and JD 193 and 30.1 ± 28.6 μg L^{-1} to 62.8 ± 3.7 μg L^{-1} on JD 201 and JD 193 for the inlet and outlet, respectively (Table 7). Mean SRP concentrations ± standard deviations at the inlet and outlet of Pond 45 for individual storm events ranged from 3.2 ± 8.0 μg L^{-1} to 83.2 ± 57.2 μg L^{-1} on JD 201 and 193 and 0.9 ± 0.01

μg L^{-1} to 9.9 ± 6.5 μg L^{-1} on JD 231 and JD 193, respectively. Mean and standard error SRP concentrations at the inlet(s) and outlet for all stormflow samples combined were 35.3 ± 3.3 μg L^{-1} and 43.0 ± 2.0 μg L^{-1} for Pond 33 and 55.1 ± 6.4 μg L^{-1}, 31.5 ± 2.6 μg L^{-1} and 3.6 ± 0.4 μg L^{-1} for Pond 45, respectively. Pond 33 BF mean and standard error results for the inlet and outlet are 44.61 ± 30.903 μg L^{-1} and 66.36 ± 8.793 μg L^{-1}, respectively. However, Pond 45 had lower mean and standard error SRP values. Inlet 1, Inlet 2 and the outlet had mean BF SRP concentrations ± standard error of 39.7 ± 4.7 μg L^{-1}, 6.8 ± 2.0 μg L^{-1} and 5.7 ± 1.1 μg L^{-1}, respectively.

The variability of SRP at the outlet of Pond 33 is greater than Pond 45 which exhibits a very small variation from the mean (Figure 7). Mean SRP variability during the entire field season is illustrated in Figure 10. Storm events had the highest SRP concentrations at the inlet(s) and outlet of each pond compared to mean BF SRP concentrations. An increase in SRP concentrations occurred slightly between JD 230 and JD 257. During this time period peak TP concentrations were around 275 μg L^{-1} and approximately 70% of this was in the dissolved form. Peak SRP concentrations between these dates are at least double the concentrations of SRP throughout the rest of the field season.

Mean temperature, pH and dissolved oxygen values taken in situ at each inlet and outlet are displayed in Table 8. Water column temperatures were between 17.8ºC and 23.8ºC from July to September and decreased to between 7.8ºC and 10.8ºC in October. Outlet water column temperatures either decreased or were similar to the inlet(s) temperatures at both ponds. Throughout the field season pH values at all locations were variable, ranging from 6.8 to 8.9. Dissolved oxygen concentrations at the outlet of Pond 33 were considered hypoxic as they were considerably lower than the inlet DO concentrations (Table 8). Dissolved oxygen concentrations at Pond 45 were not as variable between the inlets and the outlet (Table 8).

Temporal variability of SRP concentrations at the inlet(s) and outlet of Pond 33 and Pond 45 are also illustrated in Figure 8 and Figure 9. Inlet concentrations of SRP at Pond 33 and Pond

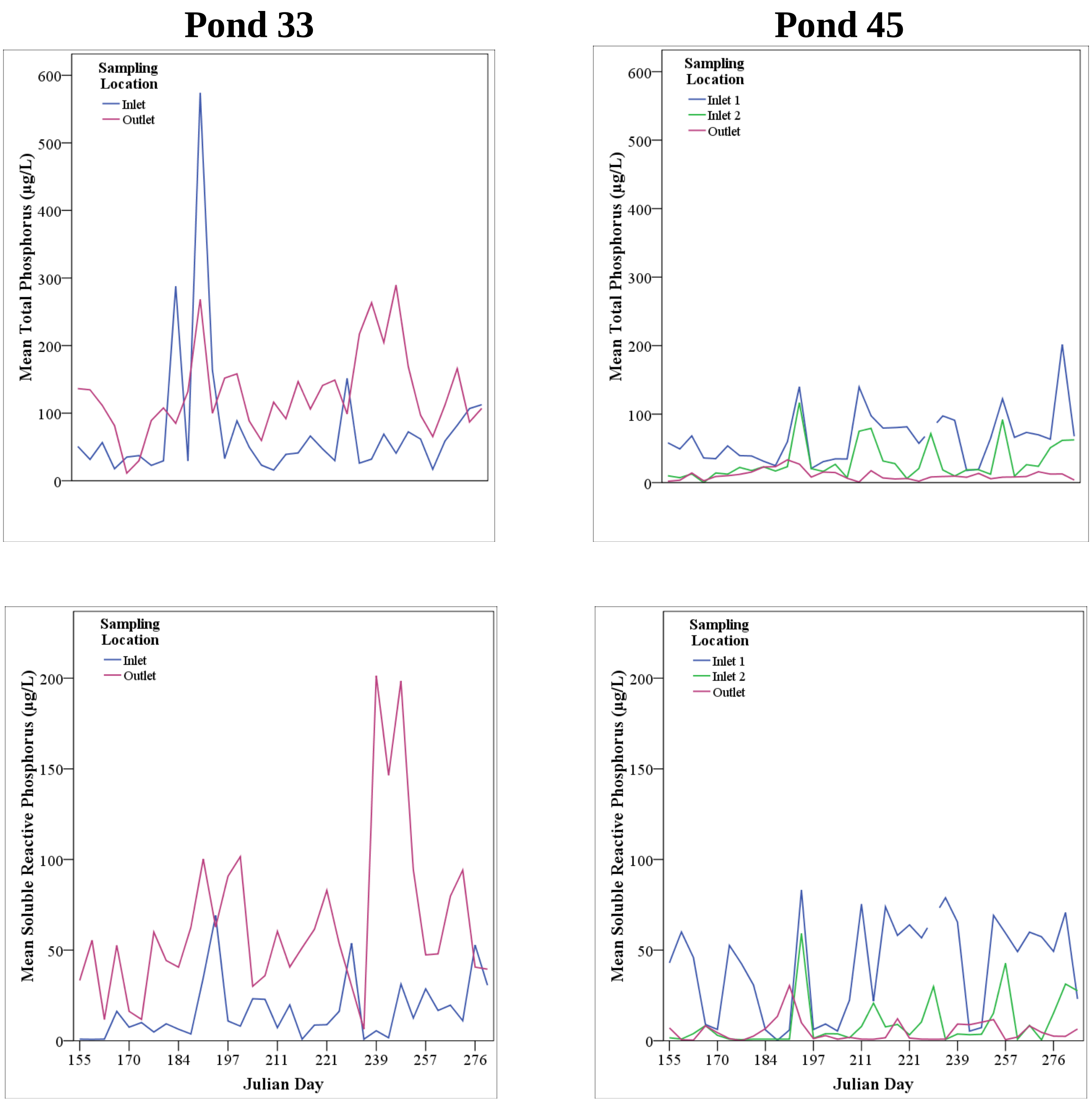


Figure 10. Mean TP and SRP concentrations at the inlet(s) and outlet of two stormwater management ponds during BF and SF sampling events.

Table 8. Average(standard deviation) BF pH, temperature (°C) and dissolved oxygen (mg L^{-1}) levels at two stormwater management ponds during the field season. n=21

Month	Average	Pond 33		Pond 45		
		Inlet	Outlet	Inlet 1	Inlet 2	Outlet
July (n=8)	pH	7.8(0.7)	7.3(0.2)	8.1(1.2)	8.9(1.2)	8.9(1.0)
	Temp	21.8(4.2)	22(2.5)	20.8(2.6)	23.8(2.0)	23.7(1.9)
	DO	9.0(1.7)	3.1(0.7)	6.5(1.7)	8.2(2.3)	6.3(1.2)
August (n=5)	pH	7.6(0.3)	7.3(0.2)	7.6(0.2)	7.9(0.2)	8.3(0.3)
	Temp	21(3.1)	19.6(2.7)	19.0(2.7)	21.4(1.6)	22.8(2.3)
	DO	7.0(1.1)	3.3(1.5)	6.7(1.8)	6.7(0.7)	6.5(1.5)
September (n=5)	pH	7.5(0.2)	7.4(0.5)	7.7(0.4)	7.9(0.4)	8.0(0.2)
	Temp	20.7(3.4)	17.8(3.2)	18.8(4.0)	21.1(2.1)	21.3(2.3)
	DO	7.2(0.8)	4.4(2.0)	7.4(2.2)	6.3(0.4)	6.5(0.6)
October (n=3)	pH	7.2(0.2)	6.8(0.9)	7.4(0.5)	7.0(0.4)	7.3(0.2)
	Temp	10.8(3.9)	7.8(4.6)	9(5.6)	10.4(3.7)	8.9(5.3)
	DO	6.5(1.5)	6.4(1.2)	8.0(1.6)	7.5(0.1)	8.4(2.0)

45 increased with increasing storm intensity and subsequently declined in concentration when storm intensity decreased. Outlet concentrations for SRP for both ponds were variable, with concentrations ranging from 11.7 to 105.7 µg L^{-1} and 0.9 to 44.5 µg L^{-1} for Pond 33 and Pond 45, respectively.

Redox and pH measurements were taken at 2 cm intervals from sediment collected from a Ponar sediment sampler. Measured pH values ranged from 6.12 to 6.77 in Pond 33 and 5.15 to 6.53 in Pond 45. Table 9 presents the redox potential (mV) measured within the collected sediment samples. The maximum depth obtained was 6 cm at the sediment forebay and secondary settling pond in Pond 33, 4 cm in the constructed wetland in Pond 33 and Inlet 2 of Pond 45 and 2 cm at the north side of Pond 45. The redox measurements obtained show a slight decrease in redox values as the probe moved deeper into the sediment substrate, however, there was an increase of 5.2 mV from 2 cm to 4 cm below the sediment surface at the constructed wetland.

Sediment porewater profiles of SRP for Pond 33 and Pond 45 are presented in Figure 11 and Figure 12, respectively. These figures show the change in porewater SRP concentrations with depth. The data indicate that Pond 33 has the highest concentration of SRP in the bottom

Table 9. Redox potential (mV) at 2 cm intervals from sediment collected from five locations in two stormwater management ponds. n=1

Depth (cm)	Pond 33			Pond 45	
	Sediment Forebay	Secondary Settling Pond	Constructed Wetland	North Side	Inlet 2
-2	18.3	28.6	33.2	47.2	34.5
-4	14.4	22.5	38.4	-	26.4
-6	14.7	19.1	-	-	-

sediments in the sediment forebay and the smallest concentration in the constructed wetland (Figure 11). The zone of anoxia is roughly around 7.5 to 15 cm below the sediment-water interface within the sediment forebay, and around 2.5 to 10 cm below the sediment-water interface for the secondary settling pond and constructed wetland. Peak SRP concentrations within the sediment forebay, secondary settling pond and constructed wetland were 563.9 µg L^{-1}, 384.7 µg L^{-1} and 126.6 µg L^{-1}, respectively. There is a very small reserve of SRP in sediment porewater in Pond 45 (Figure 12) compared to measurements obtained at Pond 33. Peak SRP concentrations in sediment porewater were 14.4 µg L^{-1} and 25.7 µg L^{-1} for PWPs near Inlet 2 and the north side of the pond, respectively.

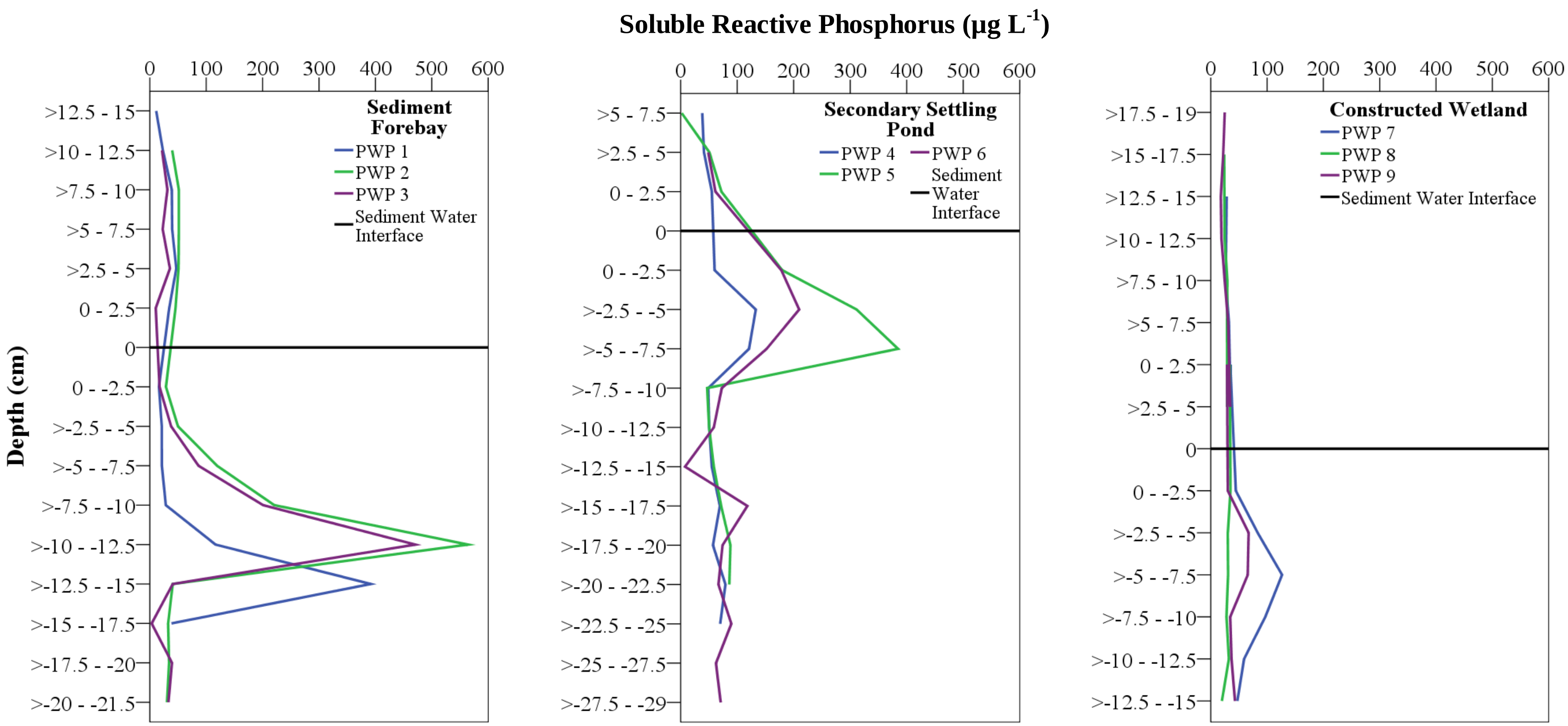


Figure 11. Sediment porewater profiles of SRP in Pond 33.

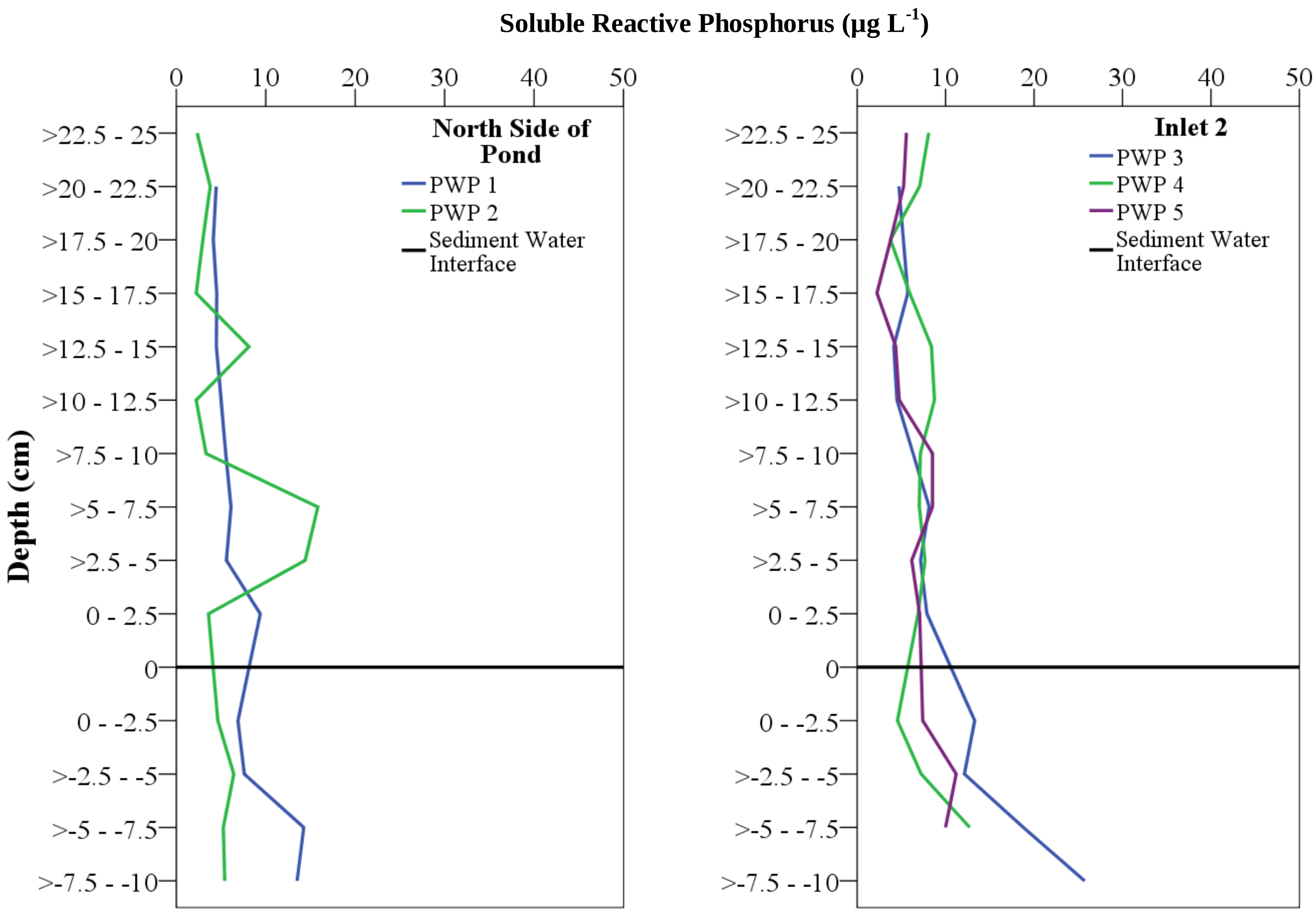


Figure 12. Sediment porewater profiles of SRP in Pond 45.

Table 10. Mean inflow and outflow TSS concentrations (± standard deviation) for six storm events in two stormwater management ponds.

Parameter	Julian Day	Pond 33				Pond 45					
		Inlet		Outlet		Inlet 1		Inlet 2		Outlet	
		n	Mean ± Std. Dev	n	Mean ± Std. Dev	n	Mean ± Std. Dev	n	Mean ± Std. Dev	n	Mean ± Std. Dev
TSS (mg L^{-1})	193	11	156.3 ± 108.8	24	31.0 ± 22.3	11	1104.6 ± 1145.0	11	91.8 ± 123.4	42	8.2 ± 11.0
	201	13	29.6 ± 36.3	13	45.4 ± 28.9	7	295.3 ± 427.5	13	10.4 ± 4.4	13	3.2 ± 1.5
	212	12	47.9 ± 55.9	24	11.3 ± 4.4	8	873.9 ± 1090.2	12	46.8 ± 62.9	24	3.2 ± 1.9
	231	6	92.4 ± 41.4	24	23.2 ± 10.1	-	-	24	16.0 ± 15.0	24	10.8 ± 18.1
	257	7	20.8 ± 6.6	24	27.1 ± 68.2	20	327.2 ± 714.4	20	37.8 ± 33.2	24	6.8 ± 7.1
	299	20	73.1 ± 76.4	13	4.5 ± 2.3	16	69.7 ± 72.7	24	16.0 ± 9.5	24	7.6 ± 18.4

3.3. Suspended and Benthic Sediment Characteristics

Suspended solid concentrations varied at the inlets of Pond 33 and Pond 45 but were generally more stable at the outlets of each pond. Mean inlet TSS concentrations ranged from 20.8 ± 6.6 μg L^{-1} to 156.3 ± 108.8 μg L^{-1} in Pond 33 on JD 257 and JD 193 for individual storm events (Table 10). However, outlet TSS concentrations ranged from 4.5 ± 2.3 μg L^{-1} to 45.4 ± 28.9 μg L^{-1} on JD 299 and JD 201, respectively. Mean and standard deviation for individual storm events for Pond 45 ranged from 10.4 ± 4.4 μg L^{-1} to 1104.6 ± 1145.0 μg L^{-1} on JD 201 and JD 193 and 3.2 ± 1.5 μg L^{-1} to 10.8 ± 18.1 μg L^{-1} on JD 201 and JD 231 at inlet 1, inlet 2 and the outlet, respectively. Table 10 presents the mean ± standard deviation TSS concentrations at the inflow and outflow of Pond 33 and Pond 45 for six storm events captured. Stormflow samples were averaged to represent the mean over the field season. Pond 33 had an average TSS concentration (standard error) of 68.6 (9.4) μg P L^{-1} and 23.5 (6.2) μg P L^{-1} at the inlet and outlet, respectively. Inlet 1, Inlet 2 and the outlet of Pond 45 had an average and standard error of 454.0 (103.9) μg P L^{-1}, 31.1 (5.2) μg P L^{-1} and 7.2 (1.0) μg P L^{-1}, respectively.

TSS concentration variability at the inlet(s) and outlet of each pond are illustrated in Figure 13 and Figure 14. The first flush of runoff sampled at the inlets show a large increase in TSS concentrations, especially at Inlet 1 in Pond 45 (Figure 14). An increase in TSS concentrations increases with higher storm intensity. The decrease in TSS concentrations at the inlets is related to sediment exhaustion. Outlet concentrations are generally constant and do not respond to changes in precipitation intensity and storm magnitude. The 2002 CCME guideline states that concentrations of TSS should not increase more than 25 mg L^{-1} from background levels. Pond 45 was consistently able to decrease TSS concentrations below this threshold. Pond

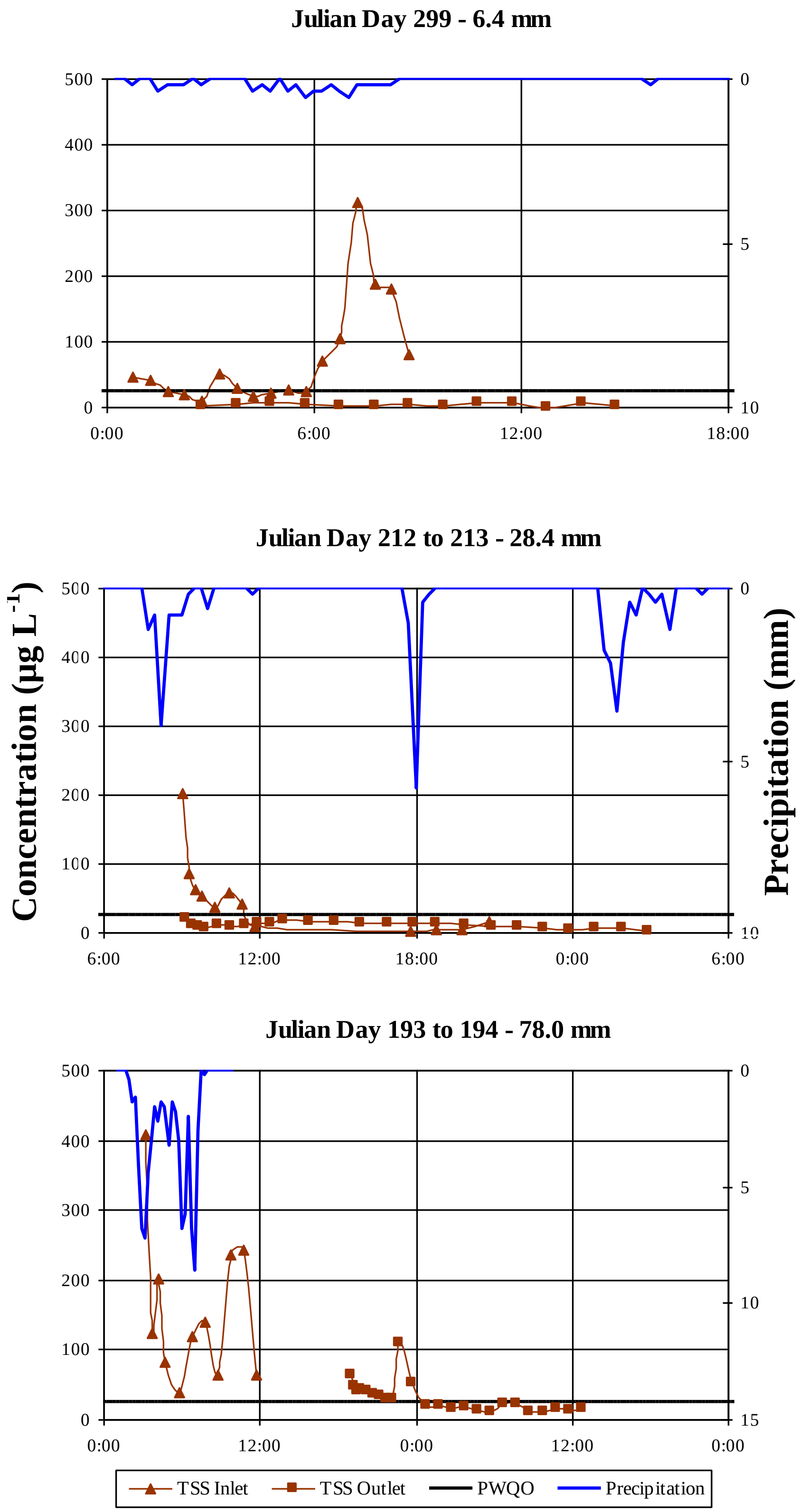


Figure 13. Temporal variability of TSS for storm events of varying magnitudes at the inflow and outflow of Pond 33. PWQO: Provincial Water Quality Objectives

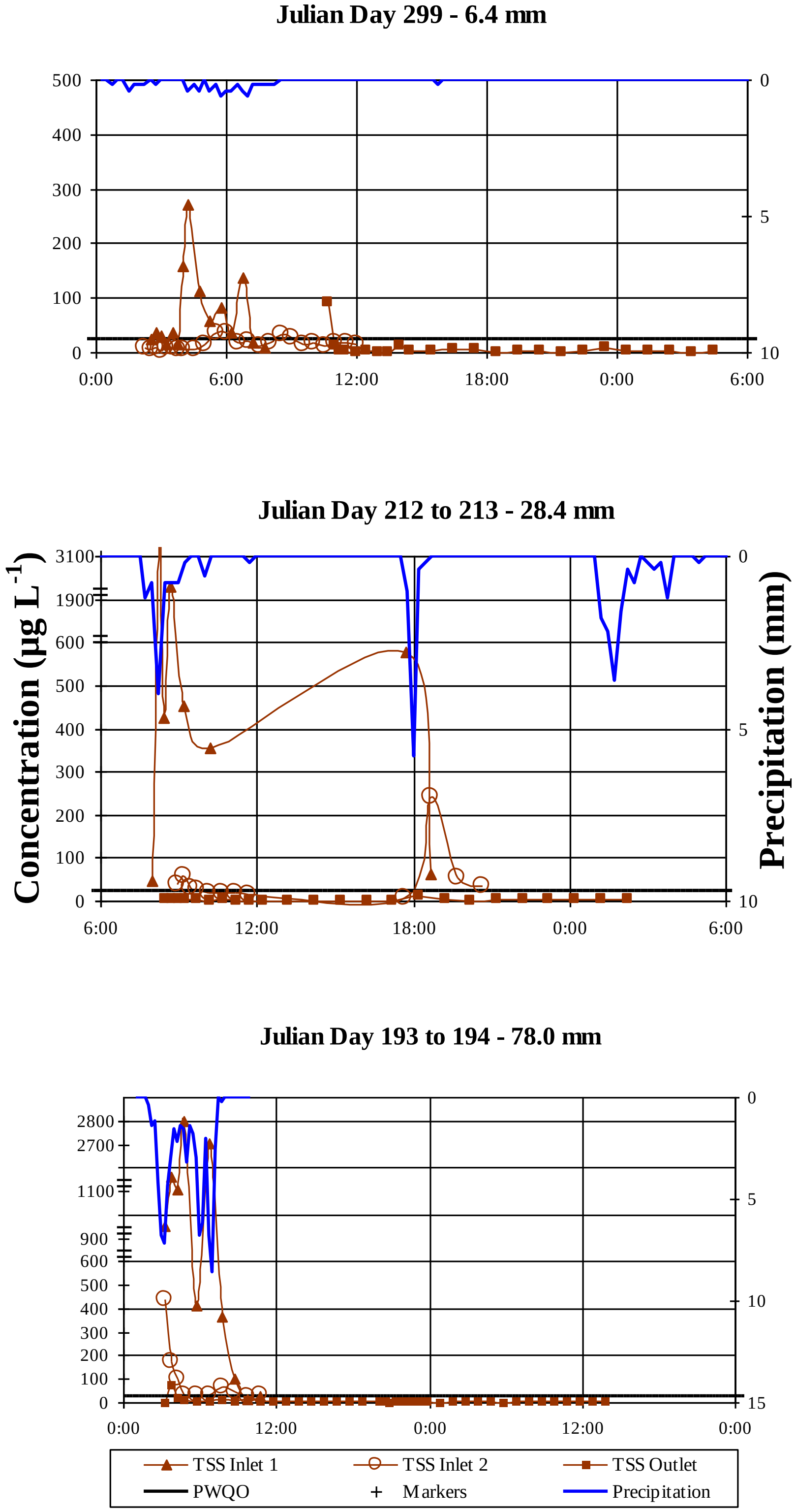


Figure 14. Temporal variability of TSS for storm events of varying magnitudes at the inflow and outflow of Pond 45. PWQO: Provincial Water Quality Objectives

Table 11. The maximum, mean D_{50} (standard deviation) and mean D_{90} (standard deviation) particle size in microns of suspended sediment in two stormwater management ponds during three storm events.

Julian Day		Pond 33				Pond 45					
		n	Inlet	n	Outlet	n	Inlet 1	n	Inlet 2	n	Outlet
193	Max		330		628		275		260		367
	D_{50}	5	4.9(0.4)	5	4.7(0.3)	5	4.2(0.3)	5	4.6(0.2)	8	4.6(0.3)
	D_{90}		13.3(2.4)		12.5(3.0)		10.8(1.0)		13.5(1.5)		12.0(3.0)
202	Max		422		283		278		716		271
	D_{50}	13	4.4(0.3)	13	4.8(0.1)	7	4.2(0.2)	13	4.3(0.1)	13	4.5(0.1)
	D_{90}		8.8(3.2)		12.5(1.3)		9.4(2.9)		9.5(1.1)		9.7(0.9)
212	Max		464		427		517		628		640
	D_{50}	12	5.2(0.7)	24	5.4(0.4)	8	5.0(0.3)	11	4.9(0.4)	24	5.0(0.2)
	D_{90}		15.8(4.1)		18.6(1.8)		15.3(1.9)		14.8(2.1)		15.2(1.7)

33 was able to decrease mean TSS concentrations below the 2002 CCME threshold three out of six storm events.

Grain size distribution of suspended sediment at the inlet(s) and outlet of each pond for three storm events is shown in Figure 15. The data show that the grain size of bottom sediment of Pond 45 is much coarser than Pond 33. The grain size distribution becomes finer across Pond 33 from the sediment forebay to the constructed wetland. Table 11 presents the maximum

particle size along with the mean ± standard deviation D_{50} and D_{90} values from image analysis of SS from three representative storms. The samples collected at the inlet and outlet appears to be dominated by fine grained sediment (≤63 µm) because the D_{90} for all locations for each storm event is less than 63 µm.

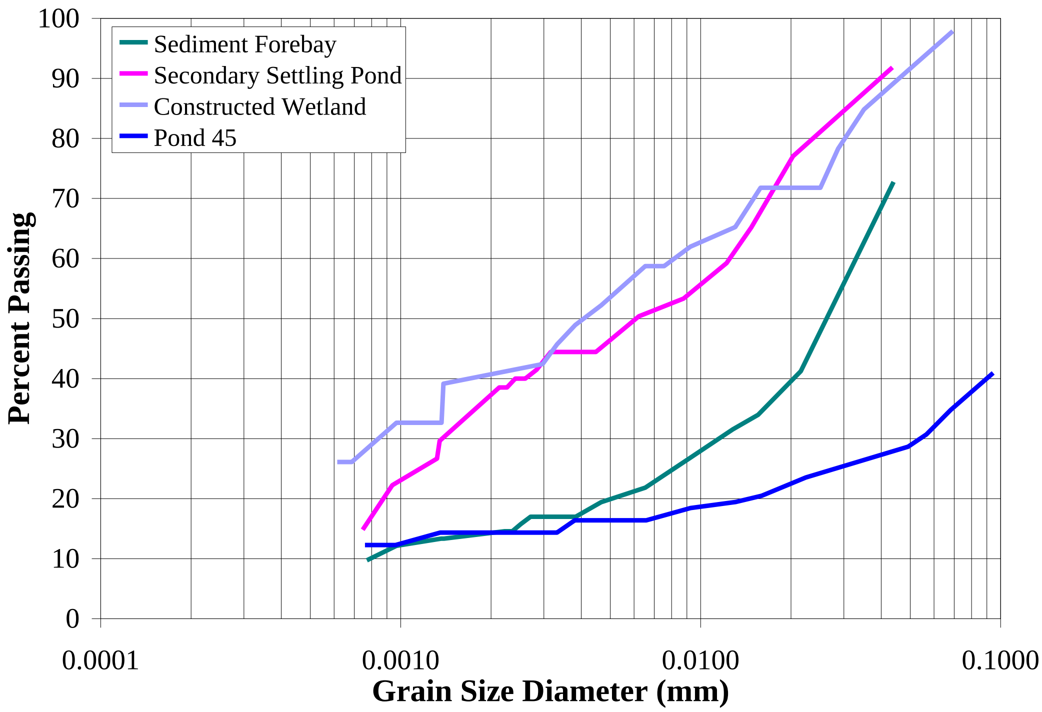


Figure 15. Grain size distribution of benthic sediment samples from two stormwater management ponds.

Grain size distributions of suspended sediment at the inlet(s) and outlet of Pond 33 and Pond 45 are presented in Figure 16 and Figure 17. Refer to Appendix B for GS distributions and representative photographs for Pond 33 and Pond 45 on JD 193 and JD 201. The grain size distributions from the inlet(s) and outlets of each pond are quite similar, however, representative photomicrographs do show differences in the nature of these materials. Inlet solids are finer grained and less flocculated compared to particles leaving the outlet of each pond. Many of the flocs formed within the pond include algae and bacteria (Figure 16 and Figure 17).

Representative grain size distributions from each location were chosen from the storm event on JD 212 to illustrate the importance of flocculation in particle transport dynamics within the water column (Figure 18). For the JD 212 event, the maximum particle size was 275 µm and constituted 23% of the total volume of sediment at the outlet of Pond 33. The largest particle at the inlet of Pond 33 was 165 µm which was 11% of the total volume. Particle sizes of 296 µm, 280 µm and 504 µm accounted for 17%, 38% and 44% of the total particle volume at the inlets 1 and 2 and outlet of Pond 45, respectively.

The major element composition of bottom sediment for Pond 33 and Pond 45 are presented in Table 12. The data show that Pond 33 has lower concentrations of SiO_2 and CaO but higher percentages of P, Al and Fe than Pond 45. The OM content was higher in Pond 33 as well as the concentrations of Al, Fe and Ca.

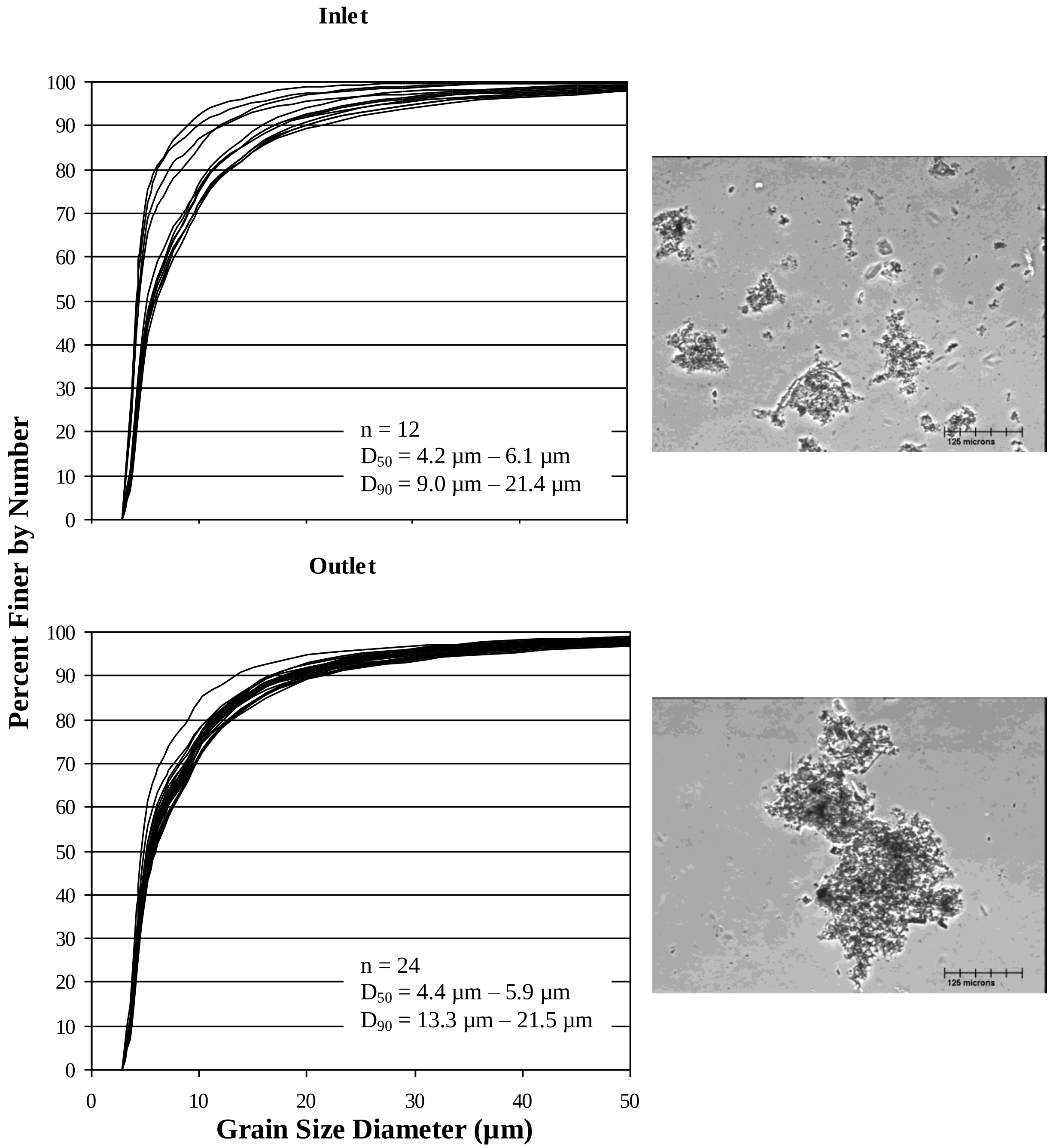


Figure 16. Grain size distribution of suspended sediment in Pond 33 during a storm event on JD 212 to 213. Scale = 125 microns for photographs.

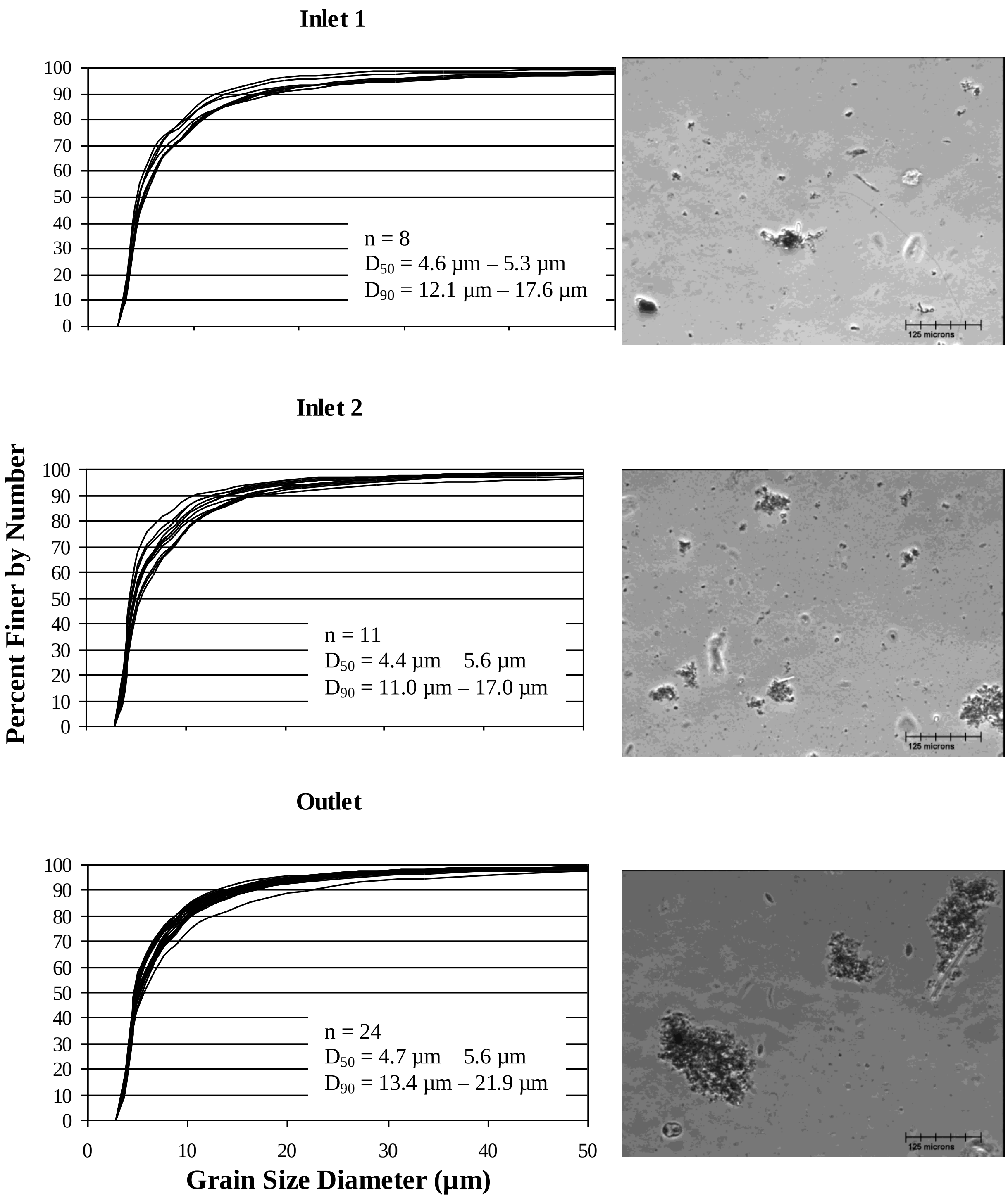
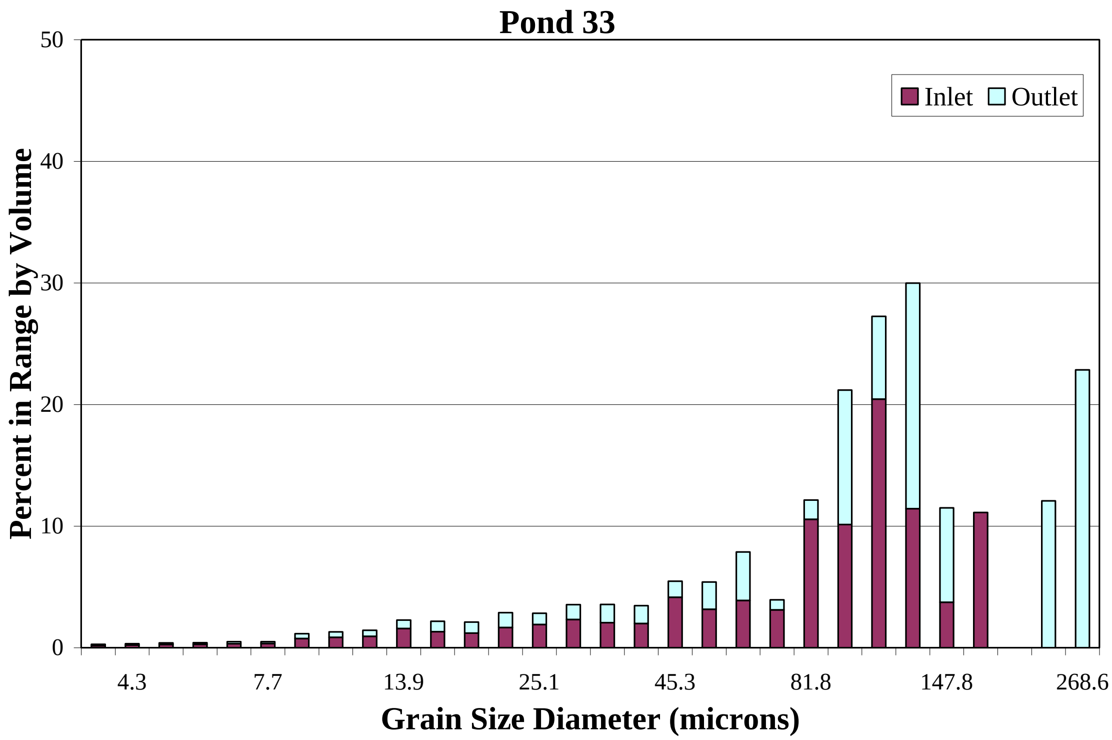


Figure 17. Grain size distribution of suspended sediment in Pond 45 during a storm event on JD 212 to 213. Scale = 125 microns for photographs.

Table 12. Major elemental composition of sediment samples from Pond 33 (sediment forebay, secondary settling pond and constructed wetland) and Pond 45

Constituent (%)	Sediment Forebay	Secondary Settling Pond	Constructed Wetland	Pond 45
SiO_2	50	45	43	66
Al_2O_3	8.01	11	11	9.8
Fe_2O_3	2.8	4.9	5.4	3.04
MnO	0.062	0.090	0.057	0.074
MgO	4.1	3.8	3.7	2.6
CaO	13	13	7.1	5.6
Na_2O	1.6	1.02	0.78	2.2
K_2O	1.7	2.4	2.4	2.2
TiO_2	0.48	0.61	0.60	0.50
P_2O_5	0.17	0.21	0.33	0.12
Cr_2O_3	0.010	0.020	0.020	0.010
LOI	18	18	26	6.8

The major mineralogical composition of pond sediment (Table 13) shows a decreasing percentage of quartz in Pond 33. Increasing clay content (chlorite and muscovite) from the sediment forebay to constructed wetland indicates a decrease in grain size which was also seen in Figure 15. Albite, a form of plagioclase feldspar, decreases from the inlet to outlet of Pond 33, while microcline increases. Pond 45 has a higher percentage of quartz than all three sections in Pond 33, which also supports grain size results from the hydrometer analysis (Figure 15). Pond 45 has less muscovite than the constructed wetland in Pond 33; however, it has over 2 times the percentage of chlorite within the composite sample than the constructed wetland in Pond 33.



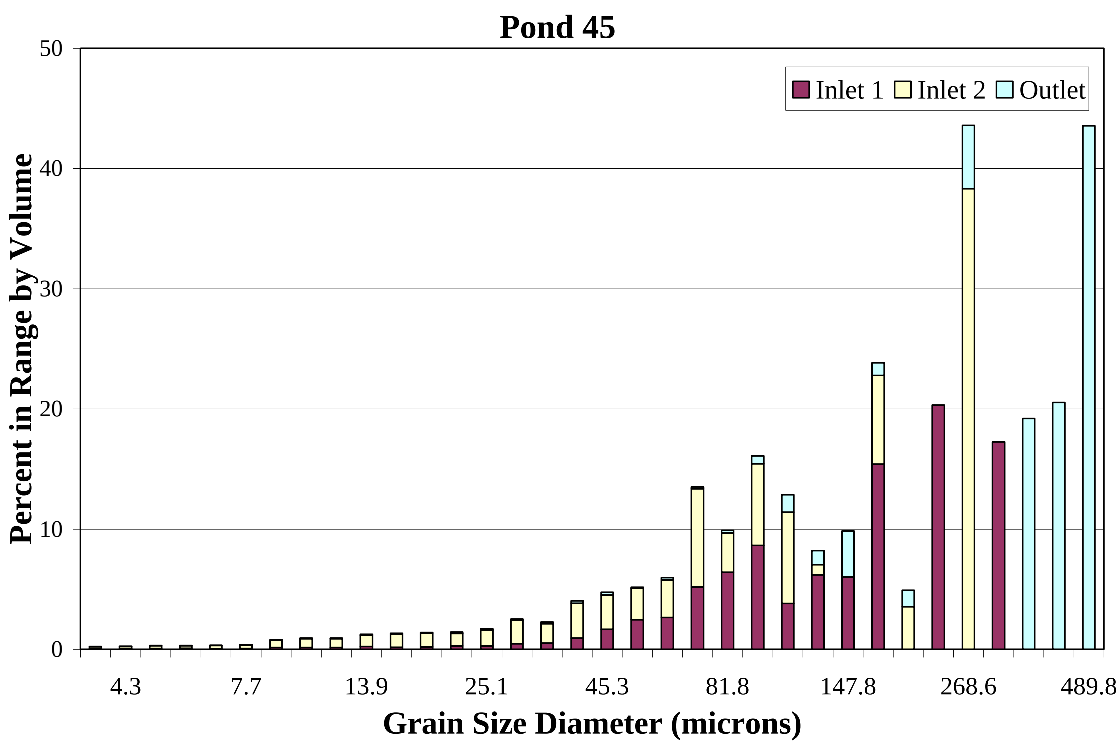


Figure 18. Volumetric distribution of particle size classes for a representative sample from the inlet(s) and outlet of two stormwater ponds on JD 212.

Table 13. Major mineralogical composition of sediment samples from Pond 33 (sediment forebay, secondary settling pond and constructed wetland) and Pond 45.

Constituent (%)	Sediment Forebay	Secondary Settling Pond	Constructed Wetland	Pond 45
Quartz	30	35	21	37
Dolomite	15	9	15	6
Calcite	6	12	6	1
Albite, calcian, ordered	20	14	12	18
Maximum Microcline	16	22	32	15
Tremolite	7	0	0	5
Chlorite	4	4	6	14
Muscovite	2	4	6	4
Zeolite	0	1	1	0

3.4. Phosphorus Buffering Capacity of SWM Pond Sediments

The P buffering capacity of sediment in Pond 33 and Pond 45 for two size fractions (≤63 µm and >63 µm) are shown in Figure 19. Grain size fractions ≤63 µm adsorbed more P sediment dry weight than the >63 µm fractions. However, at lower concentrations the amount of SRP adsorbed by both GS fractions was similar. According to the P sorption experiments, the EPC for Pond 45 is approximately 19 µg P L^{-1} and this corresponds to the outflow SRP concentrations from Figure 10 which fluctuates around this level.

In Pond 33, there is no difference in P sorption due to grain size at the sediment forebay and secondary settling pond. There is a shift in the amount of SRP adsorbed onto sediment from the sediment forebay to the constructed wetland. The sediment forebay adsorbed less SRP than the secondary settling pond. The constructed wetland was able to adsorb less SRP from the water column than the other compartments. For example, at an initial SRP concentration of 400.0 µg P L^{-1}, 18.0, 19.0 and 16.5 µg P g^{-1} sediment was adsorbed on the sediment forebay, secondary settling pond and constructed wetland, all ≤63 µm in size. The EPC for the constructed wetland is approximately around 50 µg L^{-1}, and this also corresponds to outlet SRP concentrations

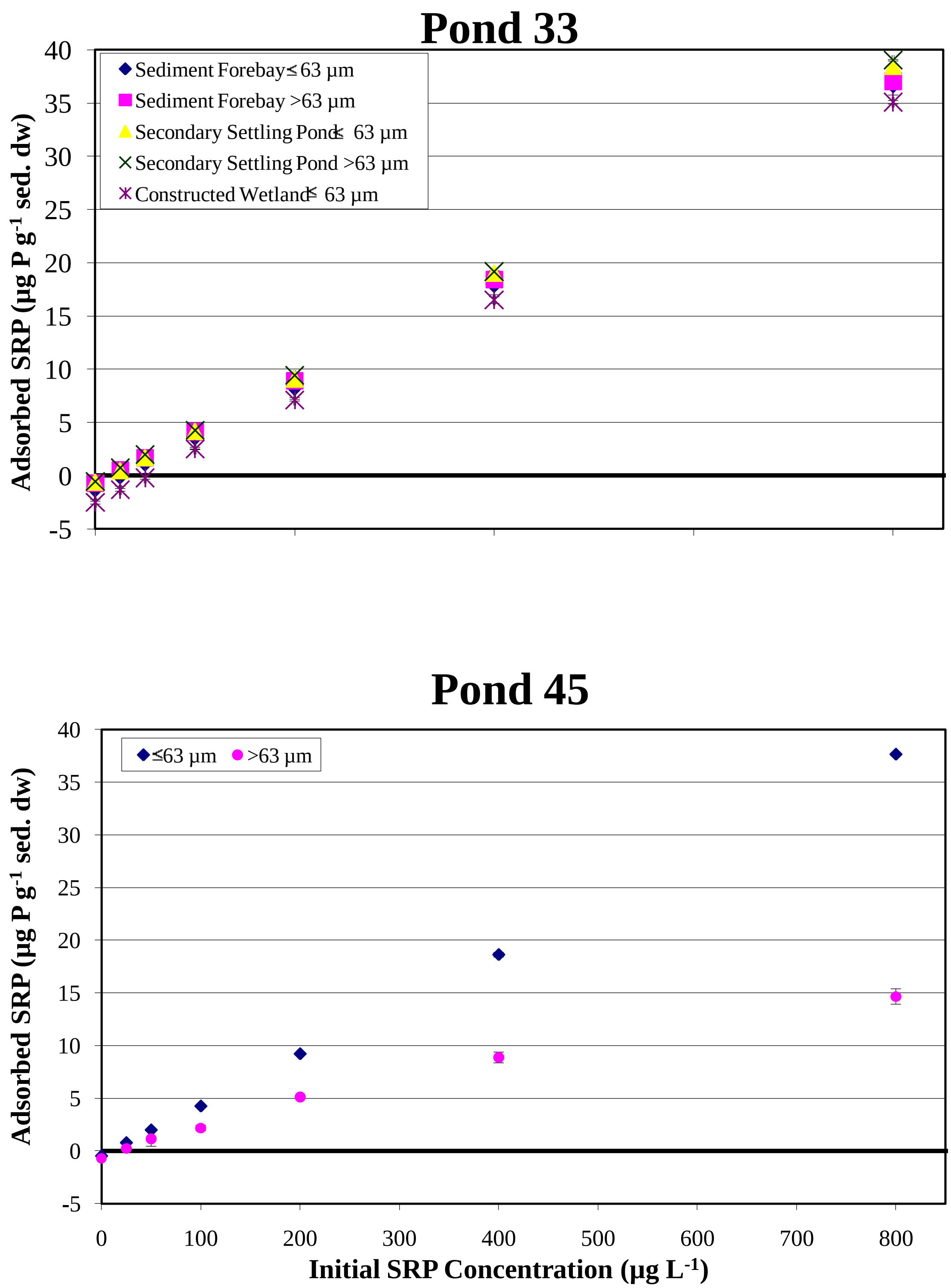


Figure 19. Mean phosphate sorption (µg P g^{-1} dry weight) of two bottom sediment size fractions (≤ 63 µm and >63 µm) in two stormwater management ponds. Standard error for each size class is less than 1 µm.

(Figure 8) measured during storm events.

3.5. Pond Trap Efficiency

Mean TP, SRP and TSS concentrations were used to determine the average trap efficiency for six storm events monitored during the field season. It was assumed that the inflow and outflow discharges were equal because of the lack of adequate flow data to calculate the change in fluxes. The TE for Pond 45 on JD 231 could not be calculated because of missing data from Inlet 1. Overall, Pond 45 performed better than Pond 33, the hybrid extended detention pond (Figure 20). Pond 45 retained over 75% and 90% of influent TP and SRP and over 90% of influent TSS. Pond 33 was a sink for TP three of the six events monitored, and retained less than 50% of influent TP for the other three storm events. Pond 33 was a source for SRP for four events and also retained less than 50% of influent SRP for events on JD 193 and JD 231. Total suspended solids retention was higher than TP and SRP. Four of the six events had a positive retention in the pond. Pond 33 was a source of TSS on JD 201 and JD 257 with a -53% and -31% retention. The TE for the stormflow samples from the entire field season for Pond 33 was -1.9%, -19.2% and 66.5% for TP, SRP, and TSS, respectively. Trap efficiencies of 92.2%, 95.8% and 98.5% for TP, SRP and TSS were calculated from all stormflow samples collected at the inlets and outlet of Pond 45. Combining SF and BF concentrations, the TE of Pond 33 increases to 24.3%, 26.7% and 66.8% for TP, SRP and TSS concentrations, respectively. The TE for Pond 45 increased slightly to 93.8% for TP and decreased to 94.2% and 98.0% for SRP and TSS concentrations when BF and SF samples were used to calculate TE.

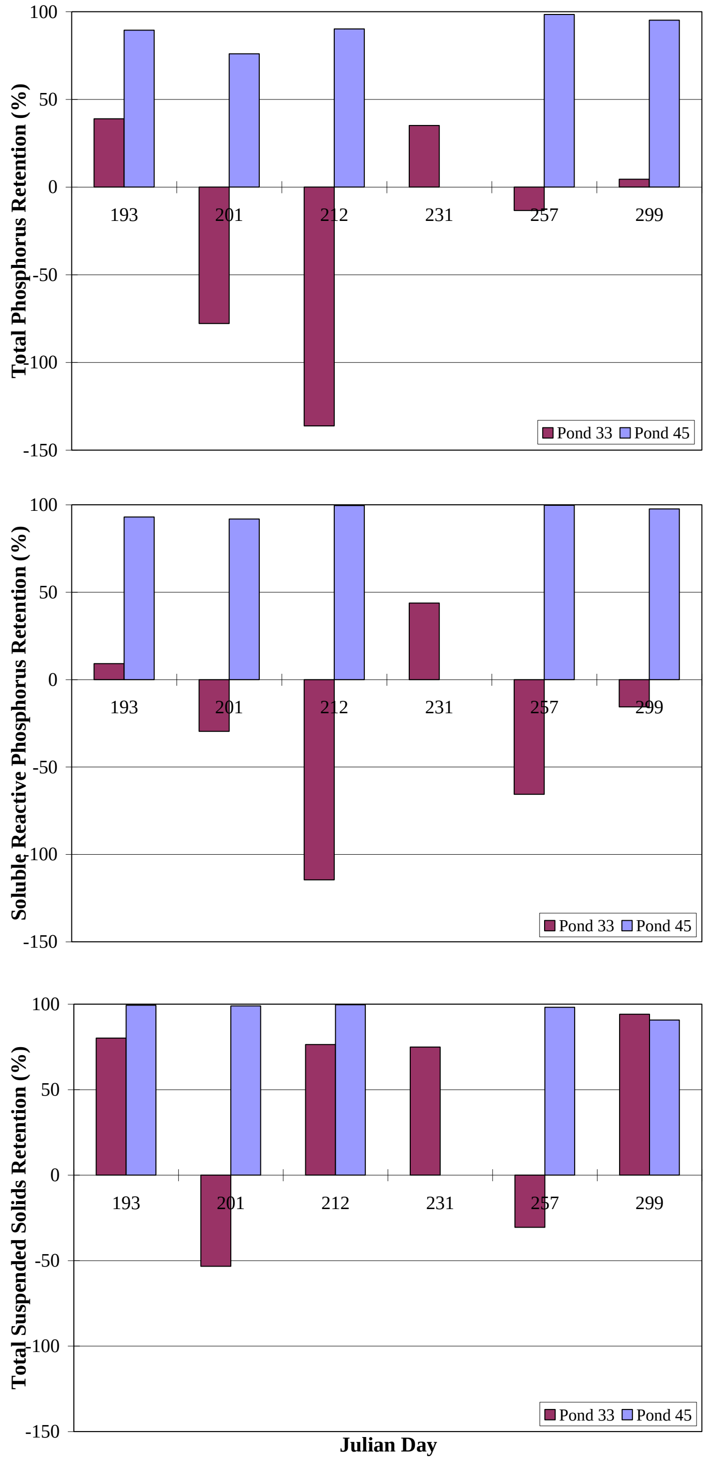


Figure 20. Trap efficiency of TP, SRP and TSS in two stormwater management ponds.

Chapter 4: Discussion

4.1. Introduction

There is an abundant literature set related to physical, chemical and biological processes that govern P mobilization, transport and speciation in freshwater and marine environments (Boström et al., 1988; Gächter and Meyer, 1993; Stone and English, 1993; Krishnappan, 2007). However, Watt and Marsalek (1994) report that few peer reviewed studies have been conducted to evaluate the performance of structurally different SWM ponds. These authors elucidate the need to conduct comprehensive monitoring programs that include measurements of chemical cycling, pond hydraulics and velocity distributions to better understand and model processes that influence their performance. Such information is required to improve pond design for water quality treatment.

The primary goal of this thesis is to conduct a comprehensive study of P cycling in a hybrid extended detention pond (Pond 33) and a conventional wet pond (Pond 45) located in Waterloo, Ontario. Trap efficiencies of three water quality parameters (TP, SRP and TSS) were evaluated and compared to evaluate hydrological and chemical processes that influence pond TE. An evaluation of the physical and chemical characteristics of the pond sediment (including measurements of sediment porewater P, sediment geochemistry and mineralogy, grain size distribution of suspended and bottom sediments) was conducted and the P buffering capacity of sediment in both ponds was assessed. In the following sections, results of the present study (summarized in Figure 21) are compared and discussed in the context of the published literature.

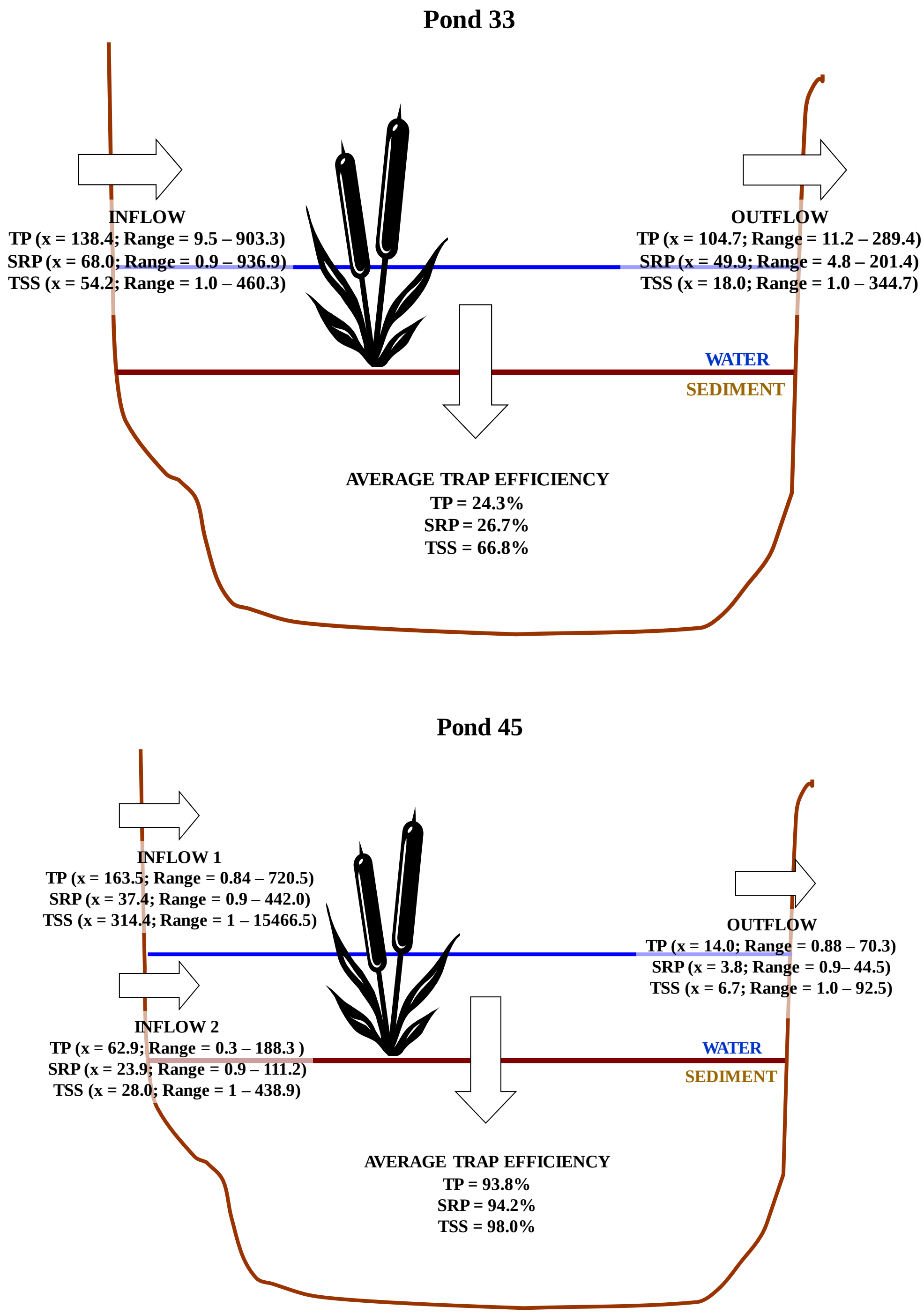


Figure 21. Average inflow and outflow concentration, range and trap efficiency for TP, SRP and TSS for two stormwater management ponds for baseflow and stormflow events. X represents the average concentration at the inlet(s) and outlet of the two stormwater management ponds.

4.2. Trap Efficiency of Pond 33 and Pond 45

Some previous studies have reported the TE of SW detention ponds for a range of water quality parameters such as TP, SRP and TSS (c.f. Van Buren et al., 1997; Borden et al., 1998; Gharabaghi et al., 2006; Yeh and Wu, 2009). The literature reports a wide range of trap efficiencies for TP, SRP and TSS. This variability ranges from -35% to 77%, -76% to 87% and -22% to >90% for TP, SRP and TSS respectively (Table 14). Gharabaghi et al. (2006) measured TSS retention in two wet ponds (Greensborough and Ballymore Ponds) located in Markham and Richmond Hill, ON, respectively. The TE of both ponds was >90%, however, the ponds were located in developing catchments where construction was still in progress. At the inlet of Ballymore Pond, the maximum SS concentration of 1500 mg L^{-1} was measured on JD 201 in 2004. Over 90% of influent SS were retained in the pond, however TSS concentrations of 250 mg L^{-1} were recorded at the outlet. Accordingly, while the reported TE for this pond was high (>90%) the effluent concentrations still exceed CCME guidelines. Inlet 1 at Pond 45 had TSS concentrations as high as the inlet concentrations at the Ballymore Pond reported by Gharabaghi et al. (2006). In the present study, neither catchment was experiencing major construction and the primary source of sediment and nutrients were from the established residential neighbourhoods. Olding et al. (2004) stated that catchments in Richmond Hill, ON took anywhere from one to five years to stabilize after construction activities ceased.

Mallin et al. (2002) measured the TE of two wet ponds located in residential areas and one other on a golf course. The golf course pond was a source of TSS, TP and SRP at the outlet. They attributed this to excessive fertilization on golf courses resulting in the pond not being able to treat the high loadings of nutrients. However, ponds treating runoff in the residential or commercial areas generally had lower TE for TP and SRP (Mallin et al., 2002). Only one SWM pond (McCarrons Wetland, Oberts and Osgood, 1991) in Table 14 had TE values for TP >75% , while four ponds retained over 50% of influent TP. Five ponds retained over 50% of influent SRP, while two ponds retained over 75%. The ability of a pond to remove a high percentage of influent TP depends on the internal processes, which are usually not accounted for in mass balance studies.

In the present study, there was a large difference in TE between the hybrid extended detention pond (Pond 33) and the conventional pond (Pond 45). Over the field season, Pond 33 was a sink for TP, SRP and TSS (24.3%, 26.7% and 66.8%). However, the pond was a source for TP and SRP (-1.9% and -19.2%) during storm events. A comparison of TE for TP, SRP and TSS in the literature and this thesis research is listed in Table 14. Reduction of influent TP and SRP concentrations in Pond 33 was near the minimum TE reported in the literature (Table 14). Total suspended solid concentration reduction in Pond 33 was above the average TSS reduction reported in Table 14. There was a significant difference (Mann Whitney U) between all stormflow samples from the inlet and outlet at Pond 33 for TP ($p = 0.01$), SRP ($p = 0.005$) and TSS ($p = >0.0001$). Pond 45 had large reductions of TP, SRP and TSS, with average reductions of 93.8%, 94.2% and 98.0% for the combined field season. Stormflow inlet and outlet samples for TP, SRP and TSS from Pond 45 during the field season displayed a significant difference (Mann Whitney U, $p = >0.0001$). The TE percentages for TP, SRP and TSS in Pond 45 are above all the TE percentages reported in Table 14. The TE can be influenced by several factors including influent pollutant concentrations, pond design, pond volume vs. catchment area, sediment mineralogy or sediment geochemistry (Driscoll, 1986 in Comings et al., 2000; Stone and English, 1993; Bradford and Gharabaghi, 2004; Gharabaghi et al., 2006).

Table 14. Summary of TSS, TP and SRP inlet and outlet concentrations and corresponding trap efficiencies.

Author	Location	Pond Name	Facility Type[1]	Parameter	Concentration[2] Inlet	Concentration[2] Outlet	Trap Efficiency[3] (%)
Mulroy, 2009	Waterloo, ON	Pond 33	HP	TSS	**70**	**24**	**67**
				TP	**95**	**97**	**-2**
				SRP	**36**	**43**	**-19**
		Pond 45	WP	TSS	**462 / 31[4]**	**7**	**99**
				TP	**135 / 73**	**16**	**92**
				SRP	**55 / 32**	**4**	**96**
Gharabaghi et al., 2006	Markham, ON	Greensborough	WP	TSS	nd[5]	nd	**>90**
		Ballymore	WP	TSS	nd	nd	**>90**
Van Buren et al. 1997	Kingston, ON	-	WP/DP	TSS	**212**	**73**	**65**
				TP	**216**	**84**	**61**
				SRP	**89**	**37**	**58**
Comings et al., 2000	Washington	Pond A	WP	TSS	**22.8**	**8.9**	**61**
				TP	**95**	**77**	**19**
				SRP	**15**	**14**	**7**
		Pond C	WP	TSS	**16.2**	**2.9**	**82**
				TP	**87**	**45**	**48**
				SRP	**26**	**10**	**62**
Oberts and Osgood 1991	Minnesota	McCarrons Wetland	HP	TSS	nd	nd	**96**
				TP	nd	nd	**77**
				SRP	nd	nd	**48**

[1]CW = constructed wetland; WP = wet pond; HP = hybrid pond and DP = dry pond

[2]Concentration Units: TSS = mg L^{-1}; TP/SRP - µg L^{-1}

[3]Trap efficiency = $1-(C_{out}/C_{in}) * 100\%$

[4]values from inlet 1 and inlet 2 of Pond 45

[5]nd = data not available

Table 15 cont. Summary of TSS, TP and SRP inlet and outlet concentrations and corresponding trap efficiencies.

Author	Location	Pond Name	Facility Type[1]	Parameter	Concentration[2] Inlet	Concentration[2] Outlet	Trap Efficiency[3] (%)
Borden et al. 1998	North Carolina	Piedmont Pond	WP	TSS	**61**	**49**	**20**
				TP	**160**	**100**	**38**
				SRP	**33**	**27**	**18**
		Davis Pond	WP	TSS	**97**	**39**	**60**
				TP	**360**	**210**	**42**
				SRP	**220**	**100**	**55**
Mallin et al. 2002	North Carolina	Ann McCrary	WP	TSS	**10.5**	**3.7**	**65**
				TP	**61**	**47**	**23**
				SRP	**28**	**26**	**7**
		Silver Stream	WP	TSS	**4.3**	**5.9**	**-37**
				TP	**141**	**60**	**57**
				SRP	**100**	**23**	**77**
		Echo Farms Country Club	WP	TSS	**3.6**	**4.4**	**-22**
				TP	**51**	**69**	**-35**
				SRP	**25**	**44**	**-76**
Line et al. 2008	North Carolina	CMS	CW	TSS	nd	nd	**64**
				TP	nd	nd	**43**
		UNC	CW	TSS	nd	nd	**83**
				TP	nd	nd	**52**
Yeh and Wu 2009	Taiwan	-	HP	TSS	**12.42**	**1.65**	**87**
				TP	**1750**	**1170**	**33**
				SRP	**1070**	**960**	**10**
Birch et al. 2004	Australia	Mean of RA-RF	CW	TSS	nd	nd	**-4**
				TP	nd	nd	**12**

4.3. Temporal and Spatial Variability of Phosphorus and Suspended Sediment Concentrations

Temporal and spatial variability of TP, SRP and TSS in SW ponds are a function of pond design, climate, time of year and pollutant delivery to the inlet, which is controlled by rainfall intensity and duration (Vaze and Chiew, 2003; Gharabaghi et al., 2006), antecedent moisture conditions (Black, 1997) and sediment availability (Stone, personal communication). The temporal and spatial variability of flow and TSS concentrations for the Ballymore pond in Markham, ON on July 19, 2004 and September 9, 2004 was investigated by Gharabaghi et al. (2006). The hyetographs and hydrographs for two sampling dates show a larger increase in discharge at the inlet compared to the outlet, indicating the pond is retaining incoming runoff and releasing it at a slower rate. This was also apparent in Pond 33 and Pond 45 for the current study. Water level fluctuations in response to precipitation at the inlet(s) and outlet of Pond 33 and Pond 45 are presented in Figure 6 and Appendix B. The outlet controls used at Pond 33 and Pond 45 were able to reduce the peak in the hydrograph and allow water to gradually leave the ponds. The Manning discharge equation is commonly used to calculate discharge for open channels however, Flowlink 5 software makes the following assumptions:

1) There is no sudden change in flow.

2) The channel is round.

3) Two or more streams or pipes do not combine into one.

4) The flow does not enter and exit at different angles.

Poor results will be calculated if the sampling location violates any of these assumptions (ISCO, 2006). The inlet at Pond 33 and Inlet 2 at Pond 45 had a sudden change in flow as they were both submerged therefore the Manning equation cannot be applied. Accordingly, Pond 33 had a 1500 mm and 1650 mm pipe flowing into a 2400 mm inlet pipe which would have changed the flow characteristics. With the two ponds violating a couple of the software assumptions of the Manning equation, level measurements could not be used to calculate flow.

The OMOE (2003) recommends detention times greater than 24 hours. However, Burge and Breen (2006) stated that temperature stratification can result from stagnant water as well as algal growth; therefore caution should be used when designing ponds with long detention times. Pond 33 and Pond 45 were designed to retain influent runoff for approximately 24 and 48 hours, respectively and therefore provide enough time to treat runoff as per design specifications of the MOE (CCL, 1996; PEIL, 1997; OMOE, 2003).

The temporal variability of TP, SRP and TSS concentrations at Pond 33 and Pond 45 during three storm events of varying magnitudes are presented in Figure 8, Figure 9, Figure 13 and Figure 14. Differences between inlet and outlet concentrations for each pond and between the inlets and outlets of the ponds were determined by the Mann Whitney U statistical test (Table 15). Significant differences were only measured on JD 212 for TP ($p = >0.0001$) and SRP ($p = 0.049$) in Pond 33 (Table 15). Outlet TP and SRP concentrations at Pond 33 for all three events were around 100 µg L^{-1} and 50 µg L^{-1}, respectively. The size of the storm event did not influence outlet TP and SRP concentrations, however, internal processes including sediment buffering of P may influence outlet concentrations at Pond 33. Significant differences between inlet and outlet TSS concentrations at Pond 33 occurred on JD 299 ($p = >0.0001$) and JD 193 ($p = >0.0001$). Inlet concentrations at Pond 33 on JD 299 and JD 193 showed increases in TSS concentrations as the storm intensity increased (Figure 13). On JD 212, TSS sediment exhaustion occurred after the first peak in storm intensity (Figure 13). Total suspended solid concentrations at 18:00 were below outlet TSS concentrations (Figure 13). Outlet TSS concentrations were less variable and

Table 15. Mann-Whitney U comparison for TP, SRP and TSS concentrations for three storm events of different magnitudes. P-values significantly different at >0.05 are listed in bold.

		JD 299	JD 212	JD193
TP	Pond 33 Inlet vs. Outlet	0.928	**>0.0001**	0.092
	Pond 45 Inlet vs. Outlet	**>0.0001**	**>0.0001**	**>0.0001**
	Inlet vs. Inlet(s)	0.204	**0.012**	0.281
	Outlet vs. Outlet	**>0.0001**	**>0.0001**	**>0.0001**
SRP	Pond 33 Inlet vs. Outlet	0.158	**0.049**	0.766
	Pond 45 Inlet vs. Outlet	**>0.0001**	**0.001**	**>0.0001**
	Inlet vs. Inlet(s)	0.121	0.083	0.955
	Outlet vs. Outlet	**>0.0001**	**>0.0001**	**>0.0001**
TSS	Pond 33 Inlet vs. Outlet	**>0.0001**	0.084	**>0.0001**
	Pond 45 Inlet vs. Outlet	**>0.0001**	**>0.0001**	**>0.0001**
	Inlet vs. Inlet(s)	**0.002**	0.116	0.56
	Outlet vs. Outlet	0.838	**>0.0001**	**>0.0001**

generally <25 mg L^{-1} for all three storm events (Figure 13). Significant differences between inlet and outlet TP, SRP and TSS concentrations in Pond 45 were measured for storm events on JD 299, JD 212 and JD 193 (Table 15). Storm magnitude did not have an effect on the TE of TP, SRP and TSS concentrations at Pond 45 (Figure 9). Pond 45 has a detention time of 48 hours (PEIL, 2006). The additional 24 hours of detention could allow internal processes reduce influent TP, SRP and TSS concentrations.

Concentrations of TP, SRP and TSS were significantly different (p = >0.0001) at the outlets of Pond 33 and Pond 45 during all storm events with the exception of TSS concentrations sampled on JD 299 (Table 15). The observed significant differences are related to the hydraulic and hydrological treatment processes occurring in each pond. Pond 45 was able to reduce inflow concentrations of TP, SRP and TSS better than Pond 33. These differences would not be found if

Pond 33 was capable of reducing outflow concentrations below PWQO and CCME guidelines. Significant differences between the inlets of Pond 33 and Pond 45 were found for TP concentrations ($p = 0.012$) on JD 212 and for TSS concentrations ($p = 0.0002$) on JD 299 (Table 15). Both ponds treat runoff from residential areas although the footprint at each sewershed is different. The amount of runoff and subsequent P and SS loading is dependent on sewershed characteristics such as drainage area and land use. The drainage area of Pond 33 is approximately 50 ha more than Pond 45 while Pond 33 is 30% smaller by volume than Pond 45. Differences in inflow TP, SRP and TSS concentrations would be because of the larger contributing area in Pond 33.

Four storm events were selected during the field season to investigate the spatial and temporal variability of inflow and outflow TP and SRP concentrations at Pond 33 and Pond 45. It was anticipated that TP and SRP concentrations would begin to increase once the *Typha sp.* began to senesce. This was not evident from the individual storm events analyzed (Appendix C). Significant differences were analyzed using the Mann Whitney U test for TP and SRP concentrations at the inlets and outlets of Pond 33 and Pond 45 for storm events on JD 201, JD 231, JD 257 and JD 299. Inlet(s) and outlet concentrations of TP were significantly different on JD 231 ($p = 0.002$) and JD 257 ($p = >0.0001$) for Pond 33 and for all four events ($p = >0.0001$) for Pond 45 (Table 16). Concentrations of SRP were significantly different on JD 257 ($p = 0.003$) for Pond 33 and on JD 231, JD 257 and JD 299 ($p = >0.0001$) for Pond 45 (Table 16). Baseflow inlet and outlet TP and SRP samples were significantly different ($p = >0.0001$) for Pond 33. Pond 45 inlet and outlet BF samples were also significantly different for TP ($p = >0.0001$) and SRP ($p = 0.010$) concentrations. The Mann Whitney U test results and Appendix C indicate that Pond 33 is less likely to reduce TP and SRP concentrations to the same degree as

Pond 45. Storm magnitude, drainage area, pond volume, detention time and sediment characteristics could be larger factors than seasonality for TP and SRP concentration reduction in a pond for individual storm events.

Table 16. Mann-Whitney U comparison for TP, SRP and TSS concentrations for four storm events sampled throughout the field season. P-values >0.05 are significantly different (bolded).

		JD 201	JD 231	JD 257	JD 299
TP	Pond 33 Inlet vs. Outlet	0.016	**0.002**	**>0.0001**	0.928
	Pond 45 Inlet vs. Outlet	**>0.0001**	**>0.0001**	**>0.0001**	**>0.0001**
	Inlet vs. Inlet(s)	**>0.0001**	**>0.0001**	**0.022**	0.204
	Outlet vs. Outlet	**>0.0001**	**>0.0001**	**>0.0001**	**>0.0001**
SRP	Pond 33 Inlet vs. Outlet	0.545	>0.129	**0.003**	0.158
	Pond 45 Inlet vs. Outlet	0.128	**>0.0001**	**>0.0001**	**>0.0001**
	Inlet vs. Inlet(s)	**>0.0001**	**>0.0001**	**0.012**	0.121
	Outlet vs. Outlet	**>0.0001**	**>0.0001**	**>0.0001**	**>0.0001**

4.4. Biological Processes and Succession in the Constructed Wetland

4.4.1. Phosphorus Assimilation and Wetland Succession

Stormwater runoff treatment is a function of pond design and physical (Boström et al., 1988; De Boer and Stone, 1999; Krishnappan et al., 1999; Stephan et al., 2005), chemical (Boström et al., 1988; Broberg and Persson, 1988; Stone and Mudroch, 1989; Cooke et al., 1993; Søndergaard et al., 2003) and biological (Reddy and Debusk, 1987; Gumbricht, 1993; Cronk and Fennessy, 2001) processes occurring within the pond. Constructed wetlands can increase sedimentation rates because plants provide quiescent conditions (Jenkins and Greenway, 2005). Specific aquatic plants, including *Typha sp.*, are used for constructed wetlands because of their high potential to assimilate nutrients from sediment porewater and the water column during the early phases of the plants life cycle (Gumbricht, 1993; Brix, 1994; Weng et al., 2006; Yeh and Wu, 2009). However, during senescence aquatic plants, such as *Typha sp.*, release P back to the water column and soil porewater (Nichols, 1983), thus negating the initial reduction of P in pond effluent.

In temperate environments, *Typha sp*. generally arrive at their peak growth period around the middle of August (Bonneville et al., 2008) and senescence usually occurs between September and November (Bayly and O'Neill, 1972). In the present study, there was a gradual increase in TP concentrations between JD 230 and JD 257 at the outlet of Pond 33, which may be attributed in part to *Typha sp.* senescence and subsequent release of excess nutrients (Figure 10). Soluble reactive P concentrations also showed an increase of approximately two to four times normal SRP concentrations between JD 230 and JD 257 (Figure 10). Plant biomass was not measured nor analyzed for P. Accordingly, more research is required to relate P mobilization to *Typha sp.* senescence in SWM ponds.

Many studies have investigated P retention in constructed wetlands on a short time scale, e.g. 5 years or less. The TE of a hybrid pond in Richmond Hill was measured after major reconstruction of the pond (TRCA, 2002) and satisfactory removal of TP and SRP were observed over the two year study period. In contrast, other studies show that P retention in wetlands can decrease during the succession of wetland plants (Richardson, 1985; Craft, 1997, Mitsch et al., 2005). Craft (1997) stated that P retention is more pronounced in the first few years of succession because plants are being established in the wetlands and bottom sediment still has available sorption sites. Consequently, when sediment sorption sites become more saturated and less P is assimilated by plants, retention of P in the wetland is decreased (Richardson, 1985; Craft, 1997). Throughout succession, aerobic conditions in the wetland can become anaerobic, therefore decreasing decomposition of organic matter (Craft, 1997) and allowing SRP to desorb from sediment surfaces to the water column (Lijklema, 1980; Peng et al., 2007).

Harvesting aquatic plants before senescence has been suggested as a management techniques to permanently remove P from a treatment systems (Richardson, 1985; Oberts and Osgood, 1991; Headley et al., 2003). Headley et al. (2003) conducted a phosphorus mass balance study in four identical reed beds in Alstonville, Australia. They found that *Phragmites australis* had three growth stages beginning in the spring where the plants grew rapidly with the help of P reserves in their rhizomes from the previous fall. Above ground plant nutrient requirements were supplied by influent P sources during the summer and upon the arrival of cooler fall and winter temperatures P was translocated to their rhizomes (Headley et al., 2003). Headley et al (2003) suggested harvesting wetland plants after their peak growth period before senescence to maximize the amount of P removed from the system. Harvesting aquatic plants during the peak growth period could also increase P uptake into below ground biomass because the plants do not have the required amount of P reserve for shoot growth the next year (Headley et al., 2003). Davies and Cottingham (1993) suggested that aquatic plants will provide zero P removal if they are not harvested because the system will be in equilibrium between assimilation and release. However, Vymazal (2007) stated that harvesting plants will remove low amounts of P in highly loaded wetlands, but could make a difference in lightly loaded (<10-20 g P m^{-2} yr^{-1}) wetlands. The time required to remove the above ground biomass from constructed wetlands each year should be weighed against the amount of P removed from the system. If the receiving water body is sensitive to P inputs, this management technique might be beneficial. However, less sensitive water bodies to P loading could require different P management techniques such as rip rap at the outlet to increase the DO concentration of the effluent or a sorption medium that effluent water passes through after leaving the pond.

4.4.2. Phosphorus in Sediment Porewater

Many wetland plants translocate O_2 from the atmosphere or water column (Kennedy and Mayer, 2002) to their roots via aerenchyma (Cronk and Fennessy, 2001). Diffusion of O_2 from plant roots to sediment porewater can result in the formation of a zone of oxidized sediment near the sediment-water interface (Brix, 1997; Cronk and Fennessy, 2001); a process known as radial oxygen loss (Copeland Michaud and Richardson, 1989). Moore et al. (1994) studied the effect of an emergent macrophyte, *Menyanthestrifoliata L.* on sediment porewater chemistry in Silver Lake, Washington. They found *Menyanthestrifoliata L.* roots created a surficial O_2 boundary layer where SRP, total Fe, Fe^{2+} and total Mn were oxidized. In the present study, sediment porewater concentrations were lower in the constructed wetland than the other two compartments in Pond 33. Possible reasons for this observation include radial oxygen loss from *Typha sp.* located in the constructed wetland, grain size distribution and sediment geochemistry and mineralogy. With respect to redox potential in sediment porewater, lower dissolved oxygen content correlates to a lower redox potential (Kadlec and Knight, 1996). The constructed wetland had a slightly higher redox potential (38.4 mV) compared to the sediment forebay (14.4 mV) and secondary settling pond (22.5 mV), which could be a result of radial oxygen loss from *Typha sp*. Higher redox values were also measured by Moore et al. (1994) in sediment porewater with *Menyanthestrifoliata* L. plants compared to sediment porewater without macrophyte growth. Redox potential measurements from both ponds ranged from 14.4 mV to 47.2 mV. These measurements were taken in early October where cooler water column temperatures (approximately 10ºC or lower) and higher DO concentrations (>6.0 mg L^{-1}) could have influenced redox potentials within both ponds. Phosphorus release from sediment occurs when redox potentials fall below +200 mV (Wetzel, 1983; Boström et al., 1988). A larger range of redox potential in the summer would most likely occur because of higher water column temperatures and lower DO concentrations, thus causing greater amounts of P release from the sediment-water interface.

In Pond 33, maximum SRP concentrations in porewater ranged from 563.9 µg L^{-1}, 384.7 µg L^{-1} and 126.6 µg L^{-1} in the sediment forebay, secondary settling pond and constructed wetland, respectively. Porewater SRP concentrations were inversely related to the density of *Typha sp.* in the compartment. The constructed wetland had the highest density of macrophyte plantings, while the secondary settling pond had *Typha sp.* around the edge of the pond and *Nymphaeaceae* (water lily) in the open water section of the compartment. In the sediment forebay, a small number of *Typha sp.* were observed. The oxidized boundary from radial oxygen loss below the sediment-water interface in the constructed wetland may have caused a decrease in SRP porewater concentrations. However, O_2 concentrations in the water column were low in the constructed wetland from July (3.1 mg L^{-1}) to September (4.4 mg L^{-1}). The development of an oxygenated microzone at the sediment-water interface most likely did not occur during this time. Soluble reactive P in sediment porewater in the constructed wetland can diffuse upward into the water column where SRP concentrations ranged from 50 to 100 µg L^{-1}. Inlet DO levels were high (BF mean = 7.6 mg L^{-1}). However, an oxygenated microzone might not have been present during the entire field season because of increased biological activity due to higher water column temperature in the summer (Lijklema, 1980). Breaking or dissolving the microzone can allow porewater with high SRP concentrations to mix with the water column (Lijklema, 1980). Bioturbation and ebullition can also increase the rate and magnitude of SRP entering the water column by disturbing the sediment-water interface in the pond (Boström et al., 1988; Jansson et al., 1988; Reddy et al., 1999).

The SRP concentrations within sediment porewater profiles were very low in Pond 45, peaking approximately around 15μg L^{-1} and 25 μg L^{-1} at the north side of the pond and Inlet 2, respectively. The concentrations of SRP began to increase around 10 cm below the sediment-water interface and this could indicate that the zone of anoxia is below this depth and not captured by the PWP. Pond 45 was also covered with a rooted submerged aquatic plant, possibly *Eurasian Milfoil*, which might be able to assimilate P from sediment porewater. *Eurasian Milfoil* is not native to Ontario but has been able to propagate with ease in surface waters in Ontario. The introduction of *Eurasian Milfoil* to Pond 45 could have occurred during a large storm event when Clair Creek and the pond outlet are hydraulically connected.

4.5. Grain Size Characteristics

Greb and Bannerman (1997) investigated particle size distribution at the inlet and outlet of a wet pond in Madison, Wisconsin. The pond was designed to retain sediment >10 µm in size and it retained on average 87% of influent material. The pond removed the majority of larger particles (sand and some silt); however, it was not as effective at removing clay particles. The relative proportion of clay from the inlet to the outlet increased from 35.8% to 72.3%, respectively. Greb and Bannerman (1997) stated that the removal efficiency in detention ponds is dependent on flow and influent grain size distributions. Higher percentages of clay in influent SW results in a lower TE compared to sediment with a greater proportion of sand (Greb and Bannerman, 1997). Failing to remove a high percentage of fine grained (<63 µm) material in SWM ponds is a problem for effective management of stormwater. Clay particles stay in suspension (Greb and Bannerman, 1997) longer than sand and silt because of their lower density and related settling velocities and therefore require a detention time >24 hours. Constructed wetlands can be designed to increase sedimentation by using vegetation and enhanced pond design to reduce turbulence and promote settling in a SWM pond by creating quiescent conditions (Nichols, 1983; Borden et al., 1998; Bavor et al., 2001; Stephan et al., 2005).

Grain size distributions were analyzed from samples taken at the inlet(s) and outlet of Pond 33 and Pond 45 for three storm events of varying magnitude. D_{50} and D_{90} particle sizes from the inlet(s) to the outlet of both ponds did not show a significant difference in sediment size class except on Julian Day 202. Mean inlet and outlet D_{50} and D_{90} values are very similar to each other for both ponds (Table 11). These results indicate that the grain size distribution between the inlet and outlet of both ponds did not exhibit a clear fining of suspended sediment grain size. Representative images of suspended solids show changes in particle morphology between the inlet(s) and outlet of each pond (Figure 16 and Figure 17). Outlet distributions consisted of large flocs bound by extra cellular polymeric substances (Krishnappan et al., 1999), while inlet particles were smaller and less flocculated. Flocs of this nature are less dense and stay in suspension longer than individual particles between 5 and 15 µm (Krishnappan et al. 1999). The majority of sediment pollutant load is transported by sediment size fractions $\leq$63 µm (Stone and Mudroch, 1989; Krishnappan and Marsalek, 2002). Stone and English (1993) found that sediment size fractions <8 µm collected from two Lake Erie tributaries had the highest percentage of NAIP, which is the most bioavailable form of PP. If a SWM pond does not retain fine grained sediment ($\leq$63 µm), especially sediment <8 µm, algae could proliferate downstream because of an increase in bioavailable P in pond effluent. Changes to pond design might not enhance sedimentation of fine particulate matter. Sedimentation of silt and clay could take days and retaining water for this length of time might cause unfavourable conditions in the pond such as temperature stratification (Burge and Breen, 2006).

A second consideration to explain the grain size distribution at Pond 33 and Pond 45 is the sampling methodology used. Inlet 1 from Pond 45 had consistently high TSS concentrations because of large pebbles collected by the ISCO automatic sampler. Inlet 2 was submerged and the sampling intake tube was approximately 35 cm above the bottom of the sewer and was likely not capable of sampling larger particles because the runoff velocity was not high enough to keep the larger particles in suspension at a height above 35 cm. The concentration of TSS and GS distribution of particles could be skewed because of sampling tube installation and sewer inlet design. Also, the methodology used to evaluate GS distribution may have skewed GS results. Each sample was carefully inverted to homogenize the distribution and a known volume of sample was poured into a cylinder for filtration. Larger particles would fall out of suspension in the ISCO bottles before a complete sample was poured into the cylinder. This could skew inlet GS distributions to appear finer, however, the outlets of both ponds consisted of fine grained material and are likely representative of actual outlet conditions.

4.6. Geochemical and Mineralogical Sediment Characteristics

The geochemical and mineralogical composition of bottom sediment samples collected from the sediment forebay, secondary settling pond and constructed wetland in Pond 33 and a composite sample from Pond 45 provide valuable information on the effectiveness of each pond to adsorb P from the water column. The hydrometer GS data indicate that there was gradation of coarse to fine sediment from the sediment forebay to the constructed wetland in Pond 33, while Pond 45 had a coarser grained sediment distribution than Pond 33 (Figure 15). Sediment composition indicates there is a lower percentage of quartz (SiO_2) in the constructed wetland than in the other two compartments of Pond 33 and Pond 45 has a higher percentage of SiO_2 than the individual compartments in Pond 33 (Table 13).

Stone and Mudroch (1989) analyzed the mineralogical composition of four particle size fractions (500 µm, 67 µm, 16 µm and ≤ 13 µm) from Big Creek and Big Otter Creek in Ontario. They found that the percentage of quartz decreased with decreasing grain size but was similar between the 16 µm and <13 µm size fractions. They also reported that the percentage of chlorite and zeolites generally increased with decreasing grain size (Stone and Mudroch, 1989). In the present study, Pond 33 had an increase in chlorite and zeolite but SiO_2 decreased (Table 13) with decreasing grain size (Figure 15) from the sediment forebay to the constructed wetland. Pond 45 had a higher percentage of chlorite than Pond 33, but lower percentages of other minerals (Table 13). Clay minerals, organic matter and Fe- and Al-oxides have a greater affinity to sorb P from the water column than quartz, amphiboles and feldspars (Stone and Mudroch, 1989). However, a dedicated clay analysis is required to quantify the type and amount of clay minerals present in the four sediment samples (ActLabs, 2009). With the right conditions, Pond 33 appears to have more favourable sediment mineralogy than Pond 45 to adsorb P, however, low redox potentials could lead to desorption of P from sediment (Lijklema, 1980; Peng et al., 2007).

Pond 33 has a decreasing percentage of silica (Si) from the sediment forebay to the constructed wetland (Table 12) which compares well to XRD and hydrometer results indicating a decrease in quartz and grain size distribution in the three compartments within Pond 33 (Table 13 and Figure 15). Pond 45 has a higher percentage of Si than Pond 33 (Table 12), which agrees with the results from XRD analysis (Table 13) and grain size distribution from the hydrometer analysis (Figure 15) indicating that Pond 45 has coarser grained material at the locations sampled. Pond 33 has increasing percentages of Fe and Al from the sediment forebay to the constructed wetland, which was higher than the composite sample from Pond 45 (Table 12). Fe and Al are most likely associated with clay minerals such as illite, kaolinite and pyrophylite

(Stone and Mudroch, 1989). Clay minerals have a high cation exchange capacity resulting in a higher affinity to adsorb P (Hillel, 1998). Percentages of Ca measured in the constructed wetland are lower than the sediment forebay and secondary settling pond because the constructed wetland had more organic matter and fine grained material, which could be associated with clay minerals. Stone and English (1993) stated that Fe, Al, Mn and organic C content is inversely proportional to grain size, therefore with decreasing grain size there is an increase in those elements.

Iron, Al and Ca are the main elements influencing adsorption and desorption of P in sediment (Lijklema, 1980; Boström et al., 1988; Stumm and Morgan, 1996; Hongve, 1997). Iron is extremely labile, and changes in redox, pH and temperature could cause Fe^{3+} to desorb from sediment to Fe^{2+} and release any P previously adsorbed onto it (Lijklema, 1980; Lake et al., 2007; Peng et al., 2007). Aluminum not incorporated into the crystalline structure can be influenced by changes in pH and also release adsorbed P to the water column (Cooke et al., 1993). Carbonates are not as reactive to minor changes in redox and pH, therefore will not release as much P to the water column (Stumm and Morgan, 1996). However, if there were equal proportions of Fe, Al and Ca in bottom sediment, P would bind with Fe^{3+} and Al^{3+} before Ca^{2+} because of their higher valency and 'pull' to stabilize their charges. A higher proportion of Al and Fe than Ca in the constructed wetland could indicate that when anoxic conditions are present P will be released from the sediment water interface to the water column. However, if aerobic conditions dominated the constructed wetland, P would be adsorbed by Fe^{2+} and Al^{3+} and not transported downstream. Dissolved oxygen measurements taken at Inlet 1, Inlet 2 and the outlet of Pond 45 were on average 6.7 mg L^{-1}, 7.6 mg L^{-1} and 6.8 mg L^{-1}, respectively. Even with lower percentages of Fe and Al, Pond 45 might have a greater ability to keep P adsorbed onto sediment surfaces because of favourable physical and chemical conditions in the water column.

Pond 45 has less than half the OM content of the sediment forebay and secondary settling pond and about a third the amount of the constructed wetland. Pond 33 has a higher density of aquatic plants than Pond 45. When these plants die off and decompose there is a higher probability that during mineralization refractory organic P will be formed (Gächter and Meyer, 1993; Søndergaard et al., 2003). Iron and Al can also bind onto OM complexes and adsorb additional P from the water column (Boström et al., 1988). However, having a higher percentage of OM could also limit the amount of P adsorbed onto sediment surfaces if the Fe/Al-bound P complexes are weak (Stuanes, 1982). Organic matter could replace P from these complexes, therefore releasing it to the water column (Stuanes, 1982).

4.7. Sediment Buffering Capacity

Froelich (1988) introduced the concept that sediment is capable of buffering SRP to an equilibrium concentration between the sediment and water column. Evaluating outflow concentrations of SRP at Pond 33 and Pond 45 reveal almost constant concentrations around 50 µg L^{-1} and 3.5 µg L^{-1}, respectively. Liu et al. (2009) stated that sediment in a multipond system in China had EPC values less than 50 µg L^{-1} which indicated the sediment had a strong buffering capacity of water column P. Sorption tests following the methods of Stone and Mudroch (1989) showed that the EPC for Pond 33 at the constructed wetland and Pond 45 were approximately 50 µg L^{-1} and 19 µg L^{-1}, respectively. These results compare well to the literature as seen in Table 17 (Stone et al., 1995; McDowell and Sharpley, 2001; McDowell and Sharpley, 2003; Haggard et al., 2004). From results presented in the literature and the current study it can be concluded that sediment in Ponds 33 and 45 have a strong buffering effect on the effluent concentrations of dissolved P.

Table 17. Summary of EPC from the literature and current study.

Author	Location	Type of Sediment / Grain Size	EPC (mg L^{-1})
Mulroy, 2009	Sediment Forebay, Pond 33	Stormwater pond sediment; <63 µm	**0.025**
	Sediment Forebay, Pond 33	>63 µm	**0.014**
	Secondary Settling Pond, Pond 33	<63 µm	**0.014**
	Secondary Settling Pond, Pond 33	>63 µm	**0.011**
	Constructed Wetland, Pond 33	<63 µm	**0.050**
	Pond 45, <63 µm	<63 µm	**0.010**
	Pond 45, >63 µm	>63 µm	**0.019**
Stone et al., 1995	Big Creek and Big Otter Creek, Ontario	Freshwater sediment; 42 - 63 µm 31 - 42 µm 19 - 31 µm 12 - 19 µm 8 - 12 µm <8 µm	**0.015 – 0.080**[1]
McDowell & Sharpley 2001	Pennsylvania, USA	River bed sediment	**0.043**
		River bank sediment	**0.02**
McDowell & Sharpley 2003[2]	Pasture & forest sediment with addition of poultry or dairy manure, Pennsylvania, USA	Forest with poultry	**0.02**
		Forest with dairy	**0.022**
		Pasture with poultry	**0.027**
		Pasture with dairy	**0.024**

Table 17 cont. Summary of EPC from the literature and current study.

Haggard et al. 2004[2]	Mud Creek	Sediment downstream of a WWTP	**0.11**
	Osage Creek		**0.13**
	Spring Creek		**5.29**
	Sager/Flint Creek, Arkansas, USA		**0.88**
Wang et al. 2006	East Taihu Lake	360 – 480 mesh	**48.2**
		240 – 360 mesh	**41.3**
		60 – 240 mesh	**34.2**
		18 – 60 mesh	**30.1**
	Yue Lake	360 – 480 mesh	**634.1**
		240 – 360 mesh	**588.5**
		60 – 240 mesh	**562.7**
		18 – 60 mesh	**531.3**
	Wuli Lake, China	360 – 480 mesh	**214.8**
		240 – 360 mesh	**194.4**
		60 – 240 mesh	**212.3**
		18 – 60 mesh	**87.9**

[1]Range of EPC values for Big Creek and Big Otter Creek, Ontario for 6 sediment size classes

[2]Values reported are an average from 3 sampling locations downstream of a wastewater treatment plant (WWTP) within each creek before Alum treatment

The results of laboratory sediment P sorption experiments for Pond 33 indicate that there was only a slight difference in sorptive capacity between the two sediment size fractions, ≤ 63 µm and >63 µm (Figure 19). Stone et al. (1995) referenced several studies indicating that there are two types of particles in river sediment, water stable aggregates and primary particles. Depending on the sample preparation prior to grain size separation, water stable aggregates (consisting of clay particles) could be sorted into larger grain size fractions even though their primary particle size is much smaller (Wall et al., 1978). This is the most probable cause for the observed lack of difference in EPC values between sediment size fractions in the three compartments in Pond 33. Pretreatment to neutralize sediments ionic charge was not conducted before wet sieving, so it is possible that some of the particles in the >63 µm size fraction consisted of aggregated clay particles. Pond 45 had more sand sized particles which corresponded to a lower sorption capacity of the >63 µm size fraction.

Differences in sorption data from the present study compared to that reported in the literature are related to differences in the land use practices, grain size, geochemical and mineralogical composition of the sediment and possible changes to sediment chemistry that occur during laboratory preparation of samples for the sorption experiments. While every attempt was made to minimize the effect of sample pretreatment on sediment chemistry, drying the samples to obtain a mass of dry weight may have caused chemical changes in sediment geochemistry and aggregate stability to occur (c.f. Worsfold et al., 2005). A drying method should be chosen depending on the type of experiment and analysis being conducted so to limit potential sediment geochemical changes (Mudroch and Azcue, 1995).

4.8. Design Criteria, Maintenance and Water Quality Targets

Stormwater management ponds are designed to meet OMOE water quality and quantity targets and the targets (level of protection) required is related to the sensitivity of the receiving water body. A certificate of approval issued by the OMOE stipulates the maximum discharge allowed from the outlet which is determined from pre-urban levels and the percent reduction of inlet TSS concentrations required by the pond. There are three levels of protection; basic, normal and enhanced where a reduction of 60%, 70% and 80% of influent SS concentrations is required, respectively (OMOE, 2003). Pond 33 and Pond 45 were designed with normal protection, therefore the ponds must retain 70% of influent SS. Pond 45 retained over 90% SS for all events and Pond 33 retained over 70% SS four out of six events. Many pollutants, including P, can sorb onto sediment (Stone and Mudroch, 1989), therefore by reducing SS concentrations the water quality of downstream aquatic systems should increase (OMOE, 2003). In the present study, results from inlet(s) and outlet TSS, TP and SRP concentrations, sediment porewater SRP concentration measurements, sediment geochemistry and mineralogy, grain size distribution and sediment buffering capacity from both ponds indicate that relying on sedimentation alone might not improve effluent water quality because of several other physical, chemical and biological processes occurring in the pond.

There are no stormwater quality targets for P, however, the MOE Provincial Water Quality Objectives (OMEE, 1994) provides an interim value of 30 µg L^{-1} for TP so excessive plant growth, including algal blooms, is limited. This value can be used to assess the quality of effluent leaving a SWM pond, however, SWM ponds are not regulated to meet this target unless incorporated into the design, such as Pond 33. Pond 45 had low TP effluent concentrations (<30 µg L^{-1}), while, outlet TP concentrations in Pond 33 exceeded the PWQO guideline 99% of the time. The CCME (2002) recommends a maximum increase of 25 mg L^{-1} TSS from background levels in surface waters for the protection of aquatic life. On average, both ponds released TSS concentrations below this limit. Pond 33 and Pond 45 released 23 mg L^{-1} and 7 mg L^{-1} of TSS on average, respectively. There were periods when a pond released higher concentrations of TSS but then quickly returned to lower concentrations.

According to the MOE, TSS removal should be calculated over a long period of time, approximately 10 years, to account for rainfall variability (OMOE, 2003). The current study was approximately five months long and both ponds generally had favourable SS removal during high and low flow periods. Inadequate settling times can account for low SS retention rates, while resuspension of sediment can occur through wave action and bioturbation (Boström et al., 1988). Vegetation in and around both ponds creates habitat for wildlife but bioturbation and animal excrement can impact the treatment of SS and P in a SWM pond (Boström et al., 1988; Mallin et al., 2000).

Stormwater management pond design can help achieve the water quality targets established by the OMOE (1994, 2003) and CCME (2002). Water quality portions of a pond are enhanced by providing high L:W ratios or baffles when the land parcel is small. These design parameters can improve water quality treatment by increasing the hydraulic efficiency of the pond (Persson et al., 1999; Wong et al., 1999). Pond 33 had a L:W ratio of 4:1 which is greater than the minimum criteria of 3:1 stipulated by the OMOE (2003). Pond 45 had a crescent moon shape and two inlets, therefore the L:W ratio was around 2.5, which is less than the OMOE L:W ratio of 3:1. Pond 45 drained approximately 29 ha with a permanent pool volume of 6925 m^3 and a detention time of 48 hours. In contrast, Pond 33 drains 80 ha into a pond with a permanent pool volume of 1685 m^3 and a detention time of approximately 24 hours. Pond 45 has a greater capacity to treat runoff because of favourable conditions in the pond and less runoff to treat in a larger sized pond. Incorporating baffles into Pond 33 will help increase the detention time and allow more contact time between the water column and sediment.

Allowing sediment in Pond 33, particularly the CW, to be exposed to air will help create aerobic conditions. If aerobic conditions dominate, P adsorbed onto the surface of particles could become incorporated into the crystalline lattice in sediment. If anaerobic conditions resume afterwards, P will not be released as easily from sediment. Harvesting *Typha sp.* plants before senescence is an option for SW managers, however, the cost required to remove the upper biomass might be too large for the amount of P removed from the pond. Other maintenance options are to install a filter medium to remove excess P from the effluent or dredge the accumulated sediment in the pond. Sediment removed from a SWM pond could have high pollutant concentrations, especially heavy metals such as Pb, Cu and Cr (Marsalek and Marsalek, 1997). Sediment disposal would be difficult with contaminated sediment as it cannot be reused or transported to a municipal landfill (Marsalek and Marsalek, 1997). The sediment would most likely have to be transported to a hazardous waste landfill which would substantially increase the overall cost of dredging the pond.

4.9. Implications to Stormwater Management

Over the past 60 years, urbanization has been increasing in North America. Natural or agricultural lands have been converted to residential, commercial or industrial land uses. Infiltration of SW has decreased because land has been paved over or compacted to impermeable conditions (Stephens et al., 2002; Xiao et al., 2007); therefore, increasing the rates and magnitude of runoff from urbanized areas (Bradford and Gharabaghi, 2004). Nutrients, fecal matter and heavy metals are washed off during a storm event and transported to a downstream water body (Mallin et al., 2002). Non-point source pollution is difficult to treat because the type of pollutant is not known. Sedimentation of particulate matter and its associated pollutants increased with the introduction of wet ponds and constructed wetlands (OMOE, 2003). If TSS concentrations are decreased, the quality of runoff will increase and therefore will have less of an impact on receiving water courses. Unfortunately, many wet ponds and constructed wetlands have low TE because of SWM pond design, size and shape (Van Buren et al., 1997). A high TE indicates a percentage of influent TSS was retained in the pond, however, the effluent concentration could still be high (Gharabaghi et al., 2006). Also, fine grained sediment (<63 µm) are known to transport the highest pollutant load because of their higher surface area compared to larger particles (Stone and Mudroch, 1989, Greb and Bannerman, 1997). A pond's ability to retain SS and pollutants such as P is dependent not only on the structural design of the facility (L:W ratio, inlet/outlet construction, detention time) but the physical, chemical and biological processes occurring within the pond. Designing a SW detention pond not only involves hydraulics but also water chemistry, aquatic science, biology, and sediment characterization.

The rates and magnitude of physical, chemical and biological processes occurring in Pond 33 and Pond 45 determined the level of TSS, TP and SRP concentration reduction. Long term studies on the effectiveness of SWM ponds to treat influent runoff are rare. This study was conducted over a five month period where BF and SF samples were collected at the inlets and outlets of a hybrid extended detention pond (Pond 33) and a conventional wet pond (Pond 45). The constructed wetland in Pond 33 has reached equilibrium with the water column and is no longer capable of assimilating large quantities of P. Chemical cycling of P has been thoroughly investigated in freshwater and marine systems (Boström et al., 1988; Cooke et al., 1993; White and Stone, 1996; Spears et al., 2007) however, Peng et al. (2007) has stated that P loading to wastewater stabilization ponds are higher and P distribution in sediments and adsorption / desorption from sediment within these systems requires more research. Measuring the change in wetland succession, sediment and organic matter accumulation and relative pollutant buildup within pond sediment would provide information on the rate these systems become saturated and can no longer effectively reduce incoming pollutants.

A paradigm shift is occurring where SWM is moving from treating SW at the end-of-pipe to a treatment train approach where SW is treated at the source, through conveyance and lastly at the end-of-pipe (OMOE, 2003; Marsalek and Schreier, 2009). Large storm events will be treated by SWM ponds and constructed wetlands because the amount of precipitation could exceed the treatment capacity of the first two control types (Marsalek and Schreier, 2009). The effectiveness of wet ponds and constructed wetlands requires improvement. Stormwater management ponds remove TSS concentrations between -37% to 95% and TP concentrations ranging from -35% to 92% of influent concentrations (c.f. Comings et al., 2000; Mallin et al., 2002; Mulroy, 2010). Ensuring built and proposed SWM ponds function as effectively as possible is a necessity because source and conveyance controls might not be implementable at all locations because of inadequate soil porosity, proximity to a groundwater aquifer or lack of public acceptance.

The MOE Stormwater Planning and Design Manual provides guidance on SWM pond design including inlet and outlet design, recommended L:W ratio, volume of the permanent pool and the required TSS removal efficiency. The next version of the manual could include updated information of TP and SRP concentrations and levels of protection for P sensitive areas. Phosphorus is the limiting nutrient for plant and algal growth for many aquatic systems in Ontario (Schueler, 1987; Chambers et al., 2001) therefore, providing P discharge targets to achieve with the use of BMPs would promote greater innovation with regard to designing these types of facilities.

Ongoing monitoring of these facilities is required to ensure their effectiveness over time (OMOE, 2003) however, this is very costly and many owners do not have the time or resources to allocate to monitoring all their SWM ponds. Lee (2007) has suggested choosing representative ponds, equipping them with automatic samplers so they can sample throughout the hydrograph so an event mean concentration (EMC) can be calculated. If this is not feasible to implement, a grab sample collected 2-3 hours into the hydrograph can provide a good estimate of the EMC and decrease costs and maintenance of the automatic samplers (Khan et al., 2006 reported in Lee et al., 2007).

Generating enough funds to allocate to the operation, maintenance and capital works of SWM ponds can be accomplished through SW financing. This has been successfully implemented in locations throughout the United States and Canada (Cameron et al., 1999). Rates can be applied based on lot size, percent imperviousness or land use type (Cameron et al., 1999). The Province of Ontario has identified areas within the Greater Golden Horseshoe, including Uptown Waterloo in the City of Waterloo that will increase in population density (OMPIR, 2006). It is estimated that by 2031, Uptown Waterloo will have 200 people and jobs combined per hectare (OMPIR, 2006). Rapid intensification will cause an increase in the amount of impervious surfaces within an area, and will require innovative means to manage SW coming from these areas. Funding to operate and maintain existing and new SWM ponds and other SW infrastructure to current standards will be necessary to ensure flood control for public and private lands and the health of downstream aquatic ecosystems.

Chapter 5: CONCLUSIONS

5.1. Conclusions

Stormwater runoff in urban areas represents a major pathway for pollutant transfer to receiving waters (Stephens et al., 2002). Stormwater management (SWM) ponds are used as a best management practice to help mitigate the negative effects of stormwater (OMOE, 2003). Despite knowledge gained from an increasing number of studies on SWM pond design and TE, few studies have focused on the physical, chemical and biological processes governing P and SS retention or release. The purpose of this study was to quantify the spatial and temporal variability of TP, SRP and TSS concentrations at the inlet(s) and outlet of two structurally different (conventional wet pond and a hybrid extended detention wet pond) SWM ponds. Sediment characterization, porewater SRP concentrations and sediment P buffering capacity was measured to evaluate the control of sediment on P mobilization in the SW ponds.

The spatial and temporal variability of TP, SRP and TSS concentrations at the inlets of Pond 33 and Pond 45 are influenced by storm magnitude, antecedent moisture conditions and sediment availability. Generally, there was no significant difference of TP, SRP and TSS concentrations between the inlets of Pond 33 and Pond 45. Both ponds receive runoff from residential areas and the range of pollutants is common to both residential areas. Outlet concentrations between both ponds for different storm magnitudes and seasonally were significantly different for TP, SRP and TSS concentrations. This is a result of the internal processes occurring in each pond and the detention time of influent runoff. Pond 45 was capable of decreasing inlet TP and SRP concentrations below the 1994 PWQO guideline of 30 µg L^{-1}, while Pond 33 was above this guideline 99% of the time. Outlet TP, SRP and TSS concentrations were less variable than inlet concentrations. The CCME (2002) has recommended an increase of up to 25 mg L^{-1} for TSS into streams and rivers. Pond 33 was designed to have outlet concentrations <25 mg L^{-1}. On average, Pond 45 had TSS concentrations below this guideline. Pond 33 had average TSS concentrations ranging from 4.5 mg L^{-1} on JD 299 to 45.4 mg L^{-1} on JD 201. A large increase in TSS concentrations could have been the result of sediment resuspension in the pond.

An increase in TP and SRP concentrations at the outlet of Pond 33 occurred during the period of *Typha sp.* senescence however, biomass measurements of *Typha sp.* plants should be conducted to quantify the amount of P translocated within the plants during the same time period so to account for this increase. Accordingly, radial oxygen loss from *Typha sp.* could explain the higher redox potential and lower porewater SRP concentrations in the bottom sediment of the constructed wetland. Higher porewater SRP concentrations were found in Pond 33 sediment forebay and secondary settling pond, which could be released into the water column if bottom sediment was disturbed by incoming runoff, waves or bioturbation. Lower porewater SRP concentrations at Pond 45 could have been because the PWPs were not placed deep enough into the sediment profile to capture the anoxic zone or because of P uptake from sediment porewater from the submerged plant, possibly Eurasian Milfoil, found within the pond.

Sediment geochemistry and mineralogy results have shown that Pond 33 has a higher percentage of Fe, Al and OM than Pond 45. If water column conditions were more favourable in Pond 33, i.e. higher redox potential and higher DO concentrations, greater amounts of P could be adsorbed onto sediment within the pond. Bottom sediment in Pond 33 became progressively finer as the parcel of water travelled from the inlet to the outlet. However, for both ponds, a high proportion of sediment leaving the outlets were flocculated clay particles, which are known to transport the majority of the pollutants including P.

The EPC at the outlet of Pond 33 was approximately 50 μg L^{-1} and 19 μg L^{-1} for Pond 45. Outlet SRP concentrations were around these values for the majority of the field season therefore indicating that the ponds are in a steady state between adsorption and desorption of P. Langmuir's maximum sorption capacity (Γ_m) indicates that bottom sediment within Pond 33 and Pond 45 are not saturated, therefore changes in pH, temperature, DO and redox potential can increase the amount of P adsorbed onto sediment particles, thus changing the EPC.

Overall, Pond 45 was capable of treating influent SW better than Pond 33. Trap efficiencies for TP, SRP and TSS concentrations at Pond 33 were 24.3%, 26.7% and 66.8% for BF and SF samples. Pond 45 had TE's of 93.8%, 94.2% and 98.0% for TP, SRP and TSS concentrations during the five month field season. Pond design, such as pond volume, L:W ratio and detention time will affect the rates and magnitude of P and SS cycling. Pond 45 had a drainage area 33% smaller than Pond 33, however the pond volume was 4 times larger than Pond 33. Despite not having tertiary treatment from a constructed wetland, Pond 45 had more favourable conditions, i.e. higher DO concentrations, which resulted in a better performance than Pond 33. Some design and maintenance considerations for Pond 33 are to incorporate baffles to increase the detention time of influent runoff and allow greater contact time between the water column and sediment, remove the accumulated sediment in the pond, allow the constructed wetland to be exposed to air so aerobic soil conditions are created, harvest *Typha sp.* plants before they senesce and/or to install a filter medium after the outlet to remove effluent TP and TSS concentrations.

5.2. Recommendations for Future Research

Stormwater management has evolved from peak shaving or flood control in the 1970s to water quality improvement in the 1990s was addressed by incorporating a permanent pool and constructed wetlands. Currently SWM is looking at the system at a watershed scale and attempting to manage the natural water balance by infiltrating and evaporating stormwater on site rather than convey it to end-of-pipe facilities. However, SWM ponds will be needed to control and treat larger events. Some ponds have been built within the last 20 – 30 years and require retrofitting to current standards.

More research is required to address the internal cycling of P in SWM ponds. A long term study focusing on the physical, chemical and biological cycling of P in a hybrid extended detention pond and conventional wet pond will provide information on seasonal changes of P in the water column and sediment porewater. Many studies have focused on either the physical, chemical or biological processes in SW ponds. It is recommended that these studies could be completed on the same study site so that climate, land use, drainage area are all the same. This will make it easier to evaluate the processes occurring within each pond. Measurements of P in upper and lower biomass of macrophytes at regular intervals during the study period will provide information on the potential release of P to the water column during plant senescence. Measuring TP and SRP concentrations in the water column and sediment porewater during the study period would help strengthen P biomass results. Benthic invertebrate analysis and a complete wildlife assessment will provide information on habitants of the ponds. Understanding the physical, chemical and biological cycling of P between the two most common SW pond designs will help water resources managers and planners effectively design SW ponds to optimize these processes.

Appendix A

Activation Laboratory Certificate of Analyses

X-Ray Fluorescence QA/QC

Analyte Symbol	SiO2	Al2O3	Fe2O3(T)	MnO	MgO	CaO	Na2O	K2O	TiO2	P2O5	Cr2O3	LOI
Unit Symbol	%	%	%	%	%	%	%	%	%	%	%	%
Detection Limit	0.01	0.01	0.01	0.001	0.01	0.01	0.01	0.01	0.01	0.01	0.01	
Analysis Method	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF	FUS-XRF
NIST 694 Meas	11.45	1.81	0.77	0.015	0.32	43.26	0.92	0.55	0.13	30.61		
NIST 694 Cert	11.2	1.8	0.79	0.0116	0.33	43.6	0.86	0.51	0.11	30.2		
DNC-1 Meas	46.71	18.37	9.93	0.147	10.1	11.08	1.99	0.23	0.49	0.08		
DNC-1 Cert	47	18.3	9.93	0.149	10.1	11.3	1.87	0.234	0.48	0.09		
AN-G Meas	46.46	29.79	3.28	0.041	1.76	15.77	1.64	0.14	0.22	0.02		
AN-G Cert	46.3	29.8	3.36	0.04	1.79	15.9	1.63	0.13	0.22	0.01		
BE-N Meas	37.98	10.01		0.2	12.92	13.8	3.22	1.34	2.65	1	0.05	
BE-N Cert	38.2	10.1		0.2	13.1	13.9	3.18	1.39	2.61	1.05	0.05	
W-2a Meas	52.33	15.28	10.81	0.171	6.32	10.74	2.24	0.63	1.07	0.12		
W-2a Cert	52.4	15.4	10.7	0.163	6.37	10.9	2.14	0.626	1.06	0.13		
MICA-Mg Meas	38.38	15.2	9.36	0.256	20.39	0.03	0.29	10.04	1.64	0.01		
MICA-Mg Cert	38.3	15.2	9.46	0.26	20.4	0.08	0.12	10	1.63	0.01		
NCS DC73304 (GBW 07106) Meas	90.22	3.52	3.22		0.05	0.27	0.12	0.64		0.22		
NCS DC73304 (GBW 07106) Cert	90.36	3.52	3.22		0.082	0.3	0.061	0.65		0.222		
Method Blank Method Blank	< 0.01	< 0.01	< 0.01	< 0.001	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	< 0.01	-0.01

X-ray Diffraction Analysis of Four Pond Sediment Samples

W.O. # A09-0943
Invoice # A09-0943
No. of pages: 4

Client: Faculty of Environment
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Date Received: February 26, 2009

Date Reported: March 11, 2009

EXPERIMENTAL

Four samples were submitted for mineral identification. The samples were identified as follows:

Client sample ID	ActLabs sample ID
Pond33-A	A09-0943-1
Pond33-B	A09-0943-2
Pond33-C	A09-0943-3
Pond45	A09-0943-4

The samples were freeze-dried and ground in a ceramic mortar prior to X-ray analysis.

X-ray diffraction was performed on a pressed portion of the sample using Phillips X'Pert PW3040-PRD diffractometer equipped with Cu X-ray source and operated at 40kV and 50 mA.

RESULTS

All samples are similar by content and have Quartz, Carbonates, Silicates and Clays. Identification of the clay minerals (muscovite and chlorite) is tentative. Dedicated clay analysis is necessary to identify clay minerals present in the samples.
The amount of the compounds present in samples was estimated using the integrated peak intensities of the strongest peak for each compound. The intensities were normalized with values of $k=I/I_{cor}$ from PDF (Powder Diffraction File) database. Normalization factor k for a compound is the ratio of its strongest peak intensity to the intensity of the strongest peak of corundum, Al_2O_3, in a sample containing 50% of the compound and 50% of corundum. The values of k measured and reported by different researchers are collected and displayed in the Powder Diffraction database. The use of k factors affords semi-quantitative estimate of sample composition
The minerals identified in the sample are listed in the Table I below; the spectra (22 pages) are enclosed.

Table I. Identified minerals and approximate concentrations in the Pond Sediment samples.

Client sample ID ActLabs Sample ID	Identified Minerals	Approximate Concentration %
Pond33-A A09-0943-1	Quartz, SiO_2	30
	Dolomite, $CaMg(CO_3)_2$	15
	Calcite, $CaCO_3$	6
	Albite, calcian, ordered, $(Na, Ca)Al(Si, Al)_3O_8$	20
	Maximum Microcline, $KAlSi_3O_8$	16
	Tremolite, $Ca_2Mg_5Si_8O_{22}(OH)_2$	7
	Chlorite, $Mg_{2.5}Fe_{1.65}Al_{1.5}Si_{2.2}Al_{1.8}O_{10}(OH)_8$	4
	Muscovite, $KAl_3Si_3O_{10}(OH)_2$	2
Pond33-B A09-0943-2	Quartz, SiO_2	35
	Dolomite, $CaMg(CO_3)_2$	9
	Calcite, $CaCO_3$	12
	Albite, calcian, ordered, $(Na, Ca)Al(Si, Al)_3O_8$	14
	Maximum Microcline, $KAlSi_3O_8$	22
	Chlorite, $Mg_{2.5}Fe_{1.65}Al_{1.5}Si_{2.2}Al_{1.8}O_{10}(OH)_8$	4
	Muscovite, $KAl_3Si_3O_{10}(OH)_2$	4
	* Zeolite X, (Ca,C3H6), $(Ca_{46}(Si_{100}Al_{92}O_{384}))(C_3H_6)_{30}$	1
Pond33-C A09-0943-3	Quartz, SiO_2	21
	Dolomite, $CaMg(CO_3)_2$	15
	Calcite, $CaCO_3$	6
	Albite, calcian, ordered, $(Na, Ca)Al(Si, Al)_3O_8$	12
	Maximum Microcline, $KAlSi_3O_8$	32
	Chlorite, $Mg_{2.5}Fe_{1.65}Al_{1.5}Si_{2.2}Al_{1.8}O_{10}(OH)_8$	6
	Muscovite, $KAl_3Si_3O_{10}(OH)_2$	6
	* Zeolite X, (Ca,C3H6), $(Ca_{46}(Si_{100}Al_{92}O_{384}))(C_3H_6)_{30}$	1
Pond45 A09-0943-4	Quartz, SiO_2	37
	Dolomite, $CaMg(CO_3)_2$	6
	Calcite, $CaCO_3$	1
	Albite, calcian, ordered, $(Na, Ca)Al(Si, Al)_3O_8$	18
	Maximum Microcline, $KAlSi_3O_8$	15
	Tremolite, $Ca_2Mg_5Si_8O_{22}(OH)_2$	5
	Chlorite, $Mg_{2.5}Fe_{1.65}Al_{1.5}Si_{2.2}Al_{1.8}O_{10}(OH)_8$	14
	Muscovite, $KAl_3Si_3O_{10}(OH)_2$	4

*Possibly present

Please do not hesitate to contact us if you have any questions.

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4

Activation Laboratories Ltd

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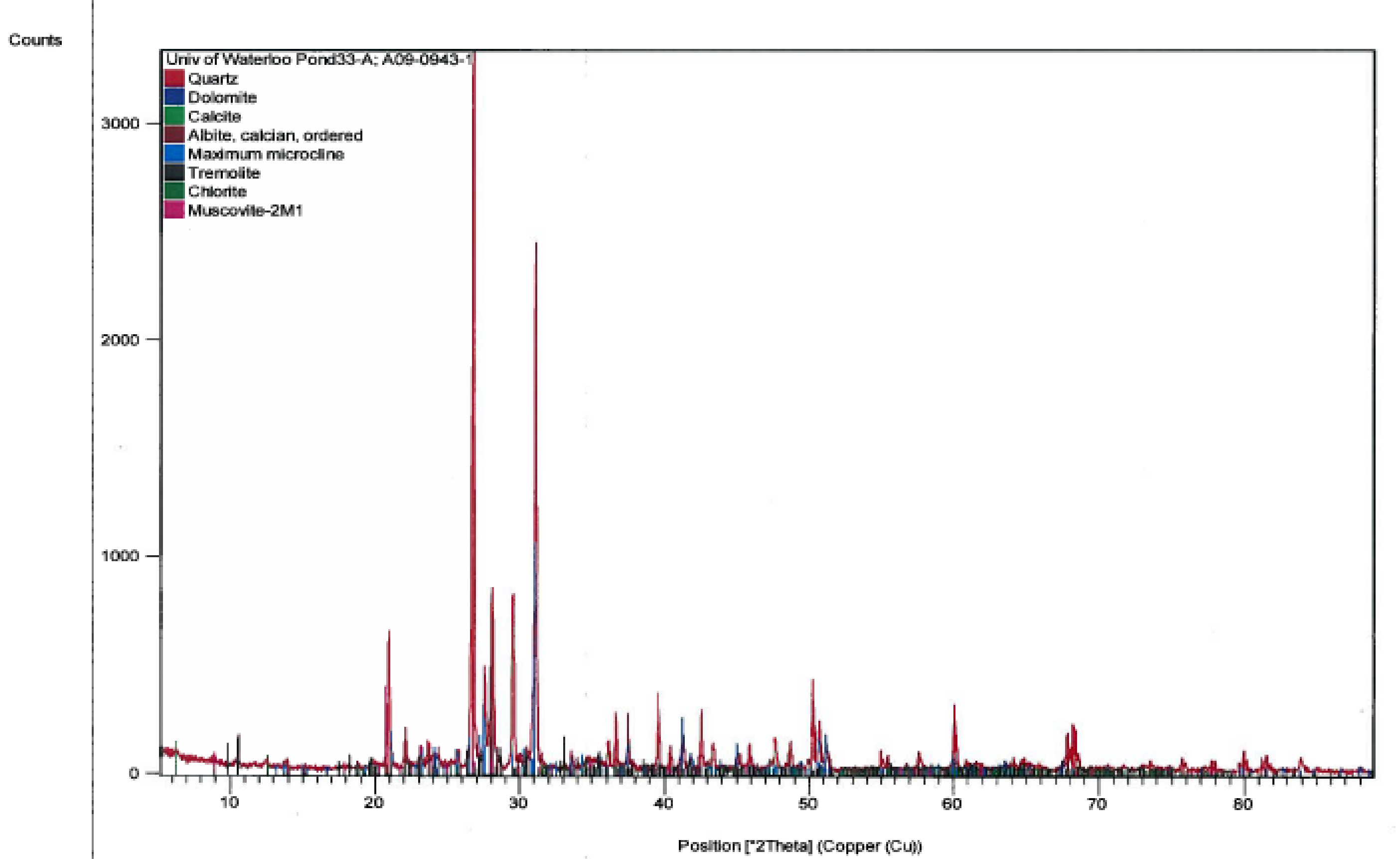

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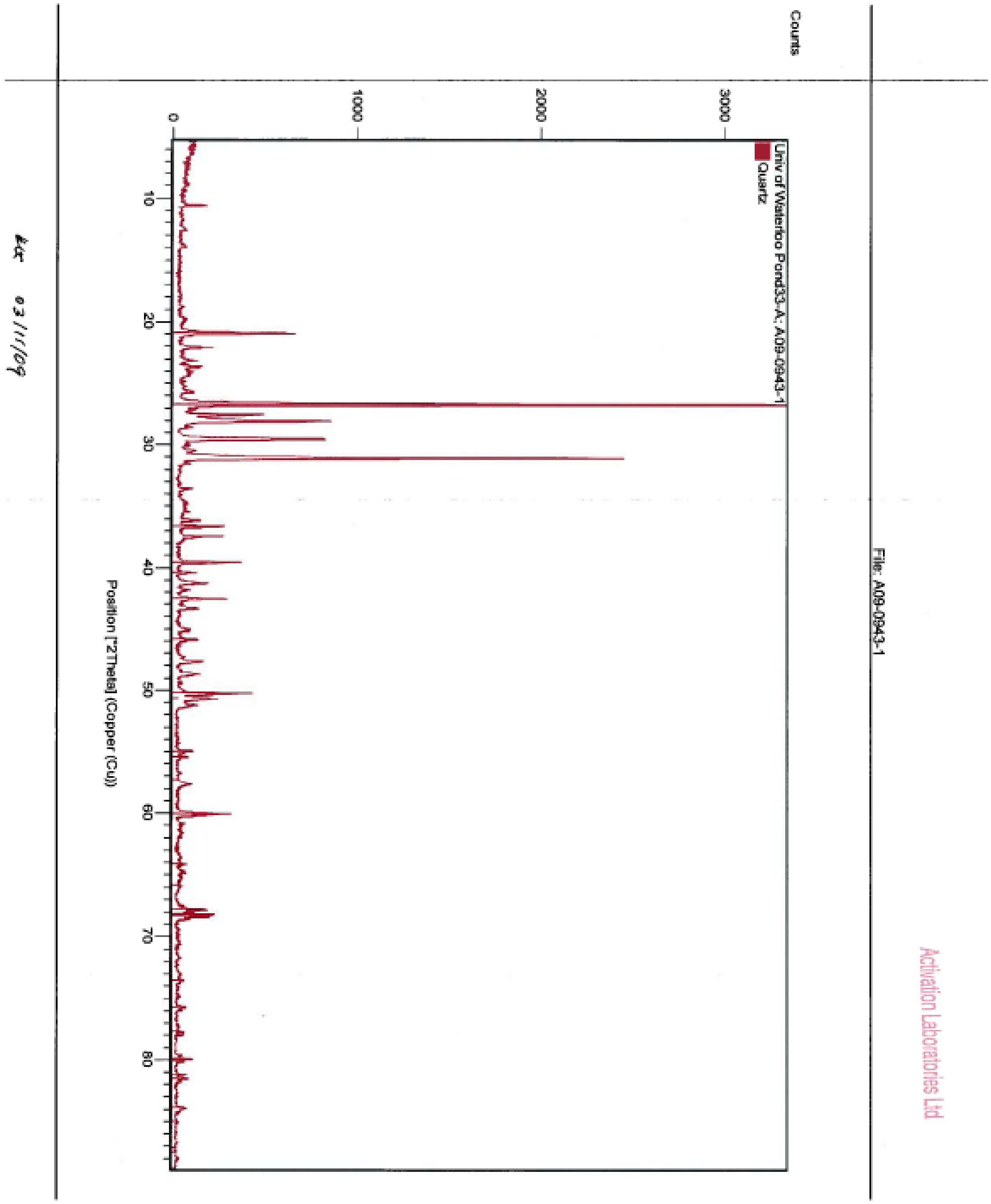
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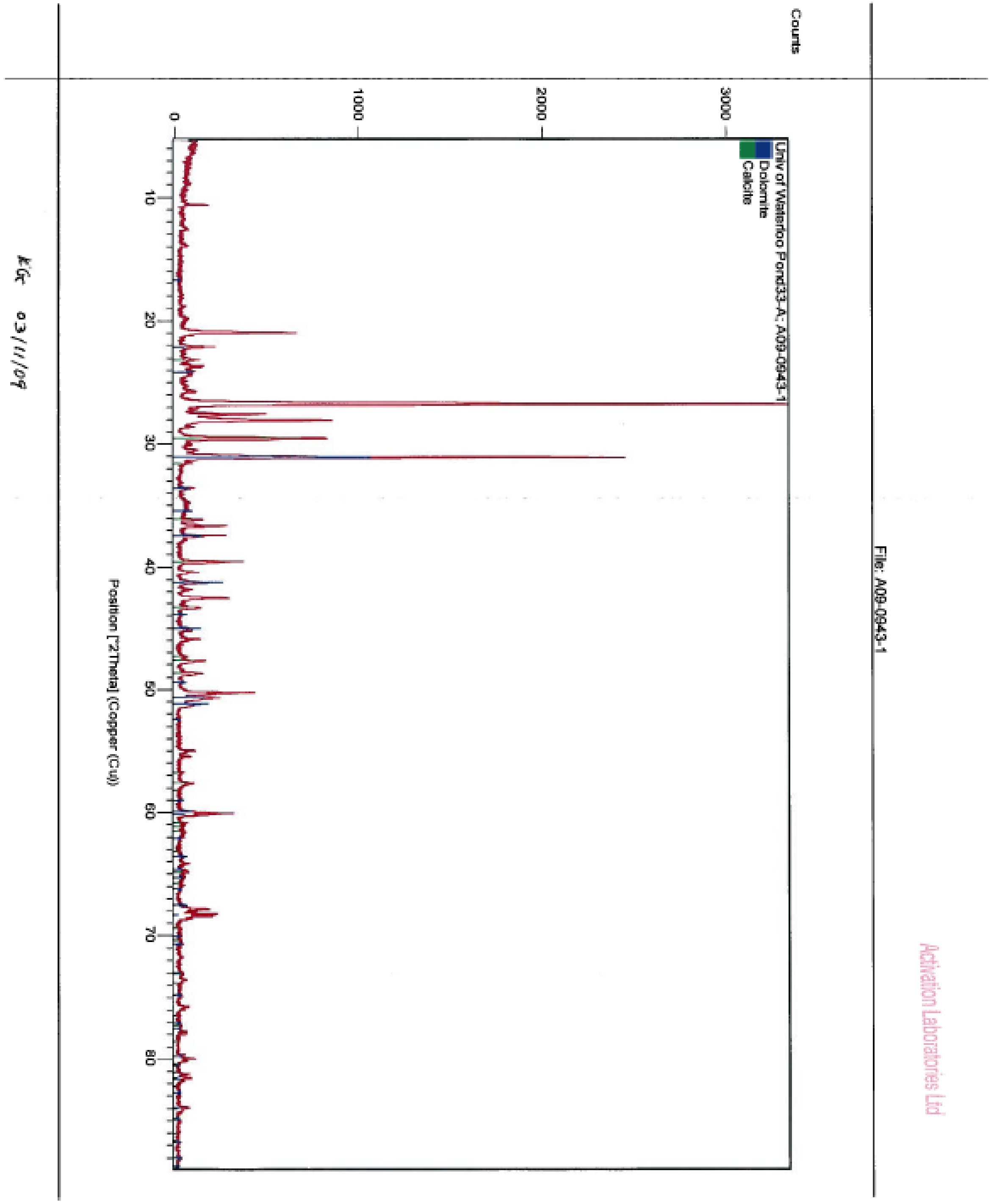
Activation Laboratories Ltd
03/11/09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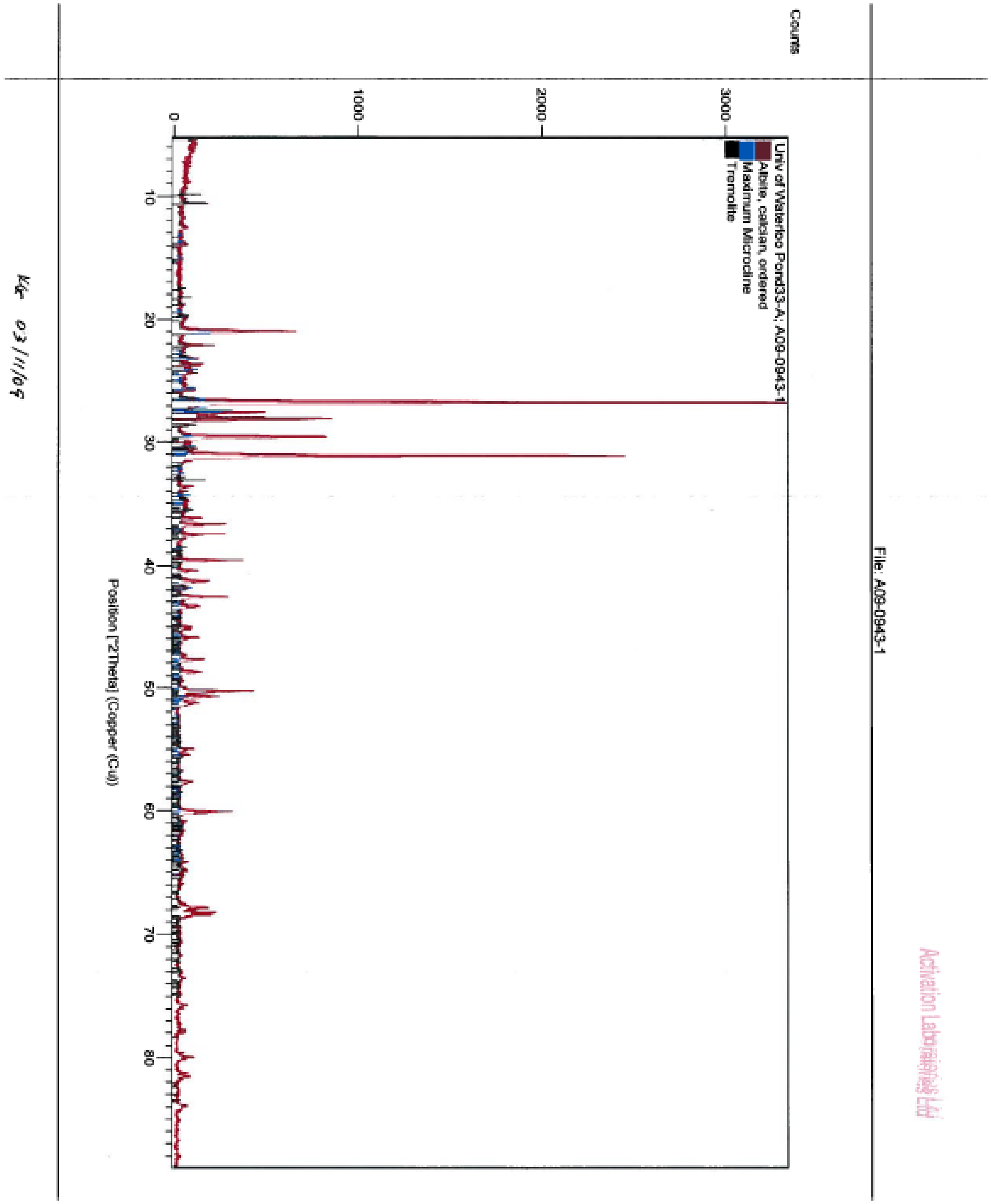
KG 03/11/09
Activation Laboratories Ltd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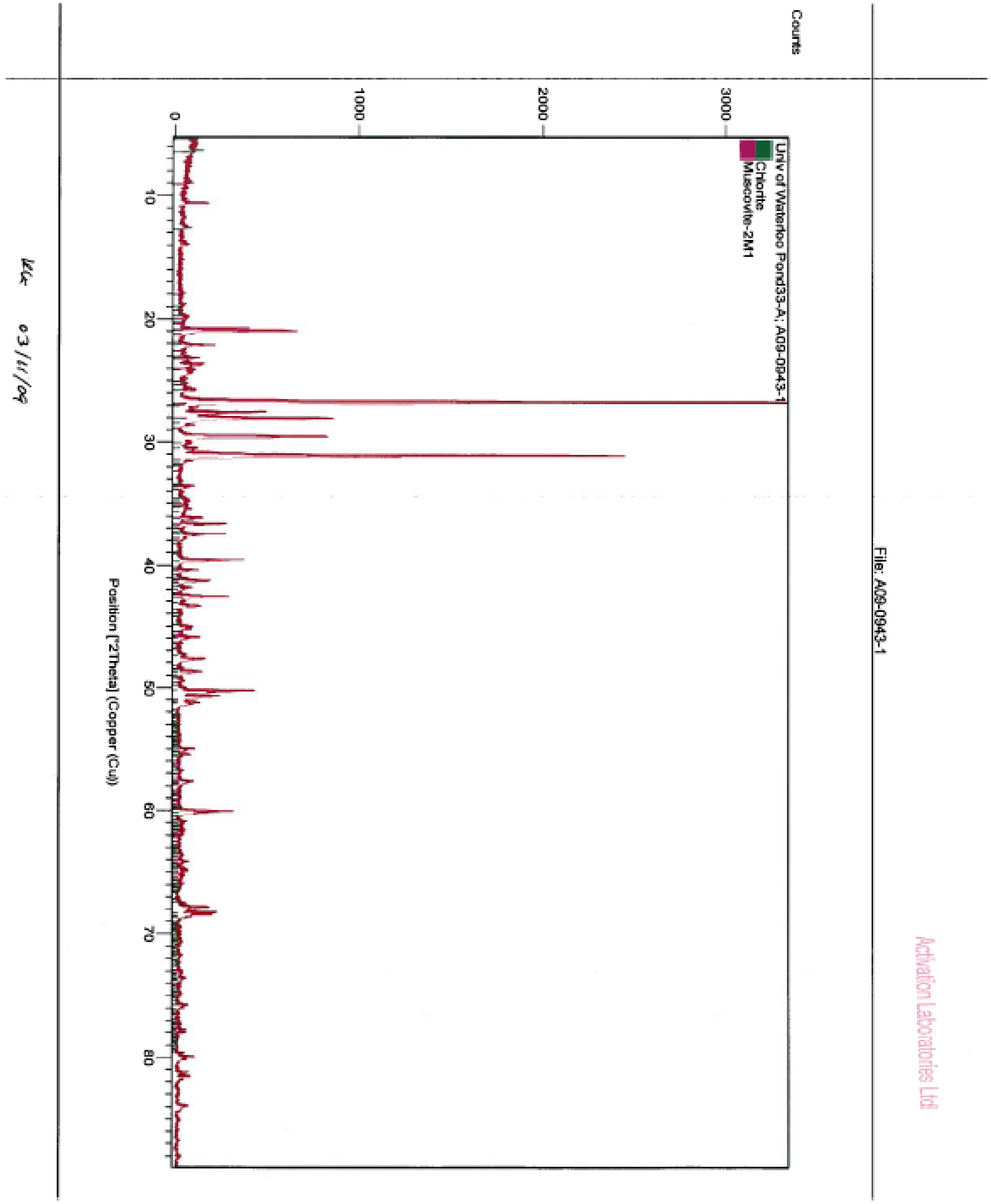
Activation Laboratories Ltd
KC 03/11/09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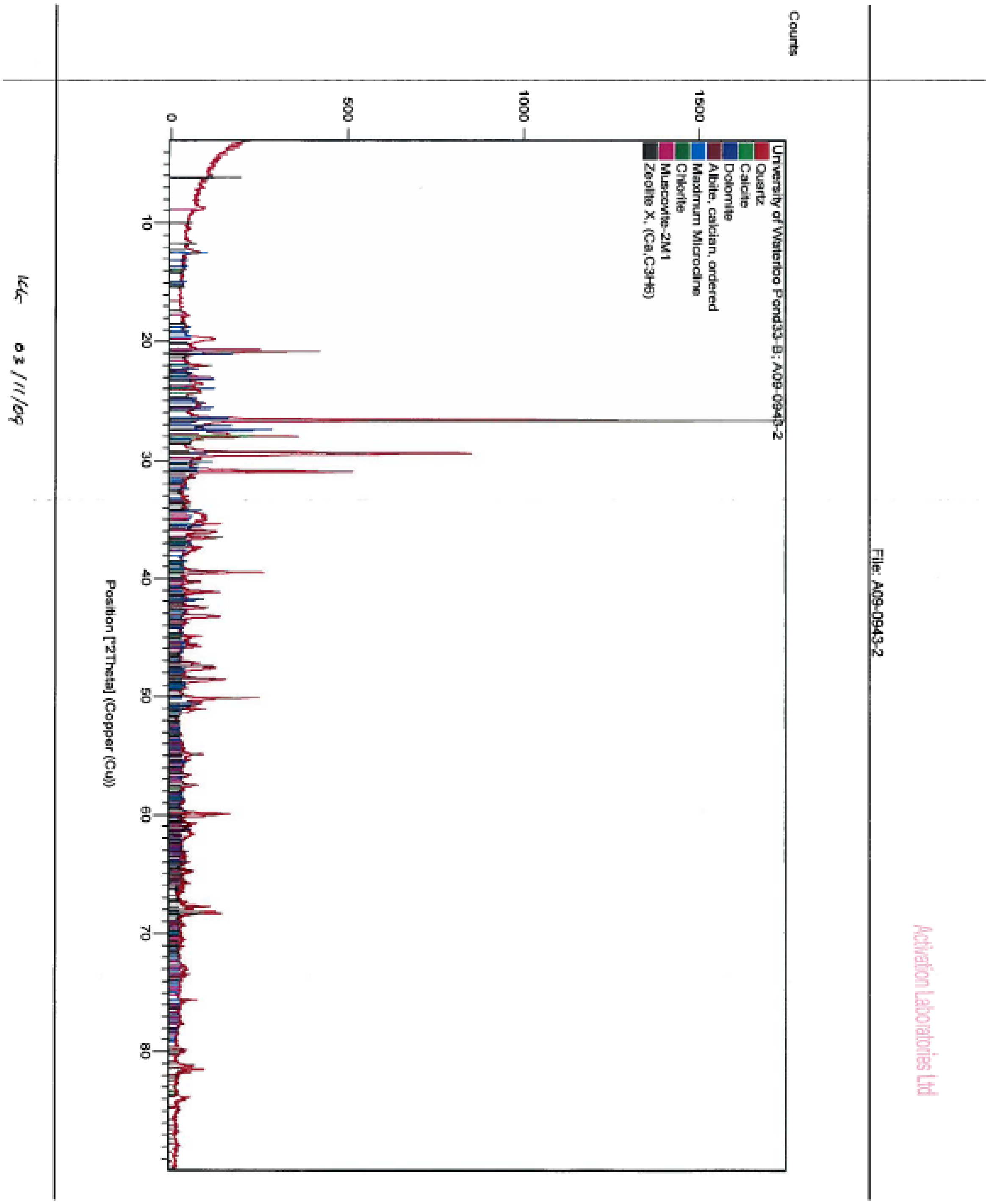
Activation Laboratories Ltd.
KL 03/11/09



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Maximum Microcline
Chlorite
Muscovite-2M1
Zeolite X, (Ca,C3H6)
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1000
1500
10
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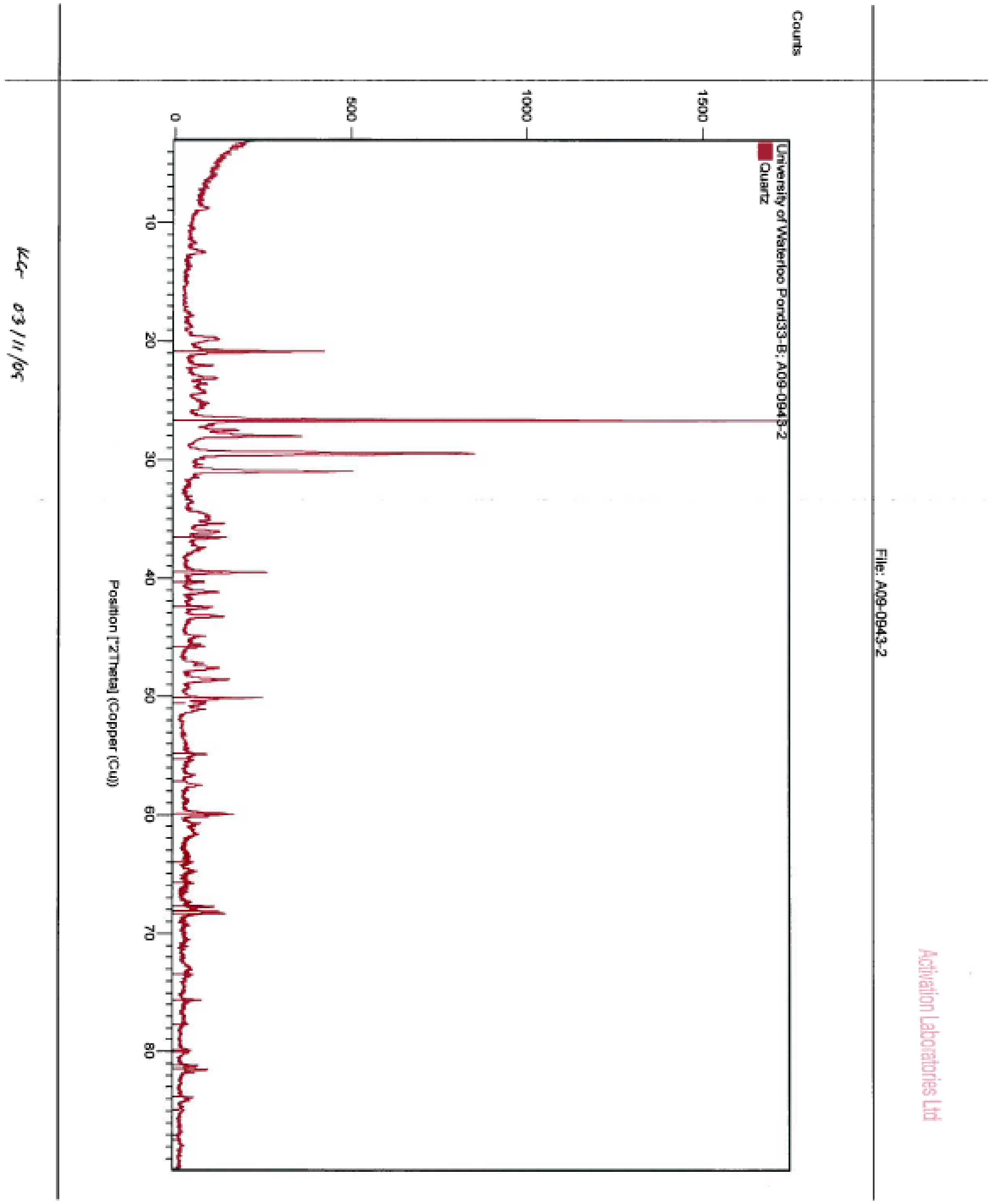
Activation Laboratories Ltd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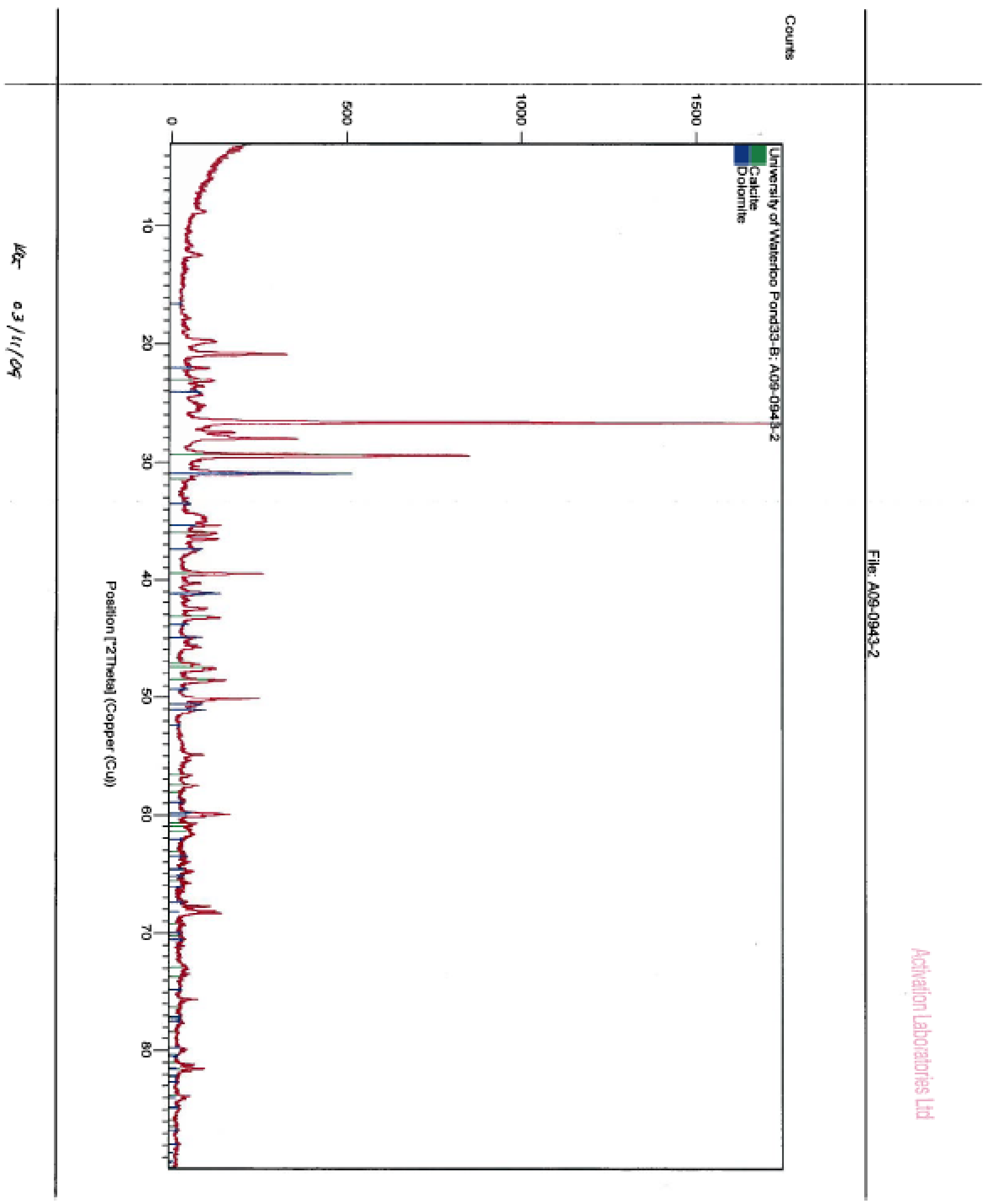
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd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Dolomite
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1500
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd

Activation Laboratories Ltd

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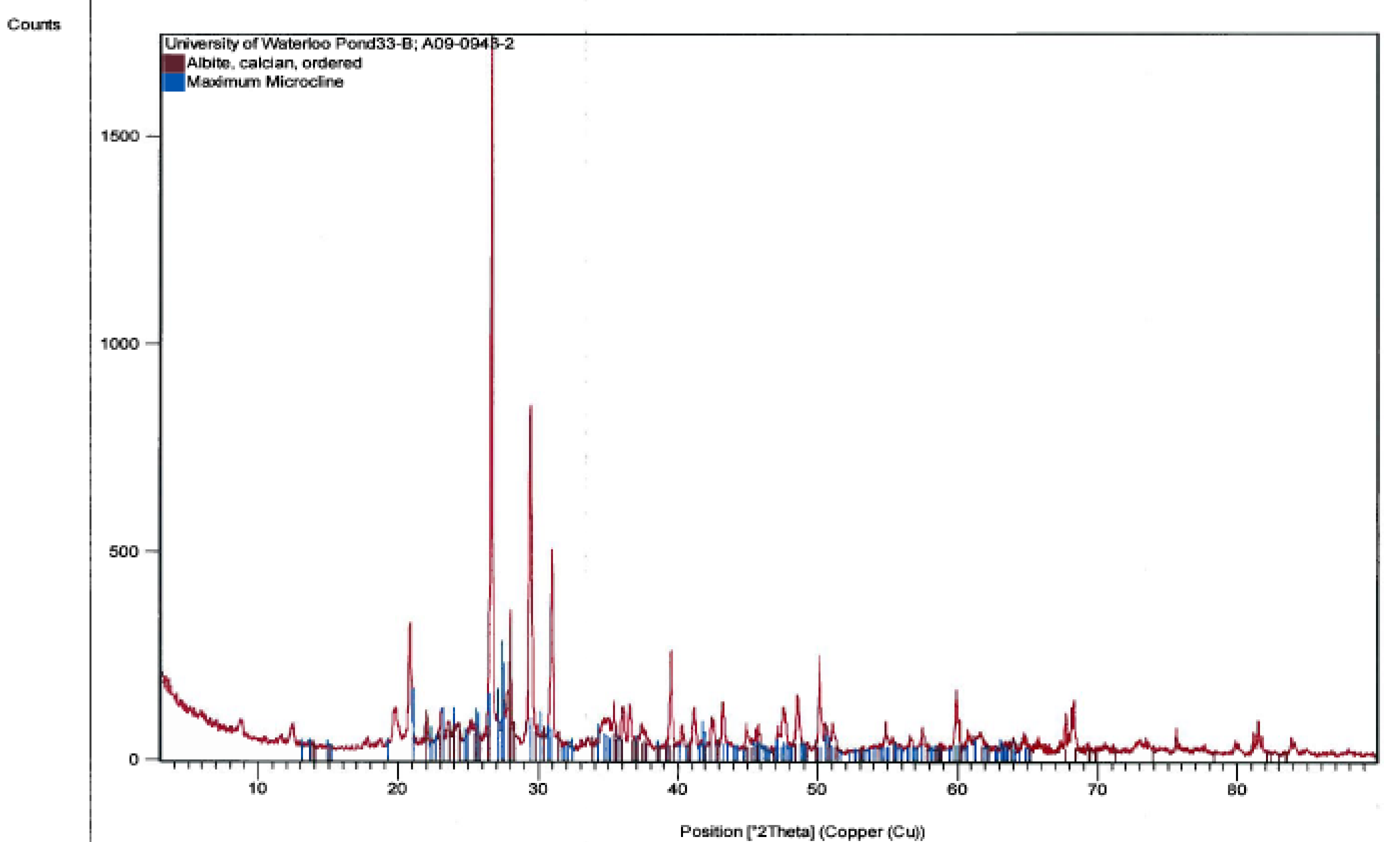

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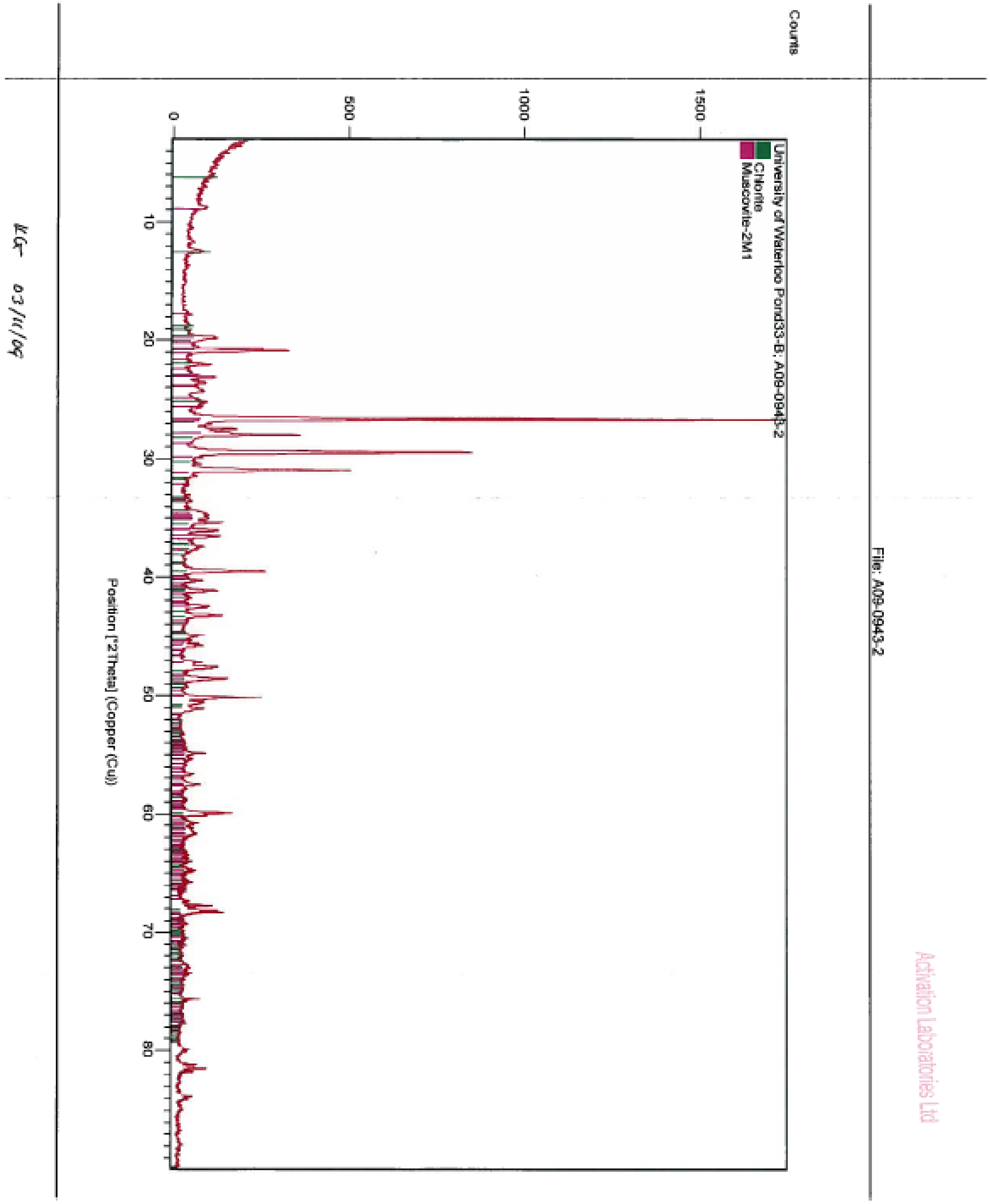

Counts
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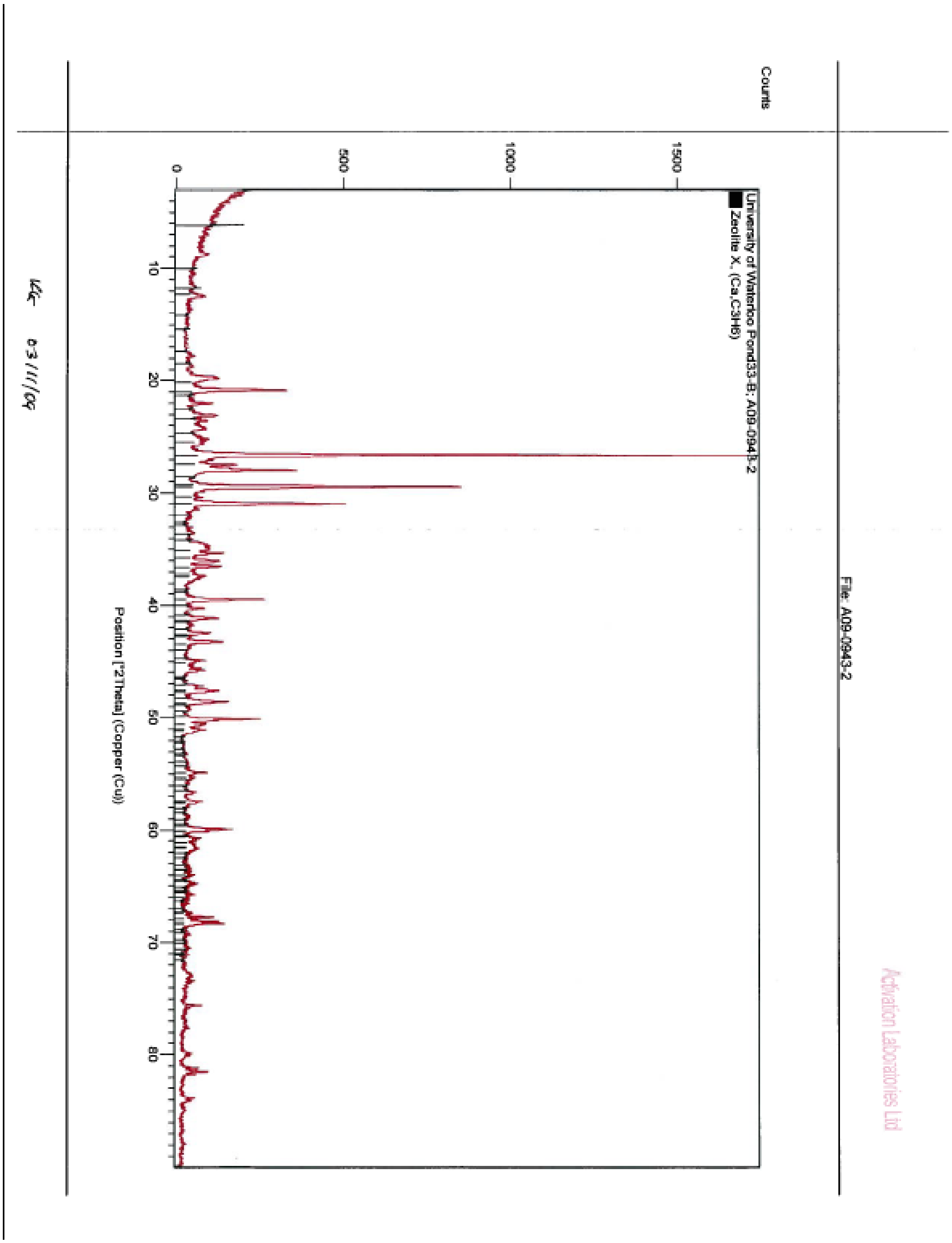
University of Waterloo Pond33-B; A09-0943-2
Chlorite
Muscovite-2M1
1500
1000
500
0
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd
KG 03/11/09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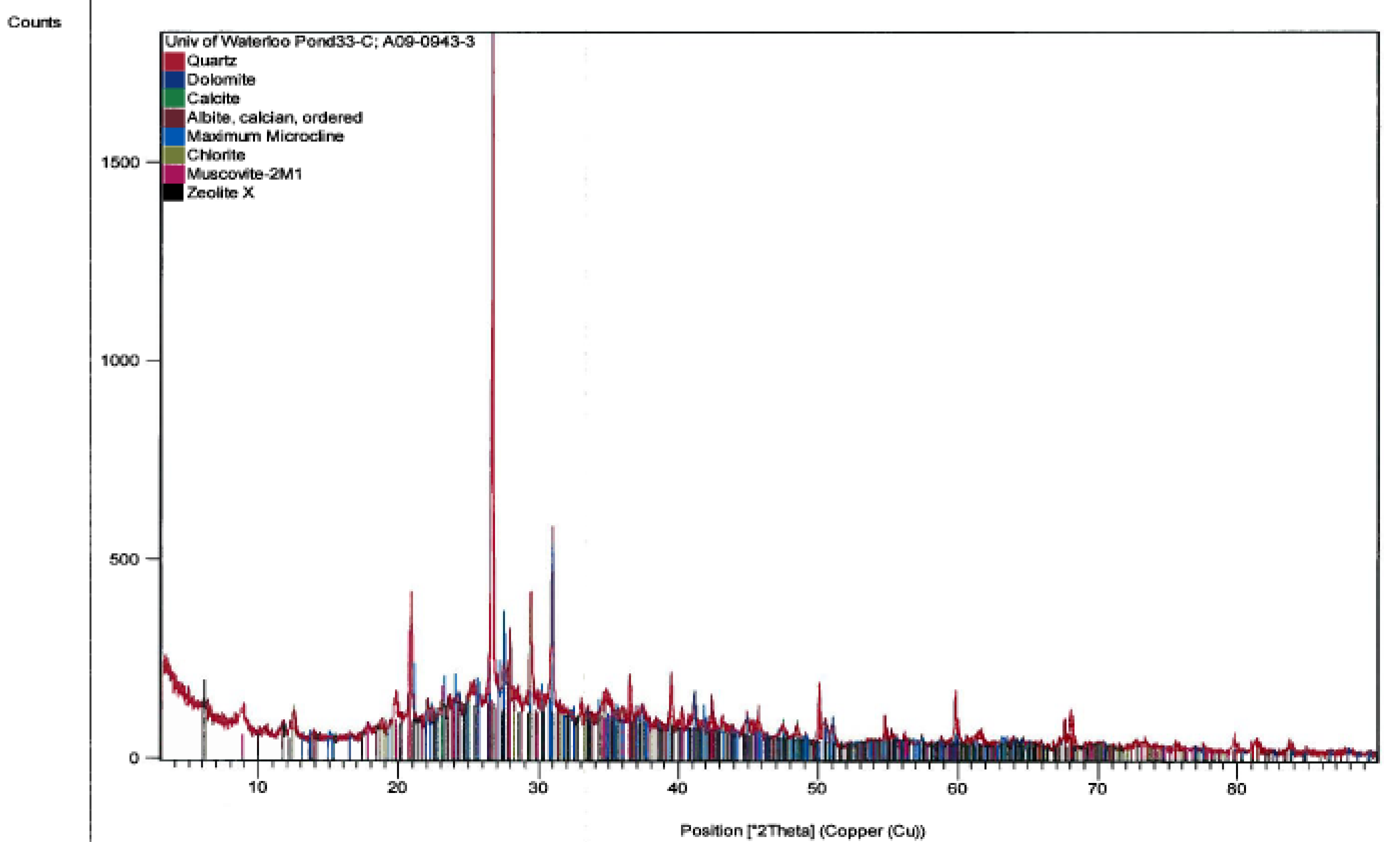
University of Waterloo Pond33-B; A09-0943-2
Zeolite X, (Ca,C3H6)
0
500
1000
1500
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd

File: A09-0943-3



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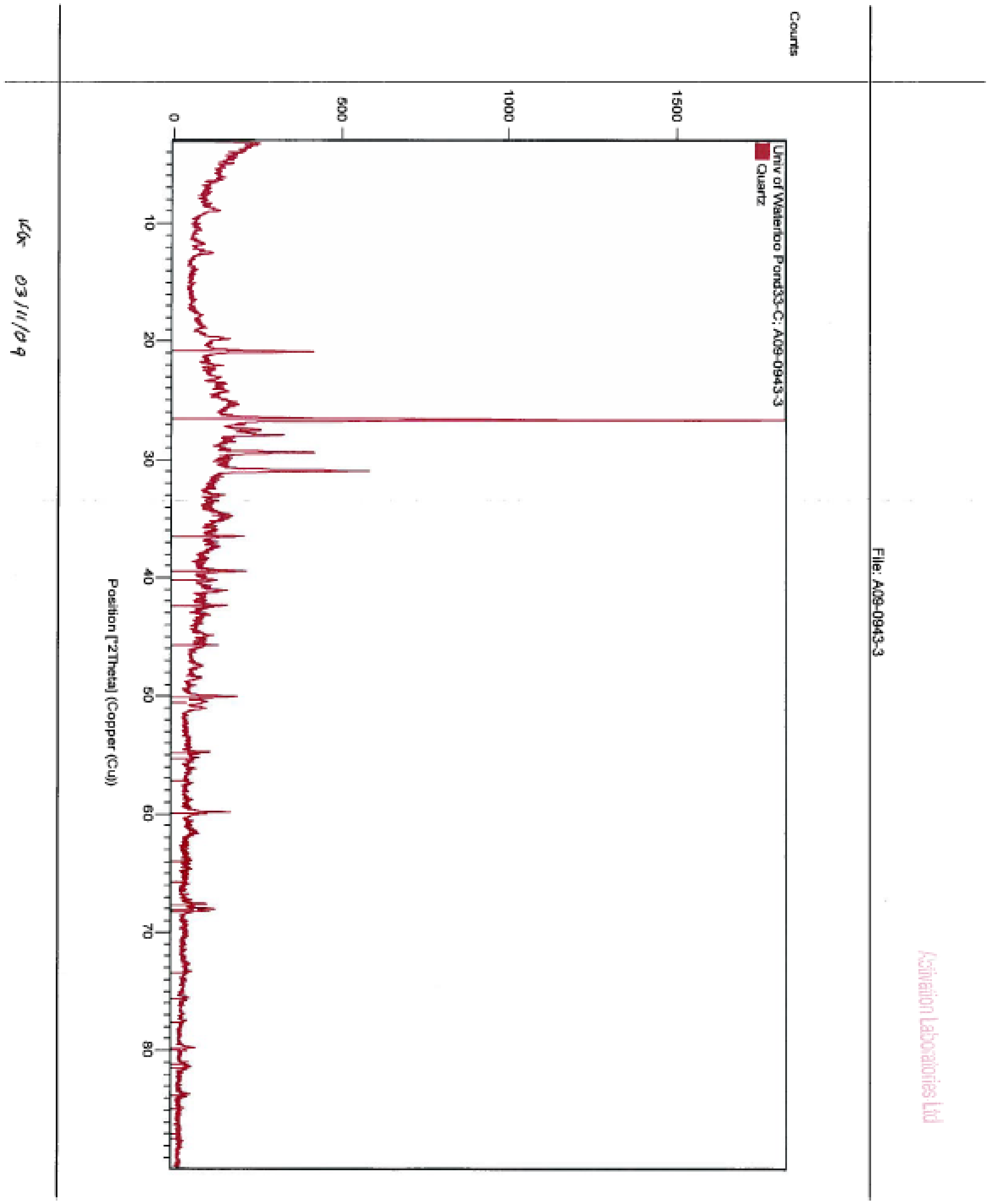
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Counts
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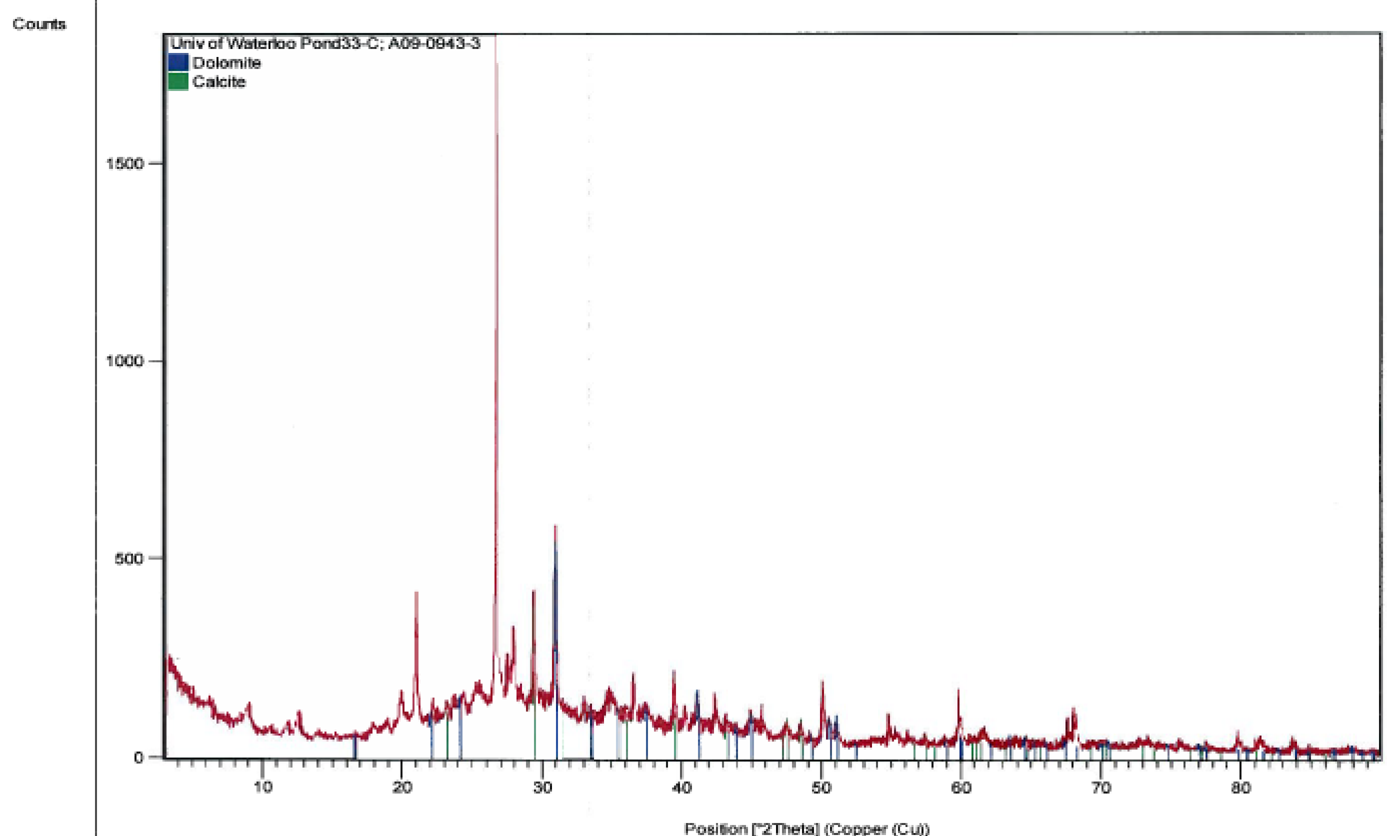
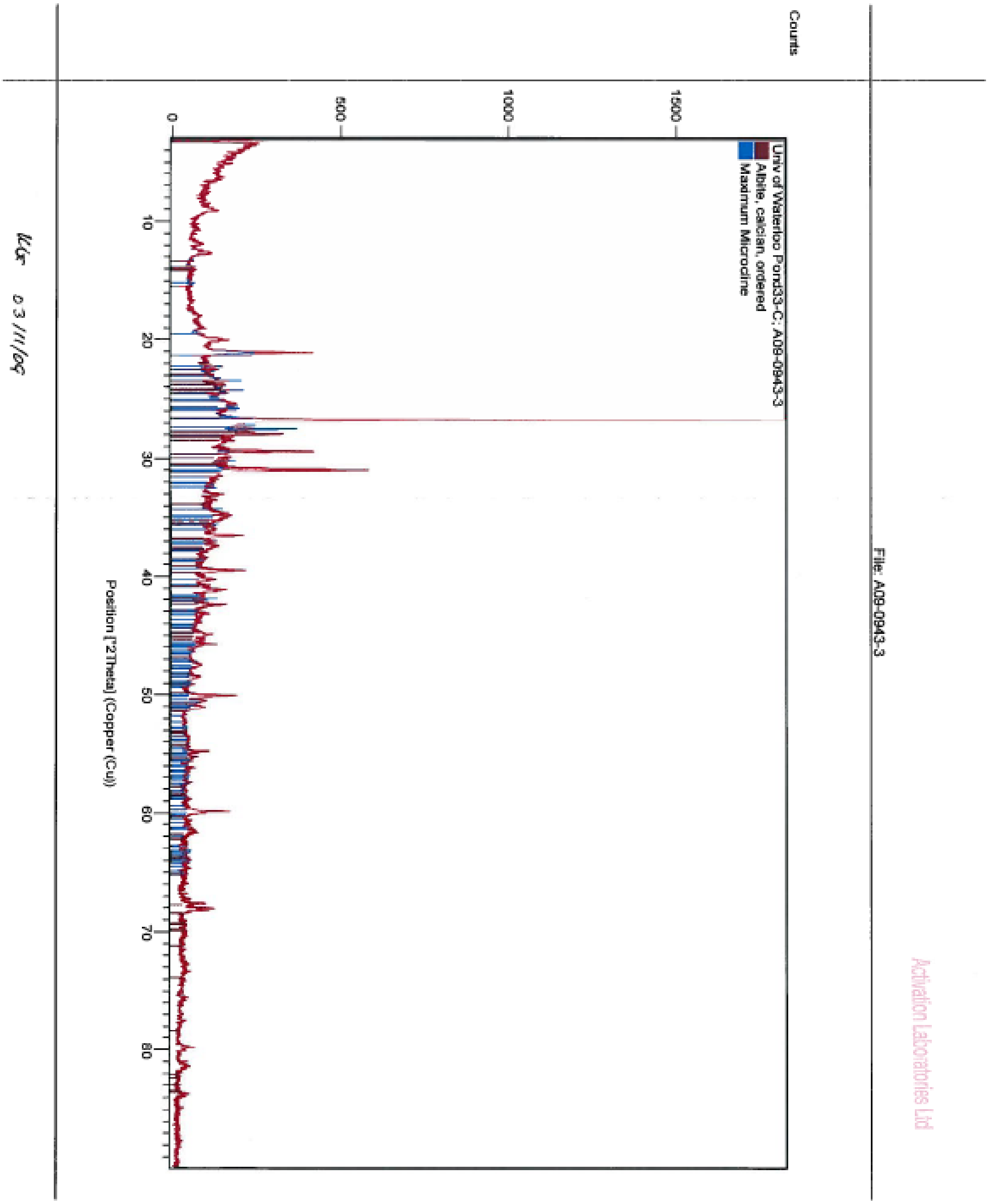
Univ of Waterloo Pond33-C; A09-0943-3
Quartz
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd

Activation Laboratories Ltd

File: A09-0943-3



KG 03/11/09


File: A08-0943-3
Counts
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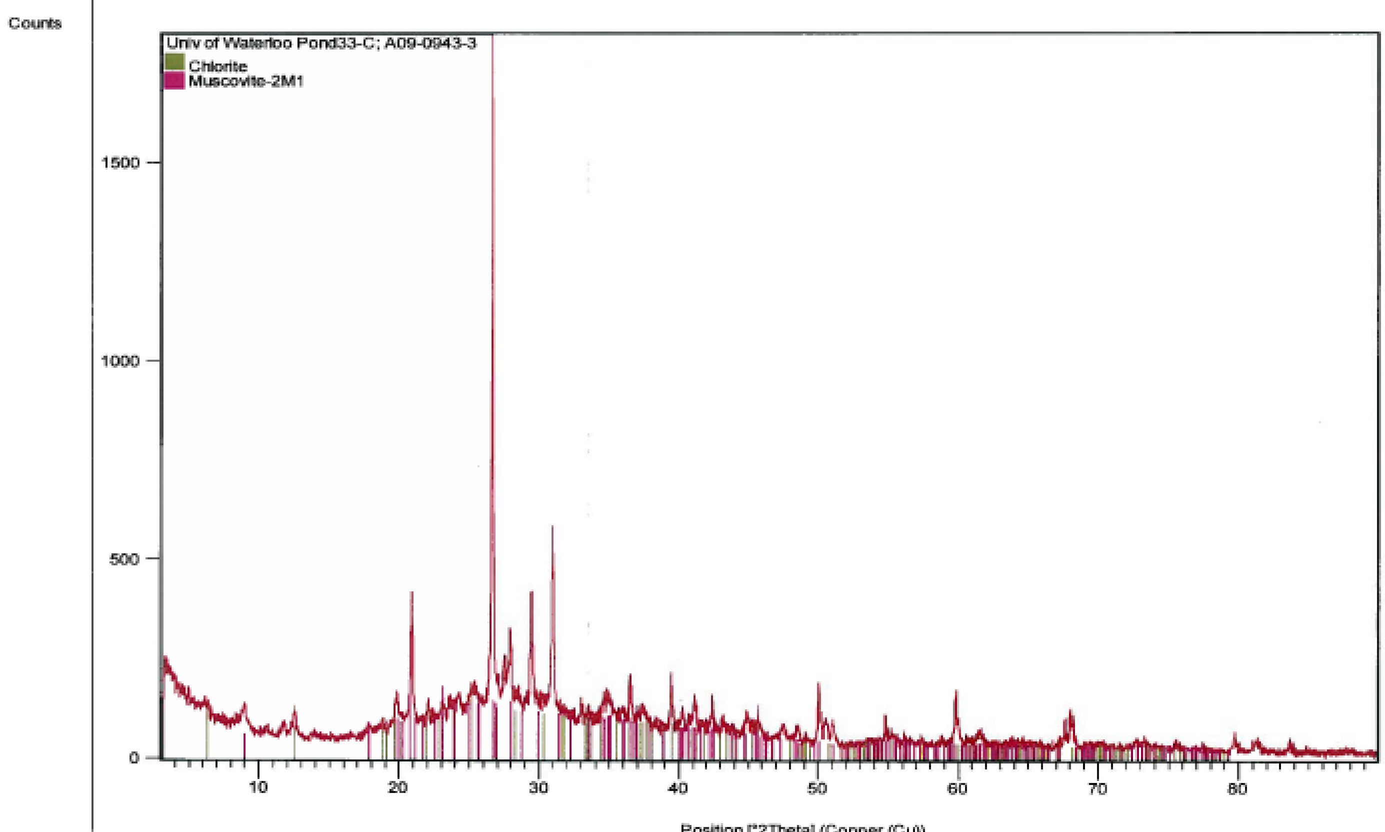
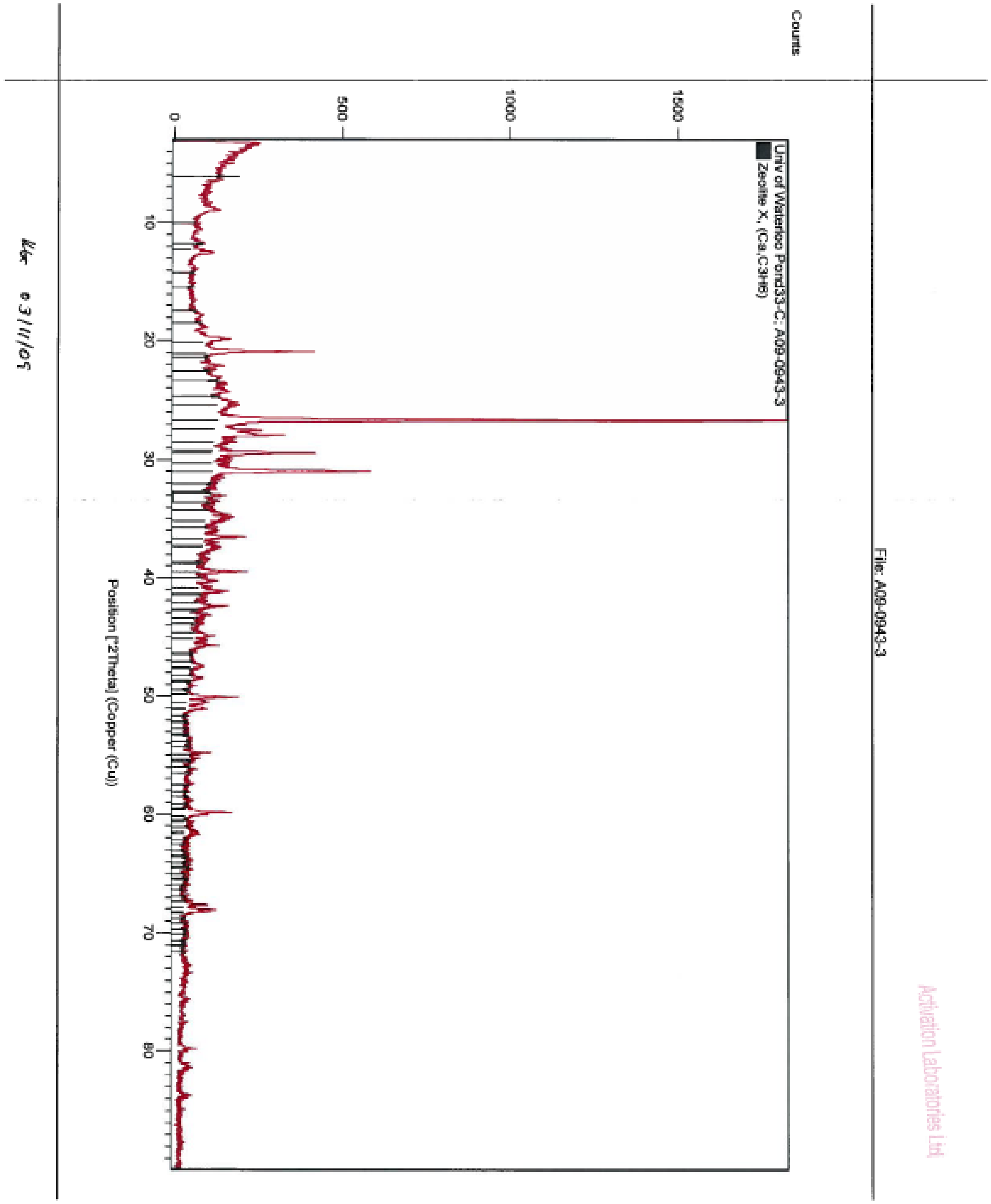
1000
500
0
Univ of Waterloo Pond33-C; A08-0943-3
Albite, calcian, ordered
Maximum Microcline
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd

Activation Laboratories Ltd

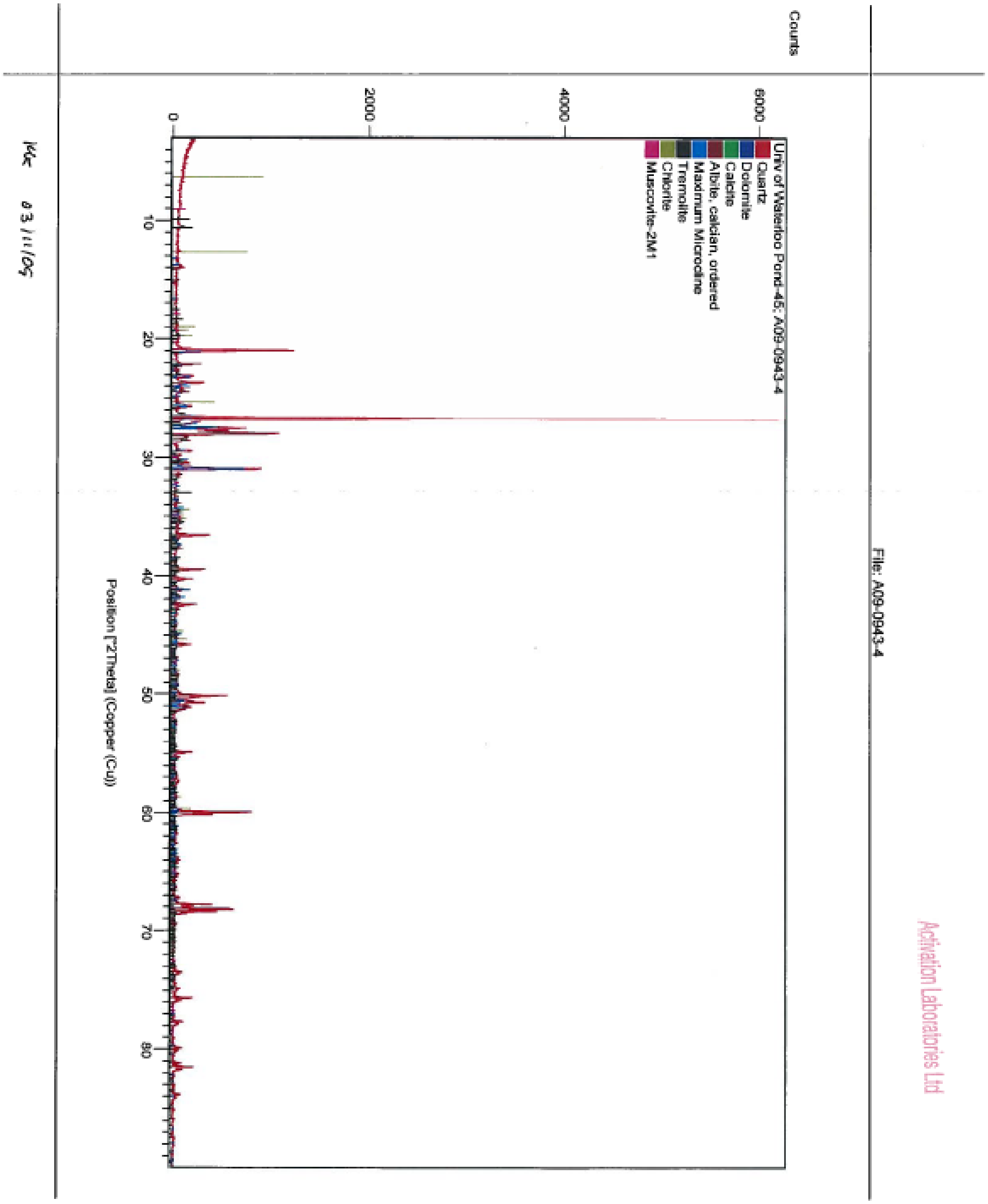
File: A09-0943-3



KG 03/11/09



File: A09-0943-3
Counts
1500
1000
500
0
Univ of Waterloo Pond33-C; A09-0943-3
Zeolite X, (Ca,C3H6)
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd.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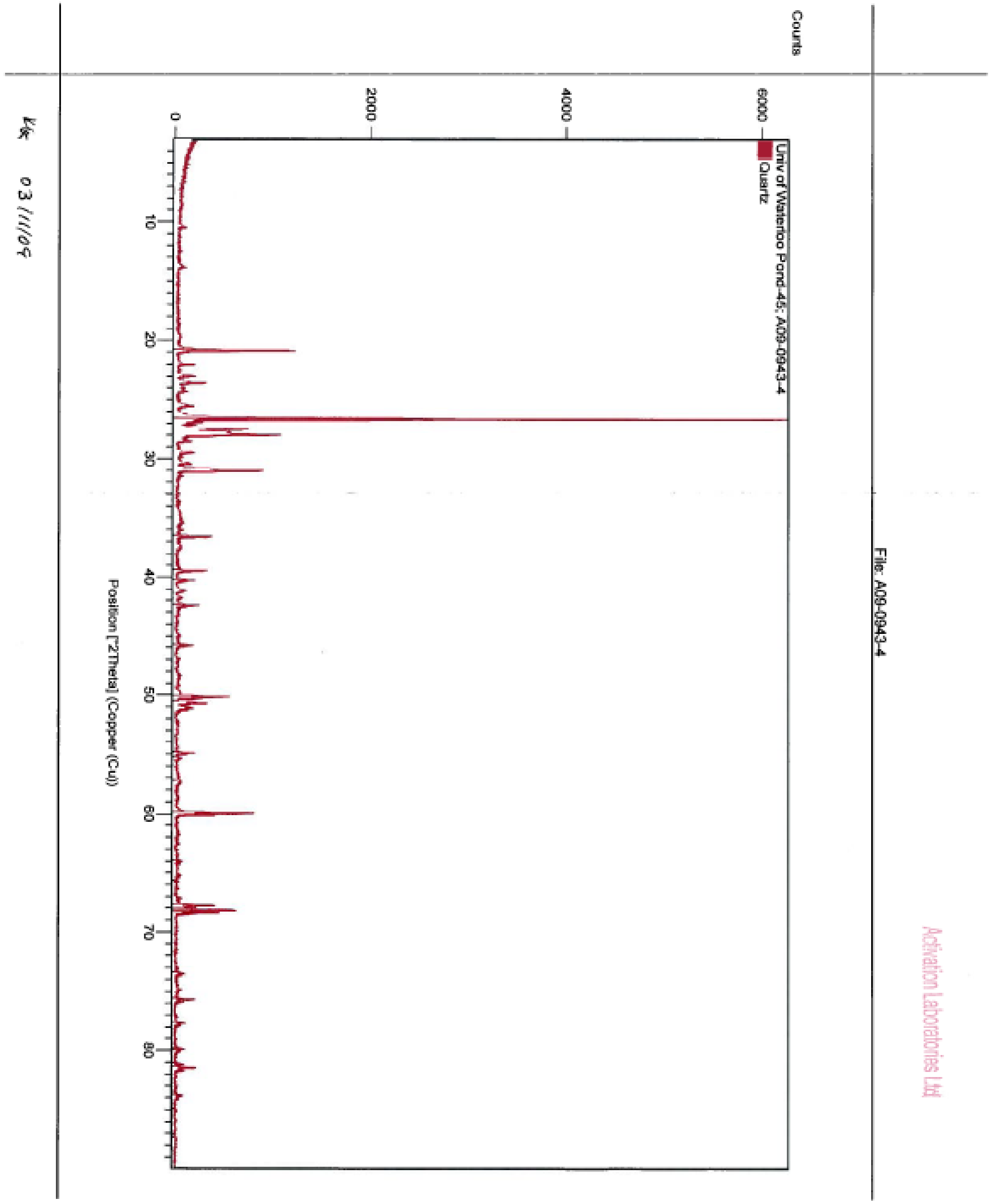
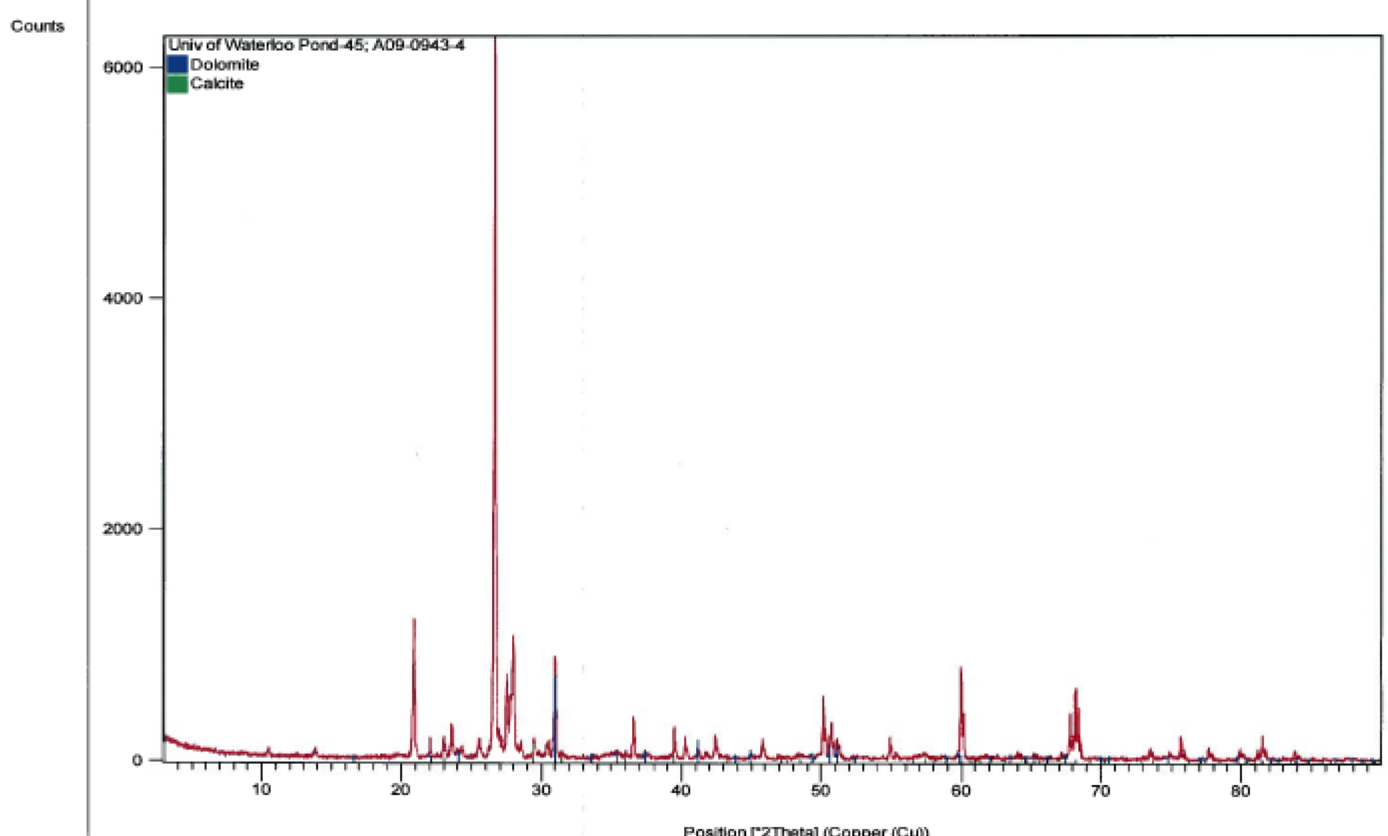
Quartz
Dolomite
Calcite
Albite, calcian, ordered
Maximum Microcline
Tremolite
Chlorite
Muscovite-2M1
6000
4000
2000
0
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd



Counts
6000
4000
2000
0
Univ of Waterloo Pond-45; A09-0943-4
Quartz
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
File: A09-0943-4
Activation Laboratories Ltd

Activation Laboratories Ltd

File: A09-0943-4



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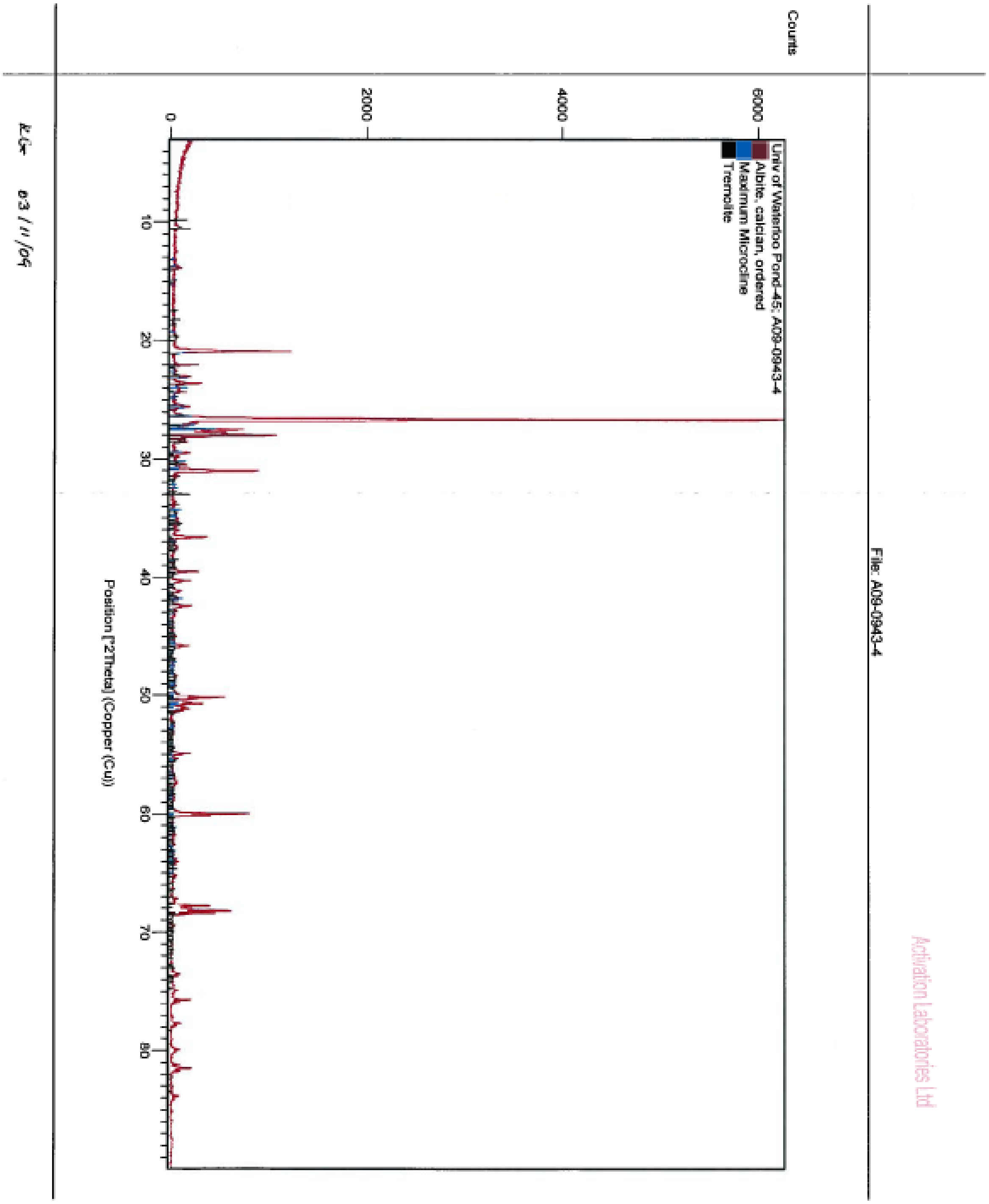
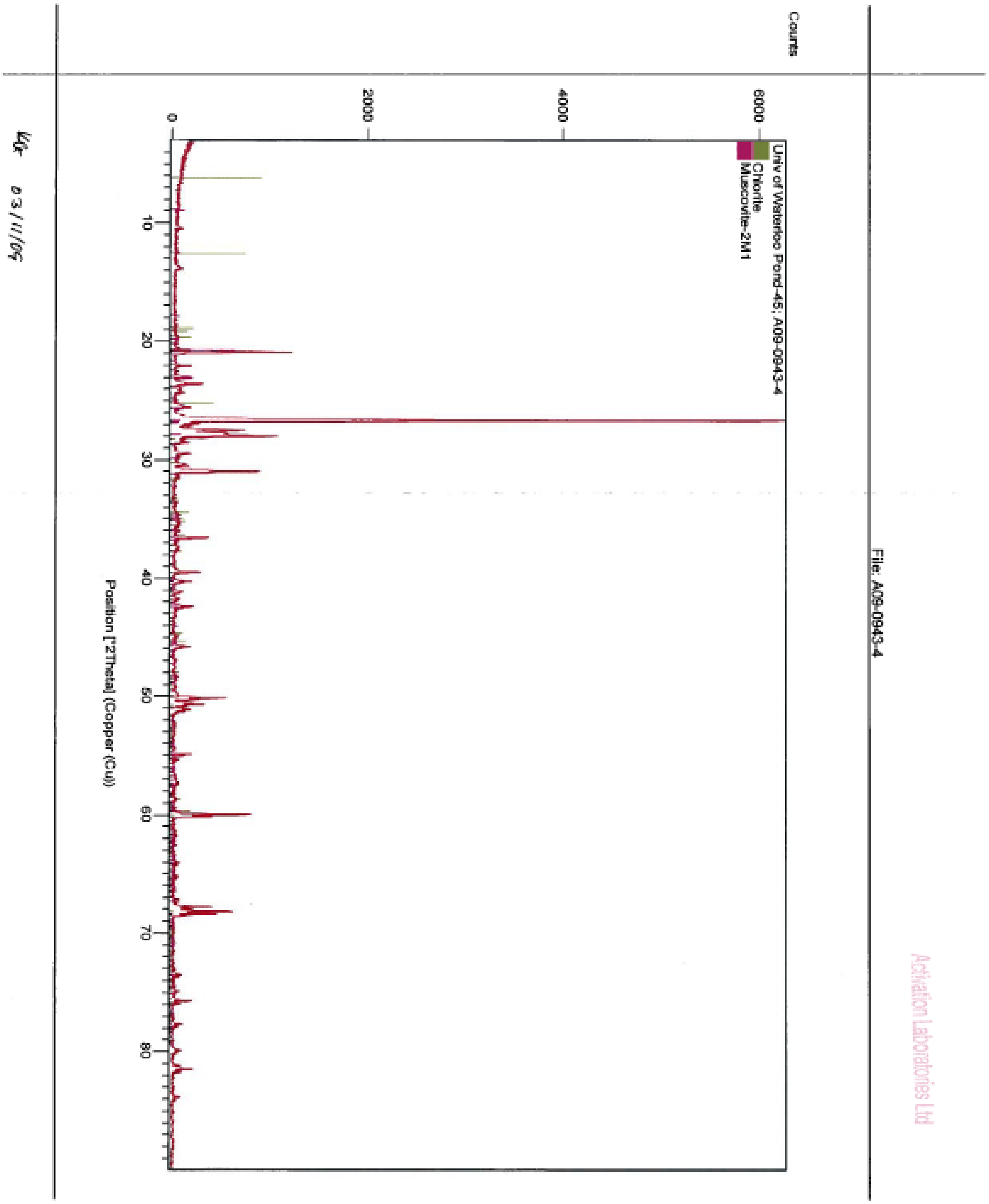
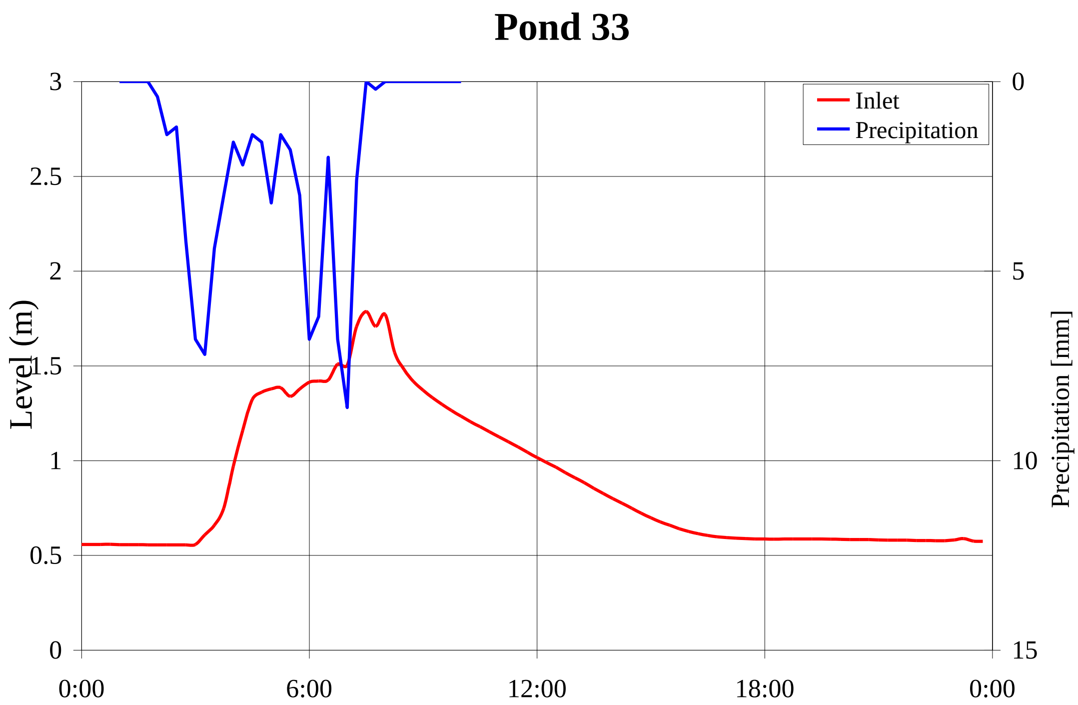
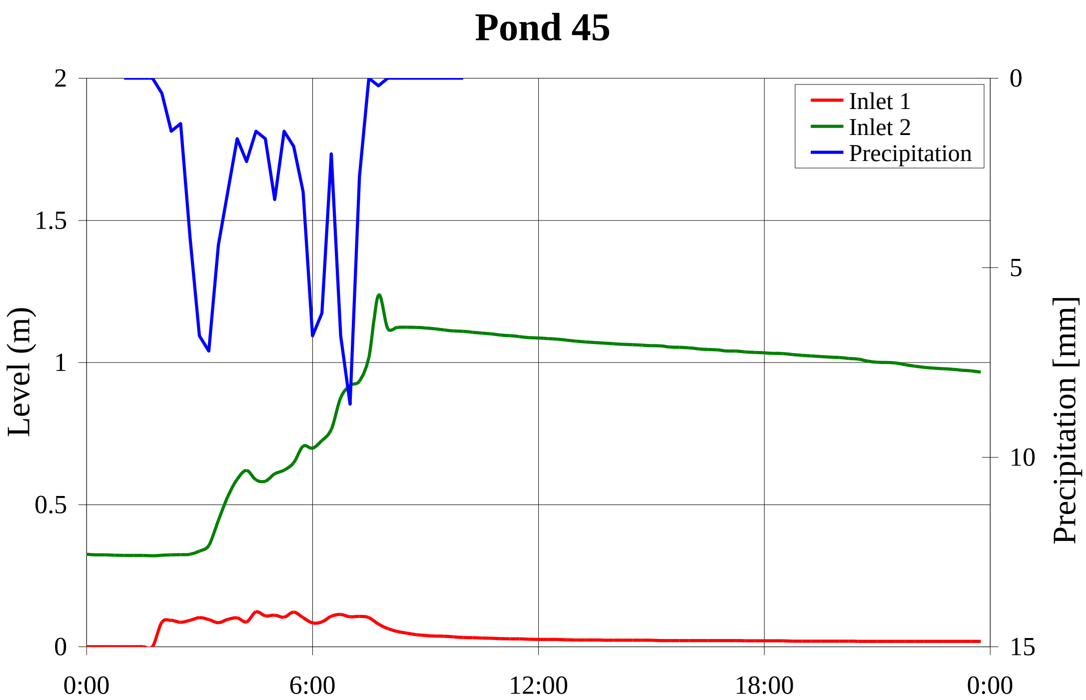

File: A09-0943-4
Counts
6000
4000
2000
0
Univ of Waterloo Pond-45; A09-0943-4
Albite, calcian, ordered
Maximum Microcline
Tremolite
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd
03/11/09

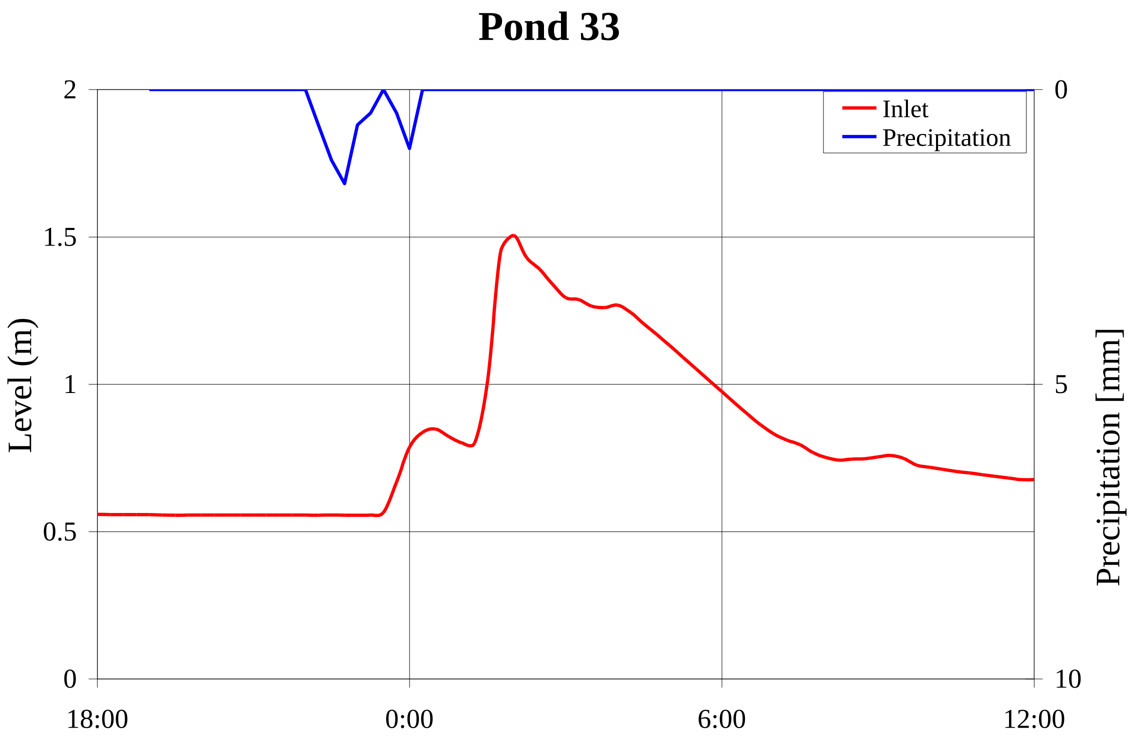

File: A09-0943-4
Counts
0
2000
4000
6000
Univ of Waterloo Pond-45; A09-0943-4
Chlorite
Muscovite-2M1
10
20
30
40
50
60
70
80
Position [°2Theta] (Copper (Cu))
Activation Laboratories Ltd

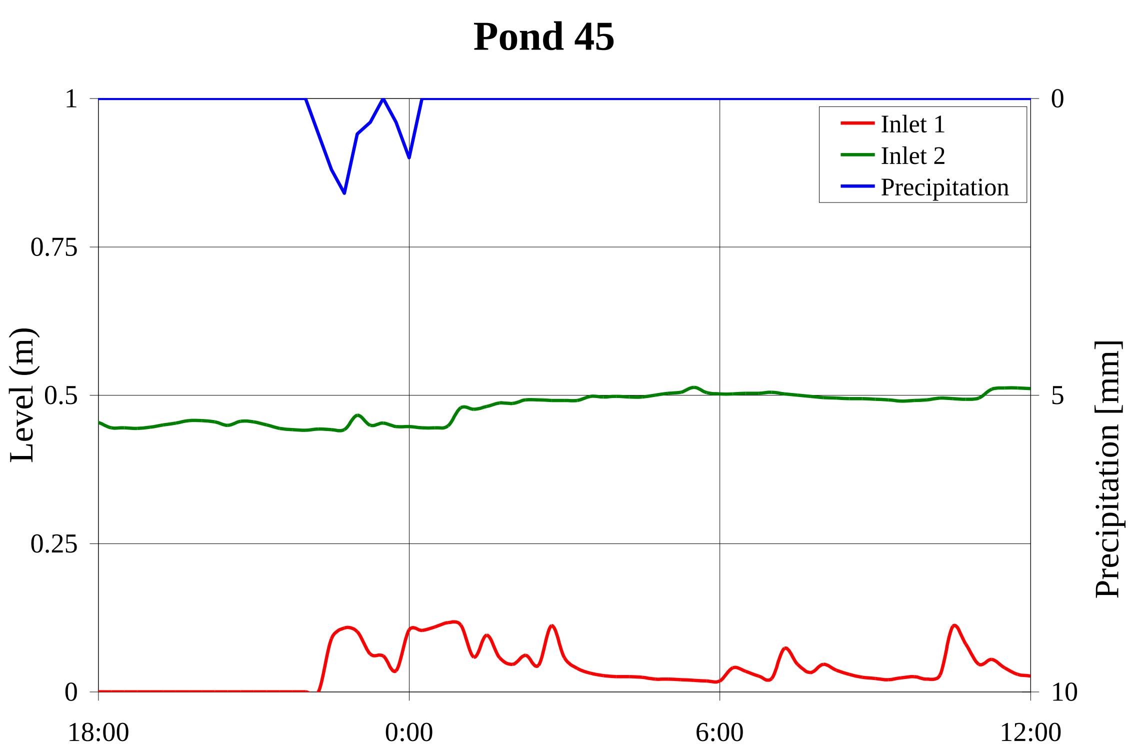
Appendix B



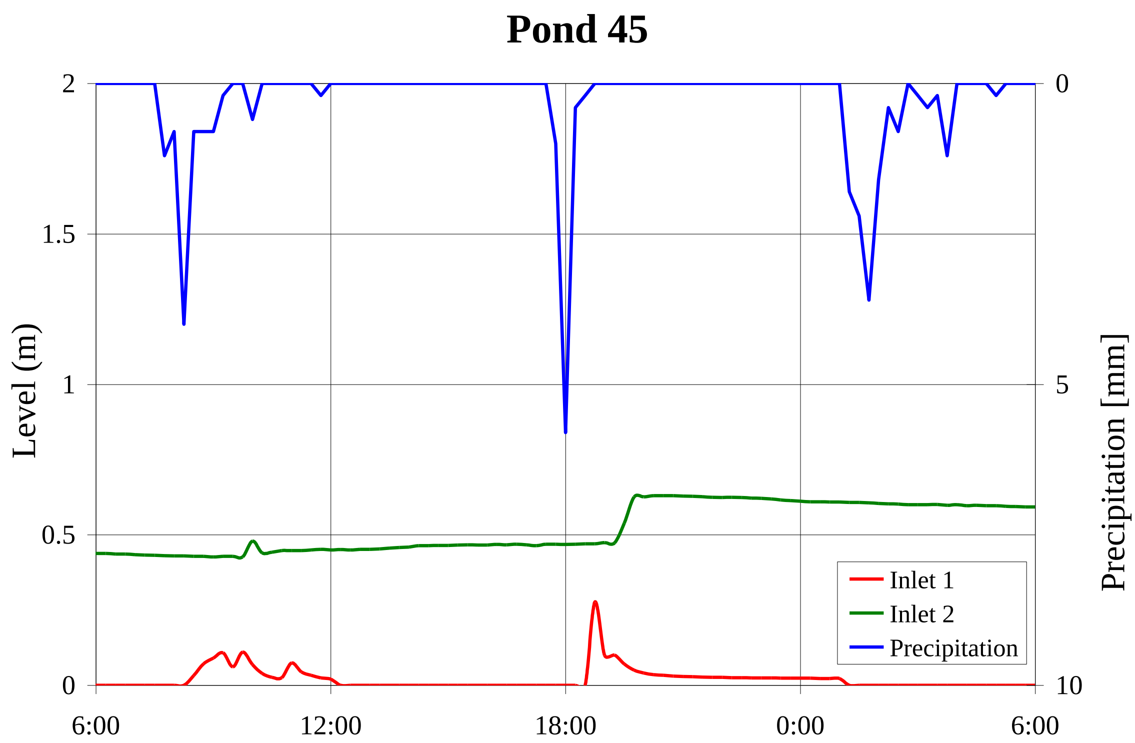


Change in water level at the inlet(s) and outlet in response to a rain event beginning on JD 193 (78 mm) for two stormwater ponds.

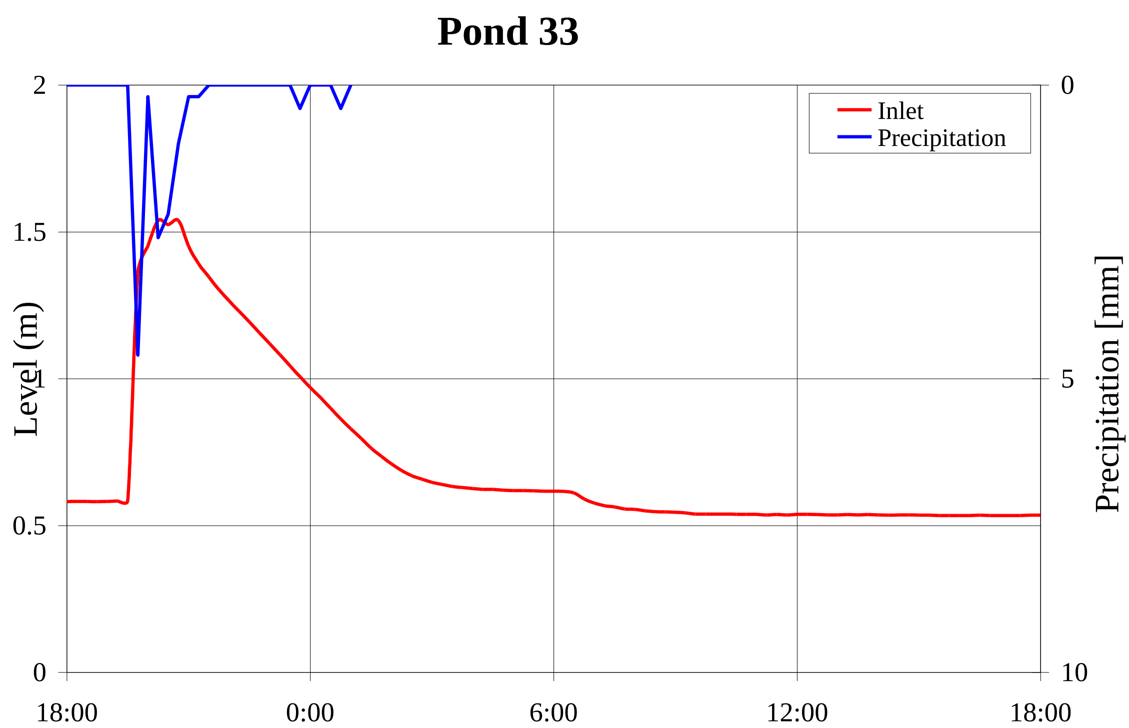


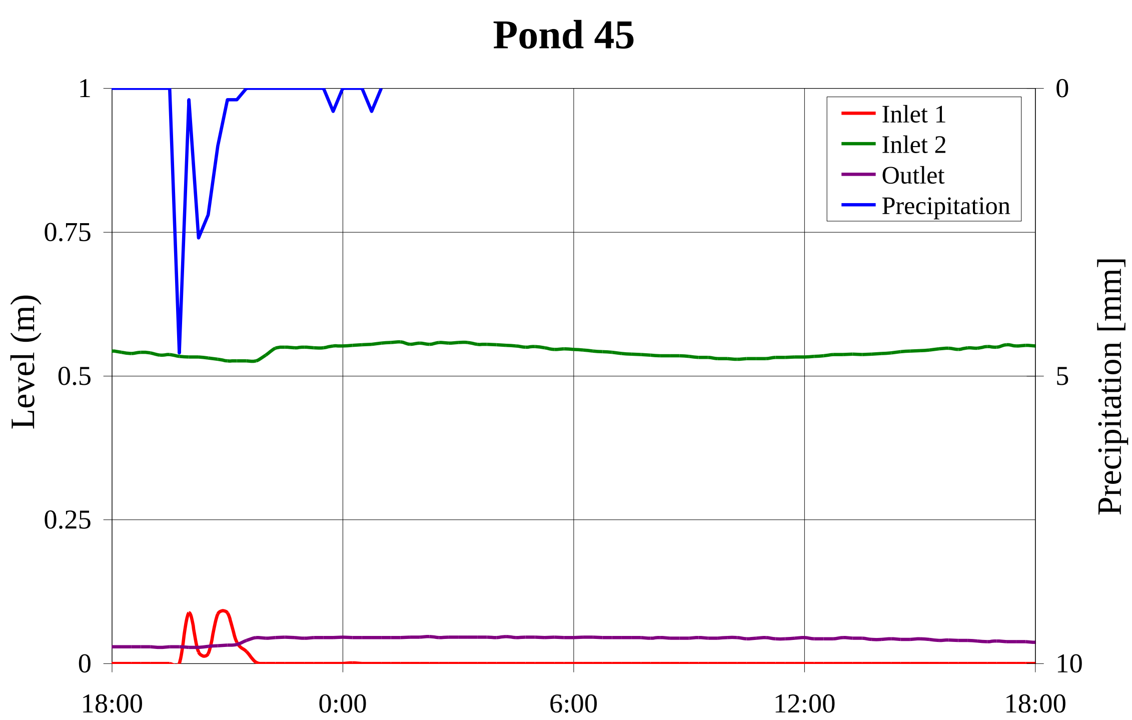


Change in water level at the inlet(s) and outlet in response to a rain event beginning on JD 201 (3.6 mm) for two stormwater ponds.

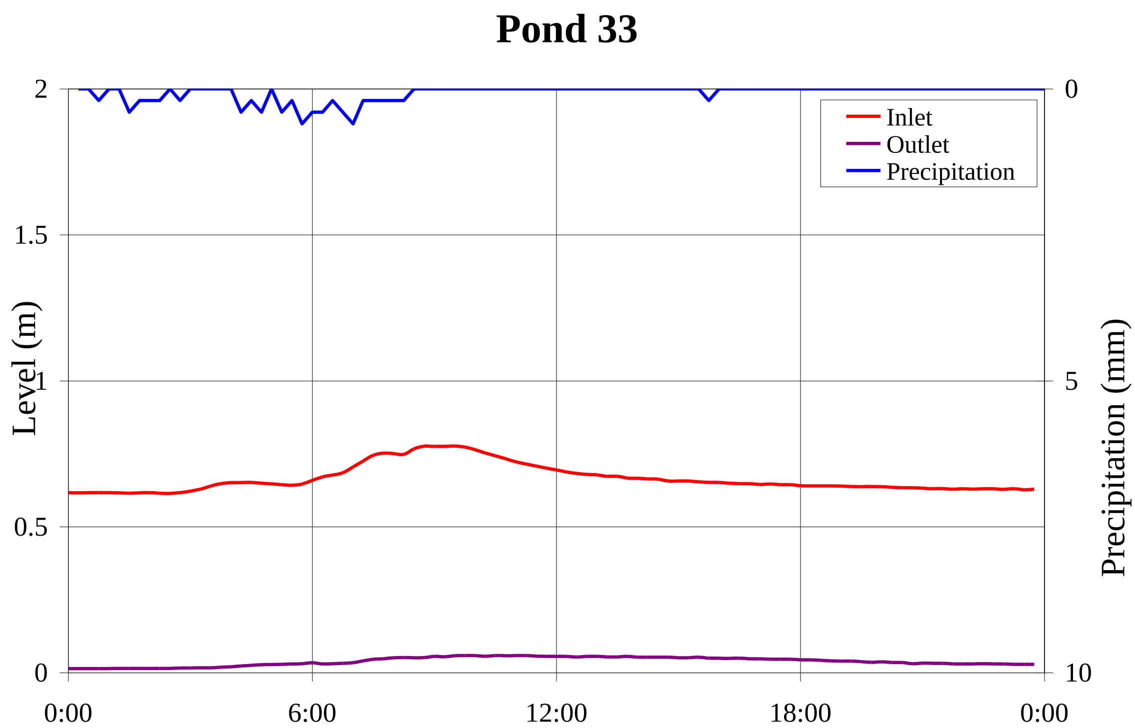


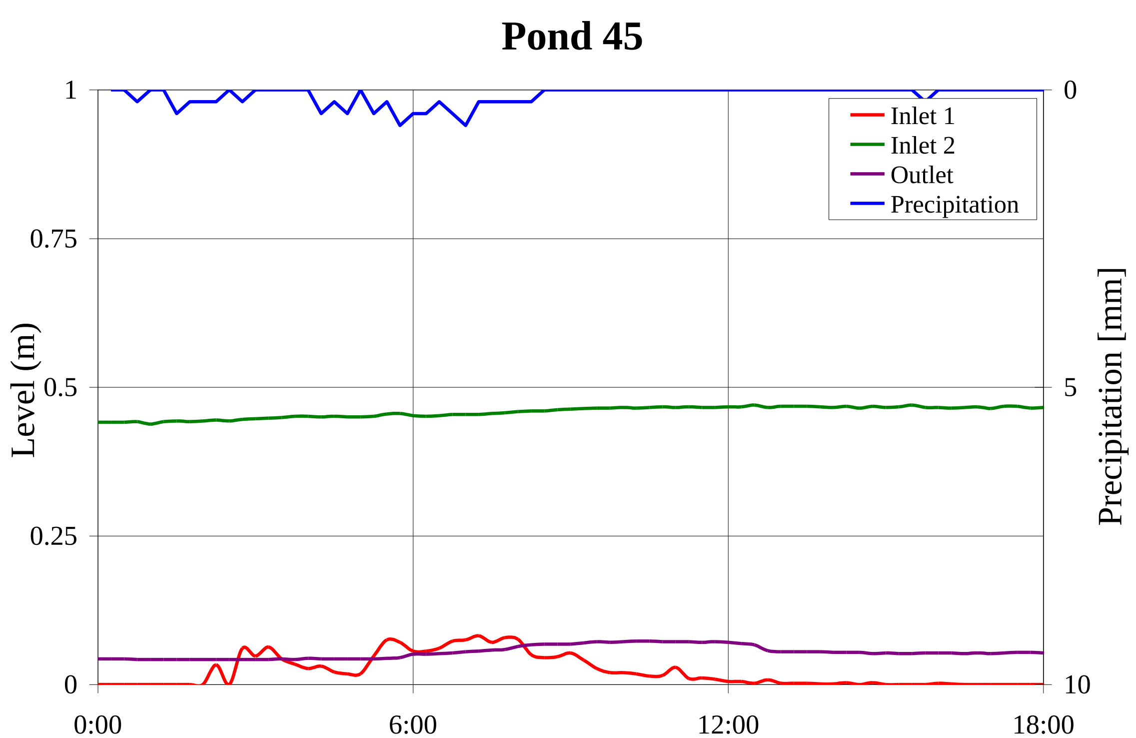
Change in water level at the inlet(s) and outlet in response to a rain event beginning on JD 212 (29.4 mm) for two stormwater ponds. Data from Pond 33 is missing due to battery failure.





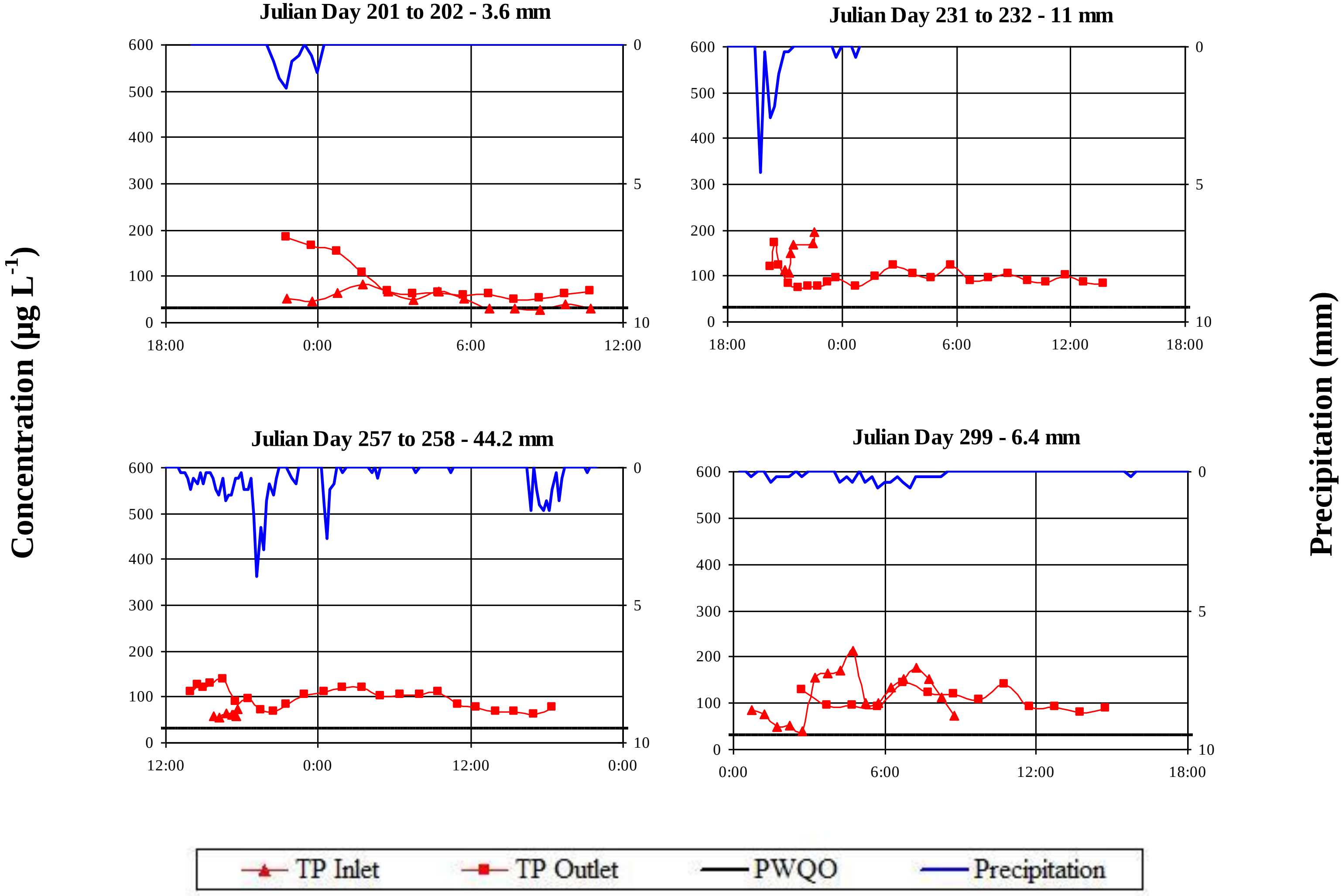
Change in water level at the inlet(s) and outlet in response to a rain event beginning on JD 231 (11.0 mm) for two stormwater ponds.



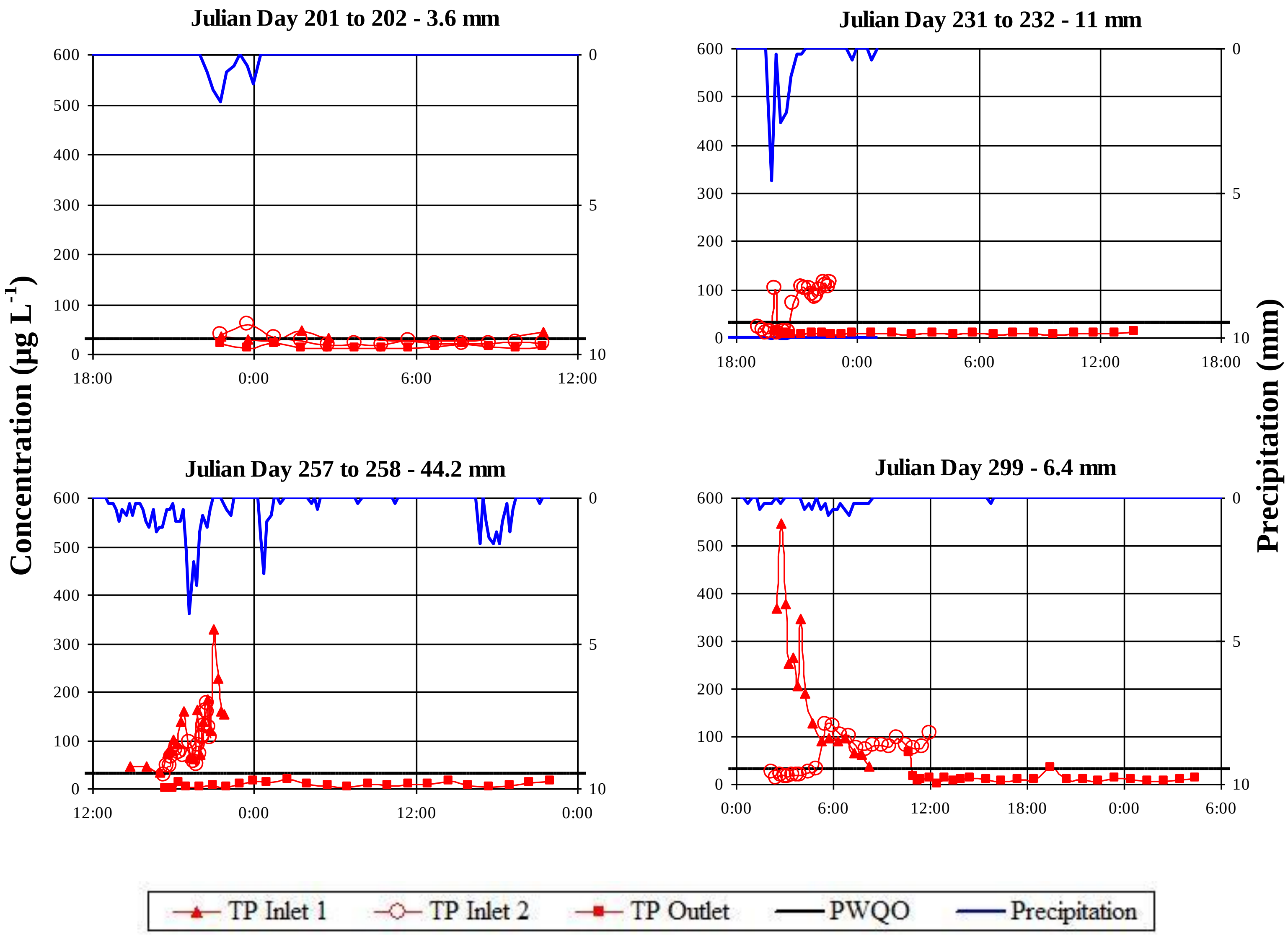


Change in water level at the inlet(s) and outlet in response to a rain event beginning on JD 299 (6.4 mm) for two stormwater ponds.

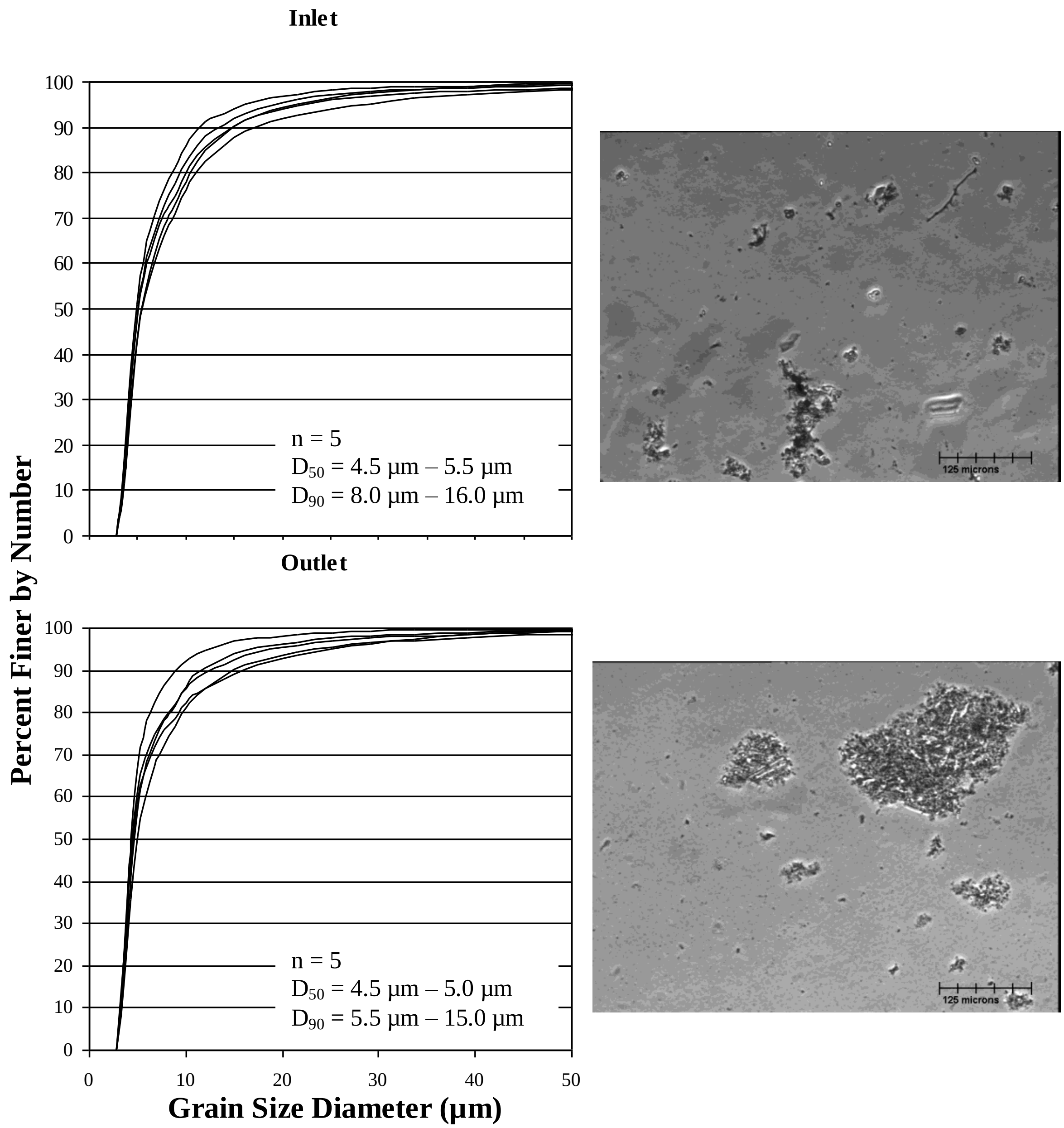
Appendix C



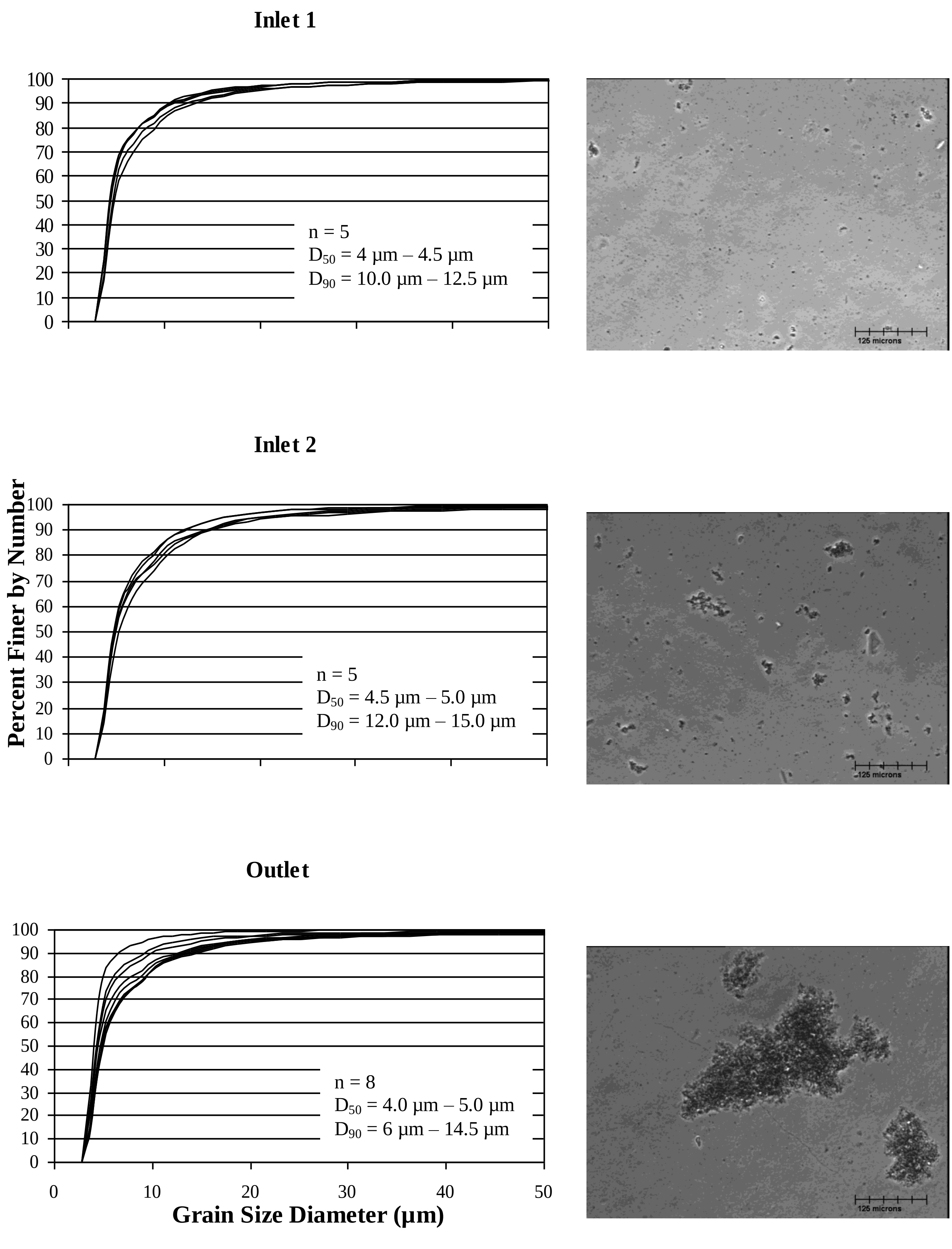
Temporal variation in total phosphorus removal in Pond 33.



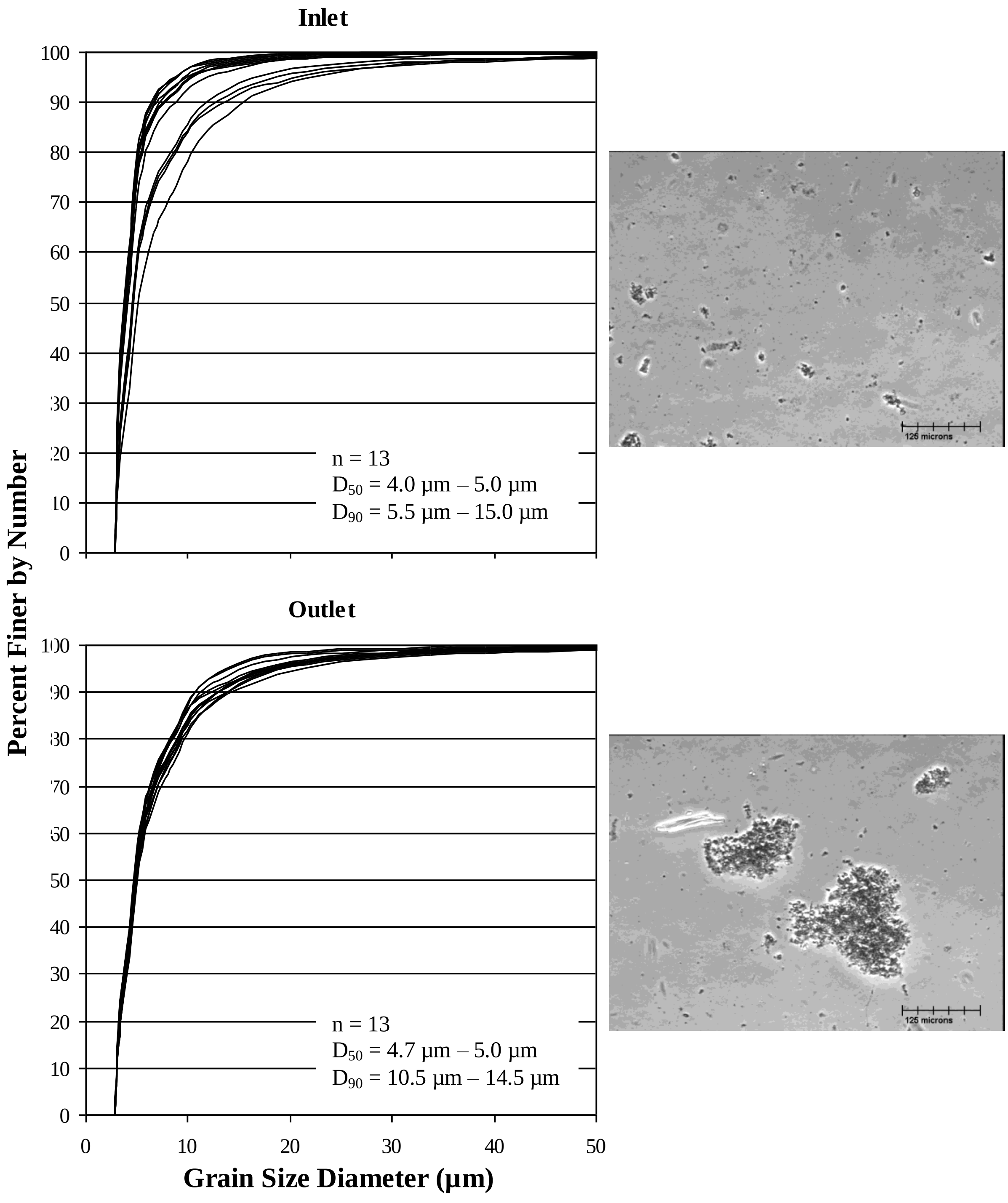
Temporal variation in total phosphorus removal in Pond 45.



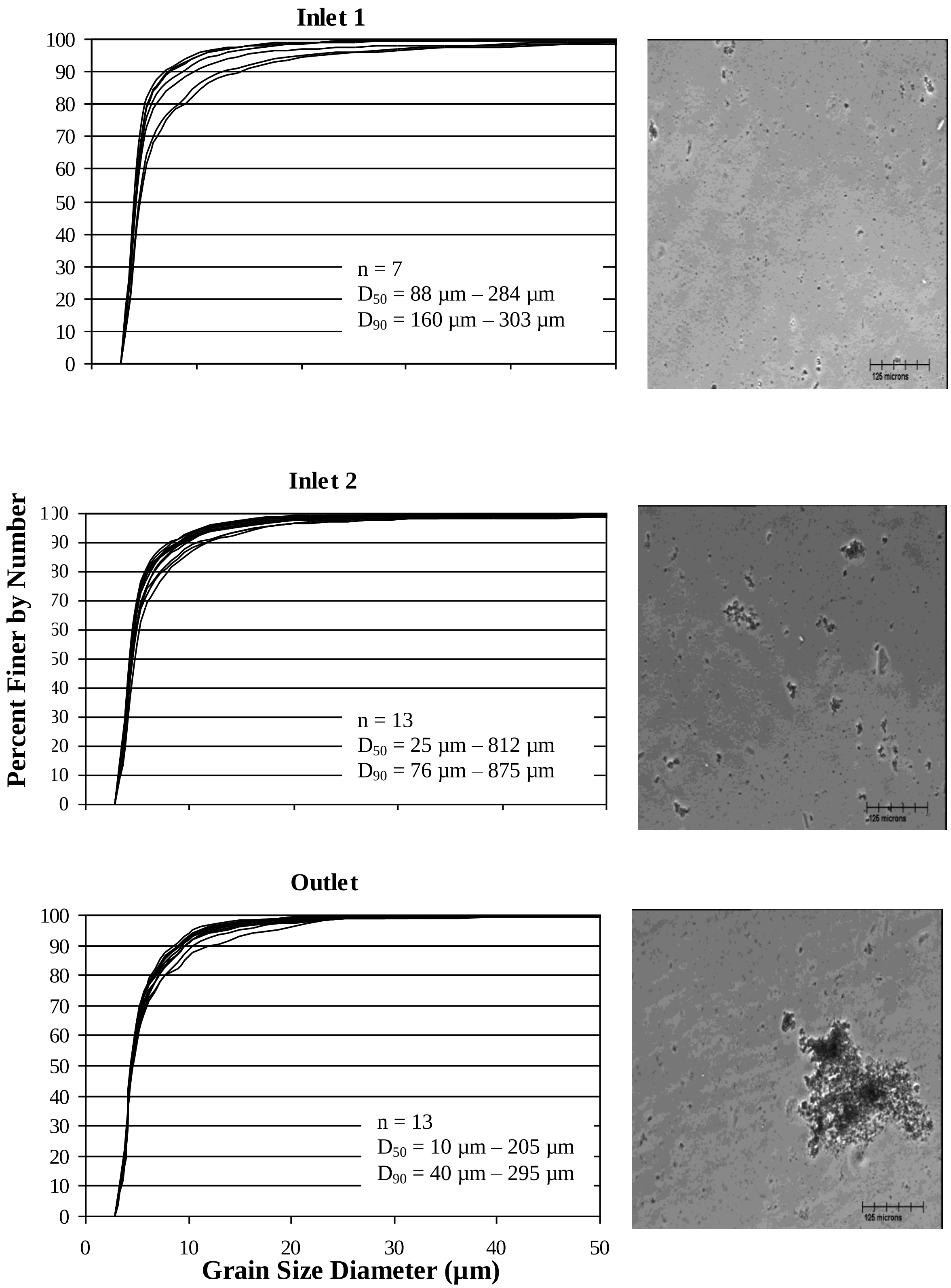
Grain size distribution of suspended sediment in Pond 33 during a storm event on JD 193. Images are representative of individual particle characteristics.



Grain size distribution of suspended sediment in Pond 45 during a storm event on JD 193. Images are representative of individual particle characteristics.



Grain size distribution of suspended sediment in Pond 33 during a storm event on JD 201 to 202. Images are representative of individual particle characteristics.



Grain size distribution of suspended sediment in Pond 45 during a storm event on JD 201 to 202. Images are representative of individual particle characteristics.

Appendix D

Site Photographs

Pond 33: a)View of Sediment Forebay; b)View of Constructed Wetland near *Typhs* peak growth period; c) Pond water flowing out of the overflow grate at the outlet during a storm event on JD 193; d) Collecting the PWPs in the sediment forebay; e) View of the constructed wetland after *Typha* senescence; and f) Outlet at Pond 33.

Pond 45: a) Inlet 1 and metal box that housed the sampling equipment; b) ISCO 6712 Automatic Sampler, 4150 Area/Velocity Flow Logger and a 12V marine deep cycle battery located at each inlet and outlet; c) Exposed pilot channel at Inlet 1; and d) Hickenbottom outlet at Pond 45; e) View of Pond 45 from the east; f) Canadian Geese who live at Pond 45.

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