Has the Redesign of Columbia Lake Improved Water Quality in Laurel Creek?

By

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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.

I understand that my thesis may be made electronically available to the public.

Abstract

Stormwater impoundments are one of many types of best management practices (BMP) designed and implemented to regulate water quantity and improve the quality of runoff from urban areas. Studies of water quality in urban impoundments have indicated that conventional designs are however, not very effective at removing solids and associated pollutants. Accordingly, many urban impoundments are being re-designed to improve downstream water quality. However, few studies have systematically monitored and quantified post-design water quality improvements of urban impoundments. This thesis examines changes in the water quality performance of an urban impoundment (Columbia Lake) in Waterloo, Ontario resulting from redesign of the lake for the pre-design period (2003 and 2004) and the post-design period (2006 and 2007). To achieve this goal, four years of water quality data collected at the inlet and outlet of Columbia Lake as part of the Laurel Creek Monitoring Program was measured. Water chemistry parameters included total phosphorus (TP), soluble reactive phosphorus (SRP), suspended solids (SS), dissolved oxygen (DO), pH and total dissolved solids (TDS). Inlet and outlet discharge (Q) were measured to determine the water retention time in the lake. Concentrations and loads of TP and SS for the post-design period (2006 and 2007) were compared to those for the pre-design period (2003 and 2004).

During the pre-design period (2003 and 2004), inflow TP concentrations ranged from 18 to 372 $\mu g\ L^{-1}$ with an average (mean ± standard error) of 56±7 $\mu g\ L^{-1}$, while outflow TP concentrations ranged from 37 to 266 $\mu g\ L^{-1}$ with an average of 116±6 $\mu g\ L^{-1}$. Post-design TP concentrations ranged from 10 to 124 $\mu g\ L^{-1}$ with an average of 53±5 $\mu g\ L^{-1}$ and from 14 to 147 $\mu g\ L^{-1}$ with an average of 44±3 $\mu g\ L^{-1}$ at the inflow and outflow, respectively. Pre-design SS concentrations ranged from 1.8 to 168.5 mg L^{-1} with a mean of 19.0±3.2 mg L^{-1} and from 4.0 to 194.7 mg L^{-1} with a mean of 66.6±4.7 mg L^{-1} at the inflow and outflow, respectively. Post-design SS concentrations varied from < 0.1 to 25.8 mg L^{-1} with an average of 8.5±0.8 mg L^{-1} and from < 0.1 to 42.5 mg L^{-1} with an average of 14.5±0.8 mg L^{-1} at the inflow and outflow, respectively. Sedimentation/resuspension dominated the TP and SS transfer via Columbia Lake. Pre-design TP loads (log-transformed) strongly correlated with SS loads at the inflow and outflow ($r = 0.661$ and 0.777, $p = 0.0001$). These parameters were more strongly correlated during the post-design period ($r = 0.794$ and 0.915, $r = 0.0001$), which indicates that particulate P (PP) was a dominant fraction of TP and that the release of dissolved phosphorus (DP) from bottom sediments was considerably decreased following the redesign. No significant difference was observed between inflow and outflow SRP concentrations. Discharge strongly affected TP and SS loads at the inflow and outflow during the pre- and post-design periods (r

> 0.79, $p = 0.000$ for all). After the redesign of Columbia Lake, the average net internal P loading rate decreased from 198% to 22% for TP. The primary factor influencing the observed decreased post-design TP and SS outputs was the removal of sediment from the lake. Bottom sediment removal and changes to the lake bathymetry reduced sediment resuspension and P desorption, which decreased the average net internal SS loading rate from 828% to 154%. The Columbia Lake Water Quality Model developed by Stantec Consulting Ltd. (2004) underestimated the post-design outflow TP and SS concentrations mainly because it did not include terms that account for factors such as bioturbation, wave induced resuspension and biological activity.

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Chapter 1 INTRODUCTION

1.1 Problem Statement

Since 1980, over 12,000 km^2 of land has been urbanized in Canada (Hofmann et al., 2005). The resulting land use change has substantially increased the impervious surface cover (ISC), which significantly affected the quantity and quality of both surface and ground water systems (Arnold and Gibbons, 1996; Hogan and Walbridge, 2007). Stormwater runoff from ISC areas often contains high levels of sediments and associated nutrients such as phosphorus (P) (Brabec et al., 2002). The transfer of these nutrient rich materials from terrestrial to aquatic environments can the eutrophication of receiving waters (Carpenter et al., 1998), which can ultimately impact ecosystem health as well as the quality of drinking water sources.

To mitigate some of the adverse effects associated with urban development, a wide range of structural and vegetative Best Management Practices (BMPs) have been implemented in Ontario through the subwatershed planning process. This planning process is used to design Stormwater Management Practices that address stormwater quality, quantity and erosion concerns as an assumed part of the development form. The implementation procedure determines the management options and the level of controls (e.g., lot level, conveyance level, end of pipe) to be used, and then tests the performance of these options on the key physical and biological attributes of the watershed (OMEE, 1994). Increasing emphasis and reliance are being placed on the performance of management options to meet targets for environmental protection, set through the recent development of subwatershed strategies and their use in planning for future land use and resource management in Ontario.

Urban impoundments are important conveyance and end-of-pipe water storage measures that are designed to mitigate downstream flooding (Van Buren, 1997; OMEE, 2003), by reducing peak runoff rates after development and providing flow augmentation to downstream reaches (Shantz et al., 2004; Shammaa et al., 2002). Although they are widely regarded as effective in enhancing water quality in watersheds (Alaoui-Mjamdi, 1996, James et al., 2004; Istvanovics and Somlyody, 1999; Salvia-Castellvi et al., 2001), their performance varies greatly from site to site, due to differences in stormwater characteristics, and in the climate, design, size, shape and mode of operation of impoundments (Van Buren *et al*., 1997; Alaoui Mhamdi et al., 2007). In many cases, water quality in these impoundments and downstream environments is often negatively impacted because of the excessive accumulation of sediments and associated nutrients and contaminants (Fridl and Wuest, 2002; Van Buren *et al*., 1997). To offset the adverse effects of urban development and improper reservoir design on water quality, there has been an increase in the number of rehabilitation and redesign efforts in North America to improve water quality.

While these engineering projects often focus on modeling water quality and redesigning impoundments to improve water quality, few studies have evaluated water quality performance of impoundments that have been redesigned to enhance water quality. This is primarily due to the fact that such data bases are often not available, and without these data the evaluation of post-design performance is not possible. Accordingly, in order to evaluate the effectiveness of impoundment design to enhance water quality, further research is required to evaluate pre- and post-design water quality performance of urban impoundments and to quantify the change in nutrient and suspended-solid concentrations and loads between pre- and post-design conditions. In addition, there is an overall lack of research on the relationship between various designs of urban impoundments and their performance (Strecker et al., 2001). The effectiveness of measures that prolong water retention and regulate flow circuits has been discussed by some literatures (Shammaa et al., 2002; Paul et al., 1998). However, few researches have been conducted to investigate bottom sediment dredging and lake bathymetry (Kleeberg and Kohl, 1999). Various water quality models have been utilized to successfully describe the hydrological and ecological functions of impoundments (Teeter et al., 2001). However, site-specific data are still required because different physical, chemical and biological factors dominate the transport of nutrients and suspended solids at each site. Water quality models based on site characteristics should be developed and validated, in order to predict the long-term performance of impoundments.

1.2 Objective

This research evaluates the water quality of an urban impoundment (Columbia Lake) in Waterloo, Ontario. A mass balance approach is used to characterize a range of physical and chemical water quality parameters at the inlet and outlet of Columbia Lake. The hydrology (instantaneous discharge) and water quality parameters (TP and SS) were monitored for two pre-design years (2003 and 2004) and compared to data collected for two post-design years (2006 and 2007). These data are used to compare pre- and post-design conditions and to quantify water quality treatment of the lake after its redesign.

The specific objectives of this study are to:

1) Assess the effectiveness of the Columbia Lake redesign project on water quality improvement, by comparing the temporal variability (in both pre- and post-design periods) of the concentrations and loads of TP and SS (at the in- and outflow), and Columbia Lake's performance (as characterized by net TP and SS internal loading rates).

2) Test the Columbia Lake Water Quality Model (developed by Stantec Consulting Ltd.) to predict outflow TP and SS concentrations.

It is hypothesized that:

1) There is no significant difference between water quality parameters (SS, TP concentrations and loads) at the outflow of Columbia Lake for the pre- and post-design periods. Outflow TP and SS concentrations and loads during the pre-design period were similar to the post-design period. Hydrological factor and sediment characters are still the dominant factors that control P and SS transfer.

2) The Columbia Lake Water Quality Model proposed by Stantec Consulting Ltd. accurately predicts outflow TP and SS concentrations.

1.3 Literature Review

1.3.1 Stormwater Management in Urban Systems

Urbanization is regarded as one of the most important causes of water quality impairment in watersheds, primarily because it increases the spatial extent of impervious surface cover (ISC). ISC is defined as those materials that prevent the infiltration of water into the soil. There is a clear link between impervious surface and the hydrologic changes that degrade water quality. ISCs convey pollutants into the waterways by preventing percolation (Brabec et al., 2002). An increasing amount of evidence has shown that several water quality and biological indicators are strongly correlated to urban density and total imperviousness (Marsh, 2005). Stream health is impacted when ISC constitutes between 10 to 30% of the total area of a watershed, and severe degradation occurs when the percentage is more than 30% (Arnold and Gibbons, 1996). The impervious threshold of eutrophication based on TSS and TP is 30% (Brabec et al., 2002). Accordingly, increases in ISC alter the hydrological cycle and stream morphology, and degrade water quality as well as aquatic habitats (OMEE, 2003). To mitigate these effects and improve water quality, recent changes in policies and regulations for land use development in Ontario have improved the practice of stormwater management (Marsalek and Chocat, 2002).

In 1991, the Ontario Ministry of the Environment created a document entitled "*Stormwater Quality Best Management Practices*" which stated that watersheds are logical planning units for land use development (OMEE, 2003). The document argued that localized solutions (as opposed to solutions at the watershed or subwatershed level), may actually accelerate the negative impacts of urban drainage by increasing downstream peak flow rates (Yeh et al., 1997). In June 1994, the Ministry of the Environment published the *Stormwater Management Practices Planning and*

Design Manual, which focused more on understanding the performance requirements of stormwater management projects on water quality improvement (MOEE, 2003). Later, the 2003 version of the manual provided an overview of the impact of urbanization on the hydrological cycle and the aquatic ecosystem. It also included sections on integrated water quality protection, erosion and water quantity control and water balance as they relate to the design of stormwater management practices.

In watershed or subwatershed level planning, integrated stormwater management practices include the use of several structural measures at the lot, conveyance and end-of-pipe level. Lot level and conveyance controls are mainly used to decrease the rates and magnitude of runoff by enhancing infiltration of surface water into the ground (Marsh, 2005; OMEE, 2002). End-of-pipe controls are flooding and erosion controls as well as water quality improvement. These facilities are designed to store excess water on or near the site and to release it slowly to receiving waters gradually over time (Marsh, 2005; OMEE, 2002).

1.3.2 Impoundment Design and Functions

Traditionally, impoundments were designed to control water quantity by decreasing peak discharge and increasing lag time, but they were less effective at mitigating the adverse effects of urban development on water quality (Van Buren et al., 1997; OMEE, 2003). Decreased flow velocities of reservoir inflows cause SS and associated pollutants to deposit on the lake bottom. Accumulated P and solids originating from high external loading can accelerate internal P loading processes due to processes such as aerobic and anaerobic bacteria in bottom sediments, groundwater seepage and decomposition of organic materials (Cooke et al., 2005). Resuspension processes due to wind and bioturbation processes can increase concentrations of SS and associated P at impoundment outflows. Anoxic environments caused by the decomposition of organic matter enhance P release from bottom sediments (Jensen and Anderson, 1992). Mineralization and microbial processes affect P uptake, storage and release (Jensen and Anderson, 1992) and often lead to eutrophication problems in impoundments and downstream reaches.

To alleviate the degradation of hydrological and ecological conditions in aquatic systems, inputs of nutrients and sediments from the watershed must be reduced. However, internal sediment and P loadings impede lake water quality enhancement (Coveney et al, 2005; Jeppesen et al., 2005). According to Sondergaard et al. (2003), it can take anywhere from 10 to more than 20 years for external loading to be reduced so as to affect lake nutrient retention. Coveney et al. (2005) indicated that external P loading to Lake Apopka (FL, USA) from farm lands decreased from 0.56 g m^{-2} $year^{-1}$ to under-detectable during eleven years, while TP in-lake concentrations only decreased from 0.23 to 0.11 mg L^{-1} during the same study period.

Since reducing external P loadings from upstream is not usually sufficient to lower P concentrations and loadings in urban impoundments, it is essential to utilize in-lake redesign methods to reduce internal P loading from bottom sediments. Environmentally sound impoundments and reservoirs are designed to prevent internal loadings and increase TP and SS retention. They are thought to consistently achieve a moderate to high level of removal for both particulate and soluble pollutants in the long term and, hence, are prevalently applied in Ontario (Tsihrintzis and Hamid, 1997). By increasing basin volume, optimizing water retention time, regulating flow direction and changing the lake configuration, sediment and pollutant retention can be optimized. However, on a broader temporal and spatial scale, the performance of impoundments on water quality treatment is highly variable, often due to ineffective designs. Fine grained particles (< 63 μm) have large cation exchange capacities and sorb nutrients and pollutants which may travel downstream without being treated (Cooke et al., 2005). In addition, an undesirable anaerobic biochemical aquatic environment may be created, which leads to high release of soluble P forms from bottom sediments. The effectiveness of various design methods reported in the literature on nutrient and SS retention are listed in Table 1-1. The basic goal of these methods is to create desirable physical conditions and in-lake biogeochemical environment to improve the ability of an impoundment to retain nutrients and sediments and to control factors that promote internal loading.

Table 1-1 Summary of literature on impoundment design methods for water quality improvement

Reference	Design Approaches	Effectiveness
Davis et al., 2006	Bioretention: a mulch/soil/plant-based stormwater best management practice.	The accumulation of P and N can be controlled by carefully managing the growth and harvest of vegetation. Case studies indicated a P-removal rate of 70 to 85%.
Gachter and Wehrli, 1998;	Hypolimnetic aeration	10 year of experience showed that hypolimnetic oxygen had no impact on internal P cycling. The sediment-water interface was still anoxic, due to unchanged high sedimentation rates of organic matter.
Oberts and Osgood, 1991	Combined detention/wetland stormwater treatment facility: detention pond that discharges into six wetland "chambers"	The removal rates of TSS and TP in the detention pond were >90% and >70%, respectively. The removal rates for the whole treatment system were about 96% and 77% for TSS and TP, respectively. The newly constructed system worked well, but long-term maintenance of the accumulation of sediments and nutrients was required.
Putz and Benndorf, 1998	Pre-reservoirs (small reservoirs with a water retention time of several days), were used to decrease input to main reservoirs	A two-stage operation: first, the conversion of dissolved P into particulate P; the second is the sedimentation process. Data from five pre-reservoirs showed that P-removal rates were 60-90% from Apr. to Oct. and 10-30% during the rest of the year.
Perkins and Underwood, 2001	Dosing of input water with ferric sulphate to control external P loading of a eutrophic reservoir	A decrease along the length of the reservoir in sediment labile P content from 0.62 to 0.08 mg P g^{-1} and iron-bound P content from 3.22 to 0.46 mg P g^{-1}.

Table 1-1 Summary of literature on impoundment design methods for water quality improvement (Cont'd)

Reference	Design Approaches	Effectiveness
Paul et al., 1998	Submerged flexible curtain (SFC)	SFC prevented hydraulic short circuits, increased the retention time of inflowing water, caused a 30% to 40% increase of the elimination of SRP, and favored sedimentation in the mouth region.
Shammaa et al., 2002	Dry ponds with two-stage facilities and multi-level outlet design.	The upper stage was designed to store large and infrequent stormwaters, and the lower stage was designed to promote sedimentation of the smaller and more frequent stormwaters. Research from two dry ponds showed low TSS removal rates, due to the low retention time, the most important factor for dry pond design.
Sondergaard et al., 2003	Sediment removal	Usually, P in the upper 10 cm of bottom sediments is regarded as part of the whole lake metabolism. However, in some cases, P mobility from lake bottom down to depths of 20 – 25 cm has been observed. Hence, sediment dredging (removing nutrient-rich sediment) is an effective physical method for preventing internal loading.
Szilagyi et al., 1990	A reservoir with a retention time of 40 days, with buffers to regulate the flow direction	As a result of reservoir operation, nutrient outputs from the basin have been significantly reduced. The removal efficiencies for SS, TP and SRP were 70%, 51% and 61% respectively in 1986, and 84%, 37% and 53% in the first nine months of 1987.
Wu, 1996	Reservoir operation: outlet position	Releasing water from a certain depth in a reservoir can improve outlet water quality. The nutrient concentration in the low layer was worse when the reservoir outlet positions were at the low layer than when the outlet was in the middle layer.

1.3.3 Phosphorus and Sediment Dynamics in Urban Impoundments

Urban stormwater runoff contains elevated levels of SS, nutrients, bacteria, heavy metals, oil and grease, and pesticides (OMEE, 2003). Among these pollutants, SS are the most crucial indicator of water quality, since both organic and inorganic matters can bind with SS and be released into the water column (Alaoui-Mjamdi, 1996). Hence, the priority of stormwater water quality management is the control of SS and a variety of sediment-associated pollutants including P (OMEE, 2003). Phosphorus is the limiting nutrient for algae growth and control of this nutrient in urban impoundments is critical to prevent eutrophication (Cooke, et al., 2005).

Phosphorus is a limiting nutrient in most freshwater systems and has two operationally-defined forms: Dissolved P (DP) and Particulate P (PP) (Reddy et al., 1999). The sum of DP and PP is Total P (TP). DP is dissolved in freshwater, while PP is mainly bound with particulate solids and colloids or confined to the geochemical matrices of solids. Both DP and PP include organic and inorganic forms of P. Soluble Reactive P (SRP) is the most bioavailable form among various forms of DP (Macrae et al., 2003; Casey and Klaine, 2001). Iron (II) (Fe^{2+})-bound P is in the dissolved form. During aerobic conditions, it can be oxidized to iron (III) (Fe^{3+})-bound P, which is not dissolved in water.

Table 1-2 summarizes several studies that investigated the rates and magnitudes of P and SS transfer in impoundments. These studies show that SRP concentrations in waters entering impoundments are typically on the order of 0.6 to 8 times higher than those released from the impoundments, whereas TP ranges from 0.4 to 2.3. Internal loading rates ranged from -86 to 68% for SRP and from -82 to 300% for TP. In terms of SS, the inflow concentrations are typically 0.4 – 2.2 times as those of the outflow and internal loading rates range from -69 to 117%. These parameters indicate that TP is more variable in impoundments than SRP, and impoundment performance on TP and SS removal varies widely.

Impoundments can act as either sinks or sources of P and SS to receiving waters, depending on land use as well as hydrological and biochemical settings. In a study of reservoirs in Europe, Salvia-Castellvi et al. (2001) found that deeper reservoirs are more effective for P removal likely because P dynamics are related to phytoplankton growth cycles. Teodoru and Wehri (2005) reported that the Iron Gates Reservoirs in Romania acted as a source of nutrients but a sink of SS. The internal P loading in this system was due to the remobilization of bottom sediments which released high levels of dissolved P (Teodoru and Wehri, 2005).

Table 1-2 Literature on the mass balance analysis of impoundments on P and SS

Reference	Location	Parameters	Input		Output		Internal loading rate[1]
			C_{IN}[2]	L_{IN}[3]	C_{OUT}[2]	L_{OUT}[3]	
Alaoui Mhamdi et al., 2007	Sahela Reservoir, Morocco	TP	n/a	31	n/a	122	300%
Istvanovics and Somlyody, 1999	The Upper Kis-Balaton Reservoir, Hungary	SRP(1986-90)	322	5475.25	47	895.73	-84%
		SRP(1991-97)	118	2370.04	15	338.94	-86%
		TP (1986-90)	562	9556.18	243	4631.10	-52%
		TP (1991-97)	290	5824.68	224	5061.45	-14%
		SS(1986-90)	59	1003.29	27	514.57	-49%
		SS (1991-97)	44	883.75	29	655.28	-26%
James et al., 2004	Peoria Lake, USA	TSS	n/a	9.84×10^4	n/a	6.83×10^4	-31%
Salvia-Castellvi et al., 2001	Deeper Pre-reservoir Bavigne	TP	n/a	285	n/a	53	-82%
		SRP		34		16	-54%
	Shallow Pre-reservoir Misere	TP	n/a	3310	n/a	1370	-60%
		SRP		479		457	-4%

[1] Internal loading rate = (outflow loads – inflow loads) / inflow loads.
[2] C_{IN} means inflow concentration; C_{OUT} means outflow concentration. The unit for TP, SRP and PO_4^{3-} concentrations is $\mu g\ L^{-1}$, and the unit for TSS and SS concentrations is $mg\ L^{-1}$.
[3] L_{IN} means inflow load; L_{OUT} means outflow load. The unit for TP, SRP and PO_4^{3-} loads is $g\ h^{-1}$, and the unit for TSS and SS loads is $Kg\ h^{-1}$.
n/a means data is not available.

Table 1-3 Literature on the mass balance analysis of impoundments on P and SS (Cont'd)

Reference	Location	Parameters	Input		Output		Internal loading rate
			C_{IN}	L_{IN}	C_{OUT}	L_{OUT}	
Shantz et al., 2004	Columbia Lake, Waterloo	TP (pre-drawdown)	33	3	85	5	67%
		SS (pre-drawdown)	15	1.2	41	2.6	117%
		TP (drawdown)	417	292	173	225	-23%
		SRP (drawdown)	12	9	9	11	22%
		SS (drawdown)	96	344	51	102	-69%
Teodoru and Wehri, 2005	The Iron Gates Reservoirs, Romania	PO_4^{3-}	32.1	6.5×10^5	51.3	11.0×10^5	68%
		TP	79.6	17.0×10^5	84.5	18.8×10^5	11%
		SS	34	10.1×10^5	17	4.4×10^5	-56%
Van Buren et al., 1997	Kingston, Ontario	TSS (event)	78 - 134	n/a	73	n/a	-42%
		TSS (baseflow)	22		34		55%
		TP (event)	110 - 106	n/a	84	n/a	-21%
		TP (baseflow)	40		53		33%

Istvanovics and Somlyody (1999) investigated differences in the Upper Kis-Balaton Reservoir (UKB) in Hungary performance between pre- and post-management conditions. The conditions managed by this case study were the external loadings from an agricultural land, which were decreased. As shown in Table 1-2, inflow TP concentrations and loads in the post-management period (1991-1997) declined to half those in the pre-management period (1986-1990). However, the TP concentrations and loads following a decrease in the external P loadings were similar to the pre-management period, due to sediment resuspension. The internal loading rates of SRP were similar in both periods. Generally, P uptake by vegetation is restricted when SRP is below 30 μg L^{-1} (Istvanovics and Somlyody, 1999). In such cases, abiotic processes, including adsorption by sediments and coprecipitation with iron controls SRP retention. Input-output load functions (L_{IN}-L_{OUT}) were also used to examine the P cycle. Since TP output is a combination of input and internal loads of TP, the study examined the function $L_{OUT} = L_{IN} + L_{int}$. The shift in the L_{OUT} vs. L_{IN} curve occurred after the external loading was decreased (Istvanovics and Somlyody, 1999).

Figure 1-1 illustrates the transfer and cycle of P in a shallow impoundment, where stratification is ignored. TP transfer is different from the sedimentation/resuspension process of SS transport in impoundments; TP transfer is much more complicated and includes the downward flux caused by sedimentation of PP and absorption of DP, the upward flux driven by sediment resuspension and DP desorption, and the transformation of DP into PP. DP is utilized by macrophytes as a nutrient from interstitial water (Reddy et al., 1999). After macrophyte senescence and decomposition, the absorbed P is released as PP into the water column (Haggard and Soerens, 2006; Salvia-Castellvi et al., 2001). Meanwhile, catabolic activities catalyze the mineralization of particulate organic P, transferring it to dissolved inorganic P, assuming sufficient oxygen is provided (Reddy et al., 1999; Ahlgren et al., 1988).

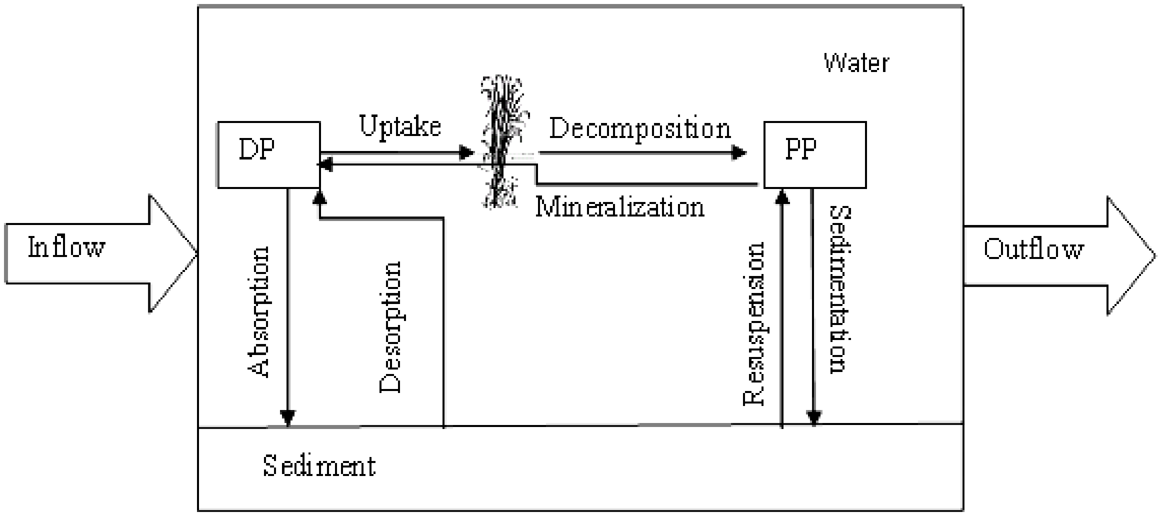

Figure 1-1 Phosphorus transfer dynamics in shallow impoundments

PP is transported by means of sedimentation/resuspension (Figure 1-1), which is controlled by the physical, geochemical and biological characteristics of sediment and hydrological factors (Van Buren et al., 1997; Alaoui-Mhamdi, 1996; Alaoui-Mhamdi et al., 2007; Teodoru and Wehrli, 2005; Istvanovics and Somlyody, 1999). Sediment characteristics include concentrations, geochemistry and particle size. A higher SS removal rate (associated with a higher inflow SS concentration) was attributed to particle-particle interactions and resulting flocculation processes (Urbonas, 1995). The coagulated particles are more inclined to settle to the lake bottom. Negative P removal rates of an on-stream stormwater pond during the baseflow period in Kingston, Ontario, were attributed to the fact that fine particles have little chance to collide when SS concentrations are low (Van Buren et al., 1997). Additionally, fine particulate fractions of sediments (<63μm) can often release nutrients and pollutants into the water column due to their relatively large surface area and geochemical composition (Stone and English, 1993).

Hydrology is another important factor affecting PP transfer. The portion of SS and PP deposited at the bottom of an impoundment and DP absorbed by bottom sediments and vegetation increases with an increase in water retention time. However, the marginal improvement in P and SS removal rates diminishes significantly because two processes (the monotonically decreased proportion of runoff that is actually processed through impoundments and the increased removal efficiency) control overall P and SS retention (Papa et al., 1999; Kennedy, 1999; Shammaa et al., 2002). Flow velocity, sometimes accelerated by wind-generated wave action and coupled with shallow morphometry, increases the resuspension of deposited SS and PP (Haggard and Soerens, 2006; James et al., 2004). A P release in a shallow and eutrophic lake, Lake Arreso in Denmark, caused by resuspension, was reported as 20 to 30 times greater than the P release from undisturbed sediment cores (Sondergaard et al., 1992). To mitigate resuspension, submerged aquatic macrophytes can be planted to obstruct sediments (James et al., 2004). In addition, bioturbation can be controlled by removing Carp. For example, Barton et al. (2000) showed that by removing Carp in Laurel Lake, Waterloo, Ontario, turbidity decreased and SS retention increased by 45%.

DP is controlled mainly by absorption/desorption processes and co-precipitation with iron (Figure 1-1). DP mobility is mainly determined by DP or SRP equilibrium concentrations in a water column (Sondergaard et al., 1992; Haggard and Soerens, 2006). Laboratory equilibration studies indicated that a buffering mechanism would likely maintain SRP equilibrium concentrations between 0.05 to 0.20 mg L^{-1} (Haggard and Soerens, 2006). When SRP concentration in surface water is below this range, the interstitial water of bottom sediments (which usually have higher DP concentrations) will act as a linkage between the sediment surface and the water column, and stimulate a diffusive flux at the sediment/water interface (Sondergaard et al., 1992; Sondergaard et al., 2003).

Anaerobic/aerobic conditions in water/sediment interface are regarded as a controlling factor for DP release from sediments (Alaoui-Mhamdi, 1996; Alaoui Mhamdi et al., 2007; Haggard and Soerens, 2006). Seasonal anoxia in the hypolimnion of six marine lakes in USA strongly correlated with internal P loadings to water column (Lake et al., 2007). Under anaerobic conditions, P release from bottom sediments is much higher than the release under aerobic conditions. Studies from Haggard and Soerens (2006) and Penn et al. (2000) indicated that sediment P releases were 3 and 4 mg P m^{-2} day^{-1} under aerobic conditions and approximately 15 and 38 mg P m^{-2} day^{-1} under anaerobic conditions, respectively. This is because aerobic conditions prevent microbial activities from releasing P through mechanisms such as ligand exchange and enzymatic hydrolysis of organic ester bonds (Pettersson, 1998; Alaoui-Mhamdi, 1996; Jensen and Anderson, 1992). Moreover, redox-dependent phosphorus can be fixed by Fe (III) in oxic condition, preventing the potential of reducing Fe^{3+}-bound P into dissolved Fe^{2+}-bound P under anoxic conditions, especially when the oxygen concentration is below 3 mg L^{-1} (Cooke *et al.,* 2005; Reddy et al., 1999; Sondergaard et al., 2003; Alaoui Mhamdi et al., 2007).

Therefore, to avoid internal P release, an oxic aquatic environment can be developed through the design of circulation patterns that fully mix the water (Ainsworth, 2001; Cooke *et al.,* 2005; Kneese and Bower, 1984), or through the addition of iron or alum to increase the sorption capacity of bottom sediments (Perkins and Underwood, 2001; Sondergaard et al., 2003). However, Gachter and Wehrli (1998) argued that anoxic sediment surface and high P release rates from lake sediments may not have a causative relationship, but may simply be two parallel systems with one common cause: excessive organic matter and P sedimentation exhausting the stock of hypolimnetic DO and exceeding the P retention capacity of the sediments after diagenesis.

Other factors, such as temperature and pH, also affect DP release. Increased temperature enhances internal loadings by stimulating mineralization process and accelerating the anaerobic dissolution of Fe-bound P (Jensen and Anderson, 1992; Perkins and Underwood, 2001). Christophoridis and Fytianos (2006) reported that under reductive conditions (-200 mV), when pH varied from 7 to 9, P release from bottom sediments increased from 1.189 to 1.510 mg m^{-2} day^{-1} and from 0.708 to 2.004 mg m^{-2} day^{-1} in two lakes respectively; this was attributed to ion-exchange with OH^- and release of DP at higher pH. Penn et al. (2000) observed internal P release at pH 6.5 was 15% higher than at pH 7.5 under anaerobic conditions, because calcium-bound P and Fe-bound P are more soluble at low pH.

1.3.4 Water Quality Modeling

Water quality models, as environmental management and land use planning tools, are widely utilized to provide information for watershed planning and management. Three types of models, empirical models, conceptual models, and physics-based models, are often utilized to characterize the transfer of sediments and associated nutrients (Merritt et al., 2003). Empirical models are the simplest of all the three types, but require a large amount of spatially and temporally distributed input data (Merritt et al., 2003). They are usually supported by coarse materials. Conceptual models are typically based on the representation of a catchment as a series of internal storages and usually incorporate the underlying transfer mechanisms of sediments and nutrients for characterizing their dynamic behavior (Merritt et al., 2003). For instance, the interaction between water quality and macrophyte levels was researched, so that methods to reduce the undesirable consequences of eutrophication problems could be predicted (Muhammetoglu and Soyupak, 2000); a conceptual model of sediment transport characterized by hydrological processes provided good predictions of stream flow across most of the Avon Basin in Australia (Viney et al., 2000). Physics-based models are based on the solution of fundamental physical equations describing streamflow, sediment and nutrient generation in a catchment (Merritt et al., 2003). In general, a large amount of parameters were involved in an equation. Therefore, physics-based models are derived at the small scale and applied under very specific physical conditions (Merritt et al., 2003).

Columbia Lake Water Quality Model is a conceptual model, which is used to simulate the water quality response at the outflow of Columbia Lake to upstream loads from Laurel Creek, and to predict the long-term effectiveness of the Columbia Lake redesign project on water quality improvement (Stantec Consulting Ltd., 2004). This model focused on hydrological impact (discharge and water retention time) of the transfer mechanisms on TP and SS, representing flow paths in the catchment as a series of storage. Water quality data at the inflow and outflow from 1997 to 2003 were analyzed and incorporated into Equation 1.1:

$$s_1 = \frac{Q_i s_i}{Q_{12} + K_1 V_1 + K_s V_1}\left[1 - e^{-\left(\frac{Q_{12}}{V_1} + K_1 + K_s\right)t}\right] + s_0 e^{-\left(\frac{Q}{V} + K + K_s\right)} \qquad 1.1$$

where

Q_{12} = outflow from Columbia Lake (m^3/s);

S_1 = parameter concentration response in Columbia Lake (mg/l);

Q_i = Laurel Creek inflow rate (m^3/s);

S_i = parameter concentration in Laurel Creek inflow (mg/l);

V_1 = Columbia Lake volume (125,000 m^3 in the post-design condition);

K_1 = pollutant decay rate in Columbia Lake (s^{-1});

K_s = pollutant settling rate (s^{-1});

T = time (s);

S_0 = the initial parameter concentration in the lake (mg/l);

Q = the outflow rate;

V= volume (m^3).

This water quality model is hydrology-related, because sediment and nutrient loadings are dominated by hydrological processes; this necessitates a clear understanding of flow and load relationships for catchments (Merritt et al., 2003). According to the model calibration, TP was a conservative substance with K_1 and K_s equal to zero (Stantec Consulting Ltd., 2004). In terms of SS, K_1 was zero and K_S varied from 3.3 to 12×10^{-7} s^{-1} in the former years (1997 - 2003).

Modeling is an important process for stormwater management, because models can predict long term performance of storm water facilities on water quality improvement and further provide advice on management and policy-making. However, by now, model performance and accuracy remain a major difficulty in model development particularly with spatially distributed models, due to the natural complexity, uncertainties in sediment generation and transport, and limitations in understanding of sediment and associated nutrient transport (Jakeman et al., 1999; Merritt et al., 2003). A series of biogeochemical processes affect the TP and SS transfer. Complicated natural impact and the limitations in understanding these mechanisms lead to the uncertainties in predications. In addition, it is difficult to obtain and verify information on sediment sources, paths, transport rate and delivery (Merritt et al., 2003).

Uncertainties in water quality predications are considerably greater than in the water quantity predictions (Merritt et al., 2003). This is because a causative relationship between controlling factors and resulting water chemistry is not always self-evident in managing water quality (Merritt et al., 2003), as well as a high degree of variability in TP and SS inflow concentrations (Stantec Consulting Ltd., 2004). Additionally, many parameters affect TP and SS transfer through an impoundment. However, most of the conceptual models can only include relevantly important parameters, which causes the differences between model predications and measurement. In terms of the Columbia Lake Water Quality Model, Stantec Consulting Ltd. (2004) demonstrated that the observed data varied beyond the ranges predicted by the model, due to the inflow water quality data set and sampling methodology, which cannot account for rapid water quality responses, such as sharp concentration spikes, produced by this system (Stantec Consulting Ltd., 2004).

By now, a large amount of work has been conducted to monitor water quality. There is no model validation has been conducted to investigate the effectiveness of the water quality model on predicting the outflow TP and SS concentrations during the post-design period in Columbia Lake by incorporating the monitored parameters. Most of the literatures which evaluated the performance of stormwater facilities, only studied the P and SS removal efficiencies during the study period (Salvia-Castellvi et al., 2001; Shammaa t al., 2002; Davis et al., 2006; Van Buren et al., 1997). No models were developed to predict long term performance on water quality improvement. Hence, it is crucial to establish water quality models to characterize long-term performance of impoundments on water treatment and further to provide technical support for planning and management.

Chapter 2 METHODOLOGY

2.1 Experimental Design

In 2002, the City of Waterloo reported that Columbia Lake was a source of sediments and P to downstream reaches (Stantec Consulting Ltd., 2004) and subsequently commissioned Stantec Consulting Ltd. to redesign the lake. The goal of the redesign project was to enhance water quality of Columbia Lake and downstream reaches. The engineering project was completed in October 2005. The Columbia Lake Water Quality Model used by Stantec Consulting Ltd. indicated that the final lake design would result in a 75 – 90 % decrease in TP and SS export downstream. Currently, no studies have been conducted to assess whether the redesign has met the proposed target reductions. The approach taken in this thesis to examine the post-design water quality performance of Columbia Lake is by comparing water quality data collected at the lake's inflow and outflow for the pre-design period (2003-2004) and post-design period (2006-2007). The study compares concentrations and loads of TP and SS to quantify the performance of Columbia Lake and incorporates these data to validate the Columbia Lake Water Quality Model proposed by Stantec Consulting Ltd. The data were collected during the pre- and post-design periods as part of the Laurel Creek Water Quality Monitoring Program and include discharge (Q), surface water temperature (T), dissolved oxygen (DO), pH, total phosphorus (TP), suspended solids (SS), and total dissolved solid (TDS) (TDS data were unavailable in 2003). During 2007, I participated in the monitoring program and in addition to the parameters described above, I sampled soluble reactive phosphorus (SRP) and grain size to investigate some physical and biogeochemical processes that influence P transfer in the lake.

A conceptual model of the physical and biogeochemical processes governing P and SS transfer in impoundments is presented in Figure 1-1. The figure shows that surface runoff and stream inflow are the primary sources of TP and SS to Columbia Lake. Bioturbation and resuspension of SS and associated P are suspected causes of the high internal nutrient loading reported during the pre-design period (Shantz et al., 2004).

This study focuses specifically on the change in measured water quality (TP and SS) and hydrology at the inflow and outflow of Columbia Lake for the pre-design (2003-2004) and post-design (2006-2007) periods (Figure 2-1). The goal of this thesis is to assess the effectiveness of the Columbia Lake redesign project on water quality and to assess the performance of Columbia

Lake on water treatment. A mass balance approach (Figure 2-1) is used to evaluate P and SS concentrations and loads at the inlet and outlet of Columbia Lake for pre- and post-design periods. Once the water quality performance (monthly, annual, pre- and post-design periods) is calculated, a positive net internal loading indicates that Columbia Lake is a source of TP and SS; whereas, negative values indicate that the lake is retaining TP and SS. A high net internal loading rate suggests degradation of water quality in the lake. The net internal loading and internal loading rate are determined using the following equations:

Net internal loading = $L_{OUT} - L_{IN}$ 2-1,

Net internal loading rate (%) = Net internal loading / $L_{IN} \times 100\%$ 2-2,

where L_{OUT} and L_{IN} are TP and SS loads in the outflow and inflow, respectively.

The terms L_{OUT} and L_{IN} are calculated using Equation 2-3:

$L_{IN/OUT} = Q_{IN/OUT} \times C_{IN/OUT}$ 2-3,

where $Q_{IN/OUT}$ means the inflow and outflow discharges, respectively; $C_{IN/OUT}$ stands for the inflow and outflow concentrations, respectively.

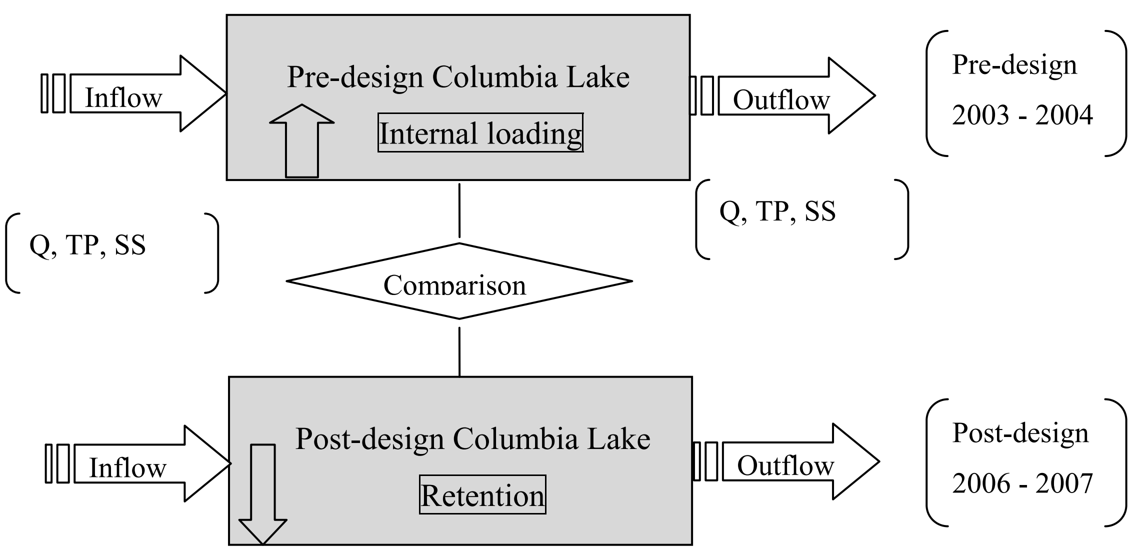

Figure 2-1 Diagram of mass balance approach to evaluate P and SS concentrations and loads at the inlet and outlet of Columbia Lake for pre- and post-design periods

2.2 Study Site Description

2.2.1 General Description

Columbia Lake is located in the Laurel Creek Watershed; a small (74 km^2) subwatershed in the central Grand River watershed (Stantec Consulting Ltd., 2004). Land use in the Laurel Creek Watershed is varied; with agriculture (cash crops and pasture land), woodlots and wetlands in the northwestern portion of the watershed and urban land use (including high to low density residential, industrial, commercial and institutional zoning) in the southeastern portion (City of Waterloo, 2004; Shantz et al., 2004). Approximately, 80% of the watershed area is urbanized and within the City of Waterloo (City of Waterloo, 2004). According to the City of Waterloo, further urban expansion is proposed in the western portion of the watershed (Winter and Duthie, 2000).

Annually, Laurel Creek receives large inputs of nutrient-rich runoff from upstream agriculture. In addition, rapid development in the northwest of the watershed is causing increasing SS concentrations in the runoff (City of Waterloo, 2004) and deposits in Columbia Lake, thus decreasing water quality in this impoundment. Aerial photographs of Laurel Creek Reservoir and Columbia Lake in 1995, 2000, and 2003 are shown in Figure 2-2. Initially, due to the high quality of surface water upstream of Laurel Creek Watershed, the turbidity of Laurel Creek Reservoir was relatively low as shown in the photographs in 1995..

Figure 2-2 Aerial photos of Laurel Creek Reservoir and Columbia Lake.

Scale: 1: 20,000. Source: Map Library, University of Waterloo. A. year 1995. B. year 2000. C. year 2003. I. Laurel Creek Reservoir. II. Columbia Lake

Columbia Lake is an urban impoundment used primarily for stormwater management. Prior to its redesign, the lake was used for flood control and recreation (Hendrickson, 1996; Barton et al., 2000). It was approximately 12 hectares (30 acres) in size, with a mean depth of 1 m, and was built as an engineered reservoir in 1967. The inflow sampling location is located on Laurel Creek between Laurel Creek reservoir and Columbia Lake adjacent to Beaver Creek Road. The outflow sampling location is near the south end of Columbia Lake, immediately downstream of the Columbia Street Dam (Figure 2-3).

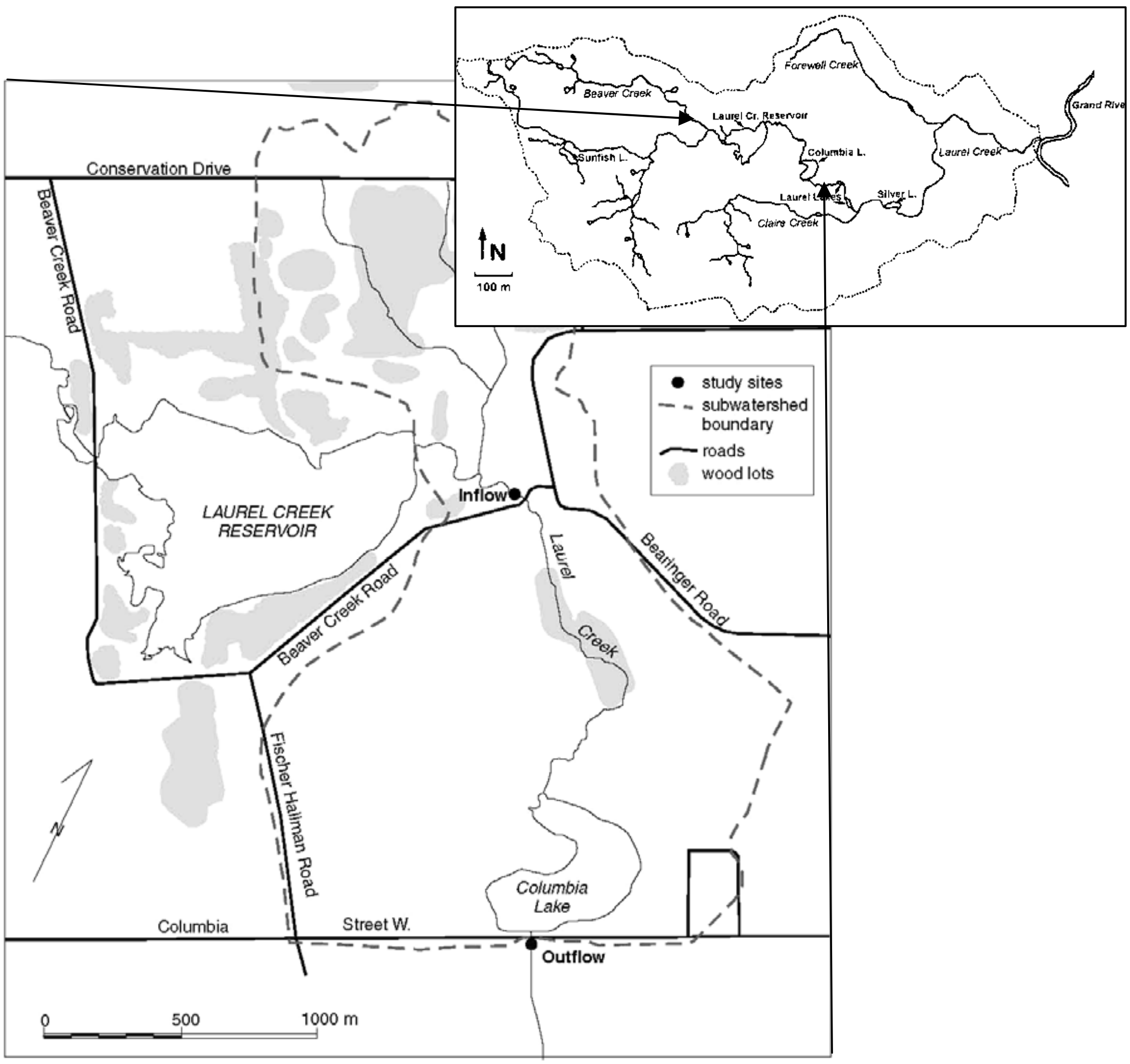

Figure 2-3 Study Site (Shantz et al., 2004) and its relative location in Laurel Creek Watershed (Winter and Buthie, 2000)

2.2.2 Redesign of Columbia Lake

Prior to the lake retrofit project, Columbia Lake was characterized as a shallow impoundment with extremely low habitat diversity, poor substrate quality, high nutrient-rich sediment preposition levels and a degraded benthic community (Stantec Consulting Ltd., 2004). To solve its water quality problem, Columbia Lake was redesigned to maximize the diversity of both aquatic and terrestrial habitats (Stantec Consulting Ltd., 2004).

To enhance water quality, changes were implemented to the design of the lake inlet, outlet and lake configuration. The lake inlet, which is comprised of a ditch inlet catch basin and a 50 m long, saw-tooth shaped spillway in the lake berm, was designed to split the flows within Laurel Creek before they were directed to the reconstructed lake, in order to minimize overflow of water not treated in the lake (Stantec Consulting Ltd., 2004). The outlet, designed with a stoplog control structure, offers the optimal combination of desired elevation control and allows better operational flexibility (Stantec Consulting Ltd., 2004). Geese and carp control methods were utilized to mitigate negative impact of bioturbation on water quality. The shoreline was reconfigured and naturalized to diversify the habitats adjacent to the lake.

The purpose of the in-lake configuration is to increase water retention time and basin capacity, thus increasing the retention of nutrients and SS. The original "online" impoundment created by the damming of Laurel Creek at Columbia Street was changed to an "off-line" configuration, to increase settling times and decrease the risk of sediment export from the lake. The volume, area, and depths of Columbia Lake in the pre- and post-design conditions are compared in Table 2-1. Appendix 15 shows the pictures of the inflow and outflow study points and the reconstructed Columbia Lake.

Table 2-1 Physical characteristics of Columbia Lake for pre- and post-design conditions

Condition	Volume (m^3)	Surface area (m^2)	Depth
Pre-design	127,000	152,000	No more than 1 m on average
Post-design	125,000	95,000	Variable depths from 0 to 3.5 m or more

Physical changes made to improve water quality in the impoundment include the following:

1. A new island was constructed along the east shore area, and a remnant of the existing west island was retained and utilized within the new containment berm structure in order to regulate flow direction and prolong water retention time and the length of flow paths.

2. Bottom nutrient sediments were removed to decrease internal loadings and provide more desirable substrate for aquatic habitats. The volume of Columbia Lake changed slightly because of the combined effects of the removal of bottom sediments and the creation of a new island.

3. Wetland and riparian vegetation were planted within and around the lake to absorb dissolved nutrients and pollutants.

4. After nutrient sediment removal, a new lake bathymetry was designed with variable depths and an undulated and natural shoreline to enhance in-lake and near-shore habitats. The lake bottom was reconfigured to create five habitat zones from the outer perimeter to the central part of the lake. Design considerations (described in detail below) included changing lake bottom topography (slopes and depths), creating a beach with sandy flat areas, a wetland shallows area, a littoral shelf, a drop-off shelf, and a deep water zone.

A. The beach/sand flats zone covers an area from a beach area to the edge of the water, with sand as the dominant substrate material. Since the prevalent wind blows from northwest to southeast, the sand flats positioned along the west side of the lake avoid erosive wave action and the potential for increased turbidity from the resuspension of fine sand substrates. Its functions are to provide shoreline habitat diversity at the land-water interface, and to provide bottom substrate in submerged areas for use by different species of fish.

B. The wetland shallows are located at several locations, mainly along the outer perimeter of the lake. These zones range in depth from 0 to 1.0 m and are mainly used to remove nutrients from the water column and provide habitat for small fish and invertebrates.

C. The littoral shelf comprises a large portion of the perimeter of the new Columbia Lake and is characterized by coarser substrates in 0.2 to 1.5 meters of water. This zone, with its gently sloping bottom, starts at the land-water interface and gradually deepens towards the central portion or offshore area of the lake. It is one of the most productive areas of the lake, with important ecological functions.

D. The drop-off shelf is designed to mimic drop-off areas in natural lakes and reservoirs. Its depth changes from 1.5 meters to 3.5 meters, with a slope of 2:1 or 3:1. It is used to provide additional habitat structure.

E. The deep water zone is restricted to the central portion of the lake and has a depth of 3.5 meters or more.

2.3 Hydrology

Instantaneous discharge was measured using the area-velocity method twice weekly, at both the inflow and the outflow. The inflow stream cross-section was divided into six panels, and the outflow section was divided into four panels. Flow velocity in each panel was measured using an Ott propeller-type current meter. Discharge was calculated according to Equation 2-4

$$Q = \sum_{i=1}^{n} V_i \times A_i \qquad \text{2-4,}$$

where Q is discharge (m^3 s^{-1}), V_i is flow velocity (m s^{-1}) and A_i means the area (m^2) of each panel. For inflow discharge, n is equal to six, and for outflow discharge, n is equal to four.

Water retention time is calculated using estimates of discharge and lake volume, as shown in Equation 2-5:

$$R_T = V / Q \qquad \text{2-5,}$$

where R_T represents water retention time; V is the volume of Columbia Lake (m^3); and Q is equal to the reservoir inflow discharge (m^3 s^{-1}).

According to Stantec Consulting Ltd. (2004), the volume of Columbia Lake changed from 127,000 m^3 in the pre-design period to 125,000 m^3 in the post-design period (Table 2-1) and these values were used in calculations of R_T for pre- and post-design conditions.

2.4 Phosphorus

Stream water was sampled at the inflow and the outflow. Water samples were collected in acid (20% H_2SO_4)-washed and triple-rinsed glass bottles, then immediately stored in coolers and transferred to the laboratory for analysis. The parameters analyzed in the surface water samples included TP and SRP. For TP analysis, the water samples were acidified by adding 1 ml 20% H_2SO_4 within 6 hours of collection and were digested in 20 ml aliquots with 0.5 ml of saturated potassium persulfate on a heating plate until a 2 - 3 ml sample remained. The digested samples were diluted up to 20 ml with de-ionized water and filtered through 0.45 µm membrane filters (Whatman, Schleicher & Schuell). The samples for SRP analysis were filtered through 0.45 µm membrane filters (Whatman, Schleicher & Schuell) upon return to the laboratory, and then refrigerated at 4° C within 2 hours of collection. Final analysis for both TP and SRP was conducted with a single-channel colorimeter (Technicon Auto-analyser II) linked to a computer running NAP analysis software, according to the stannous chloride-ammonium molybdate procedure (Environmental Canada, 1987). The minimum concentration detectable by this method is 3 μg P L^{-1} (APHA, 1995). The stannous chloride method is sensitive and more suitable for the range of 0.01 to 6 mg P L^{-1} (APHA, 1995). The colorimeter and analysis software allow a direct analysis of samples containing < 200 μg L^{-1} for TP and < 160 μg L^{-1} for SRP, respectively. Samples with higher P concentrations out of the range should be diluted to concentrations within the range before they are analyzed by the instrument.

2.5 Suspended Solid Concentrations and Particle Size Distributions

Suspended solids (SS) were sampled in glass bottles using a DH-48 depth integrating sampler. In the lab, the samples were weighed and then filtered through glass microfiber filters. The weights of glass microfibers filters before and after the filtering were measured with an advanced electronic balance (GT Series) for calculating the net weight of SS. Concentration of SS was calculated by Equation 2-6:

SS concentration (mg/L) = net weight of SS × 1000/ (water quantity) 2-6,

where the net weight of SS (g) is calculated by the difference between gross weight and filter weight; and water quantity (L) = (weight of water sample – net weight of SS) / 1000.

The particle size distributions were measured twice weekly at the inflow and outflow. Standard particle-sizing techniques cannot be easily used *in situ* and do not permit accurate analysis of individual particles or floc / aggregates (Droppo and Ongley, 1994). Accordingly, the method of Droppo and Ongley (1992) was employed for the collection of suspended solids and subsequent grain size analysis. In this method, a water sample was collected in a plankton settling chamber by submerging the chamber to a depth of 30 cm below the surface, parallel to the direction of flow. Suspended solids in the chamber are deposited onto a filter paper under the slight pressure from a hand-operated vacuum pump. The settling chamber size was determined according to the turbidity of the surface water, by turbidity to suspended-solids / column relationship (Droppo and Ongley, 1994). To avoid overlapping of the settled sediment particles and increased flocculation due to too large a volume of samples, a chamber size of 25 ml was chosen on most of the sampling days. During analysis, filters were rendered semi-transparent by the application of three drops of Stephens Scientific low viscosity immersion oil to distinguish particles from their background. Particles of SS were sized by image analysis to a lower resolution of 1 μm (10 × objective) using a Zeiss Axiovert 100 microscope fitted with a Sony XC75 CCD connected to a Pentium computer running the Northern Eclipse image analysis software.

2.6 Relevant Water Quality Parameters

Related water chemistry, including concentrations of dissolved oxygen (DO), pH, surface water temperatures (T), and electricity conductivities were measured on site. DO concentrations and surface water T were measured with an YSI portable dissolved oxygen/temperature meter (Model 55-12), and pH was measured with a portable pH/ISE meter (Model 250A, ORION Research INC.). The electrical conductivity was measured using an ORION Model 105A+ conductivity meters for calculating the concentrations of total dissolved solids (TDS). The equation for calculating TDS concentrations is shown in Equation 2-7:

$$C_{TDS} = 0.666 \times \text{Conductivity} / (1 + 0.02 \times (T - 25)) \qquad 2\text{-}7,$$

where C_{TDS} is TDS concentrations (mg L^{-1}), Cond. is electricity conductivities (μs), and T stands for surface water temperatures (°C).

2.7 Modeling Calibration and Validation

The calibration of the Columbia Lake Water Quality Model was performed by adjusting the estimated values of pollutant decay rate (K_1) and pollutant settling rate (K_S) in Columbia Lake. Estimates of K_1 and K_S were based on the previous calibration from 1997 to 2003 illustrated in Stantec Consulting Ltd. (2004). After the calibration of K_1 and K_S, the data set of the inflow and outflow discharges, inflow concentrations of water quality parameters (TP and SS), water resident time and Columbia Lake volume were utilized by the model to predict the outflow TP and SS concentrations. These data were collected from May to August in 2006 and 2007. The validation of the calibrated model was implemented by comparing the predictive outflow concentrations of these parameters to the observed ones via statistical analysis. To adjust the predictions of outflow SS concentrations, the model was recalibrated using values of -3×10^{-8}, 0, 3×10^{-8}, 3×10^{-7}, and 3×10^{-6} for K_1+K_S. After the adjustment, the model predictions were compared to the measurement.

2.8 Quality Control and Assurance

Quality control and assurance for each parameter was conducted according to Standard Methods (APHA, 1995). Once a month, de-ionized water stored in four acid (20% H_2SO_4)-washed and triple-rinsed glass bottles was taken to the field sites. With the lids of two bottles opened and another two closed, these bottles were kept in the site for one minute. Then, the same analysis process as for the samples was conducted, to determine TP and SRP in the de-ionized water. Equipment for measuring flow velocity, pH, electrical conductivity, DO and temperature was checked and calibrated regularly. Triplicate sampling was conducted on the first sampling day of each month.

2.9 Statistical Analysis

SPSS software (Version 15.0, SPSS® Inc.) was utilized to conduct statistical analysis on comparing the inflow and outflow concentrations and loads of water quality parameters, and Columbia Lake's internal loading rates in the post-design period to the same data in the pre-design period. Previous studies show that mean concentrations of many pollutants in urban runoff are not normally distributed (Van Buren et al., 1997; Oberts and Osgood, 1991; Salvia-Castellvi et al., 2001). However, concentrations of dissolved constituents, such as TDS, seemed to follow the normal distribution (Van Buren et al., 1997a). Hence, Shapiro-Wilk tests with a statistical significance threshold of $p = 0.05$ were used to examine whether or not the data were normally distributed. These normality tests showed that surface water quality and quantity data were not normally distributed. Therefore, nonparametric Kruskal-Wallis tests followed by Bonferroni multiple comparison tests in Univariate Analysis of Variance were applied (personal communication, Erin Harvey, University of Waterloo Statistics Consulting Service, 2008). These tests were used to compare the temporal differences of discharge, concentrations and loads of TP, SS and TDS, DO concentrations, temperature and pH in each study site, and water retention time and internal P and SS loading rates between the pre- and post-design periods. In addition, spatial differences of these parameters were examined to predict the P cycle in the impoundment. Monthly changes in the concentrations and loads of TP, SRP, SS and TDS, discharge, DO concentrations, and pH were examined by multiple comparison tests (Post Hoc Tests: Bonferroni) in Univariate Analysis of Variance. The confidence level is set at 95%.

Analysis of the correlation between water quality and quantity parameters was done to examine the hydrological, biochemical and physical effects on P and SS transfer via Pearson's Correlation tests. Correlation analysis requires a normally distributed data set. Van Buren et al. (1997a) concluded that SS and it associated nutrients followed log-normal distribution. Hence, the surface water data were converted to a log-normal distribution before the correlation analysis was conducted. In addition, L_{IN} vs. L_{OUT} curve was analyzed to characterize P and SS transfer via the lake.

In terms of modeling validation, a coefficient was added to the original model when the predicted values were considerably different from the actual measurements. After the model adjustment, the adjusted predictions and the original predicted values were compared with measured data, respectively, via multiple comparison tests (Post Hoc test: Bonferroni) in Univariate Analysis of Variance. Regressions of measured-predicted data were used to isolate extreme values at a confidence interval of 95%.

Chapter 3 RESULTS

3.1 Meteorology

Based on 1970-2000 data from the University of Waterloo metrological station, the average annual daily high temperature was 11.9°C and the average annual daily low temperature was 1.7 °C. For the period of record, the annual average precipitation is 904.4 mm and monthly temperature and total precipitation for the months of May to August during 2003, 2004, 2006 and 2007 are listed in Table 3-1. Monthly mean temperatures for the four study years were within ±3.0 °C of the 30 year average (1970-2000). However, monthly total precipitations were quite different from the 30-year historical average. In May, the average historical precipitation was 75.7 mm; precipitation values for May of 2003, 2004, and 2006 were much higher, with values of 122.7, 145.0 and 113.4 mm, respectively. However, the average precipitation was only 59.1 mm in May of 2007, only 78% of the historical average for this month.

The June mean precipitation for the years 2003 to 2007 (except 2005) was approximately 50, 64, 41 and 33% of the historical value (Table 3-1). Precipitation in July was more variable. In 2003 and 2007, the precipitation was about 55% of the historical average of 92.9mm. However, the average precipitation in July 2006, 152.2 mm, was extremely high compared to the 30 year average. August also had dry weather: precipitation values from 2003 to 2007 were only 49, 64, 63 and 72% of the historical average, respectively.

Table 3-1 A summary of monthly temperature and total precipitation during these study periods

	Temperature (°C)			30 year		Total precipitation (mm)	30 year (mm)
	Aver. Daily High	Aver. Daily Low	High/low Temp	Aver. Daily High	Aver. Daily Low		
May-03	18.9	6.6	22.9/-0.4	18.9	6.6	122.7	75.7
May-04	16.6	4.8	27.7/-2.9			145.0	
May-06	19.0	8.0	32.7/-1.1			113.4	
May-07	20.4	7.2	30.0/-1.1			59.1	
Jun.-03	23.6	11.3	30.2/3.8	23.6	11.3	40.0	80.0
Jun.-04	26.9	15.3	29.2/3.8			51.2	
Jun.-06	23.7	13.1	31.1/6.0			32.8	
Jun.-07	25.7	12.7	32/6.5			26.6	
Jul.-03	25.9	13.8	31.2/9.5	26	13.8	52.7	92.9
Jul.-04	27.4	15.3	29.3/9.0			108.6	
Jul.-06	26.9	16.9	31.9/10.1			152.2	
Jul.-07	25.2	13.4	31.5/7.5			50.8	
Aug.-03	24.7	12.6	30.1/7.7	24.7	12.6	39.0	87
Aug.-04	25.8	15.3	27.6/5.2			56.4	
Aug.-06	24.5	13.9	33.7/7.7			52.4	
Aug.-07	25.5	14.3	33.1/7.9			62.6	

*Data from University of Waterloo Weather Station (43° 28’ 25.6” N: 80° 33’ 27.5” W, Elevation: 334.4 m ASL)

3.2 Hydrology

3.2.1 Discharge

Discharge was monitored bi-weekly at the inflow and outflow of Columbia Lake from May to August in 2003, 2004, 2006 and 2007. For the pre-design period (2003-2004), inflow discharge ranged from 0.025 to 2.2 m^3 s^{-1}. At the outflow, discharge ranged from 0.0074 to 0.41 m^3 s^{-1}. For the post-design period (2006-2007), inflow discharge ranged from 0.0074 to 0.41 m^3 s^{-1} while outflow discharge ranged from 0.0094 to 0.48 m^3 s^{-1} (Appendix 1).

Temporal variation in the inflow and outflow discharge for the period of record is shown in Figure 3-1. Totally, there were 60 and 59 measurements at the inflow and outflow during the pre-design period, and 52 and 53 samples at the two sites during the post-design period. Discharge was measured during storm events on Julian Day (JD) 146, 162, 169 and 197, 2004 and JD 185, 2007. The figure shows that inflow and outflow were similar and generally below 1.0 m^3 s^{-1} for most of the sampling days. However, on 146 in 2004, discharge at both sites was above 2.0 m^3 s^{-1} during a storm event. Discharge in 2007 was lower than in previous years. A Shapiro-Wilk Test indicated that the discharge was not normally distributed and was skewed with a median of 0.12 m^3 s^{-1}.

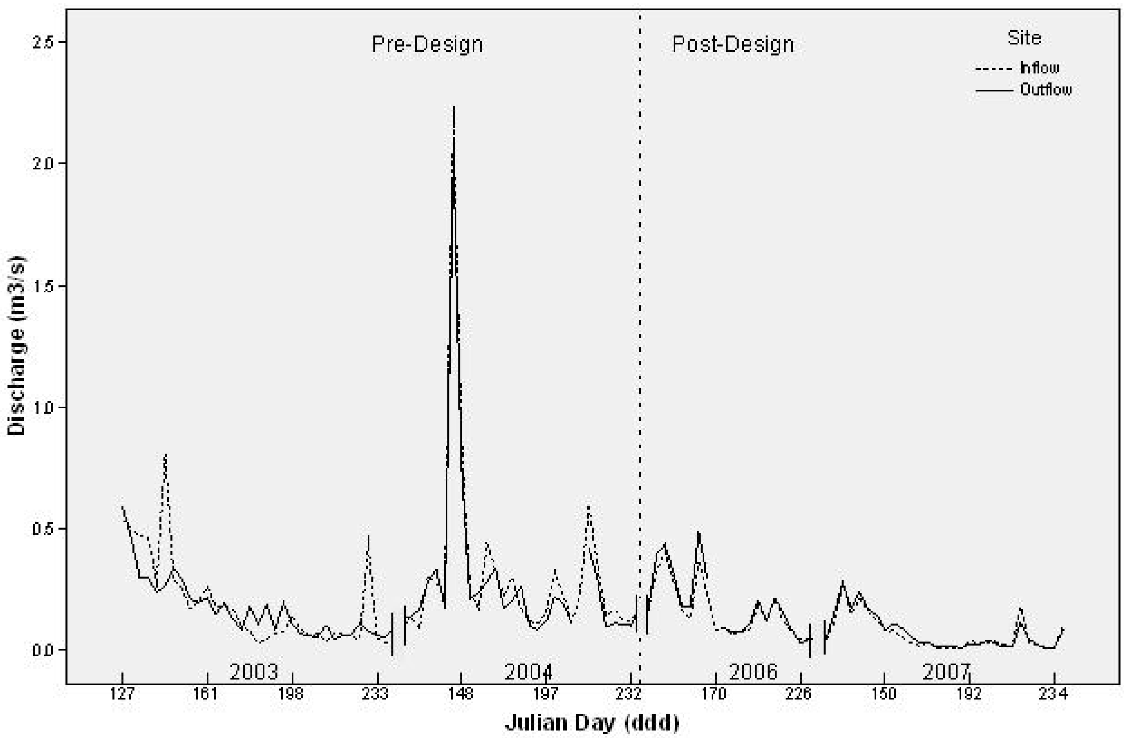

Figure 3-1 The discharge at the inflow and outflow for the pre- and post-design periods

The monthly variation in inflow and outflow discharge for pre- and post-design periods is shown in Figure 3-2. Monthly discharge during the pre-design period was higher than in the post-design period, particularly in May and June. Almost all discharge measurements were below 0.5 m^3 s^{-1}. The discharge in May was higher than in the three subsequent months.

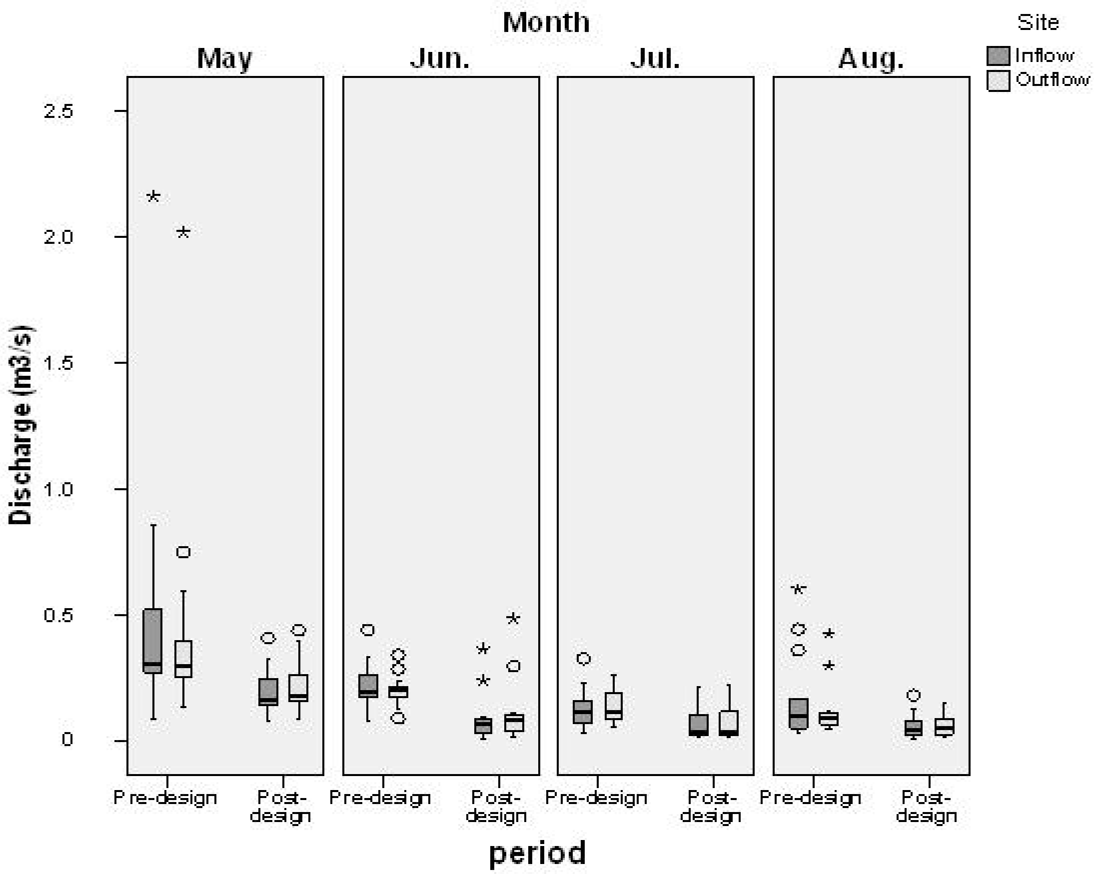

Figure 3-2 Monthly discharge changes at the inflow and outflow for the pre- and post-design periods (* = the extreme, ° = the outliner)

3.2.2 Water Retention Time (R_T)

Water retention times (R_T) of Columbia Lake during the post-design period are listed in Table 3-2. The table shows that mean R_T was much higher in the post-design period compared to the pre-design period. In the pre-design period, the mean R_T was 317 h (~ 13 days) and it ranged from 16 to 1411 h (from 0.7 to 59 days). In the post-design period, the mean R_T was 982 h (~ 41 days) and it varied from 86 to 4692 h (from 4 to 196 days). A Shapiro-Wilk Test indicated that the R_T data were not normally distributed.

Table 3-2 Water retention time (h) during the pre- and post-design periods

Period	N	Mean	SEM	Minimum	Maximum	Range	SD
Pre-design	60	317	38	16	1411	1398	298
Post-design	53	982	164	86	4692	4607	1192
Total	113	629	85	16	4692	4676	904

* N is sample size; SEM is standard error of mean; SD is standard deviation; Range = (Maximum - Minimum).

The temporal change in R_T for pre- and post-design periods is shown in Figure 3-3. Water retention time sharply increased after the lake retrofit project, particularly in 2007. On JD 178 and 232 of 2007, the water retention time was 4509 and 4692 h (about 188 and 196 days), respectively. The minimum water retention time of 16 h was measured during a storm event on JD 146, 2004.

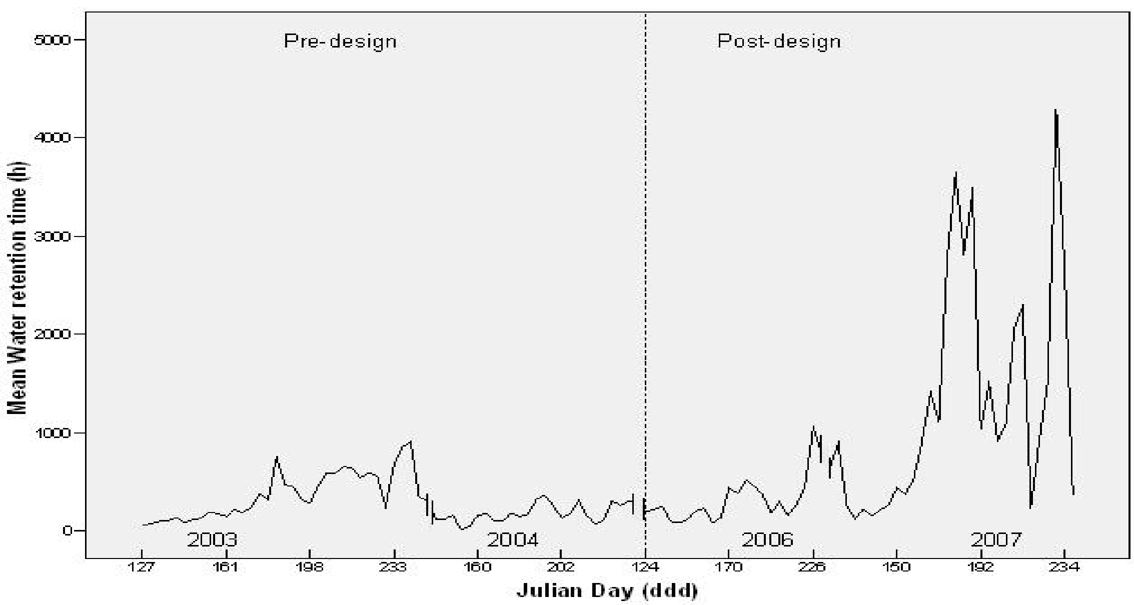

Figure 3-3 Water retention time for the pre- and post-design periods

3.3 Phosphorus

3.3.1 Total Phosphorus (TP)

3.3.1.1 Concentrations

Inflow TP concentrations ranged from 18 to 372 μg L^{-1} during the pre-design period and from 10 to 124 μg L^{-1} during the post-design period, with averages of 56±7 and 53±5 μg L^{-1} and SDs of 53 and 3 μg L^{-1}, respectively. At the outflow, TP concentrations significantly decreased during the post-design period. For the pre-design period, the outflow TP concentrations ranged from 37 to 266 μg L^{-1}, and for the post-design period, ranged from 14 to 147 μg L^{-1}, with averages of 116±6 and 44±3 μg L^{-1} and SDs of 47 and 23 μg L^{-1}, respectively (Appendix 1). The maximum TP concentration occurred at the inflow on JD 233 in 2003, with the value of 372 μg L^{-1} and the minimum TP concentration of 10 μg L^{-1} was measured at the inflow on JD 130, 2006. A Shapiro-Wilk Test indicated the data were not normally distributed, being skewed with a median of 49 μg L^{-1}.

Temporal variation in TP concentrations at the inflow and outflow of Columbia Lake for pre and post design periods is illustrated in Figure 3-4. With the exception of measurements on JD 231, 233 and 238 in 2003, when TP concentrations were more than 200 μg L^{-1}, inflow TP concentrations during the post-design period were similar to those during the pre-design period. However, outflow TP concentrations decreased considerably after the lake retrofit project. Most of the TP concentrations changed from the pre-design range of 100 – 300 μg L^{-1} to a range of 5 – 70 μg L^{-1} during the post-design period. The difference between the inflow and outflow concentrations was also significantly reduced. For the pre-design period, all the TP concentrations increased by more than 20 μg L^{-1} after the water flowed through the lake. On JD 139 in 2004, the outflow TP concentrations were more than 200 μg L^{-1} higher than the inflow. However, from 2006 to June in 2007, outflow TP concentrations were not obviously distinguishable from inflow concentrations. Subsequent TP concentrations at the outflow were significantly lower than at the inflow (Figure 3-4).

According to provincial water quality objectives (PWQO), TP concentrations in surface water should be < 30 μg L^{-1} (OMEE, 1994). The City of Waterloo (2004) set water quality targets for Laurel Creek and require that TP concentrations downstream of Laurel Creek Reservoir should not exceed 80 μg L^{-1}. For the pre-design period, only 18% and 0% of the TP concentrations were under the provincial objective at the inflow and outflow, respectively. About 28% of outflow TP concentrations were under the water quality target for Laurel Creek. Most of the measured concentrations in 2003 and 2004 exceeded both the PWQO and the city's target for the outflow (Figure 3-4). However, after the reconstruction of Columbia Lake, outflow TP concentrations decreased dramatically. Although only 29% and 25% of the inflow and outflow TP concentrations were still above the OMEE PWQO level of 30 μg L^{-1}, respectively, 94% of the monitored outflow concentrations were below 80 μg L^{-1} (Figure 3-4).

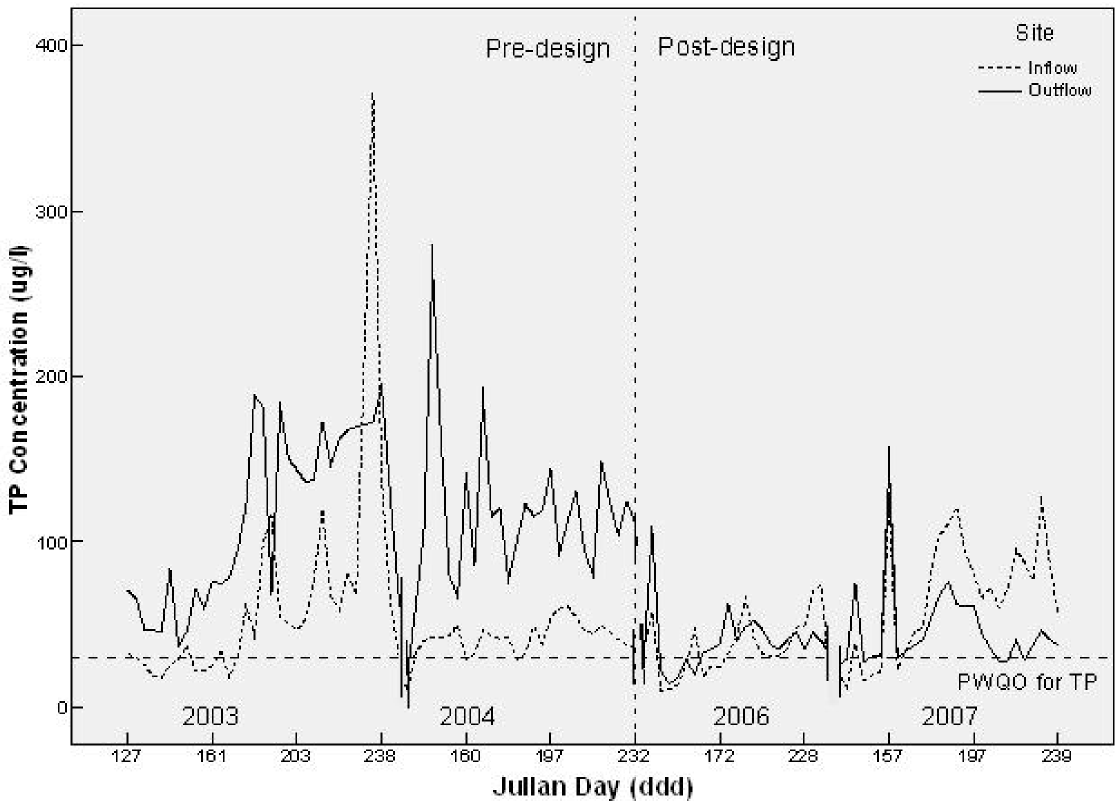

Figure 3-4 The TP concentrations at the inflow and outflow for the pre- and post-design periods

Variations in monthly TP concentration (mean and range) at the inflow and outflow of Columbia Lake for the pre- and post-design periods are illustrated in Figure 3-5. In the pre-design period, monthly average outflow concentrations were consistently higher than the inflow. Outflow TP concentrations increased from month to month, while inflow concentrations remained relatively less variable. In the post-design period, inflow concentrations gradually increased from May to August. From June to August, the mean outflow TP concentrations were lower than the inflow.

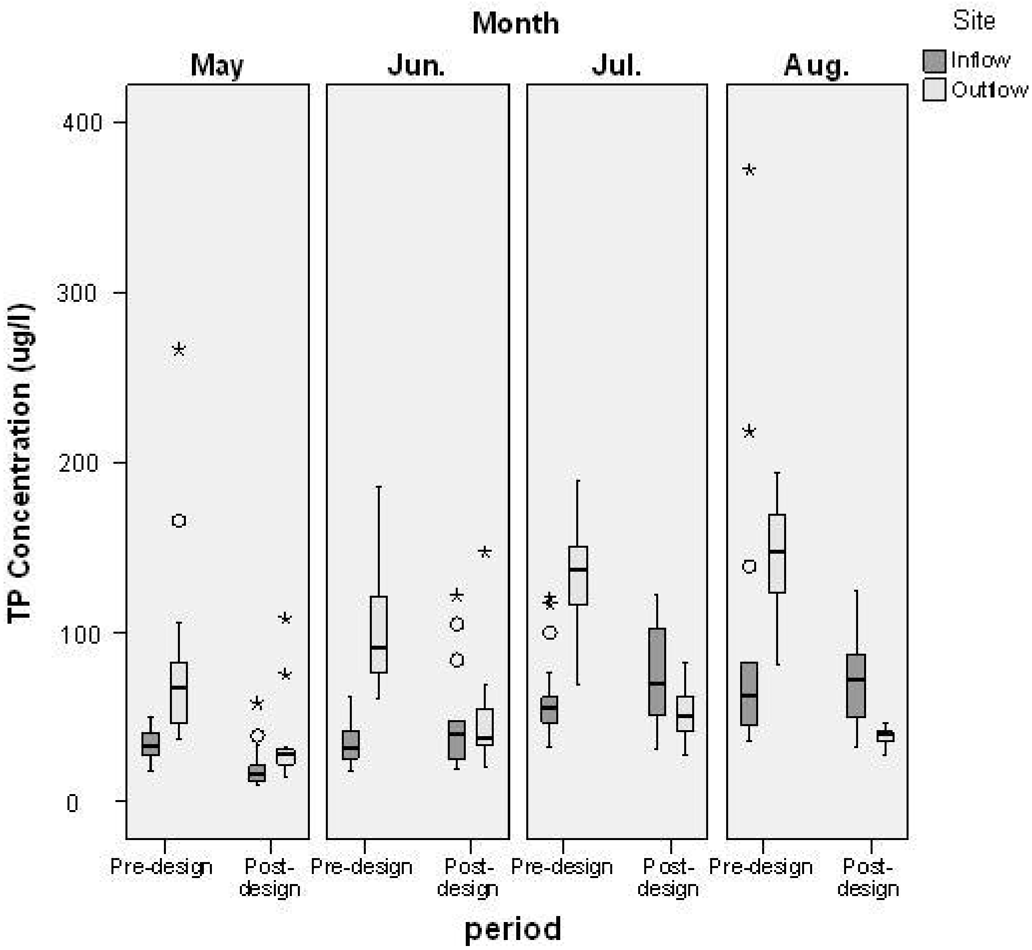

Figure 3-5 Monthly TP concentration changes at the inflow and outflow for the pre- and post-design periods (* = the extreme, ° = the outliner)

3.3.1.2 Hourly Loads

In the post-design period, TP hourly loads ranged from 3 to 84 g h^{-1} at the inflow and from 1 to 169 g h^{-1} at the outflow, with averages of 14±2 and 18±4 g h^{-1}, respectively (Appendix 1; Figure 3-6). Pre-design TP hourly loads varied between 5 to 346 g h^{-1} with an average of 43±8 g h^{-1} at the inflow, and between 22 to 582 g h^{-1} with an average of 82±11 g h^{-1} at the outflow. Figure 3-6 shows that both the inflow and outflow TP hourly loads decreased sharply after the lake was redesigned. The peak hourly load of 582 g h^{-1} occurred at the outflow on JD 146 in 2004, while the minimum value of 1 g h^{-1} was measured at the outflow on JD 211 in 2007. The distribution of TP hourly loads data was also skewed, with a median of 21 g h^{-1}.

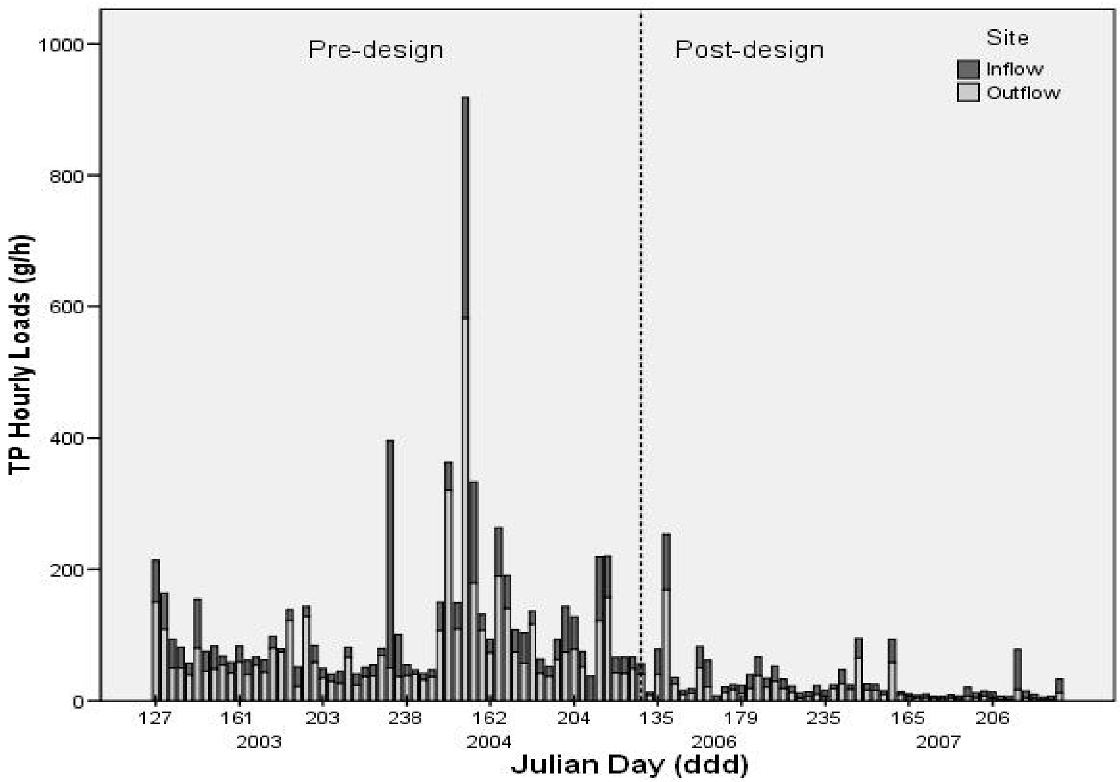

Figure 3-6 Inflow and outflow TP hourly loads for the pre- and post-design periods

The net internal TP loading (inflow – outflow TP loads) is presented in Figure 3-7. Except on JD 231 in 2003, outflow TP loads measured were consistently higher than the inflow loads during the pre-design period. The data show that Columbia Lake was a TP source during the pre-design period and the net internal loads varied from 20 to 300 g h^{-1}. However, after the redesign project, net TP internal loads decreased considerably. Except on JD 137 in 2006 when about 85 g TP h^{-1} was exported from Columbia Lake, the difference in internal TP loading was within ±20 g h^{-1} for the rest of the sampling days in the post-design period. From the initial measurements in 2006 to JD 178 in 2007, most of the net internal loads were positive, which indicated that TP was exported from the impoundment. However, negative net internal TP hourly loads indicate that Columbia Lake had become a TP sink since JD 190, 2007.

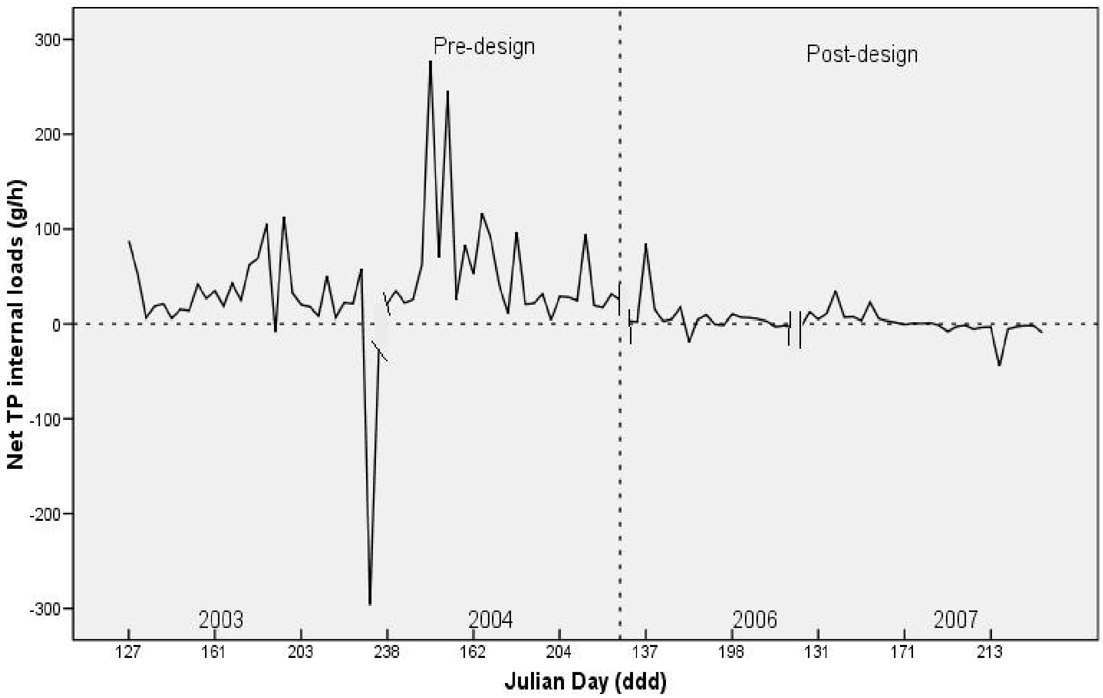

Figure 3-7 Net internal TP hourly loads from Columbia Lake in the pre- and post-design periods

The monthly change in TP hourly loads for the pre- and post-design periods is shown in Figure 3-8. During the pre-design period, the average outflow loads were consistently higher than the inflow loads during the study period (Figure 3-8). The TP loads varied more in May than in the subsequent months. In the post-design period, inflow TP hourly loads were more similar to the outflow loads. Hourly loads of TP were similar among each month.

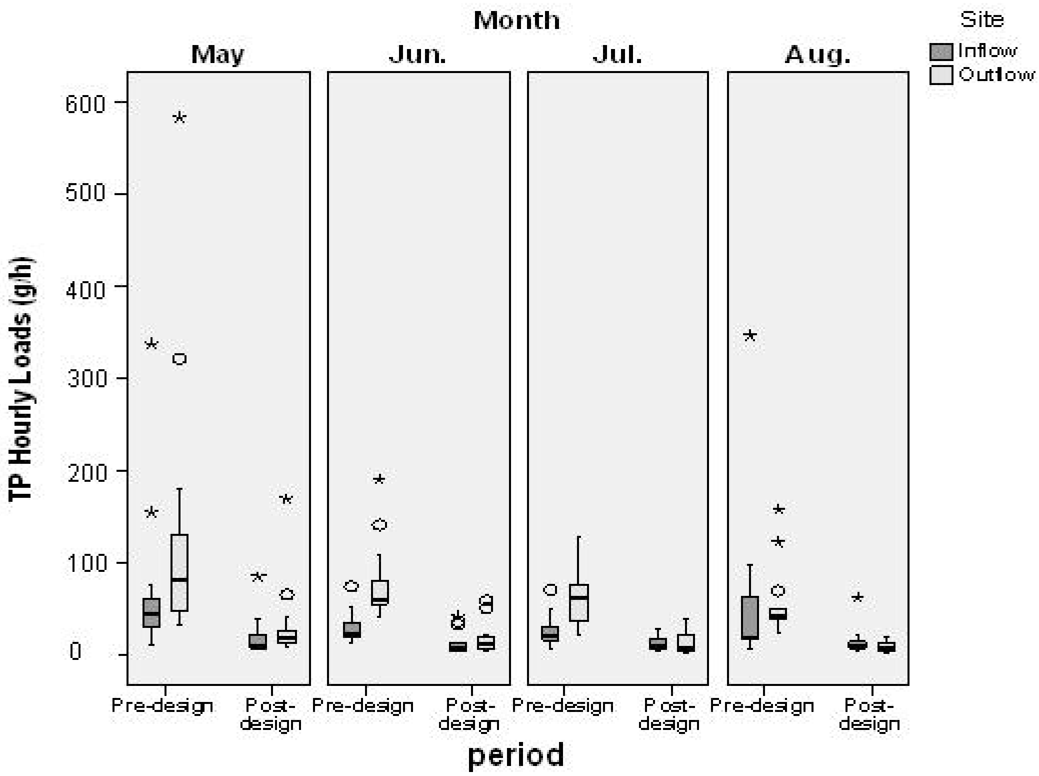

Figure 3-8 Monthly TP hourly loads at the inflow and outflow for the pre- and post-design periods (* = the extreme, ° = the outliner)

3.3.1.3 Net Internal Loading Rate

Figure 3-9 indicated the TP internal loading rates during both the pre- and post-design periods. TP internal loading rates varied between – 86% and 1521%, with an average and an SD of (198 ± 179) % and 250% during the pre-design period. After the Columbia Lake redesign, internal loading rates decreased considerably. Internal loading rates of TP changed from – 75% to 214% with an average and an SD of (22 ± 31) % and 69%. Columbia Lake was a TP source prior to the project. However, in 2006 and 2007, the lake was a TP source in May and June, but a TP sink in August.

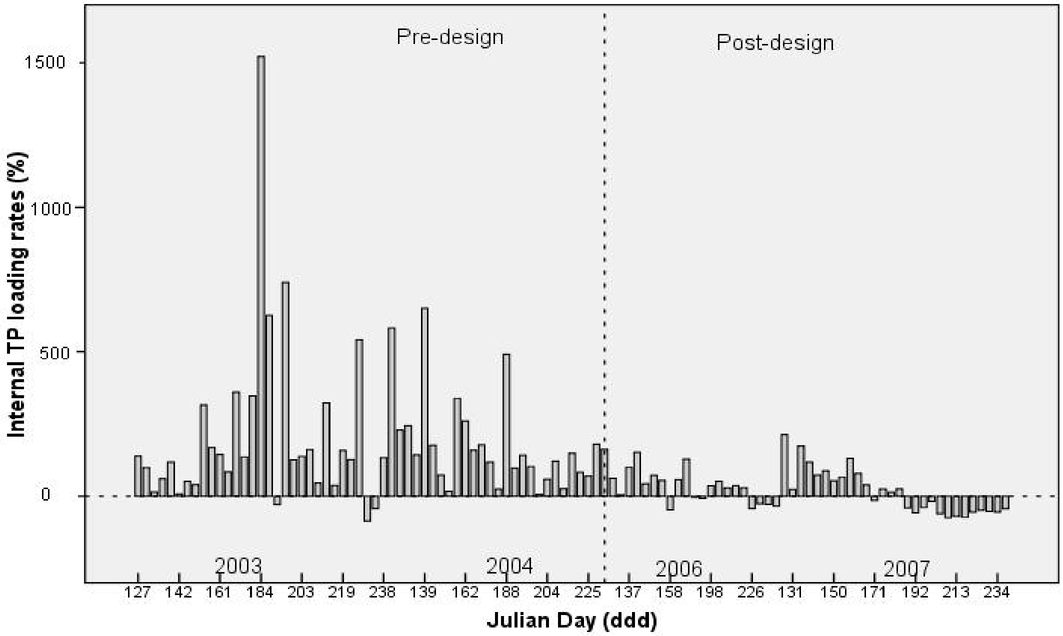

Figure 3-9 Columbia Lake performance characterized by internal TP loading rates during the pre- and post-design periods

3.3.2 Soluble Reactive Phosphorus (SRP)

3.3.2.1 Concentrations

Concentrations of SRP at the Columbia Lake inflow and outflow for the period May to August, 2007 are shown in Figure 3-10. The average SRP concentrations were 10±1 μg L^{-1} and 9±1 μg L^{-1} at the inflow and outflow, respectively. The ranges varied slightly, from 2 to 17 μg L^{-1} at the inflow and from 5 to 19 μg L^{-1} at the outflow (Appendix 1). The maximum SRP concentration of 19 μg L^{-1} occurred at the outflow on JD 239, 2007. Figure 3-10 shows that SRP concentrations varied dramatically over time but generally increased in July and August. Both a histogram and a Shapiro-Wilk Test indicated that the SRP data were not normally distributed.

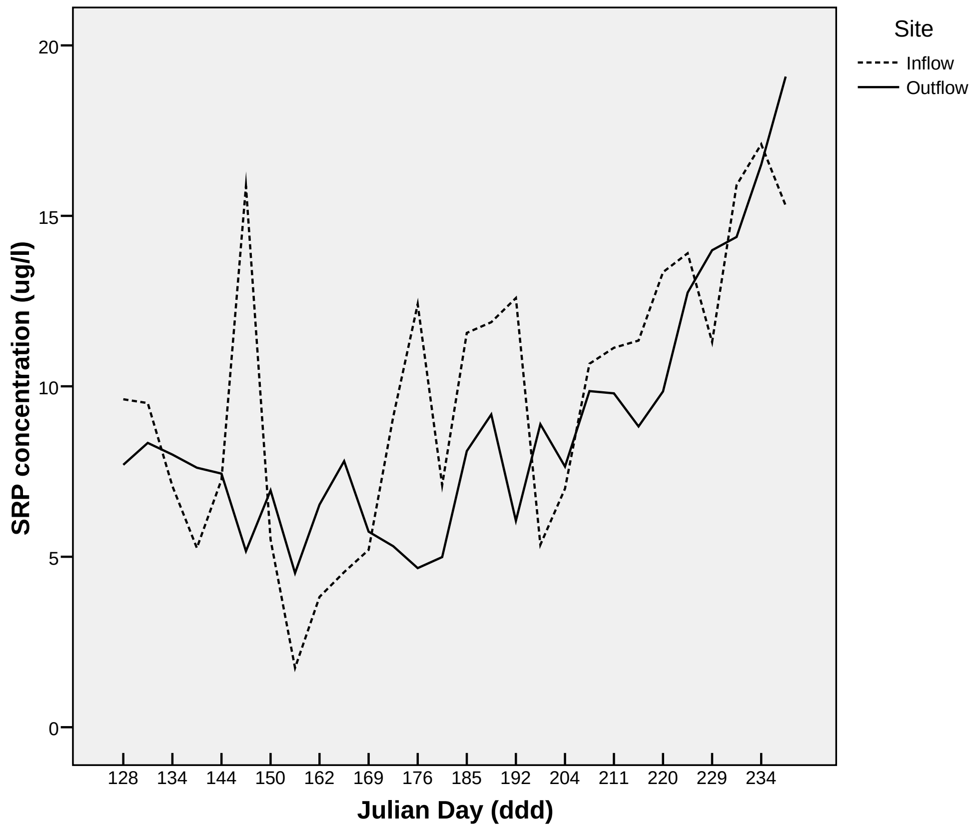

Figure 3-10 SRP concentrations at the inflow and outflow (May to August, 2007)

3.3.2.2 Hourly Loads

The hourly loads of SRP were determined for May to August in 2007 and presented in Figure 3-11. The average SRP hourly loads were 2.2± 0.5 g h^{-1} at the inflow and 2.1 ± 0.4 g h^{-1} at the outflow. The loads changed from 0.2 to 9.4 g h^{-1} and from 0.2 to 8.4 g h^{-1} at the inflow and outflow, respectively. Net internal SRP hourly loads were highly variable and alternated from negative and positive values (Figure 3-11). On JD 148 and 220, net internal loads were negative and SRP retention was 4.4 and 4.5 g h^{-1}, respectively, while on JD 142, the net SRP internal load was 2.6 g h^{-1}.

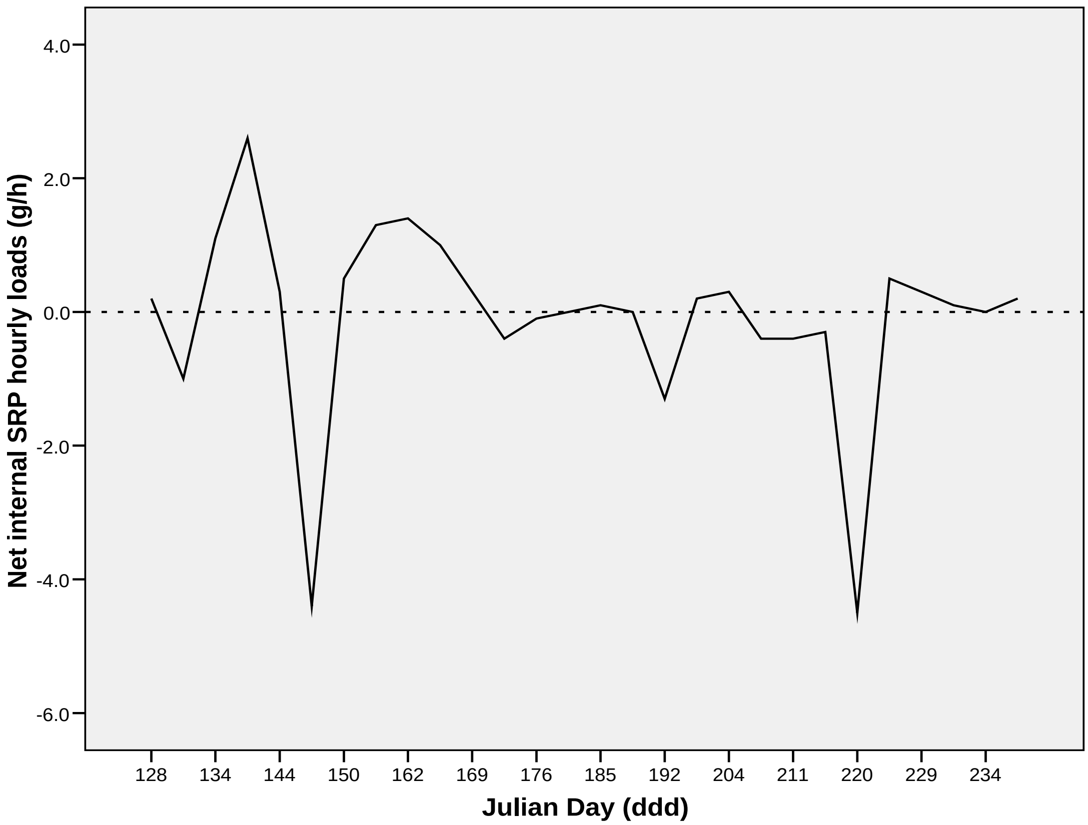

Figure 3-11 Net internal SRP hourly loads from Columbia Lake in 2007

3.3.3 P forms in Columbia Lake

The SRP and TP ratio at the inflow and outflow of Columbia Lake varied dramatically during the study period (Figure 3-12). The SRP / TP ratios for loads varied from 1% to 82% with the average of 22% and from 3% to 50% with the average of 23% at the in- and outflow, respectively. In May, the SRP / TP ratio at the inflow ranged from 10% to 80% and was considerably higher than the outflow ratios. During subsequent sampling periods, the ratios changed slightly between the inflow and outflow and remained steady at the range of 0 to 20%. In late July and August, outflow SRP / TP ratios increased from 10% to about 50% and were much higher than the inflow ratios. The peak occurred at the inflow on JD 148 when SRP was 82% of TP, while outflow SRP/TP was only less than 20% on the same day (Figure 3-12).

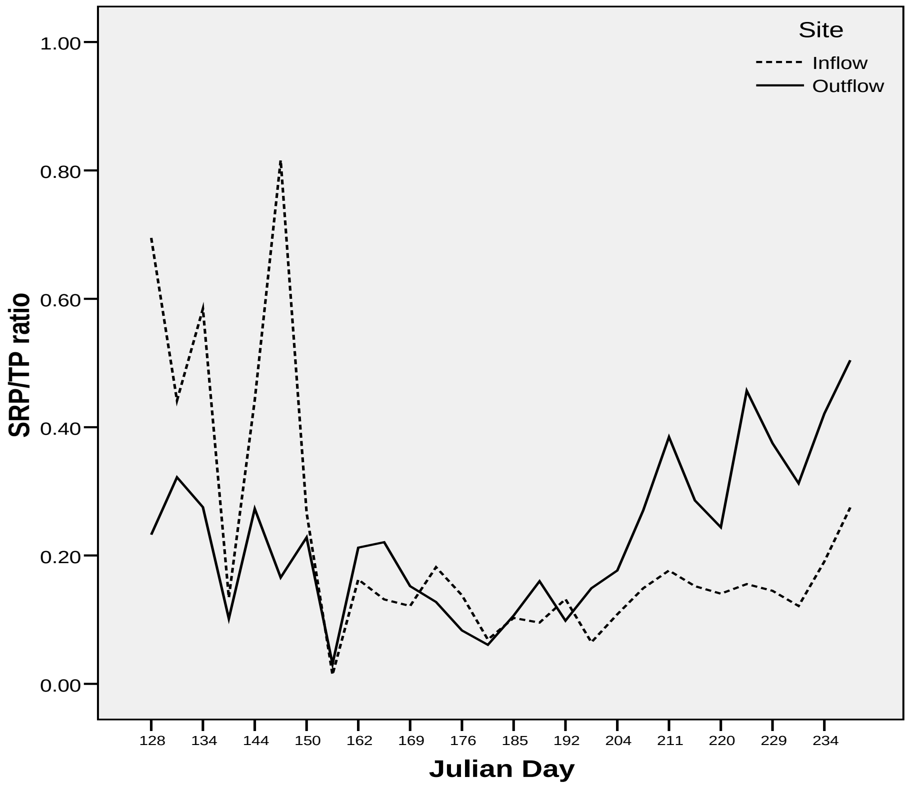

Figure 3-12 SRP / TP ratios at the inflow and outflow (May to August, 2007)

3.4 Suspended Solids and Grain Size Distribution

3.4.1 Suspended Solids (SS)

3.4.1.1 Concentrations

Concentrations of SS at the inflow and outflow of Columbia Lake for pre- and post-design periods varied considerably. In the pre-design period, SS concentrations ranged from 1.8 to 168.5 mg L^{-1} at the inflow, and from 4.0 to 194.7 mg L^{-1} at the outflow. The averages were 19.0±3.2 and 66.6±4.7 mg L^{-1}, respectively. In the post-design period, SS concentrations decreased drastically. Inflow SS concentrations varied from < 0.1 to 25.8 mg L^{-1} with a mean of 8.5±0.8 mg L^{-1}, and from < 0.1 to 42.5 mg L^{-1} with a mean of 14.5±0.8 mg L^{-1} at the outflow (Appendix 1). The maximum value of 190 mg L^{-1} was measured on JD 167 in 2004 and the minimum values of

< 0.1 mg L^{-1} were observed occasionally in the post-design period. A Shapiro-Wilk Test indicated that the SS data were not normally distributed and skewed with a median of 15.2 mg L^{-1}.

Temporal variability in SS concentrations at the inflow and outflow of Columbia Lake during the pre- and post-design periods is shown in Figure 3-13. With the exception of JD 163 (2003) and JD 132, 216 and 230 (2004) when SS concentrations were > 55 mg L^{-1}, the inflow SS concentrations for most of the samples were ≤ 25 mg L^{-1} in both pre- and post-design periods. Outflow SS concentrations were more variable. During the pre-design period, outflow SS concentrations were more than 50 mg L^{-1} higher than inflow, indicating that Columbia Lake was a source of sediments to downstream reaches. After redesign of Columbia Lake, SS concentrations were less variable at both the inflow and outflow (Figure 3-13). In the post-design period, the outflow SS concentrations never exceeded the inflow concentrations by more than 10 mg L^{-1}. In the pre-design period, most of the SS concentrations ranged from 50 to 150 mg L^{-1}. The peaks appeared on JD 217 (2003) and JD 167 (2004), when SS concentrations reached 177.9 and 194.7 mg L^{-1}, respectively. After the redesign project, except on JD 177 (2006) and JD 220 (2007) when SS concentrations reached 42.5 and 25.8 mg L^{-1}, respectively, all the measured SS concentrations were below 25 mg L^{-1}. The benchmark for SS concentration in surface water is 25 mg L^{-1} (OMEE, 1994; City of Waterloo, 2004). As shown in Figure 3-13, outflow SS concentrations exceeded this objective on 2% of the sampling days after the retrofit project, compared with only four out of 60 samples under this benchmark concentration in the pre-design period. High levels of SS can clog feeding structures, reducing feeding efficiency and therefore reducing growth rates, and even killing these organisms (Bilotta and Brazier, 2008). Bilotta and Brazier (2008) summarized that when surface water SS concentrations were above 8 mg L^{-1} during continuing 56 days, invertebrates density would reduced by 26%. During the Columbia Lake study, 33% and 52% of the inflow SS concentrations were under 8 mg L^{-1} during both the pre- and post-design periods, respectively. However, at the outflow, the SS levels were adverse to the aquatic invertebrates, with 97% and 94% of SS concentrations > 8 mg L^{-1}.

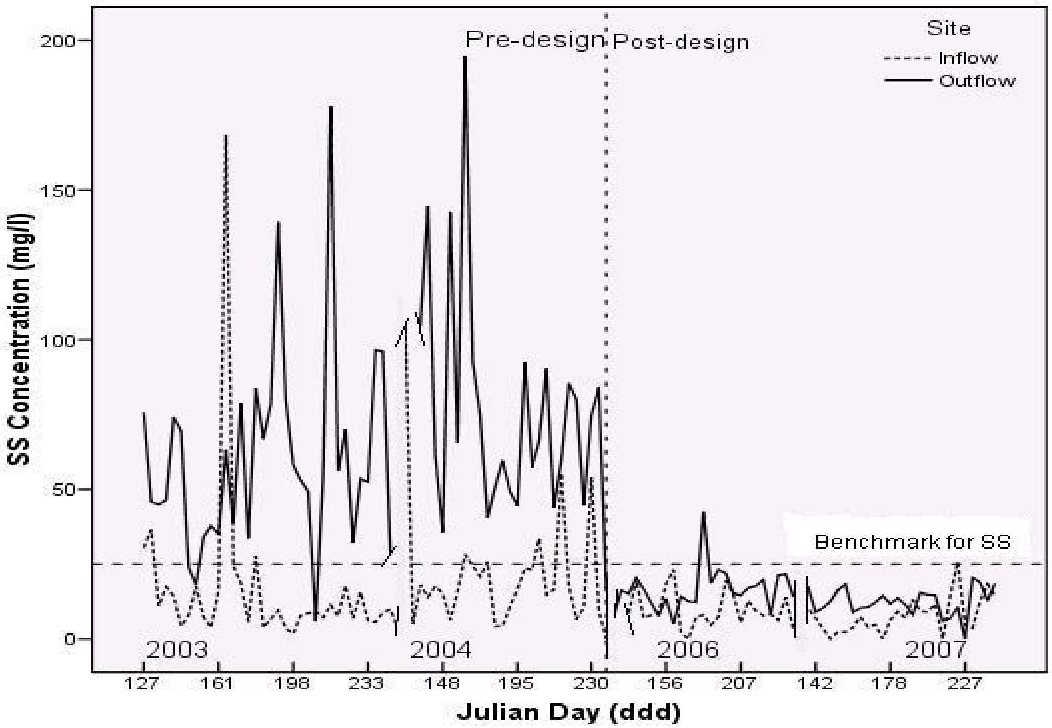

Figure 3-13 SS concentrations at the inflow and outflow for the pre- and post-design periods

The monthly changes in SS concentrations at the inflow and outflow of Columbia Lake for pre- and post-design periods are shown in Figure 3-14. In the pre-design period, monthly average SS concentrations at the outflow increased slightly from May to August, and those at the inflow fluctuated slightly. In the post-design period, there was no significant month to month difference. In each month for both periods, the outflow SS concentrations were consistently higher than the inflow. This was particularly true for the pre-design period, when differences between the averages of inflow and outflow SS concentrations varied from about 30 to more than 50 mg L^{-1} from May to August (Figure 3-14).

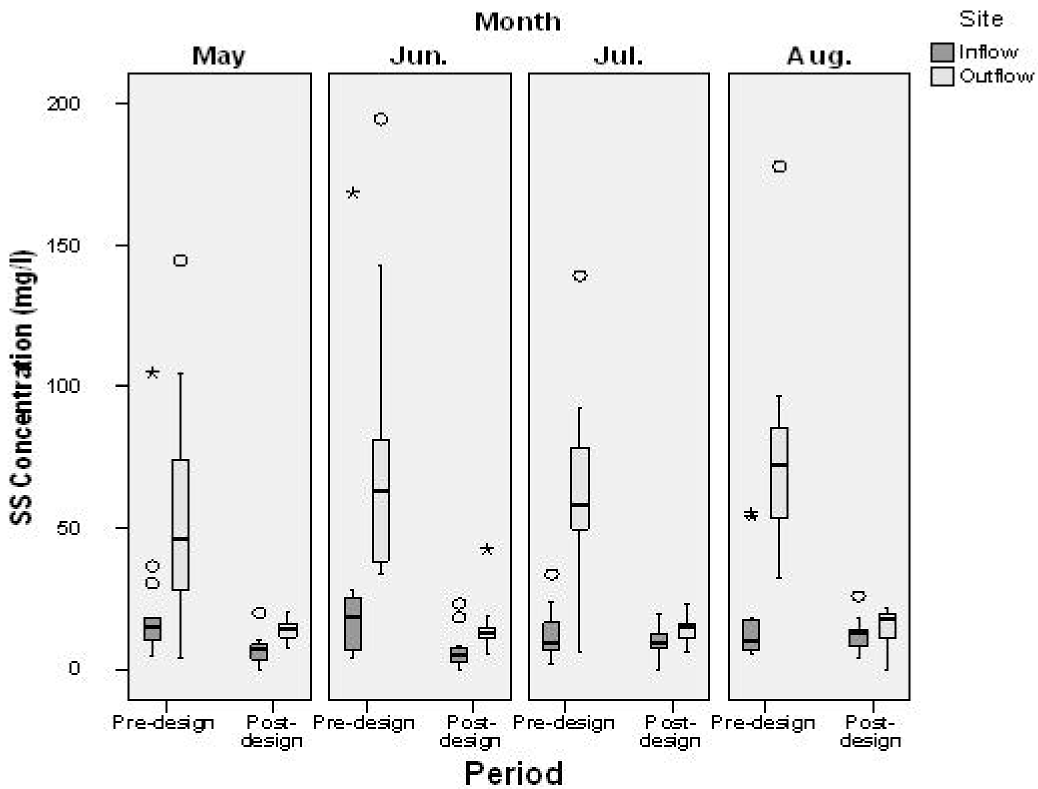

Figure 3-14 Monthly SS concentrations at the inflow and outflow for both pre- and post-design periods (* = extreme values, ° = outliner)

3.4.1.2 Hourly Loads

The SS hourly loads for pre- and post-design periods are presented in Figure 3-14. In the pre-design period, SS hourly loads ranged from 0.4 to 136.3 Kg h^{-1} at the inflow and from 1.2 to 447.3 Kg h^{-1} at the outflow, with averages of 18.6±3.6 and 52.2±8.6 Kg h^{-1}, respectively. After the Columbia Lake redesign project was completed, SS loads decreased considerably. During the post-design period, SS hourly loads ranged from < 0.1 to 29.1 Kg h^{-1} with an average of 3.9±0.8 Kg h^{-1} at the inflow and from < 0.1to 32.1 Kg h^{-1} at the outflow (Appendix 1).

Outflow SS hourly loads during the pre-design period were much higher than those in the post-design period (Figure 3-15). The maximum value of 447.3 Kg h^{-1} occurred on JD 146, 2004 during a storm event. On several sampling days in the post-design period, SS hourly loads were extremely low due to very low flows. According to a Shapiro-Wilk Test, the data were not normally distributed and were skewed with a median of 7.1 Kg h^{-1}.

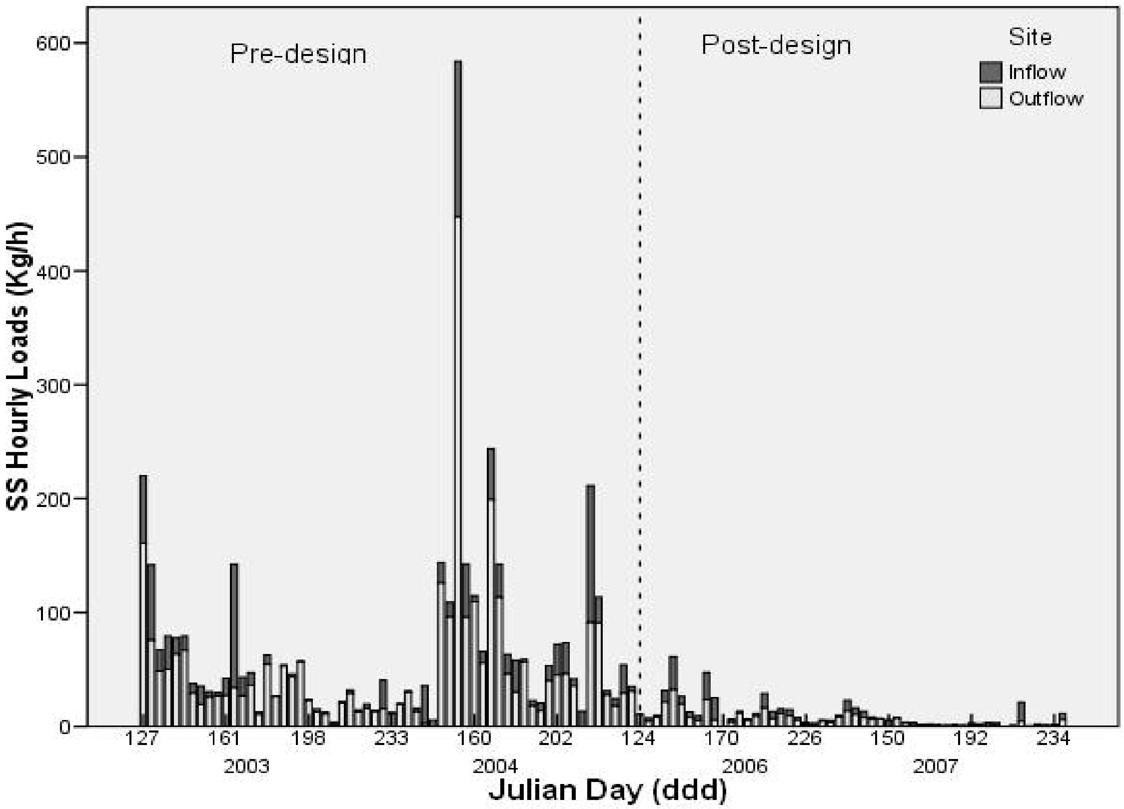

Figure 3-15 Inflow and outflow SS hourly loads during the pre- and post-design periods

Temporal variability in the net internal SS loads (difference between inflow and outflow SS hourly loads) during both the pre- and post-design periods is shown in Figure 3-16. Net internal SS loads varied more considerably during the pre-design period (from -80 Kg h^{-1} to > 300 Kg h^{-1}) than during the post-design period (Figure 3-16). On JD 163, 2003 and JD 132 and 216, 2004, net internal loadings were negative and SS were retained in the impoundment. However, on the rest of the sampling days in the pre-design period, Columbia Lake acted as a SS source, particularly on JD 146 in 2004, when more than 300 Kg SS h^{-1} were exported from the lake as a result of a storm event. However, during the post-design period, most of the net internal SS hourly loads did not vary by more than ±5 Kg h^{-1}, although the net internal loadings for most sampling days were positive. The maximum SS output in this period (12.1 Kg h^{-1}) occurred on JD 144, 2006, while the maximum SS retention happened on JD 158, 2006 and JD 220, 2007, with values of 14.4 and 12.2 Kg h^{-1}, respectively. In 2007, most of the measured net internal SS loads were about zero and Columbia Lake showed little effect on SS transfer (Figure 3-16).

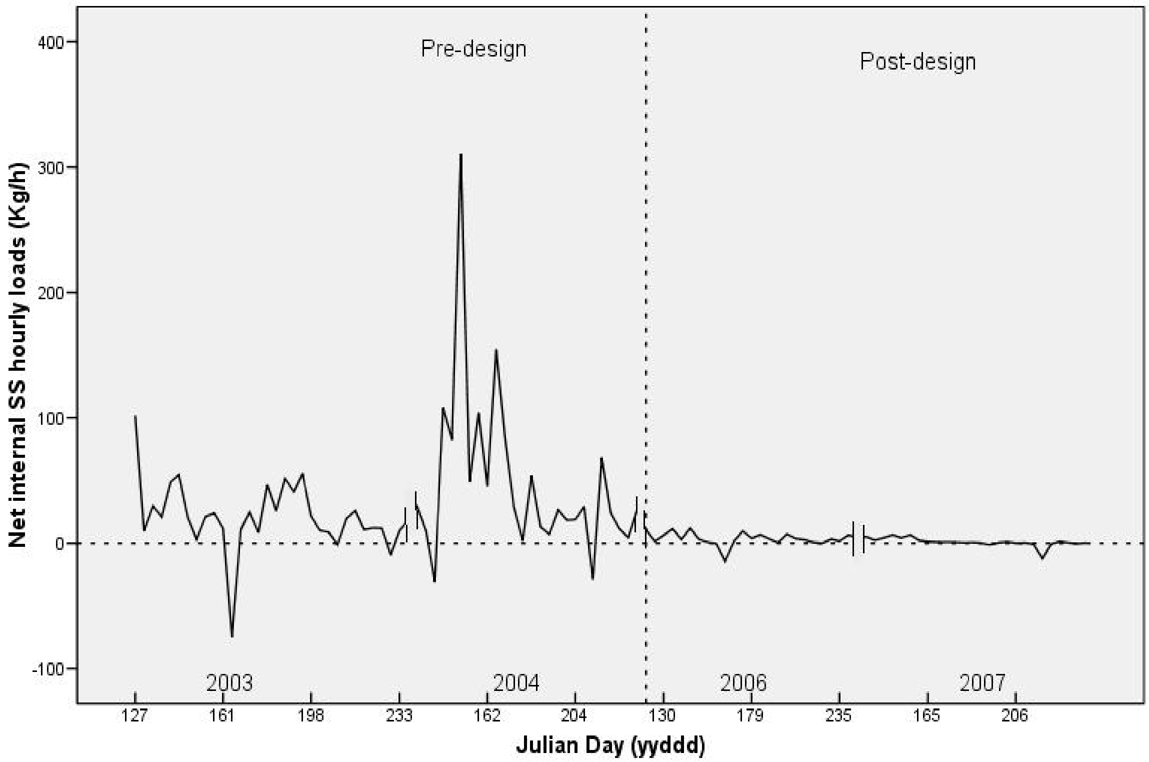

Figure 3-16 Net internal SS hourly loads from Columbia Lake during the pre- and post-design periods

Monthly differences in SS hourly loads during the pre- and post-design periods are shown in Figure 3-17. During the pre-design period, inflow SS loads were similar for each month, with outflow SS loads in May being higher than in subsequent months. Monthly averages of outflow SS loads decreased gradually from May to August (Figure 3-17). The outflow loads were significantly higher for each month than at the inflow for the pre-design period. However, after the lake retrofit project, SS hourly loads at both the inflow and outflow were similar. No significant post-design monthly differences were observed.

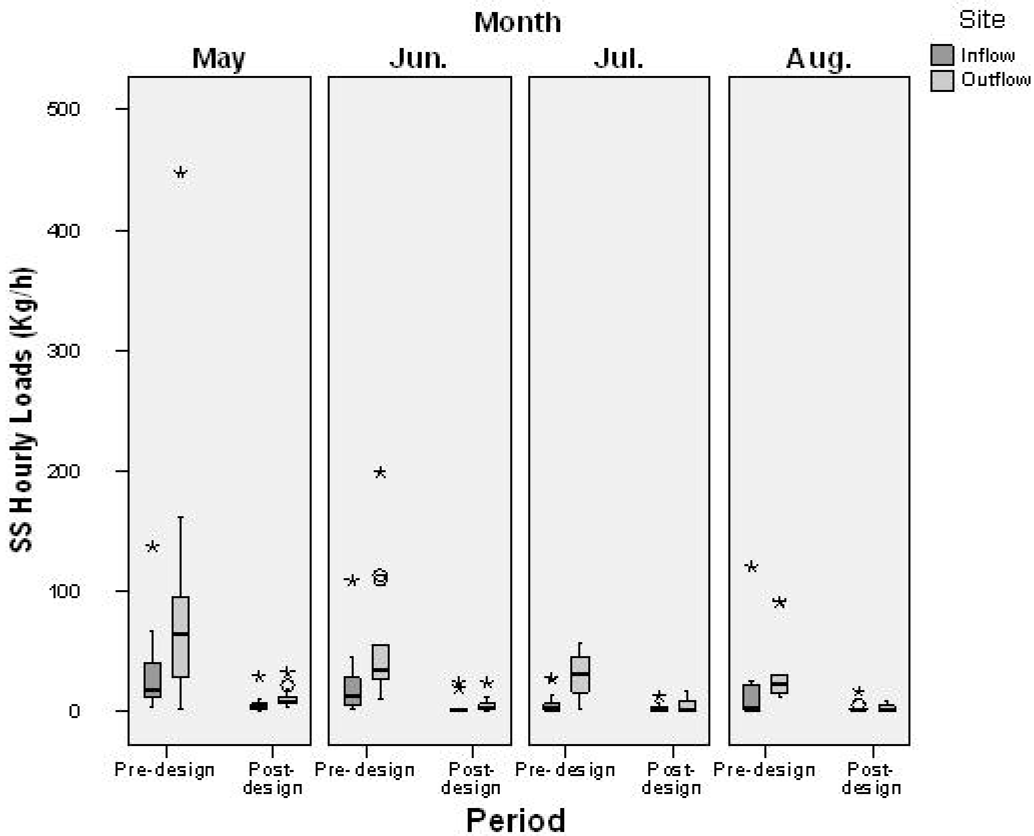

Figure 3-17 Monthly SS hourly loads at the inflow and outflow in the pre- and post-design periods (* = the extreme, ° = the outliner)

3.4.1.3 Net Internal Loading Rate

In the Columbia Lake study, impoundment performance on water quality changed considerably following the Columbia Lake redesign (Figures 3-18). SS internal loading rates changed from – 93% to 6307% with an average and an SD of (828 ± 33) % and 1360%, respectively, during the pre-design period. They varied from -100% to 1100% with an average and SD of (154 ± 10)% and 214% during the post-design period. The maximum value occurred on JD 184 in 2003, during a storm.

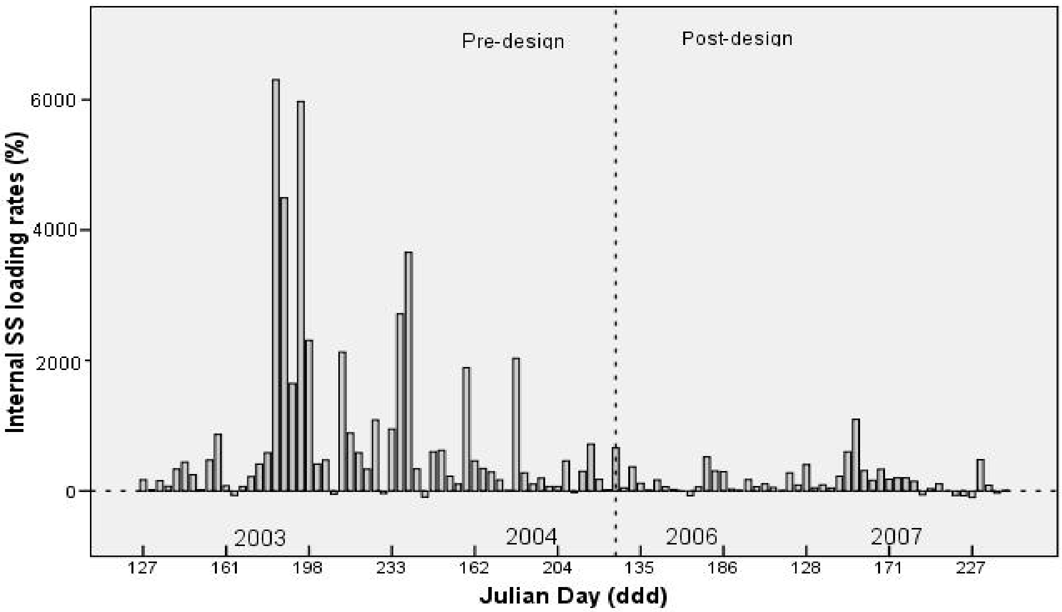

Figure 3-18 Columbia Lake performance characterized by internal SS loading rates during the pre- and post-design periods

3.4.2 Grain Size

3.4.2.1 Median Diameter (D_{50})

The grain size distribution of suspended solids was determined at the inflow and outflow of Columbia Lake from May to August, 2007. Temporal variation in the median diameter (D_{50}) is presented in Figure 3-19 . Generally, the grain size of SS at the inflow was larger than at the outflow. The values of D_{50} ranged from 4.9 to 9.0 µm with a mean of 6.4±0.2 µm at the inflow and from 4.3 to 7.1 µm with a mean of 5.4±0.1 µm at the outflow (Appendix 1). In 2007, the D_{50} data were not normally distributed, according to a Shapiro-Wilk Test.

Figure 3-19 displays the D_{50} at the inflow and outflow during the study period in 2007. Higher D_{50} values at the inflow were observed during almost the entire period except on JD 206, 211 and 213 when D_{50} values were about 1 µm higher at the outflow. The highest value of D_{50} was 9.0 µm, occurring at the inflow on JD 178, while the lowest D_{50}, 4.3 µm, was recorded at the outflow on JD 165 (Figure 3-19).

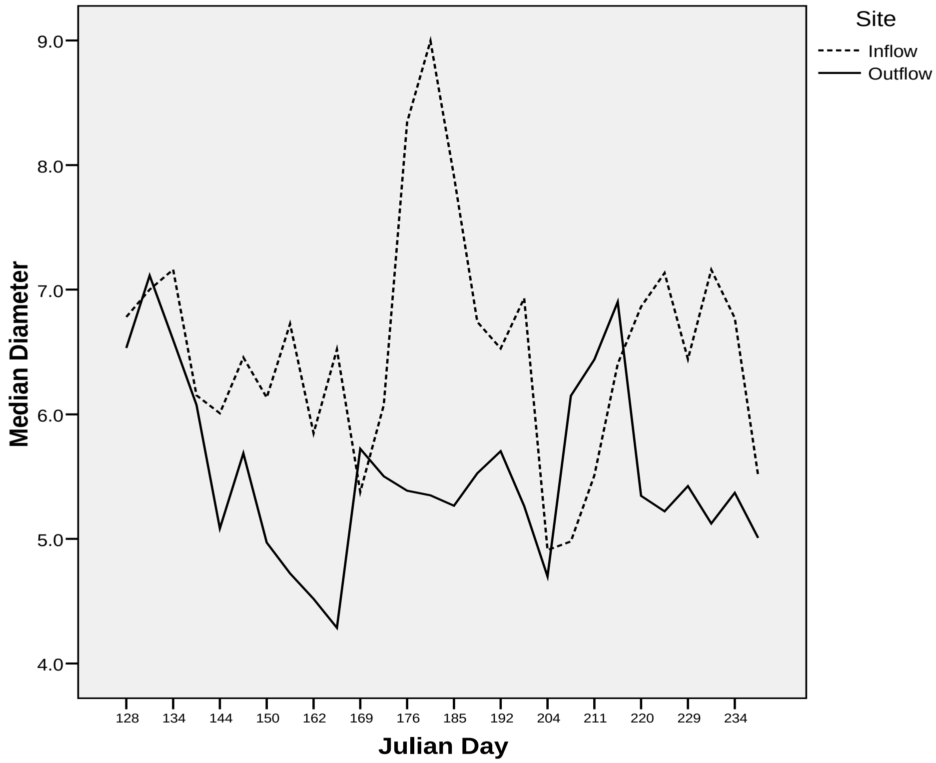

Figure 3-19 D_{50} at the inflow and outflow (May to August, 2007)

3.4.2.2 Grain Size Characteristics

The grain size data show that the SS at both study sites were predominated by fine-grained (< 63 μm) materials. For comparison, the grain size distributions of SS at the inflow and outflow of Columbia Lake are presented in Figure 3-20 for four representative days in 2007 (JD 144, 162, 178, 232). Nearly 100% of measured particles were < 63 μm and approximately 40 – 60% and 70 – 80% of the materials measured the inflow and outflow, respectively, were < 8 μm.

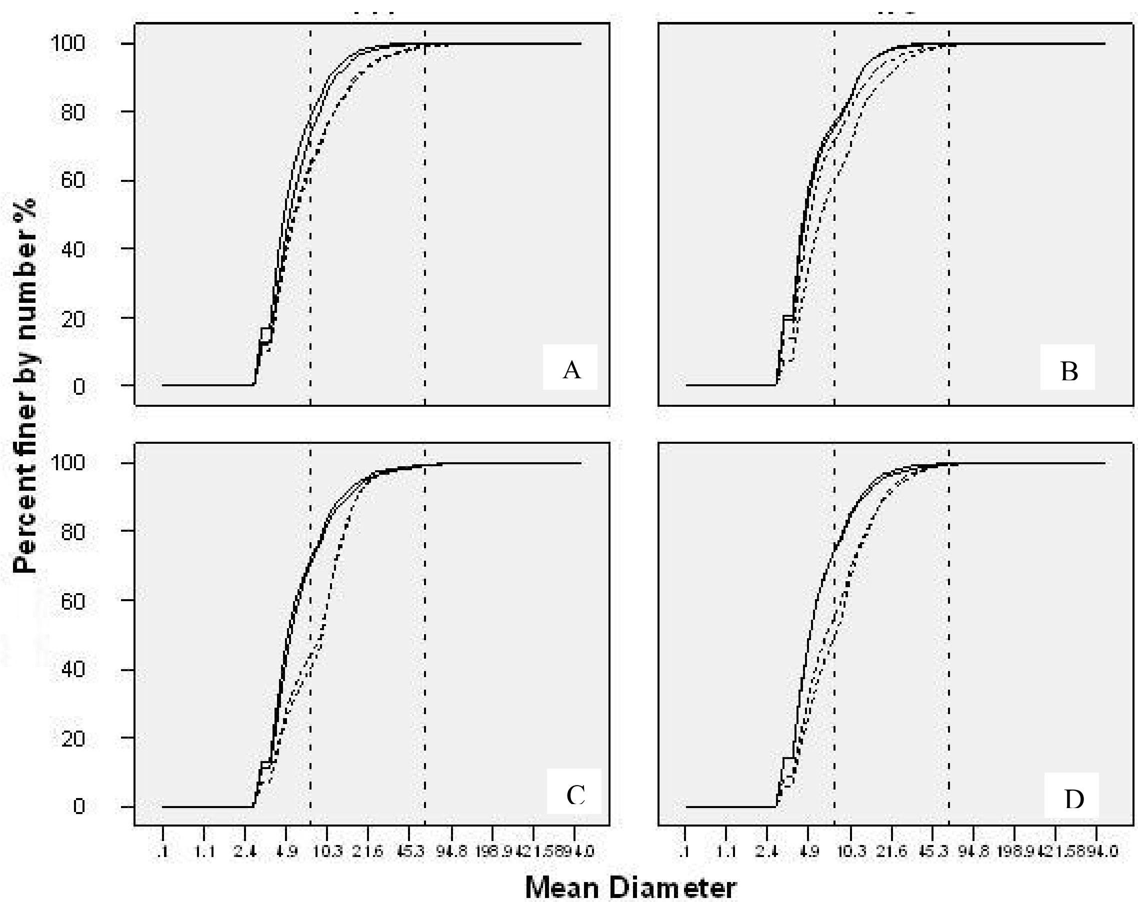

Figure 3-20 Grain size distributions at the inflow and outflow of Columbia Lake during JD 144, 162, 178, 232 (2007). ______ Outflow - - - - - Inflow

Representative photomicrographs of suspended solids measured at the inflow and outflow of Columbia Lake are shown in Figure 3-21. These images qualitatively show that inflow particles were larger and highly flocculated whereas particles in the outflow were more uniformly distributed in size and fewer large flocculated particles were present. Droppo and Ongley (1994) among other authors have shown that flocculation can change the grain size distribution and alter both the density and settling velocity of suspended particles in the water column. To demonstrate the extent of flocculation of SS in the Columbia Lake inflow, grain size data for JD 131 are presented in Figure 3-22 as both the primary (disaggregated) distribution and the actual (in situ) size distribution. Compared to the distribution of the outflow SS, the inflow sediment distribution is skewed to the right suggesting that the particles are highly flocculated at the inflow. At the outflow, about 12% of solids by volume were > 90 µm due to the existing flocs. Flocculated solids represent a small fraction of the total number of particles transported but constitute a large fraction of the SS total volume. The maximum size classes, which were 362.7 μm for the inflow and 94.8 μm for the outflow, accounted for 31.5% and 12.0% of the total volume at the two sites, respectively (Figure 3-22). Thus flocculation in Columbia Lake may significantly affect the sedimentation dynamics and concomitantly impact the sediment-associated chemical transfer in Columbia Lake.

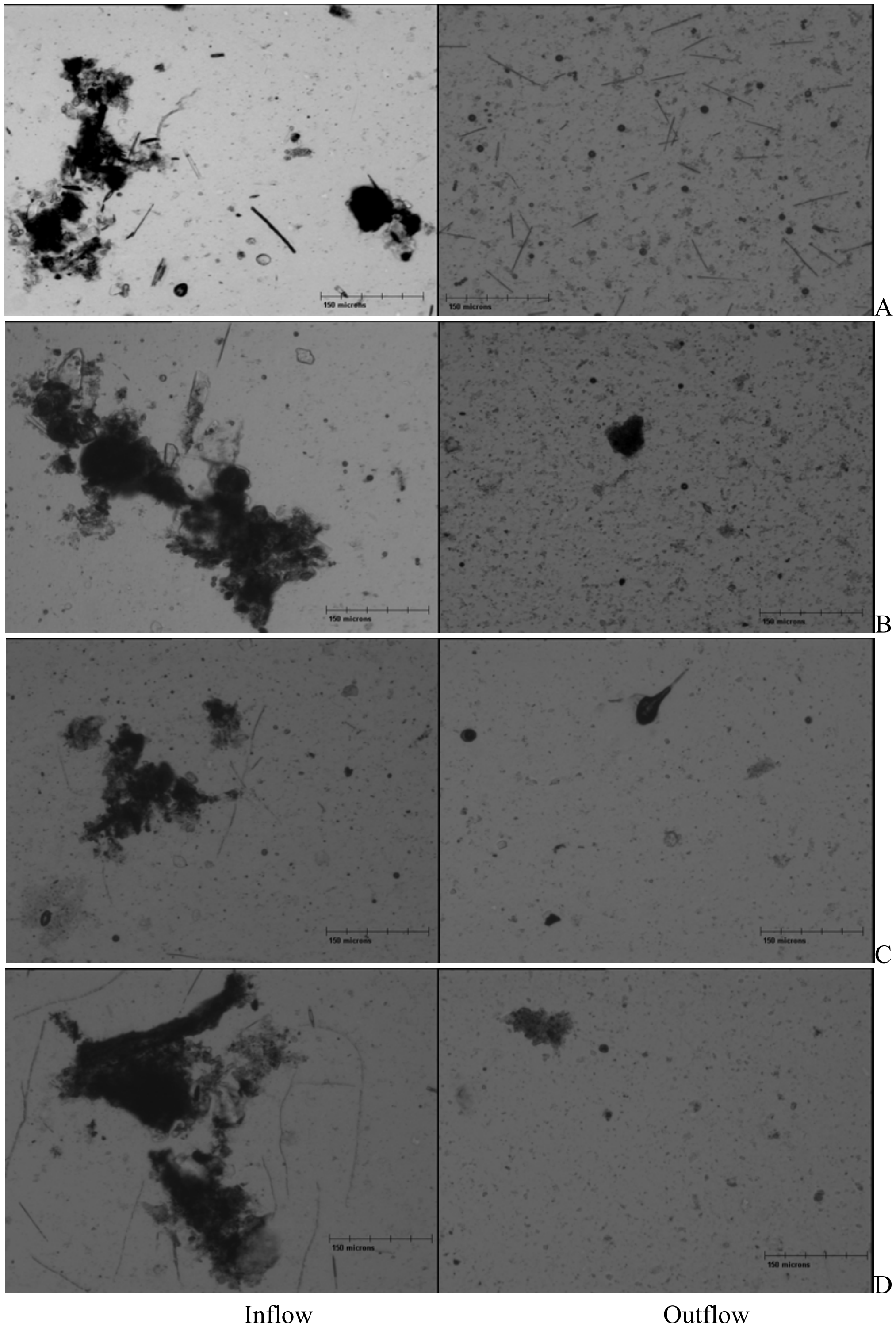

Figure 3-21 Representative photomicrographs of suspended solids at the inflow and outflow of Columbia Lake during JD 131, 157, 211, 220 (2007)

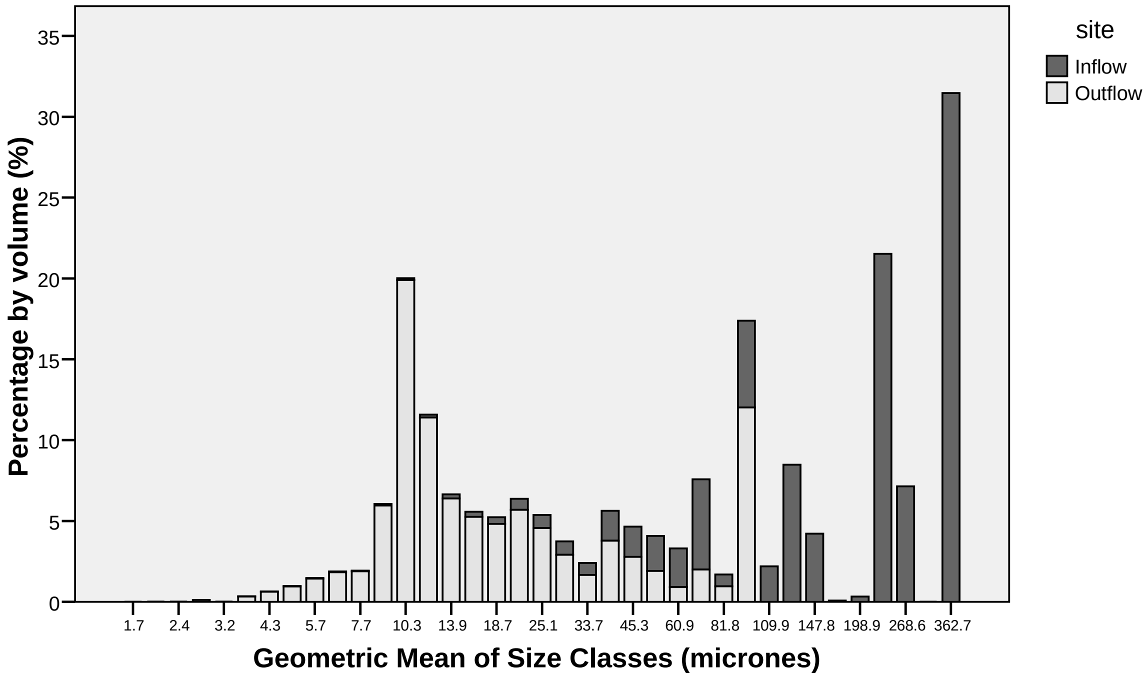

Figure 3-22 Primary size distribution of SS on Julian Day 131

3.5 Water Quality

3.5.1 Surface Water Temperature

Water temperatures were similar at both the inflow and outflow during the pre- and post-design periods (Figure 3-23). In the pre-design period, temperatures ranged from 11.7 to 26.3 °C at the inflow and from 9.7 to 28.9 °C at the outflow. In the post-design period, water temperature varied between 13.9 and 28.7 °C at the inflow and between 13.6 and 28.3 °C at the outflow (Appendix 1). The distribution of the data was not normal, but was skewed, with a median of 21.7 °C. The temperatures in 2004 and 2006 were higher than in 2003 and 2007. However, the overall trend of temperature change was similar among the study period in which temperature increased gradually from early May, then was more constant through June and July and then decreased in late August.

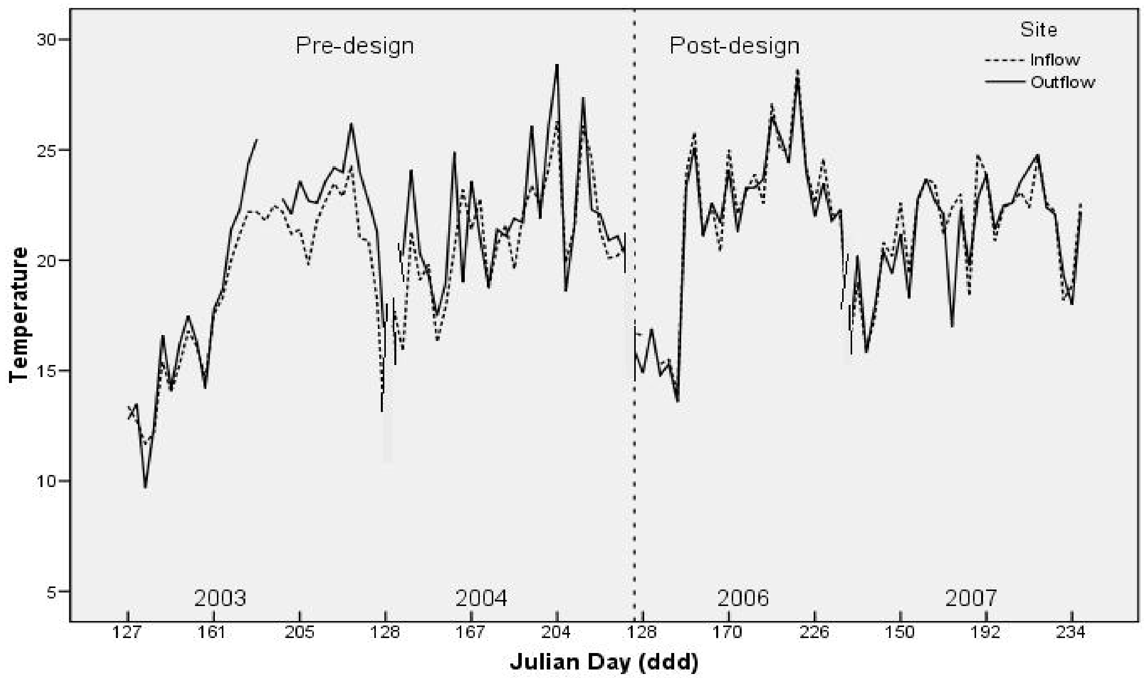

Figure 3-23 Water temperature at the inflow and outflow for the pre- and post-design periods

3.5.2 Dissolved Oxygen (DO)

The dissolved oxygen (DO) concentrations varied widely during the study periods. In the pre-design period, DO concentrations varied between 1.62 and 12.90 mg L^{-1} at the inflow and between 1.51 and 10.64 mg L^{-1} at the outflow. In the post-design period, DO concentrations ranged from 3.87 to 11.50 mg L^{-1} at the inflow and from 3.19 to 10.40 mg L^{-1} at the outflow, respectively (Appendix 1). Data on DO concentrations were not normally distributed according to a Shapiro-Wilk Test.

Figure 3-24 shows the variation in DO concentrations at the inflow and outflow of Columbia Lake for the pre- and post-design periods. The City of Waterloo (2004) reported that DO concentrations lower than 5.0 mg L^{-1} are harmful to several freshwater fish. Six out of 33 samples in 2003 and 4 out of 28 observations in 2007 had outflow DO concentrations below the water quality target of 5.0 mg L^{-1}. All of the DO concentrations were above the city's requirement in 2004 and 2006. Outflow DO concentrations were erratic in 2003. In the post-design period, there was an overall downward trend in the DO concentrations from May to early August, increasing slightly in late August.

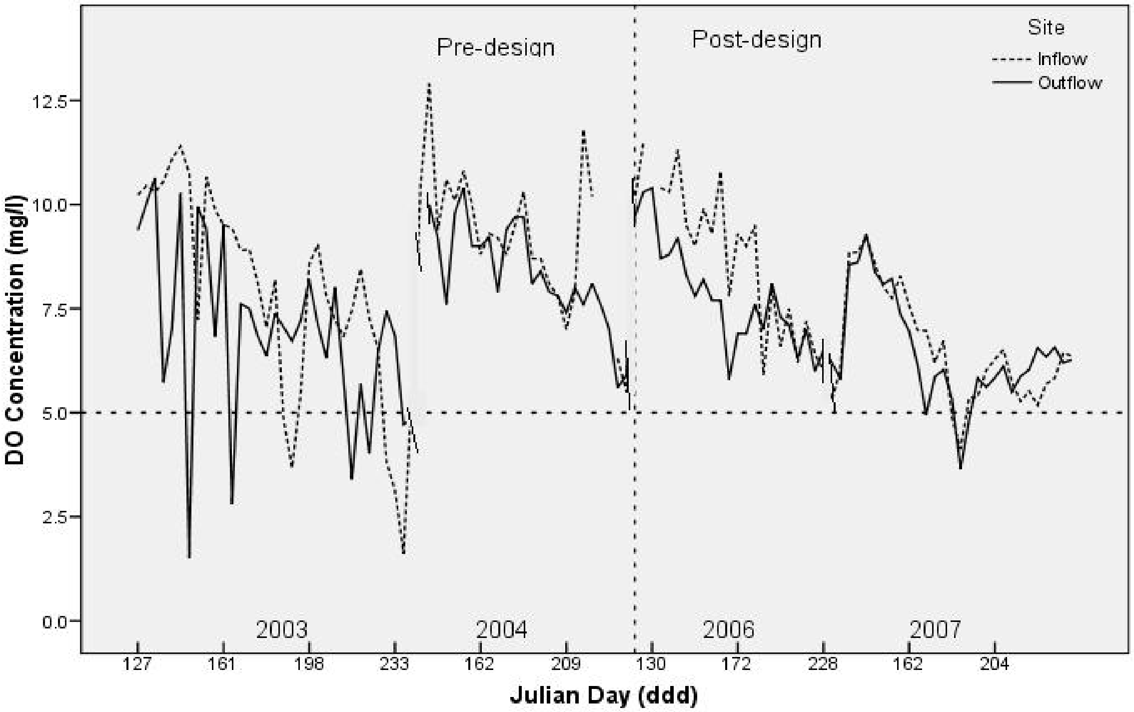

Figure 3-24 DO concentrations at the inflow and outflow for the pre- and post-design periods

During the first three months during the pre-design period, monthly averages of DO concentrations were higher than those for the post-design period at both study sites (Figure 3-25). In 2003 and 2004, the average monthly DO concentration gradually decreased in both sites from May through August. In the post-design period, the average monthly DO concentration declined from May through July and remained relatively constant in August.

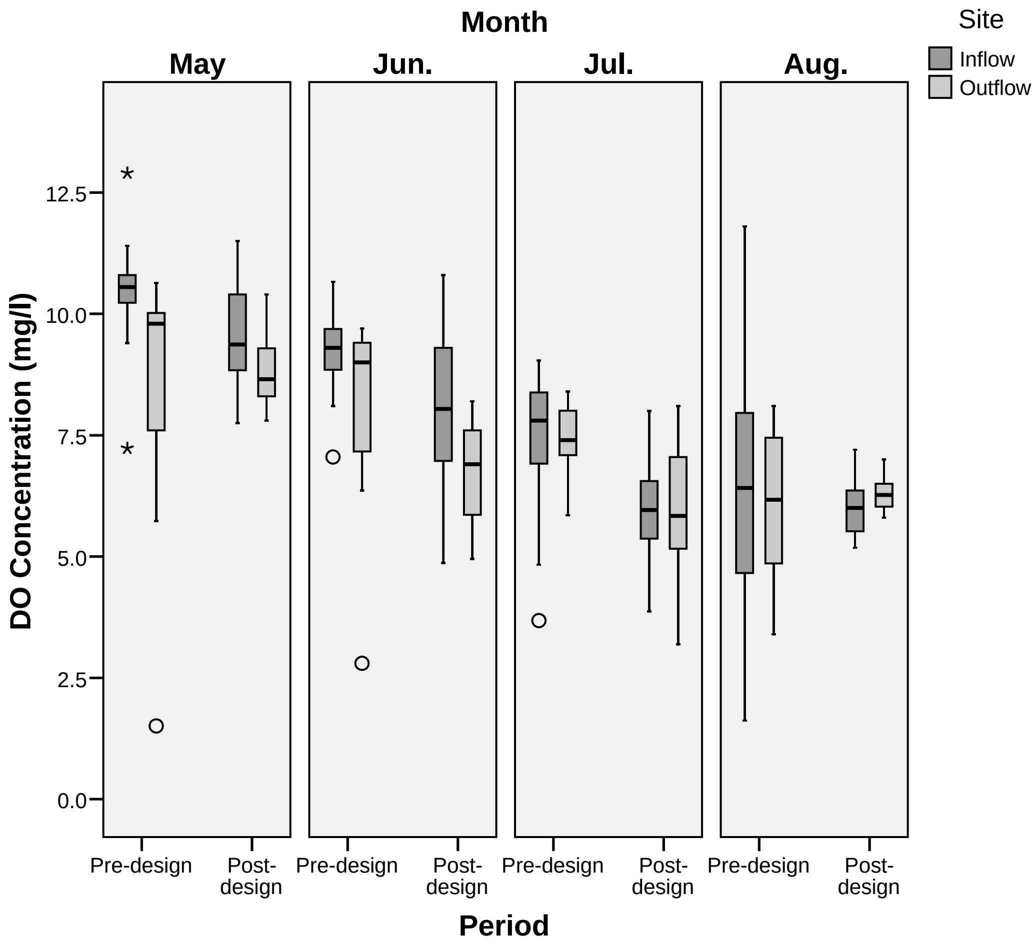

Figure 3-25 Monthly DO concentration changes at the inflow and outflow for the pre- and post-design periods (* = the extreme, ° = the outliner)

3.5.3 pH

During the pre-design period, pH ranged from 7.15 to 8.92 and from 7.52 to 8.37 at the inflow and outflow, respectively. During the post-design period, pH varied between 7.07 and 9.45 and between 7.10 and 8.91 at the inflow and outflow, respectively (Appendix 1). Data on pH values was not normally distributed according to a Shapiro-Wilk Test.

Figure 3-26 illustrates the temporal variability in pH for the study period. In May and June in 2006, inflow pH ranged from 8.50 to 9.50. The peak appeared on JD 170 in 2006, with the value of 9.45. According to Ontario provincial water quality objectives (PWQO), pH should be maintained in the range of 6.5-8.5, in order to protect aquatic habitats (OMOE, 1994). In the post-design period, however, 18 pH values out of 54 at the inflow and 15 values out of 54 at the outflow were above 8.5 (Figure 3-26).

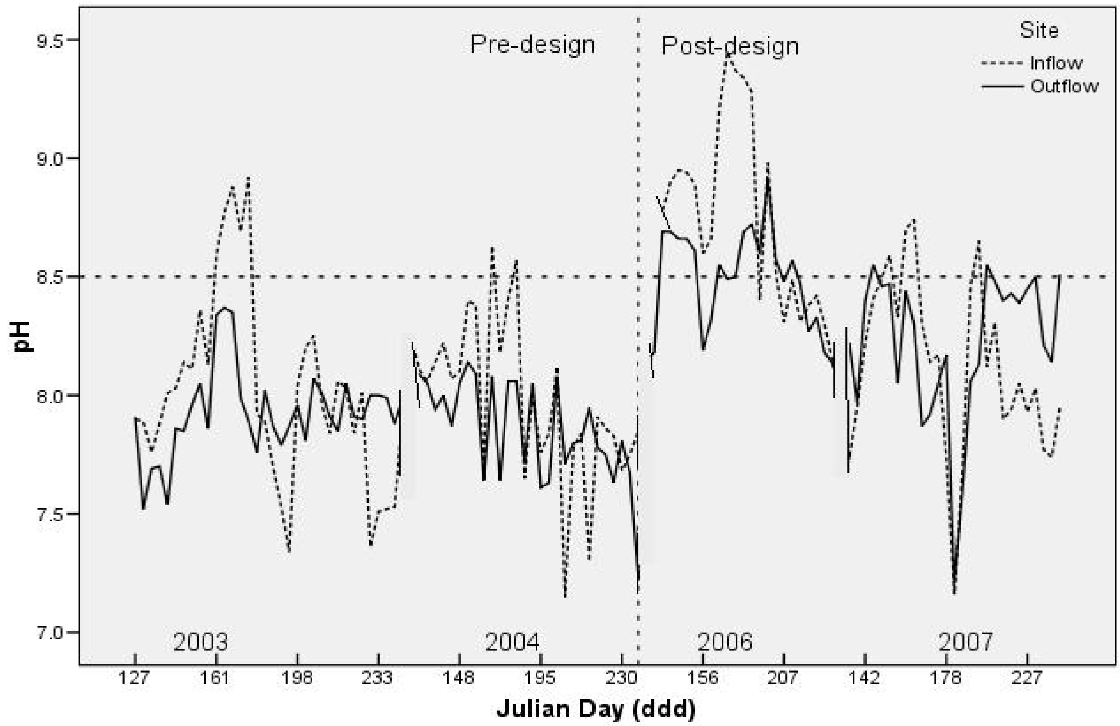

Figure 3-26 pH changes at the inflow and outflow for the pre- and post-design periods

Figure 3-27 provides the monthly changes in pH during the pre- and post-design periods. Compared with the pre-design period, pH fluctuated widely during the post-design period, and the averages were higher. The inflow pH increased from May to June, then declined during the following months. However, the averages and distributions of outflow pH in the post-design period were similar for each month. In terms of spatial variability, during the pre-design period, pH decreased after the flow passed through Columbia Lake in May and June, and then changed little in July and August. After the retrofit project, pH decreased in June and then increased in August after the water flowed through the lake.

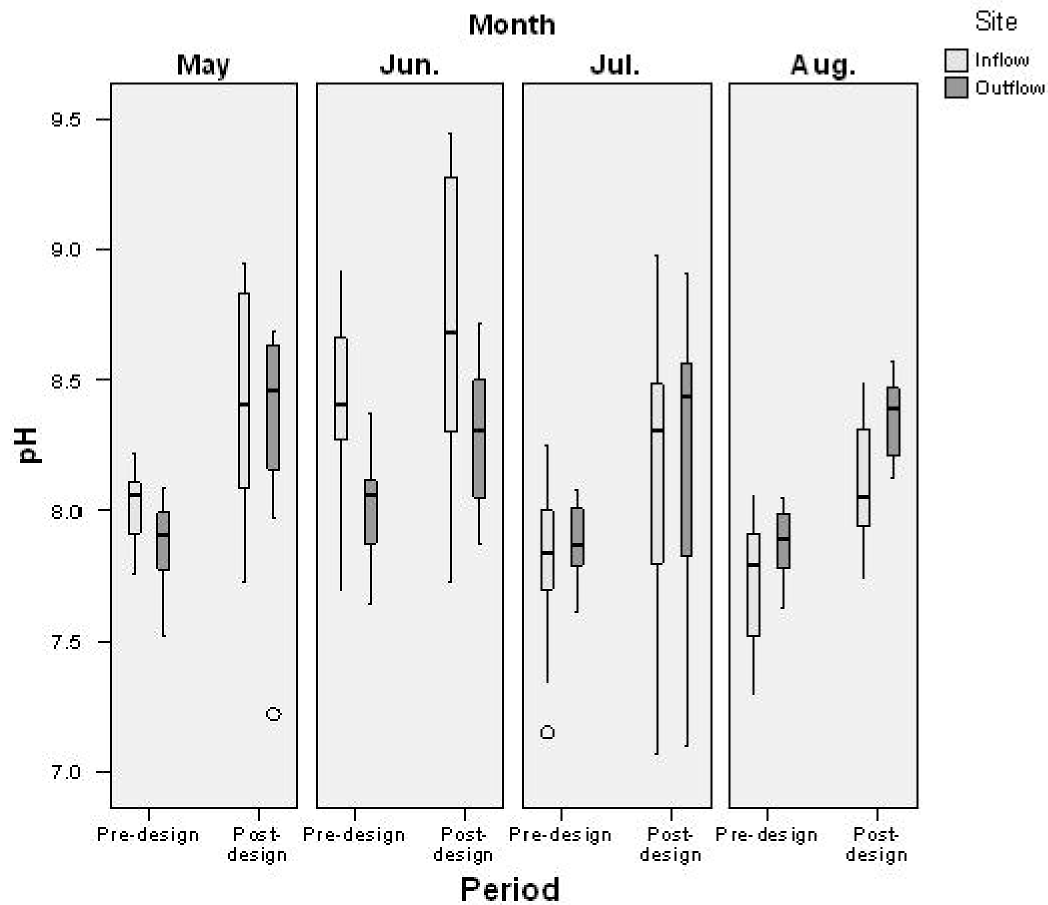

Figure 3-27 Monthly changes in surface water pH at the inflow and outflow during the pre- and post-design periods (° = the outliner)

3.5.4 Total Dissolved Solids (TDS)

3.5.4.1 Concentrations

Total dissolved solids (TDS) were monitored at the inflow and outflow of Columbia Lake from May to August for the years 2004 to 2007. During the pre-design period (2004), TDS concentrations at the inflow ranged from 189.0 to 391.0 mg L^{-1} and 189.0 to 426.0 mg L^{-1} at the outflow. During the post-design period (2006-2007), TDS concentrations varied between 207.7 and 404.5 mg L^{-1}, and between 231.5 and 482.9 mg L^{-1} at the inflow and outflow, respectively (Appendix 1). According to a Shapiro-Wink Test, TDS concentrations were not normally distributed.

In the post-design period, outflow TDS concentrations were consistently higher than those measured at the inflow (Figure 3-28). A maximum value of 482.9 mg L^{-1} occurred at the outflow on JD 185 (2007) while a minimum value of 189.0 mg L^{-1} was measured on JD 218 (2004).

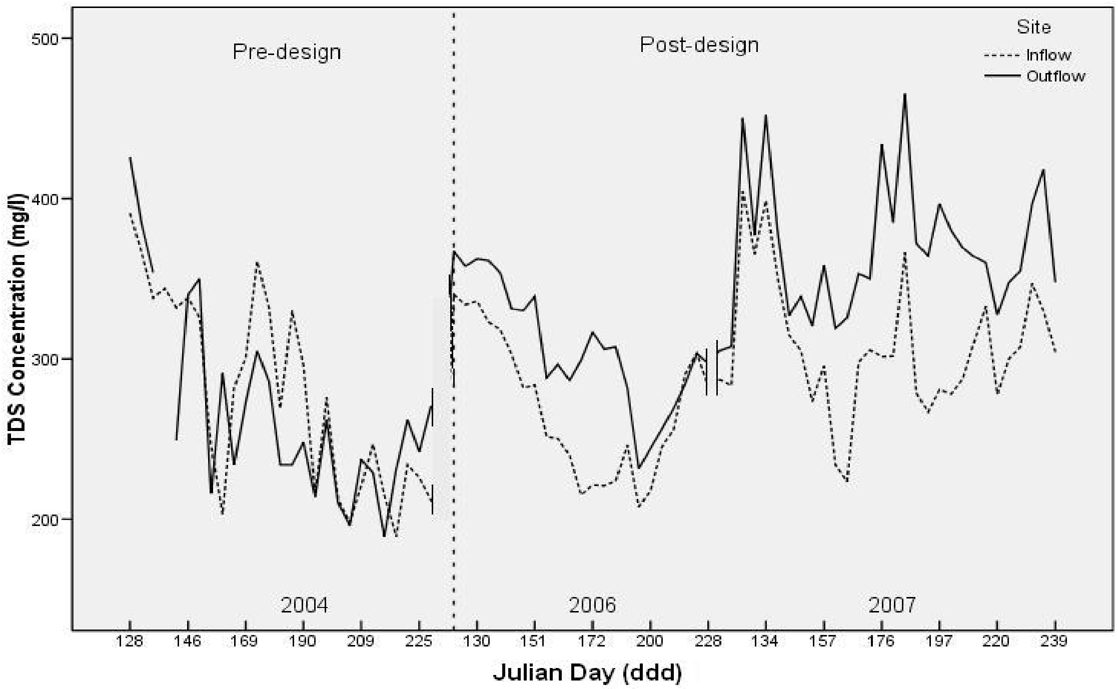

Figure 3-28 TDS concentrations at the inflow and outflow for the pre- and post-design periods

3.5.4.2 Hourly Loads

The hourly loads of TDS varied dramatically at the inflow and outflow of Columbia Lake for the pre- and post-design periods (Figure 3-29). During the pre-design period, the inflow TDS loads ranged from 86.2 to 2626.0 Kg h^{-1} while the outflow loads ranged from 66.4 to 2469.3 Kg h^{-1}. In the post-design period, the inflow loads varied from 8.4 to 464.5 Kg h^{-1} and the outflow loads ranged from 13.4 to 555.7 Kg h^{-1} (Appendix 1). The maximum values of 2626.0 and 2469.3 Kg h^{-1} were measured at the inflow and the outflow, respectively, on JD 146, 2004 during a storm event. The data of TDS hourly loads were not normally distributed and were skewed with a median of 116.8 K h^{-1}.

In the pre-design period, TDS internal loads were negative and Columbia Lake acted as a TDS sink (Figure 3-29). However, during the post-design period, Columbia Lake became a source of TDS. Except on JD 226 (2006) and JD 220 (2007) when the net internal loads were -9.0 and -42.8 Kg h^{-1}, respectively, the positive net internal loading for the remainder of the TDS measurements demonstrated that Columbia Lake was a TDS source after the redesign of Columbia Lake (Figure 3-29). The internal loadings in 2006 were higher compared to those in 2007. The TDS loads fluctuated between -20 and 180 Kg h^{-1} with the average of 42.5 Kg h^{-1} in 2006. In 2007, they ranged between - 40 and 80 Kg h^{-1}.

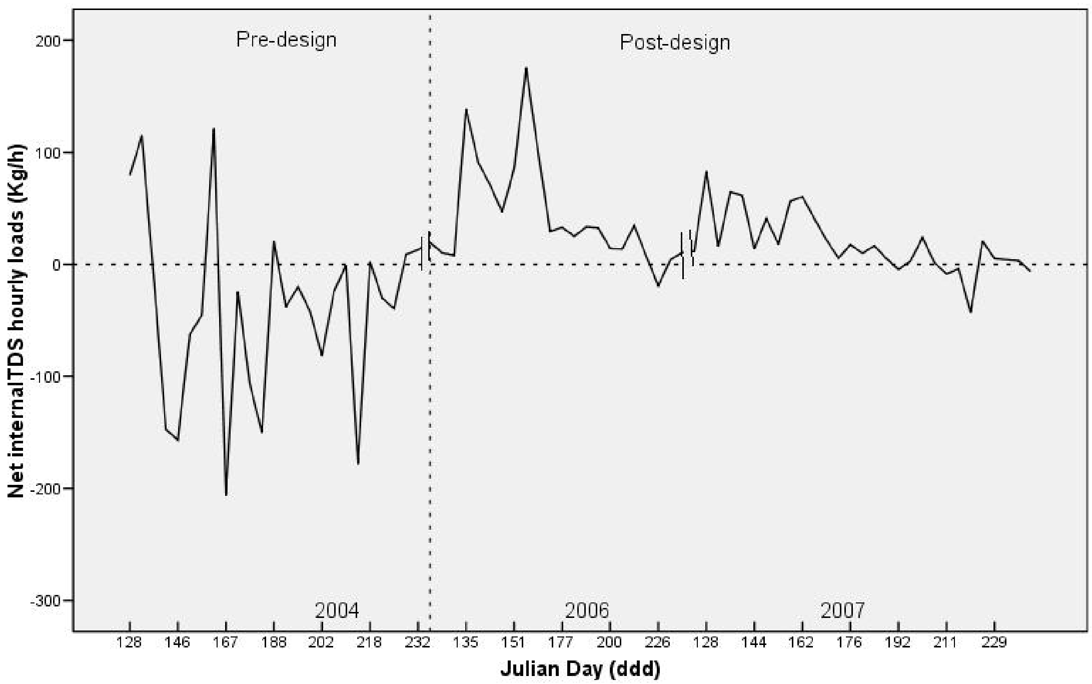

Figure 3-29 Net internal TDS hourly loads from Columbia Lake during the pre- and post-design periods

3.6 Modeling Calibration and Validation

Calibration and validation of the Stantec Model was conducted during 2006 and 2007. Pollutant decay rate (K_1) and pollutant settling rate (K_S) were estimated based on the model calibration from 1997 to 2003. TP was characterized as a conservative substance and K_1 and K_s equal to zero (Stantec Consulting Ltd, 2004). In terms of model calibration for outflow SS prediction, K_1 was zero and K_S varied from 3.3 to 12×10^{-7} s^{-1} in the former years (1997 - 2003)

(Stantec Consulting Ltd, 2004). Since average inflow SS concentrations in the post-design period significantly decreased, from 19.0 to 8.5 mg L^{-1}, the chances for particles to collide and flocculate significantly decreased (Van Buren et al., 1997). Therefore, more fine particles remained in suspension. The settling rate considerably declined. In such a case, a K_S value equal to 3.3×10^{-7} s^{-1} was selected for modeling outflow SS concentrations. Because the model calibration and validation were conducted to predict outflow concentrations in 2006 and 2007 following the reconstruction of Columbia Lake through March to October in 2005, the initial in-lake TP and SS concentrations were regarded as negligible, and given a zero value.

Comparisons of predicted outflow water parameters (TP and SS) to measured concentrations in Columbia Lake are presented in Figure 3-30 and Figure 3-31. Columbia Lake Water Quality Model underestimated the outflow TP and SS concentrations: most of the predicted values were lower than the measured values. However, the overall pattern of the model prediction was similar to direct measurements (Figure 3-30). Compared to the predicted TP and TSS concentrations from May to August in 1998 when the model predictions were more than 50 µg L^{-1} and 50 mg L^{-1} higher than the measurements in several sampling days, respectively (Stantec Consulting Ltd, 2004), most of the predicted TP and SS concentrations were no more than 20 µg L^{-1} and about 10 mg L^{-1} lower than the measured TP and SS concentrations, respectively (Figure 3-30 and 3-31). However, on JD 158 (2006) and JD 220 (2007), the predicted SS concentrations were about 5 mg L^{-1} higher than the measured concentrations.

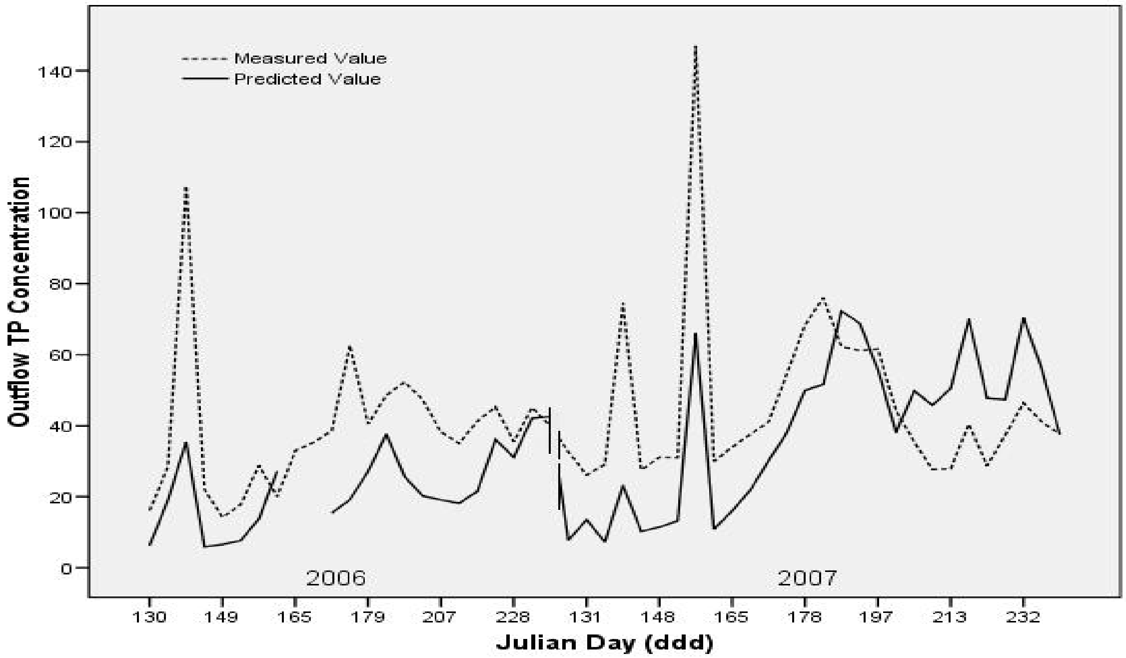

Figure 3-30 The comparison between predicted and measured outflow TP concentrations

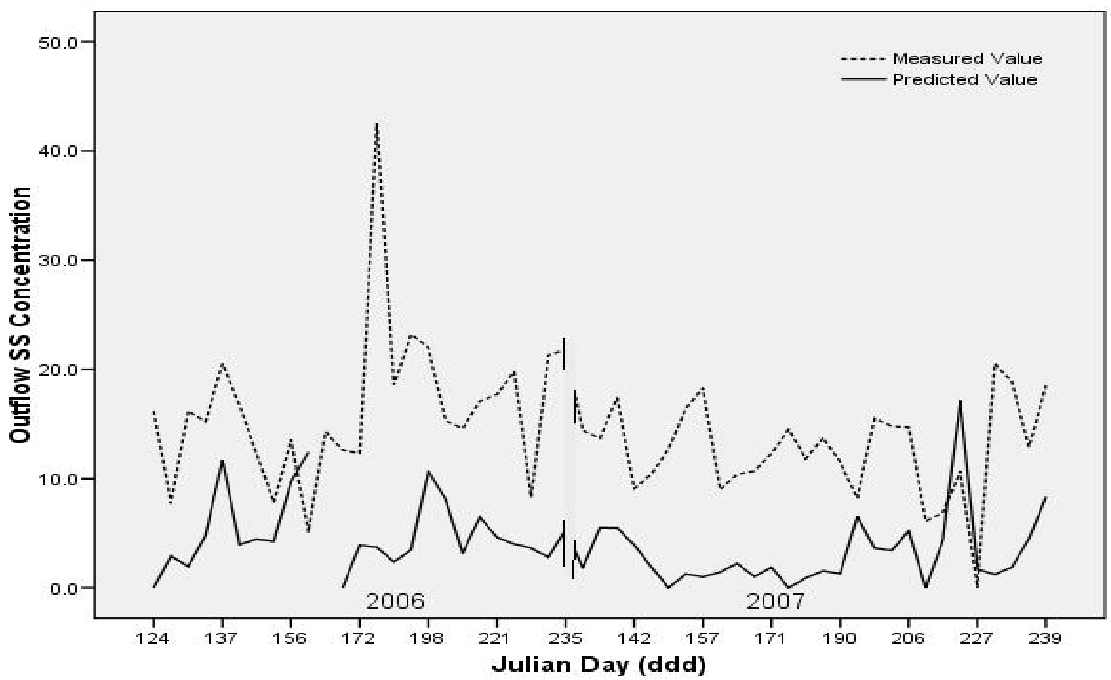

Figure 3-31 The comparison between predicted and measured outflow SS concentrations

The Stantec Water Quality Model provided reasonable long-term prediction of SS and TP in Columbia Lake (Table 3-3). Based on the investigation of the inflow and outflow TP and SS levels of Columbia Lake, the model predicted the SRP / TP ratio about 20%, which is close to the direct measurement of 22% on average (Table 3-3). The measured outflow TP and SS concentrations during the post-design period ranged more widely and were higher than the predicted ones. Table 3-3 contains a detailed comparison between model predictions and measured values:

Table 3-3 The comparisons between model predictions and measured condition

Indicator	Predicted condition	Measured condition
TSS removal via Columbia Lake	70%	The average TSS loads changed from 3.9 Kg h^{-1} at the inflow to 6.2 Kg h^{-1} at the outflow, internal loading rate was 59%.
SRP / TP ratio	20%	22%
Inflow TP removal by sedimentation	56%	The average TP loads changed from 14.1 to 17.6 g h^{-1}, internal loading rate was about 25%.
Outflow TP concentrations after the retrofit project	Change from 85 – 171 μg L^{-1} in the pre-design period to 20 – 36 μg L^{-1} in the post-design period	14 – 147 μg L^{-1} during the post-design period
Outflow TSS concentrations after the retrofit project	Change from 38 – 67 mg L^{-1} in the pre-design period to 3– 13 mg L^{-1} in the post-design period	< 0.1 – 43 mg L^{-1} in the post-design period
Reduction in TP and SS flux between pre- and post-design conditions	75 – 90% reduction in TP and SS inputs downstream reaches of Columbia Lake	Average outflow SS hourly loads decreased from 52.16 Kg h^{-1} in the pre-design period to 6.23 Kg h^{-1} in the post-design period, taking about 88% of sediment flux reduction. Outflow TP flux changed from 82.15 to 17.55 g h^{-1}, accounting for about 79% of TP reduction.

Chapter 4 DISCUSSION

4.1 Introduction

To measure the effectiveness of the Columbia Lake redesign to enhance water quality in the downstream reaches of Laurel Creek, this thesis compared TP and SS concentrations and loads at the inflow and outflow of Columbia Lake during the post-design period to those during the pre-design period. The performance of Columbia Lake, as characterized by its internal TP and SS loading rates, were compared and discussed in the context of relevant literature. The utility of the Columbia Lake Water Quality Model proposed by Stantec Consulting Ltd. was discussed.

4.2 TP Concentrations and Loads during the Pre- and Post-design Periods

Previous studies have indicated that P and SS concentrations and loads changed after external loads were decreased. Istvanovics and Somlyody (1999) detailed the temporal change in outflow P concentrations and loads following the reduction in external loads to the Upper Kis-Balaton Reservoir (UKB) in Hungary, due to improved sewage treatment. Inflow SRP and TP concentrations in the UKB decreased from 322 to 118 μg L^{-1} and from 562 to 290 μg L^{-1}, respectively, while they changed from 47 to 15 μg L^{-1}, from 243 to 224 μg L^{-1} at the outflow (Istvanovics and Somlyody, 1999). In Lake Apopka, Florida, Coveney et al. (2005) found that in-lake TP concentrations decreased from 0.23 to 0.11 mg L^{-1} following a continued decrease in P external loads from 0.56 g m^{-2} $year^{-1}$ to under-detectable during 11 years.

In Columbia Lake, the TP loads decreased by an average of 30 g h^{-1} at the inflow and the decrease was attributed to a reduction in inflow velocities and associated decrease in inflow discharge after the lake was redesigned (Stantec Consulting Ltd., 2004). Statistical analysis indicated that discharge during the post-design period was significantly lower than during the pre-design period ($p = 0.0001$). Particularly, discharge in 2007 was considerably lower than in 2006 ($p = 0.0001$) because 2007 was an extremely dry year and the monthly precipitation for each month was dramatically lower than the 30 year average (Table 3-1). However, in 2006, total precipitation in each month was similar to those in 2003 and 2004. Both the redesign and weather conditions controlled the level of discharge. The average outflow concentration during the post-design period was approximately 40% of the pre-design average concentration. The TP output from Columbia Lake diminished by 64 g h^{-1} following the reconstruction, which can be attributed to the decrease in both external and internal TP loads. Internal loads shifted from 39 g h^{-1} to only

4 g h^{-1} on average, which is only about one tenth of the pre-design internal loads. Table 4-1 indicated the significant differences of TP concentrations / loads between inflow and outflow, and the pre- and post-design periods.

Table 4-1 Statistical comparisons (Kruskal Wallis Tests, p = 0.0001) of TP concentrations loads at the inflow and outflow during the pre- and post-design periods (Appendixes 2 and 3)

	TP concentrations	TP loads
Inflow between the pre- and post-design periods	P = 0.965	**P = 0.0001**
Outflow between the pre- and post-design periods	**P = 0.0001**	**P = 0.0001**
Pre-design between the inflow and outflow	**P = 0.0001**	**P = 0.0001**
Post-design between the inflow and outflow	P = 0.291	P = 0.823

* P values in bold mean statistically significant differences

Outflow TP loads are a function of inflow loads and net internal TP loads. Istvanovics and Somlyody (1999) investigated the P cycle in the UKB Reservoir in Hungary. Their findings indicated a shift of $L_{OUT} = f(L_{IN})$ after the high external loads were decreased by one half in 1991, due to improved sewage treatment. A similar shift of the L_{OUT} vs. L_{IN} curve was observed in the Columbia Lake study after the lake was redesigned. The curve altered from $ln(L_{OUT}) = 0.423\, ln(L_{IN}) + 2.756$ ($R^2 = 0.305$, p = 0.0001, 2-tailed at 0.01 significant level) to $ln(L_{OUT}) = 1.004\, ln(L_{IN}) + 0.013$ ($R^2 = 0.630$, p = 0.0001, 2-tailed at 0.01 significant level) (Figure 4-1). Both of the decrease in external loads and the lake redesign caused the decreased TP output. According to the pre-design and post-design curves, Columbia Lake redesign considerably prevented internal TP loading and eventually leaded to a small amount of TP output (Figure 4-1).

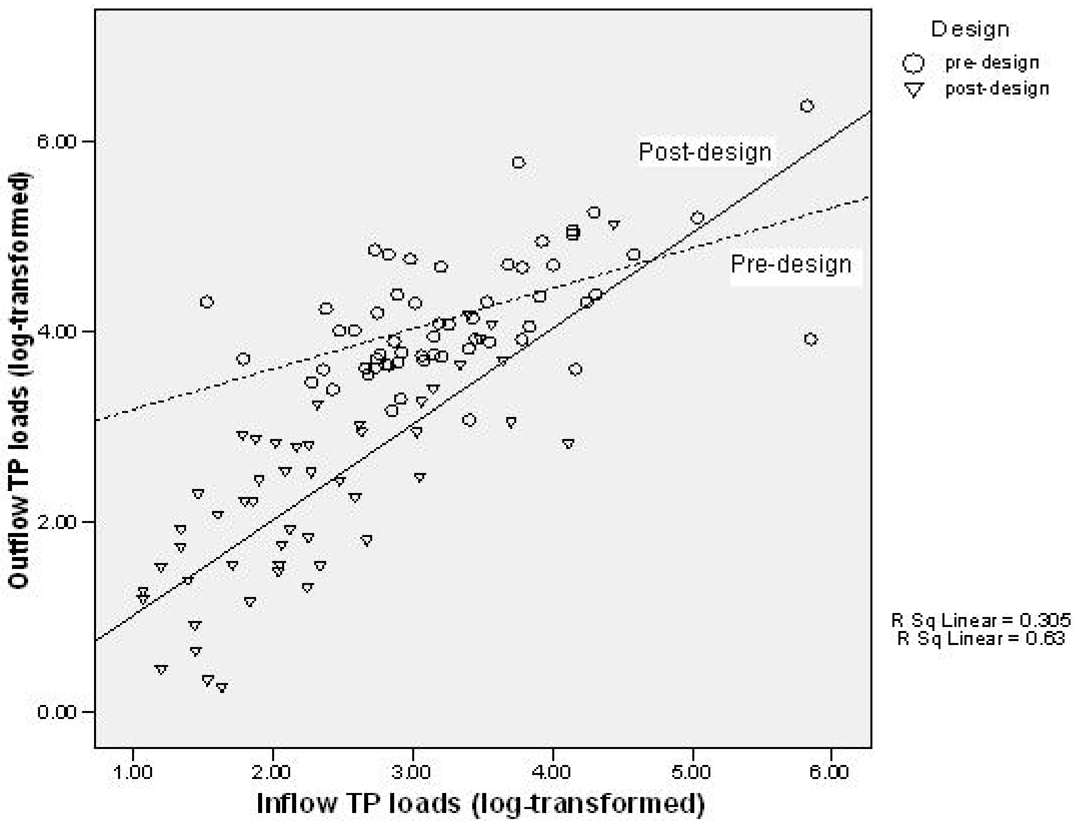

Figure 4-1 Relation between inflow TP loads and outflow TP loads (log-transformed) during the pre- and post-design periods

Several studies reported that discharge pattern dominated P transfer through impoundments. Wu (1996) showed a correlation between water quality and flow rate by means of the geometrical regression method: (TP loads) = $1.57 \times 10^{-3}Q^{1.68}$, with $r = 0.78$. Istvanovics and Somlyody (1999) found a strong correlation between non-bioavailable P (nBAP) and flow as well as SS loads. Alaoui Mhamdi et al. (2007) indicated that about 80% of P was in the particulate form and the TP output strongly correlated to outflow discharge ($r > 0.90$, $P < 0.001$). These findings were similar to the present research in Columbia Lake. The overall trend of TP hourly loads was similar to the discharge trend, and the maximum TP hourly loads resulted from high discharge during a storm on JD 146 (2004) (Figures 3-1 and 3-5). Pre-design TP hourly loads (log-transformed) correlated strongly to the pre-design discharge (log-transformed) at both the inflow and outflow ($r = 0.799$ and 0.793, $p = 0.001$, 2-tailed at 0.01 significant level). Strong correlations during the post-design period were also observed ($r = 0.758$ and 0.911, $p = 0.001$, 2-tailed at 0.01 significant level). TP loads were strongly correlated to SS loads as well. The significant correlations between *ln*(TP hourly loads) and *ln*(SS hourly loads) at the inflow and outflow in the pre-design period ($r = 0.661$ and 0.777, $p = 0.0001$, 2-tailed at 0.01 significant level) suggest that the sediment erosion and deposition process were one of the important mechanisms governing the TP transfer in Columbia Lake (Alaoui Mhamdi et al., 2007; Alaoui-Mhamdi et al., 1996). Enhanced post-design correlations between TP loads and SS loads (log-transformed) at the inflow and outflow ($r = 0.794$ and 0.915, $p = 0.0001$, 2-tailed at 0.01 significant level) suggest the increased impact of SS on TP loads. The above correlations suggested that physical processes (sedimentation / resuspension) dominated TP transfer through Columbia Lake (Van Buren, 1997; Alaoui-Mhamdi, 1996; Alaoui Mhamdi et al., 2007; Teodoru and Wehrli, 2005; Istvanovics and Somlyody, 1999; James and Berko, 1997).

SRP concentrations / loads, which accounted for 20% of TP on average, changed slightly between the inflow and outflow during the study period in 2007. SRP concentrations ranged from 2 to 17 $\mu g\ L^{-1}$ and 5 to 19 $\mu g\ L^{-1}$, with averages of 10 ± 1 and 9 ± 1 $\mu g\ L^{-1}$, for the inflow and outflow, respectively. The average SRP loads were 2.2 ± 0.5 and 2.1 ± 0.4 $g\ h^{-1}$. Kruskal Wallis Tests demonstrated that there were no significant differences between inflow and outflow SRP concentrations, and inflow and outflow SRP loads ($p = 0.266$ and 0.668, Appendix 7). Previous studies demonstrated that SRP release is mainly dependant on equilibrium conditions in a water column, which is determined by sediment geochemistry, grain size and organic and metal-oxyhydroxide coatings (Sondergaard et al., 1992; Haggard and Soerens, 2006). Decreased DO concentrations and increased pH will accelerate SRP release from bottom sediments (Jensen and Anderson, 1992; Haggard and Soerens, 2006; Penn et al., 2000; Christophoridis and Fytianos, 2006). Laboratory studies on sediment P release in aerobic and anaerobic conditions indicated that SRP buffering concentrations ranged from 50 to 200 $\mu g\ L^{-1}$ (Haggard and Soerens, 2006). Results from a water quality monitoring of 54 UK river sites across seven major lowland catchment systems found that SRP released from bottom sediments when surface water SRP concentrations were less than 50 $\mu g\ L^{-1}$ (Javie et al., 2006). The coarse substrate at the reconstructed Columbia Lake would likely minimize the potential for P release from sediments. Hence, the SRP equilibrium concentrations in Columbia Lake are likely to be much lower than reported in previous studies. There were no significant differences between inflow and outflow DO concentrations ($p = 0.793$, Appendix 4), and between inflow and outflow pH ($p = 0.208$,

Appendix 4), leading to the similarities between inflow and outflow SRP concentrations / loads. These findings suggested that SRP concentrations were probably in the range of equilibrium concentrations and SRP loads changed slightly after flow passed through the lake.

Although no significant differences were observed in SRP concentrations/loads between inflow and outflow, monthly changes in SRP/TP were observed (Appendix 5). The monthly changes were probably caused by the changes in discharge and associated PP loads. The ratio of SRP/TP gradually increased in Columbia Lake from about 10% on JD 178 to approximately 50% in late August in 2007 (Figure 3-12). Since TP is a combination of SRP and PP, the temporal increase in SRP/TP ratio resulted from the gradually increased SRP and decreased PP concentrations. The overall trend of discharge, with monthly averages of $Q_{May} > Q_{Jun.} = Q_{Aug.} > Q_{Jul}$, was similar to the monthly changes in PP loads. Therefore, the month-by-month changes in PP internal loads are related most likely to changes in discharge. Multiple comparisons indicated that in 2007, the discharge in May was significantly higher than that in August (Appendix 6). A higher discharge accelerated sediment and associated P resuspension. The SRP concentrations increased from 5 μg L^{-1} in early May to 20 μg L^{-1} in late August 2007, at both the inflow and outflow, with the change of monthly averages of $[SRP]_{Aug} > [SRP]_{Jul} > [SRP]_{May} > [SRP]_{Jun}$ at the inflow and outflow (Figure 3-10). DO concentrations decreased from 10 to 3.5 mg L^{-1}, with changes in monthly averages of $[DO]_{May} > [DO]_{Jun} > [DO]_{Aug} \approx [DO]_{Jul}$ at the inflow, which was roughly concurrent with the gradual increase of SRP concentrations (Figure 3-25). A negative correlation between SRP and DO concentrations was observed at the inflow, indicating that SRP increased with the decrease in DO concentrations ($r = -0.470$, $p = 0.012$, 2-tailed at 0.05 significant level). This correlation indicated that SRP concentrations changed oppositely with the DO concentrations, which is agreed by previous literatures (Haggard and Soerens, 2006; Penn et al., 2000; Christophoridis and Fytianos, 2006, Reddy et al., 1999; Perkins and Underwood, 2001). Moreover, the outflow SRP/TP ratios from JD 178 to the late August were consistently higher than the inflow (Figure 3-12). Similar findings by Teodoru and Wehri (2005) indicated that PO_4^{3-} and TP concentrations changed from 32 μg L^{-1} and 80 μg L^{-1} at the inflow to 51 μg L^{-1} and 85 μg L^{-1} at the outflow in the Iron Gate I Reservoir on the Danube River. The study indicates that the loss of PP and an even larger increase in DP (increasing SRP/TP ratio) are the most significant processes that control the P flux balance, which was partly a consequence of nutrient remobilization from older sediments (Teodoru and Wehri, 2005).

Monthly differences also existed for TP concentrations and loads in Columbia Lake. During the pre-design period, significant differences were only observed for outflow TP concentrations in May and July, and in May and August (Appendix 7). Figure 3-5 showed that the May average was about 50 μg L^{-1} lower than the July and August averages. The average outflow TP concentrations gradually increased from month to month. The monthly average internal loads gradually decreased from 64 g h^{-1} to 4 g h^{-1}. However, no significant differences were observed (Appendix 8). Prior to the lake redesign, Columbia Lake consistently acted as a TP source. After the lake rehabilitation, significant differences were observed in the inflow TP concentrations in May and July, and in May and August (Appendix 9). Monthly average TP internal loads consistently decreased from 15 to -6 g h^{-1}. Significant difference in TP loads was observed between May and August (Appendix 8). Columbia Lake shifted from being a TP source in May and June to being a TP sink in August in 2006 and 2007.

4.3 SS Concentrations and Loads during the Pre- and Post-design Periods

Many previous studies have indicated that, due to the sedimentation / resuspension process, SS concentrations and loads may be influenced as they move through impoundments. James et al. (2004) indicated that the establishment of submersed aquatic macrophytes and a resulting mitigation of sediment resuspension in Peoria Lake, Illinois (U.S.A.) decreased SS loads from 9.84×10^4 Kg h^{-1} to 6.83×10^4 Kg h^{-1}. Furthermore, a study in the largest impoundment on the Danube River by Teodoru and Wehrli (2005) reported a reduction of SS concentrations and loads from 34 mg L^{-1} and 10.1×10^5 Kg h^{-1} at the inflow to 17 mg L^{-1} and 4.4×10^5 Kg h^{-1} at the outflow, respectively. Historical changes in SS transfer via an impoundment were also reported. After a reduction in SS loads and the regulation of flow direction in the Kis-Balaton Reservoir in Hungary, SS concentrations changed from 59 to 44 mg L^{-1} at the inflow, and from 27 to 29 mg L^{-1} at the outflow (Istvanovics and Somlyody, 1999).

In the Columbia Lake study, SS concentrations and loads changed considerably after the lake reconstruction (Figure 3-13). Before the lake reconstruction, Columbia Lake acted as a SS source to downstream reaches of Laurel Creek. Pre-design SS loads increased by more than 30 Kg h^{-1} after the flow passed through the lake on most of the sampling days (Figure 3-16). Outflow SS concentrations and loads decreased by 78% and 88% during the post-design period (Figure 3-13 and 3-15). Moreover, average net internal TP loads decreased to only 2 Kg h^{-1}. Although

Columbia Lake consistently acted as a SS source during the entire study period, the outflow concentrations and loads significantly decreased following the lake redesign (p = 0.0001). Table 4-2 shows the statistical comparisons between inflow and outflow SS concentrations and loads during the pre- and post-design periods (Appendix 10 and 11).

Table 4-2 Statistical comparisons (Kruskal Wallis Tests, p = 0.0001) of SS concentrations and loads at the inflow and outflow during the pre- and post-design periods

	SS concentrations	SS loads
Inflow between the pre- and post-design periods	**P = 0.0001**	**P = 0.0001**
Outflow between the pre- and post-design periods	**P = 0.0001**	**P = 0.0001**
Pre-design between the inflow and outflow	**P = 0.0001**	**P = 0.0001**
Post-design between the inflow and outflow	P = **0.0001**	**P = 0.004**

* P values in bold mean statistically significant difference

Input-output function indicated that the in-lake techniques played an important role on mitigating the outflow SS loads. Outflow SS loads (L_{OUT}) are a combination of inflow SS loads (L_{IN}) and net internal loads. After the lake was redesigned, the L_{OUT} vs. L_{IN} curve shifted from $ln(L_{OUT}) = 0.33ln(L_{IN}) + 2.84$ (r = 0.487, p = 0.001, 2-tailed at 0.01 significant level) to $ln(L_{OUT}) = 0.69ln(L_{IN}) + 0.86$ (r = 0.782, p = 0.001, 2-tailed at 0.01 significant level) (Figure 4-2). The pre-design and post-design curves indicated that, sediment resuspension was dramatically mitigated, which eventually caused the decreased SS outflow, due to the rehabilitation.

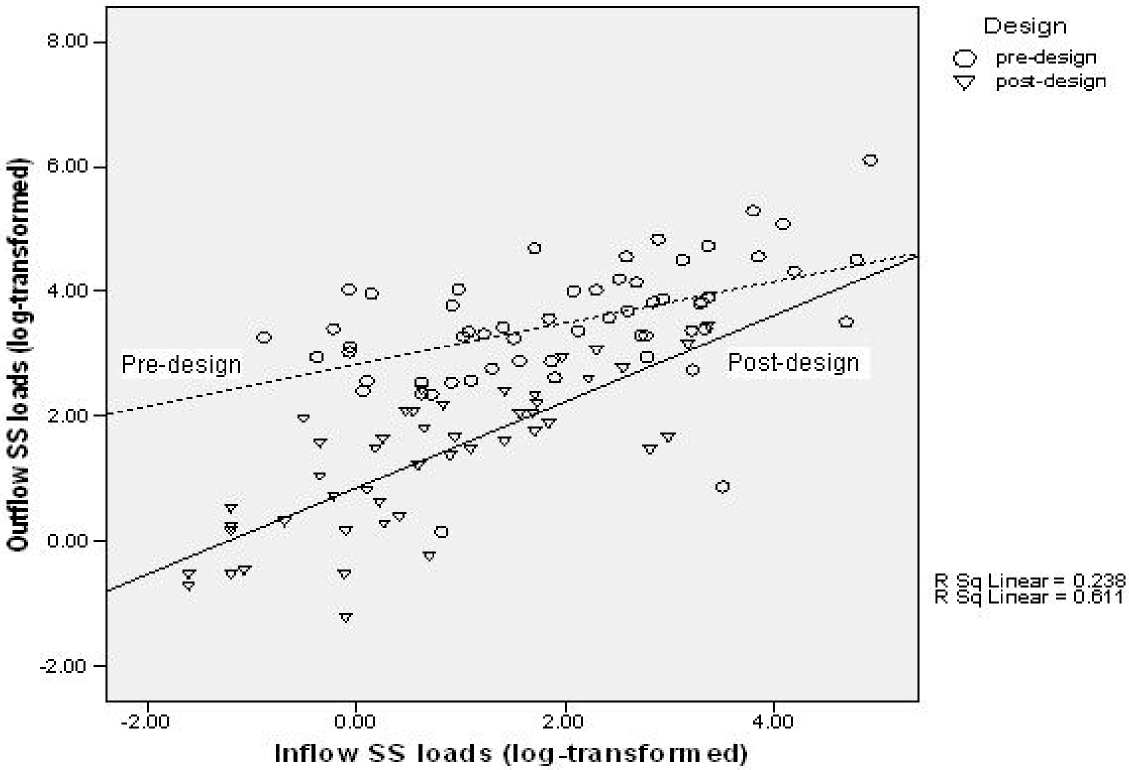

Figure 4-2 Relation between inflow and outflow SS loads (log-transformed) during the pre- and post-design periods

Discharge considerably affected SS hourly loads at the inflow and outflow during both the pre- and post-design periods. Pre-design SS loads (log-transformed) strongly correlated to the discharge (log-transformed) at the inflow and outflow ($r = 0.836$ and 0.735, $p = 0.001$, 2-tailed at 0.05 significant level). Stronger correlations between discharge (log-transformed) and SS loads (log-transformed) also occurred during the post-design period ($r = 0.894$ and 0.945, $p = 0.001$, 2-tailed at 0.01 significantly level), which indicated that sedimentation / resuspension mechanism dominated SS transfer at the input and output.

The median diameter (D_{50}) of SS at the inflow was higher than that at the outflow during most of the sampling days (Figure 3-17). Statistically significant difference was observed between inflow and outflow ($p = 0.001$). Stone and English (1993) investigated the effect of grain size and sediment geochemistry on P adsorption by river sediment, and found that sediment < 8 µm released the most bioavailable P. In Columbia Lake, about 40 – 60% of the particles were < 8 µm at the inflow, while at the outflow, 80 – 90% of the particles were < 8 µm (Figure 3-18).

Photomicrographs of SS in Columbia Lake (Figure 3-19) show that suspended particles at the inflow were larger and more flocculated than at the outflow. This suggests that the majority of the larger suspended particles (> 40 μm) tended to settle on the lake bottom, due to the post-design reduction of flow velocity and increased water retention time in Columbia Lake. Although large solids deposited at the lake bottom, concurrently, the SS concentrations and hourly loads increased after the flow passed through Columbia Lake. A likely explanation is that more fine particles were resuspended into the water column by carp and wildfowl activities or wide-generated waves, while, simultaneously, coarse SS deposited on the lake bottom (Barton et al., 2000; Stantec Consulting Ltd., 2004). In addition, the post-design SS loads were strongly correlated to the TP loads at the outflow ($r = 0.915$, $p = 0.0001$, 2-tailed at 0.01 significant level), which further supports the notion that the outflow SS was dominated by finer grained solids. Investigations above suggest that parts of the outflow SS came from the SS in the river inflow and others were solids resuspended from the edges and bottom of the lake.

The SS concentrations and internal loads decreased gradually, but were not significantly different for months during the same period. Net internal loads were consistently positive and Columbia Lake acted as a SS source during both periods. The monthly averages in net internal loads in May were considerably higher than those in June and August (Appendix 12). The changes were similar to the monthly differences in discharge (Appendix 6). This finding further suggests that discharge dominated SS transfer at the inflow and outflow of Columbia Lake.

4.4 Columbia Lake Performance on TP and SS Internal Loadings

Previous investigations have shown that stormwater impoundments can effectively reduce the downstream loads of TP and SS. The basic engineering design principle is to use strategies and structures that accelerate the physical, chemical and biological processes which can reduce internal P and SS loads and increase nutrient and solids retention. Szilagyi et al. (1990) and Paul et al. (1998) reported that by installing buffers and a submerged flexible curtain (SFC), hydraulic short circuits were prevented and the water retention time was extended. Such measurements resulted in high removal efficiencies for SS, TP and SRP. In the former study, the removal rates (equal to the negative internal loading rate) for SS, TP and SRP were 70, 51, and 61%, respectively. An increase of 30 to 40% in the SRP retention rate was observed due to the SFC in the latter study. Shammaa et al. (2002) suggested that water retention time and detention volume were the most important factors that should be considered for impoundment design. In this study, two dry ponds (surface areas of 5780 and 7500 m^2 and average depths of 3.0 and 2.5 m, respectively) in Edmonton, AB was investigated which showed that low SS removal resulted from low water retention time (Shammaa et al., 2002). It also suggested placing barriers in the pond's bottom would lengthen the flow path, reduce flow velocity and increase detention time. Other facilities, such as a combined detention/wetland stormwater treatment facility (Oberts and Osgood, 1991) or pre-reservoirs (Putz and Benndorf, 1998), were designed to prolong water retention and increase detention volume, so that more nutrients and SS could be retained in impoundments.

In the Columbia Lake study, Kruskal Wallis tests indicated that post-design TP and SS internal loading rates were significantly lower than those during the pre-design period ($p = 0.001$; Appendix 13). Several factors, including inflow SS concentrations, the creation of a new island, the removal of bottom sediments and changes to the lake bathymetry, caused the decrease in net internal TP and SS loading rates.

Inflow SS concentrations affected SS internal loading rates during both the pre- and post-design periods. Concentrations of SS (log-transformed) were strongly and negatively correlated to SS internal loading rates (log-transformed) ($r = -0.750$ and -0.726, $p = 0.001$ for the pre- and post-design, respectively, 2-tailed at 0.01 significant level). The correlations indicated that SS internal loading rates decreased with the increase in inflow SS concentrations. Similar findings were reported by Urbonas (1995) and Van Buren et al. (1997). The latter research found positive internal loading rates during baseflow but negative rates during storm events when inflow SS concentrations increased considerably (Van Buren et al., 1997). This probably resulted from higher inflow SS concentrations during storm events which provided more fine grained particles to flocculate and deposit (Van Buren et al., 1997). Accordingly, far fewer SS were exported from the impoundment.

Previous studies indicated that stormwater management facilities can dramatically increase nutrient and SS retention by prolonging water retention time (Oberts and Ogood, 1991; Paul et al., 1998; Shammaa et al, 2002). The Columbia Lake redesign included the creation of a new island along the east shore of the lake, and maintaining a remnant of the existing west island. These measures, with similar function to buffers, regulated the flow directions and prolonged the water retention time (R_T) from the average of 13 days in the pre-design period to 41 days in the post-design period (Figure 3-3). In the former period, correlations between R_T and TP internal loading rates (log-transformed), and between R_T and SS internal loading rates (log-transformed) were positive (r = 0.530 and 0.652, p = 0.001, 2-tailed at 0.01 significant level), suggesting that net internal loads increased with an increase in water retention time. However, after the Columbia Lake redesign, R_T negatively correlated to TP internal loading rates (log-transformed), and no correlation was observed between R_T and SS loading rates (r = - 0.333, p = 0.018, 2-tailed at 0.05 significant level). Thus, R_T had no impact on SS retention, but prolonging R_T slightly increased TP retention. Many investigations indicated that longer water retention time can increase TP and SS removal from impoundments, mainly by enhancing sedimentation process (Papa et al., 1999; Kennedy, 1999; Shammaa et al., 2002). In the Columbia Lake study, resuspension was the dominant mechanism for TP and SS transfer, and was strongly affected by bioturbation from carp and wildfowl activities and wind-induced wave activity. Because of these multiple impacts on resuspension / sedimentation processes and the complexity of in-lake processes, prolonging water retention time may be ineffective on enhancing TP removal. In terms of SRP, if SRP was at equilibrium concentrations, longer R_T cannot accelerate chemical and biological processes for SRP transfer (Sondergaard et al., 1992; Haggard and Soerens, 2006). Moreover, because the creation of a sediment forebay at the inlet, most of the sediments have already been captured before flow entered Columbia Lake. Therefore, long water retention time had little impact on SS retention. Other approaches, rather than simply prolonging water retention time, affected the TP and SS transfer.

The creation of a new lake bathymetry and the removal of bottom nutrient-rich sediments effectively prevented sediment resuspension and reduced internal TP and SS loading. Many investigators regarded sediment dredging as a useful technology for controlling internal P loadings (Sondergaard et al., 2003; Cooke et al., 2005; Perkins and Underwood, 2001), in particular, effectively when applied to lakes with high internal loading and a short water retention time (Van Der Does et al., 1992). The removal of bottom sediment in Columbia Lake decreased the degree of wave-related resuspension, because wave influence on bottom sediments was considerably decreased and coarse substrate materials characterized the new lake bottom required more wave energy for resuspension. After the sediment removal, the littoral shelf, drop-off shelf, and deep water zone were characterized by coarse substrates (Stantec Consulting Ltd., 2004). In addition, the depth was changed from a 1.0 m average to variable depths, from 0.2 m at the edge to 3.5 m the lake center (Stantec Consulting Ltd., 2004). Increased depth and coarse substrates impeded the resuspension of bottom sediments and associated particulate P (Szilagyi et al., 1990;

Haggard and Soerens, 2006; James et al., 2004). Sondergaard et al. (1992) reported that P release from undisturbed sediment cores was 20 to 30 times lower than the release caused by resuspension. In the Columbia Lake study, after nutrient-rich and easily-resuspended bottom sediments were removed, TP outflow loads during the post-design period decreased to 22% of the loads during the pre-design period, on average. Although the lake bottom materials were mainly slight fine sediments in wetland shallows, which are inclined to be suspended into the water column, the submerged aquatic macrophytes in this area effectively impeded resuspension (a finding supported by James et al. (2004)), and also can absorb dissolved nutrients (Stantec Consulting Ltd., 2004). In addition, beach/sand flats were created along the west side of the lake to avoid resuspension caused by wave action, since the prevalent wind blows from northwest to southwest (Stantec Consulting Ltd., 2004). In current study period (May to August), Columbia Lake was a SS source. However, according to Shantz et al. (2004) and Stantec Consulting Ltd. (2004), particulate settling in Columbia Lake during the water level drawdown period occurred, suggesting that at some point, a large amount of lake sediments may accumulate at the lake bottom and once again contribute TP and SS to the water column. It is estimated that Columbia Lake may accumulate sediments at a rate of 0.5 to 1 cm per year (Stantec Consulting Ltd., 2004). Prior to the lake redesigned, the accumulated sediment depth present within Columbia Lake ranged from depths of 0.3 m to over 1 m (Stantec Consulting Ltd., 2004). Based on the sediment depths and the predicted settling rate, Columbia Lake will return to the pre-design lake sediment condition after approximately more than 300 years, if other factors (i.e., future climate trends, watershed management, land use conditions, etc…) are constant.

There were monthly differences during the post-design period. The average TP internal loading rates from May to August were 90, 44, -11, and -36%, respectively (Figure 3-9). Significant differences on the rates were observed between May and July, May and August and June and August (Appendix 14). Both the change in DO concentration and discharge affected internal TP loading rates. In May and June, higher DO concentrations allow the fine sediments with oxic surface layers to maximize their capacity to bind P (Cooke et al., 2005; Sondergaard et al., 2003). Because of higher inflow discharge in May and June, those fine sediments which bound P were likely to be resuspended from the lake bottom and be transported downstream (Figure 3-2) (Sondergaard et al., 2003). Those processes caused the positive internal TP loading rates. During the following two months, decreased DO concentrations probably leaded to more SRP release from bottom sediments to water column. However, this process was overwhelmed by the increased PP sedimentation due to the decreased discharge, indicating that the PP retention exceeded SRP release (Haggaard and Soerens, 2006; James et al., 2004). Therefore, negative internal loading rates were observed in July and August.

4.5 Model Prediction and Validation

The utility of the Columbia Lake Water Quality Model (Equation 1.1) to predict post-design water quality changes in Columbia Lake was evaluated using measured data for the terms (Q_{12}, Q_i, s_i, V_1, t) for the post-design period (2006-2007). The predicted outflow TP and SS concentrations were compared to the measured data in Figures 3-28 and 3-29. As previously indicated, the predicted water quality was not as variable as the measured water quality, a finding supported by Stantec Consulting Ltd. (2004). In particular, the sharp TP concentration spikes monitored on JD 137 of 2006 and JD 157 of 2007 were 2-3 times the predicted TP concentrations for the same sampling days and the measured SS concentration on JD 177 of 2006 was 7 times higher than the predicated value.

As shown in Figure 3-30 and 3-31, the post-design outflow TP and SS concentrations on most of the sampling days were underestimated by the Columbia Lake Water Quality Model. Values of TP on JD 137 of 2006, and JD 142, 157 and 185 of 2007 were deleted as outliners because the measured TP concentrations were > 70 µg L^{-1} and the predicted values were much lower than the measurements, as indicated by Stantec Consulting Ltd. (2004) that the model was not sensitive to sharp concentration spikes that occurred in Columbia Lake (Stantec Consulting Ltd., 2004). In terms of SS prediction, the value on JD 177 (2006) was deleted as an outliner because the measured concentration was a peak and ten times higher than the prediction (Figure 3-31). The SS concentrations on JD 158 (2006) and JD 220 (2007) were also deleted because they were overestimated, unlike on the rest of the sampling days with predicted values more than 10 µg L^{-1} lower. After deleting these outliners, the regression line for TP (r^2 = 0.365, p = 0.0001) was closer to the reference line (y = x), although the fit was still poor. It indicated that 44 out of 50 values lay within the 95% confidence intervals (Figure 4-3).

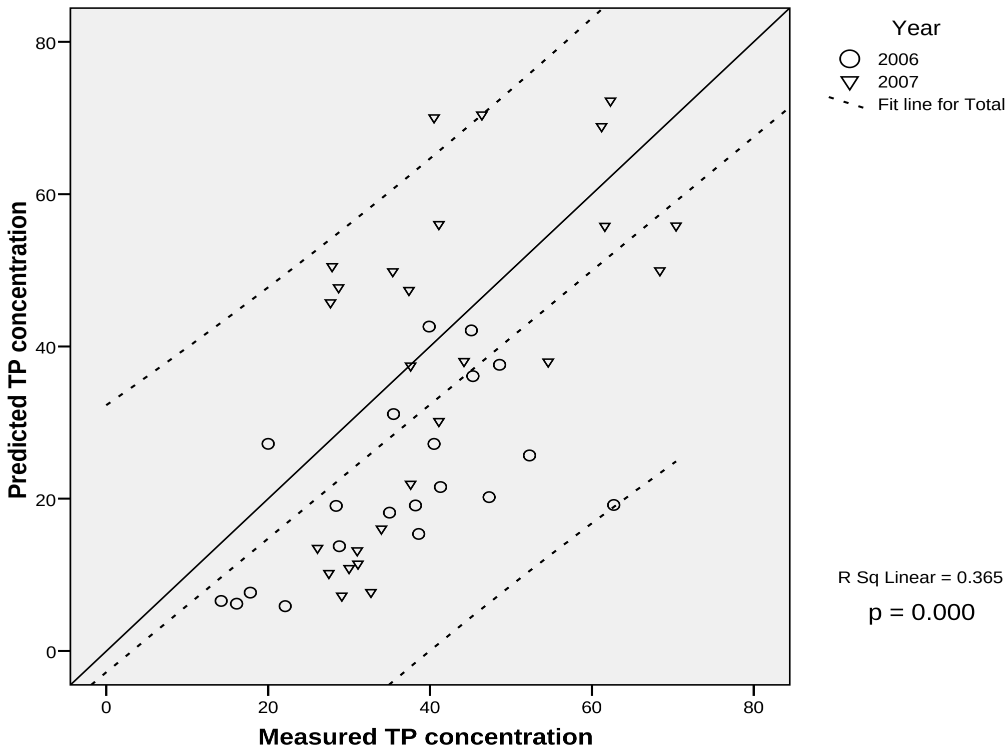

Figure 4-3 Comparison of model prediction with measured outflow TP concentrations

______ Reference line: y = x; - - - - - Regression line (r^2 = 0.365, p = 0.0001) for measured and predicted TP concentrations, confidence of 95%

About 76% of the TP concentrations were underestimated by the model, probably because the impact of resuspension of PP caused by bioturbation and wind-generated waves was not considered. Moreover, the initial TP concentration in the lake was regarded zero for model validation for the post-design period, which may be not the scenario in redesigned Columbia Lake. To mitigate the discrepancy between predicted and measured TP concentrations caused by PP resuspension and initial in-lake concentrations, a coefficient of 10 μg L^{-1} was added to the originally model. After the adjustment, the model predictions get closer to the measurements (Figure 4-4). Multiple comparisons in Univariate Analysis of Variance (Post Hoc tests: Bonfessoni) indicated that the original model predictions were significantly different from the measurements ($p = 0.021$ at 0.05 significant level), while no significant difference was observed between the adjusted model predictions and the measured outflow TP concentrations.

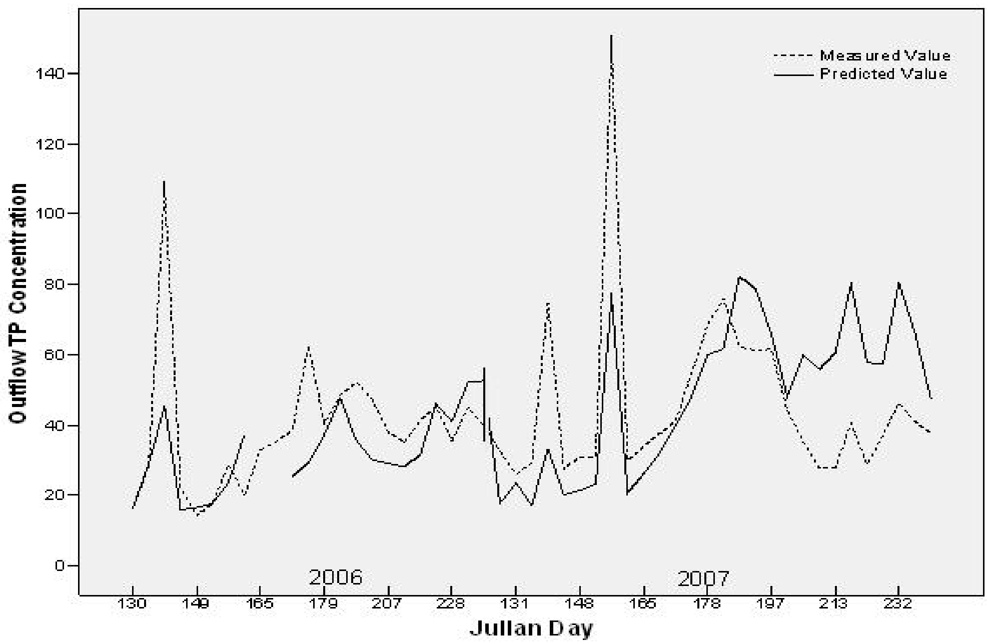

Figure 4-4 The comparison between measured and adjusted predicted (adding coefficient of 10 μg L^{-1}) TP concentrations

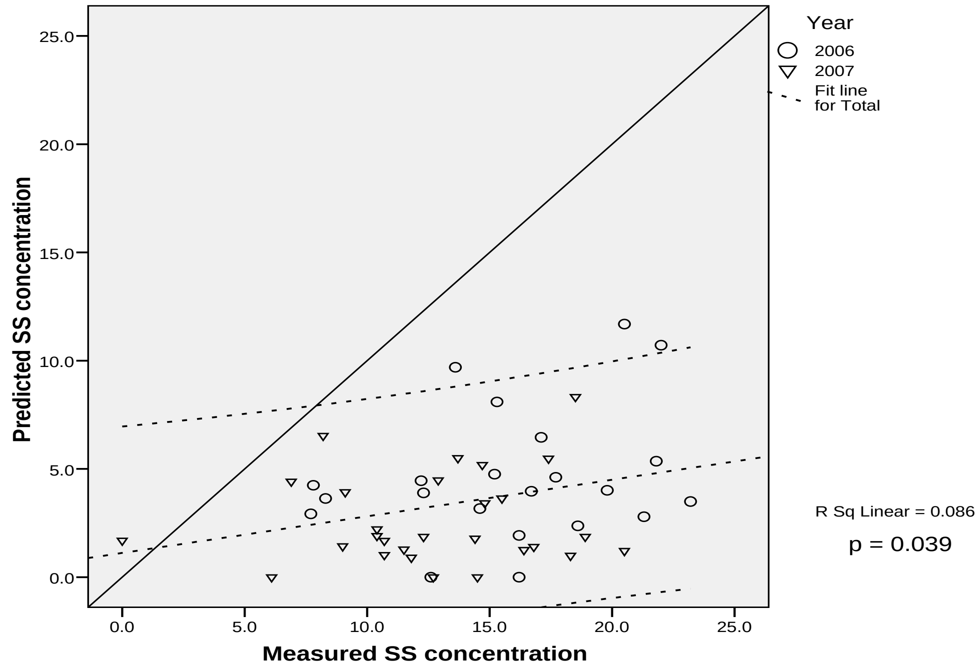

Figure 4-5 Comparison of model prediction with measured outflow SS concentrations

——— Reference line: y = x; - - - - - Regression line (r^2 = 0.086, p = 0.039) for measured and predicted SS concentrations, confidence of 95%

Figure 4-5 shows the difference between predicted and measured SS concentrations, indicating that the model underestimated outflow SS concentrations and performed poor. To calibrate the water quality model for better predicting SS outflow concentrations, the coefficient $K_1 + K_S$ values were examined. As the original value of $K_1 + K_S$ was regarded 3×10^{-7}, ten times different values, 3×10^{-8} and 3×10^{-6} were selected for model recalibration. Because Columbia Lake still acted as a SS source, settling rates of -3×10^{-8} and zero was selected as well. However, the results in Figure 4-6 indicated that SS outflow concentrations were still underestimated following the adjustment of $K_1 + K_S$.

The impact of water retention time, bioturbation and wind-generated waves accounted for the distinction between predicted and measured values. The model is based on the assumption that prolonging R_T and decreasing inflow SS loads would significantly accelerate sedimentation process and therefore, decrease outflow SS loads (Stantec Consulting Ltd., 2004). This means that significantly increased R_T (p = 0.0001; Figure 3-3) during the post-design period should sharply decrease outflow SS concentration by increasing sedimentation. However, the Columbia Lake study found that resuspension was still the dominant mechanism for SS transfer, with the average internal loading rate of 154%, and longer R_T had no impact on SS internal loading and hence, on outflow SS concentrations during the post-design period, probably due to the creation of a sediment forebay at the inlet after the lake rehabilitation and the complexity of in-lake sedimentation / resuspension process. The model assumption was different from the measured impact of R_T, which leads to the underestimated outflow SS concentrations. Resuspension caused by wind-generated waves and bioturbation from carp and wildfowl activities was regarded as important influencing factors for internal loading (Stantec Consulting Ltd., 2004; Barton et al., 2000). Moreover, initial in-lake SS concentration of zero may not be the scenario in the redesigned Columbia Lake. These factors were not considered by the water quality model, but can cause considerable variability in outflow TP and SS concentrations. Without incorporating these factors, Columbia Lake Water Quality Model underestimated the outflow TP and SS concentrations. To adjust the discrepancy caused by resuspension and underestimated initial in-lake SS concentrations, coefficient of 10 mg L^{-1} was added to the model. The adjusted predictions were showed in Figure 4-7. Compared with Figure 3-31, the differences between model predictions and measurements were decreased in most of the sampling days. Multiple comparisons in Univariate Analysis of Variance (Post Hoc tests: Bonfessoni) indicated that the original model predictions were significantly different from the measurements ($p = 0.0001$ at 0.05 significant level), while no significant difference was observed between the adjusted model predictions and the measured outflow SS concentrations.

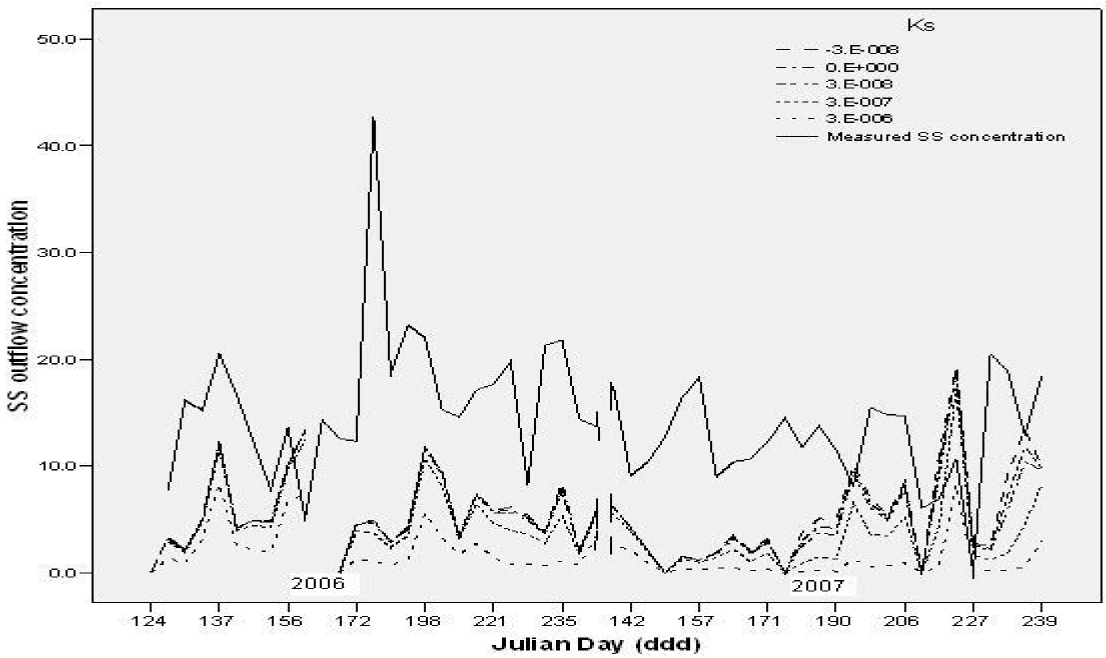

Figure 4-6 Comparison of model predictions with different K_S values to the measured SS outflow concentrations

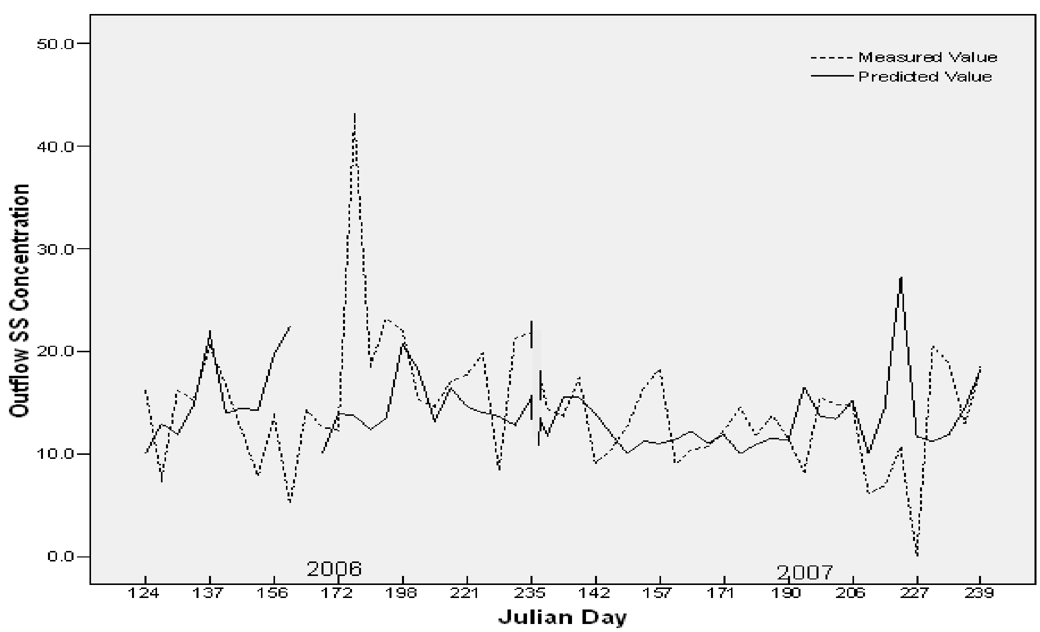

Figure 4-7 Comparison between measured and adjusted predicted (adding coefficient of 10 mg L^{-1}) SS concentrations

4.6 Planning and Management Implication

Impervious surface cover greater than 30% of a watershed causes a dramatic deterioration of water quality and increase surface runoff (Brabec et al., 2002). According to Johnson (2001), urban sprawl can degrade environmentally sensitive areas; reduce regional open space as well as increase stormwater runoff and the risk of flooding. When impervious surfaces cover more than 30% of the area of a watershed, severe degradation on water quality will occur (Brabec et al., 2002; Arnold and Gibbons, 1996).

A wide range of stormwater facilities are being increasingly used as best management practices (BMPs) in Ontario to mitigate the negative impacts of urbanization on water quality and quantity. In particular, on-line and off-line stormwater impoundments are often used to treat the quality and quantity of stormwater in Ontario. Many of the early impoundments were designed specifically to decrease the volume and extend the lag time of stormwater and less attention was directed towards improving water quality.

Modern urban stormwater impoundments have several important ecological and water quality enhancement functions for both watersheds and the surface water system (OMEE, 2003a). They are routinely used in Ontario, in order to prevent or reduce the detrimental impact of land development and practices on the environment (Marsh, 2005). The ultimate goal is to minimize the risks of loss of life, and to mitigate the negative effects of urban development on the natural environment (OMEE, 2003a). To achieve these goals, stormwater management strives to maintain a natural cycle, prevent an increased flooding risk, prevent undesirable stream erosion and protect the water quality from eutrophication (OMEE, 2003a; Hogan and Walbridge, 2007). Previous studies have examined the effectiveness of the urban impoundment to remove TP and SS. The results were highly variable with removal efficiencies ranging from -40% to 90%. This variation has been attributed to differences in the climate, geology and stormwater characteristics as well as the design, size and shape of the impoundment (Van Buren et al., 1997; Isvanovics and Somlyody, 1999; Szilagyi et al., 1990; Davis et al., 2006; Oberts and Osgood, 1991).

In the Columbia Lake study, continuing land use development in the upstream of Laurel Creek Watershed has increased the nutrient and sediment loads delivered to downstream reaches. Columbia Lake was originally designed to control floods, and accumulated a large amount of nutrient-rich sediments, which has resulted in serious internal loading. The lake had a severe eutrophication problem. Therefore, Columbia Lake was redesigned in 2005, to avoid internal nutrient and SS loading and enhance the lake's capacity to absorb pollutants and sediments. The redesign of Columbia Lake is one example of how Best Management Practices (BMPs) can be optimized. But it is necessary to understand the application of wide range of BMPs to control water quality, which means that integrated water source management and watershed planning should be incorporated.

To accomplish the OME water quality objectives, integrated watershed planning and careful design of impoundments should be considered and effectively implemented by planners and engineers. The watershed is the most practical unit for managing water, because impacts such as the P cycle and eutrophication are felt at the watershed level, rather than at the level of political boundaries (OMEE, 2004, Holas et al., 1999). Hence, integral water source management (IWSM) and environmental planning should be implemented to incorporate comprehensive river basin planning and management, as well as stormwater management (Mitchell, 2005).

From planning and engineering perspectives, detailed measures, including upstream land use development, urban impoundment design, modeling and policy-making, should be considered, in order to implement integrated watershed management. The impact of long-term land use development on water resource should be incorporated. Land in the upstream of a watershed should be barred from development and used only for agriculture and conservation purposes. The nutrient and sediment transfers and cycles in the watershed should be fully understood, so that artificial impoundments can be designed, based on an understanding of the relationship between impoundment operation and water quality influence (Kennedy, 1999). In detail, If PP is the dominant form and physical process (sedimentation/resuspension) is important, measures, such as prolonging retention time and using coarse particles to characterize a lake bottom, should be implemented, in order to increase the sedimentation processes (Stantec Consulting Ltd., 2004; Paul et al., 1998; Shammaa et al., 2002). If DP is the dominant form, then measures, such as creating bio-retention and artificial circulation by pumps, jets and bubble air, should be implemented, to avoid DP release from sediments and vegetation (Cooke, 1993; Davis et al., 2006).

Water quality monitoring and modeling are also important parts of watershed planning. Scientific data obtained from monitoring can be utilized to model the TP and SS transfer in a watershed, in order to avoid crisis management, by means of a more effective and complete use of information, and also to decrease the probability of unanticipated conflicts between management activities and resource objectives (Montgomery et al., 1996). From a policy perspective, stormwater management should require no change to either water quality or quantity after the land use development. Stormwater design manuals should set out variable design standards, explicating the adaptations required due to differences in climate, stormwater characteristics and dominant mechanisms for pollutant and SS transfer. Furthermore, stormwater management should integrate land use development, stormwater facility design, water quality monitoring and modeling, and policy making.

4.7 Conclusions

Conclusions of this study are:

1) Outflow TP concentrations and loads significantly decreased after the lake retrofit. Outflow TP concentrations and loads during the post-design period were 38% and 42% of the concentrations and loads during the pre-design period, respectively. Prior to the redesign, Columbia Lake used to be an important TP source, with outflow concentrations and loads 107% and 93% higher than those at the inflow. However, during the post-design period, outflow TP concentrations and loads were not significantly different from those at the inflow, suggesting that internal TP loading was considerably mitigated. Average outflow concentration decreased by 17%, and loads changed from 14 to 18 g h^{-1} after the flow passed through the lake. Additionally, internal SS loads changed from 39 g h^{-1} during the pre-design period to 4 g h^{-1} during the post-design period.

2) Both inflow and outflow SS concentrations and loads dramatically decreased, although Columbia Lake consistently acted as a SS source during the study period. Outflow SS concentrations and loads decreased by 78% and 88% following the lake reconstruction, while the concentrations and loads at the inflow reduced by 55% and 79%, respectively. Moreover, internal SS loads changed from 33 Kg h^{-1} during the pre-design period to 2 Kg h^{-1} during the post-design period.

3) The internal TP and SS loading rates during the post-design period were significantly different from those during the pre-design period. The average net internal loading rates changed from 198% to 22% for TP and from 828% to 154% for SS.

4) Physical processes (sedimentation / resuspension) dominated the transfer of TP and SS. The strong correlation between TP and SS loads indicated that PP was the dominant P form at the inflow and outflow during the whole study period. During the post-design period, outflow grain sizes distributed at lower size classes, while inflow sizes were larger and the particles were

flocculated. This demonstrated that coarse particles deposited at the lake bottom when flow passed through the lake. Concurrently, the outflow SS concentrations and loads increased significantly. It suggested that during the post-design period, inflow coarse sediments deposited due to the sedimentation process, while, simultaneously, fine sediments were resuspended to the water column.

5) Discharge patterns strongly correlated with the TP and SS loads at the inflow and outflow during these study periods. During the post-design period, water retention time slightly affected TP internal loading, and had no impact on SS internal loading, probably due to the complexity of the in-lake process and multiply-affected sedimentation / resuspension, as well as the creation of a sediment forebay at the inlet.

6) The outflow SRP concentrations were slightly different from those at the inflow, probably because the SRP concentrations were in the range of equilibrium concentrations. On average, SRP took about 20% of TP. The data in 2007 demonstrated that the SRP / TP ratios gradually increased from May to August, probably due to the decreased DO concentrations and discharge.

7) The Columbia Lake redesign created the lake bottom with coarse materials and removed nutrient-rich bottom sediments. These measures significantly decreased internal TP and SS loading, by effectively mitigating resuspension of SS and release of DP from bottom sediment.

8) The Columbia Lake Water Quality Model presented by Stantec Consulting Ltd. (2004) underestimated the post-design outflow TP and SS concentrations. It was because bioturbation and wind-generated waves were not incorporated in the model, and the impact of water retention time on TP and SS retention was altered after the lake redesign.

9) After the redesign project, although outflow TP concentrations were significantly decreased with an average of 44±3 μg L^{-1}, 71% and 75% of the TP concentrations at the inflow and outflow, respectively, were still beyond the provincial water quality objective of 30 μg L^{-1}. In terms of SS, after the lake redesign, except in one sample, SS concentrations in all surface water samples were under the benchmark of 25 mg L^{-1}. About 94% of the measured DO concentrations were above 5 mg L^{-1}. In terms of pH values, the surface water was slightly alkaline. About 21% and 26% of pH values were in the optimal range, 6.5 – 8.5, for aquatic habitats at the inflow and outflow, respectively. Additionally, the lake shifted from a TDS sink to a source after the lake was reconstructed.

10) Urban impoundments and reservoirs are Best Management Practices (BMPs) and prevalently utilized in Ontario. By carefully designing the in-lake configuration and inlet and outlet structure, the goal to enhance water quality and control floods and erosion can be achieved. In addition, a strategy integrating land use development, stormwater facility design, water quality monitoring and modeling and policy-making, should be planned and implemented. This strategy should focus on the watershed level, because this is where negative impact can be most easily discerned.

4.7.1 Recommendations for Future Research

From the literature review and results of the present study, the following areas of future research are suggested:

1) Columbia Lake is comprised of five different zones, which are identified by the depths and slope of the bottom. In-lake investigations can be processed to research the pathways for P and SS cycling by monitoring P and SS concentrations and flow discharge in the five parts separately. The functions of each zone on TP and SS transfer can be identified. Evaluation of the spatial variability of P and SS retention (internal loading) is necessary for better understanding the transport and fate of P and SS in Columbia Lake.

2) Since sedimentation is the dominant factor on controlling TP transport and fate, sedimentation / resuspension is the key to understand the spatial variation in load-response relationships in the lake. The sediment core should be researched, to predict sedimentation rates. By comparing sedimentation rates and net retention rates, the degree of the impact of sedimentation / resuspension on TP retention (internal loading) can be evaluated.

3) Research on long-term sediment dynamics is necessary to predict the future performance of Columbia Lake. The frequency of bottom sediment removal should be modeled, based on the sedimentation process.

4) Future research should be addressed to refine the Columbia Lake Water Quality Model, in order to better simulate outflow TP and SS concentrations. The model was primarily based on terms that describe physical process (sedimentation / resuspension), and failed to incorporate terms to describe both chemical and biological processes, such as uptake / release of P by algae, absorption / desorption reactions, bioturbation and resuspension by wind. By incorporating these terms into the model, we will provide more robust prediction and such a model will likely be applicable to a broader range of urban impoundments.

5) In a large scale, future monitoring and analysis should be conducted based on the Laurel Creek Watershed, because the Columbia Lake retrofit project can affect the whole watershed, particularly downstream reaches.

Appendices

Appendix 1

Statistical analysis for water quality and quantity parameters

In- and outflow discharge (m^3/s) in pre- and post-design periods

Design	Site	N	Mean	Std. Error of Mean	Minimum	Maximum	Range	Std. Deviation
Pre-design	Inflow	60	.25	.040	.025	2.16	2.14	.309
	Outflow	59	.22	.035	.045	2.02	1.98	.272
	Total	119	.24	.027	.025	2.16	2.14	.291
Post-design	Inflow	53	.11	.013	.0074	.40	.40	.097
	Outflow	52	.12	.016	.0094	.48	.47	.113
	Total	105	.11	.010	.0074	.48	.48	.105
Total	Inflow	113	.18	.023	.0074	2.16	2.15	.245
	Outflow	111	.17	.021	.0094	2.02	2.01	.219
	Total	224	.18	.015	.0074	2.16	2.15	.232

In- and outflow TP Concentration (μg/l) in both pre- and post-design periods

Design	Site	N	Mean	Std. Error of Mean	Minimum	Maximum	Range	Std. Deviation
Pre-design	Inflow	60	56	6.80544	18	372	354	53
	Outflow	60	116	6.09858	37	266	229	47
	Total	120	86	5.31602	18	372	354	58
Post-design	Inflow	52	53	4.60484	10	124	114	33
	Outflow	52	44	3.17280	14	147	132	23
	Total	104	48	2.82106	10	147	137	29
Total	Inflow	112	55	4.21097	10	372	362	45
	Outflow	112	82	4.95105	14	266	252	52
	Total	224	68	3.37301	10	372	362	50

In- and outflow TP Hourly Loads (g/h) in both pre- and post-design periods

Design	Site	N	Mean	Std. Error of Mean	Minimum	Maximum	Range	Std. Deviation
Pre-design	Inflow	59	43	8	5	346	342	62
	Outflow	58	82	11	22	582	561	85
	Total	117	62	7	5	582	578	76
Post-design	Inflow	51	14	2	3	84	82	16

	Outflow	50	18	4	1	169	168	26
	Total	101	16	2	1	169	168	21
Total	Inflow	110	29	5	3	346	343	49
	Outflow	108	52	7	1	582	581	72
	Total	218	41	4	1	582	581	62

SRP concentrations (µg/l) and hourly loads (g/h) at the inflow and outflow in 2007

	Site	N	Mean	Std. Error of Mean	Minimum	Maximum	Range	Std. Deviation
SRP concentration (ug/l)	Inflow	29	10	.8	2	17	15	4
	Outflow	29	9	.7	5	19	15	4
	Total	58	9	.5	2	19	1	4
SRP hourly loads (g/h)	Inflow	29	2.2	.5	.2	9.4	9.2	2.6
	Outflow	29	2.1	.4	.2	8.4	8.2	2.2
	Total	58	2.1	.3	.2	9.4	9.2	2.4

In- and outflow SS Concentration (mg/l) in pre- and post-design periods

Design	Site	N	Mean	Std. Error of Mean	Minimum	Maximum	Range	Std. Deviation
Pre-design	Inflow	61	19.0	3.2	1.8	168.5	166.7	25.1
	Outflow	60	66.6	4.7	4.0	194.7	190.7	36.5
	Total	121	42.6	3.6	1.8	194.7	192.9	39.3
Post-design	Inflow	54	8.5	.8	<0.01	25.8	25.8	5.9
	Outflow	54	14.5	.8	<0.01	42.5	42.5	6.1
	Total	108	11.5	.6	<0.01	42.5	42.5	6.7
Total	Inflow	115	14.1	1.8	<0.01	168.5	168.5	19.4
	Outflow	114	41.9	3.5	<0.01	194.7	194.7	37.4
	Total	229	27.9	2.2	<0.01	194.7	194.7	32.8

In- and outflow SS Hourly Loads (Kg/h) in pre- and post-design periods

Design	Site	N	Mean	Std. Error of Mean	Minimum	Maximum	Range	Std. Deviation
Pre-design	Inflow	60	18.6	3.6	.4	136.7	136.3	28.0
	Outflow	58	52.2	8.6	1.2	447.3	446.2	65.5
	Total	118	35.1	4.8	.4	447.3	446.9	52.6
Post-design	Inflow	53	3.9	.8	<0.01	29.1	29.1	6.1

	Outflow	52	6.2	.9	<0.01	32.1	32.1	6.6
	Total	105	5.1	.6	<0.01	32.1	32.1	6.4
Total	Inflow	113	11.7	2.1	<0.01	136.7	136.7	22.0
	Outflow	110	30.4	5.0	<0.01	447.3	447.3	52.9
	Total	223	21.0	2.8	<0.01	447.3	447.3	41.3

D_{50} at the inflow and outflow in 2007

Site	N	Mean	Median	Std. Error of Mean	Minimum	Maximum	Range	Std. Deviation
Inflow	29	6.6	6.5	.18	4.9	9.0	4.1	1.0
Outflow	29	5.5	5.4	.13	4.3	7.1	2.8	.7
Total	58	6.1	6.0	.13	4.3	9.0	4.7	1.0

In- and outflow temperature (°C), DO concentration (mg/l) and pH in pre- and post-design conditions

Design	Parameters	Site	N	Mean	Std. Error of Mean	Minimum	Maximum	Range	Std. Deviation
Pre-design	Temperature	Inflow	58	19.9	.5	11.7	26.3	14.6	3.6
		Outflow	56	20.7	.5	9.7	28.9	19.2	4.0
		Total	114	20.3	.4	9.7	28.9	19.2	3.8
	DO Conc.	Inflow	55	8.40	.31	1.62	12.90	11.28	2.28
	(mg/l)	Outflow	57	7.53	.26	1.51	10.64	9.13	1.94
		Total	112	7.95	.20	1.51	12.90	11.39	2.15
	pH	Inflow	61	8.00	.05	7.15	8.92	1.77	.38
		Outflow	61	7.91	.02	7.52	8.37	.85	.19
		Total	122	7.95	.03	7.15	8.92	1.77	.30
Post-design	Temperature	Inflow	54	21.5	.4	13.9	28.7	14.8	3.2
		Outflow	54	21.3	.4	13.6	28.3	14.7	3.3
		Total	108	21.4	.3	13.6	28.7	15.1	3.2
	DO Conc.	Inflow	53	7.47	.27	3.87	11.50	7.63	1.94
	(mg/l)	Outflow	53	6.98	.21	3.19	10.40	7.21	1.50
		Total	106	7.23	.17	3.19	11.50	8.31	1.74
	pH	Inflow	54	8.36	.07	7.07	9.45	2.38	.51
		Outflow	54	8.30	.05	7.10	8.91	1.81	.37
		Total	108	8.33	.04	7.07	9.45	2.38	.44
Total	Temperature	Inflow	112	20.7	.3	11.7	28.7	17.0	3.5
		Outflow	110	21.0	.3	9.7	28.9	19.2	3.6
		Total	222	20.8	.2	9.7	28.9	19.2	3.6
	DO Conc.	Inflow	108	7.94	.21	1.62	12.90	11.28	2.16
	(mg/l)	Outflow	110	7.26	.17	1.51	10.64	9.13	1.76
		Total	218	7.60	.13	1.51	12.90	11.39	1.99

pH	Inflow	115	8.17	.04	7.07	9.45	2.38	.48
	Outflow	115	8.09	.03	7.10	8.91	1.81	.34
	Total	230	8.13	.03	7.07	9.45	2.38	.42

In- and outflow TDS concentration (mg/l) and hourly loads (Kg/h) in pre- and post-design periods

Design	Site		N	Mean	SEM	Minimum	Maximum	Range	SD
Pre-design	Inflow	TDS Conc. (mg/l)	28	275.2	11.6	189.0	391.0	202.0	61.5
		TDS Loads (Kg/h)	27	337.3	95.2	86.2	2626.0	2539.8	494.9
	Outflow	TDS Conc. (mg/l)	27	268.2	11.3	189.0	426.0	237.0	58.6
		TDS Loads (Kg/h)	25	302.4	96.6	66.4	2469.3	2402.9	482.8
	Total	TDS Conc. (mg/l)	55	271.8	8.0	189.0	426.0	237.0	59.7
		TDS Loads (Kg/h)	52	320.5	67.2	66.4	2626.0	2559.6	484.6
Post-design	Inflow	TDS Conc. (mg/l)	54	292.1	6.3	207.7	404.5	196.8	46.6
		TDS Loads (Kg/h)	53	111.9	14.9	8.4	464.5	456.1	108.6
	Outflow	TDS Conc. (mg/l)	54	344.2	7.2	231.5	482.9	251.4	53.0
		TDS Loads (Kg/h)	52	142.2	19.0	13.4	555.7	542.3	136.8
	Total	TDS Conc. (mg/l)	108	318.1	5.4	207.7	482.9	275.2	56.1
		TDS Loads (Kg/h)	105	126.9	12.0	8.4	555.7	547.3	123.7
Total	Inflow	TDS Conc. (mg/l)	82	286.3	5.8	189.0	404.5	215.5	52.4
		TDS Loads (Kg/h)	80	188.0	35.3	8.4	2626.0	2617.6	316.0
	Outflow	TDS Conc. (mg/l)	81	318.8	7.3	189.0	482.9	293.9	65.4
		TDS Loads (Kg/h)	77	194.2	34.5	13.4	2469.3	2455.9	303.1
	Total	TDS Conc. (mg/l)	163	302.5	4.8	189.0	482.9	293.9	61.2
		TDS Loads (Kg/h)	157	191.0	24.6	8.4	2626.0	2617.6	308.8

Appendix 2

Kruskal Wallis Test on the comparison of inflow and outflow TP concentrations, and the comparisons of inflow and outflow TP hourly loads in the pre-design period

Ranks

	Site	N	Mean Rank
TP Concentration (μg/l)	Inflow	60	37.08
	Outflow	60	83.92
	Total	120	
TP Hourly Loads (g/h)	Inflow	59	41.02
	Outflow	58	77.29
	Total	117	

Test Statistics (a,b)

	TP Concentration (μg/l)	TP Hourly Loads (g/h)
Chi-Square	54.383	33.454
df	1	1
Asymp. Sig.	.000	.000

a Kruskal Wallis Test b Grouping Variable: Site

Kruskal Wallis Test on the comparison of inflow and outflow TP concentrations, and the comparison of inflow and outflow TP hourly loads in the post-design period

Ranks

	Site	N	Mean Rank
TP Concentration (ug/l)	Inflow	52	55.63
	Outflow	52	49.38
	Total	104	
TP Hourly Loads (g/h)	Inflow	51	50.35
	Outflow	50	51.66
	Total	101	

Test Statistics (a,b)

	TP Concentration (μg/l)	TP Hourly Loads (g/h)
Chi-Square	1.116	.050
df	1	1
Asymp. Sig.	.291	.823

a Kruskal Wallis Test b Grouping Variable: Site

Appendix 3

Kruskal Wallis Tests on the comparison of inflow TP concentrations between pre- and post-design periods, and the comparison of inflow TP hourly loads between pre- and post-design periods

Ranks

	Design	N	Mean Rank
TP Concentration (μg/l)	Pre-design	60	56.38
	Post-design	52	56.64
	Total	112	
TP Hourly Loads (g/h)	Pre-design	59	72.13
	Post-design	51	36.26
	Total	110	

Test Statistics (a,b)

	TP Concentration (ug/l)	TP Hourly Loads (g/h)
Chi-Square	.002	34.577
df	1	1
Asymp. Sig.	.965	.000

a Kruskal Wallis Test b Grouping Variable: Design

Kruskal Wallis Tests on the comparison of outflow TP concentrations between pre- and post-design periods, and the outflow comparison of TP hourly loads between pre- and post-design periods

Ranks

	Design	N	Mean Rank
TP Concentration (μg/l)	Pre-design	60	79.34
	Post-design	52	30.14
	Total	112	
TP Hourly Loads (g/h)	Pre-design	58	76.42
	Post-design	50	29.07
	Total	108	

Test Statistics (a,b)

	TP Concentration (ug/l)	TP Hourly Loads (g/h)
Chi-Square	63.932	61.376
df	1	1
Asymp. Sig.	.000	.000

a Kruskal Wallis Test b Grouping Variable: Design

Appendix 4
Kruskal Wallis Test on the comparisons between inflow and outflow SRP concentrations and loads in 2007

Ranks

	Site	N	Mean Rank
SRP concentration (µg/l)	Inflow	29	31.97
	Outflow	29	27.03
	Total	58	
SRP hourly loads (g/h)	Inflow	29	28.55
	Outflow	29	30.45
	Total	58	

Test Statistics(a,b)

	SRP concentration (ug/l)	SRP hourly loads (g/h)
Chi-Square	1.236	.184
df	1	1
Asymp. Sig.	.266	.668

a Kruskal Wallis Test b Grouping Variable: Site

Kruskal Wallis Test on the comparison between inflow and outflow DO concentrations and pH in 2007

Ranks

	Site	N	Mean Rank
DO Concentration (mg/l)	Inflow	28	29.07
	Outflow	28	27.93
	Total	56	
pH	Inflow	29	26.71
	Outflow	29	32.29
	Total	58	

Test Statistics(a,b)

	DO Concentration (mg/l)	pH
Chi-Square	.069	1.588
df	1	1
Asymp. Sig.	.793	.208

a Kruskal Wallis Test b Grouping Variable: Site

Appendix 5
Multiple comparisons in Univariate Analysis on monthly differences in SRP/TP at the inflow in 2007

Between-Subjects Factors

		Value Label	N
Month	5	May	7
	6	Jun.	7
	7	Jul.	8
	8	Aug.	7

Tests of Between-Subjects Effects
Dependent Variable: SRP/TP

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	.662(a)	3	.221	14.441	.000
Intercept	1.413	1	1.413	92.409	.000
Month	.662	3	.221	14.441	.000
Error	.382	25	.015		
Total	2.416	29			
Corrected Total	1.045	28			

a R Squared = .634 (Adjusted R Squared = .590)

Multiple Comparisons
Dependent Variable: SRP/TP at the inflow
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	.3659(*)	.06610	.000	.1766	.5553
	Jul.	.3664(*)	.06400	.000	.1831	.5498
	Aug.	.3146(*)	.06610	.000	.1253	.5040
Jun.	May	-.3659(*)	.06610	.000	-.5553	-.1766
	Jul.	.0005	.06400	1.000	-.1828	.1839
	Aug.	-.0513	.06610	1.000	-.2407	.1381
Jul.	May	-.3664(*)	.06400	.000	-.5498	-.1831
	Jun.	-.0005	.06400	1.000	-.1839	.1828
	Aug.	-.0518	.06400	1.000	-.2352	.1315
Aug.	May	-.3146(*)	.06610	.000	-.5040	-.1253
	Jun.	.0513	.06610	1.000	-.1381	.2407
	Jul.	.0518	.06400	1.000	-.1315	.2352

Based on observed means.
* The mean difference is significant at the .05 level.

Multiple comparisons in Univariate Analysis on monthly differences in SRP/TP at the outflow in 2007

Between-Subjects Factors

		Value Label	N
Month	5	May	7
	6	Jun.	7
	7	Jul.	8
	8	Aug.	7

Tests of Between-Subjects Effects

Dependent Variable: SRP/TP

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	.233(a)	3	.078	10.349	.000
Intercept	1.489	1	1.489	198.395	.000
Month	.233	3	.078	10.349	.000
Error	.188	25	.008		
Total	1.894	29			
Corrected Total	.421	28			

a R Squared = .554 (Adjusted R Squared = .500)

Multiple Comparisons

Dependent Variable: SRP/TP at the outflow

Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	.1015	.04631	.228	-.0312	.2342
	Jul.	.0468	.04484	1.000	-.0816	.1753
	Aug.	-.1432(*)	.04631	.029	-.2759	-.0106
Jun.	May	-.1015	.04631	.228	-.2342	.0312
	Jul.	-.0547	.04484	1.000	-.1831	.0738
	Aug.	-.2447(*)	.04631	.000	-.3774	-.1121
Jul.	May	-.0468	.04484	1.000	-.1753	.0816
	Jun.	.0547	.04484	1.000	-.0738	.1831
	Aug.	-.1901(*)	.04484	.002	-.3185	-.0616
Aug.	May	.1432(*)	.04631	.029	.0106	.2759
	Jun.	.2447(*)	.04631	.000	.1121	.3774
	Jul.	.1901(*)	.04484	.002	.0616	.3185

Based on observed means.

* The mean difference is significant at the .05 level.

Appendix 6
Multiple comparisons in Univariate Analysis on monthly differences of discharge in the pre-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	30
	6	Jun.	28
	7	Jul.	33
	8	Aug.	28

Tests of Between-Subjects Effects

Dependent Variable: Discharge (m^3/s)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	2.391(a)	3	.797	12.104	.000
Intercept	6.723	1	6.723	102.096	.000
Month	2.391	3	.797	12.104	.000
Error	7.573	115	.066		
Total	16.675	119			
Corrected Total	9.964	118			

a R Squared = .240 (Adjusted R Squared = .220)

Multiple Comparisons

Dependent Variable: Discharge (m3/s)
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	.266664(*)	.0674296	.001	.085638	.447690
	Jul.	.350488(*)	.0647335	.000	.176700	.524276
	Aug.	.332736(*)	.0674296	.000	.151710	.513762
Jun.	May	-.266664(*)	.0674296	.001	-.447690	-.085638
	Jul.	.083824	.0659334	1.000	-.093185	.260833
	Aug.	.066071	.0685824	1.000	-.118049	.250192
Jul.	May	-.350488(*)	.0647335	.000	-.524276	-.176700
	Jun.	-.083824	.0659334	1.000	-.260833	.093185
	Aug.	-.017752	.0659334	1.000	-.194761	.159257
Aug.	May	-.332736(*)	.0674296	.000	-.513762	-.151710
	Jun.	-.066071	.0685824	1.000	-.250192	.118049
	Jul.	.017752	.0659334	1.000	-.159257	.194761

Based on observed means.
* The mean difference is significant at the .05 level.

Multiple comparisons in Univariate Analysis on monthly differences of discharge in the post-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	30
	6	Jun.	25
	7	Jul.	24
	8	Aug.	26

Tests of Between-Subjects Effects

Dependent Variable: Discharge (m^3/s)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	.389(a)	3	.130	17.249	.000
Intercept	1.202	1	1.202	159.963	.000
Month	.389	3	.130	17.249	.000
Error	.759	101	.008		
Total	2.464	105			
Corrected Total	1.148	104			

a R Squared = .339 (Adjusted R Squared = .319)

Multiple Comparisons

Dependent Variable: Discharge (m3/s)

Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	.104035(*)	.0234789	.000	.040848	.167221
	Jul.	.137245(*)	.0237442	.000	.073344	.201146
	Aug.	.148294(*)	.0232313	.000	.085774	.210815
Jun.	May	-.104035(*)	.0234789	.000	-.167221	-.040848
	Jul.	.033210	.0247770	1.000	-.033470	.099891
	Aug.	.044260	.0242860	.428	-.021099	.109618
Jul.	May	-.137245(*)	.0237442	.000	-.201146	-.073344
	Jun.	-.033210	.0247770	1.000	-.099891	.033470
	Aug.	.011049	.0245426	1.000	-.055000	.077099
Aug.	May	-.148294(*)	.0232313	.000	-.210815	-.085774
	Jun.	-.044260	.0242860	.428	-.109618	.021099
	Jul.	-.011049	.0245426	1.000	-.077099	.055000

Based on observed means.

* The mean difference is significant at the .05 level.

Appendix 7
Multiple comparisons in Univariate Analysis on monthly difference of inflow TP concentration in the pre-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	15
	6	Jun.	14
	7	Jul.	17
	8	Aug.	14

Tests of Between-Subjects Effects

Dependent Variable: TP Concentration (µg/l)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	35933.291(a)	3	11977.764	5.240	.003
Intercept	187609.037	1	187609.037	82.067	.000
Month	35933.291	3	11977.764	5.240	.003
Error	128018.292	56	2286.041		
Total	352223.600	60			
Corrected Total	163951.583	59			

a R Squared = .219 (Adjusted R Squared = .177)

Multiple Comparisons

Dependent Variable: TP Concentration (µg/l)
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	-.6543	17.76771	1.000	-49.2529	47.9443
	Jul.	-28.6341	16.93741	.579	-74.9617	17.6934
	Aug.	-61.2471(*)	17.76771	.006	-109.8457	-12.6485
Jun.	May	.6543	17.76771	1.000	-47.9443	49.2529
	Jul.	-27.9798	17.25577	.663	-75.1782	19.2185
	Aug.	-60.5929(*)	18.07145	.009	-110.0223	-11.1635
Jul.	May	28.6341	16.93741	.579	-17.6934	74.9617
	Jun.	27.9798	17.25577	.663	-19.2185	75.1782
	Aug.	-32.6130	17.25577	.384	-79.8114	14.5853
Aug.	May	61.2471(*)	17.76771	.006	12.6485	109.8457
	Jun.	60.5929(*)	18.07145	.009	11.1635	110.0223
	Jul.	32.6130	17.25577	.384	-14.5853	79.8114

Based on observed means. * The mean difference is significant at the .05 level.

Multiple comparisons in Univariate Analysis on monthly difference of outflow TP concentrations in the pre-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	15
	6	Jun.	14
	7	Jul.	17
	8	Aug.	14

Tests of Between-Subjects Effects

Dependent Variable: TP Concentration (µg/l)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	35459.673(a)	3	11819.891	6.880	.001
Intercept	796240.824	1	796240.824	463.496	.000
Month	35459.673	3	11819.891	6.880	.001
Error	96202.547	56	1717.903		
Total	939022.220	60			
Corrected Total	131662.220	59			

a R Squared = .269 (Adjusted R Squared = .230)

Multiple Comparisons

Dependent Variable: TP Concentration (µg/l)
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	-16.5833	15.40241	1.000	-58.7123	25.5456
	Jul.	-51.1745(*)	14.68264	.006	-91.3348	-11.0143
	Aug.	-59.4190(*)	15.40241	.002	-101.5480	-17.2901
Jun.	May	16.5833	15.40241	1.000	-25.5456	58.7123
	Jul.	-34.5912	14.95862	.147	-75.5063	6.3240
	Aug.	-42.8357	15.66572	.050	-85.6849	.0135
Jul.	May	51.1745(*)	14.68264	.006	11.0143	91.3348
	Jun.	34.5912	14.95862	.147	-6.3240	75.5063
	Aug.	-8.2445	14.95862	1.000	-49.1597	32.6706
Aug.	May	59.4190(*)	15.40241	.002	17.2901	101.5480
	Jun.	42.8357	15.66572	.050	-.0135	85.6849
	Jul.	8.2445	14.95862	1.000	-32.6706	49.1597

Based on observed means.
* The mean difference is significant at the .05 level.

Appendix 8
Multiple comparisons in Univariate Analysis on monthly differences of net internal TP loads in the pre-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	15
	6	Jun.	13
	7	Jul.	16
	8	Aug.	14

Tests of Between-Subjects Effects

Dependent Variable: Net internal TP hourly loads (g/h)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	27860.978(a)	3	9286.993	2.111	.109
Intercept	88740.279	1	88740.279	20.175	.000
Month	27860.978	3	9286.993	2.111	.109
Error	237524.194	54	4398.596		
Total	355882.832	58			
Corrected Total	265385.172	57			

a R Squared = .105 (Adjusted R Squared = .055)

Multiple Comparisons

Dependent Variable: Net internal TP hourly loads (g/h)
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	13.9563	25.13152	1.000	-54.8774	82.7899
	Jul.	23.5081	23.83594	1.000	-41.7771	88.7932
	Aug.	59.7629	24.64600	.112	-7.7410	127.2668
Jun.	May	-13.9563	25.13152	1.000	-82.7899	54.8774
	Jul.	9.5518	24.76420	1.000	-58.2758	77.3795
	Aug.	45.8066	25.54484	.471	-24.1591	115.7724
Jul.	May	-23.5081	23.83594	1.000	-88.7932	41.7771
	Jun.	-9.5518	24.76420	1.000	-77.3795	58.2758
	Aug.	36.2548	24.27134	.846	-30.2229	102.7325
Aug.	May	-59.7629	24.64600	.112	-127.2668	7.7410
	Jun.	-45.8066	25.54484	.471	-115.7724	24.1591
	Jul.	-36.2548	24.27134	.846	-102.7325	30.2229

Based on observed means.

Multiple comparisons in Univariate Analysis on monthly differences of net internal TP loads in the post-design condition

Between-Subjects Factors

		Value Label	N
Month	5	May	13
	6	Jun.	12
	7	Jul.	12
	8	Aug.	13

Tests of Between-Subjects Effects

Dependent Variable: Net internal TP hourly loads (g/h)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	2939.892(a)	3	979.964	4.748	.006
Intercept	529.581	1	529.581	2.566	.116
Month	2939.892	3	979.964	4.748	.006
Error	9494.922	46	206.411		
Total	12983.215	50			
Corrected Total	12434.814	49			

a R Squared = .236 (Adjusted R Squared = .187)

Multiple Comparisons

Dependent Variable: Net internal TP hourly loads (g/h)
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	11.1591	5.75141	.351	-4.6985	27.0168
	Jul.	14.9241	5.75141	.076	-.9335	30.7818
	Aug.	20.6115(*)	5.63521	.004	5.0743	36.1488
Jun.	May	-11.1591	5.75141	.351	-27.0168	4.6985
	Jul.	3.7650	5.86531	1.000	-12.4067	19.9367
	Aug.	9.4524	5.75141	.643	-6.4052	25.3101
Jul.	May	-14.9241	5.75141	.076	-30.7818	.9335
	Jun.	-3.7650	5.86531	1.000	-19.9367	12.4067
	Aug.	5.6874	5.75141	1.000	-10.1702	21.5451
Aug.	May	-20.6115(*)	5.63521	.004	-36.1488	-5.0743
	Jun.	-9.4524	5.75141	.643	-25.3101	6.4052
	Jul.	-5.6874	5.75141	1.000	-21.5451	10.1702

Based on observed means.
* The mean difference is significant at the .05 level.

Appendix 9
Multiple comparisons in Univariate Analysis on monthly difference of inflow TP concentrations in the post-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	13
	6	Jun.	14
	7	Jul.	12
	8	Aug.	13

Tests of Between-Subjects Effects

Dependent Variable: TP Concentration (µg/l)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	22206.414(a)	3	7402.138	10.441	.000
Intercept	148095.898	1	148095.898	208.904	.000
Month	22206.414	3	7402.138	10.441	.000
Error	34028.095	48	708.919		
Total	202048.220	52			
Corrected Total	56234.509	51			

a R Squared = .395 (Adjusted R Squared = .357)

Multiple Comparisons

Dependent Variable: TP Concentration (µg/l)
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	-26.9808	10.25521	.068	-55.2033	1.2418
	Jul.	-52.5724(*)	10.65874	.000	-81.9055	-23.2393
	Aug.	-48.3538(*)	10.44339	.000	-77.0943	-19.6134
Jun.	May	26.9808	10.25521	.068	-1.2418	55.2033
	Jul.	-25.5917	10.47443	.110	-54.4175	3.2342
	Aug.	-21.3731	10.25521	.255	-49.5956	6.8495
Jul.	May	52.5724(*)	10.65874	.000	23.2393	81.9055
	Jun.	25.5917	10.47443	.110	-3.2342	54.4175
	Aug.	4.2186	10.65874	1.000	-25.1145	33.5517
Aug.	May	48.3538(*)	10.44339	.000	19.6134	77.0943
	Jun.	21.3731	10.25521	.255	-6.8495	49.5956
	Jul.	-4.2186	10.65874	1.000	-33.5517	25.1145

Based on observed means.
* The mean difference is significant at the .05 level.

Multiple comparisons in Univariate Analysis on monthly difference of outflow TP concentrations in the post-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	13
	6	Jun.	14
	7	Jul.	12
	8	Aug.	13

Tests of Between-Subjects Effects

Dependent Variable: TP Concentration (µg/l)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	2465.273(a)	3	821.758	1.628	.195
Intercept	98548.213	1	98548.213	195.213	.000
Month	2465.273	3	821.758	1.628	.195
Error	24231.593	48	504.825		
Total	125137.370	52			
Corrected Total	26696.865	51			

a R Squared = .092 (Adjusted R Squared = .036)

Multiple Comparisons

Dependent Variable: TP Concentration (µg/l)

Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	-12.7176	8.65399	.889	-36.5336	11.0984
	Jul.	-17.3212	8.99452	.360	-42.0743	7.4320
	Aug.	-3.3385	8.81280	1.000	-27.5915	20.9146
Jun.	May	12.7176	8.65399	.889	-11.0984	36.5336
	Jul.	-4.6036	8.83898	1.000	-28.9287	19.7215
	Aug.	9.3791	8.65399	1.000	-14.4369	33.1951
Jul.	May	17.3212	8.99452	.360	-7.4320	42.0743
	Jun.	4.6036	8.83898	1.000	-19.7215	28.9287
	Aug.	13.9827	8.99452	.760	-10.7704	38.7358
Aug.	May	3.3385	8.81280	1.000	-20.9146	27.5915
	Jun.	-9.3791	8.65399	1.000	-33.1951	14.4369
	Jul.	-13.9827	8.99452	.760	-38.7358	10.7704

Based on observed means.

Appendix 10

Kruskal Wallis Test on the comparison of inflow and outflow SS concentrations, and the comparison of inflow and outflow SS hourly loads in the pre-design period

Ranks

	Site	N	Mean Rank
SS Concentration (mg/l)	Inflow	61	35.82
	Outflow	60	86.60
	Total	121	
SS Hourly Loads (Kg/h)	Inflow	60	41.98
	Outflow	58	77.62
	Total	118	

Test Statistics (a,b)

	SS Concentration (mg/l)	SS Hourly Loads (Kg/h)
Chi-Square	63.407	32.008
df	1	1
Asymp. Sig.	.000	.000

a Kruskal Wallis Test b Grouping Variable: Site

Kruskal Wallis Test on the comparison of inflow and outflow SS concentrations, and the comparison of SS hourly loads in the post-design period

Ranks

	Site	N	Mean Rank
SS Concentration (mg/l)	Inflow	54	39.23
	Outflow	54	69.77
	Total	108	
SS Hourly Loads (Kg/h)	Inflow	53	44.48
	Outflow	52	61.68
	Total	105	

Test Statistics (a,b)

	SS Concentration (mg/l)	SS Hourly Loads (Kg/h)
Chi-Square	25.673	8.378
df	1	1
Asymp. Sig.	.000	.004

a Kruskal Wallis Test b Grouping Variable: Site

Appendix 11
Kruskal Wallis Test on the comparison of inflow SS concentrations between pre- and post-design periods, and on the comparison of SS hourly loads between pre- and post-design periods

Ranks

	Design	N	Mean Rank
SS Concentration (mg/l)	Pre-design	61	68.90
	Post-design	54	45.69
	Total	115	
SS Hourly Loads (Kg/h)	Pre-design	60	71.98
	Post-design	53	40.04
	Total	113	

Test Statistics (a,b)

	SS Concentration (mg/l)	SS Hourly Loads (Kg/h)
Chi-Square	13.891	26.758
df	1	1
Asymp. Sig.	.000	.000

a Kruskal Wallis Test b Grouping Variable: Design

Kruskal Wallis Test on the comparison of outflow SS concentrations between pre- and post-design periods, and on the comparison of SS hourly loads between pre- and post-design periods

Ranks

	Design	N	Mean Rank
SS Concentration (mg/l)	Pre-design	60	82.38
	Post-design	54	29.85
	Total	114	
SS Hourly Loads (Kg/h)	Pre-design	58	78.52
	Post-design	52	29.83
	Total	110	

Test Statistics (a,b)

	SS Concentration (mg/l)	SS Hourly Loads (Kg/h)
Chi-Square	71.792	63.886
df	1	1
Asymp. Sig.	.000	.000

a Kruskal Wallis Test b Grouping Variable: Design

Appendix 12
Multiple comparisons in Univariate Analysis on monthly differences of internal SS loads in the pre-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	14
	6	Jun.	14
	7	Jul.	16
	8	Aug.	14

Tests of Between-Subjects Effects

Dependent Variable: Net internal SS hourly loads (Kg/h)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	14465.721(a)	3	4821.907	1.918	.138
Intercept	65002.859	1	65002.859	25.856	.000
Month	14465.721	3	4821.907	1.918	.138
Error	135757.876	54	2514.035		
Total	214310.981	58			
Corrected Total	150223.596	57			

a R Squared = .096 (Adjusted R Squared = .046)

Multiple Comparisons

Dependent Variable: Net internal SS hourly loads (Kg/h)
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	23.2886	18.95120	1.000	-28.6176	75.1947
	Jul.	33.4148	18.34942	.445	-16.8431	83.6727
	Aug.	43.0171	18.95120	.163	-8.8890	94.9233
Jun.	May	-23.2886	18.95120	1.000	-75.1947	28.6176
	Jul.	10.1263	18.34942	1.000	-40.1317	60.3842
	Aug.	19.7286	18.95120	1.000	-32.1776	71.6347
Jul.	May	-33.4148	18.34942	.445	-83.6727	16.8431
	Jun.	-10.1263	18.34942	1.000	-60.3842	40.1317
	Aug.	9.6023	18.34942	1.000	-40.6556	59.8602
Aug.	May	-43.0171	18.95120	.163	-94.9233	8.8890
	Jun.	-19.7286	18.95120	1.000	-71.6347	32.1776
	Jul.	-9.6023	18.34942	1.000	-59.8602	40.6556

Based on observed means.

Multiple comparisons in Univariate Analysis on monthly differences of internal SS loads in the post-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	15
	6	Jun.	12
	7	Jul.	12
	8	Aug.	13

Tests of Between-Subjects Effects

Dependent Variable: Net internal SS hourly loads (Kg/h)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	243.266(a)	3	81.089	4.891	.005
Intercept	226.232	1	226.232	13.645	.001
Month	243.266	3	81.089	4.891	.005
Error	795.853	48	16.580		
Total	1303.090	52			
Corrected Total	1039.119	51			

a R Squared = .234 (Adjusted R Squared = .186)

Multiple Comparisons

Dependent Variable: Net internal SS hourly loads (Kg/h)
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	4.3722(*)	1.57704	.047	.0321	8.7122
	Jul.	3.8747	1.57704	.106	-.4654	8.2147
	Aug.	5.4872(*)	1.54297	.005	1.2409	9.7335
Jun.	May	-4.3722(*)	1.57704	.047	-8.7122	-.0321
	Jul.	-.4975	1.66234	1.000	-5.0723	4.0773
	Aug.	1.1151	1.63006	1.000	-3.3709	5.6010
Jul.	May	-3.8747	1.57704	.106	-8.2147	.4654
	Jun.	.4975	1.66234	1.000	-4.0773	5.0723
	Aug.	1.6126	1.63006	1.000	-2.8734	6.0985
Aug.	May	-5.4872(*)	1.54297	.005	-9.7335	-1.2409
	Jun.	-1.1151	1.63006	1.000	-5.6010	3.3709
	Jul.	-1.6126	1.63006	1.000	-6.0985	2.8734

Based on observed means.
* The mean difference is significant at the .05 level.

Appendix 13

Kruskal Wallis Test on the comparison of TP internal loads between pre- and post-design periods, and on the comparison of TP internal loading rates between pre- and post-design periods

Ranks

	Design	N	Mean Rank
Net internal TP hourly loads (g/h)	pre-design	58	74.21
	post-design	50	31.64
	Total	108	
Columbia Lake performance on TP retention	pre-design	58	70.93
	post-design	50	35.44
	Total	108	

Test Statistics (a,b)

	Net internal TP hourly loads (g/h)	Internal TP loading rate (%)
Chi-Square	49.596	34.478
df	1	1
Asymp. Sig.	.000	.000

a Kruskal Wallis Test b Grouping Variable: Design

Kruskal Wallis Test on the comparison of SS internal loads between pre- and post-design periods, and on the comparison of SS internal loading rates between pre- and post-design periods

Ranks

	Design	N	Mean Rank
Columbia Lake performance on SS retention rate	pre-design	58	64.55
	post-design	48	40.15
	Total	106	
Net internal SS hourly loads (Kg/h)	pre-design	58	75.59
	post-design	52	33.10
	Total	110	

Test Statistics (a,b)

	Internal SS loading rate (%)	Net internal SS hourly loads (Kg/h)
Chi-Square	16.552	48.655
df	1	1
Asymp. Sig.	.000	.000

a Kruskal Wallis Test b Grouping Variable: Design

Appendix 14
Multiple comparisons in Univariate Analysis on monthly changes in TP internal loading rates during the post-design period

Between-Subjects Factors

		Value Label	N
Month	5	May	13
	6	Jun.	12
	7	Jul.	12
	8	Aug.	13

Tests of Between-Subjects Effects

Dependent Variable: Columbia Lake performance on TP retention

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	12.267(a)	3	4.089	16.973	.000
Intercept	2.395	1	2.395	9.942	.003
Month	12.267	3	4.089	16.973	.000
Error	11.082	46	.241		
Total	25.798	50			
Corrected Total	23.350	49			

a R Squared = .525 (Adjusted R Squared = .494)

Multiple Comparisons

Dependent Variable: Columbia Lake performance on TP retention
Post Hoc Tests: Bonferroni

(I) Month	(J) Month	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
May	Jun.	.4679	.19649	.129	-.0739	1.0097
	Jul.	1.0150(*)	.19649	.000	.4732	1.5568
	Aug.	1.2598(*)	.19252	.000	.7290	1.7906
Jun.	May	-.4679	.19649	.129	-1.0097	.0739
	Jul.	.5471	.20038	.054	-.0054	1.0996
	Aug.	.7919(*)	.19649	.001	.2502	1.3337
Jul.	May	-1.0150(*)	.19649	.000	-1.5568	-.4732
	Jun.	-.5471	.20038	.054	-1.0996	.0054
	Aug.	.2448	.19649	1.000	-.2969	.7866
Aug.	May	-1.2598(*)	.19252	.000	-1.7906	-.7290
	Jun.	-.7919(*)	.19649	.001	-1.3337	-.2502
	Jul.	-.2448	.19649	1.000	-.7866	.2969

Based on observed means.
* The mean difference is significant at the .05 level.

Appendix 15
Picture of Columbia Lake

Study site: Columbia Lake inflow

Study site: Columbia Lake outflow

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