Effect of Hydrological Regimes on Groundwater Phosphorus Transfer in a Riparian Wetland

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

Riparian wetlands are important areas for regulating phosphorus (P) transfer in shallow groundwater. Limited knowledge is currently available regarding the effects of hydrological regimes on the transfer process of various P forms in shallow groundwater in upland-riparian-stream continuums in agricultural areas. This study focuses on P transfer in shallow groundwater within and between the riparian zone and hyporheic zone in Spencer Creek, and considers the effects on P transfer caused by flooding from the drawdown of the upstream Valens Reservoir, Hamilton, southern Ontario. A series of piezometer nests (each nest consisted of three piezometers at the depths of 50 cm, 100 cm and 150 cm) and corresponding wells were installed at the upland-riparian interface, riparian corridor and riparian-stream interface along two transects with different vegetation covers. Water chemistry was measured in groundwater and the stream from May 10 to October 20, 2006. The chemistry parameters included total phosphorus (TP), soluble reactive phosphorus (SRP), soil-extractable phosphorus, dissolved oxygen (DO), total dissolved solids (TDS) and pH. Water table elevation, hydraulic head and saturated hydraulic conductivity (K_{sat}) were measured at each location along the transects.

Groundwater TP concentrations at the two transects ranged from 11.8 μgL^{-1} to 2685.6 μgL^{-1} with an average (mean ± standard error) of 365.2 ± 25.9 μgL^{-1} and from 12.7 μgL^{-1} to 8485.5 μgL^{-1} with an average of 742.4 ± 109.0 μgL^{-1} during baseflow and flood conditions, respectively. The groundwater SRP concentrations ranged from 3.6 μgL^{-1} to 417.9 μgL^{-1} with an average of 44.0 ± 3.8 μgL^{-1} and from 2.1 μgL^{-1} to 280.4 μgL^{-1} with an average of 33.9 ± 5.2 μgL^{-1} during baseflow and flood conditions, respectively. For both transects, mean TP and SRP concentrations in piezometers tended

to increase with increasing distance from the field during both baseflow and flood. This trend was poorly correlated with depth. SRP concentrations in groundwater did not show the same decreasing trend by depth. The flood caused by the drawdown of upstream Valens Reservoir significantly increased TP concentrations ($p=0.002$) but significantly decreased SRP concentrations ($p<0.001$) in groundwater in the riparian zone. The TP concentrations at three depths were all significantly lower at transect T1 (predominantly grass cover) than T4 (predominantly forest cover) (50 cm: $p=0.001$; 100 cm: $p<0.001$; 150 cm: $p<0.001$). The SRP concentrations at 50 cm were significantly lower at transect T1 than T4 ($p=0.003$).

There is little evidence to indicate groundwater flow from the field to the stream; rather it appeared to flow laterally from T1 to T4, across the riparian zone. The estimated lateral P flux in shallow groundwater were 2218.2 and 122845.0 $\mu gm^{-2}d^{-1}$ for TP and 1295.1 and -57.1 $\mu gm^{-2}d^{-1}$ for SRP during baseflow and flood conditions, respectively.

Groundwater recharge zones characterized by zero or negative vertical hydraulic gradient (VHG's) were identified in the hyporheic zone along the two transects. The TP and SRP loads estimated from the stream channel toward the hyporheic zone at two transects ranged from 774.7 to 233005.6 $\mu gm^{-2}d^{-1}$ and from 114.8 to 18381.2 $\mu gm^{-2}d^{-1}$ during the study period. The flood may substantially increase TP and SRP loading because the VHG's are increased at the riparian-stream interface.

The results of this study suggested that hydrology was a dominant factor regulating groundwater P transfer within the upland-riparian-stream continuum while other factors such as vegetation type and substrate characteristics were less important. The effectiveness of riparian zones to reduce P transport in groundwater from agricultural fields to adjacent streams will depend highly on site-specific hydrological regimes.

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Chapter 1 INTRODUCTION

1.1 Problem Statement

Phosphorus (P) is the limiting nutrient for the eutrophication of freshwater (Gardiner and Miller, 2004). This nutrient promotes primary production which leads to algal blooms and taste and odor problems (Yli-Halla *et al*., 1995; Lyons *et al*., 1998). The main sources of phosphorus to aquatic systems are from agricultural fertilizers, industrial contaminants, atmospheric deposition and sewage treatment effluent (Goen and Notodarmojo, 1995). While a range of source controls are used to limit P concentrations in point sources, diffuse sources of P particularly from agriculture remain problematic (Mainstone and Parr, 2002; Sharpley *et al*., 2002).

Phosphorus loss from agricultural lands via surface runoff is strongly controlled by the magnitude and frequency of rainfall and snowmelt events along surface and subsurface hydrologic flowpaths (Heathwaite and Dils, 2000). While groundwater typically has low phosphorus concentrations relative to surface runoff, it represents a flow path which is often connected to most streams all year (Heathwaite and Dils, 2000). Groundwater has recently been recognized as an important pathway for P transport to streams from agricultural land (Vellids *et al*., 2003; Banaszuk *et al*., 2005; and Hoffmann *et al*., 2006) but more research is required to better understand the physical, chemical and biological processes that govern the rate and magnitude of P transport in shallow groundwater (Flores-Lopez *et al*., 2005). Such information is required to model and predict P losses from watersheds (Ulen *et al*., 2001).

Riparian zones are important areas for regulating soluble reactive phosphorus (SRP) in shallow groundwater (Carlyle and Hill, 2001). A significant exchange of nutrients can occur in the hyporheic zone which is defined as a spatially fluctuating ecotone between the stream and the groundwater (Boulton *et al*., 1998). Previous studies involving the transfer of agriculturally-derived groundwater phosphorus in riparian areas have focused mostly on the upland-riparian interface (Osborne and Kovacic, 1993; Mander *et al*., 1997; Snyder *et al*., 1998; Vellids *et al*., 2003; Banaszuk *et al*., 2005; Hoffmann *et al*., 2006) and little attention has been paid to P transport processes in shallow groundwater at the riparian-stream interface (Stainton, 2000). These studies reported high variability in the transfer of phosphorus in riparian zones and that riparian zones can act as either sinks or sources of phosphorus to receiving waters depending on the site specific hydrogeologic settings (Table 1-1). In addition, little is known about the role of flood-associated phosphorus as a potential source of phosphorus to riparian zones (Thoms *et al*., 2000; Walling *et al*., 2000; Steiger and Gurnell, 2002). Although some studies have examined groundwater P dynamics in agricultural areas in southern Ontario, these studies focused either only on riparian zones (Carlye and Hill, 2001) or only on hyporheic zones (Stainton, 2000). At present, there is a lack of knowledge regarding the effects of variable hydrological regimes on the transfer of various P forms in shallow groundwater along upland-riparian-stream and downstream gradients in riparian wetlands in agricultural areas in southern Ontario (Stainton, 2000). Collectively, these findings raise questions regarding the effects of hydrological regimes (baseflow and flood) on groundwater phosphorus transfer in riparian wetlands in agricultural landscapes. Accordingly, this study examines the effect of variable hydrologic regimes (seasonal as well as a controlled flood) on groundwater P transfer in an agricultural riparian wetland (riparian and hyporheic zones) adjacent to Spencer Creek in southern Ontario.

1.2 Literature Review

1.2.1 Nutrient regulation function of riparian zones in agricultural areas

Stream riparian zones play an important role in regulating energy and nutrient exchange between terrestrial and aquatic systems (Carlyle and Hill, 2001). They provide a range of important hydrological, biogeochemical and ecological functions which include transfer of water and dissolved and particulate matter across both longitudinal and transverse gradients in stream riparian zones. In general, riparian areas are effective in removing nutrients from surface runoff (Daniels and Gilliam, 1996). Kuusemets *et al*. (2001) suggested that riparian buffer zones have a high efficiency for P removal when input concentration exceeded 0.15 mg L^{-1}. However, on a broader temporal and spatial scale, the ability of riparian zones to reduce nutrient loads has been questioned (Norris, 1993) due to the rates of nutrient (e.g., nitrogen and phosphorus) assimilation which vary greatly due to the high temporal and spatial variability in hydrological, physical and biogeochemical settings (Price *et al*., 2005). Studies regarding groundwater phosphorus in agricultural areas (Table 1-1) showed that the differences between SRP concentrations entering riparian zones and those released ranged by -39 to 9 fold. Equivalent values for TP were by -1.1 to 1.5 fold. This indicates SRP concentrations are more variable than TP concentrations (Hoffmann *et al*., 2006). Several studies show riparian zones can act as either sinks or sources of phosphorus to receiving waters depending on individual hydrogeologic settings. Broader forest (>100 m) or mixed forest-grass strips appear to be more efficient in phosphorus removal for shallow lateral groundwater flows (Table 1-1). The observed variation results from the complexity of phosphorus inputs, transformation, release/uptake and mobilization/immobilization across riparian zones.

1.2.2 Phosphorus forms and cycle

Phosphorus is a limiting nutrient in most freshwater systems (Mitsch and Gosselink, 1993) and it is found in two operationally-defined forms: dissolved phosphorus (DP) and particulate phosphorus (PP) (Reddy *et al*., 1999). The sum of both forms comprises total phosphorus (TP) (Stainton, 2000). Both DP and PP consist of inorganic and organic fractions. The former is dissolved, while the latter is typically bound to soil particles and colloids or confined to the geochemical matrices of solids. Orthophosphate is the primary form of inorganic phosphorus in freshwater systems. Depending upon the pH, orthophosphate is found in various phosphate forms which include PO_4^{3-}, HPO_4^{2-} and H_2PO_4 (Mitsch and Gosselink, 1993). Inorganic phosphorus minerals are typically present as aluminum and iron phosphates in acidic soils, while calcium phosphates tend

Table 1-1 Summary of riparian studies on groundwater phosphorus dynamics in agricultural landscapes

Reference	Location	Adjacent Land Use	Riparian Zone Form	P Form	A ($\mu g\ L^{-1}$)	B ($\mu g\ L^{-1}$)	C ($\mu g\ L^{-1}$)	Hydrology and Soil Setting	Groundwater Flow Path
Jordan *et al.*, 1993	Maryland, USA	Agriculture	55m deciduous forest	PO_4	2*	80*	N/A	Shallow aquifer, clay at 1.2-2.5m	Shallow lateral
Osborne and Kovacic, 1993	Illinois, USA	Agriculture	16m deciduous forest	TP	58	122	N/A	Shallow aquifer over dense basal till at 0.6 to 1.3m	Shallow lateral
				DP	13	66	N/A		
			39m grass	TP	92*	91*	N/A		
				DP	20*	40*	N/A		
			unbuffered riparian crop site	TP	N/A	69	N/A		
				DP	N/A	36	N/A		
Akhmetieva, 1994	Moscow, Russia	Agriculture	Pine and deciduous forest	PO_4	20	10	N/A	Shallow aquifer sandy soils above dense loam	Shallow lateral
				PO_4	40	20	N/A		
Mander *et al.*, 1997	Southern Estonia	Agriculture	50m deciduous forest	TP	100*	100*	70	Glaciofluvial sands and gravels underlain by periodic layers of clay at 2m	Shallow lateral
			200m grass/deciduous forest	TP	300*	140*	40		N/A
			120m fen	TP	300*	310*	70		Upward flow
			2.5m grass	TP	200*	80*	80		N/A
Snyder *et al.*, 1998	Virginia, USA	Agriculture	130m forest/wetland	PO_4	20	2-4	20	Shallow sandy aquifer over confining layer	Shallow lateral

A is shallow groundwater inputs to riparian zones from upland; B is shallow groundwater discharged from riparian zones; C is concentrations in stream water. * Concentration estimated from figures in published paper; + Concentration at the riparian-stream interface (after Stainton, 2000).

Table 1-1 (continued) Summary of riparian studies on groundwater phosphorus dynamics in agricultural landscapes

Reference	Location	Adjacent Land Use	Riparian Zone Form	P Form	A (µg/L)	B	C	Hydrogeologic Setting	Groundwater Flow Path
Stainton, 2000	Waterloo, Canada	Agriculture	Unbuffered row crops to stream channel	TP	N/A	185+	49	Glaciofluvial sands and gravels underlain by hardpan creek till	Shallow lateral
				SRP	N/A	33+	21		
			162m mixed herbaceous/deciduous	TP	N/A	207+	66		
				SRP	N/A	16+	16		
Carlyle and Hill, 2001	Toronto, Canada	Agriculture	100-200m mixed conifer-deciduous forest/grass	SRP	6 -120	25 -80	4 -23	Sandy aquifer underlain by clay and silts	2-4m deep lateral
Vellids *et al.*, 2003	Tifton, USA	Agriculture	three zones from the field: 8m Bermuda grass, 20m slash pine and 10m mixed the trees and hardwoods	TP	1400	2050	N/A	Loamy sand underlain by a plinthic soil layer at 1-1.5m	0.1-2m laeral
				SRP	155	158	N/A		
Banaszuk *et al.*, 2005	Bialystok, Polland	Agriculture	100m mire with a 20m strip of tree and meadow	SRP	1400	1100	200	Shallow aquifer underlain by sandy clay	Shallow lateral
Hoffmann *et al.*, 2006	Jutland, Denmark	Agriculture	20-25 m meadow	SRP	7	18	N/A	30-50cm sandy sapric and hemic peat underlain by 1-2m medium grained sand with gravel and pebbles	Shallow lateral

A is shallow groundwater inputs to riparian zones from upland; B is shallow groundwater discharged from riparian zones; C is concentrations in stream water. * Concentration estimated from figures in published paper; + Concentration at the riparian-stream interface (after Stainton, 2000).

to dominate in alkaline soils (Gardiner and Miller, 2004; Sylvia *et al*., 2005). The inorganic component of soluble reactive phosphorus (SRP) is generally considered to be the most bio-available form of phosphorus because plants utilize this form efficiently. Soluble reactive phosphorus originates from the release of soil phosphorus, chemical fertilizer leaching and decomposition of organic matter (Yli-Halla *et al*., 1995). Phosphorus forms are in a dynamic equilibrium between soluble and particulate forms in riparian zones which is driven primarily by the redox condition of soil solutions, grain size, competitor ions and microbial activity (Qualls and Richardson, 1995; Mander *et al*., 1997).

1.2.3 Phosphorus in riparian zones

The mechanisms of phosphorus retention in riparian wetlands include both biotic and abiotic processes regulated by the physical, chemical, and biological features of the system (Reddy *et al*., 1999). Biological mechanisms include uptake by soil microorganisms and plants, as well as accretion of peat. Geochemical processes involve adsorption by Al and Fe oxides and hydroxides and the precipitation of Al, Fe, and Ca phosphates (Walbridge and Struthers, 1993; Qualls and Richardson, 1995).

Adsorption and sedimentation are the dominant processes of phosphorus retention in wetlands (Bridgham *et al*., 2001). Cations such as Fe^{3+}, Al^{3+}, and Ca^{2+} generally enhance phosphorus adsorption to soil particles (Mitsch and Gosselink, 1993). Reddy *et al*. (1998) observed significant correlations between P sorption maximum and extractable Fe and Al concentrations, suggesting that amorphous forms of Fe and Al tend to control P sorption on soil particles due to their greater reactive surface area per unit soil volume (Walbridge and Struthers, 1993). Under anaerobic conditions, PO_4^{3-} is released when Fe^{3+} is reduced to Fe^{2+} (Gilliam *et al*., 1999). Carlyle and Hill (2001) showed that higher groundwater SRP concentrations in a riparian zone were associated with low DO and high Fe^{2+} concentrations in buried channel sediments near the river bank.

Vegetation cover may also impact P retention in riparian zones. Osborne and Kovacic (1993) reported that grassed strips had a higher efficiency for DP and TP assimilation in shallow groundwater compared to forested strips. Takatert *et al*. (1999) observed lower concentrations of phosphate in groundwater in forested compared to meadow covered floodplains. These observations are attributed to more efficient uptake by woody species than herbaceous plants. Estimated annual P storage in woody biomass in forested riparian zones ranges from 0.2-1.8 kg/ha but periodic harvest is needed to maintain this function when the systems reach a balanced state (Walbridge and Struthers, 1993). However, high phosphate inputs may rapidly saturate soil microbial P pools and about 40% of microbial biomass is released as soluble P due to cell death and subsequent mineralization for a wide category of wetland soils (Walbridge *et al*., 1991; Bridgham *et al*., 2001). Plant assimilation is not a permanent P sink in riparian zones.

1.2.4 Hydrological effects on phosphorus transfer in riparian zones

Hydrologic regime is also expected to affect P transfer through its control on (a) P mobility due to soil redox status and (b) P loading and transfer between riparian areas and adjacent areas. Since groundwater levels influence the aerobic/anaerobic status of soils, hydrological processes are also an important factor in regulating groundwater phosphorus dynamics than other considerations such as vegetation composition or substrate type (Takatert *et al*., 1999). Flores-Lopez et al. (2005) reported an increase of 20 μgL^{-1} in SRP when the groundwater table rose in response to a high creek flow. Higher SRP concentrations were observed in riparian shallow groundwater in the Catskill Mountains, USA when the pore water in spodosol soils received inputs of dairy manure. They reported a high potential for lateral P transport in shallow groundwater below the Bh horizon when the water table was low (Villapando and Graetz, 2001).

Phosphorus dynamics may also be affected by loading from adjacent areas. The riparian zones downstream from reservoirs are subject to flooding and possibly P transfer from upstream sources. Reservoir drawdown has a significant impact on downstream transport of sediment and phosphorus (Shantz *et al*., 2004). During drawdown, reservoirs can release resuspended sediments and associated contaminants to downstream environments (Jansson and Erlinsson, 2000). The fine-grained sediments (< 63 µm) are the most important fraction for contaminant adsorption and transport, due to their relatively large surface area and geochemical composition (Stone and English, 1993). The concentration and load of suspended solids and total phosphorus exported from reservoir are closely related to the discharge of drawdown (Shantz *et al*., 2004). Riparian zones have an ability to trap and buffer longitudinal or downstream transfer of sediment and associated nutrient during overbank flooding (Naiman and Decamps, 1997; Steiger and Gurnell, 2002) and have been demonstrated to be important sinks for sediment associated phosphorus (Walling *et al*., 2000). The rate of TP accumulation at riparian zones is related to the P content of deposited sediment and the rate of sediment accretion which varies with floodplain morphology (Walling *et al*., 2000). Thoms *et al*. (2000) found that concentrations of TP in floodplain deposited sediments increased with increasing distance from the river channel while the higher rates of sediment deposition were observed nearer to the channel. These studies show the importance of geomorphological aspects of riparian zones for accumulation and distribution of sediment-borne phosphorus.

1.2.5 The role of hyporheic zones in P transfer

The stream hyporheic zone is the subsurface region characterized by mixing of stream water and groundwater adjacent to the stream channel (Triska *et al*., 1993). Boulton et al. (1998) defined the hyporheic zone as a spatially fluctuating ecotone between the surface stream and the deep groundwater where important ecological processes and their products are influenced at a number of scales by water movement, permeability, substrate particle size, resident biota, and the physiochemical features of the overlying stream and adjacent aquifers. The size of the hyporheic zone and extent of interaction between stream water and groundwater vary in response to geological lithology, groundwater flux magnitude, riverbed gradients and precipitation (Hill, 1997). Riparian groundwater can influence surface water P chemistry through advective P exchange processes between the riparian zone and adjacent stream channels (Surridge *et al*., 2006).

The hyporheic zone plays an important role in the processes of nutrient exchange between stream water and groundwater (Triska *et al*., 1989). These processes are controlled by hydraulic gradient and streambed porosity that may generate interstitial flow pathways in the hyporheic zone (Boulton *et al*.,1998). Variations in redox status of flooded sediments in floodplains or streambeds may mobilize/immobilize iron- or manganese-bound P which is controlled by redox thresholds, and subsequently influence P concentration in interstitial water (Boulton *et al*., 1998; Surridge *et al*., 2006). Higher SRP concentrations were observed in hyporheic zones in agricultural areas than in surface water (Carlyle and Hill, 2001; Stainton, 2000), indicating that the hyporheic zone may be a substantial source of SRP to adjacent streams. Statinton (2000) also reported higher TP concentrations in hyporheic zones than in stream water near agricultural land uses in the Laurel Creek Watershed in Southern Ontario.

1.3 Research Question

A review of literature on phosphorus transfer and transformation dynamics in riparian zones has identified a number of research needs. Hydrology is generally considered to be important in governing groundwater phosphorus transfer from uplands to receiving streams. However, limited knowledge is currently available regarding the degree and extent to which hydrological processes govern phosphorus concentrations in both intra-riparian zones and their interfaces with uplands and streams. No study has been conducted to examine the impacts of reservoir drawdown on groundwater phosphorus transfer/distribution in riparian zones in agricultural areas despite the fact that flood associated phosphorus has been recognized as a major phosphorus source to riparian zones (Thoms *et al*., 2000; Walling *et al*., 2000; Steiger and Gurnell, 2002). Such information is necessary to better understand the lack of effectiveness of riparian zones to reduce TP and SRP concentrations in groundwater (Osborne and Kovacic, 1993; Vellids *et al*., 2003).

Therefore, there is a need to better understand the role of groundwater in phosphorus transfer processes influenced by riparian hydrology in the context of agricultural landscapes. This study addresses the following broad research questions:

(1) What is the effect of hydrological regimes (baseflow and flood caused by drawdown of upstream reservoir) on phosphorus transfer in shallow groundwater across riparian zones adjacent to agricultural areas?

(2) What is the role of riparian zones (sink or source) for groundwater phosphorus transfer between agricultural fields and adjacent streams?

1.4 Objective

The main focus of this study is to evaluate the effects of hydrological variations on groundwater phosphorus transfer in a riparian wetland adjacent to an agricultural field in southern Ontario. This will be accomplished by characterizing groundwater phosphorus transfer patterns at the upland-riparian interface, riparian corridor, and riparian-stream interface within and between two transects with different soil and vegetation features under baseflow and flood conditions resulting directly from the drawdown of an upstream reservoir.

The specific objectives of this study are to:

(1) Assess the spatial and temporal variability of TP and SRP in shallow groundwater as a function of vegetation cover, distance from an agricultural field, and depth under ground in response to flow regimes in a riparian wetland adjacent to Spencer Creek;

(2) Estimate the rates and magnitudes of phosphorus exchange via groundwater pathways in the hyporheic zone of Spencer Creek.

It is hypothesized that:

(1) Groundwater TP and SRP concentrations will decrease with increasing distance from the field edge and with depth due to accessibility to P input via both surface and subsurface pathways and P loss for soil adsorption and vegetation uptake along the field to the stream gradient and from upper to lower soil horizons within riparian zones. The former trend with distance may change during flood because of P inputs by overbank flow. The latter trend with depth may be disturbed by highly variable soil profile even within small areas.

(2) Groundwater TP and SRP concentrations during flood are higher than baseflow, which result from phosphorus load of overbank flow in both dissolved and particulate

forms. Considering particulate P is the dominant form contributed by drawdown, the flood effect will be more significant for particulate P than SRP.

(3) Groundwater TP and SRP concentrations are higher at transect T4 than T1. Observable differences exist between landscape positions (transect T1 and T4) due to different vegetation types (e.g., herbaceous predominant cover versus forest predominant cover), which may result in differences mainly in organic matter content at the upper layer soils and thereby influence P retention capacity. In addition, herbaceous species along transect T1 will likely have higher phosphorus uptake efficiencies for phosphorus in shallow groundwater.

Chapter 2 METHODOLOGY

2.1 Experimental Design

A conceptual model of the physical and biogeochemical processes governing phosphorus transfer in an upland-riparian wetland-stream system is illustrated in Figure 2-1. Surface runoff from uplands during rainfall events contributes phosphorus to the riparian wetland, mainly through deposition of suspended sediment particles (Heathwaite and Johnes, 1996). Studies have shown phosphorus concentrations in groundwater flowing into riparian zones are generally higher than those in groundwater discharge from riparian zones (Tables 1-1 and 1-2), indicating that subsurface pathways exist for agriculturally-derived phosphorus entering riparian zones. Riverine wetland soils can act as either sinks or periodic internal sources of P depending on hydrological and biochemical settings (Porter and Sanchez, 1992; Reddy *et al*., 1999; Bridgham *et al*., 2001; Casey and Klaine, 2001). Overbank flow during flood events represents another phosphorus contributor to riparian zones through the deposition of sediment-associated phosphorus (Walling *et al*., 2000).

Vegetation uptake and release is an important process of phosphorus cycling in wetlands (Mitsch and Gosselink, 1993). Plants including trees and herbaceous species absorb soil pore-water phosphorus through their roots and rhizomes, particularly during the growing season (Reddy *et al*., 1999) and return phosphorus to the water column and the soil either by leaching or decomposition of detrital materials (Reddy *et al*., 1995).

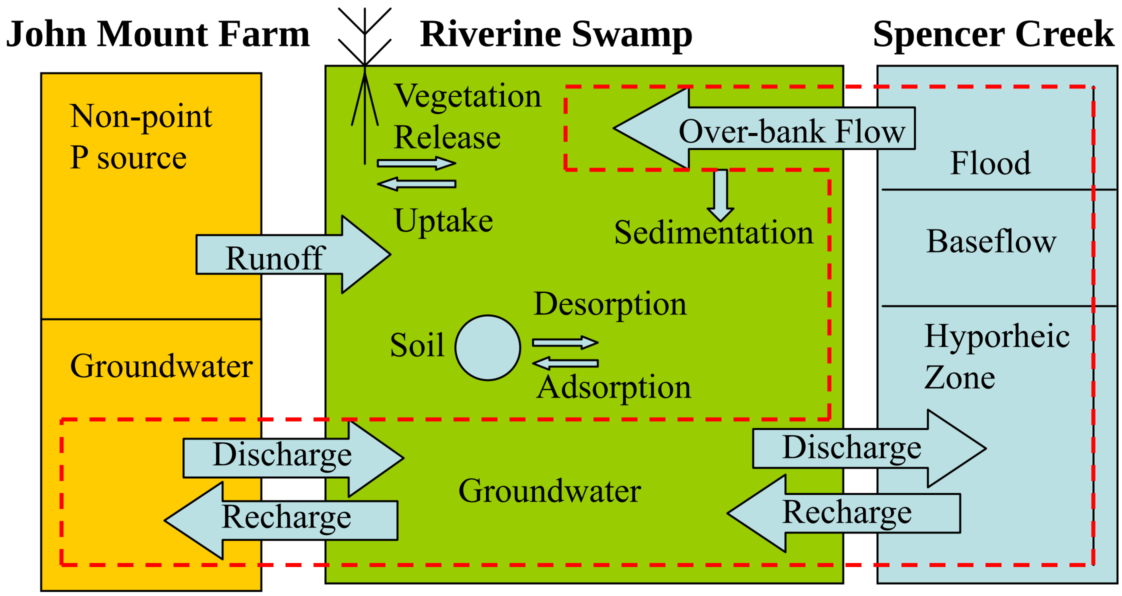


Figure 2-1 Conceptual model of phosphorus transfer in an upland-riparian wetland-stream system (dashed lines highlight the research scope)

Phosphorus can be fixed as mineral phosphates in wetland soil and also be adsorbed on to silicate clay, ferric and aluminum hydroxides, and organic matter particularly in peat soils (Mitsch and Gosselink, 1993; Qualls and Richardson, 1995; Reddy *et al*., 1998, 1999; Gardiner and Miller, 2004). Sorbed phosphates on the surface of soil particles can be released by the exchange of anions under anaerobic conditions (e.g., flooded) (Mitsch and Gosselink, 1993). This soluble phosphate will be re-precipitated when the soil is drained (Gardiner and Miller, 2004). However, some portion of phosphorus adsorption is less reversible due to the effect of hysteresis (Reddy *et al*., 1999).

This study specifically focuses on groundwater pathways for phosphorus transfer within and between the riparian zone and hyporheic zone during baseflow and flood conditions caused by reservoir drawdown. This study is designed to evaluate the variability of groundwater phosphorus concentrations at different depths in the upland-riparian interface, riparian corridor and riparian-stream interface and between landscape positions within the riparian zone under changing hydrological regimes. The study area therefore has been instrumented with piezometer nests (each nest includes 3 piezometers at depths of 50, 100 and 150 cm and a corresponding well) at distances of 6, 19, 33, 48 and 50 from the field edge to stream channel along two randomly selected transects to examine spatial and temporal variations of phosphorus concentrations, hydraulic head and groundwater elevation under changing flow conditions throughout the study period (Figure 2-2). Since there are three treatment variables or factors (distance from the agricultural field, depth under surface and flow condition), each treatment variable has three or five categories with independent measurements that are not matched (each groundwater sample is unique to the set of treatment), the data from this experimental design meet the criteria for using a factorial ANOVA (analysis of variance) which may be applied to test the effect of factors and determine whether there are significant differences in population means or an interaction between any or all of the factors (Holmes *et al*., 2006).

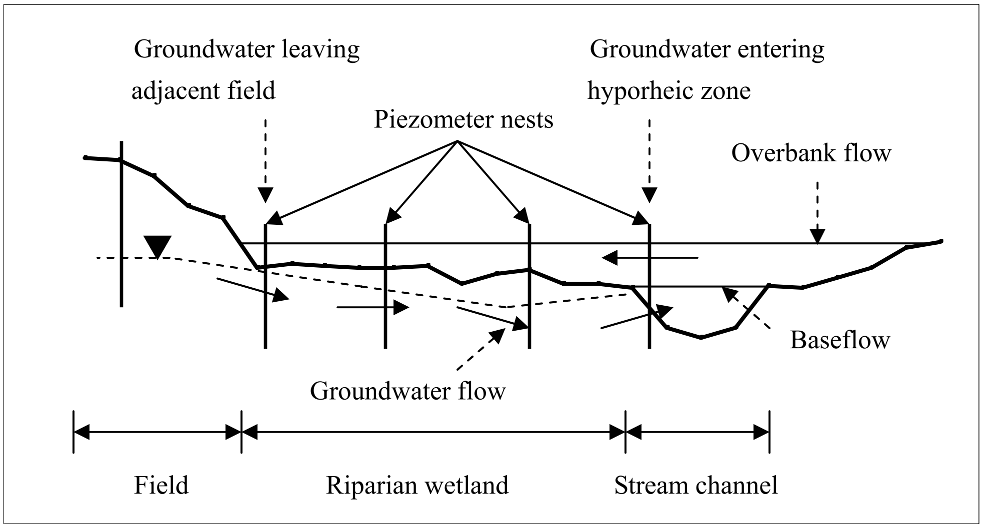


Figure 2-2 Sample transect from the field to the stream system

2.2 Study Site Description

The study site is located in a riparian zone on the north side of Spencer Creek adjacent to the John Mount Farm, near Hamilton, Ontario (Figure 2-3). Conventional tillage practices and crop rotation have been used at the site for decades. The riparian wetland is classified as riverine swamp according to the Canadian Wetland Classification System (National Wetlands Working Group, 1997). The site is approximately 30-50 m wide from the field to the stream channel. Flow in the channel is controlled by the Valens Reservoir located approximately one km upstream. The riparian wetland is characterized by emergent vegetation including herbaceous plants and woody species, supported by seasonal flooding derived from Spencer Creek caused by the annual drawdown of Valens Reservoir in the fall. The site slopes toward the stream from northeast to southwest. A detailed description of the study site is listed in Table 2-1~2.

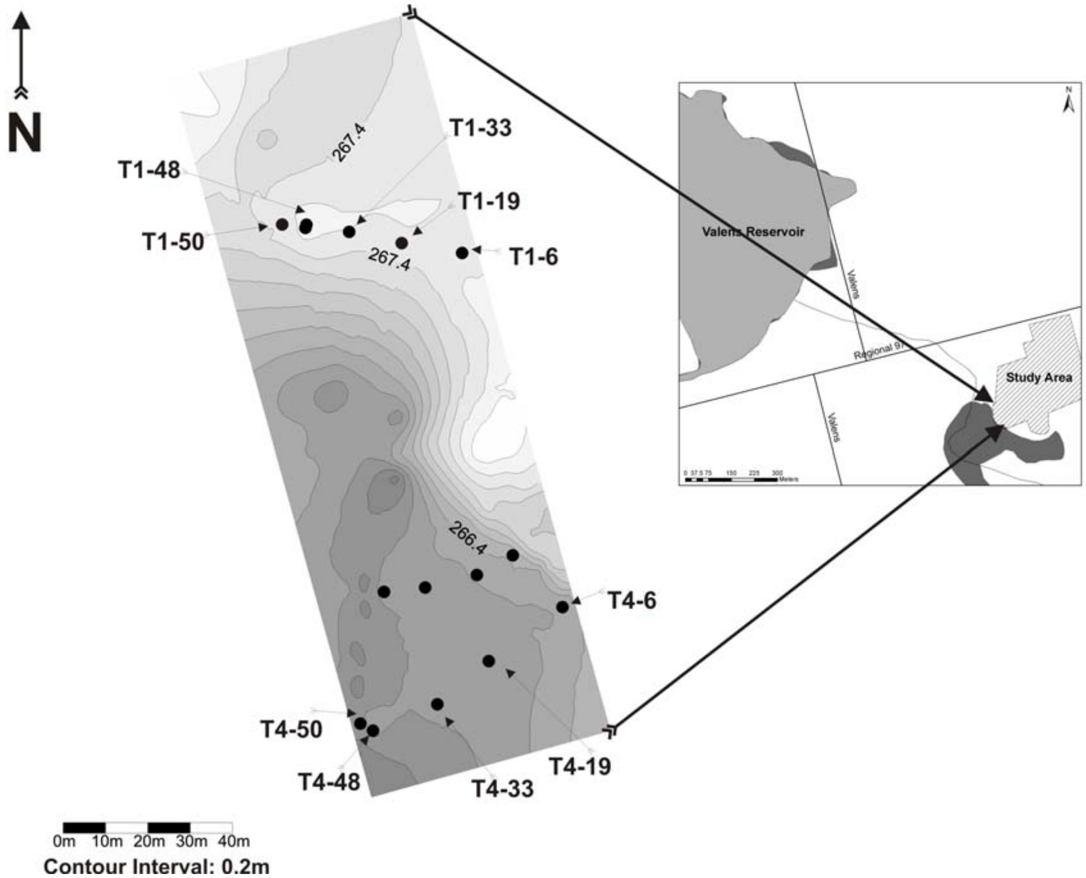


Figure 2-3 Study site in Spencer Creek watershed showing two study transects with elevation contour lines (black dots indicate piezometer nests) and relative location of upstream Valens Reservoir

Spencer Creek is one of two surface inflows to Beverly Swamp (43º22' N, 80º07' W) which is located in the northern region of the Spencer Creek watershed that drains into Cootes Paradise marsh on west end of Lake Ontario (Kaufman *et al*., 2005). Beverly Swamp is one of the largest temperate wetlands in Southern Ontario which covers an area of 20 km^2 with an elevation between 265 to 270 m (Munro *et al*., 2000). Most of Beverly Swamp is underlain at shallow depths by Guelph dolomite. Soil parent materials in this area are calcareous and have high base content (Presant *et al*., 1965). Soils in Beverly

Swamp are classified as Beverly series in the Organic Group, which have developed on level to very gently sloping areas of lacustrine silty clay loam and silty clay. The Ah horizons are usually somewhat deeper and darker, while the Ae and B horizons display some reddish-brown mottling (Presant *et al*., 1965). The soils along Spencer Creek are characterized by a combination of woody peat and incompletely decomposed organic matters (Warren *et al*., 2001). The peat depth within Beverly Swamp varies from 0.3 to 1.5 m underlain by a relatively impermeable marl layer between 0.5 to 1 m thick with an average organic content of 75% (Galloway and Branfireun, 2004). Beneath this marl layer lies sand, gravel and silty sand layers in sequence (Warren *et al*., 2001). The marl layer at about 1 m below the surface effectively separates the surface soil hydrology from that of underlying sand (Munro *et al*., 2000). The vegetation cover in Beverly Swamp is a mixed conifer-deciduous forest with heterogeneous species distribution throughout the swamp. The white cedar (*Thuja occidentalis*), tamarack (*Larix laricina*) and aspen (*Populus tremuloides*) are dominant in the northern region, while red maple (*Axwe rubrum*), white birch (*Betula papyifera*) and trembling aspen (*Populus tremuloides*) are the most common species in the central and southern regions (Warren *et al*., 2001; Galloway and Branfireun, 2004).

Soils at the study site differed from the soils described above. Field studies provided additional information on soil and vegetation in the present study site. Transect T1 passes through grass for the first 15 m from the field edge then small trees with some shorter grass beneath them to the stream bank. The vegetation becomes observably abundant during growth season. There is less exposed soil along T1 and the soil color was brown throughout the study period (see Appendix 11). In contrast, transect T4 has leafy vegetation of tall and short trees (almost no grass) for its entire length. Some exposed soils are observed near the field edge. The soil color is brown-grey compared to T1 and turns darker during wet conditions. Organic topsoils are present to a depth of approximately 0.3m depth. Below this, mineral soils are found (see Appendix 12). For example, at the site of T4-19 in the middle riparian zone, the soil profile is as follows: medium-fine sand (0.5-0.05 mm) with organic content of 0.82% at 50 cm underlain by 1 cm blue clay layer (likely leda clay); very fine sand (0.1-0.05 mm) with organic content of 0.54% at 100 cm; fine-very fine sand (0.25-0.05 mm) with organic content of 0.65% at 150 cm, while the site T4-48 in the riparian edge near the stream has a profile of mixture of slit and clay (0.05-0.002 mm) with organic content of 28.74% at 50 cm; fine-medium sand (0.1-0.5 mm) with organic content of 12.54% at 100 cm with some organic matter, root and wood chips; mixture of very fine sand and silt (0.1-0.002 mm) with organic content of 1.30% at 150 cm. The vegetation density at T4 does not change as much as T1, although the canopy gets denser during the growth season. Beverly Swamp is drained by two 2nd order streams, Spencer and Fletcher Creeks (Warren *et al.*, 2001), which join to form the larger Spencer Creek at a confluence in the centre of the swamp and continue in downstream region (Kaufman, 2005). Spencer Creek which is regulated by the Valens

Reservoir enters the swamp in its northwest side but subsequently breaks into several small unconfined channels and disappears underground for about 1 km before re-emerging upstream of the confluence (Galloway and Branfireun, 2004). This unconfined flow pattern facilitates extensive interaction between surface water and groundwater within the wetland (Warren et al., 2001). The annual drawdown of Valens Reservoir between September and November may change the water table by about 0.15 m throughout the entire wetland (Munro *et al*., 2000), which substantially influences the hydrology and water chemistry in the swamp (Warren *et al*., 2001; Galloway and Branfireun, 2004; Kaufman *et al*., 2005).

2.3 Hydrology and Chemistry

2.3.1 Hydrology

In this study, two transects (T1, T4) located approximately 80 m apart in the riparian zone, were instrumented with four piezometer nests each and corresponding monitoring wells (Figure 2-4). The piezometer nests are located along each transect from field edge to riparian edge near to the stream bank at distances of 6 m, 19 m, 33 m, and 48 m relative to the field edge. Each piezometer nest is paired with a monitoring well that included three individual piezometers at depths of 50 cm, 100 cm, and 150 cm. Two additional drive-point piezometers were installed in the stream at the depths of 50 cm and 100 cm in the hyporheic zone for each transect (Tables 2-1 and 2-2). Piezometers (except drive-point piezometers) were constructed with 3 cm inside diameter (ID) PVC pressure

Table 2-1 Location and description of piezometer nests and wells (Transect T1)

Site	Site Location	Site Character	Piezometer Location	Distance (m)	Depth (cm)	Screen Depth (cm)	Intra nest Distance (cm)	Baseflow Stream Width (cm)	Piezometer Label
Transect 1 (T1)	Upstream of the reach of Spencer Creek when it enters the riparian zone	Grass and shrubs	Field-riparian interface	6	69.5	52	A-B: 23	-	T1-6-50
					114	96.5	B-C: 23.5	-	T1-6-100
					147.5	140	C-A: 46	-	T1-6-150
					156	156	W-A: 56	-	T1-6-W
			Middle riparian (beginning)	19	67.5	50	A-B: 28	-	T1-19-50
					121	103.5	B-C: 25.5	-	T1-19-100
					160	152.5	C-A: 53.5	-	T1-19-150
					175	175	W-C: 28	-	T1-19-W
			Middle riparian (end)	33	67.5	50	A-B: 35.5	-	T1-33-50
					107.5	90	B-C: 41.5	-	T1-33-100
					160	152.5	C-A: 77	-	T1-33-150
					176	176	W-C: 42.5	-	T1-33-W
			Riparian-stream interface	48	70	52.5	A-B: 28.5	-	T1-48-50
					119	101.5	B-C: 50	-	T1-48-100
					161	153.5	C-A: 36	-	T1-48-150
					168.5	168.5	W-B: 28.5	-	T1-48-W
			Hyporheic zone	50	43	34.5	A-B: 34	180	T1-50-50
					101	92.5		176	T1-50-100

* 'A', 'B', and 'C' refers to the piezometer at depths of 50 cm, 100 cm, and 150 cm, respectively; 'W' refers to the adjacent well.
** Distance refers to the distance from the field edge.
*** Depth refers to the depth under ground surface; screen depth means the true screen depth under ground surface.

Table 2-2 Location and description of piezometer nests and wells (Transect T4)

Site	Site Location	Site Character	Piezometer Location	Distance (m)	Depth (cm)	Screen Depth (cm)	Intra nest Distance (cm)	Baseflow Stream Width (cm)	Piezometer Label
Transect 4 (T4)	Downstream of the reach of Spencer Creek when it enters the riparian zone	Forest	Field-riparian interface	6	53	45.5	A-B: 14.5	-	T4-6-50
					92	84.5	B-C: 18	-	T4-6-100
					143	135.5	C-A: 17	-	T4-6-150
					169	169	W-A: 82.5	-	T4-6-W
			Middle riparian (beginning)	19	61.5	54	A-B: 16.5	-	T4-19-50
					108	100.5	B-C: 19	-	T4-19-100
					151	143.5	C-A: 19	-	T4-19-150
					164	161	W-A: 22.5	-	T4-19-W
			Middle riparian (end)	33	56	48.5	A-B: 31	-	T4-33-50
					100	92.5	B-C: 21.5	-	T4-33-100
					133	125.5	C-A: 40.5	-	T4-33-150
					176	151	W-B: 38	-	T4-33-W
			Riparian-stream interface	48	46	38.5	A-B: 24.5	-	T4-48-50
					101.5	94	B-C: 26	-	T4-48-100
					147	139.5	C-A: 36	-	T4-48-150
					194	191	W-B: 24.5	-	T4-48-W
			Hyporheic zone	50	48	39.5	A-B: 27	322	T4-50-50
					98	89.5		296	T4-50-100

* ‘A’, ‘B’, and ‘C’ refers to the piezometer at depths of 50 cm, 100 cm, and 150 cm, respectively; ‘W’ refers to the adjacent well.

** Distance refers to the distance from the field edge.

*** Depth refers to the depth under ground surface; screen depth means the true screen depth under ground surface.

pipes. The pipe has a 15 cm slotted zone wrapped in plastic mesh and a 10 cm reservoir at the tip. Porous (60-90 μm) polyethylene filtered drive-point piezometers were constructed using stand-pipe tips 21.8 cm in length, set within a perforated 25 mm ID PVC body and connected with 25 mm ID PVC riser pipes. Monitoring wells were constructed using the same materials as piezometers but slotted along the entire length and wrapped in plastic mesh. A stainless steel auger was used to drill a hole to depth and piezometers were inserted manually. All piezometers were purged 24 hours after installation to verify seepage into the piezometer tip and to remove fine sediments. Piezometers were purged three times before the first sampling occurred. The location and elevation of each well and piezometer were determined by surveying the study area with a Wild Leitz Total Station.

To completely but effectively examine groundwater P variation in response to changing flow conditions (Warren *et al*., 2001; Vellidis *et al*., 2003), hydraulic heads in piezometers and water elevations in wells were measured manually prior to water sampling with a portable water level sensor at weekly intervals for the first 6 weeks and biweekly for the remainder of the study period for baseflow and daily during the flood event, respectively. Monitoring frequency was increased for flood period because hydrologic parameters changed more rapidly during drawdown event. Saturated hydraulic conductivities (K_{sat}) of stream sediments were measured using bail tests (Hvorslev, 1951). Groundwater was pumped from piezometers and water levels were measured as they rose back to at least 2/3 of their original levels. Hydraulic conductivity (K) was then calculated by,

$$K = \frac{r^2 \ln(L/R)}{2LT_o} \tag{1}$$

where K is the hydraulic conductivity (cms^{-1}), r is the internal radius of the pipe (cm), R is the external radius of the pipe (cm), L is the length of the slotted area of the pipe below the water table (cm), T_o is the basic time lag after the initiation of the test. T_o is determined by plotting a graph of time T on the X axis and log [(H-h)/(H-H_0)] on the Y axis. Here, H is the initial measured water level relative to a datum, h is the water level relative to the datum at time t, H_o is the water level relative to the datum immediately following pumping. For the expression log (H-h)/(H-H_0), a value of 0.37 is used to determine the appropriate time lag value T_o used in the calculating of K (Hvorslev, 1951).

Data collected from drive-point piezometers in the stream were used to calculate groundwater discharge/recharge in the hyporheic zone using Darcy's equation:

$$Q_L = -K(\Delta h/\Delta l)[1/2(w)]L \quad (2)$$

where K is the vertical hydraulic conductivity of the streambed sediments, Δh is the hydraulic head difference between the drive-point intake filter and the stream surface, Δl is the depth of the piezometer screen below the sediment-water interface of the stream, *w* is the stream width at the point of piezometer sampling and L is one meter of stream length in order to generate a per unit area discharge. The expression (Δh/Δl) is defined as the vertical hydraulic gradient (VHG's) which indicates the direction and magnitude of hydraulic potential between points in the hyporheic zone and the stream surface (Stainton, 2000). For the lateral groundwater flux at different depths along the cross-section between transect T1 and T4, K has the lowest hydraulic conductivity of the paired piezometers at two transects (e.g., T1-33-P50 and T4-33-P50), Δh is the hydraulic head difference between paired piezometers, Δl is the distance from T1 to T4, w is the riparian width from the field edge to the stream and L is the depth of groundwater layer. The expression ($\Delta h/\Delta l$) therefore represents the lateral hydraulic gradient (LHG's) which reflects the direction and magnitude of hydraulic potential between paired piezometers at two transects.

Instantaneous stream discharge was determined by using a Swoffer Model 2100 velocity meter connected to a propeller type velocity probe to determine the velocity of flow in 4 equal spaced points across the stream. The sum of the discharge times the areas of the four panels was used to calculate the discharge. Baseflow data for pre-drawdown was obtained using a Sigma portable velocity meter with a Doppler ultrasonic sensor (Allin, unpublished data, 2006).

2.3.2 Chemistry

Groundwater was collected from the piezometers using a battery-peristaltic operated pump (Portable Masterflex® Model 7533-30) with 5mm ID peristaltic tubing. Groundwater samples were collected in acid (20% H_2SO_4) washed, triple-rinsed polyethylene or glass bottles. The peristaltic tubing was rinsed by de-ionized water after the sample had been pumped from the piezometers to the bottles. The de-ionized water that remained in the tubing was removed by reversing the pumping direction after finishing each sampling. Approximately 5 ml of sample was pumped from the piezometers before collection in sample bottles in order to avoid dilution of successive samples. Groundwater samples were immediately stored in coolers then transported to the lab for analysis.

The parameters analyzed in shallow groundwater samples include total phosphorus (TP), soluble reactive phosphorus (SRP), total dissolved solids (TDS), dissolved oxygen

(DO), and pH. Concentrations of SRP and TP were analyzed with a single-channel colorimeter (Technicon® Autoanalyzer 28) and NAP analysis software, following the ammonium molybdate - stannous chloride procedure (Environment Canada, 1987). The detection limit of this method is 1 μg L^{-1}. Groundwater samples for TP were acidified by adding 1ml of 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 heating plates until about 2-3 ml sample remained. The digested samples were diluted up to 20 ml with de-ionized water and filtered through 0.45 μm membranes before the final analysis. The samples for SRP were filtered through 0.45 μm membrane filters immediately upon return to the lab then stored at 4°C in a refrigerator for analysis. Groundwater pH was determined in the lab using a calibrated conductivity meter (Accumet® Basic Fisher Scientific Model AB30) with a temperature display and a pH electrode (Accumet® Model AB15).

Dissolved oxygen (DO) was measured on site using a calibrated DO meter (ORION® Model 840) probe inserted into piezometer. Dissolved oxygen was collected daily during the flood event but every two days for baseflow condition because the equipment was not always available. Quality control and assurance for each parameter was conducted according to Standard Methods (APHA, 1995). Stream surface water grab samples were collected in the centroid of flow using acid-washed (20% H_2SO_4) washed triple-rinsed 125 ml polyethylene or glass bottles. Stream water was analyzed for total phosphorus (TP), soluble reactive phosphorus (SRP), total dissolved solids (TDS), dissolved oxygen (DO) and pH using the same equipment and QA/QC procedures described above.

2.4 Soil Chemistry

Three soil pits (30 cm deep) were dug along the two transects adjacent to corresponding piezometer nests at the field edge, middle riparian zone and riparian edge near the stream channel with distances of 2 m, 25 m, and 48 m from the field edge. Soil cores were collected prior to the drawdown of Valens Reservoir by inserting 10 cm PVC (5 cm inside diameter) cylinders laterally into individual soil horizons at the depths of 0-8 cm and 17-22 cm (i.e., centered on 5 and 20 cm) below the ground surface at each pit along two transects. Each site was sampled in triplicate. Core samples were immediately placed into polyethylene bags then put into a cooler in the field. The samples were stored at 4°C in the laboratory until analysis was performed.

The core samples were hand mixed till visually homogenized, removing roots and stones. The field moist sample was coned and quartered then a representative 5 g sub-sample was placed in 50 ml of distilled-deionized water. Samples were shaken for 1 h then passed through a 0.45 μm membrane filter then analyzed for water extractable phosphorus. Water-extractable phosphorus has been shown to be the method that most closely resembles the dissolved P in runoff (Pote *et al*., 1996). Soil extractable phosphorus was analyzed using the same equipment, software and procedure as groundwater SRP.

2.5 Statistical Analysis

Statistical analysises were conducted using SPSS software (Version 15.0, SPSS® Inc.). The analysis of variance (ANOVA) was chosen to examine controls over the spatial and temporal patterns in groundwater P concentrations at the study site. Normality test showed that groundwater quality data was not normaly distributed which was also found by previous studies (Takatert *et al*., 1999; Stainton, 2000). Natural logarithm transformations were applied for groundwater TP, SRP and DO data before using ANOVA (Holmes *et al*., 2006).

The effects of distance from the field (i.e., 6 m-the field edge; 19-33 m-the middle riparian zone; 48 m-the riparian edge; and 50 m-the hyporheic zone), depths below ground surface (50 cm, 100 cm, and 150 cm) and flow conditions (baseflow vs. flood) on groundwater TP and SRP were tested using three-way ANOVA. One-way ANOVA according to all distances at each depth or all depths at each distance was used to break down the interaction between distance and depth (personal communication, Erin Harvey, University of Waterloo Statistics Consulting Service, 2007), followed by Fisher's LSD multiple comparison test when the overall ANOVA was significant at $p < 0.05$. Differences in groundwater TP and SRP in the riparian zone (excluding the piezometers in the hyporheic zone) between transect T1 and T4 with different vegetation types (grass vs. forest) were evaluated by depth using the independent-samples T test (Holmes *et al*., 2006). Three-way ANOVA for all piezometers (in both riparian and hyporheic zones) was applied to check the effects of flow conditions, distance and depth on groundwater DO concentration, and two-way ANOVA for on land piezometers (in the riparian zone) was used to check the effects of distance and depth during flood condition. Spearman rank test was performed to examine the correlation between SRP and DO concentrations in groundwater in the riparian zone. Groundwater recharge toward the hyporheic zone in Spencer Creek was determined by measuring the saturated hydraulic conductivity and the mean vertical hydraulic gradient as well as the stream width at the point of individual piezometer (Equation 2). The P loading toward the hyporheic zone was estimated by multiplying the P concentration by the recharge for each piezometer. These analyses may permit characterization of groundwater phosphorus transfer patterns under the effects of hydrological variations, redox conditions and vegetation types in both longitudinal and lateral directions at intra/inter transect components in the study site.

2.6 Estimation of Error

Scientific error in environmental science is a function of both the precision and accuracy of a measurement. This section discusses possible measurements errors from the present study and indicates the steps taken to minimize error to an acceptable level (Whyte and Paul, 1985).

A survey of the study site was conducted with an electronic total station (PENTAX® PTS600). The elevation of the base for each piezometer nest and its distance to the bench mark were calculated automatically by the instrument. Human error due to changing prism height caused by operators was estimated at < 0.3 cm. Systematic or instrumental errors may result from changing environmental conditions (e.g., temperature, pressure, sunlight and wind speed), particularly when the distance between the prism and station is > 1 km (personal communication, Dr. Ian McKenzie, Department of Geography, University of Waterloo, 2007).

The changes in water table elevation was measured in piezometers and wells by an electronic water table sensor with a precision of ± 2 mm. Human errors in water table measurement may result from small bends in the sensor tape, personal bias of operators (e.g., a tendency always either to underestimate of overestimate the value of some readings) and possible changes in the absolute position of the piezometers or wells due to both environmental (e.g., ground surface changes due to swelling or shrinking of organic soil layers) especially peat soils (Whittington, 2005) and human factors (e.g., unintended pressing on the tip of a piezometer). The stick-up (height above ground surface) for all piezometers and wells were measured at the beginning, the middle and at the end of study period, as well as some random checks for individual piezometers throughout the study period. The average of these measurements for each piezometer or well was used for analysis to reduce the error effect from changing stick-ups. Errors due to mistakes in reading, recording and computing can not be estimated quantitatively. Given all sources of error, any difference in water table or hydraulic head < than 4 cm was not considered to be significantly different.

Hydraulic conductivity (K_{sat}) was measured with an electronic water table sensor and a stop watch with display of second, minute and hour, following the bail test procedure. Bail tests may produce more replicable estimations of K_{sat} compared to slug test (Baird *et al.*, 2004). Since the sensor is ± 2 mm, the estimated error is likely < 2% when an initial head difference is > 10 cm (Whittington, 2005). The time of each sensor measurement was record in seconds, the error associated with time reading was < 1%, but may become larger for some piezometers with very quick head recovery.

Phosphorus concentrations of TP and SRP in groundwater and stream water samples were tested using the stannous chloride colorimetric method. The stannous chloride method is appropriate for the samples in the phosphorus range of 0.01 to 6 mg L^{-1} and is more sensitive than the vanadomolybdophosphoric acid colorimetric method (APHA, 1995). The colorimeter and analysis software allow a direct analysis of the samples < 200 μg L^{-1} for TP and < 160 μg L^{-1} for SRP, respectively. Samples with higher P concentration out of this range must be diluted to the concentration within the range before analyzed by the instrument. Errors may exist throughout the preliminary sample treatment (i.e., digestion, filtration), sample dilution, reagent preparation (ammonium molybdate reagent, stannous chloride reagent, standard phosphorus solution, sulfate acid solution) and final colorimetric analysis (mainly due to interference). These are mostly from human (operators) factor except the last one which is systematic but is assumed to be constant for all samples as long as the procedures are consistent. Considering all procedures of the stannous chloride method, for the final results of P concentration, error was likely within 5-15% (APHA, 1995).

Chapter 3 RESULTS

3.1 Meteorology

The study site is characterized by a humid continental climate. The annual mean temperature is 7.6 °C, with the maximum average of 20.8 °C in July and the minimum average of -6.2 °C in January. Precipitation varies over the year, with an annual average of 890.4 mm. Rainfall contributes 83.5% to total precipitation (Warren *et al*., 2001). A summary of meteorological conditions is shown in Table 3-1. The graph of precipitation (obtained from a Hobo Weather Station situated at an open central location on the John Mount farm) during baseflow and flood are shown in Figure 3-1.

Table 3-1 A summary of monthly temperature and total precipitation

	Temperature for 2006 (°C)			30 year	Total precipitation for 2006 (mm)	30 year
	Mean	Max	Min	Mean		
May	13.7	33.1	-1.1	13.3	62.5	81.8
June	18.6	30.9	6.2	18.8	37.2	71.6
July	22.3	33.7	10.6	22.0	149.2	74.9
August	20.0	34.1	8.6	20.9	69.4	84.6
October	9.0	13.9	4.0	10	132.3	72.4

* Data are from Environmental Canada, Hamilton RBG station (43° 16' N, 79° 52' W, the nearest station to the study site 43° 22' N, 80° 07' W, 30 year is from 1971 to 2000).

The study season in temperature was normal compared with the 30 year average, and all monthly means were within ± 1.0 °C of 30 year means. The study season was dry with total precipitation in most months below the 30 year mean, but July and October had extremely high precipitation compared to the 30 year mean (Table 3-1). During flood, daily precipitation on JD290, JD291, JD292 and JD293 was 26.8 mm, 0.2 mm, 1 mm and 2.8 mm, respectively, but no precipitation fell on JD289. The largest daily precipitation occurred on JD290 during flood (Figure 3-1).

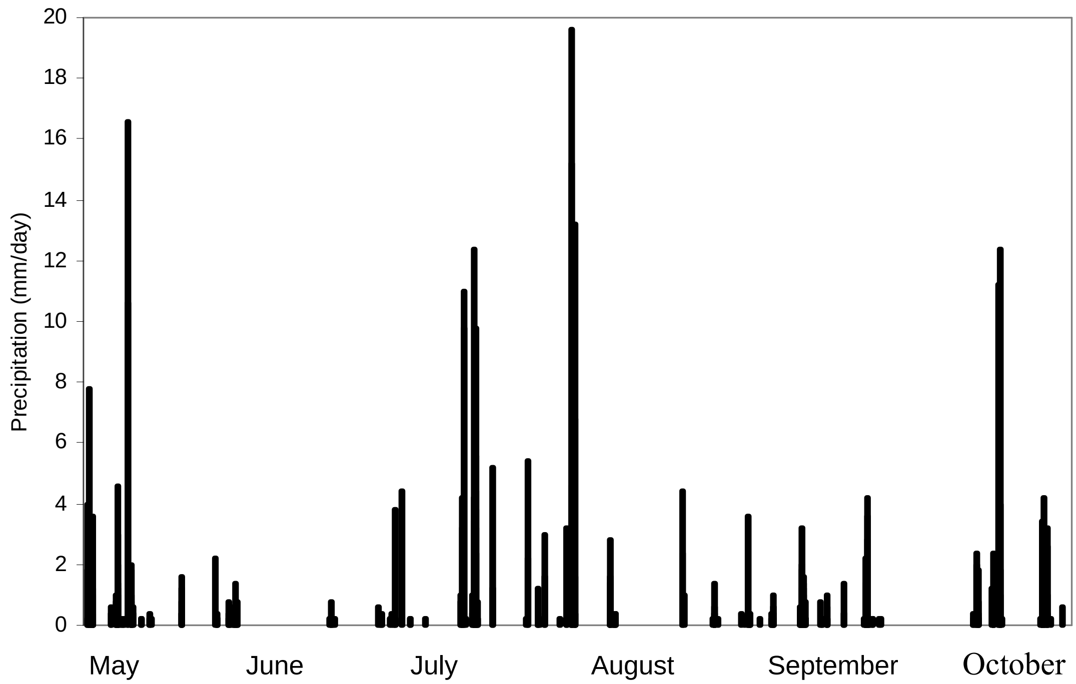


Figure 3-1 Precipitation during baseflow (May 10 (JD130) to October 15 (JD288), 2007) (The sampling days of the present study included JD130, JD138. JD145, JD157, JD171, JD187, JD199, JD213, JD222 and JD288)

3.2 Hydrology

3.2.1 Hydraulic conductivity (K_{sat})

Groundwater hydraulic conductivity varied spatially in the study site. There was no trend with distance across the riparian zone (i.e., K_{sat} does not clearly increase or decrease towards the stream). High variability in K_{sat} (3 orders of magnitude) was observed.

3.2.1.1 Intra-transect variability

Shallow soils at transect T1 had a higher K_{sat} (10^{-2} cms^{-1}) than deeper soils (10^{-5}-10^{-4} cms^{-1}), K_{sat} generally decreased with depth and trended $K_{sat(50cm)}$>$K_{sat(100cm)}$> $K_{sat(150cm)}$ with an exception at the field edge (T1-6) which increased with depth by less than an order of magnitude (Figure 3-2a, c).

At transect T4, deeper soils generally had higher K_{sat} of 10^{-3}-10^{-2} cms^{-1} compared to shallow soils, although there were some areas with lower hydraulic conductivities such as the hyporheic zone. K_{sat} in hyporheic zone (T4-50) had a similar decreasing trend with depth as transect T1. In the riparian edge (T4-48), K_{sat} was different from transect T1, and trended $K_{sat(50cm)}>K_{sat(150cm)}>K_{sat(100cm)}$. For the remaining sites in field edge (T4-6), near middle riparian (T4-19), and far middle riparian zone (T4-33) at T4, K_{sat} increased with depth and trended $K_{sat(50cm)}<K_{sat(100cm)}<K_{sat(150cm)}$ (Figure 3-2b, d).

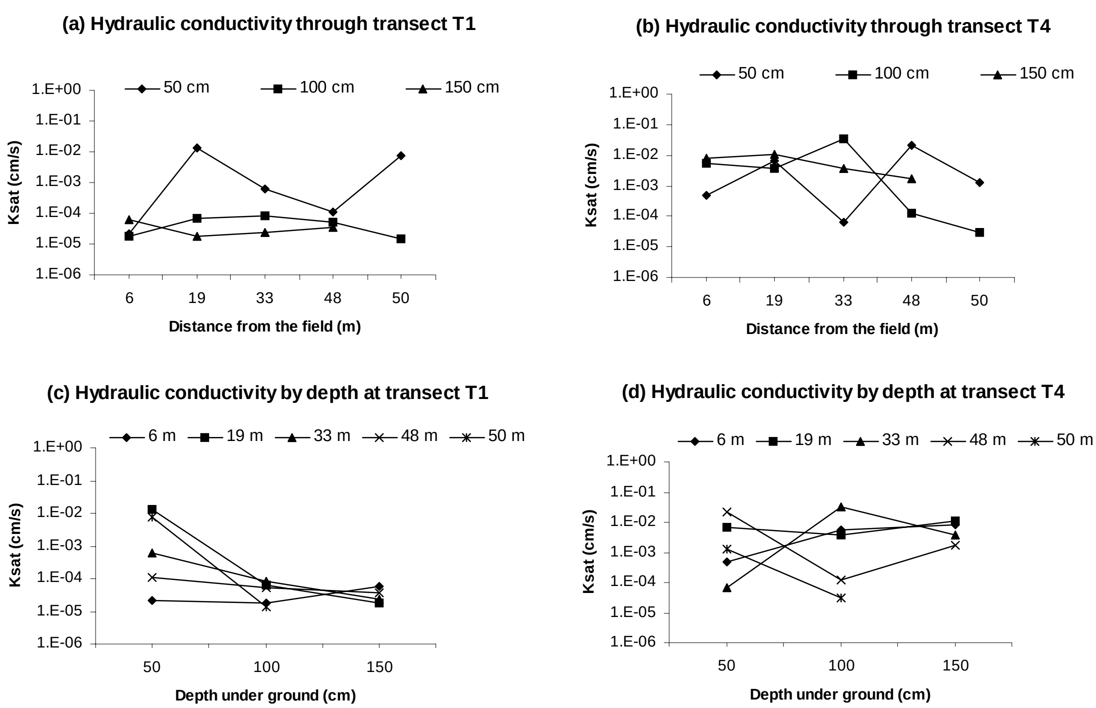


Figure 3-2 Spatial variability of K_{sat} within transects

3.2.1.2 Inter-transect variability

Hydraulic conductivities at 50 cm were similar at transect T1 and T4. At 100 cm and 150 cm, K_{sat} was generally higher at transect T4 than T1 by 2 to 3 order of magnitude (Figure 3-3).

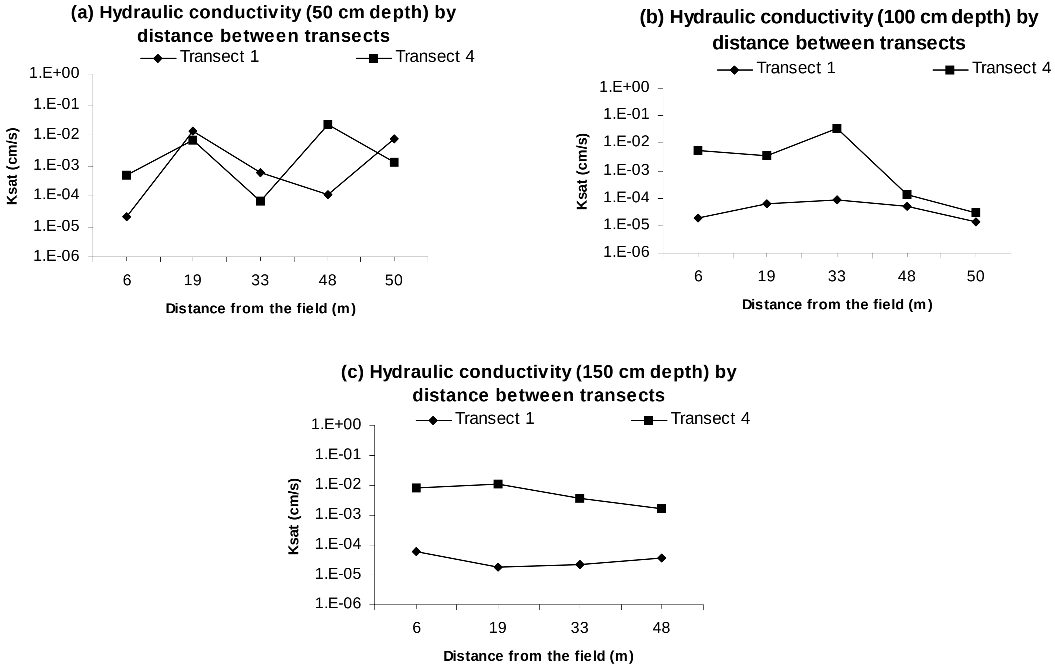


Figure 3-3 Spatial variability of K_{sat} between transects

3.2.2 Water table

Water table was highly variable over the study period, ranging from as low as 55 cm below the ground surface to greater than 28 cm above ground (Table 3-2). Water table under three hydrological conditions (dry, wet and peak flood) are shown for each transect to compare intra/inter transect differences in water table elevation (Figure 3-4). Differences in water table by transect and flow regimes are shown in Figure 3-5 and 3-6, respectively. Time-series plots are used to illustrate temporal patterns of water table for the study period in Figure 3-7.

Table 3-2 Descriptive statistics summaries of water table (cm)

Transect	Flow regime	n	Mean	SEM	Min	Max	SD	Range
1	Baseflow	49	-15.3	1.9	-55.5	5.5	13.4	61.0
	Flood	25	3.9	2.9	-54.5	28.5	14.6	83.0
4	Baseflow	49	-14.2	2.0	-51.5	8.0	14.2	59.5
	Flood	25	6.6	1.2	-6.5	17.5	5.8	24.0

* n is sample size; SEM is standard error of mean; Min and Max refer to minimum and maximum, respectively; SD is standard deviation; Range = (Max-Min).

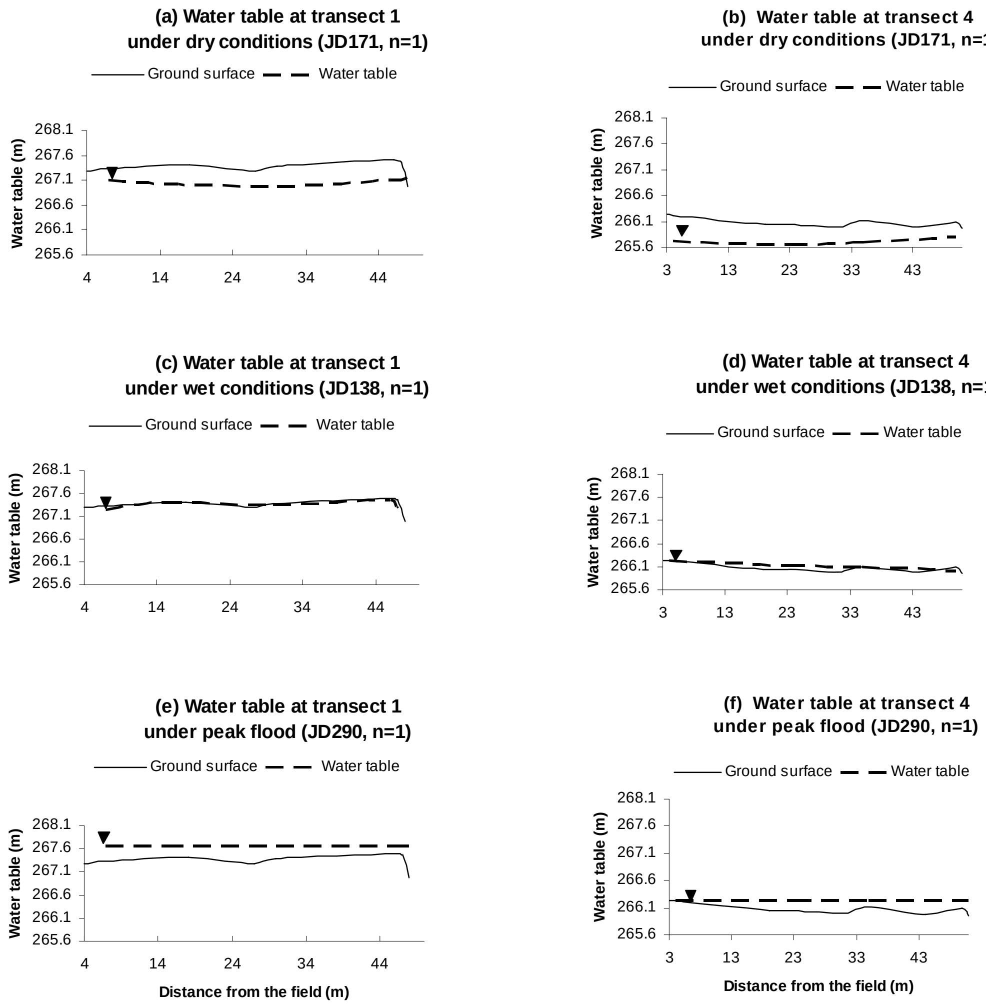


Figure 3-4 Differences in water table under different hydrological regimes

3.2.2.1 Spatial variability

1) Intra-transect pattern

At transect T1, mean water table decreased relative to ground surface with increasing distance from field edge (T1-6) to riparian edge (T1-48) during baseflow conditions. This trend was not observed at transect T4 which had a higher degree of spatial variability. During the flood caused by the drawdown of Valens reservoir, variability of mean water table across two transects was minimized (Figure 3-5).

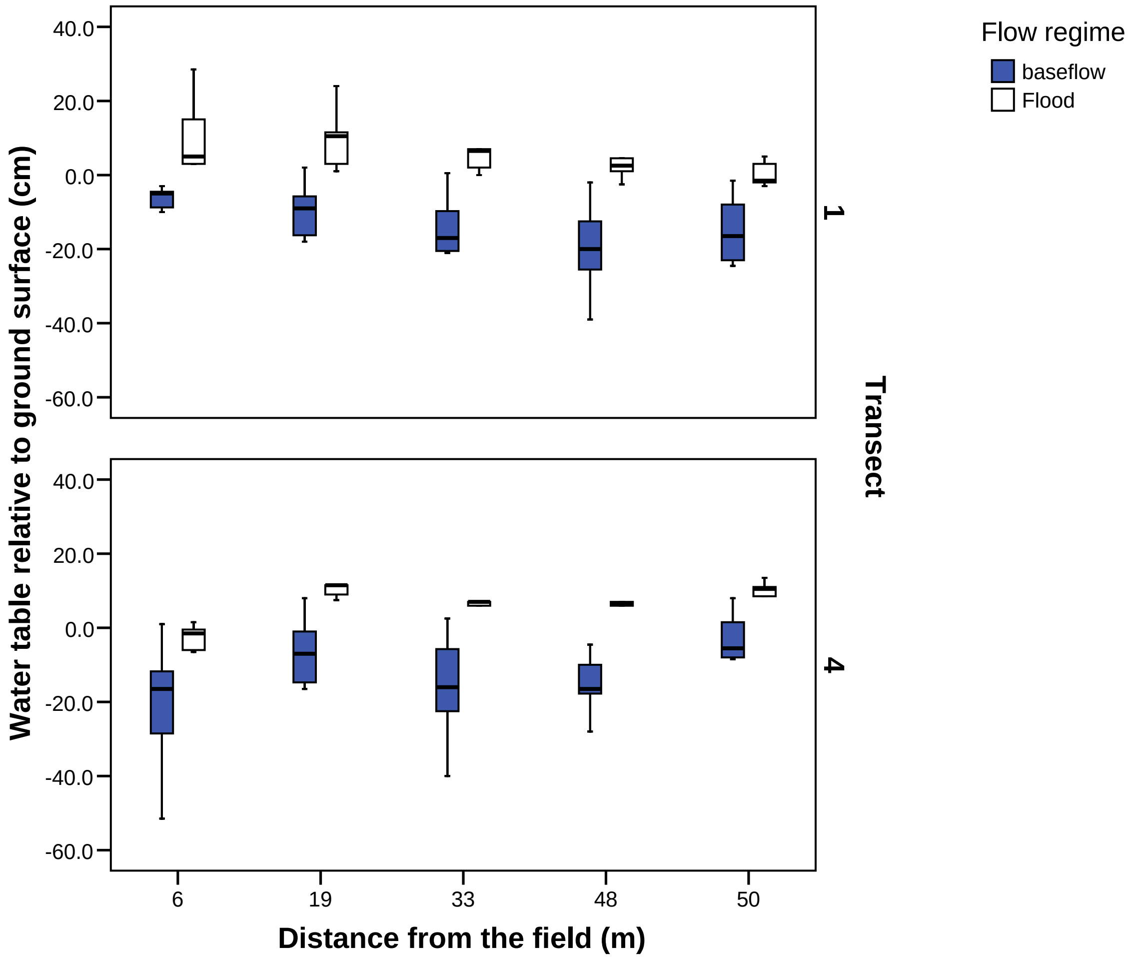


Figure 3-5 Spatial variability of water table under different flow regimes within transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

2) Inter-transect pattern

The mean water table was always lower at transect T1 than T4 during both baseflow and flood conditions except at the field edge (6 m) where mean water table was higher at transect T1 than T4 during both baseflow and flood conditions (Figure 3-6).

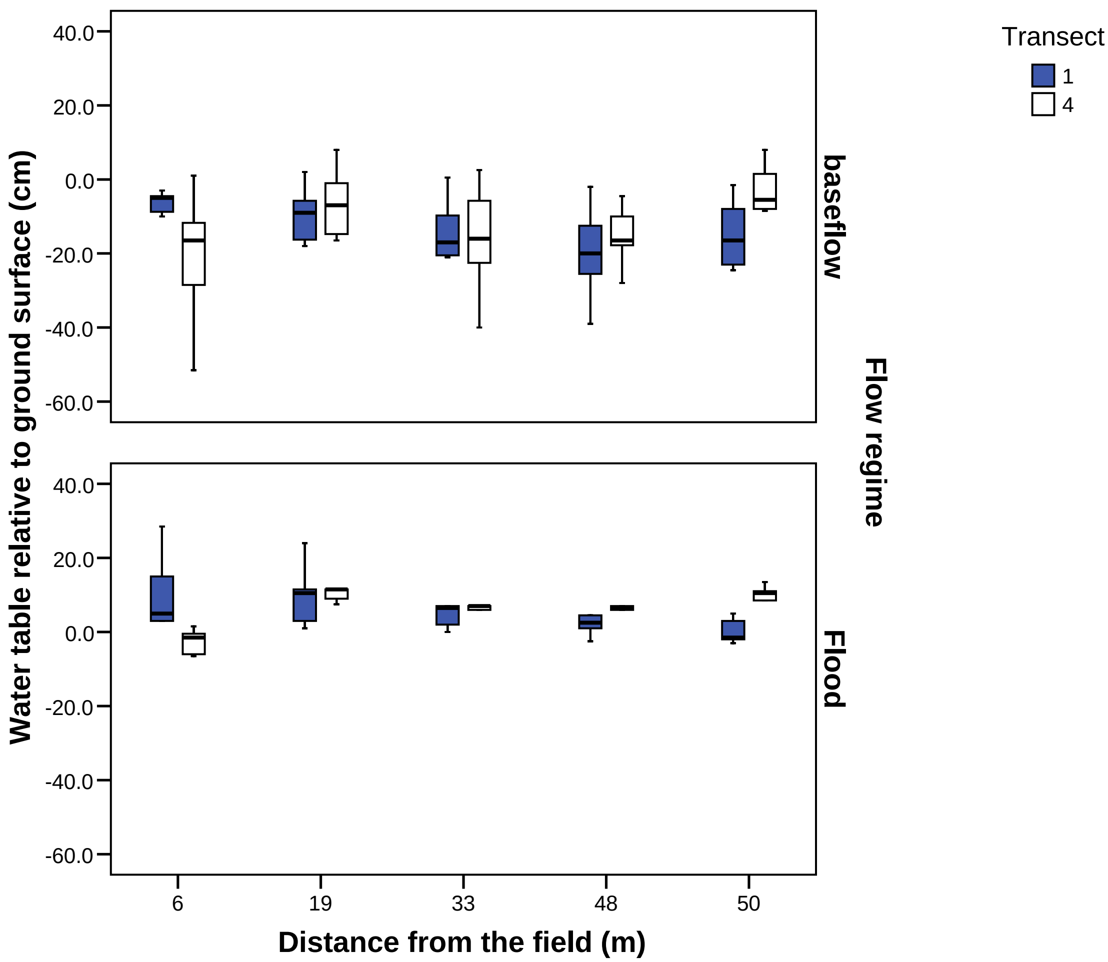


Figure 3-6 Spatial variability of water table under different flow regimes between transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

3.2.2.2 Temporal variability

The mean water tables of all sites at transect T1 and T4 had similar temporal patterns. They were higher in mid May, late July and early August than late May, June and early July in response to rain events (Figure 3-1) during baseflow. The mean water tables of all sites were much higher during flood than baseflow at the two transects with an exception at the field edge (6 m) at T4 which had a similar water table during flood as that in mid May (JD 138). Prior to flood, water tables were higher compared to the rest of the season during baseflow with an exception at the field edge (6 m) at T4 which was lower than that in mid May (JD 138) (Figure 3-7).

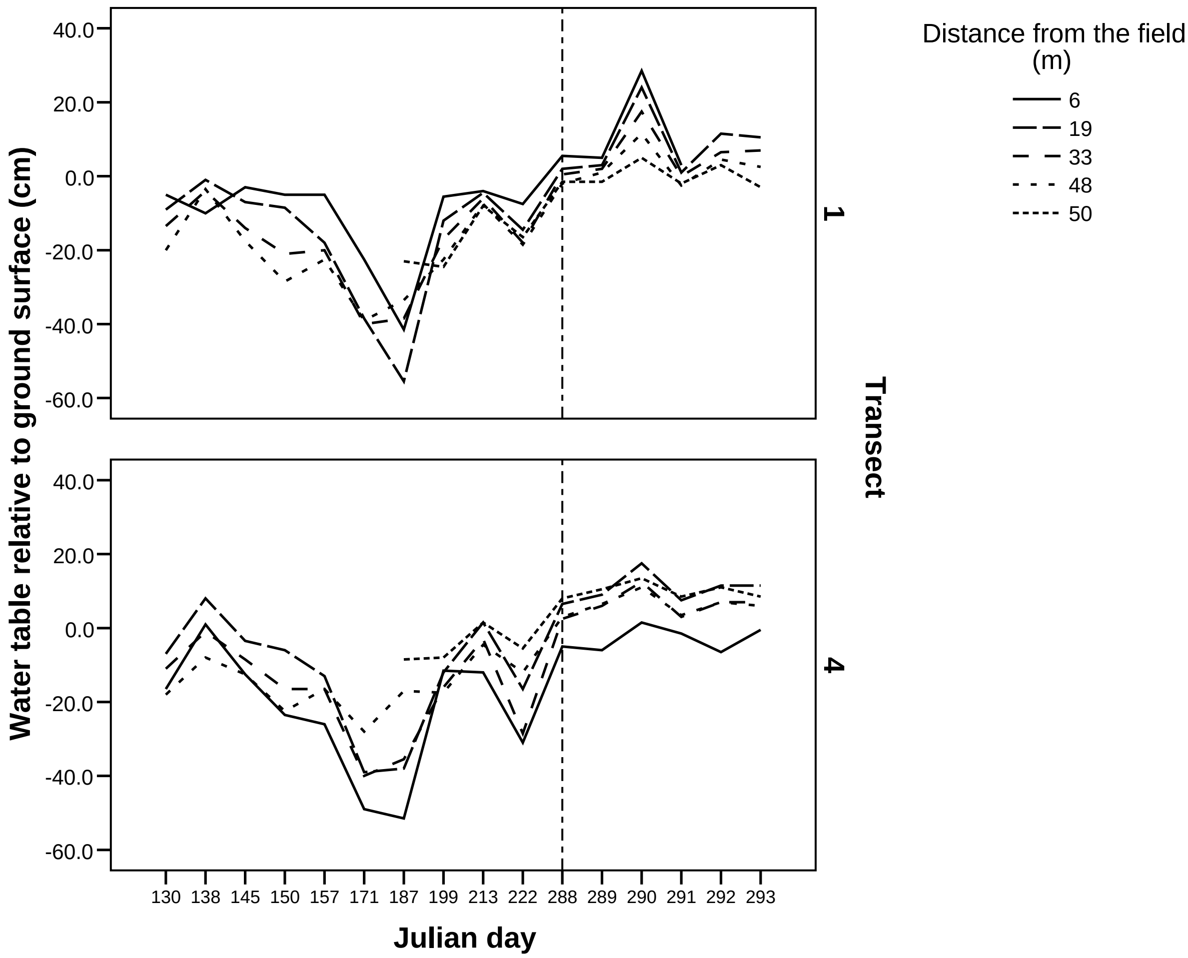


Figure 3-7 Temporal variability of water table at two transects under different flow regimes (vertical dashed line separates baseflow and flood periods, JD 288 is the pre-flood day.)

3.2.3 Hydraulic head and groundwater flow

Hydraulic head varied both spatially and temporally over the study period, ranging from 265.041 m to 267.605 m in response to flow conditions. The section and plan plots are used to illustrate the hydraulic head distribution along transect sections (Figures 3-8 and 3-9) and the representative section between two transects (Figure 3-10), and subsurface of the riparian zone (Figures 3-11 and 3-12) under different flow regimes.

3.2.3.1 Spatial variability

1) Intra-transect pattern

At transect T1, the hydraulic head generally decreased from the field edge (T1-6) to the edge of riparian zone (T1-48) on the dry day during baseflow (Figure 3-8a). In contrast, the hydraulic head tended to increase from the field edge (T1-6) to the edge of riparian zone (T1-48) on the peak flood day (Figure 3-8b). Both trends on the dry day and peak flood day were more pronounced at 50 and 100 cm than 150 cm depths. No consistent pattern with depth was observed on the dry day during baseflow. However, the hydraulic head mostly decreased with depth on the peak flood day. The flood event increased the differences between groundwater depths. The hydraulic head at 50 cm depth in the riparian edge (48 m) and hyporheic zone (50 m) was always higher than that at 100 cm during all flow conditions (Figure 3-8).

At transect T4, the hydraulic head generally increased from the field edge (T4-6) to the hyporheic zone (T4-50) on the dry day during baseflow (Figure 3-9a). There was no clear pattern with distance on the peak flood day (Figure 3-9b). The variability of hydraulic head minimized during flood compared to baseflow. No clear pattern with depth was found on the dry day during baseflow. The hydraulic head at 50 cm depth in the riparian edge (48 m) and hyporheic zone (50 m) was always higher than that at 100 cm during all flow conditions (Figure 3-9).

2) Inter-transect pattern

The hydraulic head in all sites at each depth was higher at transect T1 than T4 on both the dry day during baseflow and the peak flood day, indicating that flow was likely from transect T1 towards T4 during all flow conditions (Figure 3-10).

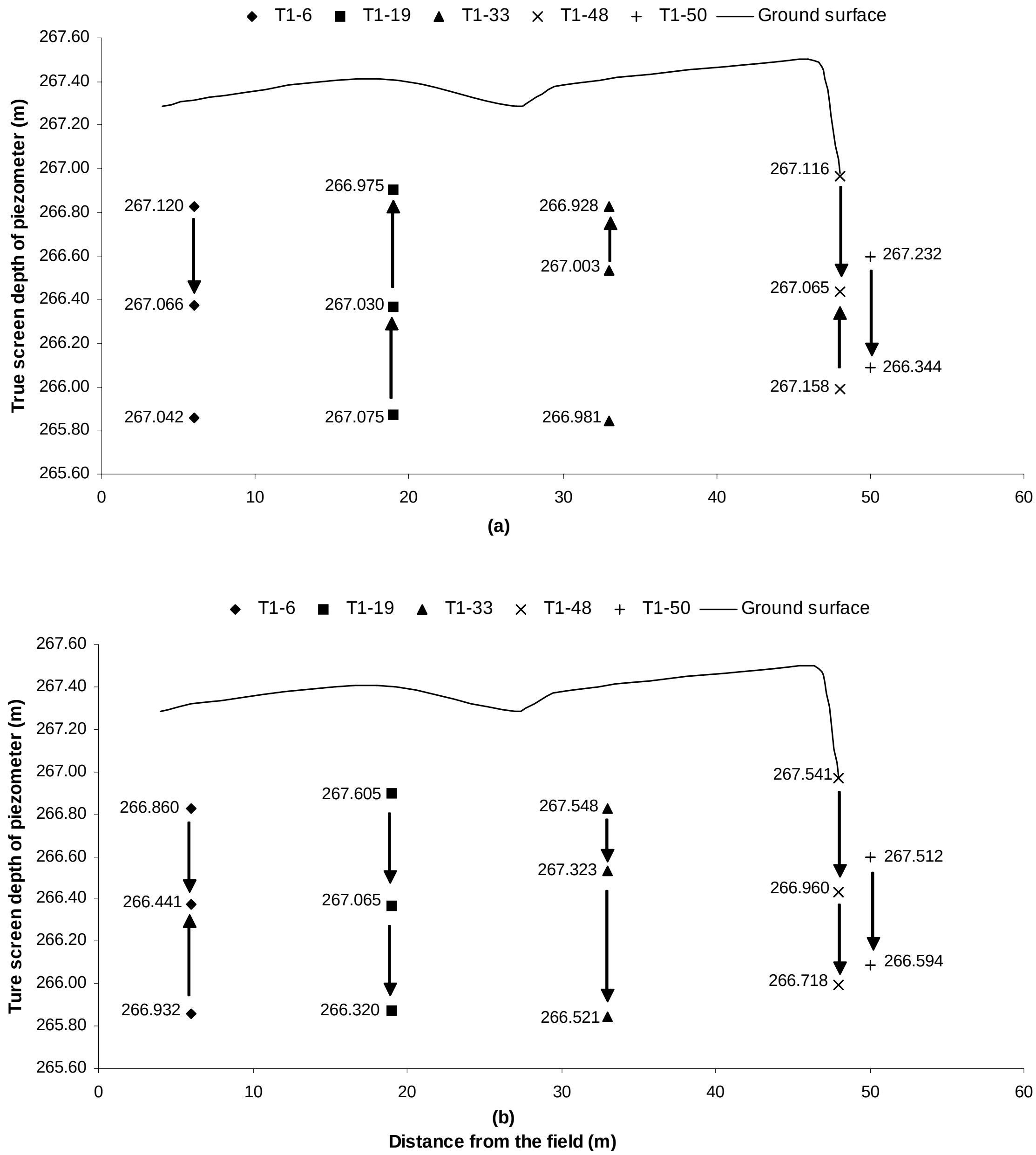


Figure 3-8 Hydraulic head in transect T1 under different flow regimes. (a) dry day during baseflow and (b) peak flood day. Arrows indicate possible flow directions.

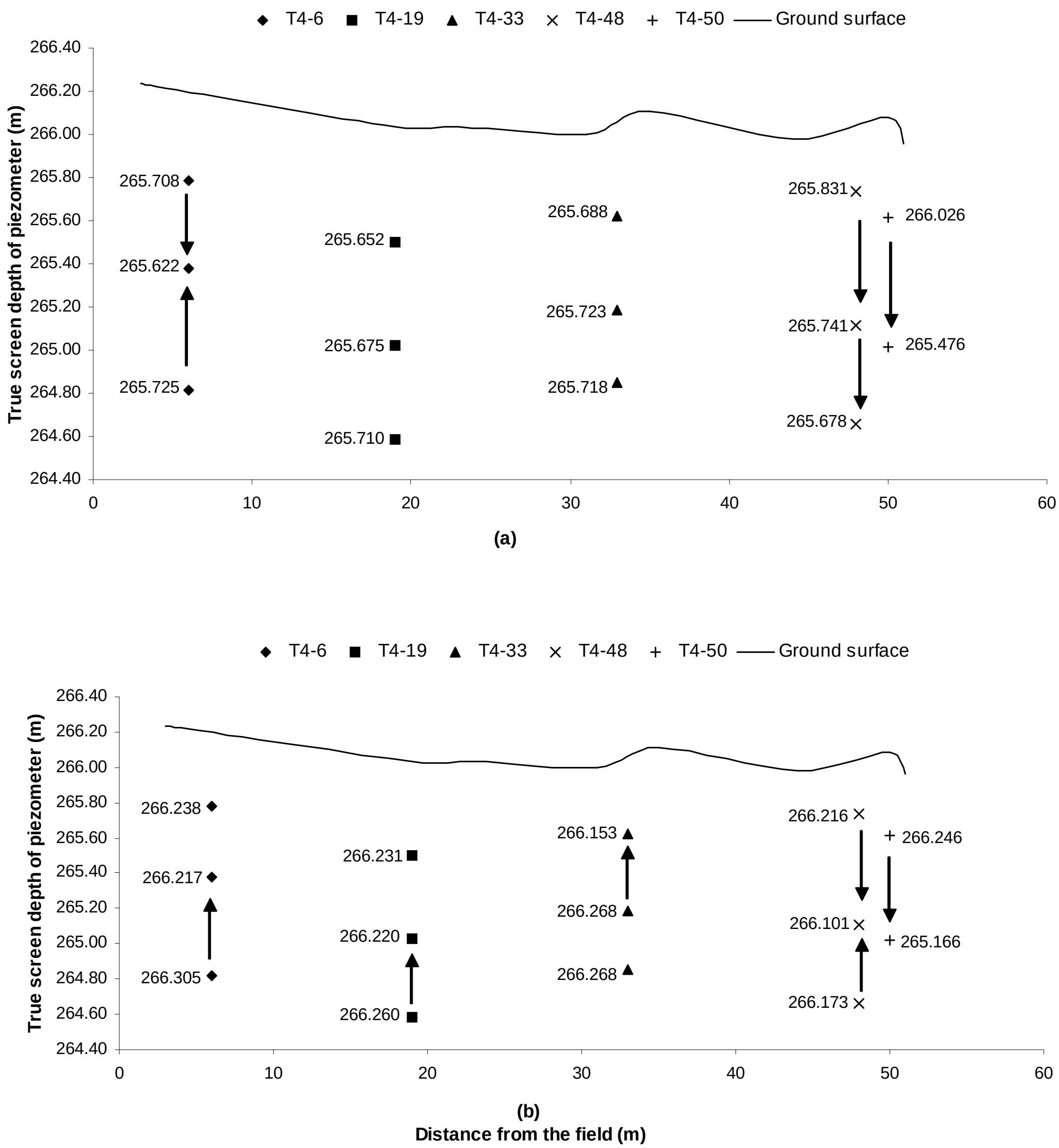
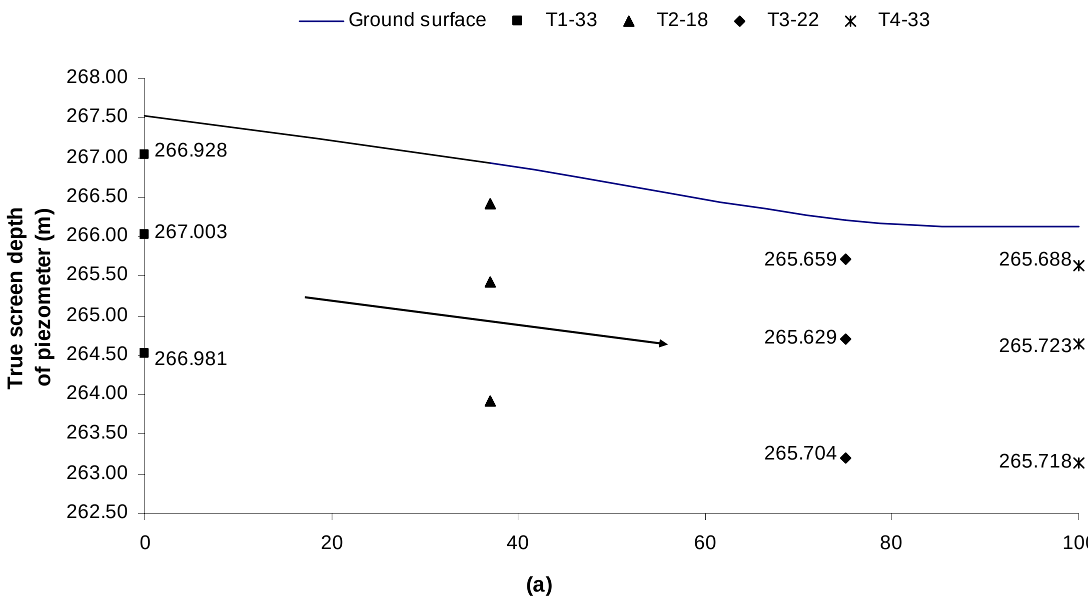


Figure 3-9 Hydraulic head in transect T4 under different flow regimes (a) dry day during baseflow and (b) peak flood day. Arrows indicate possible flow directions.



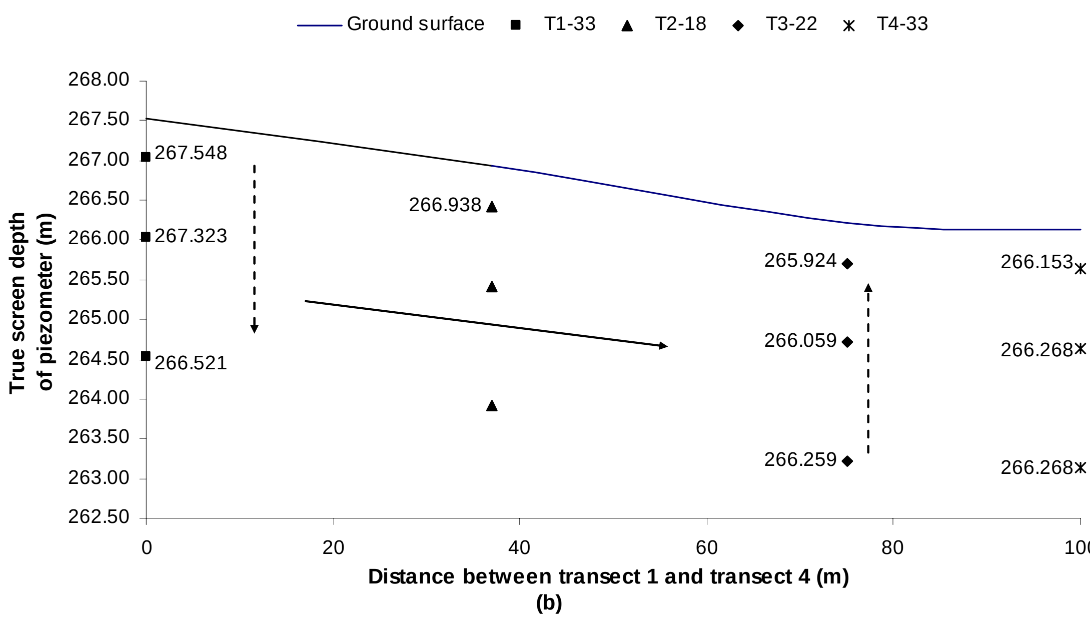


Figure 3-10 Hydraulic head in the representative cross-section between transect T1 and T4 under different flow regimes. (a) dry day during baseflow and (b) peak flood day. Arrows indicate possible flow directions.

3.2.3.2 Groundwater flow

Groundwater flow generally follows the hydraulic gradient and moves from the position with higher hydraulic head to a lower one. There was no consistent vertical flow pattern at two transects during dry conditions (Figure 3-8a and 3-9a). Groundwater recharge from upper layers toward lower layers possibly existed at most sites at transect T1 during the peak flood day (Figure 3-8b). The vertical hydraulic gradients (VHG's) in the riparian zone were consistently one order of magnitude higher on the peak flood day than during baseflow. VHG's were also higher in the hyporheic zone during peak flow compared to baseflow conditions, although not as high as in the riparian zone. At transect T4, this recharge pattern was not the same at the riparian zone but more pronounced at the hyporheic zone which likely due to a one order magnitude higher VHG during flood than baseflow (Figure 3-9b). Also, the VHG at the riparian zone during flood was substantially higher at T1 than T4. But this pattern reversed at the hyporheic zone. An effort to generate groundwater flow lines along the section of two transects was not possible due to the highly variable and irregular hydraulic head, indicating that there was no substantial groundwater flow along transect sections from the field edge (6 m) to the hyporheic zone (50 m) during baseflow and flood (Figures 3-8 and 3-9). In contrast, the hydraulic head gradient along the cross-section from transect T1 to T4 showed generally horizontal groundwater flows between two transects during both baseflow and flood (Figures 3-10, 3-11 and 3-12). Under all flow conditions, the lateral hydraulic gradients (LHG) between T1 and T4 at three depths (represented by the LHG between T1-33 nest and T4-33 nest in Figure 3-10) were all higher than the VHG at each transect with an exception of that at 150 cm depth during flood (Figures 3-8, 3-9 and 3-10). The LHG was similar during baseflow (Figure 3-10a) but more pronounced in the upper layers during flood (Figure 3-10b).

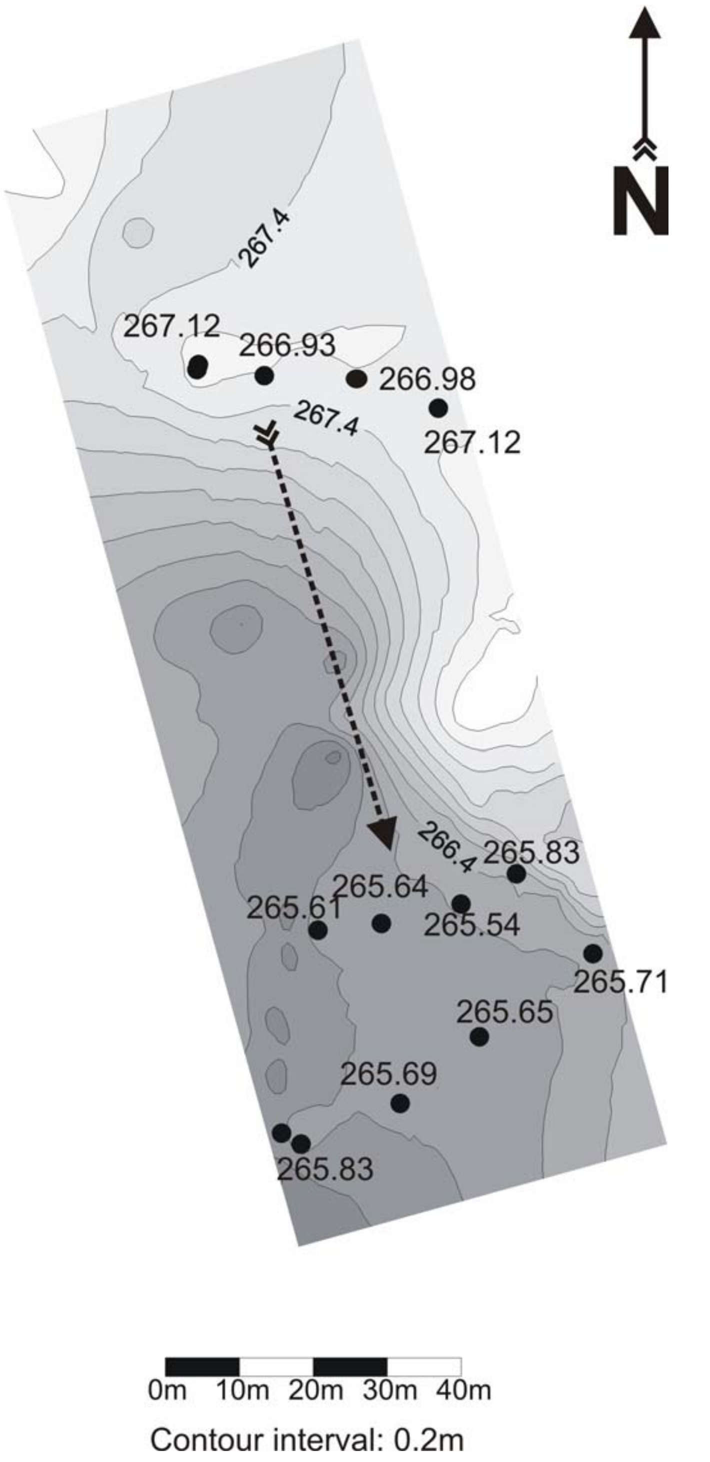


Figure 3-11 Plan view of estimated possible shallow groundwater flow between transects during dry day (baseflow conditions). Black dots indicate locations of piezometer nests and water table elevation; arrows indicate flow directions.

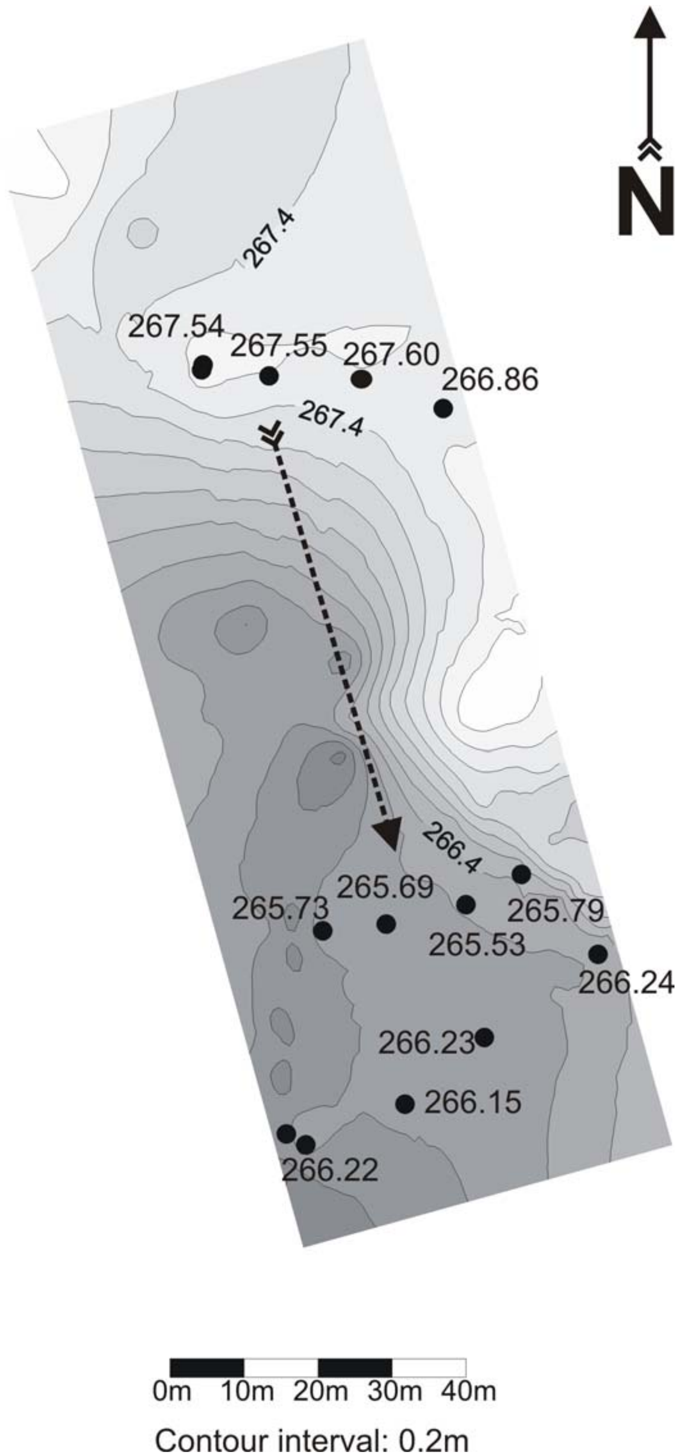


Figure 3-12 Plan view of estimated possible shallow groundwater flow between transects during peak flood (reservoir draw down). Black dots indicate locations of piezometer nests and water table elevation; arrows indicate flow directions.

3.2.4 Stream discharge

Stream discharge varied temporally from May to October (Figure 3-13). For sampling days, stream discharge ranged from 0.10 Ls^{-1} to 72.79 Ls^{-1} with an average (mean ± standard error) of 13.80 ± 0.64 Ls^{-1} during baseflow and from 52.68 Ls^{-1} to 204.03 Ls^{-1} with an average of 108.30 ± 1.97 Ls^{-1} during the flood.

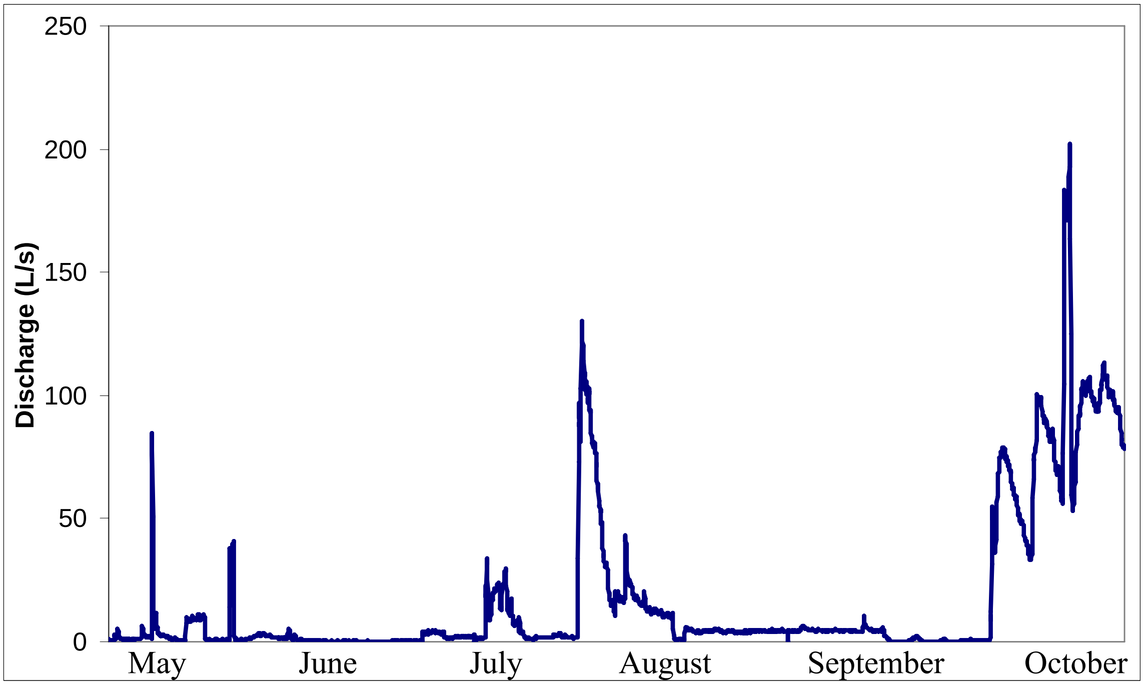


Figure 3-13 Stream discharge during the study period

Stream discharge varied in response to storm events and drought prior to reservoir draw down (Figures 3-1 and 3-13). During baseflow, it increased to a maximum of 130.28 Ls^{-1} on July 28 (Figure 3-13) as a result of the highest precipitation (Figure 3-1). The discharge reached a higher peak value of 204.03 Ls^{-1} on October 17 during flood than during baseflow, although the corresponding precipitation on October 17 was far less than on July 28 (Figure 3-1), indicating a substantial effect of reservoir release on stream discharge.

3.3 Groundwater Phosphorus

Groundwater P concentrations (TP and SRP) varied both spatially and temporally over the study period. Descriptive statistics are used to illustrate general trends of groundwater phosphorus for the study period.

3.3.1 Total phosphorus

Total phosphorus concentrations in groundwater were highly variable, ranging from 11μgL^{-1} to > 8000μgL^{-1} (Table 3-3). Groundwater TP concentrations under three extreme hydrological conditions (dry, wet and peak flood) are selected for each transect to compare intra/inter transect differences in groundwater TP concentrations. The data are presented in Table 3-3 and Figure 3-14. The TP data are presented by transect and flow regime for 50 cm, 100 cm and 150 cm depths and differences in TP concentrations by depth and flow regime are shown in Figure 3-15 and 3-16, respectively. Time-series plots are used to illustrate temporal patterns of TP concentrations for the study period in Figure 3-17.

Table 3-3 Descriptive statistics of groundwater TP concentration (μgL^{-1})

Transect	Flow regime	n	Mean	SEM	Min	Max	SD	Range
1	Baseflow	138	247.1	28.4	13.6	1829.3	333.7	1815.7
	Flood	68	294.4	43.0	12.7	1934.5	354.8	1921.8
4	Baseflow	132	488.7	41.4	11.8	2685.6	476.1	2673.8
	Flood	63	1226.0	205.9	35.1	8485.5	1634.3	8450.4

* n is sample size; SEM is standard error of mean; Min and Max refer to minimum and maximum, respectively; SD is standard deviation; Range = (Max-Min).

3.3.1.1 Spatial variability

1) Intra-transect variability

At transect T1, mean TP concentrations generally increased with increasing distance from the field edge (T1-6) to the hyporheic zone (T1-50) at 50 and 100 cm during all flow conditions. TP concentrations were consistently low at 150 cm (Figure 3-15) but higher during flood than baseflow (Table 3-3). At transect T4, the mean TP concentrations did not show consistent patterns with distance from the field edge (T4-6) to the hyporheic zone (T4-50) during all conditions with an exception of groundwater at 50 cm, which generally increased from the field edge (T4-6) to the hyporheic zone (T4-50) during baseflow (Figure 3-15). Mean TP concentrations at transect T1 generally decreased with depth and trended $TP_{50cm}>TP_{100cm}> TP_{150cm}$ for all sites during all flow conditions. However, these trends were not prevalent at T4. Variability in TP concentration generally decreased at transect T1 but increased at T4 during flood compared to baseflow (Figure 3-15).

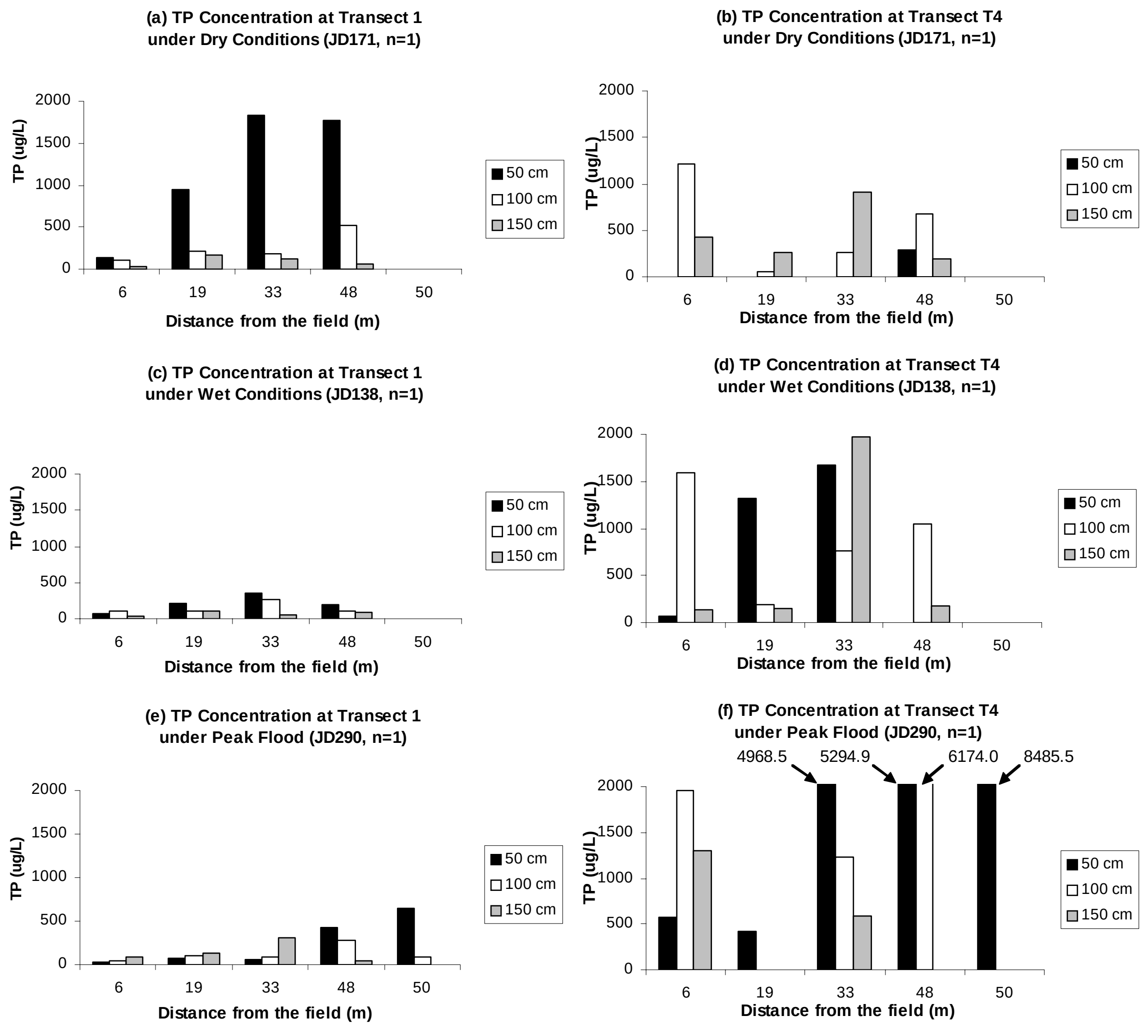


Figure 3-14 Groundwater TP concentration under different hydrological conditions over transects

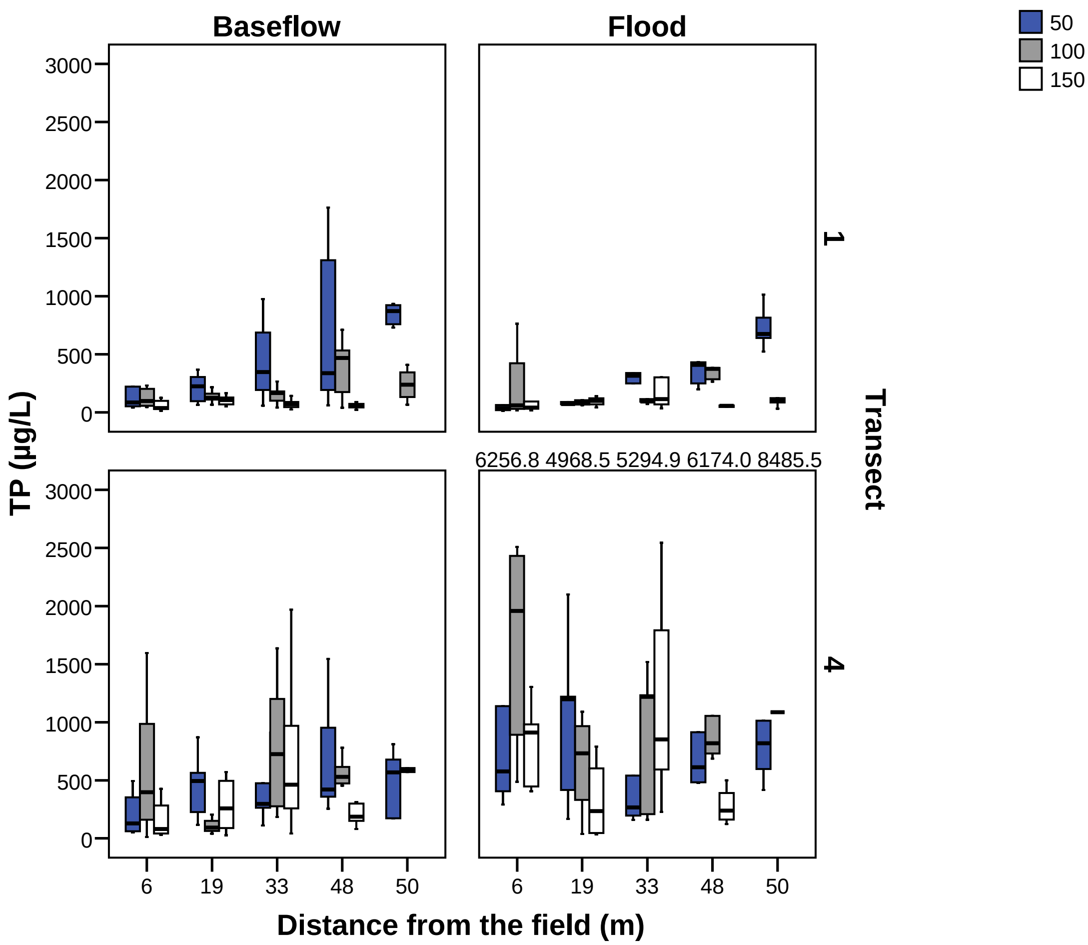


Figure 3-15 Spatial variability of groundwater TP concentration under different flow regimes within transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers. Numbers above the top edge are the outliers can not be included due to scale.)

2) Inter-transect pattern

During baseflow, TP concentrations in groundwater at 100 and 150 cm depths were always greater at transect T4 than T1. However, trends in TP concentrations at 50 cm depth during baseflow were less clear. For example, TP concentrations at T4 were higher than T1 at some sites (e.g., the field edge (6 m), near middle riparian zone (19 m) and riparian edge (48 m)), but were lower than T1 at other sites (e.g., far riparian zone (33 m) and hyporheic zone (50 m)). During flood conditions, the mean TP concentrations in all sites at all depths were higher at transect T4 than T1 (Figure 3-16). Thus, while in general, the mean TP concentrations were higher at transect T4 than T1 during all flow conditions (Table 3-3), this pattern was far more dramatic under flood conditions than during baseflow.

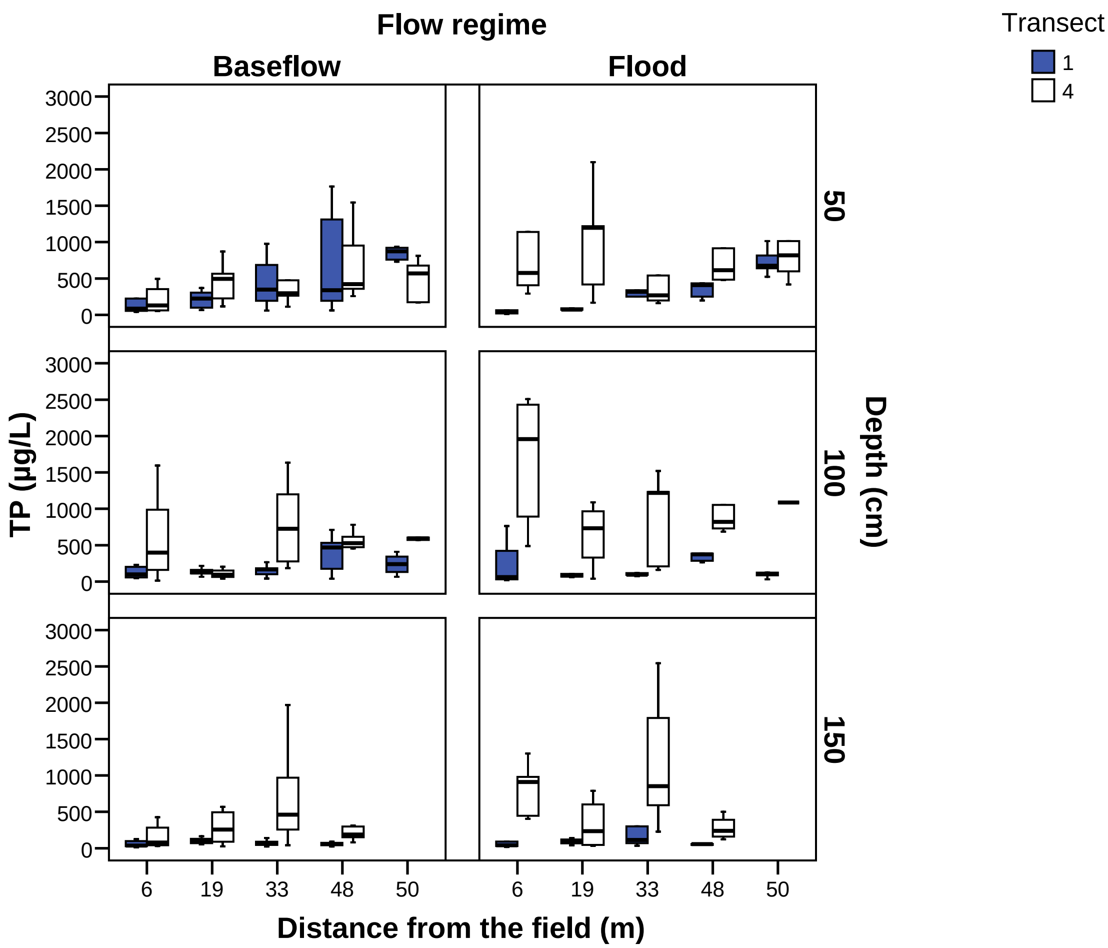


Figure 3-16 Spatial variability of groundwater TP concentration under different flow regimes between transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

3.3.1.2 Temporal variability

At transect T1, the mean TP concentration at 50 cm peaked in mid June then decreased from mid June till mid August during baseflow. No temporal variability was observed at 100 and 150 cm. At transect T4, the mean TP concentrations were markedly different over sampling dates at all depths with similar patterns that fluctuated from late

May to the date prior to flood event. The flood from reservoir release produced dramatic shift in TP concentration in groundwater at all depths. However, the mean TP concentrations at transect T1 peaked two days late after the flood (Figure 3-17). During basflow, mean TP concentrations at all depths were < 1200 μgL^{-1} at transect T1 and < 1000 μgL^{-1} at T4, respectively. During flood from reservoir draw down, they were still < 1200 μgL^{-1} at T1 but reached 4000 μgL^{-1} at 50 cm and 3000 μgL^{-1} at 100 cm depth with an average still < 1000 μgL^{-1} at 150 cm at T4.

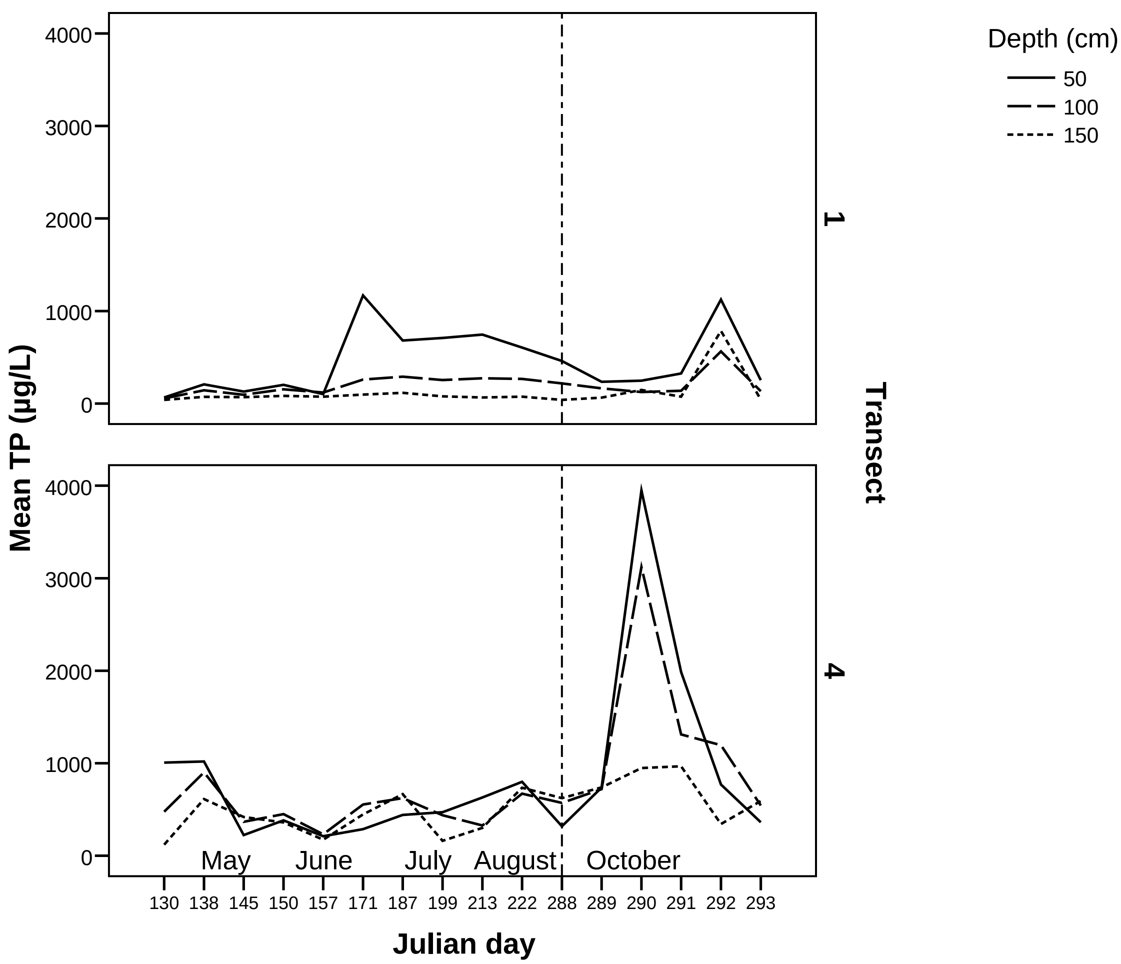


Figure 3-17 Temporal variability of groundwater TP concentration under different flow regimes (vertical dashed lines separate baseflow and flood periods.)

3.3.2 Soluble reactive phosphorus

Soluble reactive phosphorus concentrations in groundwater were less variable than TP and ranged from 2 μgL^{-1} to > 400 μgL^{-1} (Table 3-4). Groundwater SRP concentrations under three hydrological conditions (dry, wet, and peak flood) for each transect was used to compare intra/inter transect differences in groundwater SRP concentrations. The data are presented in Table 3-4 and Figure 3-18. The SRP data are presented by transect and flow regime for 50 cm, 100 cm and 150 cm depths and differences in SRP concentrations by depth and flow regime are shown in Figure 3-19 and 3-20, respectively. Time-series plots are used to illustrate temporal patterns of SRP concentrations for the study period in Figure 3-21.

Table 3-4 Descriptive statistics of SRP groundwater concentration (μgL^{-1})

Transect	Flow regime	n	Mean	SEM	Min	Max	SD	Range
1	Baseflow	140	36.7	3.7	3.6	236.6	43.6	233.0
	Flood	68	15.2	1.0	6.7	45.4	7.9	38.7
4	Baseflow	138	51.5	6.7	3.8	417.9	79.0	414.1
	Flood	62	54.4	10.3	2.1	280.4	81.2	278.3

* n is sample size; SEM is standard error of mean; Min and Max refer to minimum and maximum, respectively; SD is standard deviation; Range = (Max-Min).

3.3.2.1 Spatial variability

1) Intra-transect variability

At transect T1, there was no clear pattern of mean SRP concentrations with distance from the field edge (T1-6) to the hyporheic zone (T1-50). In contrast, mean SRP concentrations generally increased with distance from the field edge (T4-6) to the hyporheic zone (T4-50) at 50 and 100 cm but this trend was not observed at 150 cm at transect T4 (Figure 3-19). Mean SRP concentrations generally decreased with depth and trended SRP_{50cm}> SRP_{100cm}> SRP_{150cm} at transect T4 during all flow conditions. No

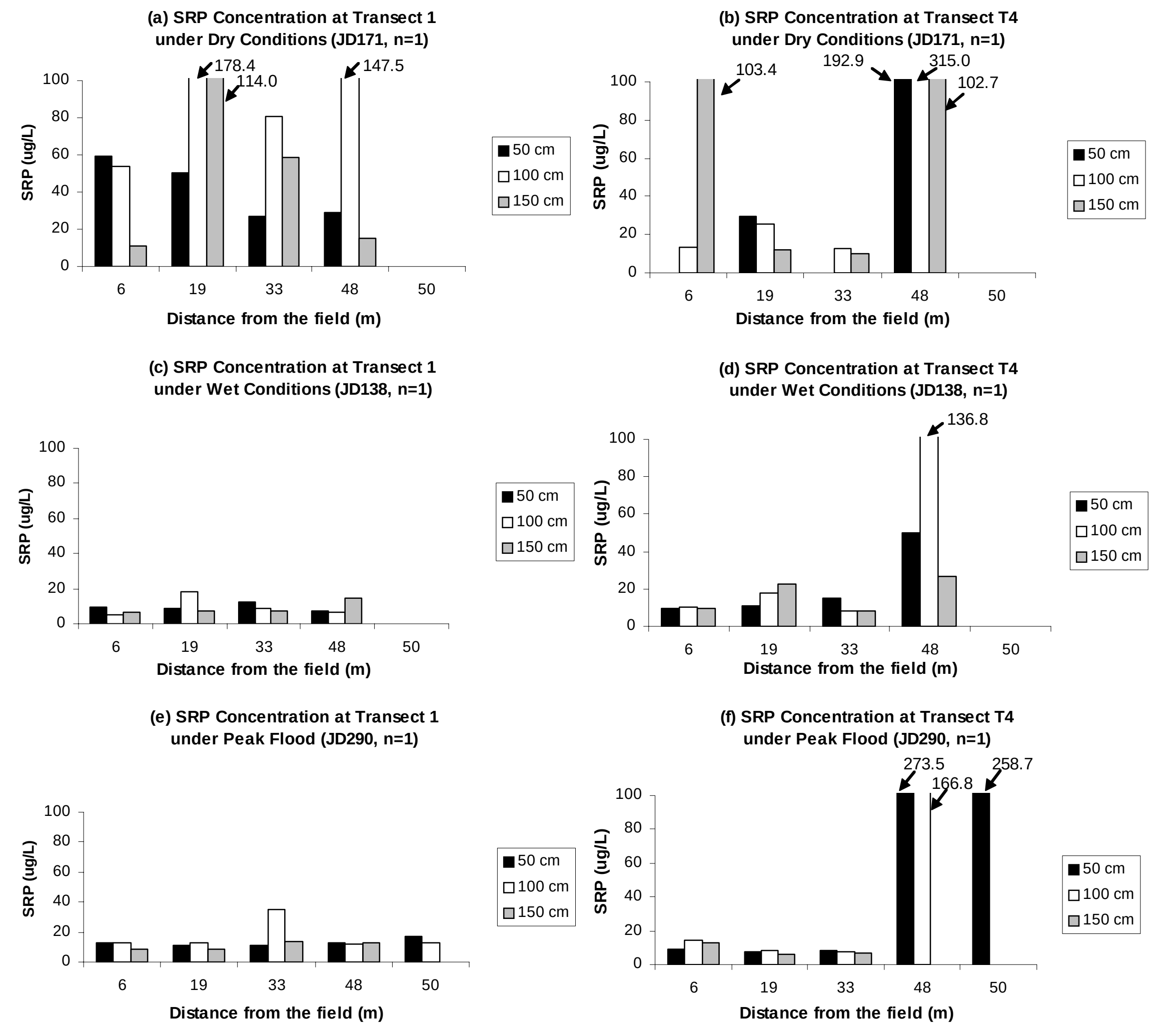


Figure 3-18 Groundwater SRP concentration under different hydrological conditions over transects

consistent pattern in SRP with depth was observed at transect T1. Variability in SRP concentrations was always larger during baseflow than flood at all depths across two transects (Figure 3-19).

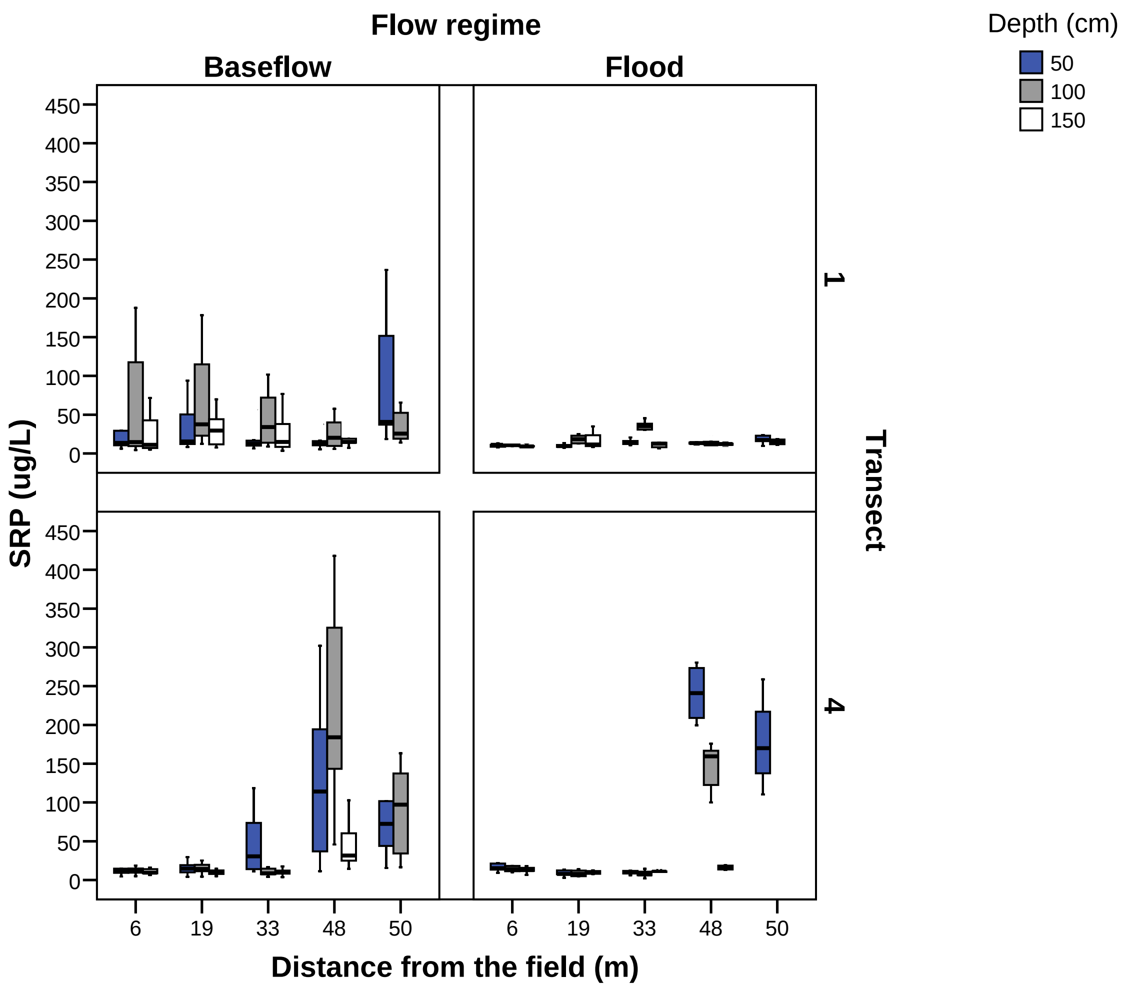


Figure 3-19 Spatial variability of groundwater SRP concentration under different flow regimes within transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

2) Inter-transect pattern

There was no observable inter-transect pattern of mean SRP concentrations at 50 cm and 100 cm depths for all sites with the exceptions of riparian edge (48 m) and the hyporheic zone (50 m) where mean SRP concentrations were higher at transect T4 than T1 during all flow conditions. The mean SRP concentrations at 150 cm were higher at transect T1 than T4 for all sites but reversed at the riparian edge (48 m) during baseflow. This trend was not observed during flood (Figure 3-20).

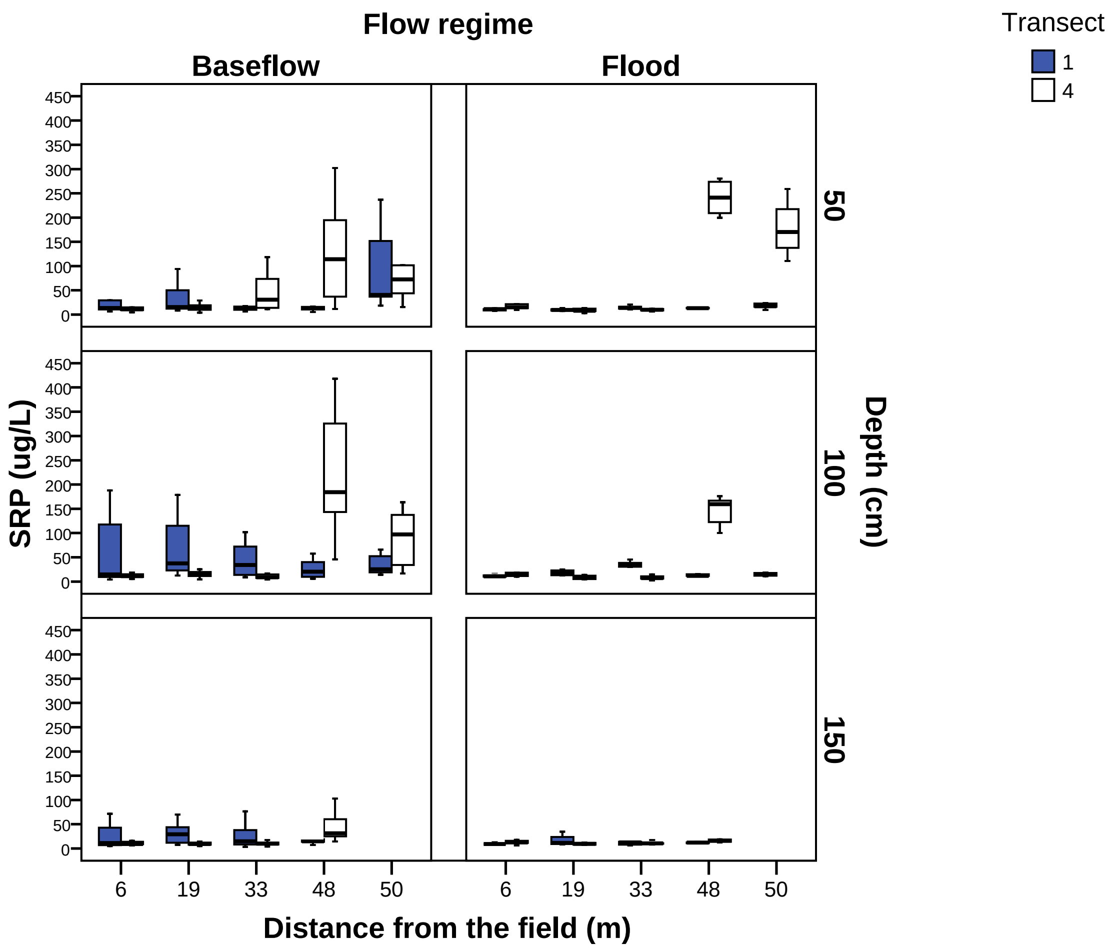


Figure 3-20 Spatial variability of groundwater SRP concentration under different flow regimes between transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

3.3.2.2 Temporal variability

The mean SRP concentration at two transects were higher in June, July and August than May except during large storm events (i.e., JD 187). The mean SRP concentration was consistently low at all depths at transect T1 but peaked at 50 and 100 cm at T4 in response to the flood (Figure 3-21).

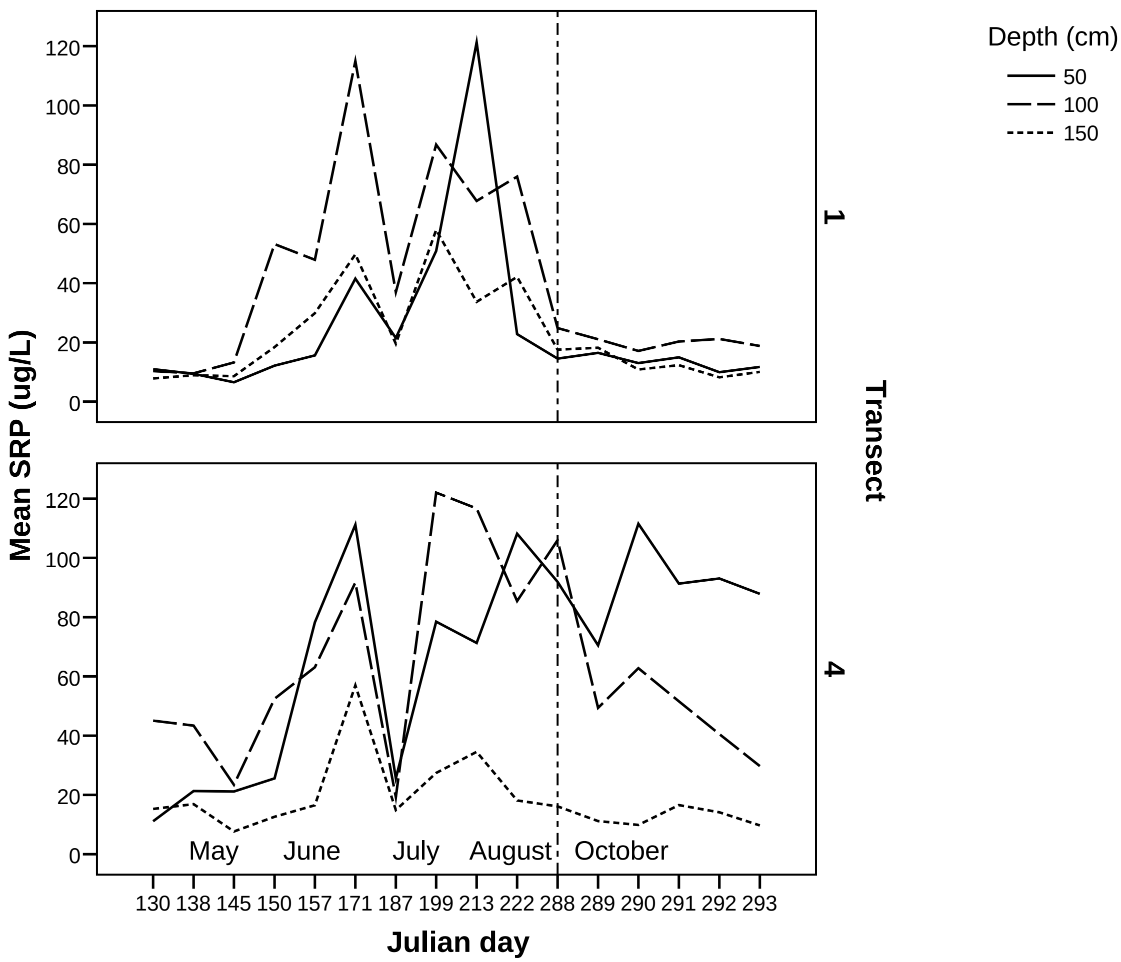


Figure 3-21 Temporal variability of groundwater SRP concentration under different flow regimes (vertical dashed lines separate baseflow and flood periods.)

3.4 Related Groundwater Chemistry

3.4.1 pH

Groundwater pH was consistently near neutral or a slightly alkaline, ranging from 6.71 to 8.37 (Table 3-5). Graphs of pH by transect and flow regime at the 50 cm, 100 cm, 150 cm depths and differences in pH by depth and flow regime are shown in Figure 3-22 and 3-23, respectively. Time-series plots are used to illustrate temporal patterns of pH for the study period in Figure 3-24.

Table 3-5 Descriptive statistics of groundwater pH

Transect	Flow regime	n	Mean	SEM	Min	Max	SD	Range
1	Baseflow	140	7.40	0.02	6.71	7.93	0.24	1.22
	Flood	69	7.57	0.04	6.89	8.37	0.36	1.48
4	Baseflow	134	7.56	0.02	6.87	8.19	0.25	1.32
	Flood	63	7.55	0.03	6.89	8.10	0.25	1.21

* n is sample size; SEM is standard error of mean; Min and Max refer to minimum and maximum, respectively; SD is standard deviation; Range = (Max-Min).

3.4.1.1 Spatial variability

1) Intra-transect pattern

At transect T1 during baseflow, the mean pH of groundwater generally increased with increasing distance from the field edge (T1-6) to the hyporheic zone (T1-50). This pattern was more pronounced in shallow groundwater than deeper layers, but was not observed during flood (Figure 3-22). At transect T4, similar trend with distance existed at 100 and 150 cm depths with inconsistent low pH at the edge of riparian zone (T4-48) during all flow conditions (Figure 3-22).

The mean pH generally increased with depth and trended $pH_{150cm} > pH_{100cm} > pH_{50cm}$ at transect T1 during all flow conditions. In contrast, they were prevalently higher in shallow groundwater than deeper layer at T4. Higher variability in groundwater pH was observed during flood at transect T1, but this was not found at T4 (Figure 3-22).

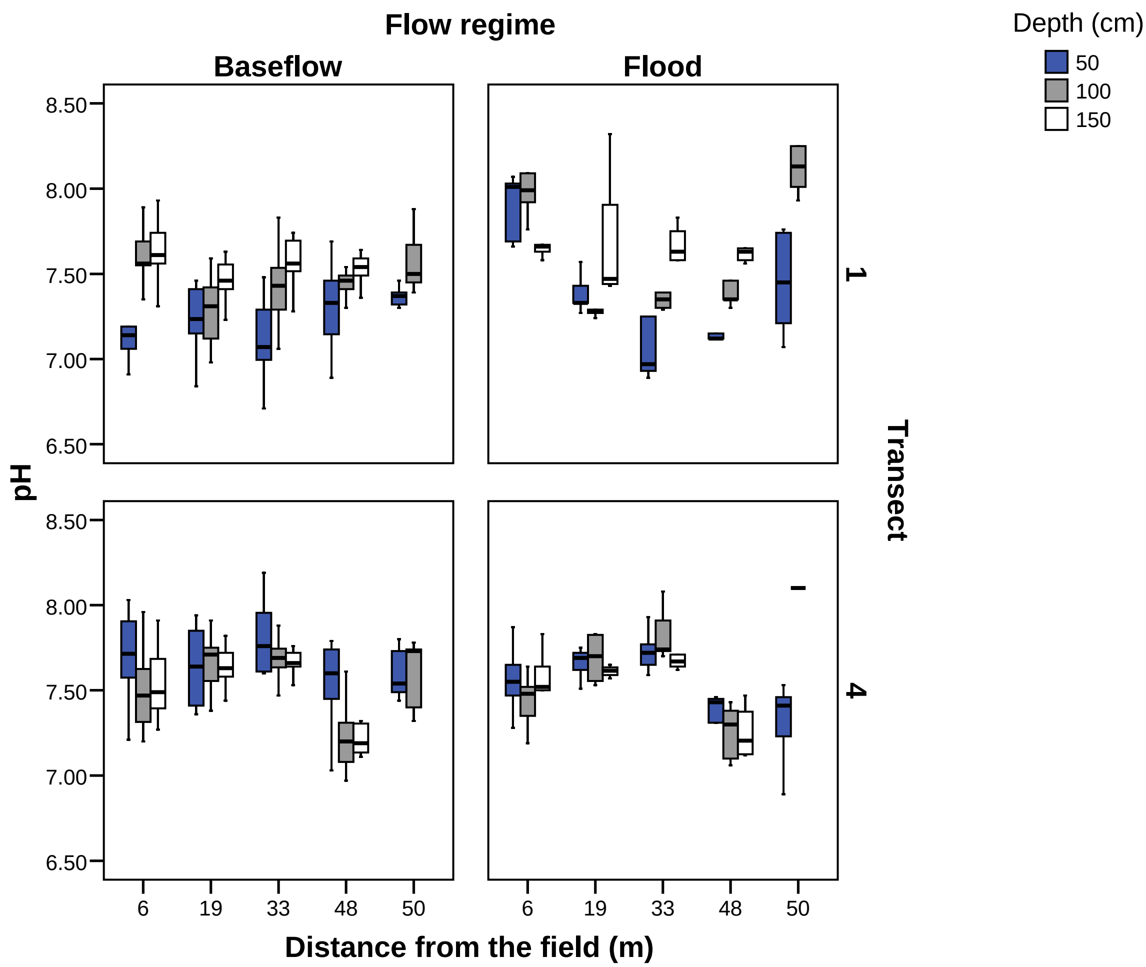


Figure 3-22 Spatial variability of groundwater pH under different flow regimes within transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

2) Inter-transect pattern

During baseflow, the mean pH at 50 cm was always higher at transect T4 than T1. This pattern was not as prevalent at 100 and 150 cm as at 50 cm. Similar trend existed but became weak even reversed at some sites (e.g., the field edge (6 m), riparian edge (48 m) and hyporheic zone (50 m)) during flood (Figure 3-23).

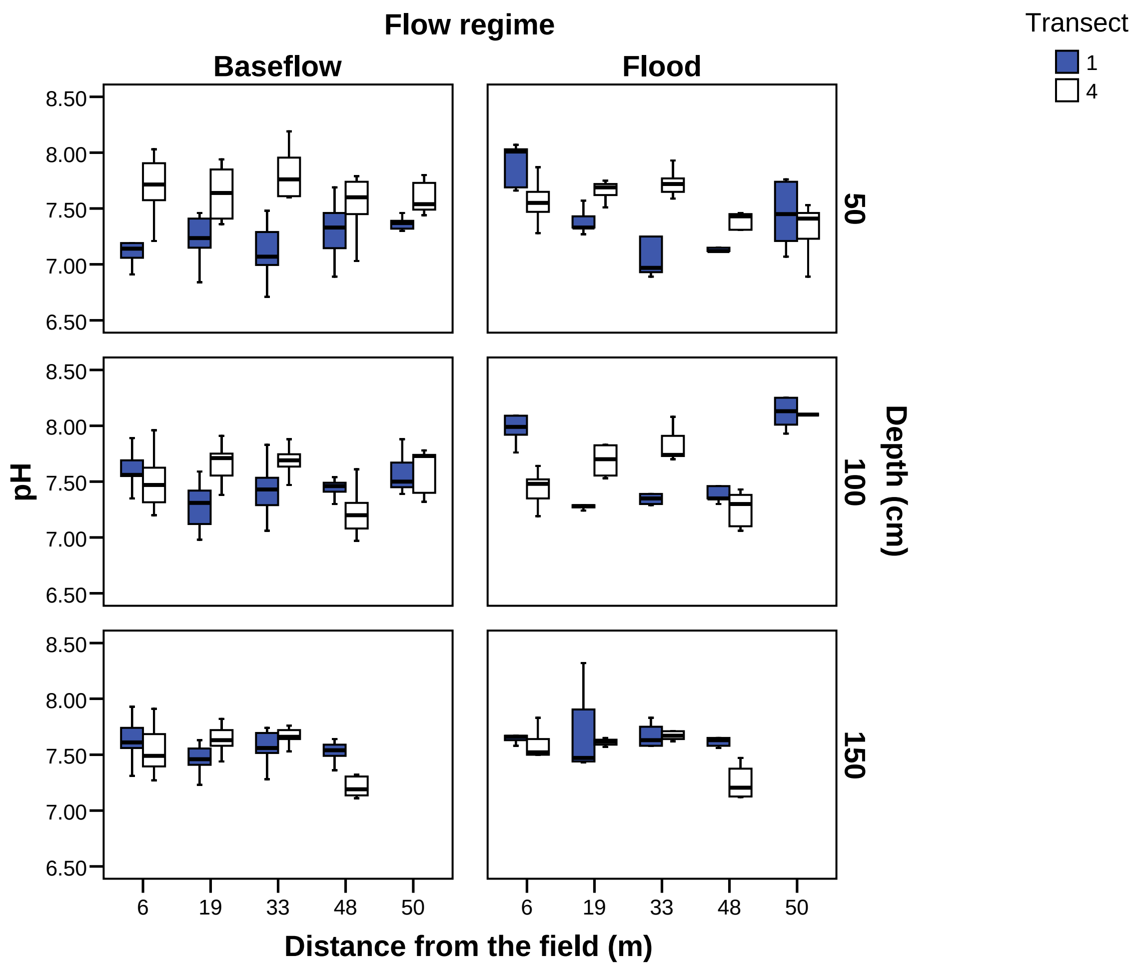


Figure 3-23 Spatial variability of groundwater pH under different flow regimes between transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

3.4.1.2 Temporal variability

Temporal patterns in mean pH at two transects were similar. During baseflow, the mean pH generally peaked in late June then dropped in mid July in response to storm events (Figures 3-1 and 3-24) and rose again from August to the pre-flood day in October. During flood, the mean pH at transect T1 consistently increased in response to flood but it tended to decrease at T4 with an exception of pH at 150 cm that peaked during flood (Figure 3-24).

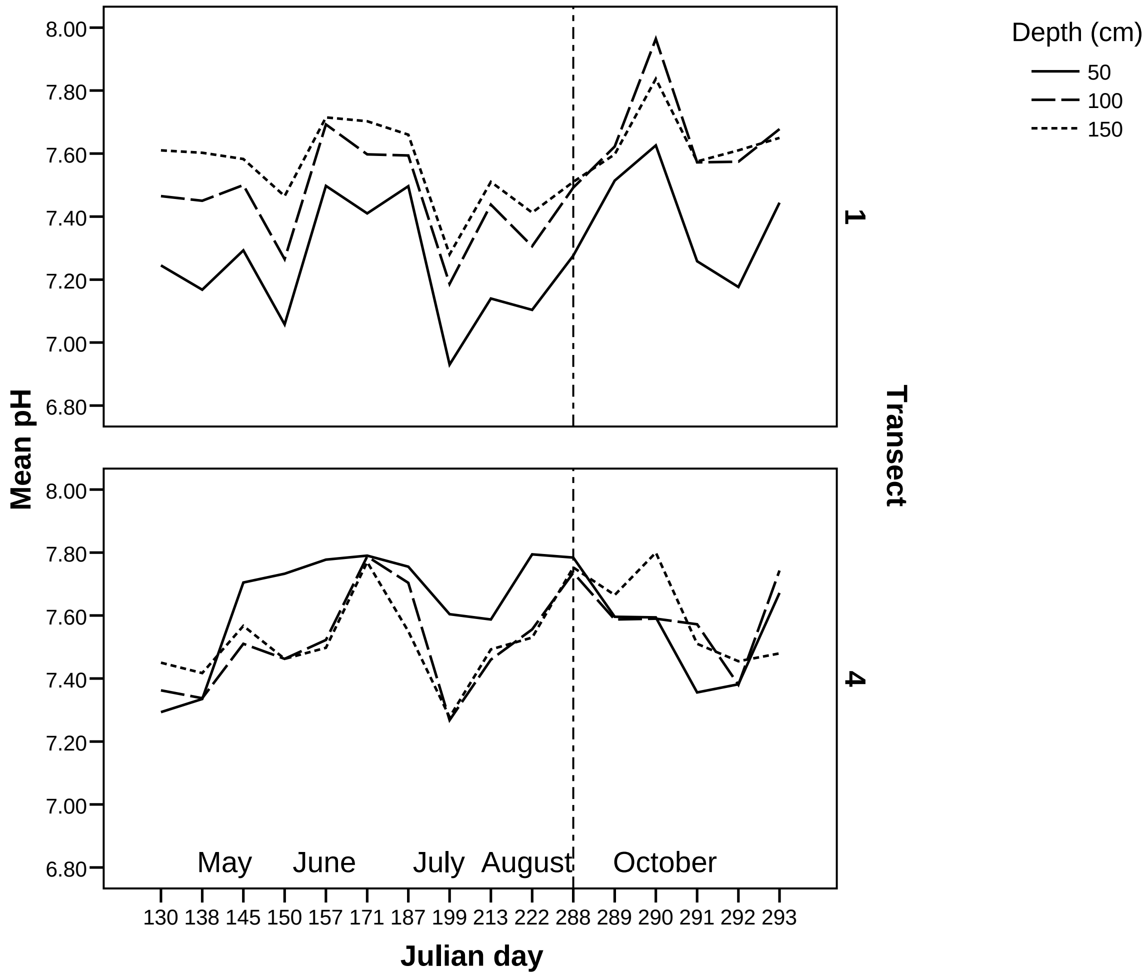


Figure 3-24 Temporal variability of groundwater pH under different flow regimes (vertical dashed lines separate baseflow and flood periods.)

3.4.2 Dissolved oxygen

Groundwater dissolved oxygen (DO) was highly variable, ranging from as low as less than 0.1 mgL^{-1} to higher than 27 mgL^{-1} (Table 3-6). Graphs of groundwater DO by transect and flow regime for 50 cm, 100 cm and 150 cm depths and differences in groundwater DO by depth and flow regime are shown in Figure 3-25 and 3-26, respectively. Time-series plots are used to illustrate temporal patterns of groundwater DO for the study period in Figure 3-27.

Table 3-6 Descriptive statistics of groundwater DO (mgL^{-1})

Transect	Flow regime	n	Mean	SEM	Min	Max	SD	Range
1	Baseflow	26	0.5	0.1	<0.1	2.9	0.6	2.9
	Flood	67	1.3	0.2	<0.1	7.5	2.0	7.5
4	Baseflow	18	1.0	0.4	0.1	6.0	1.7	5.9
	Flood	57	2.4	0.7	<0.1	27.0	5.2	27.0

* n is sample size; SEM is standard error of mean; Min and Max refer to minimum and maximum, respectively; SD is standard deviation; Range = (Max-Min). "<0.1" the measurement was less than detection limit 0.1 mgL^{-1}.

3.4.2.1 Spatial variability

1) Intra-transect pattern

Groundwater was consistently oxygen-poor at transect T1 with an exception of the hyporheic zone (T1-50). The mean DO concentration was also low at transect T4 at most sites except the riparian edge (T4-48). The flood increased DO at 50 cm and to a lesser extent 100 cm but did not affect 150 cm at two transect. Effects of flood were more dramatic at transect T4 than T1 (Three piezometers (T4-19-100, T4-19-150 and T4-48-150) were submerged from the tips by flood water during the peak flood day (JD 290). All data from these piezometers on JD290 were removed before analysis). There was no consistent pattern with depth at two transects during flood. The highest mean DO concentrations were always found at 50 cm in all sites at transect T1 with the exception of the hyporheic zone (T1-50) where the mean DO was higher at 100 cm than 50 cm, while the highest mean DO concentrations were always found at 150 cm at T4 with the exception in the riparian edge (T4-48) where the mean DO trended $DO_{50cm} > DO_{100cm} > DO_{150cm}$ (Figure 3-25).

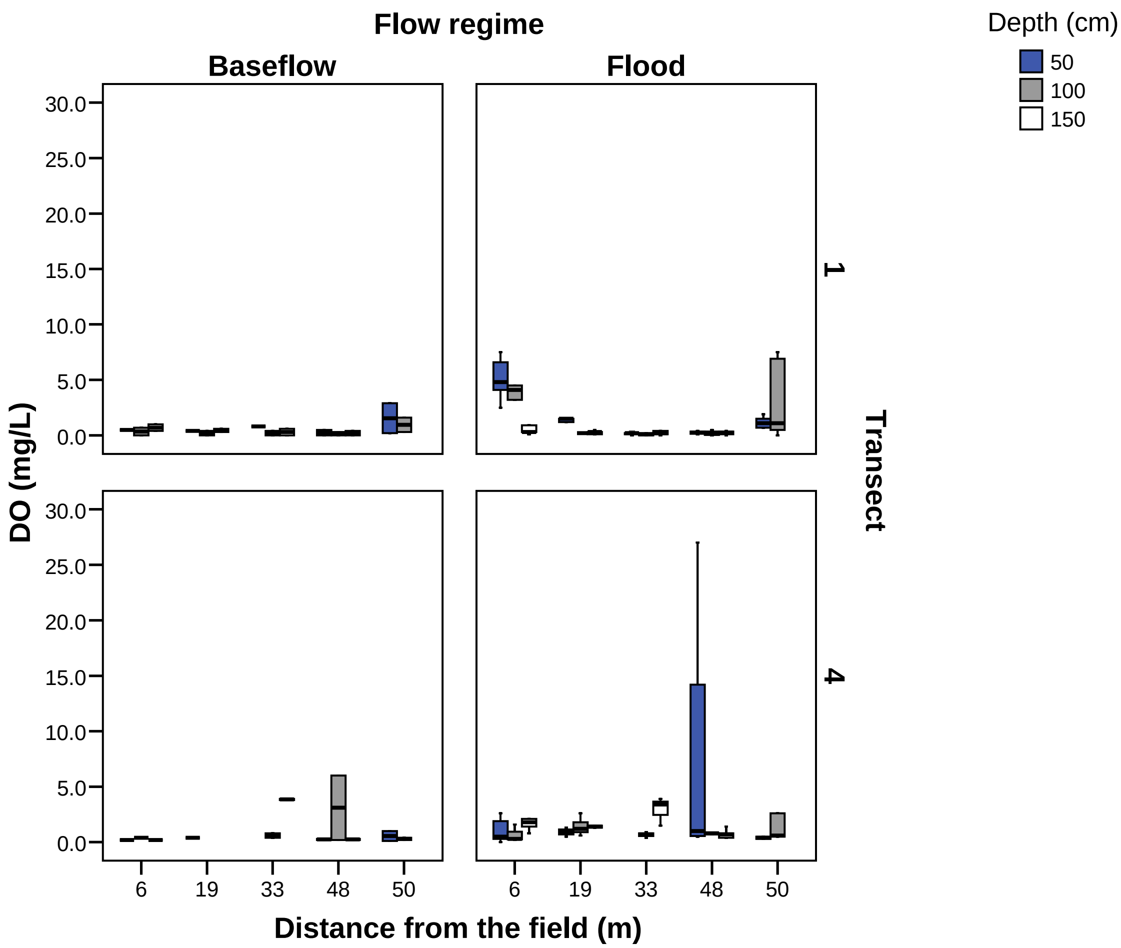


Figure 3-25 Spatial variability of groundwater DO under different flow regimes within transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

2) Inter-transect pattern

During flood, the mean DO at 100 and 150 cm was always lower at transect T1 than T4 with only one exception of the field edge (6 m) at 100 cm where this pattern reversed. However, the trends in DO at 50 cm during flood were less clear. For instance, the mean DO at transect T1 were higher than T4 at some sites (e.g., the field edge (6 m), near middle riparian (19 m) and hyporheic zone (50 m)) but were lower than T4 at other sites (e.g., far middle riparian zone (33 m) and the riparian edge (48 m)) (Figure 3-26). The inter-transect patterns during baseflow were not examined due to limited data.

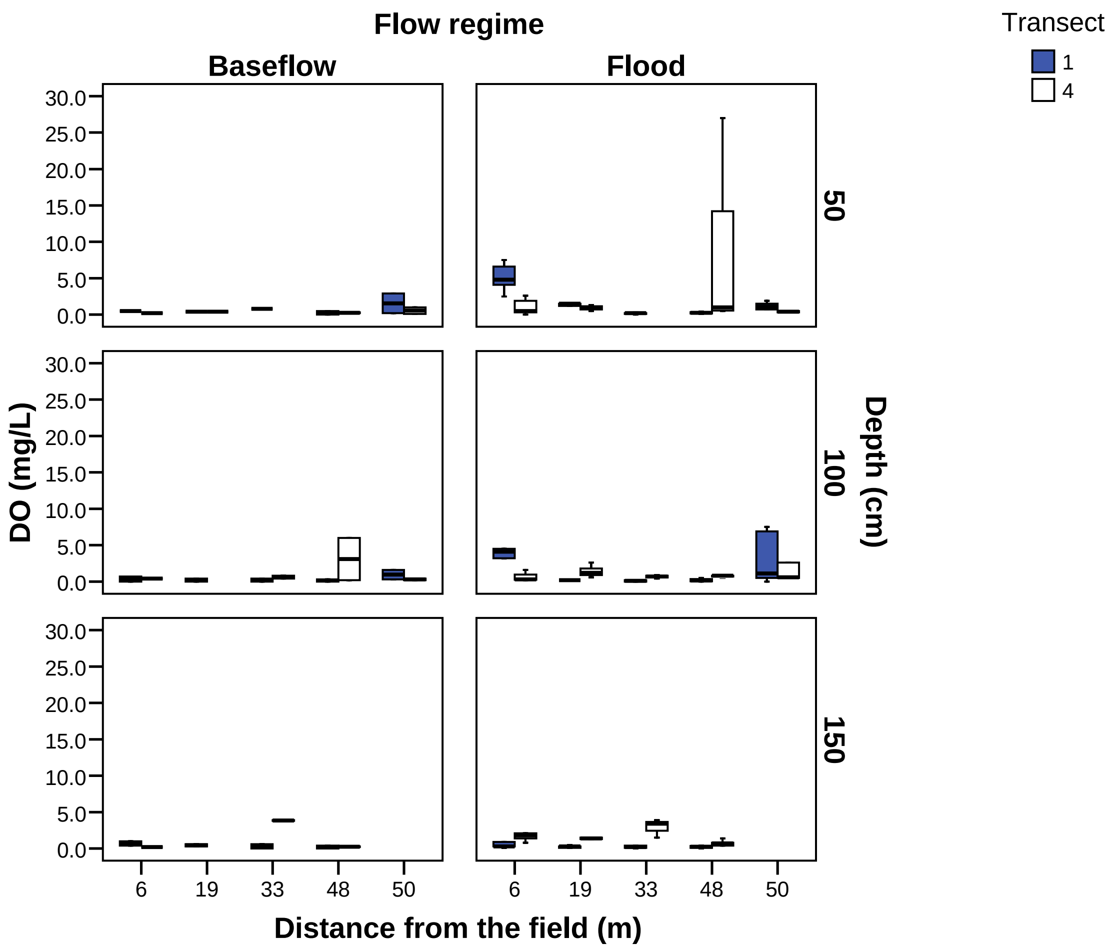


Figure 3-26 Spatial variability of groundwater DO under different flow regimes between transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

3.4.2.2 Temporal variability

There was a clear flood effect on groundwater DO concentrations although this was more apparent at transect T4 than T1. The mean DO at two transects generally increased from July to pre-flood condition in October during baseflow, and this was more pronounced at 50 cm than 100 and 150 cm. The mean DO at two transects peaked in respond to flood but the peaks were obviously higher at transect T4 than T1. Flood had larger effects on DO concentration in shallow groundwater than deeper one (Figure 3-27).

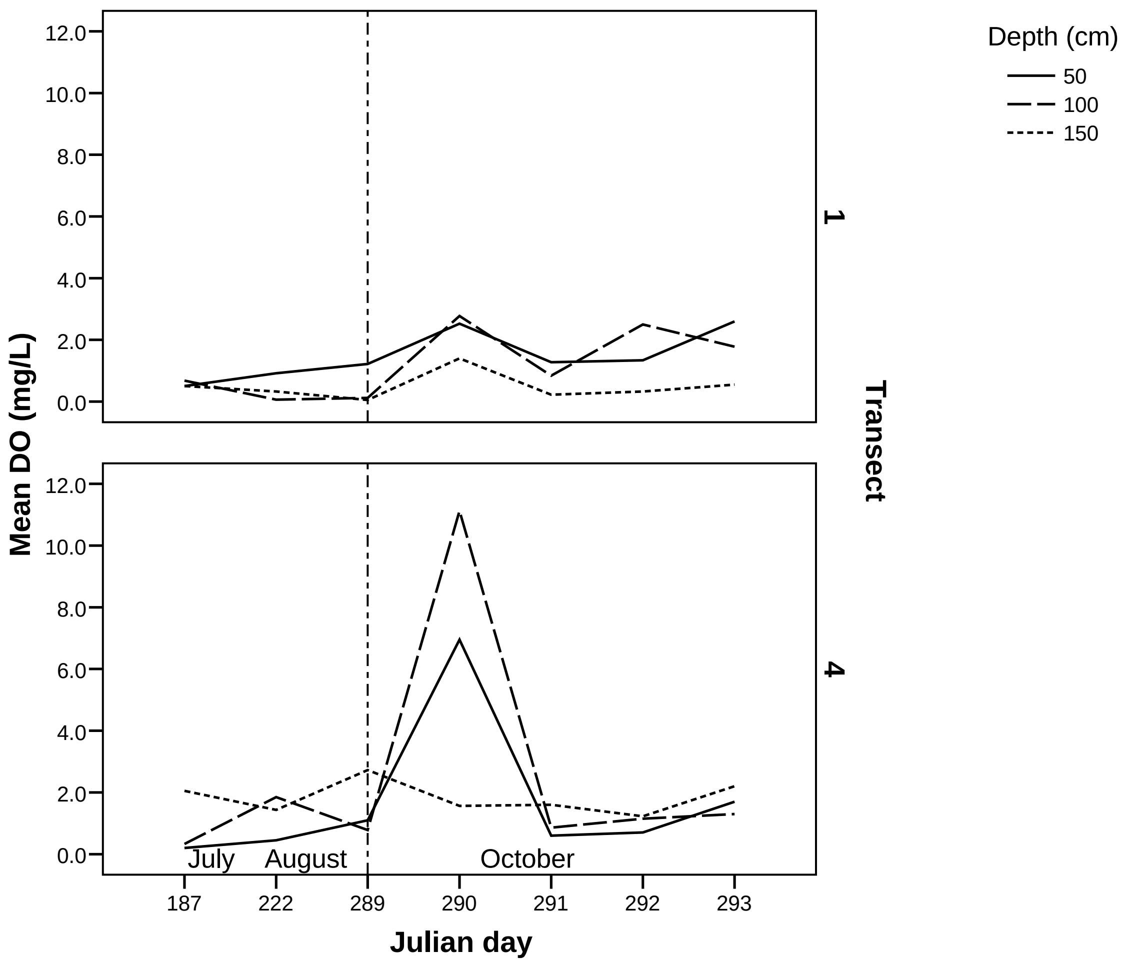


Figure 3-27 Temporal variability of groundwater DO under different flow regimes (vertical dashed lines show the start of flooding)

3.5 Stream Water Phosphorus

Stream water phosphorus concentrations in Spencer Creek varied both spatially and temporally in response to flow conditions over the study period.

3.5.1 Total phosphorus

Total phosphorus concentrations in stream water were highly variable in response to flood, ranging from < 1 μgL^{-1} to >12000 μgL^{-1} (Table 3-7). Differences in stream water TP concentrations by transect and temporal patterns during the study period are shown in Figures 3-28 and Figure 3-29, respectively.

Table 3-7 Descriptive statistics of stream water TP concentration (μgL^{-1})

Transect	Flow regime	n	Mean	SEM	Min	Max	SD	Range
1	Baseflow	11	26.0	4.4	6.0	56.9	14.7	50.9
	Flood	5	15.8	6.2	<1	36.8	13.8	36.8
4	Baseflow	9	21.6	4.3	2.4	39.8	12.9	37.4
	Flood	5	2503.7	2383.2	<1	12031.8	5328.9	12031.8

* n is sample size; SEM is standard error of mean; Min and Max refer to minimum and maximum, respectively; SD is standard deviation; Range = (Max-Min). "<1" the measurement was less than detection limit 1 μgL^{-1}.

3.5.1.1 Variability along the stream reach

During baseflow, all TP concentrations were < 60 μgL^{-1}. During flood, TP at T4 was two orders of magnitude higher than T1 due to extremely high values during the peak flood (JD290 and JD291). Otherwise, TP concentrations were always < 60 μgL^{-1} throughout the study period (Figure 3-28).

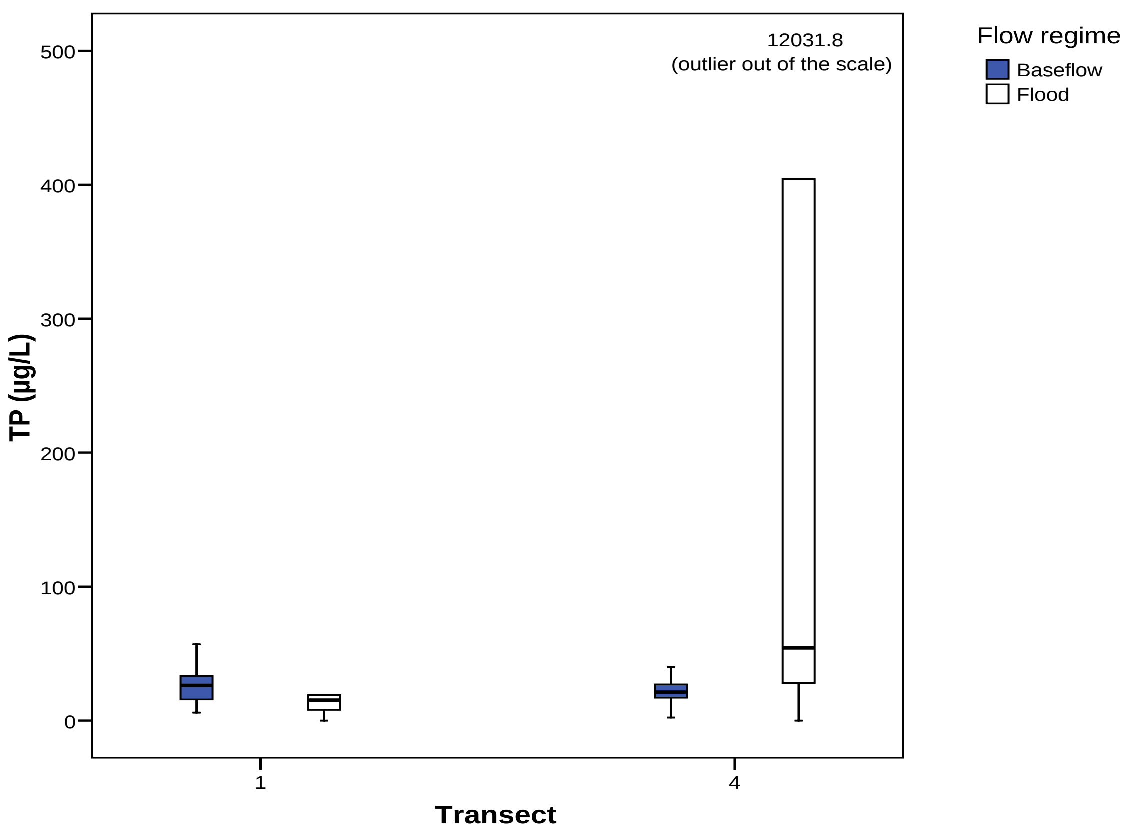


Figure 3-28 Variability of stream water TP concentration under different flow regimes along the reach (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

3.5.1.2 Temporal variability

Mean TP concentrations were consistently low at the two transects during baseflow and remained low at transect T1 during flood but increased dramatically on the peak flood day-JD290 then dropped sharply and returned to the previous low level on JD292 (Figure 3-29).

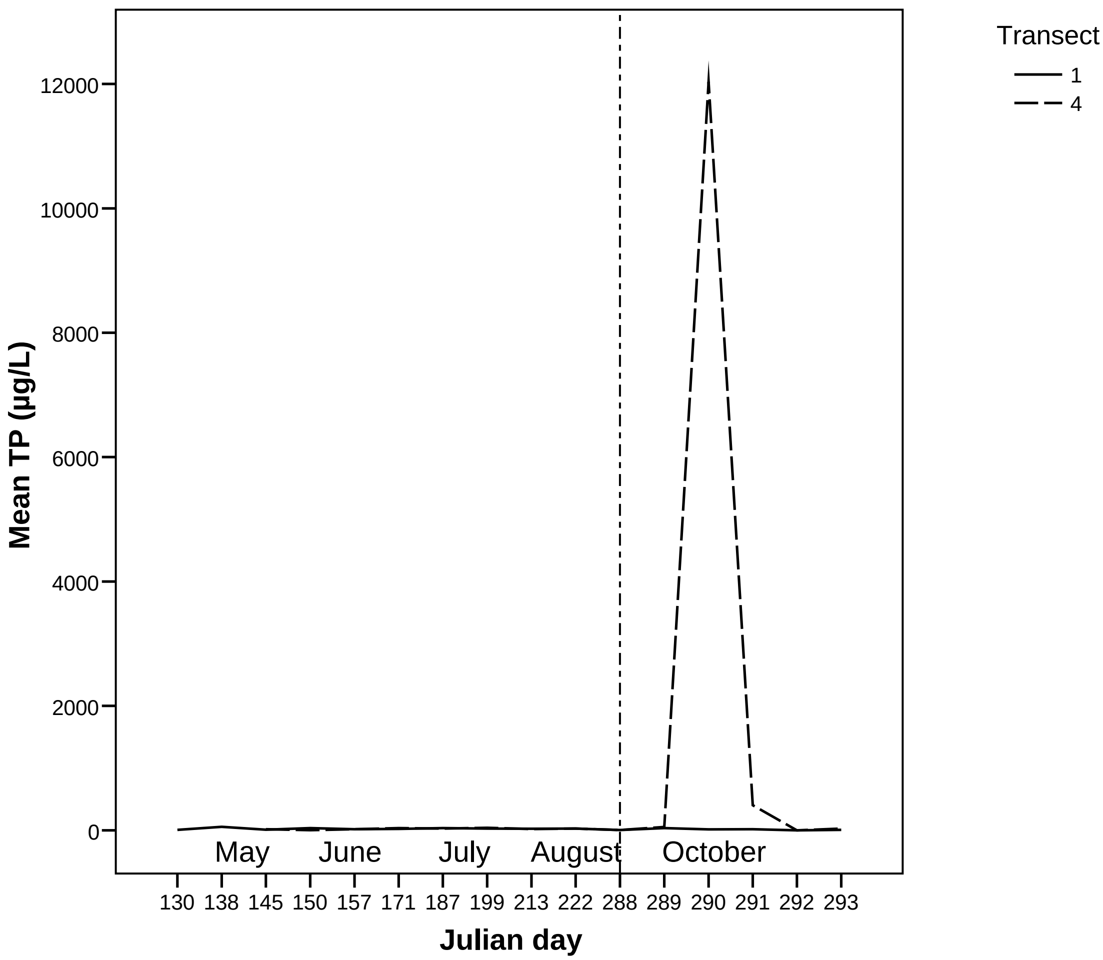


Figure 3-29 Temporal variability of stream water TP concentration under different flow regimes (vertical dashed line separates baseflow and flood periods.)

3.5.2 Soluble reactive phosphorus

Soluble reactive phosphorus concentrations in stream water were less variable compared to TP, which ranged from 1.2 µgL^{-1} to > 20.2 µgL^{-1}, although it did change in response to flood as well (Table 3-8). Graphs of stream water SRP concentrations by transect is shown in Figure 3-30. Time-series plots are used to illustrate temporal patterns of stream water SRP concentrations for the study period in Figure 3-31.

Table 3-8 Descriptive statistics of stream water SRP concentration (μgL^{-1})

Transect	Flow regime	n	Mean	SEM	Min	Max	SD	Range
1	Baseflow	11	9.5	1.0	4.2	14.4	3.4	10.2
	Flood	5	7.7	1.5	3.4	12.2	3.4	8.8
4	Baseflow	9	9.9	1.5	5.1	20.2	4.5	15.1
	Flood	5	5.9	1.3	1.2	8.6	2.9	7.4

* n is sample size; SEM is standard error of mean; Min and Max refer to minimum and maximum, respectively; SD is standard deviation; Range = (Max-Min).

3.5.2.1 Variability along the stream reach

The mean SRP concentration at transect T1 were similar with T4 during all flow conditions (Figure 3-30).

3.5.2.2 Temporal variability

The mean SRP concentration in stream water at two transects had similar temporal patterns. They were higher in late June than May, July, August during baseflow and October during flood as a result of drought. The mean SRP concentration generally decreased at two transects during flood but tended to recover the level prior to the flood event (Figure 3-31).

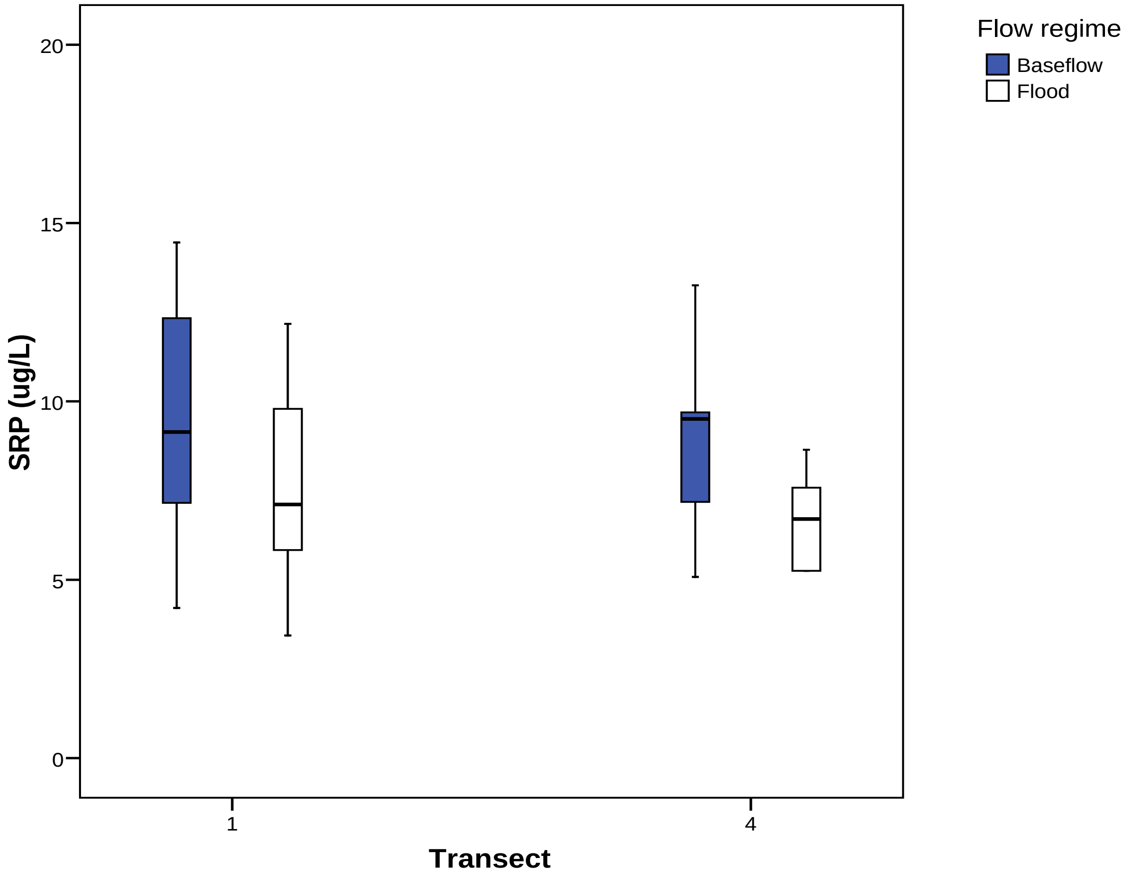


Figure 3-30 Variability of stream water SRP concentration under different flow regimes along the reach (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

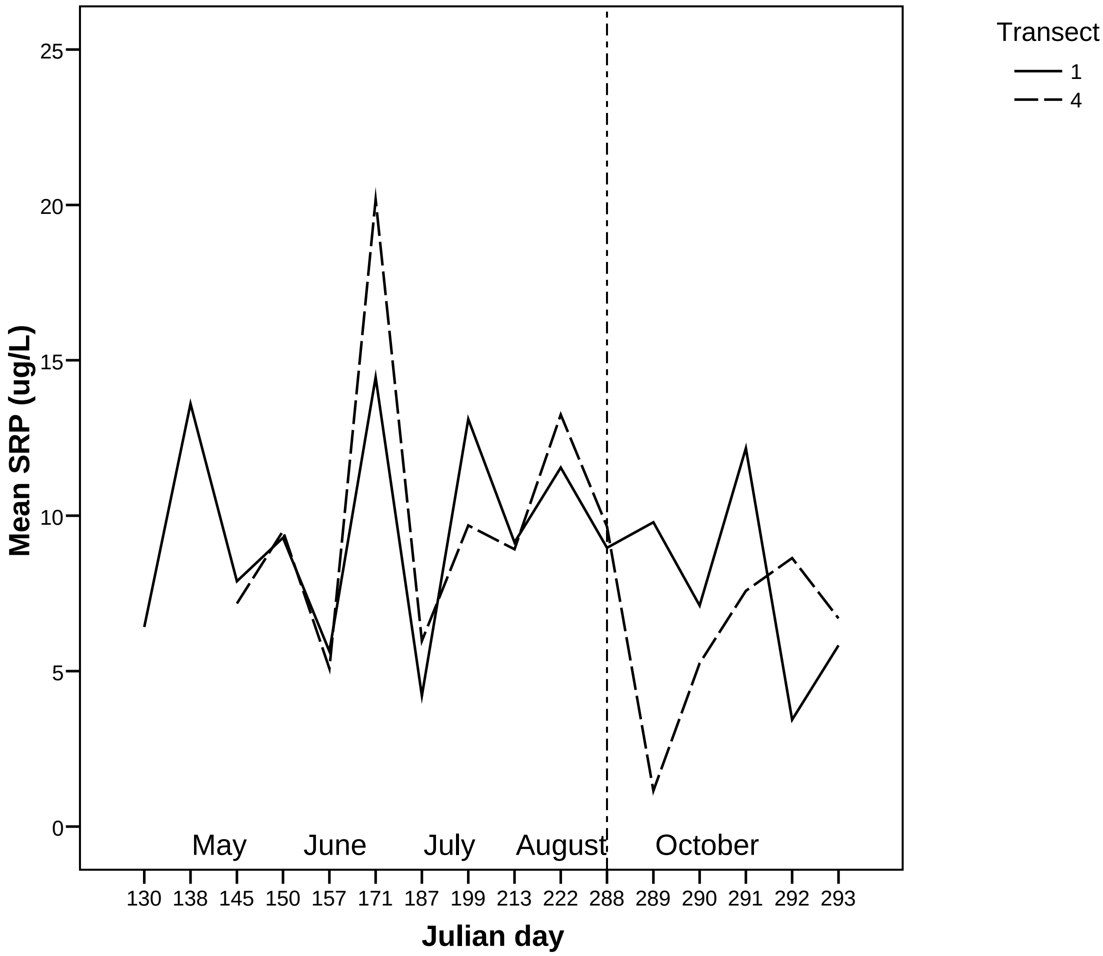


Figure 3-31 Temporal variability of stream water SRP concentration under different flow regimes (vertical dashed line separates baseflow and flood periods.)

3.6 Soil Extractable Phosphorus

Soil extractable phosphorus (soil SRP) concentrations slightly varied spatially in the study site, ranging between 0.4 μgg^{-1} soil to 4.6 μgg^{-1} soil (Table 3-9). Graphs of soil SRP concentrations by transect and by depth are shown in Figure 3-32 and 3-33, respectively.

Table 3-9 Descriptive statistics of soil SRP concentration (μgg^{-1} soil)

Transect	n	Mean	SEM	Min	Max	SD	Range
1	18	1.3	0.2	0.4	4.6	1.0	4.2
4	18	1.2	0.1	0.6	2.2	0.4	1.6

* n is sample size; SEM is standard error of mean; Min and Max refer to minimum and maximum, respectively; SD is standard deviation; Range = (Max-Min).

3.6.1 Spatial variability

1) Intra-transect pattern

There was a distance trend in soil extractable-P along transect T1, where SRP was higher at the field edge (T1-2) and lower near stream (T1-48). This trend was not observed at transect T4. There was a depth trend at transect T1 with higher P at surface (0-8 cm). This trend was seen at mid riparian (T4-25) and near stream (T4-48) but not at the field edge (T4-2) at T4 (Figure 3-32).

2) Inter-transect pattern

At surface soil (0-8 cm), SRP was higher at transect T4 than T1 except the field edge (2 m). At deeper soil (17-22 cm), SRP was also higher at transect T4 than T1 except the middle riparian zone (25 m) (Figure 3-33).

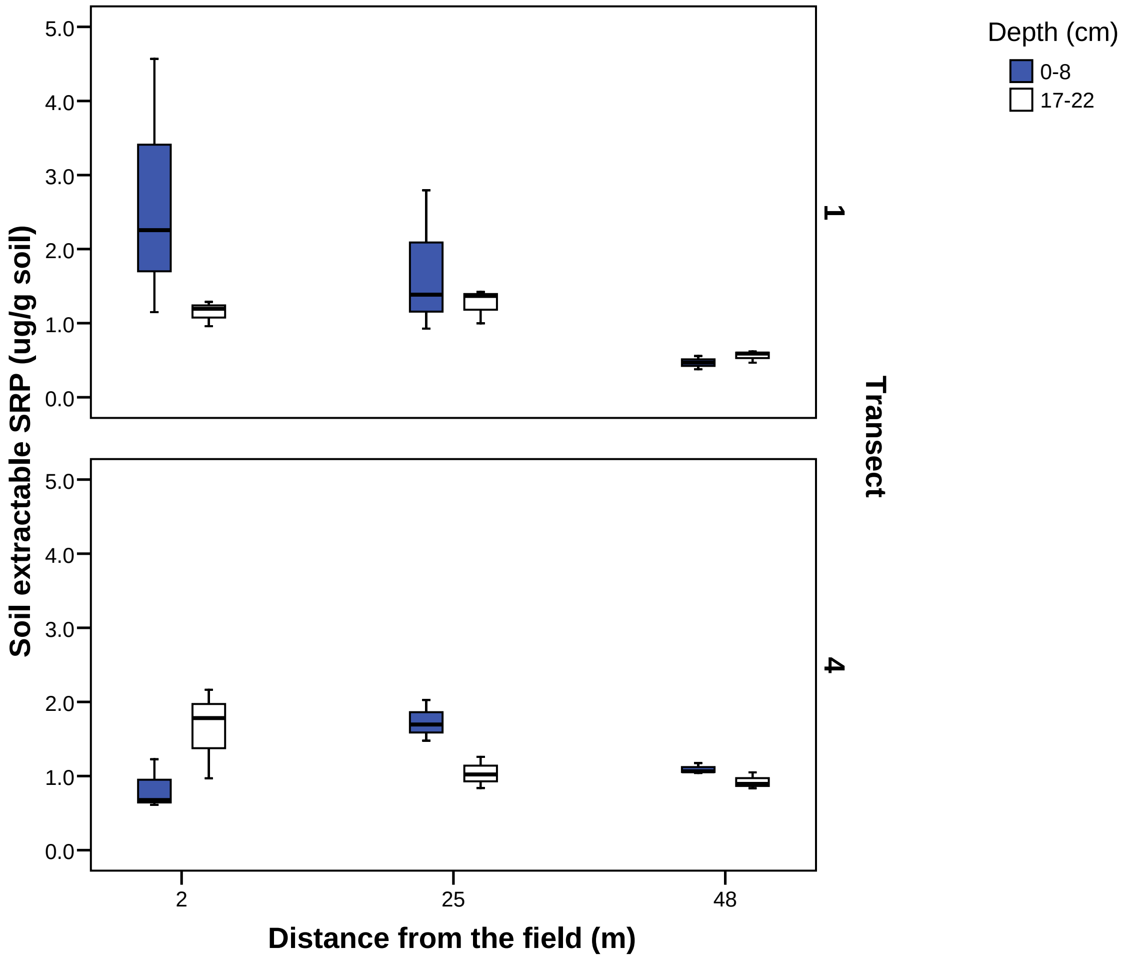


Figure 3-32 Spatial variability of soil SRP concentration within transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

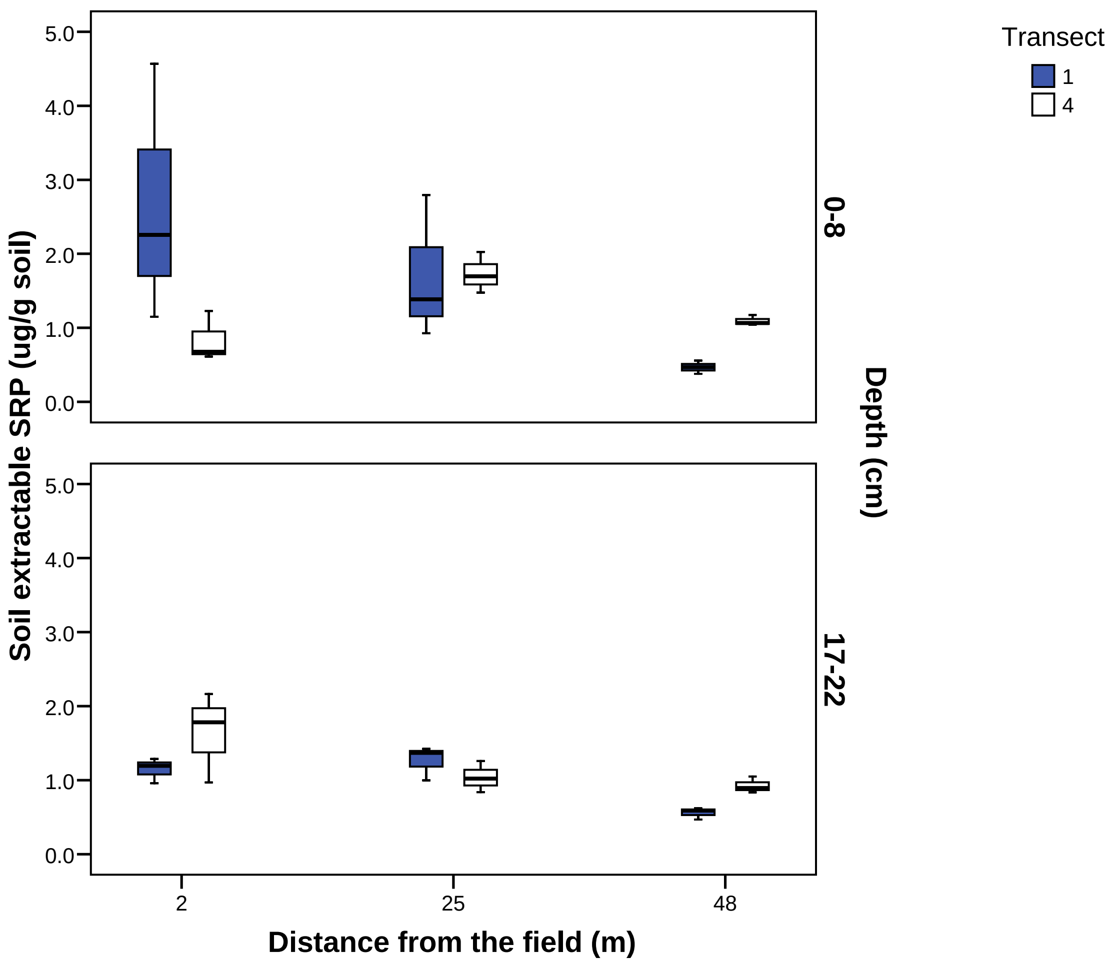
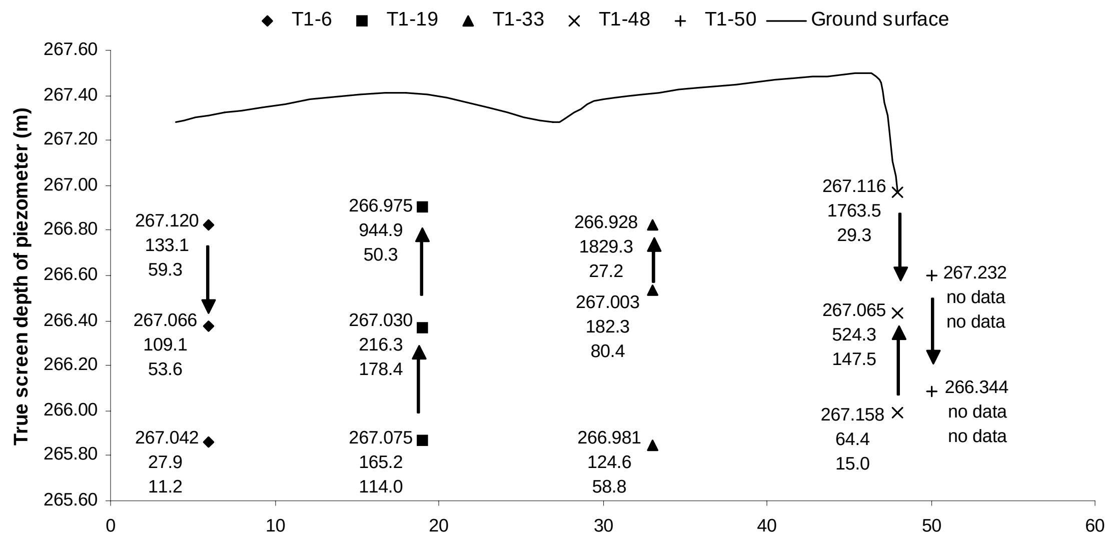


Figure 3-33 Spatial variability of soil SRP concentration between transects (Boxes show the 25, 50, and 75th percentiles. The 10th and 90th percentiles are shown by whiskers.)

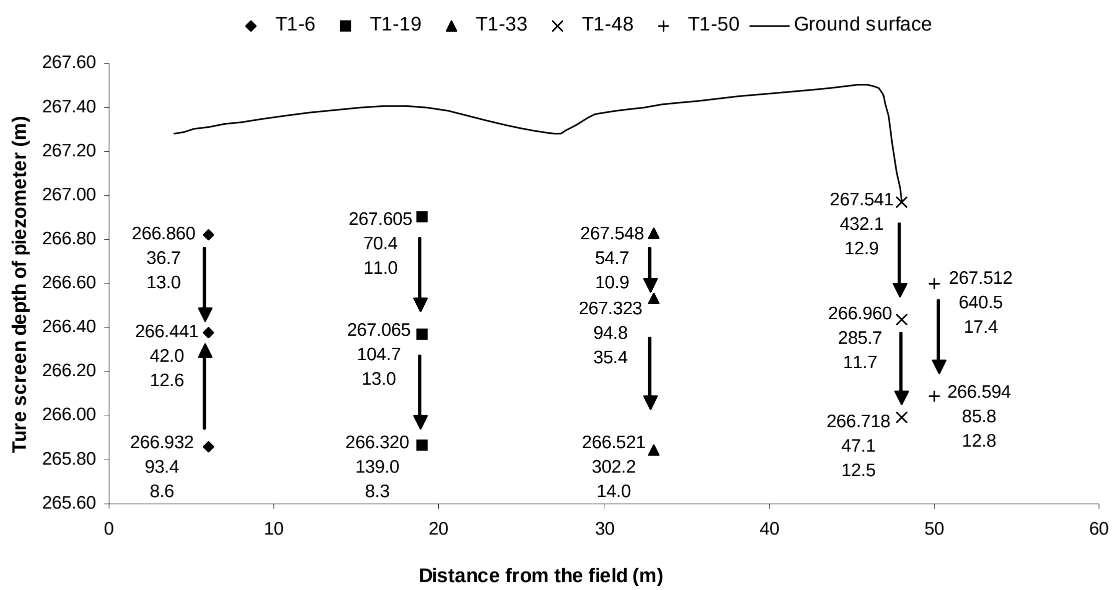
3.7 Phosphorus Mass Flux

3.7.1 Phosphorus flux through the riparian zone

Lateral groundwater flow likely existed between T1 and T4 (Figure 3-36). The P flux along the representative cross-section was estimated based on the assumption of unchanging groundwater flux from T1 to T4 (Table 3-10 and 3-11).

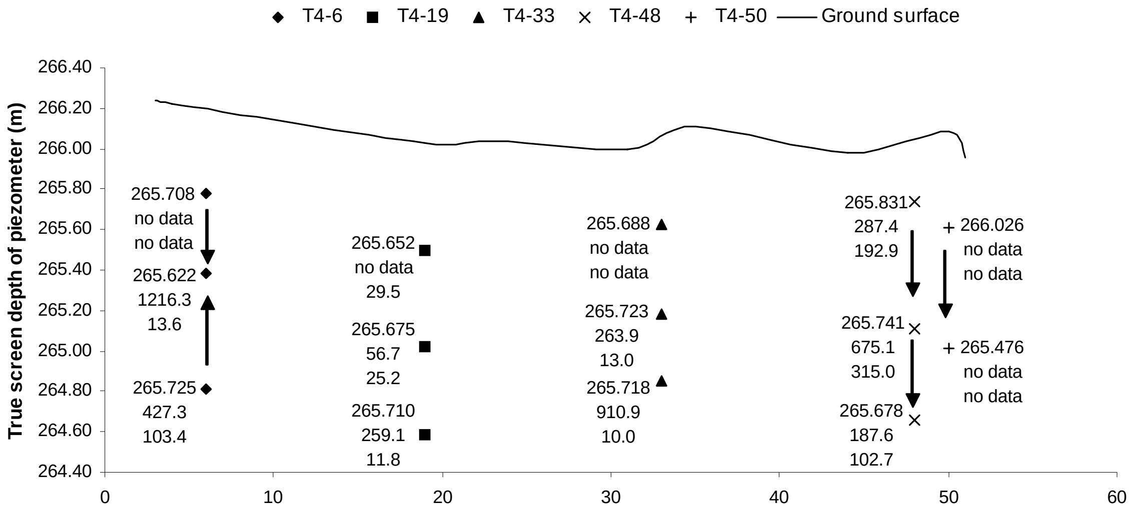


(a)

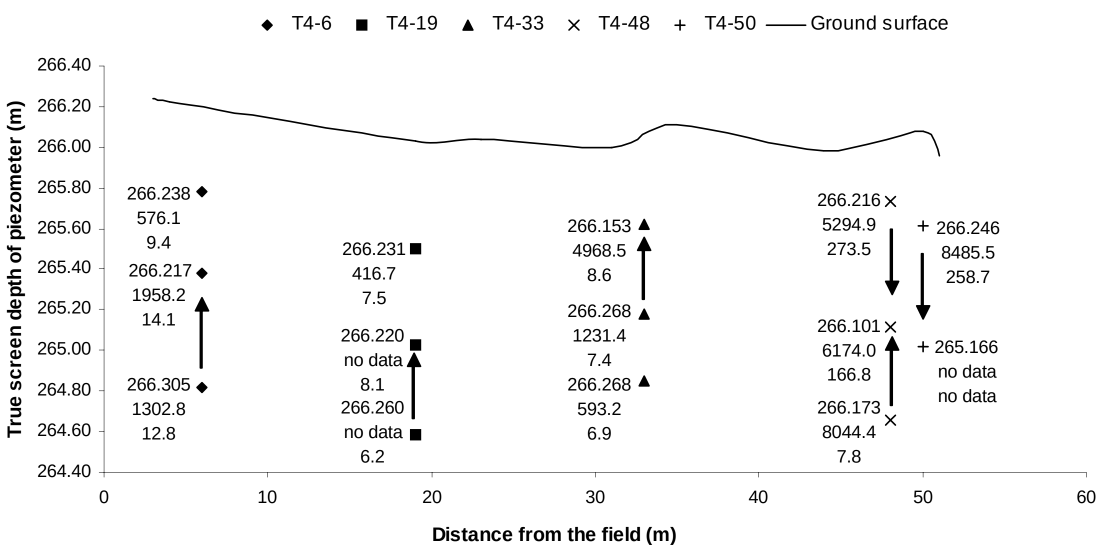


(b)

Figure 3-34 Hydraulic head and P flux in transect T1 under different flow regimes. (a) dry day during baseflow and (b) peak flood day. Arrows indicate possible flow directions. Three numbers beside each symbol are hydraulic head, TP and SRP concentrations in descending order.



(a)



(b)

Figure 3-35 Hydraulic head and P flux in transect T4 under different flow regimes. (a) dry day during baseflow and (b) peak flood day. Arrows indicate possible flow directions. Three numbers beside each symbol are hydraulic head, TP and SRP concentrations in descending order.

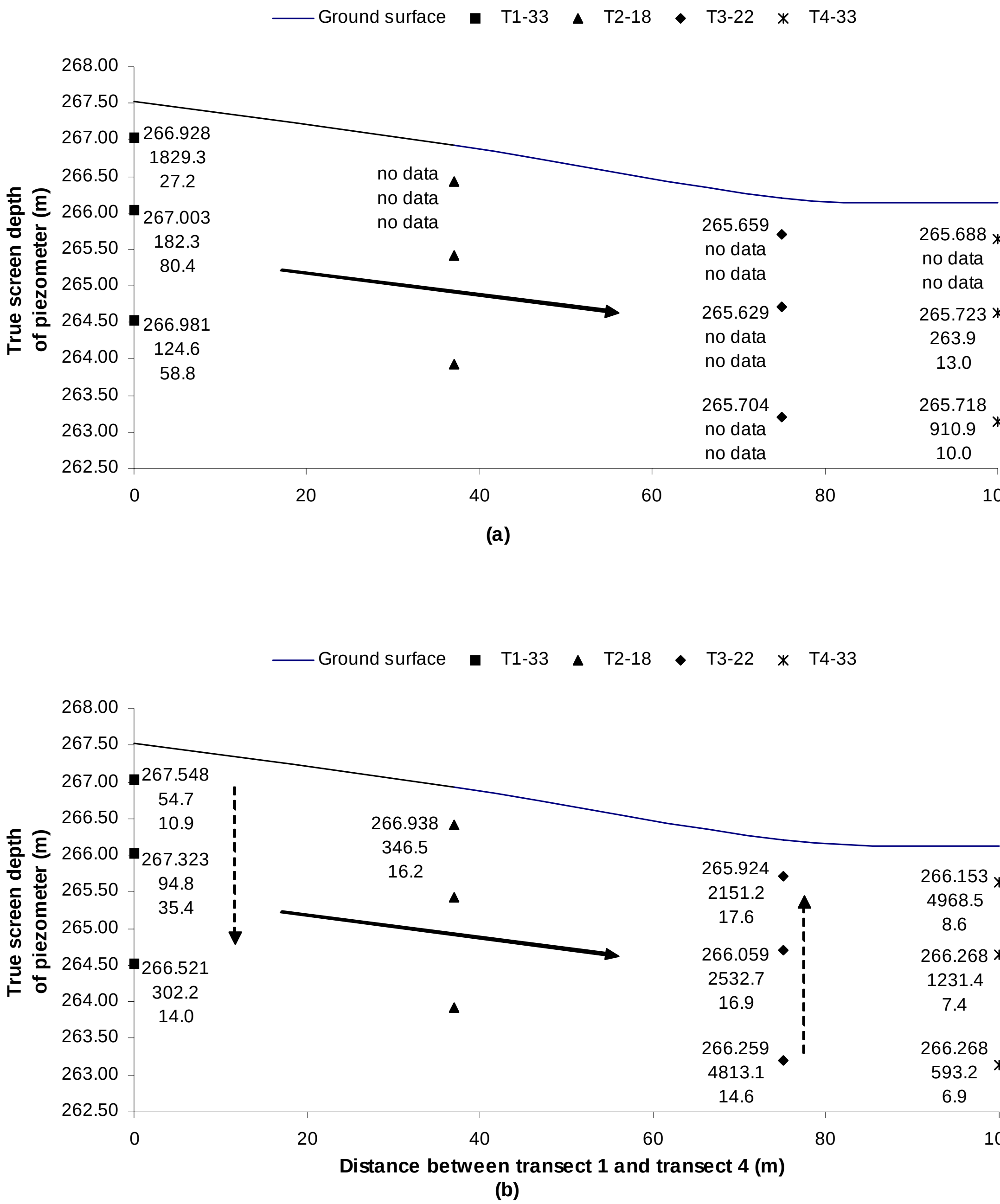


Figure 3-36 Hydraulic head and P flux in the representative cross-section between transect T1 and T4 under different flow regimes. (a) dry day during baseflow and (b) peak flood day. Arrows indicate possible flow directions. Three numbers beside each symbol are hydraulic head, TP and SRP concentrations in descending order.

Table 3-10 P flux through the riparian zone in shallow groundwater (50 cm depth) during baseflow (JD 157)

Piezometer ID	K_{sat} (cms^{-1})	LHG	Riparian width (cm)	Groundwater flux (Ld^{-1})	Groundwater Phosphorus (μgL^{-1})		P flux between T1 and T4 ($\mu gm^{-2}d^{-1}$)	
					TP	SRP	TP	SRP
T1-33-P50	6.05E-04	1.61E-02	5000	14.7	182.7	17.4	2218.2	1295.1
T4-33-P50	6.65E-05	1.61E-02	5000	14.7	333.6	105.5		

Table 3-11 P flux through the riparian zone in shallow groundwater (50 cm depth) during peak flood (JD 290)

Piezometer ID	K_{sat} (cms^{-1})	LHG	Riparian width (cm)	Groundwater flux (Ld^{-1})	Groundwater Phosphorus (μgL^{-1})		P flux between T1 and T4 ($\mu gm^{-2}d^{-1}$)	
					TP	SRP	TP	SRP
T1-33-P50	6.05E-04	1.74E-02	5000	25.0	54.7	10.9	122845.0	-57.1
T4-33-P50	6.65E-05	1.74E-02	5000	25.0	4968.5	8.6		

The lateral hydraulic gradient (LHG) between transect T1 and T4 increased during flood conditions compared to baseflow but the LHG magnitudes were similar (Tables 3-10 and 3-11). The flux of TP through the riparian zone along the representative cross-section between transects (T1 and T4) was two orders of magnitude higher during flood than baseflow (Tables 3-10 and 3-11). The SRP flux was 1295.1 $\mu gm^{-2}d^{-1}$ during baseflow but -57.1 $\mu gm^{-2}d^{-1}$ during flood conditions. This demonstrates that a SRP loss occurs when groundwater flows through the riparian zone, although this loss was considerably low and may be negligible compared to the SRP flux under baseflow condition. The lateral TP mass flux through the riparian zone from transects T1 to T4 was more pronounced during flood than baseflow conditions. This pattern reversed in SRP flux (Tables 3-10 and 3-11).

3.7.2 Phosphorus flux at the hyporheic zone

Phosphorus mass flux caused by the vertical hydraulic gradient (VHG) at the riparian-stream interface (hyporheic zone) for both transects over the study period are presented in Table 3-12 and 3-13.

Table 3-12 Mean load of phosphorus downward in hyporheic zone along 1 m long stream reach at two transects during baseflow

Piezometer ID	K_{sat} (cms^{-1})	Mean VHG	Stream width (cm)	Recharge (Ld^{-1})	Groundwater phosphorus		Mean load	
					TP (μgL^{-1})	SRP (μgL^{-1})	TP ($\mu gm^{-2}d^{-1}$)	SRP ($\mu gm^{-2}d^{-1}$)
T1-50-P50	7.30E-03	0.00	180	0.00	843.5	96.9	0.0	0.0
T1-50-P100	1.41E-05	-0.30	176	3.25	238.3	35.3	774.7	114.8
T4-50-P50	1.27E-03	-0.01	322	13.41	480.3	92.9	6440.1	1245.3
T4-50-P100	2.99E-05	-0.58	296	22.28	541.0	89.8	12056.4	2000.4

Table 3-13 Mean load of phosphorus downward in hyporheic zone along 1 m long stream reach at two transects during flood

Piezometer ID	K_{sat} (cms^{-1})	Mean VHG	Stream width (cm)	Recharge (Ld^{-1})	Groundwater phosphorus		Mean load	
					TP (μgL^{-1})	SRP (μgL^{-1})	TP ($\mu gm^{-2}d^{-1}$)	SRP ($\mu gm^{-2}d^{-1}$)
T1-50-P50	7.30E-03	-0.03	180	164.47	733.7	17.9	120671.7	2945.4
T1-50-P100	1.41E-05	-1.15	176	12.38	105.4	14.9	1305.0	184.9
T4-50-P50	1.27E-03	-0.06	322	102.80	2266.5	178.8	233005.6	18381.2
T4-50-P100	2.99E-05	-0.89	296	33.90	nd	nd	nd	nd

* ‘nd‘ means no data available.

3.7.2.1 Vertical hydraulic gradient

The hydraulic head and associated vertical hydraulic gradient (VHG) of the in-stream piezometers showed that these locations are groundwater recharge zones, i.e. the area where groundwater flows were either lateral with the mean VHG of zero or downward from the stream to the hyporheic zone with negative mean VHG. VHG increased at all sites during flood compared to baseflow. The VHG was stronger at 100 cm than 50 cm at all sites. At transect T1, the mean VHG in the hyporheic zone was zero and -0.30 at 50 cm and 100 cm during baseflow, respectively but during flood, it was -0.03 and -1.15, respectively. At transect T4, the mean VHG in the hyporheic zone was -0.01 and -0.58 at 50 cm and 100 cm during baseflow, respectively. During flood, it was -0.06 and -0.89, respectively. In general, VHG was greater at transect T4 than T1 (Tables 3-12 and 3-13).

3.7.2.2 Mean phosphorus load

Phosphorus mass flux in the hyporheic zone was estimated for the two transects under different flow regimes (Tables 3-12 and 3-13). At transect T1, there was no vertical phosphorus transfer through the hyporheic zones at 50 cm during baseflow. The mean TP load of 774.7 $\mu gm^{-2}d^{-1}$ and mean SRP load of 114.8 $\mu gm^{-2}d^{-1}$ occurred in the hyporheic zone at 100 cm at transect T1. During flood, the mean TP and SRP loads toward the hyporheic zone were two orders of magnitude and one order of magnitude larger at 50 cm than at 100 cm, respectively. At transect T4, both mean TP and SRP loads toward hyporheic zone were larger at 100 cm than 50 cm during baseflow. Overall, the downward mass flux phosphorus in hyporheic zone was more pronounced at transect T4 than T1 during both baseflow and flood (Tables 3-12 and 3-13).

Chapter 4 DISCUSSION

4.1 Introduction

Several factors have been shown to influence the subsurface transport of phosphorus (TP and SRP) in stream riparian zones (Osborne and Kovacic, 1993; Heathwaite and Dils, 2000; Banaszuk *et al*., 2005). These include: 1) distance from the field (Banaszuk *et al*., 2005); 2) depth (Heathwaite and Dils, 2000); 3) flow regimes (Takatert *et al*., 1999) and 4) vegetation type (Osborne and Kovacic, 1993). In this chapter, the effects of above factors on subsurface transport of phosphorus (P) in the study area will be discussed in the context of the literature.

4.2 Vertical and Horizontal Variation in P Concentrations in Shallow Groundwater

Previous studies have shown that P concentrations in shallow groundwater varied both vertically (Heathwaite and Dils, 2000) and horizontally (Banaszuk *et al*., 2005) according to factors such as depth (Heathwaite and Dils, 2000) and distance from the field (Banaszuk *et al*., 2005). Banaszuk *et al*. (2005) reported that with increasing distance from cropland there was a slight increase in groundwater SRP in the riparian zone of Kurowka River in Poland. The SRP concentrations ranged from 1.0 ± 0.7 mgL^{-1} (mean ± standard deviation) at the field edge to 1.6 ± 0.6 mgL^{-1} at the riparian edge. Heathwaite and Dils (2000) found that TP concentrations in shallow groundwater decreased with depth and ranged from 500 ± 184 μgL^{-1} (mean ± standard error) at 50 cm to 210 ± 32 μgL^{-1} at 150 cm in Pistern Hill catchment in England.

In the Spencer Creek riparian zone, there were significant interactions between the effects of distance and depth on TP (F=3.922, df=7, p<0.001, Appendix 1) and SRP (F=2.584, df=7, p=0.013, Appendix 2) which suggested that distance and depth did not affect groundwater P concentrations independently. These interactions could possibly be the results of complexity in soil physical and geochemical properties observed by field studies that affect soil P sorption capacity (Bridgham *et al*., 2001) and highly complex groundwater flow regimes in the riparian zone (Figures 3-8, 3-9 and 3-10) that may cause irregular P transfer in groundwater and consequently influence spatial distribution of P concentration both vertically and horizontally.

For both transects, mean TP concentration tended to increase with increasing distance from the field to the hyporheic zone during both baseflow and flood, but this trend became weak by depth (Appendix 3). Statistically significant differences in TP concentration between distances with increasing distance from the field edge to the hyporheic zone existed between every two selected sites (6 m, 19 m, 33 m, 48 m and 50 m) at 50 cm depth but not for those closer to each other (Appendix 3). Significant differences existed between 6 m and 48 m, 19 m and 33 m, 19 m and 48 m at 100 cm depth, but this was only found between 6 m and 33 m at 150 cm depth (Appendix 3). The mean SRP concentration of two transects tended to increase with increasing distance from the field to the hyporheic zone during both baseflow and flood. Similarly, this trend became weak by depth (Appendix 4).

Statistically significant differences in SRP concentration between distances with increasing trend from the field edge to the hyporheic zone existed between every two selected sites at 50 cm depth but not among 6 m, 19 m and 33 m (Appendix 4). These significant differences in SRP existed between 6 m and 48 m, 6 m and 50 m, 19 m and 48 m, 33 m and 48 m at 100 cm depth, but no differences between distances were found at 150 cm depth (Appendix 4). The results of SRP of the present study was similar to the findings of Banaszuk *et al*. (2005) who studied groundwater SRP transport in a riparian zone of Kurowka river in Poland. They also reported an increase in groundwater SRP with increasing distance from the field edge of cropland but did not test the statistical significance. The pattern of increasing SRP concentration in groundwater with increasing distance from the field at two transects can be partially explained by varying DO levels in groundwater which were generally higher near the field edge than near the riparian edge (Figure 3-26), particularly in the riparian zone during flood (F=6.130, df=3, p=0.001, Appendix 5). Aerobic conditions in this environment can promote P sorption on soil particles (Gillian *et al*., 1999) and enhance phosphate mineralization (Reddy *et al*., 1999), while anaerobic conditions enhance P desorption and release from metal hydroxide particle surfaces to solution (Surridge *et al*., 2006). There was no significant difference between the distances of 19 m and 33 m at 50 and 150 cm depths for TP and at 50 and 100 cm depths for SRP, and between 33 m and 48 m at 50 and 100 cm depths for TP concentrations (Appendix 3 and 4), which might be partially attributed to the similar patterns of DO concentrations (no significant difference between distances of 33 m and 48 m) within the riparian zone (Appendix 5). Groundwater SRP concentrations at two transects were inversely correlated with DO concentrations during flood (Spearman rank correlation, r=-0.216, p=0.021, 2-tailed at 0.05 significant level). The increased correlation (r=-0.275, p=0.006, 2-tailed at 0.01 significant level) was found when analyzed by excluding the piezometers in the hyporheic zone (50 m), indicated that groundwater SRP concentration was more influenced by the redox conditions in the riparian zone than the hyporheic zone which was typically anaerobic for most sampling period. Carlyle and Hill (2001) reported a similar negative relationship between groundwater SRP concentrations (log transformed) and DO concentrations in the Boyne River riparian zone in southern Ontario. Other factors than DO may also influence the spatial variation of P concentrations in riparian zones. pH is one of the factors that regulate P adsorption/desorption processes in aquatic systems (Stone, 1993). Lower pH may decrease SRP in groundwater by promoting phosphate adsorption to surfaces of iron, aluminum and manganese minerals in soils (Gardiner and Miller, 2004). In the Spencer Creek, groundwater pH generally increased with increasing distance from the field (Figure 3-22), reflecting a more favorable environment for P adsorption in the area near the field than near the stream. This observation was consistent with the overall increasing pattern of groundwater SRP from the field edge to the hyporheic zone. Many soil microbes such as bacteria, fungal, yeast and actinomycete species are capable of transforming soil P to soluble inorganic P (Kucey *et al*., 1989). Roden and Edmonds (1997) found that bacterial sulfate reduction can cause release of SRP when iron sulfide (FeS) is formed in the sediments. Microbial processes were not included in the present study. However, sulfur (H_2S) was smelled when groundwater samples were pumped from the piezometers at several sites (piezometers of T1-48-150 and T4-19-100 on JD 157; T4-19-100 and T4-19-150 on JD 171; T1-33-100, T1-33-150, T4-19-100, T4-33-100, T4-48-50, T4-48-150 on JD 187; T4-19-50, T4-19-100 and T4-19-150 on JD 199). This qualitative observation provided indirective evidence of anaerobic conditions that may facilitate release of SRP. Consequently, elevated P concentrations with distance from the field may also be simultaneously associated with bacteria distribution in riparian soils.

For both transects, the mean TP concentration at the distances of 19 m, 48 m and 50 m decreased significantly by depth during both baseflow and flood, although there were no significant differences between 100 cm and 150 cm at 19 m, and between 50 cm and 100 cm at 48 m. They were consistently higher at 50 cm than 150 cm in distances of 6 m and 33 m, but no significant differences between depths were found in these sites (Appendix 6). Overall, TP concentration of two transects were significantly different by depth at distances of 19, 48 and 50 m with an overall trend of $TP_{50cm}>TP_{100cm}>TP_{150cm}$ during both baseflow and flood (Appendix 6). The mean SRP concentration of two transects was significantly higher at 50 cm and 100 cm than 150 cm with no significant difference between 50 cm and 100 cm in the distance of 48 m. No significant differences in SRP concentration between depths were found at other distances (Appendix 7). Overall, SRP concentration of two transects did not show a pronounced decreasing trend by depth during both baseflow and flood except the site at the riparian edge (Appendix 7). The pattern of decreasing TP concentrations with depth may be attributed to P loss for soil adsorption from upper to lower soil horizons. Mineral soils in the deeper layer tended to have stronger adsorption capacity than organic soils near ground surface because of Al and Fe oxides and hydroxides predominantly found in mineral soils (Qualls and Richardson, 1995; Gardiner and Miller, 2004). This finding was consistent with the spatial patterns of groundwater TP concentrations by depth reported by Heathwaite and Dils (2000). They showed similar mean values but smaller ranges at depths of 50 cm, 100 cm and 150 cm in Pistern Hill catchment in UK compared to baseflow data of the present study (Appendix 8). However, no clear pattern by depth was found in their research for SRP concentrations as reflected at the riparian edge (48 m) in the present study. Macrae *et al*. (2005) reported that total dissolved phosphorus concentrations (TDP) in groundwater showed no pattern with depth in a forested wetland in north-central Alberta. The contrast in spatial patterns of SRP between the riparian edge site of the present study and previous research is probably due to highly site-specific soil properties (e.g., soil organic matter, amorphous Fe and Al, equilibrium phosphate concentration, soil moisture, pH and bulk density) that directly or indirectly control soil adsorption capacity (Reddy *et al*., 1995;

Darke and Walbridge, 2000; Bridgham *et al*., 2001). This range of factors can influence groundwater P concentrations particularly the soluble form (SRP). Higher vertical heterogeneity in organic content of soils at the riparian edge (48 m) (e.g., site T4-48 had an organic content of 28.74%, 12.54% and 1.30% at the depths of 50, 100 and 150 cm, respectively) compared to other sites (e.g., site T4-19 had an organic content of 0.82%, 0.54% and 0.65% at the depths of 50, 100 and 150 cm, respectively) may account for the significant decrease in SRP by depth at some locations because organic matter can block P sorption sites on soil metal oxides and therefore reduce P loss in groundwater (Holtan *et al*., 1988). Observations from this study was supported by Macrae et al. (2005) who found that higher SRP concentrations occurred in groundwater in organic horizons near the surface.

4.3 The Effect of Flood on TP and SRP Concentrations in Shallow Groundwater

Several investigators have examined the effects of factors that regulate P transport in groundwater such as distance from the field (Banaszuk *et al*., 2005), depth (Heathwaite and Dils, 2000), redox chemistry (Carlyle and Hill, 2001) and vegetation type (Osborne and Kovacic, 1993). Takatert *et al*. (1999) investigated SRP concentrations in groundwater at 2 m depth in the riparian area of the Rhine Plain in France. The SRP concentration varied between 10 μgL^{-1} to 180 μgL^{-1} during an annual hydrological cycle but there was no observable change in SRP concentrations during flood period.

In the Spencer Creek riparian zone, the mean TP concentrations of two transects were significantly higher during flood than baseflow ($F=9.319$, $df=1$, $p=0.002$, Appendix 1) although this was more pronounced at transect T1 than T4, while the mean SRP concentrations of two transects were significantly lower during flood than baseflow ($F=12.553$, $df=1$, $p<0.001$, Appendix 2) likely due to higher DO during flood ($F= 4.288$, $df=1$, $p=0.040$, Appendix 9). During drawdown (JD289 to JD291), a total of 918.9 kg TP was exported from Valens Reservior (Allin, unpublished data, 2006) and both transects (T1 and T4) were completely flooded (Three piezometers including T4-19-100, T4-19-150 and T4-48-150 were submerged from the tip by flood water on JD290. Data on JD290 from these piezometers were removed before statistical analysis) (Figures 3-4e and 3-4f). Flooding increased TP due to mobilization and flushing of particulate P but decreased SRP because of the influx of oxygen-rich flood water. In riparian zones, high water tables can cause leaching of decayed vegetation which increases TP concentrations in particulate form in groundwater (Takatert *et al*., 1999). Flood conditions provided a better connectivity between the riparian zone and Spencer Creek which facilitated mixing of groundwater and surface water (Kaufman *et al*., 2005). During the peak flood day (JD 290), similar extremely high TP concentrations were found in flood water within the riparian zone at transect T4 (14.6-41.6 mgL^{-1}) and in stream water near the riparian edge of T4 (12.0 mgL^{-1}) which were three orders of magnitude higher than TP concentration in stream water at T1. This fact suggested that TP was mobilized by flood water in the riparian zone and the much higher TP in stream at T4 than T1 was likely caused by mixing with the TP-loaded plume that returned from the riparian zone to the stream channel. A similar flushing effect on sulphate dynamics between groundwater and stream water at a 400 m downstream transect in Spencer Creek was observed by Warren et al. (2001). Higher TP in stream was also associated with P-rich groundwater export from the riparian zone to the stream during flood. This groundwater export process was suggested by previous study in Spencer Creek (Munro *et al*., 2000).

Using the data of Heathwaite and Dils (2000), the ratio of SRP to TP in groundwater was calculated and showed a decrease by depth in Pistern Hill catchment in England but no other analysis is available in the literature. In the Spencer Creek riparian zone, the ratio of SRP to TP in groundwater at two transects were significantly lower during flood than baseflow ($F=13.994$, $df=1$, $p<0.001$, Appendix 10). They were significantly different by depth ($F=3.048$, $df=2$, $p=0.049$, Appendix 10) but not by distance from the field ($F=2.010$, $df=3$, $p=0.112$, Appendix 10). The ratio of SRP to TP estimated from the literature (Heathwaite and Dils, 2000) at depths of 50 cm, 100 cm and 150 cm were 0.46 to 9.54 fold higher than the values obtained at the same depths for the present study (Appendix 8), reflecting that the ratio of SRP to TP is a highly site-specific factor. An influx of stream water during drawdown with significantly lower SRP concentration ($F=12.553$, $df=1$, $p<0.001$, Appendix 2) caused a dilution effect to occur in the riparian zone. SRP fractions in groundwater at 50 cm were significantly lower than at 100 cm and 150 cm (Appendix 10), due to the input of elevated particulate P either inputted by drawdown of Valens Reservior (Allin, unpublished data, 2006) or from leaching of decayed vegetation detritus tissues in ground surface (Walbridge *et al*., 1991). Decrease in SRP fraction in groundwater during flood may be explained by significantly increased DO concentrations in groundwater during flood compared to baseflow ($F=4.288$, $df=1$, $p=0.040$, Appendix 9) which likely caused SRP concentrations to decrease because of P sorption on metal oxide surface of soil particles under aerobic condition (Qualls and Richardson, 1995).

4.4 The Effect of Vegetation on P Concentrations in Shallow Groundwater

Many studies have reported P assimilation by plants, however, the effects of vegetation uptake on P concentrations in groundwater in riparian zones were highly variable even for the same plant species (Osborne and Kovacic, 1993; Mander *et al*., 1997; Takatert *et al*., 1999).

In the Spencer Creek riparian zone, TP concentrations at three depths were all significantly higher at transect T4 than T1 (Independent-samples T test, 50 cm: $p<0.001$; 100 cm: $p<0.001$; 150 cm: $p<0.001$, 2-tailed). The SRP concentrations at 50 cm were significantly higher at transect T4 than T1 (Independent-samples T test, $p<0.003$, 2-tailed). At transect T4, they were higher but not significantly higher than T1 at both 100 cm ($p=0.627$, 2-tailed) and 150 cm ($p=0.335$, 2-tailed) depths. The significant differences in TP and SRP concentrations at 50 cm between transect T1 and T4 could be attributed to the different types of vegetation at the two transects. However, other unfound factors such as soil type and their influence on mobilization of particulate P at transect T4 during flood may work as well. Herbaceous species (grass and short shrubs) that predominate at transect T1 have a higher uptake efficiency for both dissolved P and total P assimilation in shallow groundwater compared to the predominant forest cover at T4 (Osborne and Kovacic, 1993). Other studies support this finding that grasses have higher P assimilation ability than young alder in the riparian zone of Viiratsi Ditch, Estonia (Kuusemets *et al*., 2001). Both trees (e.g., grey alder) and grass (e.g., wet meadow) can utilize SRP in groundwater (Mander *et al*., 1997), but incorporated P in plant biomass in the growth season may be released after plant die off in the dormant season (Uusi-Kamppa *et al*., 1997). The data suggested that the vegetation cover had much higher effects on SRP concentration than TP concentration in shallow groundwater in the riparian zone of Spencer Creek. This is reasonable given that SRP is bio-available. The present study was mainly conducted during the growing season when the plant uptake effect was relatively high (Reddy *et al*., 1999). Considering that vegetation is only a periodic P sink, factors other than vegetation may contribute to significantly lower TP concentrations at transect T1 than T4. A consistent hydraulic gradient between two transects (Figure 3-36) allowed a possible groundwater flow from transect T1 toward T4 during the study period, which facilitated particulate P movement through the groundwater pathway (Heathwaite and Dils, 2000). The P retention function of riparian vegetation likely has a relatively small impact on groundwater P concentrations compared to groundwater P transfer controlled by biogeochemical and hydrological process (Takatert *et al*., 1999). This is because biomass P will return to soil pore water and groundwater due to leaching of decayed plants during the dormant season (Walbridge *et al*., 1991).

4.5 Phosphorus Mass Flux

The hyporheic zone is an important interface between territorial and aquatic systems which governs a range of physical and biogeochemical processes. Hill (1997) suggested that the hyporheic zone is a SRP source to stream water, but Stainton (2000) demonstrated that the hydrogeological setting largely governs TP and SRP loading over the hyporheic zones in Laurel Creek watershed in southern Ontario.

In the Spencer Creek, estimates of saturated hydraulic conductivity (K_{sat}) in the hyporheic zone were similar in magnitude at the same depth at the two transects but highly variable by depths within each transect (Tables 3-12 and 3-13). This suggested there is a larger vertical substrate heterogeneity than in the horizontal direction. The saturated hydraulic conductivity in the hyporheic zone was higher at 50 cm than 100 cm by two and one order of magnitude at transect T1 and T4, respectively. This suggested that there may be higher conductivity substrates in the upper parts of the hyporheic zone in Spencer Creek, which will facilitate more groundwater flux. The higher K_{sat} at 50 cm in the hyporheic zone of two transects (1.27×10^{-5} ms^{-1} and 7.30×10^{-5} ms^{-1}) were one order of magnitude higher than the K_{sat} (2.43×10^{-6} ms^{-1}) at the same 50 cm depth in another riparian site (400 m far from the present study site) at downstream of Spencer

Creek (Warren *et al*., 2001), but were similar in magnitude to those at 108 cm and 113 cm depths (2.34×10^{-5} ms^{-1} and 2.51×10^{-5} ms^{-1}) reported by Stainton (2000) for the buffered riparian zone of Clair Creek in southern Ontario. A saturated hydraulic conductivity in the range of 10^{-5} ms^{-1} is characteristic of a silt-sand substrate (Freeze and Cherry, 1979), which is consistent with the textural composition at this depth in Spencer Creek (Bourbonniere and Macrae, unpublished data, 2007). The K_{sat} at 100 cm (1.41×10^{-7} ms^{-1} to 2.99×10^{-7} ms^{-1}) in Spencer Creek is similar to values estimated for 89 cm depth (5.76×10^{-7} ms^{-1}) at the urban un-buffered site for Clair Creek in southern Ontario (Stainton, 2000). In the present study, K_{sat} at 50 cm ($\times 10^{-5}$ ms^{-1}) and 100 cm ($\times 10^{-7}$ ms^{-1}) was two and four orders of magnitude higher than similar depths in Beaver Creek (55 cm, K_{sat} $=3.41 \times 10^{-9}$ ms^{-1}) and Laurel Creek (94 cm, K_{sat} $=1.18 \times 10^{-9}$ ms^{-1}), respectively (Stainton, 2000).

The present study indicated that the differences in K_{sat} or VHG's had a larger influence on P loading than P concentration at the riparian-stream interface. The difference in SRP was less than 3 μgL^{-1} between 50 cm and 100 cm in the hyporheic zone at transect T1 during flood. However, SRP loading was one order of magnitude higher at 50 cm than 100 cm due to the same magnitude difference in groundwater recharge (Table 3-13). Similarly, much higher TP and SRP loading at all depths at both transects during flood relative to baseflow (the difference were as larger as two orders of magnitudes) were primarily caused by large VHG's during flood than baseflow, given the K_{sat} was assumed to be constant during different flow conditions in the present study and most P concentrations during flood had the same magnitude as those during baseflow. Although the hyporheic zone in the Spencer Creek tended to recharge P according to its hydrologic gradients, the overall hydrology and P transfer are likely between transect T1 and T4. Stream water seemed to recharge the groundwater in the riparian zone and then to flush the riparian zone at transect T4 when the water returned to the stream during the flood, however, groundwater simultaneously discharged from the riparian zone to the stream channel. This indicated that the hydrology is the main factor driving P transfer between the wetland and Spencer Creek, which was supported by other studies on climate, sulphate dynamics or hydrogeomorphic regimes in Beverly Swamp (Munro *et al*., 2000; Warren *et al*., 2001; Kaufman *et al*., 2006)

4.6 Implications for Management

Riparian buffer zones have several important ecological and water quality enhanced functions for both surface water and groundwater systems (Uusi-Kamppa *et al*., 1997; Boulton *et al*., 1998). Since the vegetation covers in riparian areas have the ability to reduce nutrients from surface runoff (Daniels and Gilliam, 1996), riparian buffer zones have been recognized as an effective measure to control P transported by overland flow (Kuusemets *et al*., 2001). This process was not included in the present study, but qualitative evidence from field observations showed that sediment transported by overland flow during storm events did not move far away from the field edge.

Although riparian buffer zones have been adopted as management tools for mitigating the adverse impacts of agriculture practices on the aquatic system (Osborne and Kovacic, 1993), questions still remain regarding the efficiency of riparian buffer zones to reduce shallow groundwater transport of agricultural nutrients to adjacent streams at the landscape scale (Norris, 1993). Stainton (2000) found that a vegetated riparian buffer zone between the agricultural field and the stream channel did not effectively control P concentrations in groundwater reaching the stream in Laurel Creek watershed in southern Ontario. Considering that the efficiency of buffer zones depends on the mechanisms by which P is transported from agricultural fields toward streams (Muscutt *et al*., 1993), it is reasonable that the riparian zone of Spencer Creek did not reduce P input in groundwater from the agricultural field because there was no consistent groundwater flow along the transverse section of the riparian zone between the field and stream (Figures 3-8 and 3-9).

Shantz *et al*. (2004) demonstrated that reservoir drawdown has a significant impact on the downstream transport of P. Other investigators have also shown that flooding is an important mechanism for the transfer of fluvial derived P to riparian zones (Thoms *et al*., 2000; Steiger and Gurnell, 2002). During flood, the riparian wetland in the present study received significant amounts of P from Spencer Creek during the drawdown of upstream Valens Reservoir. The present study contrasts the findings of Walling *et al*. (2000) that riparian zones buffer longitudinal or downstream transfer of sediment bound P. However, in Spencer Creek the riparian zone is a P source to the adjacent stream. From a management perspective, the present study revealed that whether the riparian zones functions as groundwater P buffers between agricultural fields and adjacent streams is highly dependent on the local hydrological regimes, although the width (Heathwaite, 1998), substrate composition (Carlyle and Hill, 2001) and vegetation type (Osborne and Kovacic, 1993) of the riparian zone have been shown to be important factors influencing groundwater P retention of riparian wetlands. For riparian wetlands periodically flooded by the drawdown of upstream reservoirs, hydrological variability is a dominant factor regulating groundwater P transfer within upland-riparian-stream continuum, while other considerations such as vegetation type and substrate characteristics may also be important but require further studies (Takatert *et al*., 1999; Withers and Lord, 2002). The degree to which riparian zones can reduce P transport in groundwater from agricultural fields to adjacent streams will depend highly on site-specific hydrological regimes.

Chapter 5 CONCLUSION

5.1 Conclusions

The present study examined P transfer in groundwater under varying hydrological regimes (low flow, high flow and flooding) and vegetation cover in a riparian zone in Spencer Creek in southern Ontario. The data provide further understanding of the role of riparian wetlands on groundwater P regulation in agricultural landscapes. The conclusions of this study are presented as follows:

1) The hydrological regimes in the study site were relatively complex compared to previous studies in the riparian zones. During baseflow, there was no consistent groundwater flow from the field to the stream along two transects or vertical exchanges between depths within each transect. Recharge seemed to predominate at transect T1 not at T4 during flood, but there was no substantial transverse flow across the riparian zone. Instead, a general longitudinal groundwater flow existed from transect T1 to T4 during both baseflow and flood conditions. High water table above surface during flood provided a better connectivity between the riparian zone and Spencer Creek, facilitating groundwater interaction with surface water.

2) Factors such as distance from the agricultural field and depth had significant effects on both TP and SRP concentrations in groundwater. The distance and depth did not affect TP and SRP concentration independently, possibly due to heterogeneity of soil physiochemical characteristics that affect soil P sorption capacity and highly complex

groundwater flow regimes in the riparian zone. Both TP and SRP concentrations generally increased with increasing distance from the field to the hyporhiec zone during both baseflow and flood. However, this increasing pattern with distance was poorly connected to depth. TP concentrations decreased with depth, but this trend with depth for SRP was only found in the riparian edge. Redox conditions combined with soil characteristics related to P adsorption may have accounted for most P concentration patterns in groundwater in the riparian zone. However, soil microbial processes may also affect P concentration variation through assimilation or releasing bio-available P in groundwater.

3) The flood caused by the upstream reservoir drawdown significantly increased TP concentrations and significantly decreased SRP concentrations in groundwater in the riparian zone. The lower ratio of SRP to TP during flood indicated that particulate P imported by overbank flow contributed to the significant increase in TP concentration during flood relative to baseflow, while the higher water table may have simultaneously increased TP concentration by leaching decayed vegetation detritus. An influx of oxygen-rich water introduced by flood flow into the riparian zone may lower SRP concentration in groundwater by promoting soil P sorption and dilution effect.

4) TP and SRP concentrations were significantly lower at 50 cm depth at transect T1 than T4. Herbaceous species (grass and short shrub) that predominated T1 have been shown by previous studies to have higher uptake capacity for both SRP and TP than the

forest cover at T4. Considering that vegetation is only a periodic P sink, possible longitudinal groundwater flow that facilitated P movement from transect T1 to T4 may be partially responsible for the differences in P concentration between transect T1 and T4. This difference may be due to soil type and also mobilization of particulate P at transect T4 during flood.

5) The variability of saturated hydraulic conductivity (K_{sat}) within and between transects reflected a larger intra-transect substrate heterogeneity which implies that the upper parts of the hyporheic zone with higher K_{sat} in Spencer Creek may facilitate more groundwater flux. The groundwater recharge zones characterized by zero or negative vertical hydraulic gradient (VHG's) were identified in the hyporheic zone at two transects, while the lateral hydraulic gradient (LHG) indicated generally horizontal groundwater flows. Both TP and SRP mass flux from transect T1 to T4 through the riparian zone and from stream channel toward the hyporheic zone were highly variable during the study period. The flood may substantially increase TP and SRP loading by increasing VHG's at the hyporheic zone. At the riparian zone, flood enlarged TP flux but reduced SRP flux from T1 to T4.

6) The present study has revealed some implications for utilizing riparian wetlands to reduce groundwater P transfer from agricultural fields to adjacent streams. Given that P concentration was always higher at transect T4 than T1 under a general longitudinal groundwater flow approximately parallel with the stream channel instead of

a transverse flow from the agricultural field to the stream, the riparian wetland of Spencer Creek was more likely a P source to downstream area or Spencer Creek rather than a sink for agricultural-derived P in groundwater from the field toward the stream. The function of riparian zones as groundwater P buffers between agricultural fields and adjacent streams is highly dependent on the local hydrological regimes. The effectiveness of riparian zones to reduce P transport in groundwater from agricultural fields to adjacent streams will be highly dependent on site-specific hydrological regimes.

5.2 Recommendations for Future Research

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

1) An estimate of plant uptake and release including a detailed investigation of riparian vegetation species should be conducted to examine P accumulation in the above ground biomass and P retention efficiency of different riparian plant communities. This may help to evaluate variability of SRP distribution in groundwater in riparian zones.

2) Additional hydrogeological information such as a complete study of soil composition and stratigraphy is necessary to better understand and interpret flow patterns in the riparian zone.

3) Considering the significant influence of upstream reservoirs and drawdown on P transfer in the riparian system, further understanding of the transport and sedimentation mechanisms of flood-associated P and their relationship with groundwater P in riparian zones is required. Flood water hydrology and chemistry should be measured

simultaneously with groundwater at each site during drawdown events for better examining the processes of P mobilization and transfer in the riparian zone. Also, there is a need to evaluate the geochemical composition and P speciation of suspended particle matter in the river during flood to determine the loading of particle P from the stream to the riparian zone.

4) Future research should attempt to develop a predictive model that can be applied to simulate the groundwater P transport process between agricultural field and adjacent stream. This may facilitate further studies regulating groundwater P transfer within agricultural field-riparian zone-stream continuum, providing information to support more effective stream water protection.

Appendix 1

Analysis of Variance (ANOVA) of TP concentrations according to distance, depth and flow condition (dependent variable: natural logarithm of TP, significant difference at p<0.05)

Tests of Between-Subjects Effects

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	249.571(a)	28	8.913	8.875	.000
Intercept	9705.056	1	9705.056	9663.154	.000
Flow	9.360	1	9.360	9.319	.002
Transect	96.487	1	96.487	96.070	.000
Distance	27.667	4	6.917	6.887	.000
Depth	38.199	2	19.100	19.017	.000
Flow * Depth	.778	2	.389	.387	.679
Flow * Distance	6.082	4	1.521	1.514	.197
Distance * Depth	27.576	7	3.939	3.922	.000
Flow * Distance * Depth	6.373	7	.910	.906	.501
Error	373.613	372	1.004		
Total	12464.470	401			
Corrected Total	623.184	400			

a R Squared = .400 (Adjusted R Squared = .355)

Estimated Marginal Means

1. Flow regime

Flow regime	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
Baseflow	5.364(a)	.064	5.239	5.488
Flood	5.708(a)	.088	5.534	5.882

a Based on modified population marginal mean.

2. Transect

Transect	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
1	5.043(a)	.073	4.900	5.186
4	6.029(a)	.075	5.881	6.177

a Based on modified population marginal mean.

Appendix 2

Analysis of Variance (ANOVA) of SRP concentrations according to distance, depth and flow condition (dependent variable: natural logarithm of SRP, significant difference at p<0.05)

Tests of Between-Subjects Effects

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	120.732(a)	28	4.312	4.988	.000
Intercept	2772.937	1	2772.937	3207.833	.000
Flow	10.852	1	10.852	12.553	.000
Transect	2.898	1	2.898	3.353	.068
Distance	55.409	4	13.852	16.025	.000
Depth	10.917	2	5.459	6.315	.002
Flow * Depth	1.198	2	.599	.693	.501
Flow * Distance	2.595	4	.649	.751	.558
Distance * Depth	15.636	7	2.234	2.584	.013
Flow * Distance * Depth	6.056	7	.865	1.001	.430
Error	327.618	379	.864		
Total	4175.992	408			
Corrected Total	448.349	407			

a R Squared = .269 (Adjusted R Squared = .215)

Estimated Marginal Means

1. Flow regime

Flow regime	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
Baseflow	3.181(a)	.058	3.067	3.295
Flood	2.815(a)	.084	2.650	2.980

a Based on modified population marginal mean.

2. Transect

Transect	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
1	2.913(a)	.067	2.781	3.046
4	3.083(a)	.070	2.944	3.221

a Based on modified population marginal mean.

Appendix 3

Analysis of Variance (ANOVA) of TP concentrations for all distances at each depth (dependent variable: natural logarithm of TP, significant difference at p<0.05)

Tests of Between-Subjects Effects (50 cm depth)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	46.572(a)	5	9.314	8.458	.000
Intercept	4613.503	1	4613.503	4189.304	.000
Distance	31.885	4	7.971	7.238	.000
Transect	13.892	1	13.892	12.614	.001
Error	143.163	130	1.101		
Total	4831.150	136			
Corrected Total	189.735	135			

a R Squared = .245 (Adjusted R Squared = .216)

Multiple Comparisons (Post Hoc Tests: LSD, distance from the field)

(I) Distance from the field (m)	(J) Distance from the field (m)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
6	19	-.5024	.27804	.073	-1.0524	.0477
	33	-.7756(*)	.27575	.006	-1.3211	-.2301
	48	-1.1524(*)	.27804	.000	-1.7024	-.6023
	50	-1.4728(*)	.30724	.000	-2.0806	-.8650
19	6	.5024	.27804	.073	-.0477	1.0524
	33	-.2732	.27328	.319	-.8139	.2674
	48	-.6500(*)	.27559	.020	-1.1952	-.1048
	50	-.9704(*)	.30502	.002	-1.5739	-.3670
33	6	.7756(*)	.27575	.006	.2301	1.3211
	19	.2732	.27328	.319	-.2674	.8139
	48	-.3768	.27328	.170	-.9174	.1639
	50	-.6972(*)	.30294	.023	-1.2965	-.0979
48	6	1.1524(*)	.27804	.000	.6023	1.7024
	19	.6500(*)	.27559	.020	.1048	1.1952
	33	.3768	.27328	.170	-.1639	.9174
	50	-.3204	.30502	.295	-.9239	.2830
50	6	1.4728(*)	.30724	.000	.8650	2.0806
	19	.9704(*)	.30502	.002	.3670	1.5739
	33	.6972(*)	.30294	.023	.0979	1.2965
	48	.3204	.30502	.295	-.2830	.9239

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

Tests of Between-Subjects Effects (100 cm depth)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	63.534(a)	5	12.707	13.318	.000
Intercept	3862.269	1	3862.269	4048.147	.000
Distance	25.730	4	6.433	6.742	.000
Transect	36.719	1	36.719	38.486	.000
Error	127.847	134	.954		
Total	4416.234	140			
Corrected Total	191.382	139			

a R Squared = .332 (Adjusted R Squared = .307)

Multiple Comparisons (Post Hoc Tests: LSD, distance from the field)

(I) Distance from the field (m)	(J) Distance from the field (m)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
6	19	.4433	.25016	.079	-.0515	.9381
	33	-.2943	.24823	.238	-.7853	.1966
	48	-.8108(*)	.24823	.001	-1.3018	-.3199
	50	.0479	.30888	.877	-.5631	.6588
19	6	-.4433	.25016	.079	-.9381	.0515
	33	-.7376(*)	.24615	.003	-1.2245	-.2508
	48	-1.2541(*)	.24615	.000	-1.7410	-.7673
	50	-.3954	.30722	.200	-1.0031	.2122
33	6	.2943	.24823	.238	-.1966	.7853
	19	.7376(*)	.24615	.003	.2508	1.2245
	48	-.5165(*)	.24419	.036	-.9995	-.0335
	50	.3422	.30565	.265	-.2623	.9467
48	6	.8108(*)	.24823	.001	.3199	1.3018
	19	1.2541(*)	.24615	.000	.7673	1.7410
	33	.5165(*)	.24419	.036	.0335	.9995
	50	.8587(*)	.30565	.006	.2542	1.4632
50	6	-.0479	.30888	.877	-.6588	.5631
	19	.3954	.30722	.200	-.2122	1.0031
	33	-.3422	.30565	.265	-.9467	.2623
	48	-.8587(*)	.30565	.006	-1.4632	-.2542

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

Tests of Between-Subjects Effects (150 cm depth)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	66.109(a)	4	16.527	16.484	.000
Intercept	3033.277	1	3033.277	3025.330	.000
Distance	10.285	3	3.428	3.419	.020
Transect	55.635	1	55.635	55.489	.000
Error	120.315	120	1.003		
Total	3217.085	125			
Corrected Total	186.424	124			

a R Squared = .355 (Adjusted R Squared = .333)

Multiple Comparisons (Post Hoc Tests: LSD, distance from the field)

(I) Distance from the field (m)	(J) Distance from the field (m)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
6	19	-.2947	.25447	.249	-.7986	.2091
	33	-.7661(*)	.25033	.003	-1.2617	-.2705
	48	-.1558	.25234	.538	-.6554	.3438
19	6	.2947	.25447	.249	-.2091	.7986
	33	-.4714	.25447	.066	-.9752	.0325
	48	.1389	.25644	.589	-.3688	.6467
33	6	.7661(*)	.25033	.003	.2705	1.2617
	19	.4714	.25447	.066	-.0325	.9752
	48	.6103(*)	.25234	.017	.1107	1.1099
48	6	.1558	.25234	.538	-.3438	.6554
	19	-.1389	.25644	.589	-.6467	.3688
	33	-.6103(*)	.25234	.017	-1.1099	-.1107

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

Appendix 4

Analysis of Variance (ANOVA) of SRP concentrations for all distances at each depth (dependent variable: natural logarithm of SRP, significant difference at p<0.05)

Tests of Between-Subjects Effects (50 cm depth)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	55.826(a)	5	11.165	13.150	.000
Intercept	1402.341	1	1402.341	1651.584	.000
Distance	41.400	4	10.350	12.190	.000
Transect	14.697	1	14.697	17.309	.000
Error	115.476	136	.849		
Total	1555.142	142			
Corrected Total	171.302	141			

a R Squared = .326 (Adjusted R Squared = .301)

Multiple Comparisons (Post Hoc Tests: LSD, distance from the field)

(I) Distance from the field (m)	(J) Distance from the field (m)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
6	19	.1526	.23805	.523	-.3182	.6233
	33	-.1046	.23805	.661	-.5754	.3662
	48	-.8890(*)	.23805	.000	-1.3598	-.4183
	50	-1.3229(*)	.26783	.000	-1.8525	-.7932
19	6	-.1526	.23805	.523	-.6233	.3182
	33	-.2572	.23405	.274	-.7200	.2057
	48	-1.0416(*)	.23405	.000	-1.5044	-.5787
	50	-1.4754(*)	.26428	.000	-1.9981	-.9528
33	6	.1046	.23805	.661	-.3662	.5754
	19	.2572	.23405	.274	-.2057	.7200
	48	-.7844(*)	.23405	.001	-1.2473	-.3216
	50	-1.2183(*)	.26428	.000	-1.7409	-.6957
48	6	.8890(*)	.23805	.000	.4183	1.3598
	19	1.0416(*)	.23405	.000	.5787	1.5044
	33	.7844(*)	.23405	.001	.3216	1.2473
	50	-.4339	.26428	.103	-.9565	.0888
50	6	1.3229(*)	.26783	.000	.7932	1.8525
	19	1.4754(*)	.26428	.000	.9528	1.9981
	33	1.2183(*)	.26428	.000	.6957	1.7409
	48	.4339	.26428	.103	-.0888	.9565

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

Tests of Between-Subjects Effects (100 cm depth)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	31.543(a)	5	6.309	5.463	.000
Intercept	1330.975	1	1330.975	1152.501	.000
Distance	31.542	4	7.886	6.828	.000
Transect	.001	1	.001	.001	.972
Error	155.906	135	1.155		
Total	1649.813	141			
Corrected Total	187.449	140			

a R Squared = .168 (Adjusted R Squared = .137)

Multiple Comparisons (Post Hoc Tests: LSD, distance from the field)

(I) Distance from the field (m)	(J) Distance from the field (m)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
6	19	-.2746	.27082	.312	-.8102	.2610
	33	-.1758	.26866	.514	-.7071	.3555
	48	-1.2556(*)	.26866	.000	-1.7869	-.7243
	50	-.7091(*)	.34435	.041	-1.3901	-.0281
19	6	.2746	.27082	.312	-.2610	.8102
	33	.0988	.27082	.716	-.4368	.6344
	48	-.9810(*)	.27082	.000	-1.5166	-.4454
	50	-.4345	.34604	.211	-1.1188	.2499
33	6	.1758	.26866	.514	-.3555	.7071
	19	-.0988	.27082	.716	-.6344	.4368
	48	-1.0798(*)	.26866	.000	-1.6111	-.5485
	50	-.5333	.34435	.124	-1.2143	.1477
48	6	1.2556(*)	.26866	.000	.7243	1.7869
	19	.9810(*)	.27082	.000	.4454	1.5166
	33	1.0798(*)	.26866	.000	.5485	1.6111
	50	.5465	.34435	.115	-.1345	1.2275
50	6	.7091(*)	.34435	.041	.0281	1.3901
	19	.4345	.34604	.211	-.2499	1.1188
	33	.5333	.34435	.124	-.1477	1.2143
	48	-.5465	.34435	.115	-1.2275	.1345

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

Tests of Between-Subjects Effects (150 cm depth)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	4.541(a)	4	1.135	2.127	.082
Intercept	902.776	1	902.776	1691.236	.000
Distance	4.021	3	1.340	2.511	.062
Transect	.481	1	.481	.902	.344
Error	64.056	120	.534		
Total	971.037	125			
Corrected Total	68.597	124			

a R Squared = .066 (Adjusted R Squared = .035)

Appendix 5

Analysis of Variance (ANOVA) of DO concentrations during flood according to distance and depth (dependent variable: natural logarithm of DO, significant difference at p<0.05)

Tests of Between-Subjects Effects

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	65.248(a)	12	5.437	3.597	.000
Intercept	25.018	1	25.018	16.551	.000
Distance	27.800	3	9.267	6.130	.001
Depth	2.171	2	1.086	.718	.490
Transect	20.358	1	20.358	13.468	.000
Distance * Depth	8.664	6	1.444	.955	.460
Error	133.019	88	1.512		
Total	225.071	101			
Corrected Total	198.267	100			

a R Squared = .329 (Adjusted R Squared = .238)

Estimated Marginal Means

1. Transect

Transect	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
1	-.978	.163	-1.302	-.653
4	-.047	.193	-.430	.336

Multiple Comparisons (Post Hoc Tests: LSD, distance from the field)

(I) Distance from the field (m)	(J) Distance from the field (m)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
6	19	.6421	.33162	.056	-.0169	1.3011
	33	1.5667(*)	.35492	.000	.8614	2.2720
	48	1.0001(*)	.33830	.004	.3278	1.6725
19	6	-.6421	.33162	.056	-1.3011	.0169
	33	.9246(*)	.35772	.011	.2137	1.6355
	48	.3580	.34124	.297	-.3201	1.0362
33	6	-1.5667(*)	.35492	.000	-2.2720	-.8614
	19	-.9246(*)	.35772	.011	-1.6355	-.2137
	48	-.5666	.36393	.123	-1.2898	.1567
48	6	-1.0001(*)	.33830	.004	-1.6725	-.3278
	19	-.3580	.34124	.297	-1.0362	.3201
	33	.5666	.36393	.123	-.1567	1.2898

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

Appendix 6

Analysis of Variance (ANOVA) of TP concentrations for all depths at each distance (dependent variable: natural logarithm of TP, significant difference at p<0.05)

Tests of Between-Subjects Effects (6 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	50.597(a)	3	16.866	10.569	.000
Intercept	2275.907	1	2275.907	1426.145	.000
Depth	8.783	2	4.391	2.752	.069
Transect	42.109	1	42.109	26.387	.000
Error	137.243	86	1.596		
Total	2447.368	90			
Corrected Total	187.840	89			

a R Squared = .269 (Adjusted R Squared = .244)

Tests of Between-Subjects Effects (19 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	19.666(a)	3	6.555	7.164	.000
Intercept	2381.030	1	2381.030	2601.903	.000
Depth	9.712	2	4.856	5.307	.007
Transect	10.118	1	10.118	11.057	.001
Error	78.700	86	.915		
Total	2468.212	90			
Corrected Total	98.366	89			

a R Squared = .200 (Adjusted R Squared = .172)

Multiple Comparisons (Post Hoc Tests: LSD, depth under ground)

(I) Depth under ground (cm)	(J) Depth under ground (cm)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
50	100	.7030(*)	.24713	.006	.2117	1.1943
	150	.6906(*)	.24912	.007	.1954	1.1858
100	50	-.7030(*)	.24713	.006	-1.1943	-.2117
	150	-.0124	.24500	.960	-.4994	.4746
150	50	-.6906(*)	.24912	.007	-1.1858	-.1954
	100	.0124	.24500	.960	-.4746	.4994

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

Tests of Between-Subjects Effects (33 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	39.683(a)	3	13.228	14.245	.000
Intercept	2993.350	1	2993.350	3223.482	.000
Depth	4.407	2	2.204	2.373	.099
Transect	35.923	1	35.923	38.685	.000
Error	83.575	90	.929		
Total	3100.794	94			
Corrected Total	123.258	93			

a R Squared = .322 (Adjusted R Squared = .299)

Tests of Between-Subjects Effects (48 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	61.463(a)	3	20.488	31.256	.000
Intercept	3022.351	1	3022.351	4610.840	.000
Depth	41.665	2	20.832	31.781	.000
Transect	19.421	1	19.421	29.628	.000
Error	57.683	88	.655		
Total	3130.143	92			
Corrected Total	119.146	91			

a R Squared = .516 (Adjusted R Squared = .499)

Multiple Comparisons (Post Hoc Tests: LSD, depth under ground)

(I) Depth under ground (cm)	(J) Depth under ground (cm)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
50	100	.0989	.20757	.635	-.3137	.5114
	150	1.4795(*)	.20916	.000	1.0638	1.8952
100	50	-.0989	.20757	.635	-.5114	.3137
	150	1.3806(*)	.20403	.000	.9752	1.7861
150	50	-1.4795(*)	.20916	.000	-1.8952	-1.0638
	100	-1.3806(*)	.20403	.000	-1.7861	-.9752

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

Tests of Between-Subjects Effects (50 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	15.163(a)	2	7.581	7.668	.002
Intercept	1203.281	1	1203.281	1217.096	.000
Depth	13.074	1	13.074	13.224	.001
Transect	1.164	1	1.164	1.177	.286
Error	31.637	32	.989		
Total	1317.953	35			
Corrected Total	46.800	34			

a R Squared = .324 (Adjusted R Squared = .282)

Estimated Marginal Means

Depth under ground (cm)

Depth under ground (cm)	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
50	6.574	.222	6.121	7.027
100	5.333	.259	4.806	5.861

Appendix 7

Analysis of Variance (ANOVA) of SRP concentrations for all depths at each distance (dependent variable: natural logarithm of SRP, significant difference at p<0.05)

Tests of Between-Subjects Effects (6 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	2.093(a)	3	.698	.934	.428
Intercept	672.243	1	672.243	900.344	.000
Depth	.798	2	.399	.534	.588
Transect	1.282	1	1.282	1.716	.194
Error	66.452	89	.747		
Total	742.051	93			
Corrected Total	68.545	92			

a R Squared = .031 (Adjusted R Squared = -.002)

Tests of Between-Subjects Effects (19 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	15.472(a)	3	5.157	9.824	.000
Intercept	704.950	1	704.950	1342.793	.000
Depth	3.163	2	1.581	3.012	.054
Transect	11.998	1	11.998	22.854	.000
Error	46.199	88	.525		
Total	767.353	92			
Corrected Total	61.671	91			

a R Squared = .251 (Adjusted R Squared = .225)

Tests of Between-Subjects Effects (33 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	5.705(a)	3	1.902	2.635	.055
Intercept	732.010	1	732.010	1014.161	.000
Depth	2.861	2	1.431	1.982	.144
Transect	2.817	1	2.817	3.903	.051
Error	65.683	91	.722		
Total	804.189	95			
Corrected Total	71.388	94			

a R Squared = .080 (Adjusted R Squared = .050)

Tests of Between-Subjects Effects (48 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	85.287(a)	3	28.429	47.168	.000
Intercept	1196.215	1	1196.215	1984.698	.000
Depth	16.273	2	8.136	13.500	.000
Transect	68.209	1	68.209	113.169	.000
Error	54.245	90	.603		
Total	1327.647	94			
Corrected Total	139.531	93			

a R Squared = .611 (Adjusted R Squared = .598)

Multiple Comparisons (Post Hoc Tests: LSD, depth under ground)

(I) Depth under ground (cm)	(J) Depth under ground (cm)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
50	100	-.3796	.19565	.055	-.7683	.0091
	150	.6515(*)	.19719	.001	.2597	1.0432
100	50	.3796	.19565	.055	-.0091	.7683
	150	1.0310(*)	.19565	.000	.6423	1.4197
150	50	-.6515(*)	.19719	.001	-1.0432	-.2597
	100	-1.0310(*)	.19565	.000	-1.4197	-.6423

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

Tests of Between-Subjects Effects (50 m)

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	13.726(a)	2	6.863	9.259	.001
Intercept	479.983	1	479.983	647.508	.000
Depth	1.543	1	1.543	2.082	.159
Transect	10.754	1	10.754	14.507	.001
Error	22.980	31	.741		
Total	534.752	34			
Corrected Total	36.706	33			

a R Squared = .374 (Adjusted R Squared = .334)

Appendix 8

Comparisons of TP and SRP concentrations and ratio of SRP to TP at different depths reported by Heathwaite and Dils (2000) with findings from the present study

	TP (μgL^{-1})			DIP or SRP (μgL^{-1})		
	A	B	C	A	B	C
50 cm						
Mean	500	500.15	998.71	66 (0.132)	44.88 (0.090)	52.04 (0.052)
Std. Error	184	54.15	236.91	6	6.49	11.80
Minimum	68	41.90	12.68	7	4.21	2.94
Maximum	931	2685.57	8485.49	108	302.05	280.43
N	7	86	50	7	92	50
100 cm						
Mean	252	353.31	709.35	74 (0.294)	61.04 (0.173)	31.26 (0.044)
Std. Error	53	36.29	157.97	7	8.33	6.69
Minimum	76	11.83	18.22	3	4.26	2.09
Maximum	727	1635.25	6174.05	164	417.94	175.99
N	24	96	44	24	98	43
150 cm						
Mean	210	246.38	435.30	62 (0.295)	24.14 (0.098)	12.32 (0.028)
Std. Error	32	40.14	90.33	9	2.75	0.83
Minimum	66	13.58	17.23	2	3.56	6.71
Maximum	539	1969.10	2544.26	176	113.96	34.87
N	34	88	37	34	88	37

DIP: dissolved inorganic P equivalent to SRP; A: data during storm conditions (rainfall >10 mm in 24 h) reported by Heathwaite and Dils (2000); B: results from the present study obtained during baseflow; C: results from the present study obtained during flood. The values inside brackets are the ratio of SRP to TP concentrations.

Appendix 9

Analysis of Variance (ANOVA) of DO concentrations according to distance, depth and flow condition (dependent variable: natural logarithm of DO, significant difference at p<0.05)

Tests of Between-Subjects Effects

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	91.697(a)	28	3.275	1.989	.005
Intercept	56.017	1	56.017	34.022	.000
Flow	7.059	1	7.059	4.288	.040
Transect	18.501	1	18.501	11.236	.001
Distance	11.036	4	2.759	1.676	.159
Depth	1.604	2	.802	.487	.615
Flow * Depth	1.168	2	.584	.355	.702
Flow * Distance	14.544	4	3.636	2.208	.071
Distance * Depth	8.730	7	1.247	.757	.624
Flow * Distance * Depth	5.264	7	.752	.457	.864
Error	223.924	136	1.646		
Total	375.320	165			
Corrected Total	315.621	164			

a R Squared = .291 (Adjusted R Squared = .144)

Estimated Marginal Means

1. Flow regime

Flow regime	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
Baseflow	-.956(a)	.205	-1.362	-.551
Flood	-.464(a)	.119	-.700	-.228

a Based on modified population marginal mean.

2. Transect

Transect	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
1	-1.060(a)	.144	-1.345	-.775
4	-.361(a)	.172	-.700	-.021

a Based on modified population marginal mean.

Appendix 10

Analysis of Variance (ANOVA) of ratio of SRP to TP concentrations according to distance, depth and flow condition (dependent variable: ratio of SRP to TP concentrations, significant difference at p<0.05)

Tests of Between-Subjects Effects

Source	Type III Sum of Squares	df	Mean Square	F	Sig.
Corrected Model	2.093(a)	23	.091	2.250	.001
Intercept	9.101	1	9.101	224.988	.000
Distance	.244	3	.081	2.010	.112
Depth	.247	2	.123	3.048	.049
Flow	.566	1	.566	13.994	.000
Distance * Depth	.179	6	.030	.740	.618
Distance * Flow	.016	3	.005	.129	.943
Depth * Flow	.234	2	.117	2.893	.057
Distance * Depth * Flow	.185	6	.031	.763	.600
Error	13.834	342	.040		
Total	28.866	366			
Corrected Total	15.927	365			

a R Squared = .131 (Adjusted R Squared = .073)

Estimated Marginal Means

Flow regime

Flow regime	Mean	Std. Error	95% Confidence Interval	
			Lower Bound	Upper Bound
Baseflow	.213	.013	.188	.238
Flood	.128	.019	.091	.165

Multiple Comparisons (Post Hoc Tests: LSD, depth under ground)

(I) Depth (cm)	(J) Depth (cm)	Mean Difference (I-J)	Std. Error	Sig.	95% Confidence Interval	
					Upper Bound	Lower Bound
50	100	-.08683(*)	.025929	.001	-.13783	-.03583
	150	-.07852(*)	.025929	.003	-.12952	-.02752
100	50	.08683(*)	.025929	.001	.03583	.13783
	150	.00831	.025440	.744	-.04173	.05835
150	50	.07852(*)	.025929	.003	.02752	.12952
	100	-.00831	.025440	.744	-.05835	.04173

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

Appendix 11

Vegetation types at the two study transects

Grass and short shrubs dominant at transect T1

Forest dominant at transect T4

Appendix 12

Soil profile at transect T4 showing organic A horizon underlain by silt-clay B horizon

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