Auckland
Regional Council
TE RAUHITANGA TAIAO

Small headwater streams of the Auckland Region Volume 2: Hydrology and water quality

August 2006 TP312

Auckland Regional Council
Technical Publication No. 312, 2006
ISSN 1175-205X
Volume 1: TP 313, Vol 2: TP 312, Vol 3: TP311, Vol 4: TP310
ISBN-13 : 978-1-87741644-6
ISBN-10 : 1877416-44-4
Printed on recycled paper

Small headwater streams of the Auckland Region Volume 2: Hydrology and water quality

Lucy McKergow
Stephanie Parkyn
Rob Collins
Pete Pattinson

Prepared for
Auckland Regional Council

NIWA Client Report: HAM2006-091
July 2005

NIWA Project: ARC06211

National Institute of Water & Atmospheric Research Ltd
Gate 10, Silverdale Road, Hamilton
P O Box 11115, Hamilton, New Zealand
Phone +64-7-856 7026, Fax +64-7-856 0151
www.niwa.co.nz

Contents

Reviewed by:

J. Cooke

Approved for release by:

M. Scarsbrook

Formatting checked

A. Bartley

1 Executive Summary

Small headwater streams can be highly vulnerable to modification from land use and management changes (e.g., urbanisation, cultivation, deforestation), and re-engineering (e.g., piping and damming). Currently, streams providing year-round habitat for fish, invertebrates or aquatic plants are given greater protection under the Proposed Auckland Regional Plan: Air, Land and Water[1] than streams that dry up for part of the year. The Auckland Regional Council (ARC) requires information on the value of small headwater streams in terms of their function and natural values, to aid development of management options.

The objective of this study was to quantify the hydrology and water quality of headwater streams under dry-stock grazing, a predominant land use in the Auckland Region. The influence of riparian conditions on headwater stream hydrology and water quality are also important, so sites were selected in pasture with and without riparian protection and with a range of riparian vegetation.

Hydrology and water quality of four headwater streams were monitored during a 2 year period (May 2004-May 2006) on the headwaters of the Puhinui Stream at Totara Park. Four sites were selected at Totara Park to cover a range of riparian vegetation types. Cattle have full access to the 0.7 ha Swamp and 2.1 ha Redoubt Rd sites. Cattle are excluded from the Bush (4.3 ha) site by a fenced, mature native bush riparian buffer, and from the lower part of the Gorse (5.6 ha) site by young riparian buffer of mixed gorse and regenerating vegetation. Continuous water level records were collected at the four sites. Water quality sampling included fortnightly grab samples and automated storm monitoring of four storms.

Two of the sites (Bush, Gorse) were perennial and flowed continuously during the monitoring period. The Swamp site ceased to flow for 10 days in the 777 day record, while flow at the Redoubt Rd site became intermittent during the summer months. The baseflow and Q90/Q50 indices suggest that groundwater or subsurface flow dominates the contributions to streamflow at all four sites. At the Redoubt Rd site this groundwater contribution is seasonal, occurring during the winter-spring period.

Suspended sediment (SS) concentrations were generally low during baseflow conditions, but increased to high levels (>400 g/m^3) during storm events. Median grab sample SS concentrations were 5 g/m^3 at the Bush sites, and around 20 g/m^3 at the Redoubt Rd and Gorse sites.

Escherichia coli concentrations ranged over several orders of magnitude, but were generally lower at the Bush site (median 227 MPN/100ml) that the other three sites. All median values were above that considered acceptable for contact recreation (126 *E.coli*/100ml) and stock drinking water (100 faecal coliforms/100ml). High numbers (10^3 to $>10^6$ MPN/100 ml) were measured during storm events, particularly at the Swamp site. *E. coli* exports were also lowest from the Bush site (4 x 10^{10}/ha/ y).

[1] These provisions are subject to a number of appeals and may change.

Nitrogen (N) was dominated by nitrate-N at all sites except the Redoubt Rd site, where organic forms of nitrogen dominate. Nitrate-N exports were similar at the Bush and Swamp sites (7 kg/ha/y) and higher than exports from the Gorse (3 kg/ha/y) and Redoubt Rd (<0.35 kg/ha/y) sites. Seasonal variations in nitrate-N concentrations from the Swamp suggest that during dry periods nitrate removal occurs. Ammonium concentrations and loads were low from all sites.

Particulate forms of P, rather than filterable reactive phosphorus (FRP) dominated total Phosphorus (P) transport. Total P and FRP concentrations and loads were higher at Redoubt Rd and a possible explanation is direct deposition of fertiliser onto the intermittently flowing channel.

The headwater streams in this study had a range of flow duration, ranging from permanent to intermittent. The different levels of riparian protection are reflected in faecal contamination levels and excluding stock from headwater streams could yield improvements in bacterial water quality. Removal of direct fertiliser inputs to intermittent stream channels may also reduce P exports. For other water quality parameters the concentrations and loads do not appear to be strongly influenced by the level of riparian protection, possibly because of the influence of groundwater flow to these streams containing leached pollutants from the surrounding or historical land use.

2 Introduction

2.1 Background

Streams that do not flow all year around have received relatively little research attention in New Zealand or internationally. Headwater streams are highly sensitive to modification of land use (e.g., urbanisation, cultivation) and re-engineering (e.g., piping and damming).

The Auckland Regional Council (ARC) requires information on the value of headwater streams in terms of their function and natural values, to aid development of management options. There are several components to this headwater streams project, including their spatial extent, natural values and hydrological and water quality functions. This report describes the hydrology and water quality functions of headwater streams. A large volume of both surface and subsurface runoff flows through headwater streams. As a consequence, these systems have the potential to modify the flow regime downstream, and influence pollutant transport including sediment, nutrients and faecal material.

2.2 Framework and aims

ARC identified that research should be focussed on the predominant land use within the Auckland Region, dry stock farming. The influence of riparian conditions on headwater stream hydrology and water quality were also important, so sites were selected in pasture with and without riparian protection and with a range of riparian vegetation.

Four hydrogeological areas (HGA) dominate the region – Franklin volcanics, Awhitu/Kaipara sand country, Waitemata sandstones, and Dairy Flat/Wellsford mudstone. Each HGA has different landform, hydrogeological characteristics and runoff rates.

The objective of this study was to quantify the hydrology and water quality of headwater streams under dry-stock grazing in a representative HGA of the Auckland Region.

2.2.1 Definitions

There is considerable inconsistency in the terminology for streams that only flow a part of the year. Temporary, transient, intermittent, and ephemeral are all terms used to describe streams with irregular flow. Flow duration is generally used to differentiate the different stream types. Perennial and intermittent stream types flow well beyond storm events. Under normal circumstances, perennial streams flow all year. Intermittent streams cease to flow for portion of a year, typically during summer and early autumn (Wigington et al. 2005). Ephemeral channels may flow during storm events but typically not for extended periods afterwards. Wigington et al. (2005) further differentiate between ephemeral streams with defined and undefined channels, and use the terms swale for ephemeral streams with gentle, concave drainage ways rather than defined channels.

2.2.2 Hydrology

Streamflow is the result of complex natural processes that operate at the catchment scale. Conceptually a catchment can be perceived as a series of interlinked reservoirs, each of which has components of recharge, storage and discharge. Recharge to the system is largely controlled by rainfall, whereas storage and discharge are complex functions of catchment physiography (Smakhtin, 2001).

Streamflow is typically divided into two reservoirs, 'stormflow' and 'baseflow' for analytical purposes. Stormflow is the water in the stream during and immediately after a storm event, while baseflow is the steady flow between peaks (Davie, 2003). There is general consensus that the major source of baseflow is groundwater, while overland flow, shallow subsurface flow and groundwater can all contribute to stormflow. The amount of groundwater flow depends on the hydrogeological characteristics of the aquifer, streambed materials and the hydraulic gradient. Groundwater hydraulic gradients can change seasonally and may be controlled by topography. When the groundwater level drops below the base of the streambed, groundwater flows will cease and the stream will start to dry up.

During extended wet periods the saturation of soils can increase the size of the channel network significantly. Soil saturation is more likely in soils of moderate hydraulic conductivity, in areas of reduced soil moisture storage and on lower slope angles (Burt and Pinay, 2005). For example, Wigington et al. (2005) found that the stream channel network in five catchments in Oregon increased by an order of magnitude in winter. During summer the perennial channels drainage densities ranged from 0.24 to 0.66 km/km^2, whereas winter drainage densities ranged between 2.9 and 8 km/km^2. Network expansion during the winter season occurred primarily in areas with very low relief and slow-infiltration soils. Intermittent streams, ephemeral streams and swales were important parts of the winter drainage network.

Wetlands may also form part of the channel network. Emergent wetland vegetation can be expected to develop where water velocities following rainfall are not high enough to scour a channel. Seepages (as defined by Johnson and Gerbeaux, 2004) are areas of slope with surface and groundwater flow that is "less than that which would be considered as a stream or spring" and which receives periodic flushes of water from rainfall. In New Zealand they can vary in size from a few square metres to a 1000 m^2 and are well vegetated with pasture grasses, sedges and rushes (Collins, 2004). These wetlands primarily occur where groundwater diffuses to the surface, especially at a change of slope or where an impermeable layer creates a perched water table (Johnson and Gerbeaux, 2004). Seepage wetlands are potentially a critical source area with respect to the delivery of agricultural pollutants to pastoral streams as a disproportionately large fraction of total flow can pass through them (Cooper, 1990).

2.2.3 Water quality

Headwater streams can be source areas and provide transport pathways for sediment, faecal contamination and nutrients. In order to put our research in Auckland headwater streams in a wider context, this section summarises the role of riparian protection, provides an overview of the sources and transport of agricultural contaminants and recent New Zealand research on headwater streams in drystock catchments.

Riparian protection can serve several different water quality functions depending on the site characteristics and pollutant sources and pathways. In water quality terms riparian buffers can: (1) filter surface runoff, (2) provide suitable conditions for nutrient uptake or transformations, (3) stabilise streambank morphology and (4) move sediment and nutrient generating activities away from streams. The first two filtering functions may prevent pollutants generated on hillslopes from reaching the channel system, while the latter two may reduce pollutant generation within riparian areas. It is important to recognise that not all of these functions can operate simultaneously in a given environment and some are incompatible.

Uncontrolled grazing of riparian areas by livestock, both sheep and cattle, can cause degradation of streams and their water quality (Trimble and Mendel, 1995, Belsky et al. 1999). The impact of livestock grazing will depend on the type and density of animal, and bank characteristics. Cattle are attracted to riparian areas and may spend time in and around streams (Trimble and Mendel, 1995).

Recent New Zealand studies have assessed the effect of sheep and cattle grazing on contaminant losses from wetlands, hillslopes, and headwater catchments. Much of this research has been conducted at the Whatawhata Research Station, west of Hamilton (Table 1).

2.2.3.1 Sediment

Sediment in headwater streams can be sourced from within riparian areas or from hillslopes. Sources may include stream bank erosion, rills and gullies and surface wash processes on hillslopes.

Grazing can increase suspended sediment concentration in runoff, particularly in winter when soils can be damaged by grazing (Elliot and Carlson, 2004; Nguyen et al. 1998). Elliot and Carlson (2004) used a rainfall simulator to measure high SS concentrations immediately after grazing (500-2200 g/m^3). After several months without grazing concentrations were less than 100 g/m^3. Sediment mobilised during events after treading damage may not be delivered to the channel network immediately, but may be intercepted downslope by pasture grasses, stock tracks, riparian vegetation and wetlands. In the long term, however, the mobilised and trapped sediments can enter stream systems if pasture grasses are continually grazed, when storm events wash sediments from stock tracks, or in extreme events that scour riparian wetlands as occurred at Whatawhata after a one in 20 year storm event in 1998. Nguyen et al. (1999) found that while small wetlands can be sediment sinks during low flows, fine sediment and particulate organics can be remobilised during storm events. Stock damage to seepage wetlands may also release sediment.

Table 1:

Recent headwater stream research at Whatawhata Research Station, Waikato.

Reference	Location/scale/landuse/duration	Interest	Key results
Elliot and Carlson (2004)	Whatawhata/hillslope/sheep/ experiments	treading	Winter grazing increased 13-16 x for sediment and particulate nutrients and 33-76 times for FRP and NH_4-N
Collins (2004)	Whatawhata/wetland/cattle/	*E. coli*	Baseflow concentrations 10^1 and 10^3 MPN/100ml; stormflow 10^3 to 10^6 MPN/100 ml
Nguyen et al. (1999)	Whatawhata/wetland/sheep/6 months	nutrient removal, residence times	Groundwater= baseflow + >75% stormflow; wetland NO_3-N (54%) and FRP (26%) sink, but NH_4-N and PN source. Sediment sink at low flow and source at high flows.
Quinn and Stroud (2002)	Whatawhata/headwater streams/native, pine, pasture/2-5 years	land use	Pasture stream exports 2.5 to 7 fold higher for SS, TP, TKN, NO_3-N, NH_4-N than native forest; seasonal variability in exports high
Burns and Nguyen (2002)	Whatawhata/wetland/sheep/24 days	nitrate removal	>90% NO_3-N over 1m; removal rate limited by NO_3-N supply; small storms likely to transport NO3-N to streams
Nguyen et al. (1998)	Whatawhata/hillslope/cattle/experiment	treading	treading damage reduced infiltration; on steep areas infiltration -46%,SS +88%, +89% TKN and TP +94% than untreaded area
Rutherford and Nguyen (2004)	Whatawhata/wetland/sheep/experiments	nitrate removal	in dry weather 24% NO_3-N added removed over 1.5 m; 1 day sufficient to remove NO_3-N; vertical mixing may be important to increase removal of upwelling water high in NO_3-N

At the catchment scale SS concentrations and exports are higher from pasture catchments than native forest catchments. Quinn and Stroud (2002) measured median SS concentrations <5 g/m³ (range 0 - 64.5) in two native bush catchments, but medians >6.2 g/m³ (range 2-1270) in three pasture catchments. Exports were also lower (320 kg/ha/y) in native bush than pasture (>990 kg/ha/yr).

2.2.3.2 *E. coli*

Grazing livestock can be an important source of faecal contamination in pastoral streams (e.g., Wilcock et al. 1999). Important pathways to streams include overland flow, subsurface flows and direct deposition by livestock. There is uncertainty about how bacteria are transported - as individual cells, in clumps of cells or attached to soil particles, and the relative importance of each mechanism (Muirhead et al. 2005). In this study *E. coli* are used as an indicator of faecal contamination.

Collins (2004) measured *E. coli* exports from two small wetlands in paddocks grazed by drystock at Whatawhata. During baseflow *E. coli* concentrations typically ranged between 10^1 and 10^3 MPN/100ml and showed a weak correlation with flow. During the monitored events *E. coli* concentrations increased with increasing flow and peak concentrations were higher, typically 10^3 to 10^6 MPN/100 ml. Cattle were attracted to the small, shallower wetlands for grazing in both summer and winter. Field observations showed >20 pats were deposited directly onto a shallow wetland over a 3 day period.

2.2.3.3 Nutrients

Nitrogen can be transported by water in several different forms, including dissolved inorganic N (nitrate, ammonium and nitrite), dissolved organic N and particulate N. The main routes for nitrogen transfer from hillslopes to streams are nitrate leaching, direct inputs of animal excreta to streams and transport of excreta by surface runoff, and soil erosion (Heathwaite, 1993). In New Zealand, the major potential sources of N by leaching in cattle systems are N from urine and dung patches and applied N fertiliser (Ledgard and Menneer, 2005)

Phosphorus may come many sources, including from fertilisers, unfertilised soils, plants, microbial biomass and grazing animals. In headwater streams, direct deposition of P fertiliser into the stream channel can constitute a significant proportion (~8%) of total P exported (Cooke, 1988). Soils that are dried and rapidly rewetted may also release a considerable quantity of P into the soil solution, including mobile P from the soil microbial biomass (Turner and Haygarth, 2001). Phosphorus may be transported in soluble and particulate forms, with particulate P including P sorbed by soil particles and organic matter.

Grazing, accompanied by treading damage can increase TN and TP losses. For example, Nguyen et al. (1998) measured 89% increases in TN losses and >94 % TP losses on steep land, compared with undamaged areas, largely due to increased SS mobility.

Nitrate removal from seepage wetlands has been studied at Whatawhata. Burns and Nguyen (2002) injected tracer at a depth of 10-20 cm and measured removal of >90% over 60 cm and >99% over 1 m. Rutherford and Nguyen (2004), concerned about the surface runoff visible on the wetland, applied tracer to the wetland surface to measure nitrate removal. In dry weather 24% of the added NO_3-N was removed over 1.5m, largely by denitrification.

At Whatawhata, TN in headwater streams was reasonably evenly split between nitrate-N and organic N with ammonium making up a smaller component. Quinn and Stroud (2002) measured lower median concentrations (< 230 mg TN/m^3,100 mg NO_3-N/m^3) in native catchments than in adjacent pasture catchments, where medians > 500 mg TN/m^3 and >360 mg NO_3-N/m^{3}were recorded. Ammonium was a small part of TN, with median concentrations less than 20 mg/m^3. Total P concentrations were similar in both native and pasture headwater streams at Whatawhata, with medians round 30-60 mg TP/m^3 (Quinn and Stroud, 2002). FRP made a minor contribution (<16%) to the TP export in the pasture catchments monitored and emphasises the importance of erosion driven processes in nutrient exports from steep hill country at Whatawhata.

3 Methods

3.1 Site selection

The research was conducted on the headwaters of the Puhinui Stream at Totara Park. Totara Park is a Manukau City Park and is grazed by drystock. Two hundred 1-2 year old heifers graze the pasture as set stock (15 per paddock). Super-phosphate is applied to the pasture each autumn at a rate of 400 kg/ha.

The HGA underlying the sites is Waitemata sandstone, the most extensive HGA in the region. The Waitemata sandstones are a series of rocks or formations that make up the Waitemata Group. These rocks underlie a large proportion of the region. The generally consist of alternating sandstones and mudstones, which are overlain by a thick weathering profile consisting of silty clays. The Waitemata Group rocks have low to moderate hydraulic conductivity, and hence have low inflows to streams.

Four sites were selected at Totara Park to cover a range of riparian vegetation types and levels of protection (Figure 1-6, Table 2).

Figure 1:
Totara Park showing the monitoring sites, catchment boundaries, vegetation cover and NZMS 260 20 m contours and stream network.

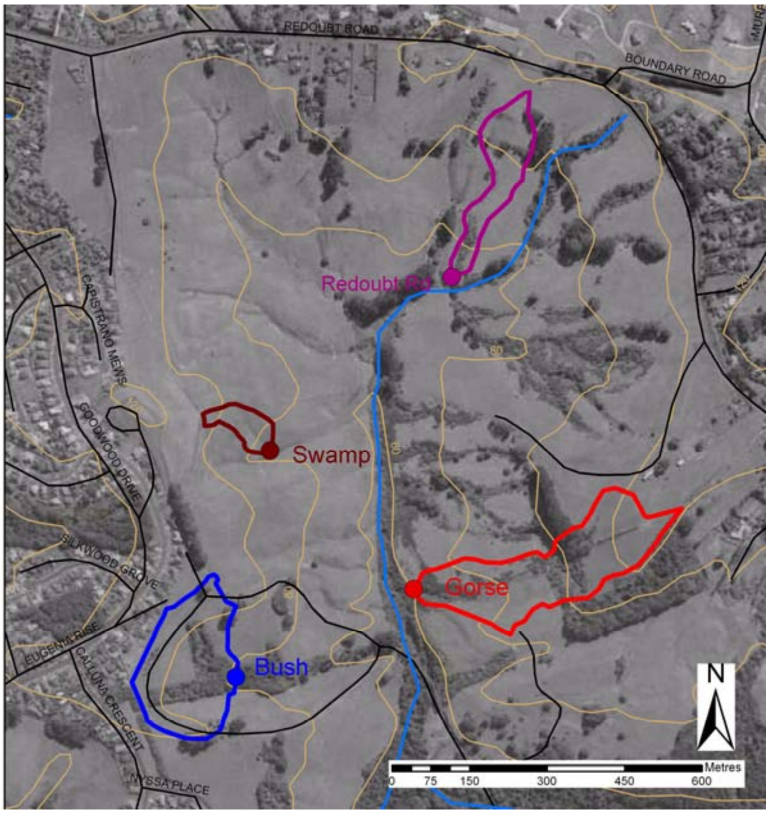


Table 2:

Site characteristics.

Site	Bush	Swamp	Gorse	Redoubt Rd
catchment area	4.3 ha	0.7 ha	5.6 ha	2.1 ha
land use	dry stock	dry stock	dry stock	dry stock
riparian vegetation	native forest	pasture	gorse/scrub and pasture	pasture
stock access to channel	No	Yes	No	Yes

Cattle have full access to the Swamp and Redoubt Rd sites and hence they provide a baseline or worst-case with respect to water quality. The Swamp site has a small seepage wetland. The wetland is 15m long and 5m wide, with soil depths of between 50 and 90cm down the centre line of the wetland.

Cattle are excluded from the Bush site by a fenced, mature native bush riparian buffer. The bush area covers 1 ha and the buffer is at least 20 m wide on both sides of the stream.

The Gorse site is intermediate relative to the others, with a young riparian buffer (< 5 yrs) of mixed vegetation in the lower part of the catchment. The buffer has a variable width, ranging between 5 and 20 m and is a mix of gorse and regenerating native vegetation. The buffer was fenced and cattle excluded when sampling began. In the upper reaches of the catchment stock have access to the channel and these paddocks are generally grazed by horses. A culvert under the road joins the upper and lower parts of the catchment. In December 2005 gorse was cleared from and area inside the fenced buffer and piled for future burning (see Figure 5).

Figure 2:
The Swamp site and catchment (Photo: Ron Ovenden, 6 Sep 2005).

Figure 3:

The Bush site (Photo: Ron Ovenden, 6 Sep 2005).

Figure 4:

The Gorse site (Photo: Ron Ovenden, 6 Sep 2005).

Figure 5:

Looking south-east towards the lower Gorse catchment in December 2005. Note the cleared gorse near the top of the photo.

Figure 6:

Redoubt Rd catchment (Photo: Ron Ovenden, 30 May 06).

3.2 Flow and water quality monitoring

V-notch weirs were installed at all four sites (Table 3). Water levels were monitored at 2.5 min intervals by Hydrologger (Unidata, model 2001) shaft encoders (resolution 1 mm) and data loggers. Water levels were converted to flow using standard rating curves for sharp-crested v-notch weirs.

Rainfall data from two sites is available. Long term rainfall records exist for the ARC site Puhinui at Botanics (740815). A short term record, starting 13 October 2005, was also measured at the Swamp site. The swamp rainfall data were logged at 2.5 minute intervals, to a resolution of 0.2 mm.

Table 3:

Site monitoring details.

Site	Bush	Swamp	Gorse	Redoubt Rd
flow start date	22 Apr 04	22 Apr 04	22 Mar 05	26 Sep 05
flow end date	8 June 06	8 June 06	8 June 06	8 June 06
length of record (days)	777	777	443	255
control	90° v-notch weir, 453 mm range	½ 90° v-notch weir, range 305mm	½ 90° v-notch weir, range 510mm	½ 90° v-notch weir, range 750mm
fortnightly samples start	10 Aug 04	10 Aug 04	22 Mar 05	30 Sep 05
fortnightly samples end	16 May 06	16 May 06	16 May 06	16 May 06
length of record (days)	754	754	420	225
no. events monitored	2	2	2	0

Water quality samples were taken at fortnightly intervals and during a small number of events. Fortnightly grab samples were stored on ice immediately after collection.

In addition to the fortnightly sampling, storm event samples were also collected at three of the sites using automatic samplers. The samplers were triggered when the water level reached a predetermined level. Samples were collected on a time basis, typically 10 to 15 minute intervals. Automatically collected samples were retrieved the following day and stored on ice during transport to the lab.

All samples were sent to NIWA's Chemistry Lab in Hamilton for analysis. All samples were analysed for suspended sediment (SS), *E. coli*, total nitrogen (TN), nitrate (NO_3-N), ammonium (NH_4-N), total phosphorus (TP) and filterable reactive phosphorus (FRP). The term FRP is used here, rather than dissolved or soluble, as the filtrate could be a mixture of dissolved and P attached to colloidal material that passes through the 0.45 μm filter. Table 4 summarises the analysis methods and detection limits.

Table 4:

Water quality analysis methods.

Analyte	Method description	Units and detection limit	Method code
suspended sediment	Gravimetric determination after filtration & drying at 104°C	0.5 g/m³	APHA 2540D
E. coli	IDEXX Laboratories Inc Colilert Test Kit	1 MPN/100ml	APHA 9223B
total nitrogen (TN)	Persulphate digest, auto cadmium reduction, FIA	10 mg/m³	Lachat
nitrate and nitrite N (NO_3-N)	flow injection analysis	1 mg/m³	Lachat
ammonium-N (NH_4-N)	flow injection analysis	1 mg/m³	Lachat
total phosphorus (TP)	Acid persulphate digestion, molybdenum blue colorimetry	1 mg/m³	NWASCO 38
filterable reactive phosphorus (FRP)	flow injection analysis	1 mg/m³	Lachat

3.2.1 Data analysis

Various graphical and statistical methods were used in this report to summarize data and facilitate comparisons between the four sites. Boxplots are used to show the distribution of the water quality data in a format that allows for easy comparison between sites. Line plots are used to show data variability over time and scatter plots are used to show the relation between two variables. LOWESS (locally weighted scatterplot smoothing) smooth lines are added to scatter plots to identify trends. The LOWESS trend lines illustrate relationships between concentrations and flow that are difficult to discern in a simple scatterplot. This smoothing technique is useful as no assumptions regarding linearity of the data are required (Helsel and Hirsch, 2002). A smoothness factor of 0.5 was used, which means that the closest 50 % of the data points were used to calculate each smoothed value.

3.2.1.1 Statistical analysis

Statistical analyses used in this report include basic descriptive statistics and Kruskall-Wallis and Kolmogorov-Smirnov tests. All statistical analysis was conducted using SYSTAT version 11. Exploratory data analysis included inspections for normality. The majority of fortnightly water quality datasets (33/36) were non-normal (Shapiro-Wilk $p<0.05$) and so non-parametric statistical methods were used. The median is used as the measure of central tendency and the inter-quartile range (IQR) as a measure of spread. For each parameter a Kruskall Wallis One Way ANOVA on Ranks was used to detect differing distributions between sites. If the distributions were significantly different a Kolmogorov-Smirnov Two Sample Test (K-S) was used to identify differing sites.

The additional advantage of using non-parametric statistics is the inclusion of data below, and, above (for *E coli* dilution series) the detection limits. A small proportion of FRP and *E. coli* values were above or below the detection limit. For FRP one fortnightly sample (1/135) and six storm samples (6/110) were below the detection limit. Three *E. coli* samples were below the detection limit and 19 were above the detection limit of the dilution used.

3.2.1.2 Hydrological analysis

Tideda version 4 was used for hydrological analysis, with secondary data analysis in Excel and SYSTAT. Instantaneous flows were analysed, as at least 5 years of record is recommended before using mean daily flows (Davie, 2002). Statistics were calculated by PSUMMARY and flow duration was analysed using PDIST. Specific discharge ($l/s/km^2$) was calculated by dividing the PDIST results by the catchment area.

Flow analysis includes Q90/Q50 and percentage of time the stream is at zero-flow. The index Q90/Q50 may be interpreted as an index representing the proportion of streamflow originating from groundwater stores (excluding the effects of catchment area), while the percentage of time at zero flow illustrates the degree of intermittency (Smakhtin, 2001).

Baseflow separation was done using a Lyne and Hollick digital filter on daily flow using xlr8tr (Moore, 2002) with the default parameters ($\alpha = 0.925$ and 3 passes). The baseflow index (BFI) was estimated for the period of record and is defined as the volume of total baseflow divided by the total streamflow volume. In catchments with high groundwater contributions to streamflow BFI may be close to 1, but it is close to zero for ephemeral streams (Smakhtin, 2001).

Runoff:rainfall ratios were calculated on a monthly basis for all sites for common periods of flow and rainfall. Rainfall volumes were estimated using the total rainfall depth spread over the contributing catchment area.

3.2.1.3 Load estimation

All load estimation approaches assume that the error in estimating flows are small and that the sample collected is representative of average conditions. With these assumptions in mind it is important to recognise the short time period over which data has been collected.

In this report loads are calculated using linear interpolation, averaging estimators and regression models. Not every one of these methods is suitable for any particular dataset. A basic description of each method and its suitability are summarized in Table 5. By using several approaches a likely range of load estimates can be predicted and if agreement between the estimates is good this provides confidence in the estimated loads.

Table 5:

Load estimation methods and their suitability (Degens and Donohue, 2002, Helsel and Hirsch, 2002). C denotes concentration and Q discharge.

Method	Description	Suitability
Linear interpolation	C converted to a continuous variable by interpolation and load calculated for each Q	best when true temporal pattern represented
Averaging	average C x average Q	temporally intensive data
		little variation in nutrient concentration
		underestimates when < 50 samples/yr when conc positively related to flow and overestimates when conc negatively related to flow
Regression	Regression of C or LnC, with Q or LnQ and seasonal functions	depends on single functions to explain the Q-C relationship
		best when: linear model fits, > 30 sample points, model is not used to extrapolate beyond the range of data

Loads of sediment and nutrients were calculated as the product of mean daily streamflow and estimated concentrations using the LOADEST model (Runkel et al. 2004). This model is a log-linear regression analysis of constituent concentrations against functions of streamflow and time.

The basic regression equation used is:

$$\ln C(t) = \beta_0 + \beta_1 \ln Q(t) + \varepsilon$$

where, C(t) and Q(t) are the concentration and discharge at time t, respectively and β_0 and β_1 are coefficients estimated by linear regression and ε is the model residual. Additional explanatory variables were added to the model to improve residual plots and model predictive capabilities. These include decimal time, seasonality functions, $\sin(2\pi T)$ and $\cos(2\pi T)$ (where, T is the date in years), and $\ln Q^2$ (see Appendix 1). The sine and cosine terms, which account for seasonality, were always included together if either was selected.

LOADEST performs calibration procedures and makes load estimates using three statistical estimation methods: Adjusted Maximum Likelihood Estimation (AMLE), Maximum Likelihood Estimation (MLE) and Least Absolute Deviation (LAD). The “best” model is chosen using the Akaike Information Criterion, which selects a model that fits well but has a minimum number of parameters. The output includes the probability plot correlation coefficient and residuals. Residuals must be normally distributed to use AMLE and MLE, so if the residuals do not adhere to this assumption the load estimates from the LAD model can be used instead. The model residuals were analysed in Excel and checked for non-uniformity, lack of equal variance, seasonality and non-normality. In this study, the coefficient of efficiency, E, was also used as a model performance measure. The coefficient of efficiency, expresses the proportion of variance of the observed concentration that is explained by the model (Nash and Sutcliffe, 1970). The load estimation methods used for each site and the period covered are summarised in Table 6.

All fortnightly samples were used to develop the regression models. LOADEST requires a minimum of 12 or more non zero observations for constituents and so models were not developed for the Redoubt Rd site.

Peak flows at the Bush site were not measured during some events due to instrumentation problems. As a result the load estimates and flow statistics may be underestimates at that site.

Table 6:

Load estimation methods used. (ID= insufficient data, NM =no model, AMLE = Adjusted Maximum Likelihood Estimation and LAD = Least Absolute Deviation).

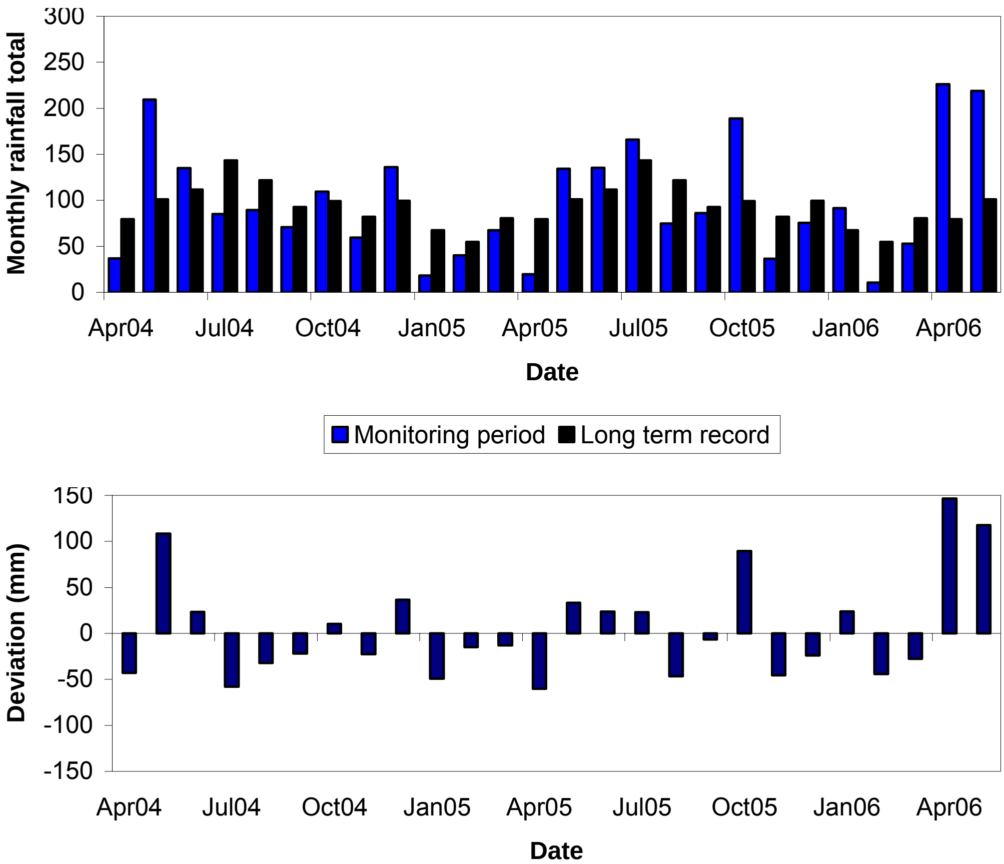
Site	Bush	Swamp	Gorse	Redoubt
Periods	1May04-1May06	1May04-1May06	1May05-1May06	1Oct05-1May06
Interpolation	✓	✓	✓	✓
Averaging estimator	✓	✓	✓	✓
SS model	AMLE	AMLE	AMLE	ID
EC model	AMLE	NM	AMLE	ID
TN model	LAD	NM	AMLE	ID
NO_3-N model	LAD	NM	LAD	ID
NH_4-N model	AMLE	AMLE	AMLE	ID
TP model	AMLE	AMLE	AMLE	ID
FRP model	AMLE	NM	AMLE	ID

4 Results

4.1 Monitoring period rainfall characteristics

The annual mean rainfall at Puhinui at Botanics is 1210 mm. In 2004 the annual total was 1312 mm, while in 2005 the total was 1043 mm. A more useful comparison for the 26 month monitoring period is to evaluate monthly totals. Rainfall totals were generally below the long term monthly medians (Figure 7). The exceptions were May04, Oct 05, Apr and May 06 which all had >90 mm more than the long term record (Figure 7).

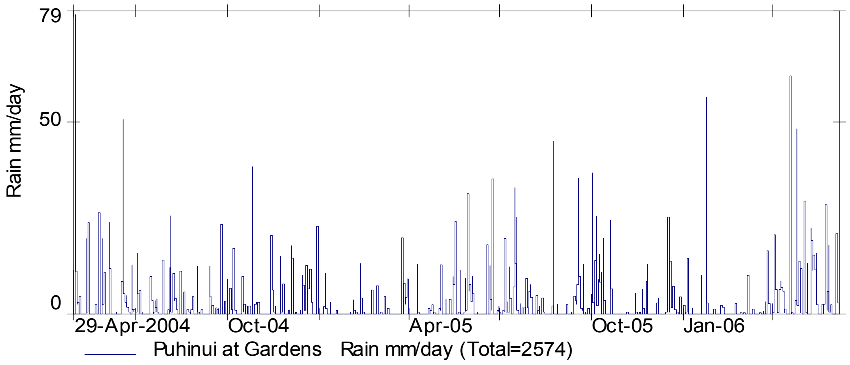
Figure 7:
Monitoring period and long term record median monthly rainfall (mm) (Puhinui at Botanics) and deviation from the long term median.



Daily rainfalls were variable and there were 225 days with a total rainfall >2 mm, 139 days > 5mm, 85 days >10 mm and 29 days >20 mm and 4 days >50mm during the monitoring period (22 April 04 - 1 June 06). The longest run of dry days in the long term record is 28 days. During the monitoring period the longest dry run was 14 days. The maximum daily rainfall recorded was 78 mm on 1 May 2004.

Figure 8:

Daily rainfall totals for Puhinui at Botanics.

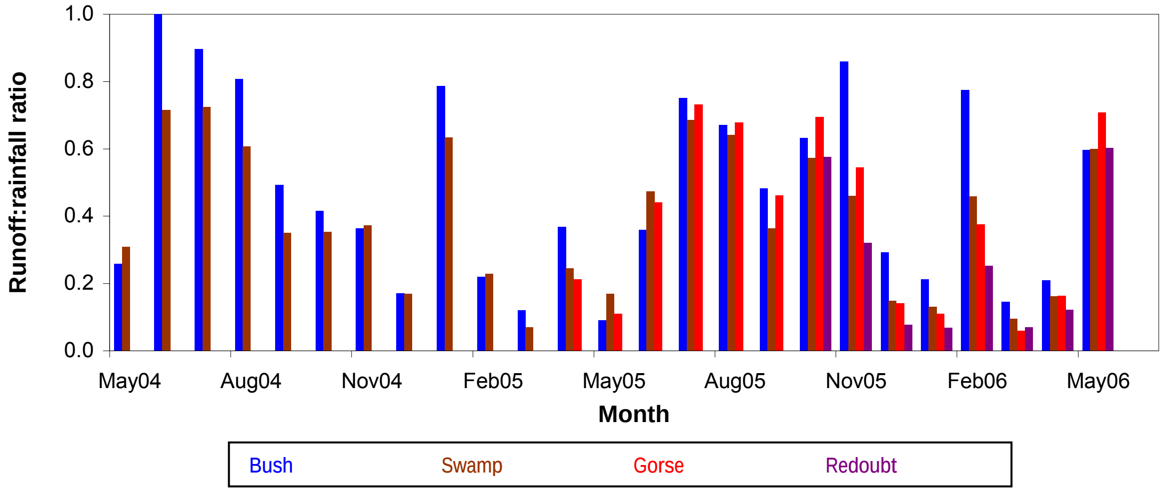


4.2 Runoff ratios

On an annual basis (May-May) between 37 and 48% of rainfall is measured as runoff at the Bush, Swamp and Gorse sites. On a monthly basis there is considerable variability. Generally, 10 to 20% of monthly rainfall became runoff during summer, while between July and December up to 100% of rainfall may become runoff.

Figure 9:

Monthly runoff:rainfall ratios for the four monitoring sites.



4.3 Flow dynamics

The Bush and Gorse sites were perennial during the monitoring period. Flows ranged between 0.066 and 145 l/s at the Bush site, with a mean flow of 0.8 l/s (Table 7). During the 14 month monitoring period the Gorse mean flow was 0.995 l/s and the median 0.385 l/s.

There were periods of no flow at both the Swamp and Redoubt Rd sites. Flows at the Swamp site ranged between 0 and 46.6 l/s, with a mean flow of 0.099 l/s (Figure 10, Table 7). The site stopped flowing for a 10 day dry period in March 2005 and flow resumed after a 16.2 mm rainfall event. The Redoubt Rd site was intermittent, with only storm flows during the period Jan- mid April 2006. Ten percent of the flow record was zero flow days, and 21% of the flows were less than 0.001 l/s during the 9 month flow record. This is clearly illustrated by the hydrograph, with flows less than 1 ml/s not included on the log scale (Figure 10).

Baseflow dominated the hydrographs at all sites (Figure 11). The flow duration curves all have low slopes, which suggest low flow variability and high baseflows (Figure 12). Baseflow index values were in excess of 0.65 (Table 7) and Q90/Q50 ratios less than 0.3. The Gorse flow duration curve and flow indices suggest that groundwater or subsurface flow dominate flow, but not quite to the same extent as the Bush or Swamp sites (Table 7). The Redoubt Rd flow duration curve has a steeper slope and for 45% of time flow was below 0.01 l/s (Figure 12). The Q90/Q50 index and BFI at Redoubt Rd were lower than at the other sites (Table 7). This suggests that the Redoubt Rd flow volume is seasonally dominated by groundwater or subsurface flow.

Table 7:

Flow statistics for the periods of record.

	Units	Bush	Swamp	Gorse	Redoubt
Min	l/s	0.066	0	0.025	0
Max	l/s	145	46.6	198	48.6
Mean	l/s	0.762	0.099	0.995	0.284
Std. Dev.	l/s	2.68	0.422	3.13	1.10
Median	l/s	0.41	0.046	0.385	0.015
IQR	l/s	0.547	0.095	1.04	0.249
25%ile	l/s	0.179	0.019	0.106	0.002
75%ile	l/s	0.726	0.114	1.14	0.251
95%ile	l/s	1.963	0.279	3.18	1.11
Q90/Q50	-	0.26	0.22	0.15	0.07
% time zero flow	-	0	1	0	10
Total volume	m^3	53470	7310	38120	6530
Baseflow volume	m^3	44760	6090	28400	4230
Baseflow index (BFI)	-	0.84	0.83	0.74	0.65
Stormflow volume	m^3	8710	1210	9720	2300

Figure 10:

Instantaneous hydrographs for the monitored sites. Note the differing x-axis scales, particularly for the Gorse and Redoubt sites.

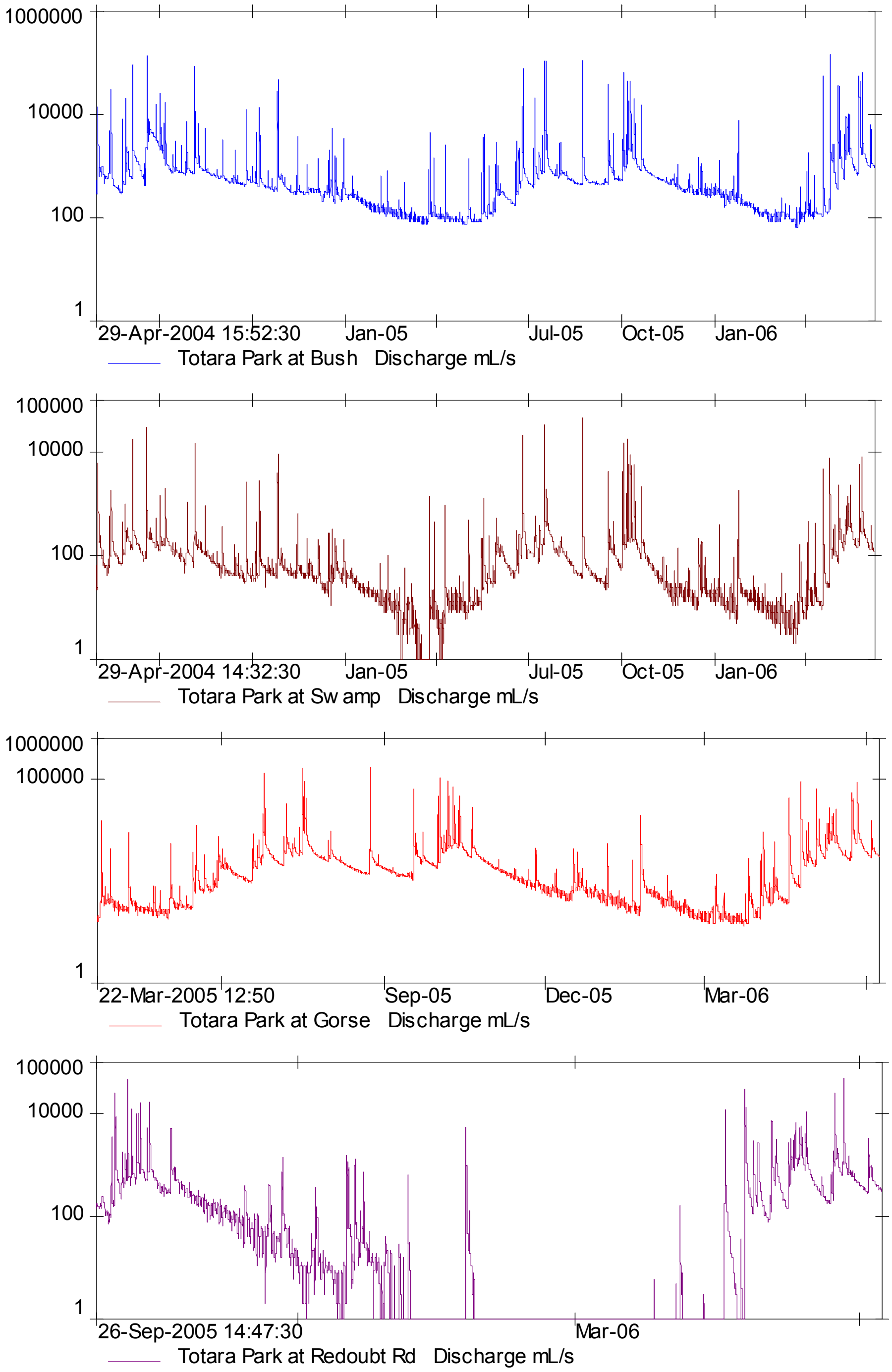


Figure 11:
Daily flow and baseflow hydrographs for the four sites. Note the differing x-axis scales.

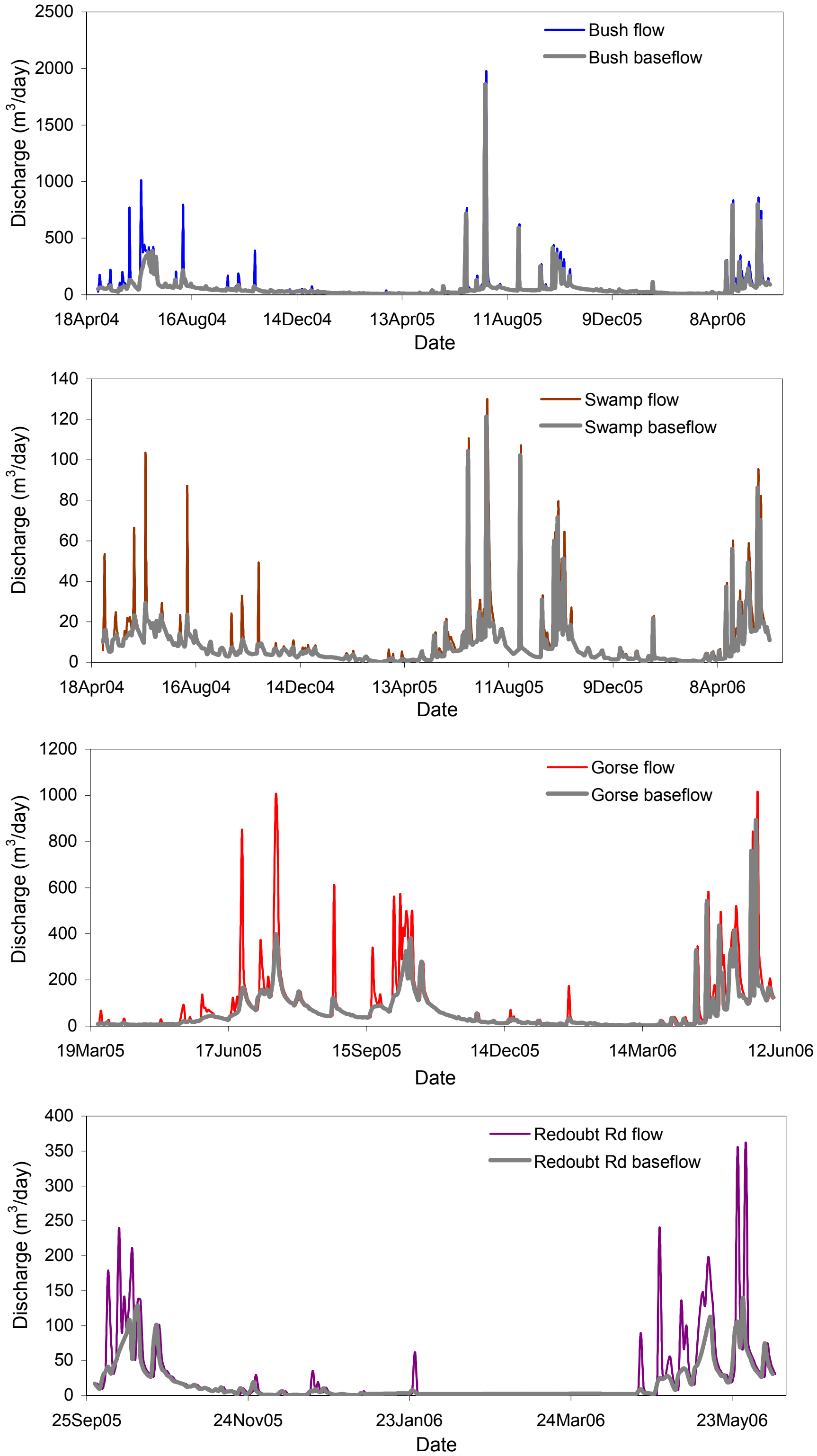
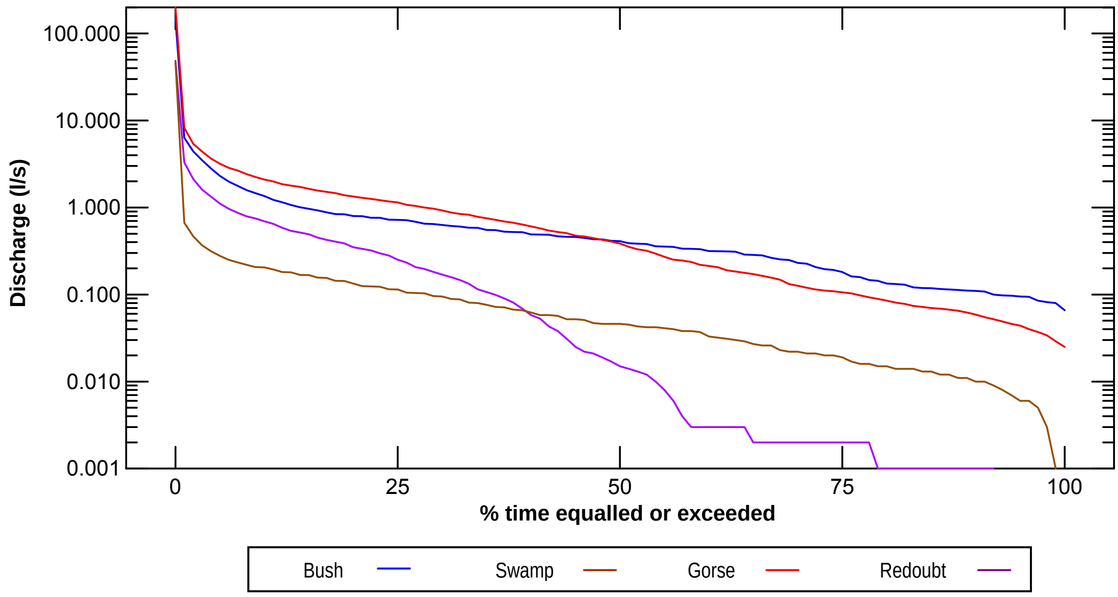


Figure 12:

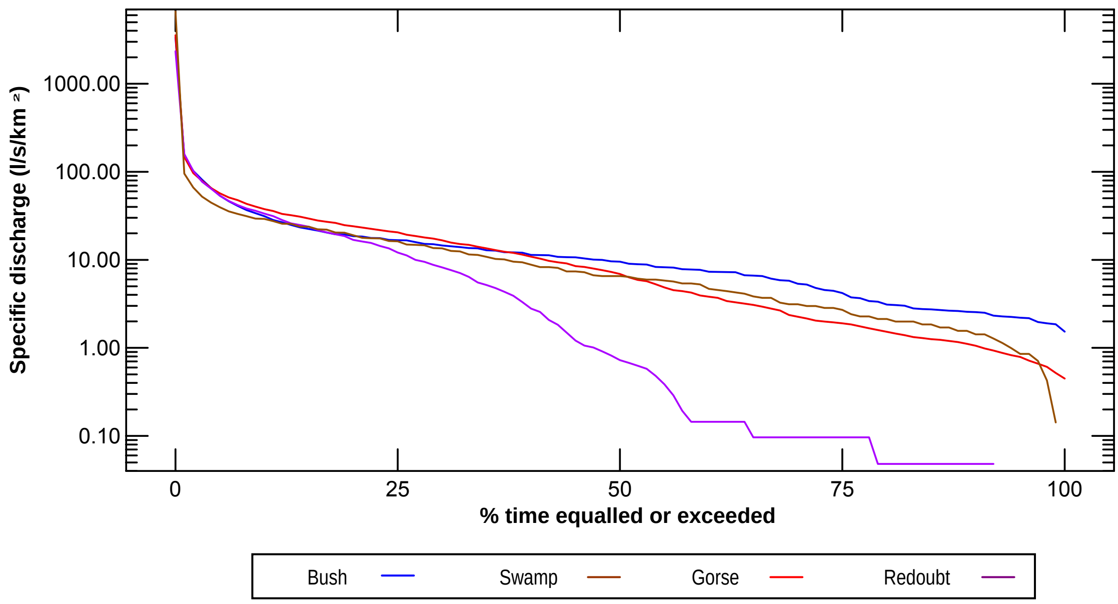
Instantaneous flow duration curves over the period of record for the four sites.



Comparison of specific discharge flow duration curves suggests that three of the sites behave similarly (Figure 13). The Bush, Gorse and Swamp curves are similar in shape and vertical position indicating that their specific discharges are similar. The Redoubt Rd site by comparison has a lower specific discharge for 75% of the time and flow is less reliable.

Figure 13:

Specific discharge flow duration curves (l/s/km²) for the four sites.



The sites were all responsive to rainfall inputs, with many short duration peaks in the hydrographs. A total of 76 peaks (over 2 l/s) were recorded during the monitoring period at the Gorse site (Table 8). The maximum flow recorded as 198 l/s and there were nine peaks with flows greater than 50 l/s. The peak flows at the Swamp and Redoubt Rd site were smaller, just under 50 l/s (Table 8).

Table 8:

Frequency distribution of peak discharges greater than 2 l/s and with a minimum separation of 60 minutes. Refer to Table 3 for the period of record.

Class	Bush	Swamp	Gorse	Redoubt
2-5	55	13	49	18
5-10	14	10	12	8
10-25	13	5	3	7
25-50	7	3	3	5
>50	11	0	9	0
Total	100	31	76	38

4.4 Water quality

4.4.1 Fortnightly sample concentrations

Sampling under the fortnightly monitoring program has primarily been at low flow, with fewer samples collected at above the median flow (Table 9). The exception is the Redoubt Rd site, as on six of the site visits there was insufficient flow to collect a sample.

4.4.1.1 Suspended sediment

Suspended sediment concentrations were typically low, with median values of between 5 and 11 g/m^3 for the Bush, Gorse and Swamp sites (Table 10, Figure 14). The Redoubt Rd site had a higher median concentration (20 g/m^3) and larger spread of values (IQR 26.5). The Bush site had significantly lower SS concentrations than the other three sites (K-S, $p=0.000$).

A time series plot (Figure 15a) of the SS concentration data shows that a seasonal pattern is evident at the Swamp but not at other sites. The SS-discharge plot (Figure15b) suggests that SS concentrations are stable at all flows at the Bush and Gorse sites.

Figure 14:

Boxplots of fortnightly sample SS and E.coli concentrations. Box represents the median with 25th and 75th percentiles, whiskers are the 10th and 90th percentiles and outliers are dots. Sites identified by the same letter are not significantly different (K-S, $p < 0.05$).

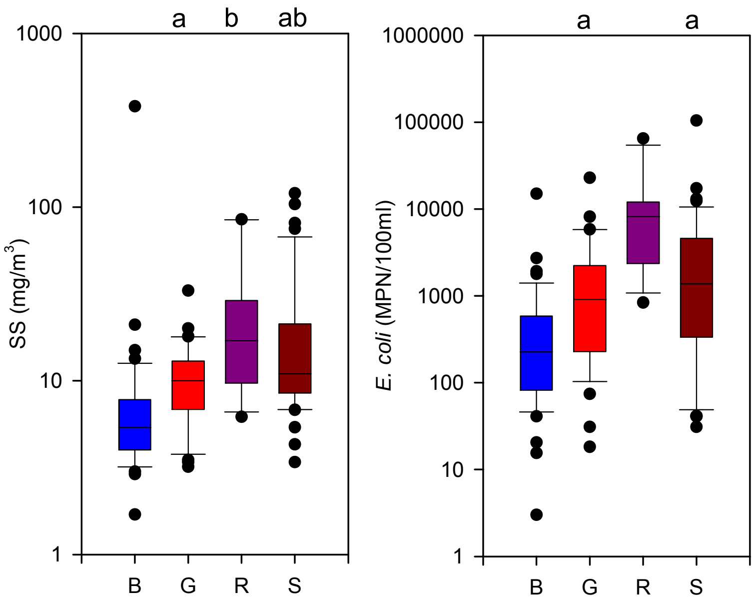


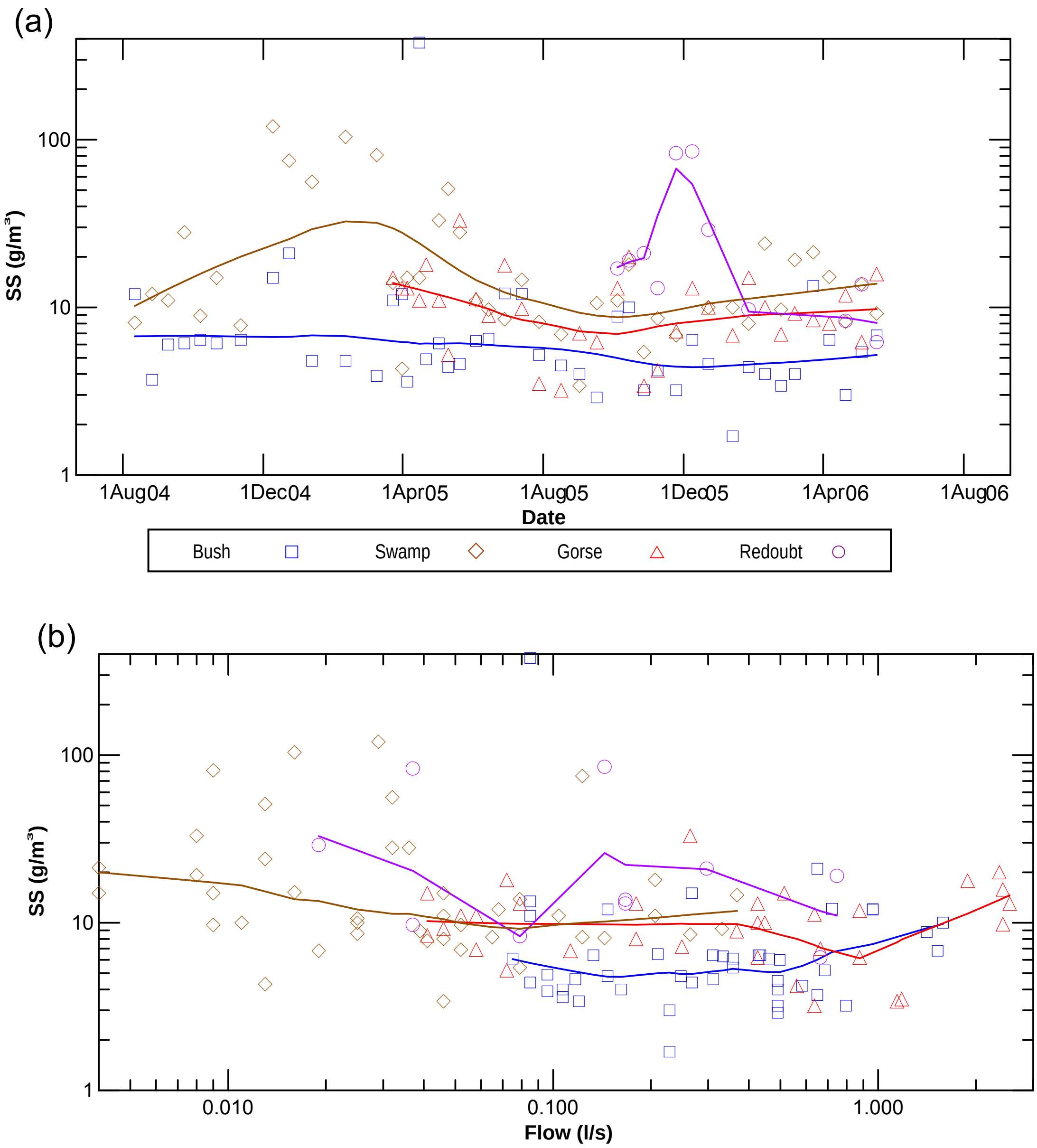
Table 9:

Water quality statistics for the fortnightly monitoring.

Site		SS g/m³	E. coli MPN/100 ml	TN mg/m³	NO_3-N mg/m³	NH_4-N mg/m³	TP mg/m³	FRP mg/m³
Bush	median	5.4	227	1920	1705	13.5	28	10.5
	IQR	3.3	491	810	760	9	13	6
	min	1.7	3	1090	853	5	12	5
	max	380	15000	3280	3200	79	273	25
	n	45	46	46	46	46	46	46
	n>median flow	19	19	19	19	19	19	19
Gorse	median	17	915	862	322	43	32.5	6
	IQR	6.1	1905	417	403	47	22	4
	min	3.2	18	522	11	8	9	2
	max	33	22820	2390	2080	410	74	11
	n	33	34	34	34	34	34	34
	n>median flow	18	18	18	18	18	18	18
Redoubt	median	20	8212	1230	55	41	142	33
	IQR	26.5	8910	246	74	24	53	33
	min	6.2	2359	735	4	21	49	21
	max	85	64880	1310	351	84	158	62
	n	11	11	11	11	11	11	11
	n>median flow	10	10	10	10	10	10	10
Swamp	median	11	1380	2420	1715	63	32	4
	IQR	12	4206	1810	2344	101	38	4
	min	3.4	31	462	3	22	12	<1
	max	120	104624	3880	3580	1560	307	26
	n	43	46	45	46	46	46	46
	n>median flow	15	15	15	15	15	15	15

Figure 15:

Suspended sediment fortnightly sample (a) time series and (b) relationship with flow, with LOWESS smooth.



4.4.1.2 E.coli

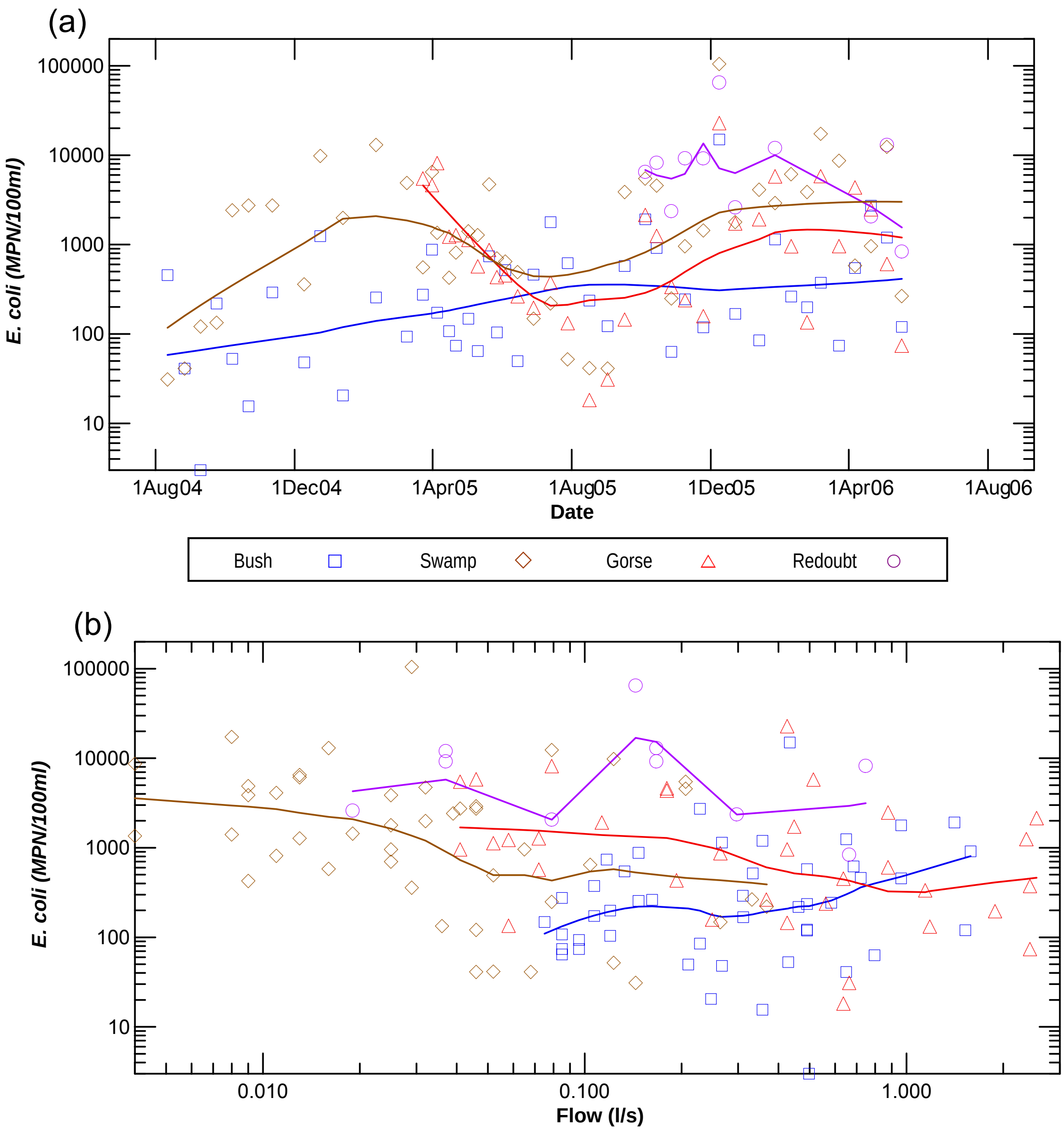
E. coli numbers ranged over several orders of magnitude, with a maximum of 104624 MPN/100ml at the Swamp site. This was several orders of magnitude greater than the freshwater recreational guideline of 126 MPN/100ml. The median concentration increased with decreasing levels of riparian protection from 208 MPN/100ml at the

Bush site to 8710 MPN/100ml at Redoubt Rd (Table 10). The Gorse and Swamp site distributions were not statistically significantly different (K-S, p=0.393) despite the Gorse site having greater riparian protection.

Both the Gorse and Swamp sites had seasonal declines in *E. coli* numbers (Figure 16a) from maximums in March to minimums in September. Figure 16 shows that concentrations at Redoubt Rd were generally higher than the other sites on each sampling date. There are no clear relationships between *E. coli* and flow evident for the four sites (Figure 16b).

Figure 16:

E. coli fortnightly sample (a) time series and (b) relationship with flow, with LOWESS smooth.

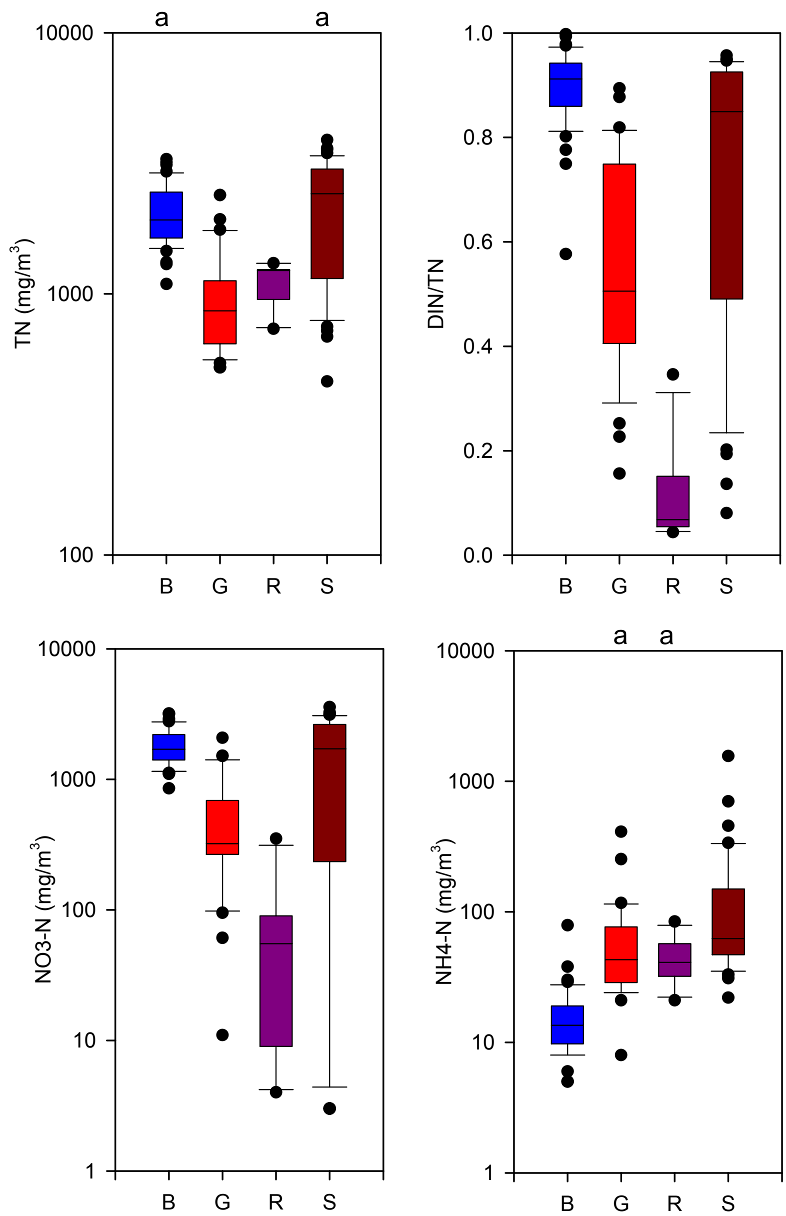


4.4.1.3 Nitrogen

Total N concentrations at Gorse and Redoubt Rd were generally lower than those at Swamp and Bush (Figure 17). The Bush site had a small TN range, between 1090 and 3280 mg/m^3, with a median of 1920 mg/m^3 (Table 9). By comparison the Swamp had the largest range of TN concentrations (Figure 17).

Figure 17:

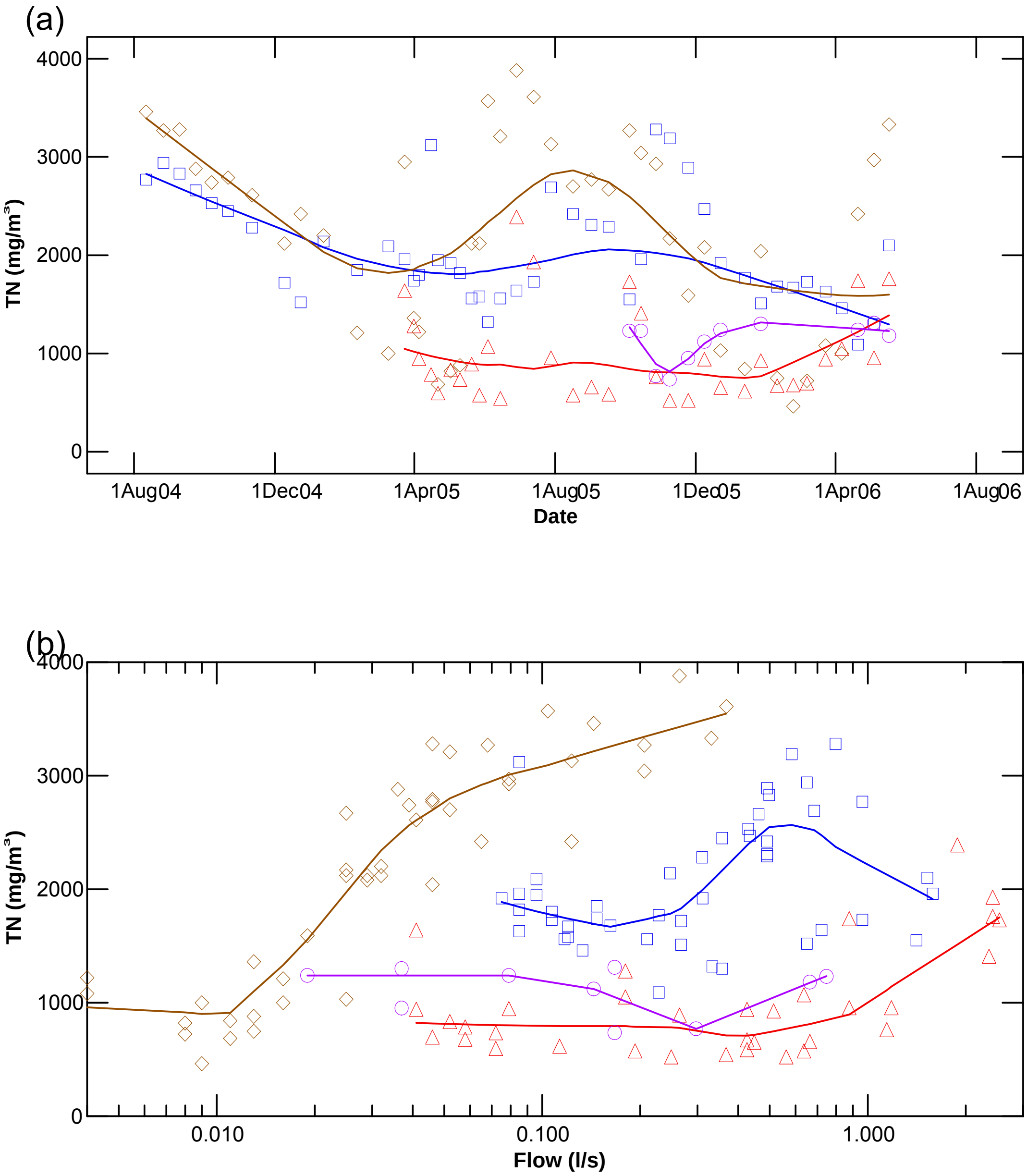
Boxplots of fortnightly sample N concentrations. Box represents the median with 25th and 75th percentiles, whiskers are the 10th and 90th percentiles and outliers are dots. Sites identified by the same letter are not significantly different (K-S, $p < 0.05$).



Both the Bush and Swamp sites have seasonal TN patterns, with a general decrease in concentration from August through to March in both years monitored. While there is a shorter period of data for the Gorse site, it does not show the same pattern (Figure 18a). Total N concentrations at the Swamp and Bush sites also have positive relationships with flow (Figure 18b). At the Swamp site the TN concentration increases with $\log_{10}$ flow (Figure 18b).

Figure 18:

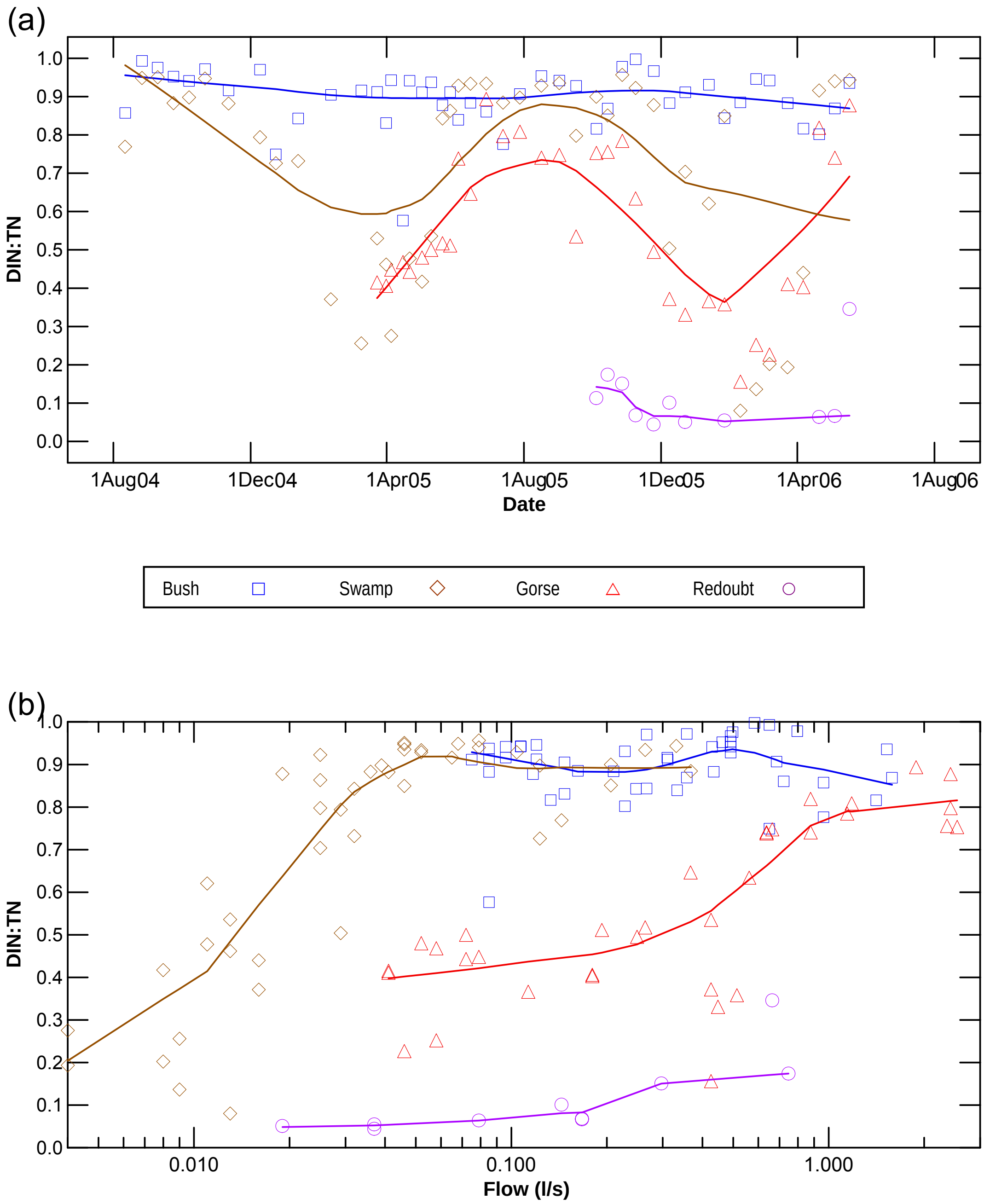
Total N fortnightly sample (a) time series and (b) relationship with flow, with LOWE SS smooth.



Total N is dominated by DIN at the Bush and Swamp sites. The ratio of DIN (sum of NO_3-N and NH_4-N) to TN has a small range for the Bush site (Figure 17) with a median of 0.91. While the range of DIN:TN ratio is larger for the Swamp, the median is also high at 0.85. At the Gorse site the DIN:TN ratio ranges between 0.15 and 0.9, with a median of 0.5 (Figure 17, Figure 19). At Redoubt Rd nitrogen concentrations were dominated by organic N, with nitrate-N and ammonium-N contributing less than 0.34 to TN (Figure 17, Figure 19).

Figure 19:

DIN:TN fortnightly sample (a) time series and (b) relationship with flow, with LOWESS smooth.

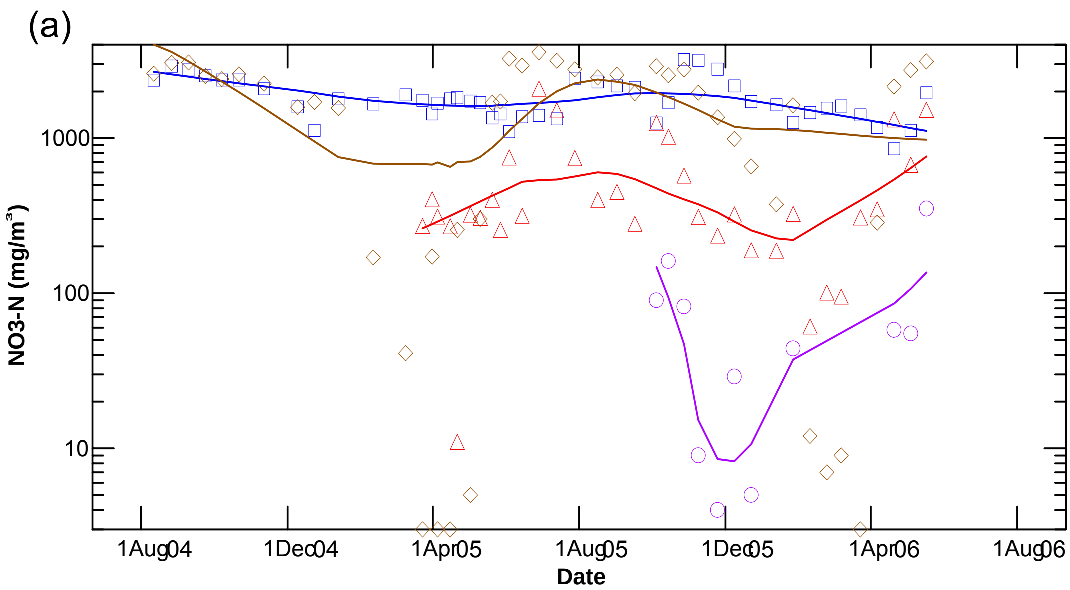


Nitrate-N concentrations at the four sites come from different distributions (Figure 17, K-S, $p<0.005$) with concentrations decreasing in the order B>S>G>R. Median nitrate-N concentrations were similar at the Bush (1705 mg/m³) and Swamp sites (1715 mg/m³), but the Swamp site had a larger range of concentrations (Figure 17). The median concentration at the Gorse site was significantly lower (321 mg/m³, IQR 403). Nitrate-N concentrations at Redoubt Rd were low, with a median of 55 mg/m³.

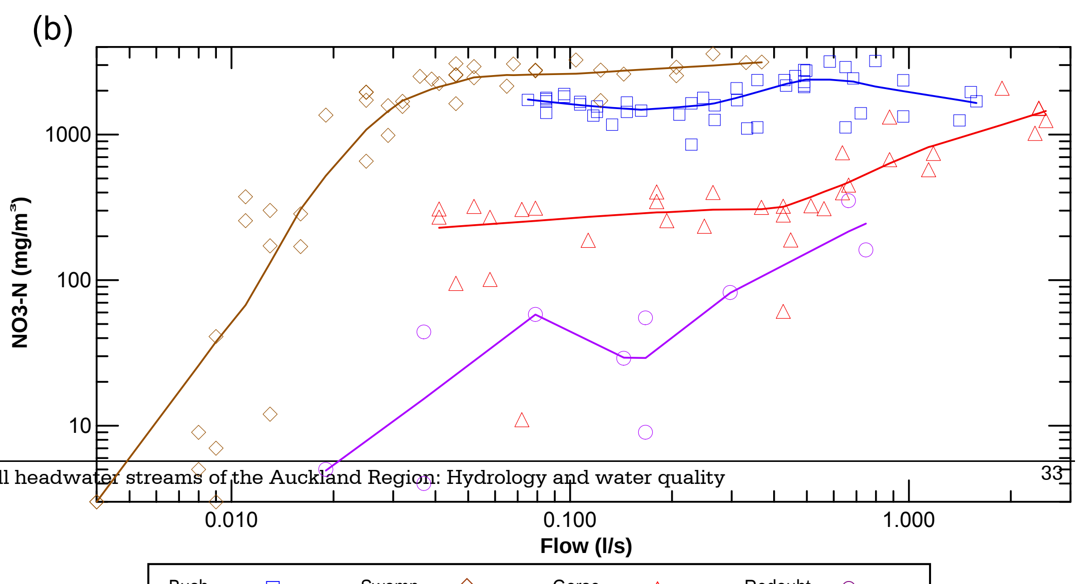
Time series plots of nitrate-N reveal some interesting patterns (Figure 20). At the Swamp site nitrate-N concentrations decreased between November and March each year.

Figure 20:

Nitrate-N fortnightly sample (a) time series and (b) relationship with flow, with LOWESS



smooth



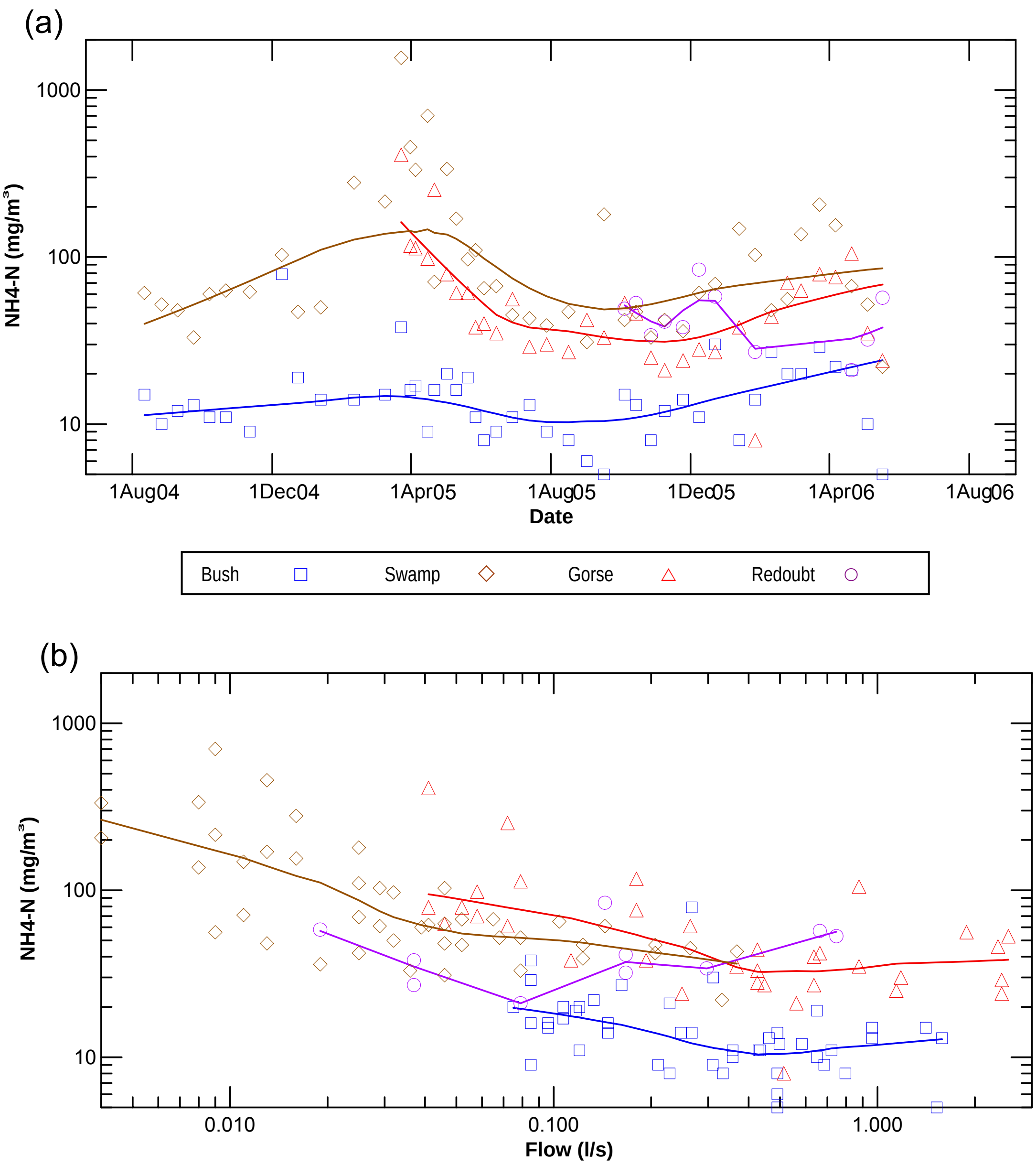
Nitrate-N concentrations at the Swamp site increase with increasing flow until a flow of around 0.025 l/s is reached (Figure 20b). Above 0.025 l/s the nitrate concentration does not change with flow. At the Bush site nitrate-N concentration does not change with flow (Figure 20b). At both the Gorse and Redoubt Rd sites concentrations increase with increasing flow (Figure 20b).

Ammonium-N concentrations were typically a small fraction of TN, with concentrations an order of magnitude smaller than nitrate-N. The Bush site has a different distribution from the other three sites (Figure 17, K-S, $p=0.000$). It has a median NH_4-N concentration of 13.5 mg/m^3, while the other three other sites all have median concentrations > 40 mg/m^3.

Data for the Gorse and Swamp sites show a seasonal trend in NH_4-N concentration, with the peak concentration in March/April and minimum concentration in August/September (Figure 21a). A weak seasonal pattern is evident in the Bush NH_4-N concentration data. Ammonium-N concentrations decrease with increasing flow in both the Gorse and Swamp catchments, but remain low and constant with flow at the Bush site (Figure 21b).

Figure 21:

Ammonium-N fortnightly sample (a) time series and (b) relationship with flow, with LOWESS smooth.

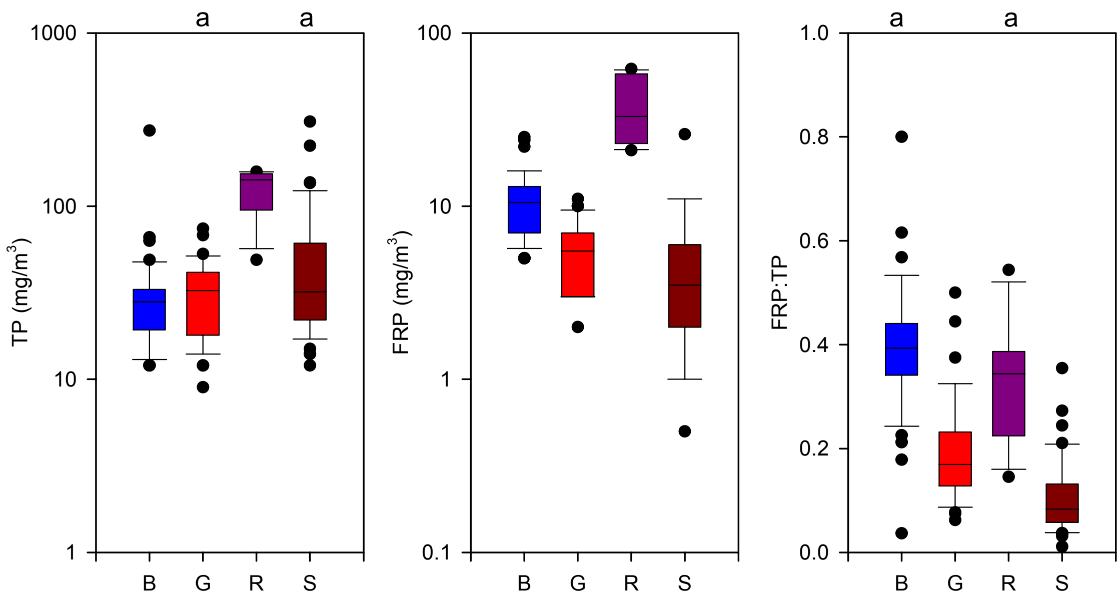


4.4.1.4 Phosphorus

Total P concentrations ranged between 9 and 307 mg/m³. The median concentrations for the Bush, Gorse and Swamp sites are significantly lower, around 30 mg/m³, than that recorded at the Redoubt site (median 142 mg/m³, Table 9). Concentrations come from three different distributions, with concentration increasing in the order B<G=S<<R (Figure 22, K-S, $p<0.05$).

Figure 22:

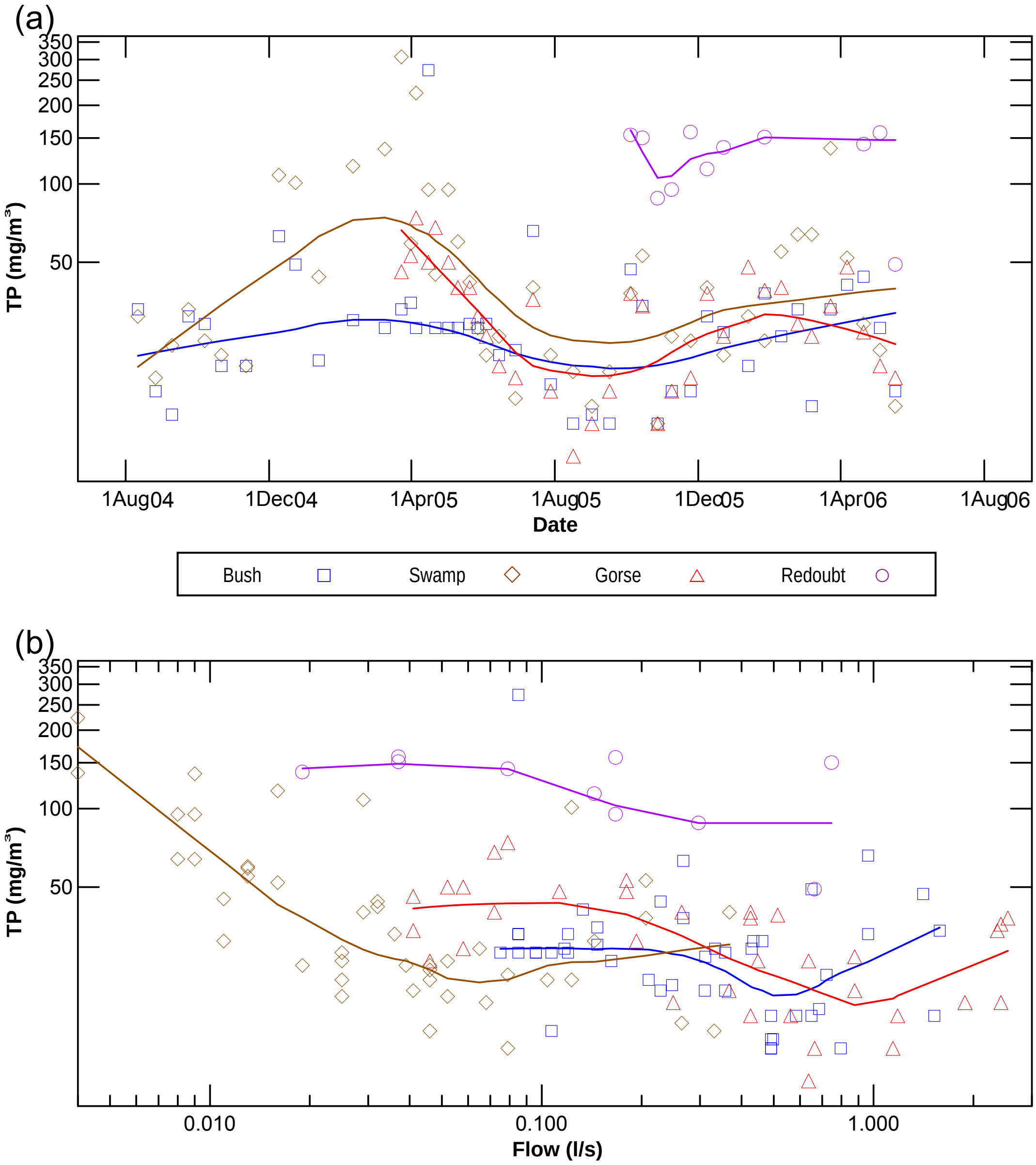
Boxplots of fortnightly P concentration data. Box represents the median with 25th and 75th percentiles, whiskers are the 10th and 90th percentiles and outliers are dots. Sites identified by the same letter are not significantly different, K-S, $p < 0.05$.



The Gorse and Swamp data shows a seasonal trend in TP concentration, with the peak in March/April and minimum in August/September (Figure 23a). A weaker seasonal pattern is evident in the Bush TP concentration data. The Redoubt Rd site TP concentrations are noticeably higher on every sampling occasion (Figure 23a). Total P concentrations decrease with increasing flow in both the Gorse and Swamp catchments, but remain constant with flow at the Bush site (Figure 23b).

Figure 23:

Total P fortnightly sample (a) time series and (b) relationship with flow, with LOWESS smooth.



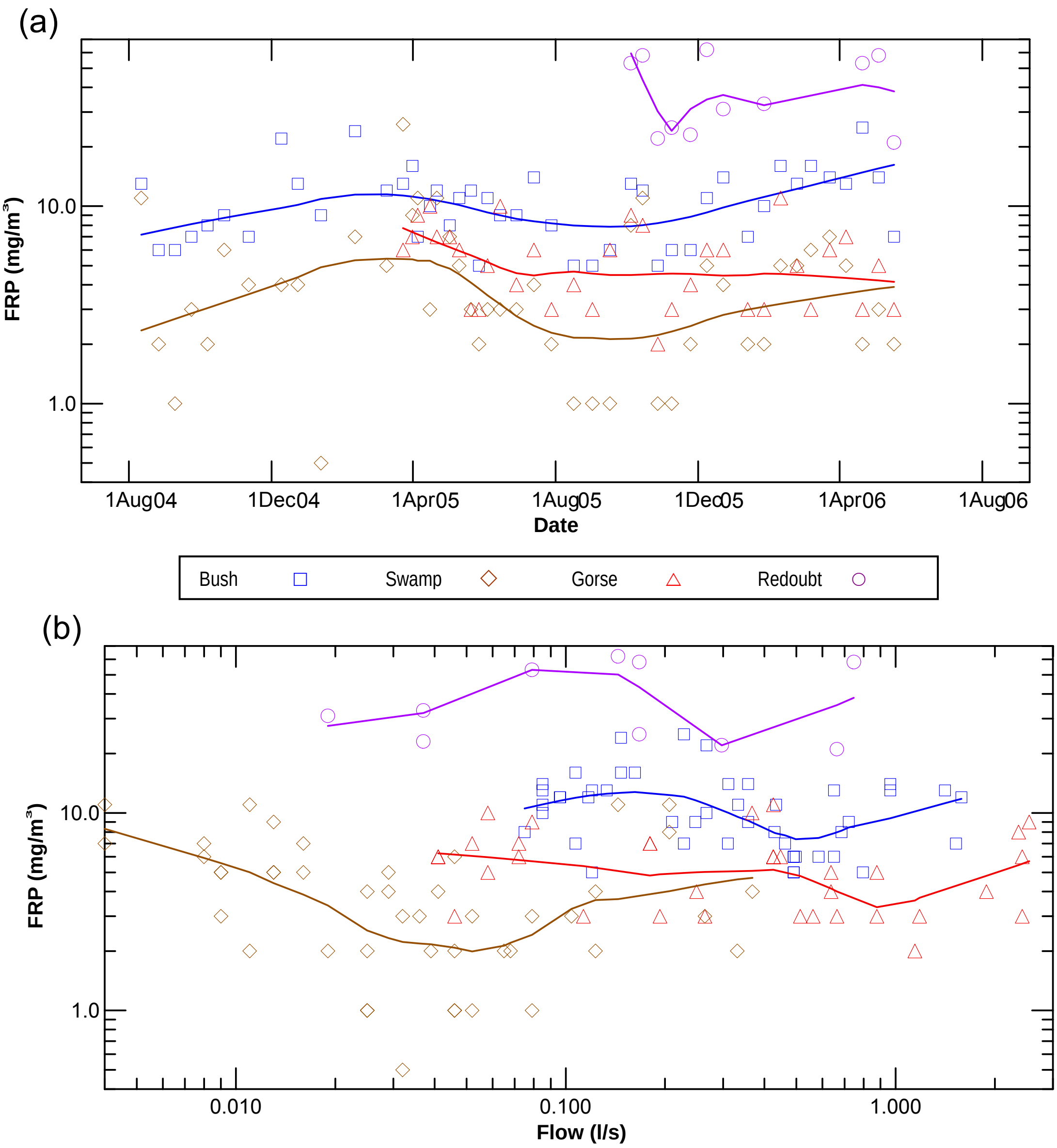
Total P is dominated at all sites by particulate forms of P. Filterable reactive P makes up about 0.2-0.4 of TP. At the Bush and Redoubt Rd sites the median FRP:TP ratio is around 0.3 (Figure 22, KS, p=0.152), while for the Swamp the median FRP:TP ratio is 0.08. The sites can be ranked in the following order of decreasing FRP:TP ratio, B=R>G>S.

FRP concentrations are high at Redoubt Rd, with a median of 33 mg/m^3 while the medians for the other sites are all < 11 mg/m^3. Only the Swamp and Gorse sites have similar distributions and so there are three groupings for the FRP data (R>>B>G=S).

A seasonal pattern is evident in the time series plots for both the Bush and Swamp sites, with minimum concentrations generally occurring in August/September and maximums in December/January (Figure 24a).

The relationship between FRP and flow is complex for the Swamp site, with concentrations decreasing with flow to 0.01 l/s and then levelling out (Figure 24b). No linear relationship between flow and FRP is evident for the Bush or Gorse sites (Figure 24b).

Figure 24:
Filterable reactive P fortnightly sample (a) time series and (b) relationship with flow, with LOWESS smooth.



4.4.2 Storm events

Six storm events were monitored, two each at the Bush, Swamp and Gorse sites and the storm event characteristics are summarised in Table 10. On 16 May 2005 the storm was sampled at both the Swamp and Gorse sites, while on 6 July 2005 the storm was sampled at the Bush and Gorse sites.

Table 10:

Monitored storm event characteristics.

	Bush		Swamp		Gorse	
Date	6Jul05	28Jul05	15May05	2Aug05	15May05	6Jul05
Storm number	2	3	1	4	1	2
Number of samples	22	21	22	7	24	14
Comment			last sample at first peak	small event	last sample at peak flow	No SS, TP, TN
Total rain (mm)	17.6	4.5	14.9	4.4	14.9	17.6
Peak flow (l/s)	2.2	1.1	0.2	0.2	1.6	4.0
Flow (m^3)	37	36	5.6	20.5	32	139

Storm 1 had a total rainfall of 14.9 mm and a duration of 7.5 hours (Table 10). The preceding day had 9.9 mm of rainfall, but the 11 days preceding that had minimal rainfall (4 mm/day on two days). Total runoff volumes at the Swamp and Gorse sites were 5.6 and 32 m^3, respectively. Samples were collected on the first peak at both sites, with only rising limb samples at the Swamp site (Figure 26), and rising and falling limb samples at the Gorse site (Figure 25). At both sites the peak sampled concentrations typically preceded the peak discharge by a short period and concentrations increased with increasing flow. *E. coli* concentrations were an order of magnitude higher at the Swamp site than the Gorse site (Figure 25, Figure 26). Suspended sediment concentrations were 5 times higher at the Gorse site and TP concentrations were also higher at the Gorse site.

Storm 2 (6 July 2005) was sampled at the Bush and Gorse sites (Table 10). It was preceded by 10 days of low rainfall (<3 mm/day). Peak concentrations generally coincided with the peak flows with the exception of nitrate-N at the Bush site (Figure 27, Figure 28). At the Bush site the peak nitrate-N concentration of 1580 mg/m^3 occurred early on the rising limb and then concentrations decreased with increasing flow.

Only the Bush site was sampled during storm 3 (28 July 2005, Table 10). Both peaks were sampled during this event and the total runoff volume was 36 m^3. Concentrations were generally higher on the first peak, with the exception of *E. coli,* which peaked on the second flow peak (Figure 29). Phosphorus was dominated by particulate P and nitrate dominated N transport. The nitrate-N concentration decreased on the falling limb of the first peak, suggesting that dilution of the source by rainfall inputs occurred (Figure 29).

Storm 4 (2 August) was sampled at the Swamp site (Table 10). Rainfall > 6 mm was recorded on the preceding two days. The peak flow for this event was low, but due to an attenuated hydrograph the flow volume was 20 m³. Only *E. coli* and dissolved nutrients were analysed for this event (see Figure 31 in Appendix 2).

Figure 25:
Flow, rainfall and water quality time series and hysteresis plots for the Gorse storm on 15-16 May 2005.

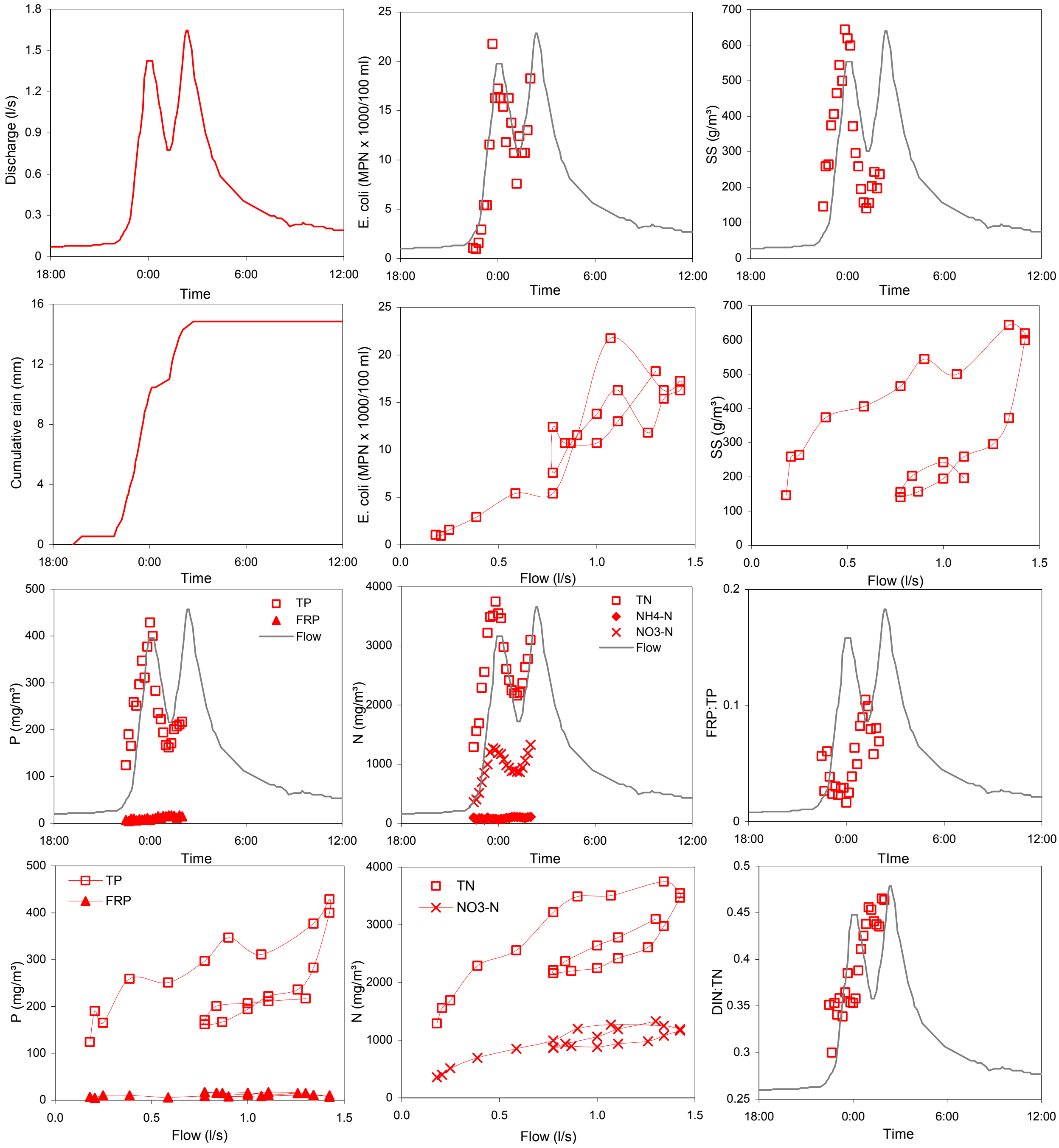


Figure 26:

Flow, rainfall and water quality time series and hysteresis plots for the Swamp storm on 15-16 May 2005.

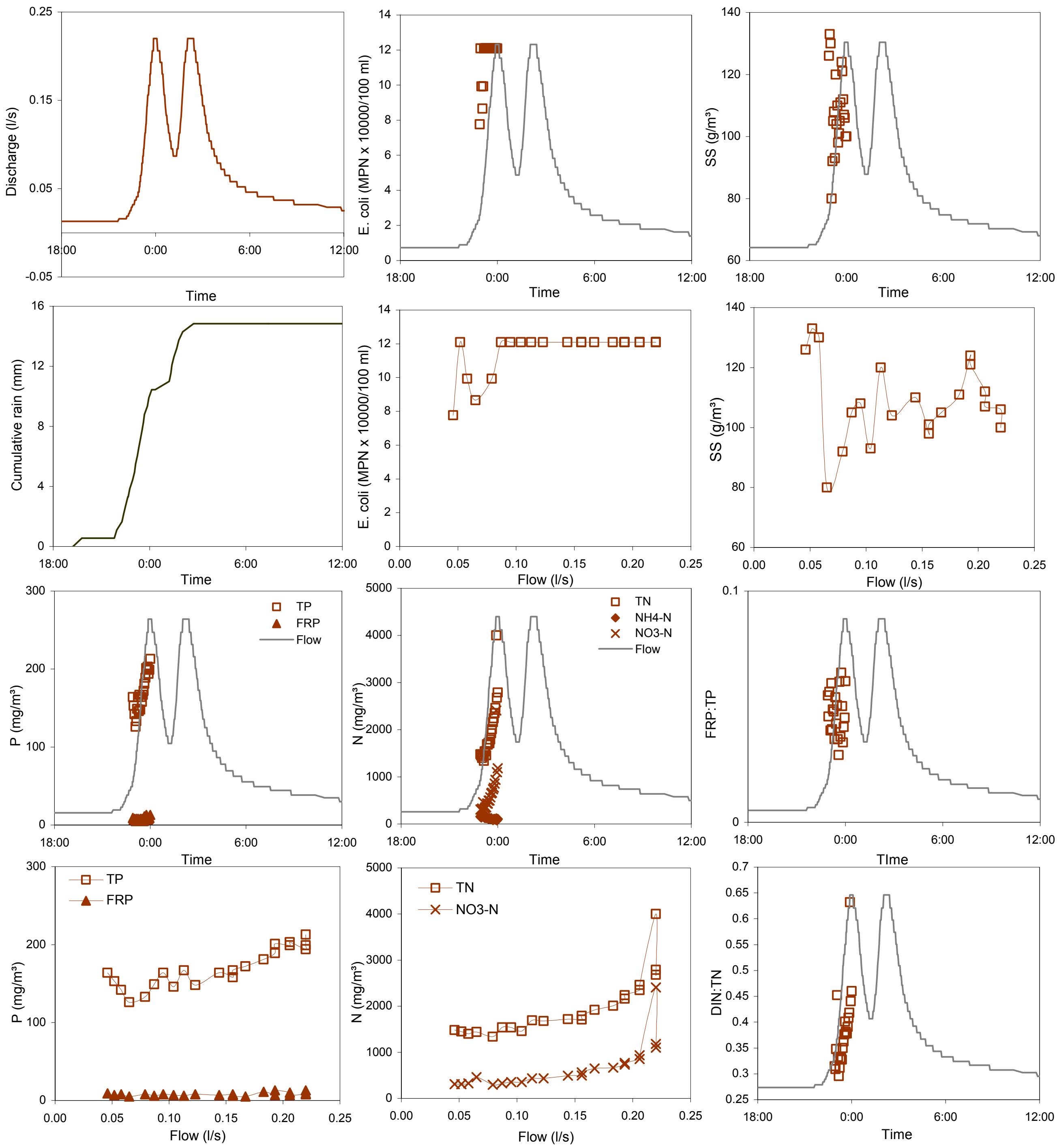


Figure 27:

Flow, rainfall and water quality time series and hysteresis plots for the Gorse storm on 6 July 2005.

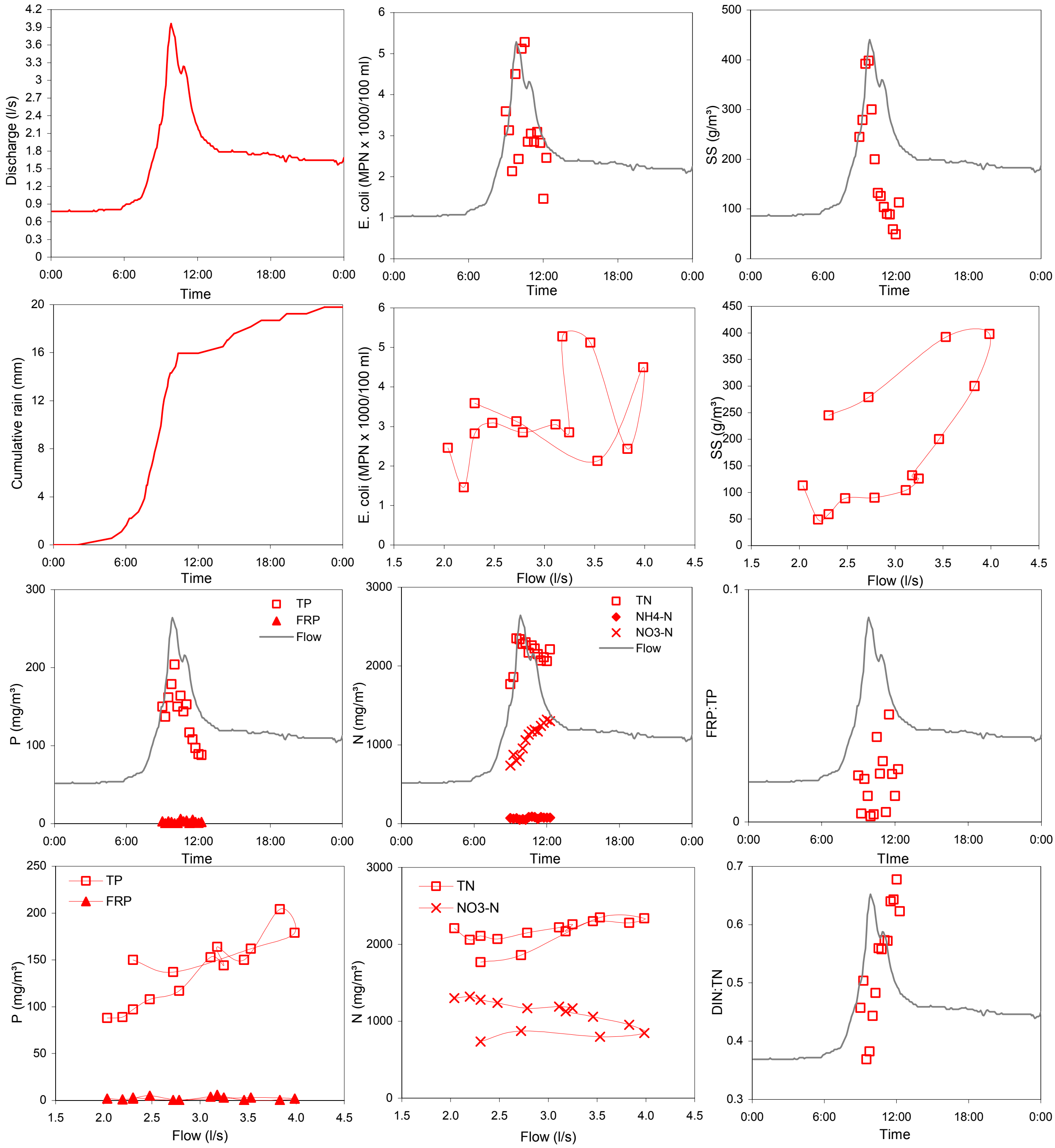


Figure 28:

Flow, rainfall and water quality time series and hysteresis plots for the Bush storm on 6 July 2005.

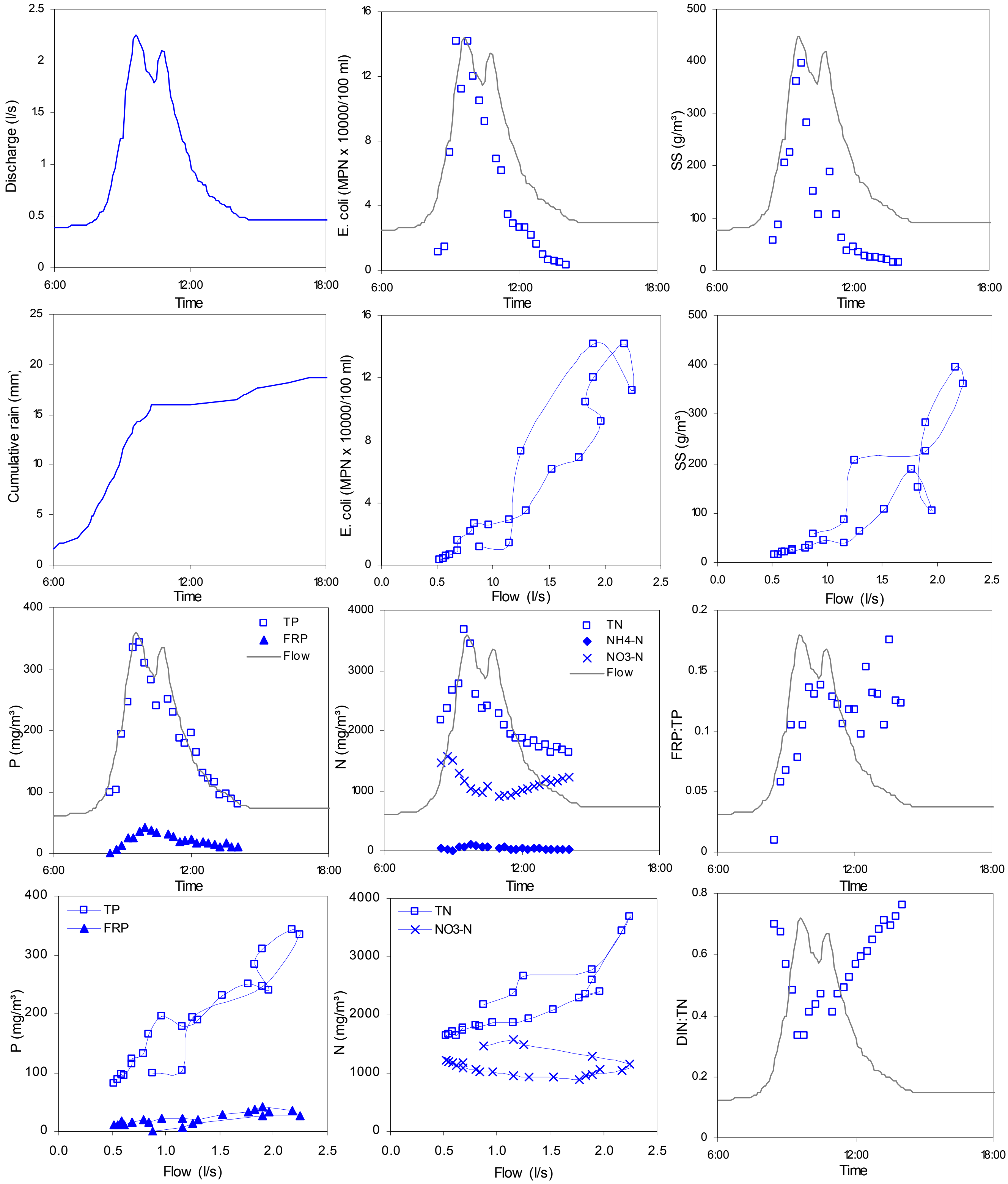
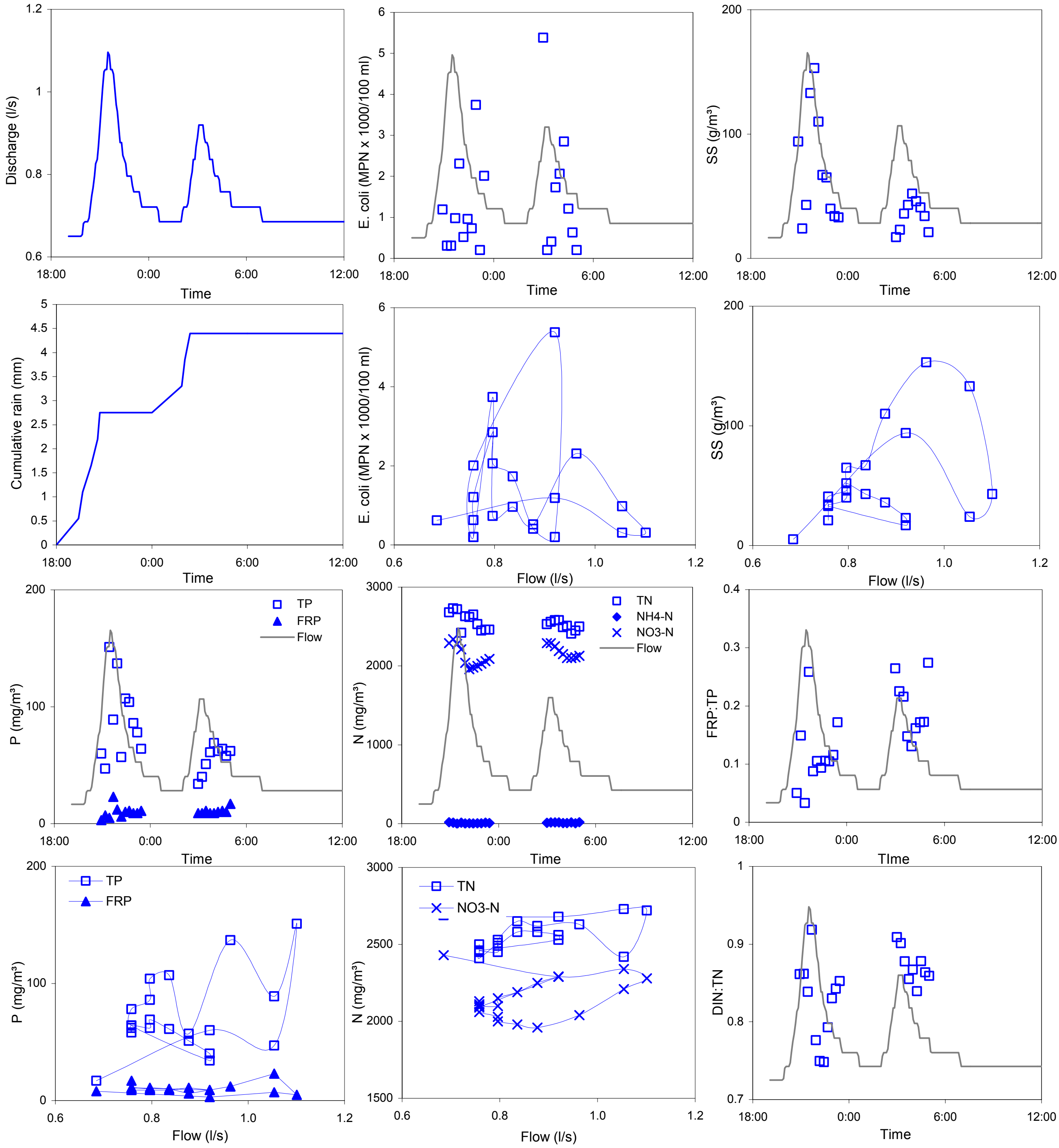


Figure 29:

Flow, rainfall and water quality time series and hysteresis plots for the Bush storm on 28-29 July 2005.



4.4.3 Loads

Load estimates calculated by the three methods are generally in agreement (Table 11). The greatest differences are for TN and NO_3-N exports, with the LOADEST estimates being considerably higher than the other two methods. Caution must be applied to these LOADEST exports as the estimated concentrations exceed the data concentration range and/or are LAD model estimates without residual analysis or coefficient of efficiency estimates. The *E. coli* LOADEST estimates have low coefficients of efficiency (<0.1) and therefore may not be reliable estimates. The Bush SS load is the only other load with a coefficient of efficiency (E) less than 0.3, the remaining estimates all have E > 0.5 (see Table 15 in Appendix 1) and so there is a good fit between the observed and estimated concentrations.

For comparative purposes it is necessary to examine exports per unit area (Table 12). Suspended sediment exports are similar for all sites, with a range of loads predicted by the different analysis methods. SS exports are generally in the order of tens rather than hundreds of kg/ha/y. *E.coli* exports are similar at the Redoubt Rd and Swamp sites (>2.3×10^{11}), and higher than exports from the Bush and Gorse sites (≈4×10^{10}).

Total N exports were lowest at the Redoubt Rd site, around 4.5 kg/ha/yr. Exports from the Bush sites were similar to those from the Swamp, in excess of 8 kg/ha/y. Nitrate-N is a small part of the Redoubt Rd export (≈0.25 kg/ha/y) and significantly lower than NO_3-N export from the other sites which are between 3 and 4 kg/ha/y at Gorse and > 6.5 kg/ha/y at the Bush and Swamp sites. Ammonium exports were lowest at the Bush site (<0.1 kg/ha/y) and highest at the Swamp site (up to 0.65 kg/ha/y).

Total P and FRP exports at Redoubt Rd (≈0.5 kg TP/ha/y and 0.1 kg FRP/ha/y) are significantly higher than those estimated for the other three sites (<0.25 kg TP/ha/y and <0.06 kg FRP/ha/y).

Table 11:

Daily load estimates for the four sites.

Site	Parameter	Units	Interpolation	Averaging estimator	LOADEST
Bush	SS	g/d	389	1024	2634
	EC	number/d	$4.7x10^8$	$5.2x10^8$	$7.1\ x10^{8\#}$
	TN	g/d	94	141	512‡
	NO_3-N	g/d	77	126	539*‡
	NH_4-N	g/d	0.57	1.10	0.69
	TP	g/d	1.44	2.29	2.71
	FRP	g/d	0.41	0.75	0.82
Swamp	SS	g/d	111	191	184
	EC	number/d	$4.7x10^8$	$4.4x10^8$	-
	TN	g/d	18	19	-
	NO_3-N	g/d	14	14	-
	NH_4-N	g/d	0.04	1.26	0.45‡
	TP	g/d	0.23	0.47	0.49‡
	FRP	g/d	0.03	0.04	-
Gorse	SS	g/d	1304	927	1410
	EC	number/d	$8.4x10^8$	$2.0x10^9$	$9.2x10^{8\#}$
	TN	g/d	102	85	284*‡
	NO_3-N	g/d	59	45	305*‡
	NH_4-N	g/d	2.81	5.79	4.61
	TP	g/d	2.25	2.82	2.54
	FRP	g/d	0.35	0.47	0.4
Redoubt	SS	g/d	331	680	-
	EC	number/d	$1.5x10^9$	$2.9x10^9$	-
	TN	g/d	24	27.5	-
	NO_3-N	g/d	1.20	2.0	-
	NH_4-N	g/d	0.53	1.10	-
	TP	g/d	2.55	3.11	-
	FRP	g/d	0.55	0.98	-

coefficient of efficiency < 0.1

‡ estimation concentration exceeds calibration concentration

* LAD model without residual analysis

Table 12:

Specific exports calculated using interpolation (I), averaging estimator (AE) and LOADEST (L) methods.

	Method	Bush	Swamp	Gorse	Redoubt
SS	I	33	58	85	58
(kg/ha/y)	AE	87	99	61	120
	L	173	96	92	-
E. coli	I	3.95×10^{10}	2.42×10^{11}	5.47×10^{10}	2.72×10^{11}
(#/ha/y)	AE	4.42×10^{10}	2.30×10^{11}	1.28×10^{11}	5.14×10^{11}
	L	$4.59 \times 10^{10\#}$	-	$6.00 \times 10^{10\#}$	-
TN	I	7.93	9.39	6.67	4.17
(kg/ha/y)	AE	11.98	9.81	5.56	4.84
	L	$33.58^{‡}$	-	$18.62^{*‡}$	-
NO_3-N	I	6.50	7.32	3.86	0.21
(kg/ha/y)	AE	10.70	7.13	2.97	0.35
	L	$35.35^{*‡}$	-	$20.0^{*‡}$	-
NH_4-N	I	0.05	0.02	0.18	0.09
(kg/ha/y)	AE	0.09	0.65	0.38	0.19
	L	0.05	$0.23^{‡}$	0.3	-
TP	I	0.12	0.12	0.15	0.45
(kg/ha/y)	AE	0.19	0.24	0.18	0.55
	L	0.18	$0.25^{‡}$	0.17	-
FRP	I	0.03	0.02	0.02	0.10
(kg/ha/y)	AE	0.06	0.02	0.03	0.17
	L	0.05	-	0.03	-

$^{\#}$ coefficient of efficiency < 0.1

‡ estimation concentration exceeds calibration concentration

* LAD model without residual analysis

5 Discussion

This study examines the hydrology and water quality of four headwater streams in the Waitemata HGA grazed by drystock. The streams drain adjacent catchments, a design which provides data on the influence of riparian protection, from full stock access to wide riparian buffers, on stream water quality.

None of the headwater streams monitored were part of the blue-line stream network on NZMS260. They are all small tributaries of the blue-line stream (Figure 1). The stream network at Totara Park is well defined and cut into the steep slopes.

The Bush and Gorse sites are both perennial headwater streams, with year round flows. In contrast, the Swamp and Redoubt Rd sites were intermittent, particularly Redoubt Rd. The channels are well defined at the Bush and Gorse sites, with unvegetated widths in excess of 1 m. At Redoubt Rd the channel is narrow, around 0.2 m wide and vegetated with grass. The Swamp site is a seepage wetland (Johnson and Gerbeaux, 2004) as it has a moderate groundwater flow and surface water inputs. Downslope of the Swamp site, the flow goes underground. So despite flowing continuously there is no defined surface channel downslope of the seepage wetland. The stream network is comprised of a swale disrupted by potholes, suggesting subsurface flowpaths also occur (Figure 30).

Figure 30:
Drainage network downslope of the Swamp site.

Median suspended sediment concentrations were high at the Redoubt Rd, Swamp and Gorse catchments compared to longer term water quality records in larger Auckland streams (ARC, 2000). The Bush site by comparison had a low median concentration for fortnightly samples. However, during the two storms monitored at the Bush site, concentrations up to 400 g/m^3 were measured (Figure 27, Figure 28). Similar concentrations were measured at both the Bush and Gorse sites during the 6 July storm, illustrating the importance of including storm event monitoring for SS.

Suspended sediment exports are low by comparison with those measured in headwater streams at Whatawhata (Table 13), but higher than those measured at Purukohukohu. These catchments are larger than the headwater streams monitored in this study, but there has been very little water quality monitoring of pasture catchments less than 6 ha previously in New Zealand (e.g., McColl, 1978).

Table 13:

Small catchment exports (kg/ha/y) of sediment and nutrients from catchments at Whatawhata (Quinn and Stroud, 2002), Purukohukohu (Dons, 1987) and Scotsman's Valley (Cooke, 1988, Cooke and Cooper, 1988).

Site and land use	Area (ha)	SS	TN	NO_3-N	NH_4-N	TP	FRP
Whatawhata P3 (1995-96)	259	3212	23.2	9.4	10.4	3.24	0.31
Whatawhata P3 (1996-97)	259	988	10	4.37	0.34	1.5	0.25
Whatawhata Mixed M2	266	2632	6.8	2.55	0.26	1.33	0.26
Whatawhata Native N2	300	320	20.7	0.55	0.12	0.58	0.27
Purukohukohu pasture	22.5	22	11.9	1.19	0.48	1.67	0.37
Purukohukohu native	37.2	27	3.67	2.84	0.056	0.12	0.017
Scotsman's Valley	16		7.2	0.94		1.30	

Sediment sources were visible at all four sites. Fine organic material was visible at both the Swamp and Redoubt Rd sites, particularly when stock had recently been in the paddocks. Organic debris is a possible sediment source at the Bush site. Tree riparian buffers can be sediment source areas (e.g., Smith, 1992) due to low ground cover. Leaves and organic debris clogged the Bush weir until a leaf guard was added to prevent interference with water level measurements. Stream bank erosion is probably the dominant sediment source at the Gorse site, with channel changes observed upstream of the monitoring sites, although much of the material in the bed is coarse aggregates rather than fine suspended sediment. Channel adjustments following fencing and increased shade with tree growth are predicted to release material stored in grassy streambank vegetation over a long period (Parkyn et al. 2005). The banks will eventually stabilise to a new steady-state forest stream morphology.

All four sites had median *E. coli* concentrations greater than 126 MPN/100 ml, the recreation freshwater guideline value (Department of Health, 1992). New Zealand guidelines for stock drinking water recommend a median value of less than 100 faecal coliforms or E.coli/100ml (NWQMS 2000). The Bush site had the lowest *E. coli* concentrations and exports and this is consistent with the high level of riparian protection and the Redoubt Rd site was particularly contaminated with values in excess of 2300 MPN/100ml. Although the Swamp and Gorse sites had similar median concentrations and distributions for the fortnightly sampling, during storm 1, *E. coli* concentrations were several orders of magnitude higher at the Swamp site than the Gorse site. This suggests that stock access, combined with high flows in the wetland can flush high numbers of *E. coli* from seepage wetlands. Collins (2004) measured higher concentrations from wetlands at Whatawhata during events and concluded that excluding stock from shallow wetlands could yield improvements in bacterial water quality.

ANZECC/ARMCANZ (2000) default trigger levels for nitrogen in slightly disturbed lowland river ecosystems are 614 mg TN/m^3, 444 mg NO_x-N/m^3 and 21 mg NH_4-N/m^3. Total N and nitrate-N concentrations were consistently above these values, with the exception of nitrate-N at Redoubt Rd. Nitrate-N concentrations in the Bush and Swamp catchments were high compared to larger Auckland streams (ARC, 2000), while Redoubt Rd concentrations were low. Nitrogen exports are similar to those measured in all catchments at Whatawhata and Purukohukohu (Table 13).

Surprisingly the Bush and Swamp sites have similar TN concentrations and exports, suggesting that they have the same source. The large difference in nitrate-N concentrations and exports between the western and eastern sides of the catchment suggest that there was at some time an additional N source on the western slopes. The groundwater currently being discharged is unlikely to be recent recharge and so different historical land uses may be the cause for the difference.

Seasonal patterns in nitrate-N concentrations were evident at the Swamp and Gorse sites. Several processes may contribute to seasonal fluctuations in nitrate concentrations, including winter leaching and wetland removal. Nitrate accumulates in topsoil during dry periods from mineralisation of soil organic matter and stock inputs (dung and urine), but there is typically little leaching due to limited water for transport. As soil moisture increases in the winter, the accumulated nitrate can be transported and flushed from the soil.

There is evidence for nitrate-N retention at the Swamp site. During the winter months the Bush and Swamp sites have similar TN concentrations, DIN:TN ratios and NO_3-N concentrations. However, during the summer, nitrate-N is still high at the Bush site, but is low ($<$10 mg/m^3) at the Swamp. This suggests that under wetter conditions the seepage wetland was less effective in reducing nitrate concentrations than during drier periods. This is consistent with the experimental findings at Whatawhata where high nitrate-N removal from throughflow was measured over short distances in during dry periods, with little removal from surface runoff during rain events when there is limited soil contact time (Burns and Nguyen, 2002; Rutherford and Nguyen, 2004).

Ammonium concentrations are low at all four sites, but particularly the Bush site. Ammonium concentrations increased during the summer months at the Swamp site, Nguyen et al. (2000) found that 62 m^2 seepage wetland monitored at Whatawhata was a source of NH_4-N. This was attributed to the low potential for nitrification from organic N mineralisation and also dissimilatory reduction of nitrate-N to NH_4-N.

Median P concentrations are low by comparison with larger streams in the Auckland Region (ARC, 2000) and below the ANZECC/ARMCANZ (2000) default trigger values for slightly disturbed ecosystems (33 mg TP/m^3 and 10 mg FRP/m^3), with the exception of Redoubt Rd. At Redoubt Rd TP and FRP concentrations were greater than the default trigger value on all sampling occasions, with minimum concentrations of 49 mg TP/m^3 and 21 mg FRP/m^3. The catchments are all fertilised with superphosphate each autumn and it is possible that direct deposition of fertiliser in the intermittent channel has raised the P concentrations and exports from this catchment.

Phosphorus loads are low by comparison with other small New Zealand pasture streams (Table 13). The exports are similar to those measured in the native forest catchments at Whatawhata and Purukohukohu (Table 13), despite the use of superphosphate at Totara Park.

It is important to note that headwater streams are prone to contamination because of small dilution capacity and limited opportunities for attenuation between the source and monitoring site. McColl (1978) noted that the mean concentration of most elements were highest in the nested Pukeiti stream (1.44 ha) than further downstream at the Puketitoi weir where mean concentrations were lower and less variable. Once flow has channelised there are limited opportunities for improving water quality, unless there is significant in-stream attenuation.

Section 2 of the Resource Management Act (1991) defines a river as being permanently or intermittently flowing and so all sites could be classified as rivers under this definition. The current definition of Category 1 streams in the Proposed Auckland Regional Plan: Air, Land, Water[2] has continual flow or stable natural pools and certain aquatic biota (see Appendix 3) while Category 2 streams are dry for part of the year. Under this definition the Bush and Gorse sites would be Category 1, and Redoubt Rd Category 2.

This study shows that riparian protection for Category 1 streams is valuable for reducing *E. coli* loads and concentrations by excluding stock. The riparian protection at the Bush and Gorse sites appears to have also reduced direct fertiliser application to the channel, compared with Redoubt Rd. Extending fertiliser guidelines to exclude application to Category 2 streams could remove this additional P source. At the time that fertiliser is applied (typically autumn) these streams may not be flowing, but identification of Category 2 streams as winter transport pathways on farms could reduce P concentrations and exports.

The potential of riparian areas to mitigate pollutants transported by subsurface flows, such as nitrate, will be intimately linked to the hydrogeologic setting. The presence of a confining layer, high permeability gravel layers or sediments with low hydraulic conductivities will determine flowpaths, water residence times and removal rates. At a landscape scale, indicators could be developed to help identify riparian saturation and potential for nitrate removal. For example, Vidon and Hill (2004) identified riparian hydrologic types by relating depth of permeable sediment and topography to predict levels of riparian saturation.

Seepage wetlands fall outside the definitions of category 1 and 2 streams, despite being part of the flow network. This direct linkage to the stream network means that

[2] These provisions are subject to a number of appeals and may change

they can provide rapid transport pathways for pollutants during storm events, similar to well defined channels. For example, similar storm event concentrations were measured at the Swamp and Gorse sites during storm 1. Recognition of these wetlands, on a par with Category 1 and 2 streams, is required. However, such small wetlands can be controversial. Tools and methodologies are currently being developed to assist with their identification, functional analysis and management negotiations (e.g., Merot et al. 2006).

Channel flow is a mixture of different source waters and so efforts to improve water quality of headwater streams must take into account pollutant sources, pathways that different pollutants can take and transformations that may occur during transport. Headwater streams are good candidates as priority targets for moderating the entry of surface derived nutrients, sediment and faecal matter to higher order streams. However, the results of this study support the case for such targeting with respect to faecal contamination but provide less convincing support for nutrients and sediment.

6 Conclusions and Recommendations

This report is part of four volumes of research on the values of headwater streams and overall conclusions and recommendations are summarized below.

6.1 Implications for Management

6.1.1 Values of headwater streams

Collier (1993), in his review of the conservation of freshwater invertebrates, advocated a habitat- rather than species-based approach to conserving biodiversity. The protection of a range of rare, endangered, or representative habitats is most likely to ensure the protection of a wide range of invertebrate species, as well as maintain natural ecosystem processes.

Our research on the natural values of headwater streams has shown that there are significant biodiversity values associated with headwater habitats that dry up or contract in length for part of the year and are often not mapped as blue lines on topographic maps (Parkyn et al. 2006). For all land uses assessed, additional taxa occurred in the mud, pools, and flowing habitats that were not found in the perennial streams sampled. Therefore, protection of these habitats would enhance the overall biodiversity of stream communities.

However, our research also showed that despite the presence of additional taxa, the overall community composition and structure, and invertebrate metrics of ecosystem health were not significantly different between perennial stream habitats and the smaller headwater habitats. Mud samples were the most different from perennial samples as might be expected, but surprisingly, mud also contained communities of freshwater invertebrates. It seems likely that mud can act as a short-term refugium for some species, but other species may have adapted to exploit this habitat more permanently.

Based on the invertebrate species composition, there does not seem to be a rationale to separate Category 1 and 2 streams. However, it seems reasonable to suggest that stream reaches that are completely dry would have less value than streams with moisture, at a given point in time. In order to rank the differences between streams that all have a dry phase we would need to know the proportion of time that streams are wet and able to support aquatic life. Hydrological studies in one area of Auckland (Totara Park, Waitemata sandstones) indicated that 2 of the 4 streams ceased flowing for part of the year at the point where the weirs were placed (McKergow et al. 2006). In the smallest pasture catchment (0.7 ha) the stream stopped flowing for only 10 days in summer, while in a larger pastoral catchment (2.1 ha) stream flow dried up to occur only as storm flow between January and mid-April. Because of the influence of groundwater on these headwater areas, it is difficult to predict flows based on catchment areas. Currently we have estimates of the stream length per hectare that is intermittently flowing (or changing in length) from the Spatial Extent survey (Wilding & Parkyn 2006), but little understanding of how flow varies over time for these headwater systems.

The main differences in natural values occurred between land uses. Clearly, riparian vegetation improved the conditions of the streams towards that of native forest and allowed the existence of aquatic species associated with native forest streams. This suggests that riparian planting is a valuable method for managing headwater streams and it also shows that headwater streams with existing vegetation could be valuable sources of recolonists for stream restoration. Small, vegetated gullies are often pockets of refugia for native forest stream species within a pastoral catchment. Protecting these areas could be particularly valuable as source areas for restoration downstream and could mean that successful restoration is achieved after several years rather than several decades. For instance, if the headwater streams in a catchment were piped and filled (e.g., during urban development of a pastoral area), and only the perennial streams were restored with riparian planting, then it would take much longer for the recolonisation of stream communities to occur as there would be no upstream source of recolonists. Retaining headwater streams that already have riparian vegetation would improve the speed and success of the restoration process.

6.1.2 Recommendations for current management

Small headwater streams and wetlands are extensive in the Auckland region compared to the length of higher-order streams. Management of these areas is complex and decisions on the protection of these areas may ultimately depend upon socio-economic factors as well as ecological factors. An important question that remains unanswered is that of the cumulative effect of widespread loss or deterioration of headwater stream habitat. However, our research does provide information to help with management of rural and urban headwater streams.

Rural

There are several ways that headwater streams and wetlands could be managed under dry stock agriculture. One way is to fence all small waterways and plant them with native riparian plants, as was the case in the PR streams that we studied. This clearly has biodiversity benefits, particularly in summer, when even the remaining moist mud habitat was able to support EPT taxa. Communities of invertebrates in pastoral streams have changed from that of the original forested condition, but significant improvements in habitat and biodiversity of pastoral streams could be gained by fencing and planting riparian buffers. When there is adequate shade from riparian vegetation, water temperatures are lower and dissolved oxygen levels are higher during the summer months, creating healthier conditions for the invertebrate communities. Shade and cover from planted buffers also provides habitat for fish and koura (no koura were found in non-perennial pastoral headwater streams, but they were common in native forest).

The other important function of riparian buffers is for water quality in most stream systems. Fencing stock out of streams at Totara Park produced lower annual loads of *E.coli* than in streams open to stock (McKergow et al. 2006). However, headwater stream flow is greatly influenced by groundwater and subsurface flow. This means that the water can be carrying leached pollutants from the surrounding land use or historical land uses that have bypassed the riparian zone. Nitrogen loads in the riparian protected stream at Totara park were similar in the protected (Bush) and open (Swamp) sites.

Significant processing of nitrate and phosphorus (>90%) can occur in headwater wetlands under base flow conditions but this function can be reduced by stock access (Sukias & Nagels 2006). Hoof prints can create holes in wetlands that allow subsurface water to flow up and over the surface of the wetland where negligible denitrification occurs. Stock can also eat vegetation that would have naturally added to the organic build up in the wetland and therefore, stock reduce the processing capacity. Where headwater wetlands occur, best practice would be to fence stock out and allow wetland vegetation to develop. Planting with taller tree species is not recommended if the goal is to reduce nitrogen loads, as wetlands will revert to streams once shaded. Storm flows contribute significant amounts of pollutants and reduce the functioning of the wetlands. Efforts to extend protection or rough vegetation (e.g., encourage long grasses above wetlands by electric fencing in winter) may help to slow flood flows and give time for settling and infiltration of contaminants from the water flow.

The consequences of not managing these areas by removing stock are a continued export of sediments and faecal bacteria that will contribute to pollution and, in the case of sediment, accumulation downstream. With no riparian buffers on headwater streams, direct fertilizer additions and open access to stock, exports of nitrogen and phosphorus will remain high.

If fencing and/or planting headwater streams is not feasible then an alternative could be to construct wetlands at the base of catchments before the streams enter significant waterbodies (e.g., lakes, estuaries). However, this option would provide no biodiversity protection for the headwaters and may impede fish passage. Another alternative could be strategic protection of some of the headwater stream network. Existing tools for predicting fish assemblages could assist with this process, at least for fish biodiversity (John Leathwick, NIWA. pers. comm.). We recommend further research in order to make predictions about the placement, or the percentage, of streams that should be protected.

Recommendations

Based on research carried out over the last three years, we strongly recommend fencing stock out of headwater streams and wetlands for water quality improvements. For wetlands, fencing could take the form of hotwire fences that could be removed for stock grazing if the wetland dried up in summer.

For biodiversity goals in headwater streams, we recommend riparian protection with planted buffers of native trees. While shaded buffers reduce the nutrient processing capacity of headwaters, they provide multiple ecological benefits.

It may not be necessary to protect every headwater tributary to achieve improved biodiversity and water quality. We recommend further research into catchment-based approaches to assess the cumulative impacts of not managing all pastoral headwater streams and potential methods to select important or representative reaches.

Urban

When catchments are converted to urban land use there is potential for severe loss of stream function through piping and infilling (Rowe et al. 2006, Wilding 1996). Effectively all habitat values are lost and functions such as natural attenuation of contaminants, connectivity for species dispersal, food webs etc., are impaired. Urbanisation of catchments can also mean a loss of groundwater recharge from the increased impervious area. Therefore, it is likely that streams in urbanized catchments dry up for longer periods of time in summer and/or over a greater length.

Our research has shown that temporary headwater streams have similar aquatic invertebrate communities to those in perennial streams, but can also provide habitats that add additional species to the overall biodiversity of the catchment. The consequences of losing these streams will be loss of habitat values and a decline in overall biodiversity. Furthermore, urbanization that increases the duration of the dry period may decrease the biodiversity values of these headwater streams.

While intercepting nitrogen and phosphorus in urban streams may not be as necessary as it would be in pasture, it is worth noting that groundwater flow to these streams may still be carrying nutrients from historical land use, and simply piping them would transport these nutrients downstream without any instream attenuation. In addition, headwater streams may be just as important for the processing of stormwater contaminants as for rural contaminants, and incorporating natural stream functioning into urban design could make these streams important resources for treating urban runoff.

Recommendations

Our recommendation is that headwater streams be protected with riparian planting when catchments are converted to urban land use, for the sake of instream habitat, biodiversity, and ecosystem functioning – i.e., contaminant processing.

We recommend further research into the cumulative effects of the loss of headwater streams and better spatial modeling of the impact of urban development on catchment biodiversity and stream functioning.

6.2 Recommendations for future research

From the state of the science currently, we have concluded that intermittently flowing headwater streams do have values similar to that of perennial streams and their management should therefore be similar. However, we recognize that it may not be feasible for all headwater streams to be protected. Thus, there are a number of additional research areas that could allow us to differentiate between streams of higher and lower ecological value or provide a process for sustaining ecological and economic values.

Cumulative effects

Currently, the ARC has to deal with applications to alter headwater streams and wetlands on a piecemeal basis. There are no tools available to assess the cumulative effects of changing land use, or piping and damming streams. How many waterways can be lost (to infilling, piping or damming) in a catchment before this has impacts on catchment functions such as downstream water quality and quantity, or habitat provision? Conversely, is there a proportion or spatial arrangement of streams in a catchment that could be restored to enhance habitat and biodiversity, and improve water quality but still be affordable for the region?

This will be a difficult question to answer but one that is very important to consider. The first step would be to ascertain whether it is possible to assess the cumulative effects of stream loss and to consider the wide-ranging implications from species protection and habitat provision through to downstream effects on water quality and quantity and ecosystem functioning.

Variation through time

Can the length of time that headwater streams are wet be used to rank or value the headwater streams? At present, we have a widespread estimate of the amount of stream length that is intermittently flowing or changing in length (Wilding & Parkyn 2006), but no widespread estimates of the variability in flow through time of these headwater systems. Are headwater streams typically dry for a matter of days or a matter of months through the year, and how does this period differ between years? Do the streams typically dry out at the same time each year? Is this the best time of year to make a stream valuation?

These questions could be answered by incorporating monitoring of the weirs installed at Totara Park into a monitoring network and by investigating means to economically survey the temporal variation in hydrology of a wide range of headwater streams.

Urban headwaters

Traditional urban development creates large areas of impervious surfaces, which means a large proportion of rainfall can no longer infiltrate and extensive stormwater systems are required. This can have a profound impact on stream hydrology, resulting in a stream flow regime that is more flashy, has a higher risk of flooding in lowland areas. Water quality is also affected, as pollutants that accumulate on impervious surfaces enter streams more rapidly and effectively (Brydon et al. 2006). Headwater wetlands can provide water detention and water storage during rain events, and water release during dry periods. Headwater streams and swales could be managed to slow flood flows and trap contaminants to reduce downstream effects. Together with measures to reduce impervious area in urban catchments, headwater streams and wetlands could be managed as important resources to ameliorate the effects of stormwater run-off and they could also provide significant areas of natural and biodiversity values within an urban context. To further the management of headwater streams in urban areas, studies of the present values and functions of urban headwater streams are needed and, in particular, investigation of the effects of low–impact urban design on the values and functions of urban headwater streams.

7 References

ANZECC/ARMCANZ (2000). 'Australian and New Zealand guidelines for fresh and marine water quality: national water quality management strategy.' Australian and New Zealand Environment and Conservation Council, and Agriculture and Resource Management Council of Australia and New Zealand, Canberra.

ARC (2000). 'Year 2000 Summary Report. Baseline water quality. Stream, lake and saline waters.' Auckland Regional Council, TP132.

Belsky, A.J.; Matzke, A.; Uselman, S. (1999). Survey of livestock influences on stream and riparian ecosystems in the western United States. *Journal of Soil and Water Conservation 54*: 419-431.

Brydon, J.; Roa, M.C.; Brown, S.J.; Schreier, H. (2006). Integrating wetlands into watershed management: effectiveness of constructed wetlands to reduce impacts from urban stormwater. J. Krecek and M. Haigh (*eds*.), Environmental Role of Wetlands in Headwaters, 143-154. Springer, Netherlands.

Burns, D.; Nguyen, L. (2002). Nitrate movement and removal along a shallow groundwater flow path in a riparian wetland within a sheep-grazed pastoral catchment: results of a tracer study. *New Zealand Journal of Marine and Freshwater Research 36*: 371-385.

Burt, T.P.; Pinay, G. (2005). Linking hydrology and biogeochemistry in complex landscapes. *Progress in Physical Geography 29*: 297-316.

Collier, K.J. (1993). Review of the status, distribution, and conservation of freshwater invertebrates in New Zealand. *New Zealand Journal of Marine and Freshwater Research 27*: 339-356.Collins, R.C. (2004). Fecal contamination of pastoral wetlands. *Journal of Environmental Quality 33*: 1912-1918.

Cooke, J.G. (1988). Sources and sinks of nutrients in a New Zealand hill pasture catchment II: Phosphorus. *Hydrological Processes 2*: 123-133.

Cooke, J.G.; Cooper, A.B. (1988). Sources and sinks of nutrients in a New Zealand hill pasture catchment III: Nitrogen. *Hydrological Processes 2*: 135-149.

Cooper, A.B. (1990). Nitrate depletion in the riparian zone and stream channel of a small headwater catchment. *Hydrobiologia 202*: 13-26.

Davie, T. (2002). 'Fundamentals of hydrology.' (Routledge: London).

Degens, B.P.; Donohue, R.D. (2002). 'Sampling mass loads in rivers - a review of approaches for identifying, evaluating and minimising estimation errors.' Water and Rivers Commission.

Department of Health (1992). 'Provisional microbiological water quality guidelines for recreation and shellfish gathering in New Zealand.' Department of Health.

Dons, A. (1987). Hydrology and sediment regime of a pasture, native and pine forest catchment in the Central North Island. *New Zealand of Forestry Science 17*: 161-178.

Elliott, A.H.; Carlson, W.T. (2004). Effects of sheep grazing episodes on sediment and nutrient loss in overland flow. *Australian Journal of Soil Research 42*: 213-220.

Heathwaite, A.L. (1993). Nitrogen cycling in surface waters and lakes. In 'Nitrate: processes, patterns and management'. (Eds TP Burt, AL Heathwaite and ST Trudgill) pp. 99-140. (John Wiley and Sons: Chichester).

Helsel, D,R.; Hirsch, R.M. (2002). 'Statistical methods in water resources.' (US Geological Survey).

Johnson, P.; Gerbeaux, P. (2004). 'Wetland types in New Zealand.' (Department of Conservation: Wellington).

Ledgard, S.F.; Menneer, J.C. (2005). Nitrate leaching in grazing systems and management strategies to reduce losses. In 'Developments in fertiliser application technologies and nutrient management. Occasional Report No. 18. Fertilizer and Lime Research Centre, Massey University, Palmerston North.' Palmerston North. (Eds LD Currie and JA Hanly) pp. 79-92.

McColl, R.H.S. (1978). Chemical runoff from pasture: The influence of fertiliser and riparian zones. *New Zealand Journal of Marine and Freshwater Research 12*: 371-380.

Merot, P.; Hubert-Moy, L.; Gascuel-Odoux, C.; Clement, B.; Durand, P.; Baudry, J.; Thenail, C. (2006). A method for improving the management of controversial wetland. *Environmental Management 37*: 258-270.

Moore, G. (2002). xlr8tr. In. (University of Melbourne: Melbourne).

Muirhead, R.W.; Collins, R.P.; Bremer, P.J. (2005). Erosion and subsequent transport state of Escherichia coli from cowpats. *Applied and Environmental Microbiology 71*: 2875-2879.

Nash, J.E.; Sutcliffe, J.V. (1970). River flow forecasting through conceptual models. I. A discussion of principles. *Journal of Hydrology 10*: 282-290.

Nguyen, L.; Downes, M.T.; Mehlhorn, J.; Stroud, M.J. (1999). Riparian wetland processing of nitrogen, phosphorus and suspended sediment inputs from a hill country sheep-grazed catchment in New Zealand. In 'Second Australian Stream Management Proceedings: The challenge of rehabilitating Australia's streams'. Adelaide. (Eds I Rutherfurd and R Bartley) pp. 481-485. (Cooperative Research Centre for Catchment Hydrology).

Nguyen, M.L.; Sheath, G.W.; Smith, C.M.; Cooper, A.B. (1998). Impact of cattle treading on hill land. 2. Soil physical properties and contaminant runoff. *New Zealand Journal of Marine and Freshwater Research 41*: 279-290.

NWQMS (2000). Primary industries – Rational and back-ground information. Chapter 9. In: Australian and New Zealand guidelines for fresh and marine water quality Volume 3. Biological parameters. Australia and New Zealand Environment and Conservation Council and Agriculture and Resource Management council of Australia and New Zealand. Pp 9.3.2 – 9.3.7.

Parkyn, S.; Davies-Colley, R.J.; Cooper, A.B.; Stroud, M.J. (2005). Predictions of stream nutrient and sediment yield changes following restoration of forested riparian buffers. *Ecological Engineering 24*: 551-558.

Parkyn, S; Wilding, T.; Croker, G. (2006). Small headwater streams of the Auckland Region Volume 4: Natural Values. Auckland Regional Council Technical Publication No. 310, 2006.

Quinn, J.M.; Stroud, M.J. (2002). Water quality and sediment and nutrient export from New Zealand hill-land catchment of contrasting land use. *New Zealand Journal of Marine and Freshwater Research 36*: 409-429.

Rowe, D; Quinn, J.; Parkyn, S.; Collier, K.; Hatton, C.; Joy, M.; Maxted, J.; Moore, S. 2005. Stream Ecological Valuation (SEV): a method for scoring the ecological performance of Auckland streams and for quantifying mitigation. ARC Technical Publication No. 302, 2006.

Runkel, R.L.; Crawford, C.G.; Cohn, T.A. (2004). 'Load estimator (LOADEST): A Fortran program for estimating constituent loads in streams and rivers.' US Geological Survey.

Rutherford, J.C.; Nguyen, M.L. (2004). Nitrate removal in riparian wetlands: Interactions between surface flow and soils. *Journal of Environmental Quality 33*: 1133-1143.

Smakhtin, V.U. (2001). Low flow hydrology: a review. *Journal of Hydrology 240*: 147-186.

Smith, C.M. (1992). Riparian afforestation effects on water yields and water quality in pasture catchments. *Journal of Environmental Quality 21*: 237-245.

Sukias, J; Nagels, J. (2006). Small headwater streams of the Auckland Region Volume 3: Nitrate & Phosphate Removal. Auckland Regional Council Technical Publication No. 311, 2006Trimble, S.W.; Mendel, A.C. (1995). The cow as a geomorphic agent - A critical review. *Geomorphology 13*: 233-253.

Tyrrel, S.F.; Quinton, J.N. (2003). Overland flow transport of pathogens from agricultural land receiving faecal wastes. *Journal of Applied Microbiology 94*: 87-93.

Vidon, P.G.F.; Hill, A.R. (2004). Landscape controls on the hydrology of stream riparian zones. *Journal of Hydrology 292*: 210-228.

Wigington, P.J.J.; Moser, T.J.; Lindeman, D.R. (2005). Stream network expansion: a riparian water quality factor. *Hydrological Processes 19*: 1715-1721.

Wilcock, R.J.; Nagels, J.W.; Rodda, H.J.E.; O'Connor, M.B.; Thorrold, B.S.; Barnett, J.W. (1999). Water quality of a lowland stream in a New Zealand dairy farming catchment. *New Zealand Journal of Marine and Freshwater Research 33*: 683-696.

Wilding, T.K. (1996). Effects of urban development on small stream ecosystems of Auckland and the influence of riparian vegetation. Unpublished MSc thesis, University of Auckland.

Wilding, T.K.; Parkyn, S.M. (2006). Small headwater streams of the Auckland Region Volume 1:Spatial Extent. Auckland Regional Council Technical Publication No. 313, 2006

8 Appendix 1: LOADEST model performance

LOADEST runs nine different models under the default option (Table 14). Models that include variables that are related to one or more of the other variables (models 2, 5, 6, 7, 8 and 9) use centred variables to eliminate the collinearity.

Model performance is frequently based on the coefficient of determination (r^2), which indicates the proportion of the total variation "explained" by the regression line. However, there is no general rule for what is too low an r^2 for a useful regression equation (Helsel and Hirsch, 1992). In this study, the coefficient of efficiency, E, is also used as a model performance measure. The coefficient of efficiency, expresses the proportion of variance of the observed concentration that is explained by the model (Nash and Sutcliffe, 1970). It provides a measure of the closeness of the plot of predicted concentration (in actual units e.g., mg/ l) versus the observed (in actual units, e.g., mg/l) to the 1:1 line. The coefficient of efficiency is therefore a more realistic indicator of model performance for predicting the observed concentration compared with r^2 which is a measure of the best-fit line. Unfortunately LOADEST does not produce residual output for the MLE or LAD models and so E cannot be calculated for the LAD estimates.

Table 14:

Model form and number.(LnQ = ln(streamflow) – centre of ln (streamflow); dtime = decimal time – centre of decimal time).

1	$a_0 + a_1 \ln Q$
2	$a_0 + a_1 \ln Q + a_2 \ln Q^2$
3	$a_0 + a_1 \ln Q + a_2 dtime$
4	$a_0 + a_1 \ln Q + a_2 \sin(2\pi dtime) + a_3 \cos(2\pi dtime)$
5	$a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 dtime$
6	$a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi dtime) + a_4 \cos(2\pi dtime)$
7	$a_0 + a_1 \ln Q + a_2 \sin(2\pi dtime) + a_3 \cos(2\pi dtime) + a_4 dtime$
8	$a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi dtime) + a_4 \cos(2\pi dtime) + a_5 dtime$
9	$a_0 + a_1 \ln Q + a_2 \ln Q^2 + a_3 \sin(2\pi dtime) + a_4 \cos(2\pi dtime) + a_5 dtime + a_6 dtime^2$

Table 15:

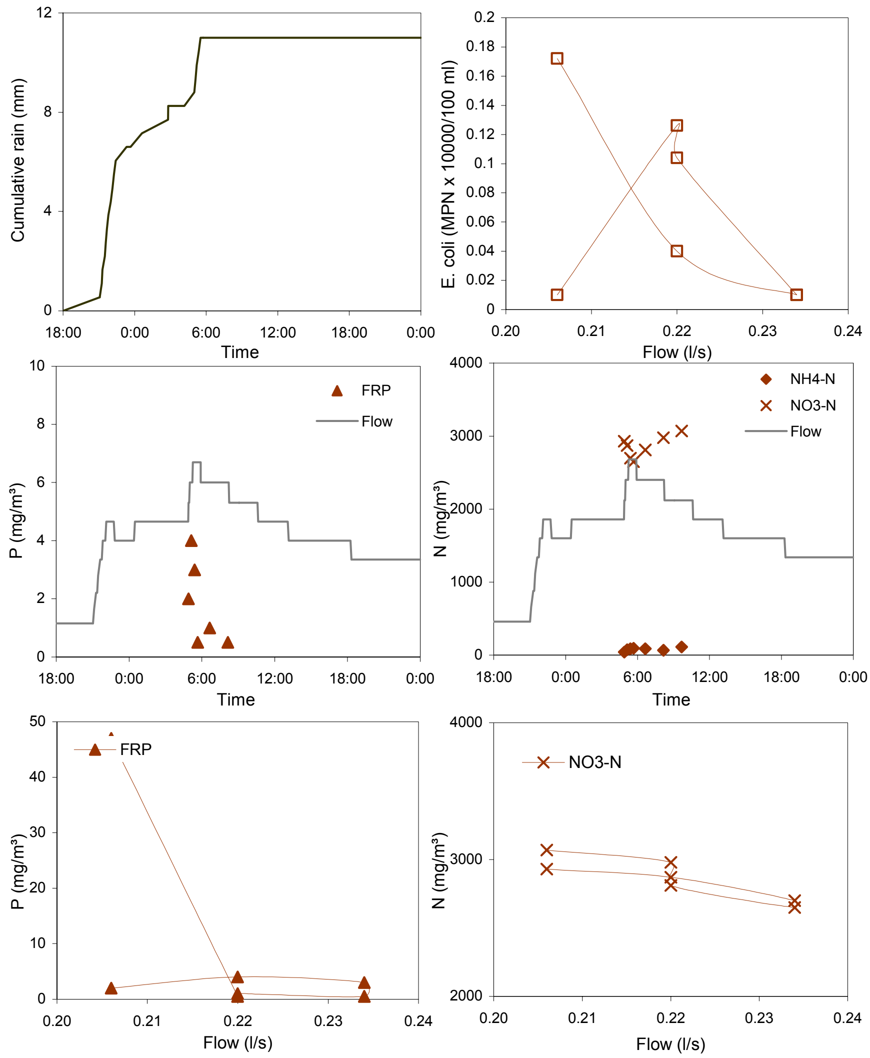
LOADEST model selection, parameters and extrapolation requirements.

Site	Model	L model	AMLE residual plot comments	L model E	L model r^2	L model PPCC	Estimated concentration > calibration concentration
Bush	SS	5, AMLE		0.28	0.61	0.86	N
	EC	3, AMLE		0.05	0.5	0.99	N
	TN	6, LAD	seasonal pattern	-	-	-	Y, 90th %ile
	NO3	6, LAD	seasonal pattern	-	-	-	Y, 90th %ile
	NH4	4, AMLE		0.56	0.71	0.97	N
	TP	7, AMLE		0.51	0.71	0.92	N
	FRP	4, AMLE		0.64	0.84	0.99	N
Gorse	SS	8, AMLE		0.84	0.92	0.98	N
	EC	7, AMLE		0.07	0.59	0.98	N
	TN	8, AMLE		0.95	0.98	0.98	Y, 90th %ile
	NO3	2, LAD	seasonal pattern	-	-	-	Y, 90th %ile
	NH4	9, AMLE		0.57	0.83	0.98	N
	TP	8, AMLE		0.80	0.93	0.99	N
	FRP	3, AMLE		0.71	0.89	0.99	N
Swamp	SS	7,AMLE		0.71	0.74	0.98	N
	NH4	9, AMLE		0.78	0.69	0.98	Y, 99th %ile
	TP	8, AMLE		0.70	0.84	0.99	Y, 99th %ile

9 Appendix 2: Storm event concentrations

Figure 31:

Flow, rainfall and water quality time series and hysteresis plots for the Swamp storm on 2 August 2005.



10 Appendix 3: Auckland Regional Plan: Air, Land, Water stream definitions

The following definitions of a river/stream are taken from the Proposed Auckland Regional Plan: Air, Land and Water (Variation 1, June 2002, downloaded September 2005). This can be accessed at http://www.arc.govt.nz/arc/publications/proposed-arp-alw.cfm and following the links to Section 12, Definitions And Abbreviations. This terminology is the subject of appeals to the Plan and may change.

Definitions and Abbreviations – 12
Proposed Auckland Regional Plan; Air, Land and Water Plan

Category 1 River or Stream
A river or stream which meets any one or more of the following criteria:
(a) has continual flow; or
(b) has natural stable pools having a depth at their deepest point of not less than 150 millimetres and a surface area of not less than 0.5 square metres present throughout the period commencing 1 February and ending 30 April of any year;
(c) has any of the following aquatic biota at any time of year:
- eels
- kokopu
- crayfish
- shrimp
- mayflies, stoneflies or caddisflies
- oxygen weed species Elodea sp., Egeria sp. and Lagarosiphon sp.
- pondweed species Potamogeton sp.

Notes:
(1) This definition does not include:
a. any artificial watercourse (including an irrigation canal, water supply race, canal for the supply for electricity power generation, and farm drainage canal); or
b. any stream which does not meet criterion (a) or (b) of the definition and which only meets criterion (c) because there is a dam or artificial pond (on the stream) containing any of the listed fauna and flora.
(2) Most, but not all, streams which appear as blue lines on Map Series 1 of the Proposed Auckland Regional Plan: Air, Land and Water are Category 1 rivers or streams. In addition some Category 1 rivers or streams do not appear on this map series.
(3) Where there is uncertainty over the status of any stream the ARC will provide assistance and advice concerning the steps involved in making that determination.

Category 2 Stream
Any stream that is not a Category 1 stream.
Note:
This definition does not include any artificial watercourse (including an irrigation canal, water supply race, canal for the supply for electricity power generation, and farm drainage canal).