Critical Thresholds of Ecological Function and Recovery Associated with Flow Events in Urban Streams

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The Urban Water Security Research Alliance (UWSRA) is a $50 million partnership over five years between the Queensland Government, CSIRO's Water for a Healthy Country Flagship, Griffith University and The University of Queensland. The Alliance has been formed to address South East Queensland's emerging urban water issues with a focus on water security and recycling. The program will bring new research capacity to South East Queensland tailored to tackling existing and anticipated future issues to inform the implementation of the Water Strategy.

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Cover Photograph:

Description: Urban Stream - Iron floc in a tributary to Blunder Creek, Daintree Close, Forest Lake, Brisbane.
Photographer: Fran Sheldon

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FOREWORD

Water is fundamental to our quality of life, to economic growth and to the environment. With its booming economy and growing population, Australia's South East Queensland (SEQ) region faces increasing pressure on its water resources. These pressures are compounded by the impact of climate variability and accelerating climate change.

The Urban Water Security Research Alliance, through targeted, multidisciplinary research initiatives, has been formed to address the region's emerging urban water issues.

As the largest regionally focused urban water research program in Australia, the Alliance is focused on water security and recycling, but will align research where appropriate with other water research programs such as those of other SEQ water agencies, CSIRO's Water for a Healthy Country National Research Flagship, Water Quality Research Australia, eWater CRC and the Water Services Association of Australia (WSAA).

The Alliance is a partnership between the Queensland Government, CSIRO's Water for a Healthy Country National Research Flagship, The University of Queensland and Griffith University. It brings new research capacity to SEQ, tailored to tackling existing and anticipated future risks, assumptions and uncertainties facing water supply strategy. It is a $50 million partnership over five years.

Alliance research is examining fundamental issues necessary to deliver the region's water needs, including:

- ensuring the reliability and safety of recycled water systems.
- advising on infrastructure and technology for the recycling of wastewater and stormwater.
- building scientific knowledge into the management of health and safety risks in the water supply system.
- increasing community confidence in the future of water supply.

This report is part of a series summarising the output from the Urban Water Security Research Alliance. All reports and additional information about the Alliance can be found at http://www.urbanwateralliance.org.au/about.html.

C. Davis

Chris Davis
Chair, Urban Water Security Research Alliance

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EXECUTIVE SUMMARY

Flow has a major effect on stream ecosystems. In urban environments, flow characteristics differ substantially from natural conditions due to the direct piping of water into stream networks and the increased amount of impervious (non-porous) surfaces in catchment areas. Changed hydrological conditions have a direct impact on macroinvertebrate assemblages through dislodgement of individuals and an indirect impact through reduced habitat diversity and degraded water quality. This project explored the ecological impacts of urbanisation on streams in the Brisbane region through: (i) exploring broad scale impacts of urbanisation and total impervious area on macroinvertebrate ecosystem health metrics; (ii) undertaking a more focussed spatial study of impervious impacts and direct connection on macroinvertebrate assemblages; (iii) focussing more specifically on three sites for a twelve-month period, covering a wet and dry phase, to detect differences associated with urbanisation in hydrology, water quality and macroinvertebrate assemblages; and (iv) exploring specific impacts of urbanisation on water quality, particularly pH and conductivity.

Catchment scale measures of urbanisation and measures of percent total impervious area (%TIA) for the highly urbanised sites were used to explore how much variation in the macroinvertebrate ecosystem health indicators, measured as part of the Healthy Waterways partnership ecosystem health monitoring program (EHMP), could be explained by some aspect of urbanisation. The %TIA was not a major predictor for any of the macroinvertebrate response variables, however, catchment scale measures of urbanisation were. The importance of urbanisation, and particularly lumped measures of urbanisation, suggest the influence of impervious area connection on the health of streams, as measured by these macroinvertebrate and fish indicators, may be significant.

To further focus on the influence of percent of upstream impervious area and direct connection of stormwater, a more intensive survey of macroinvertebrate assemblages in 26 streams across Brisbane was undertaken in 2010. This study suggested a high degree of between-site variability with minimal differences between site 'groups' classified as "forested with little or no direct stormwater connection", "Water Sensitive Urban Design (WSUD)" or "urban with direct stormwater connection". Taxa in the insect orders Ephemeroptera, Plecoptera and Trichoptera (EPT) had lower relative abundances, both in streams with direct urban connection and in streams with greater upstream catchment imperviousness. However, correlation between biotic and environmental variables in the directly-connected sites was lacking in comparison with other sites. This spatial study showed a strong impact of urbanisation on the sensitive groups of macroinvertebrates, namely EPT taxa, but did not provide a specific mechanism.

Our conceptual model of the impacts of urbanisation on streams suggests that, if flow is a driving mechanism, then impacts should be increased during the summer months when rainfall is highest. Three sites (one forested, one urban and one water sensitive urban design (WSUD)) were chosen for a more intense temporal study, instrumented with pressure gauge transducers to measure daily flow and sondes to log water quality continuously, with macroinvertebrate assemblage samples collected every six (6) weeks for a twelve-month period.

There were hydrological differences between sites that could be related to urbanisation. The two more urbanised sites, Stable Swamp Creek and the tributary of Blunder Creek, had a higher number of flow rises across nearly all months, higher average daily flows and higher rates of fall during flow recession. The one parameter that was markedly different to that expected was the minimum daily flow and the base-flow index; the highly urbanised Stable Swamp Creek had continual baseflow throughout the year which would have mitigated many of the water quality impacts of urbanisation.

As with the hydrological metrics, there were clear water quality differences between sites that could also be related to urbanisation. The two more urbanised sites, Stable Swamp Creek and the tributary of Blunder Creek, had larger daily ranges in dissolved oxygen (DO) and temperature. In comparison, water quality parameters in the forested Tingalpa Creek, while occasionally harsh, were more often typical of forested catchments, with low daily ranges in DO and temperature and low conductivity.

The focussed temporal study of macroinvertebrates across the highly urbanised Stable Swamp Creek compared with forested Tingalpa Creek suggested differences between the two that could be explained by urbanisation, but also differences that were intrinsic to the different stream types. Species richness and other summary metrics were similar across both sites, suggesting differences were likely to be at the assemblage composition level, rather than in terms of broad summary metrics. The assemblage of the urbanised Stable Swamp Creek was dominated by taxa common in degraded streams, including the midges (Chironominae) and worms (Oligochaeta). In comparison, the forested Tingalpa Creek assemblage was dominated by mayflies (Family Leptophlebiidae) and other insect orders. When the assemblage data was compared with the logged water quality and hydrology data, it was stream electrical conductivity, daily temperature range and rates of rise and fall of the runoff hydrograph that explained the most variation in assemblage differences between the two sites. These physical parameters are known to be heavily influenced by urbanisation.

During both the broad spatial study and the focussed temporal study of urbanisation, an increase in the presence of iron floc (rusty red deposit on the bed of the streams) was apparent in the urbanised streams, particularly in the tributary to Blunder Creek at Daintree Close downstream of Forest Lake. At this site, pH was often low and conductivity was often high. A broader investigation of the incidence of this iron floc in urban streams found that it is likely to be exacerbated by stream incision, with water needing to pass through deeper, iron rich soils to reach the stream in incised areas, compared with non-incised areas.

1. INTRODUCTION

Flow has a major effect on stream ecosystems. In urban environments, flow characteristics differ substantially from natural conditions due to the direct piping of water into stream networks and the increased amount of impervious (non-porous) surfaces in catchment areas (Walsh *et al.*, 2005). This can lead to sudden and large pulses of water in urban streams, events which may cause large scale erosion of instream habitat and dislodge organisms like invertebrates. Subsequently, dissolved oxygen levels may decline such that life can no longer be supported. In this way, the changes in stream hydrology caused by urbanisation can impact invertebrates directly or through changes in water quality and habitat availability (Figure 1-1). Degradation of urban streams is a worldwide problem (Paul and Meyer, 2001) with restoration attempts covering physical habitat restoration (instream and riparian) as well as flow modification (Bunn and Arthington, 2002). More recently, the focus has shifted to the role of stormwater removal in urban streams to restore more natural flow magnitudes and durations and reduce instream flow velocities during storm events (see Walsh *et al.*, 2005).

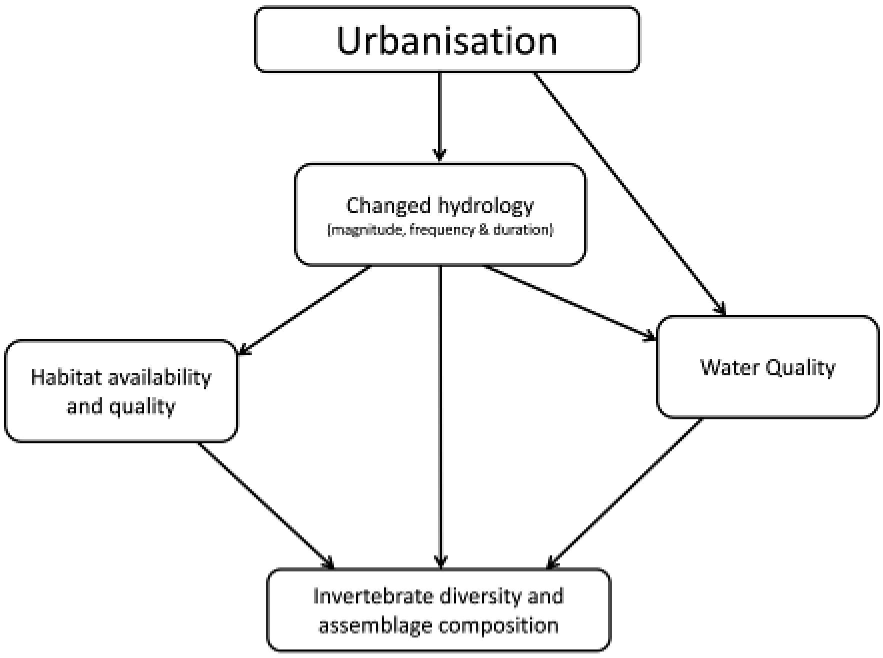


Figure 1-1: Conceptual model of the impacts on instream aquatic invertebrates caused by urbanisation.

Our conceptual understanding of the function of urban streams in sub-tropical Queensland suggests that the combined impact of a flashy hydrograph, where extreme velocities dislodge invertebrates, combined with inter-flow periods where water quality would become extremely poor provided a mechanism for poor urban stream health (Figure 1-2). Given this background we expected urban streams to be in the 'poorest' health from spring – autumn where the combination of higher air temperatures and more frequent flows would provide a mechanism for impact, and be in comparatively better health over the winter months.

This report has three sections with the following aims:

1. Can impacts of urbanisation be detected at broad spatial scales and can we distinguish between impacts associated with total upstream impervious area or impacts associated with direct stormwater inputs?
2. Given broad scale spatial patterns can we detect specific changes in hydrology, water quality of macroinvertebrate assemblages over a 12 months period (wet phase and dry phase) that can be attributed to urbanisation?
3. Does the process of urbanisation cause specific changes in water quality that can be associated with degraded macroinvertebrate assemblages?

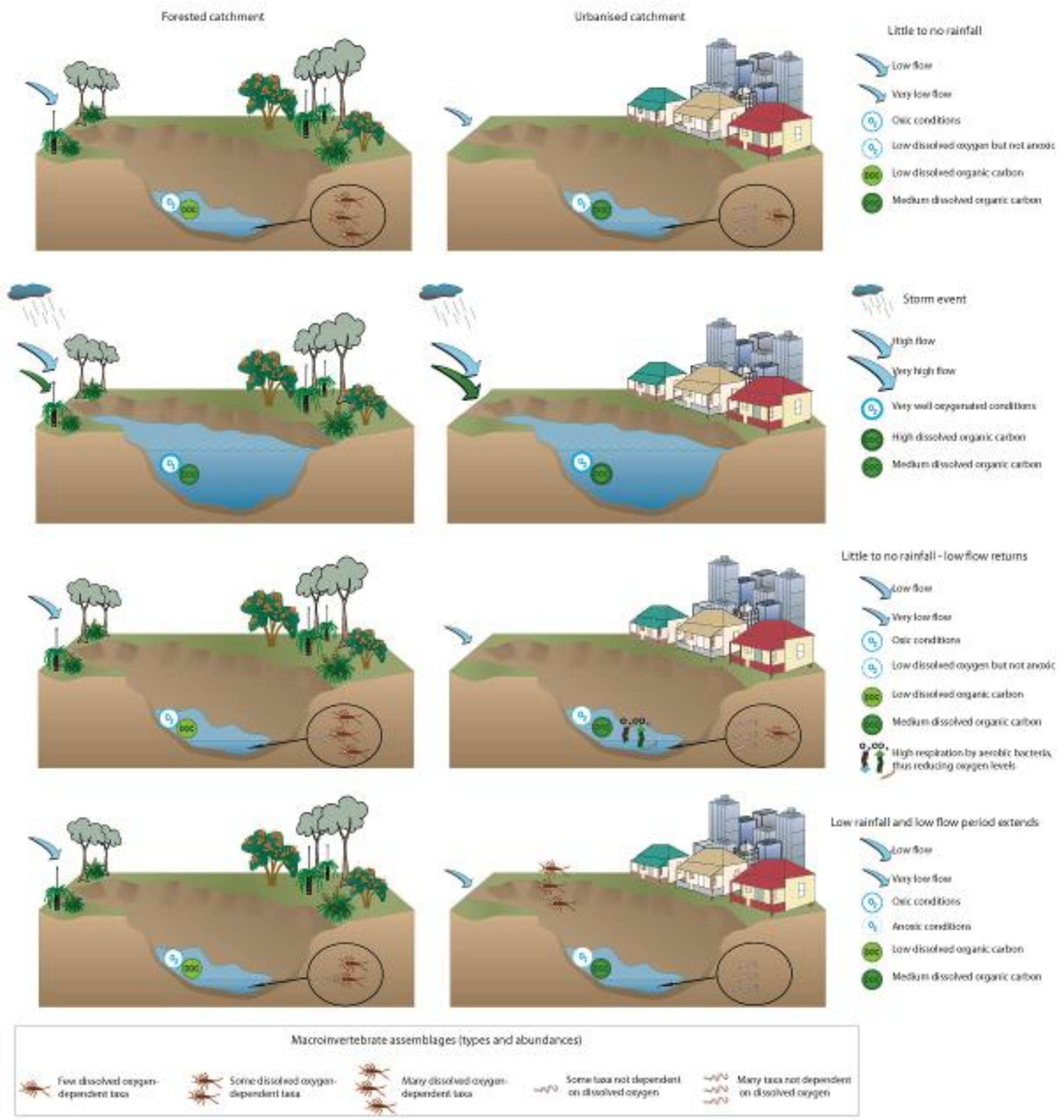


Figure 1-2: Conceptual model outlining the impacts on streams macroinvertebrates from urbanisation as mediated through flow and water quality changes.

2. BROAD SCALE DIFFERENCES ACROSS URBAN STREAMS IN BRISBANE

2.1. Spatial Scale of Land Use Influencing Ecological Indicators of Ecosystem Health in Urban Streams in South East Queensland

2.1.1. Background

In a recent analysis of the South East Queensland (SEQ) Ecosystem Health dataset, Sheldon *et al.* (2012) showed that urbanisation within the upstream catchment of a site was a significant driver of poor ecosystem health. This analysis focussed on the overall EHMP score and the separate Indicator scores and didn't provide information on which aspects of the five separate Indicators could be responding negatively to urbanisation, nor did it tease apart the impact of impervious area specifically as an element of urban land use.

The aim of this study was to extend the work of Sheldon *et al.* (2012) by focussing on percentage total impervious area (%TIA) as a driver of negative ecosystem health. In particular the aim of this study was to determine to extent to which %TIA is a significant driver of reduced aquatic ecosystem health, and to compare the influence of urban land area in aggregate on EHMP score and component indicators versus the influence of % TIA as the component of urban land area.

In both cases, as with the analysis of Peterson *et al.* (2011), the influence of the land-use (urban or %TIA) was treated as having a uniform effect regardless of distance from stream or catchment outlet, and then as having an effect of size which is inversely weighted with distance away from stream and, separately, with an effect inversely weighted with distance away from catchment outlet to mimic the reducing likelihood of direct hydraulic connection with streams as imperviousness is located further away. Doing so enabled the study to tease apart whether any ecosystem health impact was a function of directly connected urban area / impervious area or not, following work by several authors which has documented the ecological importance of hydraulic connectivity to streams rather than simply the presence of urban area or imperviousness (Fletcher *et al.* 2008).

2.1.2. Remote Sensing Methods

A set of geo-referenced colour aerial ortho-photos, at 0.5m resolution, were converted from tiff image format to the native format of the image analysis software being utilised, i.e. ERDAS Imagine v9.1 (Leica Geosystems, Atlanta, Georgia, USA.), format. Mosaics were then created for each catchment area. Training data was developed for each mosaic by taking signature samples from recognised targets. Although the spectral resolution of the imagery was restricted to red, green, and blue visible wavebands, with no infra-red wavebands, the spatial resolution (0.5m) allowed for correct visual identification of surface features, without the need for ground truthing surveys.

As infra-red wavebands were not available to differentiate between features such as green rooves and green grass, or pavements and high-albedo compacted soils, it was necessary to obtain samples from a wide variety of surface features with a range of different roof colours, road types, grassed surfaces, trees and other common surface types. The signature sampling was the most critical aspect to obtaining accurate classification results and it was therefore given a significant proportion of the total effort.

An initial supervised classification, using the parallelepiped non-parametric rule in ERDAS Imagine, was run for each mosaic using the respective signature sets, created for each mosaic, as training data. During the supervised classification process, results were checked against the original mosaics to identify errors, and additional signature sampling was required where spectral confusion occurred, such as for different shades of road surface and where roof colours were confused with non-impervious areas, i.e. a green roof and green grass. The classification was then re-run and the result was checked again followed by further sampling of signatures (if necessary) and a re-run of the classification to correct false negatives/positives if necessary.

Once a classification result was deemed final, the image was converted to ArcGIS (Environmental Systems Research Institute, Redlands, California, USA.) GRID format. The GRID was then re-classified into impervious and pervious classes. Total impervious area (TIA) was estimated for each catchment by viewing summary statistics to obtain the number of impervious grid cells (each cell = 0.25 m^2 for reclassified catchment grid) and subtracting number of false positive areas from the digital cadastre comparison.

The supervised classification method gave poor results, i.e. large areas of false positives, for one catchment, which had little impervious area in it. An unsupervised classification was then applied to this catchment, using 5 output classes, with the resulting output image reclassified from 5 to 2 classes, i.e. pervious and impervious. This method yielded a more accurate result for this catchment.

In some catchments there were large non-urban areas with significant area of false positives, which needed to be eliminated from the final TIA using cadastral data. The Queensland Digital Cadastral Database (DCDB) GIS layer was clipped to the extent of each catchment using ArcGIS v9.3. The resulting DCDB catchment layers were then used to identify the large parcels of non-urban areas, which were selected and removed, leaving DCDB polygons only covering the urban areas. Each of the image mosaics were then clipped to the extent of the new DCDB catchment layers to create images that only covered the urban areas of each catchment.

2.1.3. Statistical Methods

We aimed to understand and model the relationship between %TIA within an upstream catchment on various ecological heath indicators. The TIA metrics considered were based on various methods of spatially aggregating the rasters of impervious area within catchments to mimic potential variations in hydraulic connection with the stream and catchment outlet. Six spatial aggregation metrics were used: (i) Lumped; (ii) iFLO; (iii) iFLS; (iv) HA-iFLO; (v) HA-iFLS; and (vi) iEuc. Figure 2-1 below from Peterson *et al.* (2011) illustrates the metrics and how they differentially weight cells according to their location. By summing the weighted TIA values for all cells in the landscape, under each weighting a total TIA and consequently a weighted % TIA, or TIA metric was obtained for use in statistical analysis with the EHMP score and component indicators.

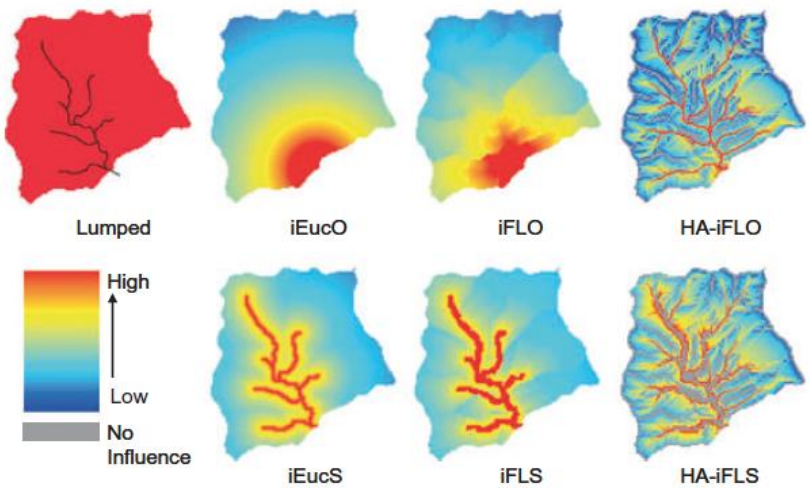


Figure 2-1: Landscape weighting metrics from Peterson *et al.* (2011).

The metrics are fully described in Peterson *et al.* (2011) but briefly:

- Lumped – all cells across a landscape are equally weighted.
- iFLO – TIA inversely weighted according to flow length (or distance) from cell to catchment outlet (where flows are measured).
- iFLS – TIA inversely weighted according to the distance from cell to the stream.
- HA-iFLO – TIA is weighted according to the product of flow accumulation at that cell and the inverse of the flow length to the catchment outlet.
- HA-iFLS – TIA is weighted according the product of flow accumulation at that cell and the inverse of the distance from the cell to the stream.
- iEUC – TIA is weighted according to the inverse of the Euclidean distance of the cell from the catchment outlet.

We examined the relationship of these TIA metrics with six indicators of ecosystem health from the EHMP dataset for SEQ: (i) FishOE; (ii) MacroRich; (iii) PET; (iv) PONSE; (v) PropAlien; and (vi) SIGNAL. Descriptions of these indicators and the type of data are outlined in Table 2-1; see also Bunn *et al.* (2010). We used data from 48 EHMP monitoring sites, covering various levels of urbanization within southeast Queensland. For each site, we used data for the six response variables measured in Spring and Autumn for the years 2002 – 2010.

In order to meet some of the assumptions of the linear model, it was necessary to apply transformations to some of the indicators, so that the residuals were roughly normally distributed. We used a logit transform for PropAlien and square root transforms for the count data: MacroRich, PET and PONSE. FishOE and SIGNAL were left untransformed for the analyses.

Table 2-1: Description of Response Variables.

Response Variable Name	Description
Fish OE	Ratio of observed to expected native fish species. Values are strictly positive.
MacroRich	Macroinvertebrate richness (number of taxa observed). Count data (integer values).
PET	Number of families in sample belonging to order Plecoptera, Ephemeroptera and Trichoptera. Count data (integer values).
PONSE	Percentage of native species expected. Values are strictly positive (values above 100% are in the data).
PropAlien	Proportion of alien fish observed. Values are integers from 0 to 100.
SIGNAL	Stream invertebrate grade number (average level). Values are strictly positive.

A number of important environmental covariates were also collected for each of the EHMP sites. These covariates included other land use metrics (see Sheldon *et al.* 2012), physico-chemical measurements, modelled stream flow data and seasonal information (Table 2-2). By incorporating some of these covariates into the statistical model, it was hoped that the linear model error variance could be reduced, increasing the ability of our models to detect significant relationships between the calculated TIA metrics and the ecological response variables.

We modelled the data using linear mixed effect models, with EHMP sites and the observation time (the data collection "run") as random intercepts. Many of the potential covariates in the model were dependent. For example, covariates for the same land use at various spatial scales (e.g. for Mid-Dense Forest (MDF) the different scales would be MDFBuf, h2oMDFper, MDF iFLO, MDF iFLS, MDF HA-iFLO and MDF HA-iFLS) are highly correlated. In addition, given the potential number of covariates, there were huge numbers of candidate models – far too many to exhaustively explore. For this reason we adopted the methodology of Sheldon *et al.* (2012) which uses a Bayesian Model Averaging (BMA) approach to explore the model space. The methodology employs a Markov Chain Monte Carlo (MCMC) search of the model space, with a Metropolis-Hastings step to decide whether to accept or reject the proposed new model at each iteration. For each of the response variables, we ran the MCMC scheme with 10 independent chains, each with a burn-in phase of 1000 steps followed by 10,000 recorded steps. Models were forced to include the random intercepts, but all other fixed effects were free to be included or left out of the model through the MCMC procedure. We allowed the models considered by the MCMC procedure to include interactions between Season and land use variables as well as variables within the Flow group and land use variables.

As described in Sheldon *et al*. (2012), we use groups of variables that are considered to be dependent and for which we would only consider including a single member variable in the model at a given iteration of the MCMC procedure. These groups of variables are described in Table 2-2. Note that: (i) there was a "None" option available within each group, indicating that none of the variables in the group were included in the model; and (ii) that the TIA metrics are included with the urban land uses, because of a perceived dependence between these variables. Note that under the MCMC procedure, if a variable was removed or swapped for a different variable within its group, then any corresponding interaction terms in the model were also either removed or swapped respectively. This rule was used to ensure that we did not allow interaction terms to be included in the model if the main effects were not also present.

Following the MCMC run, the posterior probabilities of the explanatory variables were computed along with the model-averaged coefficients for each of the variables in the linear model. Denoting the largest posterior probability encountered by the chain as pmax, only models whose posterior probabilities were larger than pmax /20 were retained and used to calculate the inclusion probability of an explanatory variable and the model-averaged coefficients. This approach is the "Occam's Window" approach of Madigan and Raftery (1994) and is the rationale for retaining only those models with a posterior probability above 0.0001 in the work presented by Sheldon *et al*. (2012).

Table 2-2: Explanatory variables used in each analysis.

Group	Explanatory Variable Options
Urban (G1)	UrbBuf, h2oUrbPer, iFLOurb, iFLSurb, HA-iFLOurb, HA-iFLSurb, PerResid, TIALumped, TIA iFLO, TIA iFLS, TIA HA-iFLO, TIA HA-iFLS, TIAEuc, None
Crop (G2)	cropBuf, h2oCropPer, iFLOcrop, iFLScrop, HA-iFLocrop, HA-iFLScrop, Percrop, None
Pasture (G3)	pastBuf, h2oPastPer, iFLOpast, iFLSpast, HA-iFLOpast, HA-iFLSpast, None
Dense Forest (G4)	dfBuf, h2oDFper, iFLOdf, iFLSdf, HA-iFLOdf, HA-iFLSdf
Medium Dense Forest (G5)	mdfBuf, h2oMDFper, iFLOmdf, iFLSmdf, HA-iFLOmdf, HA-iFLSmdf, None
Sparse Forest (G6)	sfBuf, h2oSFper, iFLOsf, iFLSsf, HA-iFLOsf, HA-iFLSsf, None
Very Sparse Forest (G7)	vspBuf, h2oVSPper, iFLOvsp, iFLSvsp, HA-iFLOvsp, HA-iFLSvsp, None
Conservation (G8)	conBuf, h2oConPer, None
Agriculture (G9)	agBuf, h2oAgPer, None
Flow (G10)	flowProb30, flowProb60, flowProb90, None
Season (G11)	Season, None
pH (G12)	pH, None
Electrical Conductivity (G13)	Cond, None
Maximum Temperature (G14)	TempMax, None
Temperature Range (G15)	TempRange, None
Minimum Dissolved Oxygen (G16)	DOMin, None
Dissolved Oxygen Range (G17)	DORange, None
Interactions (G18)	All possible interactions between: (i) the variables in G11 and G1-G9; and (ii) the variables in G10 and G1-G9. For all of these interactions, there is also an option to not include each interaction.

Table 2-3: Description of explanatory variables used in the analysis.

Explanatory Variable	Description
Season	The season in which the measurements were taken (Autumn or Spring)
flowProb30	The empirical probability of total stream discharge over the past 30 days being less than the observed discharge.
flowProb60	The empirical probability of total stream discharge over the past 60 days being less than the observed discharge.
flowProb90	The empirical probability of total stream discharge over the past 90 days being less than the observed discharge.
cropBuf	Lumped riparian crop cover
h2oCropPer	Lumped watershed crop cover
iFLOcrop	Inverse distance weighted metric to the site for crop cover
iFLScrop	Inverse distance weighted metric to the stream for crop cover
HA-iFLOcrop	Hydrologically active inverse distance weighted metric to the site for crop cover
HA-iFLScrop	Hydrologically active inverse distance weighted metric to the stream for crop cover
Percrop	Lumped crop cover along the reach
pastBuf	Lumped riparian pasture cover
h2oPastPer	Lumped watershed pasture cover
iFLOpast	Inverse distance weighted metric to the site for pasture cover
iFLSpast	Inverse distance weighted metric to the stream for pasture cover
HA-iFLOpast	Hydrologically active inverse distance weighted metric to the site for pasture cover
HA-iFLSPast	Hydrologically active inverse distance weighted metric to the stream for pasture cover
dfBuf	Lumped riparian dense forest cover
h2oDFper	Lumped watershed dense forest cover
iFLOdf	Inverse distance weighted metric to the site for dense forest cover
iFLSdf	Inverse distance weighted metric to the stream for dense forest cover
HA-iFLOdf	Hydrologically active inverse distance weighted metric to the site for dense forest cover
HA-iFLSdf	Hydrologically active inverse distance weighted metric to the stream for dense forest cover
mdfBuf	Lumped riparian medium dense forest cover
h2oMDFper	Lumped watershed medium dense forest cover
iFLOmdf	Inverse distance weighted metric to the site for medium dense forest cover
iFLSmdf	Inverse distance weighted metric to the stream for medium dense forest cover
HA-iFLOmdf	Hydrologically active inverse distance weighted metric to the site for medium dense forest cover
HA-iFLSmdf	Hydrologically active inverse distance weighted metric to the stream for medium dense forest cover
sfBuf	Lumped riparian sparse forest cover
h2oSPFper	Lumped watershed sparse forest cover
iFLOspf	Inverse distance weighted metric to the site for sparse forest cover
iFLSspf	Inverse distance weighted metric to the stream for sparse forest cover
HA-iFLOspf	Hydrologically active inverse distance weighted metric to the site for sparse forest cover
HA-iFLSspf	Hydrologically active inverse distance weighted metric to the stream for sparse forest cover
vsfBuf	Lumped riparian very sparse forest cover
h2oVSPper	Lumped watershed very sparse forest cover
iFLOvsp	Inverse distance weighted metric to the site for very sparse forest cover
iFLSvsp	Inverse distance weighted metric to the stream for very sparse forest cover
HA-iFLOvsp	Hydrologically active inverse distance weighted metric to the site for very sparse forest cover
HA-iFLSvsp	Hydrologically active inverse distance weighted metric to the stream for very sparse forest cover
conBuf	Lumped riparian conservation cover
h2oConPer	Lumped watershed conservation cover
agBuf	Lumped riparian agriculture cover
h2oAgPer	Lumped watershed agriculture cover

Explanatory Variable	Description
Pertree	Lumped watershed tree cover
Pergrass	Lumped watershed grass cover
TIALumped	Lumped watershed TIA cover
TIAiFLO	Inverse distance weighted metric to the site for TIA cover
TIAiFLS	Inverse distance weighted metric to the stream for TIA cover
TIAHA-iFLO	Hydrologically active inverse distance weighted metric to the site for TIA cover
TIAHA-iFLS	Hydrologically active inverse distance weighted metric to the stream for TIA cover
TIAEuc	Euclidean distance TIA cover
urbBuf	Lumped riparian urban cover
h2oUrbPer	Lumped watershed urban cover
iFLOurb	Inverse distance weighted metric to the site for urban cover
iFLSurb	Inverse distance weighted metric to the stream for urban cover
HA-iFLOurb	Hydrologically active inverse distance weighted metric to the site for urban cover
HA-iFLSurb	Hydrologically active inverse distance weighted metric to the stream for urban cover
Perresid	Lumped watershed residential cover
pH	pH of water quality sample
Cond	Electrical conductivity of water quality sample
TempMax	Maximum temperature of water
TempRange	Temperature range of water
DOMin	Minimum dissolved oxygen
DORange	Range of dissolved oxygen

2.1.4. Results and Discussion

The inclusion probabilities for the various land use metrics in the models for each of the six response variables are shown in Figure 2-2. For the fish response metrics no land-use measure at the scales included had a strong inclusion probability for modelled FishOE (Figure 2-2a), DF close to the site (HA-iFLS DF) had the highest inclusion probability for modelled PONSE (Figure 2-2d) while the presence of very sparse forest (VSF) in the riparian zone or close to the site influenced the proportion of alien species present (Figure 2-2e). Of the TIA metrics, TIA iEuc was the only metric of importance, with an inclusion probability of 0.4 for PONSE (Figure 2-2d).

For the macroinvertebrate response metrics, measures of urbanisation were important for family richness (MacroRich), with urbanisation anywhere in the catchment or in the riparian zone important (Figure 2-2b). The macroinvertebrate metric SIGNAL, which relies heavily on the presence of insect taxa was mostly influenced by dense forest at the site (DF iFLO) (Figure 2-2f), which intuitively makes sense as many of the insect taxa have a terrestrial adult phase which requires riparian cover.

Overall, TIA metrics were included in the models approximately 38% of the time. TIA metrics were also present as explanatory variables for modelling FishOE, with an overall inclusion of almost 7%, which were again spread mostly between the iEuc and HA-iFLO metrics. Inclusion probabilities of TIA metrics in the remaining models were very small.

Whilst the TIA metrics may not have featured heavily in the models, urban land use metrics featured strongly in modelling MacroRich and PropAlien with member variables included in of 99% and 94% of models respectively. For both response variables, lumped riparian, lumped watershed and iFLS metrics were included most frequently, and these lumped measures of urbanisation may be reflective of impervious connection which can occur anywhere in the catchment.

Figure 2-3 shows the inclusion probabilities for the water quality (physico-chemical) variables that were useful covariates for modelling each of the response variables. For MacroRich, temperature range was included in 47% of models. Minimum dissolved oxygen was included in approximately 35% of models for PET and 95% of the models for SIGNAL. For PONSE, pH was included as a predictor in 52% of models. PropAlien had a number of useful predictor variables, with pH, temperature range and minimum dissolved oxygen included in 21%, 20% and 33% of models respectively.

Figure 2-4 shows the inclusion probabilities of season and the three flow variables in the models. For FishOE, flow variables were included in approximately 26% of the models, whilst season was rarely included. For MacroRich, season was included in approximately 16% of models, whilst flow variables were included in nearly 61% of models. For PET, Flow variables were included in almost 25% of the models, but season was included in less than 5% of models. For PONSE, both flow and season variables were highly important predictors, with season included in almost 66% of models and flow variables in over 92% of models. For PropAlien, inclusion probabilities were higher yet, with season present in roughly 84% of models and flow variables present in almost 98% of models. For SIGNAL, season and flow variables were not particularly important.

Appendix Tables A1 – A6 provide the inclusion probabilities and model averaged coefficients from the models for each of the six response variables.

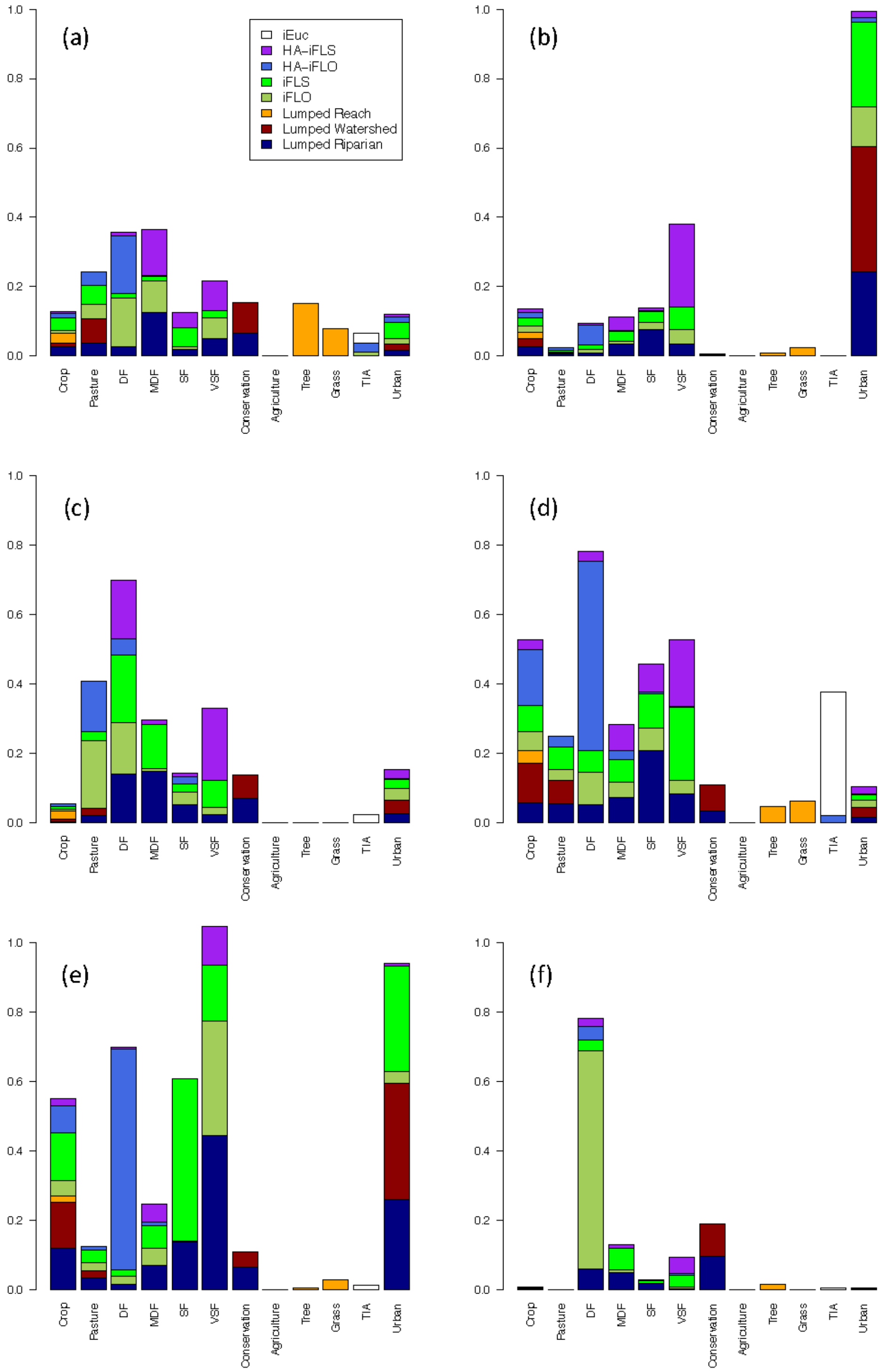


Figure 2-2: Inclusion probabilities of various land use metrics for: (a) FishOE; (b) MacroRich; (c) PET; (d) PONSE; (e) PropAlien; and (f) SIGNAL.

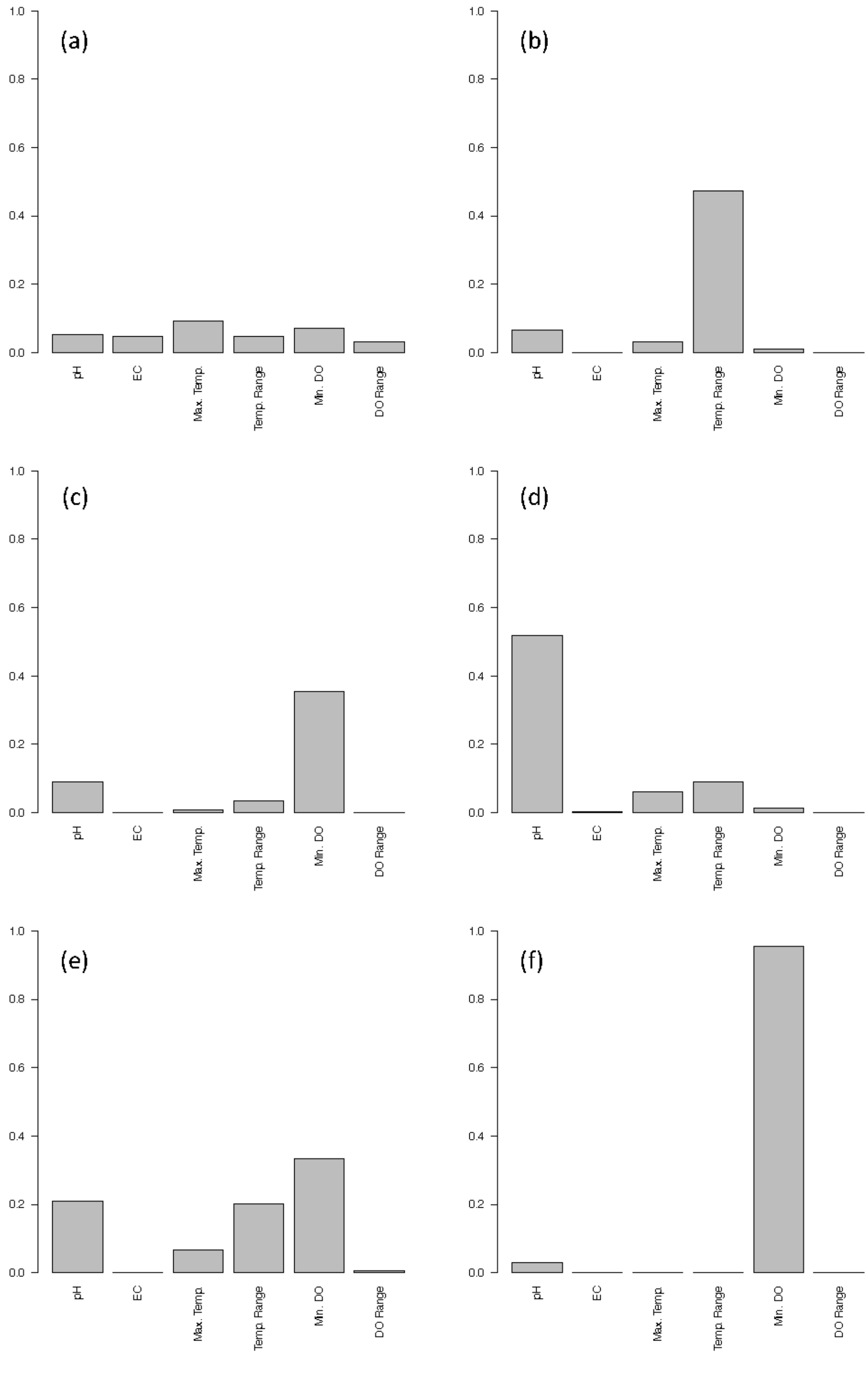


Figure 2-3: Inclusion probabilities of various physico-chemical variables for: (a) FishOE; (b) MacroRich; (c) PET; (d) PONSE; (e) PropAlien; and (f) SIGNAL.

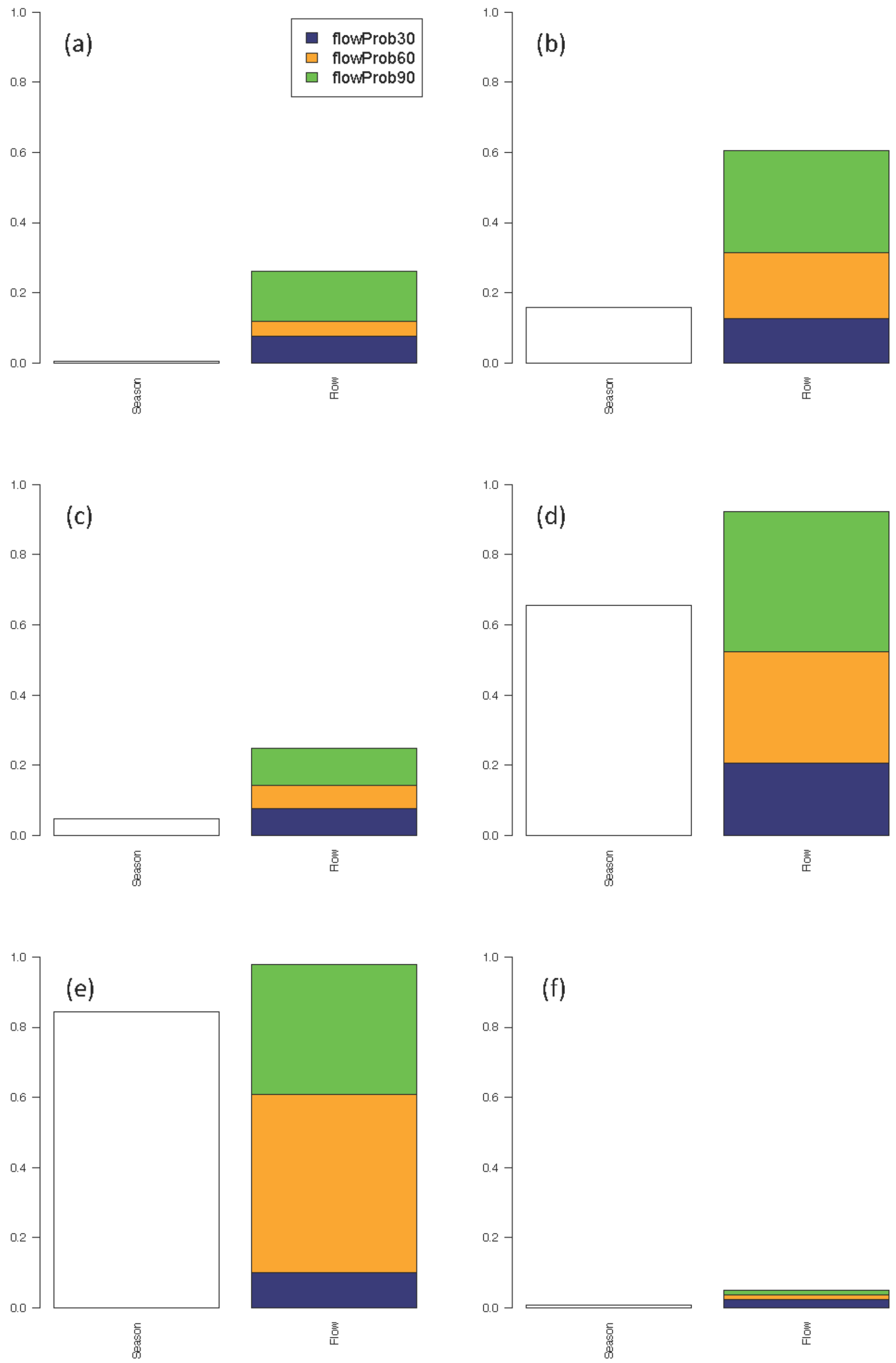


Figure 2-4: Inclusion probabilities of season and flow variables: (a) FishOE; (b) MacroRich; (c) PET; (d) PONSE; (e) PropAlien; and (f) SIGNAL.

2.1.5. Conclusion

While %TIA was not a major driving predictor for any of the ecological response variables the importance of urbanisation and particularly the lumped measures of urbanisation suggest the influence of connection of impervious areas on the health of streams, as measured by these macroinvertebrate and fish indicators, may be significant. This insight is new and begs the question – why does the aggregate measure of urbanisation show a negative impact ecologically on streams, whilst the generally accepted hydraulically relevant measure (Fletcher *et al.* 2008), impervious area (IA), weighted to both represent undifferentiated TIA as well as directly connected or hydraulically active IA, not show any negative impact? There are a number of possible explanations which are characterised below:

(1) 'Impervious area not in the right place' – the imperviousness in the urban catchments studied could have been generally located either far away from streams or variably so from catchment to catchment. Such variation could have masked the influence of TIA, which may still be exerting the expected negative impact from changes to stream hydrology occurring from changes to runoff.

(2) 'Not enough TIA' – the EHMP scheme was not explicitly set up to monitor the impact of urbanisation and so the number and range of urban sites may not best suit the question being asked. In addition only a limited number of the urban sites could be analysed to characterise TIA due to labour and cost constraints.

(3) 'Urbanisation does the damage' – perhaps in a sub-tropical climate such as in SEQ the on-going impacts of changed hydrology from urban IA are relatively minor compared to the initial, significant stream morphology, habitat and biotic impacts which occur during the process of construction? Streams in SEQ often have naturally significantly high peak flows due to the high intensity summer rainfall, and in larger catchments such as the Tingalpa studied in this project, the frequency of high flows can be high. Such catchments vary qualitatively in their behaviour from catchments in more temperate climates, where much of the research into the relationships between urbanisation and stream hydrology has been undertaken. If high flows and high peak flows are the norm, then perhaps the relative hydrologic impact of urbanisation is less significant in SEQ than in, say, Melbourne.

So, how might one decide between these alternative explanations? First of all one might ask whether there is a relationship between the total % of urban cover in the catchments examined and the value of the hydrologically active and inverse stream flow metrics HA-iFLS and iFLS, which best characterise directly connected or hydraulically active urban areas? Does urban cover and consequently IA tend to occur near or far from streams and the catchment outlet? Figure 2-5 below shows the relationships between lumped urban area (x-axis) and the four key metrics – iFLO, iFLS, HA-iFLO and HA-iFLS.

Figure 2-5 shows a variable but generally increasing trend towards more urban areas near streams (iFLS), near the catchment outlets (iFLO) and near the hydrologically active area around streams (HA-iFLS) and the catchment outlet (HA-iFLO) with increasing total urban area. The relationship between total urban area and urban areas near the hydrological active area around the catchment outlet (HA-iFLO) is variable and not clear. This suggests that generally as urban area increases more urban area is found near locations where it is likely to play an active role in modifying hydrology, and consequently that one would expect, if all urban area is equally impervious, that imperviousness is in the right location to exert a hydrological impact. Of course not all urban area is equally impervious and there is, as shown in Figure 2-5, notable variability in the location of urban area (and consequently imperviousness) across the 36 EHMP catchments assessed. This variability could well be masking the impact of IA, and in particular TIA, which will likely have a weaker relationship between increasing total TIA and the key hydrologically relevant TIA metrics – iFLS, iFLO, HA-iFLS & HA-iFLO.

The data in Figure 2-5 covers the 36 EHMP catchments which have greater than or equal to 10% total urban area. The actual TIA analysis only included a smaller number, the 9 EHMP catchments with greater than 50% total urban area, primarily because of the cost of the digital photographs for TIA characterisation and the length of the characterisation process. To further understand the direct role of TIA on stream health, over and above the general impact of urbanisation, at broad spatial scales we need a more mechanistic way of calculating TIA in upstream catchments. This analysis included only a portion of the total sites available as part of the EHMP dataset due to the laborious manner in which TIA needs calculating. If all 130 EHMP sites could be used and TIA calculated then a better understanding of the role of impervious area on stream health would be obtained, remembering of course that within urban areas it is not just impervious area that influences stream degradation, a better predictor may be connected impervious area as TIA that is directly connected to streams via the stormwater network may have a greater negative impact on stream health than unconnected areas.

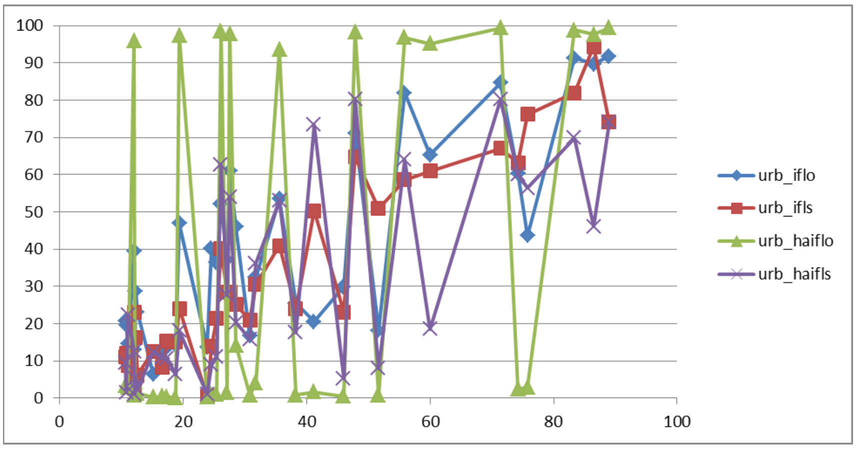


Figure 2-5: Relationships between total catchment urban area (% area, x-axis) and four urban area metrics (weighted % area, y-axis) across the 36 most urbanised EHMP catchments.

Interestingly, one of the ways in which urbanisation manifests its impacts on macroinvertebrates and fish is through changes to water quality, and particularly through reduced dissolved oxygen owing to the higher organic load in runoff draining urban areas. In this analysis minimum dissolved oxygen was an important predictor for both PET richness and SIGNAL score. Both these metrics rely heavily on the presence and abundance of sensitive insect taxa, mainly the Ephemeroptera (Mayflies), Plecoptera (Stoneflies) and Trichoptera (caddisflies). So there is sufficient evidence that urbanisation and potentially impervious area specifically are causing ecological impacts.

But are these impacts (i) an on-going consequence of the changes to hydrology following the building of impervious area as part of urbanisation, or (ii) were the impacts primarily caused by the process of construction – levelling ground, clearing vegetation, changing channel morphology – at the time of construction or shortly afterwards? That is, are the stream systems still adjusting to a new dynamic equilibrium in structural, hydrologic and ecological terms as under (i) or have they been in dynamic equilibrium for some time and no longer significantly being impacted by changed hydrology as under (ii)? Based on the evidence we have analysed here we can say that TIA, even when weighted to proxy the effect of directly connected impervious area is unlikely to be exerting a significant impact ecologically. This would suggest (ii); that the EHMP catchments are not continuing to degrade ecologically in adjustment to hydrological changes from urbanisation, but rather they have already adjusted.

However as shown in Figure 2-5 and explained above we cannot be certain of this – urban land use and in particular IA location is variable across catchments in relation to hydraulic connectivity to streams, and this variability may be masking the hydrological and consequently ecological impacts created. To clarify the situation a fuller analysis of urban land use and TIA across all EHMP catchments is required, and ideally, in addition, research to better understand how hydrological changes occur during and shortly after construction processes. Are there rapid changes or more strung out temporally? What changes happen quickly and which ones slowly? How do those hydrologic changes influence habitat, water quality and consequently ecology? These are important questions for they suggest qualitatively different interventions – construction phase control (e.g. sediment erosion control) or water sensitive urban design.

2.2. The Role of Indirect and Direct Connection on Macroinvertebrate Indices

An extended version of this study has been submitted to the peer reviewed journal Freshwater Sciences as the following manuscript:

Leigh C., Dunlop, J.E. and F. Sheldon (in review) Effects of urbanisation on macroinvertebrate assemblages: scale, tolerance and implications for biological recovery. Freshwater Sciences. Submitted September 2012.

2.2.1. Background

Physical and chemical characteristics of urban streams, including their flow regimes, differ substantially from natural conditions due, in part, to comparatively high proportions of impervious surface cover in urban areas and the direct piping of urban runoff into stream networks. These factors drive changes in biotic assemblages of urban streams and may lead to assemblages of generalist, tolerant taxa that no longer respond to environmental variation. Impacts of urbanisation on stream biota may be reduced, however, in systems such as in the tropics and subtropics where adaptation to 'flashy' flow events, typical of urban streams, is more likely. In this study we investigated effects of urbanisation at the catchment (% total imperviousness) and site scales of influence (localised, direct connection to stormwater drainage systems) on macroinvertebrate assemblage characteristics in subtropical streams in Australia, using family-level datasets collected under different sampling regimes (quantitative and semi-quantitative rapid bioassessment protocols).

2.2.2. Study Area and Design

Quantitative samples of benthic macroinvertebrates were collected from 16 sites across 12 different streams in the study area in May 2010 (austral autumn) during baseflow conditions (Figure 2-6). All sites were stream reaches within the greater Brisbane area. There were two scales of urbanisation relevant:

(i) the catchment scale, where sites were selected along a gradient of upstream catchment imperviousness. Estimates of the percentage of total imperviousness (%TI) in each site's upstream catchment area were calculated using the automated image analysis method described by Chowdhury *et al*. (2010) and provided by the Urban Water Security Research Alliance (Brisbane, Australia).

(ii) the local-scale, where sites were categorised as to their local-scale connection to stormwater drainage systems by the presence or absence of stormwater pipe entrances in the 100 m upstream from the sampling location. If a stormwater pipe was observed, the site was categorised as being directly connected (DC) and if no stormwater pipes were observed upstream for 100 m, sites were designated as non-directly connected (NDC).

Overall, the two groups of sites had similar ranges of catchment size (DC: 122-15,410 km2; NDC: 252-11,763 km2) and some overlap in their ranges of %TI (DC: 6-41%; NDC: 1-28%).

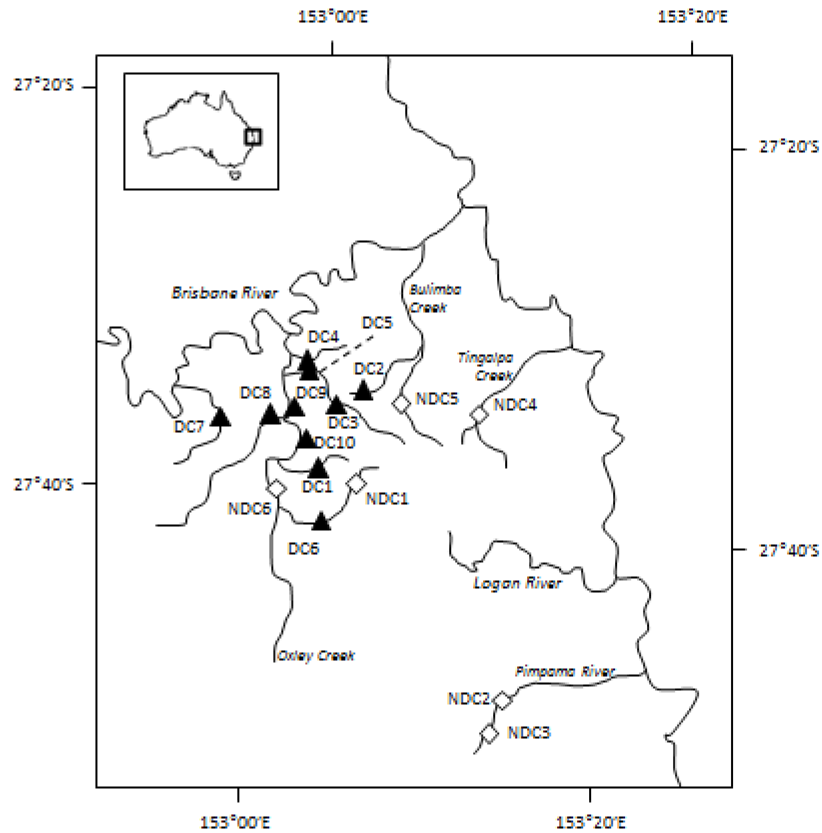


Figure 2-6: Sites sampled in May 2010 in the Brisbane area. Sites were located on stream reaches and categorised according to their local-scale connection to stormwater drainage systems by the presence (DC1-10, closed triangles) or absence (NDC1-6, open diamonds) of stormwater pipe entrances in the 100m upstream from the sampling points. From Leigh et al. (in review).

2.2.3. Field Sampling

Sites on the same stream were more than 2 km apart by stream flowpath. A modified suction sampler (similar to Growns 1990) was used to collect samples by placing it over a defined bed area at three locations (individual riffles, or runs if riffles were not present; pools were not sampled as they were not found consistently across all sites) along each site (up to ~50 m apart) and 'digging' or disturbing the substrate while collecting invertebrates by pumping the water into a volumetric flask. Samples were then preserved in 70% aqueous methanol until identified in the laboratory. Using this method, comparable macroinvertebrate densities could be estimated (individuals per sampling area) for each location. Samples represented the in-situ benthic fauna only as the method is unlikely to capture drifting or surface-dwelling macroinvertebrates.

The AUSRIVAS protocol of family-level taxonomic identification was followed, except for taxa identified as Porifera, Nematoda, Oligochaeta, Cladocera, Copepoda, Cladocera, Acarina or Collembola. Taxa that occurred in less than 5% of samples were excluded.

We calculated macroinvertebrate diversity variables from abundances (N) and densities (D). These were total macroinvertebrate family-level richness (S), EPT (Ephemeroptera, Plecoptera and Trichoptera) family-level richness (EPTs), relative richness (EPTrs, proportion) and abundance (EPTra, proportion), Chironomidae relative abundance (ChironRA, proportion), Oligochaeta relative abundance (OligoRA, proportion), and the Berger-Parker index (BP), a relative measure of dominance by the most abundant taxon, having positive values only and a maximum value of 1 (representing 1-taxon assemblages) (Magurran 1988). SIGNAL2 scores were also calculated as a measure of the macroinvertebrates' sensitivity to water pollution (Chessman 2003), along with SPEARpesticides, a recently derived, trait-based index adapted for southeast Australian fauna, calculated as the percentage of macroinvertebrate taxa at risk from pesticides and organic toxicants (see Schäfer *et al.* 2011). SIGNAL2 scores are based on grades specific for Australian invertebrates, which range from 1 (least sensitive) to 10 (most sensitive), and use a weighting system for abundances. SPEARpesticides is based on both physiological (direct toxicity of organic toxicants) and biological traits (characteristics of reproduction, dispersal and life stages) that are each defined as either 'sensitive' or 'tolerant'. SPEARpesticides is then calculated as the percentage of taxa in a sample that are classified as 'sensitive' for all traits. SPEARpesticides and SIGNAL2 calculations were based only on taxa for which scoring data are provided (thus, microcrustacean orders were not included).

2.2.4. Environmental and Water Quality Data

Dissolved O_2 concentration (DO, mg/L) and water temperature (Temp, °C) were recorded at each of the three sampling locations at each site using a handheld TPS WP-82Y meter fitted with a YSI 5739 DO probe, along with electrical conductivity (Cond, µS/cm) and pH at 2 of the locations using a handheld TPS WP-81 meter. Conductivity and pH readings were sufficiently stable between locations within sites such that measures were averaged to give one value for each variable per site. Canopy cover (%) at the thalweg was estimated at each location within each site, following Dixon *et al.* (2006).

2.2.5. Results

Sharp declines in the distributions of macroinvertebrate taxa were detected along the gradient of catchment imperviousness, in both datasets, with several being effectively lost from assemblages at low thresholds of imperviousness (1-17%). Ephemeroptera (families - Leptophlebiidae, Baetidae) and Trichoptera (family - Ecnomidae,) taxa had lower relative abundances in streams with higher proportions of imperviousness in their upstream catchment areas (Figure 2-7). Other sensitive taxa included the beetle families Psephenidae, Dytiscidae and Hydrophilidae and the dragonfly family Gomphidae (Figure 2-7). Sites locally connected to drainage systems were typified by high suspended solids and maximum diel temperatures, and low dissolved O_2 and canopy cover. There were few relationships between biotic and environmental variables in these sites compared with the many and strong relationships observed across all sites as a whole and in sites without localised connection to drainage systems.

2.2.6. Conclusion

Macroinvertebrate assemblages of urban streams, particularly of those directly connected to stormwater drainage systems at the site scale, that are dominated by generalist, tolerant taxa may be unlikely to respond to variation in the environmental conditions of their aquatic habitat, even in regions with naturally 'flashy' hydrographs and at least in the short-term. Substantial time and improvement in hydrology, riparian zone condition, and instream water quality may be required before positive biotic responses to rehabilitation efforts within urban streams are detected.

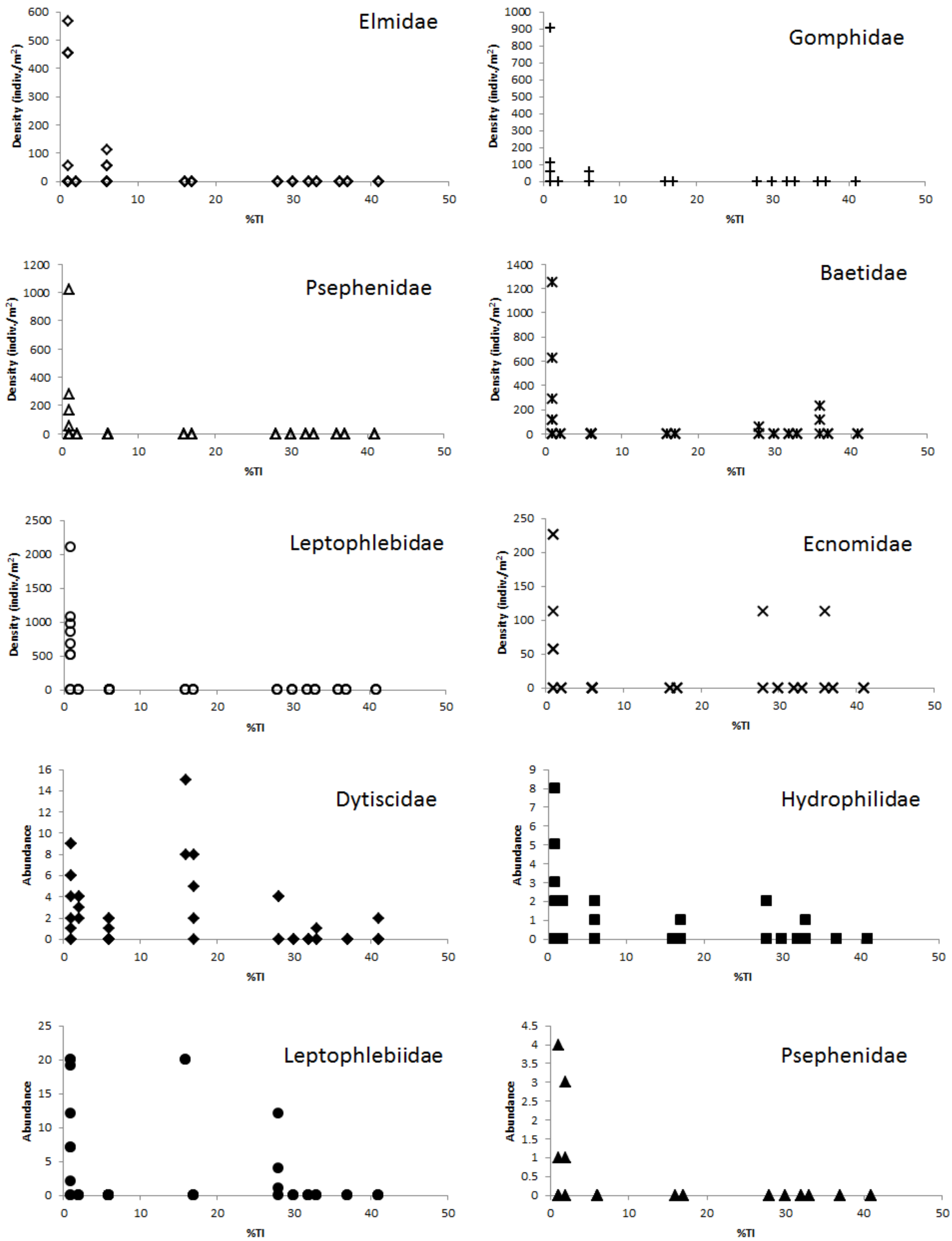


Figure 2-7: Taxa with decline in abundance (closed symbols; 2007-2008 data) or density (individuals/m^2; open symbols, 2010 data) along the gradient of catchment imperviousness (%TI). From Leigh et al. (in review).

3. FOCUSSED TEMPORAL STUDY ACROSS THREE URBAN STREAMS

3.1. Background

Flow has a major effect on stream ecosystems. In urban environments, flow characteristics differ substantially from natural conditions due to the direct piping of water into stream networks and the increased amount of impervious (non-porous) surfaces in catchment areas (Walsh and Kunapo, 2009). This can lead to sudden and large pulses of water in urban streams, events which may catastrophically dislodge organisms like invertebrates. Subsequently, dissolved oxygen levels may decline such that life can no longer be supported. However, the critical thresholds of these events are unknown. By monitoring dissolved oxygen levels and invertebrate diversity in streams within natural (forested) and urbanised (modified) catchments, we aim to determine these thresholds and provide knowledge critical for the future management and health of stream ecosystems in urban settings, where the majority of Australia's population reside.

The broader spatial studies (Section 2) suggested a strong influence of urbanisation on aspects of stream health (Section 2.1) as well as an influence of direct connection of impervious areas to the stream (Section 2.2). This section focusses on three (3) of the twelve creeks originally selected in the broader study (Section 2.2); in these three catchments the following detailed ecological studies were undertaken. The ecological studies were designed to provide support (or otherwise) for the conceptual models outlined in Section 1. The aims were to explore: (i) how standard hydrological metrics varied between the three sites and if the differences could be related to urbanisation; (ii) how water quality varied between the three sites and if the differences could be related to urbanisation; and (iii) how macroinvertebrate assemblages varied across 12 months and if changes observed in the assemblage reflected signals of urbanisation through water quality and hydrology.

Based on our conceptual model we predicted that (i) diversity in the urban site (Stable Swamp) would be lower than in either Tingalpa Creek or the tributary to Blunder Creek and dominated by more tolerant taxa (worms, snails etc), in contrast Tingalpa Creek would have the highest diversity and be dominated by EPT taxa; (ii) all sites should be more similar in the winter when flows are similar and should diverge more in summer when the urban site is subject to a greater frequency of high flows; and (iii) the abundance of specific fauna (eg. EPTs) on the same microhabitat across all three sites should be different in the summer when the urban sites are subjected to a higher frequency of high flow.

3.2. Study Sites

From the broader spatial study (Section 2.2) three sites were chosen for a more focussed temporal investigation of water quality and macroinvertebrate changes throughout a 12 month period, from dry to wet conditions (winter through summer). These sites were chosen based on (i) similarities in riparian cover - we were aiming to reduce riparian cover as a driving variable in differences, and (ii) the fact that they all had hydrological monitoring stations, collecting both daily flow data and water quality. The three sites included Tingalpa Creek (Forested treatment), Blunder Creek tributary (Water-Sensitive Urban Design (WSUD) treatment) and Stable Swamp Creek (Urban treatment).

Tingalpa Creek (Figure 3-2a) is an upland creek in the Redlands Shire Council region, the upstream portion of the creek is completely forested, the channel comprises bedrock riffles, runs and pools with complex microhabitats of snags (fallen timber), tree roots and macrophytes. Stable Swamp Creek Figure 3-2c) and the tributary to Blunder Creek (Figure 3-2b) are both in the Oxley-Blunder Creek subcatchment that drains into the Brisbane River. The upstream reaches of Stable Swamp Creek are highly urbanised, the channel is degraded and greatly incised, pools have deep silt with poor habitat quality, mostly dominated by thick introduced macrophytes, the one riffle below the road culvert was likely placed in the channel during early attempts at channel restoration. The WSUD site on the tributary to Blunder Creek is downstream of Forest Lake, a large stormwater retention basin; the channel is deeply incised and degraded with mostly poor habitat quality. The presence of good riparian vegetation on both banks, however, has allowed some riffle development along the channel.

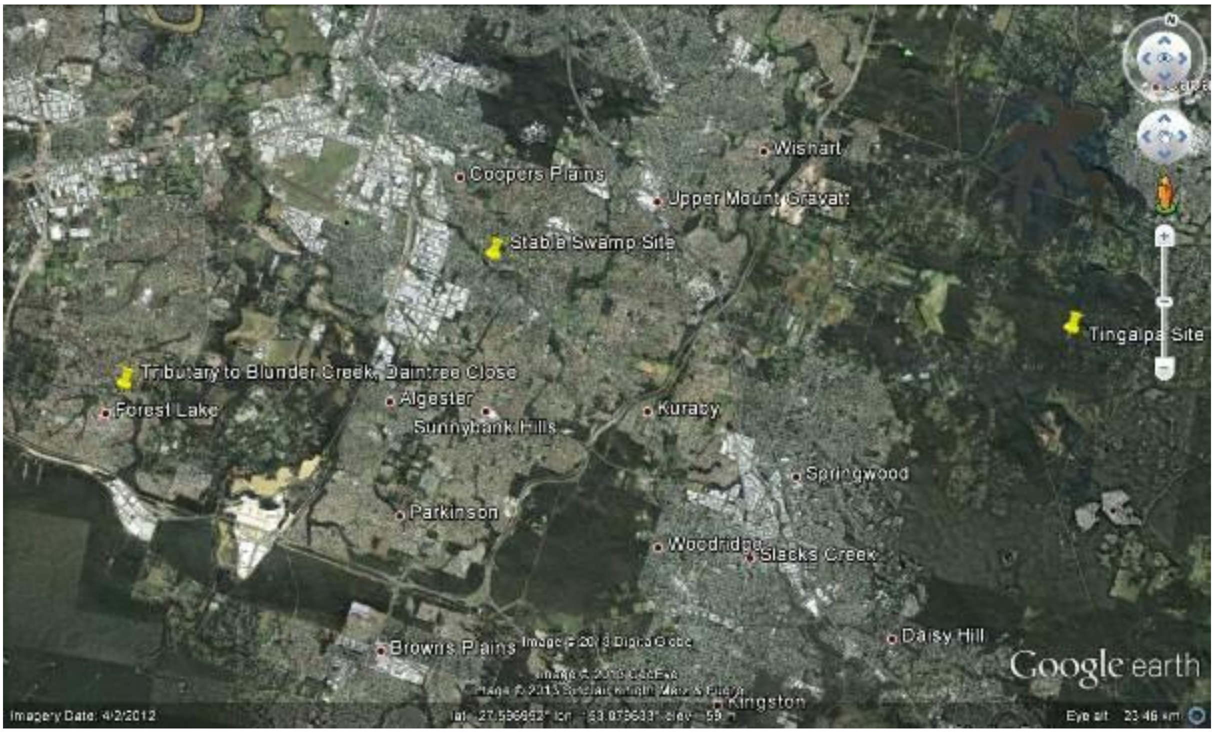


Figure 3-1: Location of the three study sites, Tingalpa Creek (forested), Stable Swamp Creek (urban) and the tributary to Blunder Creek (WSUD).

(a)

(b)

(c)

Figure 3-2: Study Sites (a) Forested site – Tingalpa Creek at Sheldon; (b) WSUD Site - tributary to Blunder Creek and Daintree Close, Forest Lake; (c) Urban site – Stable Swamp Creek at Sunnybank.

3.3. Hydrological Differences

3.3.1. Methods

3.3.1.1 Data Collection

Hydrological data for the three catchments was obtained from a larger dataset of 12 catchments which had been instrumented (for full details see Chowdhury *et al.* 2013). A tipping bucket raingauge (0.2 mm) and pressure transducer with data logger measured continuous six minute rainfall and water height data respectively at each site for a period from 2009 until the end of 2012.

3.3.1.2 Analysis

The Time Series Module in the River Analysis Package (RAP) (Marsh, 2004) was used to calculate a number of hydrological metrics deemed to drive declining ecological health in urban streams (Walsh and Kunapo, 2009). These metrics were calculated on daily data but summarised by calendar month for the period Summer 2009 (January) until Summer 2011 (December). Principal Components Analysis using Varimax Rotation was used to summarise the calculated hydrological metrics and determine which metrics were driving differences, if any, between both catchments and months (seasons). Metrics seen to be significant drivers of differences were further explored using Two-Way Analysis of Variance with both site and month as fixed factors. All analyses were performed in SPSS Statistics (version 20).

3.3.2. Results and Discussion

3.3.2.1 Background Hydrological Setting

There were a total of 75 missing daily flow values in the 2009-2011 period for Tingalpa, 16 gaps in total, longest gap was 34 days between 02/07/2009 and 04/08/2009. There 141 days of missing values within the Sunnybank data set for the same period, 8 gaps in total with the longest being 70 days between 15/12/2009 and 22/2/2010 and the next longest being 43 days between 20/07/2011 and 31/08/2011. Blunder creek contained 42 missing daily values for the period 2009-2011. There were 9 gaps in total with the longest being 33 days between 09/10/2009 and 10/11/2009. Gaps in the data were filled using linear interpolation, which basically fills the gaps in the data by drawing a straight line between the ends of the data where the gap occurs.

The line is defined by the following equation. If the gap is from t1 to t2, and the values of the time series at the start and end of the gap are x1 and x2, then the value of x over the interval t1 <t< t2 is approximated as:

> *The line is defined by the following equation. If the gap is from t_1 to t_2, and the values of the time series at the start and end of the gap are x_1 and x_2, then the value of x over the interval $t_1 < t < t_2$ is approximated as:*

$$x(t) = \frac{x_2 - x_1}{t_2 - t_1} \times t + \frac{x_1 t_2 - x_2 t_1}{t_2 - t_1}$$

(RAP, Marsh 2004)

Hydrographs for the three sites for the study period are shown in Figure 3-3 to Figure 3-5) with the same hydrographs plotted on the same scale in Figure 3-6. The urbanised Stable Swamp Creek and the WSUD tributary to Blunder Creek both had higher total discharge and more frequent discharge events compared with the forested Tingalpa Creek (Figure 3-6).

Table 3-1: Description of the variables calculated from the daily flow data (Marsh 2004).

Variable Name	Variable Acronym	Variable Description
Minimum	Min	Minimum is the smallest value for flow recorded for the time period.
Maximum	Max	Maximum is the largest value for flow recorded for the time period.
Percentile 10	P 10	The 10th percentile is the value that is exceeded by 10% of the records.
Percentile 90	P 90	The 90th percentile is the value that is exceeded by 90% of the records.
Mean Daily Flow	MDF	The Mean daily flow is a measure of central tendency and is calculated as the average of the records (sum of values/number of days in the time period).
Median Daily Flow	Med	The Median is the "middle" value for the entire record: it is the value exceeded 50% of the time. For flow data, The median is usually much lower than the mean daily flow because the distribution of discharge data is negatively skewed with a lower limit of zero and no upper limit.
Coefficient of Variation	CV	The CV of daily flow is the mean of all daily flow values divided by the standard deviation for the daily flow values.
Standard deviation	STD	The standard deviation is a measure of how widely the values are dispersed from the mean value. The Standard Deviation has the same units as the input data.
Skewness	Skw	Skewness is a measure of how different the mean and median are. Skew = mean/median.
Number of zero flow days	Zer	The number of zero days counted. Note that the number of zero flow days does not include days with a missing record unless they are filled with zero values.
Number of High Spell (5)	HS(5)Num	Number of times the flow exceeded five times the mean flow value for the time period.
Number of High Spell(10)	HS(10)Num	Number of times the flow exceeded ten times the mean flow value for the time period.
Number of Low Spell (0.5)	LS(0.5)Num	Number of times the flow fell below half the mean flow value for the time period.
Number of Low Spell (0.1)	LS(0.1)Num	Number of times the flow fell below one tenth of the mean flow value for the time period.
Longest Low Spell(0.5)	LS(0.5)Long	Duration (in days) of longest low flow event below half the mean flow value for the time period.
Mean of Low Spell(0.5) troughs	LS(0.5)Peak	Mean of low spell troughs. Low spell threshold was set at half the mean flow value for the time period.
Mean Duration of Low Spell(0.5)	LS(0.5)MeanDur	Mean duration (in days) of low spell events. Low spell threshold was set at half the mean flow value for the time period.
LS(0.5)TotDur	LS(0.5)TotDur	Total duration (in days) of low spell events for the time period. Low spell threshold was set at half the mean flow value for the time period.
Number of Rises	NumRise	Number of continuous periods of rise.
Mean magnitude of Rises	MMagRise	Mean difference in the flow values between the start and end of the rise.
Mean duration of Rises	MDurRise	Mean duration of periods of rise.
Total duration of Rises	TotDurRise	Total duration of periods of rise.
Mean rate of Rise	MRateRise	Mean rate of rise.
Greatest rate of Rise	GreatRatRise	The fastest rate of rise.
Number of Falls	NumFall	Number of continuous periods of fall.
Mean magnitude of Falls	MMagFall	Mean difference in the flow values between the start and end of the fall.
Mean duration of Falls	MDurFall	Mean duration of periods of fall.
Total duration of Falls	TotDurFall	Total duration of periods of fall.
Mean rate of Fall	MRateFall	Mean rate of fall.
Greatest rate of Fall	GreatRateFall	The fastest rate of fall.
Baseflow Index	BFI	Ratio of base flow to total flow in a period.
Flood Flow Index	FFI	Ratio of non-base flow to total flow in a period.
Mean Daily Baseflow	MDBF	Mean of base flow in a period.

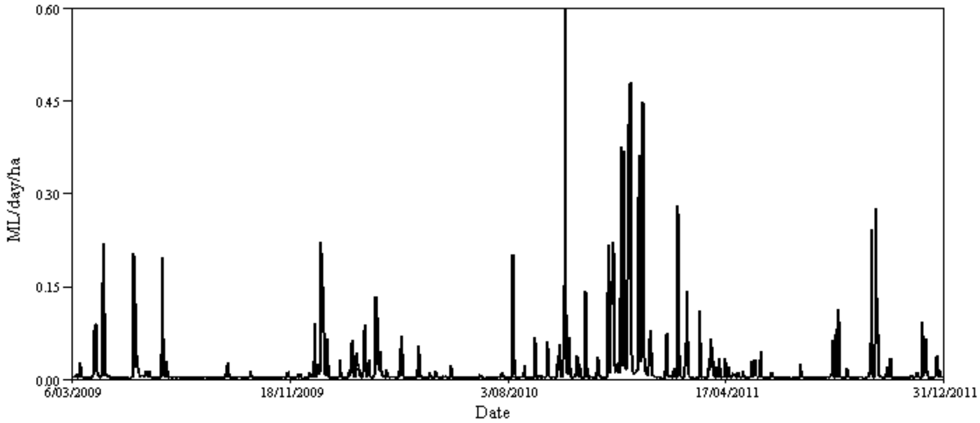


Figure 3-3: Hydrograph of daily flow (ML/day per hectare of catchment) for Blunder creek at Daintree Close, Forest Lake, over 2009-2011.

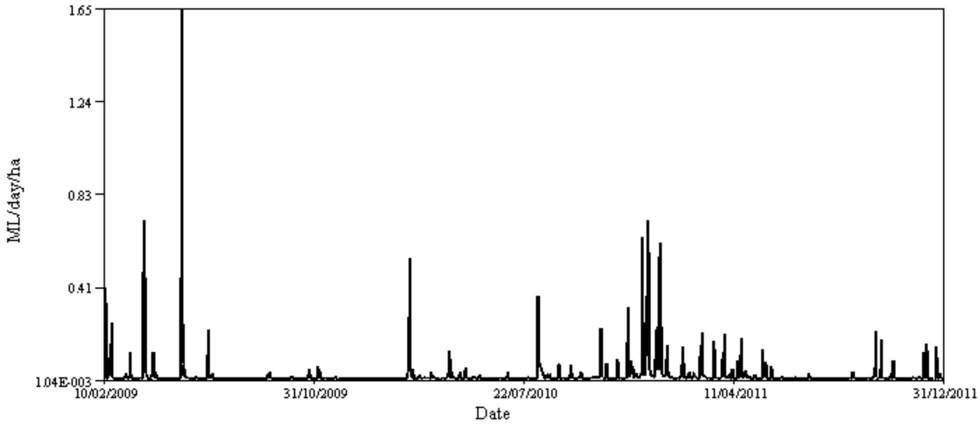


Figure 3-4: Hydrograph of daily flow (ML/day per hectare of catchment) for Stable Swamp Creek, Sunnybank, over 2009-2011 period.

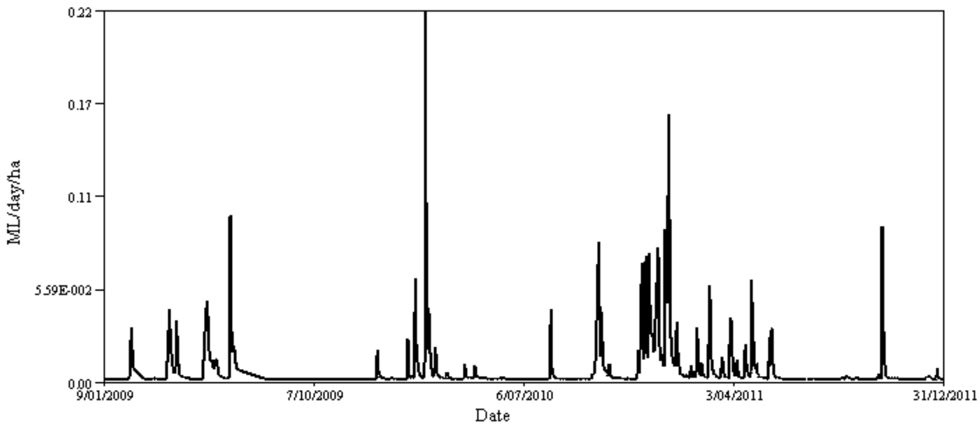


Figure 3-5: Hydrograph of daily flow (ML/day per hectare of catchment) for Tingalpa Creek, Sheldon, over 2009-2011 period.

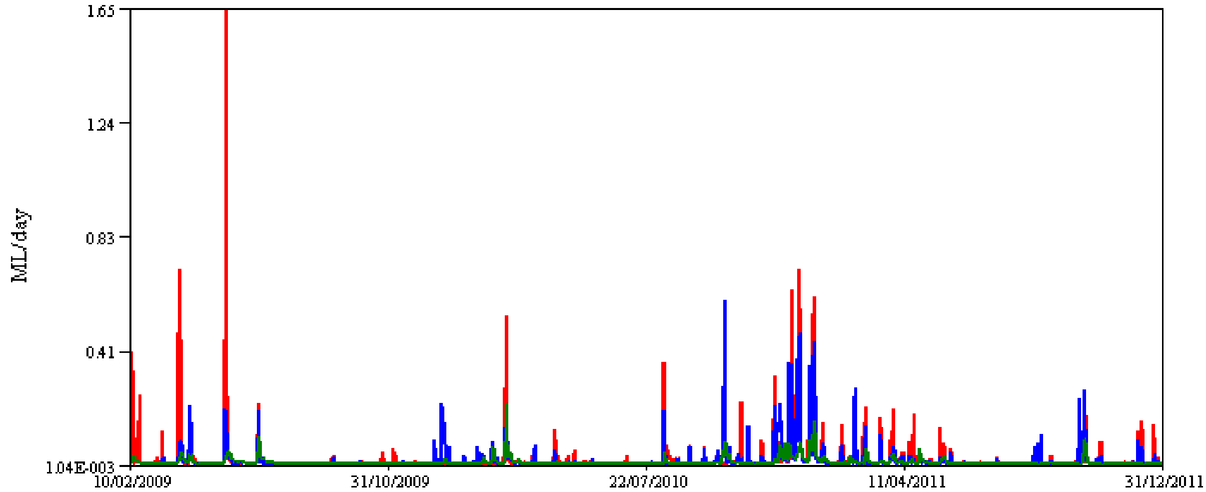


Figure 3-6: Hydrograph of daily flow (ML/day per hectare of catchment) for Tingalpa Creek (Green), tributary to Blunder Creek (Blue) and Stable Swamp Creek (Red), between 2009 and 2011.

3.3.2.2 Hydrological Drivers between Catchments

Principal Components Analysis (PCA) was used to summarise the hydrological metrics and explore differences between sites and months (seasons). In the initial PCA, PC1 explained 37% of the variation in the data and was associated with the number of high flow events and the magnitude and rate of rise and the magnitude of the fall (Table 3-2). PC2 explained a further 12% of the variation and was associated with base flow index (BFI) and flood flow index (FFI); PC3 explained 9% of the variation and was associated with the number of rises, whole PC4 explained a further 7% and was associated with the duration of both rises and falls (Table 3-2). This suggests that the hydrological metrics distinguishing months across the sites are largely related to the number of high flow events as well as their rate of rise and fall and duration. Hydrological metrics describing hydrology in December 2010 and January 2011 from both Stable Swamp Creek and the Blunder Creek tributary were responsible for the distribution of sample months along PC1 (Figure 3-7; Figure 3-8); these sample months represent extreme rainfall events in the Brisbane region and while Tingalpa is not represented in this plot, the more eastern catchments in the Brisbane region did not receive similar rainfall intensity and so its absence may reflect differing rainfall patterns, rather than the forested vs. urban categorisation.

In the reduced PCA with the high flow months (Dec 2010 and January 2011) from Blunder Creek and Stable Swamp Creek removed, the separation of sample months was more apparent. PC1 explained 35% of the total variation and was associated with the magnitude of the rates of rise and fall, while the forested Tingalpa Creek didn't have any hydrological months high on PC1 there was no clear separation of the sites suggesting similar magnitudes of flow rise and fall in all three streams (Figure 3-9). PC2 explained a further 14% of the variation and was associated with aspects of minimum flow (Min and P10), with the highly urbanised Stable Swamp Creek clustering higher on PC2 than either the forested Tingalpa Creek of the tributary to Blunder Creek, suggesting Stable Swamp creek has much higher base flow conditions than either Tingalpa Creek or Blunder Creek. PC3 explained a further 9% of the variation and was associated with base flor (BFI) and flood flow (FFI) indices (Table 3-3).

Table 3-2: Rotated component matrix for the initial PCA including the variation explained by each principal component, high component loadings are highlighted in grey.

Rotated Component Matrix[a]

	Component							
	1	2	3	4	5	6	7	8
Variation Explained	37.264	12.304	9.664	7.083	6.500	5.121	3.811	3.054
Min	.246	.093	.168	.854	.106	-.121	.086	-.012
Max	.390	.167	.058	.108	.820	.055	-.006	.037
P 10	.410	.061	.187	.741	.086	-.069	.119	.301
P 90	.687	.252	-.020	.087	.159	-.063	-.138	-.110
MDF	.746	.175	.026	.277	.459	-.008	-.051	-.212
Med	.641	.178	-.019	.583	.042	-.120	.026	-.145
CV	.060	.583	.081	-.138	.348	.490	.078	-.191
STD	.878	.195	.141	.116	.263	.004	.037	.071
Skw	.012	.074	-.073	-.110	.151	.913	-.119	-.015
Zer	-.079	-.076	-.138	-.038	-.107	.943	-.020	-.026
HS(5)Num	.663	.343	-.067	.110	.141	-.055	.002	.475
LS(0.5)Num	.268	.723	.323	.217	.074	-.101	-.072	.034
LS(0.5)Long	-.364	-.787	-.062	-.240	-.070	.158	.060	-.275
LS(0.5)Peak	.471	.259	.251	.556	.019	-.080	.015	.163
LS(0.5)MeanDur	-.331	-.829	-.145	-.223	-.107	.158	.044	-.168
LS(0.5)TotDur	-.509	-.504	.204	-.181	-.010	.148	.032	-.448
NumRise	.018	.039	.933	.059	.090	-.171	.010	.023
MMagRise	.931	.165	.002	.127	.168	.004	.053	.085
MDurRise	.019	-.046	-.520	-.017	-.023	-.124	.774	-.074
TotDurRise	.100	.003	.300	.066	.062	-.301	.851	-.001
MRateRise	.780	.228	.078	.105	.243	.063	.062	.248
GreatRatRise	.523	.176	.151	.021	.712	.023	.032	.069
NumFall	.020	.047	.943	.069	.071	-.135	-.072	.041
MMagFall	.918	.171	.028	.142	.216	-.006	.063	.101
MDurFall	-.073	-.096	-.782	-.056	-.024	-.088	-.349	-.022
TotDurFall	-.015	-.014	-.138	-.105	.017	-.153	-.885	-.028
MRateFall	.892	.146	.103	.165	.003	.059	.090	.188
GreatRateFall	.273	.069	.056	.052	.929	.036	.001	.004
BFI	-.296	-.720	.087	.333	-.128	-.303	-.094	.117
FFI	.296	.720	-.087	-.333	.128	.303	.094	-.117
MDBF	.301	.018	.195	.285	-.027	-.048	-.046	.716
HS(10)Num	.671	.124	-.112	.033	.268	-.061	.126	.415
LS(0.1)Num	-.032	.320	.305	-.660	-.038	-.024	-.041	-.307

Extraction Method: Principal Component Analysis.

Rotation Method: Varimax with Kaiser Normalization.

a. Rotation converged in 12 iterations.

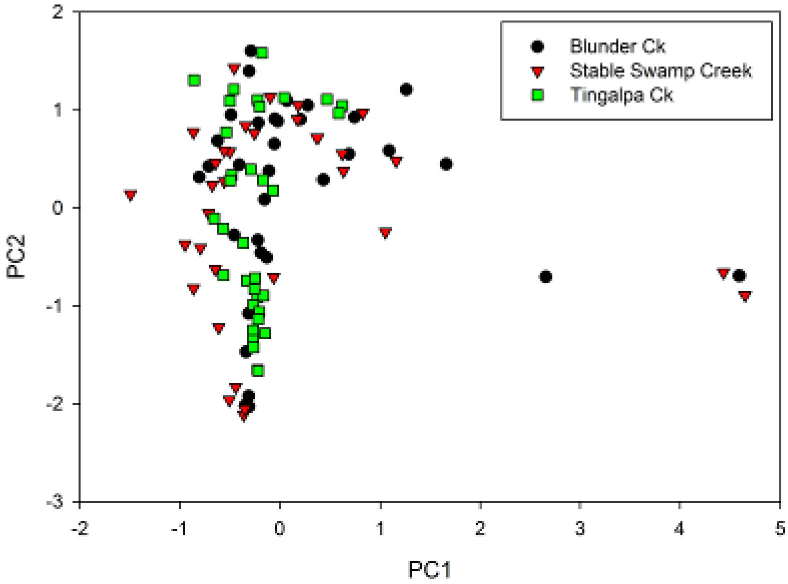


Figure 3-7: Initial solution for Principal Components Analysis (Varimax Rotation) showing PC1 and PC2 with summary hydrological data for each month coded by site. PC1 is positively associated with mean daily flow, rates of rise and fall, whereas PC2 is associated with Base Flow Index (BFI) and Flood Flow Index (FFI).

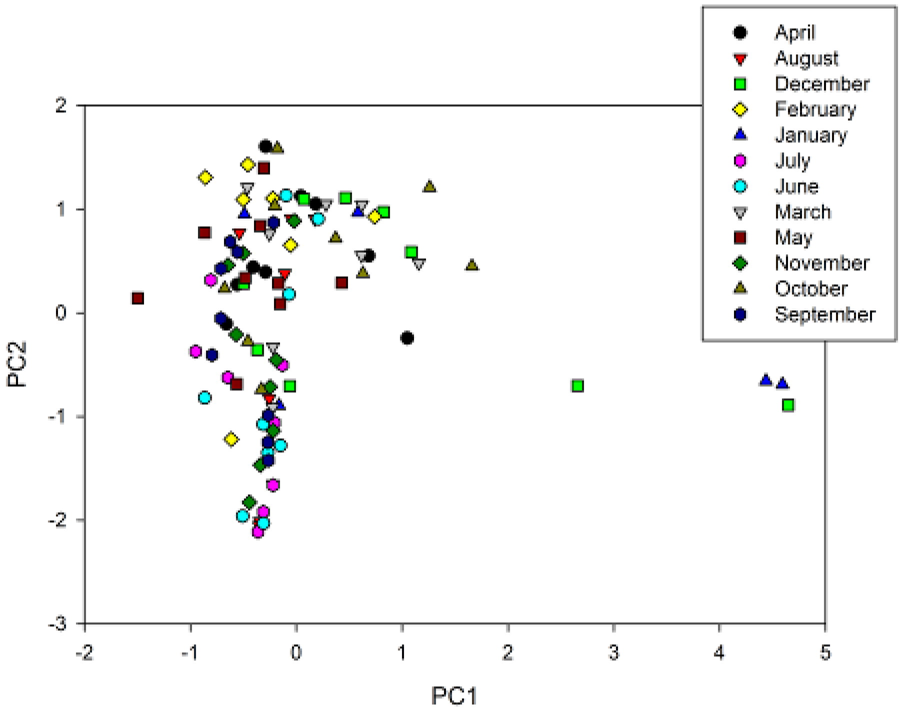


Figure 3-8: Initial solution for Principal Components Analysis (Varimax Rotation) showing PC1 and PC2 with summary hydrological data for each month coded by month.

Table 3-3: PCA with Varimax Rotation on reduced dataset with Dec 2010 and Jan 2011 from both Blunder Creek and Stable Swamp Creek removed from the dataset.

Rotated Component Matrix[a]

	Component							
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**
Variation Explained	35	14	9	7.4	6.9	5.6	3.6	3.3
Min	.218	.836	-.184	.087	.142	.053	.146	-.043
Max	.153	.111	.204	.912	.051	.090	-.003	.032
P 10	.230	.895	-.121	.097	.070	-.043	.132	-.055
P 90	.668	.198	.185	.279	.120	.364	-.110	-.027
MDF	.480	.265	.243	.728	.097	.183	-.058	-.002
Med	.233	.723	.125	.098	.023	.326	.068	-.107
CV	.212	-.041	.666	.351	.029	-.127	.066	.397
STD	.837	.103	.186	.327	.159	.135	.013	-.012
Skw	.076	-.134	.171	.156	-.068	-.046	-.124	.892
Zer	-.087	-.124	.034	-.093	-.114	-.053	-.018	.942
HS(5)Num	.583	.220	.156	.110	-.062	.625	-.045	-.016
LS(0.5)Num	.538	.437	.569	.051	.193	-.069	-.101	-.156
LS(0.5)Long	-.341	-.435	-.591	-.054	-.060	-.444	.106	.171
LS(0.5)Peak	.298	.677	.163	.029	.187	-.035	.020	-.094
LS(0.5)MeanDur	-.381	-.430	-.648	-.093	-.116	-.302	.078	.187
LS(0.5)TotDur	-.233	-.300	-.329	.003	.137	-.798	.084	.115
NumRise	.132	.150	-.011	.083	.921	-.067	.015	-.171
MMagRise	.885	.147	.206	.242	-.006	.199	.075	.003
MDurRise	-.031	-.057	.013	-.020	-.548	-.037	.766	-.143
TotDurRise	.093	.082	-.008	.047	.273	-.003	.855	-.310
MRateRise	.549	.232	.364	.350	.026	.116	.051	.045
GreatRatRise	.615	.012	.041	.698	.112	.060	.019	.015
NumFall	.050	.169	.037	.079	.930	-.094	-.075	-.144
MMagFall	.884	.140	.170	.282	.014	.188	.075	-.018
MDurFall	-.056	-.132	-.096	-.044	-.803	-.048	-.360	-.098
TotDurFall	-.038	-.080	-.023	.035	-.126	-.023	-.883	-.157
MRateFall	.743	.251	.337	-.039	.099	.115	.104	.066
GreatRateFall	.244	.007	.006	.928	.051	.061	.002	.031
BFI	-.302	.228	-.751	-.140	.051	-.237	-.073	-.239
FFI	.302	-.228	.751	.140	-.051	.237	.073	.239
MDBF	-.065	.712	-.012	.019	.159	.134	-.141	-.089
HS(10)Num	.380	-.010	-.033	.261	-.028	.735	.126	-.009
LS(0.1)Num	.145	-.511	.540	-.023	.283	-.241	-.074	-.152

Extraction Method: Principal Component Analysis.
Rotation Method: Varimax with Kaiser Normalization.
a. Rotation converged in 9 iterations.

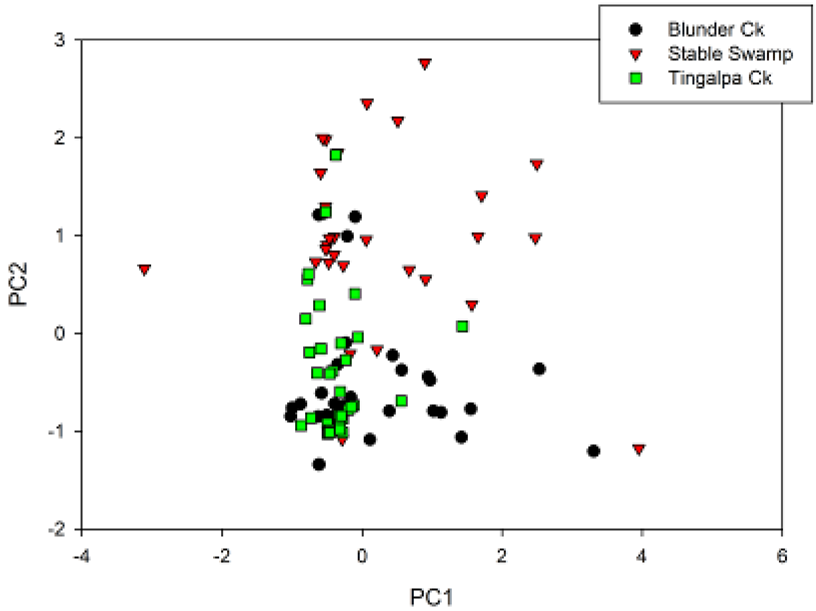


Figure 3-9: **Solution for reduced Principal Components Analysis (Varimax Rotation) showing PC1 and PC2 with summary hydrological data for each month coded by site. PC1 is positively associated with magnitude of rate of rise and fall while PC2 with aspects of minimum flow (Table 3-3).**

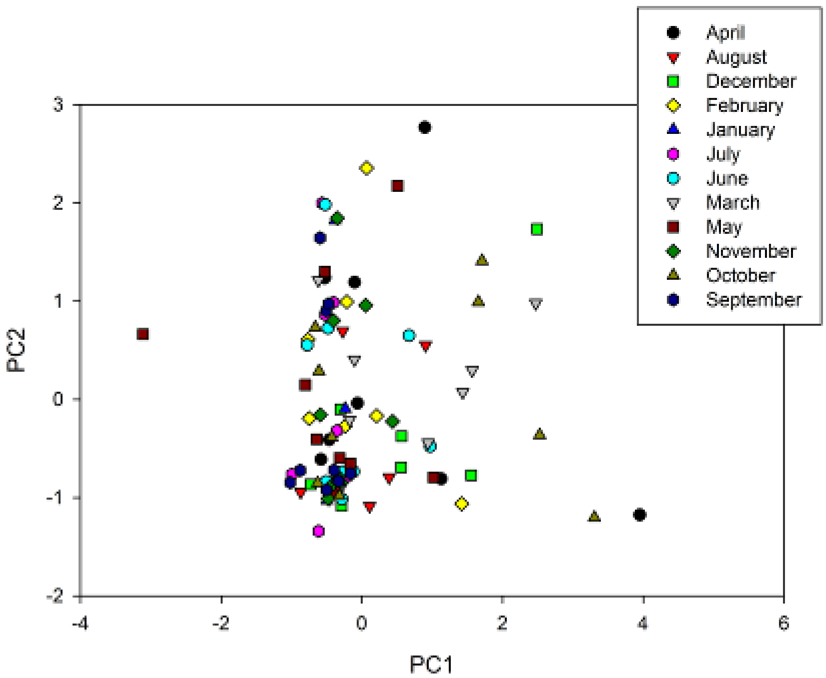


Figure 3-10: **Solution for reduced Principal Components Analysis (Varimax Rotation) showing PC1 and PC2 with summary hydrological data for each month coded by month.**

When individual metrics were explored separately there was no significant difference in the number of rises in each month ($F_{11,60}$=1.507, p>0.05) but a significant difference between sites ($F_{2,60}$=16.283, p<0.001) with no significant interaction ($F_{21,60}$=1.14, p>0.05). The forested Tingalpa Creek had significantly fewer rises than both the urbanised Stable Swamp Creek and the tributary to Blunder Creek (Figure 3-11). For mean daily flow there was no significant interaction between site and month ($F_{21,60}$=0.809, p>0.05), but a significant difference between months ($F_{11,60}$=2.15, p=0.03) and between sites ($F_{2,60}$=5.828, p<0.01). The mean daily flow was significantly higher in Stable Swamp Creek than either Tingalpa Creek or the tributary to Blunder Creek (Figure 3-12).

For the metrics that describe the rise and fall of flow events there were significant differences between months ($F_{11,60}$=2.185, p<0.05) and sites ($F_{2,60}$=3.791, P<0.05) but no significant interaction ($F_{21,60}$=0.537, p>0.05), with Tingalpa Creek having significantly smaller magnitudes of flow pulse rises than Stable Swamp creek (Figure 3-13). Across all sites the magnitude of the rises was smaller during the drier months (May – September) compared with the seasonally wet months (December – March). A similar pattern was found for the mean magnitude of the falling limb of a flow pulse with a significant difference between sites ($F_{2,60}$=4.161, p<0.05) and between months ($F_{11,60}$=2.597, p<0.01) but no significant interaction between site and month ($F_{21,60}$=0.690, p>0.05). Again the difference was between Tingalpa Creek and Stable Swamp Creek, with Tingalpa having the reduced magnitude of fall (Figure 3-14). In both cases the magnitudes of rise and falls are following that expected of urban streams, with the highly urbanised Stable Swamp Creek having significantly greater magnitudes of rise and fall than the forested Tingalpa Creek.

Component 2 of the PCA separated sample months on their low flow conditions with Stable Swamp Creek having much higher 'low flows' than either Tingalpa Creek or the Blunder Creek tributary. This was confirmed when focussing on the two low flow metrics specifically. For minimum flow (Min) there was a significant difference between sites ($F_{2,60}$=65.966, p<0.001) but not between months ($F_{11,60}$=0.289, p>0.05) with no interaction ($F_{21,60}$=0.726, p>0.05); with Stable Swamp Creek having significantly higher minimum flows then either Tingalpa Creek or the Blunder Creek tributary (Figure 3-15). The same pattern was observed for the 10th percentile of flows (P10). Interestingly it is the higher base flows in Stable Swamp creek during the drier winter months that are dominant.

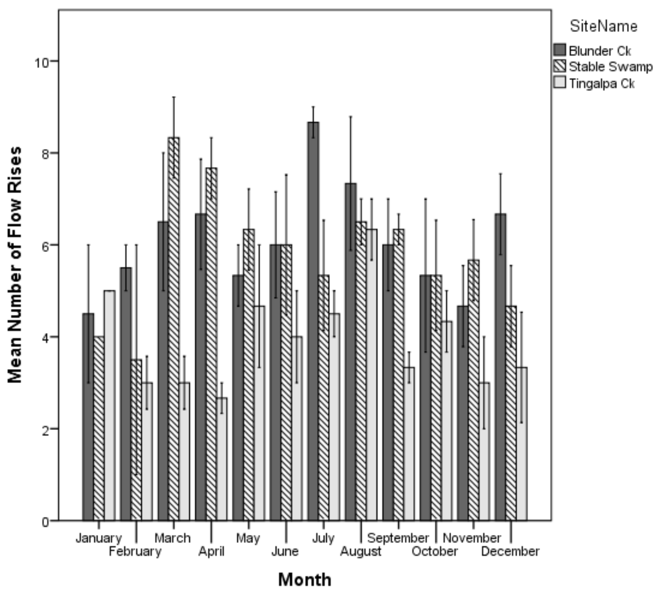


Figure 3-11: Mean number of rises (±SE) for months by site.

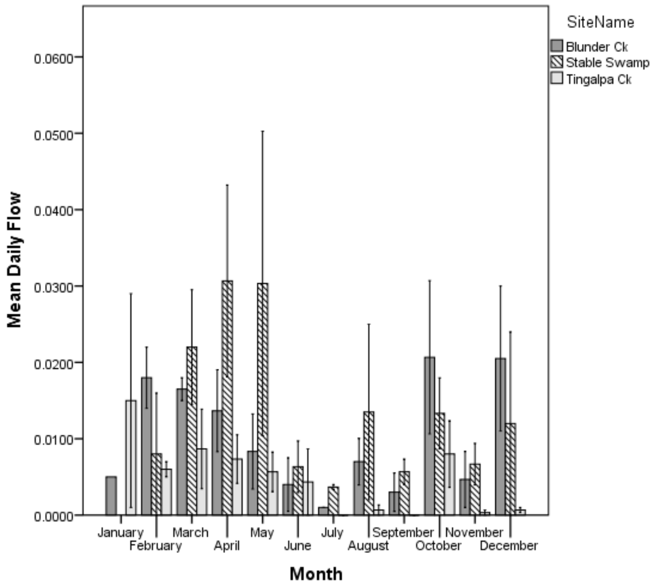


Figure 3-12: Mean daily flow (±SE) for months by site (ML/day).

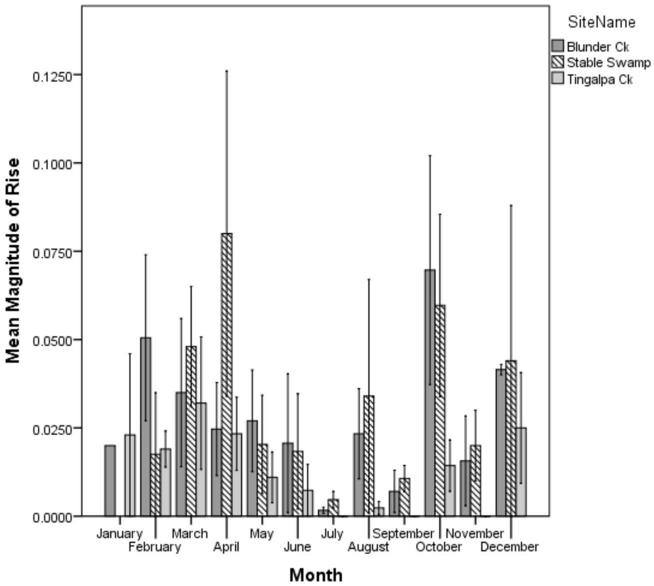


Figure 3-13: Mean magnitude of pulse rise (ML/day) (±SE) for months by site.

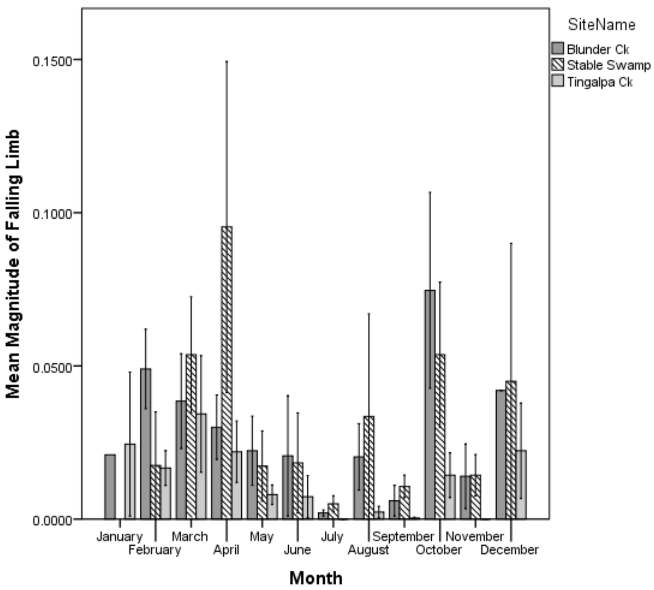


Figure 3-14: Mean (±SE) magnitude of falling limb (ML/day).

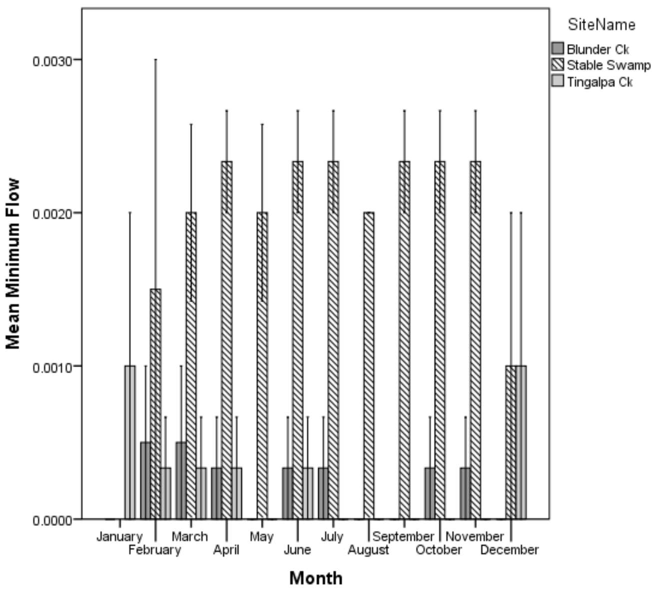


Figure 3-15: Mean (±SE) minimum flow by month and site (ML/day).

3.3.3. Conclusions

There were differences between sites that could be related to urbanisation. The two more urbanised sites, Stable Swamp Creek and the tributary of Blunder Creek have a higher number of flow rises across nearly all months, higher average daily flows and higher rates of fall during flow recession. The one parameter that was markedly different to that expected was the minimum daily flow and the base-flow index; the highly urbanised Stable Swamp Creek had continual baseflow throughout the year which would have mitigated many of the water quality impacts of urbanisation. It isn’t clear what is causing the continual baseflows at the Stable Swamp Creek site, however there is a wetland not far upstream from the gauging station which may act like a ‘sponge’ holding water during wet times and continually releasing it downstream (Figure 3-16). There may also be considerable water entering the creek from local urban use.

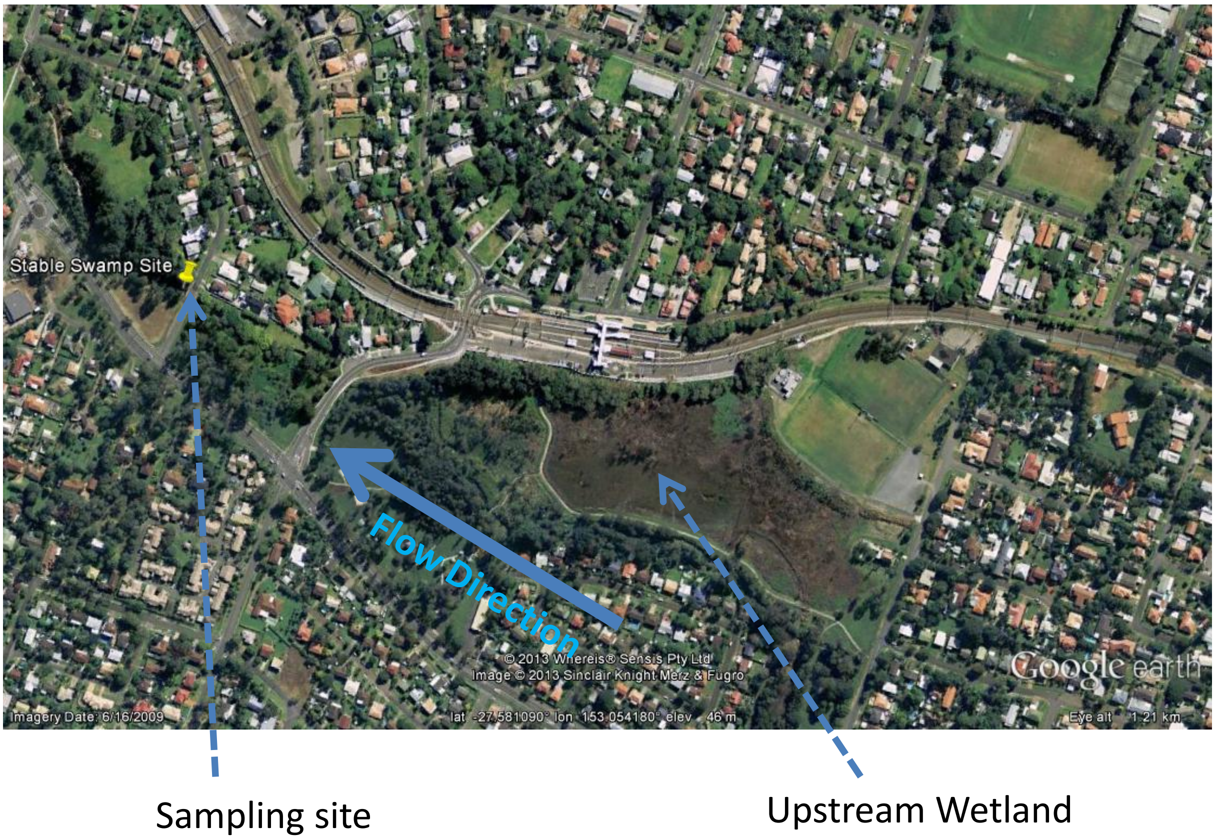


Figure 3-16: Position of the Stable Swamp Creek sampling site and gauging station in comparison with the upstream wetland.

3.4. Water Quality Changes

3.4.1. Background

Urbanisation is known to have a strong influence on the water quality of streams. Increased impervious area, direct piping of stormwater to the channel and reduced riparian cover act to increase the background levels of nutrients, suspended sediments and pollutants to urban streams as well as increase the daily range in dissolved oxygen and temperature (Walsh *et al*. 2005).

Our conceptual model (Figure 1-2) suggested water quality changes between forested and urbanised catchments across most flow conditions. Under the low flow conditions typical of spring and early summer we would expect reduced dissolved oxygen conditions in all streams, however, urbanised streams are likely to have lower dissolved oxygen than forested streams due to higher loads of organic carbon and consequently higher respiration rates. Under the higher flows of autumn and early winter (and the lower water temperatures in winter) we would expect higher dissolved oxygen levels across all sites.

In this section we used the daily logged water quality data from the three sites for a period between 2009 and 2012 to explore differences in a number of water quality parameters. We hypothesised that our two more urban sites (the tributary on Blunder Creek and Stable Swamp Creek) would have poorer water quality as measured by conductivity, pH, turbidity and dissolved oxygen levels and more variable 24 hour ranges of both dissolved oxygen and temperature, consistent with the conceptual understanding of urbanisation.

3.4.2. Methods

A Sonde was installed in each catchments at the same site as the gauge (for full details see Chowdhury *et al.* 2013); these Sondes measured pH, dissolved oxygen, turbidity and electrical conductivity at hourly intervals.

3.4.3. Results and Discussion

3.4.3.1 Dissolved Oxygen

Dissolved oxygen levels at the logger varied throughout the year with mean daily levels generally much lower in the summer months (spring and summer) compared with the winter months across all sites (Figure 3-17; Figure 3-18). Interestingly, the highly urbanised Stable Swamp Creek had much higher mean daily dissolved oxygen levels across most of the year compared to the other sites (Figure 3-18), possibly reflecting the higher baseflow at this site. The daily range in dissolved oxygen is often used as an indicator of stream health (Bunn *et al.* 2010), with lower daily ranges seen to be more typical of healthy streams. Across our sites the Blunder Creek tributary WSUD site and the urbanised Stable Swamp Creek had larger daily ranges across all seasons compared with the forested Tingalpa Creek (

Figure 3-19; Figure 3-20). These dissolved oxygen results are consistent with how we understand water quality in urban creeks; generally dissolved oxygen is much lower across the seasons due to higher organic loads from impervious surfaces and the general inputs from urbanisation (Blunder Creek) unless water flow is maintained (Stable Swamp Creek). Interestingly, daily mean dissolved oxygen levels in the forested Tingalpa Creek were often extremely low, particularly in the summer (Figure 3-17) and this possibly reflects the position of the logger in a pool within the creek, the high organic load from riparian leaf litter and the often low flow conditions between storm events. This suggests that at times during the year conditions in the forested catchment in terms of dissolved oxygen levels will be just as severe as in the urbanised catchments. However, in the urbanised catchments there are likely to be other drivers of poor health, including increased loads of organic carbon, higher conductivity, heavy metals and other pollutants that will exacerbate low levels of dissolved oxygen.

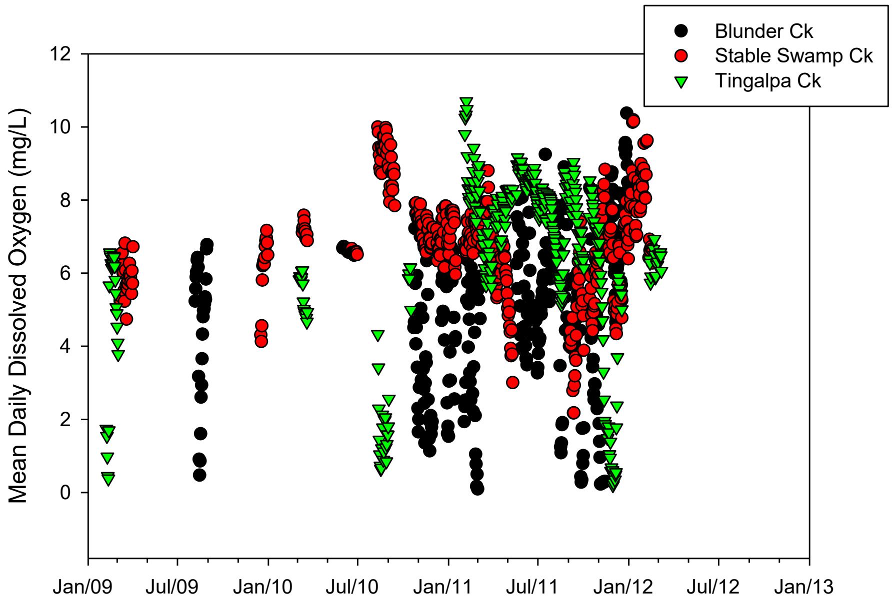


Figure 3-17: Mean daily dissolved oxygen (mg/L) levels across all three sites for the period of data logging (January 2009 – June 2012), note that continual logging of all sites only commenced in January 2011.

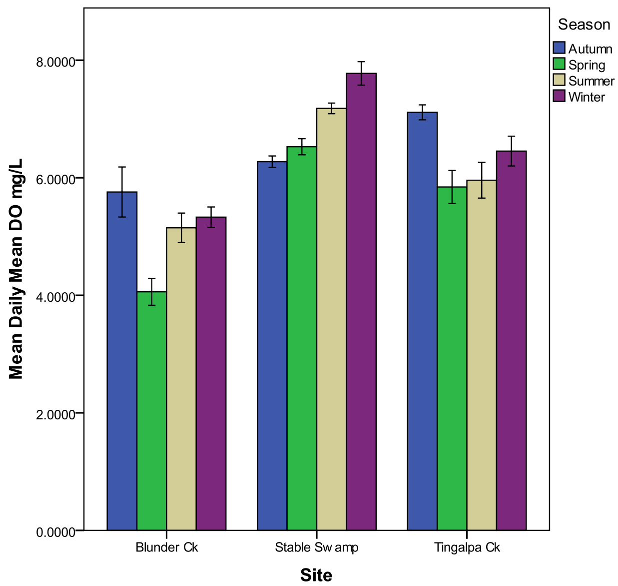


Figure 3-18: Mean daily dissolved oxygen (mg/L) levels across all three sites grouped by season (Autumn: March-May; Winter: June-August; Spring: September-November; Summer: December-February).

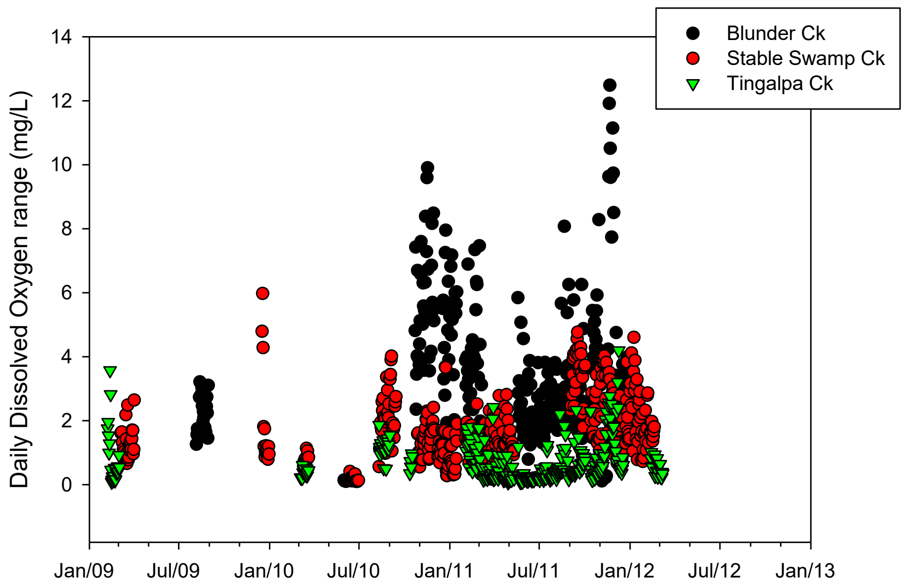


Figure 3-19: Daily dissolved oxygen range (mg/L) across all three sites for the period of data logging (January 2009 – June 2012), note that continual logging of all sites only commenced in January 2011.

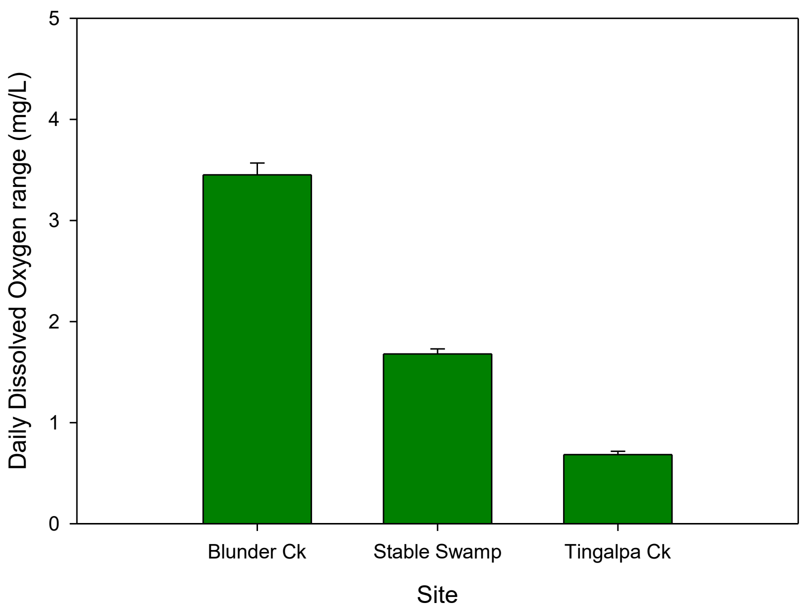


Figure 3-20: Mean daily dissolved oxygen range (mg/L) (± SE) across all three sites for the period of record.

3.4.3.2 Water Temperature

Water temperature at the three sites varied across the year as expected, with maximum daily temperatures around 10-15°C during the winter months and summer temperatures between 25-30°C (Figure 3-21). Across the three sites higher maximum daily temperatures were found at the two urban sites, the Blunder Creek tributary and Stable Swamp Creek compared with forested Tingalpa Creek (Figure 3-22). Daily temperature range is also considered to be a good indicator of stream health (Bunn *et al.* 2010), with higher temperature ranges indicative of poorer stream health. Higher temperature ranges are usually the result of reduced riparian cover which is common in urban streams. Although the three sites in this study were chosen as they had good riparian cover the expectation of lower daily temperature range in forested streams was still met (Figure 3-23) with forested Tingalpa Creek having a mean daily temperature range of nearly 1.5°C lower than the two urban sites (Figure 3-24). The much higher daily temperature range in the Blunder Creek tributary and Stable Swamp Creek possibly reflects discontinuous riparian cover, although the actual sample and logger sites had good riparian cover, water temperature was likely influenced by a lack of upstream cover and the inability of the available cover at the site to actively cool the water temperature.

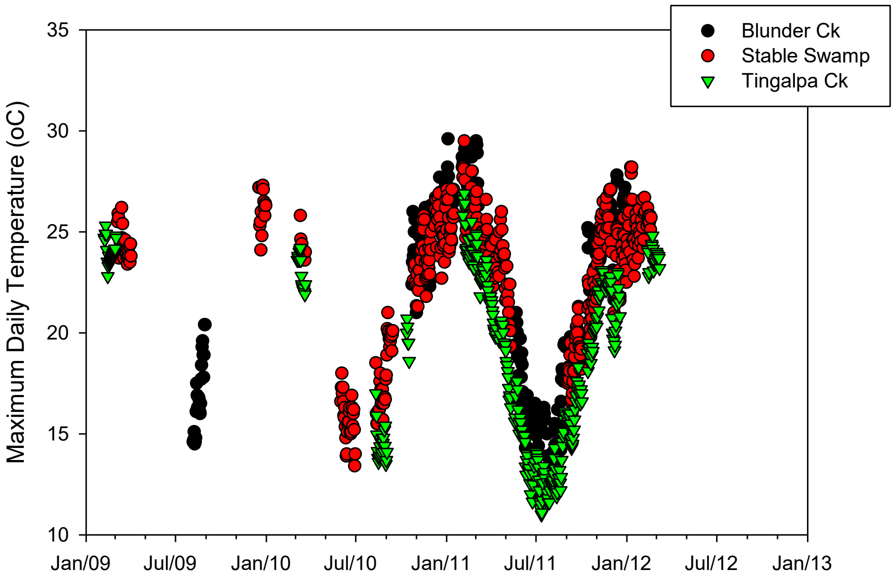


Figure 3-21: Maximum daily temperature (°C) across all three sites for the period of data logging (January 2009 – June 2012), note that continual logging of all sites only commenced in January 2011.

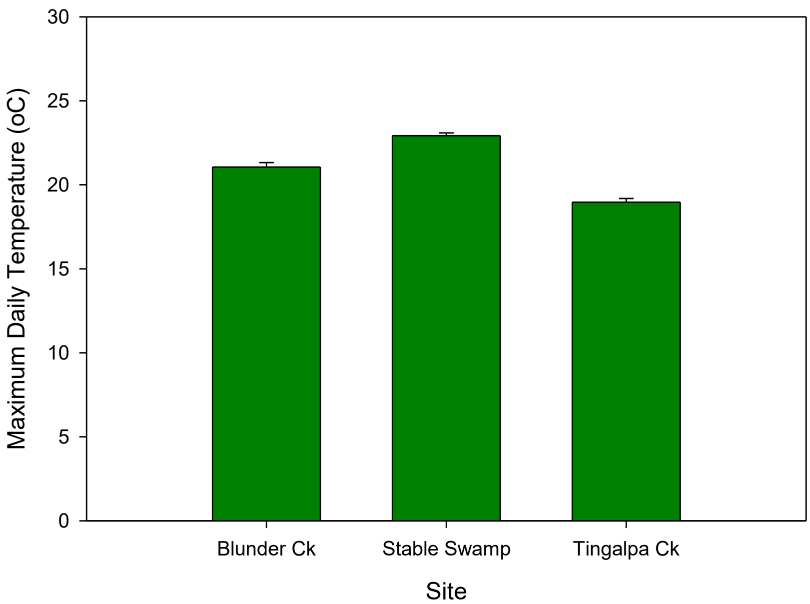


Figure 3-22: Maximum daily temperature (°C) (± SE) across all three sites for the period of record.

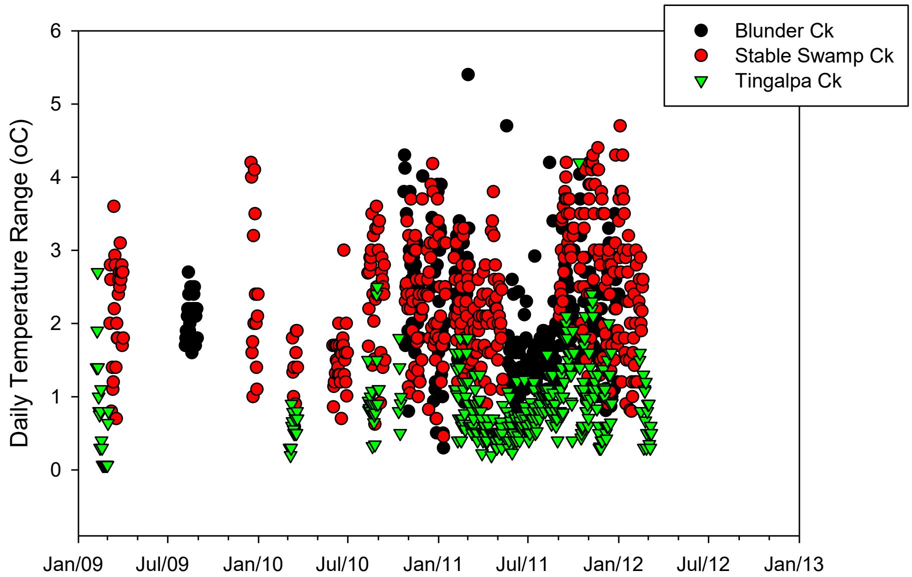


Figure 3-23: Daily temperature range (°C) across all three sites for the period of data logging (January 2009 – June 2012), note that continual logging of all sites only commenced in January 2011.

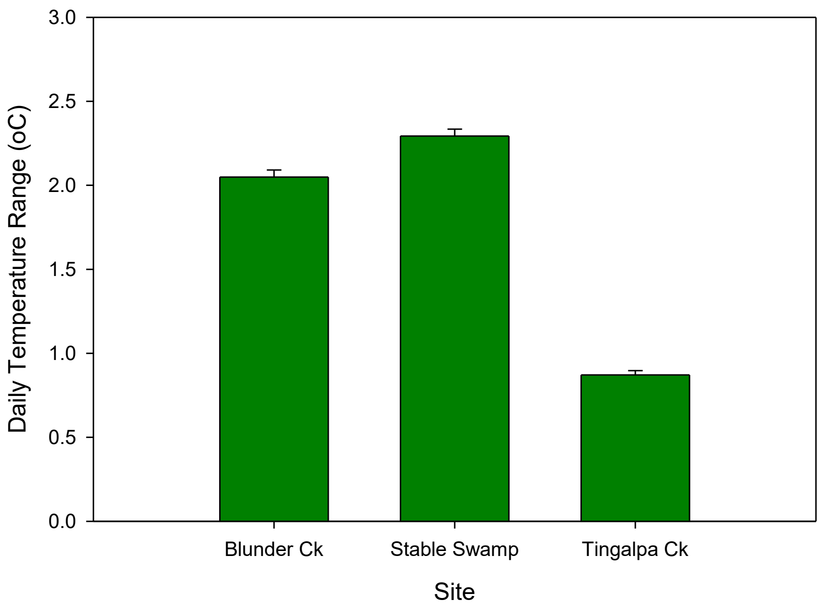


Figure 3-24: Mean daily temperature range (°C) (± SE) across all three sites for the period of record.

3.4.3.3 Turbidity

High turbidity levels in streams are often a reflection of catchment disturbance after high flow events and can impact streams through increased in siltation of instream habitat when flow ceases and / or clogging of gills of aquatic fauna during periods of high turbidity. Across the three sites turbidity levels were generally low; however, the increased rainfall in the summer of 2011 saw a peak in turbidity in the forested Tingalpa Creek and not in the urban streams. This is interesting as upstream of the water quality logger is natural forest, and suggests that under conditions of extreme soil saturation even natural forested catchments can suffer erosion and cause sedimentation in streams. Interestingly, peaks were not seen in either Stable Swamp Creek or the Blunder Creek tributary during this high rainfall period.

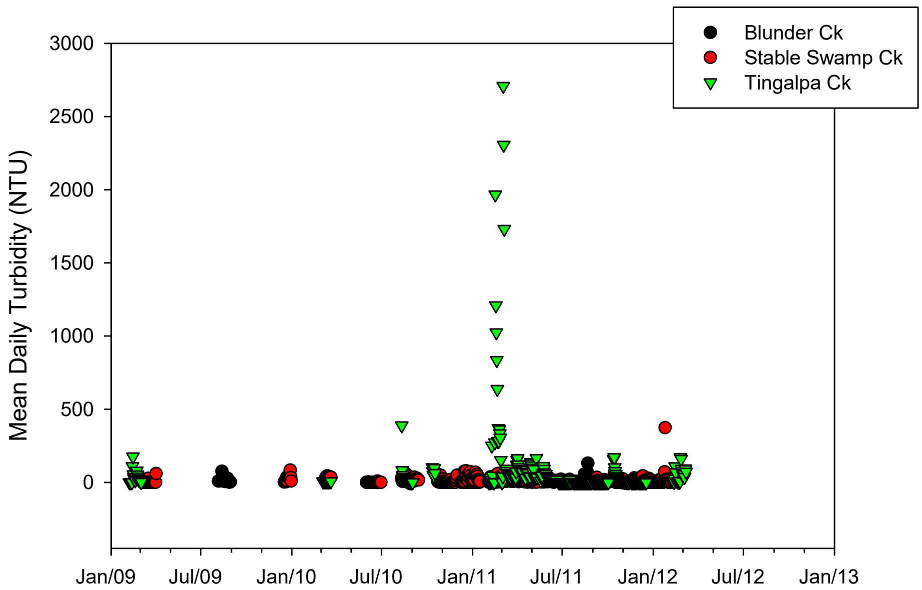


Figure 3-25: Mean daily turbidity (NTU) across all three sites for the period of data logging (January 2009 – June 2012), note that continual logging of all sites only commenced in January 2011.

3.4.3.4 Conductivity

High levels of water conductivity can also pose risks for stream health. Increased conductivity has been associated with landuse change, from forested to agricultural or urban landscapes, and we would therefore expect higher conductivities in the more urbanised streams. Across the three sites the role of urbanisation in increased stream conductivity was not clear. Highest conductivity values were observed in the tributary to Blunder Creek (Figure 3-26; Figure 3-27) with little apparent difference across the logged sampling time between Stable Swamp Creek and Tingalpa Creek, however, when overall means are compared Stable Swamp Creek has a long-term lower conductivity than Tingalpa Creek (Figure 3-27). This lower overall conductivity in the highly urbanised Stable Swamp Creek most likely reflects the continual baseflows in that system (Section 3.3) compared with forested Tingalpa Creek where low flows and isolation of instream pools is common during the drier months. The much higher conductivity observed in the tributary to Blunder Creek is likely the result of considerable iron floc entering the stream at that point (Figure 3-28). A more detailed investigation into the occurrence of iron floc in urbanised Brisbane streams can be found in Section 4.

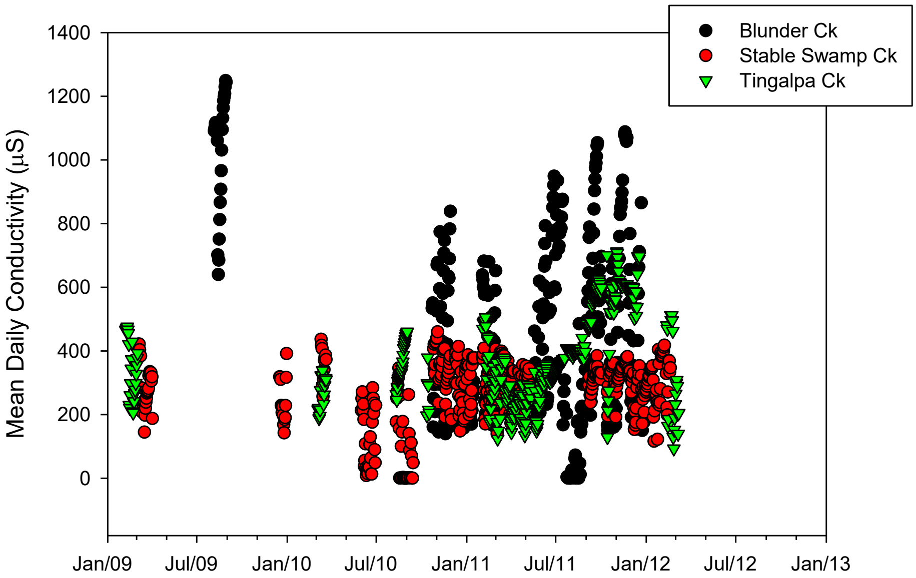


Figure 3-26: Mean daily conductivity (µS) across all three sites for the period of data logging (January 2009 – June 2012), note that continual logging of all sites only commenced in January 2011.

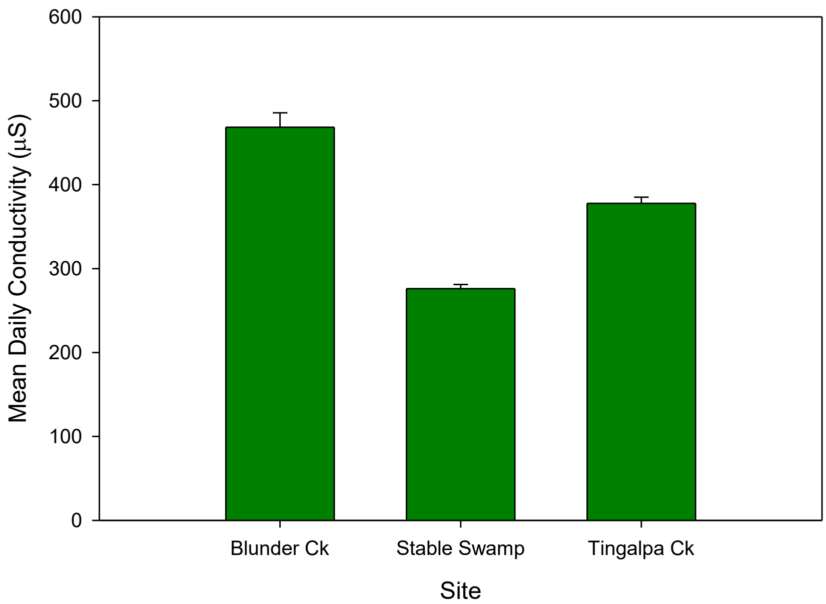


Figure 3-27: Mean daily conductivity (µS) (± SE) across all three sites for the period of record.

Figure 3-28: Iron (Fe) floc in the tributary to Blunder Creek upstream of the water quality logger, June 2012.

3.4.3.5 pH

pH was not expected to vary greatly between the three streams as pH is not a strong driver of stream ecosystem health in SEQ (Bunn *et al*. 2010; Sheldon *et al*. 2012) and stream acidification is not a major issue. However, given the issue of iron floc at the Blunder Creek tributary site we expected to see low pH levels at that site. However, this was not the case (Figure 3-29), the water quality logger at the Blunder Creek tributary site was positioned about 100 m downstream of the macroinvertebrate sampling point (Figure 3-28) and it appears that the extremely low pH levels recorded at that site (pH<6.5) did not last for a great distance downstream, unlike the influence of increased conductivity (Figure 3-26). Interestingly, low pH values were more often observed in the forested Tingalpa Creek compared to the other two sites (Figure 3-29); the logger at the Tingalpa site was positioned in a pool and it is likely that under low flow conditions increased respiration within the pool may have caused these drops in pH.

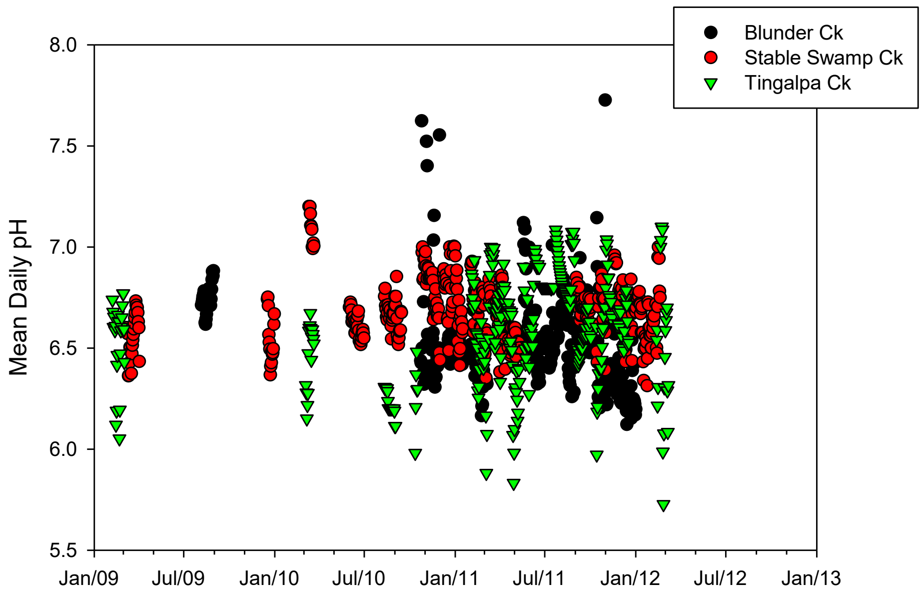


Figure 3-29: Mean daily pH across all three sites for the period of data logging (January 2009 – June 2012), note that continual logging of all sites only commenced in January 2011.

3.4.4. Conclusion

As with the hydrological metrics, there were clear water quality differences between sites that could be related to urbanisation. The two more urbanised sites, Stable Swamp Creek and the tributary of Blunder Creek had larger daily ranges in dissolved oxygen (DO) and temperature. The higher conductivity observed at the site on the tributary to Blunder Creek was likely associated with groundwater inputs containing dissolved Fe which becomes insoluble in contact with oxygen, the associated iron rich salts in the water causing a local increase in conductivity. The multiple water quality impacts at the Blunder Creek tributary site, high daily ranges in DO and temperature, high conductivity and locally high pH make it a potentially a harsh environment for macroinvertebrates. Water quality parameters in the forested Tingalpa Creek, while occasionally harsh (pH: Figure 3-29) are more often typical of forested catchments with low daily ranges in DO and temperature and low conductivity.

3.5. Macroinvertebrate Community Composition

3.5.1. Background

Urban streams are considered harsh environments for the survival of in-stream biota. Hydrologically they differ from their forested counterparts through increased magnitudes of peak flows and ‘flashy’ hydrographs (Paul and Meyer 2001), these hydrological changes are associated with the direct piping of water into stream networks and the high amounts of impervious surfaces in upstream catchments (Walsh *et al.* 2005; Walsh and Kunapo 2009). These hydrological changes are often associated with a reduction in the diversity of instream habitat through increased erosion. The runoff associated with draining impervious surfaces and with stormwater through the drain network affects water quality and can introduce a range of pollutants urban streams including heavy metals, petroleum aromatic hydrocarbons, pesticides, nutrients and sediments (Wenger *et al.* 2009). Thus, urban streams are often typified by depauperate assemblages of tolerant macroinvertebrate taxa (Walsh *et al.* 2005, Utz *et al.* 2009). In this section we explored changes in macroinvertebrate composition across a 12-month period, covering a wet and dry phase, in a forested stream (Tingalpa Creek) and an urbanised stream (Stable Swamp Creek).

3.5.2. Methods

3.5.2.1 Sampling Methods

At each site distinct microhabitats were identified, these included “riffle”, “pool”, “macrophyte (aquatic plants)” and “snag (woody debris)”. Replicate samples, where possible, were then sampled from each available microhabitat by sweeping a 250 µm mesh pond net over an area approximating 5 m^2 for 20 seconds. Samples were preserved in 70% ethanol and in the laboratory the organisms were hand-sorted, enumerated and identified as far as practicable. Unidentified specimens were recorded as separate taxa (e.g. “tiny Zygoptera”).

3.5.2.2 Analysis

Due to the extreme water quality conditions at the Blunder Creek tributary site there was reduced macroinvetebrate diversity, so to explore the questions associated with urbanisation we focussed on Stable Swamp Creek and Tingalpa Creek.

For all samples combined, Analysis of Variance (ANOVA) models were used to explore differences for each site between sampling months for a range of summary diversity metrics. For ‘pool’ and ‘riffle’ habitats separately differences between the factors of “month” and “site” were explored using two-way ANOVA for the same summary diversity metrics. All analyses were conducted in the SPSS Software Package Version 20.

To explore changes in assemblage composition between sites and through time, data (abundance of each macroinvertebrate family in each sample) were square root transformed before analysis to reduce the influence of extremely abundant and rare taxa and the Bray-Curtis similarity coefficient used to measure assemblage differences between samples. All taxa occurring in one or more samples were retained in the subsequent analyses. The multivariate statistics were conducted in the Primer-E Software package. Initially Analysis of Similarities (ANOSIM) was used to explore differences between habitat types with these differences displayed visually using Non-metric Multi-Dimensional Scaling (MDS). Based on this analysis ‘Pool’ and ‘Riffle’ habitats were then compared separately for each stream and month, again using ANOSIM with differences displayed using MDS. The Similarity Percentages procedure (SIMPER) was used to extract the different taxa responsible for the observed differences between sites and sampling months.

To explore the influence of background water quality and hydrology on assemblage composition, monthly median values for a range of water quality parameters and hydrology metrics were calculated (Table 3-4). The BIOENV routine in Primer-E was used to explore which variables from each of the two environmental datasets best explained the observed patterns in the macroinvertebrate assemblage dataset.

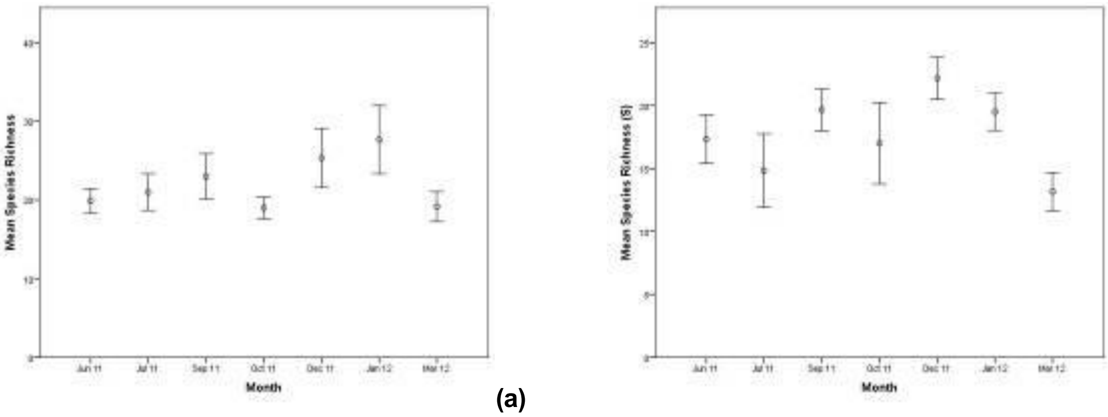
Table 3-4: Summary water quality parameters and hydrology metrics for which monthly medians were calculated from the longer term logged data.

Summary Water Quality Parameters	Summary Hydrology Metrics
Daily Mean DO mg/L	Max
Daily Minimum DO mg/L	MDF
Daily range DO mg/L	CV
Mean electrical conductivity (µS)	NumRise
Daily mean pH	MMagRise
Daily Mean Turbidity (NTU)	MRateRise
Daily Maximum Turbidity (NTU)	NumFall
Mean Temperature (°C)	MMagFall
Maximum Temperature (°C)	MRateFall
Range Temperature (°C)	BFI
	FFI

3.5.3. Results

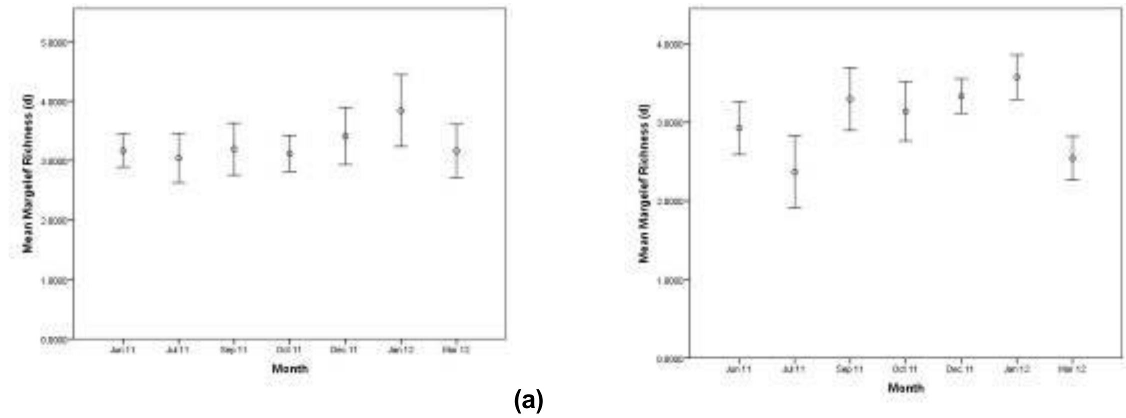
3.5.3.1 Diversity of all Habitats

Across all habitats, there was no significant difference in species richness between months for Stable Swamp Creek ($F_{6,45}$=1.498, p>0.05) or Tingalpa Creek ($F_{6,45}$=1.898, p>0.05) (Figure 3-30). A similar pattern was seen for Margalef richness (d), with no significant difference between sampling months for either Stable Swamp Creek ($F_{6,45}$=0.4, p>0.05) or Tingalpa Creek ($F_{6,45}$= 1.491, p>0.05) (Figure 3-31). Again, there was no significant difference between sampling months for Pielou Evenness (J') in either Stable Swamp Creek ($F_{6,45=}$ 1.69, p>0.05) or Tingalpa Creek ($F_{6,45}$= 2.113, p>0.05) (Figure 3-32), or for the Shannon-Weiner measure of diversity (H') (Stable Swamp: $F_{6,45}$= 1.177, p>0.05; Tingalpa: $F_{6,45}$= 1.278, p>0.05) (Figure 3-33), or for the Simpsons measure of diversity (1-λ) (Stable Swamp: $F_{6,45}$= 1.273, p>0.05; Tingalpa: $F_{6,45}$= 1.625, p>0.05) (Figure 3-34).



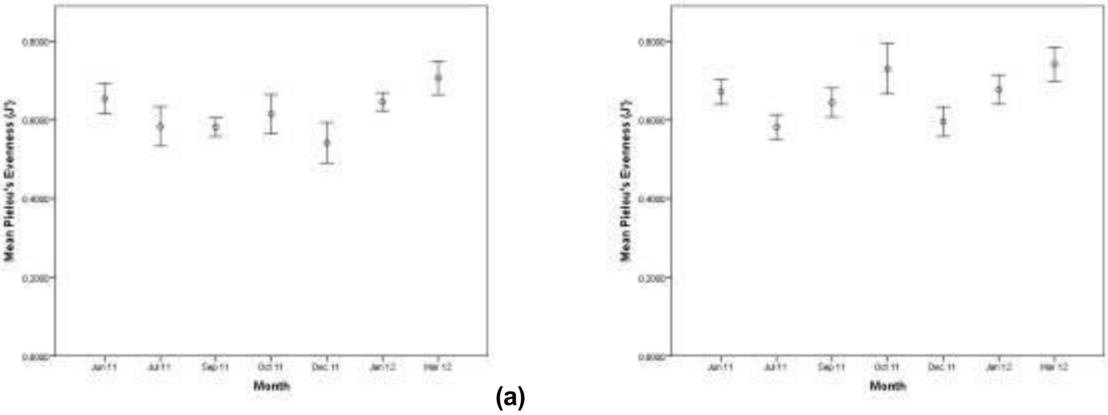
(a) (b)

Figure 3-30: Mean (±SE) Species Richness (S) between months across all habitats in (a) Stable Swamp Creek and (b) Tingalpa Creek.



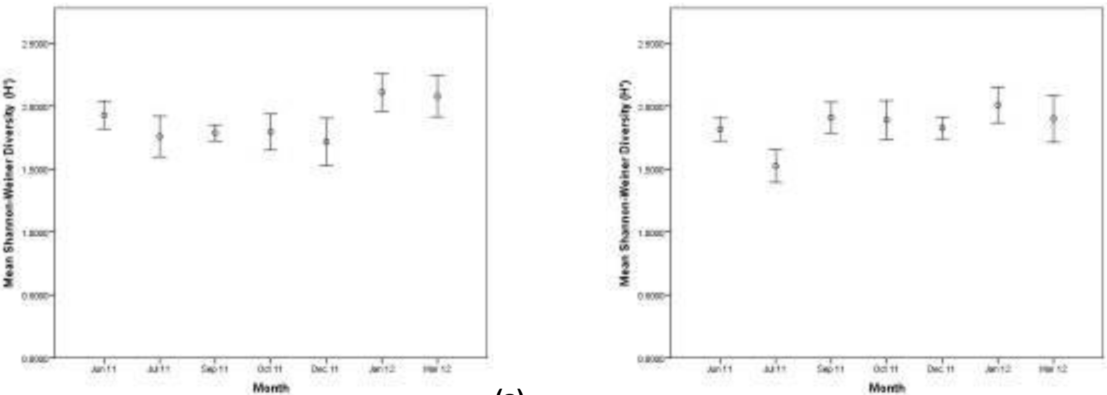
(a) (b)

Figure 3-31: Mean (±SE) Margalef Richness (d) between months across all habitats in (a) Stable Swamp Creek and (b) Tingalpa Creek.



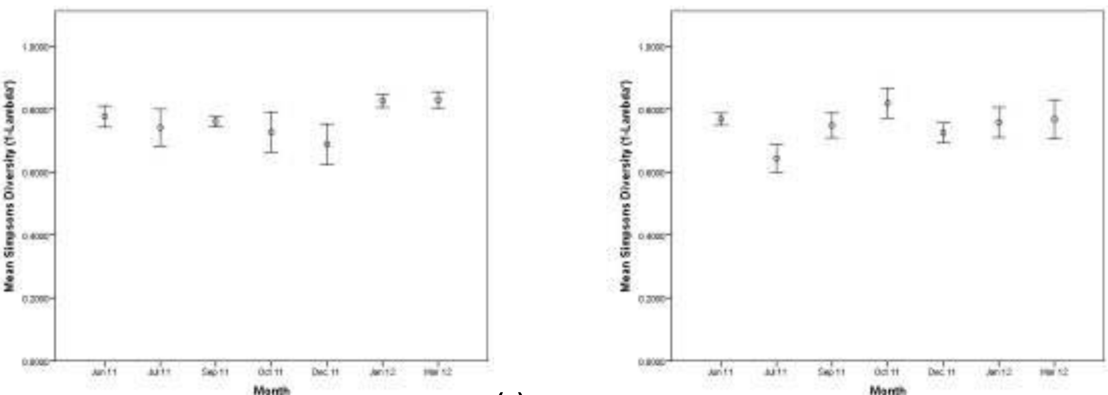
(a) (b)

Figure 3-32: Mean (±SE) Pielou's Evenness (J') between months across all habitats in (a) Stable Swamp Creek and (b) Tingalpa Creek.



(a) (b)

Figure 3-33: Mean (±SE) Shannon-Weiner diversity (H') between months across all habitats in (a) Stable Swamp Creek and (b) Tingalpa Creek.



(a) (b)

Figure 3-34: Mean (±SE) Simpsons diversity (1-λ) between months across all habitats in (a) Stable Swamp Creek and (b) Tingalpa Creek.

3.5.3.2 Diversity across Riffle Habitats

When comparing riffle habitats only between sites and months for Species Richness (S) there was a significant difference between months ($F_{6,42} = 2.729$, $p<0.05$), but not between sites ($F_{1,42} = 0.514$, $p>0.05$) with no site by month interaction ($F_{6,42} = 1.381$, $p>0.05$) (Figure 3-35). Riffle habitats in Tingalpa Creek had, in general, greater variation in species richness between sampling months compared with Stable Swamp Creek, July and October 2011 had markedly reduced species richness with December 2011 having the highest richness (Table 3-5). The highest total abundances were found in Stable Swamp creek in July and September 2011 and January 2012 (Table 3-5).

There was no significant differences across months or sites for Pielou's Evennes (Figure 3-36); however, there was a significant difference between sampling months for Shannon-Weiner Diversity (H') for riffle habitats only ($F_{6,42} = 4.361$, $p<0.01$), with no difference between sites ($F_{1,42} = 0.255$, $p>0.05$) and a non-significant interaction between site and sampling month ($F_{6,42} = 2.054$, $p>0.05$) (Figure 3-37). In Tingalpa Creek the summer months of January and March had higher diversity than the winter months, particularly July.

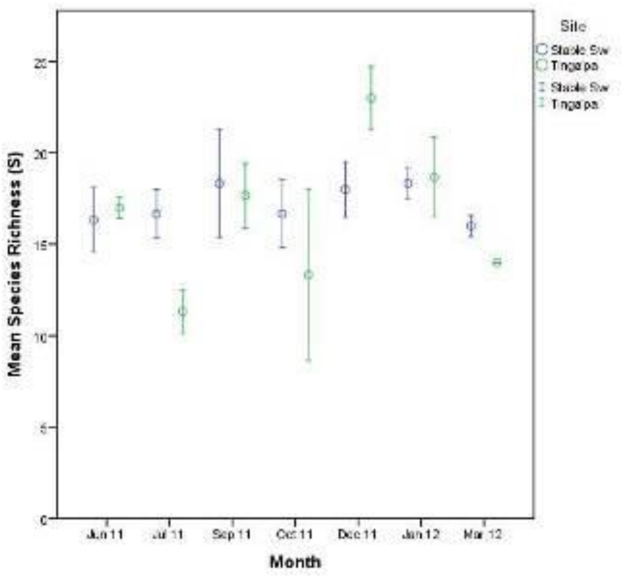


Figure 3-35: Mean (±SE) for Species Richness (S) between sampling months for riffle habitats only of Stable Swamp Creek and Tingalpa Creek.

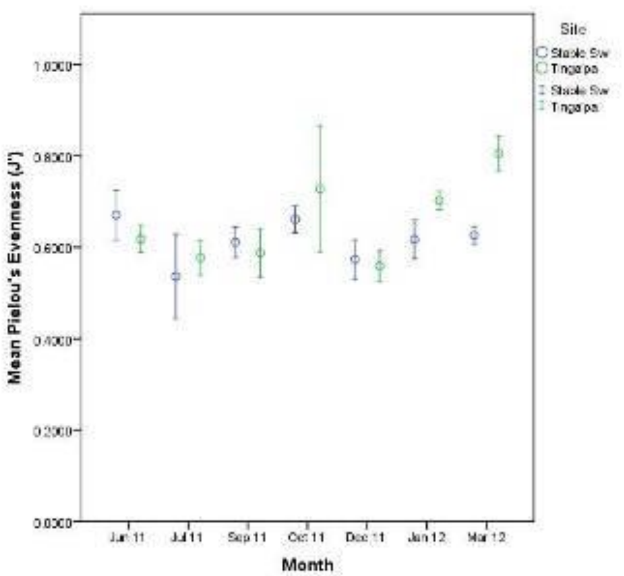


Figure 3-36: Mean (±SE) for Pielou's Evenness (J') between sampling months for riffle habitats only of Stable Swamp Creek and Tingalpa Creek.

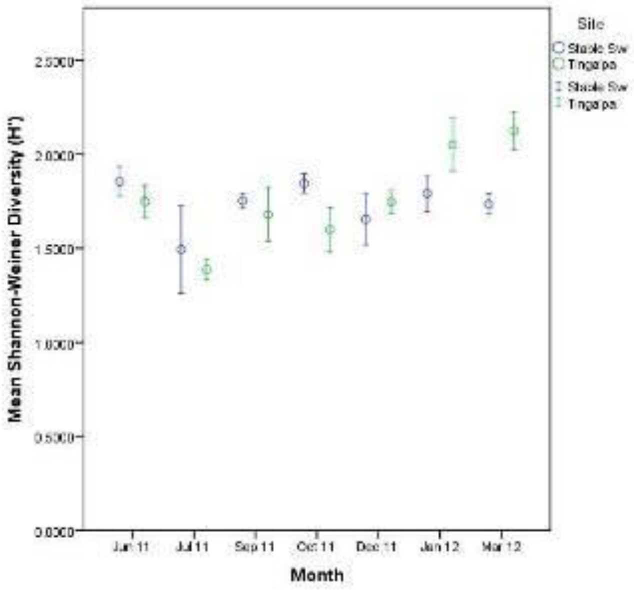


Figure 3-37: Mean (±SE) for Shannon-Weiner Diversity (H') between sampling months for riffle habitats only of Stable Swamp Creek and Tingalpa Creek.

Table 3-5: Mean (±SE) for a range of diversity measures for each site by sampled month for riffle habitats only.

	Species Richness (S)	Abundance (N)	Margalef Richness (d)	Pielou's Evenness (J')	Shannon-Weiner Diversity (H')	Simpsons Diversity (1-λ)
Stable Swamp Creek						
June 2011	16.3 (1.76)	481 (46)	2.48 (0.25)	0.67 (.05)	1.85 (.08)	0.79 (0.03)
July 2011	16.7 (1.33)	1498 (446)	2.16 (0.13)	0.54 (0.09)	1.49 (0.23)	0.67 (0.11)
September 2011	18.3 (2.96)	1427 (463)	2.39 (0.3)	0.61 (0.03)	1.75 (0.04)	0.77 (0.02)
October 2011	16.7 (1.85)	452 (59)	2.58 (0.35)	0.66 (0.03)	1.84 (0.05)	0.78 (0.02)
December 2011	18.0 (1.53)	867 (127)	2.52 (0.21)	0.57 (0.04)	1.65 (0.14)	0.72 (0.05)
January 2012	18.3 (0.88)	1394 (620)	2.51 (0.12)	0.62 (0.04)	1.79 (0.09)	0.78 (0.02)
March 2012	16 (0.58)	926 (272)	2.22 (0.08)	0.63 (0.02)	1.73 (0.05)	0.77 (0.01)
Tingalpa Creek						
June 2011	17.0 (0.57)	699 (206)	2.48 (0.11)	0.62 (0.03)	1.75 (0.08)	0.74 (0.04)
July 2011	11.33 (1.2)	524 (167)	1.67 (0.14)	0.58 (0.04)	1.39 (0.05)	0.62 (0.04)
September 2011	17.67 (1.76)	821 (315)	2.52 (0.13)	0.58 (0.05)	1.68 (0.15)	0.68 (0.06)
October 2011	13.3 (4.7)	299 (203)	2.64 (0.25)	0.73 (0.14)	1.59 (0.12)	0.79 (0.11)
December 2011	23.0 (1.73)	571 (102)	3.51 (0.38)	0.56 (0.03)	1.74 (0.06)	0.67 (0.03)
January 2012	18.7 (2.18)	233 (109)	3.42 (0.53)	0.70 (0.02)	2.05 (0.14)	0.79 (0.02)
March 2012	14 (0)	202 (51)	2.49 (0.14)	0.81 (0.04)	2.12 (0.10)	0.84 (0.03)

3.5.3.3 Diversity across all Pool Habitats

When comparing pool habitats only for differences between sites and months, there was a significant difference in species richness (S) between months ($F_{6,42}$=3.176, p<0.05) and between sites ($F_{1,42}$ = 25.373, p<0.001) with no interaction ($F_{6,42}$=1.143, p>0.05); however, Tingalpa Creek generally had lower species richness in pool habitats compared with Stable Swamp Creek (Figure 3-38). Although there was no significant difference between sites ($F_{1,42}$=3.215, p>0.05) or sampling months ($F_{6,42}$ = 0.249, p>0.05), evenness tended to be lower in Stable Swamp Creek compared with Tingalpa Creek, suggesting more of the abundance in the urban creek was occurring in fewer taxa (Table 3-6; Figure 3-39; Figure 3-40).

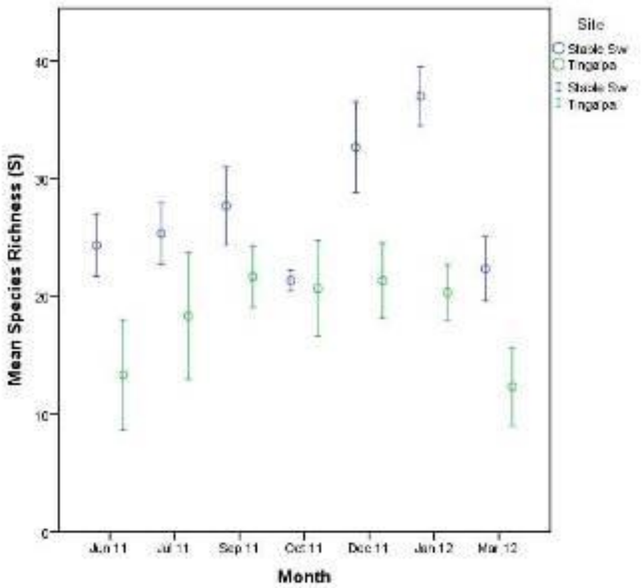


Figure 3-38: Mean (±SE) for Species Richness (S) between sampling months for pool habitats only of Stable Swamp Creek and Tingalpa Creek.

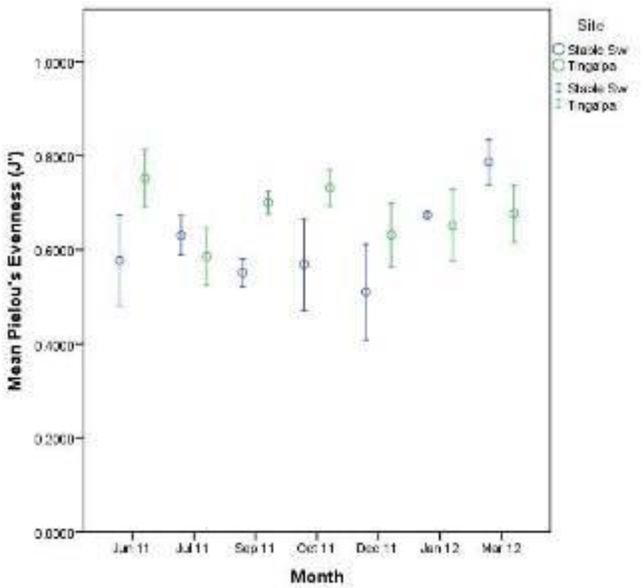


Figure 3-39: Mean (±SE) for Pielou's Evenness (J') between sampling months for pool habitats only of Stable Swamp Creek and Tingalpa Creek.

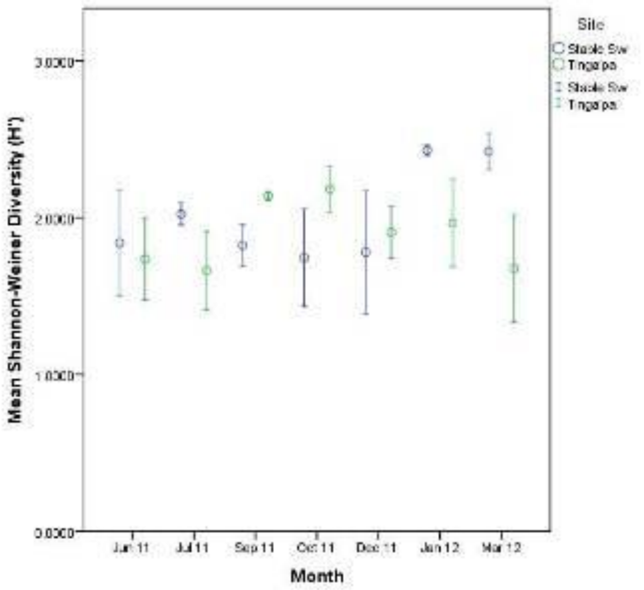


Figure 3-40: Mean (±SE) for Shannon-Weiner Diversity (H') between sampling months for pool habitats only of Stable Swamp Creek and Tingalpa Creek.

Table 3-6: Mean (±SE) for a range of diversity measures for each site by sampled month for pool habitats only.

	Species Richness (S)	Abundance (N)	Margalef Richness (d)	Pielou's Evenness (J')	Shannon-Weiner Diversity (H')	Simpsons Diversity (1-λ)
Stable Swamp						
June 2011	24.3 (2.67)	530 (210)	3.88 (0.58)	.57 (0.09)	1.84 (0.33)	0.72 (0.09)
July 2011	25.3 (2.6)	507 (108)	3.92 (0.28)	0.63 (0.04)	2.02 (0.07)	0.81 (0.02)
September 2011	27.7 (3.33)	807 (112)	3.98 (0.47)	0.55 (0.03)	1.82 (0.13)	0.75 (0.03)
October 2011	21.3 (0.8)	304 (91)	3.65 (0.22)	0.57 (0.09)	1.74 (0.31)	0.67 (0.13)
December 2011	32.7 (3.84)	1649 (225)	4.30 (0.56)	0.51 (0.10)	1.78 (0.39)	0.66 (0.13)
January 2012	37.0 (2.52)	1105 (241)	5.16 (0.23)	0.67 (0.01)	2.43 (0.03)	0.86 (0.01)
March 2012	22.3 (2.73)	283 (162)	4.09 (0.34)	0.78 (0.05)	2.42 (0.11)	0.88 (0.02)
Tingalpa						
June 2011	13.3 (4.7)	162 (70)	2.39 (0.74)	0.75 (0.06)	1.73 (0.26)	0.78 (0.03)
July 2011	18.3 (5.36)	273 (103)	3.07 (0.74)	0.58 (0.06)	1.66 (0.25)	0.67 (0.08)
September 2011	21.7 (2.6)	159 (16)	4.07 (0.43)	0.70 (0.02)	2.14 (0.03)	0.81 (0.02)
October 2011	20.7 (4.05)	225 (55)	3.63 (0.63)	0.73 (0.04)	2.18 (0.14)	0.84 (0.02)
December 2011	21.3 (3.18)	880 (367)	3.15 (0.25)	0.63 (0.07)	1.91 (0.17)	0.77 (0.04)
January 2012	20.3 (2.33)	181 (37)	3.72 (0.30)	0.65 (0.07)	1.96 (0.28)	0.72 (0.09)
March 2012	12.3 (3.28)	77 (25)	2.58 (0.59)	0.67 (0.06)	1.67 (0.34)	0.69 (0.11)

3.5.3.4 Assemblage Composition: General Patterns

Two-way PERMANOVA with factors of 'site' and 'habitat' nested within 'site' suggested no significant difference between the assemblage composition of the two sites (pseudo $F_{1,89} = 3.22$, $p>0.05$). However, there was a significant difference in the assemblage composition of the habitats within each site (pseudo $F_{4,89} = 10.58$, $p<0.001$) (Figure 3-41). Samples from Stable Swamp Creek, regardless of habitat, had a within site similarity of 56% and were dominated by chironomids, with the Chironominae contributing 16% to the within site similarity and the Orthocladiinae contributing a further 12%, worms (Oligochaeta) contributed a further 11%, with the Simuliidae (blackfly larvae) contributing a further 8%, these four taxa contributed a cumulative 46% to the within site similarity for Stable Swamp. In contrast, the within sites similarity of all samples from Tingalpa Creek was 47%, with the Leptophlebiidae mayflies contributing 27% to the within-group similarity and the Aytidae shrimps a further 11%. Of the long-term logged water quality variables, median daily mean electrical conductivity (µS) and median daily temperature range (°C) explained 48% of the variation in assemblage composition. The long-term hydrology metrics, however, explained little variation in the assemblage composition, with only 14.6% explained by a combination of the rate of pulse rise (MRateRise), rate of pulse fall (MRateFall) and the Base Flow Index (BDI).

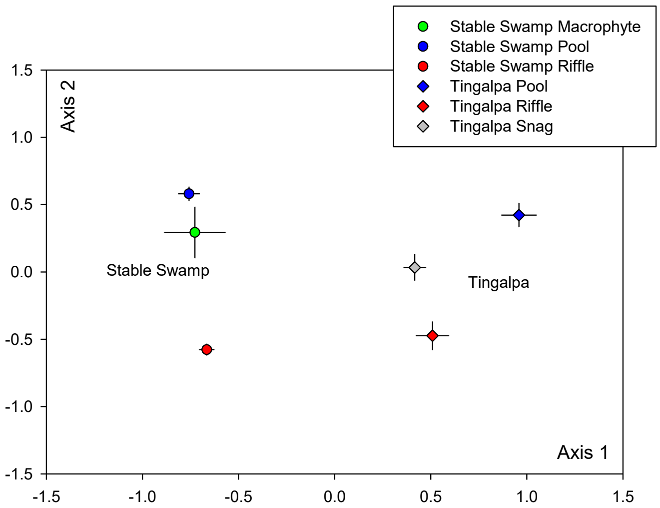


Figure 3-41: Two-dimensional MDS Ordination plot based on the Bray-Curtis Similarity measure showing the centroids (mean X and Y coordinates) for the different habitats within each site. Stress = 0.14.

3.5.3.5 Exploring Differences between Riffle Assemblages

Exploring differences across riffle habitats only, ANOSIM suggested a significant difference between Tingalpa Creek and Stable Swamp Creek (Global R = 0.921, p=0.001) and a significant difference between months across both sites (Global R = 0.404, p<0.001). In Stable Swamp Creek, the summer months (December, January and March) tended to cluster together and be quite distinct to the samples taken during winter, however the pattern was not as distinct in Tingalpa Creek (Figure 3-42). Riffle samples from Stable Swamp creek had an average similarity of 68% and were dominated by orthoclad chironomids (16%), the hydropsychid caddis Cheumatopsyche sp. (15%), blackfly larvae (Simuliidae) (13%) and the oligochaeta (13%), with these taxa cumulatively contributing to 57% of the riffle sample similarity at Stable Swamp. In contrast, riffle samples from Tingalpa Creek had an overall similarity of 58% and were dominated by mayflies in the family Leptophlebiidae (28%), all three subfamilies of chironomids (Chironominae, 9.5%; Tanypodinae, 9.5%; and Orthocladiinae 9.3%) and the hydropsychid caddis Cheumatopsyche sp. (9.1%), with these taxa cumulatively contributing to 65% of the within habitat similarity at Tingalpa Creek. For riffle habitats only, long-term water quality variables of mean conductivity (µS) and median daily temperature range (°C) explained 59.7% of the variation in assemblage composition. In the riffle habitats only, again 13.2% of the variation in assemblage composition was explained by a combination of the long-term hydrology metrics, rate of pulse rise (MRateRise), rate of pulse fall (MRateFall) and the Base Flow Index (BDI).

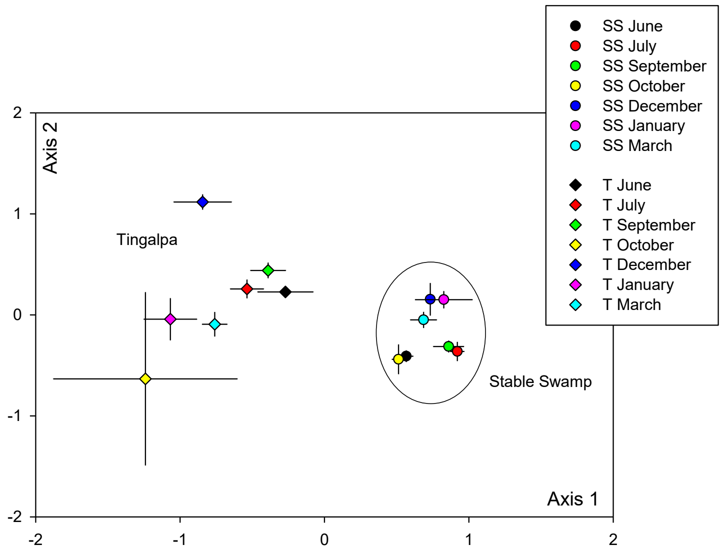


Figure 3-42: Two-dimensional MDS Ordination plot based on the Bray-Curtis Similarity measure showing the centroids (mean X and Y coordinates) for each sampling month at each site, riffle habitats only. Stress = 0.09.

3.5.3.6 Exploring Differences between Pool Assemblages

Exploring differences across pool habitats only, ANOSIM suggested a significant difference between Tingalpa Creek and Stable Swamp Creek (Global R = 0.968, p=0.001) and a significant difference between months across both sites (Global R = 0.44, p<0.001). In Stable Swamp Creek, the summer months (January and March) tended to cluster together as did the winter samples (June, July and September) (Figure 3-43). In Tingalpa Creek, the summer months (December and January) tended to be very distinct while all other months tended to be similar in assemblage composition (Figure 3-43). Pool samples from Stable Swamp Creek had an average similarity of 33% and were dominated by Hyroptilid caddisflies (46%), Leptocerid caddisflies (29%) and Ecnomid caddisflies (13%), with these taxa cumulatively contributing to 88% of the pool sample similarity at Stable Swamp. In contrast, pool samples from Tingalpa Creek had an overall similarity of 54% and were dominated by mayflies in the family Leptophlebiidae (57%), Leptocerid caddisflies (29%) and the Baetid mayflies (10%), with these taxa cumulatively contributing to 96% of the within habitat pool similarity at Tingalpa Creek. For pool habitats only, again the long-term water quality variables of mean conductivity (µS) and median daily temperature range (°C) explained 63.5% of the variation in assemblage composition. In the pool habitats only, 21.5% of the variation in assemblage composition was explained by a combination of the long-term hydrology metrics, rate of pulse rise (MRateRise) and the Base Flow Index (BDI).

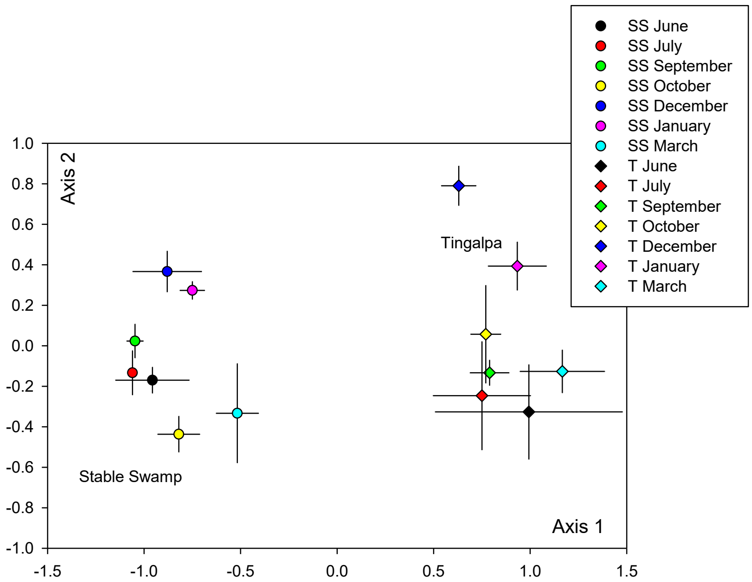


Figure 3-43: Two-dimensional MDS Ordination plot based on the Bray-Curtis Similarity measure showing the centroids (mean X and Y coordinates) for each sampling month at each site, pool habitats only. Stress = 0.09.

3.5.3.7 Diversity of EPT Taxa

The proportion of the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies) and Tripchoptera (caddisflies) – more commonly known as the EPT taxa – is a commonly used metric for assessing the health of streams. A higher proportion of EPT taxa present in the sample suggests 'better' stream health. The EPT taxa are relatively more sensitive to a range of pressures associated with poor stream health, including habitat degradation, reduced water quality, increased pollutants and changed hydrological regime. Thus, the diversity of the EPT taxa throughout the sampling period for both sites was compared. We predicted that the forested Tingalpa Creek would have a higher diversity of EPT taxa when compared with the urbanised Stable Swamp Creek, and that the diversity in Stable Swamp creek would be reduced by higher more variable flows in the summer months.

Across all sampled habitats, there were more EPT taxa in the forested Tingalpa Creek when compared with the urbanised Stable Swamp Creek (Figure 3-44). To explore the potential impact of flow and water quality, each site and habitat was considered separately. We expected that at the urban site the proportion of EPT taxa in the riffles would be reduced during the summer months when flows are likely to be higher, compared with the forested site where summer flows should have less impact. In the more stable 'pool' habitats of both streams we expected little change throughout the year. For pool habitats this was the case; there was no significant difference in species richness between the sampling months for either Stable Swamp Creek ($F_{6,13} = 0.995$, $p>0.05$) or for Tingalpa Creek ($F_{6,14} = 0.695$, $p>0.05$) (Figure 3-45). For the 'riffle' habitats, our predictions were correct, there were no significant changes in the number of EPT taxa in samples from different months from Tingalpa Creek ($F_{6,14} = 0.695$, $p>0.05$), however, there was a significant difference between sample months for the urbanised Stable Swamp Creek ($F_{6,14} = 4.111$, $p<0.05$) (Figure 3-46). Our predictions suggested that in Stable Swamp Creek there would be reduced EPT richness in the summer months, however, we found reduced richness in the winter months when flows are more stable, suggesting the possible influence of water quality as a driver for reduced diversity.

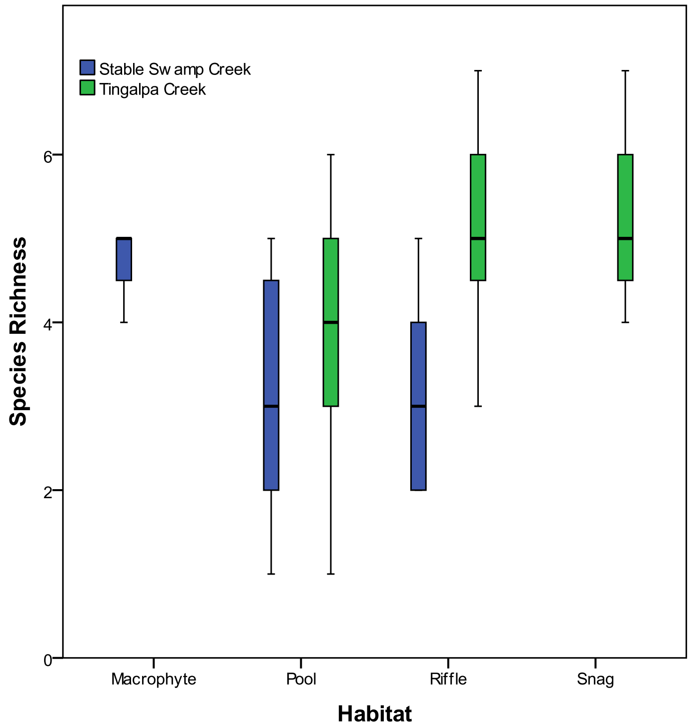


Figure 3-44: Species richness box plots (showing minimum, first quartile, median, third quartile, and maximum values) for the two sites (Tingalpa Creek – forested and Stable Swamp Creek – Urban) for each habitat type.

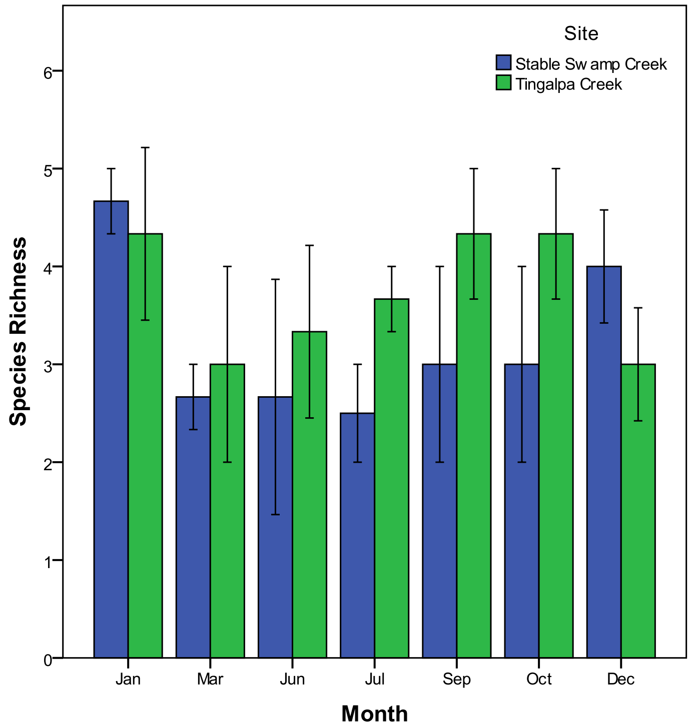


Figure 3-45: Mean (± SE) of species richness for the 'pool' habitat in the two streams across the sampling months.

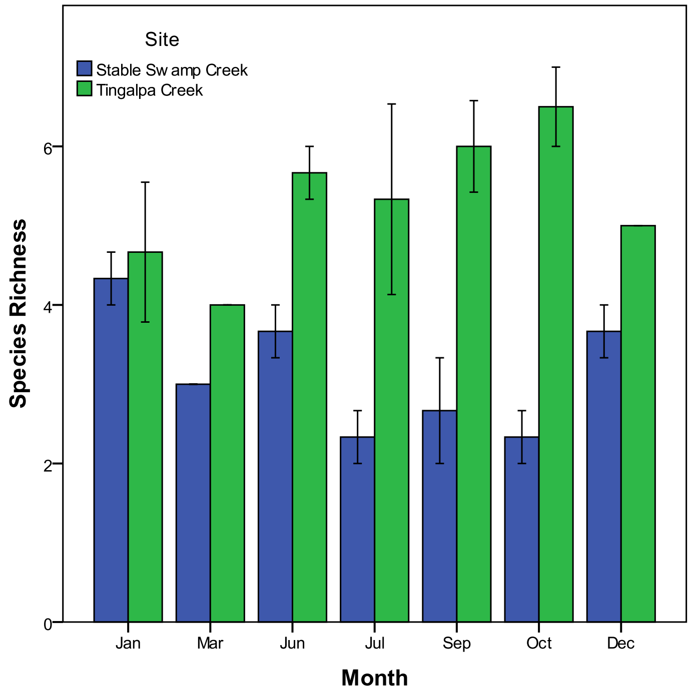


Figure 3-46: Mean (± SE) of species richness for the 'riffle' habitat in the two streams across the sampling months.

3.5.3.8 Habitat Availability

Urban streams are often characterised by extremely degraded instream habitat which can make it difficult to distinguish between reduced macroinvertebrate diversity owing to degraded habitat alone, or the combination of degraded habitat and changed hydrology. In this section, we surveyed the proportion of riffle and pool habitat in 1 km sections of streams across an urban gradient. We predicted that in urban streams there would be lower proportions of riffle habitat, but that where riffles did occur in urban streams insescts of the orders Trichoptera and Ephemeroptera would be found. This work was undertaken by Mr Lars Pelzer, a visiting undergraduate student in the International Water Centre.

Methods: 20 approximately 1km segments of different streams were surveyed using to estimate the ratio of riffles and pools. This was undertaken by walking along the stream and counting steps of a standard distance (10 steps = 10 meters) (Figure 3-47). Whenever riffles were found the presence or absence of EPT taxa was measured.

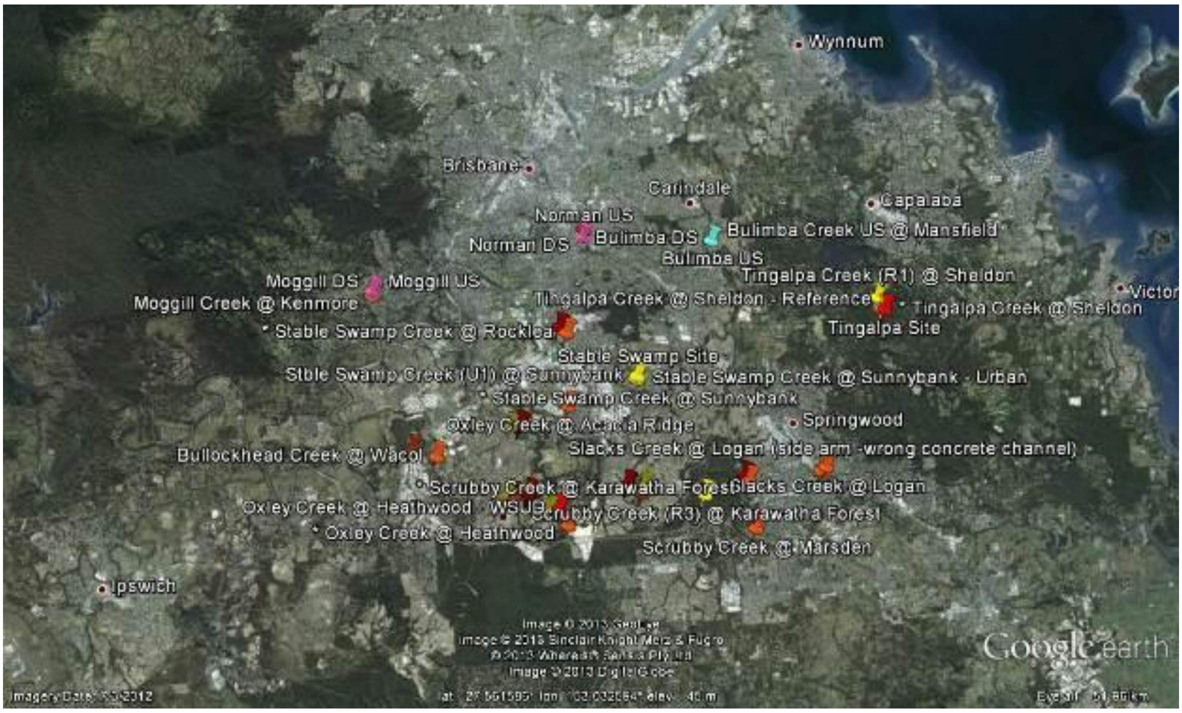


Figure 3-47: Position of sites in the urban area of Brisbane surveyed for pool-riffle habitat proportions.

Results: There was no significant difference in the proportion of riffles ($F_{1,12} = 0.005$, $p = 0.94$) or pools ($F_{1,12} = 0.52$, $p = 0.48$) between urban and forested streams (Figure 3-48). There was also no clear pattern in the presence of EPT taxa in riffles from different stream types. Ephemeroptera were observed in riffles from one forested site (Tingalpa Creek, Sheldon) and two urban sites (Norman Creek and Bulimba Creek). Trichoptera were observed in riffles at two of the three forested sites (Tingalpa Creek at Sheldon and Moggill Creek and Kenmore) and six of the urban sites (Table 3-7).

Table 3-7: Summary of the average percent riffle and pool habitats in 1km sections of streams across the Brisbane region.

Landuse	Site	EPT	Pool%	Riffle%
Reference	Tingalpa@Sheldon	E,T	91.99	8.01
Reference	Scrubby Creek@Karrawatha		100.00	0.00
Reference	Moggill Creek@Kenmore	T	86.64	13.36
Urban	Stable Swamp@Sunnybank	T	58.98	41.02
Urban	Stable Swamp@Rocklea		100.00	0.00
Urban	Boss Creek@Durack		0.00	0.00
Urban	Sheepstation Creek@Parkinson	T	75.48	24.52
Urban	Norman Creek@Greenslopes	E,T	21.99	2.33
Urban	Bulimba@Mansfield	E,T	97.94	2.06
Urban	Oxley Creek@Acacia Ridge		100.00	0.00
Urban	Scubby Creek@Marsden	T	99.37	0.63
Urban	Slacks Creek@Logan		100.00	0.00
Urban	Bullockhead Creek@Wacol		98.86	1.14
Urban	Sandy Creek@Wacol		100.00	0.00
WSUD	Blunder Creek@Forest lake		88.41	11.59
WSUD	Oxley Creek@Heathwood	T	98.81	1.19

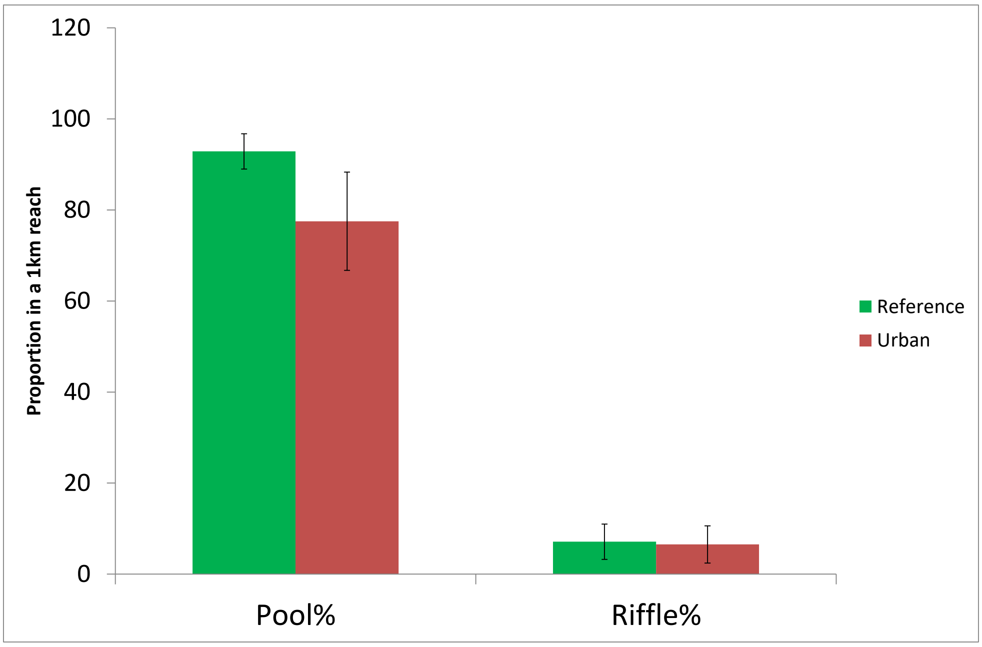


Figure 3-48: Comparative proportion of pools and riffles in 1km sections of urban and forested streams in the Brisbane area.

3.5.4. Conclusion

The focussed temporal study of macroinvertebrates across the highly urbanised Stable Swamp Creek compared with forested Tingalpa Creek suggested differences between the two that could be explained by urbanisation, but also differences that were intrinsic to the different stream types.

Species richness and other summary metrics were similar across both sites suggesting differences were likely to be at the assemblage composition level, rather than in terms of broad summary metrics. The assemblage composition of the two sites did differ, with the urbanised Stable Swamp Creek dominated by taxa common in degraded streams, including the midges (Chironominae) and worms (Oligochaeta). In comparison, the forested Tingalpa Creek assemblage was dominated by mayflies (Family Leptophlebiidae) and other insect orders. When the assemblage data was compared with the logged water quality and hydrology data, it was stream conductivity, daily temperature range and aspects of hydrograph shape (rates of rise and fall) that explained the most variation in assemblage differences between the two sites. These physical parameters are known to be heavily influences by urbanisation (Section 3.3 and Section 3.4).

When the 'pool' and 'riffle' habitats were explored separately, there was a greater turnover of species (variation in assemblage composition between months) in Tingalpa Creek compared with Stable Swamp Creek, which showed a similar assemblage composition throughout the year. This relative stability in Stable Swamp creek was contrary to our predictions where we expected extreme flashy summer flows to have a marked impact on assemblage composition. However, the Stable Swamp creek site has a much higher base-flow (Section 3.3) and a large upstream wetland which may protect the section of stream sampled in this study from the full impacts of flashy hydrology.

The number of EPT taxa present in the assemblages was very different between the two streams. EPT are known as the 'sensitive' insect orders and their absence from sites suggests some form of physical or hydrological harshness. Forested Tingalpa Creek had a higher diversity of EPT taxa across both 'pool' and 'riffle' habitats, and also through time compared with Stable Swamp Creek. The urbanised Stable Swamp Creek showed significantly lower abundance during the drier winter months, which suggests an influence on water quality.

While the habitat mapping did not suggest a lack of riffle habitat in urban streams, there may be differences in riffle 'quality' between urban and forested streams; urban stream riffles were observed to have smaller particle sizes (sands and gravels) compared to forested streams where riffles comprised boulders and cobbles. This suggests a possible mechanism for the poor health of urban streams, while flow itself may be partially responsible by directly dislodging invertebrates, its impact on sediment delivery to the stream through increased erosion may change habitat quality and therefore availability.

This focussed temporal study did not suggest strongly that changed hydrology itself was playing a direct role in reducing the diversity of macroinvertebrates in urban streams through a mechanism of direct dislodgement. Rather, we suggest reduced diversity in urban streams is more likely linked to a combination of reduced water quality in winter when flows are low and pollutants are likely to be concentrated and reduced habitat quality. Interestingly, the stable base flows in Stable Swamp Creek appeared to be having a positive impact on the health of the stream by maintaining some flow during the winter months when water quality normally declines and potentially mediating the impacts of extreme flashy flows during the summer months.

4. INCREASED OCCURRENCE OF FERRIC OXYHYDROXIDE PRECIPITATES IN URBAN STREAMS: IMPLICATIONS FOR WATER QUALITY

4.1. Background

Iron precipitates are a natural and common occurrence even in pristine streams (Abesser *et al.* 2006, Duckworth *et al.* 2009). Iron precipitates occur where rapid oxidation of ferrous iron from groundwater enters streams and comes in contact with the oxic environment (Figure 4-1; Schwertmann 1991, Rhoton *et al.* 2002). The iron minerals found in stream precipitates are a range of amorphous or poorly ordered ferric oxyhydroxides ($FeO(OH){\cdot}nH_2O$) and ferrihydrite ($Fe_2O_3{\cdot}0.5H_2O$) (Gotoh and Patrick 1974, Rhoton *et al.* 2002, James and Ferris 2004). Iron precipitates found in streams themselves are actually a formation of insoluble ferric iron precipitate combined with microbial biomass (Duckworth *et al.* 2009). The bacteria of the iron precipitate-microbial mats can be actively involved in mediating the oxidation of groundwater-sourced ferrous iron, using this reaction as an energy source (James and Ferris 2004, Duckworth *et al.* 2009).

Figure 4-1: Iron oxide precipitates in a tributary of Blunder Ck, Forest Lake. (Photo: M.Newham)

The microbial mediation of iron oxidation can also enhance the solubility and availability of ferric iron in the oxic stream environment. Bacteria need to prevent the precipitation of ferric iron on their cell surfaces so as not to become entombed. This is achieved by releasing chelating agents, siderophores, which promote the solubility of ferric iron (Sobolev and Roden 2001, Duckworth *et al.* 2009). A similar process is achieved by organic acids found in leachates from soils and organic matter (Albert *et al.* 2005). Dissolved organic matter or excreted chelating agents may also chelate ferrous iron, making it resistant to oxidation in streams allowing it to be transported downstream in this form (Theis and Singer 1974). Chelating agents excreted by bacteria and from dissolved organic matter can therefore enhance the biological availability and transport of iron to downstream waters by providing a pool of soluble iron (Rose and Waite 2003). In the coastal waters of Pumicestone Passage of southeast Queensland, Australia, there has been a strong correlation observed between dissolved organic carbon and soluble iron concentrations (Albert *et al.* 2005). The researchers suggest that dissolved carbon in runoff from soils chelates ferric iron and is transported to the estuary. Here, the iron can potentially enhance the growth of *Lyngbya majuscula*, a toxic blooming blue-green algae which responds to increases of iron, phosphorus and dissolved organic matter with rapid growth (Albert *et al*. 2005, Ahern *et al*. 2008).

Ferrous iron oxidation in streams can have negative impacts at the reach level where it occurs, not just at downstream coastal waters. The oxidation and hydrolysis reactions of ferrous iron produces H+ ions which can cause acidification of stream waters (Singer and Stumm 1970):

$$Fe^{2+} + \frac{1}{4} O_2 + H^+ = Fe^{3+} + 1/2H_2O$$

$$Fe^{3+} + 3H_2O = Fe(OH)_3(s) + 3H^+$$

The release of ferrous iron from soils can also be accompanied by the release of other substances such as heavy metals and phosphorous (Carlyle and Hill 2001). Excesses of ferrous iron oxidation in streams could result in the effects of acidification and release of metals reaching harmful levels. In fact, more that 20 years ago it was known that high levels of iron in streams results in reduced macroinvertebrate abundance and density (Rasmussen and Lindegaard 1988).

As noted above, the release of ferrous iron into streams and its oxidation are natural processes. However, in excess, these processes can contribute to water quality and habitat degradation. Circumstances which may enhance the release of ferrous iron into streams and result in its downstream transport include forest clearing and reduced evapotranspiration, which increases runoff and DOM loads, leading to a greater iron load (Albert *et al*. 2005). Industrial and urban effluents and runoff may also contribute to excess iron loads to streams, causing degradation and in extreme cases smothering the benthic habitat (Nedeau *et al*. 2003).

From observations made during fieldwork for several urban stream research projects, we suggest that symptoms of urbanisation on streams may also lead to increased loads of iron and precipitates in affected streams. In particular, we notice that that incision and downcutting of streams, caused by increased runoff from impervious surfaces, exposes previously buried sub-soils rich in iron minerals. As sub-surface flow is released from these deeper soil layers into the stream channel, it may carry with it a greater load of iron than water released from shallower soil layers before incision occurred (Figure 4 2).

This section of the report explores the prevalence of iron producing soils in the urban areas of Brisbane and suggests a mechanism for its liberation and potential impacts of stream water quality.

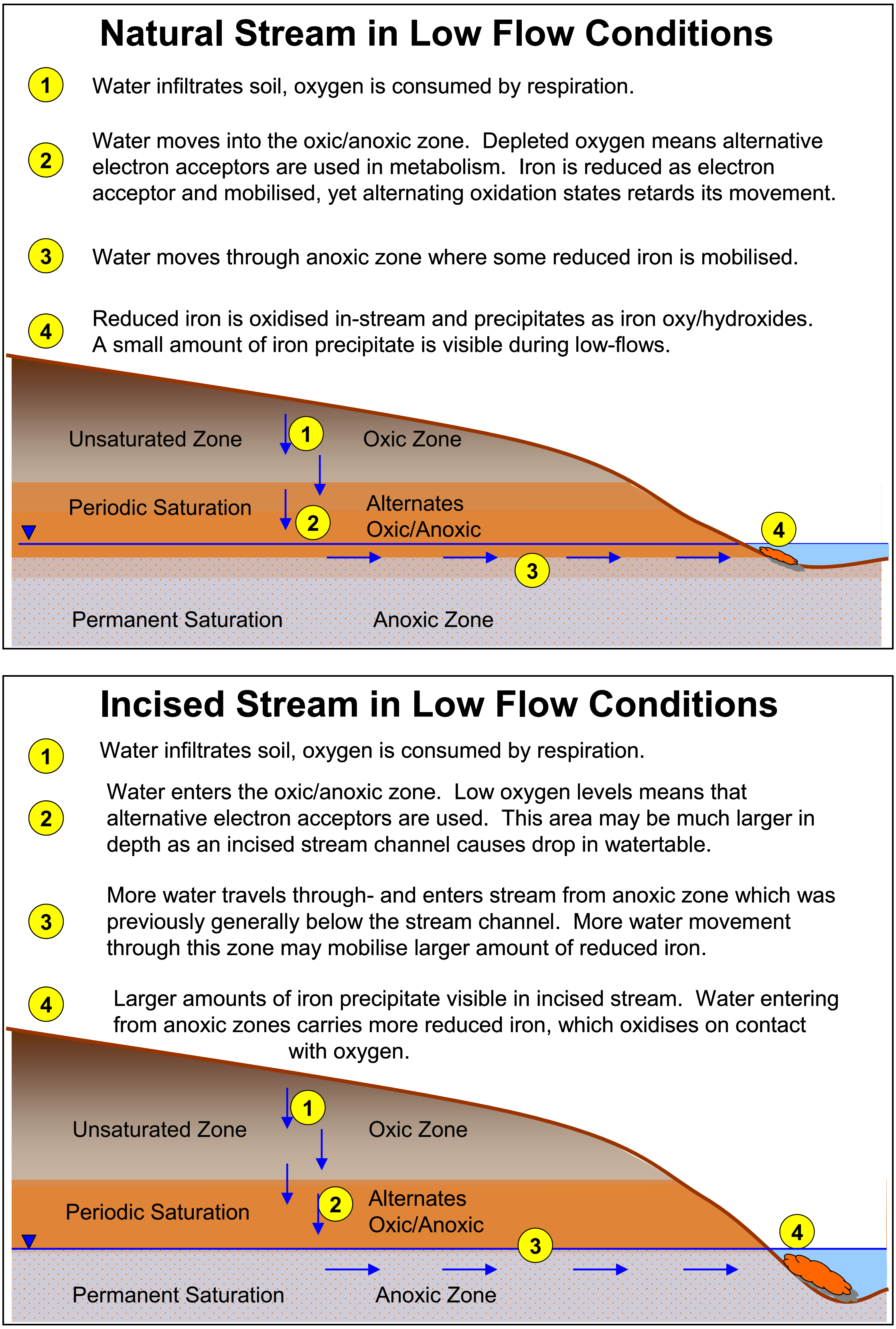


Figure 4-2: Conceptual model of iron precipitation in a natural stream, and exacerbated conditions in an incised urban stream.

4.2. Methods

4.2.1. Study Region/Sites

This study was conducted largely in the Lower Brisbane River Catchment, SEQ, Australia. The area experiences relatively dry winters, and warm, wet summers with frequent intense rainfall events. SEQ is rapidly developing with growing areas of urbanisation (Abal *et al.* 2005). This urbanisation is placing ever increasing degrading pressures on the streams of the region. This has been shown to reduce water quality and diversity of sensitive macroinvertebrate taxa (Leigh *et al.* in review).

The 12 sites for this study are located in three different sub-catchments and are spread across three geological formations (Whitaker and Green 1980). Most sites are within the Oxley Creek Catchment. The lower to middle reaches of this creek flow through areas of Quaternary Alluvium, deposited during the Holocene epoch when sea levels were higher. The catchment flows through areas of the Oxley Group and the Woogaroo Subgroup. The Oxley Group was formed from deposits of Tertiary continental basins and composed of claystone, sandstone, shale, basalt, conglomerate, siltstone and limestone. The Woogaroo Subgroup was deposited during the Triassic/Jurassic and is composed of sandstone, conglomerate, siltstone, shale and coal. The headwaters of Oxley Creek drain an area of the Marburg Subgroup, which was deposited during the Jurassic and is composed of sandstone, siltstone, shale, conglomerate, coal and oolitic ironstone. Two sites are found outside the Oxley Creek Catchment, draining directly into Moreton Bay, yet both are in areas of Oxley Group deposits. The basalt and oolitic ironstone in the two geological formations described could contribute a source of iron that, over time, may be released to soils, alluvial deposits and made biologically available through redox processes.

4.2.2. Soil Collection

One soil core was excavated from each of the 12 study sites chosen. The core was excavated from one side of the stream, at the top of the bank or at approximately 4 m (the maximum auger length) above the stream water level at sites where banks were higher than this. Soil cores were excavated with a stainless steel auger (AMS soil auger) with diameter of 70 mm. Soil was separated into depth increments every 50 cm, or where an obvious change in soil profile occurred. Each depth increment was thoroughly mixed using a plastic trowel with visible stones and roots removed. Three replicate samples of soil were then taken for iron content analysis (see below) and a sub-sample placed in a zip-lock bag and transported in an insulated container back to the laboratory. On return to the lab, moisture content of soils was measured by oven-drying a sample at 105°C overnight. The remaining soil was oven-dried at 40°C, and then stored in new zip lock bags.

4.2.3. Soil Analysis

Soil particle size was measured by sieve and hydrometer. First, oven-dried soil samples were weighed, then passed through a sieve with 2 mm mesh size. The fraction retained was weighed and formed the gravel component (>2 mm). Of the <2 mm fraction, 30 g was used in measurement of smaller particle sizes using hydrometer. Soil particles were dispersed using a solution of 10 mL of Calgon® (sodium hexametaphosphate), and 10 mL of 1 M sodium hydroxide in a total volume of 1 L. The hydrometer method provides further particle size classes of sand (2-0.06 mm) and silt (0.06-0.002 mm). The clay size fraction (<0.002 mm) is calculated by difference.

Soil electrical conductivity (EC) and pH were determined by a 1:5 suspension of dry soil:deionised water. EC and pH of the suspensions were measured using a TPS WP-81 Meter with Conductivity and pH probes. Total carbon and nitrogen content of soils was analysed using a LECO CHN Analyser. Prior to analysis, soils were ground with a mortar and rubber pestle to break apart aggregates without grinding down large particles, then passed through a sieve with 2 mm mesh size.

4.2.4. Iron Content - Ferrozine Method

The iron content of soils and stream sediments was measured using 0.5 M HCl extraction and ferrozine analysis (Stookey 1970, Lovely and Phillips 1987). 0.5 M HCl extractable iron is the organic bound and poorly crystalline iron fractions and is a good representation of the iron content of soil that is available for microbial processing (Lovely and Phillips 1987). Total and ferrous iron was measured from HCl extraction of each sample of soil collected and total iron from HCl extraction of stream sediments.

For ferrous iron analysis, soils were extracted with a 0.5 M HCl solution. Approximately 1 g dry weight equivalent of soil was placed into pre-prepared vials of 25 mL HCl solution in the field at time of soil collection. This minimised time available for oxidation or other transformation of iron to occur after soil disturbance and exposure to air. Extraction for total iron was in a combined solution of 0.25 M HCl and 0.25 M hydroxylamine hydrochloride. Hydroxylamine hydrochloride reduces ferric iron to ferrous iron, which can then be analysed as total extractable iron. On return to the lab, the extract vials were centrifuged at 3000 rpm for 15 minutes. From the vials, 0.1 mL of the supernatant was added to 10 mL of ferrozine (1 g/L) in 50 mM HEPES buffer (pH 7), the absorbance of this was measured at 562 nm.

4.2.5. Stream Channel Characteristics

Channel characteristics at each site were measured at the point of soil sampling in the riparian zone. These characteristics included the current width of wetted channel, bankfull width, maximum bankfull depth, horizontal distance from bank top to toe and an estimation of bank stability on a scale of 1-5, extremely unstable to stable.

4.3. Results and Discussion

4.3.1. Soils

Soils at the study sites varied in composition and characteristics, despite being largely in catchments on similar geological formations. Soil texture classes varied from clay through to sand, though most riparian soil profiles consisted mostly of sand clay to sandy loams. We did not find abrupt changes of soil texture to smaller particle size at depth at any site, showing that the sampled soil depths were not hydraulically separated.

Soil carbon and nitrogen contents varied with depth. Greatest carbon and nitrogen content of soils was found in the shallowest soils (Figure 4-3). Carbon and nitrogen content tended to decrease with soil depth, though a second peak in content of both commonly occurred at the transition to sub-surface saturation.

Ferrous and total iron content of soils did not follow the same pattern as for carbon and nitrogen. Ferrous iron had very low content in surface and shallow soils, but at depth made up a significant proportion of the total iron (Figure 4-4A). Total iron content for about half the sites was largest in the surface soil, though often a second peak of iron was found at deeper soil layers, though not as high as the surface content. At five sites, however, the highest iron concentrations (>0.5%) were found at depths below 50cm (Figure 4-4B). In the soil samples from all depths and sites, there appears to be a negative relationship between soil pH and conductivity (µS cm-1). At pH levels below 5, conductivity increases with further decreases in pH (Figure 4-5). Conductivity or pH do not show a correlation with soil depth or other measured parameters; high conductivity or low pH appear to be more site dependent than influenced by depth of a sample within the profile.

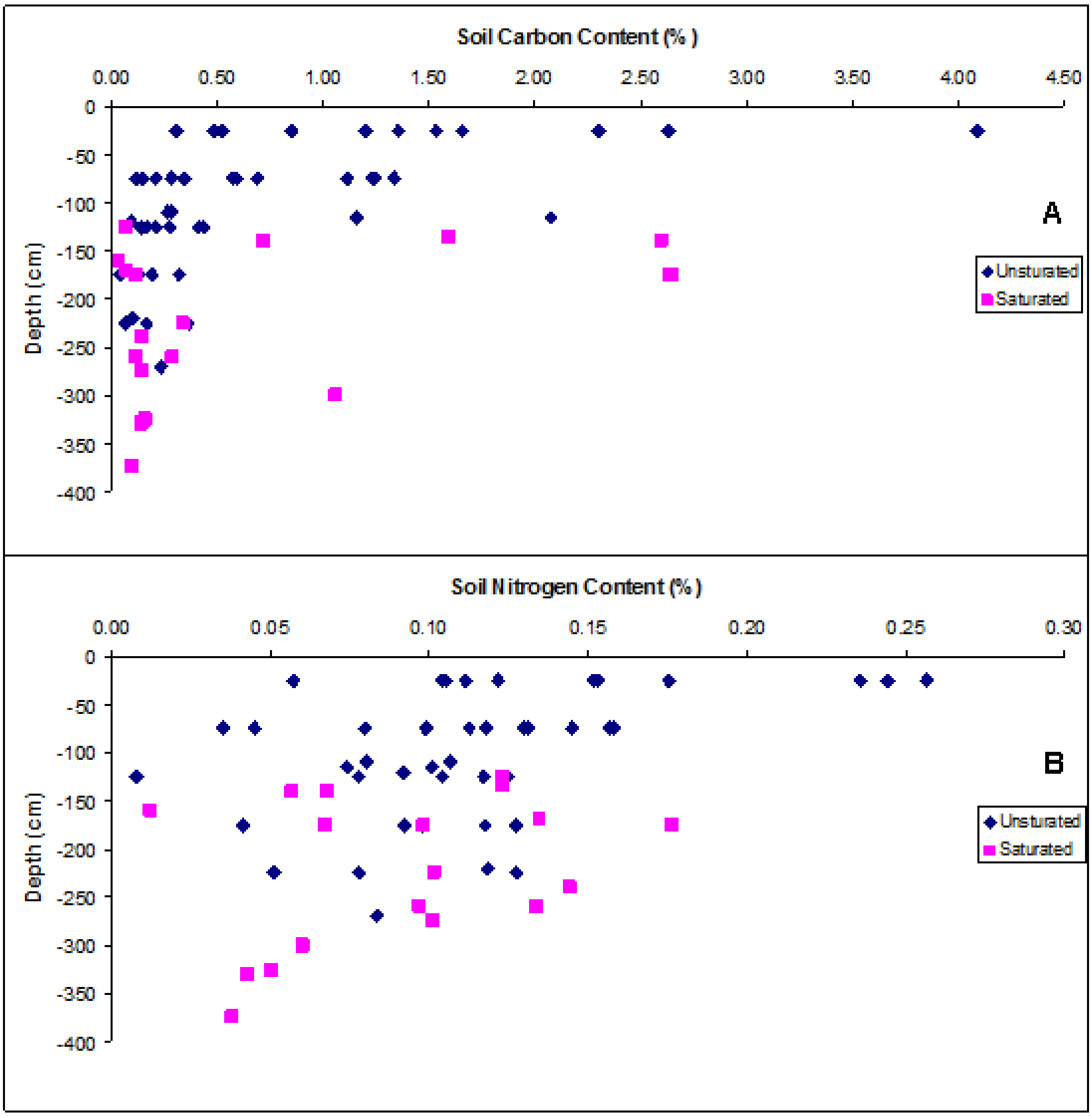


Figure 4-3: Soil total (A) Carbon and (B) Nitrogen content determined by LECO analysis, from riparian sites of the Lower Brisbane Catchment. Each point is the result from a single soil interval. 'Saturated' samples are those sampled below the sub-surface water level.

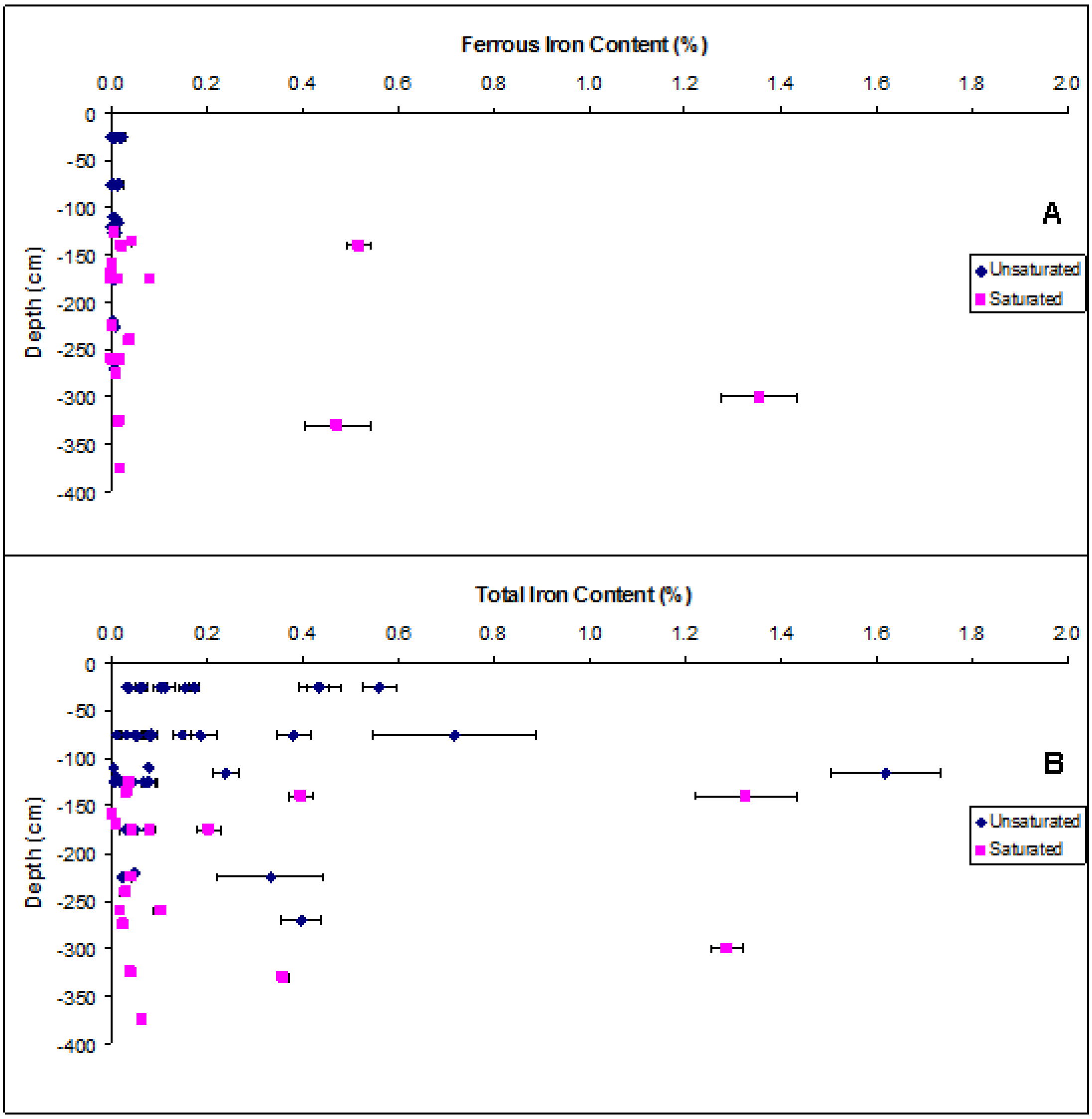


Figure 4-4: Soil total (A) Ferrous Iron and (B) Total Iron content determined by ferrozine method, from riparian sites of the Lower Brisbane Catchment. Each point is the mean (±SE) from three sub-samples of a sampled soil increment. 'Saturated' samples are those sampled below the sub-surface water level.

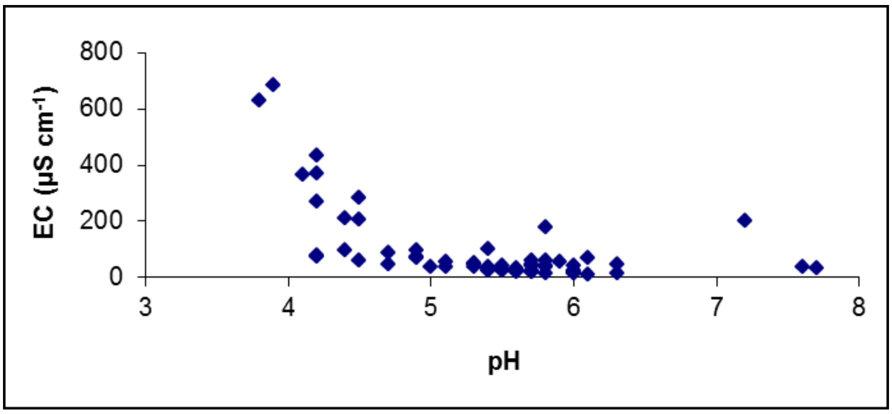


Figure 4-5: Soil pH and electrical conductivity, EC (µS cm^{-1}) for riparian soils from south Brisbane catchments. pH and EC measured in 1:5, dry soil:DI water solutions.

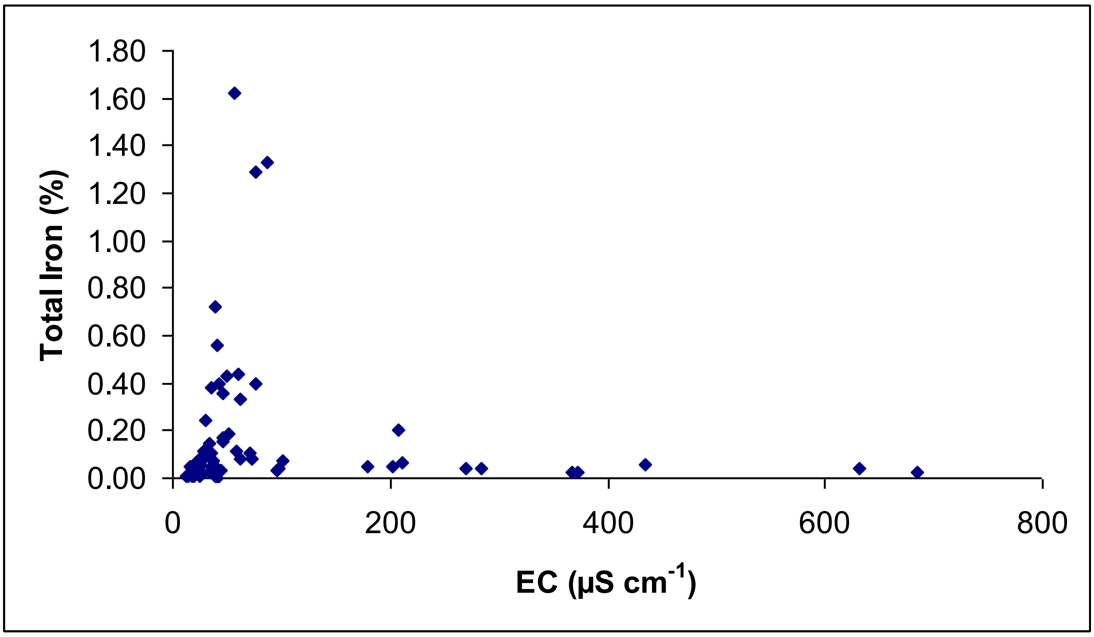


Figure 4-6: Total Iron (%) and conductivity, EC (μS cm^{-1}) for soil samples from riparian soils from south Brisbane catchments. Total iron was measured by the ferrozine method with HCl extraction, EC was measured on 1:5 soil suspension.

Table 4-1: Water quality parameters from all study sites (n=12) across the Lower Brisbane Region.

	Temp (°C)	pH	EC (μS cm^{-1})	DO (ppm)
Mean	22.61	6.95	369.32	6.15
SE	0.57	0.20	54.57	0.65
Median	23.05	7.00	311.00	6.02
Min.	19.60	5.73	118.50	0.28
Max.	26.80	8.49	693.00	8.64

Regression Tree analysis was used to isolate which of the measured predictor variables were strongest at explaining soil total iron concentration. The two strongest predictors of the percent of total soil nitrogen were percent organic carbon content of the soil and pH (Figure 4-7). A similar regression tree was attempted for soil ferrous iron alone; however, it did not group predictions satisfactorily. Only two groups were found, and one of these were all samples from a single site, suggesting there are other unmeasured predictors.

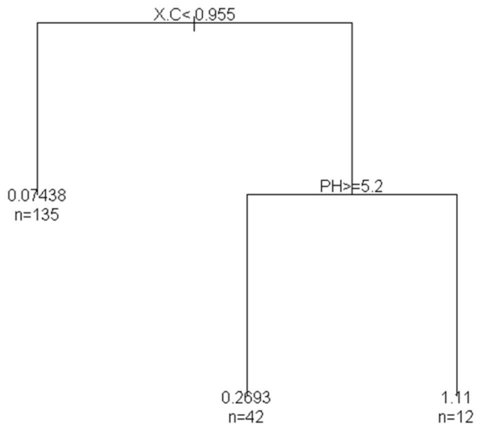


Figure 4-7: Regression tree for Soil Total Iron concentration. The predicted variable is soil total iron (%), the grouping predictor variables are soil carbon content (%) and soil pH (1:5 soil:water suspension).

Regression tree analysis was also used to predict which measured soil sample parameters could predict instream benthic total iron levels. This analysis suggested there was higher iron content at sites with deeper soil samples (ie larger depth to water), depth to groundwater, substrate type and availability of soil organic carbon were also important (Figure 4-8).

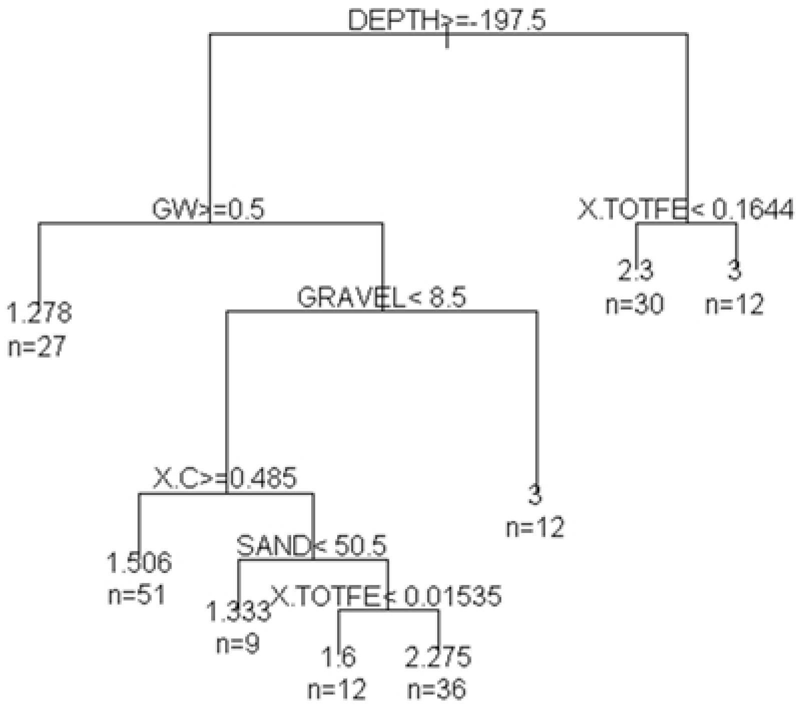


Figure 4-8: Regression tree predicting level of benthic total iron found in stream, predicted from soil sample parameters.

4.3.2. Local Management Case Study

An engineered structure to prevent the movement of a sand slug was the likely cause of a considerable decline in water quality in the tributary to Blunder Creek used in the focussed temporal study (Section 3). At this site, it appears the entire bed of the stream has been bituminised immediately before a road culvert, presumably in an attempt to stabilise the creek bed and prevent a sand slug from moving downstream and blocking the road culvert (Figure 4-9). This has caused water from upstream to move downstream through the sand slug as alluvial groundwater, or hyporheic water, as this water is passing through the sand (under the bitumen) it is presumably become anoxic very quickly and dissolving ferrous iron which then becomes insoluble and flocs out as a precipitate when it emerges from under the bitumen (see Section 4.1). The impact of this structure can be seen by comparing water quality parameters measured immediately upstream of the bitumen with those of the water entering the stream after passing through the sand slug – a distance of approximately 20 m (Figure 4-10). Downstream of the bitumen section of the stream pH was lower, conductivity was higher and turbidity higher. These conditions were most obvious during dry times and were less apparent after rainfall when flow presumable washed the iron floc downstream where its impact was lessened by dilution (Figure 4-10).


(a)

(b)

Figure 4-9: Photos of upstream (a) and downstream (b) of a bitumen structure in the base of a tributary to Blunder Creek. The orange colour in the stream is precipitated ferrous iron.

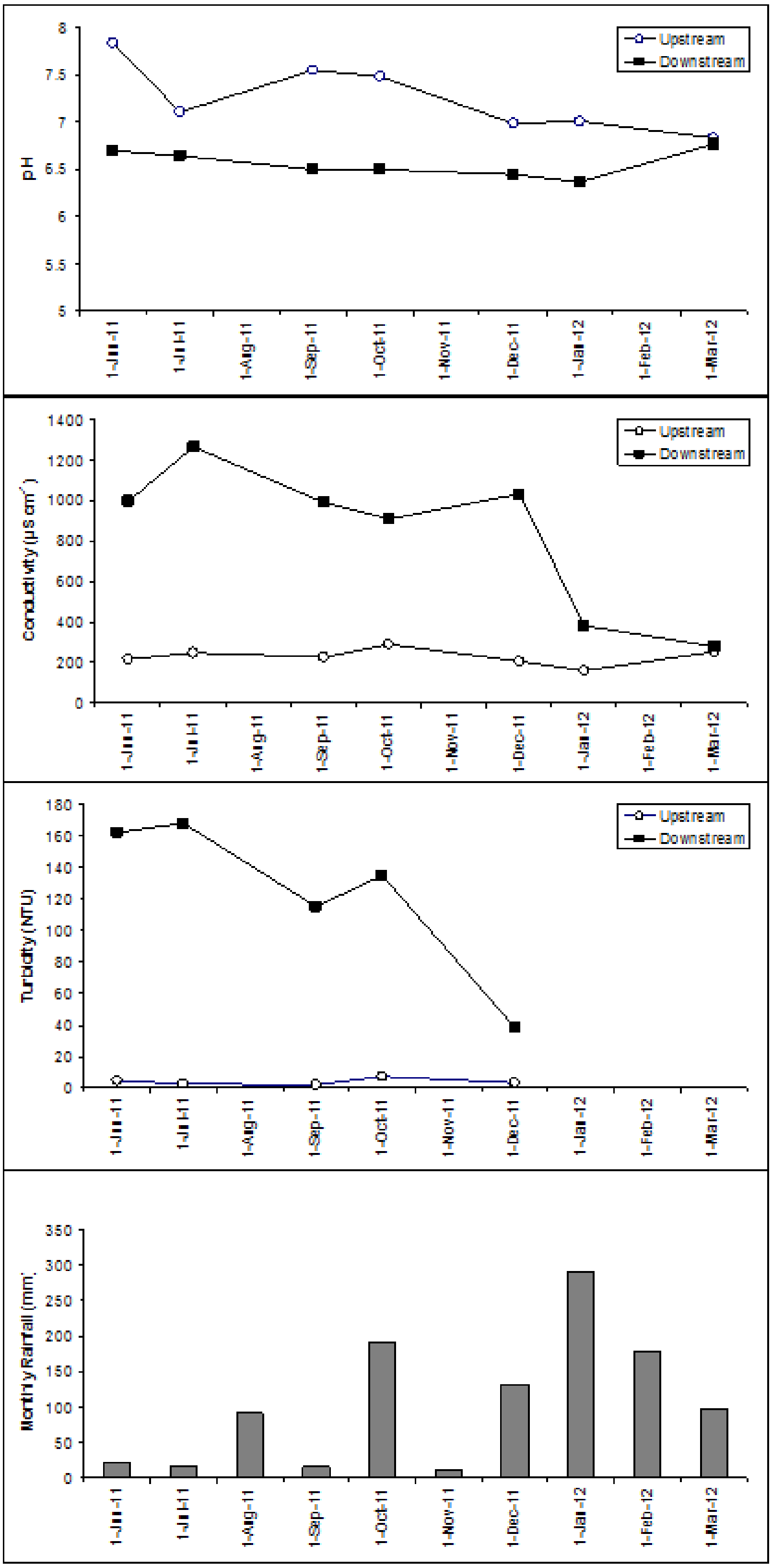


Figure 4-10: Water quality impacts of an engineered structure, a bitumen covered sand-slug, causing excess iron precipitation in-stream. Data shows (A) pH, (B) conductivity and (C) turbidity upstream and downstream of the structure. Monthly rainfall (D) is shown to indicate impacts of flow regime. Rainfall data collected at Heathwood weather station (available www.bom.gov.au/qld).

4.4. Conclusions

It was apparent from this survey that where soils are rich in iron and groundwater must travel through these soils to get to the stream, there is likely to be water quality implications, however, this will also likely result in complete removal of excess nitrogen. To completely understand the issues associated with iron floc within streams a more thorough sampling of benthic iron is required. The heterogeneous distribution of iron precipitates, even within small areas makes it difficult to sample and characterise the level of iron precipitates occurring within a stream reach. It is apparent that the incidence of iron floc in urban streams is likely to be exacerbated by stream incision, with water needing to pass through deeper iron rich soils to reach the stream in incised areas compared with non-incised areas.

It is possible that this process associated with urbanisation could be contributing more iron to coastal waters and therefore increasing the incidence of Lyngbia blooms. Although our data suggest the reduced pH caused by the iron floc does not extend 100 m downstream (Section 3.4.3) the iron itself could travel further as a precipitate, or be chelated by organic substances and travel far enough to reach coastal receiving waters. The movement of this iron when in the streams is worthy of further exploration.

This study showed that engineered structures can produce localised sources of stream iron. This was observed near several of the studied sites, though these were specifically avoided in this study, further investigation is warranted. Of particular note is a bitumen culvert downstream of the ‘Forest Lake’ site. At engineered structures, water is forced downwards to flow through anoxic zones where ferrous iron is mobilised and transported to the stream when water flow returns to the surface. At the Forest Lake site, it appears that during low flow the entire stream volume is flowing beneath the bitumen surface through an anoxic, iron rich matrix of sand and gravel. When water surfaces downstream, it carries with it a very large load of iron which seriously affects stream water quality and discolours the stream water and sediment for hundreds of meters downstream.

5. SUMMARY

Our conceptual understanding of the function of urban streams in sub-tropical Queensland suggests that the combined impact of a flashy hydrograph, where extreme velocities dislodge invertebrates, combined with inter-flow periods where water quality would become extremely poor, provided a mechanism for poor urban stream health (Figure 1-2). Given this background, we predicted that we could detect broad-scale impacts of urbanisation using macroinvertebrate eocystem health metrics and that urban streams would be in the 'poorest' health from spring – autumn where the combination of higher air temperatures and more frequent flows would provide a mechanism for impact, and be in comparatively better health over the winter months. The outcomes of this study have allowed us to update the conceptual model to include mechanisms for impact on invertebrate assemblage composition at each level (Figure 5-1).

We found that, although calculated %TIA was not a major predictor of any of the macroinvertebrate response variables at broad spatial scales, catchment scale measures of urbanisation were (Section 2.1). The importance of urbanisation, and particularly lumped measures of urbanisation, suggest the influence of impervious area connection on the health of streams, as measured by these macroinvertebrate and fish indicators, may be significant. The more focussed spatial study measuring both upstream impervious area and direct connection suggested a high degree of between-site variability with minimal overall differences between site 'groups' classified as "forested with little or no direct stormwater connection", "Water Sensitive Urban Design (WSUD)" or "urban with direct stormwater connection". Taxa in the insect orders Ephemeroptera, Plecoptera and Trichoptera (EPT) had lower relative abundances, both in streams with direct urban connection and in streams with greater upstream catchment imperviousness. However, correlation between biotic and environmental variables in the directly-connected sites was lacking in comparison with other sites. This spatial study showed a strong impact of urbanisation on the sensitive groups of macroinvertebrates, namely EPT taxa, but did not provide a specific mechanism (Section 2.2).

From or initial conceptual model we predicted that, if flow was a driving mechanism of the impacts of urbanisation, then these impacts should be increased during the summer months when rainfall is highest. From the focussed temporal study of three sites (one forested, one urban and one WSUD), we found differences that could be associated with urbanisation. The two more urbanised sites, Stable Swamp Creek and the tributary of Blunder Creek, had a higher number of flow rises across nearly all months, higher average daily flows and higher rates of fall during flow recession. The one parameter that was markedly different to that expected was the minimum daily flow and the base-flow index; the highly urbanised Stable Swamp Creek had continual baseflow throughout the year which would have mitigated many of the water quality impacts of urbanisation.

As with the hydrological metrics, there were clear water quality differences between sites that could also be related to urbanisation. The two more urbanised sites, Stable Swamp Creek and the tributary of Blunder Creek, had larger daily ranges in dissolved oxygen (DO) and temperature. In comparison, water quality parameters in the forested Tingalpa Creek, while occasionally harsh, were more often typical of forested catchments with low daily ranges in DO and temperature and low conductivity.

There were also differences in macroinvertebrate assemblage composition that could be related to urbanisation. The assemblage of the urbanised Stable Swamp Creek was dominated by taxa common in degraded streams, including the midges (Chironominae) and worms (Oligochaeta). In comparison, the forested Tingalpa Creek assemblage was dominated by mayflies (Family Leptophlebiidae) and other insect orders. When the assemblage data was compared with the logged water quality and hydrology data, it was stream electrical conductivity (EC), daily temperature range and aspects of hydrograph shape (rates of rise and fall) that explained the most variation in assemblage differences between the two sites. These physical parameters are known to be heavily influences by urbanisation. Interestingly, we predicted the greatest difference between forested and urbanised sites to occur during the summer months when more frequent flashy flows dominated in urban streams, however, we found the greatest difference during the winter months, suggesting water quality and habitat availability may be stronger drivers of degradation, the influence of both more likely to be increased during periods of comparatively lower flow.

One water quality parameter that seems to be strongly associated with urbanisation was an increase in the incidence of iron floc (rusty red deposit) on the bed of urban streams. This was very apparent in the tributary to Blunder Creek at Daintree Close downstream of Forest Lake (cover photo). At this site, pH was often low and conductivity was often high. A broader investigation of the incidence of this iron floc in urban streams found that the incidence in urban streams is likely to be exacerbated by stream incision, with water needing to pass through deeper, iron-rich soils to reach the stream in incised areas compared with non-incised areas.

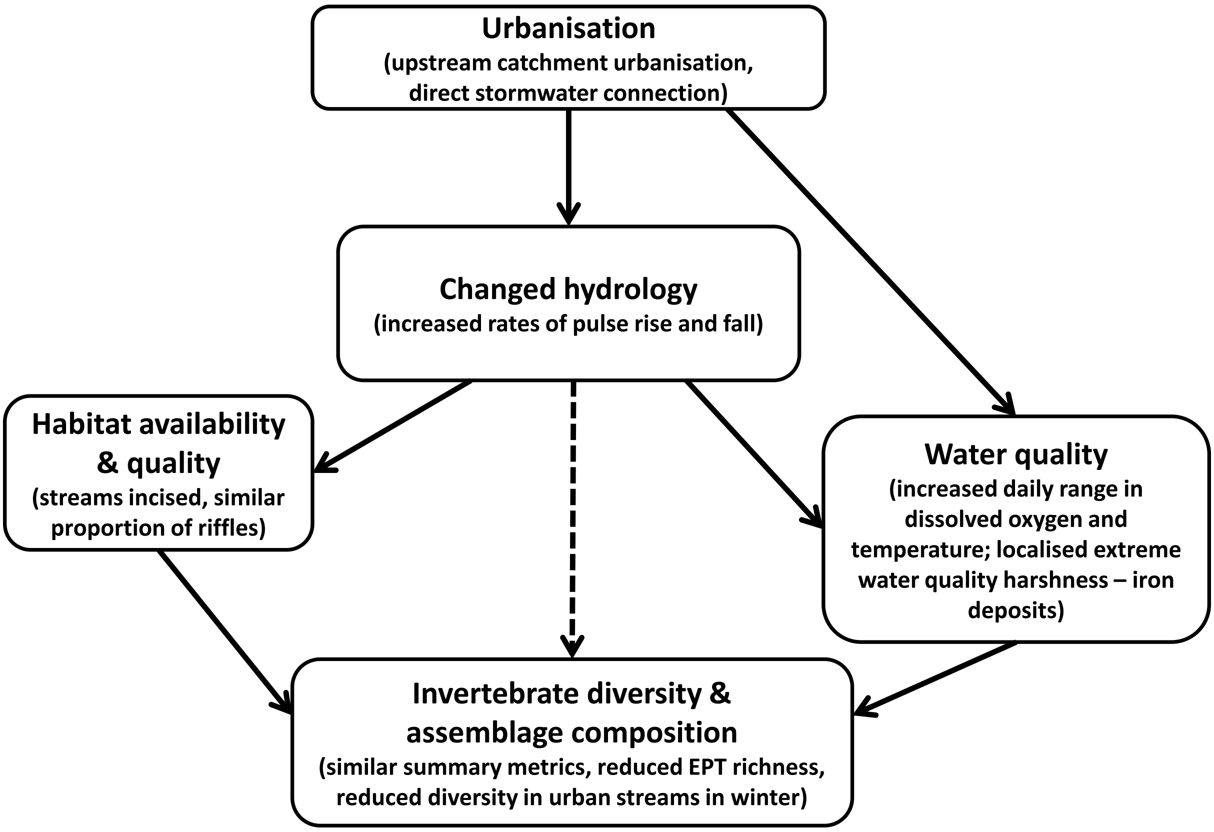


Figure 5-1: Updated conceptual model with the outcomes from this study summarising the impacts of urbanisation in invertebrate assemblages in streams in the Brisbane region.

APPENDICES

Table A1. Inclusion probabilities and model averaged coefficients for variables used to model FishOE.

Variable	Inclusion Probability	Averaged Coefficient
Season	0.0057	0
flowProb90	0.1425	-0.042259517
flowProb60	0.042	-0.012994529
flowProb30	0.0773	-0.000656423
cropBuf	0.0269	0.001978602
h2oCropPer	0.0091	-0.002819876
iFLOcrop	0.0096	0.006157575
invFLScrop	0.0348	0.000485926
idwfaOcrop	0.0134	0.001488953
idwfaScrop	0.0051	-0.004935331
Percrop	0.0292	-0.006410299
pastBuf	0.036	-0.001180618
h2oPastPer	0.0704	-0.001074555
iFLOpast	0.0419	-0.000192449
invFLSpast	0.0555	-0.001082202
idwfaOpast	0.0377	-4.71E-05
idwfaSPast	0.0361	-0.000263229
dfBuf	0.0266	-0.001177338
h2oDFper	0.0681	-0.00150525
iFLOdf	0.1403	-0.00092607
invFLSdf	0.0132	-0.001194099
idwfaOdf	0.1668	-0.035723974
idwfaSdf	0.0089	-0.001267088
mdfBuf	0.1246	0.004840399
h2oMDFper	0.0288	0.001436815
iFLOmdf	0.0914	0.000227983
invFLSmdf	0.0148	0.004339185
idwfaOmdf	8.00E-04	-0.00030136
idwfaSmdf	0.1319	0.003838033
sfBuf	0.0492	0.001774368
h2oSPFper	0.1003	-0.000919651
iFLOspf	0.0093	0.003350377
invFLSspf	0.0547	0.000177086
idwfaOspf	0	0
idwfaSspf	0.0432	-0.003104963
vsfBuf	0.0187	-0.038322578
h2oVSPper	0.0611	-0.058383374
iFLOvsp	0.0596	0.013756094
invFLSvsp	0.0232	-0.035054377
idwfaOvsp	0	0
idwfaSvsp	0.0844	0.17990138
conBuf	0.0669	0.001255289
h2oConPer	0.0869	0.00096112
agBuf	0.1817	-0.000558946

Variable	Inclusion Probability	Averaged Coefficient
h2oAgPer	0.0269	0.000983816
Pertree	0.1509	0.001217257
Pergrass	0.0775	-0.001022519
TIALumped	0	0
TIAiFLO	0.0124	9.52E-05
TIAiFLS	0	0
TIAHAiFLO	0.0246	0.00062841
TIAHAiFLS	0	0
TIAEuc	0.0299	0.009675136
urbBuf	0.0165	-6.94E-05
h2oUrbPer	0.0174	0.002043813
iFLOurb	0.0164	-0.000219862
invFLSurb	0.0463	-0.000444501
idwfaOurb	0.015	-0.001002553
idwfaSurb	0.0086	0.000465038
Perresid	0.2007	0.073986588
pH	0.052	0.006019994
Cond	0.0473	-3.35E-05
TempMax	0.0919	-0.001487654
TempRange	0.048	-0.008380243
DOMin	0.0712	0.000610539
DORange	0.0325	-0.000489022
Season:cropBuf	0.0173	0.00584205
Season:h2oCropPer	0.0091	0.007011185
Season:iFLOcrop	0	0
Season:invFLScrop	0.0125	0.005435228
Season:idwfaOcrop	0	0
Season:idwfaScrop	0.0028	0.003339666
Season:Percrop	0.0014	0.002802401
Season:pastBuf	0	0
Season:h2oPastPer	0	0
Season:iFLOpast	0	0
Season:invFLSpast	0.0045	-0.000116601
Season:idwfaOpast	0	0
Season:idwfaSPast	0.008	4.24E-05
Season:dfBuf	0	0
Season:h2oDFper	0	0
Season:iFLOdf	0	0
Season:invFLSdf	0	0
Season:idwfaOdf	0	0
Season:idwfaSdf	0	0
Season:mdfBuf	0	0
Season:h2oMDFper	0	0
Season:iFLOmdf	0	0
Season:invFLSmdf	0	0
Season:idwfaOmdf	0	0
Season:idwfaSmdf	0	0
Season:sfBuf	0.0378	0.001967916
Season:h2oSPFper	0	0
Season:iFLOspf	0.0084	-0.000684038

Variable	Inclusion Probability	Averaged Coefficient
Season:invFLSspf	0.006	0.002072086
Season:idwfaOspf	0	0
Season:idwfaSspf	0	0
Season:vsfBuf	0	0
Season:h2oVSPper	0	0
Season:iFLOvsp	0	0
Season:invFLSvsp	0	0
Season:idwfaOvsp	0	0
Season:idwfaSvsp	0.0232	0.06537731
Season:conBuf	0.0023	-0.000648367
Season:h2oConPer	0	0
Season:agBuf	0.0727	-9.43E-06
Season:h2oAgPer	4.00E-04	-0.000667758
Season:Pertree	0	0
Season:Pergrass	0	0
Season:TIALumped	0	0
Season:TIAiFLO	0	0
Season:TIAiFLS	0	0
Season:TIAHAiFLO	0	0
Season:TIAHAiFLS	0	0
Season:TIAEuc	0.0244	0.001524983
Season:urbBuf	0	0
Season:h2oUrbPer	0.0027	2.47E-05
Season:iFLOurb	0	0
Season:invFLSurb	0.0024	8.49E-05
Season:idwfaOurb	0.0021	-0.000151479
Season:idwfaSurb	0.0042	0.000854132
Season:Perresid	0.0081	0.006879176
flowProb90:cropBuf	0.0053	-0.007566925
flowProb90:h2oCropPer	0	0
flowProb90:iFLOcrop	0.0044	-0.006571061
flowProb90:invFLScrop	0.0013	-0.012453998
flowProb90:idwfaOcrop	0.0129	-0.005613823
flowProb90:idwfaScrop	0	0
flowProb90:Percrop	0	0
flowProb90:pastBuf	0	0
flowProb90:h2oPastPer	0	0
flowProb90:iFLOpast	0	0
flowProb90:invFLSpast	0.001	0.001135342
flowProb90:idwfaOpast	0	0
flowProb90:idwfaSPast	0	0
flowProb90:dfBuf	0	0
flowProb90:h2oDFper	0.0187	-2.48E-05
flowProb90:iFLOdf	0	0
flowProb90:invFLSdf	0	0
flowProb90:idwfaOdf	0	0
flowProb90:idwfaSdf	0	0
flowProb90:mdfBuf	0	0
flowProb90:h2oMDFper	0	0
flowProb90:iFLOmdf	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb90:invFLSmdf	0	0
flowProb90:idwfaOmdf	0	0
flowProb90:idwfaSmdf	0	0
flowProb90:sfBuf	0	0
flowProb90:h2oSPFper	0.0195	-0.013858745
flowProb90:iFLOspf	0	0
flowProb90:invFLSspf	0.0012	-0.008578251
flowProb90:idwfaOspf	0	0
flowProb90:idwfaSspf	0	0
flowProb90:vsfBuf	8.00E-04	-0.123943376
flowProb90:h2oVSPper	0.0118	-0.178351577
flowProb90:iFLOvsp	0.0032	-0.036804762
flowProb90:invFLSvsp	0	0
flowProb90:idwfaOvsp	0	0
flowProb90:idwfaSvsp	0.0061	-0.041110756
flowProb90:conBuf	0.0222	0.001029037
flowProb90:h2oConPer	0	0
flowProb90:agBuf	0.003	0.004071386
flowProb90:h2oAgPer	0	0
flowProb90:Pertree	0	0
flowProb90:Pergrass	0	0
flowProb90:TIALumped	0	0
flowProb90:TIAiFLO	0	0
flowProb90:TIAiFLS	0	0
flowProb90:TIAHAiFLO	0	0
flowProb90:TIAHAiFLS	0	0
flowProb90:TIAEuc	0	0
flowProb90:urbBuf	0.0068	0.001641029
flowProb90:h2oUrbPer	0	0
flowProb90:iFLOurb	0	0
flowProb90:invFLSurb	0	0
flowProb90:idwfaOurb	0	0
flowProb90:idwfaSurb	0	0
flowProb90:Perresid	0	0
flowProb60:cropBuf	0	0
flowProb60:h2oCropPer	0	0
flowProb60:iFLOcrop	6.00E-04	-0.00545054
flowProb60:invFLScrop	0.0049	-0.007892896
flowProb60:idwfaOcrop	5.00E-04	0.000102564
flowProb60:idwfaScrop	0	0
flowProb60:Percrop	0.001	-0.004736163
flowProb60:pastBuf	0	0
flowProb60:h2oPastPer	0.0085	-0.001365663
flowProb60:iFLOpast	0	0
flowProb60:invFLSpast	0.0045	0.00021252
flowProb60:idwfaOpast	0	0
flowProb60:idwfaSPast	0.0017	-0.000276079
flowProb60:dfBuf	0	0
flowProb60:h2oDFper	0	0
flowProb60:iFLOdf	0.0018	0.003450723

Variable	Inclusion Probability	Averaged Coefficient
flowProb60:invFLSdf	0	0
flowProb60:idwfaOdf	0	0
flowProb60:idwfaSdf	0	0
flowProb60:mdfBuf	0	0
flowProb60:h2oMDFper	0	0
flowProb60:iFLOmdf	0	0
flowProb60:invFLSmdf	0	0
flowProb60:idwfaOmdf	0	0
flowProb60:idwfaSmdf	0	0
flowProb60:sfBuf	0	0
flowProb60:h2oSPFper	0	0
flowProb60:iFLOspf	0	0
flowProb60:invFLSspf	0	0
flowProb60:idwfaOspf	0	0
flowProb60:idwfaSspf	0	0
flowProb60:vsfBuf	0	0
flowProb60:h2oVSPper	0.0062	-0.180075251
flowProb60:iFLOvsp	0	0
flowProb60:invFLSvsp	0	0
flowProb60:idwfaOvsp	0	0
flowProb60:idwfaSvsp	0	0
flowProb60:conBuf	0.0021	0.001321507
flowProb60:h2oConPer	0	0
flowProb60:agBuf	0.004	-0.002356721
flowProb60:h2oAgPer	4.00E-04	-0.003220802
flowProb60:Pertree	0	0
flowProb60:Pergrass	0.0053	-0.001719799
flowProb60:TIALumped	0	0
flowProb60:TIAiFLO	0	0
flowProb60:TIAiFLS	0	0
flowProb60:TIAHAiFLO	0	0
flowProb60:TIAHAiFLS	0	0
flowProb60:TIAEuc	0	0
flowProb60:urbBuf	0	0
flowProb60:h2oUrbPer	0.002	0.002298996
flowProb60:iFLOurb	0	0
flowProb60:invFLSurb	0.0084	0.000538099
flowProb60:idwfaOurb	0	0
flowProb60:idwfaSurb	0.0042	0.002527862
flowProb60:Perresid	0.0021	0.020319995
flowProb30:cropBuf	4.00E-04	-0.011321241
flowProb30:h2oCropPer	0	0
flowProb30:iFLOcrop	0	0
flowProb30:invFLScrop	0	0
flowProb30:idwfaOcrop	0	0
flowProb30:idwfaScrop	0	0
flowProb30:Percrop	0	0
flowProb30:pastBuf	0	0
flowProb30:h2oPastPer	0	0
flowProb30:iFLOpast	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb30:invFLSpast	0	0
flowProb30:idwfaOpast	0	0
flowProb30:idwfaSPast	0	0
flowProb30:dfBuf	0	0
flowProb30:h2oDFper	0	0
flowProb30:iFLOdf	0	0
flowProb30:invFLSdf	0	0
flowProb30:idwfaOdf	0	0
flowProb30:idwfaSdf	0	0
flowProb30:mdfBuf	0	0
flowProb30:h2oMDFper	0.0043	-0.000217001
flowProb30:iFLOmdf	0.0137	0.000903663
flowProb30:invFLSmdf	0	0
flowProb30:idwfaOmdf	0	0
flowProb30:idwfaSmdf	0	0
flowProb30:sfBuf	0	0
flowProb30:h2oSPFper	0	0
flowProb30:iFLOspf	0	0
flowProb30:invFLSspf	0	0
flowProb30:idwfaOspf	0	0
flowProb30:idwfaSspf	0	0
flowProb30:vsfBuf	0	0
flowProb30:h2oVSPper	0	0
flowProb30:iFLOvsp	4.00E-04	-0.01660037
flowProb30:invFLSvsp	0	0
flowProb30:idwfaOvsp	0	0
flowProb30:idwfaSvsp	0.0205	0.048515493
flowProb30:conBuf	4.00E-04	0.000793575
flowProb30:h2oConPer	0	0
flowProb30:agBuf	4.00E-04	0.004836683
flowProb30:h2oAgPer	0	0
flowProb30:Pertree	0	0
flowProb30:Pergrass	0	0
flowProb30:TIALumped	0	0
flowProb30:TIAiFLO	0	0
flowProb30:TIAiFLS	0	0
flowProb30:TIAHAiFLO	0	0
flowProb30:TIAHAiFLS	0	0
flowProb30:TIAEuc	0	0
flowProb30:urbBuf	4.00E-04	0.000919236
flowProb30:h2oUrbPer	0	0
flowProb30:iFLOurb	0	0
flowProb30:invFLSurb	0	0
flowProb30:idwfaOurb	0	0
flowProb30:idwfaSurb	0	0
flowProb30:Perresid	0	0

Table A2. Inclusion probabilities and model averaged coefficients for variables used to model MacroRich.

Variable	Inclusion Probability	Averaged Coefficient
Season	0.15965	0
flowQuantile90	0.29187	-2.442704306
flowProb60	0.18902	-1.922450601
flowProb30	0.12531	-1.102012936
cropBuf	0.02605	0.033468961
h2oCropPer	0.02411	0.035779117
iFLOcrop	0.01914	-0.067009841
invFLScrop	0.02173	0.036282494
idwfaOcrop	0.01598	-0.050249178
idwfaScrop	0.0099	-0.039457351
Percrop	0.01783	-0.099583395
pastBuf	0.00536	-0.018549678
h2oPastPer	0.00244	-0.019400752
iFLOpast	0.00309	-0.012658053
invFLSpast	0.0055	-0.020080185
idwfaOpast	0.007	-0.010110062
idwfaSPast	0.00532	-0.016944678
dfBuf	0.00732	0.030044008
h2oDFper	0.00477	0.028002941
iFLOdf	0.01184	0.034107395
invFLSdf	0.01167	0.026632837
idwfaOdf	0.05868	0.28797833
idwfaSdf	0.00448	0.020865841
mdfBuf	0.03501	0.060024966
h2oMDFper	0.01302	0.04480035
iFLOmdf	0.00805	0.042780582
invFLSmdf	0.02764	0.065288137
idwfaOmdf	0.00237	0.012085316
idwfaSmdf	0.03898	0.05481992
sfBuf	0.03363	-0.163648587
h2oSPFper	0.02654	-0.203611779
iFLOspf	0.01968	-0.008395413
invFLSspf	0.03181	-0.192341513
idwfaOspf	0.00327	0.017204217
idwfaSspf	0.00776	-0.055634662
vsfBuf	0.07691	-0.065313537
h2oVSPper	0.07982	-0.734658384
iFLOvsp	0.04209	-0.449399345
invFLSvsp	0.06448	-0.092562345
idwfaOvsp	0.00193	-0.013131624
idwfaSvsp	0.23835	2.689384344
conBuf	0.00363	0.021962019
h2oConPer	0.00168	0.018618731
agBuf	0.82632	-0.2134047
h2oAgPer	0.08272	-0.20552399
Pertree	0.00981	0.031454257
Pergrass	0.02407	-0.048377841

Variable	Inclusion Probability	Averaged Coefficient
TIALumped	0	0
TIAiFLO	0	0
TIAiFLS	0	0
TIAHAiFLO	0	0
TIAHAiFLS	0	0
TIAEuc	0.0016	-0.529701306
urbBuf	0.24369	-0.094371598
h2oUrbPer	0.35926	-0.092720309
iFLOurb	0.11449	-0.08769098
invFLSurb	0.2448	-0.094419057
idwfaOurb	0.01212	-0.071570864
idwfaSurb	0.02016	-0.104517095
Perresid	4.00E-04	-1.643169058
pH	0.06688	-0.425693909
Cond	0	0
TempMax	0.03245	-0.103026661
TempRange	0.47271	-0.441282253
DOMin	0.00977	0.011243455
DORange	0	0
Season:cropBuf	0	0
Season:h2oCropPer	0	0
Season:iFLOcrop	0	0
Season:invFLScrop	0	0
Season:idwfaOcrop	0	0
Season:idwfaScrop	0	0
Season:Percrop	0	0
Season:pastBuf	0	0
Season:h2oPastPer	0	0
Season:iFLOpast	0	0
Season:invFLSpast	0	0
Season:idwfaOpast	0	0
Season:idwfaSPast	0	0
Season:dfBuf	0	0
Season:h2oDFper	0	0
Season:iFLOdf	0	0
Season:invFLSdf	0	0
Season:idwfaOdf	0.00082	1.910797008
Season:idwfaSdf	0	0
Season:mdfBuf	0	0
Season:h2oMDFper	0	0
Season:iFLOmdf	0	0
Season:invFLSmdf	0	0
Season:idwfaOmdf	0	0
Season:idwfaSmdf	0	0
Season:sfBuf	0	0
Season:h2oSPFper	0	0
Season:iFLOspf	0	0
Season:invFLSspf	0	0
Season:idwfaOspf	0	0
Season:idwfaSspf	0	0

Variable	Inclusion Probability	Averaged Coefficient
Season:vsfBuf	0	0
Season:h2oVSPper	0	0
Season:iFLOvsp	0.00037	-0.778692131
Season:invFLSvsp	0	0
Season:idwfaOvsp	0	0
Season:idwfaSvsp	0.00073	0.688401594
Season:conBuf	0	0
Season:h2oConPer	0	0
Season:agBuf	0	0
Season:h2oAgPer	0	0
Season:Pertree	0	0
Season:Pergrass	0	0
Season:TIALumped	0	0
Season:TIAiFLO	0	0
Season:TIAiFLS	0	0
Season:TIAHAiFLO	0	0
Season:TIAHAiFLS	0	0
Season:TIAEuc	0	0
Season:urbBuf	0	0
Season:h2oUrbPer	0	0
Season:iFLOurb	0	0
Season:invFLSurb	0	0
Season:idwfaOurb	0	0
Season:idwfaSurb	0	0
Season:Perresid	0	0
flowProb90:cropBuf	0	0
flowProb90:h2oCropPer	0	0
flowProb90:iFLOcrop	0	0
flowProb90:invFLScrop	0	0
flowProb90:idwfaOcrop	0	0
flowProb90:idwfaScrop	0	0
flowProb90:Percrop	0	0
flowProb90:pastBuf	0	0
flowProb90:h2oPastPer	0	0
flowProb90:iFLOpast	0	0
flowProb90:invFLSpast	0	0
flowProb90:idwfaOpast	0	0
flowProb90:idwfaSPast	0	0
flowProb90:dfBuf	0	0
flowProb90:h2oDFper	0	0
flowProb90:iFLOdf	0	0
flowProb90:invFLSdf	0	0
flowProb90:idwfaOdf	0.00102	0.219237089
flowProb90:idwfaSdf	0	0
flowProb90:mdfBuf	0	0
flowProb90:h2oMDFper	0	0
flowProb90:iFLOmdf	0	0
flowProb90:invFLSmdf	0	0
flowProb90:idwfaOmdf	0	0
flowProb90:idwfaSmdf	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb90:sfBuf	0	0
flowProb90:h2oSPFper	0	0
flowProb90:iFLOspf	0	0
flowProb90:invFLSspf	0	0
flowProb90:idwfaOspf	0	0
flowProb90:idwfaSspf	0	0
flowProb90:vsfBuf	0.00056	-0.47920815
flowProb90:h2oVSPper	0.00395	-2.232503724
flowProb90:iFLOvsp	0.00127	0.880347581
flowProb90:invFLSvsp	0.00025	-0.669908067
flowProb90:idwfaOvsp	0	0
flowProb90:idwfaSvsp	0.03933	-6.266216166
flowProb90:conBuf	0	0
flowProb90:h2oConPer	0	0
flowProb90:agBuf	0.00626	0.156175224
flowProb90:h2oAgPer	0	0
flowProb90:Pertree	0	0
flowProb90:Pergrass	0	0
flowProb90:TIALumped	0	0
flowProb90:TIAiFLO	0	0
flowProb90:TIAiFLS	0	0
flowProb90:TIAHAiFLO	0	0
flowProb90:TIAHAiFLS	0	0
flowProb90:TIAEuc	0	0
flowProb90:urbBuf	0	0
flowProb90:h2oUrbPer	0	0
flowProb90:iFLOurb	9.00E-05	-0.03553941
flowProb90:invFLSurb	0	0
flowProb90:idwfaOurb	0	0
flowProb90:idwfaSurb	0	0
flowProb90:Perresid	0	0
flowProb60:cropBuf	0	0
flowProb60:h2oCropPer	0	0
flowProb60:iFLOcrop	0	0
flowProb60:invFLScrop	0	0
flowProb60:idwfaOcrop	0	0
flowProb60:idwfaScrop	0	0
flowProb60:Percrop	0	0
flowProb60:pastBuf	0	0
flowProb60:h2oPastPer	0	0
flowProb60:iFLOpast	0	0
flowProb60:invFLSpast	0	0
flowProb60:idwfaOpast	0	0
flowProb60:idwfaSPast	0	0
flowProb60:dfBuf	0	0
flowProb60:h2oDFper	0	0
flowProb60:iFLOdf	0	0
flowProb60:invFLSdf	0	0
flowProb60:idwfaOdf	0.00067	0.879972763
flowProb60:idwfaSdf	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb60:mdfBuf	0	0
flowProb60:h2oMDFper	0	0
flowProb60:iFLOmdf	0	0
flowProb60:invFLSmdf	0	0
flowProb60:idwfaOmdf	0	0
flowProb60:idwfaSmdf	0	0
flowProb60:sfBuf	0	0
flowProb60:h2oSPFper	0	0
flowProb60:iFLOspf	0	0
flowProb60:invFLSspf	0	0
flowProb60:idwfaOspf	0	0
flowProb60:idwfaSspf	0	0
flowProb60:vsfBuf	0.00126	0.196016414
flowProb60:h2oVSPper	0.00046	-1.547103206
flowProb60:iFLOvsp	0.00024	1.067790434
flowProb60:invFLSvsp	0.00031	-0.224604764
flowProb60:idwfaOvsp	0	0
flowProb60:idwfaSvsp	0.01342	-5.324546855
flowProb60:conBuf	0	0
flowProb60:h2oConPer	0	0
flowProb60:agBuf	0.00487	0.061121071
flowProb60:h2oAgPer	0.00048	-0.133049331
flowProb60:Pertree	0	0
flowProb60:Pergrass	0	0
flowProb60:TIALumped	0	0
flowProb60:TIAiFLO	0	0
flowProb60:TIAiFLS	0	0
flowProb60:TIAHAiFLO	0	0
flowProb60:TIAHAiFLS	0	0
flowProb60:TIAEuc	0	0
flowProb60:urbBuf	0	0
flowProb60:h2oUrbPer	0	0
flowProb60:iFLOurb	0	0
flowProb60:invFLSurb	0	0
flowProb60:idwfaOurb	0	0
flowProb60:idwfaSurb	0	0
flowProb60:Perresid	0	0
flowProb30:cropBuf	0	0
flowProb30:h2oCropPer	0	0
flowProb30:iFLOcrop	1.00E-05	0.337449336
flowProb30:invFLScrop	0.00012	0.479917381
flowProb30:idwfaOcrop	8.00E-05	1.037922887
flowProb30:idwfaScrop	0	0
flowProb30:Percrop	0	0
flowProb30:pastBuf	0	0
flowProb30:h2oPastPer	0	0
flowProb30:iFLOpast	0	0
flowProb30:invFLSpast	0	0
flowProb30:idwfaOpast	0	0
flowProb30:idwfaSPast	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb30:dfBuf	0	0
flowProb30:h2oDFper	0	0
flowProb30:iFLOdf	0	0
flowProb30:invFLSdf	0	0
flowProb30:idwfaOdf	0.00304	2.348146722
flowProb30:idwfaSdf	0	0
flowProb30:mdfBuf	0	0
flowProb30:h2oMDFper	0	0
flowProb30:iFLOmdf	0	0
flowProb30:invFLSmdf	0	0
flowProb30:idwfaOmdf	0	0
flowProb30:idwfaSmdf	0	0
flowProb30:sfBuf	0	0
flowProb30:h2oSPFper	0	0
flowProb30:iFLOspf	0	0
flowProb30:invFLSspf	0	0
flowProb30:idwfaOspf	0	0
flowProb30:idwfaSspf	0	0
flowProb30:vsfBuf	5.00E-04	-0.605651363
flowProb30:h2oVSPper	0.00078	-2.695115657
flowProb30:iFLOvsp	0.00144	0.849920787
flowProb30:invFLSvsp	0.00018	-0.81674699
flowProb30:idwfaOvsp	0	0
flowProb30:idwfaSvsp	0.00753	-4.560043795
flowProb30:conBuf	0	0
flowProb30:h2oConPer	0	0
flowProb30:agBuf	0.00223	0.065181206
flowProb30:h2oAgPer	0	0
flowProb30:Pertree	0	0
flowProb30:Pergrass	0	0
flowProb30:TIALumped	0	0
flowProb30:TIAiFLO	0	0
flowProb30:TIAiFLS	0	0
flowProb30:TIAHAiFLO	0	0
flowProb30:TIAHAiFLS	0	0
flowProb30:TIAEuc	0	0
flowProb30:urbBuf	0	0
flowProb30:h2oUrbPer	0	0
flowProb30:iFLOurb	0	0
flowProb30:invFLSurb	0	0
flowProb30:idwfaOurb	0	0
flowProb30:idwfaSurb	0	0
flowProb30:Perresid	0	0

Table A3. Inclusion probabilities and model averaged coefficients for variables used to model PET.

Variable	Inclusion Probability	Averaged Coefficient
Season	0.04645	0
flowProb90	0.10662	-0.29225798
flowProb60	0.06612	-0.415785431
flowProb30	0.07606	-0.431783132
cropBuf	0.002	-0.00852945
h2oCropPer	0.00903	-0.073595287
iFLOcrop	0.00527	0.001356047
invFLScrop	0.00798	-0.024096573
idwfaOcrop	0.00586	0.158008468
idwfaScrop	0	0
Percrop	0.02373	-0.090715897
pastBuf	0.02129	0.024151283
h2oPastPer	0.0203	0.024315015
iFLOpast	0.19659	0.030321794
invFLSpast	0.02363	0.024427179
idwfaOpast	0.14778	0.019982717
idwfaSPast	0.03788	0.022707884
dfBuf	0.14	0.091413181
h2oDFper	0.18804	0.10513226
iFLOdf	0.14786	0.14353046
invFLSdf	0.19592	0.097630204
idwfaOdf	0.0476	0.505654404
idwfaSdf	0.16917	0.093351544
mdfBuf	0.14784	0.0551809
h2oMDFper	0.02786	0.03625539
iFLOmdf	0.00733	0.035374534
invFLSmdf	0.12852	0.052772455
idwfaOmdf	0	0
idwfaSmdf	0.01359	0.040214374
sfBuf	0.02423	0.038450177
h2oSPFper	0.03429	0.154365099
iFLOspf	0.03626	0.133802082
invFLSspf	0.0225	0.025708511
idwfaOspf	0.02003	0.024663558
idwfaSspf	0.01105	-0.016361271
vsfBuf	0.05356	0.845496735
h2oVSPper	0.11397	1.236327347
iFLOvsp	0.0198	0.139343006
invFLSvsp	0.0789	0.949460276
idwfaOvsp	0	0
idwfaSvsp	0.20672	3.090091853
conBuf	0.07099	0.052029295
h2oConPer	0.06679	0.050215651
agBuf	0.02514	-0.039500508
h2oAgPer	0.01952	-0.055903389
Pertree	0	0
Pergrass	0	0

Variable	Inclusion Probability	Averaged Coefficient
TIALumped	0	0
TIAiFLO	0	0
TIAiFLS	0	0
TIAHAiFLO	8.00E-04	-0.014109649
TIAHAiFLS	0	0
TIAEuc	0.02314	-0.160218177
urbBuf	0.02517	-0.023962449
h2oUrbPer	0.04091	-0.024780324
iFLOurb	0.03323	-0.024563998
invFLSurb	0.02627	-0.024568781
idwfaOurb	0.00319	-0.01959758
idwfaSurb	0.0239	-0.027733229
Perresid	0.03155	-0.437916565
pH	0.0911	-0.319294303
Cond	0	0
TempMax	0.00792	-0.02473603
TempRange	0.03455	-0.095562054
DOMin	0.35511	0.016323117
DORange	0	0
Season:cropBuf	0	0
Season:h2oCropPer	0	0
Season:iFLOcrop	0	0
Season:invFLScrop	0	0
Season:idwfaOcrop	0	0
Season:idwfaScrop	0	0
Season:Percrop	0	0
Season:pastBuf	0	0
Season:h2oPastPer	0	0
Season:iFLOpast	0	0
Season:invFLSpast	0	0
Season:idwfaOpast	0	0
Season:idwfaSPast	0	0
Season:dfBuf	0	0
Season:h2oDFper	0	0
Season:iFLOdf	0	0
Season:invFLSdf	0	0
Season:idwfaOdf	0	0
Season:idwfaSdf	0	0
Season:mdfBuf	0	0
Season:h2oMDFper	0	0
Season:iFLOmdf	0	0
Season:invFLSmdf	0	0
Season:idwfaOmdf	0	0
Season:idwfaSmdf	0	0
Season:sfBuf	0	0
Season:h2oSPFper	0	0
Season:iFLOspf	0	0
Season:invFLSspf	0	0
Season:idwfaOspf	0	0
Season:idwfaSspf	0	0

Variable	Inclusion Probability	Averaged Coefficient
Season:vsfBuf	0	0
Season:h2oVSPper	0	0
Season:iFLOvsp	0	0
Season:invFLSvsp	0	0
Season:idwfaOvsp	0	0
Season:idwfaSvsp	0	0
Season:conBuf	0	0
Season:h2oConPer	0	0
Season:agBuf	0	0
Season:h2oAgPer	0	0
Season:Pertree	0	0
Season:Pergrass	0	0
Season:TIALumped	0	0
Season:TIAiFLO	0	0
Season:TIAiFLS	0	0
Season:TIAHAiFLO	0	0
Season:TIAHAiFLS	0	0
Season:TIAEuc	0	0
Season:urbBuf	0	0
Season:h2oUrbPer	0	0
Season:iFLOurb	0	0
Season:invFLSurb	0	0
Season:idwfaOurb	0	0
Season:idwfaSurb	0	0
Season:Perresid	0	0
flowProb90:cropBuf	0	0
flowProb90:h2oCropPer	0	0
flowProb90:iFLOcrop	0	0
flowProb90:invFLScrop	0	0
flowProb90:idwfaOcrop	0	0
flowProb90:idwfaScrop	0	0
flowProb90:Percrop	0	0
flowProb90:pastBuf	0	0
flowProb90:h2oPastPer	0	0
flowProb90:iFLOpast	0	0
flowProb90:invFLSpast	0	0
flowProb90:idwfaOpast	0	0
flowProb90:idwfaSPast	0	0
flowProb90:dfBuf	0	0
flowProb90:h2oDFper	0	0
flowProb90:iFLOdf	0	0
flowProb90:invFLSdf	0	0
flowProb90:idwfaOdf	0	0
flowProb90:idwfaSdf	0	0
flowProb90:mdfBuf	0	0
flowProb90:h2oMDFper	0	0
flowProb90:iFLOmdf	0	0
flowProb90:invFLSmdf	0	0
flowProb90:idwfaOmdf	0	0
flowProb90:idwfaSmdf	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb90:sfBuf	0	0
flowProb90:h2oSPFper	0	0
flowProb90:iFLOspf	0	0
flowProb90:invFLSspf	0	0
flowProb90:idwfaOspf	0	0
flowProb90:idwfaSspf	0	0
flowProb90:vsfBuf	0	0
flowProb90:h2oVSPper	0	0
flowProb90:iFLOvsp	0	0
flowProb90:invFLSvsp	5.00E-04	-0.310095531
flowProb90:idwfaOvsp	0	0
flowProb90:idwfaSvsp	0.02873	-4.027616193
flowProb90:conBuf	0	0
flowProb90:h2oConPer	0	0
flowProb90:agBuf	0	0
flowProb90:h2oAgPer	0	0
flowProb90:Pertree	0	0
flowProb90:Pergrass	0	0
flowProb90:TIALumped	0	0
flowProb90:TIAiFLO	0	0
flowProb90:TIAiFLS	0	0
flowProb90:TIAHAiFLO	0	0
flowProb90:TIAHAiFLS	0	0
flowProb90:TIAEuc	0	0
flowProb90:urbBuf	0	0
flowProb90:h2oUrbPer	0	0
flowProb90:iFLOurb	0	0
flowProb90:invFLSurb	0	0
flowProb90:idwfaOurb	0	0
flowProb90:idwfaSurb	0	0
flowProb90:Perresid	0.00069	0.082391789
flowProb60:cropBuf	0	0
flowProb60:h2oCropPer	0	0
flowProb60:iFLOcrop	0	0
flowProb60:invFLScrop	0	0
flowProb60:idwfaOcrop	0	0
flowProb60:idwfaScrop	0	0
flowProb60:Percrop	0	0
flowProb60:pastBuf	0	0
flowProb60:h2oPastPer	0	0
flowProb60:iFLOpast	0	0
flowProb60:invFLSpast	0	0
flowProb60:idwfaOpast	0	0
flowProb60:idwfaSPast	0	0
flowProb60:dfBuf	0	0
flowProb60:h2oDFper	0	0
flowProb60:iFLOdf	0	0
flowProb60:invFLSdf	0	0
flowProb60:idwfaOdf	0	0
flowProb60:idwfaSdf	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb60:mdfBuf	0	0
flowProb60:h2oMDFper	0	0
flowProb60:iFLOmdf	0	0
flowProb60:invFLSmdf	0	0
flowProb60:idwfaOmdf	0	0
flowProb60:idwfaSmdf	0	0
flowProb60:sfBuf	0	0
flowProb60:h2oSPFper	0	0
flowProb60:iFLOspf	0	0
flowProb60:invFLSspf	0	0
flowProb60:idwfaOspf	0	0
flowProb60:idwfaSspf	0	0
flowProb60:vsfBuf	0.00067	-0.028411596
flowProb60:h2oVSPper	0.00049	-0.56760908
flowProb60:iFLOvsp	0	0
flowProb60:invFLSvsp	0.00042	0.000757178
flowProb60:idwfaOvsp	0	0
flowProb60:idwfaSvsp	0.00876	-3.231702346
flowProb60:conBuf	0	0
flowProb60:h2oConPer	0	0
flowProb60:agBuf	0	0
flowProb60:h2oAgPer	0	0
flowProb60:Pertree	0	0
flowProb60:Pergrass	0	0
flowProb60:TIALumped	0	0
flowProb60:TIAiFLO	0	0
flowProb60:TIAiFLS	0	0
flowProb60:TIAHAiFLO	0	0
flowProb60:TIAHAiFLS	0	0
flowProb60:TIAEuc	0	0
flowProb60:urbBuf	0	0
flowProb60:h2oUrbPer	0	0
flowProb60:iFLOurb	0	0
flowProb60:invFLSurb	0	0
flowProb60:idwfaOurb	0	0
flowProb60:idwfaSurb	0	0
flowProb60:Perresid	0	0
flowProb30:cropBuf	0	0
flowProb30:h2oCropPer	0	0
flowProb30:iFLOcrop	0	0
flowProb30:invFLScrop	0	0
flowProb30:idwfaOcrop	0	0
flowProb30:idwfaScrop	0	0
flowProb30:Percrop	0	0
flowProb30:pastBuf	0	0
flowProb30:h2oPastPer	0	0
flowProb30:iFLOpast	0	0
flowProb30:invFLSpast	0	0
flowProb30:idwfaOpast	0	0
flowProb30:idwfaSPast	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb30:dfBuf	0	0
flowProb30:h2oDFper	0	0
flowProb30:iFLOdf	0	0
flowProb30:invFLSdf	0	0
flowProb30:idwfaOdf	0	0
flowProb30:idwfaSdf	0	0
flowProb30:mdfBuf	0	0
flowProb30:h2oMDFper	0	0
flowProb30:iFLOmdf	0	0
flowProb30:invFLSmdf	0	0
flowProb30:idwfaOmdf	0	0
flowProb30:idwfaSmdf	0	0
flowProb30:sfBuf	0	0
flowProb30:h2oSPFper	0	0
flowProb30:iFLOspf	0	0
flowProb30:invFLSspf	0	0
flowProb30:idwfaOspf	0	0
flowProb30:idwfaSspf	0	0
flowProb30:vsfBuf	0	0
flowProb30:h2oVSPper	0	0
flowProb30:iFLOvsp	0	0
flowProb30:invFLSvsp	0	0
flowProb30:idwfaOvsp	0	0
flowProb30:idwfaSvsp	0.00661	-3.151681414
flowProb30:conBuf	0	0
flowProb30:h2oConPer	0	0
flowProb30:agBuf	0	0
flowProb30:h2oAgPer	0	0
flowProb30:Pertree	0	0
flowProb30:Pergrass	0	0
flowProb30:TIALumped	0	0
flowProb30:TIAiFLO	0	0
flowProb30:TIAiFLS	0	0
flowProb30:TIAHAiFLO	0	0
flowProb30:TIAHAiFLS	0	0
flowProb30:TIAEuc	0	0
flowProb30:urbBuf	0	0
flowProb30:h2oUrbPer	0	0
flowProb30:iFLOurb	0	0
flowProb30:invFLSurb	0	0
flowProb30:idwfaOurb	0	0
flowProb30:idwfaSurb	0	0
flowProb30:Perresid	0.00027	-0.038816745

Table A4. Inclusion probabilities and model averaged coefficients for variables used to model PONSE.

Variable	Inclusion Probability	Averaged Coefficient
Season	0.65561	0
flowProb90	0.39981	-6.129481295
flowProb60	0.31682	-3.383033505
flowProb30	0.20748	-1.418200669
cropBuf	0.05794	0.691722539
h2oCropPer	0.1149	1.693841565
iFLOcrop	0.05629	0.842699152
invFLScrop	0.07504	0.7586024
idwfaOcrop	0.16052	-0.456805251
idwfaScrop	0.02822	-0.065783562
Percrop	0.03504	0.026217843
pastBuf	0.05592	-0.282474634
h2oPastPer	0.06543	-0.300202708
iFLOpast	0.03351	-0.257579391
invFLSpast	0.06472	-0.303704774
idwfaOpast	0.03132	-0.219410147
idwfaSPast	0.02118	-0.23829223
dfBuf	0.05166	-0.853679103
h2oDFper	0.08125	-1.005784404
iFLOdf	0.09439	-1.150189919
invFLSdf	0.06292	-0.848247672
idwfaOdf	0.5448	-9.865186335
idwfaSdf	0.02935	-0.585524225
mdfBuf	0.07342	0.414935308
h2oMDFper	0.04424	-0.139288115
iFLOmdf	0.04324	-0.131195373
invFLSmdf	0.06697	0.360995881
idwfaOmdf	0.02563	0.130129776
idwfaSmdf	0.07487	0.354274085
sfBuf	0.08413	-0.760138737
h2oSPFper	0.14237	-1.578135236
iFLOspf	0.06639	-0.896361554
invFLSspf	0.09773	-0.846951368
idwfaOspf	0.00575	-0.085566631
idwfaSspf	0.08003	-0.736842525
vsfBuf	0.20736	7.104709941
h2oVSPper	0.30772	0.517236951
iFLOvsp	0.03734	4.665086959
invFLSvsp	0.21258	6.954560026
idwfaOvsp	0.00122	0.246690443
idwfaSvsp	0.19333	9.264143244
conBuf	0.03513	0.219048931
h2oConPer	0.074	-0.158909583
agBuf	0.09242	0.587746922
h2oAgPer	0.13448	0.854739242
Pertree	0.04798	0.136621466
Pergrass	0.06156	-0.306216703

Variable	Inclusion Probability	Averaged Coefficient
TIALumped	0	0
TIAiFLO	7.00E-04	0.036239756
TIAiFLS	0	0
TIAHAiFLO	0.0199	0.256724188
TIAHAiFLS	0	0
TIAEuc	0.35772	2.172461105
urbBuf	0.01602	0.338487522
h2oUrbPer	0.0282	0.365205253
iFLOurb	0.02005	0.323197604
invFLSurb	0.01719	0.293080694
idwfaOurb	0.00139	0.190687068
idwfaSurb	0.02254	0.393849892
Perresid	0.31186	3.675267918
pH	0.51708	-3.889593622
Cond	0.00359	-0.002723891
TempMax	0.06025	0.185617979
TempRange	0.08894	-0.679667826
DOMin	0.0143	0.077751387
DORange	0.00032	-0.001048228
Season:cropBuf	0	0
Season:h2oCropPer	0.00052	0.148466072
Season:iFLOcrop	0	0
Season:invFLScrop	0.00137	0.282890941
Season:idwfaOcrop	0.00597	1.178578256
Season:idwfaScrop	0.00243	0.286541457
Season:Percrop	0.00068	0.35753777
Season:pastBuf	0	0
Season:h2oPastPer	0	0
Season:iFLOpast	0	0
Season:invFLSpast	0	0
Season:idwfaOpast	0	0
Season:idwfaSPast	0	0
Season:dfBuf	0	0
Season:h2oDFper	0.00297	0.177541074
Season:iFLOdf	0.00082	0.275404979
Season:invFLSdf	0	0
Season:idwfaOdf	0.08748	2.130287318
Season:idwfaSdf	0.00125	0.124117284
Season:mdfBuf	0.00011	0.055091948
Season:h2oMDFper	0.00132	0.127275326
Season:iFLOmdf	0.00027	0.169885643
Season:invFLSmdf	0	0
Season:idwfaOmdf	0	0
Season:idwfaSmdf	0	0
Season:sfBuf	0.00033	-0.727388159
Season:h2oSPFper	0.00356	-0.472802456
Season:iFLOspf	0	0
Season:invFLSspf	0.00253	-0.155398751
Season:idwfaOspf	0	0
Season:idwfaSspf	0	0

Variable	Inclusion Probability	Averaged Coefficient
Season:vsfBuf	0.09756	-8.810346469
Season:h2oVSPper	0.11841	-8.934133013
Season:iFLOvsp	0.00566	-5.37378676
Season:invFLSvsp	0.08644	-8.524554768
Season:idwfaOvsp	0	0
Season:idwfaSvsp	0.04771	3.225370901
Season:conBuf	0	0
Season:h2oConPer	0	0
Season:agBuf	0	0
Season:h2oAgPer	0	0
Season:Pertree	0	0
Season:Pergrass	0	0
Season:TIALumped	0	0
Season:TIAiFLO	0	0
Season:TIAiFLS	0	0
Season:TIAHAiFLO	0	0
Season:TIAHAiFLS	0	0
Season:TIAEuc	0.02041	0.529930835
Season:urbBuf	0	0
Season:h2oUrbPer	0	0
Season:iFLOurb	0	0
Season:invFLSurb	0	0
Season:idwfaOurb	0	0
Season:idwfaSurb	0	0
Season:Perresid	0.04421	2.900082766
flowProb90:cropBuf	0.00045	-0.784419917
flowProb90:h2oCropPer	0.00102	0.546269586
flowProb90:iFLOcrop	0.00075	-0.794491692
flowProb90:invFLScrop	0.00198	-0.849143162
flowProb90:idwfaOcrop	0.02578	-1.980023652
flowProb90:idwfaScrop	0.00033	-0.322396402
flowProb90:Percrop	0.00037	-0.834514516
flowProb90:pastBuf	0	0
flowProb90:h2oPastPer	1.00E-05	-0.247964078
flowProb90:iFLOpast	8.00E-05	-0.19459517
flowProb90:invFLSpast	0.0015	-0.103107229
flowProb90:idwfaOpast	0	0
flowProb90:idwfaSPast	0	0
flowProb90:dfBuf	0.0015	-0.364885283
flowProb90:h2oDFper	0.00605	-0.215320983
flowProb90:iFLOdf	0.00657	-0.636538291
flowProb90:invFLSdf	0.00015	0.124335857
flowProb90:idwfaOdf	0.07497	-4.036622828
flowProb90:idwfaSdf	0	0
flowProb90:mdfBuf	0	0
flowProb90:h2oMDFper	0	0
flowProb90:iFLOmdf	0	0
flowProb90:invFLSmdf	0	0
flowProb90:idwfaOmdf	0	0
flowProb90:idwfaSmdf	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb90:sfBuf	0.00164	0.286103551
flowProb90:h2oSPFper	0.00581	-0.779526635
flowProb90:iFLOspf	0.00137	0.231439672
flowProb90:invFLSspf	0.00296	-0.19526338
flowProb90:idwfaOspf	0	0
flowProb90:idwfaSspf	0.0019	0.577890115
flowProb90:vsfBuf	0.0698	-28.62101354
flowProb90:h2oVSPper	0.11299	-27.89369538
flowProb90:iFLOvsp	0.00494	-4.958663211
flowProb90:invFLSvsp	0.05284	-19.81846648
flowProb90:idwfaOvsp	0	0
flowProb90:idwfaSvsp	0.04165	11.8771463
flowProb90:conBuf	0	0
flowProb90:h2oConPer	0	0
flowProb90:agBuf	0.00073	0.261496671
flowProb90:h2oAgPer	0.01073	0.751709193
flowProb90:Pertree	0	0
flowProb90:Pergrass	0	0
flowProb90:TIALumped	0	0
flowProb90:TIAiFLO	0	0
flowProb90:TIAiFLS	0	0
flowProb90:TIAHAiFLO	0.00115	0.318304704
flowProb90:TIAHAiFLS	0	0
flowProb90:TIAEuc	0.0888	3.126611594
flowProb90:urbBuf	0.00256	0.416051979
flowProb90:h2oUrbPer	0.00715	0.397186036
flowProb90:iFLOurb	0.00252	0.357793673
flowProb90:invFLSurb	0.00478	0.378964255
flowProb90:idwfaOurb	0	0
flowProb90:idwfaSurb	0.00052	0.484223645
flowProb90:Perresid	0.08025	8.674835351
flowProb60:cropBuf	0.00068	-0.472765464
flowProb60:h2oCropPer	0.00313	-1.431445484
flowProb60:iFLOcrop	0.00329	-1.045825891
flowProb60:invFLScrop	0.00053	-0.971347877
flowProb60:idwfaOcrop	0.00553	-1.255405902
flowProb60:idwfaScrop	0.00078	-0.37392223
flowProb60:Percrop	0.00145	-0.182462276
flowProb60:pastBuf	0.00021	-0.193626962
flowProb60:h2oPastPer	0.00179	-0.255978693
flowProb60:iFLOpast	0	0
flowProb60:invFLSpast	0.00074	-0.193581388
flowProb60:idwfaOpast	0	0
flowProb60:idwfaSPast	0	0
flowProb60:dfBuf	0.00062	-0.05347837
flowProb60:h2oDFper	0.00227	-0.263051149
flowProb60:iFLOdf	0.00231	-0.485796661
flowProb60:invFLSdf	1.00E-05	0.154217479
flowProb60:idwfaOdf	0.05925	-1.778844896
flowProb60:idwfaSdf	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb60:mdfBuf	0	0
flowProb60:h2oMDFper	0	0
flowProb60:iFLOmdf	0	0
flowProb60:invFLSmdf	0	0
flowProb60:idwfaOmdf	0	0
flowProb60:idwfaSmdf	0	0
flowProb60:sfBuf	0.00276	0.02078551
flowProb60:h2oSPFper	0.0152	-1.047512301
flowProb60:iFLOspf	0.00202	0.871126283
flowProb60:invFLSspf	0.0028	-0.935981672
flowProb60:idwfaOspf	4.00E-04	0.383136361
flowProb60:idwfaSspf	0.00088	0.263104672
flowProb60:vsfBuf	0.05434	-24.70145567
flowProb60:h2oVSPper	0.07983	-28.0024257
flowProb60:iFLOvsp	0.00555	-5.371527827
flowProb60:invFLSvsp	0.06784	-24.93555406
flowProb60:idwfaOvsp	0	0
flowProb60:idwfaSvsp	0.03886	11.00291606
flowProb60:conBuf	0	0
flowProb60:h2oConPer	0	0
flowProb60:agBuf	0.00155	0.485932772
flowProb60:h2oAgPer	0.00083	0.636337525
flowProb60:Pertree	0	0
flowProb60:Pergrass	0	0
flowProb60:TIALumped	0	0
flowProb60:TIAiFLO	0	0
flowProb60:TIAiFLS	0	0
flowProb60:TIAHAiFLO	0.00258	0.306634559
flowProb60:TIAHAiFLS	0	0
flowProb60:TIAEuc	0.08747	2.91434926
flowProb60:urbBuf	0.00206	0.318013657
flowProb60:h2oUrbPer	0.00211	0.333855993
flowProb60:iFLOurb	0.00111	0.325009544
flowProb60:invFLSurb	8.00E-05	0.344053035
flowProb60:idwfaOurb	9.00E-05	0.22946547
flowProb60:idwfaSurb	0.00382	0.447899044
flowProb60:Perresid	0.05462	5.826473702
flowProb30:cropBuf	0	0
flowProb30:h2oCropPer	0.00523	-0.007261444
flowProb30:iFLOcrop	0.00037	-0.697398
flowProb30:invFLScrop	0.00028	-0.161635792
flowProb30:idwfaOcrop	0.00179	-1.526232711
flowProb30:idwfaScrop	0	0
flowProb30:Percrop	0.00031	-0.629126909
flowProb30:pastBuf	0	0
flowProb30:h2oPastPer	0	0
flowProb30:iFLOpast	0	0
flowProb30:invFLSpast	5.00E-05	-0.060943331
flowProb30:idwfaOpast	0	0
flowProb30:idwfaSPast	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb30:dfBuf	0	0
flowProb30:h2oDFper	0	0
flowProb30:iFLOdf	0.00113	-0.466674261
flowProb30:invFLSdf	0	0
flowProb30:idwfaOdf	0.04495	3.448804235
flowProb30:idwfaSdf	6.00E-05	-0.159625795
flowProb30:mdfBuf	0	0
flowProb30:h2oMDFper	0	0
flowProb30:iFLOmdf	0	0
flowProb30:invFLSmdf	0	0
flowProb30:idwfaOmdf	0	0
flowProb30:idwfaSmdf	0	0
flowProb30:sfBuf	0.00069	0.112347405
flowProb30:h2oSPFper	0.00561	-0.341884025
flowProb30:iFLOspf	0.00062	0.684545286
flowProb30:invFLSspf	0.00023	-0.551655709
flowProb30:idwfaOspf	9.00E-04	0.309229828
flowProb30:idwfaSspf	0.00087	0.456163367
flowProb30:vsfBuf	0.03695	-29.05849313
flowProb30:h2oVSPper	0.04713	-34.51006762
flowProb30:iFLOvsp	0.00314	-5.330373947
flowProb30:invFLSvsp	0.0467	-26.38010733
flowProb30:idwfaOvsp	0	0
flowProb30:idwfaSvsp	0.01916	3.199030234
flowProb30:conBuf	0	0
flowProb30:h2oConPer	0	0
flowProb30:agBuf	0.00119	0.681132455
flowProb30:h2oAgPer	0.00084	0.778886543
flowProb30:Pertree	0	0
flowProb30:Pergrass	0.00245	-0.155849061
flowProb30:TIALumped	0	0
flowProb30:TIAiFLO	0	0
flowProb30:TIAiFLS	0	0
flowProb30:TIAHAiFLO	0	0
flowProb30:TIAHAiFLS	0	0
flowProb30:TIAEuc	0.03571	1.962677728
flowProb30:urbBuf	0	0
flowProb30:h2oUrbPer	0	0
flowProb30:iFLOurb	0.00014	0.219221824
flowProb30:invFLSurb	0.00088	0.188324146
flowProb30:idwfaOurb	0.00039	0.178758857
flowProb30:idwfaSurb	6.00E-05	0.297902364
flowProb30:Perresid	0.01432	2.454371172

Table A5. Inclusion probabilities and model averaged coefficients for variables used to model PropAlien.

Variable	Inclusion Probability	Averaged Coefficient
Season	0.84434	0
flowProb90	0.36821	-2.05509875
flowProb60	0.50785	-2.230542446
flowProb30	0.10163	10.77965309
cropBuf	0.11914	2.585200807
h2oCropPer	0.13333	-0.123349906
iFLOcrop	0.04485	0.051379014
invFLScrop	0.13759	2.690176222
idwfaOcrop	0.07752	-0.618595943
idwfaScrop	0.02087	-0.253514234
Percrop	0.01761	-0.045673564
pastBuf	0.03425	-0.165147921
h2oPastPer	0.02016	-0.032182866
iFLOpast	0.02352	-0.130402416
invFLSpast	0.03645	-0.060622463
idwfaOpast	0.00931	-0.096645155
idwfaSPast	0.07141	-0.269367564
dfBuf	0.01517	-0.147551507
h2oDFper	0.02308	-0.265206713
iFLOdf	0.02393	-0.19420997
invFLSdf	0.01899	-0.197161024
idwfaOdf	0.63639	8.959788256
idwfaSdf	0.00538	-0.049792944
mdfBuf	0.06934	0.400132255
h2oMDFper	0.07437	0.445757027
iFLOmdf	0.05136	0.383659787
invFLSmdf	0.06355	0.403522465
idwfaOmdf	0.01078	0.041455292
idwfaSmdf	0.05177	0.312640754
sfBuf	0.4454	2.764330479
h2oSPFper	0.06702	3.196990007
iFLOspf	0.0016	0.615057384
invFLSspf	0.46694	2.88769026
idwfaOspf	0	0
idwfaSspf	0	0
vsfBuf	0.13905	-12.99143474
h2oVSPper	0.23328	-12.53546052
iFLOvsp	0.32811	-8.404839743
invFLSvsp	0.16326	-17.38742126
idwfaOvsp	0.00012	-0.162235262
idwfaSvsp	0.11199	5.60144916
conBuf	0.0659	-0.101715898
h2oConPer	0.04369	-0.118991393
agBuf	0.12709	0.695790271
h2oAgPer	0.1641	0.709029833
Pertree	0.00532	-0.070442258
Pergrass	0.0292	-0.124129365

Variable	Inclusion Probability	Averaged Coefficient
TIALumped	0	0
TIAiFLO	0	0
TIAiFLS	0	0
TIAHAiFLO	0	0
TIAHAiFLS	0	0
TIAEuc	0.01349	3.979222317
urbBuf	0.2604	0.836444467
h2oUrbPer	0.33518	0.821956078
iFLOurb	0.03256	0.767091625
invFLSurb	0.30513	0.838149938
idwfaOurb	0.00122	0.193054691
idwfaSurb	0.0055	0.686368575
Perresid	0.02654	2.01329728
pH	0.21036	1.065602311
Cond	0	0
TempMax	0.06705	-0.280617461
TempRange	0.20074	-0.834173395
DOMin	0.33352	-0.15099811
DORange	0.00619	-0.026074964
Season:cropBuf	0.08047	-2.451766085
Season:h2oCropPer	0.07259	-1.724700745
Season:iFLOcrop	0.00511	-0.868361344
Season:invFLScrop	0.0936	-2.39780654
Season:idwfaOcrop	0.00389	-0.676726318
Season:idwfaScrop	0.00131	-0.341371397
Season:Percrop	0.00058	-0.154776607
Season:pastBuf	0	0
Season:h2oPastPer	0	0
Season:iFLOpast	0.00059	0.199362758
Season:invFLSpast	0	0
Season:idwfaOpast	3.00E-05	0.132092833
Season:idwfaSPast	0.00058	0.165220307
Season:dfBuf	0.00043	0.304365304
Season:h2oDFper	0.00454	0.52534763
Season:iFLOdf	0.00152	0.510358648
Season:invFLSdf	0.00111	0.317840854
Season:idwfaOdf	0.19509	1.930733277
Season:idwfaSdf	0	0
Season:mdfBuf	0.00185	0.208437398
Season:h2oMDFper	0.00218	0.200457678
Season:iFLOmdf	0.00048	0.229642541
Season:invFLSmdf	0.00105	0.268649514
Season:idwfaOmdf	0.00047	-0.065446174
Season:idwfaSmdf	6.00E-05	-0.009232411
Season:sfBuf	0.00752	0.077522348
Season:h2oSPFper	0.00846	0.36022324
Season:iFLOspf	0.00081	0.221200226
Season:invFLSspf	0.01084	-0.24284074
Season:idwfaOspf	0	0
Season:idwfaSspf	0	0

Variable	Inclusion Probability	Averaged Coefficient
Season:vsfBuf	0.04854	7.036702867
Season:h2oVSPper	0.10151	3.830345856
Season:iFLOvsp	0.06151	1.703218627
Season:invFLSvsp	0.08016	7.603661966
Season:idwfaOvsp	0	0
Season:idwfaSvsp	0.04512	-5.757571853
Season:conBuf	0.00553	0.171150291
Season:h2oConPer	0.00742	0.175459115
Season:agBuf	0.0072	-0.735465979
Season:h2oAgPer	0.00414	-0.378458721
Season:Pertree	0	0
Season:Pergrass	0	0
Season:TIALumped	0	0
Season:TIAiFLO	0	0
Season:TIAiFLS	0	0
Season:TIAHAiFLO	0	0
Season:TIAHAiFLS	0	0
Season:TIAEuc	0.00333	-2.25087927
Season:urbBuf	0.08514	-0.164160014
Season:h2oUrbPer	0.11055	-0.165549839
Season:iFLOurb	0.00379	-0.138549474
Season:invFLSurb	0.09527	-0.151906617
Season:idwfaOurb	0	0
Season:idwfaSurb	0	0
Season:Perresid	0.0044	-1.59942254
flowProb90:cropBuf	0.00901	-2.487854232
flowProb90:h2oCropPer	0.01189	0.732749522
flowProb90:iFLOcrop	0.00116	0.835741812
flowProb90:invFLScrop	0.01241	-1.700450913
flowProb90:idwfaOcrop	0.00409	2.055452069
flowProb90:idwfaScrop	0	0
flowProb90:Percrop	0	0
flowProb90:pastBuf	0.00169	-0.605079138
flowProb90:h2oPastPer	0.00248	-0.608501289
flowProb90:iFLOpast	0.00167	-0.525149917
flowProb90:invFLSpast	0.00992	-0.572493325
flowProb90:idwfaOpast	0	0
flowProb90:idwfaSPast	0	0
flowProb90:dfBuf	0	0
flowProb90:h2oDFper	4.00E-04	0.738680668
flowProb90:iFLOdf	0.00039	0.706035965
flowProb90:invFLSdf	0	0
flowProb90:idwfaOdf	0.14914	-11.10185419
flowProb90:idwfaSdf	0.00067	-0.256298631
flowProb90:mdfBuf	0	0
flowProb90:h2oMDFper	0.00036	-0.052292239
flowProb90:iFLOmdf	0	0
flowProb90:invFLSmdf	0.00026	0.079082405
flowProb90:idwfaOmdf	0	0
flowProb90:idwfaSmdf	0.00209	0.244527211

Variable	Inclusion Probability	Averaged Coefficient
flowProb90:sfBuf	0.05253	1.988964065
flowProb90:h2oSPFper	0.00412	0.791825408
flowProb90:iFLOspf	0	0
flowProb90:invFLSspf	0.06517	1.537025018
flowProb90:idwfaOspf	0	0
flowProb90:idwfaSspf	0	0
flowProb90:vsfBuf	0.03638	20.35474401
flowProb90:h2oVSPper	0.08112	26.28493702
flowProb90:iFLOvsp	0.09857	14.65729206
flowProb90:invFLSvsp	0.06324	23.40631369
flowProb90:idwfaOvsp	0	0
flowProb90:idwfaSvsp	0.03125	2.30175702
flowProb90:conBuf	0.00104	-0.137386953
flowProb90:h2oConPer	0.00094	-0.125854245
flowProb90:agBuf	0.00964	-2.086617911
flowProb90:h2oAgPer	0.00506	-1.659926322
flowProb90:Pertree	0	0
flowProb90:Pergrass	0	0
flowProb90:TIALumped	0	0
flowProb90:TIAiFLO	0	0
flowProb90:TIAiFLS	0	0
flowProb90:TIAHAiFLO	0	0
flowProb90:TIAHAiFLS	0	0
flowProb90:TIAEuc	0.00104	-4.122993435
flowProb90:urbBuf	0.00265	0.08087042
flowProb90:h2oUrbPer	0.00967	-0.295751598
flowProb90:iFLOurb	0	0
flowProb90:invFLSurb	0.00297	0.073686698
flowProb90:idwfaOurb	0	0
flowProb90:idwfaSurb	0	0
flowProb90:Perresid	0.00963	0.384613869
flowProb60:cropBuf	0.00917	-1.249171522
flowProb60:h2oCropPer	0.01889	-1.060804824
flowProb60:iFLOcrop	0.0035	1.084614976
flowProb60:invFLScrop	0.02145	-1.21945448
flowProb60:idwfaOcrop	0.01231	2.51253492
flowProb60:idwfaScrop	0.00193	0.224574088
flowProb60:Percrop	0.00202	0.287085693
flowProb60:pastBuf	0.00687	-0.719482923
flowProb60:h2oPastPer	0.00307	-0.651944432
flowProb60:iFLOpast	9.00E-05	-0.342614161
flowProb60:invFLSpast	0.00134	-0.749364548
flowProb60:idwfaOpast	0	0
flowProb60:idwfaSPast	0	0
flowProb60:dfBuf	0	0
flowProb60:h2oDFper	0.00088	0.775251149
flowProb60:iFLOdf	0.00124	0.973652561
flowProb60:invFLSdf	0	0
flowProb60:idwfaOdf	0.22147	-11.40101634
flowProb60:idwfaSdf	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb60:mdfBuf	7.00E-05	-0.073419233
flowProb60:h2oMDFper	0.00032	-0.078440145
flowProb60:iFLOmdf	0.00017	0.552591974
flowProb60:invFLSmdf	0.00046	-0.085205138
flowProb60:idwfaOmdf	0	0
flowProb60:idwfaSmdf	0	0
flowProb60:sfBuf	0.087	1.862162346
flowProb60:h2oSPFper	0.00246	0.021845863
flowProb60:iFLOspf	0	0
flowProb60:invFLSspf	0.0815	1.768109631
flowProb60:idwfaOspf	0	0
flowProb60:idwfaSspf	0	0
flowProb60:vsfBuf	0.04441	17.67007969
flowProb60:h2oVSPper	0.08347	21.45250648
flowProb60:iFLOvsp	0.2079	14.271999
flowProb60:invFLSvsp	0.06109	26.32283724
flowProb60:idwfaOvsp	0	0
flowProb60:idwfaSvsp	0.01612	-2.017096923
flowProb60:conBuf	0	0
flowProb60:h2oConPer	0	0
flowProb60:agBuf	0.02225	-1.946451135
flowProb60:h2oAgPer	0.00893	-1.760736295
flowProb60:Pertree	0	0
flowProb60:Pergrass	0	0
flowProb60:TIALumped	0	0
flowProb60:TIAiFLO	0	0
flowProb60:TIAiFLS	0	0
flowProb60:TIAHAiFLO	0	0
flowProb60:TIAHAiFLS	0	0
flowProb60:TIAEuc	8.00E-04	-4.839959575
flowProb60:urbBuf	0.00363	0.131352996
flowProb60:h2oUrbPer	0.00701	-0.081238916
flowProb60:iFLOurb	1.00E-04	0.144897205
flowProb60:invFLSurb	0.00204	-0.118039783
flowProb60:idwfaOurb	0	0
flowProb60:idwfaSurb	0.00071	-0.080564475
flowProb60:Perresid	0.00416	1.08662407
flowProb30:cropBuf	0.01519	-4.631174983
flowProb30:h2oCropPer	0.00867	-4.223082958
flowProb30:iFLOcrop	0	0
flowProb30:invFLScrop	0.0209	-4.727519896
flowProb30:idwfaOcrop	0.00118	-0.266569086
flowProb30:idwfaScrop	0	0
flowProb30:Percrop	0	0
flowProb30:pastBuf	0.00466	-0.521746167
flowProb30:h2oPastPer	0.001	-0.484244609
flowProb30:iFLOpast	0	0
flowProb30:invFLSpast	0.00091	-0.523530911
flowProb30:idwfaOpast	0	0
flowProb30:idwfaSPast	0	0

Variable	Inclusion Probability	Averaged Coefficient
flowProb30:dfBuf	0.00013	0.319377739
flowProb30:h2oDFper	0	0
flowProb30:iFLOdf	3.00E-05	0.424810877
flowProb30:invFLSdf	0	0
flowProb30:idwfaOdf	0.03586	-12.9068934
flowProb30:idwfaSdf	0	0
flowProb30:mdfBuf	0.00057	0.092059902
flowProb30:h2oMDFper	0	0
flowProb30:iFLOmdf	0.00081	0.312229335
flowProb30:invFLSmdf	0	0
flowProb30:idwfaOmdf	0	0
flowProb30:idwfaSmdf	0.00039	0.345535089
flowProb30:sfBuf	0.01136	0.774711502
flowProb30:h2oSPFper	0.00081	1.558568553
flowProb30:iFLOspf	0	0
flowProb30:invFLSspf	0.01163	1.613999103
flowProb30:idwfaOspf	0	0
flowProb30:idwfaSspf	0	0
flowProb30:vsfBuf	0.01076	0.699021194
flowProb30:h2oVSPper	0.02084	2.67476552
flowProb30:iFLOvsp	0.0065	5.606025708
flowProb30:invFLSvsp	0.00452	2.649359322
flowProb30:idwfaOvsp	0	0
flowProb30:idwfaSvsp	0.01712	-0.395848766
flowProb30:conBuf	0	0
flowProb30:h2oConPer	0	0
flowProb30:agBuf	0.00132	-2.406611155
flowProb30:h2oAgPer	0	0
flowProb30:Pertree	0	0
flowProb30:Pergrass	0	0
flowProb30:TIALumped	0	0
flowProb30:TIAiFLO	0	0
flowProb30:TIAiFLS	0	0
flowProb30:TIAHAiFLO	0	0
flowProb30:TIAHAiFLS	0	0
flowProb30:TIAEuc	0.00207	-1.891582799
flowProb30:urbBuf	0.00079	0.18522093
flowProb30:h2oUrbPer	0.00062	0.175103436
flowProb30:iFLOurb	0	0
flowProb30:invFLSurb	0.00315	0.188351923
flowProb30:idwfaOurb	0	0
flowProb30:idwfaSurb	0	0
flowProb30:Perresid	0.00103	1.373083304

Table A6. Inclusion probabilities and model averaged coefficients for variables used to model SIGNAL.

Variable	Inclusion Probabilities	Averaged Coefficients
Season	0.00804	0
flowProb90	0.01369	0.093166525
flowProb60	0.01257	0.076228054
flowProb30	0.02384	0.124882136
cropBuf	0.00263	0.002692059
h2oCropPer	0.00764	0.011558326
iFLOcrop	0.00136	0.001949263
invFLScrop	0.00129	0.002741036
idwfaOcrop	0.00043	-0.004220087
idwfaScrop	0	0
Percrop	0.00255	-0.009771057
pastBuf	0.00132	0.00109922
h2oPastPer	0	0
iFLOpast	0	0
invFLSpast	0.00121	0.000482546
idwfaOpast	0.00185	0.000225424
idwfaSPast	0	0
dfBuf	0.08594	0.018997158
h2oDFper	0.07676	0.020906123
iFLOdf	0.56107	0.031300743
invFLSdf	0.06592	0.019833412
idwfaOdf	0.05905	0.272782086
idwfaSdf	0.04259	0.017396786
mdfBuf	0.08222	0.00821314
h2oMDFper	0.0018	0.00544206
iFLOmdf	0.02437	0.006513406
invFLSmdf	0.06189	0.008617746
idwfaOmdf	0	0
idwfaSmdf	0.01676	0.005508997
sfBuf	0.00264	-0.009814817
h2oSPFper	0	0
iFLOspf	0	0
invFLSspf	0.00307	-0.010948134
idwfaOspf	0	0
idwfaSspf	0	0
vsfBuf	0.01367	0.012848908
h2oVSPper	0.02147	0.0295356
iFLOvsp	0.01392	-0.035934293
invFLSvsp	0.01104	0.014511482
idwfaOvsp	0.00018	-0.00296856
idwfaSvsp	0.0288	0.163330339
conBuf	0.05548	0.008626488
h2oConPer	0.07778	0.008550749
agBuf	0.00141	-0.011308419
h2oAgPer	0.0022	-0.010158593
Pertree	0	0
Pergrass	0.00188	-0.003336311

Variable	Inclusion Probabilities	Averaged Coefficients
TIALumped	0	0
TIAiFLO	0	0
TIAiFLS	0	0
TIAHAiFLO	0	0
TIAHAiFLS	0	0
TIAEuc	1.00E-04	-0.012058121
urbBuf	0	0
h2oUrbPer	0	0
iFLOurb	0	0
invFLSurb	0	0
idwfaOurb	0	0
idwfaSurb	0	0
Perresid	0.0049	0.009274065
pH	0.02835	-0.055760335
Cond	0.0014	-6.82E-05
TempMax	0.00067	0.003115329
TempRange	0	0
DOMin	0.95481	0.00409095
DORange	0	0
Season:cropBuf	0	0
Season:h2oCropPer	0	0
Season:iFLOcrop	0	0
Season:invFLScrop	0	0
Season:idwfaOcrop	0	0
Season:idwfaScrop	0	0
Season:Percrop	0	0
Season:pastBuf	0	0
Season:h2oPastPer	0	0
Season:iFLOpast	0	0
Season:invFLSpast	0	0
Season:idwfaOpast	0	0
Season:idwfaSPast	0	0
Season:dfBuf	0	0
Season:h2oDFper	0	0
Season:iFLOdf	0	0
Season:invFLSdf	0	0
Season:idwfaOdf	0	0
Season:idwfaSdf	0	0
Season:mdfBuf	0	0
Season:h2oMDFper	0	0
Season:iFLOmdf	0	0
Season:invFLSmdf	0	0
Season:idwfaOmdf	0	0
Season:idwfaSmdf	0	0
Season:sfBuf	0	0
Season:h2oSPFper	0	0
Season:iFLOspf	0	0
Season:invFLSspf	0	0
Season:idwfaOspf	0	0
Season:idwfaSspf	0	0

Variable	Inclusion Probabilities	Averaged Coefficients
Season:vsfBuf	0	0
Season:h2oVSPper	0	0
Season:iFLOvsp	0	0
Season:invFLSvsp	0	0
Season:idwfaOvsp	0	0
Season:idwfaSvsp	0	0
Season:conBuf	0	0
Season:h2oConPer	0	0
Season:agBuf	0	0
Season:h2oAgPer	0	0
Season:Pertree	0	0
Season:Pergrass	0	0
Season:TIALumped	0	0
Season:TIAiFLO	0	0
Season:TIAiFLS	0	0
Season:TIAHAiFLO	0	0
Season:TIAHAiFLS	0	0
Season:TIAEuc	0	0
Season:urbBuf	0	0
Season:h2oUrbPer	0	0
Season:iFLOurb	0	0
Season:invFLSurb	0	0
Season:idwfaOurb	0	0
Season:idwfaSurb	0	0
Season:Perresid	0	0
flowProb90:cropBuf	0	0
flowProb90:h2oCropPer	0	0
flowProb90:iFLOcrop	0	0
flowProb90:invFLScrop	0	0
flowProb90:idwfaOcrop	0	0
flowProb90:idwfaScrop	0	0
flowProb90:Percrop	0	0
flowProb90:pastBuf	0	0
flowProb90:h2oPastPer	0	0
flowProb90:iFLOpast	0	0
flowProb90:invFLSpast	0	0
flowProb90:idwfaOpast	0	0
flowProb90:idwfaSPast	0	0
flowProb90:dfBuf	0	0
flowProb90:h2oDFper	0	0
flowProb90:iFLOdf	0	0
flowProb90:invFLSdf	0	0
flowProb90:idwfaOdf	7.00E-04	0.272647148
flowProb90:idwfaSdf	0	0
flowProb90:mdfBuf	0	0
flowProb90:h2oMDFper	0	0
flowProb90:iFLOmdf	0	0
flowProb90:invFLSmdf	0	0
flowProb90:idwfaOmdf	0	0
flowProb90:idwfaSmdf	0	0

Variable	Inclusion Probabilities	Averaged Coefficients
flowProb90:sfBuf	0	0
flowProb90:h2oSPFper	0	0
flowProb90:iFLOspf	0	0
flowProb90:invFLSspf	0	0
flowProb90:idwfaOspf	0	0
flowProb90:idwfaSspf	0	0
flowProb90:vsfBuf	0	0
flowProb90:h2oVSPper	0	0
flowProb90:iFLOvsp	0	0
flowProb90:invFLSvsp	0	0
flowProb90:idwfaOvsp	0	0
flowProb90:idwfaSvsp	0	0
flowProb90:conBuf	0	0
flowProb90:h2oConPer	0	0
flowProb90:agBuf	0	0
flowProb90:h2oAgPer	0	0
flowProb90:Pertree	0	0
flowProb90:Pergrass	0	0
flowProb90:TIALumped	0	0
flowProb90:TIAiFLO	0	0
flowProb90:TIAiFLS	0	0
flowProb90:TIAHAiFLO	0	0
flowProb90:TIAHAiFLS	0	0
flowProb90:TIAEuc	0	0
flowProb90:urbBuf	0	0
flowProb90:h2oUrbPer	0	0
flowProb90:iFLOurb	0	0
flowProb90:invFLSurb	0	0
flowProb90:idwfaOurb	0	0
flowProb90:idwfaSurb	0	0
flowProb90:Perresid	0	0
flowProb60:cropBuf	0	0
flowProb60:h2oCropPer	0	0
flowProb60:iFLOcrop	0	0
flowProb60:invFLScrop	0	0
flowProb60:idwfaOcrop	0	0
flowProb60:idwfaScrop	0	0
flowProb60:Percrop	0	0
flowProb60:pastBuf	0	0
flowProb60:h2oPastPer	0	0
flowProb60:iFLOpast	0	0
flowProb60:invFLSpast	0	0
flowProb60:idwfaOpast	0	0
flowProb60:idwfaSPast	0	0
flowProb60:dfBuf	0	0
flowProb60:h2oDFper	0	0
flowProb60:iFLOdf	0	0
flowProb60:invFLSdf	0	0
flowProb60:idwfaOdf	0.00026	0.259615513
flowProb60:idwfaSdf	0	0

Variable	Inclusion Probabilities	Averaged Coefficients
flowProb60:mdfBuf	0	0
flowProb60:h2oMDFper	0	0
flowProb60:iFLOmdf	0	0
flowProb60:invFLSmdf	0	0
flowProb60:idwfaOmdf	0	0
flowProb60:idwfaSmdf	0	0
flowProb60:sfBuf	0	0
flowProb60:h2oSPFper	0	0
flowProb60:iFLOspf	0	0
flowProb60:invFLSspf	0	0
flowProb60:idwfaOspf	0	0
flowProb60:idwfaSspf	0	0
flowProb60:vsfBuf	0	0
flowProb60:h2oVSPper	0	0
flowProb60:iFLOvsp	0	0
flowProb60:invFLSvsp	0	0
flowProb60:idwfaOvsp	0	0
flowProb60:idwfaSvsp	0	0
flowProb60:conBuf	0	0
flowProb60:h2oConPer	0	0
flowProb60:agBuf	0	0
flowProb60:h2oAgPer	0	0
flowProb60:Pertree	0	0
flowProb60:Pergrass	0	0
flowProb60:TIALumped	0	0
flowProb60:TIAiFLO	0	0
flowProb60:TIAiFLS	0	0
flowProb60:TIAHAiFLO	0	0
flowProb60:TIAHAiFLS	0	0
flowProb60:TIAEuc	0	0
flowProb60:urbBuf	0	0
flowProb60:h2oUrbPer	0	0
flowProb60:iFLOurb	0	0
flowProb60:invFLSurb	0	0
flowProb60:idwfaOurb	0	0
flowProb60:idwfaSurb	0	0
flowProb60:Perresid	0	0
flowProb30:cropBuf	0	0
flowProb30:h2oCropPer	0	0
flowProb30:iFLOcrop	0	0
flowProb30:invFLScrop	0	0
flowProb30:idwfaOcrop	0	0
flowProb30:idwfaScrop	0	0
flowProb30:Percrop	0	0
flowProb30:pastBuf	0	0
flowProb30:h2oPastPer	0	0
flowProb30:iFLOpast	0	0
flowProb30:invFLSpast	0	0
flowProb30:idwfaOpast	0	0
flowProb30:idwfaSPast	0	0

Variable	Inclusion Probabilities	Averaged Coefficients
flowProb30:dfBuf	0	0
flowProb30:h2oDFper	0	0
flowProb30:iFLOdf	0	0
flowProb30:invFLSdf	0	0
flowProb30:idwfaOdf	0	0
flowProb30:idwfaSdf	0	0
flowProb30:mdfBuf	0	0
flowProb30:h2oMDFper	0	0
flowProb30:iFLOmdf	0	0
flowProb30:invFLSmdf	0	0
flowProb30:idwfaOmdf	0	0
flowProb30:idwfaSmdf	0	0
flowProb30:sfBuf	0	0
flowProb30:h2oSPFper	0	0
flowProb30:iFLOspf	0	0
flowProb30:invFLSspf	0	0
flowProb30:idwfaOspf	0	0
flowProb30:idwfaSspf	0	0
flowProb30:vsfBuf	0	0
flowProb30:h2oVSPper	0	0
flowProb30:iFLOvsp	0	0
flowProb30:invFLSvsp	0	0
flowProb30:idwfaOvsp	0	0
flowProb30:idwfaSvsp	0	0
flowProb30:conBuf	0	0
flowProb30:h2oConPer	0	0
flowProb30:agBuf	0	0
flowProb30:h2oAgPer	0	0
flowProb30:Pertree	0	0
flowProb30:Pergrass	0	0
flowProb30:TIALumped	0	0
flowProb30:TIAiFLO	0	0
flowProb30:TIAiFLS	0	0
flowProb30:TIAHAiFLO	0	0
flowProb30:TIAHAiFLS	0	0
flowProb30:TIAEuc	0	0
flowProb30:urbBuf	0	0
flowProb30:h2oUrbPer	0	0
flowProb30:iFLOurb	0	0
flowProb30:invFLSurb	0	0
flowProb30:idwfaOurb	0	0
flowProb30:idwfaSurb	0	0
flowProb30:Perresid	0	0

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