Impacts of stormwater treatment wetlands on stream macroinvertebrates

A study of four wetlands constructed by Melbourne Water
on streams in eastern Melbourne.

Christopher J. Walsh
Water Studies Centre
Monash University 3800 AUSTRALIA

Water Studies Centre

MONASH University

A report prepared for Melbourne Water

October 2004

Summary

The ecological condition of four streams in the east and south-east of Melbourne that receive water from constructed stormwater treatment wetlands was assessed: Olinda Creek and the Hull Rd wetland, Scotchmans Creek and the Huntingdale Rd wetland, Dandenong Creek and the (North) Heatherton Rd wetland, and Hampton Park drain and the Hampton Park wetland. Three of the four streams (Hampton Park Drain, Dandenong, Scotchmans) were in poor condition (failing SEPP biological objectives) upstream of their wetlands, and one (Olinda) was in moderately good condition (just passing SEPP biological objectives). No difference in the ecological condition downstream of the wetlands was evident in Hampton Park or Scotchmans, but the ecological condition of Olinda Creek and Dandenong Creek degraded slightly downstream relative to upstream.

Despite the small differences in ecological condition upstream and downstream of each wetland, there were strong upstream-downstream differences in the composition of macroinvertebrate assemblages in the three streams for which a detailed longitudinal study was conducted (Olinda, Scotchmans and Dandenong). The differences in composition were driven in all streams by the increased abundance of three groups of invertebrates:

- wetland vagrants (i.e. species common to ponds and slow-moving waters likely to be abundant in the upstream wetland)
- opportunistic feeders on plankton washed out of the wetland, and
- periphyton grazers

The increased occurrence of the last group suggests increased growth of palatable benthic algae downstream of the wetlands, consistent with observations in the Hampton Park wetland of increased baseflow concentrations of bioavailable nutrients.

The observed decline in the ecological indicators SIGNAL and number of EPT families downstream of the wetlands was driven in part by the increased abundance of families belonging to the above groups (most of which are tolerant of disturbance) and in part by the less frequent occurrence of sensitive families downstream of the wetlands in Olinda and Dandenong creeks.

The stormwater treatment wetlands appear to degrade the ecological condition of streams that are in good condition and make no difference to the condition of already degraded streams. In the three most degraded streams studied here (Hampton Park drain, Scotchmans Creek and Dandenong Creek), it is likely that the effect of the wetlands was small compared to the catchment-wide impacts of stormwater runoff. The restoration of streams such as these would be more achievable using more dispersed approaches to stormwater management, closer to source. The construction of stormwater treatment wetlands may be appropriate at small scales, but their design must carefully consider the catchment context, and they should only be installed in train with dispersed stormater management methods in the catchment.

Introduction

In the last two decades, the use of constructed ponds and wetlands as a strategy for ecological restoration has become increasingly popular. Constructed ponds and wetlands have been employed widely in urban areas with the primary aim of improving water quality of stormwater runoff (e.g. Moshiri 1993; Breen *et al.* 1994). A secondary aim has often been to restore stream values in urban areas and to conserve flora and fauna (Lawrence and Breen 1998), but the effectiveness of constructed wetlands in achieving this aim has rarely been assessed. Others have warned that wetlands may have negative downstream impacts on streams as they offer only a limited capacity for temporary storage of contaminant inputs and may actually produce more toxic or bioavailable forms of some contaminants (Helfield and Diamond 1997).

Water quality improvement aims for wetlands are usually focussed on reducing loads of pollutants (e.g. Lawrence and Breen 1998). Pollutant loads are critical for the management of large downstream receiving waters such as lakes, estuaries or coastal embayments, however their relevance to the functioning of stream ecosystems is arguable. Almost all of pollutant loads are transported during large storm events, which in urban areas in particular, occur only a small proportion of time. The effects of constructed stormwater treatment wetlands on *baseflow* conditions are likely to be more critical to stream biota, and it is possible that wetlands that store high flow waters (carrying high pollutant concentrations) for treatment may worsen water quality during baseflow.

The catchment context is critical for stream management, and the effectiveness of constructed wetlands will depend strongly on the form and density of urban development in the catchment. Helfield and Diamond (1997) argued that the circumstances under which constructed wetlands may be effective or appropriate approaches to stormwater management were limited.

This study aims to assess the effect of constructed stormwater treatment wetlands on the ecological condition of streams using macroinvertebrate assemblage composition. Four wetlands in the eastern and south-eastern suburbs of Melbourne were used for study.

Methods

Site description

Four wetlands were studied (Fig. 1), in catchments of varying size and urban development:

- The Hull Rd wetland, off Olinda Creek, Lilydale, completed in February 2001 takes most of the baseflow from the creek and also receives runoff from part of a small development to the east (Fig. 2). The wetland spills back into Olinda Creek some 600 m downstream of the offtake. The catchment of Olinda Creek is sparsely urbanized. Total imperviousness upstream of the wetland is ~4.1% and ~4.5% downstream, but as most of the impervious surfaces in the catchment are drained by informal drainage systems, effective imperviousness at these two points is only 1.4% and 1.6% respectively.
- The Huntingdale Rd wetland is a retrofitted retarding basin on Scotchmans Creek, Oakleigh. Constructed in 2002, the wetland drains a highly urbanized catchment with ~37-38% effective imperviousness upstream and downstream.
- The wetland upstream of Heatherton Rd, constructed in 2000, is a large off-stream wetland receiving baseflow from Dandenong Creek, Dandenong North. A large proportion of baseflows and all high flows bypass the wetland. The catchment of

Dandenong Creek at this point is largely urbanized except in its headwaters, and effective imperviousness is about 24% at this point.

- The Hampton Park wetland is a recently constructed wetland sited on the confluence of two tributaries of the Hampton Park Drain (also known as River Gum Creek). It drains the suburbs of Narre Warren Nth and Sth. Effective imperviousness of the catchment of this wetland is to be determined as part of a current CRC FE project, but has been tentatively estimated as 28% in this report.

The first three wetlands were chosen to allow a longitudinal assessment of macroinvertebrates in comparable stream reaches of similar morphology upstream and downstream of the wetland. Four sites upstream and downstream of each wetland were sampled with ~200 m between each site.

The last wetland was chosen to complement a concurrent water quality study, but because it drains two discrete tributaries a longitudinal study was not possible. So at Hampton Park a single site was sampled in each tributary upstream together with a third site in the channel 200 m downstream of the wetland.

Sampling methods

Samples were collected from the 8 sites in each creek in Autumn 2003 (23 March, Scotchmans Ck; 31 March, Olinda Ck; 17 April, Hampton Park Drain; 1 May, Dandenong Creek) and Spring 2003 (10 November, Olinda Creek; 11 November, Scotchmans Creek; 14 November, Hampton Park Drain; 19 November, Dandenong Creek).

The sampling technique was adapted from the EPA rapid biological assessment (RBA) protocol (Tiller and Metzeling 1998). Only the edge (littoral) samples were taken at all sites.

Samples were preserved in 5% formaldehyde solution and identified to family (except for mites [Class Acarina], and the phyla Nematoda and Nemertea, which were not identified further, and the midge family chironomidae, which was identified to sub-family). All samples were sub-sampled to a minimum of 10% or, if 10% contained less than 300 individuals, to 300 animals. Additionally all large animals (eg. dragonfly larvae, diving beetles) were sorted from the whole sample. One sample from upstream and one from downstream of each wetland was also sorted in the field according to the EPA protocol (Tiller and Metzeling 1998) to permit assessment against the State Environment Protection Policy (Waters of Victoria) (SEPP: EPA Victoria 2003). The field-sorted data from these samples were combined with laboratory-sorted data from the residual material using the methods of Walsh (2004) to produce a sample that was comparable with all the other samples.

Walsh (in review) assessed the sensitivity of indicators used in the SEPP to an urban gradient in streams of eastern Melbourne, and found that only SIGNAL score (a biotic index sensitive to organic pollution: Chessman 1995) and number of families belonging to the orders Ephemeroptera, Plecoptera and Trichoptera (EPT: groups sensitive to disturbance) were adequately sensitive to detect moderate levels of urban-related degradation. Therefore in this study, the condition of reaches upstream and downstream of each wetland were assessed against the SEPP biological objectives using only these two indicators. SIGNAL and number of EPT families were calculated for live-picked sample data combined for the two seasons as required by the SEPP (EPA Victoria 2003). Objectives for bioregion 2 were applied to Olinda Creek and for bioregion 4 to the other creeks. Urban objectives were applied to Dandenong and Scotchmans creeks and Hampton Park Drain, and non-urban regional objectives were applied to Olinda Creek as its catchment was less urbanized than the SEPP definition of ‘urban’.

Statistical methods

An exploratory analysis of macroinvertebrate assemblages showed that the differences in composition between streams were much greater than differences between sites within streams. Consequently, analyses were conducted separately on each stream. Compositional similarity between samples was calculated using the Bray-Curtis similarity coefficient, firstly using log(x+1)-transformed relative abundance data (hereafter referred to as relative abundance data) and secondly using presence-absence data to assess the consistency of patterns. Patterns of similarity among sites were portrayed using non-metric multidimensional scaling (NMDS).

The primary hypothesis to be tested was that the composition of macroinvertebrate assemblages differed immediately downstream of the wetland in each stream, compared to sites upstream of the wetland. To test this hypothesis, multivariate compositional similarity was assessed as the first axis score of an NMDS for each stream in each season, and by several univariate indices: SIGNAL, total number of families and number of EPT families. Patterns in the relative abundance of families that occurred in at least 2 of the 8 sites in each stream were also assessed. Families that contributed strongly to the overall dissimilarity between upstream and downstream samples based on presence-absence were identified using SIMPER analysis (Clarke and Warwick 1994).

Differences in assemblage composition and relative abundance of individual families upstream and downstream of each wetland were assessed using a linear model with a correction for first-order serial correlation among the sites moving downstream. Such a correction is necessary because the closely spaced, linearly arranged samples are likely to be autocorrelated in this study. The effect of each wetland (W) was modelled in two ways:

i) a pulse impact, whereby the impact of the wetland is felt at the site immediately downstream of the wetland but is dissipated further downstream ($W_{1\text{-}8}$ = 0, 0, 0, 0, 1, 0, 0, and 0, respectively, where W_d is the wetland effect at site *d* moving from 1 upstream to 8 downstream)

ii) a press impact, where the effect of the wetland is constant downstream ($W_{1\text{-}8}$ = 0, 0, 0, 0, 1, 1, 1, and 1, respectively)

The WinBUGS Bayesian analysis program was used to estimate the joint posterior probability distributions of model parameters with the data for each variable (Spiegelhalter *et al.* 2003). The models used were regressions with first-order autocorrelated errors along each stream (Congdon 2001), with the BUGS script based on the "Reagan" model described by Jackman (2004). For all variables modelled,

$$Y_d \sim \text{Normal}(\mu_d, \tau),$$

where Y_d was the observed indicator value at site *d*, assumed to be a sample from a normally distributed population with a 'true' population mean μ_d and precision τ. The prior distribution used for τ was a vague gamma distribution (mean and precision both 0.05).
For the most upstream site (d = 1)

$$\mu_1 = \alpha + \beta W_1,$$

but as $W_1 = 0$ for both models this reduces to

$$\mu_1 = \alpha$$

For sites further downstream (d>1)

$$\mu_d = \alpha + \beta(W_d - \rho W_{d-1}) + \rho Y_{d-1}$$

where ρ corrects for correlation with the upstream value of the variable, and has a uniform prior distribution between –1 and 1. α and β were given vague normal prior distributions (mean =0, precision = 0.001). Models were run with 3 Markov chains, a 'burn-in' of 5000 iterations, followed by 5,000 iterations thinned every 15 (see Appendix 1).A pulse model and a press model were constructed for each variable for each stream in each season. In each case, results are reported for the model with the lower deviance information criterion (DIC: Spiegelhalter *et al.* 2003). The posterior distribution of β indicates the likely effect of the wetland. A positive mean with >90% of the distribution >0 was considered a substantial increase in the variable downstream of the wetland, and a negative mean with <90% of the distribution <0 was considered a substantial decrease.

Results

SEPP objectives and the four streams

Macroinvertebrate assemblages in Dandenong and Scotchmans creeks and Hampton Park Drain were typical of assemblages found in degraded streams throughout the metropolitan area: species-poor and numerically dominated by a few disturbance tolerant taxa. The sites sampled using the SEPP protocol, upstream and downstream of the wetland on each stream, failed to meet the urban SIGNAL objective of 5.5 (Table 1). The numbers of EPT families recorded in each of these sites were very low (but no EPT objective is applicable for bioregion 4: Table 1).

When SIGNAL and number of EPT families were calculated in the same way (using combined seasons data) for all laboratory-processed samples most samples in these three sites produced SIGNAL scores of less than 5, which is indicative of moderate pollution (Fig. 3a). All three streams supported 0-3 EPT families, also suggesting degraded conditions (Fig. 3b).

In contrast, assemblages in Olinda Creek were more diverse. The SEPP-protocol samples were close to the biological objective thresholds of 5.7 for SIGNAL and 7 for EPT families (Table 1). While the upstream site just met the objectives, the downstream site just failed (Table 1). When the laboratory processed samples from all eight sites were considered (with seasons combined), upstream-downstream differences in SIGNAL and number EPT families were not apparent (Fig. 3).

The differences in SIGNAL and number of EPT families between sites were well explained by effective catchment imperviousness (Fig. 3). The better condition of Olinda Creek assemblages is likely to be the result of the low level of conventionally drained urban land use in this catchment.

Differences in assemblage composition upstream and downstream of wetlands

The remainder of the results considers only Olinda, Dandenong and Scotchmans creeks, in which eight sites were sampled in each season.

There was a strong upstream-downstream difference in assemblage compositional similarity in all three streams (Figs. 4I, 5I, 6I, Table 2), with a press model being the better-fit model in all cases. This suggests a change in composition downstream of each wetland with no tendency to revert to the upstream condition within the length of the study reach.

These strong differences in composition were not reflected in strong changes in variables commonly used as indicators of stream health. SIGNAL scores downstream of wetlands were not substantially different from scores upstream in Scotchmans Creek (Fig. 6IIa, Table 2), but in Olinda Creek in autumn (Fig. 4IIa, Table 2) downstream sites had lower SIGNAL scores and in Dandenong Creek in both seasons there was a high probability of a reduced SIGNAL score downstream (Fig. 5IIa, Table 2). Numbers of EPT families were also not substantially different downstream of wetlands except in Olinda Creek, where there was a pulse of more EPT families downstream in both seasons (Fig. 4IIc, Table 2) and in Dandenong Creek where there were no EPT families in any sites downstream in spring (Fig. 5IIc, Table 2). Total numbers of families were also unchanged downstream in all sites except for Olinda Creek in autumn, where there was a large pulse increase in the number of families downstream of Hull Rd wetland (Fig. 4IIb, Table 2), and in Scotchmans Creek, where there was a small increase in number of families downstream (Fig. 6IIb, Table 2).

Thus, although all three streams supported assemblages of different composition upstream and downstream of wetlands, the differences either had no effect or a small negative effect on standard indicators of stream health.

The upstream-downstream differences in assemblage composition were driven by a diverse group of families with varying responses.

The most common trend was for families to occur more abundantly or frequently downstream of the wetlands.

- Filter-feeding simuliid blackflies had higher relative abundances downstream in all three streams in autumn (Figs. 7j, 8f, 9f, Tables 3, 4, 5), and occurred in more downstream sites in Dandenong and Scotchmans creeks in autumn (Table 6). In all cases, the best-fit was a pulse model. Simuliids occurred less frequently in spring, but were still more abundant downstream of Hull Rd wetland in Olinda Creek in spring (Fig. 7j, Table 3).
- The sessile planktivore *Hydra* (family Hydridae) was more abundant downstream in Scotchmans Creek: a pulse model was the best fit in autumn, press in spring (Fig. 9a, Table 5). *Hydra* was uncommon in Olinda Creek and common both upstream and downstream in Dandenong Creek.
- In Olinda Creek, the predatory flatworms *Dugesia* (family Dugesiidae) were more abundant downstream in autumn (press: Fig. 7a, Table 3) and occurred more frequently downstream in spring (Table 6). They also occurred more frequently downstream in Dandenong Creek in autumn (pulse: Fig. 8a, Table 4). They were abundant in all sites in Scotchmans Creek.
- Atyid shrimps (*Paratya australiensis*), which primarily graze on periphyton, had higher relative abundances downstream in Olinda Creek in spring (press: Fig. 7i, Table 3), and in Dandenong Creek in autumn (pulse: Fig. 8e, Table 4), and occurred more abundantly and frequently downstream in Scotchmans Creek in autumn (Fig. 9k, Tables 4, 6). Atyidae were abundant both upstream and downstream in Olinda Creek in autumn (Fig. 7i). In spring they were not collected in Scotchmans Creek and only in two sites in Dandenong Creek.
- Grazing planorbid snails occurred more abundantly and frequently downstream in Dandenong Creek (press: Fig. 8b, Tables 4, 6). They were less common in the other streams.
- In Olinda Creek, grazing lymnaeid snails and predatory water-striding veliid bugs were more abundant downstream in both seasons (Fig. 7c, l, Table 3), predatory glossiphoniid leeches and detritivorous calamoceratid caddis fly larvae were more abundant downstream in autumn (Fig. 7f, m, Table 3), and detritivorous leptocerid caddisfly larvae were more abundant downstream in spring (Fig. 7n, Table 3). Predatory notonectid backswimmers occurred more frequently downstream in spring (Table 6).

Only one family showed a decline in relative abundance downstream of a wetland across two seasons—deposit feeding lumbriculid worms in Scotchmans Creek (press: Fig 9d, Table 5). Grazing ancylid snails and conoesucid caddisfly larvae were less abundant downstream in Olinda Creek in autumn (Fig. 7d, o, Table 3, 6). Predatory nemertean worms occurred less frequently downstream in Dandenong Creek in spring (Table 6). Mosquito larvae (Culicidae) occurred less commonly downstream in Scotchmans Creek in spring (Table 6).

Several families showed inconsistent trends between seasons or between streams.

- In both seasons, omnivorous corixid bugs (water boatmen) occurred more abundantly downstream in Scotchmans Creek (Fig. 9j, Table 5) and more frequently downstream in Olinda Creek (Table 6). However, in Dandenong Creek, they were less abundant downstream in spring and abundant both upstream and downstream in autumn (Fig. 8g, Table 4).
- Naidid worms (which is a mix of predatory species such as *Chaetogaster* and periphyton grazers such as *Nais*) occurred more frequently downstream in both seasons in Scotchmans Creek (Fig. 9c, Table 5), in autumn in Olinda Creek (Fig. 7h, Table 3) and in spring in Dandenong Creek (Fig. 8d, Table 4). However, they occurred less abundantly downstream in Olinda Creek in spring and Dandenong Creek in autumn (Fig. 8d, Table 4), and were not substantially different upstream-downstream in Olinda Creek in spring (Fig. 7h, Table 3).
- Grazing physid snails were more abundant downstream both seasons in Dandenong Creek (Fig. 8c, Table 4). They were less abundant downstream in Olinda Creek in autumn and were not substantially different in spring (Fig. 7e, Table 3).
- In Olinda Creek, deposit feeding tubificid worms were more abundant downstream in autumn, but less abundant downstream in spring (Fig. 7g, Table 3).
- The diverse non-biting midge subfamilies of Tanypodinae, Chironominae and Orthocladiinae also showed inconsistent trends (Figs. 7k, 9g, h, i, Tables 3, 5).

The autocorrelation parameter ρ was substantially greater than zero in only one of the models that showed a substantial upstream-downstream difference (the autumn pulse model for Atyidae in Dandenong Creek: $\rho = 0.88$ (95% credible limits, 0.26 – 1.00), and only three models without a substantial upstream-downstream difference. This suggests that spatial autocorrelation was not a strong factor in driving the observed patterns in assemblage composition or family relative abundance.

Discussion

Possible causes of upstream-downstream differences

Changes in assemblage composition downstream of stormwater treatment wetlands were observed at all sites. The strength of possible inference as to the cause of these changes is limited in this study by the lack of data on longitudinal patterns of composition in each stream before construction of the wetlands. It is conceivable that at least some of the differences in pattern observed along the streams were unrelated to the effects of the wetlands themselves.

The models assume an equivalent distance between all sites sampled along a stream. For Olinda, Scotchmans and Dandenong creeks, the distance between the last upstream site and the first downstream site (i.e. the distance through or past the wetland) was 3.5, 4.5 and 6 times the average distance between other adjacent sites, respectively. Therefore, the upstream-downstream differences modelled by the parameter β may include some stochastic variation associated with autocorrelation downstream. However, because the autocorrelation parameter ρ was not substantially different from zero in all but four models, it can be assumed that the effect of spatial correlation was small for most families, and that the wetlands were the most likely cause of most observed differences.

The biology of the families that were observed to increase in abundance downstream also lends confidence to the inference of the wetlands being the primary cause behind compositional change. Most families increasing in abundance downstream fell into one of three categories that could be affected by the upstream wetland:

- wetland vagrants (i.e. species common to ponds and slow-moving waters likely to be abundant in the upstream wetland, such as Veliidae, Notonectidae, Corixidae, and perhaps Glossiphoniidae),
- opportunistic feeders on plankton washed out of the wetland (e.g. Simuliidae and Hydridae), and
- periphyton grazers (e.g. Atyidae, Planorbidae, Physidae and some Naididae).

The increased occurrence of filter feeders downstream of lakes has been widely recorded (e.g. Harding 1994), presumably feeding primarily on plankton washed out of the outlet. The increased numbers of the predatory *Hydra* downstream in Scotchmans Creek were likely to be feeding on zooplankton from the Huntingdale wetland. In most cases, the increased abundance of Simuliidae and Hydridae was best explained by a pulse model, suggesting that the increased densities of plankton may be localized to immediately downstream of the wetland.

The increased occurrence of periphyton-grazers suggests increased growth of palatable benthic algae downstream of the wetlands. Such a pattern would be explicable if the wetlands discharge nutrient-enriched water during baseflows. The finding of increased ammonia concentrations during baseflow conditions out of the Hampton Park wetland (Fletcher and Poelsma 2003) supports this hypothesis. Thus, it is possible that stormwater treatment wetlands designed to reduce loads of nutrients to downstream waters by treating high-nutrient flood waters may actually cause nutrient enrichment of their primary receiving streams.

Inconsistencies in pattern observed for some families may have been caused by several factors. A major area of uncertainty for families such as Naididae and Chironomidae is the coarse taxonomic resolution used this study. The members of these families have diverse ecological traits and consistent patterns of some species may be masked by different responses of other species in the same family. A second factor in observed inconsistencies such as trends observed in one stream or one season, but an absence or lower abundance or absence of the family in other streams or seasons may concern the use of relative abundances in this study, which may not always reflect patterns of absolute abundance. A third major factor may be spatial and temporal stochasticity of recruitment processes. It has been argued that such processes can play a large part in the structuring of stream macroinvertebrate assemblages (Bunn and Hughes 1997; Bunn and Davies 2000). It is possible that in streams of urbanized catchments with frequent disturbance regimes that stochastic recruitment may be particularly important.

Effects of wetlands on stream health

In Scotchmans Creek, which was in poor ecological condition upstream of the Huntingdale Rd wetland, no change in condition (as indicated by SIGNAL or number of EPT families) was detected. Dandenong Creek, upstream of the Heatherton Rd wetland was also in poor ecological condition, but samples from these sites contained a few irregular occurrences of sensitive families such as Conoesucidae, Leptoceridae, Hydroptilidae, Leptophlebiidae and Scirtidae, which elevated SIGNAL scores marginally. No such taxa were present in sites downstream of the wetland, which drove the observed decline in SIGNAL and number of EPT families. Olinda Creek was in better condition than these first two streams, and a pronounced decline in ecological condition was apparent in autumn, but not in spring. This decline was primarily driven by the increased occurrence of families that are tolerant to disturbance (e.g. the average SIGNAL grade for families showing a downstream increase in Table 3 was 3.2), but also the reduced occurrence of sensitive families such as Conoesucidae.

Thus the stormwater treatment wetlands studied here appear to degrade the ecological condition of streams that are in good condition and make no difference to the condition of already degraded streams. The degradation is indicated by an increase in the abundance of disturbance tolerant invertebrates and the reduced occurrence of sensitive invertebrates.

This finding is consistent with the work of Maxted *et al.* (in review) who found degradation of stream macroinvertebrate assemblages downstream of ponds in rural streams. Maxted *et al*. (in review) identified increased water temperatures and decreased dissolved oxygen (DO) in streams downstream of ponds as the primary cause of altered macroinvertebrate assemblages. Thermal pollution has also been reported downstream of small farm dams (Lessard and Hayes 2003) and urban ponds (Pluhowski 1970).

To test if altered temperature and DO could be a factor in the changed composition observed in Olinda Creek, a DO and temperature logger was placed upstream and downstream of the Hull Rd wetland for 3 days in March 2004. While the upstream DO probe malfunctioned, the downstream results were sufficient to show that DO depletion downstream of the wetland was unlikely to be a stressor for stream biota (Fig. 10a). However, water temperature downstream of the wetland was consistently 3-4 C^o higher than upstream, and the diel variation in temperature was ~1 C^o greater downstream (Fig. 10b). These differences in temperature may be enough to stress some species adapted to cold water streams and may explain the reduced abundance of conoesucid or leptocerid caddis flies in autumn. This effect is likely to have been less pronounced in the spring sampling period (preceded by an extended period of cooler weather).

However, increases in water temperature are commonly recorded in streams subject to stormwater impacts (Walsh *et al.* 2001; Hatt *et al.* 2004), and it is unlikely that the added thermal effect of stormwater ponds would be major stressor to macroinvertebrate assemblages in more degraded streams.

Conclusions and Recommendations

While constructed ponds and wetlands may be effective at reducing the loads of certain pollutants exported from urban catchments to receiving waters, this study suggests that they are likely to have little effect on the ecological condition of their primary receiving stream if that stream is already in poor condition, and they may actually degrade streams in good condition. Therefore the aim of Lawrence and Breen (1998) to use wetlands to restore stream values in urban areas and to conserve flora and fauna may sometimes be ill-founded.

In the three most degraded streams studied here (Hampton Park drain, Scotchmans Creek and Dandenong Creek), it is likely that the effect of the wetlands was small compared to the catchment-wide impacts of stormwater runoff. The restoration of streams such as these would be more achievable using more dispersed approaches to stormwater management, closer to source (Walsh *et al.* in review).

The Hull Rd wetland takes baseflow from Olinda Creek, which is in moderately good condition upstream, with little stormwater impacts. The wetland would be likely to be more effective if it was retrofitted to receive only runoff from the development to the east of the stream. A further redesign of the wetland to accept runoff from all of the subdivision to its east (including the area outlined in red in Fig. 2, which currently bypasses the wetland) is desirable. The aim should be for the wetland to spill into the creek infrequently rather than for it to supply a large portion of baseflow downstream, as is the case at the moment.

The construction of stormwater treatment wetlands is likely to be appropriate at small scales, but their design must carefully consider the catchment context, and they should only be installed in train with dispersed stormater management methods in the catchment. Dispersed stormwater management, if adequately applied across a catchment, is likely to protect stream ecosystem health *and* meet objectives for the reduction of pollutant loads (Walsh *et al.* in review). Such a strategy may greatly reduce the need for wetlands for stormwater management.

Acknowledgements

Edward Tsyrlin and Monica Tewman conducted the sampling and identified the macroinvertebrates for this study. Kylie Waller and Alena Glaister are thanked for their assistance with identifications. Tim Fletcher, Tony Ladson, Graham Rooney, and Angus Webb are thanked for their contributions and comments on drafts of this report. Graham Rooney and Rhys Coleman are thanked for their support for this project.

References

Breen P. F., Mag V., and Seymour B. S. (1994). The combination of a flood-retarding basin and a wetland to manage the impact of urban runoff. *Water Science and Technology* **29**, 103-109.

Bunn S. E., and Davies P. E. (2000). Biological processes in running waters and their implications for the assessment of ecological integrity. *Hydrobiologia* **422/423**, 61-70.

Bunn S. E., and Hughes J. M. (1997). Dispersal and recruitment in streams: evidence from genetic studies. *Journal of the North American Benthological Society* **16**, 338-346.

Chessman B. C. (1995). Rapid assessment of rivers using macroinvertebrates: a procedure based on habitat-specific sampling, family level identification and a biotic index. *Australian Journal of Ecology* **20**, 122-129.

Clarke K. R., and Warwick R. M. (1994). 'Change in Marine Communities: An Approach to Statistical Analysis and Interpretation.' (Natural Environment Research Council, Plymouth Marine Laboratory: Plymouth, UK.)

Congdon P. (2001). 'Bayesian Statistical Modelling.' (John Wiley and Sons: Chichester.)

EPA Victoria (2003). 'Biological objectives for rivers and streams -- ecosystem protection.' Environment Protection Authority Victoria, Information Bulletin, Publication No. 793.1 (Melbourne, Australia.)

Fletcher T. D., and Poelsma P. (2003). 'Hampton Park Wetland monitoring report. Edition 1.00.' Department of Civil Engineering, Monash University and the Cooperative Research Centre for Catchment Hydrology, Report prepared for Melbourne Water (Melbourne.)

Harding J. S. (1994). Variations in benthic fauna between differing lake outlet types in New Zealand. *New Zealand Journal of Marine and Freshwater Research* **28**, 417-427.

Hatt B. E., Fletcher T. D., Walsh C. J., and Taylor S. L. (2004). The influence of urban density and drainage infrastructure on the concentrations and loads of pollutants in small streams. *Environmental Management* **34**, 112-124.

Helfield J. M., and Diamond M. L. (1997). Use of constructed wetlands for urban stream restoration: a critical analysis. *Environmental Management* **21**, 329-341.

Jackman S. (2004). 'Estimation and Inference via Markov chain Monte Carlo: a resource for social scientists.' Department of Political Science, Stanford University. http://tamarama.stanford.edu/mcmc/

Lawrence I., and Breen P. (1998). 'Design Guidelines: Stormwater Pollution Control Ponds and Wetlands.' (Cooperative Research Centre for Freshwater Ecology: Canberra.)

Lessard J. L., and Hayes D. B. (2003). Effects of elevated water temperature on fish and macroinvertebrate communities below small dams. *River Research and Applications* **19**, 721-732.

Maxted J. R., McCready C. H., Scarsbrook M. R., and Spigel R. H. (in review). Effects of small ponds on water quality and macroinvertebrate communities in Auckland streams, New Zealand. *New Zealand Journal of Marine and Freshwater Research*.

Moshiri G. A. (Ed.) (1993). 'Constructed wetlands for water quality improvement.' (Lewis Publishers: Boca Raton, Florida.)

Pluhowski E. J. (1970). 'Urbanization and its effect on the temperature of the streams on Long Island, New York.' United States Government Printing Office, Geological Survey Professional Paper 627-D (Washington, DC.)

Spiegelhalter D., Thomas A., Best N., and Lunn D. (2003). 'WinBUGS User Manual Version 1.4.' MRC Biostatistics Unit, Cambridge, UK. http://www.mrc-bsu.cam.ac.uk/bugs

Tiller D., and Metzeling L. (1998). 'Rapid bioassessment of Victorian streams: the approach and methods of the Environment Protection Authority.' Environment Protection Authority Victoria, Publication No. 604 (Melbourne: Australia.)

Walsh C. J. (2004). Protection of in-stream biota from urban impacts: minimise catchment imperviousness or improve drainage design? *Marine and Freshwater Research* **55**, 317-326.

Walsh C. J. (in review). The relative sensitivity of macroinvertebrate-based ecological indicators of stream condition to a gradient of urban disturbance. *Marine and Freshwater Research*.

Walsh C. J., Fletcher T. D., and Ladson A. R. (in review). Stream restoration in urban catchments through re-designing stormwater systems: looking to the catchment to save the stream. *Journal of the North American Benthological Society*.

Walsh C. J., Sharpe A. K., Breen P. F., and Sonneman J. A. (2001). Effects of urbanization on streams of the Melbourne region, Victoria, Australia. I. Benthic macroinvertebrate communities. *Freshwater Biology* **46**, 535-551.

Table 1. Assessment of sites upstream and downstream of wetlands in four streams of eastern Melbourne against the biological objectives for ecosystem protection of rivers and streams in the Victorian State Environment Protection Policy (Waters of Victoria).

Stream	Location	Upstream/ Downstream of wetland	SIGNAL score	SEPP objective	Number of EPT families	SEPP objective
Dandenong Creek	650 m downstream of Brady Rd, Endeavour Hills	U	4.5	FAIL	0	N/A
Dandenong Creek	300m downstream of Heatherton Rd	D	4.9	FAIL	0	N/A
River Gum Creek	End of Kiandra mews, Hampton Park	U	5.0	FAIL	1	N/A
Hampton Park Drain	End of Valley view rise, Hampton Park	U	4.5	FAIL	1	N/A
Hampton Park Drain	150m downstream intersection with Hallam Rd, Hampton Park	D	5.0	FAIL	1	N/A
Scotchmans Creek	400m downstream of Stephenson Rd, Mt Waverley	U	4.3	FAIL	1	N/A
Scotchmans Creek	350m upstream of Park Rd bridge, Chadstone	D	4.8	FAIL	2	N/A
Olinda Creek	200m upstream Hull Rd, Lilydale	U	5.8	PASS	7	PASS
Olinda Creek	Immediately downstream of Bellbird Rd, Lilydale	D	5.7	PASS	6	FAIL

Table 2. Models of change in assemblage composition upstream and downstream of wetlands in three streams. For each measure of composition, the best model (press or pulse) is indicated with the mean (and 95% credible limits) of the coefficient β, which indicates the size and direction of the effect (for SIGNAL, number of EPT families, and number of families, positive = an increase downstream of the wetland, negative = a decrease). P is the proportion of the posterior distribution of β <0 if mean is negative, or >0 if positive. Models for which P>0.90 are indicated in bold.

		Olinda			Dandenong			Scotchmans		
Variable	Season	Model	Mean of β (95% credible limits)	P	Model	Mean of β (95% credible limits)	P	Model	Mean of β (95% credible limits)	P
First MDS axis (p-a)	Autumn	Press	1.5 (0.6 - 2.2)	**1.00**	Press	0.8 (0.5 - 1.5)	**1.00**	Press	1.5 (0.4 - 2.9)	**0.99**
	Spring	Press	1.6 (2.7 - 0.8)	**1.00**	Press	1.1 (0.8 - 1.7)	**1.00**	Press	1.0 (0.0 - 1.6)	**0.98**
First MDS axis (log[x+1])	Autumn	Press	-1.7 (-2.3 - -1.0)	**1.00**	Press	1.1 (0.4 - 2.1)	**0.99**	Press	-1.2 (-2.2 - 0.0)	**0.98**
	Spring	Press	-1.0 (-1.9 - -0.3)	**1.00**	Press	1.4 (0.4 - 2.0)	**0.99**	Press	-1.7 (-3.0 - -0.6)	**0.99**
SIGNAL	Autumn	Press	-0.5 (-1.2 - 0.1)	**0.96**	Pulse	-0.6 (-1.3 - 0.3)	**0.93**	Pulse	0.0 (-0.6 - 0.5)	0.63
	Spring	pulse	-0.2 (-1.0 - 0.7)	0.72	Pulse	-0.7 (-1.2 - 0.0)	**0.98**	Press	0.6 (-0.6 - 1.5)	0.88
Number of EPT families	Autumn	pulse	3.4 (-0.8 - 7.5)	**0.94**	Press	-0.8 (-2.6 - 1.6)	0.78	Press	0.9 (-1.2 - 2.5)	0.88
	Spring	pulse	2.0 (-0.6 - 5.4)	**0.95**	Press	-0.7 (-1.6 - 0.2)	**0.96**	Press	0.2 (-0.6 - 1.1)	0.73
Number of families	Autumn	pulse	10.7 (-0.8 - 22.5)	**0.96**	Press	1.8 (-4.6 - 5.7)	0.78	Press	1.4 (-3.1 - 5.7)	0.81
	Spring	pulse	2.5 (-3.8 - 9.4)	0.80	Press	1.9 (-3.8 - 8.1)	0.78	Press	2.3 (-1.3 - 6.2)	**0.94**

Table 3. Olinda Creek. Families that exhibited a strong difference upstream and downstream of the Hull Rd wetland in autumn or spring. For each family, the best model (press or pulse) is indicated with the mean (and 95% credible limits) of the coefficient β, which indicates the effect on percentage relative abundance (positive = an increase downstream of the wetland, negative = a decrease). P is the proportion of the posterior distribution of $\beta<0$ if mean is negative, or > 0 if positive. Where no statistics are presented, the family occurred in less than 3 sites. Models for which P>0.90 are indicated in bold

	Autumn			Spring		
Family	Best model	Mean of β (95% credible limits)	P	Best model	Mean of β (95% credible limits)	P
Dugesiidae	Press	7.0 (4.8 - 8.7)	**1.00**			
Microturbellaria	Press	2.1 (0.1 - 4.9)	**0.98**			
Lymnaeidae	Press	0.3 (-0.1 - 0.7)	**0.95**	Press	0.5 (0.1 - 0.9)	**0.98**
Ancylidae	Press	-4.8 (-11.4 - 2.4)	**0.94**			
Physidae	Press	-2.7 (-4.5 - -0.7)	**0.99**	Press	1.8 (-3.4 - 6.1)	0.83
Glossiphoniidae	Press	2.1 (-0.2 - 3.9)	**0.97**			
Tubificidae	Press	18.5 (-5.6 - 39.4)	**0.95**	Press	-10.0 (-25.3 - 4.0)	**0.92**
Naididae	Pulse	2.8 (1.2 - 4.6)	**1.00**	Press	-3.4 (-15.6 - 8.8)	0.78
Atyidae	Press	7.0 (-14.4 - 34.4)	0.75	Press	10.0 (-0.3 - 20.6)	**0.97**
Simuliidae	Pulse	6.2 (5.5 - 6.9)	**1.00**	Pulse	1.6 (-0.1 - 3.3)	**0.97**
Chironominae	Pulse	1.4 (-0.4 - 3.3)	**0.95**	Press	8.1 (1.8 - 15.9)	**1.00**
Veliidae	Press	2.1 (0.6 - 3.5)	**0.99**	Pulse	4.7 (4.3 - 5.1)	**1.00**
Calamoceratidae	Pulse	7.7 (6.4 - 9.0)	**1.00**			
Leptoceridae	Press	-0.2 (-0.8 - 0.4)	0.82	Pulse	6.3 (4.0 - 8.8)	**1.00**
Conoesucidae	Press	-12.7 (-26.7 - 2.9)	**0.95**	Press	-12.1 (-33.6 - 14.4)	0.84

Table 4. Dandenong Creek. Families that exhibited a strong difference upstream and downstream of the Heatherton Rd wetland in autumn or spring. For each family, the best model (press or pulse) is indicated with the mean (and 95% credible limits) of the coefficient β, which indicates the effect on percentage relative abundance (positive = an increase downstream of the wetland, negative = a decrease). P is the proportion of the posterior distribution of $\beta < 0$ if mean is negative, or >0 if positive. Where no statistics are presented, the family occurred in less than 3 sites. Models for which P>0.95 are indicated in bold

	Autumn			Spring		
Family	Best model	Mean of β (95% credible limits)	P	Best model	Mean of β (95% credible limits)	P
Dugesiidae	Pulse	11.0 (6.4 - 16.5)	**1.00**	Press	0.5 (-0.9 - 1.5)	0.88
Planorbidae	Press	1.4 (-2.9 - 5.1)	0.81	Press	0.9 (-0.5 - 2.2)	**0.92**
Physidae	Pulse	8.4 (4.6 - 11.9)	**1.00**	Pulse	2.1 (-1.3 - 5.6)	**0.92**
Naididae	Pulse	-14.9 (-34.2 - 7.9)	**0.93**	Press	10.4 (1.7 - 21.4)	**0.99**
Atyidae	Pulse	1.7 (1.1 - 2.4)	**1.00**			
Simuliidae	Pulse	12.1 (8.8 - 15.4)	**1.00**			
Corixidae				Press	-1.0 (-2.2 - 0.3)	**0.95**

Table 5. Scotchmans Creek. Families that exhibited a strong difference upstream and downstream of the Heatherton Rd wetland in autumn or spring. For each family, the best model (press or pulse) is indicated with the mean (and 95% credible limits) of the coefficient β, which indicates the effect on percentage relative abundance (positive = an increase downstream of the wetland, negative = a decrease). P is the proportion of the posterior distribution of $\beta<0$ if mean is negative, or >0 if positive. Where no statistics are presented, the family occurred in less than 3 sites. Models for which P>0.90 are indicated in bold

	Autumn			Spring		
	Best model	Mean of β (95% credible limits)	P	Best model	Mean of β (95% credible limits)	P
Hydridae	Pulse	5.0 (2.7 - 7.0)	**1.00**	Press	2.7 (-0.5 - 5.8)	**0.96**
Tubificidae	Press	-4.9 (-21.6 - 17.6)	0.75	Press	-22.8 (-41.5 - -2.8)	**0.98**
Naididae	Press	14.6 (-3.8 - 33.3)	**0.95**	Pulse	23.4 (-8.6 - 52.0)	**0.95**
Lumbriculidae	Press	-2.0 (-4.1 - -0.2)	**0.98**	Press	-3.9 (-8.6 - 0.1)	**0.97**
Ceratopogonidae	Pulse	0.5 (0.0 - 1.0)	**0.98**			
Simuliidae	Pulse	6.4 (4.1 - 8.8)	**1.00**			
Tanypodinae	Press	-2.7 (-5.3 – 0.8)	**0.95**	Pulse	1.4 (0.9 - 1.7)	**1.00**
Orthocladiinae	Press	-2.5 (-13.1 - 3.9)	0.75	Press	7.6 (-2.7 - 13.8)	**0.95**
Chironominae	Press	-29.2 (-40.0 - -12.6)	**1.00**	Press	3.1 (-0.3 - 6.9)	**0.97**
Corixidae	Pulse	22.0 (17.0 - 27.4)	**1.00**	Press	0.5 (0.0 - 1.1)	**0.97**
Atyidae	Press	0.9 (-0.8 - 2.1)	**0.90**			

Table 6. Families identified as strong discriminators of upstream and downstream sites in each stream in each season based on presence-absence. The number of sites in which each species occurred is indicated.

			Occurrence in N sites	
Stream	Season	Family	Upstream	Downstream
Olinda	Autumn	Corixidae	0	3
		Lymnaeidae	0	3
	Spring	Corixidae	0	4
		Dugesiidae	1	4
		Glossiphoniidae	1	4
		Notonectidae	0	3
		Veliidae	1	4
		Calamoceratidae	0	3
		Ancylidae	4	1
Dandenong	Autumn	Simuliidae	0	4
		Planorbidae	0	3
	Spring	Planorbidae	0	4
		Nemertea	4	1
Scotchmans	Autumn	Atyidae	0	4
		Simuliidae	1	4
		Hydroptilidae	0	3
	Spring	Corixidae	0	3
		Tanypodinae	0	3
		Culicidae	3	0
		Microturbellaria	3	0

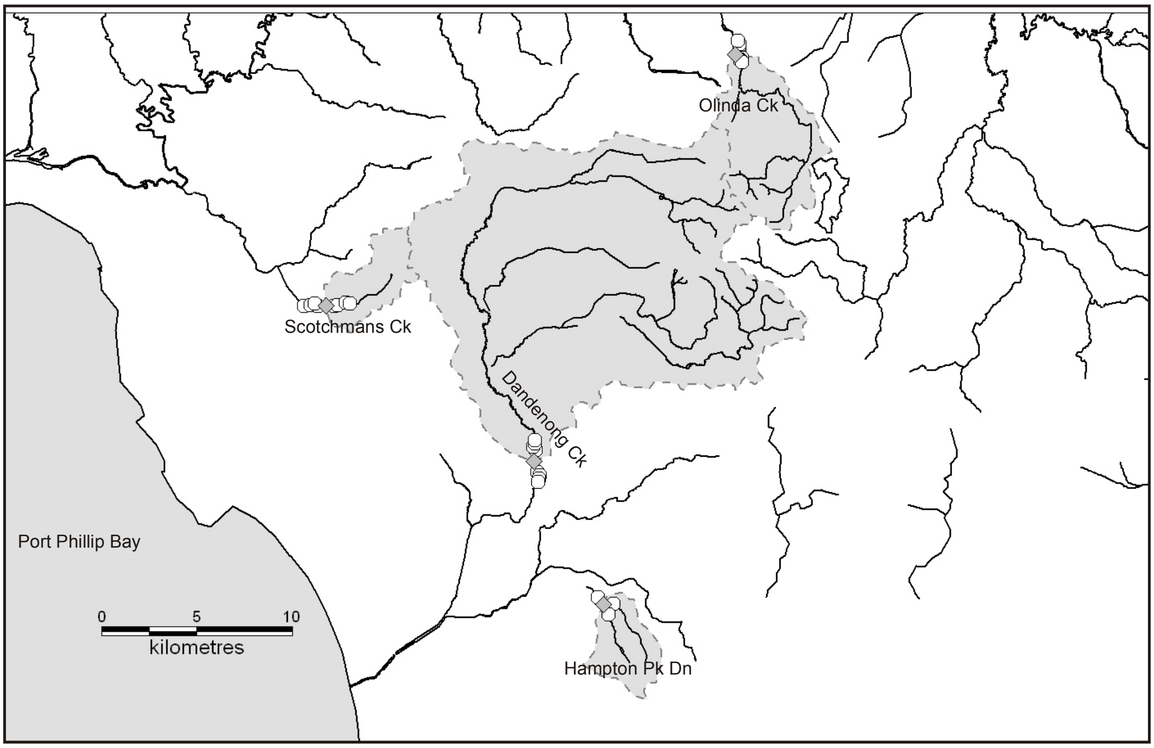

Fig. 1. Map of the study area showing the catchments of the four wetlands (shaded with dashed boundaries). Each wetland is indicated by a shaded diamond and the sampling sites upstream and downstream are indicated by open circles.

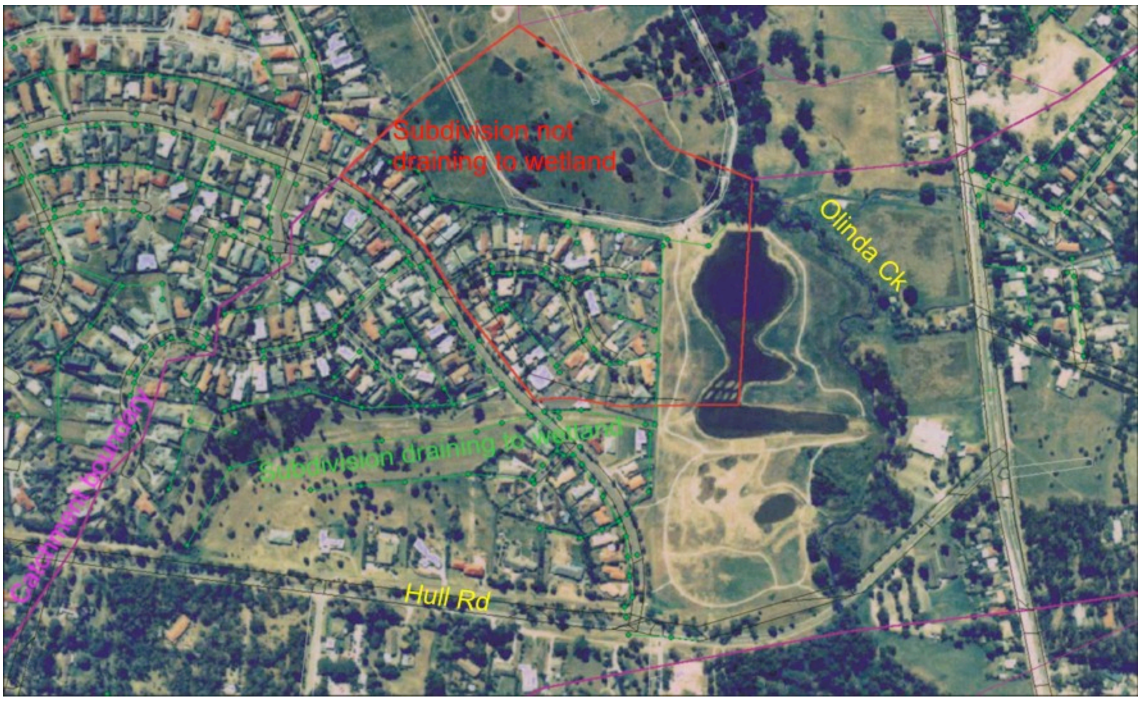

Fig. 2. Aerial orthophotograph of Hull Rd wetland under construction in February 2000. Green lines are stormwater drains. Purple lines are the subcatchment boundaries of sampling sites. An urban subdivision that drains to the creek downstream of the wetland is outlined in red.

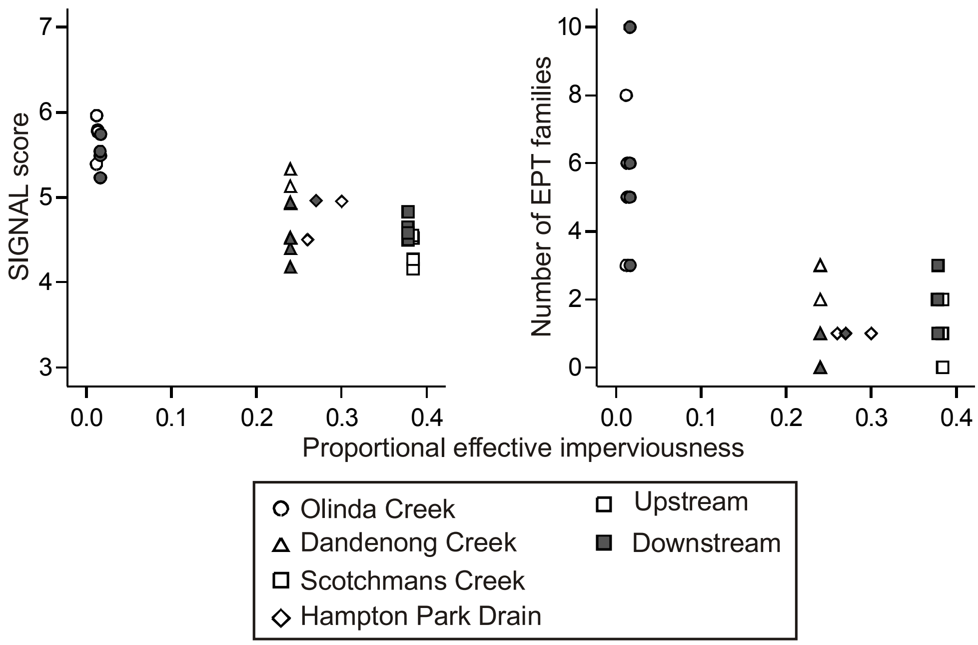

Fig. 3. SIGNAL and number of EPT families (calculated using data combined for two seasons) in all samples plotted against effective imperviousness. Symbol shapes represent streams and the fill represents position upstream or downstream of a wetland.

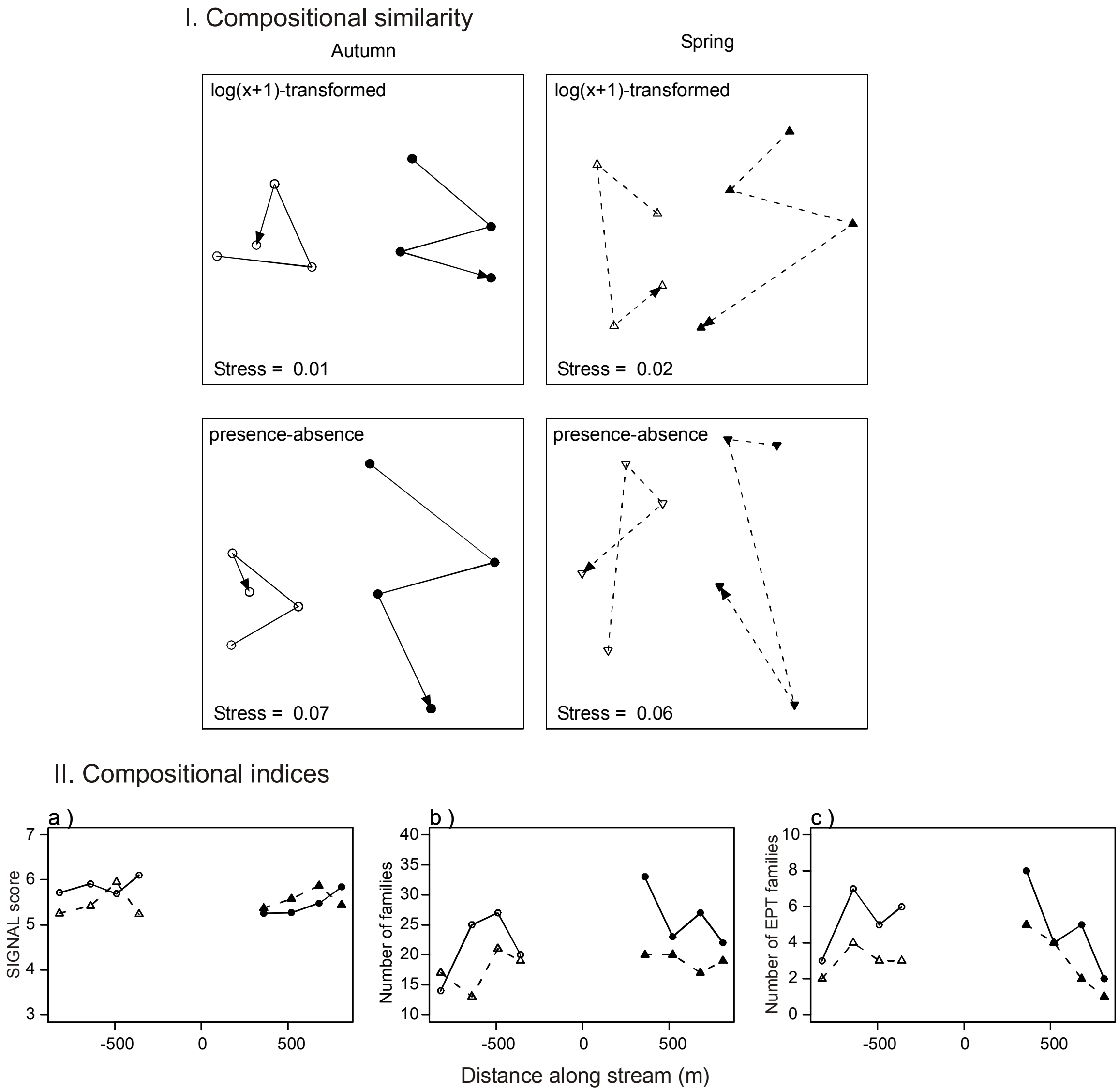

Fig. 4. Macroinvertebrate assemblage composition in Olinda Creek, upstream (closed symbols) and downstream (open symbols) of Hull Rd wetland in autumn (circles and solid lines) and spring 2003 (triangles and dashed lines). I. Assemblage compositional similarity in each season portrayed as NMDS ordinations based on log(x+1)-transformed relative abundance data and presence-absence data respectively. Arrows indicate order of sites moving downstream. II. Compositional indices: a) SIGNAL scores, b) Number of families and c) number of families of the orders Ephemeroptera, Plecoptera and Trichoptera

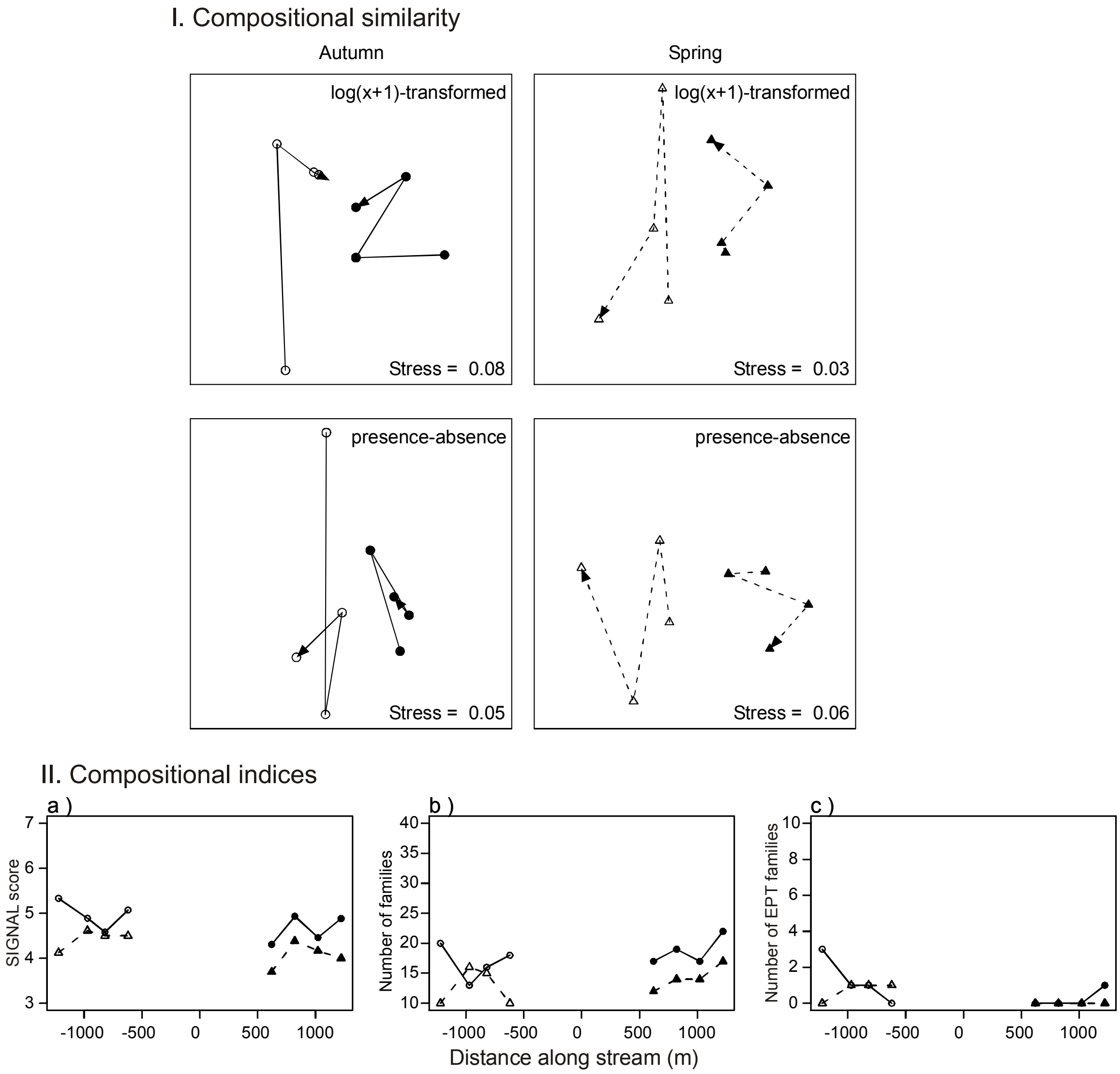

Fig. 5. Macroinvertebrate assemblage composition in Dandenong Creek, upstream (closed symbols) and downstream (open symbols) of Heatherton Rd wetland in autumn (circles and solid lines) and spring 2003 (triangles and dashed lines). I. Assemblage compositional similarity in each season portrayed as NMDS ordinations based on log(x+1)-transformed relative abundance data and presence-absence data respectively. Arrows indicate order of sites moving downstream. II. Compositional indices: a) SIGNAL scores, b) Number of families and c) number of families of the orders Ephemeroptera, Plecoptera and Trichoptera plotted against distance from the centre of the wetland.

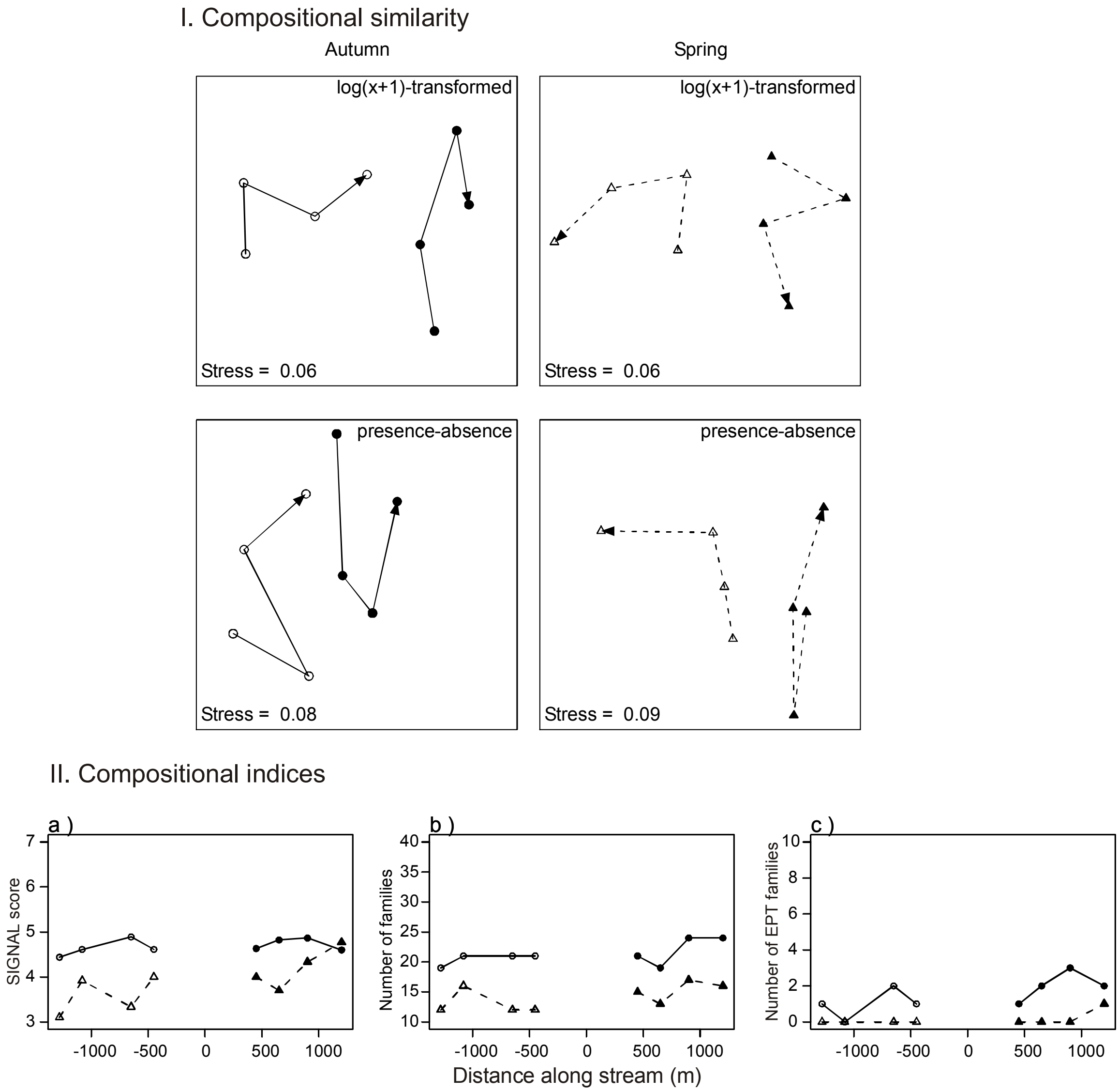

Fig. 6. Macroinvertebrate assemblage composition in Scotchmans Creek, upstream (closed symbols) and downstream (open symbols) of Huntingdale Rd wetland in autumn (circles and solid lines) and spring 2003 (triangles and dashed lines). I. Assemblage compositional similarity in each season portrayed as NMDS ordinations based on log(x+1)-transformed relative abundance data and presence-absence data respectively. Arrows indicate order of sites moving downstream. II. Compositional indices: a) SIGNAL scores, b) Number of families and c) number of families of the orders Ephemeroptera, Plecoptera and Trichoptera plotted against distance from the centre of the wetland.

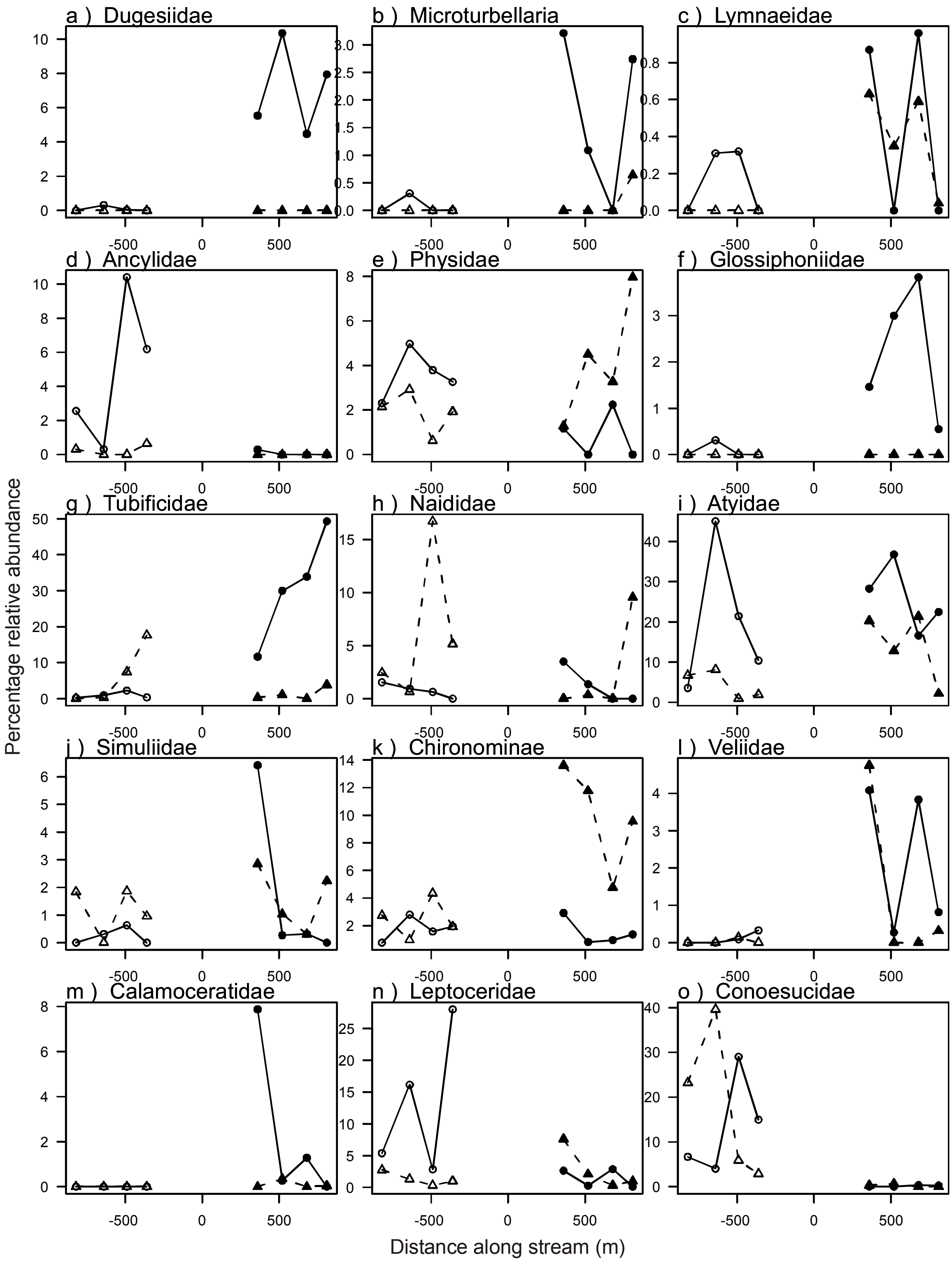

Fig. 7. Relative abundance of macroinvertebrate families in Olinda Creek, upstream (closed symbols) and downstream (open symbols) of Hull Rd wetland in autumn (circles and solid lines) and spring 2003 (triangles and dashed lines). Only those families showing a strong upstream-downstream difference in either season are illustrated.

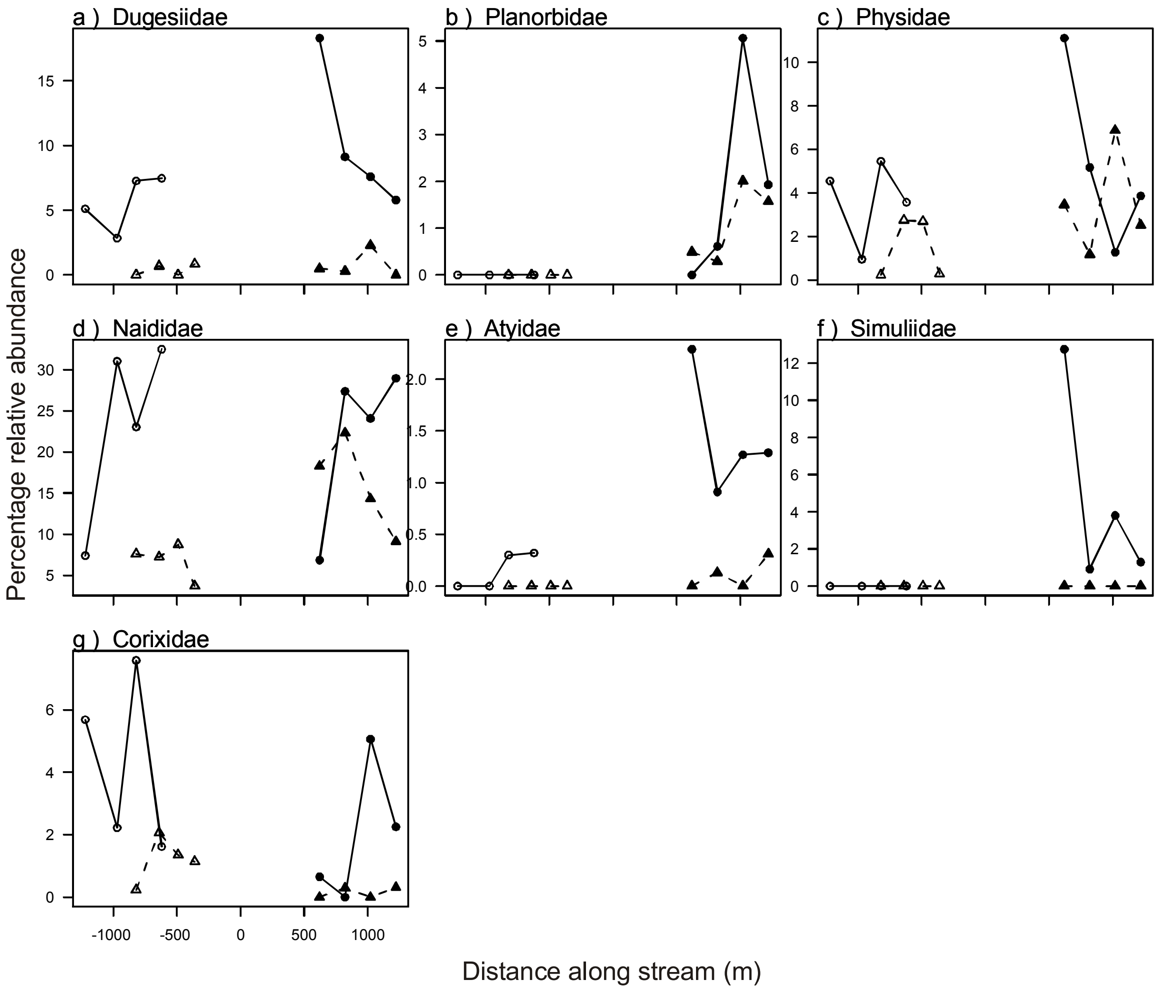

Fig. 8. Relative abundance of macroinvertebrate families in Dandenong Creek, upstream (closed symbols) and downstream (open symbols) of Heatherton Rd wetland in autumn (circles and solid lines) and spring 2003 (triangles and dashed lines). Only those families showing a strong upstream-downstream difference in either season are illustrated.

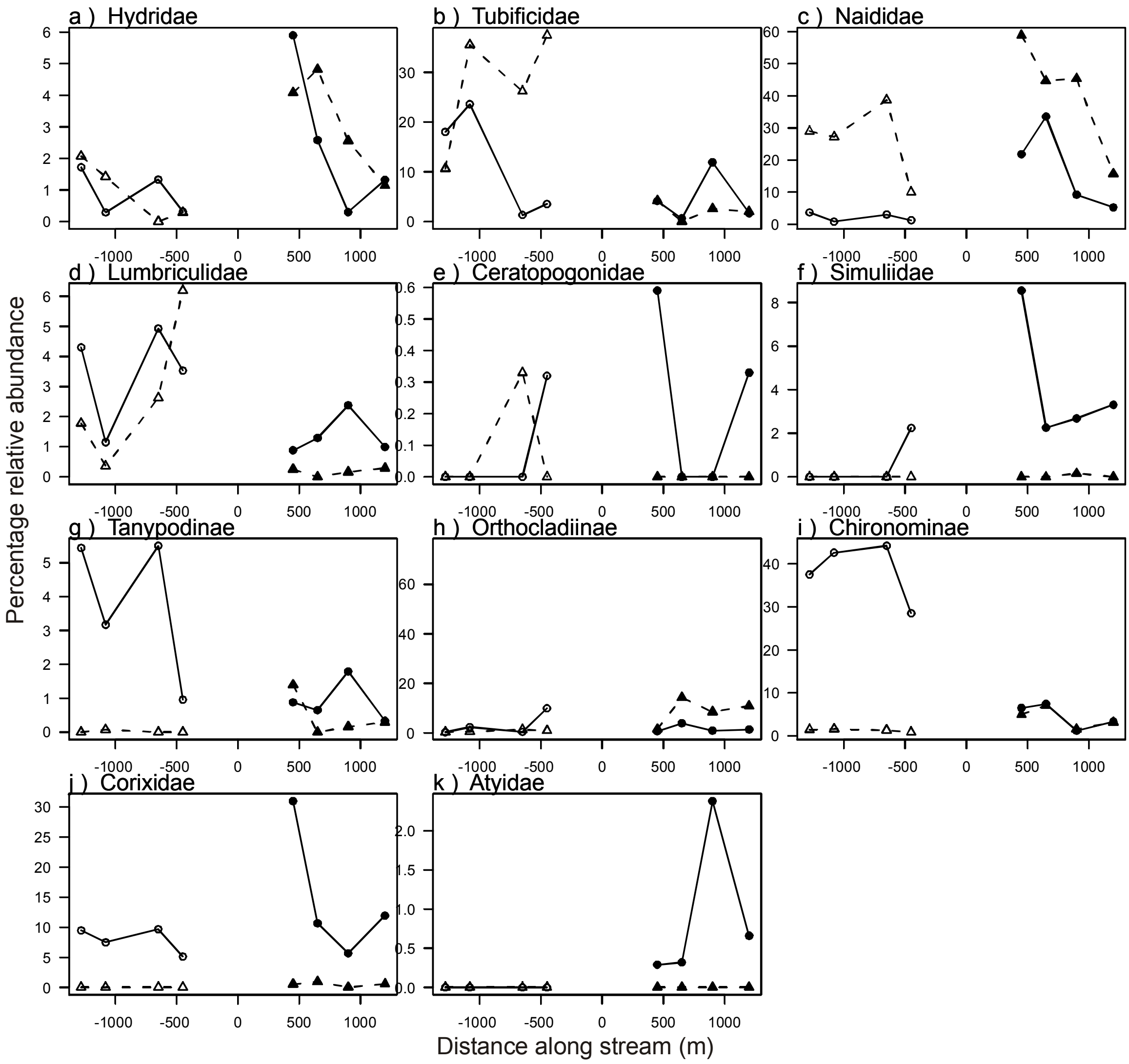

Fig. 9. Relative abundance of macroinvertebrate families in Scotchmans Creek, upstream (closed symbols) and downstream (open symbols) of Huntingdale Rd wetland in autumn (circles and solid lines) and spring 2003 (triangles and dashed lines). Only those families showing a strong upstream-downstream difference in either season are illustrated.

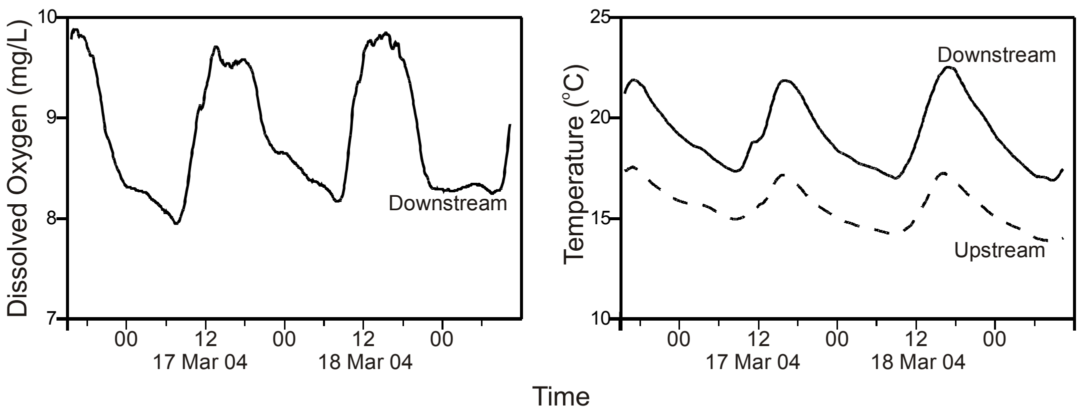

Fig. 10. a) Dissolved oxygen concentration in Olinda Creek immediately downstream of the outlet of Hull Rd wetland over three days in March 2004. b) Temperature in the same location over the same period and upstream of Hull Rd and the wetland over the same period.

Appendix 1.

WinBUGS script for the autoregressive model used in this study

```
model {
      mu[1] <-  a + b*pond[1]
      score[1] ~ dnorm(mu[1],tau.u)

  for (d in 2:8){           ## loop over sites 2 to 8 along stream
    mu[d] <- a*(1-rho)
    + b*(pond[d] - rho*pond[d-1])
    + rho*score[d-1];
    score[d] ~ dnorm(mu[d], tau.e);
  }

  sigma.e <- 1/tau.e                ## convert precision to variance
  sigma.u <- sigma.e/(1-pow(rho,2)) ## regression error variance
  tau.u <- 1/sigma.u

  ## priors
  rho ~ dunif(-1,1);               ## uniform prior on stationary interval
  a ~ dnorm(0, 0.001);
  b ~ dnorm(0, 0.001);              ## vague normal priors
  tau.e ~ dgamma(.05, .05);        ## vague prior on sigma
      }
```

R script using the package R2WinBUGS

```
## the above model should be saved as a txt file and linked as
## an R object as shown in the R2WinBUGS example
press <- c(0,0,0,0,1,1,1,1)
pulse <- c(0,0,0,0,1,0,0,0)
## Autumn Olinda SIGNAL data as an example
score <- c(5.714285714,5.909090909,5.692307692,6.105263158,
           5.258064516,5.272727273,5.48,5.842105263)
parameters <- c("a","b","rho","tau.e")
inits <- function()
            {
            list(a = runif(1, 0, 10), b = rnorm(1, 0, 0.1),
                 rho = runif(1, -1, 1),tau.e=rgamma(1,0.1,0.1))
             }
pond <- press  ## (or pulse)
data <- list("score","pond")
os <- bugs(data,inits,parameters,model.file,
                    n.chains=3,n.iter=10000,
                    bugs.directory = "c:/Program Files/WinBUGS14/",
                    working.directory ="c:/…")
##  enter correct working directory in last line
```