Stewart, B.J., Strehlow, K. and Davis, J.A. (2009) Impacts of deep open drains on water quality and biodiversity of receiving waterways in the Wheatbelt of Western Australia. Hydrobiologia, 619 (1). pp. 103-118.

Impacts of deep open drains on water quality and biodiversity of receiving waterways in the Wheatbelt of Western Australia

Barbara Stewart[1], Karin Strehlow [2], Jenny Davis[2]

[1]Centre of Excellence in Natural Resource Management, University of Western Australia, Albany, Australia

[2]School of Environmental Science, Murdoch University, Murdoch, Australia

Abstract

Extensive networks of deep drains are being built in Western Australia to reduce the effects of dryland salinity on agricultural lands. Most of these drains discharge into natural river and wetland systems, with little consideration given to the environmental impacts. This study examined the downstream ecological impacts of one of the oldest deep drain networks in Western Australia, located in the Wakeman subcatchment near Narembeen. Twelve sites were sampled bi-monthly from October 2004 to September 2006. On each occasion, water quality parameters were measured and the macro-invertebrate fauna was sampled. Significant differences in water quality and macro-invertebrates were observed between the untreated sites and those affected by the drain discharge. Surface water at untreated sites was always fresh (<3 ppt), alkaline (pH 7.6–8.9) and turbid (49–600 NTU), whereas treatment sites were always saline (28–147 ppt), acidic (pH 1.9–3.8) and mostly clear (0–100 NTU). No recovery of water quality was observed with distance from discharge point (20 km). Invertebrates reflected differences in water quality, with drain discharge resulting in a sharp decline in species richness, and significant changes in macro-invertebrate community composition. Sites affected by drain discharge were dominated by fly larvae such as Orthocladiinae and Ceratopogonidae. Microcrustaceans were far more abundant at sites unaffected by drainage. The ecological values of

Wheatbelt streams are likely to be further compromised by discharge of poor water quality from deep drainage.

Keywords: Secondary salinisation; Downstream impacts; Water quality; Acidity Biodiversity; Macro-invertebrates

Introduction

Secondary salinisation is particularly acute in Western Australia, and has been largely attributed to the clearing of deep-rooted perennial vegetation to grow shallow-rooted and annual agricultural plants (e.g. Clarke et al., 2002). This has resulted in decreased interception of rainwater by the vegetation canopy, leading to rises in groundwater levels, so that salt previously stored in soil has been mobilised. It has been suggested that a third of cleared land in the agricultural region of south-western Australia will become saline as a result of these processes (Clarke et al., 2002), leading to the loss of a significant number of plant and animal species. To date, dryland salinity has been estimated to affect over 1 million ha of previously productive farmland, and to cost Western Australia in the region of $1.5 billion in lost agricultural productivity (Kay et al., 2001).

A number of biological and engineering management options, including tree planting, use of salt resistant plants, deep drainage and groundwater pumping have been proposed to reduce impacts of salinity and water-logging on agricultural land. Of these, deep drains (2–3 m) are seen by farmers as the most likely solution to succeed, and it has been estimated that over 12,000 km of drains and banks have been installed (Dogramaci & Degens, 2003; Ali et al., 2004). Almost all of these drains discharge into streams and rivers and little consideration has previously been given to the ecological impacts of these discharges on already altered aquatic ecosystems. One of the potential hazards of deep drainage is the release of acid groundwater, which has the potential to release metals and other elements potentially harmful to the biota and ecology of the receiving waterways and wetlands. In particular, a review of historical groundwater records (Rogers & George, 2005) has demonstrated that groundwaters in the eastern wheatbelt are generally strongly acidic ($pH < 3.5$) and typically very saline (6,000–8,000 mS/m). Drains in this region were found to have high levels of iron, aluminium, cobalt, copper, zinc and lead, and during periods of low flow, exhibit extreme acidity (pH < 3) and salinity (10,000–20,000 mS/m). Dogramaci & Degens (2003) noted that there was a lack of specific information on the effects of this low quality water on downstream communities, whilst Rogers & George (2005) recommended that an assessment of the impact of these highly acidic and saline drain discharges on receiving environments was critical. In addition, the possibility that potential ecological benefits might be provided by discharging to already degraded systems needs to be explored (Dogramaci & Degens, 2003).

The Government of Western Australia has initiated an evaluation aimed at improving the use and understanding of engineering options such as deep drains to alleviate salinity in agricultural landscapes. This initiative through its three main programs addresses the performance of specific engineering options at farm scale, evaluates regional drainage plans, and also recognises the importance of identifying safe disposal options. As part of this process, the downstream impacts of drains and pumping have been assessed in four catchments, including in the Wakeman Creek subcatchment near the Wheatbelt town of Narembeen. Deep drains have been constructed at various times during the period 1998–2001 in the Wakeman subcatchment near Narembeen (Ali et al., 2004), resulting in an extensive local deep drain network of approximately 100 km in length. During 2005–2006, daily flow rate from this drainage complex usually ranged between 0 and 15 ML/day, but ‘unseasonal’ rainfall in summer 2006 resulted in a high of 25 ML/day being recorded during this period. Drain flow is a mixture of surface and groundwater discharge, as the drain is an open system that receives both surface runoff and groundwater. As is the case for many deep drains in the Wheatbelt, these have been constructed to alleviate the effects of salinity on both township infrastructure and agricultural productivity. However, many of these drains have been constructed with limited planning and design, and with little or no understanding of their effectiveness or downstream impacts (Ali et al., 2004). In the case of the Narembeen drains, their effectiveness in lowering groundwater levels has been assessed by scientists (Ali et al., 2004), but the downstream effects of these drains has not received attention. Since the discharge from these drains was found to be acidic (pH ranging from 2 to 4), highly saline (electrical conductivity values ranging from 4,000 to 10,000 mS/m), and carrying significant levels of heavy metals such as iron and aluminium (about 110–120 mg/l at some sites) (Ali et al., 2004), the lack of information on downstream impacts is of concern.

This research reports on the results of an investigation of the ecological impacts of the deep drain network situated in the Wakeman subcatchment near Narembeen. More specifically, the objectives of this study were (i) to evaluate the water quality characteristics of upstream, drain and downstream sites in the Wakeman subcatchment, (ii) to quantify the ecological values of upstream, drain and downstream sites in the Wakeman subcatchment, with a view to evaluating the impacts of deep drainage on the ecology of receiving waterbodies and (iii) to provide recommendations regarding the management of discharge waters from engineering options such as deep drains.

Methods

Study sites

As a result of the absence of collection of data prior to the construction of the drain network, multiple sites were selected in the Wakeman subcatchment near the town of Narembeen in order to make inferences about the disturbance based on differences in spatial patterns. Four categories of sites were selected: (i) ‘comparison’ sites, located on neighbouring Wheatbelt creeks of comparable stream order, but not impacted by deep drainage, (ii) ‘upstream’ sites, located upstream of the constructed drains, (iii) ‘drain’ sites, located along the two main branches of the drain complex and (iv) ‘downstream’ sites, located on Wakeman Creek at various distances downstream of the termination of the drain complex. Wakeman Creek is a tributary of the Salt River. A large proportion of the valley floor soils in the Wakeman subcatchment was cleared by the 1930s, including riparian zones along the local waterways. Average rainfall for the subcatchment is about 325 mm per annum, and the predominant landuse in the area involves the growing of wheat, barley, pastures and lupins. Accordingly, a total of 12 sites, consisting of two ‘comparison’ (C1 and C2), two ‘upstream’ (U1 and

U2), four ‘drain’ (D1–D4) and four ‘downstream’ (DS1–DS4) sites were sampled approximately bi-monthly (22 September and 15 December 2004, 2–3 March, 2–3 June, 16–17 August, and 16–17 November 2005, 7 June and 27 September 2006) during the period September 2004 to September 2006 (Table 1; Fig. 1). Because of the extensive nature of the drain network, very few suitable upstream sites could be found. The selection of comparison sites also proved difficult, as suitable sites in this category were also scarce. Eventually, only two comparison sites (C1 and C2), located on a neighbouring system (Yawerlin Creek) not impacted by deep drainage, were selected. Yawerlin Creek is a second order tributary of the Salt River. The location of each site was determined using a hand-held Magellan Meridian GPS using datum GDA 1984.

Measurement of water quality variables

Electrical conductivity (mS/cm), salinity (parts per thousand), pH, temperature (°C), dissolved oxygen content (mg/l and % saturation), oxidation reduction potential (mV) and turbidity (NTU) were measured, in-situ, using a Yeo-Kal 611 multi-parameter water analyzer. The conversion between conductivity and salinity was performed using a conversion factor of 0.64. Following Pinder et al. (2004), water was considered ‘fresh’ when salinity was <3 ppt, ‘subsaline’ when 3–10 ppt, and saline when >10 ppt. Values greater than 100 ppt signify hypersaline conditions. Seawater is approximately 35 ppt.

Ecological sampling

Macro-invertebrates have been selected as the key indicator group for assessment of the health of Australia’s rivers using biological indicators under the National River Health Program (Smith et al., 1999). Macro-invertebrates are found in most habitats, generally have limited mobility, are easy to catch using standard collecting methods, and have a range of sensitivities to changes in water quality and habitats, and are thus good indicators of ecological ‘health’.

For the sampling of macro-invertebrates (defined in this study as those invertebrates retained by a 250-μm sieve), a 10 m stretch of stream located at the centre of a study reach was selected. After disturbing the benthos using a combination of kick sampling and loosening of stones and large woody debris (if present) by hand, a 250-μm mesh net was used to sweep over 10 m^2 of streambed (see Kay et al., 2001). After rinsing off the leaves, twigs and other debris, these were discarded. Half of each sample was placed immediately in sampling containers with 70% ethanol, and returned to the laboratory for further processing, whilst the remaining half was washed through a stack of sieves with mesh apertures of 2 mm, 500 μm and 250 μm. The contents of each sieve were emptied into large white trays for sorting. Using pipettes, animals were picked out from all three trays for a total of either 60 min (one operator) or 30 min each (two operators). Care was taken to maximise the diversity of animals sampled by starting with common, abundant taxa for the first 5 min, then switching attention to finding the less common, inconspicuous taxa. Animals were placed in labelled vials containing 70% ethanol, and stored for later laboratory processing when all specimens were identified to species level and counted. The retention of half of each sample for later processing in the laboratory was followed in order to check the accuracy and suitability of the 'live pick' method.

Specimens were identified to species level where possible, using a variety of resources. Consistency of identification with previous studies of Wheatbelt systems (e.g. Pinder et al., 2004) was achieved by examination of a voucher collection based within the Department of Environment and Conservation (formerly Department of Conservation and Land Management). Species codes for undescribed species were used as per this voucher collection.

Statistical analyses

Analysis of variance (ANOVA) and Tukey's multiple comparison tests were used to test for significant differences among treatment groups (upstream, downstream, drain and comparative sites) for selected water quality variables. Scatter plots and linear regressions were conducted to establish correlations between selected environmental variables and species richness. All univariate analyses were undertaken using GenStat (Lane et al., 1988).

Patterns in the environmental data across sampling sites were investigated by a multivariate ordination. Multivariate data analyses were conducted using PRIMER (Plymouth Routines in Multivariate Ecological Research) (Clarke, 1993). As the physico-chemical variables measured were on mixed measurement scales, each variable was first normalized (by subtracting the mean and dividing by the standard deviation) to put them on a common, dimensionless measurement scale, and standard Euclidean Distance was used as a measure of dissimilarity between every pair of samples for each sampling occasion. Samples were clustered and a dendrogram plot generated. In addition, samples were also ordinated using Principal Components Analysis (PCA), a common approach for the multivariate analysis of environmental data. Data were initially transformed ($\log(1 + V)$), and a two-dimensional plot produced. Sites were grouped according to 'treatment', and selected individual water quality parameters were displayed in relation to overall patterns using bubble plots.

For the biological data, similarities were calculated between every pair of sites for each sampling occasion, using the Bray–Curtis coefficient following transformation of the original relative abundance values by taking the square root of every entry in the data matrix. Non-metric multidimensional scaling (MDS) was used to display the biological relationships among the sampling sites on each sampling occasion, such that the distances between pairs of samples reflected their relative dissimilarity of species composition. Thus, points that are close together represent samples that are very similar in species composition, and points that are further apart correspond to very different communities. The adequacy of these MDS representations were assessed by considering the stress function, whereby the following 'rule-of-thumb' for two-dimensional ordinations were followed as proposed by Clarke & Warwick (1994): stress < 0.05, an indication of an excellent representation; stress < 0.1, good ordination with no real prospect of a misleading interpretation; stress < 0.2, a potentially useful two-dimensional picture needing confirmation through other analyses, and stress > 0.3, points close to being arbitrarily placed, and thus potential for misinterpretation. Based on these values, MDS plots with stress values above 0.2 were considered likely to be poor representations, and were thus deemed unreliable. For higher values of stress, Clarke & Warwick (1994) recommend that both MDS and cluster analyses be undertaken, and that the results of both are viewed in combination. Accordingly, samples were clustered using group average linkage, and the output plotted as a dendrogram. The corresponding cluster analysis results were then superimposed onto the MDS plots in the form of polygons corresponding to given similarity thresholds.

In order to test the null hypothesis that there were no assemblage differences between the four categories of samples (comparative, upstream, drain and downstream), an analysis of similarities (ANOSIM), a rough analogue of the standard univariate ANOVA test, was conducted.
Pairwise R values which result from these analyses give an absolute measure of how separated the groups are, on a scale of 0 for groups which are indistinguishable to a value of 1, where all similarities within groups are less than any similarity between groups. The Rstatistic is thus a useful comparative measure of the degree of separation of sites. For cases of few replicates (and hence often low significance levels), these pairwise R values can be interpreted as follows: $R > 0.75$, groups well separated; $R > 0.5$, groups overlapping but clearly different, and $R < 0.25$, groups barely separable at all (Clarke & Warwick, 1994). Clarke & Warwick (1994) have noted that R values are never more than about 0.15 by chance.

Taxa primarily responsible for the observed assemblage differences between groups were identified using the similarity percentages (SIMPER) routine in PRIMER. This routine calculates the overall percentage contribution each taxon makes to the average dissimilarity between two groups, and lists taxa in decreasing order of their importance in discriminating two groups.

Results

Water flow

Although there was usually always water present at drain (D1–D4) and downstream (DS1–DS4) sites on all sampling occasions, with one exception, upstream (U1–U3) and comparative (C1–C2) sites were dry when visited in September and December 2004, March and November 2005 and September 2006. Site C2 did have surface water present when sampled in November 2005. Below average winter rainfall was recorded for this region during the study period (R. Ali, personal communication).

Patterns in environmental data across sites

Patterns in environmental data across sites were investigated using PCA. Figure 2 displays the first two axes (PC1 and PC2) of a PCA ordination on the transformed water quality data. The first component accounted for 62.7%, and the second, 17.0%, of the variability in the data matrix, and thus together, these two components account for 79.7%, providing a reasonable summary of the sample relationships. Broadly speaking, PC1 represents an axis of increasing pH and turbidity, and decreasing conductivity and ORP, whilst oxygen decreased up the PC2 axis. Drain and downstream sites always formed a cohesive group, regardless of the time of sampling, and were well differentiated from the upstream and comparative sites, which formed a separate grouping. The formation of these two main groupings was confirmed by the cluster analysis (Fig. 3). The group consisting of the drain and downstream sites was characterised by waters which were markedly acid, strongly saline, but with low turbidity. On the other hand, comparative and upstream sites were far more alkaline, less saline, but often more turbid. The cluster analysis identified the existence of two subclusters within the drain and downstream group, with sites sampled in June and August 2005 plotting separately from sites sampled in the drier months (September to March). Values for pH were extremely low, and salinity, very high, for these sites during this period.

pH

On all sampling occasions when water was present, the surface water in both upstream and comparative sites was always alkaline, with values ranging from 7.78 to 8.44 for upstream sites (mean of 7.99) and 7.63 to 8.89 for comparative sites (mean of 8.44). Values for pH were always significantly higher at these sites than for drain and downstream sites (ANOVA, $P < 0.01$). Water at drain and downstream sites was similar (ANOVA, $P > 0.05$), and was always extremely acidic, ranging from 2.10 to 3.65 for drain sites (mean of 2.92), and 1.94 to 3.81 for downstream sites (mean of 2.84) over the study period. Water in drain and downstream sites was most acidic in the drier months sampled (Fig. 4).

There was no evidence of recovery of pH with increasing distances downstream of the drain. For example, when sampled in June 2005 (plot not shown) and August 2005, although initially alkaline at upstream sites, water was strongly acidic at drain sites, and remained so at all downstream sites sampled, including at site DS4 which is located near the junction of Wakeman Creek with the Salt River, approximately 20 km downstream of the drain (Fig. 5). There are no major tributaries flowing into the Salt River upstream of the junction of Wakeman Creek and the Salt River.

Conductivity

When surface water was present at upstream and control sites, this water was always fresh (equivalent to salinity values < 3 ppt) (Fig. 4). Conductivity levels ranged from 0.2 to 0.9 mS/cm (equivalent to 0.1–0.4 ppt) at comparative sites (mean of 0.6 mS/cm), and 0.1–0.5 mS/cm (0.1–0.2 ppt) at upstream sites (mean of 0.2 mS/cm). In contrast, drain and downstream sites were always saline (>10 ppt). The lowest conductivity value recorded for a drain site was 43.7 mS/cm (about 28.1 ppt) for site D4 sampled in June 2006, and the highest, 175.3 mS/cm (about 113.1 ppt) for site D2 sampled in March 2005. The latter value is more than three times the salinity of seawater. Drain sites had a mean conductivity value of 82.4 mS/cm for all sites and sampling occasions. Downstream sites were equally saline (ANOVA, $P > 0.01$). Conductivity ranged from 61.7 mS/cm at site DS1 in August 2005 (about 41.5 ppt) to as high as 221.6 mS/cm (about 146.8 ppt) at site DS2 in November 2005, with a mean value of 107.7 mS/cm for all sampling occasions. For the 20 km 'downstream' reach investigated, there was no evidence of a reduction in conductivity with increasing distances downstream of the drain (Fig. 5).

Turbidity

Both upstream and comparative sites were significantly more turbid than drain and downstream sites (ANOVA, $P < 0.001$; Fig. 4). Turbidity values were similar for the former two categories of sites, and ranged from 49.6 (site C2 in June 2006) to 600 NTU (site U1 in June and August 2005). Water at drain sites was generally less turbid (mean of 9 NTU, range of 0–62.7 NTU), as was the case for downstream sites (mean of 19.0 NTU, range of 1–100.3 NTU). Although relatively clear and thus lacking suspended solids, water at downstream sites was often noticeably yellow in colour. This is in contrast to other findings for sites downstream of engineering works (deep drains or groundwater pumping) near Dumbleyung and Tammin, where flocculent orange plumes composed of ferric hydroxides resulted in highly turbid water at these sites (pers. obs.).

Oxygen reduction potential and dissolved oxygen

When interpreting oxidation reduction potential (ORP), values between 200 and 250 mV are considered ‘standard’ for uncontaminated water. The ORP values recorded for this study are considerably higher than this ‘standard’ range, particularly at drain (values of 630–751 mV) and downstream (665–794 mV) sites. Values for comparative (469–541 mV) and upstream (374–476 mV) sites also exceeded the recommended range.

With one or two exceptions, water at all sites on all sampling occasions was well oxygenated.

Macro-invertebrate taxa richness

As no submerged or emergent vegetation was present at any of the sites on any of the sampling occasions, all macro-invertebrates collected were associated with the stream bed. The total number of macro-invertebrate species found at all sites for all sampling occasions ranged from one (site D3 in June 2006) to 32 (site C2 in June 2006). Mean total species richness across all sampling occasions differed significantly among treatments (comparative, drain, upstream and downstream) (ANOVA, $P < 0.001$). Mean species richness at upstream sites (mean of 11.5) was similar to that at comparative sites (mean of 17.5) (ANOVA, $P > 0.05$), and both were significantly higher than at drain (mean of 6.24) and downstream (mean of 3.39) sites (ANOVA and Tukey’s test, $P < 0.05$). Although taxa richness was generally poorly correlated with either pH or conductivity ($Y = 1.9022X - 0.7944$, $R^2 = 0.543$ for pH, $Y = -1.6213\mathrm{Ln}(X) + 12.158$, $R^2 = 0.440$ for conductivity, plots not shown), there was a tendency for taxa richness to increase with increasing pH levels, and to decrease with increasing conductivity levels.

Macro-invertebrate community composition

Insects and crustaceans dominated the invertebrate community (Fig. 6). Microcrustaceans were present at all but two sites (DS2 and DS4) in August 2005, and dominated the communities at the comparative (C1 and C2) and upstream (U1 and U2) sites. The depauperate communities at downstream sites (between two and six species recorded in August 2005) were made up of mainly dipteran fly species, springtails (Collembola; associated with the banks and water surface) and a microcrustacean species. Mayflies (Order Ephemeroptera), stoneflies (Order Plecoptera) and caddisflies (Order Trichoptera) are generally more sensitive to pollution and degradation, and have been used to develop the so-called 'EPT' index (Lenat, 1988). Although mayflies and stoneflies were absent, caddisflies (Trichoptera) were collected from sites U1 and U2.

Patterns in community assemblages and abundances

When the biological relationships among the 12 sites sampled approximately bi-monthly during the period September 2004 to September 2006 were displayed using a two-dimensional MDS ordination (Fig. 7), distinct clusters emerged. Drain and downstream sites grouped together, whilst upstream and comparative sites were more similar to each other than the drain and downstream sites. These clusters were in general agreement with those from a matching cluster analysis (Fig. 8).

An ANOSIM analysis indicated that there was a significant difference among the four categories (drain, upstream, downstream and comparative) of sites (Global R statistical value of 0.531, $P < 0.1\%$), with drain sites being different from comparative ($R = 0.851$) and upstream ($R = 0.946$) sites (Table 2). Similarly, downstream sites differed significantly from comparative sites ($R = 0.960$) and upstream sites ($R = 0.994$). However, drain and downstream sites were barely separable ($R < 0.25$), and upstream and comparative sites showed large overlap ($R = 0.365$).

Taxa most responsible for differentiation between comparative and drain and downstream sites included several microcrustaceans—small, unidentified Ostracoda sp. A, calanoid and cyclopoid copepods, cladocerans (water fleas), as well as specimens of *Paralimnophyes pullulus* Skuse 1889 (family Chironomidae, subfamily Orthocladiinae) (Table 3). On the whole, microcrustaceans were far more abundant at the comparative than at the drain and downstream sites (Fig. 9). In contrast, *P. pullulus* larvae were far more abundant at drain and downstream sites than at comparative (and upstream) sites.

For example, calanoid copepods and cladocerans were far more common at upstream than at drain and downstream sites. The chironomid fly species *Procladius paludicola* Skuse, 1889 was also abundant at upstream sites, but rare or absent at drain and downstream sites. The bug species, *Micronecta gracilis* Hale 1922 was another species, which was abundant at upstream (and comparative) sites, but was found less frequently or not at all at drain and downstream sites. In contrast, biting midges belonging to the family Ceratopogonidae (Ceratopogonid sp. A) favoured the drain and downstream sites.

The main differences found among the comparative and upstream sites were due to the presence of several ostracod species at the former sites. For example, Ostracoda sp. B was moderately abundant at the comparative sites, but absent at the Wakeman Creek sites.

Discussion

Impacts of high salinity and acidity on biodiversity

Water in the drain, and downstream of the drain was moderately to strongly acidic, saline, but generally clear and well oxygenated. These water quality conditions were tolerated by only very hardy species, as increased salinity and acidity of sites impacted by deep drainage led to a sharp decline in species richness, and significant changes in macro-invertebrate community composition downstream of the drain.

Kay et al. (2001) showed a general tendency for family richness in Wheatbelt streams to decrease with increasing salinity, whilst Pinder et al. (2004, 2005) further demonstrated that salinity has a major influence on the distribution of aquatic invertebrates in the Wheatbelt. They showed that the salinity range of 10–20 ppt (g l^{-1}) is associated with a major shift in community composition.

Consequently, saline systems are characterised by the presence of a limited suite of invertebrates comprising two main elements—the most salt-tolerant of what are regarded as 'freshwater' species (halotolerant), and a restricted subset of halophilic fauna. Halophiles are species that primarily inhabit salt lakes and other naturally saline systems. Of the 957 aquatic invertebrate taxa recorded by Pinder et al. (2004, 2005) from the Wheatbelt, 134 were classified as halophiles, whilst the remaining 752 species were considered 'freshwater' species, albeit that some of these were eurytolerant. Only 17% of the 752 freshwater species were collected at salinities >10 g l^{-1} by Pinder et al. (2004, 2005). Drain and downstream sites on Wakeman Creek were always saline (>10 ppt), resulting in the occurrence of only the most hardy of the eurytolerant species at these sites.

However, high salinity values alone cannot account for the scarcity or absence of certain species from drain and downstream sites. For example, the eurytolerant midge, *P. paludicola*was mostly absent, or occurred at very low densities at the drain and downstream sites. This midge was collected from water with TDS values up to about 250 ppt (g l^{-1}) when sampled as part of the Wheatbelt aquatic fauna survey (Pinder et al., 2005), and thus salinity levels at drain and downstream sites on Wakeman Creek were well within the tolerance limits of this species.

Of significance is the fact that these sites were also markedly acidic. Recognition of this problem has led to an investigation of potential treatment options aimed at amelioration of acidity and trace element reduction (e.g. Douglas & Degens, 2006). Whilst the response of Wheatbelt aquatic fauna to extreme levels of acidity is yet to be investigated in any detail, studies in other parts of the world suggest that unnaturally acidic conditions can detrimentally effect aquatic biodiversity (e.g. Feldman & Connor, 1992; Rosemond et al., 1992; Berezina, 2001; Petrin et al., 2007), resulting in a decline in diversity, and a disruption in ecological processes in temperate systems. Berezina (2001) demonstrated that macro-invertebrate communities (sourced from the Upper Volta basin in Russia) formed in experimental mesocosms had a lower species richness for pH conditions of either <4 or >9, confirming earlier findings (e.g. Cranston et al., 1997) that water bodies with pH < 4 harbour communities of very few species. In Berezina's (2001) study, mollusks and oligochaetes were generally absent at these low pH levels, crustaceans, caddisflies, mayflies and odonates were rare, leaving only a few mainly highly tolerant, ceratopogonid and chironomid fly species. In particular, acidity has been shown to strongly influence mayfly and caddisfly assemblages (Feldman & Connor, 1992; Rosemond et al., 1992; Petrin et al., 2007).

Although levels of acidity were not the focus of their investigation, Pinder et al. (2004) did observe that the 13 Wheatbelt lakes they investigated which had pH < 4, tended to have particularly depauperate communities. Numerous salt lake species common in more alkaline lakes were absent or scarce in these systems, including several microcrustacean species.

A study by Cranston et al. (1997) suggests that responses to acidity in northern tropical waters may be different from those in southern temperate regions. These authors examined community structure in a seasonally flowing tropical stream (Rockhole Mine Creek, Northern Territory), in which acidic mine water forms a point source of pollution (Cranston et al., 1997). These authors reported an increase in chironomid species richness and abundance at sites downstream of the incoming mine effluent, and attributed this to the presence of a large pool of vagile, acid-adapted taxa in the region, associated with a range of naturally acidic habitats. This is not the situation for most Wheatbelt streams which are usually circumneutral to alkaline in character, making it unlikely that a large pool of acidophilic taxa exist in the Wheatbelt.

Extreme levels of acidity in what are usually circumneutral to alkaline surface water systems are likely to have significant impacts on macro-invertebrate community structure. Whilst high salinities appear to favour crustaceans (Bunn & Davies, 1992), our results show that high acidity levels result in the virtual absence of crustaceans in macro-invertebrate communities associated with deep drainage. It is unknown whether or not naturally occurring aquatic acidophiles occur in the study area, or would be able to thrive in drainage effluent. The Wakeman Creek subcatchment is located in the Lockhart catchment, Avon River basin, an area characterised by naturally occurring acidic groundwaters, and apparently a broad range of pH values can occur in surface waters across the Avon Basin (Douglas & Degens, 2006). However, in addition to having low pH, this drainage discharge is likely to contain high dissolved metal concentrations of toxic species such as Al, Fe and other metals, suggesting that it is not a suitable habitat for most biota.

Retention of ecological values in degraded Wheatbelt streams

It is sometimes argued that Wheatbelt streams have lost much of their ecological value, and would thus not be adversely affected by receiving discharge from deep drains. Due to widespread landclearing and other agricultural activities, the Wheatbelt is an extensively modified landscape, often degraded, and it is true that Wheatbelt streams have borne the brunt of this degradation, resulting in simplified aquatic ecosystems that have macro-invertebrate communities consisting of families capable of tolerating a broad range of environmental conditions (Kay et al., 2001). Wakeman Creek represents an example of such a Wheatbelt stream. Like many others, this creek was ephemeral, flowing only during the winter months, or after major cyclonic summer storms. Despite the presence of fresh, alkaline water at undrained sites, this stream lacks riparian trees, and is subject to other degrading processes such as erosion and sedimentation. The latter processes probably account for the relatively high values of turbidity recorded for these sites. Turbidity, or 'muddiness' of water is caused by the presence of suspended particles such as clay, sand, silt, phytoplankton and detritus, and measures of turbidity are often good indicators of the presence of erosion and sedimentation.

In addition to providing food, and shade (which regulates temperature and algal production), riparian trees also drop large and small woody and leaf debris, important habitat for fish and invertebrates in streams. Although the Wheatbelt has been shown to have a rich aquatic invertebrate fauna, with over 950 different species recorded from the region (Pinder et al., 2004), poor riparian condition is thus likely to contribute to low in-stream faunal diversity in degraded Wheatbelt streams, including Wakeman Creek.

Although depauperate in terms of macro-invertebrate diversity compared to systems that are undisturbed (total species richness varying from only 10 to 13 at upstream sites), undrained reaches of Wakeman Creek have retained some ecological in-stream values. For example, two species of caddisflies were collected from the upstream section of the creek. Along with mayflies (Ephemeroptera) and stoneflies (Plecoptera), caddisflies are relatively sensitive to pollution gradients, and the presence of these three insect groups has been used to develop an index of stream health that is used world-wide. In addition to being indicators of river 'health', aquatic invertebrates are also important as prey species for other invertebrate and vertebrate predators, such as frogs, fish and birds. The presence of aquatic invertebrates in Wheatbelt streams is thus essential for the ecological functioning of these systems.

Recommendations

The present investigation examined downstream sites located up to approximately 20 km downstream of the drain complex. No recovery of salt levels or pH values had occurred within this reach, suggesting that the quality of water entering the Salt River from the Wakeman Creek is very poor, resulting in compromised ecological values. For example, degradation of the Seagroatt Nature Reserve has been attributed to the discharge of this acid groundwater from the Narembeen deep drainage network, leading to an investigation supported by government agencies, local government and regional bodies of the feasibility of diverting drainage discharge from the Wakeman Creek into Lake Kurrenkutten. In addition, the Conservation Council of Western Australia has expressed concern regarding the push for saving agricultural land at the cost of losing biodiversity assets. In view of this, it is recommended that future monitoring of the impacts of the Narembeen deep drainage network investigate additional sites downstream of the junction of Wakeman Creek with the Salt River.

Although invertebrate species richness will continue to decline, and community composition will be simplified, as streams become increasingly hypersaline, the most significant impact from discharge waters from deep drains is likely to be from the low pH of these waters. As yet, limited research has been undertaken in Australia on the sensitivity of invertebrates to pH. Although recent research (e.g. Pinder et al., 2004, 2005) has provided data on ‘threshold levels’ for salinity, no such threshold levels have been identified for pH values. In addition, macrophytes and algae are known to be highly sensitive to acidification (e.g. Mulholland et al., 1986; Ormerod et al., 1987; Farmer, 1990). Knowledge of such threshold levels can be used to set acceptable targets for the disposal and potential treatment of highly acidic waters resulting from deep drainage. It is recommended that an investigation of pH tolerances of aquatic biota be undertaken in order to identify threshold levels associated with significant changes in species richness and community composition in Wheatbelt streams.

As yet, no monitoring and evaluation framework aimed at assessing environmental impacts of deep drainage exists. Such a tool which describes appropriate study design, selection of appropriate indicators, data analyses techniques and interpretation of results would assist proponents of deep drainage to minimise negative environmental impacts of drainage. Data resulting from the investigation of the Narembeen deep drainage network, and other deep drainage networks can be used to inform the development of such a monitoring and evaluation plan, or template.

Acknowledgements

Funding for this project was provided under the National Action Plan for Salinity and Water Quality through the Engineering Evaluation Initiative managed by the Department of Water (formerly Department of Environment). Geraldine Janicke is thanked for assistance in the field and processing of samples in the laboratory. Drs Riasat Ali and Richard Silberstein are thanked for their input into valuable discussion on the Narembeen deep drainage project.

References

Ali, R., T. Hatton, R. George, J. Byrne & G. Hodgson, 2004. Evaluation of the impacts of deep open drains on groundwater levels in the wheatbelt of Western Australia. Australian Journal of Agricultural Research 55: 1159–1171.

Berezina, N. A., 2001. Influence of ambient pH on freshwater invertebrates under experimental conditions. Russian Journal of Ecology 32: 343–351.

Bunn, S. E. & P. M. Davies, 1992. Community structure of the macroinvertebrate fauna and water quality of a saline river system in south-western Australia. Hydrobiologia 248: 143–160.

Clarke, K. R., 1993. Non-parametric multivariate analyses of changes in community structure. Australian Journal of Ecology 18: 117–143.

Clarke, K. R. & R. M. Warwick, 1994. Changes in Marine Communities: An Approach to Statistical Analysis and Interpretation. Plymouth Marine Laboratory, Natural Environment Research Council, Plymouth.

Clarke, C. J., R. J. George, R. W. Bell & T. J. Hatton, 2002. Dryland salinity in south-western Australia: its origins, remedies, and future research directions. Australian Journal of Soil Research 40: 93–113.

Cranston, P. S., P. D. Cooper, R. A. Hardwick, C. L. Humphrey & P. L. Dostine, 1997. Tropical acid streams-the chironomid (Diptera) response in northern Australia. Freshwater Biology 37: 473–483.

Dogramaci, S. & B. Degens, 2003. Review of Engineering and Safe Disposal Options. Water and Rivers Commission, Salinity and Land Use Impacts Series, Rep. No. SLUI20: 22 pp.

Douglas, G. & B. Degens, 2006. A Synopsis of Potential Amendments and Techniques for the Neutralization of Acidic Drainage Waters in the Western Australian Wheatbelt. Cooperative Research Centre for Landscape Environments and Mineral Exploration: 22 pp.

Farmer, A. M., 1990. The effects of lake acidification on aquatic macrophytes—a review. Environmental Pollution 65: 219–240.

Feldman, R. S. & E. F. Connor, 1992. The relationship between pH and community structure of invertebrates in streams of the Shenandoah National Park, Virginia, U.S.A. Freshwater Biology 27: 261–276.

Kay, W. R., S. A. Halse, M. D. Scanlon & M. J. Smith, 2001. Distribution and environmental tolerances of aquatic macroinvertebrate families in the agricultural zone of southwestern Australia. Journal of the North American Benthological Society 20: 182–199.

Lane, P. W., N. W. Galwey & N. G. Alvey, 1988. Genstat 5: An Introduction. Oxford University Press, Oxford.

Lenat, D. R., 1988. Water quality assessment using a qualitative collection method for benthic macroinvertebrates. Journal of the North American Benthological Society 7: 222–233.

Mulholland, P. J., J. W. Elwood, A. V. Palumbo & R. J. Stevenson, 1986. Effect of stream acidification on periphyton composition, chlorophyll and productivity. Canadian Journal of Fisheries and Aquatic Sciences 43: 1846–1858.

Ormerod, S. J., K. R. Wade & A. S. Gee, 1987. Macro-floral assemblages in upland Welsh streams in relation to acidity, and their importance to invertebrates. Freshwater Biology 18: 545–557.

Petrin, Z., H. Laudon & B. Malmqvist, 2007. Does freshwater macroinvertebrate diversity along a pH-gradient reflect adaptation to low pH? Freshwater Biology 52: 2172–2183.

Pinder, A. M., S. A. Halse, J. M. McRae & R. J. Shiel, 2004. Aquatic invertebrate assemblages of wetlands and rivers in the wheatbelt region of Western Australia. Records of the Western Australian Museum Supplement 67: 7–37.

Pinder, A. M., S. A. Halse, J. M. McRae & R. J. Shiel, 2005. Occurrence of aquatic invertebrates of the wheatbelt region of Western Australia in relation to salinity. Hydrobiologia 543: 1–24.

Rogers, S. & R. George, 2005. WA wheatbelt drainage—acidic groundwater, not just a salt issue. Focus on Salt 33: 8–9.

Rosemond, A. D., S. R. Reice, J. W. Elwood & P. J. Mulholland, 1992. The effects of stream acidity on benthic invertebrate communities in the south-eastern United States. Freshwater Biology 27: 193–209.

Smith, M. J., W. R. Kay, D. H. D. Edward, P. J. Papas, K. S. J. Richardson, J. C. Simpson, A. M. Pinder, D. J. Cale, P. H. J. Horwitz, J. A. Davis, F. H. Jung, R. H. Norris & S. A. Halse, 1999. AusRivAS: using macroinvertebrates to assess ecological condition in rivers in Western Australia. Freshwater Biology 41: 269–282.

Table 1

Location of 12 sites along Wakeman Creek, Narembeen and Yawerlin Creek sampled between September 2004 and September 2006

Code	Description	Northing	Easting	% Of visits with flow present
Upstream sites				
U1	Tudor Rd	6466446	636010	25
U2	Mt Walker North Rd	6472142	666575	25
Drain sites				
D1	Soldiers Rd, eastern drain	6454801	645317	100
D2	Soldiers Rd, northern drain	6454917	637200	100
D3	Narembeen townsite	6451019	631767	100
D4	Dixon Rd	6464258	661536	100
Downstream sites				
DS1	Dayman/Bows Rds	6448271	626462	100
DS2	Bristow Butler Rd	6444880	623385	88
DS3	Cumminin/Koolberrin Rd	6445541	617900	86
DS4	Boundary Rd	6445619	613839	86
Comparative sites				
C1	Bruce Rock South Rd	6456623	611307	25
C2	Cumminin Rd	6457635	616793	57

Table 2

Results of global and pairwise test comparisons between the groups of Narembeen sites generated from an ANOSIM

Test	**Groups**	***R* statistic**	**Significance level (%)**
Global	D, US, DS,CO	0.531	0.1
Pairwise	CO, D	0.851*	0.1
	CO, DS	0.960*	0.1
	CO, US	0.365	8.6
	D, DS	0.207	0.1
	D, US	0.946*	0.1
	DS, US	0.994*	0.2

D, drain; US, upstream; DS, downstream; CO, control sites. Groups well separated ($R > 0.75$) are indicated with “*”

Table 3

Taxa most responsible for differentiation among groups of sites

Taxa	Comparative	Upstream	Drain	Downstream
Ostracoda sp. A	☻☻☻	☻	☻	☻
Paralimnophyes pullulus	☻	☻	☻☻☻	☻☻☻
Calanoida spp.	☻☻☻	☻☻☻	☻	☻
Cladocera spp.	☻☻	☻☻☻	☻	☻
Cyclopoida spp.	☻☻	☺	☻	☻
Procladius paludicola	☻☻	☻☻☻	☻	☺
Micronecta gracilis	☻☻	☻☻	☻	☺
Ceratopogonidae sp. A	☻	☻	☻☻	☻☻
Ceratopogonidae sp. B	☺	☻☻	☺	☻
Hypogastruridae	☻	☻	☻☻	☻
Ostracoda sp. B	☻☻	☺	☺	☺
Ostracoda sp. C	☻☻	☻	☺	☺
Ostracoda sp. D	☻	☻☻	☺	☺

☻☻☻, mean relative score of 2.00 and above; ☻☻, scores of 1.00–1.99; ☻, score of <1.00, ☺, absent

Fig. 1

Location of upstream (U1 and U2), drain (D1–D4) and downstream (DS1–DS4) sites on Wakeman Creek and comparative (C1 and C2) sites on Yawerlin Creek. Site D3 is located in the township of Narembeen. Arrow along Salt River shows direction of flow

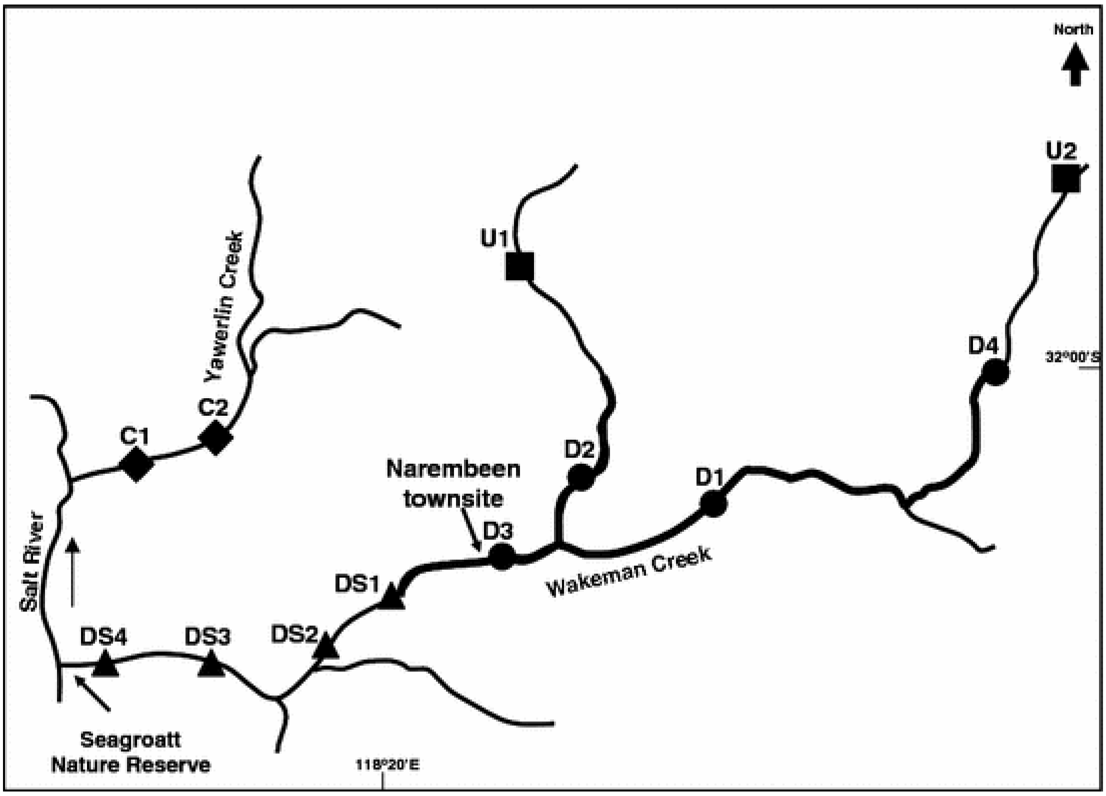


Fig. 2

Two-dimensional PCA ordination of six environmental variables (transformed and normalized) for sites sampled during the period December 2004 to September 2006

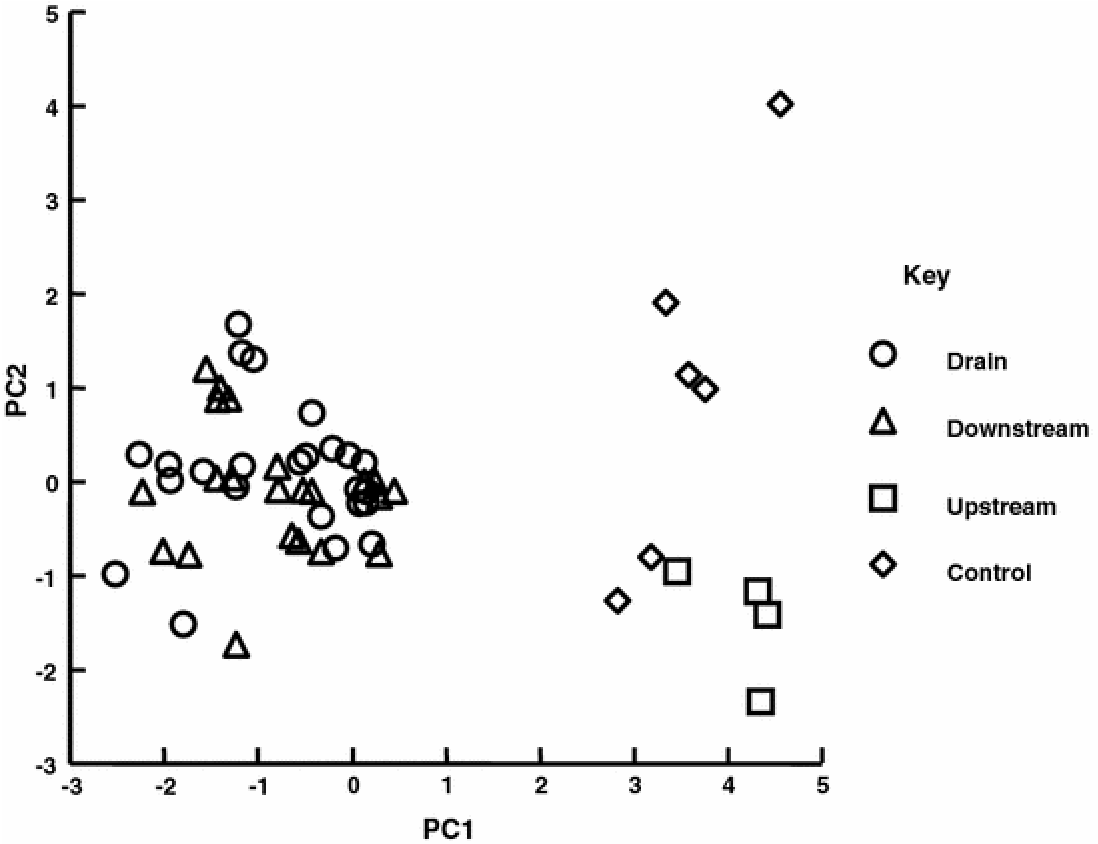


Fig. 3

Dendrogram resulting from a cluster analysis based on six environmental variables (transformed and normalised) for sites sampled during the period December 2004 to September 2006

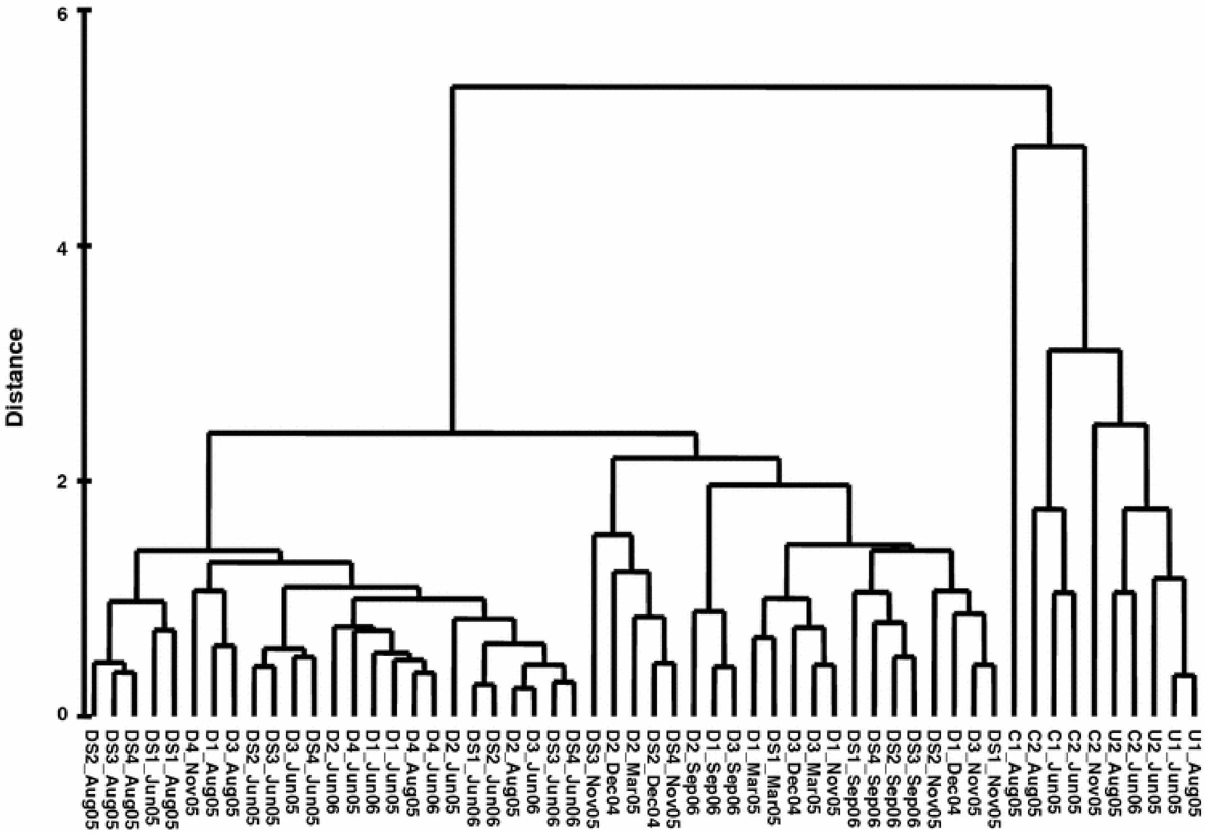


Fig. 4

Two-dimensional PCA ordination of six environmental variables (transformed and normalised) for sites sampled during the period December 2004 to September 2006, with values for (**a**) pH, (**b**) conductivity and (**c**) turbidity superimposed as bubble plots

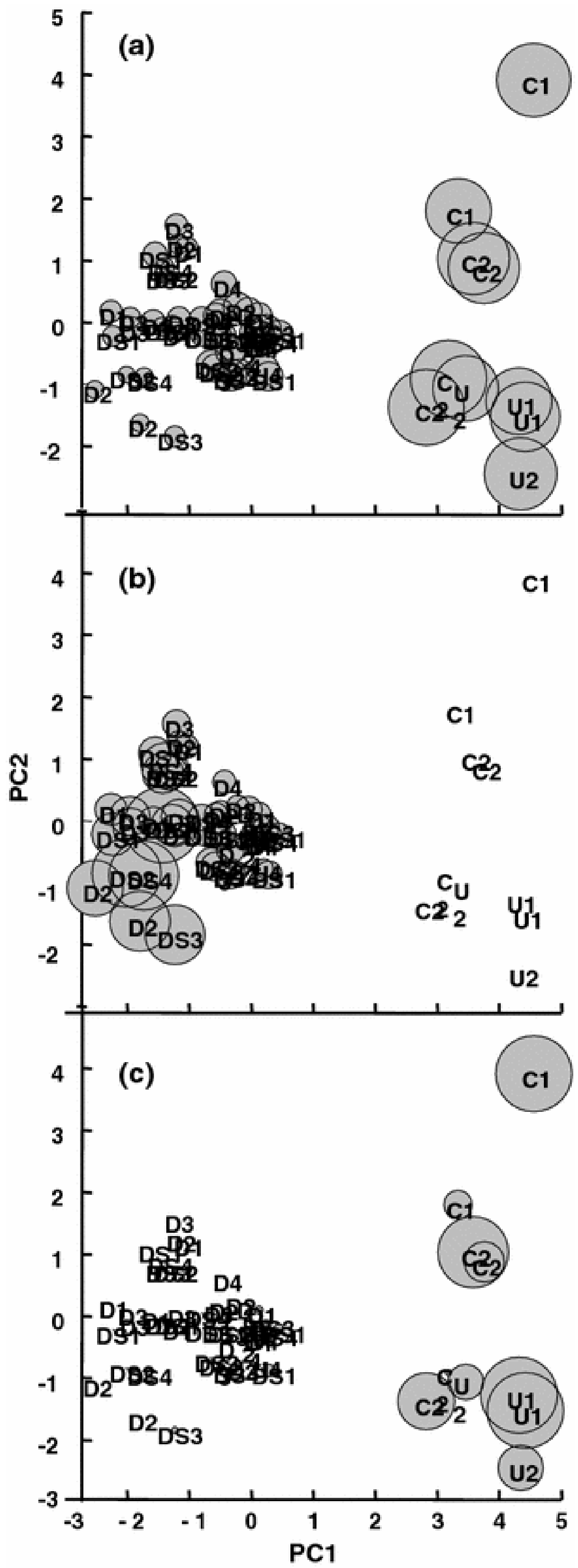


Fig. 5

Plot of distance from source (in km) versus (**a**) pH and (**b**) conductivity for sites sampled in August 2005. Junction of Wakeman Creek and Salt River occurs at approximately 90 km from source. Dashed lines demarcate three categories of sites: (i) sites upstream of the drain (U1 and U2), (ii) sites located in the drain complex (D1–D4), and (iii) sites located downstream of the drain complex (DS1–DS4)

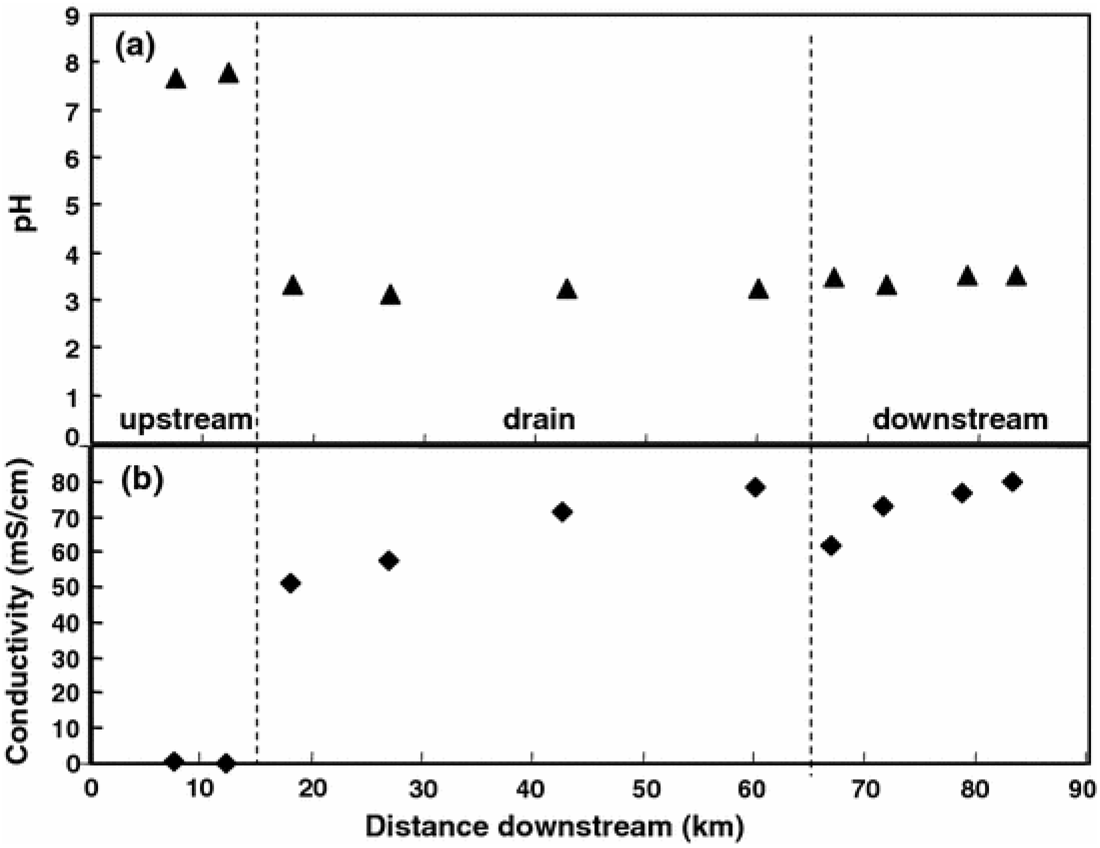


Fig. 6

Macro-invertebrate community composition at 12 sites sampled in August 2005

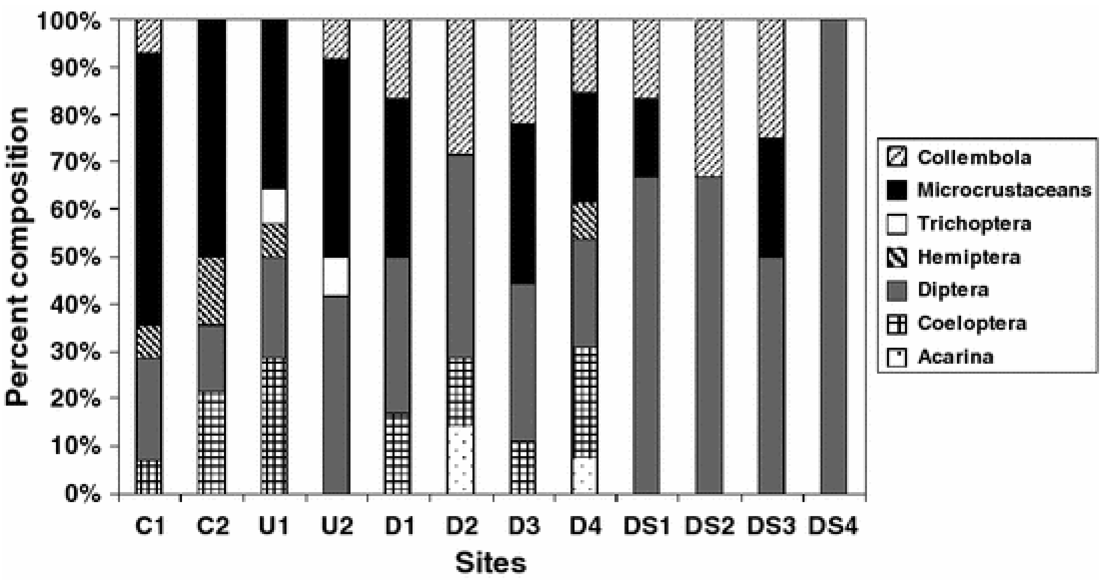


Fig. 7

Two-dimensional MDS configuration for sites sampled in the period September 2004 to September 2006, with superimposed clusters from a cluster analysis, at a similarity level of 20%

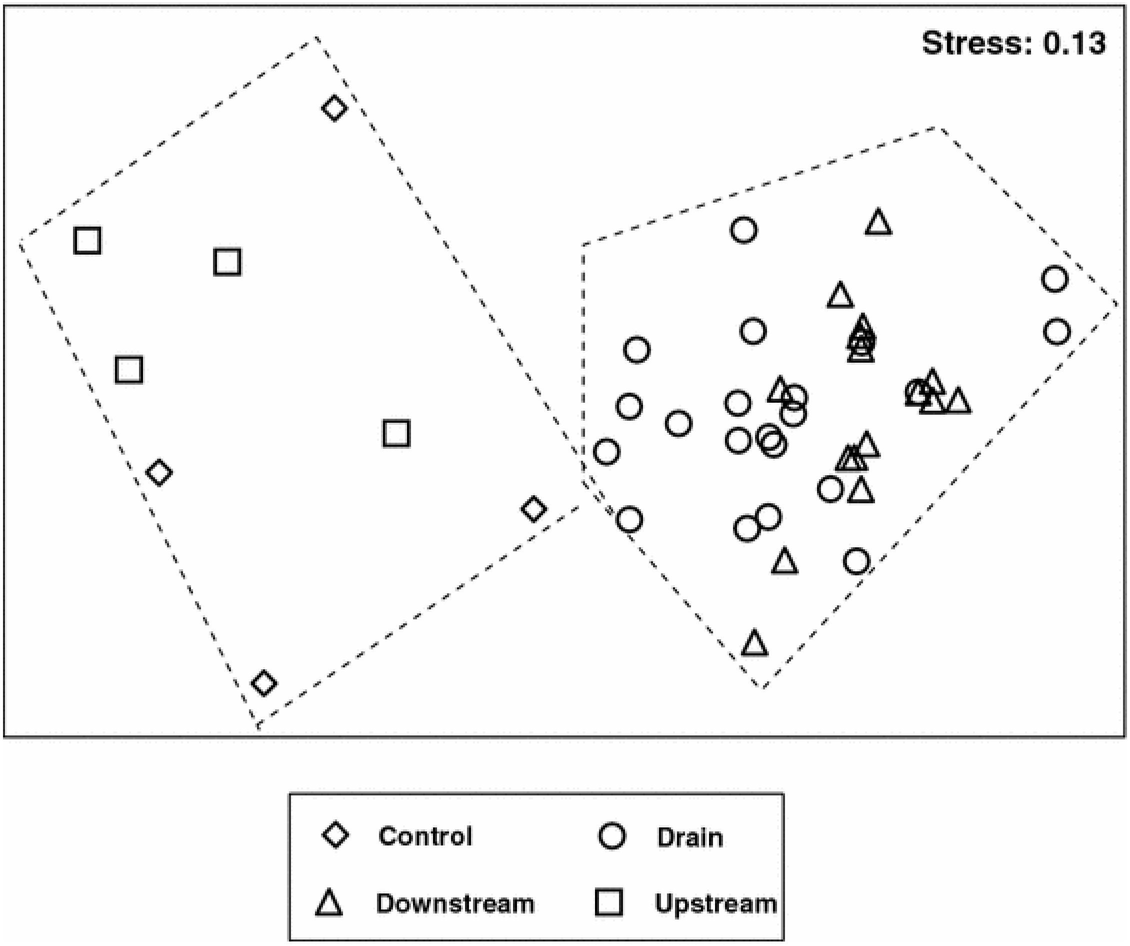


Fig. 8

Dendrogram resulting from a cluster analysis based on macro-invertebrate data for sites sampled during the period September 2004 to September 2006

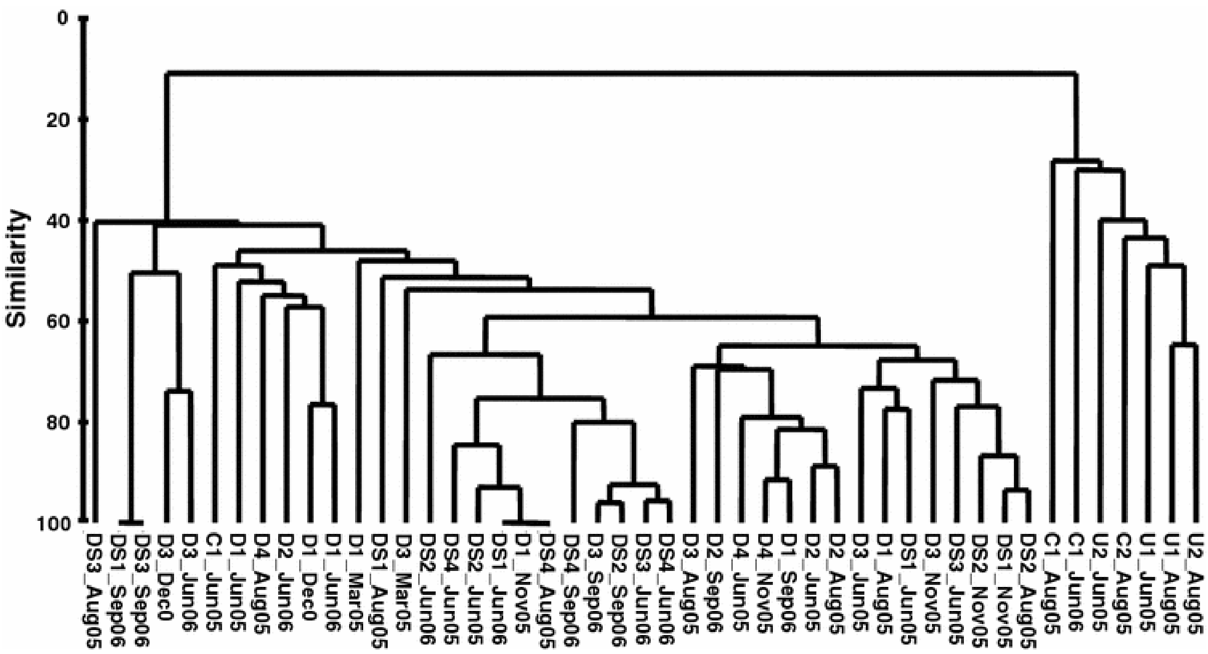
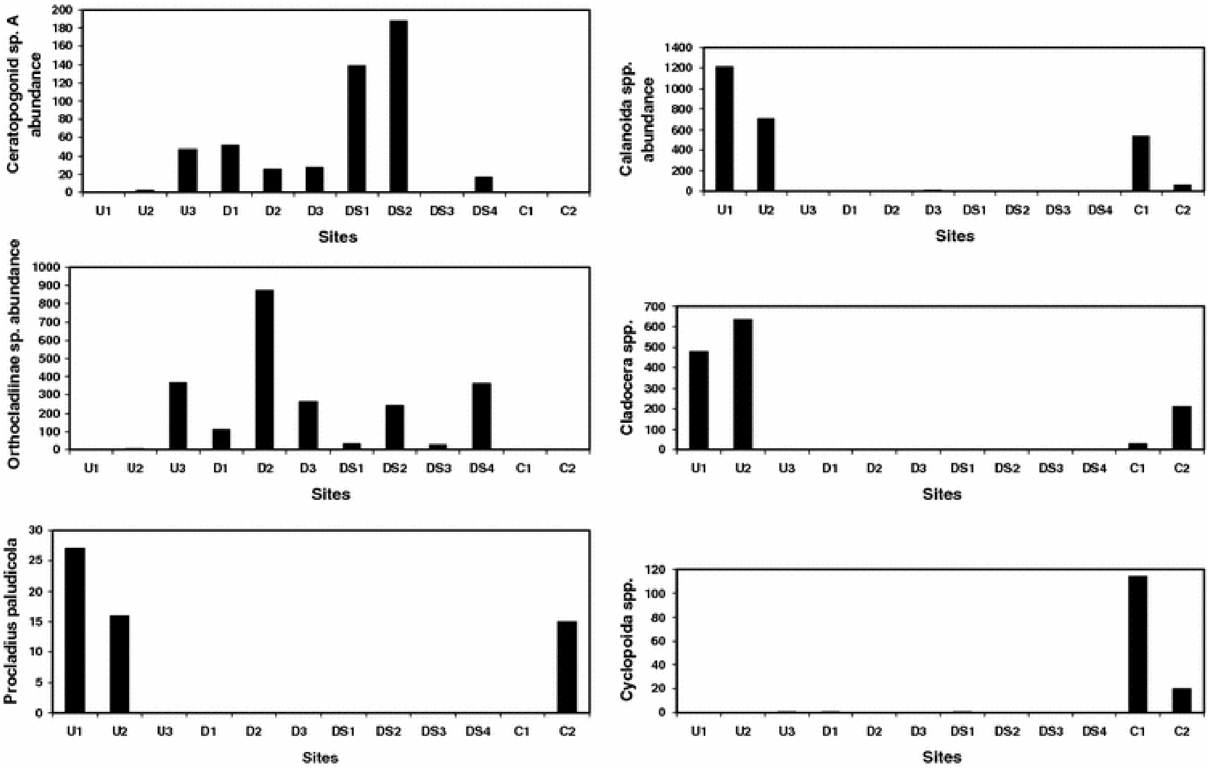


Fig. 9

Relative abundances of six selected taxa. U1–U2, upstream sites; D, drain sites; DS1–DS4, downstream sites; C, control sites



Appendix A

Macroinvertebrate species collected from comparative sites (C1-C2) on Yawerlin Creek and from upstream (U1-U2), downstream (DS1-DS4) and drain (D1-D4) sites on Wakeman Creek during the period September 2004 to September 2006. Ticks represent presence of a species at a site.

			Comparative		**Upstream**		**Drain**				**Downstream**			
			C1	**C2**	**U1**	**U2**	**D1**	**D2**	**D3**	**D4**	**DS1**	**DS2**	**DS3**	**DS4**
Mites	**Acarina**	Acarina sp.							√					
	Orbatidae	Orbatid sp.						√		√				
	Unioncolidae	Unioncolid sp.	√											
Beetles	**Dytiscidae**	*Liodessus inornatus*	√	√				√						
		Antiporus gilberti	√											
		Antiporus sp.		√										
		Eretes australis	√											
		Lancetes sp.		√										
		Megaporus howitti	√											
		Necterosoma penicillatus	√		√		√	√	√	√	√	√	√	√
		Rhantus suturalis	√					√						
		Sternopriscus multimaculatus	√				√							
	Hydrophilidae	*Berosus dallasae*	√		√		√							√
		Enochrus sp.	√							√				
		Limnoxenus zelandicus	√											
Biting midges	**Ceratopogonidae**	Ceratopogonid sp. A			√	√					√	√	√	
		Ceratopogonid sp. B	√			√	√	√	√	√	√	√	√	√
Mosquitoes	**Culicidae**	Culicid. sp								√				
Flies	**Empididae**	Empidid sp.							√			√		
	Ephydridae	Ephydrid sp. 2 (CALM sp.3)						√	√					
		Ephydrid sp. 5 (CALM sp. 6)							√					
	Muscidae	Muscid sp.					√							
	Stratiomyidae	Stratiomyid sp.					√							
	Tabanidae	Tabanid sp.					√				√			

			C1	C2	U1	U2	D1	D2	D3	D4	DS1	DS2	DS3	DS4
Midges	**Chironominae**	*Chironomus sp.*	√			√								
		Chryptochironomus nr *griseidorsum.*			√									
		Kiefferullus affin mentum	√						√				√	
		Tanytarsus barbitarsis	√											
		Chironomus alternans	√	√										
	Orthocladiinae	*Paralimnophyes pullulus*	√			√	√	√	√	√	√	√	√	√
	Tanypodinae	*Coelopynea priunosa*			√	√								
		Procladius paludicola	√		√	√	√							
Bugs	**Corixidae**	*Micronecta gracilis*	√	√	√	√				√				
		Micronecta robusta	√											
		Corixid sp. 2	√	√	√	√								
	Notonectidae	*Paranisops inconstans*	√			√								
		Anisops thienemanni	√			√		√						
		Anisops stali	√											
Caddisflies	**Leptoceridae**	*Triplectides australis*	√											
		Oecetes sp. 3	√		√	√								
Dragonflies	**Aeshnidae**	*Hemianax papuensis*	√											
	Hemicorduliidae	*Hemicordulia tau*	√											
Clam shrimps	**Cyzicidae**	*Caenestheriella packardi*	√	√	√									
Water fleas	**Cladocera**	Cladoceran sp.	√	√	√	√		√	√	√	√			
Copepods	**Calanoida**	Calanoid spp.	√	√	√	√	√	√	√	√			√	
	Cyclopoida	Cyclopoid spp.	√	√			√	√	√	√	√			
	Harpacticoida	Harpacticoid sp.	√	√		√			√					
Ostracods	**Ostracoda**	Ostracod sp. D (constriction)	√		√	√								
		Ostracod sp. B (large white)	√											
		Ostracod sp. A (small)	√	√	√		√		√				√	
		Ostracod sp. C (small green)	√		√	√								
		Ostracod sp. (small transparent)		√										
		Ostracod sp. (small white transparent)	√											

			C1	C2	U1	U2	D1	D2	D3	D4	DS1	DS2	DS3	DS4
Springtails	**Hypogastruridae**	Hypogastrurid sp.	√			√	√	√	√	√			√	
	Isotomidae	Isotomid sp.			√		√	√	√	√				
Roundworms	**Nematoda**	Nematoda sp.	√											