

## Effects of catchment clearing and sedimentation on macroinvertebrate communities of cobble habitat in freshwater streams of southwestern Australia

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### Abstract

The removal of riparian vegetation from along first order streams of the northern jarrah forest reduced ecological health, as assessed by an examination of cobble communities. Macroinvertebrate diversity was significantly lower in cleared compared to uncleared reaches. There was also an associated decrease in the biomass and a change in the composition of epilithon communities, from those dominated by the angiosperm *Potamogeton ?crispus* to thin slimes dominated by diatoms. Extensive growths of filamentous algae were not observed on cobbles in cleared reaches. Species of Odonata, Ephemeroptera, Trichoptera and Chironomidae were the most notable absences from cobbles in cleared reaches. Grazer abundance also was reduced. Few species in cobble habitats appeared to benefit from catchment clearing although there was a significant increase in the abundance of the undescribed chironomid Orthoclaudiinae V61, which may be a useful indicator species. We also tested the effect of short term elevations in suspended sediment through experimental addition of sediment in an attempt to separate the effects of sediment from others related to catchment clearing. There was no significant difference in macroinvertebrate diversity between control and sediment-added cobbles, and both had higher diversity than cobbles in cleared reaches. We concluded that the changes in the epilithic cover in cleared reaches caused indirectly the changes in the macroinvertebrate community, either through a change in the composition of food sources or loss of refugia. Sedimentation in cobble environments might have greater impact in the longer term than noted in this study, and also might have impact on other stream meso-habitats.

**Keywords:** macroinvertebrate communities, freshwater streams, cobble habitats, catchment clearing, southwestern Australia

### Introduction

The importance of riparian zones is well recognised in stream ecology. They provide a critical link between terrestrial and aquatic ecosystems (Lynch & Catterall 1999), and largely control the functioning of the latter. Riparian cover has a strong influence on stream productivity (Bunn *et al.* 1999a), providing shade that maintains lower water temperature (Bunn *et al.* 1999b) and reduces light penetration, thereby controlling algal growth (Mosisch *et al.* 1999). Riparian zones also provide an important allochthonous food source for some aquatic animals (Lake 1995) and increase habitat heterogeneity (Likens & Bormann 1974; Lake & Marchant 1990; Gregory *et al.* 1991). Vegetated stream banks are more stable than those without vegetation (Abernathy & Rutherford 1999). They moderate the flow of sediment and nutrients into the water column (Gregory *et al.* 1991; Waters 1995), and filter nutrients from surface and groundwater flows (Prosser *et al.* 1999). In eastern Australia, even narrow riparian buffers are effective at trapping overland flows of sediment (Prosser *et al.* 1999).

Loss of or disturbance to riparian vegetation therefore has many effects, one of the more obvious being the periodic increase in turbidity and bed-load sediment transport, particularly during rainfall events (Brown

1972; Karr & Schlosser 1978; Lemly 1982; Campbell & Doeg 1989; Hogg & Norris 1991; Quinn *et al.* 1992). Increased sediment loads are known to have detrimental effects on both aquatic macroinvertebrates (e.g. McLelland & Brüsven 1980; Lenat *et al.* 1981; Lemly 1982) and aquatic plants and algae (Lewis 1973a,b; Nutall 1972; Graham 1990). In many instances, stream macroinvertebrates respond to increased sediment levels by drifting (e.g. Gray & Ward 1982; Doeg & Milledge 1991).

Removal of streamside vegetation can have different effects among mesohabitat scale communities (Lenat *et al.* 1981; Lemly 1982). The present study focussed on the cobble meso-habitat (*sensu* Cummins 1962), which has been identified as a discrete 'natural unit' due to a 'well defined, abrupt boundary' (Davies 1994). Cobbles are characterised by a comparatively high projection and occur in regions of relatively high shear velocity in the stream channel (Davies 1994). A unique feature of cobbles is the provision of a stable substrate which enables growth of epilithic algae and aquatic plants. Cobbles also provide a stable platform from which sessile animals, such as filter feeders, may extract suspended food particles from the water column (Wallace & Merritt 1980).

In the northern jarrah forest, macroinvertebrate and epilithic communities of cobbles are distinct from those of other stream habitats, being dominated by animals in

scraper and filterer feeding guilds in contrast to the dominance of collectors in riffles, and collectors and shredders in organic depositional habitats (Davies 1994). Cobbles are also characterised by a greater proportion of autochthonous (instream-derived) vegetative material compared with riffle and organic depositional habitats. The production to respiration ratio of cobble habitat is typically greater than one, indicative of autotrophy (*i.e.*, having a carbon source predominantly derived from instream photosynthetic processes; Davies 1994). In upland streams of the northern jarrah forest, the low nutrient status of soils and shading by the riparian canopy minimise the standing crop of epilithon compared with streams in eastern Australia (Bunn and Davies 1990). These characteristics make cobbles a useful unit for assessing the effects of removal of streamside vegetation on both macroinvertebrates and epilithic vegetation.

The construction of the North Dandalup Dam (south of Perth, Western Australia) provided an opportunity to investigate the effects of riparian vegetation removal on aquatic macroinvertebrate communities in northern jarrah forest streams. Dam construction involved the total clearing of 430 ha of riparian and upslope vegetation within the reservoir basin. It was predicted that elevated sedimentation associated with clearing would be detrimental to both the fauna and epilithon of cobbles. We compared the diversity of cobble macroinvertebrates and the composition of epilithon between cleared and uncleared reaches. It was also predicted that elevated sediment levels in the water column would have the greatest effect on the cobble habitat, since they are exposed above the streambed. Thus, we examined the short term effect of suspended sediment addition on the aquatic macroinvertebrate communities of cobble substrates in an attempt to distinguish this from other effects resulting from the removal of riparian vegetation.

## Methods

### Study area

The study area, situated c. 100 km south of Perth, consisted of three first order streams (Foster and Wilson Brooks and the North Dandalup River; Fig 1) in the North Dandalup catchment, a system of about 40 km in length. The headwater streams arise in swamps on the Darling Range (300 m above sea level), the south-western edge of the Great Plateau of Western Australia (Jutson 1950). The streams flow over lateritic soils and granite bedrock before descending through an incised river valley to the Swan Coastal Plain. The headwaters are located in the northern jarrah forest, a dry sclerophyll forest dominated by jarrah *Eucalyptus marginata* Sm. and to a lesser extent marri *Corymbia calophylla* (Lindl.) K.D. Hill & L.A.S. Johnson. The riparian understory is composed of dense sclerophyllous shrubs, including *Agonis linearifolia* (DC.) Sweet, *Lepidosperma tetraquetrum* Nees and *Hypocalymma angustifolium* (Endl.) Schauer. The upper catchment has been selectively logged since European settlement and was the site of a small gold strike in the late 1800s. The area has a Mediterranean climate with wet, mild winters, hot, dry summers, and an annual rainfall of approximately 1100 mm. The

general geology and flora of the area has been summarised by Bunn (1986) and Storey *et al.* (1990).

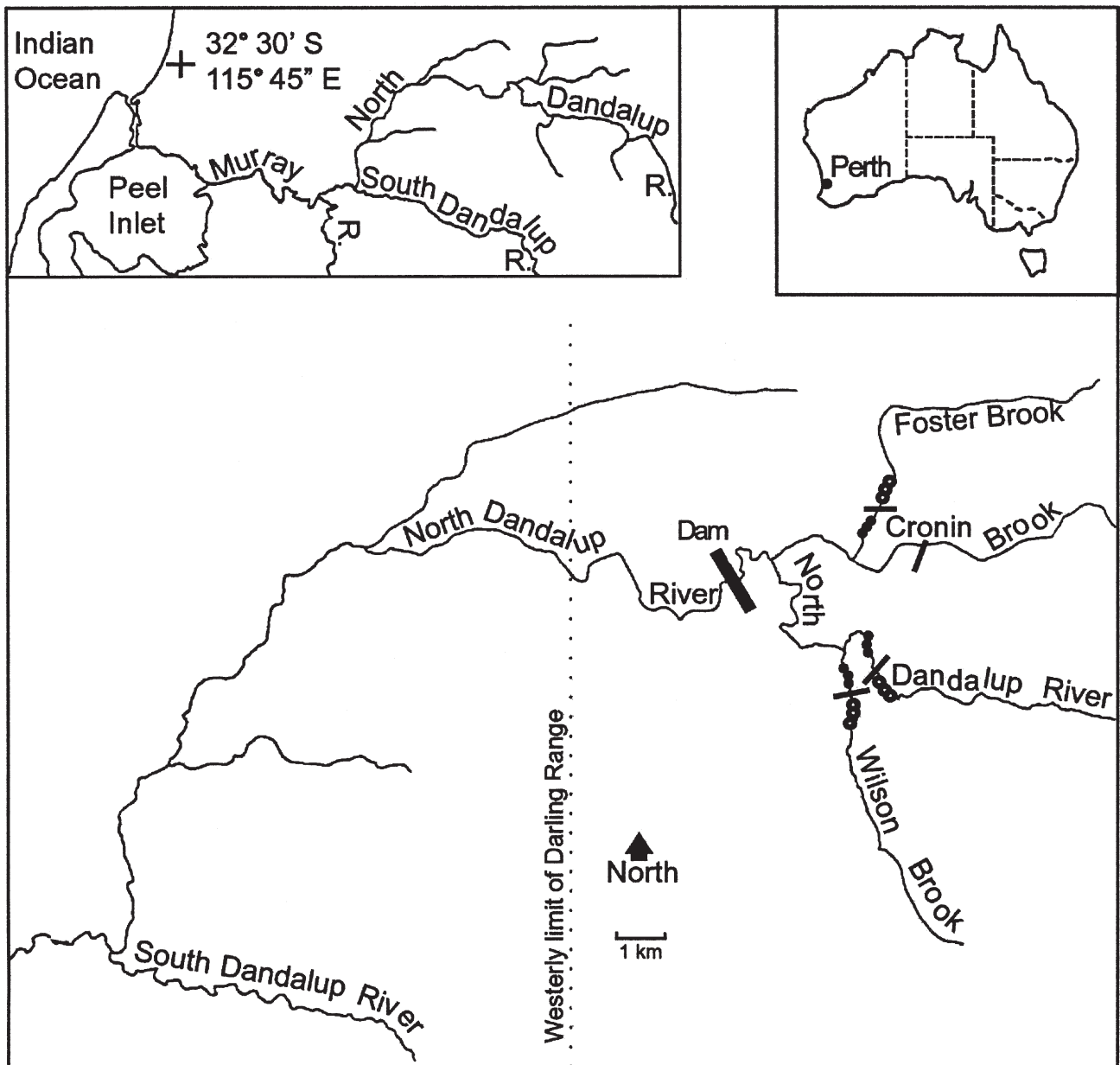
The North Dandalup system has been regulated from 1971 by a small pipehead dam (capacity 20 000 m<sup>3</sup>). However, to serve the growing demand from the Perth metropolitan area, the pipehead dam was replaced in October 1994 by the North Dandalup Dam, a 62 m high rock-fill dam wall, forming a major water supply reservoir (catchment area 153 km<sup>2</sup>, 510 ha surface area at full storage and a capacity of 75 GL), supplying 10% of the annual water consumption of Perth.

### Sampling

The experimental design comprised three treatments: undisturbed upstream controls in uncleared reaches (CON), sediment addition in uncleared reaches (SED), and cleared or 'impact' reaches (IMP). Each treatment was replicated at three sites on each of the three streams. Each replicate consisted of three cobbles sampled at random from the stream bed, giving a total of nine cobbles per treatment per stream and a total of 81 cobbles sampled. Storey *et al.* (1990) have shown that macroinvertebrate communities of northern jarrah forest streams are generally homogeneous within the same stream order, and therefore it was anticipated that the three streams would be comparable replicates. All impact sites necessarily were located downstream of uncleared areas but within the same stream order. Sediment and control treatments were interspersed within the uncleared reaches. The sediment addition experiment was performed once only for each cobble between the 29th July–3rd of August and the 10th–11th of August, 1995. Other cobbles were sampled whilst moving upstream at this time, so that no unsampled cobbles were left downstream.

The short term elevation in turbidity was mimicked by the slow addition of fine (<2 mm) sediments to channel reaches containing cobbles. Approximately 60 kg of lateritic soil was washed into the water column over a two-hour period at distance of c. 1 m above the cobbles to be sampled. Total suspended sediment level was measured from water samples (dry weight per litre) collected downstream of the point of sediment addition to assess the effectiveness of the treatment. Cobbles were sampled two hours after the commencement of the sediment addition by removal of the cobble, using a D-frame pond net (250 mm mesh aperture) placed immediately downstream.

Within the cleared reaches, there were numerous cobbles that had only recently entered the stream through bank erosion/slumping or through the influence of heavy machinery. These lacked an obvious layer of epilithon, were not embedded in the stream and therefore were not sampled; only cobbles of lateritic composition with an established epilithic layer were sampled. Attached fauna were removed into a basin by scrubbing the cobble with a soft brush, and epilithon was removed with a harder brush and immediately preserved in 5% formalin. Macroinvertebrate specimens were identified to lowest possible taxonomic level (species, where possible), and vouchers lodged with the School of Animal Biology, The University of Western Australia. Taxa were then allocated to their appropriate functional feeding groups (*i.e.* filterer and grazer; *sensu* Cummins



**Figure 1.** Location of the North Dandalup catchment and first order streams sampled in Western Australia. Bars indicate extent of catchment clearance in each stream, closed circles indicate sites where impact cobbles were sampled and open circles where control and sediment treatment cobbles were sampled.

1973, Merritt & Cummins 1978, Bunn 1985, 1986). To standardise the abundance of fauna between different-sized cobbles, the surface area of each cobble was determined in the field by wrapping it in aluminium foil and relating the weight of the foil to surface area by regression. Only the proportion of the rock surface exposed to the water was measured.

Physico-chemical variables were measured at each cobble before sampling. Conductivity (EC) dissolved oxygen (DO) and pH were measured with WTW conductivity, dissolved oxygen and pH meters respectively, and stream velocity at 5 cm above the substrate with a Marsh McBirney portable water flow meter (model 201). Total Phosphorus (by potassium persulphate digestion with subsequent orthophosphate

assay using the molybdenum blue method; Wetzel & Likens 1979) and total Kjeldahl nitrogen (by digestion with ammonia subsequently assayed using an ammonia-sensitive electrode; Bremner & Mulvaney 1982) were measured from each stream. Two samples from the water surface were taken at the downstream end of each treatment area.

#### Analysis

For statistical analysis, the type of epilithic cover was recorded as either 'algae', 'diatom matrix', '*Potamogeton*', '*Nostoc*' or 'no epilithon'. Chi-square analysis was used to detect departures in observed frequency of occurrence for each epilithon type in each treatment. Two-way ANOVA was used to test for treatment and

sub-catchment (stream) differences in epilithon biomass (per square metre), as well as macroinvertebrate community parameters (abundance, richness, Shannon-Weiner diversity (Shannon & Weiner 1949), chironomid richness and abundance, and the abundance of filterers and grazers), with 'treatment' and 'stream' as fixed factors. Where there were significant main effects, Tukey's HSD multiple range tests were applied *a posteriori* to detect treatment and sub-catchment differences in community parameters. All data were  $\log_{10}(x+1)$  transformed prior to analysis to achieve equality of variances (Cochran's C test for homoscedasticity). All the above analyses were undertaken in the SAS statistical package (SAS, 1994). Ordination of samples based on macroinvertebrate community composition was undertaken using hybrid non-metric Multi Dimensional Scaling (MDS) in the PATN analysis program (Belbin 1995), with pairwise similarities amongst samples calculated using the Bray-Curtis similarity index. The separation of treatments in ordination space was tested using Analysis of Similarity (ANOSIM) (Belbin, 1995). Gradients in environmental variables (DO, EC, pH and stream velocity) and community descriptors were fitted through ordination plots using the Principle Axis Correlation (PCC) option in PATN, with Monte Carlo randomisations ( $n=100$ ) of the data performed to test the significance of the gradients. A separate ordination was performed for each subcatchment (stream) due to the observed differences between streams. Ordinations were performed to minimise dimensions and stress, and plotted to display those dimensions that illustrated the greatest separation of treatments.

## Results

### Physico-chemical

The turbidity of the water column two metres downstream from the point of sediment addition increased significantly during sediment addition compared with control and cleared sites (two-way ANOVA:  $F_{2,25} = 4.5$ ,  $p = 0.03$ ; means  $\pm$  S.E.: SED  $26.8 \pm 4.0$  mg L<sup>-1</sup>; CON  $12.0 \pm 4.6$  mg L<sup>-1</sup>; IMP  $11.0 \pm 2.4$  mg L<sup>-1</sup>). Bedload transport was also observed to be much higher (as might be expected) but was not quantified. Total phosphorus and total Kjeldahl nitrogen were both  $< 10$  mgL<sup>-1</sup> for all treatments (K. Armstrong unpublished data).

### Epilithon

Four dominant types of epilithic growth were recorded. Most cobbles in uncleared reaches supported growths of the aquatic angiosperm *Potamogeton ?crispus* (Potamogetonaceae). When present, this species dominated the cobble epilithon community. On cobbles where *Potamogeton ?crispus* was absent, chlorophytes were the dominant epilithic growth. In this study, chlorophytes were classified into two main types; 'Nostoc' which had a nodular growth form similar to that of *Nostoc* and 'algae' which included all remaining Chlorophyta. The majority of cobbles in cleared reaches lacked an epilithon layer although many were covered in a thin, slimy matrix of diatoms, bacteria and accumulated sediments. The sediment addition was a short term treatment, and therefore had no effect on epilithon.

There was a significant departure from expected frequency for the types 'Potamogeton', 'diatom matrix' and 'no epilithon' (Table 1). In each instance, the departure from expected occurred in the impact reach, with more than the expected number of cobbles dominated by 'diatoms' or with 'no epilithon', and fewer than expected cobbles being dominated by 'Potamogeton' (Table 1). Associated with the change in epilithon type was a decrease in biomass in the impact treatment (Table 2).

### Macroinvertebrate fauna

The total number of taxa and total abundance per m<sup>2</sup> recorded in controls (98 taxa and 26 556 individuals m<sup>-2</sup>) was greater than in both sediment and impact treatments (87 taxa and 26 748 individuals m<sup>-2</sup>, and 72 taxa and 12 498 individuals m<sup>-2</sup> respectively). The proportion of taxa in common between control and impact treatments was 66 %, and between control and sediment addition treatments was 86 %. Notable absences from impact cobbles compared to controls included the mayfly *Bibulmena kadjina*, the trichopteran *Plectrotarsus minor*, the dragonfly *Argiolestes minimus*, the caddisfly *Ecnomina sentosa* type group and the chironomid *Riethia ?zeylandica*.

All community parameters showed a significant treatment effect, and the majority a significant stream effect, with no significant interactions (Table 2). In general, the impact treatment had the lowest taxonomic richness and total densities of any parameter (Figure 2). There were 20 taxa with significant subcatchment and/or treatment differences in abundance (Table 3). Of these, 10 species had a higher abundance in control than impact

Table 1

Analysis of the frequency of occurrence of dominant epilithon types on cobbles from Control, Impacted and Sedimented reaches, giving  $\chi^2$  statistic, significance level and Observed/Expected number of occurrences in each treatment. \*: indicates the cell that has the greatest contribution to the total  $\chi^2$ -statistic.

| Epilithon Type | $\chi^2$ | P        | Control | Impact    | Sediment |
|----------------|----------|----------|---------|-----------|----------|
| Algae          | 0.75     | ns       | 3 / 3   | 4 / 3     | 2 / 3    |
| Diatom matrix  | 15.32    | 0.0005   | 0 / 2.3 | 7 / 2.3*  | 0 / 2.3  |
| Nostoc         | 1.28     | ns       | 6 / 4.3 | 3 / 4.3   | 4 / 4.3  |
| Potamogeton    | 32.34    | < 0.0001 | 18 / 13 | 1 / 13*   | 20 / 13  |
| No epilithon   | 24.37    | < 0.0001 | 0 / 4.3 | 12 / 4.3* | 1 / 4.3  |

**Table 2**

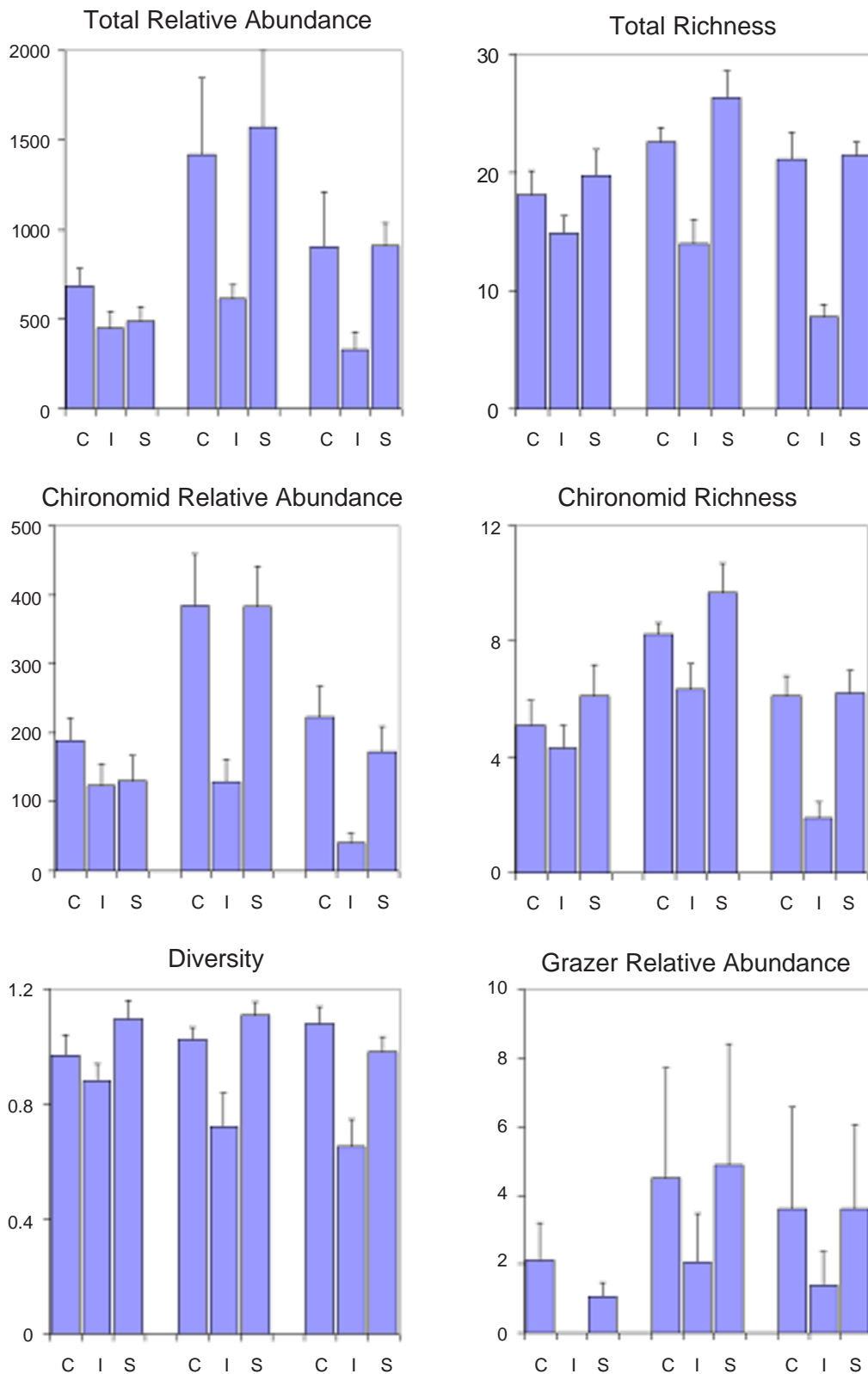
Analysis of various community parameters by sub-catchment (F: Foster, W: Wilson and N: Nth Dandalup) and treatment (C: Control, I: Impact, S: Sedimented). Tukey's multiple comparison test was used to locate between-level differences for significant main effects. The order of levels within each parameter has been standardised according to that indicated for the first parameter, and a common line joins levels not significantly different. Significance of main effects and interaction terms are indicated by: ns: not significant, \*:  $P < 0.05$ , \*\*:  $P < 0.01$  and \*\*\*:  $P < 0.001$ .

| Parameter                              | Sub-catchment<br>(df=2) | Treatment<br>(df=2) | Interaction | Sub-catchment | Treatment   |
|--|-------------------------|---------------------|-------------|---------------|-------------|
| Epilithon biomass (m <sup>2</sup> )    | 7.0**                   | 9.4***              | 0.7ns       | W _ N _ F     | _ C _ S _ I |
| Abundance (m <sup>2</sup> )            | 7.4**                   | 10.7***             | 1.7ns       | _____         | _____       |
| Number of taxa                         | 4.3*                    | 26.2***             | 2.5ns       | _____         | _____       |
| Shannon-Weiner Diversity               | 0.9ns                   | 16.7***             | 1.7ns       | _____         | _____       |
| Chironomid richness                    | 14.5***                 | 11.9***             | 1.3ns       | _____         | _____       |
| Chironomid abundance (m <sup>2</sup> ) | 2.5ns                   | 3.4*                | 1.1ns       | _____         | _____       |
| Grazer abundance (m <sup>2</sup> )     | 18.9***                 | 18.2***             | 2.5ns       | _____         | _____       |
| Filterer abundance (m <sup>2</sup> )   | 3.8*                    | 5.1**               | 0.03ns      | _____         | _____       |

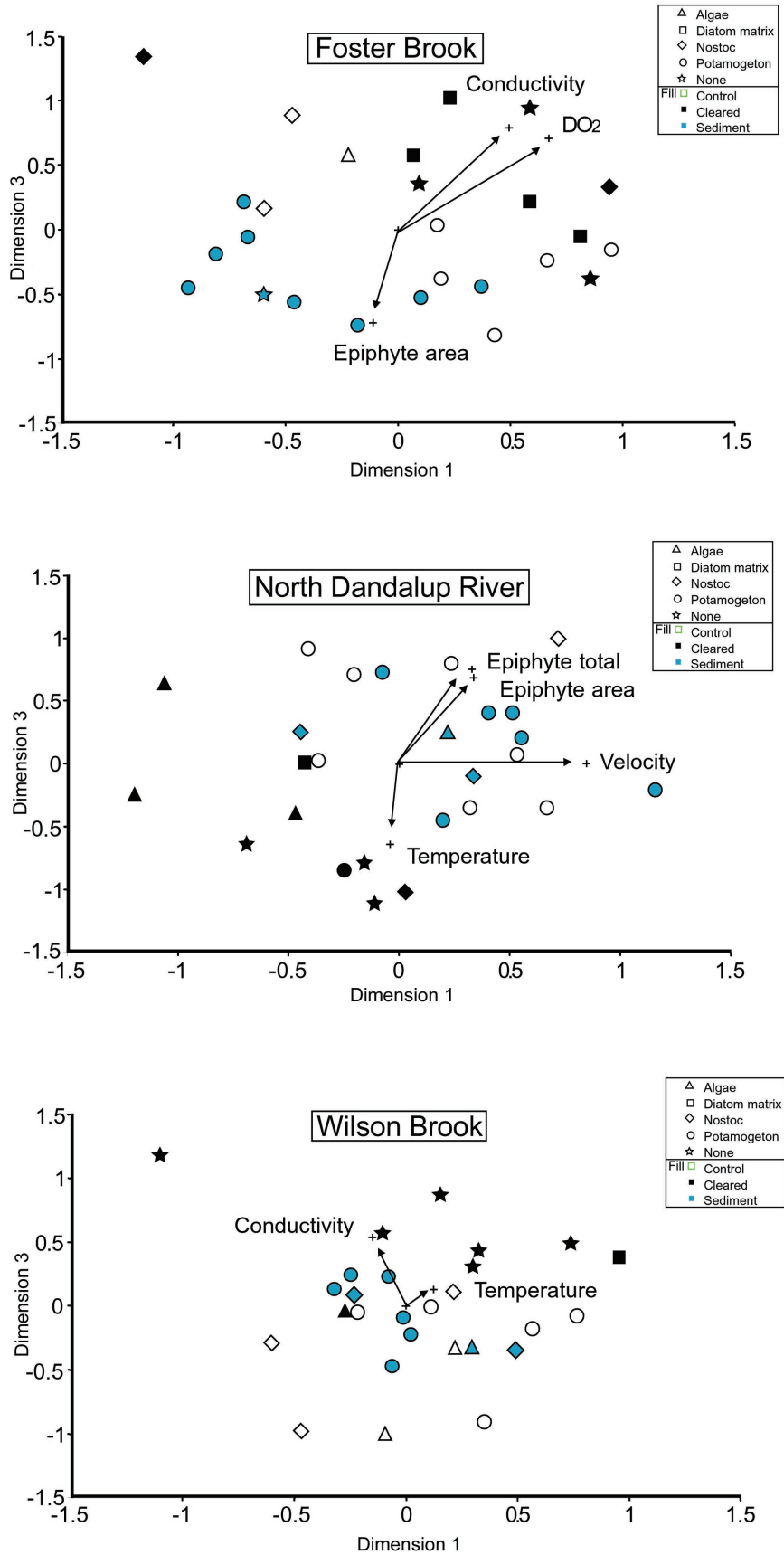
**Table 3**

Differences in the abundance of selected invertebrate species (F: Foster, W: Wilson and N: Nth Dandalup) and treatment (C: Control, I: Impact, S: Sedimented) on  $\log_{10}(x+1)$  abundance of each taxon (ns:  $p > 0.05$ , \*:  $p < 0.05$ , \*\*:  $p < 0.01$ , \*\*\*:  $p < 0.0001$ ). The order of levels within each parameter has been standardised according to that indicated for the first parameter, and levels of main effects not significantly different, as determined by Tukeys multiple range ( $\alpha < 0.05$ ) are joined by a common line. Voucher specimens are lodged in the Aquatic Research Laboratory, School of Animal Biology, The University of Western Australia.

| Taxon  | Sub-catchment | Treatment | Interaction | Sub-catchment | Treatment   |
|--|---------------|-----------|-------------|---------------|-------------|
| A090102  | ns            | *         | ns          | W _ N _ F     | _ C _ S _ I |
| <i>Austrosimulium</i> sp. X                            | *             | **        | ns          | _____         | _____       |
| <i>Ceratopogonidae</i> sp. A                           | *             | **        | ns          | _____         | _____       |
| <i>Ecnomus turgidus</i>                                | ns            | **        | ns          | _____         | _____       |
| ND 307 Hydracarina ND ?                                | *             | *         | ns          | _____         | _____       |
| <i>Neboisiphlebia</i> sp. <i>Ngunurra</i> sp. combined | ***           | ***       | ns          | _____         | _____       |
| Ostracoda sp.  | *             | ns        | ns          | _____         | _____       |
| <i>Smicrophylax australica</i>                         | **            | ns        | ns          | _____         | _____       |
| <i>Tasmanocoenis tillyardi</i>                         | ***           | ns        | ns          | _____         | _____       |
| <i>Botryocladus bibulmun</i>                           | ns            | **        | ns          | _____         | _____       |
| <i>Thienemanniella</i> sp.V19                          | ***           | ns        | ns          | _____         | _____       |
| <i>Nilothauma</i> sp.V21                               | **            | ns        | ns          | _____         | _____       |
| ? <i>Paralimnophyes</i> sp.V31                         | ***           | ***       | ns          | _____         | _____       |
| <i>Limnophyes pullulus</i> V42                         | **            | ns        | ns          | _____         | _____       |
| <i>Dicrotendipes</i> sp.V47                            | **            | ns        | ns          | _____         | _____       |
| <i>Riethia</i> sp. nov. V5                             | ns            | ***       | ns          | _____         | _____       |
| <i>Tanytarsus</i> spp.V6                               | ***           | **        | ns          | _____         | _____       |
| Orthoclaadiinae V61                                    | ns            | *         | ns          | _____         | _____       |
| <i>Stempellina ?australiensis</i>                      | ns            | *         | ns          | _____         | _____       |
| Orthoclaadiinae VND1                                   | **            | ns        | ns          | _____         | _____       |



**Figure 2.** Comparison of some community descriptors between treatments. Within each plot, histograms are grouped by stream (Foster, North Dandalup and Wilson, respectively) and labelled according to treatment (control (C), impact (I) and sediment addition (S)). Values are means  $\pm$  1 S.E.



**Figure 3.** MDS ordination plots from PATN illustrating the similarity of each cobble sample. Physical variables appear as gradients. Stress values: Foster: 0.19; North Dandalup: 0.16; Wilson: 0.14.

treatment. The chironomid Orthoclaadiinae V61 showed a positive response to catchment clearing, increasing significantly in abundance in the impact reaches.

### Ordination

Initial analyses (two-way ANOVAs and ordinations) showed significant differences between streams. Therefore, streams were treated in isolation for subsequent multivariate analysis. For each stream there was a clear separation of impact cobbles from both control and sediment addition cobbles, indicating differences in the composition of macroinvertebrate communities following clearing (Figure 3). ANOSIM testing between treatments was significant in the case of all three streams (Foster and North Dandalup:  $P < 0.0001$ ; Wilson:  $P < 0.01$ ). There was some separation of control and sediment addition cobbles in Foster Brook, however this pattern was not observed in the other streams. In Foster Brook and North Dandalup River, epiphyte area was positively correlated with uncleared cobble treatments, whereas an increase in the variables conductivity, DO and temperature were associated with cobbles in cleared reaches generally. These patterns suggest that the removal of riparian vegetation resulted in a loss of epilithon mass and a slight increase in conductivity, dissolved oxygen and temperature. The composition of epilithon was also related to treatment. Cobbles in cleared reaches generally had either a diatom matrix or no epilithon while cobbles in uncleared reaches were dominated by *Potamogeton ?crispus* and green algae (*Nostoc* type and other chlorophytes).

### Discussion

Macroinvertebrate communities of cobbles in cleared reaches had lower macroinvertebrate diversity compared with uncleared controls, which was likely the result of the removal of riparian vegetation. Contributing to the decrease in overall macroinvertebrate diversity was a significant decrease in the abundance of grazers and filterers. This was indicative of a change in the energy base of cobbles, since the composition and biomass of epilithic communities was also altered by clearing. Most cobbles in uncleared reaches supported crops of algae and the angiosperm *Potamogeton*, while most cobbles in cleared reaches were encrusted with a diatomaceous matrix or had essentially no measurable epilithic layer. No difference in any macroinvertebrate community parameter was detected between control and sediment treatment cobbles in the uncleared reaches, suggesting that isolated events involving elevated levels of suspended sediment have little overall effect on the diversity of cobble communities in the short term. This was consistent with the study of Lenat *et al.* (1981), who observed decreases in density but no significant change in community structure of rocky substrates. The effects of elevated sediment on cobble communities might therefore be more important in the longer term and less so than changes in other variables such as accumulations of deposited sediment and others that affect the epilithon.

The magnitude of changes for physical variables such as conductivity, DO and temperature was generally small and no firm conclusions about the relationship between the diversity of fauna and physical variables can be

made. There was also no significant increase in nutrient inputs as a result of clearing. Increased nutrient input has been associated with riparian removal in other studies with the consequence of marked changes in ecosystem function (see Lake & Marchant 1990). In a cleared pasture stream, Mosisch *et al.* (1999) reported increased nutrients (mainly nitrogen) to be more important in promoting the growth of epilithic algae than light, although filamentous algae have higher light requirements than unicellular microalgae (Richardson *et al.* 1983; Langdon 1998; Hill 1996). The results of the present study are not surprising, however, given poor nutrient status of jarrah forest soils (Bunn & Davies 1990) and the short (<1 km) length of the cleared reach.

The sediment addition treatment in this study was designed to mimic elevated turbidity levels in the water column that would be expected during heavy rains in the catchment when unconsolidated soils washed into streams (pers. obs.). Several elevations of turbidity over a number of days may have a greater effect on cobble communities than the single addition performed for this treatment. Other factors such as the long term effect of deposited sediment on or around cobbles may also be partly responsible for the differences observed between impact and control cobble communities. For example, in the long term, decomposition of compounds associated with materials in the bedload may depress pH and eliminate acid-sensitive species of Plecoptera and Ephemeroptera (Lemly, 1982).

Cobbles are not exposed to sediment deposition to the same extent as the surrounding substrate, however, increases in bedload transport and sediment deposition from the water column have the effect of sealing or smothering microhabitats of zero water velocity at rock-water interfaces at the edges of, or behind cobbles (McClelland & Brüsven 1980). The present study was designed to detect possible changes in the macroinvertebrate community due to elevated turbidity, but might also be used to detect the effects of short-term sediment deposition on and around cobbles since it was observed that sand from the added sediment was deposited from the water column in these areas. A greater period of time between sediment addition and sampling may have allowed some individuals to either detach from, or crawl onto the cobble substrate from surrounding microhabitats. Gazey (1994) found that the ephemeropterans *Neboissophlebia occidentalis*, *Nyungara bunnii* and *Bibulmena kadjina*, and the trichopteran *Smicrophylax australica* began to drift in significant numbers 3–24 hours after experimental sediment addition due to the effects of deposited sediment rather than elevated turbidity, and therefore, effects in the present study may be delayed. Campbell and Doeg (1989) noted that deposited rather than suspended sediment may be more important in reorganising community structure because the effects may be long term if water discharge during spates is not sufficient to cleanse the substrate. Deposition of suspended solids occurs in streams where boundary friction slows the current, however, it is only in low velocity areas that deposits form and elsewhere the sediment is deposited and resuspended at greater rates.

The results from this study suggest that changes in the composition of the epilithic community were the



main cause of changes in the composition of the macroinvertebrate community. Most cobble substrates in uncleared reaches supported a growth of the aquatic plant, *Potamogeton ?crispus*, which extended the height of roughness projection of the cobble and provided a relatively complex microhabitat. The epilithon on cleared cobble substrates did not support the same microhabitat. The microhabitat amongst the epilithon may have provided protection in the short-term from elevated turbidity levels by allowing fauna to avoid any adverse effects such as the clogging of respiratory or food capturing devices. These fauna may have otherwise responded by drifting. Such effects were identified by Lemly (1982) as being the result of long term bedload transport on filtering species, and refuge-seeking behaviour was one of the strategies employed by some macroinvertebrate species in a New Zealand braided stream which was exposed to periodic flood events (Scrimgeour & Winterbourn, 1989).

The loss of *Potamogeton ?crispus* in cleared reaches might be attributable to increased light availability. Limitation in growth and survival of epilithon communities after an increase in sediment levels has also been documented previously (silt accumulation, Graham 1990; sediment abrasion, Lewis 1973a,b). Generally, it appears that shading limits the growth of larger algal forms, with algal assemblages in shaded areas being dominated by diatoms (Duncan & Blinn 1989). When light levels are increased, a greater biomass of green and blue-green algae results, with the growth of diatoms being inhibited by high light conditions (Hill & Knight 1988; Steinman *et al.* 1989; Wellnitz *et al.* 1996). Loss of riparian vegetation in other Australian ecosystems also resulted in a decline in the health of forest streams, and instream primary producers shifted from unicellular algae to filamentous green algae and macrophytes (Bunn *et al.* 1999a). However, there are exceptions to this pattern. In a subtropical stream surrounded by pasture, diffusion surfaces exposed to full sunlight were dominated by diatoms (Mosisch *et al.* 1999), and a layer of diatoms and filamentous algae was present on cobbles in a braided stream in New Zealand, the outcome of scouring and rolling (Scrimgeour & Winterbourn 1989). The cobble epilithon of the jarrah forest streams of the present study resembled the latter unshaded streams.

In Australian tropical and subtropical streams, the epilithic unicellular algae contributes to the majority of consumption by macroinvertebrates on cobbles, but in the northern jarrah forest unicellular algae contributes little more than 50 % of carbon assimilated by cobble fauna (Bunn *et al.* 1999a). Jarrah forest stream ecosystems are almost entirely dependent on terrestrial sources of carbon (Davies 1994; Bunn *et al.* 1999a) and there is a corresponding low diversity of algal grazers (Bunn & Davies 1990). This suggests that measuring only the change in abundance of grazers on cobbles does not have the same utility in the jarrah forest than other stream ecosystems for detecting effects of disturbances. However, the present study has shown that measures of overall macroinvertebrate diversity on cobbles are useful for detecting changes in ecosystem health, and grazers, while making up only part of the community, showed the same response. Furthermore, a simpler measure of the change in macroinvertebrate community structure

would be indicated by obvious changes in the epilithon of cobbles.

Despite previous studies of northern jarrah forest streams that have shown communities within the same stream order to be generally homogeneous (Storey *et al.* 1990), differences in some macroinvertebrate parameters were found between streams. This was probably due to the differences in epilithon cover between streams. Lush growths of *Potamogeton* sp. were not present on every cobble in uncleared reaches, and although not tested in the present study, the variability in epilithic cover in uncleared reaches between streams may have contributed to the variation in macroinvertebrate communities between streams. Cobbles in jarrah forest streams may therefore be more heterogeneous than previously reported (Davies 1994) and this would be a consideration when using cobbles as an indicator habitat for assessing changes in stream ecosystem function.

Changes in chironomid community structure, as observed in the present study, is often used as an indicator of change in the stream ecosystem (Storey & Edward 1989, Smith & Cranston 1994, Edward *et al.* 2000). Other species might have some use in this regard. A number of the taxa on impact cobbles were unique to this treatment, although the proportion of unique taxa in uncleared areas was greater. Species unique to cleared cobble communities included the simuliid *Cnephia tonnoiri* and the coleopterans *Hyphydrus* sp. and *Psychodidae* sp., however, they were relatively uncommon and therefore may need to be used in conjunction with other indicators such as changes in epilithon. Unique taxa may be present on cleared cobbles for one of two reasons. Firstly, they could be relatively rare and therefore collected by chance alone, or secondly, some species such as the water beetle *Sternopriscus marginata* may have preferred other stream meso-habitats prior to clearing, but increased in numbers in cobble habitats as they became more favourable in some respect.

In conclusion, complete removal of riparian vegetation adversely affects biodiversity and composition of macroinvertebrate fauna and epilithic vegetation on cobbles. The most important changes are likely to include reduced bank stability, and increased light and sediment levels. Short-term increases in suspended sediment levels appear to have minimal effect, but the longer term effect, coupled with that of sediment deposition could be greater. Cobble communities as a whole are sensitive to changes in the aquatic ecosystem and are recommended for use when assessments of ecosystem health are being made. The most useful indicator of changes in ecosystem health may be the change in composition and biomass of epilithon. In the jarrah forest, clearing causes a general change from *Potamogeton ?crispus* to diatomaceous layers. Removal of riparian vegetation is not recommended if aquatic biodiversity and normal ecosystem functioning is to be maintained. This has implications in the south-west of Western Australia for logging practices, whereby streamside vegetation should be maintained on all streamlines.

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## References

- Abernathy B & Rutherford ID 1999 Where along a river's length will vegetation most effectively stabilize stream banks? *Geomorphology* 23: 55–75.
- Belbin L 1995 Pattern Analysis Package PATN. CSIRO Division of Wildlife and Ecology, Canberra.
- Bremner JM & Mulvaney CS 1982 Ammonia electrode modification of the Kjeldahl method. In: *Methods of Soil Analysis, Part 2* (ed AL Page). American Society of Agronomy, Soil Science Society of America, Madison, Wisconsin, USA.
- Brown JAH 1972 Hydrologic effects of a bushfire in a catchment in south-eastern New South Wales. *Journal of Hydrology* 15: 77–96.
- Bunn SE 1985 Community structure and functional organization of streams of the northern jarrah forest, Western Australia. PhD thesis, Department of Zoology, The University of Western Australia, Perth.
- Bunn SE 1986 Spatial and temporal variation in the macroinvertebrate fauna of streams of the northern jarrah forest, Western Australia: functional organization. *Freshwater Biology* 16: 621–632.
- Bunn SE & Davies PM 1990 Why is the stream fauna of south-western Australia so impoverished? *Hydrobiologia* 194: 169–176.
- Bunn SE, Davies PM & Mosisch TD 1999a Ecosystem measures of river health and their response to riparian and catchment degradation. *Freshwater Biology* 41: 333–345.
- Bunn SE, Mosisch TD & Davies PM 1999b Temperature and light. In: *Riparian Land Management Technical Guidelines Vol I*. (eds S Lovett & P Price). Land and Water Resources Research and Development Corporation, Canberra, 17–24.
- Campbell IC & Doeg TJ 1989 Impact of timber harvesting and production on streams: a review. *Australian Journal of Marine and Freshwater Research* 40: 519–539.
- Cummins KW 1962 An evaluation of some techniques for the collection and analysis of benthic samples with special emphasis on lotic waters. *The American Midland Naturalist* 67: 477–504.
- Cummins KW 1973 Trophic relations of aquatic insects. *Annual Review of Entomology* 18: 183–206.
- Davies PM 1994 Ecosystem ecology of upland streams of the northern jarrah forest, Western Australia. PhD thesis, Department of Zoology, The University of Western Australia, Perth.
- Doeg TJ & Milledge GA 1991 Effect of experimentally increasing concentrations of suspended sediment on macroinvertebrate drift. *Australian Journal of Marine and Freshwater Research* 42: 519–526.
- Duncan SW & Blinn DW 1989 Importance of physical variables on the seasonal dynamics of epilithic algae in a highly shaded canyon stream. *Journal of Phycology* 25: 455–461.
- Edward DH, Storey AW & Smith MJB 2000 Assessing river health in South-Western Australia: Comparison of macroinvertebrates at family level with Chironomidae at species level. *Verhandlungen Internationale Vereinigung Limnologie* 27: 2326–2335.
- Gazey P 1994 The impact of sedimentation on the benthic macroinvertebrate fauna of streams and rivers. Honours thesis, Department of Zoology, The University of Western Australia, Perth.
- Graham AA 1990 Siltation of stone-surface periphyton in rivers by clay-sized particles from low concentration in suspension. *Hydrobiologia* 199: 107–115.
- Gray LJ & Ward JV 1982 Effects of sediment releases from a reservoir on stream macroinvertebrates. *Hydrobiologia* 96: 177–184.
- Gregory SV, Swanson FJ, McKee WA & Cummins KW 1991 An ecosystem perspective of riparian zones: focus on links between land and water. *BioScience* 41: 540–551.
- Hart BT, Davies SH & Thomas PA 1992 Transport of iron, manganese, cadmium, copper and zinc by Magela Creek, Northern Territory. *Water Research* 16: 605–612.
- Hill WR 1996 Factors affecting benthic algae – effects of light. In: *Algal Ecology – Freshwater Benthic Ecosystems* (eds RJ Stevenson ML Bothwell & RL Lowe). Academic Press, San Diego, 128–141.
- Hill WR & Knight AW 1988 Nutrient and light limitation of algae in two northern California streams. *Journal of Phycology* 24: 125–132.
- Hogg ID & Norris RH 1991 Effects of runoff from land clearing and urban development on the distribution and abundance of macroinvertebrates in pool areas of a river. *Australian Journal of Marine and Freshwater Research* 42: 507–518.
- Jutson JT 1950 The physiography (geomorphology) of Western Australia. *Bulletin* 95, Geological Survey of Western Australia, Perth.
- Karr JR & Schlosser IJ 1978 Water resources and the land-water interface. *Science* 201: 229–234.
- Lake PS 1995 Of floods and droughts: river and stream ecosystems of Australia. In: *River and Stream Ecosystems* (eds CE Cushing, KW Cummins & GW Minshall). Elsevier, Amsterdam, 659–694.
- Lake PS & Marchant R 1990 Australian upland streams: ecological degradation and possible restoration. *Proceedings of the Ecological Society of Australia* 16: 79–91.
- Langdon C 1998 On the causes of interspecific differences in the growth irradiance relationship for phytoplankton. II. A general review. *Journal of Plankton Research* 10: 1291–1312.
- Lemly AD 1982 Modification of benthic insect communities in polluted streams: combined effects of sedimentation and nutrient enrichment. *Hydrobiologia* 87: 229–245.
- Lenat, DR Penrose DL & Eagleson KW 1981 Variable effects of sediment addition on stream benthos. *Hydrobiologia* 79: 187–194.
- Lewis K 1973a The effect of suspended coal particles on the life forms of the aquatic moss *Eurhynchium riparioides* (Hedw.). I. The gametophyte plant. *Freshwater Biology* 3: 251–257.
- Lewis K 1973b The effect of suspended coal particles on the life forms of the aquatic moss *Eurhynchium riparioides* (Hedw.). II. The effect on spore germination and regeneration of apical tips. *Freshwater Biology* 3: 391–395.
- Likens GE & Bormann FH 1974 Linkages between terrestrial and aquatic ecosystems. *BioScience* 24: 447–456.
- Lynch RJ & Catterall CP 1999 Riparian wildlife and habitats. In: *Riparian Land Management Technical Guidelines Vol I*. (eds S Lovett & P Price). Land and Water Resources Research and Development Corporation, Canberra, 121–136.
- McClelland WT & Brusven MA 1980 Effects of sedimentation on the behaviour and distribution of riffle insects in a laboratory stream. *Aquatic Insects* 2: 161–169.
- Merritt RW & Cummins KW 1978 An Introduction to the Aquatic Insects of North America. Kendall/Hunt Publishing Company, Dubuque, Iowa.
- Mosisch TD, Bunn SE, Davies PM & Marshall CJ 1999 Effects of shade and nutrient manipulation on periphyton growth in a subtropical stream. *Aquatic Botany* 64: 167–177.
- Nutall PM 1972 The effects of sand deposition upon the macroinvertebrate fauna of the River Camel, Cornwall. *Freshwater Biology* 2: 181–186.
- Prosser I, Bunn S, Mosisch T, Ogden R & Karssies L 1999 The delivery of sediment and nutrients to streams. In: *Riparian Land Management Technical Guidelines Vol I* (eds S Lovett

- & P Price). Land and Water Resources Research and Development Corporation, Canberra, 37-60.
- Quinn JM, Williamson RB, Smith RS & Vickers ML 1992 Effects of riparian grazing and channelization on streams in Southland, New Zealand. 2. Benthic macroinvertebrates. *New Zealand Journal of Marine and Freshwater Research* 26: 259-273.
- Richardson K, Beardall J & Raven JA 1983 Adaptation of unicellular algae to irradiance: an analysis of strategies. *New Phytologist* 93: 157-191.
- SAS (1994). SAS for personal computers, Version 7. SAS Institute Inc. Cary, NC, USA, 1292 pages
- Scrimgeour GJ & Winterbourne MJ 1989 Effects of floods on epilithon and benthic macroinvertebrate populations in an unstable New Zealand stream. *Hydrobiologia* 171: 33-44.
- Shannon C & Weiner W 1949 The mathematical theory of communication. University of Illinois Press, Urbana, Illinois.
- Smith MJ & Cranston PS 1994 'Recovery' of an acid mine-waste impacted tropical stream: the chironomid story. In: *Chironomids: From Genes to Ecosystems* (ed PS Cranston). CSIRO, Melbourne, 161-173.
- Steinman AD, McIntire CD, Gregory SV & Lamberti GA 1989 Effects of irradiance and grazing on lotic algal assemblages. *Journal of Phycology* 25: 478-485.
- Storey AW, Bunn SE, Davies PM & Edward DHD 1990 Classification of the macroinvertebrate fauna of two river systems in South-western Australia in relation to physical and chemical parameters. *Regulated Rivers: Research and Management* 5: 217-232.
- Storey AW & Edward DHD 1989 Longitudinal variation in community structure of Chironomidae (Diptera) in two south-western Australian river systems. *Acta Biologica Debrecen Oecologica Hungarica* 3: 315-328.
- Wallace JB & Merritt RW 1980 Filter feeding ecology of aquatic insects. *Annual Review of Entomology* 25: 103-132.
- Waters TF 1995 *Sediment in Streams: Sources, Biological Effects and Control*. American Fisheries Society, Bethesda, Maryland, USA.
- Wellnitz TA, Rader RB & Ward JV 1996 Light and a grazing mayfly shape periphyton in a Rocky Mountain stream. *Journal of the North American Benthological Society* 15: 496-507.
- Wetzel RG & Likens GE 1979 *Limnological analysis*. WB Saunders and Co, Philadelphia.