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Effects of Forestry Activities (Clearfelling) on Stream Macroinvertebrate Fauna in South-western Australia

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Abstract

The effects of forestry activities on macroinvertebrate community structure were examined in the headwaters of Carey Brook in the south-west of Australia. The fauna at four sites on an upland stream that ran through a logging coupe were compared, before and after clearfelling, with the fauna at four nearby undisturbed sites. Mean species richness and mean total abundance declined at the treatment sites relative to the control sites after the commencement of clearfelling activities. The composition of the macroinvertebrate fauna in the disturbed stream changed in comparison with that in the undisturbed sites after logging started but returned to the pre-logging composition after winter and spring rains had stopped. The observed changes in the macroinvertebrate fauna occurred during the periods of high loads of suspended inorganic solids at the treatment sites. The possible reasons for the observed results are discussed.

Introduction

Lotic systems are inherently linked to the terrestrial ecosystems that surround them (Hynes 1970), and therefore disturbances that arise within a catchment frequently affect the physical, chemical and biological components of stream systems. Forestry activities are such disturbances and can affect the fauna within a stream as a result of increased concentrations of nutrients (Murphy *et al.* 1981; Hawkins *et al.* 1982), increased loads of suspended or deposited sediments (Borg *et al.* 1987; Doeg *et al.* 1987; Doeg and Milledge 1991), increased stream flow (Campbell and Doeg 1989), increased water temperatures due to the removal of riparian vegetation, increased salinity (Graynoth 1979; Hart *et al.* 1990), or the synergistic effects of these factors (Lemly 1982). However, most of the available literature on the effects of forestry activities has dealt primarily with water quality and quantity (e.g. Likens *et al.* 1977). Relatively few studies worldwide concern the effects on stream fauna, particularly macroinvertebrates, although the potential for such effects has been known for some time in Australia and overseas (Tebo 1955; Michaelis 1984). The literature concerning the effects of forestry activities on streams has recently been reviewed by Campbell and Doeg (1989).

Most studies that have examined the effects of forestry activities on the macroinvertebrate fauna of streams have compared the faunas of streams in clearfelled areas with those in nearby undisturbed streams up to 40 years after logging ceased (Murphy and Hall 1981; Murphy *et al.* 1981; Silsbee and Larson 1983). The results from these studies are valuable, but they can be criticized because the macroinvertebrate fauna in the disturbed streams was not described before the logging took place, so the differences between streams could be attributable to other factors apart from the forestry activity itself.

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To date, only two published Australian studies have examined short-term responses of macroinvertebrate fauna to forestry activities. Robinson (1977) found that a lower diversity of macroinvertebrate fauna was present in streams draining two clearfelled catchments in comparison with an undisturbed stream. Similarly, Richardson (1985) noted changes in the benthic communities downstream of logging operations, mainly due to poor roading and causeway construction, that were present for at least nine months after increases in siltation and turbidity were recorded in the stream. Another recent study (Growns and Davis 1991) showed that the invertebrate fauna in streams in the south-west of Australia can be affected by logging activities for up to eight years after forestry activities have ceased.

The present study examined the effects of forestry activities on stream macroinvertebrate communities in the southern forests of Western Australia. The fauna at four sites on a stream that ran through a logging coupe was compared, before and after clearfelling, with the fauna at four nearby undisturbed sites. Changes in macroinvertebrate communities were examined and related to changes in water quality that occurred as a result of the logging disturbance.

Study Sites and Forestry Activities

Sampling sites were situated on Carey Brook, a tributary of the Donnelly River, approximately 300 km from Perth in the south-west of Western Australia (34°22′S,155°51′E) (Fig. 1). All forestry operations that occurred in the logging area during the study period were managed by the Department of Conservation and Land Management.

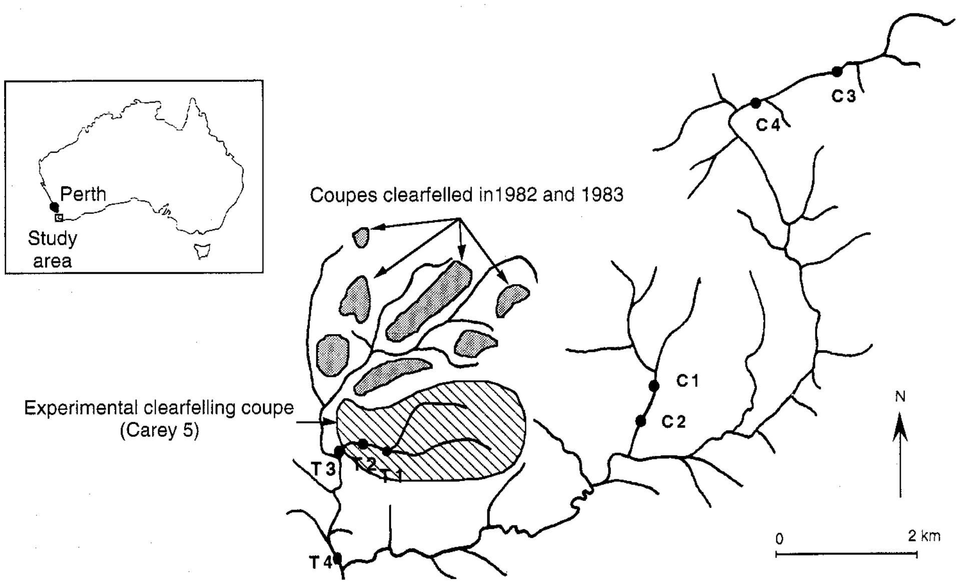

Fig. 1. Treatment (T1–T4) and control (C1–C4) study sites on Carey Brook.

The logging coupe, situated on deep red-loam soils, comprised mainly karri forest (Eucalyptus diversicolor). Several roads were constructed in 1988 to facilitate the clearfelling operation. Most of the karri forest was clearfelled during May and June 1990. Logging operations ceased during the remaining of the winter and spring of 1990 and were completed in the summer of 1991. A wildlife corridor of riparian vegetation that acted as a buffer strip (approximately 100 m wide) was left along the stream banks of the northern arm of the tributary throughout the clearfelling operations. Buffer strips were situated along both the northern (100 m width) and the southern arms (<30 m width) of the Carey Brook tributary, but 'windows' were created by skidders that were driven through the stream at several points despite the presence of nearby roads.

Eight sampling sites were chosen for the study: two within the logging coupe (T1 and T2), two downstream of the logging coupe (T3 and T4), and four on nearby undisturbed streams (C1–C4). The sites were between 1 and 3 m wide, on second- or third-order streams, and of low gradient with fine sand to gravel substrata. Altitudes ranged between 100 and 220 m above sea level and catchment sizes ranged from $1 \cdot 6$ to $3 \cdot 8$ km^2 except that Site T4 had a catchment area of $14 \cdot 4$ km^2. Bankfull stream widths ranged from $1 \cdot 4$ to $3 \cdot 2$ m and bankfull depths were all less than 40 cm. None of the sites dried out during this study, but they become very shallow (<4 cm) during late summer. The control sites were situated between 3 and 8 km from the treatment sites (Fig. 1).

Materials and Methods

Macroinvertebrate communities were sampled at the eight sites on 12 occasions between November 1988 and July 1990, except that Sites C1 and C2 were not sampled in January 1989 owing to equipment failure. Six replicate benthic samples were taken randomly from each site on each sampling occasion with a modified Boulton (1985) sampler (see Growns 1990). Macroinvertebrate samples were stored in 70% ethanol until they were sorted. Organic matter from each sample was divided into coarse particulate organic matter (CPOM, >1 mm) and fine particulate organic matter (FPOM, $0 \cdot 5$ to 1 mm), oven-dried to constant weight, and ashed at 550°C to obtain ash-free dry weight (AFDW). Invertebrates were identified to the lowest practicable taxon. The number of individuals of each taxon, the total abundance of individuals, and the richness (number of taxa per sample) were recorded for each sample.

Concentrations of inorganic and organic suspended sediments, conductivity and pH were measured from 500-mL water samples taken at each site on each sampling occasion. The samples were kept cool until conductivity and pH were measured with appropriate meters. Suspended solids were collected by filtering each water sample through a Whatman No. 541 ashless filter paper with a pore size of 25 μm (predried at 50°C for 24 h and weighed before use). The collected sediment and filter paper were then redried to constant weight and ashed at 550°C to obtain the AFDW of organic matter. The weight of the organic fraction was adjusted (assuming 15% ash content of the organic matter) to obtain the original weight of the organic suspended solids and subtracted from the total weight to obtain the weight of inorganic suspended sediments.

Rainfall data were provided by the Bureau of Meteorology from three meteorological centres (Nannup, Manjimup and Pemberton) that lay within 40 km of the sampling sites. Rainfall at the study site was assumed to be the average of the rainfall at these three centres.

Data Analysis

The macroinvertebrate data were classified by using two-way indicator species analysis (TWINSPAN) (Hill 1979). This polythetic approach to classification is widely used in aquatic studies and usually produces robust patterns with invertebrate data (e.g. Barmuta 1989; Storey *et al.* 1990; Growns *et al.* 1992). Ordination of the data was carried out by using hybrid multidimensional scaling (SSH), a robust ordination technique (Faith 1991) available in the PATN software package (Belbin 1988). The Kulczynski coefficient was used as a measure of dissimilarity between samples, as recommended by Faith *et al.* (1987). The abundance data for each macroinvertebrate taxon were standardized by subtracting the minimum abundance of each taxon and dividing by the associated range (a range transformation) to reduce the weighting of abundant taxa by the Kulczynski association measure (Belbin 1988). Five dimensions were chosen for the SSH analysis on the basis of the stress parameter. To reduce the chance of local optima (Faith 1990), 500 random starts were carried out for each analysis.

The ordination was carried out by using the mean abundances of invertebrates in each taxon from all samples taken at each site on each sampling occasion. This ordination contained 94 objects, each point representing one of eight sites on one of 12 sampling occasions, except for Sites C1 and C2 that were not sampled in January 1989. Greater separation between the pre- and post-disturbance clusters in the treatment samples in comparison with the separation between the pre- and post-disturbance clusters in the control samples was considered to indicate that the macroinvertebrate community structure had changed at the treatment sites as a result of clearfelling.

Community parameters at treatment and control sites were compared before and after clearfelling by using repeated-measures analysis of variance (ANOVA) in which particular orthogonal contrasts of interest could be examined (Norusis 1985). The contrasts that were of most interest were the differences in the richness and total abundance (log-transformed) of macroinvertebrates in samples collected before and after clearfelling. The hypothesis that richness and total invertebrate abundance would change at the treatment sites after clearfelling suggests that a significant interaction should occur between sites with and without the clearfelling disturbance over time (a significant Time by Treatment interaction). Contrasts were tested by using the MANOVA procedure of the SPSSx statistical software package (Anon. 1986). Prior to the examination of the results of the repeated-measures ANOVA, the assumption of compound symmetry of the variance–covariance matrix was tested by using Mauchly's criterion. Where this assumption was rejected, the results were corrected by using the Greenhouse–Geisser (1959) epsilon (Potvin *et al.* 1990).

The mean total abundance (log-transformed) and mean richness of six samples at each site on each sampling occasion were used as inputs to the repeated-measures ANOVA. In this sense, the replicate samples were being used to obtain a better indication of the population means at each site and time. Sites C1 and C2 were not sampled on the second sampling occasion, so data from all sites on that date were not used in the repeated-measures ANOVA. This left three pre-treatment sampling occasions and eight post-treatment occasions for both of the community variables. Changes in the abundances of individual taxa as a result of the logging treatment could not be examined by using repeated-measures ANOVA because transforming the data did not achieve homoscedasticity, mainly because of the large proportion of zero abundances in the data matrix.

To examine the behaviour of individual taxa in response to clearfelling, the data set was divided into four groups: pre-impact treatment and control and post-impact treatment and control. Descriptive statistical information about the contribution of individual taxa to the discrimination of these pre-defined groups was assessed by using Cramer values (between-group variance/total variance), which are measures of the statistical contribution of each taxon to the discrimination of each pre-defined group (Williams *et al.* 1987). Values range from 0, implying that there is no discrimination among groups for a taxon, to a value of 1, suggesting that there is no overlap of species abundances between groups (Belbin 1991). The Cramer values of taxa that showed either an increase or a decrease from pre- to post-impact at treatment sites in comparison with the situation at control sites were tested for statistical significance by using 100 Monte Carlo randomizations of the data set. The Cramer value of the original data was compared with that of each randomized data set for each taxon. A Cramer value for a particular taxon was considered to be significant if it was greater than 95 Cramer values from randomized data sets, indicating that the observed degree of overlap in the abundance of a species among the pre-defined groups did not occur by chance alone.

Results

Environmental Variables

Most rainfall during the study period occurred in late winter and early to mid spring 1989. Little rain fell during late spring 1989, throughout summer 1989–90, or in early autumn in both years, and rainfall increased again towards the end of the study. An average of 100 mm was recorded in each six-weekly period from late autumn 1990 to the end of the study.

All sites were shallow (<35 cm deep) during sampling. Conductivities ranged between 300 and 450 $\mu S\ cm^{-1}$ at the treatment sites and between 200 and 300 $\mu S\ cm^{-1}$ at the control sites. Temperature, mean velocity, depth and pH varied over time but differed little between treatment and control sites. Larger mean amounts of CPOM and FPOM were recorded at the control sites than at the treatment sites throughout the study, but no consistent differences were apparent in these variables before and after clearfelling. Thus, none of these environmental variables could be used to explain differences in macroinvertebrate communities that may have occurred before and after clearfelling.

The inorganic component of suspended sediments recorded at the treatment sites increased in comparison with that at the control sites from August 1989 until November 1989 (Fig. 2*a*). An increase in the organic component was not observed over this period (Fig. 2*b*). The first suspended sediments to cause an observable opaqueness of the water occurred on 30 June after a rainfall event averaging more than 25 mm. The stream was opaque at the treatment sites on the three sampling occasions between August and November 1989 but was clear in December. The water was clear in the streams during late winter 1988 before macroinvertebrate sampling began. Another increase in the inorganic component of the suspended solids was observed on the last macroinvertebrate sampling occasion in July 1990 at the treatment sites. For most of the time little inorganic sediment was present in these streams.

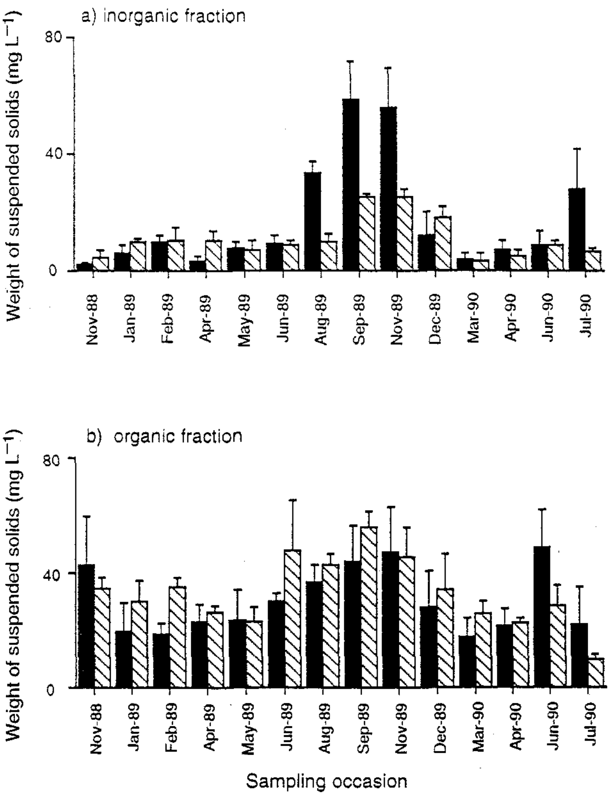

Fig. 2. Weight of suspended sediments recorded from water samples taken at (solid bars) treatment and (hatched bars) control sites on each sampling occasion and in February 1989 and March 1990.

Invertebrate Taxa

In all, 160 taxa were collected during the study. Six of these taxa were recorded at the treatment sites before logging began and were not recorded either after the commencement of logging or during the periods of high loads of suspended inorganic solids (Table 1). These taxa were *Baetis soror*, *Dicrotendipes* spp., Leptoceridae sp. 10, *Diplectrona* sp., Oxus sp. and Tanypodinae sp. 3. Most of these taxa were recorded at various control

Table 1. Responses of selected macroinvertebrate taxa to clearfelling disturbances in treatment and control sites

Sample groupings: TB and CB, treatment and control sites before clearfelling; TA and CA, treatment control sites after clearfelling. $^{*}P < 0{\cdot}05$ and $^{**}P < 0{\cdot}01$, based upon 100 Monte Carlo randomization tests

Taxon	Present in treatment sites after clearfelling commenced?	Present during increase in suspended sediments? Treatment sites	Control sites	Mean abundance in sample groups	Cramer value for the contribution of taxon to discrimination of sample groups	Presence in 94 sites/ sampling occasions (%)
Species absent from treatment sites after the commencement of clearfelling or during periods of high loads of suspended sediments						
Baetis soror	Yes	No	Never present	$TB \approx TA$	—	6
Dicrotendipes spp.	Yes	No	Yes	$TB > TA$ and $CB > CA$	—	7
Diplectrona sp.	Yes	No	Yes	$TB \approx TA$ and $CB < CA$	—	38
Leptoceridae sp. 10	No	No	Never present	Present only in TB	—	2
Oxus sp.	Yes	No	Yes	$TB < TA$ and $CB < CA$	—	23
Tanypodinae sp. 3	Yes	No	Yes	$TB \approx TA$ and $CB > CA$	—	61
Species indicated by descriptive statistics as showning an increase in abundance after clearfelling at the treatment sites in comparison with the control sites						
Ceratopogonidae sp. 7	Yes	Yes	Yes	$TB \ll TA$ and $CB = CA$	0·32**	19
Dolichopodidae sp.	Yes	Yes	Never present	$TB \ll TA$	0·42**	12
Hurleya sp. nov.	Yes	Yes	Yes	$TB < TA$ and $CB \approx CA \approx 0$	0·37**	26
Newmanoperla exigua	Yes	Yes	Yes	$TB \ll TA$ and $CB \approx CA$	0·28*	20
Platyhelminthes	Yes	Yes	Yes	$TB \ll TA$ and $CB \approx CA$	0·37**	34
Tanypodinae sp. 1	Yes	Yes	Yes	$TB < TA$ and $CB > CA$	0·35**	90
Tipulidae sp. 2	Yes	Yes	Yes	$TB \ll TA$ and $CB < CA$	0·30**	54
Tipulidae sp. 8	Yes	Yes	Yes	$TB \ll TA$ and $CB < CA$	0·55**	40
Species indicated by descriptive statistics as showing a decrease in abundance after clearfelling at the treatment sites in comparison with the control sites						
Australiobates sp.	Yes	No	Yes	$TB > TA$ and $CB < CA$	—	51
Candonocypris novaezelandiae	Yes	Yes	Yes	$TB \gg TA$ and $CB > CA$	0·40**	36
Ceratopogonidae sp. 13	Yes	No	Yes	$TB > TA$ and $CB < CA$	0·33*	11
Empididae sp. 1	Yes	No	Yes	$TB \gg TA$ and $CB < CA$	0·50**	58
Odonata sp. juv.	Yes	Yes	Yes	$TB > TA$ and $CB < CA$	0·34*	70
Taschorema pallescens	Yes	No	Yes	$TB > TA$ and $CB \approx CA \approx 0$	0·36*	20

sites during the study period (Table 1). Fourteen taxa changed significantly in abundance at the treatment sites after the commencement of logging, in comparison with the control sites. Eight taxa increased in average abundance after logging, and six taxa decreased in abundance after logging (Table 1).

Richness and Total Invertebrate Abundance

Mean richness and mean total abundance declined at the treatment sites relative to the control sites after the commencement of clearfelling activities (Fig. 3). The observed decreases clearly occurred during the periods of high loads of suspended inorganic solids at the treatment sites. However, repeated-measures ANOVA indicated that there was no statistically significant interaction between treatments over time for either parameter, indicating that clearfelling activity within the catchment had little or no effect on richness or total abundance over the study period. However, the small number of treatment and control sites in the present study would have reduced the power of the analysis to detect a statistically significant interaction.

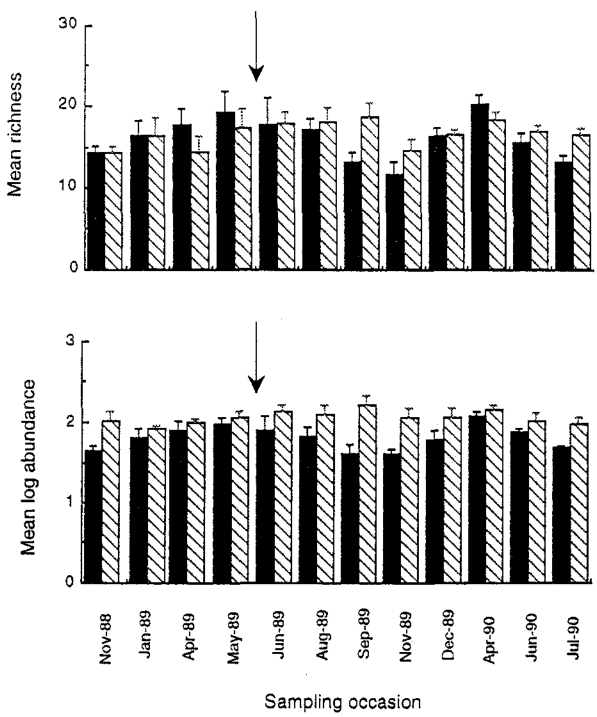

Fig. 3. Mean taxonomic richness and mean log abundance (+ s.e.) measured at (solid bars) treatment and (hatched bars) control sites on each sampling occasion. Arrows indicate the start of clearfelling disturbance.

Classification

The first division of the classification (TWINSPAN) separated all the Site C3 samples from the remaining sampling occasion/site combinations (Fig. 4). This indicates that the macroinvertebrate community at this site differed from those at the other sites throughout this study. The second division of the classification separated 80% of the pre-disturbance samples at the treatment sites from the other site/time combinations, and the third division separated 88% of the post-disturbance samples at the treatment sites from the majority

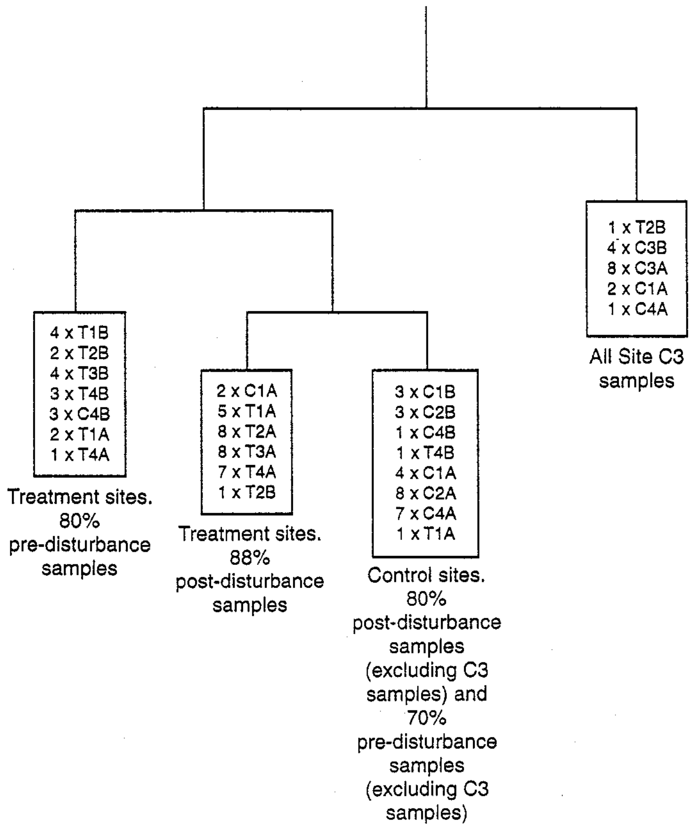

Fig. 4. Classification of samples by TWINSPAN. The number of treatment and control sampling occasions is indicated. T1 to T4, treatment sites; C1 to C4, control sites; B, before-clearfelling sampling occasions; A, after-clearfelling sampling occasions.

of the remaining pre- and post-disturbance samples at the control sites (Fig. 4). This indicates that the community composition changed at the treatment sites after clearfelling commenced in comparison with the fauna at the control sites.

Ordination

The majority of post-impact samples at the treatment sites separated from the cluster of the remaining samples at the control and treatment sites at an angle to the second and fourth vectors of the first SSH solution. The samples at the treatment sites or the sixth sampling occasion (D6 in Fig. 5) (late winter 1989) separated first from the remaining samples at the treatment sites (Fig. 5). The samples at the treatment sites on the seventh and eighth sampling occasions (early and late spring 1989) also had a similar distance from the remaining samples at the treatment sites. However, by summer on the ninth sampling occasion, the macroinvertebrate community structure was again similar to the pre-disturbance community structure. The samples taken during the 1990 sampling occasions (tenth, eleventh and twelfth) at the treatment sites move away, in sequence, from the cluster of the remaining samples in a similar direction as the samples taken on the sixth, seventh and eighth sampling occasions. The increasing distance of the treatment centroids is clearly associated with the increased loads of suspended inorganic sediment in the stream at those times (Fig. 2).

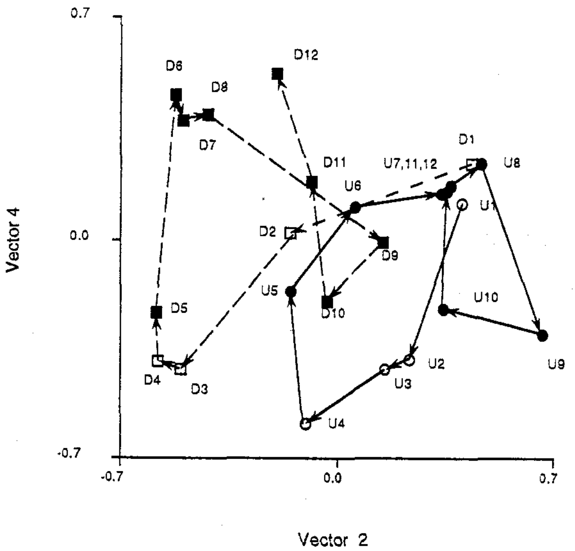

Fig. 5. Time series of centroids of samples at treatment and control sites (pre- and post-impact) SSH ordination space. Squares, treatment site centroids; circles, control site centroids; open symbols, pre-impact sampling occasions; closed symbols, post-impact sampling occasions. D, treatment centroids; U, conttrol centroids; numbers refer to sampling occasions.

The post-impact samples at the control sites showed a similar increase in distance on the fifth and sixth sampling occasions as did the treatment sites. However, by the seventh and eighth sampling occasions, the macroinvertebrate community structure at the control sites was similar to that in the pre-impact samples. This pattern was observed in the control sites again in 1990.

Discussion

Richness and total invertebrate abundance of the macroinvertebrate community in Carey Brook decreased as a result of the clearfelling disturbance at the treatment sites during periods of high loads of suspended sediment. This result is consistent with the findings of Robinson (1977), who demonstrated that invertebrate density was much lower in a stream in a catchment that had been clearfelled than at nearby undisturbed sites. However, the fauna in the present study appeared to recover quickly and to return to densities and richness comparable to those of undisturbed sites once loads of suspended sediment had returned to pre-logging values.

Differences between the compositions of macroinvertebrate communities at the treatment and control sites, as revealed by ordination and classification, also showed that clearfelling had an effect on the fauna. The abundances of several macroinvertebrate taxa declined at the treatment sites in comparison with control sites after clearfelling commenced, whereas the abundances of others increased. This result supports the findings of Richardson (1985), who demonstrated changes in abundance of the dominant species inhabiting sand and shingle substrata downstream of logging activity that were evident for at least nine months after increases in siltation and turbidity were recorded in the stream. The differing responses of macroinvertebrates to clearfelling as a result of increased suspended solids in the present treatment sites are also consistent with the findings of Rosenberg and Wiens (1978), who found that the responses of invertebrates to increases in suspended sediment depend on the individual taxon and the time of year. In a later study, Rosenberg and Wiens (1980) suggested that the responses of specific taxa could be related to differences in the microhabitats that the invertebrates occupy.

Explanation of the responses of individual taxa is hampered by a lack of knowledge of their general ecology and their behaviour under stressful conditions. One possible mechanism for the observed increases in the abundance of particular taxa in the treatment sites during clearfelling may be reduced competition for a limiting resource if another taxon that once used the resource was adversely affected by the clearfelling. Increased suspended sediment may also suppress the actions of predators on prey taxa. Walde (1986) demonstrated this effect but also demonstrated that in some situations the increased suspended sediment could actually enhance the effects of predators on prey taxa, leading to a decrease in the abundance of the prey taxon. The abundance of a filter feeder, *Diplectrona* sp., decreased in abundance during the periods of increased suspended sediments in the present treatment sites. This invertebrate is a net-spinning filter feeder, so it is likely that the increase in suspended solids may have affected its abundance through inorganic suspended sediments clogging the nets. However, the abundances of other filter-feeding taxa appear to have been unaffected by the logging disturbance.

The changes that occurred within the fauna of Carey Brook occurred at the times of high loads of suspended sediment. However, the responses of the macroinvertebrate community were smaller than those found by other studies. The sediment loads that occurred at the treatment sites in between the six-weekly sampling occasions would probably have been much higher than the maximum average of 60 mg L^{-1} measured during this study. Campbell and Doeg (1989) noted that peak loads of sediment are carried during the initial stages of increase in flow after a rain event, and such events would have occurred numerous times in between the present sampling occasions. Gammon (1970) found that loads of suspended sediment of between 40 and 120 mg L^{-1} resulted in a decrease in the population density of macroinvertebrates of between 25% and 60% in an American stream. In contrast, the response of the fauna in the present study appears to be somewhat reduced. This may indicate that the fauna in Carey Brook is more resistant to high loads of suspended sediment than is comparable American fauna. This could be a by-product of adaptations by the fauna to historical drought events that occurred in Western Australia (De Deckker 1986; Bunn and Davies 1990). De Deckker suggested that the largest of these droughts occurred 18 000 years ago and severely disrupted aquatic systems throughout Australia. Bunn and Davies (1990) suggest that this problem was accentuated in Western Australia by the inherent dryness of the state. The fauna that survived these droughts may be inherently resilient to environmental disturbance.

Little work has been done on the effects of high loads of suspended sediment on Australian stream fauna. However, Doeg and Milledge (1991) showed that a single pulse of inorganic suspended sediments of up to 135 mg L^{-1} increased the drift of macroinvertebrates by sevenfold in an artificial stream in Victoria. The substratum of this artificial stream was comprised primarily of cobbles and gravel, and so the smaller response shown by the macroinvertebrates in Carey Brook may be partly due to the substrata present in the study streams (mostly fine sands to gravels). Such substrata are usually mobile under high water velocities, and it may be that the fauna is used to small local disturbances resulting from moving sediments during periods of high flow. Also, Halse and Blyth (1992) suggested that the effects of sedimentation resulting from logging disturbance will depend largely on two factors: firstly, whether fine sediment penetrates the stream bed, how long it remains, and whether it changes the structure or composition of the substratum; secondly, whether the stream fauna is dependent on certain physical features of the interstitial spaces of the substratum, e.g. dependent on interstitial spaces free of fine inorganic sediment at certain life stages or for use as refugia in dry periods, or dependent on high ratios of organic to inorganic fine sediments for food. The nature of the substratum in the upland sections of Carey Brook means that additional fine sediment from logging activities will probably not penetrate the stream bed to any great extent. Also, fine substrata have relatively few interstitial spaces and so they will not be as important as refugia as they are in other streams with substrata made up of larger particles.

The fauna in the present study changed at the treatment sites owing to increases in suspended solids but returned to a fauna similar to that in pre-clearfelling samples after the suspended solids decreased at the beginning of summer. At the end of the present study, the suspended sediments had started to increase again at the start of the winter rainfall and the fauna at the treatment sites appeared to be responding in a fashion similar to the way it responded to the increases in suspended solids the previous year. The changes in the macroinvertebrate fauna would probably show the same annual pattern, depending on the amounts of suspended solids that are washed into the stream from the clearfelling area during periods of high rainfall. Borg *et al.* (1987) showed that suspended solids arising from clearfelling activities in Western Australia decreased a few years after logging ceased owing to stabilization of the soil by regeneration of vegetation in the logged area. The effects of clearfelling on the invertebrate fauna during periods of high rainfall observed in the present study may therefore decrease after several years. However, there may be longer-term effects on the fauna from clearfelling possibly owing to increased conductivity that may result from forestry activities (Wood 1924; Peck and Hurle 1973; Borg *et al.* 1987). Borg *et al.* (1987) showed that a gradual increase in salinity can occur after the end of clearfelling and that the salinity continues to increase for up to eight years. It is probably this factor that results in altered macroinvertebrate community structure in the longer term (Growns and Davis 1991). Resampling of the sites in the present study may indicate if the communities have also been affected in the longer term by increases in salinity.

Finally, the response of the fauna to the clearfelling might have been greater if a strip of riparian vegetation had not been present over the major part of the stream length. Other studies have shown that buffers are effective in ameliorating logging disturbance to stream water quality and quantity and fauna (Graynoth 1979). Growns and Davis (1991) found that a 100-m-wide buffer left along a stream edge appeared to reduce changes in the stream fauna, at least eight years after logging had ceased. A narrower buffer would no doubt offer some protection to stream fauna, but the width of buffer that would be effective requires further study.

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