

Impacts of timber harvest intensity on invertebrate biodiversity

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Summary

The intensity of timber harvest is believed to have significant impacts on biodiversity values and the overall health of forest habitats. The extent and importance of these impacts is an emotive issue and objective reporting of effects of timber harvesting and silviculture on biodiversity is a significant challenge to forest managers. Rapid assessment of invertebrate biodiversity may provide a solution to this need. Pitfall trap samples of litter invertebrates from pastures, unlogged forest compartments, forest compartments previously harvested at 50% of usable timber and forest compartments with standard harvest conditions, involving the removal of 70–90% of all stems, were compared. There were significant differences in invertebrate abundance, richness and composition between forest and pasture habitats, but no detectable differences between forest treatment categories. Strong site effects on abundance, richness and composition were apparent. Pastures and each forest treatment category contributed unique morpho-species to the sample set. More than 20 y of post-harvest recovery has allowed invertebrate assemblage composition to converge to the extent that harvest effects are not readily detectable within this component of biodiversity. The value of invertebrate assessments as a solution for objective reporting of biodiversity value is discussed.

Keywords: biodiversity; assessment; surveys; logging effects; invertebrates; ants

Introduction

Forests NSW (formerly State Forests of NSW) is a state trading enterprise that produces around 3×10^6 m³ of sawlogs and pulpwood annually. Most of this production comes from a publicly-owned pine and hardwood plantation estate worth an estimated A\$1.5 billion in 2003 (SFNSW 2004). Prudent environmental management for the benefit of biodiversity and maintenance of healthy and sustainable habitats is part of the agency public charter. A commitment has been made to support management practices that can sustain healthy forest habitats. This makes sound business sense. Healthy forests will produce timber more efficiently and are more likely to have and sustain high conservation value. These are outcomes that contribute directly to the triple bottom line of economic, social and environmental returns.

Forests NSW has an obligation to monitor and report on the status of the environment in its estate, as do many government agencies

in Australia. Unlike most other agencies, Forests NSW, as a government trading enterprise, is also required to deliver specific products (wood, water, biodiversity, recreation) and an annual dividend to its 'stockholders', the people of New South Wales (NSW). Reporting is prescribed for Forests NSW native forest harvest activities in the regulations, terms and licence conditions contained within the *Forestry and National Parks Estate Act 1998*. At present, these obligations are met to a minimum degree by monitoring the status of vertebrates and vascular plants (SFNSW 2004). This approach has been applied extensively in the past but has been expensive in cash and human resources (SFNSW 2004). Although compliance may require a measure of impact on endangered species, resource managers are aware of the ecological need for managing all wildlife, and for fine-tuning silviculture, rotation and harvesting practices to improve the quality and reliability of the timber yield. They know that this requires information on how the local ecology will affect timber growth and quality, as a result of current, past and future practices.

A challenge faced by commercial forestry enterprises, including Forests NSW, is to find which of the many management options for harvest, rotation, silviculture and overall forest management will sustain production and achieve local conservation outcomes. This requires a capability to measure the environmental performance of management interventions, especially the intensity of the timber harvest. Pressure is also exerted on forest managers from a wide range of stakeholders, who have views that range from deep ecology and environmentalism, through traditional rural farming values to the commercial imperatives of timber production. Information is needed to underpin a holistic approach to forest management. However, this comes at a time when resources and expertise to gather usable information remain a constraint and when information needs may conflict.

One solution is to go beyond the traditional assessments of vertebrates and vegetation and measure invertebrates to generate data on biodiversity values. Invertebrates are sensitive to both extensive and subtle environmental change (Oliver *et al.* 2000a; Werner and Raffa 2000), as well as being diverse, abundant and easy to sample. Invertebrates make up more than 90% of all non-microbial species (Pimentel *et al.* 1992); consequently studies of invertebrates are more likely to reflect true biodiversity patterns than traditional measures of vertebrates and vegetation. They follow similar patterns to vertebrates, but on smaller scales of space and time (Disney 1986; Rosenberg *et al.* 1986). It is also

possible to detect these changes at taxonomic ranks higher than species, which makes identification easier and more cost-effective (Pik *et al.* 1999).

Invertebrates provide an excellent surrogate for biodiversity patterns and there is an extensive list of reasons and justifications for their use in assessment and monitoring programs (Paoletti 1999). They are highly diverse — 85% of animal species in coniferous forests of the Pacific North-West USA are arthropods (Asquith *et al.* 1990) and they have a high biomass. In the Amazon, ant biomass alone is four times greater than that of all vertebrate species combined (Fittkau and Klinge 1973). They contribute directly to system processes such as pollination (Westman 1990), decomposition (Moldenke and Lattin 1990), soil fertility (Lavelle *et al.* 1994) and can even be ecosystem engineers, altering resource flows to change landscapes (Dangerfield *et al.* 1998). This means that while they can provide an index of overall patterns in biodiversity, they are also the organisms most likely to affect key system properties and especially those that relate to system health and productivity (Werner and Raffa 2000).

Traditionally, surveys involving diverse groups such as invertebrates faced significant financial and time constraints, mainly as a result of difficulties with identification (New 1996). The use of morpho-species (Beattie and Oliver 1994), taxonomic sufficiency (Pik *et al.* 1999) and some information technology solutions (Oliver *et al.* 2000b) has overcome this key constraint. Morphospecies are taxa readily separated by morphological differences obvious to individuals with less taxonomic training than specialist taxonomists (Oliver and Beattie 1993). Counts of morphospecies and formally identified species for several of the arthropod taxa collected in pitfall traps have been found to show a high degree of correlation (Oliver and Beattie 1996).

In April 2003 a project was undertaken to describe differences in the patterns of invertebrate biodiversity between forest sites classified by their level of harvest impact. Pitfall trap samples of selected invertebrate groups were taken on Forests NSW Long-Term Ecological Research sites (Kavanagh 1987). Comparisons of invertebrate biodiversity and assemblage composition were made between unlogged, 50% logged and standard logged forest compartments with cleared sites managed for grazing. Forest sites harvested in the Southern Hemisphere summer of 1983–1984 were chosen. These sites have remained undisturbed by harvest operations since then. We adopted these rapid biodiversity assessment approaches (Oliver and Beattie 1993) and used the Biotrack® system (Oliver *et al.* 2000b; Pik *et al.* 2002a) to rapidly sort specimens to species level units, complete analyses and make interpretations for diverse, yet poorly known, groups of invertebrates. The key objective was to determine if invertebrate biodiversity differences could be detected between sites grouped into forest treatment categories 20 y after the impact.

The objective of this paper is to discuss the capacity of invertebrates to distinguish difference between sites, timber harvest category and land use; and to assess the potential value of such information for compliance reporting, demonstration of biodiversity value and adaptive management outcomes.

Methods

The study was conducted in hilly country forming part of the Great Dividing Range in eastern Australia, about 20 km south-east of Bombala, south-eastern NSW. Forests in this area are interspersed as patches in a mixed agricultural landscape, are characterised by six main tree species (*Eucalyptus cypellocarpa*, *E. fastigata*, *E. obliqua*, *E. ovata*, *E. radiata* and *E. viminalis*) and occur in at least seven forest types that have been described elsewhere (e.g. Webb 1991; Goldingay and Kavanagh 1993). Mean annual rainfall is 1167 mm and air temperatures range from a minimum of 0°C in winter to a maximum of 24°C in summer (SFNSW 1994). A series of forest compartments, centered on the upper Waratah Creek catchment of the South-East Forests National Park (formerly Coolangubra State Forest), and pastures were surveyed.

Sites were selected from an existing network of environmental impact assessment sites established by Forests NSW in the early 1980s. They provided random and replicated samples of the environment to enable scientific evaluation of logging impacts on flora and fauna. At Eden, these sites have been monitored regularly for birds, mammals, reptiles and amphibians since their establishment (Shields 1990; SFNSW 2004). Six replicate plots of unlogged, five plots of 50% logged and four plots of standard logged areas were sampled. Replicate sites within a treatment category were 2–20 km apart. Standard logging at the time of harvest included the removal of all merchantable stems as either sawlogs or pulp. One seed tree per 15 ha, trees in riparian buffers of 40 m, consolidated stands of good quality stems (between 20 cm and 40 cm diameter at breast height) for growing stock, and trees in inaccessible steep, rocky or swamp areas were not harvested. Under this regime, between 70% and 90% of the available stems were harvested. In addition to the forest compartment, four pasture sites, which carried similar forest types prior to clearing, were also sampled. The study plots at these sites were 500 m × 100 m and generally accessible by road.

Undisturbed forests in the area are tall (> 30 m canopy height) and have multiple (up to four) vegetation layers (SFNSW 1994). The dominant trees are eucalypts (typically including *E. viminalis*, *E. fastigata* and *E. cypellocarpa*), with *Acacia* spp. in the subcanopy; a variety of woody plants in the shrub layer (e.g. *Grevillea* spp., *Banksia* spp.), and ferns and grasses dominate the ground layer (SFNSW 1994). There is a thick layer of forest litter, and the sites generally conform to the definitions of ‘wet sclerophyll forest’ and ‘tall closed forest’ (Baur 1989). Standard logging procedures at the time removed all merchantable stems (for pulp and sawlogs) in the logging area, leaving mature regrowth (‘growers’), streamside filter strips, and ‘seed trees’ at the rate of one per 15 ha (SFNSW 1994). After logging, these sites were generally open, with patches of regrowth and occasional mature trees. The 50% logging treatment aimed to leave one half of the merchantable stems (pulp and sawlog) in the net logging area. After logging, this area resembled woodland or tall open forest. The pasture sites were cleared in the late 1800s or early 1900s (SFNSW 1994).

Ground-active invertebrates were sampled once in the autumn using pitfall traps. Each trap was a 6.5 cm diameter × 10 cm deep plastic container buried flush with the soil surface and contained 200 ml of preservative solution (50% ethanol and 50% ethylene glycol). Each pitfall trap was given a unique collection barcode before installation, with an additional, identical, barcode placed on a field data sheet to record the location of the sample. The field data sheets were used to prepare and customise the relational database Biota® (Colwell 1996) for entry of specimen data.

At each site, a transect of nine pitfall traps were set 50 m apart along existing vertebrate survey lines. Where possible, traps were placed 5 m from existing bird survey points. All sites were visited and traps installed over a 48-hour period in April 2003. After one week, all pitfall traps were collected and 175 pitfall samples processed in the laboratory.

Contents of each pitfall were flushed with tap water through a 0.4 mm sieve to remove silt and other fine particles. Invertebrates and debris were transferred to a Petri dish. With the aid of a standard dissecting microscope, all invertebrate specimens were picked, identified into higher taxa groupings and placed in separate vials with 75% ethanol. In this case, higher taxa refers to ordinal level taxonomic rank for the majority of groups but also includes some splitting (e.g. Hymenoptera into ants and other Hymenoptera) and the use of non-taxonomic categories such as larvae.

Initially, all specimens from a random selection of 42 samples (18% of the total) representing all sites were identified to higher taxa and their abundance recorded. From analysis of rank-abundance curves, compositional difference through ordination and a consideration of biological characteristics, six taxa from the sample set were selected: spiders (Araneae), beetles (Coleoptera), flies (Diptera), ants (Formicidae), true bugs (Hemiptera) and grasshoppers (Orthoptera). Beetles, ants and flies were both abundant and well represented in the majority of samples; spiders provided additional indicator values as a result of their trophic position; beetles and ants were already known as good indicators of habitat condition (Oliver *et al.* 2000a; Andersen and Majer 2004). Recording only the abundance of these taxa in the remaining samples enhanced the speed and efficiency of sorting.

Standard keys were used to identify flies, beetles and adult spiders to family, while ants were identified to genus following Shattuck (1999). These taxa were further identified to morphospecies using Biotrack® protocols. Morphospecies were used as a surrogate for formal species-level classification, and have been shown to be an effective measure of species level information in aquatic and terrestrial habitats (Oliver and Beattie 1993; Pik *et al.* 1999).

The relational database software, Biotrack Images (www.biotrackaustralia.com.au), was used to assist in the identification of specimens to morphospecies (Oliver *et al.* 2000b; Pik *et al.* 2002a). Initially a subset of specimens from a family or genus was grouped into morphospecies using morphological characters relevant to species-level taxonomy by an experienced biodiversity technician with knowledge of traits specific to the group. Specimens were placed in a Petri dish under a high-power microscope and split into morphospecies on the basis of selected characters with observable character states. Individual morphospecies were cross-referenced against one another to determine and assign a morphospecies number consistent across the entire

project. Once an individual designated as the voucher specimen was assigned as a morphospecies, high quality digital images were taken of it. The digital images of morphospecies, including the whole specimen and relevant characters, together with character state information were entered into the relational database. Unknown specimens were identified by entering character state information as a query to the database, which delivered images of likely matches to a computer screen. The operator matched by eye these images to the specimen under the microscope.

Comparisons between average abundance and richness of higher taxa and all morphospecies were made using analysis of variance routines, with post-hoc multiple comparisons, in Statistica® (StatSoft 1997). Differences in higher taxa and morphospecies composition between samples grouped by management categories were made using the numerical classification package PRIMER® (Clarke and Warwick 1994). A standard multi-dimensional scaling (MDS) ordination, based on the Bray–Curtis measure of similarity, was used for simplicity and general applicability. MDS also makes few assumptions regarding the form of the data and the inter-relationships of the samples and efficiently preserves distance relationships in low-dimensional ordination space (Clarke and Warwick 1994). The analysis of similarities randomisation tests (ANOSIM), based on the Bray–Curtis measure of association, were used to determine the significance of assemblage differences between management categories. ANOSIM is a non-parametric permutation procedure that makes minimal assumptions about the normality of the data and uses the assemblage relationships between samples as summarised in the ranks of the biotic similarity matrix (Clarke and Gorley 2001).

To avoid possible confounding by replicate effects, the examination of biodiversity difference as a result of land management was conducted using a two-way nested ANOSIM. The two-way ANOSIM allows the effect of replicates to be considered before the main effect of category is analysed. Sites are grouped into categories, with replicates nested within each category. The procedure tests two hypotheses. Firstly, H_1 postulates that there may be a general difference between the location of each category, but within each category there is no pattern in the allocation of replicates to sites. One-way ANOSIM results are computed for each replicate site within each category. R values, calculated from the differences between the average rank similarities arising from all pairs of replicates between different sites and the average of all rank similarities within sites, are averaged and compared with R values from all possible permutations of sample labels. Secondly, if the significance test pools the replicates within a site, H_2 postulates that there is no difference between categories, and tests using six ‘new’ replicates formed from the combining of information from the original replicates (Clarke and Warwick 1994).

Results

A total of 14 145 terrestrial invertebrates was processed from 175 pitfall trap samples. These specimens included representatives of 29 higher taxa, dominated numerically by beetles, ants, flies and spiders which accounted for 65% of the total abundance. On average, pitfall traps on forest sites captured more than double the number of invertebrates than traps in grazed paddocks, mostly because beetles and flies were more abundant in forests (Table 1a).

Abundance of ants and spiders was similar between forest sites and paddocks.

There were some significant differences in average abundance for all invertebrates and individual taxa between harvest categories (Table 1a). However, average ant and fly abundances did not differ between forest harvest categories. Average abundance of all invertebrates in pitfalls was slightly elevated in 50% logged sites (Table 1a).

A total of 207 morphospecies (49 ant, 67 beetle, 64 fly and 27 spider) were identified from 4601 specimens from selected families. The ten most abundant morphospecies (four beetles, four ants and two flies) made up 61% of the total abundance. More than half of the morphospecies identified (125) were represented by fewer than five specimens. This is typical of local species-abundance patterns for invertebrates, where most of the taxa are infrequent in samples and a few taxa make up the bulk of the abundance.

Average morphospecies richness in the forest sites was 50% higher than the paddocks, mainly because of greater richness of beetles and flies (Table 1b). Average ant and spider richness did not differ significantly between forests and paddocks or between forest categories. There were also no significant differences in species richness for beetles and flies between forest categories. Replicate effects were significant for ants, beetles and total richness (Table 1b).

Richness is a relatively crude measure of diversity as it ignores compositional differences and relative abundance. Non-parametric multi-dimensional scaling ordinations, based on Bray–Curtis measure of similarity in composition between samples, showed distinct grouping of samples when characterised as either forest or pastures for both higher taxa and combined morphospecies (Fig. 1). There was slightly greater overlap in the groupings of samples for higher taxa than for morphospecies, where there was

Table 1. Average abundance (\pm 1 s.d.) and average richness for invertebrates caught in 204 pitfall trap samples from separate sites (nine samples per site) from four treatment categories (unlogged, 50% logged, standard logged and pasture). ANOVA results are on $\log(N+1)$ -transformed data. Values with the same letter superscript are not significantly different, when tested by a Student–Neuman–Kuel (SNK) *post hoc* test of pairwise comparisons between means.

Taxa	Forest categories			Cleared (pasture) (5 sites)	ANOVA (<i>F</i> values)	
	50% logged (5 sites)	Standard logged (4 sites)	Unlogged (6 sites)		Category effect	Replicate effect
(a) Abundance						
Ants	13.3 \pm 15.5 ^a	6.6 \pm 8.1 ^a	8.8 \pm 15.2 ^a	10.1 \pm 10.3 ^a	1.11 ^{NS}	2.01*
Beetles	20.4 \pm 13.9 ^b	19.3 \pm 14.5 ^{bc}	17.1 \pm 11.2 ^c	5.1 \pm 4.2 ^a	11.9***	2.5 ^{NS}
Flies	10.6 \pm 7.1 ^b	12.5 \pm 15.6 ^b	11.3 \pm 6.9 ^b	4.3 \pm 4.8 ^a	3.88**	0.34 ^{NS}
Spiders	2.6 \pm 1.9 ^b	2.0 \pm 1.5 ^c	2.4 \pm 2.4 ^{bc}	1.9 \pm 2.2 ^c	2.87*	1.46 ^{NS}
Total	81.9 \pm 56.1 ^{bc}	45.4 \pm 28.4 ^c	68.6 \pm 45.5 ^{bc}	25.6 \pm 16.0 ^a	9.44***	1.83 ^{NS}
(b) Richness						
Ants	2.9 \pm 1.8 ^a	2.1 \pm 1.6 ^a	2.8 \pm 1.7 ^a	2.5 \pm 2.2 ^a	0.51 ^{NS}	5.27***
Beetles	4.5 \pm 2.4 ^b	4.4 \pm 2.3 ^b	4.3 \pm 2.6 ^b	1.3 \pm 1.1 ^a	15.2***	5.1***
Flies	4.1 \pm 1.7 ^b	4.1 \pm 1.6 ^b	4.3 \pm 2.0 ^b	1.9 \pm 1.5 ^a	13.9***	2.4 ^{NS}
Spiders	1.8 \pm 1.0 ^a	1.5 \pm 1.2 ^a	1.7 \pm 1.4 ^a	1.2 \pm 0.9 ^a	2.9 ^{NS}	0.7 ^{NS}
Total	15.3 \pm 4.4 ^b	14.4 \pm 4.8 ^b	16.1 \pm 5.1 ^b	9.5 \pm 2.7 ^a	14.5***	3.8***

*** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$; ^{NS} $P > 0.05$

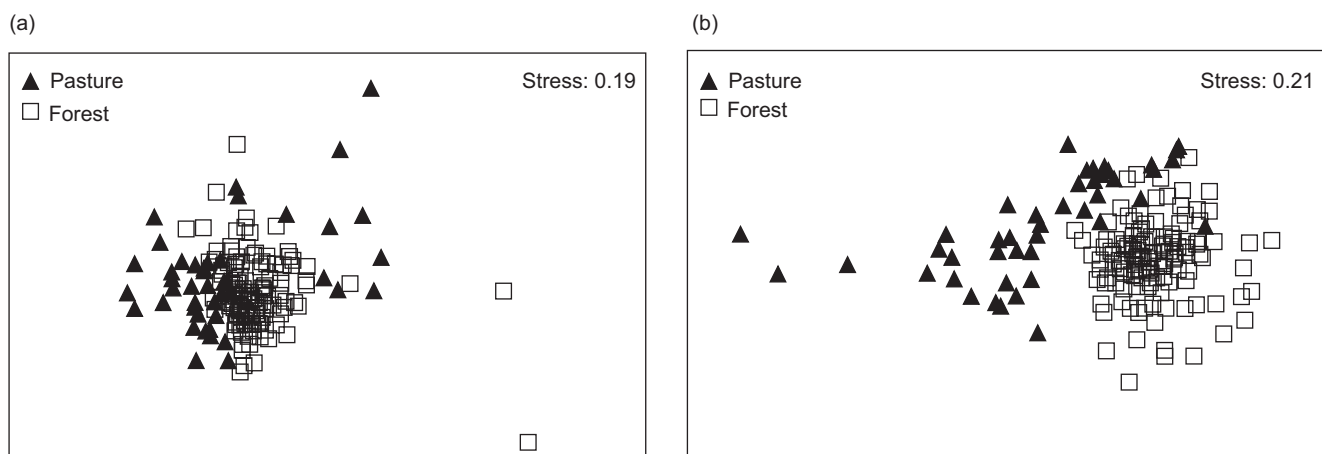


Figure 1. Multi-dimensional scaling plots of Bray–Curtis similarity measures for (a) higher taxa and (b) morphospecies composition in pitfall trap samples from forest and pasture sites in the Bombala region of NSW. Data are for abundance with a $\log(N+1)$ transformation.

near complete separation. Distinct groups of samples, classified as either forest or pasture habitat, were highly significant for higher taxa, combined morphospecies composition and for individual taxa of morphospecies (Table 2).

In contrast to this strong effect of land use on composition there was no observable difference in composition between samples from within-forest sites when grouped by harvest treatment either for higher taxa or morphospecies (Fig. 2, Table 2). There were also no significant differences in composition between pairwise comparisons of forest harvest categories, except for higher taxa composition between 50% and standard logged categories (Table 2).

Exploration of possible replicate effects with a two-way nested ANOSIM showed that both within forest and pasture, and also within the three forest treatment categories (50% logged, standard logged and unlogged), previously significant effects of treatment were confounded by strong site effects (Table 3). Therefore, there was no general difference in the location of each category, but a strong pattern in the allocation of samples to sites. Within a forest treatment category, samples from any given site were more closely grouped than would be expected by chance (strong within-forest replicate effect, Table 3).

When all treatment categories (pasture, unlogged, 50% logged and standard logged) were considered there was also no effect of category but there were strong site effects (Table 3). Only for higher taxa was there a significant category effect resulting from differences in composition between pastures and forest treatment categories that persisted despite strong site effects. Composition of morphospecies from pitfall traps appeared to be strongly site specific.

Finally, a one-way ANOSIM analysis on composition within a category was completed to explore the extent of site effects without comparisons between categories. In these analyses, two data transformations were included to allow a more explicit focus on differences in composition (presence/absence), relative abundance (untransformed) and moderately abundant taxa ($\log(N+1)$). Almost all analyses were significant, except in the standard logged category, where around half the analyses were significant (Table 4). This suggests that the site effects were a result of both composition and relative abundance changes.

Samples from grazed paddocks produced 22 morphospecies (12% of the total) that were not sampled in the forest sites. Seven of these morphospecies were sampled from more than one site, and three of them were found on three of the four pasture sites.

Table 2. Global R values from one-way analysis of similarities for higher taxa and morphospecies composition between all forest and pasture sites, and within forest treatment categories; and pairwise R values between forest treatment categories. Results are based on Bray–Curtis similarities on $\log(N+1)$ -transformed data.

Taxa	Global R		Pairwise R values		
	Forest and pasture	Within forests	50% logged and unlogged	Standard logged and unlogged	Standard logged and 50% logged
Higher taxa	0.429***	0.016 ^{NS}	0.006 ^{NS}	-0.005 ^{NS}	0.059*
Ants	0.472***	0.01 ^{NS}	0.016 ^{NS}	0.012 ^{NS}	0.002 ^{NS}
Beetles	0.615***	0.041 ^{NS}	0.015 ^{NS}	0.037 ^{NS}	0.091 ^{NS}
Spiders	0.17**	0.015 ^{NS}	0.019 ^{NS}	0.023 ^{NS}	-0.001 ^{NS}
Flies	0.057**	0.014 ^{NS}	0.0 ^{NS}	0.031 ^{NS}	0.014 ^{NS}
All morphospecies	0.623***	0.028 ^{NS}	0.008 ^{NS}	0.035 ^{NS}	0.057 ^{NS}

*** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$; ^{NS} $P > 0.05$

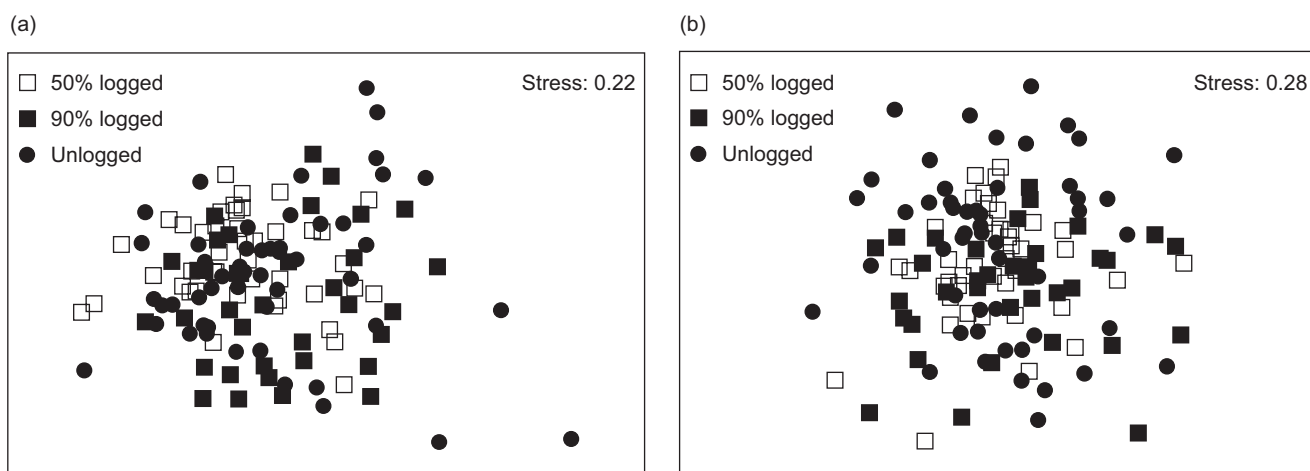


Figure 2. Multi-dimensional scaling plots of Bray–Curtis similarity measures for (a) higher taxa and (b) morphospecies composition in pitfall trap samples from forest sites that were unlogged, 50% logged and standard logged in the early 1980s. Data are for abundance with a $\log(N+1)$ transformation.

Table 3. Global R values from two-way analysis of similarities for higher taxa and morphospecies composition between all forest and pasture sites, and within forest treatment categories; and pairwise R values between forest treatment categories. Results are based on Bray–Curtis similarities on $\log(N+1)$ -transformed data.

Taxa	Global R				Pairwise R values		
	Forest and pasture		Within forests		Between treatments		
	Category	Replicate	Category	Replicate	50% logged and unlogged	Standard logged and unlogged	Standard logged and 50% logged
Higher taxa	-0.003 ^{NS}	0.227***	-0.036 ^{NS}	0.159***	-0.059 ^{NS}	-0.083 ^{NS}	0.069 ^{NS}
Ants	0.131 ^{NS}	0.333***	-0.516 ^{NS}	0.26***	-0.456 ^{NS}	-0.46 ^{NS}	-0.738 ^{NS}
Beetles	-0.112 ^{NS}	0.244***	-0.487 ^{NS}	0.23***	-0.453 ^{NS}	-0.488 ^{NS}	-0.581 ^{NS}
Spiders	-0.315 ^{NS}	0.019 ^{NS}	-0.383 ^{NS}	0.009 ^{NS}	-0.232 ^{NS}	-0.284 ^{NS}	-0.675 ^{NS}
Flies	-0.205 ^{NS}	0.161***	-0.401 ^{NS}	0.115***	-0.333 ^{NS}	0.353 ^{NS}	-0.606 ^{NS}
All morphospecies	-0.116 ^{NS}	0.395***	-0.481 ^{NS}	0.307***	0.432 ^{NS}	-0.508 ^{NS}	-0.588 ^{NS}

*** $P < 0.001$; ** $P < 0.01$; * $P < 0.05$; ^{NS} $P > 0.05$

Table 4. Global R values from one-way analysis of similarities procedures for comparing biological composition between replicates within pasture and forest treatment categories

		Pasture	Forest		
			Unlogged	50% logged	Standard logged
Binary	Higher taxa	0.226***	0.15**	0.026 ^{NS}	0.019 ^{NS}
	Ants	0.315**	0.349***	0.278***	0.09*
	Beetles	0.245**	0.254***	0.176***	0.045 ^{NS}
	Spiders	0.009 ^{NS}	0.069 ^{NS}	0.1*	-0.008 ^{NS}
	Flies	0.263*	0.092*	0.05 ^{NS}	0.074*
	All morphospecies	0.564***	0.33***	0.354***	0.069 ^{NS}
	Log ($N+1$)	Higher taxa	0.449***	0.224***	0.155***
Ants		0.292**	0.354***	0.299***	0.131*
Beetles		0.264**	0.264***	0.137**	0.076*
Spiders		0.02 ^{NS}	0.097*	0.107*	-0.01 ^{NS}
Flies		0.261*	0.146**	0.025 ^{NS}	0.05 ^{NS}
All morphospecies		0.603***	0.429***	0.346***	0.066 ^{NS}
Untransformed		Higher taxa	0.429***	0.205***	0.122***
	Ants	0.242***	0.311***	0.292***	0.083*
	Beetles	0.273**	0.24***	0.083*	0.077*
	Spiders	0.029 ^{NS}	0.105*	0.105*	-0.012 ^{NS}
	Flies	0.223*	0.178**	0.019 ^{NS}	0.033 ^{NS}
	All morphospecies	0.586***	0.44***	0.311***	0.06 ^{NS}

*** $P < 0.001$, ** $P < 0.01$, * $P < 0.05$, ^{NS} $P > 0.05$.

Samples from forest sites generated 125 morphospecies (68%) not found on paddocks and there were 38 morphospecies (21%) that were common to both habitats (Fig. 3). Further assigning the forest morphospecies by treatment category produced 29 morphospecies (18% of the total from forests) that were found only in the unlogged sites, along with 23 (14%) and 18 (11%) morphospecies that were unique to the 50% logged and standard logged sites respectively (Fig. 3). Only two of these morphospecies (3%) occurred on more than one site within a category. In the confines of this particular sample set over 40% of morphospecies sampled from forests were unique to a site rather than a forest treatment category.

A slightly smaller proportion of ant (33%) and beetle (32%) than spider (42%) and fly (64%) morphospecies was found only in one of the three management categories. The largest single

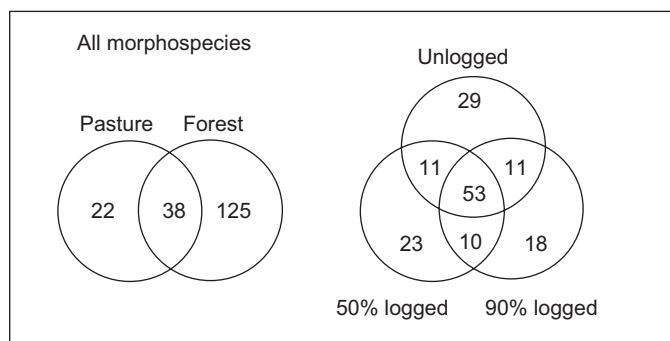


Figure 3. Venn diagram of the number of sampled morphospecies shared and unique to pasture and forest habitats, and the number of sampled morphospecies shared and unique to each forest treatment category

percentage was for flies in unlogged sites (29%). There were 15 ant, 17 beetle, 11 fly and 10 spider morphospecies that occurred in at least one site from each of the forest treatment categories. This is collectively around one-third of the total found in forests and, on average, each of these species was present on 6.3 sites (42% of all sites). Only one ant, three beetle and one fly morphospecies were sampled from all 15 forest sites. This suggests that even the widespread morphospecies were sampled only once in every two or three sites.

Discussion

Sampling of ground-active invertebrates from the Forests NSW Long Term Ecological Research sites in the Bombala region produced more than 14 000 specimens from 175 pitfall traps. Fewer than 60 specimens per trap is a low return for this sampling method in NSW, given that previous studies have produced averages of 114 individuals per trap in regenerating secondary woodland (Schnell *et al.* 2003), 330 in cypress forests (Dangerfield and Pik 2001) and 425 in urban bushland remnants (Pik *et al.* 2002b). Catches from the forest sites were up to three times greater than from sites under pasture management, a result consistent with previous studies (Vasconcelos *et al.* 2000). Average richness of morphospecies in forest sites was around 50% higher than in the pasture sites, which would be expected given the greater numbers of specimens. These patterns in richness and abundance were reflected in composition, which differed between forests and pasture sites at higher taxa, combined morphospecies and individual taxa morphospecies levels. The average Bray–Curtis similarity between forest and pasture sites was 9%, suggesting that there was little overlap in composition, compared with 16% similarity between samples from within pasture sites and 24% from samples within forest sites.

Major structural changes to vegetation occur when forests are cleared for grazing, with inevitable consequences for biodiversity. The focus on the biodiversity change is often on the vegetation and vertebrate components, to the extent that invertebrates are not often recorded. One consequence of the biodiversity difference between forest and pasture sites is that pasture sites contributed 12% of the 185 morphospecies recorded in all samples and a further 21% that were shared with the forest sites. In this study, clearing occurred many decades prior to sampling, so the comparison was between established habitat types that were both making a contribution to the overall invertebrate biodiversity across the landscape.

Despite the range of forest treatment, extending from undisturbed controls to 90% of harvestable timber removed at a site, any differences in invertebrate abundance, richness and composition between forest treatment categories that might exist were hard to detect. There were no significant differences in average morphospecies richness per trap, collectively or by individual taxon. Average abundance was slightly depressed in standard logged sites but there was no detectable difference in invertebrate composition between forest treatment categories. The average Bray–Curtis similarity estimates between samples from within unlogged, 50% logged and standard logged categories, at 21%, 30% and 24% respectively, were not different to the average similarity between categories of 24 (unlogged and 50% logged),

21 (unlogged, standard logged) and 26% (50% logged and standard logged). The chances of a single sample from a site being similar in composition to another sample from any site within the same category, was no different to that for a sample from a site in any other forest treatment category.

Each site within a category had a detectable signature of invertebrate composition. Samples within a site were more similar in composition than those from other sites within the sample forest treatment category (Table 4). This result can be interpreted as a confounding effect that masks the effects of treatment because sites within a category are not true replicates. This is certainly a weakness of this type of *post-hoc* inference or of quasi-experimental studies (Bennett and Adams 2004). Difference, or similarity, between categories can be as much a result of the selection of sites as it is the consequence of management practice. The likelihood is that no amount of site selection stratification can overcome the problems of type II error, where we continue to accept the hypothesis of ‘no difference’ when it is, in fact, false. The only short-term solution to this inevitable impasse is to repeat this comparative approach on similar combinations of sites around NSW and conclude from a meta-analysis. Far-sighted managers may also wish to install monitoring programs as harvests happen to help future generations understand and fine tune harvest practices. These can easily be made to follow Before–After Control–Impact designs (Underwood 1994).

An alternative view is to consider the apparent site-specific assemblage composition as an opportunity. The results presented here show that it is possible to provide a reliable biodiversity signature of a forest compartment or a grazing paddock using a subset of invertebrate morphospecies from a single sample of nine pitfall traps separated at 50 m intervals along a transect. This sampling effort will generate 350 to 700 specimens from up to 15 higher taxa. A selection of ants, beetle families and fly families sorted to morphospecies would generate an average of 11 morphospecies per trap and up to 40 morphospecies for a forest compartment, or six per trap and around 25 in total for a paddock. These 40 morphospecies are likely to be half to two thirds of the total present in the habitat that can be sampled with a pitfall trap method and are sufficient to provide biodiversity signatures that can be compared between compartment, management tactics and land uses. The opportunity is to consider that the tests of assemblage difference are sensitive enough and that, after 20 y, the effects of forest harvesting practices no longer override site-specific effects.

Ants have an established history as biological indicators, especially of disturbance (Majer 1983; Andersen and Majer 2004), and have been used extensively in previous surveys in NSW (Pik *et al.* 2002b; Schnell *et al.* 2003). Several ant genera enter, or increase their abundance in, disturbed sites and there is a consistent succession of species as these sites recover (Andersen and Majer 2004). Generally, ant species richness is expected to decline by up to 60% on conversion of forest to pasture (Vasconcelos *et al.* 2000). In the present study, ant richness averaged around 2.5 morphospecies per trap in both forested and pasture sites and there were no detectable compositional differences. As with the combined taxa, there was a very strong replicate effect on ant diversity in all treatment categories. That is, the local site histories

and specific habitat properties, such as fire, soils and moisture, had greater influence in defining the sampled ant assemblage than the effects of forest treatment.

Beetles are known to be sensitive to logging practices (Oliver *et al.* 2000a; Werner and Raffa 2000). Their sensitivity and huge abundance and diversity make them ideal to help monitor forestry management impacts. In northern hardwood forests in the USA, forest management affects more beetle species than do tree species dominance or canopy structure; but high variability exists among sites and among traps within sites (Werner and Raffa 2000). In New England blackbutt forest and messmate/forest ribbon gum forest, fire history and correlated habitat variables were important explanatory variables for beetle richness and composition when logging intensity was low (Oliver *et al.* 2000a). Local habitat effects, such as drainage and aspect, produce variation in biodiversity signatures of beetles.

There was no significant effect of logging on average beetle richness in the Bombala region and no effect on composition of beetle morphospecies. Twenty years after logging the effects of habitat variation are stronger than the original impact of logging. Local beetle communities may define a forest compartment, but not a forest treatment category.

In the Bombala region, the composition and abundance of flies responded strongly to local habitat conditions, perhaps because habitat variables are a consequence of site profile and history. A larger number of fly families were sorted to morphospecies than for beetles, so this may also have contributed to greater species turnover. Silviculture practices can affect habitat parameters such as moisture, but site management itself has often been shown not to be the primary cause of biodiversity difference in flies. For example, the abundance and composition of fly species in Scottish birch and oak woodlands was greater than in pine plantations, but there were also strong differences between sites that were correlated with a number of habitat factors related to soil, moisture and habitat structure (Woodcock *et al.* 2003).

The general conclusion from theory and field data is that a variety of management regimes will enhance biodiversity at landscape scales (Werner and Raffa 2000). Much of this is due to an increase in the types and incidence of habitats in the landscape and a disturbance mosaic that maintains the presence of early successional species. At a more local scale, forest management can enhance species richness and change composition by creating disturbance and more open patches within closed forests (Hamer *et al.* 2003). The evidence from the Bombala region supports this pattern, with 12%, 15% and 19% of morphospecies unique to standard logged, 50% logged and unlogged sites respectively. Within this particular data set, logging treatments added 41 morphospecies not found in unlogged or pasture sites, that is 22% of all morphospecies.

The results from the current survey showed strong replicate effects on biodiversity difference, with no significant effects of forest treatment. The likely explanation for the weak response to treatment is the convergence of composition over 20 y of recovery, and the heterogeneity in specific site histories. The pasture study sites had been cleared decades earlier and the forest sites had not been logged or otherwise disturbed by management for 20 y. The cleared sites, then, were unlikely to have disturbance specialists

present. It has consistently been shown that 20 y is a long enough period for habitats to stabilise to background levels of invertebrate diversity (Wong 1986; Nummelin 1998).

The logical extension to this survey method is to use a range of sites logged more recently and with a range of logging treatments, especially if sites can be surveyed both before and after harvest. This would provide a very robust measure of both the initial impact and the time taken for sites to recover from management interventions.

Based on this sampling effort, and using Biotrack staff and technologies, an initial survey including image capture would cost \$3500 plus fieldwork expenses per site at 2004 prices. Subsequent samples of the same site would cost around \$2250 plus fieldwork expenses. Cost of fieldwork varies according to site location and the number of sites to be included in a survey, unit costs decreasing as the number of sites is increased but is around \$500 per site. An accurate initial biodiversity signature can be obtained for \$4000 and monitoring information for \$2750 per compartment. This converts to site data for as little as \$69 per morphospecies and \$3 per individual. This compares favourably with costs per species in vertebrate surveys and complements traditional surveys with data generated at a scale more relevant to forest management.

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References

- Andersen, A.N. and Majer, J.D. (2004) Ants show the way down under: invertebrates as bioindicators in land management. *Frontiers in Ecology and Environment* **2**, 291–298.
- Asquith, A., Lattin, J.D. and Moldenke, A.R. (1990) Arthropods: the invisible diversity. *Northwest Environmental Journal* **6**, 404–412.
- Baur, G.N. (1989) *Forest Types in New South Wales*. Research Note No. 17, Forestry Commission of New South Wales.
- Beattie, A.J. and Oliver, I. (1994) Taxonomic minimalism. *Trends in Ecology and Evolution* **9**, 448–490.
- Bennett, L.T. and Adams, M.A. (2004) Assessment of ecological effects due to forest harvesting: approaches and statistical issues. *Journal of Applied Ecology* **41**, 585–598.
- Clarke, K.R. and Gorley, R.N. (2001) *PRIMER v5: User Manual/Tutorial*. Primer-E Ltd, Plymouth.
- Clarke, K.R. and Warwick, R.M. (1994) *Change in Marine Communities: An Approach to Statistical Analysis and Interpretation*. Hitchings and Mason, Plymouth.
- Colwell, R.K. (1996) *Biota. The Biodiversity Database Manager*. Sinauer Associates, Sunderland, MA, USA.

- Dangerfield, J.M. and Pik, A. (2001) *Biological Width of Ephemeral Streams*. Key Centre for Biodiversity and Bioresources, Macquarie University, Sydney.
- Dangerfield, J.M., McCarthy, T.S. and Ellery, W.N. (1998) The mound-building termite *Macrotermes michaelseni* as an ecosystem engineer. *Journal of Tropical Ecology* **14**, 507–520.
- Disney, R.H.L. (1986) Assessments using invertebrates: posing the problem. In: Usher, M.B. (ed.) *Wildlife Conservation Evaluation*. Chapman and Hall, London, pp. 271–293.
- Fittkau, E.J. and Klinge, H. (1973) On biomass and trophic structure of the Central Amazonian rain forest ecosystem. *Biotropica* **5**, 2–14.
- Goldingay, R.L. and Kavanagh, R.P. (1993) The yellow-bellied glider: a review of its ecology, and management considerations. In: Lunney, D. (ed.) *Conservation of Australia's Forest Fauna*. Royal Zoological Society of New South Wales, Mosman, pp. 365–375.
- Hamer, K.C., Hill, J.K., Benedick, S., Mustafa, N., Sherratt, T.N., Maryati, M. and Chey, V.K. (2003) Ecology of butterflies in natural and selectively logged forests of northern Borneo: the importance of habitat heterogeneity. *Journal of Applied Ecology* **40**, 150–162.
- Kavanagh, R.P. (1987) Floristic and phenological characteristics of a eucalypt forest in relation to its use by arboreal marsupials. MSc thesis, The Australian National University, Canberra.
- Lavelle, P., Dangerfield, J.M., Fragoso, C., Lopez-Hernandez, D., Eschenbrenner, V., Brussard, L. and Pashanasi, B. (1994) The relationship between soil macrofauna and tropical soil fertility. In: Woomer, P.L. and Swift, M.J. (eds) *The Biological Management of Tropical Soil Fertility*. John Wiley and Sons, Chichester, pp. 137–169.
- Majer, J. (1983) Ants: bio-indicators of minesite rehabilitation, land-use, and land conservation. *Environmental Management* **7**, 375–383.
- Moldenke, A. and Lattin, J.D. (1990) Dispersal characteristics of old-growth soil arthropods: the potential for loss of diversity and biological function. *Northwest Environment Journal* **6**, 408–409.
- New, T.R. (1996) Taxonomic focus and quality control in insect surveys for biodiversity conservation. *Australian Journal of Entomology* **35**, 97–106.
- Nummelin, M. (1998) Log-normal distribution of species abundances is not a universal indicator of rain forest disturbance. *Journal of Applied Ecology* **35**, 454–457.
- Oliver, I. and Beattie, A.J. (1993) A possible method for the rapid assessment of biodiversity. *Conservation Biology* **7**, 562–568.
- Oliver, I. and Beattie, A.J. (1996) Invertebrate morpho-species as surrogates for species: a case study. *Conservation Biology* **10**, 99–109.
- Oliver, I., MacNally, R. and York, A. (2000a) Identifying performance indicators of the effects of forest management on ground-active arthropod biodiversity using hierarchical partitioning and partial canonical correspondence analysis. *Forest Ecology and Management* **139**, 21–40.
- Oliver, I., Pik, A., Britton, D., Dangerfield, M.J., Colwell, R.K. and Beattie, A.J. (2000b) Virtual biodiversity assessment systems. *BioScience* **50**, 441–450.
- Paoletti, M.G. (1999) Using bioindicators based on biodiversity to assess landscape sustainability. *Agriculture, Ecosystems and Environment* **74**, 1–18.
- Pik, A.J., Oliver, I. and Beattie, A.J. (1999) Taxonomic sufficiency in ecological studies of terrestrial invertebrates. *Australian Journal of Ecology* **24**, 555–562.
- Pik, A., Beattie, A.J. and Dangerfield, J.M. (2002a) Biotrack simplifies monitoring of invertebrates on restored sites. *Ecological Restoration* **20**, 67–68.
- Pik, A.J., Dangerfield, J.M., Bramble, R.A., Angus, C. and Nipperess, D.A. (2002b) The use of invertebrates to detect small-scale habitat heterogeneity and its application to restoration practices. *Environmental Monitoring and Assessment* **75**, 179–199.
- Pimentel, D., Stachow, U., Takacs, D.A., Brubacker, H.W., Dumas, A.R., Meaney, J.J., O'Neil, J.A.S., Onsi, D.E. and Corzilius, D.B. (1992) Conserving biological diversity in agricultural/forest systems. *BioScience* **42**, 354–362.
- Rosenberg, D.M., Danks, H.V. and Lehmkuhl, D.M. (1986) Importance of insects in environmental impact assessment. *Environmental Management* **10**, 773–783.
- Schnell, M.R., Pik, A. and Dangerfield, J.M. (2003) Ant community succession within eucalypt plantations on used pasture and implications for taxonomic sufficiency in biomonitoring. *Austral Ecology* **28**, 553–565.
- Shattuck, S.O. (1999) *Australian Ants: Their Biology and Identification*. Monographs on Invertebrate Taxonomy, Vol. 3. CSIRO Publishing, Collingwood.
- Shields, J.M. (1990) The effects of logging on bird populations in south-eastern New South Wales. PhD thesis, University of Washington, Seattle.
- SNFSW (1994) *Eden Management Area Environmental Impact Statement*. Forests NSW, Pennant Hills NSW.
- SNFSW (2004) *Social, Environmental and Economic Report 2002/03*. State Forests of NSW, Pennant Hills NSW.
- StatSoft (1997) *Statistica*, Release 5. StatSoft, Tulsa, Oklahoma.
- Underwood, A. (1994) On beyond BACI: sampling designs that might reliably detect environmental disturbances. *Ecological Applications* **4**, 3–15.
- Vasconcelos, H.L., Vilhena, J.M.S. and Caliri, G.J.A. (2000) Responses of ants to selective logging of a central Amazonian forest. *Journal of Applied Ecology* **37**, 508–514.
- Webb, G.A. (1991) The effects of logging on populations of small ground-dwelling vertebrates in montane eucalypt forest in south-eastern New South Wales. MSc thesis, The Australian National University, Canberra.
- Werner, S.M. and Raffa, K.F. (2000) Effects of forest management practices on the diversity of ground-occurring beetles in mixed northern hardwood forests of the Great Lakes Region. *Forest Ecology and Management* **139**, 135–155.
- Westman, W.E. (1990) Managing for biodiversity. *BioScience* **40**, 26–33.
- Wong, M. (1986) Trophic organisation of understorey birds in a Malaysian dipterocarp forest. *Auk* **103**, 100–116.
- Woodcock, B.A., Watt, A.D. and Leather, S.R. (2003) Influence of management type on Diptera communities of coniferous plantations and deciduous woodlands. *Agriculture, Ecosystems and Environment* **95**, 443–452.