

A PRAGMATIC APPROACH TO CHARACTERISING INSECT COMMUNITIES IN NEW ZEALAND: MALAISE TRAPPED BEETLES

Summary: Insect communities from a range of successional vegetation stages on the central North Island volcanic plateau were characterised and compared using Malaise trapped beetle samples. Results were derived from sampling series conducted in a total of ten sites over three separate summers. Divisive classification successfully grouped samples according to four main habitat types despite temporal and spatial separation of samples within these groups. A four-week period in early summer was found to be optimum for sample discrimination according to the main vegetation types. Component species and the functional (trophic) structure of samples showed a relationship with generic habitat types as well as reflecting attributes of the individual sites sampled. The most useful information was derived from the identity of the biological entities rather than simply from numerical relationships. Use of this pragmatic methodology has potential to improve our ecosystem understanding and the sustainable management of the dominant area of terrestrial biodiversity provided sufficient attention is given to standardisation, particularly of taxonomy.

Keywords: biodiversity; insect community characterisation; Malaise trapped Coleoptera; vegetation succession; New Zealand; techniques.

Introduction

The 1991 New Zealand Resource Management Act requires sustainable management of our genetic resources or 'biodiversity'. Although the great majority of these resources are represented by insects (Southwood, 1978; Berry, 1982; Watson *et al.*, 1995), land areas are generally perceived and managed in terms of their vegetation systems. Understanding of the relationship between these two subsets of biotic systems is still in its infancy due in part to our poor knowledge of non-pest insect species. It is estimated that only about half of the over 20,000 species of New Zealand insects are currently described (Watt, 1982; Emberson, 1994). Understanding the habitat relationships of insect communities is not achievable in any single study and so a standardised methodology is required which is informative, practical and cumulative. A potential approach is the use of Malaise trapped beetles.

Coleoptera (beetles) are the most species rich order of insects and are estimated to provide close to 50% of insect species in New Zealand (Watt, 1982) and elsewhere (e.g., Lawrence and Britton, 1991). The order is ubiquitous in terrestrial systems and ranges across all insect functional groups. Taxonomy and the dynamics of sampling are also

better understood than for other species rich insect groups with a full trophic range such as the Diptera (flies) or the Hymenoptera (ants, bees and wasps). Malaise traps (Townes, 1972) are cheap, passive, and independent of both the habitat and the researcher, in contrast with e.g., branch clipping or sweep netting. When erected correctly they sample ground moving, emerging and low-flying insects.

Hutcheson (1990) sampled Coleoptera from spring to summer at two sites using three Malaise traps at each site. Divisive classification (TWINSPAN (Hill, 1979)) grouped weekly catches by site, time of season and trap-site in descending order of influence. Early summer catches were found to be most characteristic for sites, and two hypotheses were derived from the study: (a) a short sampling series in early summer is sufficient to provide distinguishable, representative samples from insect communities, and (b) insect communities are related to habitat type as well as to site. We tested these hypotheses using the results from three sampling series, conducted over three separate summers. Indigenous habitats sampled included wetland, heathland, tall diverse shrubland, and mature podocarp/broadleaf forest in the central North Island. Samples were included from similar habitats separated in space from 5m to 100 km, or in time by four years.

Methods

Trap-site information

Two sample series were taken four years apart from the Waipapa Ecological Area about 50 km northwest of Lake Taupo (38°25'S, 175°35'E). A third series sampled a heathland frostflat adjacent to the headwaters of the Rangitaiki river (38°54'S, 170°22'E) and approximately 100 km southeast of the Waipapa sites. Both areas lie on the central North Island volcanic plateau at c. 540 m asl. Climatic data over the sampling periods were extracted from NZ Meteorological Service archives collected 8 km southwest of the Waipapa, and 15 km northwest of the Rangitaiki site.

Successional stages of native vegetation ranging from monoao (*Dracophyllum subulatum* Hook. f.) dominated heathland through diverse shrubland to tall podocarp forest are present in the Waipapa Ecological Area (Leathwick, 1987). The first insect sampling from tall diverse shrubland and mature forest used three traps at each of two sites (Hutcheson 1990). The shrubland site (traps S1, S2, S3 in this paper) was approximately 200 m from the high forest edge and had a major component of *Pseudopanax* spp. The forest site (traps F1, F2, F3 in this article) was within one kilometre of the shrubland site and approximately 150 m from the forest edge. The second Waipapa series, taken four years later, used one trap in each of seven sites including the centres of the two sites sampled previously. Habitat types included in this second series were monoao heathland, tall diverse shrubland with *Pseudopanax* spp. as a major component, and mature podocarp/broadleaf forest.

The Rangitaiki series used one trap at each of three heathland sites. These were: a small gully 200 m from the heathland edge, an excessively drained area over 1 km from the heathland edge, and a boundary of the heathland and a permanent wetland area about 1 km from either of the other two traps.

The standard 'recce plot' procedure of Allen and McLennan (1983) as modified by Leathwick (1987) was used to document site characteristics and vegetation structure and composition. This procedure records vascular species and their cover classes within 6 fixed vertical tiers, within plots that are undefined but related to vegetation height. The method is rapid, semi-quantitative and provides an immediately interpretable table of the vegetation composition and structure. It has been used extensively in documenting New Zealand vegetation (Allen, 1992).

Beetle sampling

Beetles were sampled using Malaise traps of the dimensions outlined by Townes (1972). Smaller,

commercially available traps have been shown to be unable to provide samples of sufficient size to allow clear discrimination between habitat types (Dugdale and Hutcheson, 1997). Each Malaise trap requires a reasonably flat, log free area of about 2m². Traps consist of a fine gauze screen c. 2 m long x 1.2 m high pegged to the ground. They utilise the tendency of many insects to move upward when intercepted. Gauze end walls and sloping roof panels guide captives into a jar containing 70% ethanol (Cresswell, 1995). The jar attachment used (Hutcheson, 1991a) does not damage the trap peak and allows it to be tied up to existing vegetation, thereby minimising trap-site disturbance.

In the analyses reported here, catches were included from the major southern hemisphere adult insect activity period (weeks 47 to 3) of the first Waipapa series, and from weeks 47 to 52 of the two other series. Coleoptera were curated according to Walker and Crosby (1988).

Hutcheson (1990) tested data subsets using divisive cluster analysis and found good discrimination of communities required taxonomic resolution to recognisable taxonomic units (RTUs). These units are putative species and provide a practical approach to the problems generated by the unfinished nature of entomological taxonomy (Ramsay, 1986; Oliver and Beatie, 1993, 1996).

While it was possible to identify the majority of specimens to named species, many had to be coded within family, subfamily or genera. Identification utilised specimens in the Forest Research Institute and National Arthropod Collections and assistance of specialists. The general use of the term "species" in this article refers to RTU.

The majority of families were assigned into simplified functional groups defined as: detritivores (including fungivores and scavengers), herbivores (including all live plant feeders), predators and aquatic species. The latter were separated to distinguish terrestrial and aquatic processes. Weevils were assigned to trophic group at the RTU level, and staphylinids at the level of subfamily or tribe.

Analysis

Sample affinities were assessed using Sørensen's similarity index (Krebs, 1978) and polythetic divisive classification (TWINSPAN). Sørensen's index (K) compares species presence/absence in pairs of traps ($K = 2c/a+b$, where a, b, and c respectively represent species unique to each trap and those which are shared). Average K values within and between habitat types were then calculated. The TWINSPAN procedure groups catches into clusters based on component species

and their abundance within classes. Abundance classes found to provide best discrimination of catches to their associated sample group by Hutcheson (1990) were used. These were defined at cut levels of 0, 2, 5, 10 and 20 specimens. A range of sampling period subsets were tested with the classification program to determine the period with the most consistent grouping of weekly catches to their associated traps. Subsequent analyses then focused on catches taken during this period.

Average trap catches were compared in terms of numbers of individuals, species and families, and species abundance class distributions. Species richness (S) was compared with diversity measured as Shannon's index (H'), evenness (J') and summed abundance classes (SAC). Relative functional proportions for an average catch are compared for traps in terms of individuals, species and diversity (SAC), and for habitats in terms of individuals and species.

Results and discussion

The data showed (1) that catches were best discriminated into groups correlating to the habitat types when only a four week period in December was used (supporting hypothesis a), and (2) that classification was influenced more by habitat type than by site (supporting hypothesis b).

Sample affinities and the optimum sampling period

Subsets of the data were subjected to the classification procedure to test for optimum discrimination of catches from the various traps. Catches from the early summer tended to group by habitat types, although varying numbers of catch 'misclassifications' occurred with different subsets (Table 1).

Highest eigenvalues (= amount of variance in the data accounted for by a division) at the first division were obtained when catches from a single week were used. However, as occasional catches were misclassified with most permutations, and a single catch/trap cannot provide corroboration of classification groupings, a small group of catches from each trap was necessary. The four catches/trap from weeks 49-52 were most effectively discriminated in accordance with vegetation type (Fig. 1). There was no discrimination between the year of capture for sites sampled four years apart.

When more than four, weekly catches/trap were included, this increased the incidence of 'misclassified catches' and returned a lower eigenvalue at the first division. Sample discrimination was more successful within weeks 49-52 than over longer periods because the smaller seasonal changes of communities within the shorter period provided a greater homogeneity of the

Table 1: *Subsets of sampling weeks, number of catches, eigenvalues of divisions and the number of misclassified catches in each cluster. In the classification, the hierarchy of divisions corresponded to : Level 1. Wetland and heathland, from tall diverse shrubland and forest. Level 2. (a) Wetland from heathland; (b) tall shrubland from forest. Level 3. Only the heathland catches were able to be divided in any consistent fashion at this level, and then only when weeks 49-52 were included. Catches then separated into those from Rangitaiki and those from Waipapa (sites 100km apart). Level 4. This same subset also allowed heathland sites in Rangitaiki to be separated.*

Week (s) n^1	Eigenvalue ² at level 1	Misclassifications ³	Level 1	Level 2 a/b	Level 3	Level 4
49	16	0.753	3	*/*	*	*
50	16	0.819	1	*/0	*	*
51	16	0.775	0	0/2	*	*
52	16	0.743	0	*	*	*
49,50	32	0.768	0	0/1	*	*
51,52	32	0.724	0	0/4	*	*
50-52	48	0.738	0	0/*	*	*
49-51	48	0.735	1	*/0	*	*
# 49-52	64	0.718	0	0/3	0	1
47-52	96	0.689	0	*/3	*	*
47-52	114	0.653	1	0/3	*	*

¹ n = number of catches (1 catch = 1 trap-week).

²Eigenvalues indicate the amount of the variance accounted for by a division.

³Misclassifications are sample units not classified with the majority from their trap group.

* Indicates classification did not divide catches in an ecologically meaningful manner.

Period giving the most ecologically meaningful classification

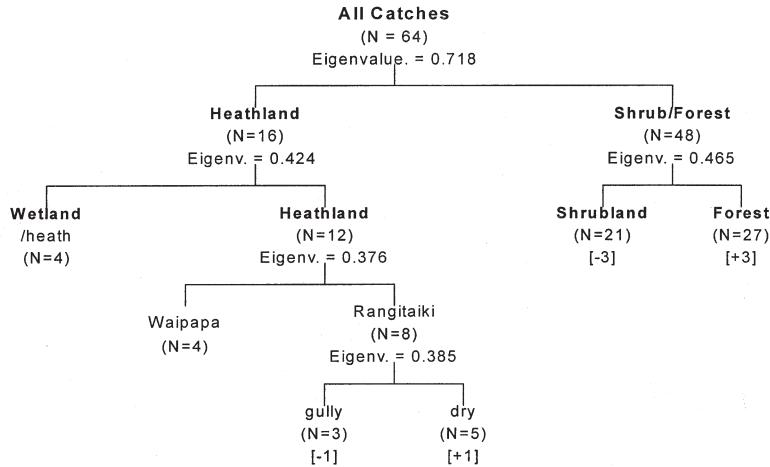


Figure 1: Classification of Malaise trapped beetle catches from a four week period withing December showing grouping in accord with four basic vegetation types. Forest and shrubland groups include catches taken four years apart. Waipapa and Rangitaiki heathland are . 100 km apart.

N = number of catches included in the group
[] = number of misclassified catches.

catches from each trap. This allowed catch discrimination to be more strongly influenced by the spatial (or habitat) variation. Hutcheson (1990) reported that low catches from early and late in the season were less successfully classified and the dispersal of more mobile species later in the season also influenced classification.

Further discussion of catch attributes is limited to samples from this optimum sampling window. Forest and shrubland habitats are therefore represented here by 6 traps (in 4 sites x 4 weeks = 24 catches each), heathland by 3 traps (in three sites x 4 weeks = 12 catches) and wetland by 1 trap (x 4 weeks = 4 catches).

Although the four consecutive weekly catches provided by each trap are not statistically independent, they do provide a relative measure of the significance of the variation between trap-sites. Where TWINSPAN divisions of catches are grouped by trap rather than week of capture, between-trap (or spatial) variation is greater than within-trap (or temporal) variation.

Relationship of beetle catch to meteorological conditions

Weekly means for temperature and rainfall over the sampling period showed no consistent relationship with total catch of individuals and species for each sampling series. Hutcheson (1990) demonstrated a moderate relationship between species richness and

temperature only during the early part of the season, suggesting the main influence of climate may occur during the earlier, formative period in beetle development. Although peak activity may vary from year to year, the recommended sampling period apparently falls within an inevitable period of higher insect emergence and activity in New Zealand. Although beetle activity may be influenced by climate on a day to day basis, as the traps operate continuously, results appear relatively independent of the weather over the actual sampling period. The effects of species abundance variations on classification results are reduced through the use of the defined abundance classes. A range of systems and bioclims, from northern coastal sites to high altitude plains in the South Island, have been sampled satisfactorily using this period (Dugdale and Hutcheson, 1997; Hutcheson and Jones, in press, J. Hutcheson, *unpubl. data*).

Beetle catch

(a) Trap effects

Average abundances of families, species and individuals varied according to trap orientation and successional stage of the vegetation (Table 2). In the original study the traps were orientated in a random manner resulting in traps F2 and F3 having their collection jar more southward than northward. Malaise traps depend upon positive phototropism of potential captives, so the orientation of trap

Table 2: Average numbers of families, species and individuals per trap (n=4) with associated mean and standard error for each habitat type grouping.

Trap	Habitat	Families	Species	Individuals
W1	Wetland/Heathland	9.3	17.3	241.5
H1	Heathland	8.3	11.0	66.5
H2	Heathland	5.8	6.0	83.5
H3	Heathland	4.0	4.3	14.5
Mean		6.0	7.1	54.8
<i>s.e.</i>		1.2	2.0	20.8
S1	Five-finger shrubland	11.0	19.5	52.5
S2	Five-finger shrubland	13.5	23.0	56.8
S3	Five-finger shrubland	12.3	24.8	99.5
S4	Five-finger shrubland	14.8	25.3	41.5
S5	Lancewood shrubland	9.0	18.5	29.0
S6	Lancewood shrubland	12.5	18.5	50.8
Mean		12.2	21.6	55.0
<i>s.e.</i>		0.8	1.3	9.8
F1	podocarp/broadleaf forest	13.5	26.5	54.3
F2#	podocarp/broadleaf forest	7.8	11.8	18.0
F3#	podocarp/broadleaf forest	7.0	9.3	11.8
F4	podocarp/broadleaf forest	13.3	26.5	52.3
F5	podocarp/broadleaf forest	11.3	18.8	38.0
F6	podocarp/broadleaf forest	11.8	21.5	35.0
Mean		10.8	19.0	34.9
<i>s.e.</i>		1.1	3.0	7.1

= Forest traps orientated more southward rather than northward

collection jars towards the light is necessary. Reduced light levels in the forest may make traps in this habitat particularly sensitive to this effect. Apart from the two forest traps subject to such orientation effects, average trap catches within each habitat type were remarkably consistent.

Although the heathland trap H3 was within 50 m of *Pseudopanax* sp. dominated shrubland, this did not influence catch discrimination, whereas contrasting vegetation within 20 m has (Dugdale and Hutcheson, 1997). This suggests that the Malaise trap (measured as influence on catch classification) may draw insects from about 0.75 ha, which approaches the spatial scale found to allow correlation of saproxylic beetles to habitat attributes by Økland *et al.* (1996).

The possibility exists that results from the forest traps were influenced by their being erected at ground level and thus not adequately sampling canopy herbivores. However, subsequent within-canopy Malaise trap sampling (Hutcheson, 1996, and *unpubl. data*) has returned relatively small catches, with similar trophic proportions to those from traps erected directly below on the forest floor. Moeed and Meads (1984) also found higher activity closer to the forest floor in New Zealand. However, they reported little difference between catches from

12 m and those from 3 m (their lowest sampling height), suggesting that the greatest decrease in activity may occur between ground level and 3 m.

(b) Habitat effects

Early successional heathland had the lowest abundance and species richness (Table 2). The forest habitat did not have greater average species richness or abundance than shrubland, even when the incorrectly orientated forest traps F2 and F3 were removed from analyses. The wetland site (which was within the species poor heathland) had as high a species richness as did the later successional stages, and trap catches were extremely high. No distinction was made in the classification between catches from shrubland trap-sites with fivefinger (*Pseudopanax arboreus* (Murray) Philipson) and those with lancewood (*P. crassifolius* Sol. Ex A. Cunn.). Several beetles recorded in association with this plant genus appear more influenced by branch size or physiological status than plant species (Hutcheson, 1991b; Kuschel, 1970). Therefore, although general habitat types (or successional stages) have characteristic insect communities, details of botanical composition do not appear to have as strong an influence as many ecologists may have presumed.

Similarity of beetle assemblages

Averaged Sørensen's similarity indices showed a direct relationship between increasing presence/absence similarity of beetle assemblages, and the temporal proximity of vegetation successional stages (Table 3). The siting of the wetland trap on the edge of the heathland was reflected in the relatively high similarity index for this habitat pairing. The average value from within each habitat type was remarkably similar, although the relatively low figure (c. 0.37) reflects the substantial variation in the species composition captured. This variation is partly a function of the sampling process, with rarer species less likely to be consistently captured. In general however, beetle catches were dominated by a limited number of common species (Fig. 2). While Malaise catches are most influenced by the local habitat, the inclusion of species from a relatively large area enhances successful classification into groups correlating to generic vegetation types.

Table 3: Averaged Sørensen's similarity measure of catches within and between pairs of habitat types. Within habitat similarity of catches is shown in bold characters.

	Wetland	Heathland	Shrubland	Forest
Heathland	0.36	0.37		
Shrubland	0.08	0.10	0.37	
Forest	0.02	0.06	0.29	0.38

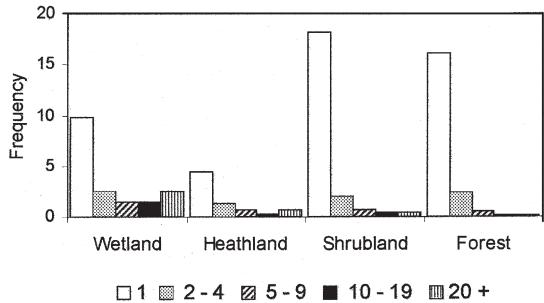


Figure 2: Distribution of abundance classes by habitat, showing the average number of species/catch in each of the defined abundance classes.

Diversity

The concept of diversity includes the genetic variation both between and within species. Diversity indices approximate the latter by including abundance in some fashion and have traditionally been used for community comparisons (Tokeski, 1993). Summed abundance classes (SAC) (Hutcheson, 1996) provide a simple measure of diversity which gives similar relative patterns to other indices. However SAC is more influenced by species richness than the commonly used Shannon's H' which is very responsive to abundant species (Fig. 3). A benefit of SAC is that calculation of the comparative diversity

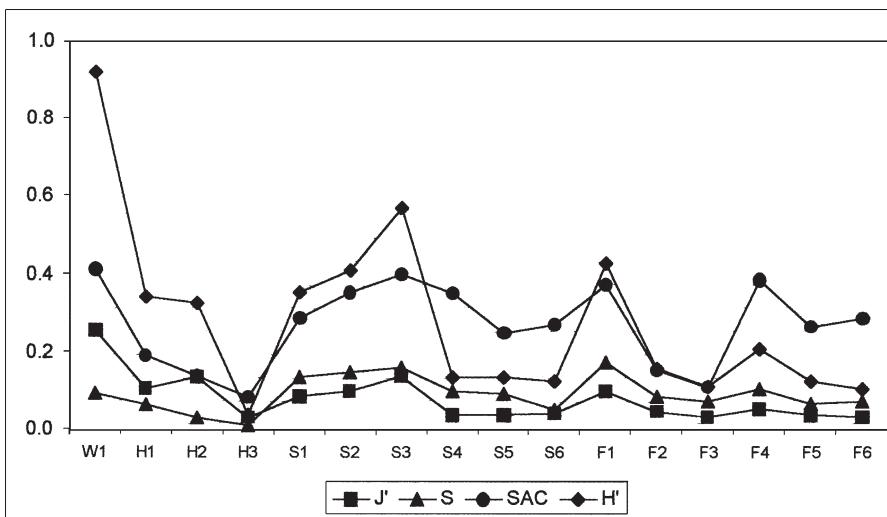


Figure 3: Comparison of a range of diversity measures of average trap catches. S = species richness, H' = Shannon's index, SAC = summed abundance classes, J' = evenness. Both S and SAC have been transformed to the same scale as other indices by dividing by 100. Trap codes are: W = wetland, H = heathland, S = shrubland and F = forest.

of data subsets based on biological attributes, such as function, is relatively simple.

Although diversity indices deliver a single figure that is apparently useful for comparing assemblages, most useful information is lost in the data compression process. Traditional diversity indices confound species richness, abundance, function and identity. Interpretations of community data of use to managers and ecologists require information such as the degree of endemism and the functional structure of the community. These depend upon species identity. Tokeshi (1993) notes that the imbalance of attention given to diversity indices is due more to conceptual appeal than to any scientific rigour or superiority. Insect populations respond to the

availability of appropriate resources (e.g., Hosking and Hutcheson, 1986; Hutcheson, 1991b; White, 1993; Stamps and Linit 1998). The identity and life history of the species within communities, and particularly of those that are dominant, can therefore provide the greatest amount of knowledge of system relationships in terms of resource variation resulting in particular community configurations.

Dominant species

Only a limited number of species were dominant in the catches (Fig. 2) and these were not always consistent in time or space within the main habitat groupings. Presence of some dominant species

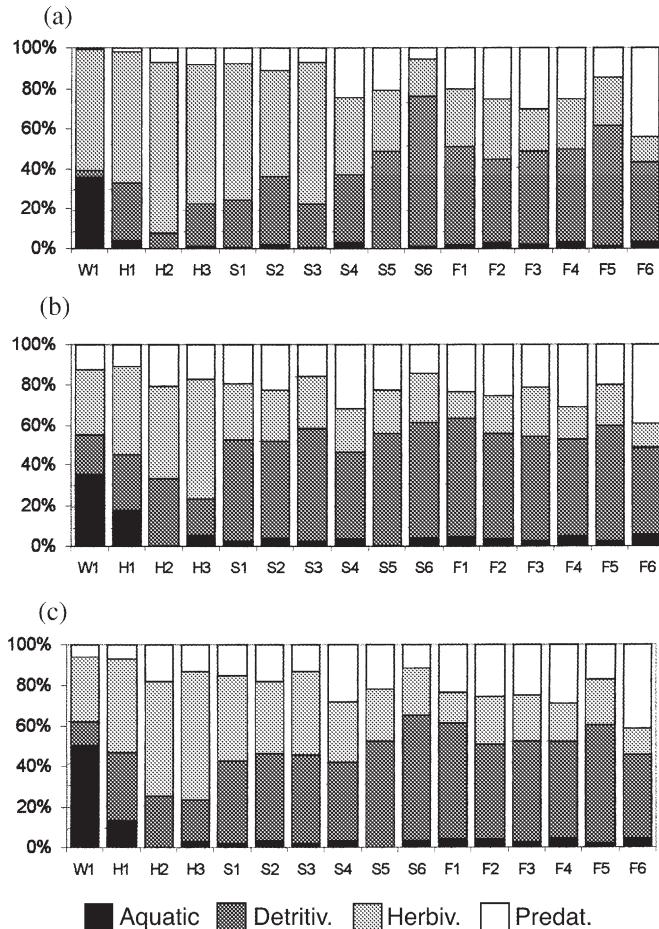


Figure 4: Average functional composition by trap, (a) individuals, (b) species, (c) diversity measured as summed abundance classes. Abundance classes formed by cut levels of 0, 2, 5, 10 & 20 specimens. Trap codes as in Fig. 3. Trophic groups: Aquatic, Detritivores (including scavengers and fungivores), Herbivores (all live plant feeders) and Predators (incl. parasites).

reflected obvious relationships with habitats, e.g. host specificity or physical factors such as the presence of water. However, several species known to be generalist herbivores, predators and detritivores were also found to be consistently associated with particular habitat types. This indicated that community types are characterised by more complicated factors than obvious obligate interspecies relationships.

Functional structure of the beetle communities

Average trophic composition was calculated for each trap (Fig. 4) and habitat group (Fig. 5) in terms of individuals, species and (for traps) diversity (SAC). Comparison of the trophic proportions of abundance and species richness (Figs. 4 & 5, a & b) clearly depicted where higher proportional abundance of a trophic group was provided by a much smaller proportion of species or vice-versa. A general trend of increasing proportions of detritivores and predators at the expense of the herbivore component, was recorded through the successional stages of the vegetation (Fig. 5). Proportional diversity of the trophic categories (measured as SAC) inevitably provided a pattern intermediate between those derived from individuals and species (Fig. 4c).

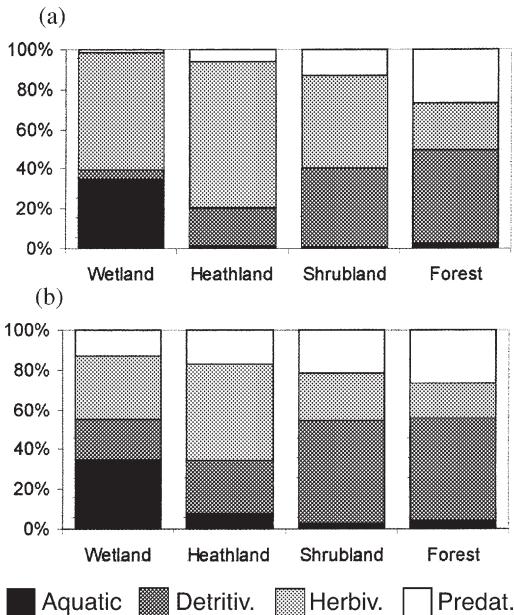


Figure 5: Average functional composition by habitat, (a) individuals, (b) species. Key as in Fig. 4.

The functional structure of the community was also revealed to be sensitive to individual site processes. For example, trap-site S6 had been modified by deer browsing and antler-rubbing of nearby pole age stems, and this was reflected by an increase in the proportional abundance (more than species richness) of detritivores. The individualistic nature of the habitats sampled is thus reflected in, and may be interpreted from, the life histories of dominant species in samples.

There is a growing recognition of biodiversity being a manifestation of the processes occurring within habitats (Reice, 1994; Attiwill, 1994). Indications from other insect studies (Hammond, 1990; Hutcheson, 1996; Hutcheson and Jones, in press) are that habitat processes have a strong influence on the abundance and composition of dominant species within local communities. For example, herbivores respond to changes in plant physiology associated with environmental stress (e.g., Hosking and Hutcheson, 1986; Hutcheson, 1991b; White, 1993; Hutcheson, 1996). Systems with large amounts of accumulated cellulose such as the late successional stages in this study, and particularly those with rapid carbon accumulation such as exotic pine plantations (Hutcheson and Jones, in press), have been found to be dominated by detritivore populations.

In the present study, tall forest was not sampled in areas of recent disruption such as tree-falls or areas of declining forest health. However, the shrubland site with deer-related damage showed a corresponding variation in functional abundance. Other studies have also shown that while beetle communities may be classified in accordance with generic vegetation types, local abundance and functional structure varies in a logically interpretable fashion when knowledge of species life histories is available (Hutcheson, 1996; Hutcheson and Jones, in press).

Relatively high species richness and abundance were recorded from all the shrubland sites and this may be related to the higher turnover of vegetation occurring within this successional stage. The extremely high abundance and relatively high species richness of the wetland catches may also be related to habitat processes. A major component of the wetland catches were Scirtidae, the larvae of which are semi-aquatic filter-feeding detritivores. This suggests that a significant portion of the nutrient stripping capacity of wetlands may be contributed by insect activity.

Methodology

Current knowledge of the relationships between insect communities and their habitat is in its infancy. Attempts at replication, particularly of natural

systems, are thus more a reflection of the ecologist's (usually botanical) perceptions than of influences that are necessarily of importance to the insects. Data must be allowed to suggest hypotheses, which may then be more widely tested for repeatability. The collation of many studies will be required and it is therefore imperative that methodology be standardised. Descriptive studies form an initial approach in all branches of science and are valid provided inferential statistics are not applied to the data (Hurlbert, 1984; Hurlbert and White, 1993). Design of insect community studies may thus be guided by criteria which may vary from those used for more narrowly focused questions on single populations, which unlike communities, are bounded by the normal curve.

The use of large, point-sourced samples is felt to be an appropriate approach for describing the relationship between insect biodiversity and habitats as perceived by ecologists. Malaise trap samples are large enough to be amenable to multivariate analyses, which are efficient at revealing community affinities and relationships. Component species are apparent in results, and species autecology may be used for interpreting the relationships of the sample with the particular site of origin.

Analyses that neglect species identity, such as species abundance models and diversity indices, are not particularly helpful for interpreting insect community data. Tokeshi (1993) showed that the geometric and the log series may be nested inside the log normal series, suggesting that the different models may be a function of definition of community boundaries (i.e., scale) rather than community assembly rules. While the log normal curve has been found to fit many datasets (Hughes, 1986), if not most (Zak, 1992), May (1975) concluded that this pattern was essentially a statistical consequence of large numbers. Rank-abundance curves may depict relative dominance patterns (e.g., Samways, 1984), but these were unable to discriminate Malaise trapped beetle samples from different communities (Hutcheson, 1990).

A high level of taxonomic information is crucial for conservation management questions of immediate concern such as the relative endemicity of communities. However, insect community attributes show great potential for improving our understanding of the dynamic nature of ecological systems. Insects permeate ecosystems, performing a wide variety of functional roles. They provide a large portion of the selection processes operating on individual plants in various environments (e.g., Hosking and Hutcheson, 1986; Hutcheson, 1991b; White, 1993). They are involved in pollination, decomposition, soil formation, and predation, and

provide the dominant food for birds, lizards, and most freshwater fish (Watt, 1975). Because of their species richness, and the wide variety of functional roles they perform, beetle assemblages provide broadly based, detailed and 'real' knowledge of the responses of the largest portion of terrestrial biodiversity to a wide range of influences.

Species are indicative of their environmental requirements, rather than of the remainder of putatively 'fixed' communities (Hutcheson *et al.*, 1997). Taxonomic groups with narrow environmental requirements are thus unable to be indicative of the current status of entire communities (Prendergast *et al.*, 1993; Lawton *et al.*, 1998). However, as the largest known group of terrestrial organisms, with a complete functional range, beetles have perhaps the strongest claim to reflecting the 'biodiversity' of ecosystems. For interpretations to be useful however, it is necessary that (a) methodology has been demonstrated to deliver samples 'characteristic' of the land managers perception of ecosystem types; and (b) that interpretations are based on the biological attributes of the component species, rather than simply on numerical summations of the entire group. This study shows it is possible to obtain characteristic beetle assemblages from vegetation systems as these are recognisable to managers, and that the autecology of assemblage components provide insights into ecosystem processes.

In Cameroon, documentation of bird assemblages required an order of magnitude less scientific resources than beetles did (Lawton *et al.*, 1998). In the current study, documentation of the vascular plants required several orders of magnitude less time than the beetles. Attempts to derive a more detailed picture using survey approaches such as fogging (e.g., Kitching *et al.*, 1993; Basset, 1990; Erwin, 1983; Majer and Recher, 1988) and full season sampling (Moeed and Meads, 1984, 1985, 1986, 1987; Hutcheson, 1990), while appropriate for some questions, are not pragmatic options for widespread comparison of insect communities of various ecosystems. The definition of "best sampling time" in this study enables extremely useful information to be acquired for a (relatively) modest level of resources.

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CORRIGENDUM

Hutcheson, J.A.; Kimberley, M.O. 1999. A pragmatic approach to characterising insect communities in New Zealand: malaise trapped beetles. *New Zealand Journal of Ecology* 23 (1): 69-79.

Table 1: The column headings are misplaced. The first column is Week(s), the second is n', the third Eigenvalue² at level 1, and the fourth to seventh are the Misclassifications at levels 1 to 4.

The reference on page 77 to Hutcheson *et al.* 1997, should be Hutcheson, Walsh and Given (1999) and may be found as referenced in press but now published in *Science for Conservation* 109, 97 pp.