

Characteristics of Mapara insect communities as depicted by Malaise trapped beetles: changes with time and animal control

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CONTENTS

Abstract	5
<hr/>	
1. Introduction	6
<hr/>	
2. Background	6
2.1 Insects and biodiversity	6
2.2 Characterisation of insect communities	7
<hr/>	
3. Methods	9
3.1 Site documentation	9
3.2 Characteristics of the Mapara sites (habitats)	9
3.3 Insect sampling	10
3.4 Taxonomy	10
3.5 Analyses	11
3.5.1 Sample affinities	11
3.5.2 Numbers of individuals, species and families, and species abundance distributions	11
3.5.3 Diversity	11
3.5.4 Functional (trophic) structure	12
3.5.5 Functional diversity of communities	12
3.5.6 Dominant species	12
<hr/>	
4. Results and discussion	13
4.1 Sample affinities	13
4.1.1 Divisive Cluster Analysis (TWINSPAN)	13
4.1.2 Species overlap (Sørensen's similarity index)	14
4.2 Numbers of individuals, species and families	14
4.2.1 Species abundance distributions	15
4.3 Diversity	15
4.4 Functional (trophic) structure of assemblages	15
4.5 Functional diversity of communities	16
4.6 Dominant species	17
<hr/>	
5. Conclusions	18
<hr/>	
6. Acknowledgements	19
<hr/>	
7. References	19
<hr/>	

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Abstract

Insect communities from mahoe-fuchsia forest in Mapara reserve and adjacent grazed forest were sampled in 1989, 1990 and 1997 using standardised protocols based on Malaise trapped beetles. Over the 8-year period that the study encompassed, mid-level woody vegetation increased in the reserve where browsing mammals were controlled, and decreased in the adjacent grazed forest.

Divisive classification (TWINSPAN) showed greatest sample differentiation between the earlier two series, and those from 1997 when fewer specimens were captured from both sites. The second level of division then clearly distinguished between samples from the grazed and reserve forest. Sample diversity was higher in the reserve in 1997. Early grazed forest samples included species involved in the breakdown of debris, while early reserve samples included species thought to be associated with dead animals. Component species autecology thus suggested that the decreased catch in 1997 was due to a stabilisation of habitat processes in both sites.

Approximately 86% of the 229 species captured are known to be endemic, with five of the 6 known adventives being captured only in the grazed site. While disturbance may encourage the incursion of adventives, these still only comprised a low percentage of the total in this grazed native bush remnant, where canopy and forest floor were still largely intact.

The study provides further evidence that insect biodiversity is associated with resource availability and habitat processes and that the most useful interpretation is based on component species identity rather than just on numerical models.

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1. Introduction

Part of the program to enhance the kokako population of Mapara reserve has been control of grazing and browsing mammals, which has encouraged the regeneration of mid-level woody vegetation. This provided an opportunity to compare the effects of such habitat management on the insect communities over time.

Insect communities were sampled within and outside the animal control zone during December of 1989, 1990 and 1997. Characteristics of the trap-sites and samples are presented here. The interpretations of the relationships between the insect communities and their habitats have been helped by a context of results from other standardised sampling of a range of habitat types.

2. Background

2.1 INSECTS AND BIODIVERSITY

Terrestrial biodiversity (i.e. genetic variety) is totally dominated by insects, whether measurement is made at genotypic (Berry 1982), phenotypic / character (Williams & Humphries 1996) or species levels (Gaston 1996). This dominance is a reflection of the involvement of insects in virtually all terrestrial ecological processes. In addition to their consumption of live plant material, insects are involved in pollination, leaf, litter and log breakdown, soil formation, general scavenging, parasitism and predation, as well as providing the dominant food for birds, lizards, and most freshwater fish (Watt 1975). The insects' high diversity and wide contribution to ecosystem function thus make them ideal for providing information on both the status of biodiversity within terrestrial systems, and on processes that are occurring in these systems.

The relationship of New Zealand insect communities to various vegetation systems, and the controlling influences on their communities are still poorly understood. This is partly due the incomplete nature of the taxonomic and autecological information available. It is estimated that only about half of our insects are currently described (Watt 1982, Emberson 1994), while the ecology of most species is presently only able to be assumed from their taxonomic relationships. Given the pervasive and integral roles that insects perform in all ecosystems, this represents a major knowledge gap for the broader conservation management of New Zealand's endemic biodiversity.

Most New Zealand insect groups display a high degree of endemism (e.g. Watt 1975, Klimaszewski & Watt 1997), and therefore the great majority of New Zealand's contribution to global terrestrial biodiversity is provided by our endemic insects. However, most estimates have been compiled from unquantified historical collections, which can therefore give little idea of any subsequent loss of endemic species, or of the effects of various current land management regimes on this part of our biological heritage.

Endemism occurs at both regional and local levels (Kuschel 1990) and biodiversity conservation is therefore required across the landscape. Over 60% of New Zealand's land area has now been radically modified from the original forest vegetation cover (McGlone 1989). Several authors have commented on the difficulties our native (mostly forest) insect fauna has in persisting in modified environments (e.g. Dugdale 1975, Watt 1982, Kuschel 1990, Hutcheson & Hosking 1994). It is only in low rainfall areas such as Canterbury and Otago that significant portions of endemic insect fauna persist within the modified (grazed) landscape, indicating that it is only in such areas that substantial grassland existed in prehuman times (Watt 1982).

Browsing mammals, and particularly domestic livestock, are now recognised as being the greatest threat to continued forest cover (Hutcheson & Hosking 1994). All trees die eventually and without the successful development of regeneration there can be no future forest. Domestic stock provides the most effective means of removing regeneration, and was routinely used after logging or fire for land clearance in the past (Buchanan 1869, Bagnall 1976, Buick 1976). Consequently, control of grazing and browsing mammals forms a major and vital part of conservation management.

Understanding how insect communities vary with habitat processes is essential to meaningful interpretations of data gathered to evaluate the effects of land management on biodiversity. The present work contributes to a program of standardised studies conducted by the author (e.g. Hutcheson 1990, Hutcheson 1996, Dugdale & Hutcheson 1997, Hutcheson & Kimberley 1999, Hutcheson & Jones 1999, J. Hutcheson unpubl. data) which provide a broader ecological context for the interpretations reported here.

2.2 CHARACTERISATION OF INSECT COMMUNITIES

The large number of insect species, together with their small size, mobility, seasonally variable and cryptic lifestyles, make the study and characterisation of their communities difficult. It is impossible to document all insect species for any large habitat, there are simply too many species and insufficient taxonomic expertise (Disney 1986). Disney's conclusions were based on the situation in Britain, where most insect species are known, keys exist for their identification, and there are a large number of entomologists. None of these conditions exist in New Zealand. In addition, while a documentation of species is sufficient for systematic purposes, information on relative species abundance is necessary for ecological interpretations that are useful to management.

Standardised sampling, taxonomy and analyses are necessary for data to be accumulated and compared. However, several traditional approaches to community characterisation have been found to be less than ideal for this type of ecological data. Sub-samples taken from the same site are pseudoreplicates, are not independent, and may not be legitimately analysed using inferential statistics (Hurlbert 1984, Hurlbert & White 1993). Separation of sub-samples over different sites, while providing independence, obscures relationships between the insect communities and the site-related processes that can influence their composition (Hutcheson & Kimberley 1999). In contrast, large point-source samples retain community-habitat associations, and are large

enough to provide robust classification with multivariate analysis techniques. Objective analysis of community affinities is therefore possible while retaining the individualistic nature of the community data. Hypotheses arising from actual data, rather than from theoretical speculation, may then be tested at sites widespread in space and time.

Because of New Zealand's unique faunal composition, many approaches to insect community characterisation developed overseas are inappropriate here (see extended discussion Hutcheson et al. 1999). The use of Malaise trapped beetles has however been demonstrated to be an informative, repeatable and pragmatic sampling methodology in New Zealand vegetation systems (Hutcheson 1990, 1996, Dugdale & Hutcheson 1997, Hutcheson & Kimberley 1999, Hutcheson & Jones 1999).

Coleoptera (beetles) are estimated to provide about half of the insect species in this country (Watt 1983) and similar proportions elsewhere (Lawrence & Britton 1991). They are multi-trophic and cover the complete range of functions provided by the insect community. Their taxonomy, ecology, and the dynamics of their sampling are also better understood than for other species-rich, multi-trophic groups such as Diptera (flies) and Hymenoptera (ants, bees and wasps). Beetles have been regarded by many authors as representative of the 'richness' of the insect fauna in several systems (e.g. Neumann 1979, Tanaka & Tanaka 1982, Moeed & Meads 1985), and review of a range of sampling methods has shown that the correlation of beetle abundance to total insect catch is related to sampling effort. High correlations are achieved with weekly Malaise trap samples whereas methods such as pitfall trapping require an inordinate sampling investment to achieve similar results (Hutcheson et al. 1999).

Malaise traps are simply gauze screens that, when erected correctly, intercept emerging, crawling and low-flying insects. They are passive (compared with for example, light-trapping or baited traps), and are independent of the habitat, weather or researcher (compared with, for example, branch clipping or sweep netting). Traps may be erected virtually anywhere, and thus comparable information may be obtained from most terrestrial vegetation systems. The vertical catching zone (within about 1 m of the ground) has also been demonstrated to be within the zone of greatest insect activity in indigenous forest habitats (Hutcheson 1996). The species that are present in the samples cover a wide range in the degree of their mobility, and habitat aspects closer to the trap influence catch composition to a greater degree (Dugdale & Hutcheson 1997). More common species of a trap-site are represented in higher numbers, while rare species are much less likely to be captured.

Spring to autumn sampling of beetles with Malaise traps suggested habitat to be of primary importance in sample composition, while time of year and trap position were of secondary and tertiary importance respectively (Hutcheson 1990). Subsequent work has verified that beetle communities are characteristic for generic habitat types, and has shown that reducing the seasonal sample variation by taking only four consecutive weekly catches per trap-site in early summer, enhances the ability to discriminate between communities (Hutcheson & Kimberley 1999). The four weekly catches taken per trap also provide a relative measure of within-site variation, against which between-site variation may be objectively evaluated using cluster analysis.

Studies from a range of habitats have now shown ecologically meaningful groupings of samples when data is analysed with divisive cluster analysis. The life histories of the beetle species which comprised the assemblages have also been able to be interpreted in terms of the attributes of their habitats of origin (Hutcheson 1990, 1996, Dugdale & Hutcheson 1997, Hutcheson & Jones 1999, Hutcheson & Kimberley 1999, J. Hutcheson unpubl. data).

As shown in the current study, the approach can provide comparisons of insect biodiversity over time and space, provided sufficient attention is given to standardisation. It therefore provides a means of evaluation of specific management questions, while at the same time cumulatively increasing our understanding of the major controlling influences on this large but cryptic part of our heritage.

3. Methods

3.1 SITE DOCUMENTATION

The same two sites were used in all three sampling series. The standardised 'rece plot' procedure of Allen & McLennan (1983) as modified by Leathwick (1987) was used to document site characteristics and vegetation structure and composition. The procedure records site attributes and vascular species and their cover classes within 6 fixed vertical tiers. The method is rapid and 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).

3.2 CHARACTERISTICS OF THE MAPARA SITES (HABITATS)

Sampling sites were similar in topography and aspect. Both had intact mahoe-fuchsia canopy, although understorey vegetation varied with the intensity and history of animal browse.

The grazed trap-site was located in low mahoe-fuchsia gully forest within logged-over tawa forest, on Mr David Bolton's farm, north of and adjacent to the Mapara reserve. The area is browsed by stock, although an old mudstone rockfall in the immediate vicinity of the trap reduced browse intensity. Consequently there was still a reasonably intact ground cover in 1997, although this consisted mostly of less palatable species. The site is about 50 m up-slope from the forest edge, which begins approximately 10 m from the willow-lined Mapara stream. Browsing has removed much of the forest edge growth and understorey. The canopy comprises a higher proportion of *Fuchsia excortica* (J.R. and G. Forst.) Linn.f. in the general area than the site in the reserve. Ground cover in the vicinity of the trap is dominated by *Asplenium bulbiferum* Forst. f. An increase in weed species in the general area was noticeable in 1997. Mid-level woody vegetation dying off in 1989 and 1990, and had virtually gone by 1997.

The reserve trap-site was located about 2 km south of the farm site in a similar mahoe-fuchsia dominated gully in logged over tawa forest. The reserve was fenced about a year prior to the 1989 sampling, although some feral stock control had occurred over the previous 5 years. The trap-site is approximately 20 m up-slope from a small tributary of the Mapara stream, and about 100 m in from the forest edge at the base of the valley which divides the reserve. Regrowth in the area was well advanced by the 1997 sampling and formed a light but vertically continuous subcanopy, particularly in the gully floor area. An increase in the abundance and stature of the highly palatable pate (*Schefflera digitata* J.R. & G. Forst.), was particularly noticeable in the reserve site by 1997. Mudstone boulders similar to those at the farm site are present, but were more buried within the substrate. Ground cover at the reserve site was dominated by *Asplenium bulbiferum* Forst. f., *Blechnum fluviatile* (R. Br.) Salomon, *Blechnum colensoi* (Hook.f.) Wakef., and *Microlaena avenacea* Raoul (Hook.f.).

Details of vegetation composition and structure for both sites in 1989 and 1997 are provided in appendix 1. Application of the recording system to insect community sampling sites was in the developmental phase during the first series, and NR has been used to denote attributes that were not recorded in 1989.

3.3 INSECT SAMPLING

Insect communities were sampled over four weeks in December of each sampling year using one Malaise trap/site. Beetles were selected from samples, curated according to Walker and Crosby (1988), and used to characterise the insect communities as described in Hutcheson (1990, 1996) Dugdale & Hutcheson (1997) Hutcheson & Kimberley (1999), and Hutcheson & Jones (1999).

Malaise traps of the dimensions outlined by Townes (1972) were used, because smaller commercially available traps have been shown to be unable to provide samples of sufficient size to allow clear discrimination between habitat types (Dugdale & Hutcheson 1997, J. Hutcheson, pers. comm.). Each Malaise trap requires a reasonably flat, log free area of about 2 m². Traps consist of a fine gauze screen c. 2 m long × 1.2 m high pegged to the ground. They utilise the movement toward light of potential captives, a feature common to many insects during dispersal phase. End walls and a sloping roof guide intercepted insects upward into a jar containing 70% ethanol. The collection attachment of the trap was as modified by Hutcheson (1991a). This does not damage the trap peak and allows it to be tied up to existing vegetation, thereby minimising trap-site disturbance. The collecting fluid used was 70% ethanol (Cresswell 1995).

3.4 TAXONOMY

Hutcheson (1990) tested data subsets and found that in order to reduce the period of sampling to a minimum and still be able to discriminate between samples from two sites, taxonomic resolution needed to be at least to

recognisable taxonomic units (RTUs). These utilise species where possible, and approximate species from exterior morphology where full determination is impracticable. They provide a practical approach to the problems of taxonomy (Ramsay 1986, Oliver & Beattie 1993, 1996). RTUs allow taxonomy to be standardised across sampling series, and to be revised as knowledge increases. Non-standardised taxonomy prevents data being accumulative, effectively isolating any study and thus limiting its usefulness. Dr G. Kuschel, New Zealand's foremost coleopterist, collaborated in taxonomy of the 1997 study.

3.5 ANALYSES

Data were analysed and summarised using a range of approaches because of the information-rich nature of insect samples. Analyses retaining the identities of the component species were found most useful. Previous investigations have found that analyses concentrating solely on numerical relationships do not deliver information helpful to meaningful discrimination or interpretation of samples (Hutcheson 1990, Hutcheson et al. 1999, Hutcheson & Kimberley (1999).

3.5.1 Sample affinities

Weekly catch data in the form of species and their abundances were subjected to the divisive cluster analysis procedure TWINSpan (Hill 1979), to test the ability to discriminate samples from the different trap-sites and / or years. TWINSpan objectively clusters subsamples (weekly catches) into dissimilar groups based on the species present and their abundance within classes. Hutcheson (1990) determined optimum classes for community discrimination with cuts at 0,2,5,10 & 20 specimens.

Sample affinities were also assessed using Sørensen's (presence / absence) similarity index (K). $K = 2c/a+b$, where a, b, and c respectively represent species unique to each trap, and those which are shared (Krebs 1978). This is effectively a measure of species overlap between pairs of traps, and thus can give some idea of variation, at the species level, of assemblages within and between various habitats.

3.5.2 Numbers of individuals, species and families, and species abundance distributions

Samples are compared in terms of the numbers of individuals, species and families, and species abundance distributions.

3.5.3 Diversity

Although species richness is often used interchangeably with diversity, true diversity includes the genetic variation occurring within species as well as that found between species. A variety of methods exist for combining this intra-specific variation (which is derived from the abundance of each species) with species richness, to form 'diversity indices'. Such indices have traditionally been used for the comparison of communities. However, several studies have noted the inability of such indices to discriminate between communities (e.g.

Hutcheson 1990, Tonhaska 1994, Hutcheson 1996). This is because they confound species richness, abundance, function and identity. Tokeshi (1993) commented that use of such diversity indices was based on their conceptual appeal rather than on any scientific rigor. A comparison of species richness and three diversity indices—Shannon's (H'), Evenness (J') and a summing of the abundance classes (SAC) (Hutcheson 1996)—was made for the present study. H' gives greater emphasis to more abundant species, while SAC gives greater emphasis to interspecific variation. The latter is more in keeping with a general acceptance that inter-species variation is more important than intra-species variation. However, the major advantage is the simplified calculation of the diversity of data subsets grouped by biological attributes, such as trophic groups.

3.5.4 Functional (trophic) structure

Species were assigned into the simple trophic groups: detritivores (including scavengers and fungivores), herbivores (including all live plant feeders) and predators. Examination was then made of the functional structure of the samples at the levels of individuals, species and diversity (measured as SAC). These very simple groups were used because autecological knowledge is sparse, and because many functional assignments may be both artificial and constraining. For example, even with autecological knowledge of species, it is often difficult to obtain a clear definition of resource use along gradients between debilitated live plants and totally dead plant material, or the latter and mycelium of wood decay fungi.

3.5.5 Functional diversity of communities

Several aspects of the samples were combined to provide comparison of the functional diversity of the communities sampled. This used average attributes from weekly catch groups defined by the TWINSPAN classification. The diversity of the trophic components of these were compiled and displayed as summed abundance classes (see 3.5.3 above) so communities could be compared easily.

3.5.6 Dominant species

The greatest amount of interpretable information about the relationship between samples and their habitat is accessed via the identity of dominant component species within the characteristic sample groups. Species identity gives access to life histories, or what they 'do' in the habitats in question. Although some of this information is available from knowledge of the families that species belong to, this is insufficient for interpretation of species from larger multi-trophic families such as the Curculionidae (weevils).

4. Results and discussion

A total of 1263 beetle individuals from 229 species were captured in the three sampling series. Approximately 86% of all species are known to be endemic, with about 3% known to be adventive. The remainder were of unknown provenance. Five of the 6 known adventives were captured only in the farm site.

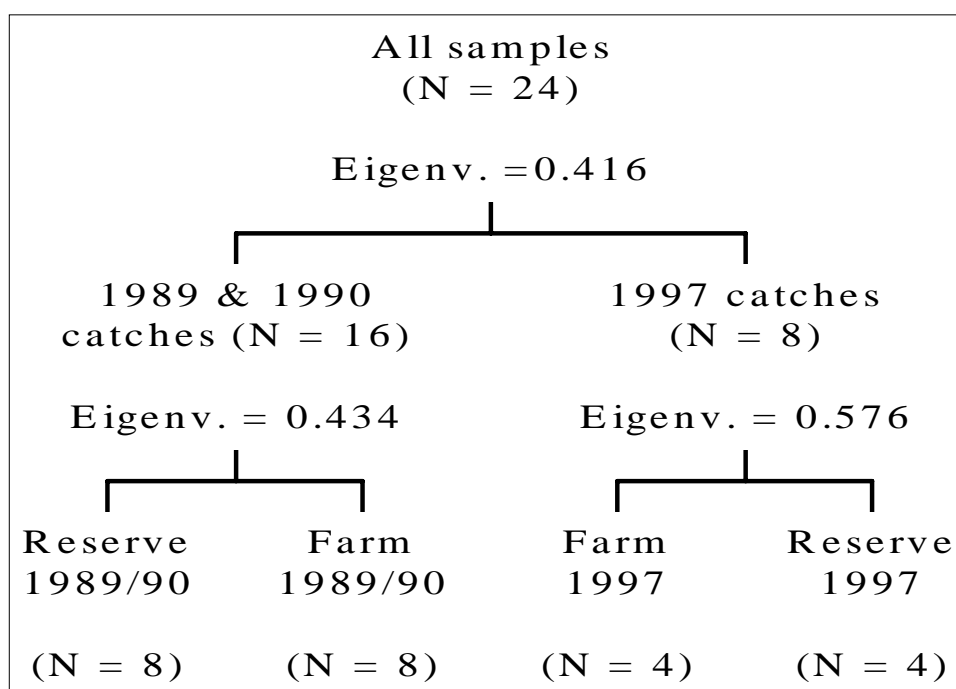
4.1 SAMPLE AFFINITIES

4.1.1 Divisive Cluster Analysis (TWINSPAN)

TWINSPAN clearly discriminated the weekly beetle catches by year (1989/90 from 1997) and then by habitat type (farm from reserve) (Fig. 1). Catches from 1989 and 1990 within the two habitats were not discriminated in the analysis. Separation of the reserve catches to the extreme left and right at the lowest level shown in figure 1 indicate that assemblages from the reserve site changed to a greater extent than those from the farm site over the 7-8 year period.

The clear discrimination of samples in the TWINSPAN analysis, from the earlier and later times and for the farm and reserve sites, objectively shows that samples were characteristic for the habitats as recognised by the manager / ecologist. This allows us to examine the attributes of the samples with some confidence that these reflect qualities of the communities that are related to the habitats at the sampling times.

Figure 1. TWINSPAN division of weekly Malaise trapped beetle catches from Mahoe / fuchsia gully forest in the Mapara area from 1989, 1990 and 1997. Eigenvalues indicate the amount of the variation accounted for by the division below the value. Four weekly catches were collected from each trap in each year and N = number of weekly catches included in the classification group.



4.1.2 Species overlap (Sørensen's similarity index)

While TWINSpan includes species abundances (within a set of classes), Sørensen's similarity index uses simply species presence / absence. Values from each trap / year show the greater species overlap between similar habitats in 1989 and 1990, but are less helpful in discriminating between the communities as defined by TWINSpan (Table 1). Previous studies have returned values from within heterogeneous indigenous systems in the region of 0.4, and from relatively homogeneous exotic systems in the region of c. 0.6. The highest similarity value was provided by assemblages from the farm site in 1989 and 1990, while the lowest was between the farm site in 1989 and the reserve in 1997.

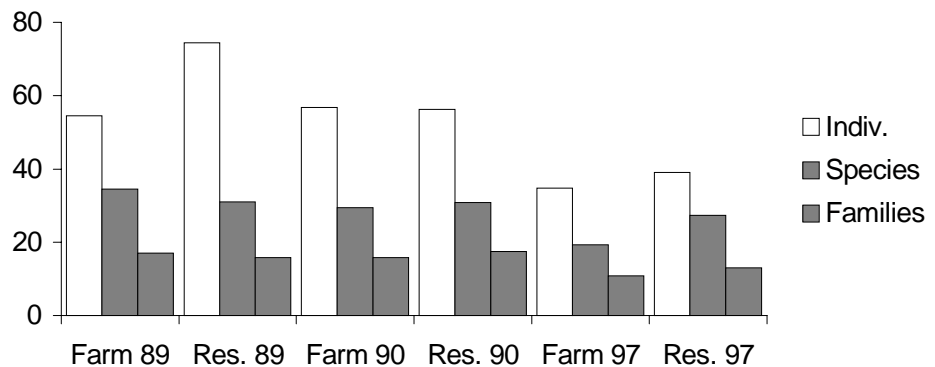
TABLE 1. SØRENSEN'S SIMILARITY INDEX BETWEEN MALAISE TRAPPED BEETLE ASSEMBLAGES FROM THE GRAZED AND RESERVE TRAP-SITES IN THE THREE SEPARATE YEARS.

	Farm 89	Res. 89	Farm 90	Res. 90	Farm 97
Res. 89	0.407				
Farm 90	0.547	0.456			
Res. 90	0.395	0.471	0.431		
Farm 97	0.317	0.275	0.305	0.270	
Res. 97	0.241	0.342	0.250	0.340	0.368

4.2 NUMBERS OF INDIVIDUALS, SPECIES AND FAMILIES

Lower overall activity in 1997 was apparent from the numbers of individuals, species and families captured (Fig 2). While species richness was slightly higher in the farm site in 1989, the number of individuals was higher in the reserve site. Although overall totals fell for both sites, by 1997 all three taxonomic levels were higher in samples from the reserve site than in the grazed site.

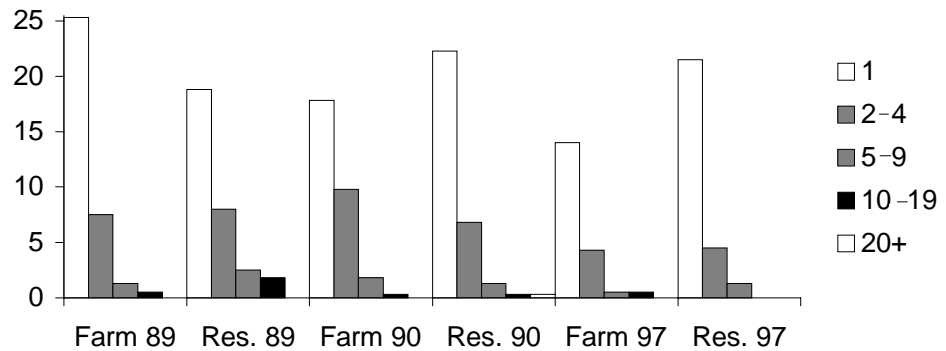
Figure 2. Numbers of individuals, species and families of beetles captured in the farm and reserve site for the years 1989, 1990 and 1997.



4.2.1 Species abundance distributions

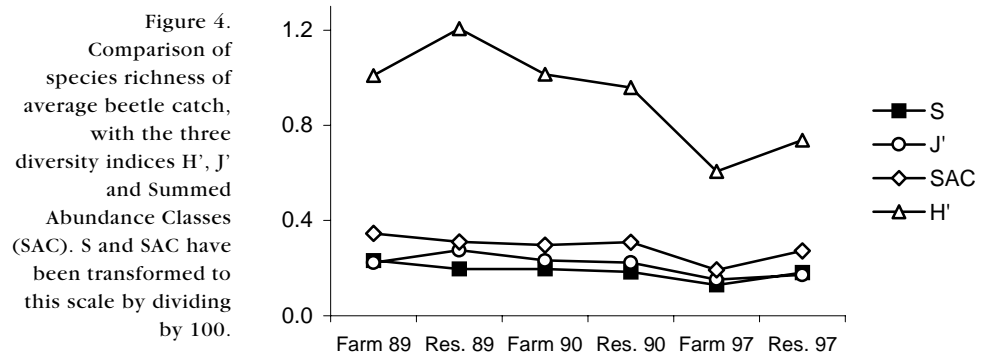
Average species abundance distributions give indications of the relative numbers of common and 'rare' species, and how these compare between times and sites (Fig. 3). Species in higher abundance were more prevalent in the earlier samples, while by 1997 samples were more dominated by species of lower abundance.

Figure 3. Average species abundance distributions for Malaise trapped beetle samples from the grazed and reserve sites for the three sampling years 1989, 1990 and 1997.



4.3 DIVERSITY

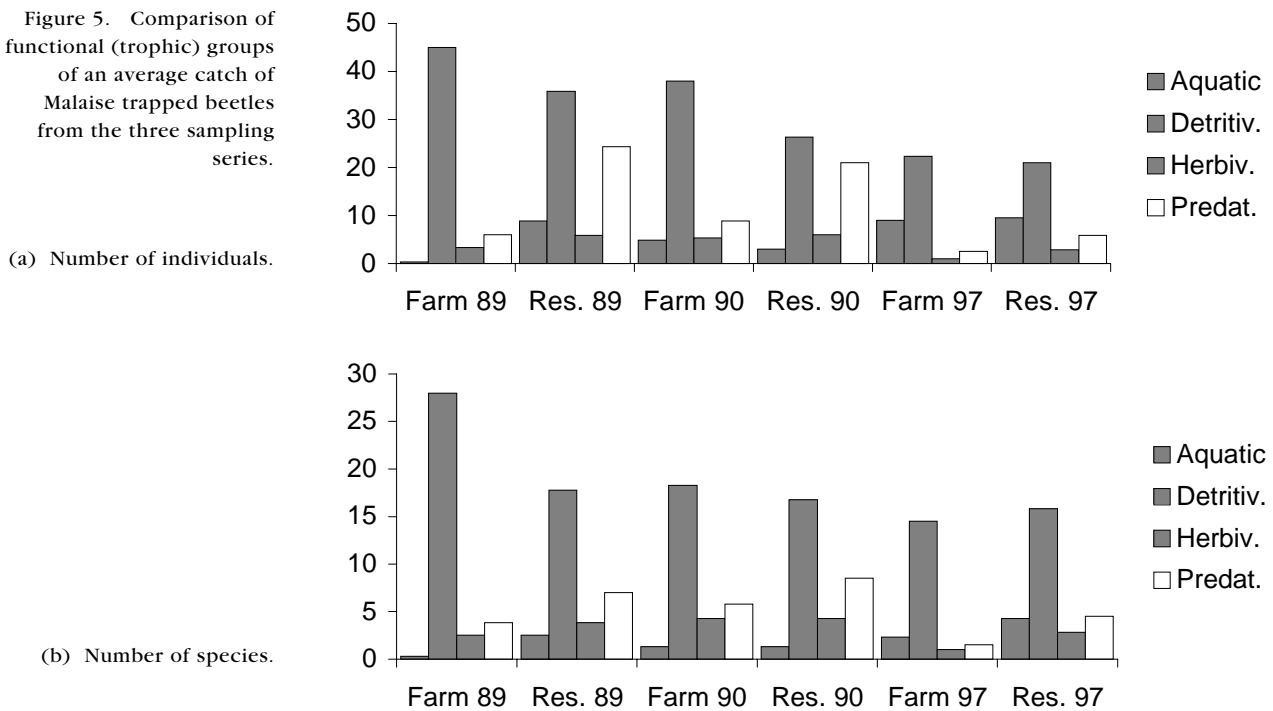
Comparison of species richness with the three diversity indices H' , J' and SAC show that they all deliver similar relative patterns across the traps. H' responds to a greater extent to higher abundances while SAC is more influenced by species richness (Fig. 4). Although the indices provide a single figure for comparisons, ecological interpretation of the comparison is difficult because species richness, abundance, function and identity are all subsumed within this figure.



4.4 FUNCTIONAL (TROPIC) STRUCTURE OF ASSEMBLAGES

Examination of the trophic structure of the samples (Fig. 5) provided a more interpretable insight into the nature of the differences between the communities of the different sites and years. At the level of individuals (Fig. 5a) all sampling series were dominated by detritivores. This has been found to be true of samples from all habitats with successional advanced vegetation. However, the dominance of detritivore individuals occurred to a much greater extent in the 1989 farm samples than in all others. Together with the information from the habitat records, this indicated that the processing of debris was occurring to a greater extent at this time in this site. By 1997

Figure 5. Comparison of functional (trophic) groups of an average catch of Malaise trapped beetles from the three sampling series.



detritivore numbers had fallen in the farm site. This was commensurate with the disappearance of most of the mid-level woody vegetation, which had been dying off during the earlier sampling series. A high number of predatory and scavenger individuals were apparent in the 1989 and 1990 reserve samples, but these occurred in much lower numbers from the 1997 sampling.

Examination of the trophic composition of species in samples shows a more stable structure than was apparent for individuals. It also revealed higher species richness in the reserve samples in most trophic groups in most years (Fig. 5b). The most noticeable exceptions were detritivore species in the early samples.

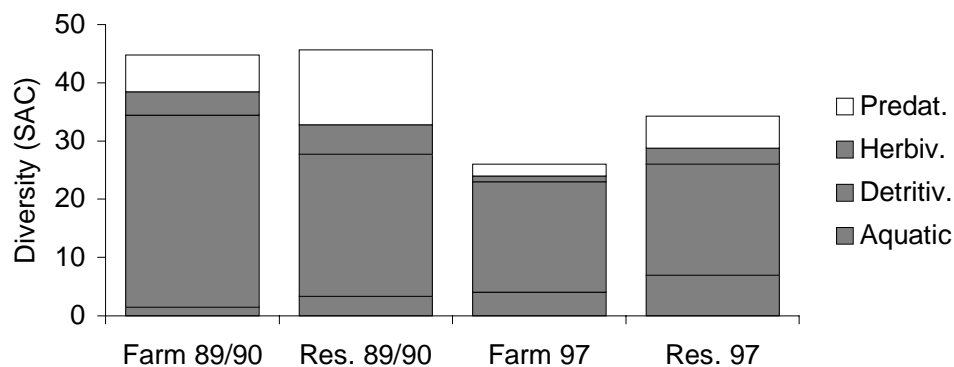
Comparison of figures 5a and 5b shows that species in all trophic groups were represented by relatively low numbers of individuals. However, some variation in these ratios was noticeable in different trophic groups. In particular, the predatory group in the early reserve samples consisted of many individuals from much lower numbers of species. Also, although numbers of aquatic individuals were similar for both sites in 1997, fewer species occurred in the farm site.

4.5 FUNCTIONAL DIVERSITY OF COMMUNITIES

Functional diversity was derived from summed abundance classes (SAC) of the trophic components in an average catch of the catch groups defined by TWINSpan (fig. 6).

Figure 6 combines several aspects of previous analyses of the data to give summaries of the communities from the two habitat types. These reveal that overall diversity was higher for the reserve samples than for the farm samples for both the early and the later series. The earlier analyses, however, showed that this was due to higher abundance in the earlier samples, and greater species richness in the later samples. The greater dominance of the detritivores in the early farm samples is clearly apparent, as are the changes in these and other groups by 1997.

Figure 6. Average functional (trophic) diversity of Malaise trapped beetle catch groups defined by TWINSpan.



4.6 DOMINANT SPECIES

Species which occurred as 5 or more specimens are shown in table 2. Life histories of these species reveal the resources enhancing the major population increases in the species present in the habitats in greater detail. These in turn can indicate the major processes shaping the insect communities. Much more could be learned of the inter-relationships between ecosystems and their insect communities if more were known about the life histories of the species involved.

TABLE 2. DOMINANT SPECIES AND THEIR ABUNDANCE CLASSES IN THE CATCH GROUPS DEFINED BY TWINSpan.

FAMILY	SPECIES	TROPHIC GROUP	FARM 89/90	RES. 89/90	FARM 97	RES. 97
Cantharidae	<i>Asilis</i> sp-01	P		3		
Carabidae	<i>Ctenognathus adamsi</i>	P		5		
Corticariidae	<i>Melanophtalma zealandica</i>	D	3			
Corylophidae	<i>Sacina oblonga</i>	D	4		3	3
Cryptophagidae	<i>Atomaria lewisi</i>	D	3	4		
Cryptophagidae	<i>Micrambina helmsi</i>	D	3	4	3	
Cryptophagidae	<i>Micrambina insignis</i>	D	4	3		
Curculionidae	<i>Microcryptorhynchus</i> spp.	D		3		
Curculionidae	<i>Pbloeophagasoma thoracium</i>	D	3			
Leiodidae	<i>Colon hirtale</i>	D		3		
Leiodidae	<i>Paracatops phyllobius</i>	D		3		
Scirtidae	<i>Amplectopus</i> spp.	A				3
Scirtidae	<i>Cyphon genalis</i>	A		4		
Scirtidae	<i>Cyphon</i> sp-08	A	4			
Scirtidae	<i>Cyphon</i> sp-46	A		3		
Scirtidae	<i>Mesocyphon</i> sp-10	A			4	3
Scraptiidae	<i>Nothotelus usitatus</i>	D	3			
Staphylinidae	<i>Aleocharinae, Falagria</i> sp-71	P		4		
Staphylinidae	<i>Atheta atriceps</i>	P		4		
Staphylinidae	<i>Tramiatbaea cornigera</i>	P	3	4		

Abundance classes: 3 = 5-9 specimens, 4 = 10-19 specimens and 5 = 20+ specimens.

Trophic groups: A = aquatic species, D = Detritivores, P = predatory.

5. Conclusions

Species richness, abundance and diversity were higher in the reserve than the farm site in 1997. Although overall numbers in samples from both sites were lower than from the two earlier sampling series, this probably reflects a stabilisation of the degeneration and mammal control processes occurring in the grazed and reserve habitats respectively. Mid-level woody plant species in the farm plot decreased over the study period and were not replaced because of the understorey grazing. Few remained by 1997, resulting in a smaller amount of woody debris resource for detritivore beetles to utilise. Lower numbers of beetles in the reserve in 1997 were due to a decrease in the abundance of beetle species thought to be associated with mammals and their carcasses, and these were no longer present in the reserve by the end of the study. Self-thinning of the regeneration resulting from the removal of grazing animals from the reserve had not yet begun to occur by 1997.

It is now apparent from this and other studies (Hutcheson 1990, 1996, Hutcheson & Kimberley 1999, Hutcheson & Jones 1999, J. Hutcheson unpubl. data) that in addition to insect communities being demonstrably characteristic for various vegetation types, they are also exceedingly dynamic. Component species and their abundances are constantly changing in response to the quantity and quality of the various resources available (Hutcheson 1991b, White 1993). These resources are associated not just with the plant species present, but with all the processes occurring within the systems. The integration of the insects with all aspects of terrestrial ecosystem status and dynamics shows why insect communities are able to convey such a large amount of information about the responses of ecosystems to external influences such as management.

The study shows that 'species richness' or other single figure indices should not be the only criteria used in the evaluation of ecosystem biodiversity. While single 'one off' studies of insect communities may provide a 'snapshot' of the current status of the community ecology of an ecosystem, a broader understanding is required to enable meaningful interpretations for extrapolation over time and space.

As the interpretations derived here fit into an understanding of community ecology which is based on wider ecological principles (e.g. resource-based or 'bottom up' controls), the study provides further confidence in the methodology applied. It is through the application of a standardised approach to characterising insect communities that it is possible to compare them and to obtain meaningful interpretations of the interrelationships between system management and the group of organisms that forms 'the heart of biodiversity'.

Implications for the kokako recovery plan are derived more from the general understanding of community ecology to which this study contributes, than from the immediate data. These relate to the nature of adjacent land use as well as to the reserve itself. Habitat disturbance within reserves due to environmental influence (e.g. wind blow) should not be regarded as having negative effects on overall insect biodiversity, **provided the forest system is retained**. In effect, such disturbances 'provide jobs' for the insects involved in recycling. The removal of regeneration, and therefore the future forest, by grazing and browsing mammals is of far greater long-term importance.

Even degenerating forest remnants adjacent to kokako habitat may thus provide valuable food resources for the birds through a long breakdown period. Although larvae of many of the insect species within debris may not be directly available to kokako, others species associated with the general processes (e.g. fungal-feeding or predatory species) may be accessible and utilised by the birds.

The retention of high insect endemicity within the grazed native bush area suggests that, despite their damaged condition, such remnants have high potential to act as nuclei for the spread of endemic biodiversity into adjacent areas where land-use regimes provide appropriate habitat.

Application of this approach to a broader series of management questions has the potential to contribute strongly to the problem of how best to sustain the main source of New Zealand's endemic biodiversity through appropriate land management across both the conservation and the production estates in New Zealand. Such a program would also provide a very necessary focus for systematic and autecological studies by the currently extremely limited and under-resourced entomological community in this country. The interpretations made in this report were made possible by the use of standardised methodology (including taxonomy), knowledge of the life histories of insect species captured, and a long period studying New Zealand ecosystem dynamics in detail. Only about half of our insect species are presently described, and therefore a cost-effective program is dependent on the redevelopment of widespread collaboration within the New Zealand research community (Hutcherson 1997).

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