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by

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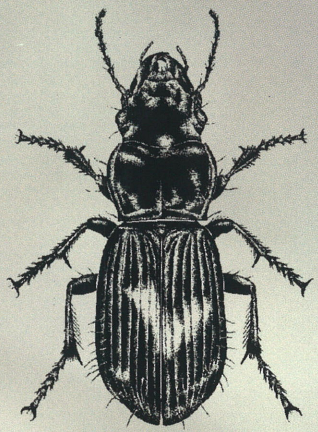

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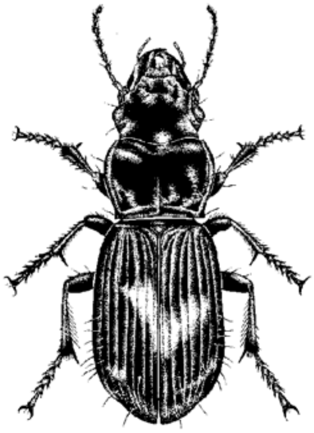

by

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TABLE OF CONTENTS

ABSTRACT

Ecological restoration in New Zealand has an emphasis on the islands due to its feasibility of mammal control. As a refuge of local, rare and endangered species, ecological restoration has been undertaken on Quail Island for 19 years from 1998. To evaluate the response in biodiversity, the invertebrate community was used as a bio-indicator to assess the restoration success. In this study, we examined the change in the terrestrial invertebrate community by pitfall traps across five different habitats including exotic grassland, restoration plantings of two ages, mixed shrubland, and pine and macrocarpa woodland. Species diversity was tested by Shannon index, Simpson's index, beetle richness and mite richness. We used the general linear model to examine the change in time and identify the influence of environmental variables. Species-level association with habitat structure was tested by pairwise comparison. We saw an apparent increase in Shannon index, Simpson's index and beetle richness, while mite richness fluctuated. Habitat differences illustrated the species preference for habitat structure. Restoration trajectories indicated a promising recovery of the invertebrate community, especially for cave weta (*Pleioplectron simplex*) and ground weta (*Hemiandrus* n. sp.). The catch of *Megadromus guerinii*, the first Bank Peninsula endemic found on Quail Island, showed its potential to be the suitable habitat for local species. The analysis of the species abundance with the environmental factors indicated their requirement of physical characteristics. The comprehensive results of 19 years restoration revealed the current state of biodiversity that contribute to the future restoration plan. Although the nature of long-term monitoring and methodologies used raised several uncertainties and concerns of the results, continuous monitoring is recommended to ensure the succession of the ecological community is under control to reach the final goal.

1. INTRODUCTION

Ecosystem collapse happened worldwide due to proliferation of human-modified landscape. Habitats in various degree of degradation lead to the loss of biodiversity. Restoration has been a primary conservation initiative to tackle the challenge. Due to early colonial history and the catastrophic effect of mammalian predator, definable boundary of ecological restoration on island is readily substantial to its effectiveness. Endangered species management focus on island further contributed to the establishment of high profile landmarks (Waterhouse, 1991). With the potential of island restoration to biodiversity conservation (Atkinson, 1988), the value of island restoration has been widely accepted in New Zealand.

Quail Island (Ōtamahua) located in Lyttelton Harbour, east of Christchurch, New Zealand. At 81 ha, is Canterbury's largest island. The majority of original vegetation was removed before European arrival, and with only a few original species remaining, the ecological communities are threatened (Jackson et al., 2006). As a Recreational Reserve managed by the Department of Conservation, ecological restoration has been undertaken by the Quail Island Ecological Restoration Trust in partnership with Te Hapu o Ngāti Wheke of Rāpaki.

To reverse the dominant exotic vegetation to native flora, initial planting programme began in the early 1980s. However, only a few 1983 plantings survived due to the browsing mainly by rabbits, although it showed a strategic restoration with pest eradication could succeed. Meurk (1990) conceived the idea to ecologically restore Quail Island and the first restoration plan arose from an unpublished Master's thesis by Ray Genet (Jackson et al., 2006). As restoration is a controlled succession process, a clear objective is required to measure the performance. Currently, the purpose of the restoration programme is to smooth the path for indigenous flora and fauna and provide a sanctuary for locally extinct, rare and endangered species of Bank Peninsula region. With a combination of restoration planting, weed control and pest control, the ecological restoration has gradually changed the habitat and inhabitants.

To assess the restoration programme, recognizing current state and ecosystem dynamics is of importance to identify the transition between restoration states. Plant-related restoration work has also been recorded in 2010 (Burrows et al., 2011) and the results were evaluated as promising to restructure an indigenous system for native inhabitants. The inventory of the invertebrate fauna has been investigated to provide basis information for the further introduction and habitat requirement (Bowie, 2001; Bowie et al. 2003). The success of mammal eradication (Bowie, 2008; Bowie et al., 2010) have been well recorded, however, the integrated response in biodiversity is less documented, especially for invertebrates (Bellingham et al., 2010).

Simply adding native planting will not necessarily bring back native fauna. To ensure that the ecological restoration leads to desired biodiversity gain, a long-term monitoring programme with a control site as reference is essential to verify the species recovery. It can not only provide feedback but also quantify the recovery of flora and fauna for continuous improvement. However, a thorough monitoring programme for all the biotic and abiotic factors to evaluate the whole ecosystem is nearly impossible. Therefore, a comprehensive consideration is necessary for the selection of a representative bio-indicator.

Invertebrates are critical component of terrestrial biodiversity and play a key role in ecosystem services (McGeoch, 1998). Their abundance, sensitivity to environmental change, fast generation times and performance in a wide range of ecological functions and services make them ideal for measuring ecological changes and are commonly used as indicators in monitoring biodiversity or ecological processes (Lindenmayer et al. 2015; Ruiz-Jaen & Mitchell Aide, 2005). Those characteristics also allow invertebrates to indicate different restoration objectives (Lindenmayer et al., 2015). In addition, island invertebrates often show high level of endemism, which disproportionately contribute to global biodiversity (St Clair et al., 2011). A high proportion of endemic species, especially invertebrates, has been found in Bank Peninsula due to its geographical isolation for nearly 20-million-year (Wilson, 2013). As a refuge of local flora and fauna, using invertebrate as a bio-indicator is congruous with the restoration objective of Quail Island.

In this study, we analysed the change of terrestrial invertebrate fauna using pitfall traps in five habitat types on Quail Island across 19 years including: exotic grassland, restoration plantings of two ages, mixed shrubland, and pine and macrocarpa woodland. We include analysis of habitat structure parameters based on field investigation and data collection of environmental factors such as the percentage of canopy cover and soil moisture. Our aims were to: (1) investigate the influence of ecological restoration on invertebrates in terms of species richness, species abundance and species composition; (2) further examine species-level association within habitat types; (3) record the restoration trajectory for future management reference; and (4) understand impact of environmental factors on invertebrate communities.

2. METHODS

2.1 STUDY SITE

The study was carried out over the southern half of the island in five habitat types: Introduced grassland (G); Restoration plantings undertaken in 1998 (R(1998)) (Burrows & Wilson, 2011); Restoration plantings undertaken in 1983 (R(1983)); Original native patches of Shrubland/regenerating scrubland (S) consisting mainly of kanuka (*Kunzea robusta* de Lange & Toelken), bracken (*Pteridium esculentum* (G.Forst.) Cockayne), (*Coprosma crassifolia* Colenso) and (*C. propinqua* A. Cunn.); and pine (*Pinus radiata* D. Don) and macrocarpa (*Cupressus macrocarpa* Hartweg) patches (PM). Pitfall traps were used in all five habitats, with six replicates for each site. To avoid random variation, replicates fairly well distributed in each habitat considering its variable nature. Six Grassland sites were distributed at the west part of Quail Island over the southern half from 1999 to 2002. Due to the restoration progress, grassland replicates had been relocated to the east part near the old jetty in 2002 (G4 and G5) and 2012 (G3 and G6). R replicates located along a transect across the centre of restoration site. Since R(1983) restoration patches are relatively small and fragmented, two sites are near the coast and the other four sites are on the higher hill.

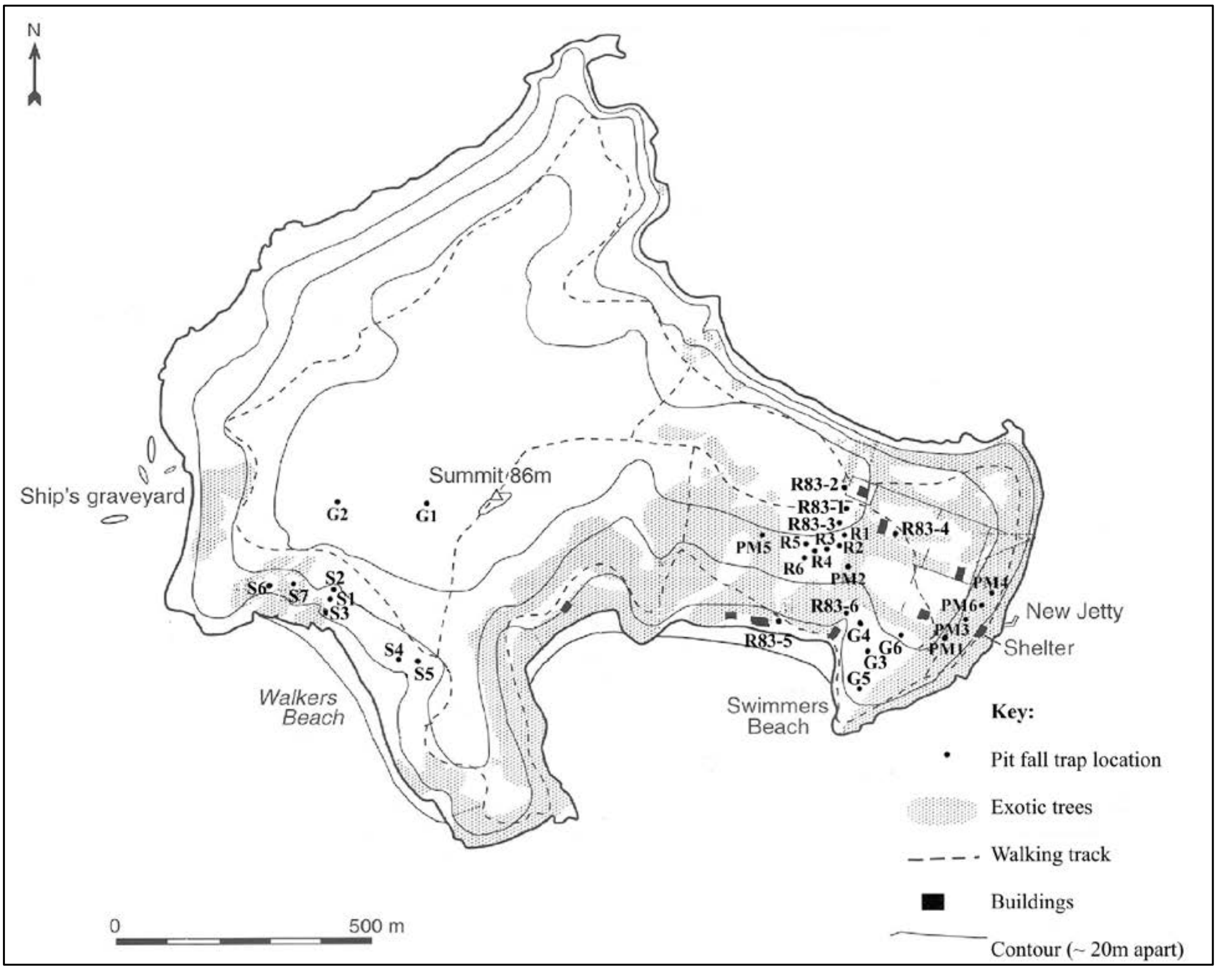


Figure 1. Location of pitfall traps on Quail Island. G=grassland pitfalls; R=pitfalls in restoration planted in 1998; R(1983)= pitfalls in restoration planted in 1983; PM=pitfalls in pine & macrocarpa patches; S= pitfalls in scrubland areas.

Replicates of Shrubland were mainly spread on the west side. Pine/Macro traps were set up around the middle of the plantations in the east and middle part of the island. All the replicates were located at least 10m from each other to minimise the spatial effects.

2.2 Pitfall trap Sampling method

Pitfalls used for analysis from 1999 to 2018, are shown below (Table 1). Years 2007 and 2013-2017 were omitted from the analysis as there were too many missing replicates. Trapping took place for approximately one month over the summer period.

Table 1. Periods of pitfall traps put out on Quail Island, trap dates and total traps collected.

Year	Period	Trap date	G	R	R(1983)	S	PM	Total
1999	07/12/1998-07/01-1999	31	4	4	4	4	4	20
2000	08/12/1999-11/01/2000	34	6	6	6	6	6	30
2001	13/12/2000-11/01/2001	29	6	6	6	6	6	30
2002	13/12/2001-11/01/2002	29	6	6	6	6	6	30
2003	18/12/2002-16/01/2003	29	6	6	6	6	6	30
2004	08/12/2003-06/01/2004	29	6	6	6	6	6	30
2005	16/12/2004-16/01/2005	31	6	6	6	6	6	30
2006	13/12/2005-09/01/2006	27	4	3	5	6	4	22
2008	12/12/2007-11/01/2008	30	3	5	6	6	6	26
2009	18/12/2008-19/01/2009	32	6	6	6	6	6	30
2010	14/12/2009-14/01/2010	31	5	5	6	6	6	28
2011	21/12/2010-18/01/2011	28	6	6	6	5	6	29
2012	17/12/2011-20/01/2012	34	5	6	5	5	4	25
2018	11/12/2017-12/01/2018	32	6	6	6	6	6	30

Pitfall traps were constructed from 350 ml honey pots (# NA6628, Stowers) with a collecting diameter of 8 cm and were filled with c. 100 ml of monopropylene glycol (antifreeze) to preserve specimens and c. 0.5 ml of detergent to reduce the surface tension. To prevent rain and plant debris from filling the pitfalls, each trap was covered with galvanised metal roofs 20 x 20 cm supported by four wire legs. The catches were sorted by clearing plant and soil debris and the remained dead organisms were preserved in 70% alcohol.

The samples were sorted under a binocular microscope. Specimens were counted into known species or recognisable taxonomic units (Appendix). In some taxa where they were extremely abundant and difficult to identify. Flies, for example, were divided into craneflies (Tipulidae), Phoridae, large Diptera (Tachnidae, Muscidae and Calliphoridae), while the remainder were combined. Insect larvae separated into Lepidoptera and Coleoptera (mainly Scarabidae and Carabidae). In smaller beetles (except for those had already listed in Appendix 1) and mites, only their presence was recorded rather than their abundance for 2000, 2005, 2010 and 2018 only.

2.3 Environmental factors data collection

Site characteristics were surveyed on 11 December 2017 and 12 January 2018, including soil moisture, litter depth, canopy cover, vegetation cover and inclination. Soil moisture and litter depth were randomly sampled around the trap for two times within a radius of 0.5 m. Vegetation cover, including the percentage of litter cover and percentage of woody debris, were recorded by visual estimation within a radius of 5 m. Canopy cover was recorded by visual estimation, using photographs taken straight upward from 15 cm above the pitfall traps. Two types of inclination are measured, which are the general slope and local slope. The measurement of general slope considered the average slope around the trap in a radius of 10 m which measured by SUUNTO clinometer. A radius of 1 m was applied on the measurement of the local slope by the metal protractor with bubble level fitted.

2.4 Data analysis

All statistical analysis was performed using the free software environment R 3.4.3 (R Core Team, 2017).

2.4.1 Species diversity

Simpson and Shannon index were used to evaluate the species diversity among habitats. General linear model was applied to examine the trends in time using Grassland as a reference habitat. Year, the quadratic effect of year, habitat and interaction between year and habitat were considered in the full model. AIC is used to choose the fittest model, but the year and habitat variables need to be included in the model as minimum factors. Therefore, the effect of interaction between year and habitat only included in the model of Shannon index.

Beetles and mites richness were calculated by the abundance of different beetles and mites which were present in habitats in 2000, 2005, 2010 and 2018. Trends in time are tested by using general linear model including two factors, which are year and habitat, with poisson distribution and Grassland as reference habitat. Pairwise.t.test (with adjusted p-value by Bonferroni method) and TukeyHSD function were used to examine the pairwise difference among habitats.

2.4.2 Habitat difference

Pairwise comparisons of species abundance among habitats were done by pairwise.t.test (with adjusted p-value by Bonferroni method) and TukeyHSD function. We saw environmental variables collected in 2018 as features of each habitat since those sites did not change except for G3 to G6 in Grassland.

2.4.3 Restoration trajectories

Restoration trajectory was analysed by comparing the change of each species in R and R(1983) to Grassland using general linear model with poisson distribution. Variables include year, the quadratic effect of year, habitat and interaction between year and habitat. Stepwise elimination was used to find the best model for each species by checking AIC. The minimum of variables is year and habitats.

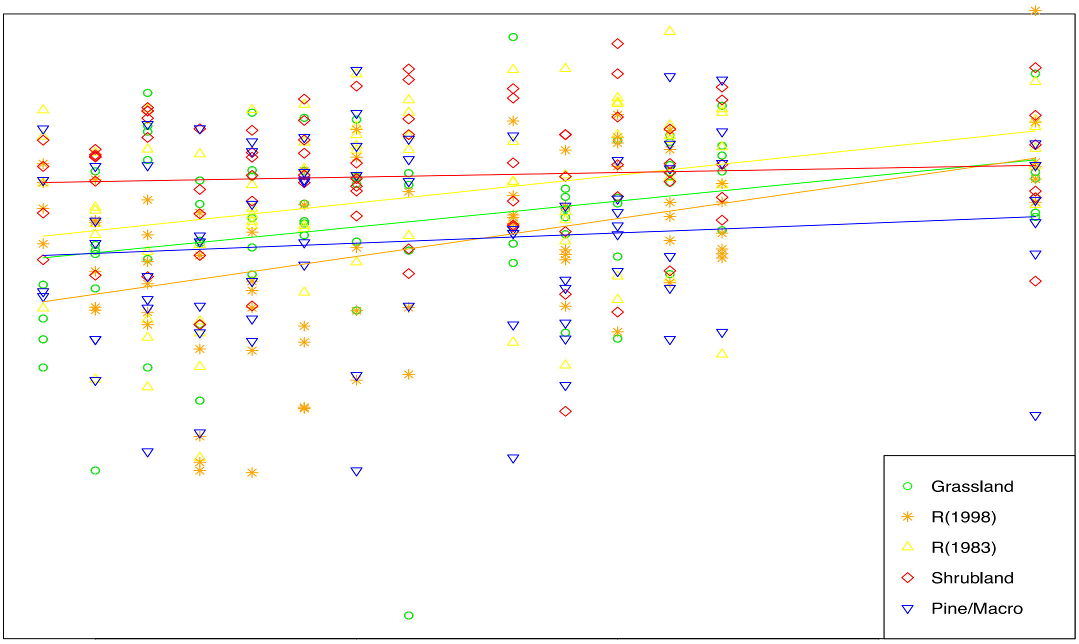
Environmental factors collected in 2018 were used to form an overall understanding of habitat features since no significant change related to those habitats except for the unexpected move of traps in Grassland.

2.4.4 The influence of environmental factors

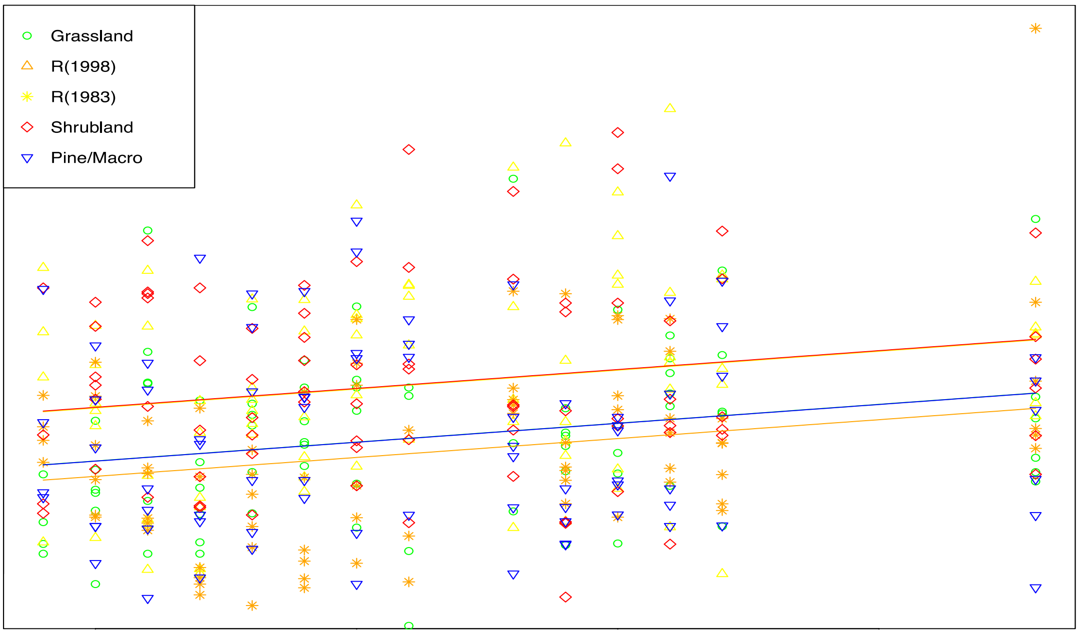
Data collected on 11 December 2017 and 12 January 2018 have been averaged and the mean of each habitat had been calculated. Correlations between each factor were tested. The influence of environmental factors was analysed by general linear mixed effect modelling approach with poisson distribution. Variables include habitat, soil moisture, litter depth, the percentage of litter cover, the percentage of canopy cover, the percentage of woody debris, general slope and local slope. We used the full model to start and find the best model for each species applying stepwise elimination by dropping the least significant variable until all the remaining variables are significant ($p<0.05$). Given that the data only across the trapping period in 2018, data were only included to assess the influence on the abundance of species trapped in 2018 instead of the overall trend analysis across years. However, data were used as references of the habitat characteristics for analysis since no significant change related to those habitats except for the unexpected move of traps in Grassland.

3. Results

3.1 Species diversity



(A)



(B)

Figure 2. (A) Trends of Shannon index among habitats. (B) Trends of Simpsons index among habitats.

The factors, the year and the habitats, best explained Simpsons index, whereas the addition of the quadratic effect of year and the interactions between the year and the habitats better explained the change in Simpsons index. Increasing trends showed in both Shannon and Simpsons index among all the habitats (Figure 2). They were all significant (p<0.05) since the increases of both index in Grassland, which was the reference site, were significant (p<0.01 and p<0.001) and other habitats, except for Simpsons index in R(1993) and S, were not significantly different (p>0.05) from it (Table 2). Lower Simpsons index in R(1993) and higher Simpsons index in Shrubland were significant (p<0.001) compared to the index in Grassland (Table 2).

Table 2. The result of standard error and p-value in examination of Shannon index, Simpsons index, beetles diversity and mites diversity from GLM model including year and treatment as factors

	Shannon index		**Simpsons index**		**Beetles Richness**		**Mites Richness**	
Factors	Std. Error	Pr(>|t|)	Std. Error	Pr(>|t|)	Std. Error	Pr(>|z|)	Std. Error	Pr(>|z|)
(Intercept)	32.3265	0.0071	35.0933	0.0001	5601.9579	0.0006	5071.0186	0.0624
Year	0.0161	0.0052	0.0175	0.0000	5.5741	0.0006	5.0462	0.0634
I(Year^2)	NA	NA	NA	NA	0.0014	0.0006	0.0013	0.0645
R(1998)	45.5165	0.3513	0.2898	0.3115	0.1609	0.0012	0.1397	0.0456
R(1983)	45.5996	0.8885	0.2871	0.0004	0.1433	0.2501	0.1320	0.3692
Shrubland	45.7152	0.1000	0.2871	0.0004	0.1449	0.1515	0.1350	0.1324
Pine/Macro	45.7735	0.2317	0.2888	0.9935	0.1495	0.0301	0.1393	0.0228
Year: R(1998)	0.0227	0.3539	NA	NA	NA	NA	NA	NA
Year: R(1983)	0.0227	0.8848	NA	NA	NA	NA	NA	NA
Year: Shrubland	0.0228	0.1017	NA	NA	NA	NA	NA	NA
Year: Pine/Macro	0.0228	0.2302	NA	NA	NA	NA	NA	NA

The pairwise comparison of both index by TukeyHSD and pairwise.t.test showed the same results in terms of significance (p<0.05) difference (Table 3). In Shannon index, the number was significantly (p<0.05) higher in Shrubland compared to other habitats except for R(1983). The index of R(1983) was significantly higher (p<0.01) than R and Pine/Macro. On the other hand, R(1983) and Shrubland performed significantly better (p<0.01) than other three habitats in Simpsons index. The mean of both index in Shrubland was higher than in R(1983), but the difference was not significant(p>0.05).

Table 3. Habitat pairwise comparison of Shannon and Simpsons index by TukeyHSD and Pariwise.t.test.

	TukeyHSD				**Pairwise.t.test**	
	Shannon index		Simpsons index		Shannon index	Simpsons index
Habitat pairwise comparison	Difference in mean levels	adjusted p-value	Difference in mean levels	adjusted p-value	adjusted p-value	adjusted p-value
R(1998)-R(1983)	-0.44	0.00282	-1.31	0.00009	0.00306	0.00009
Shrubland- R(1983)	0.17	0.62425	0.01	1.00000	1.00000	1.00000
Grassland- R(1983)	-0.22	0.37790	-1.02	0.00479	0.73425	0.00529
Pine/Macro- R(1983)	-0.40	0.01009	-1.02	0.00425	0.01152	0.00469
Shrubland- R(1998)	0.61	0.00001	1.32	0.00008	0.00001	0.00008
Grassland-R(1998)	0.22	0.37128	0.28	0.87150	0.71628	1.00000
Pine/Macro-R(1998)	0.05	0.99573	0.29	0.86664	1.00000	1.00000
Grassland-Shrubland	-0.39	0.01380	-1.03	0.00436	0.01602	0.00481
Pine/Macro-Shrubland	-0.57	0.00004	-1.03	0.00387	0.00004	0.00425
Pine/Macro-Grassland	-0.18	0.60322	0.00	1.00000	1.00000	1.00000

60 different beetles and 56 different mites were identified from the samples. Beetle richness in habitats each year were considerably growing, with all the numbers reached the peak in 2018, except for the number in Pine/Macro (Figure 3. (A)). In Table 2, the increasing trend in Grassland was proved to be significant ($p<0.001$), and the other habitats, except for R and Pine/Macro, showed the same growth due to no significant difference ($p>0.05$) had been detected. The beetle richness in R had been discovered to be significantly lower ($p<0.01$) than Grassland. In contrast to beetle richness, mite richness fluctuated during this period. Remarkably high richness presented in 2005 with a drop in 2010 (Figure 3. (B)). The increasing trend was not significant ($p>0.05$) when considering both year and habitat factors (Table 2). The same phenomenon between beetle and mite richness was only the richness in Pine/Macro did not reach the peak in 2018. In addition, no significant difference ($p>0.05$) had been found under pairwise comparison in both beetle and mite richness.

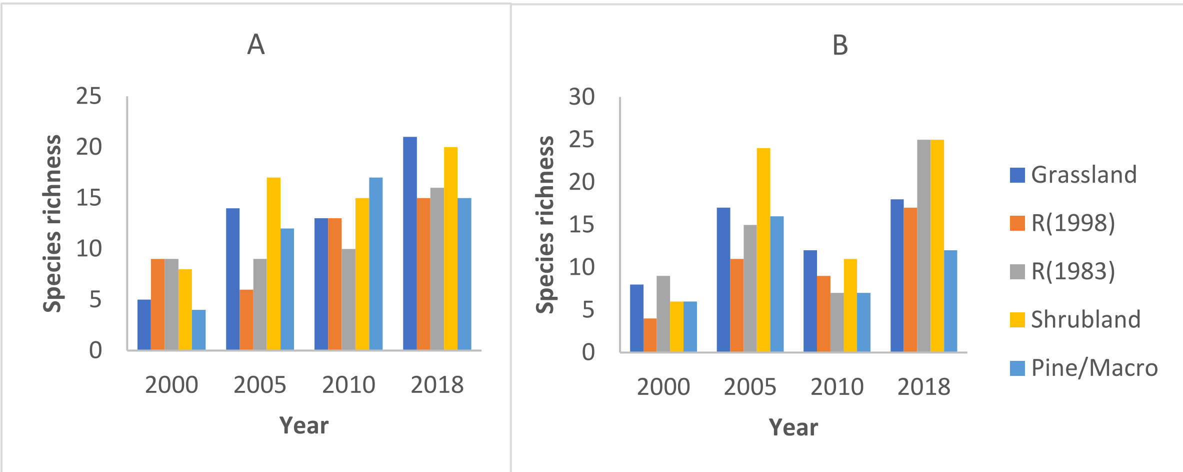


Figure 3. Species richness in 2000, 2005, 2010 and 2018 among five habitats for (A) Beetles and (B) Mites

3.2 Habitat differences

The results of Tukey HSD (Table 4) and pairwise.t.test were similar in terms of significance in all the pair comparison except for the comparison of the abundance of cockroach in Shrubland and R(1983) which was only significant ($p<0.05$) under Tukey HSD test.

3.2.1 Grassland species

The abundance of ants and thrips were significantly higher ($p<0.05$) in Grassland compared to other habitats, except for Shrubland. The number of *Baeus* spp. (Hymenoptera) caught in Grassland was higher, but no significant difference ($p>0.05$) was found compared with the number in R. Although cockroach, moths, other spiders and phorids were significantly more ($p<0.05$) abundant in Grassland, they were not only significantly more ($p<0.05$) in one habitat.

3.2.2 Restoration (1998) species

Cave weta (*Pleioplectron simplex*) had a similar abundance in R compared to R(1983) and Shrubland, but it was significantly higher ($p<0.05$) than the abundance in Grassland and Pine/Macro. In addition, the number of earthworms counted was significantly higher ($p<0.05$) in R except for Pine/Macro. Scarab was found to be significantly less ($p<0.05$) in R compared to Grassland and R(1983).

3.2.3 Restoration (1983) species

The number of snails caught was significantly higher ($p<0.05$) in R(1983) compared to other habitats. Though caterpillars were also found more in R(1983), the difference was not significant ($p>0.05$) when compared to the number in Shrubland. In addition to the highest abundance found in Pine/Macro, *Metaglymma moniliferum* also showed significantly higher ($p<0.05$) abundance in R(1983) compared to Grassland, R and Shrubland.

3.2.4 Shrubland species

The abundance of European harvestman (*Phalangium opilio*), *Holcaspis* and *Mimopeus opaculus* were significantly higher ($p<0.05$) in Shrubland among habitats. Though craneflies and wasps were significantly more abundant ($p<0.001$) in Shrubland, the numbers were not significantly high ($p>0.05$) compared to Grassland. By contrast, the count number of larger flies in Shrubland were only not significantly more ($p>0.05$) than the count number in R(1983). Besides, the abundance of cockroach and true bugs were significantly higher ($p<0.05$) in Shrubland compared to R(1983) and Pine/Macro.

3.2.5 Pine/Macro species

Bristletails (*Nesomachilis* sp.) were found more in Pine/Macro than any other habitats. While the millipede and pseudoscorpion abundance were both primarily higher in Pine/Macro, their abundance was not significantly higher ($p>0.05$) compared to Shrubland. The most common ground beetle, *Metaglymma moniliferum*, were recorded more frequently in Pine/Macro than Grassland, R and Shrubland. In contrast, ground weta (*Hemiandrus* n. sp) was rarely found in Pine/Macro than other habitats apart from Grassland.

Table 4. The result of pairwise comparison of species in each habitats by TukeyHSD.

	Ants		***Baeus***		**Bristletails**		**Caterpillars**		**Cave weta**		**Cockroach**	
	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value
R(1998) - Grassland	-2.52	0.0013	-2.04	0.0821	0.06	0.9978	-0.15	0.8877	1.88	0.0159	-0.93	0.0000
R(1983) - Grassland	-2.78	0.0002	-2.83	0.0037	0.31	0.5418	0.47	0.0341	1.06	0.3836	-0.82	0.0000
Shrubland - Grassland	-1.20	0.3505	-2.61	0.0094	0.51	0.0897	0.17	0.8282	1.42	0.1184	-0.39	0.1200
Pine/Macro - Grassland	-2.65	0.0006	-3.89	0.0000	1.35	0.0000	-0.22	0.6705	-0.04	1.0000	-0.97	0.0000
R(1983) - R(1998)	-0.27	0.9939	-0.79	0.8556	0.25	0.7383	0.63	0.0014	-0.82	0.6363	0.11	0.9590
Shrubland - R(1998)	1.32	0.2459	-0.57	0.9496	0.45	0.1781	0.33	0.2665	-0.45	0.9385	0.55	0.0064
Pine/Macro - R(1998)	-0.13	0.9996	-1.85	0.1375	1.28	0.0000	-0.07	0.9941	-1.91	0.0119	-0.04	0.9992
Shrubland - R(1983)	1.59	0.0965	0.21	0.9988	0.20	0.8566	-0.30	0.3458	0.36	0.9717	0.44	0.0488
Pine/Macro - R(1983)	0.13	0.9996	-1.06	0.6598	1.03	0.0000	-0.70	0.0002	-1.09	0.3389	-0.15	0.8832
Pine/Macro - Shrubland	-1.45	0.1598	-1.27	0.4847	0.83	0.0004	-0.40	0.1114	-1.46	0.0971	-0.59	0.0026
	Craneflies /tipulids		**Earthworm**		**European harvestman**		**Ground weta**		***Holcaspis***		**Larger flies**	
	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value
R - Grassland	-0.37	0.1252	0.17	0.0248	-0.01	0.9873	1.71	0.0623	0.04	0.9829	-0.37	0.9360
R(1983) - Grassland	-0.34	0.1944	-0.04	0.9395	-0.03	0.8620	1.13	0.3873	0.00	1.0000	0.60	0.7077
Shrubland - Grassland	0.34	0.2029	-0.04	0.9395	0.09	0.0141	2.38	0.0019	0.25	0.0037	1.56	0.0083
Pine/Macro - Grassland	-0.31	0.2793	0.04	0.9691	-0.03	0.8646	-0.78	0.7400	-0.04	0.9789	-0.39	0.9246
R(1983) - R(1998)	0.04	0.9994	-0.21	0.0015	-0.01	0.9889	-0.58	0.8930	-0.04	0.9770	0.96	0.2328
Shrubland - R(1998)	0.71	0.0001	-0.21	0.0015	0.10	0.0023	0.68	0.8227	0.21	0.0210	1.93	0.0004
Pine/Macro - R(1998)	0.06	0.9943	-0.13	0.1239	-0.01	0.9891	-2.49	0.0010	-0.08	0.7938	-0.02	1.0000
Shrubland - R(1983)	0.68	0.0001	0.00	1.0000	0.11	0.0003	1.25	0.2691	0.25	0.0026	0.96	0.2247
Pine/Macro - R(1983)	0.03	0.9998	0.08	0.6134	0.00	1.0000	-1.91	0.0214	-0.04	0.9824	-0.98	0.2141
Pine/Macro - Shrubland	-0.65	0.0004	0.08	0.6134	-0.11	0.0003	-3.16	0.0000	-0.29	0.0003	-1.94	0.0003

	Metaglymma moniliferum		Millipedes		*Mimopeus opaculus*		Moths		Other spiders		Phorids	
	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value
R(1998) - Grassland	-0.16	0.9257	-0.29	0.9833	0.00	1.0000	-1.49	0.0000	-1.61	0.7594	-1.38	0.8903
R(1983) - Grassland	0.59	0.0149	-0.08	0.9999	0.00	1.0000	-0.04	0.9999	-6.46	0.0000	-6.74	0.0001
Shrubland - Grassland	-0.11	0.9811	0.30	0.9802	0.71	0.0004	-1.02	0.0100	-8.83	0.0000	-6.01	0.0006
Pine/Macro - Grassland	0.61	0.0124	1.72	0.0121	0.01	1.0000	-1.29	0.0004	-5.83	0.0002	-9.54	0.0000
R(1983) - R(1998)	0.75	0.0007	0.21	0.9944	0.00	1.0000	1.45	0.0000	-4.85	0.0030	-5.36	0.0030
Shrubland - R(1998)	0.05	0.9989	0.59	0.7990	0.71	0.0003	0.47	0.5426	-7.22	0.0000	-4.64	0.0156
Pine/Macro - R(1998)	0.77	0.0006	2.01	0.0017	0.01	1.0000	0.20	0.9676	-4.22	0.0158	-8.16	0.0000
Shrubland - R(1983)	-0.70	0.0017	0.37	0.9524	0.71	0.0003	-0.98	0.0132	-2.38	0.3784	0.72	0.9877
Pine/Macro - R(1983)	0.01	1.0000	1.80	0.0063	0.01	1.0000	-1.25	0.0006	0.63	0.9898	-2.80	0.3175
Pine/Macro - Shrubland	0.71	0.0014	1.42	0.0555	-0.70	0.0004	-0.27	0.9020	3.00	0.1619	-3.53	0.1188

	Pseudoscorpions		Scarab adults (beetles)		Snails		Thrips		True bugs		Wasps	
	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value	Difference in mean	adjusted p-value
R(1998) - Grassland	0.00	1.0000	-0.60	0.0088	0.07	0.9890	-0.27	0.0138	-2.50	0.1518	-2.07	0.4402
R(1983) - Grassland	0.26	0.6935	0.03	0.9998	0.59	0.0009	-0.32	0.0014	-4.61	0.0002	-2.73	0.1603
Shrubland - Grassland	0.42	0.2180	-0.31	0.4291	-0.04	0.9990	-0.12	0.5997	0.41	0.9956	2.80	0.1420
Pine/Macro - Grassland	0.74	0.0024	0.05	0.9989	0.15	0.8677	-0.27	0.0130	-5.22	0.0000	-3.52	0.0328
R(1983) - R(1998)	0.26	0.6837	0.63	0.0042	0.51	0.0051	-0.05	0.9700	-2.12	0.2829	-0.67	0.9811
Shrubland - R(1998)	0.42	0.2091	0.30	0.4679	-0.11	0.9448	0.15	0.3897	2.91	0.0551	4.87	0.0006
Pine/Macro - R(1998)	0.74	0.0021	0.65	0.0031	0.07	0.9881	0.00	1.0000	-2.72	0.0896	-1.46	0.7496
Shrubland - R(1983)	0.16	0.9224	-0.34	0.3178	-0.63	0.0002	0.20	0.1098	5.02	0.0000	5.54	0.0000
Pine/Macro - R(1983)	0.48	0.1076	0.02	1.0000	-0.44	0.0246	0.05	0.9714	-0.60	0.9804	-0.79	0.9654
Pine/Macro - Shrubland	0.32	0.4878	0.36	0.2680	0.18	0.7233	-0.15	0.3803	-5.63	0.0000	-6.32	0.0000

3.3 Restoration trajectories

In this section, we compared the trend of species abundance in R(1998) and R(1983) to Grassland (Figure 4).

3.3.1 Restoration (1998) compared to Grassland

In R, the number of ants, cave weta (*Pleioplectron simplex*), cockroach, large craneflies, larger flies, ground weta, millipedes, native harvestman, pseudoscorpion, rove beetles, scarab, snails and wasps had increased in the past 19 years. The trends of the abundance of ants, cave weta (*Pleioplectron simplex*), ground weta *(Hemiandrus* n. sp), rove beetle, scarab, snail and wasps were found to be significantly different ($p<0.05$) from the trends in Grassland. In contrast, only ants, cave weta (*Pleioplectron simplex*) and wasps changed in the same direction with Grassland but others were not. *Baeus* spp., earwig, flatworms, moths, other spiders, phorids and true bugs had decreased during the period, all the trends were significantly different ($p<0.01$) from Grassland except for the trend of earwigs. Among them, only the trend of true bugs was in the opposite direction compared to Grassland. No significant change ($p>0.05$) had been detected in the number of caterpillars. The trend of *Metaglymma moniliferum* in Grassland showed an insignificant decrease ($p>0.05$), however, the trend in R was significantly different ($p<0.05$) which was a concave curve reach its peak at around 2008 and 2009.

3.3.2 Restoration (1983) compared to Grassland

In R(1983), there were increasing trends in millipedes, cave weta (*Pleioplectron simplex*), cockroach, true bugs, wasps, craneflies, flatworms, ground weta *(Hemiandrus* n. sp), caterpillars, larger flies, native harvestman, other spiders, phorids, pseudoscorpions and rove beetles. Millipede showed a concaved curve which was significantly different ($p<0.001$) from a smooth increase in Grassland. The trends in craneflies, flat worms and ground weta *(Hemiandrus* n. sp) were significantly different ($p<0.05$) from the trend in Grassland because of its opposite direction. The number of *Baeus* spp., earwig, *Metaglymma moniliferum*, moths, scarab showed the same decreasing trends with Grassland, but only the trend of *Baeus* spp. was significantly different ($p<0.001$). By contrast, the trend of ants had decreased which was significantly different ($p<0.05$) from the increasing trend in Grassland. The number of snails fluctuated, which was different from the decreasing trend in Grassland, during the past 19 years, but the difference was not significant ($p>0.05$).

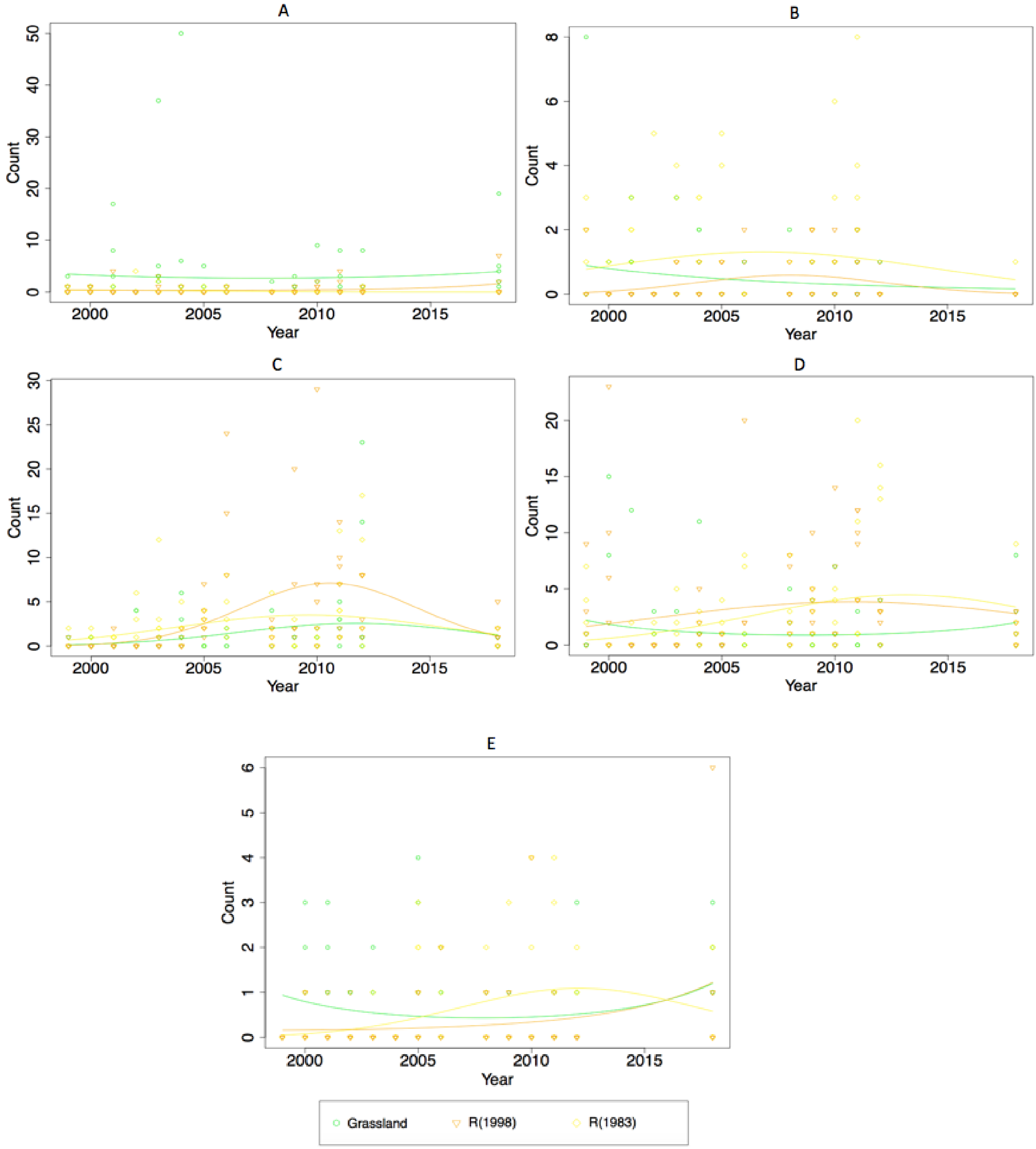


Figure 4. Restoration trajectories from 1999 to 2018 of (A) Ants, (B) Metaglymma moniliferum, (C) Cave weta (Pleioplectron simplex), (D) Ground weta (Hemiandrus n. sp.*) and (E) Millipedes.*

3.4 The influence of environmental factors

Table 4 presented the mean of environmental data collected at each trap site by habitat type. There were little differences in soil moisture, litter depth and canopy cover among R, R(1983) and Shrubland., whereas soil moisture in Grassland and Pine/Macro were fairly lower than the other three habitats. Canopy cover in Pine/Macro was slightly lower than R, R(1983) and Shrubland, while the number in Grassland showed a huge difference. By contrast, higher litter depth was found in Pine/Macro and Grassland. No clear difference could be found about the percentage of litter cover in habitats, but there was a gap between the number in R and others. The highest percentage of woody debris were found in Pine/Macro, followed by R(1983) were both over 10%. The other habitats had the percentage of woody debris that was lower than 5%, while the lowest percentage was found in Grassland with a percentage not up to 1%. R(1998), Shrubland and Pine/Macro had relatively steep slope than Grassland and R(1983) in both general and local slope.

Table 4. The mean of environmental factors collected at each trap site by habitat type.

Treatment	Soil Moisture	Litter depth	% of Litter cover	Canopy cover	% of Woody debris	General Slope	Local Slope
Grassland	19.54	3.75	85.00	34.17	0.83	14.50	11.33
Restoration(1998)	30.25	1.58	69.17	87.50	4.00	23.33	16.00
Restoration(1983)	27.33	2.08	84.50	92.50	13.33	13.67	7.33
Shrubland	26.17	1.50	82.50	92.50	4.83	28.50	20.33
Pine/Macro	12.75	6.08	82.00	80.00	17.50	24.17	18.00

Correlations of environmental factors were listed in Table 5. High correlations had been found in between soil moisture and litter depth and between general slope and local slope, which were 61% and -58%, respectively. Negative correlations around 30% found between litter depth and canopy cover and % of woody debris and soil moisture. On the other hand, soil moisture was positively correlated to canopy cover for 31%, so did litter depth and % of woody debris in 29%.

Table 5. The correlation coefficients among environmental factors.

	Soil Moisture	Litter depth	% of Litter cover	Canopy cover	% of Woody debris	General Slope	Local Slope
Soil Moisture	1.00	NA	NA	NA	NA	NA	NA
Litter depth	-0.58	1.00	NA	NA	NA	NA	NA
% of litter cover	-0.15	0.25	1.00	NA	NA	NA	NA
Canopy cover	0.31	-0.27	-0.13	1.00	NA	NA	NA
% of woody debris	-0.30	0.29	0.06	0.27	1.00	NA	NA
General slope	0.11	-0.14	-0.26	0.29	0.03	1.00	NA
Local slope	-0.12	0.02	-0.21	0.25	0.22	0.61	1.00

The best model of each species in 2018 count data analysis considering environmental factors and the significant ($p<0.05$) positive and negative effect had been listed in Table 6. Exclusion of species was due to no sufficient data for analysis. Further discussion between environmental factors and the abundance of species in 2018 had been done in section 4.4.

Table 6. The best model for environmental factors analysis in 2018 count data for each species with the summaries of significant (p<0.05) positive or negative impacts

Taxa	Model	Soil Moisture	Litter depth	% of litter cover	Canopy cover	% of woody debris	General slope	Local slope
Ants	Habitats + Soil moisture +% of litter cover + % of woody debris + Local slope	–		–		+		–
Baeus spp.	Habitats + Local slope							–
Bristletails	Habitats + General slope						+	
Caterpillars	Habitats							
Cave weta	Habitats + litter depth + % of woody debris + Local slope		–			+		–
Centipedes	Local slope							
Ground weta	Habitats + % of litter cover + Canopy cover + % of woody debris			–	+	+		
Millipedes	Habitats + Soil moisture + litter depth + % of woody debris + General slope + Local slope	–	+			+	+	–
Moths	Habitats							
Native harvestman	Habitats + Soil moisture + % of litter cover + Canopy cover + General slope	+		+	–		–	
Spiders other than trapdoors	Habitats + Litter depth + % of litter cover + Canopy cover + General slope		–	–	+		–	
Phoridae	Habitats + Soil moisture + % of woody debris + General slope	+				+	–	
Pseudoscorpions	Soil moisture + Litter depth + canopy cover + % of woody debris + General slope + Local slope	–	–		+	+	+	–
Rove beetles	Habitats + Canopy cover				–			
Scarabidae	% of litter cover							
Snails	Habitats							
Thrips	Canopy cover				–			
True bugs	Habitats + Soil moisture + Litter depth + General slope	+	–				–	
Wasps	Habitats + Soil moisture + Litter depth + Canopy cover + General slope	–	–		+		+	

4. Discussion

4.1 Species diversity

Both Shannon and Simpsons index showed obvious rising trends in all the habitats which indicate the increase of biodiversity on Quail Island. Successful pest eradication since 1997 could be the possible reason. Bowie et al. (2010) summarised the eradication work undertaken on Quail Island (Table 7.). While rabbit, cat, hedgehog and rats were removed, mice and mustelids still present. In terms of Shannon Index, it is clear that the trend of R increase noticeably than other habitats, followed by the trend of R(1983). Though the differences in inclination are not significant ($p>0.05$), it suggests that the restoration planting may effectively raise the biodiversity in the habitats. By contrast, Simpson's index shows R has the lowest diversity among habitats. However, given that R(1983) and Shrubland have significantly higher ($p<0.001$) biodiversity compared to Grassland, the continued growth of biodiversity in R could be expected. Besides, Simpsons Index gives more weight on dominant species which may entails that there are more dominant species in R(1983) and Shrubland whereas the development of invertebrate community in R is still at the early stage and the environment is more competitive.

The results of pairwise comparison illustrate that the biodiversity is the highest in Shrubland, the habitat that closest to the original vegetation on Quail Island, followed by R(1983), which is the oldest restoration site. These findings lead us to believe that the establishment of native flora contributes to biodiversity. However, it indicates that the biodiversity in R is not only lower than those two habitats but also lower than Grassland. Relatively lower percentage of litter cover compared to other habitats may not provide sufficient hiding space for species. It could have resulted from the immature native patches, but further research is needed to understand if there are any potential biotic or abiotic limitations.

Increasing numbers of different beetles suggest a more diversified beetle community in all habitats where pest eradication is considered to be the main factor. However, the beetle richness in R is the lowest as the same as the biodiversity detected by Shannon and Simpsons Index. Moreover, it is recognised as significantly lower ($p<0.01$) than Grassland. Relatively lower litter cover found in R could indicate less suitable habitats for beetles. On the other hand, the lack of data of species abundance can overestimate the increase of beetle richness. It is possible that some beetles are rarely found and are not able to contribute much when considering the other aspects of diversity such as species abundance. The drop of richness in Pine/Macro may indicate the existence of biotic or abiotic limitations that hinder the growth. Further identification and abundance check of species are recommended for an in-depth understanding of the change of beetle community.

In contrast to beetle richness, mite richness has no obvious trend. The richness in 2005 is apparently higher than both 2000 and 2010. Early pest eradication work done between 2000 and 2005 could change the predator-prey relationship in the food web, but the following meso-predator release could bring in unpredictable change (Zavaleta et al., 2001). The diversity of mites found is high, therefore further identification and abundance check may explain their change. There is also a potential to provide an insight on micro-habitat change since Oribatid mites are found to be sensitive to subtle differences (Nielsen et al. 2010).

Table 7. The eradication of mammalian pest undertaken on Quail Island (Bowie et al. , Kavermann & Ross, 2010).

Pest	Eradication
Rabbits	1997-2006 **(Eradicated)**
Cats	1998 **(Eradicated)**
Hedgehogs	2000-2003 (**Eradicated**)
Rats	2002 **(Eradicated)**
Mice	2002-2004, 2009 – still present
Mustelids	2001-ongoing

4.2 Habitat differences

4.2.1 Grassland

Ants, *Baeus*, thrips, cockroach, moths, other spiders and phorids seem to prefer Grassland. *Baeus* is a parasitoid of some spider eggs, therefore their presence is thought to follow spider abundance. The known relationship is consistent with our study that spiders are predominantly more in both Grassland and R(1998), as did *Baeus*. Although there is a high diversity of spider species, identification was not conducted in the limited time available. Given that spiders as predators that play an ecosystem role in the invertebrate community, further research of their composition and role in the ecosystem are recommended.

Relative low canopy cover in Grassland lead to lower soil moisture. The simple structure of vegetation with much less woody plant surrounded compared to other habitats offers less variety of micro-habitat for species (Curry, 1994). However, Grassland shows that it owns the highest beetle richness among habitats. Due to the characteristic of the habitat, we suggest that beetle composition in Grassland might be different and consist more adaptable herbivore species. However, further research is required.

4.2.2 Restoration (1998)

Cave weta (*Pleioplectron simplex*) shows a strong preference in native patches such as R(1998), R(1983) and Shrubland with the highest abundance in R(1998). Higher soil moisture and canopy cover fits their requirements of dark, damp places. Such places including under bark or inside hollow logs, the percentage of woody debris also has a strong positive effect on the abundance of cave weta (*Pleioplectron simplex*). Besides, the flat local slope can influence the soil hydrology that may provide a higher moisture.

The higher abundance of earthworm in this restoration site could indicate the better soil structure and decomposition process. The study from Snyder & Hendrix (2008) pointed out the key role that earthworm played in soil development that enhances plant restoration which could further promote the succession of invertebrate community.

There was mainly two genus of Scarabaeidae found on Quail Island (Bowie et al. 2003), but no further identification was done in this study. In contrast to *Costelytra* that live in grassland, *Odontria* prefers bush habitat. The significantly less ($p<0.01$) Scarabeidae found in R could indicate that young restoration sites have not matured sufficiently to provide suitable habitat, especially compared to Grassland and R(1983)).

4.2.3 Restoration 1983 (R(1983))

Snails, in a lower level on the food web, are an important food source for a variety of animals. Their higher abundance in R(1983) could indicate a high demand for leaf litter decomposition. The role they played in nutrient cycle drive the substantial ecosystem function in restoration.

More *Metaglymma moniliferum* found in R(1983) except for Pine/Macro could be influenced by the higher percentage of woody debris. Since habitat patches of R(1983) are relatively small and fragmented than other habitats, high edge effect should be taken into consideration. In this case, woody debris in R(1983) is mainly from adjacent Pine/Macro plantation that provides a different micro-habitat compared to other native plantations such as R(1983) and Shrubland. It is consistent with the suggestion from Larochelle & Larivière (2001) that they prefer dry area such as logs or stones to hide during the day.

4.2.4 Shrubland

Shrubland is the oldest native stand with dense understory layers. In general, it provides a dark, damp environment with high soil moisture, shallow litter depth, highly covered ground by litter and a few woody debris. Multi-layers allow the formation of various micro-habitats. *Holcaspis* spp. and *Mimopeus opaculus* seem to strongly prefer this type of habitat. European harvestman, commonly as an ambush predator, could benefit from the habitat structure. Flying invertebrates such as large craneflies, larger flies and wasps also found more in the dense scrubland, but large craneflies and wasps also have a high occurrence in Grassland which could be due to different species has different habitat preference. In contrast, the similar occurrence of large flies in R(1983) may indicate their native bush preference. A high presence of true bugs may result from a rich food source and various micro-habitats that created different niches.

4.2.5 Pine/Macro (PM)

Bristletails were most abundant in Pine/Macro, which is consistent with their preference of staying hidden in logs, under rocks or in the layers of leaf litter. A high percentage of woody debris and multiple litter layers provide a suitable habitat for them. Leaf litter layers also attract the millipede population, which has the highest abundance among other habitats. Pseudoscorpions, a solitary predator of small invertebrates, were found more in Pine/Macro than Grassland and R. This may be due to less leaf litter, bark trees or woody debris that they usually live in (Del-Claro & Tizo-Pedroso, 2009).

As mentioned in 4.2.3 R(1983), a higher percentage of woody debris may attract *Metaglymma moniliferum monoliferum* than Grassland, R(1998) and Shrubland. Pine plantation is also recorded to be one of their habitats (Larochelle & Larivière, 2001). On the other hand, the thick leaf layer may prevent Pine plantation to be a suitable habitat for cave weta (*Pleioplectron simplex*) since they mainly live in tunnels in the soils. It could be the same reason that they are also relatively less in Grassland compared to other three native habitats.

4.3 Restoration Trajectories

4.3.1 Ants

Ants significantly increased ($p<0.05$) in Grassland and R while decreased in R(1983). The concept of using ants as bio-indicator has obtained widespread acceptance throughout the world since ant species richness in mine site was found to have a strong association with richness and/or abundance of other taxonomic group in Australia (Majer et al., 2007). Except for the trend in R(1983), the increasing trends in Grassland and R are consistent with Shannon and Simpson's index.

4.3.2 Carabids

Metaglymma moniliferum decreased in Grassland whereas fluctuated in R(1998) and R(1983) with an increase until 2008 to 2009 and a decrease afterwards. Only the trend in R(1998) was significantly different ($p<0.05$) from Grassland could indicate that they struggled to survive in both grassland and restoration site. The initial increase was regarded to be the result of pest eradication, especially the removal of hedgehog, the major predator of carabids (Jones & Toft, 2006). However, not all the pests had been eradicated in the early restoration stage (Table 7) which could potentially cause competitive release (Zavaleta et al., 2001). Remaining pest such as mice and mustelids could possibly expand their realised niche and gradually became a new major threat after a period of time. The other reason could be lack of enough suitable habitat. Dead wood was regarded as an important shelter for saproxylic invertebrates that could influence their abundance and diversity (Sands, 2013). From the pairwise comparison, it shows that *Metaglymma moniliferum* prefers R(1983) and Pine/Macro which have a higher percentage of woody debris.

On the other hand, the number of *Holcaspis* caught in Grassland, R(1998) and R(1983) in 14 years is only 12 specimens, which shows insufficient data for the trend analysis. They seem to significantly prefer ($p<0.05$) Shrubland compared to any other habitats (Table 4). It indicates the more mature site with high-density bushes, which is consistent with the result of the previous study (Stokvis et al., 2015), could be necessary for their restoration.

Megadromus guerinii is a Banks Peninsula endemic ground beetle that had been reintroduced to Quail Island in 2004 (Bowie, 2008) near ER4 and ER5, trap sites in Shrubland. However, no presence had been recorded until the five catches in 2018 with one caught in ER6, which is approximately 400 m distance from their release site. It indicates the nearly mature habitat has the capacity to support a *Megadromus* population. As the first Banks Peninsula endemic species successfully reintroduced on to Quail Island, it shows the potential for Quail Island to be a refuge for other endangered, local and endemic carabid species. Further reintroductions can be considered, but the reason/s for being undetected for nearly thirteen years may need more research but could be the problems associated with sampling rare populations (Driscoll, 2010).

4.3.3 Millipedes

The number of millipedes is on the increase in all three habitats but no significance ($p>0.05$) has been found in the trend of R(1998). Given that Millipedes are decomposers which mainly feeds on decaying leaves, it could indicate the increase of leaf litter that resulted from the growth or restoration planting. Other studies (Margules, 1993; Baker, 1998) also found that they moved to plant plantation after the disturbance in the native forest, which indicates their mobility and adaptability that respond to environmental change. Since their population was growing in other habitats at the same time, it suggests the restoration may promote a suitable habitat as it was before. In addition, millipede is a useful physiochemical bio-indicator that represents the better nutrient cycle that enhances soil development that is substantial for the plant growth (Snyder & Hendrix, 2008). Hence, it could indicate a healthier soil environment to promote the restoration.

4.3.4 Weta

Cave weta (*Pleioplectron simplex*) is generally increasing in Grassland, R(1998) and R(1983), but the trend rises significantly more ($p<0.01$) in R(1998) and not significantly more ($p>0.05$) in R(1983) compared to Grassland. On the other hand, the number of ground weta (*Hemiandrus* n. sp.) in Grassland seems to fluctuate whereas the number in R(1998) and R(1983) show significantly rise ($P<0.05$). The overall increase except for ground weta (*Hemiandrus* n. sp.) in Grassland could be contributed by the eradication of hedgehog, which regards weta as one of the major diet (Jones & Toft, 2006). Since mice have not been eradicated from the Island (Table 7), relatively openness in Grassland could make ground weta (*Hemiandrus* n. sp.) more susceptible. In the contrary, the number of cave weta (*Pleioplectron simplex*) is growing in Grassland may be resulted from the variation of trap site. Although trap was set up in grassland, G3 to G6 are beside native shrubs which indicates the potential of micro-habitat created by small patches. With the growth of vegetation in R(1998) and R(1983), the possible increase of canopy cover, litter cover and soil moisture may explain the increases of both weta species. It shows the restoration work could provide a more suitable habitat for them.

4.4 The influence of environmental factors

Higher canopy cover seems to contribute to soil moisture (31% positive correlated), while litter depth has a stronger negative effect (58%). Among them, the negative correlation between canopy cover and litter depth indicate the slow leaf litter decomposition may strongly be influenced by these three factors. Although there is high canopy cover (80%) in Pine/Macro, nearly no understory could be found due to its characteristic of openness. It leads to a relatively dry condition in Grassland, which is a typical open space, and Pine/Macro which enhance the accumulation of leaf litter. In our study, only millipedes were found to strongly prefer thick leaf litter layers which is consistent with their high occurrence in Pine/Macro. Since the colour pattern of pine needles is similar to millipedes, it could be a reason for their preference to pine plantation. Other species were negatively affected by litter depth, but we suggest that the litter composition may be the main reason because the lower litter depth all comes from native patches.

Medium correlation can be found between the percentage of woody debris and soil moisture, litter depth and canopy cover. Since the highest percentage of woody debris is in Pine/Macro which indicate a relatively old and dry environment, low soil moisture, deep leaf litter and high canopy cover are expected. However, the percentage of woody debris in R(1983) was raised due to smaller patches near pine plantation. The edge effect, therefore, bias the correlation. In terms of the effect on species abundance, only positive effects on species abundance from the percentage of woody debris had been detected which indicate it is able to enhance habitat variability. In contrast, local slope seems to discourage species abundance, especially for those ground invertebrates.

The percentage of litter cover was found to be relatively low in R(1998) which indicates younger plantation. It has a negative correlation with the abundance of ants, ground weta (*Hemiandrus* n. sp.) and other spiders, but a positive correlation with harvestman. In addition, general slope seems to be a factor that influences most of the species. This may be related to their movement in the habitat.

5. IMPLICATIONS

Species diversity tested by index shows a promising increase. It may suggest a progress of ecosystem structure, but it does not necessarily entail a successful restoration of ecosystem function (Cortina et al., 2006). The use of diversity index is also controversial since it only considers species richness and abundance but not species composition. Barrantes & Sandoval (2009) indicated the conceptual issue of using diversity index that the loss of species identity cause their functional roles remain unknown in the community. Ensuring the functional biodiversity is key to a self-sustaining ecosystem. In addition, Gardner-Gee et al. (2015) discovered the failing of invertebrate succession by compositional differences between restoration site and target site that could be an unrevealed problem in the ecological community in Quail Island. Therefore, food web or trophic level based research that recognises intraspecific difference among species is recommended to reveal the whole picture of restoration impacts on interactions in the ecosystem.

The finding of habitat preference of species give a further understanding of their ecology, which can be beneficial for the design of future restoration plan. However, the paucity of environmental data in the past prevents us from filtering out the key attributes in habitat that contribute to species abundance. The same circumstances applied on the examination of restoration trajectory that no specific evidence of what has been changed in those habitats. Habitat monitoring is included as an important component to assess the restoration progress and is able to predict the potential species recovery (Bried et al. 2014). In addition, the use of pitfall trap is considered to be strongly influenced by habitat structure (Melbourne, 1999; Phillips & Cobb, 2005; Buchholz & Hannig, 2009). Vegetation cover can influence the mobility of species and microclimate on the soil surface which have further trapping impact. Ruiz-Jaen and Mitchell Aide (2005) further indicated that using two of the three major attributes, which are diversity, vegetation structure and ecological processes, for measuring restoration success are more appropriate. Hence, including the measurement of attributes of habitat structure in the future monitoring programme is strongly recommended.

Though a wide range of invertebrates had been sampled, the understanding of the change or impact of some species, such as true bugs and spiders, occupied various niches and have different ecology are hard without further identification. It can also be used to investigate species composition or similarities for further understanding the differences of communities in each habitat that may reveal more specific ecological succession.

Ongoing invasion of pest could be the main potential threat or limitation to the recovery of invertebrate communities. Long-term monitoring of pest species is necessary not only for prevention of reinvasion but also for understanding any unpredictable impact from competitive release. The reason for the detected struggle of *Metaglymma moniliferum* may need further research to find out the possible limitations. On the other hand, *Megadromus guerinii* survived after reintroduction, but the reason for their absence for thirteen years has not been known. The further monitoring of their survival is recommended to make sure any concern for their survival before next reintroduction.

Since using pitfall traps is widely accepted in terms of monitoring epigaeic fauna, the limitation of the methodology should not be overlooked. The population density was found to have only weak or highly variable relationship with pitfall catches considering taxa, habitats and time of the season (Lang, 2000). Habitat structure can influence on catches, therefore, a direct comparison has its concern if there are habitat disparities (Samways et al., 2010). Besides, the catch from pitfall trap has been emphasized to be influenced by species-specific differences in locomotion. Engel et al. (2017) further indicated how the bias could shape misperception in community-level diversity metrics. Therefore, applying two methods to collect data for estimation of population density and activities of organisms was suggested by Lang (2000). Applying different monitoring techniques could validate and lead to a more credible result.

Although the uncertainty and concerns outlined above, the ecological restoration promotes the invertebrate communities on Quail Island in general. Pest eradication and restoration planting yielded a rich harvest, especially for both cave weta (*Pleioplectron simplex*) and ground weta (*Hemiandrus* n. sp.). The discovery of *Megadromus guerinii*, the first Banks Peninsula endemic ground beetle on the island, shows its huge potential to translocate future absent Banks Peninsula species. Since the ecological restoration is a long-term programme, continuous monitoring is encouraged in the future to track the progress of the restoration trajectory and ensure the succession of the ecological community is progressing favourably.

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8. References

Atkinson, I.A.E. (1988). Presidential address: opportunities for ecological restoration. *New Zealand journal of Ecology*, 1-12.

Baker, G.H. (1998). Recognising and responding to the influences of agriculture and other land-use practices on soil fauna in Australia. *Applied Soil Ecology 9*(1-3), 303–310.

Barrantes, G., & Sandoval, L. (2009). Conceptual and statistical problems associated with the use of diversity indices in ecology. *Revista de biología tropical, 57*(3), 451-460.

Bellingham, P.J., Towns, D.R., Cameron, E.K., Davis, J. J., Wardle, D.A., Wilmshurst, J.M., & Mulder, C.P. (2010). New Zealand island restoration: seabirds, predators, and the importance of history. *New Zealand Journal of Ecology, 34*(1), 115.

Bowie, M.H. (2001). *Restoration of Quail Island invertebrates*. Wildlife Management Report no. 22. Lincoln University. 15pp.

Bowie, M.H., Marris, J.W.M., Emberson, R.M., Andrew, I.G., Berry, J.A., Vink, C.J., White, E.G., Stufkins, M.A.W., Oliver, E.H.A., Early, J.W., Klimazewski, J., Johns, P.M., Wratten, S.D., Mahlfeld, K., Brown, B., Eyles, A.C., Pawson, S.M. and Macfarlane, R.P. (2003). A terrestrial invertebrate inventory of Quail Island (Ōtamahua): towards the restoration of the invertebrate community. *New Zealand Natural Sciences 28*: 81-109.

Bowie, M.H. (2008). Ecological restoration of the invertebrate fauna on Quail Island (Ōtamahua). In: Clarkson, B.; Kurian, P.; Nachowitz, T. and Rennie, H. (Eds.*). Conserv-Vision Conference: the Next 50 Years. Proceedings of Conserv-Vision Conference, Waikato University 4-7th July 2007* (pp.2-4),

Bowie, M.H., Kavermann, M., & Ross, J. G. (2010). The Quail Island story-thirteen years of multi-species pest control: successes, failures and lessons learnt. In: Veitch, C.R., Clout, M.N., Towns, D.R. (Eds.) *Island Invasives: Eradication and Management* (pp. 157-161.). IUCN, Gland, Switzerland and The Centre for Biodiversity and Biosecurity (CBB), Auckland, New Zealand

Bried, J., Tear, T., Shirer, R., Zimmerman, C., Gifford, N., Campbell, S., & O'Brien, K. (2014). A framework to integrate habitat monitoring and restoration with endangered insect recovery. *Environmental management, 54*(6), 1385-1398.

Buchholz, S., & Hannig, K. (2009). Do covers influence the capture efficiency of pitfall traps? *European Journal of Entomology, 106*(4).

Burrows, C.J. & Wilson, H.D. (2011). *The Island's Plants: Ecorestoration of Otamahua / Quail Island, 1997-2010*. Christchurch, N.Z. Ecological Restoration Trust. 132pp.

Cortina, J., Maestre, F. T., Vallejo, R., Baeza, M. J., Valdecantos, A., & Pérez-Devesa, M. (2006). Ecosystem structure, function, and restoration success: Are they related?. *Journal for Nature Conservation, 14*(3-4), 152-160.

Curry, J.P. (1994). *Grassland invertebrates: ecology, influence on soil fertility and effects on plant growth*. London ; Chapman & Hall

Del-Claro, K., & Tizo-Pedroso, E. (2009). Ecological and evolutionary pathways of social behavior in Pseudoscorpions (Arachnida: Pseudoscorpiones). *Acta Ethologica, 12*(1), 13-22.

Driscoll, D.A. (2010). Few beetle species can be detected with 95% confidence using pitfall traps. *Austral Ecology, 35*(1), 13-23.

Engel, J., Hertzog, L., Tiede, J., Wagg, C., Ebeling, A., Briesen, H., & Weisser, W. W. (2017). Pitfall trap sampling bias depends on body mass, temperature, and trap number: insights from an individual-based model. *Ecosphere, 8*(4).

Gardner-Gee, R., Stanley, M.C., & Beggs, J.R. (2015). Re-forestation restores native dominance in an island beetle fauna. *Restoration ecology, 23*(3), 268-276.

Jackson, P.J., Zealand, N., & Quail Island Restoration Trust (NZ). (2006). *Otamahua/Quail Island: A Link with the Past*. Christchurch, NZ.

Jones, C., & Toft, R. (2006). *Impacts of mice and hedgehogs on native forest invertebrates: a pilot study*. Science & Technical Publishing, Department of Conservation.

Lang, A. (2000). The pitfalls of pitfalls: a comparison of pitfall trap catches and absolute density estimates of epigeal invertebrate predators in arable land. *Anzeiger für Schädlingskunde. Journal of Pest Science, 73*(4), 99-106.

Larochelle, A., & Larivière, M.C. (2001). *Carabidae (Insecta: Coleoptera): catalogue. Fauna of New Zealand, 43*. Lincoln, Canterbury, N.Z.: Manaaki Whenua Press, Landcare Research.

Lindenmayer, D., Barton, P., & Pierson, J. (2015). *Indicators and surrogates of biodiversity and environmental change.* CSIRO PUBLISHING.

Majer, J.D., Brennan, K.E., & Moir, M. L. (2007). Invertebrates and the restoration of a forest ecosystem: 30 years of research following bauxite mining in Western Australia. *Restoration Ecology, 15*(s4).

Margules, C.R. (1993). The Wog Wog habitat fragmentation experiment. *Environmental Conservation 19*(4), 316–325.

McGeoch, M.A. (1998). The selection, testing and application of terrestrial insects as bioindicators. *Biological Reviews, 73*(2), 181-201.

Melbourne, B.A. (1999). Bias in the effect of habitat structure on pitfall traps: An experimental evaluation. *Austral Ecology, 24*(3), 228–239.

Meurk. C.D. (1990). *A reappraisal of and proposal for management of Quail Island.* DSIR informal report presented to North Canterbury Conservation Board, 1992.

Nielsen, U.N., Osler, G. H., Campbell, C.D., Burslem, D.F., & van der Wal, R. (2010). The influence of vegetation type, soil properties and precipitation on the composition of soil mite and microbial communities at the landscape scale. *Journal of Biogeography, 37*(7), 1317-1328.

Phillips, I.D., & Cobb, T.P. (2005). Effects of habitat structure and lid transparency on pitfall catches. *Environmental Entomology, 34*(4), 875-882.

R Core Team (2017). *R: A language and environment for statistical computing.* R Foundation for Statistical Computing, Vienna, Austria. URL https://www.R-project.org/.

Ruiz-Jaen, M.C., & Mitchell Aide, T. (2005). Restoration success: how is it being measured?. *Restoration ecology, 13*(3), 569-577.

Sands, R.J. (2013). Effect of woodstack structure on invertebrate abundance and diversity. *Bioscience Horizons: The International Journal of Student Research, 6*.

Samways, M.J., McGeoch, M.A., & New, T.R. (2010). *Insect conservation: a handbook of approaches and methods*. Oxford University Press.

Snyder, B.A., & Hendrix, P.F. (2008). Current and potential roles of soil macroinvertebrates (earthworms, millipedes, and isopods) in ecological restoration. *Restoration Ecology, 16*(4), 629-636.

St Clair, J.J., Poncet, S., Sheehan, D.K., Székely, T., & Hilton, G.M. (2011). Responses of an island endemic invertebrate to rodent invasion and eradication. *Animal Conservation, 14*(1), 66-73.

Stokvis, E., Bowie, M.H., Barber, K., Vink, C., & Marshall, B. (2015). *Measuring the restoration trajectory of invertebrates in the Winstone Aggregates Hunua quarry site.* Lincoln University. Faculty of Agriculture and Life Sciences. Department of Ecology.

Waterhouse, E.J. (1991). *Ecological restoration in New Zealand* (Doctoral dissertation, Lincoln University).

Wilson, H.D. (2013). *Natural history of Banks Peninsula*. Canterbury University Press.

Zavaleta, E.S., Hobbs, R.J. & Mooney, H.A. (2001). Viewing invasive species removal in a whole-ecosystem context. *Trends Ecology & Evolution, 16*(8), 454–459.

9. APPENDIX

Common name	Order	Family	Genus/Species
Earthworm	ANNELIDA	Acanthodrilidae/Lumbricidae	
Trapdoor spiders	ARANEAE	Idiopidae	*Cantuaria borealis*
Other spiders	ARANEAE		
Cockroach	BLATTODEA	Blattidae	
Centipede	CHILOPODA	Henicopidae	
Megadromus	COLEOPTERA	Carabidae	*Megadromus guerinii*
Weevils	COLEOPTERA	Curculionidae	
Rove beetle	COLEOPTERA	Staphylinidae	
Click beetle	COLEOPTERA	Elateridae	
Darkling beetle	COLEOPTERA	Tenebrionidae	*Mimopeus opaculus*
Darkling beetle	COLEOPTERA	Tenebrionidae	*Mimopeus granulosus*
Huhu beetle	COLEOPTERA	Cerambycidae	*Prionoplus reticularis*
Scarab	COLEOPTERA	Scarabaeidae	*Costelytra* & *Odontria*
Beetles larvae	COLEOPTERA	Scarabae and Carabidae	
Ground beetles	COLEOPTERA	Carabidae	*Metaglymma monoliferum*
Ground beetles	COLEOPTERA	Carabidae	*Holcaspis*
Pristoderus bakewelli	COLEOPTERA	Zopheridae	*Pristoderus bakewelli*
Zophoridae	COLEOPTERA	Zopheridae	
Earwig	DERMAPTERA	Labiduridae/Forflculidae	*Chaetospania brunneri* *Forficula auricularia*
Millipedes	DIPLOPODA		7+ species
Larger flies	DIPTERA	Calliphoridae, Muscidae, Tachinidae	
Craneflies /tipulids	DIPTERA	Tipulidae	14+ species
Scuttle flies	DIPTERA	Phoridae	12+ species
True bug	HEMIPTERA		
Wasps	HYMENOPTERA		
Baeus	HYMENOPTERA	Scelionidae	*Baeus* spp.
Ants	HYMENOPTERA	Formicidae	
Moths	LEPIDOPTERA		
Caterpillar	LEPIDOPTERA		
European harvestman	OPILIONES	Phalangiidae	*Phalangium opilio*
Native harvestman	OPILIONES	Triaenonychidae	
Ground weta	ORTHOPTERA	Anostostomatidae	*Hemiandrus* n. sp.
Cave weta	ORTHOPTERA	Rhaphidophoridae	*Pleioplectron simplex*
Crickets	ORTHOPTERA	Gryllidae	
Pseudoscorpion	PSEUDOSCORPIONES		
Fleas	SIPHONAPTERA	Ceratophyllidae	
Snail	STYLOMMATOPHORA		

Slug	STYLOMMATOPHORA		
Thrips	THYSANOPTERA	Thripidae	
Bristletail	THYSANURA	Meinertellidae	*Nesomachilis* sp.
Flatworm	TUBELLARIA	Geoplanidae	
Skink	SQUAMATA	Scincidae	*Oligosoma* spp.