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RESEARCH

A case study suggests positive ecological changes associated with conversion of pasture to mānuka (*Leptospermum scoparium*) forests in Aotearoa, New Zealand

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Abstract: In Aotearoa | New Zealand, conversion of low-yield high hill-country pasture farms to mānuka (Leptospermum scoparium) forests to support the harvesting of high quality monofloral mānuka honey can provide an economic benefit for rural landowners. However, the effects of such conversions on the local ecosystem and biodiversity are largely unknown. In a cross-sectional (space-for-time substitution) case study, we undertook biodiversity surveys at a single central North Island property that contained four sites representing different time points in the transition from pasture to mānuka forest: pasture (34 ha), three-year-old (36 ha) and five-year-old (23 ha) planted mānuka, and > 30-year-old naturally regenerated mānuka forest (57 ha). Terrestrial invertebrate surveys using window traps and bat and bird audio surveys were performed at each site. Freshwater health assessments were undertaken at three stream sites that were independent of the terrestrial sites. We found that invertebrate and bird abundance and diversity were lower in the three-year-old planted mānuka than in the pasture site. However, both invertebrate and bird abundance and diversity were higher in the five-year-old planted mānuka than the pasture, and were comparable to the > 30-year-old naturally regenerated mānuka forest in terms of community composition and diversity. Calls from the critically endangered long tailed bat (Chalinolobus tuberculatus) were detected more frequently at the five-year-old and > 30-year-old naturally-regenerated sites than at the other sites. Compared to the pasture catchment, macroinvertebrate scores and other stream health measures were higher in streams whose surrounding land use was native forest, naturally regenerated mānuka, or planted mānuka. Although site age was not replicated, our results suggest that there could be considerable positive changes in biodiversity that occur between three to five years after planting mānuka, with the five-year-old mānuka in our study supporting a more diverse and abundant community that was similar to that of the > 30-year-old naturally regenerated mānuka forest. Overall, our study provides supporting evidence that planting of mānuka forests in former pasture likely leads to a positive change in the ecosystem, however replication at different locations is required to confirm these case-study observations. Results from such surveys may aid in developing management systems to further enhance ecological outcomes of planted mānuka forests.

Keywords: acoustic monitoring, biodiversity studies, mānuka forest, mānuka honey, native forestry

Introduction

Leptospermum scoparium (mānuka) is a New Zealand native pioneer species from the Myrtaceae family (Butz Huryn 1995; Chagné et al. 2023). It is tolerant of environmental conditions that restrict growth in other plant species, and commonly grows in areas with low-fertility soils, disturbed ground, high winds, and exposed ridges (Stephens et al. 2005). Planted mānuka forests in New Zealand potentially have high commercial value owing to the nectar being harvested by European honey bees (*Apis mellifera*) to make mānuka honey, which had an export revenue to New Zealand of approximately $400 million in 2022 (Ministry for Primary Industries 2022).

Beekeepers have traditionally targeted the production of mānuka honey by placing hives on public and private lands that have areas of remnant or naturally-regenerated mānuka forest. As the demand for, and value of, mānuka honey has increased so has the competition for placement of beehives on native mānuka resources, particularly in areas known to produce higher unique mānuka factor (UMF^{TM}, a quality assurance rating assessing potency, authenticity, freshness, and purity; Unique Mānuka Factor Honey Association 2024), monofloral mānuka honey (Ministry for Primary Industries 2022). There has been a recent increase in New Zealand in the conversion of marginal high hill-country sheep and beef pasture to mānuka forests with the aim of improving the yield and quality of mānuka honey. Comvita, one of New Zealand's largest mānuka honey companies, is a major contributor to mānuka reforestation, having planted > 7

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million mānuka seedlings (i.e. non-clonal forests) over > 7500 hectares of land since 2016 (Comvita 2023).

Mānuka forests for honey production are generally planted as single species blocks in order to maximize the floral resource available for mānuka nectar collection by honey bees and to reduce dilution of nectar by other floral sources (Boffa Miskel Limited 2017). Investment in these forests is based on the anticipation of high yields of high quality monofloral mānuka honey (Ministry for Primary Industries 2017) and a more secure supply than relying on unmanaged forests (Hamilton et al. 2013; Millner et al. 2016). Research into the impact of mānuka forest planting to date has largely focussed on the economics, with modelling estimating high returns from honey production (McPherson 2016) and carbon credits from carbon sequestration (Scott et al. 2000; Ross et al. 2009). However, mānuka are also known to reduce the erodibility of pasture on slopes, especially on high hill-country pasture (Watson & O'loughlin 1985; Bergin et al. 1995; Marden et al. 2020) and contribute to improved freshwater quality by reducing the potential nitrate leaching into waterbodies (Halford et al. 2021). Furthermore, as mānuka is an early-successional New Zealand native plant species, the conversion of pasture to mānuka forests is likely to have wider benefits for biodiversity and ecosystem health than is currently recognised. These benefits are likely to be enhanced by the forests' primary use as a nectar source for honey bees, as opposed to other planted forests which are primarily for harvesting of timber, leading to the periodic loss of forest cover. Currently, the environmental impact of this globally unique practice of conversion from pastoral land to planted mānuka forests has not been studied.

The value of single species production forest for native biodiversity depends on many factors, including the proportions and abundances of native or exotic understorey plants and the type of forest management, including pruning regime, previous land use, and the age of plantings (Brockerhoff et al. 2008a; Pawson et al. 2010). The introduced Monterey pine (*Pinus radiata*) comprises approximately 90% of New Zealand's plantation forest, providing a significant portion of the forest and acting as ecological corridors between native forest fragments (Pawson et al. 2010). Some endemic species, such as *Holcaspis brevicula*, an endangered ground beetle, are now thought to exist only in pine plantations (Brockerhoff et al. 2005). Whilst New Zealand planted pine forests are reported to have a higher diversity of insects than agricultural land (Brockerhoff et al. 2008a), and biodiversity in exotic plantations often increases with time as more species have time to move in and settle (Tudge et al. 2023), little is known about the impact of single species plantings of native trees such as mānuka. Owing to the long evolutionary and ecological association between mānuka and the biota and ecosystems of New Zealand, it is reasonable to assume that single-species mānuka forests are likely to provide better ecosystem services and habitat for other native species than exotic single-species forests (Brockerhoff et al. 2008b) and pasture land that is dominated by exotic grasses. As a pioneer species, mānuka is often the first species to re-establish in degraded agricultural habitats, and it is commonly one of the most abundant species used in restoration plantings (Auckland Council 2024), suggesting that the ecological outcomes of large-scale mānuka forest planting for honey production may be similar to those seen in early-stage restoration plantings or naturally-regenerating forests.

The objective of this study was to investigate the ecological changes associated with the conversion of high hill-country pasture to mānuka forests by conducting a case study on a high hill-country farm estate that has been planted with mānuka. We conducted surveys and observations at four different sites representing different timepoints in the transition from pasture to mature mānuka forest: currently grazed pasture, three-year-old planted mānuka, five-year-old planted mānuka, and a naturally-regenerating stand of mānuka that had been left to grow for more than 30 years. Our hypothesis was that the two planted mānuka forest sites would exhibit characteristics intermediate to the pasture and the naturally-regenerating forest, and provide an indication of likely timelines to achieve similar ecological and biodiversity characteristics.

Methods

Study site description

This case study was conducted on a private 4500 ha high hill-country station in the central North Island of New Zealand, north-west of Mount Ruapehu between state highways 43 and 4. The study area was a sheep and beef farm, and had mānuka forests planted on it from 2016–2019. Overall, the property was comprised of retired pasture, 767 ha of planted mānuka of different ages (three to six years) that were still grazed at a low level for weed control, and an area of naturally regenerating mānuka forest (Fig. 1a). In addition, we included a pasture site that is still grazed by sheep at normal stocking rates on a neighbouring property with comparable characteristics.

The four sites on the property (Fig. 1a), that were chosen for bird, bat and terrestrial invertebrate assessment were located within areas of sheep-grazed pasture ('pasture', 34 ha; Fig. 1b), three-year-old planted mānuka ('young planting', 36 ha; Fig. 1c), five- to six-year-old planted mānuka ('mature planting', 23 ha; Fig. 1d), and more than 30-year-old regenerated mānuka forest ('old regen', 57 ha; Fig. 1e). As this was a case study, sites were not replicated on other properties. The areas given for these sites are the area of the contiguous patches for each description, but the wider property contained numerous other patches of similar or different land use and age of growth. At the time of planting, planted mānuka forest sites were prepared via stock removal and chemical weed control to reduce competing plant species, followed with manual planting of mānuka seedlings. Compared to the young planting site, seedlings were planted at a higher density at the mature planting site over two consecutive winters (Fig. 1f). All sites were at an altitude between 370 m and 500 m, slopes of 10–40°, and varying aspects (Appendix S1 in Supplementary Material). All sites were visited for data collection on two occasions: once during Austral summer (12–25 January) and once during Austral Autumn (17–25 May).

Mānuka tree characteristics

Two 25 m transects were established at each site, perpendicular to a stream at each site, and extending up each bank, apart from at the mature planting site, where the transects extended up from the stream on the same bank, approximately 100 m apart, due to constraints with accessing parts of the site. A mānuka tree at the end of a transect was chosen as the starting point, before following along the general transect line, randomly choosing a selection of mānuka trees from the left and right of the transect, until ten trees had been selected. This was repeated for all transects at each mānuka planting or natural regeneration site. The diameter at breast height (1.30 m), height (estimated in seven height classes: 0.5, 1, 1.5, 2, 2.5, 3, and 3 + metres), and the distance between nearest mānuka trees were measured.

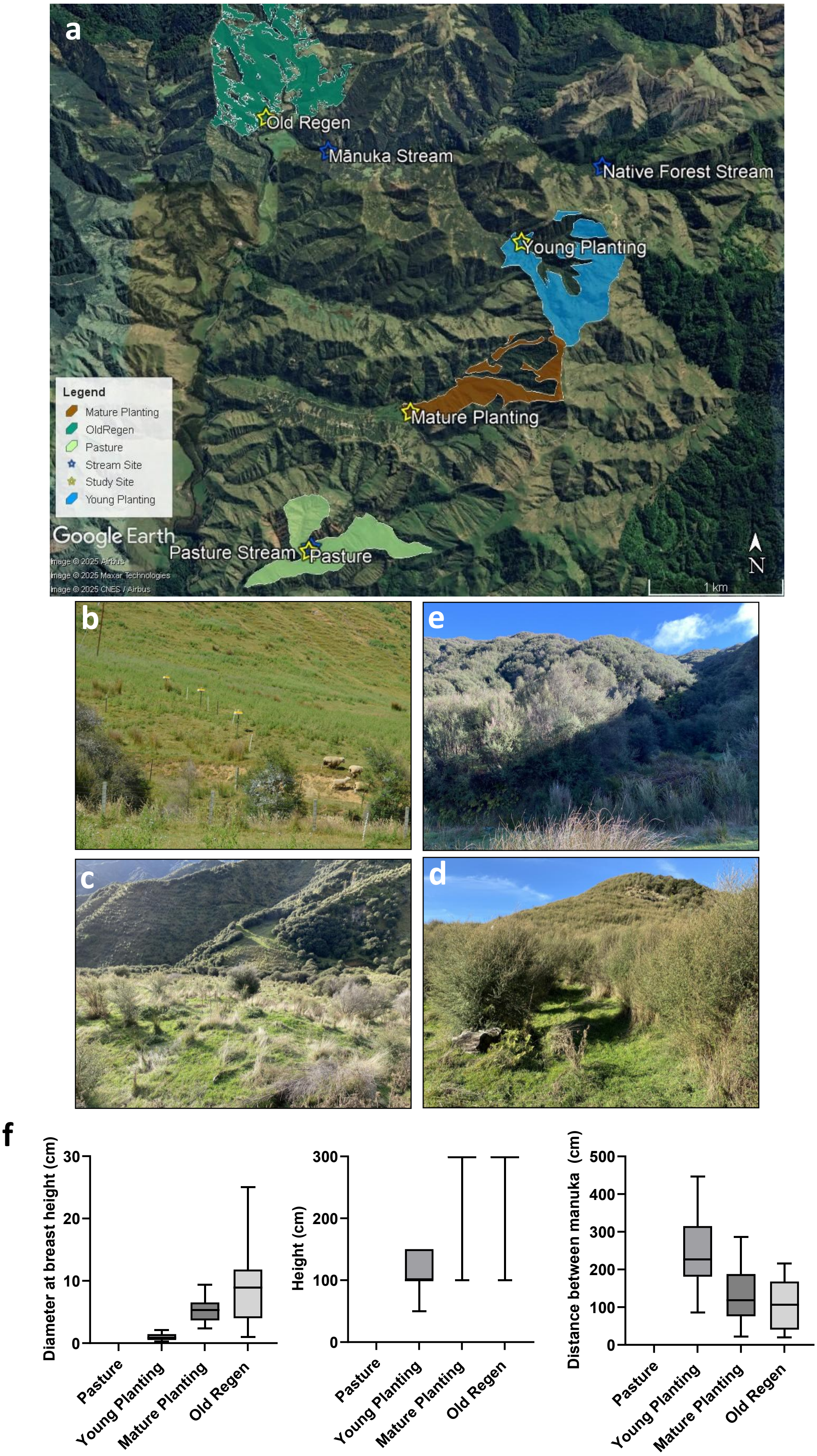

Figure 1. Property description. Property map showing survey sites and stream sites (a), pasture site (b), young planting site (c), mature planting site (d), old regen site (e), and mānuka characteristics showing the quartiles and means of the diameter at breast height (cm), height (cm, truncated at 300 cm), and distance between mānuka trees at each site (cm) (f).

Terrestrial invertebrates

Six window traps per site were used to sample aerial terrestrial invertebrates between the 18th and 22nd of January 2022. Three window traps were set up c. 8.33 m apart along each transect; each window trap consisted of two intersecting perspex panes (368.50 × 270.50 × 6 mm and 245.50 × 270.5 × 6 mm) set into a yellow container (290 × 415 × 130 mm). Each trap was set into a white tray (435 × 332 × 68 mm) that was screwed onto a square of plywood, which was fixed onto an angle bracket attached to a metal Y fence post (1.5 m long). The posts were set into the ground with the traps at c. 1 m above ground (Howlett et al. 2009). The collection fluid for the window traps was 500 mL of mono-propylene glycol, 500 mL of 70% ethanol, a pinch of salt, a squirt of dish detergent, and 1 L of tap water. Extra water was added periodically to counteract evaporation to ensure that the bases of the containers remained covered in liquid over the five days that they were set. The invertebrates collected in the window traps were sorted to Order level before Coleopterans and Dipterans were further sorted into the lowest taxonomic level possible. Bumble bees and native bees were dried and then identified to morphospecies using the keys given in Donovan (2007). Data are presented as the sum of the six traps per site across all sampling days. Introduced bumble bees that appeared to be *Bombus ruderatus* are referred to as *B. ruderatus* throughout, even though there is a small chance that *B. hortorum* were present, as it is difficult to distinguish these species once caught in window traps (Donovan 2007; Ashley et al. 2019).

Bird diversity and bat activity

Two audio recording devices (AR4 V1.4, Department of Conservation) were used to monitor birds and bats at each site. The recorders were zip tied to a mānuka plant at around 1 m height at the top and bottom of a transect. If no appropriate plants were available, a 1 m steel Y fence post was substituted (Colbourne et al. 2020). They were set to record bats overnight and the setting "Tier 1 birds" was used to record during the day, which records from 07.00 h through to 13.00 h on the "high" acoustic setting. The recorders were set for four days and five nights in the January trip to collect bird and bat data, and for 20 nights in the May trip for bats (excluding gaps in the data).

Five bird species were chosen as indicator species to focus on: fantail | pīwakawaka (*Rhipidura fuliginosa*), North Island robin | toutouwai (*Petroica longipes*), tui | tūī (*Prosthemadera novaeseelandiae*), bellbird | korimako (*Anthornis melanura*), grey warbler | riroriro (*Gerygone igata*), and New Zealand falcon | kārearea (*Falco novaeseelandiae*). These species were chosen as they are predators of a wide range of insects targeted in our sampling methods, with the exception of the New Zealand falcon, which is known to eat the insectivorous birds chosen. In addition, the bat recordings were assessed to determine the presence and call frequency of long-tailed bats | pekapeka (*Chalinolobus tuberculatus*).

Freshwater stream health

Stream health was assessed at three stream sites (Fig. 1a) in May 2022 (Austral autumn) using the stream health management and assessment kit-2 (SHMAK) (NIWA 2019). There was insufficient water flow in January 2022 to conduct summer stream health assessments. The stream sites were independent from the terrestrial invertebrate/bird/bat sites, with one stream passing through a catchment with predominantly native bush ('native forest stream'), the second through mānuka forest ('mānuka stream') and the third through grazed pasture ('pasture stream') (Fig. 1a).

The mean width and depth of each stream were calculated from multiple measures along a 20 m reach. Temperature (HANNA EC Tester-HI98303) (°C), and electrical conductivity (HANNA EC Tester-HI98303) (µS cm^{-1}), were recorded at each stream site. Stream water samples were analysed for phosphate (HANNA Handheld Colorimeter Phosphate LR - HI7 13-0 (Mg L^{-1} P, Mg L^{-1} P-P) and nitrate-nitrogen (microtest Nitrate- N NED^{HS}) (Mg L^{-1} N-N). Possible detection readings for phosphorus were 0.2 mg L^{-1}, 0.4 mg L^{-1}, and 0.8 mg L^{-1}. Stream visual assessments and scoring were undertaken as per SHMAK kit-2 instructions and included assessments of stream bed, bank vegetation, sediment deposits, habitat for aquatic animals, flow types, bank stability and erosion, riparian buffer width and intactness, shade, and channel alteration as part of the physical health rating.

The C1 protocol (Stark et al. 2001) was followed for semi-quantitative sampling of the hard-bottomed streams (native forest stream and mānuka stream). Macroinvertebrates were caught by agitating the rocks on the stream bed and having the net downstream to collect the organisms. This was repeated until a total area of 1 m^2 riffle habitat was sampled. The C2 protocol (Stark et al. 2001) was followed for semi-quantitative sampling of the soft-bottomed stream (pasture stream). Submerged woody debris, bank margins, and macrophytes were swiped/jabbed with the net to dislodge and collect organisms. A total of 3 m^2 of soft-bottomed stream was sampled. Samples from both types of streams were stored in 70% ethanol, taken back to the lab, identified down to genera where possible, and assigned taxa values based on whether they were caught in a hard-bottomed stream (Stark 1985) or a soft-bottomed stream (Stark & Maxted 2004, 2007).

Data and statistical analysis

The AviaNZ (AviaNZ, v. 3.2) software was used to assess the data for target bird species. The AviaNZ manual selection was used to review the bird recordings for five minutes out of every 30 minutes for the six hours of recordings per day, for the four days of recordings. The presence of a target bird in a five minute period was recorded as a bird call, and each five minute period with a presence was tallied. The data from the two AR4s per site were pooled to get one data set per site. This equates to a potential maximum of 96 bird calls per bird per site (Two five minute samples per hour, six hours per day, four days, two recorders = 96 sampling points).

The BATSEARCH3 (Department of Conservation, v. 3.23) computer program was used to assess the data for bat calls. Spectrograms were reviewed for bat calls (defined as a spectrogram showing identifiable bat echolocation calls), and the number of bat calls per night was tallied for each bat monitor at each site. Due to the proximity of the paired monitors, there is a high likelihood that the same individual would be recorded on both monitors at each site within a short period of time, so we took the maximum nightly value from each night at each site, and then derived a mean and standard error for the nightly number of bat calls detected per night.

R 4.2.0 (R Core team 2022) was used to analyse the data for terrestrial invertebrates, birds, and site characteristics, whilst GraphPad-prism (10.0.3) was used to produce the figures. General data manipulation was done using *data.table* (Dowle & Srinivasan 2021).

The Shannon-Wiener diversity index, the Gini-Simpson index, and Bray-Curtis dissimilarity diversity metrics were used to assess community composition of insects and birds at each site. The Shannon-Wiener diversity index and Gini-Simpson index (referred to as Shannon's diversity (H') and Simpsons index (D), respectively, in the R program) were calculated using the R package "vegan" for the invertebrates caught in window traps. Using the R package "betapart" and the code set out in Baselga (2017), the Bray-Curtis total abundance-based multiple-site dissimilarity across sites was calculated for the insect data and bird call data.

The total macroinvertebrate abundance, taxon richness, number of Ephemeroptera, Plecoptera and Trichoptera taxa (EPT richness), percentage contribution of EPT taxa to the total macroinvertebrate taxa per stream site (% EPT), Macroinvertebrate Community Index (MCI) score for New Zealand macroinvertebrate species, and Average Score Per Metric (ASPM, a combination of EPT richness, %EPT, and MCI) were calculated for each stream site. The MCI and ASPM scores were then compared to the bands set out in the New Zealand national policy statement (New Zealand Government 2020).

Results

Mānuka tree diameter at breast height increased and distance between trees decreased from the young planting to old regen sites. Mānuka tree heights were similar at the mature planting and old regen sites, and the trees were shorter at the young planting (Fig. 1f) site. No mānuka plants were present near the transects at the pasture site.

Terrestrial invertebrates

A total of 5715 invertebrates were collected in the window traps. These were identified to be from 13 orders (Fig. 2a–b). Hymenoptera accounted for over half of the invertebrates caught, dominating samples from pasture and young planting (Fig. 2a). This was found to be mainly driven by a high number of honey bees (*Apis mellifera*) being present in samples from these sites. To aid in visualisation of non-*A. mellifera* taxa, *A. mellifera* were removed from Figure 2b. With *A. mellifera* removed, the total number of individual invertebrates collected were more similar among sites (ranging from 582 to 676),

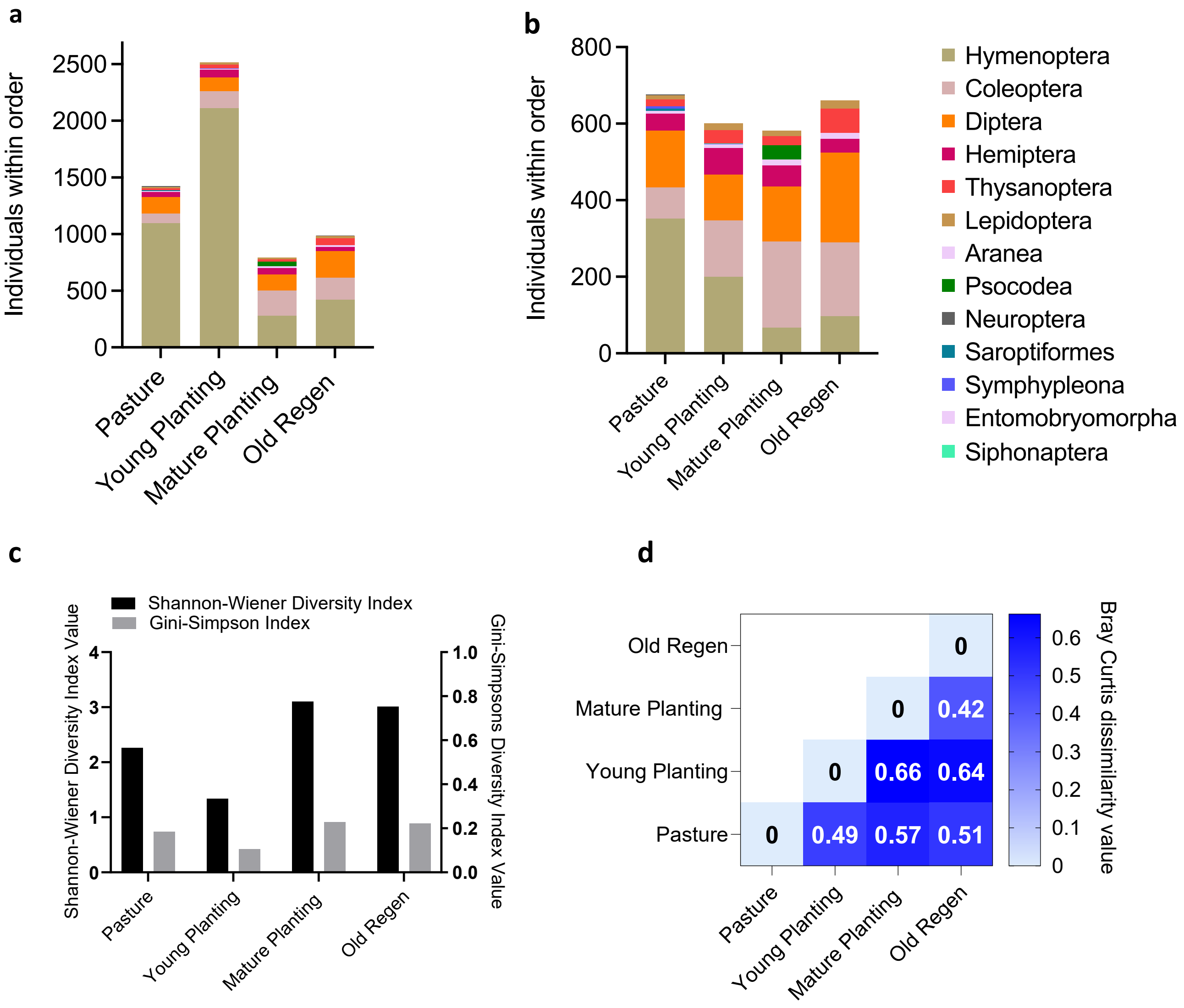

Figure 2. Terrestrial invertebrate community composition. The total number of terrestrial invertebrates caught in all window traps per site across all sampling days, identified to Order (a), identified to Order with *A. mellifera* removed from samples (b), the Shannon-Wiener diversity index and Gini-Simpson index for each site (c), and the terrestrial invertebrate Bray-Curtis dissimilarity matrix (d).

however community composition at each site varied. Numbers of Coleoptera had a general trend of increasing from pasture through to old regen (Figs. 2b & 3a). There was a larger number of non-*A. mellifera* Hymenoptera at the pasture site compared to the other sites, accounting for around half of the individuals caught at this site. Diptera numbers were highest at old regen and pasture, with young planting and mature planting having similar numbers to each other.

The Shannon-Wiener diversity index indicated that mature planting and old regen were very similar and had the highest diversity index values, whereas young planting had the lowest value (Fig. 2c). A similar pattern was seen in the Gini-Simpson index (Fig. 2c). The Bray-Curtis dissimilarity values indicated that mature planting was most like old regen and least similar to young planting (Fig. 2d). Pasture was most like young planting and least similar to mature planting. Other than mature planting and old regen (42%), all other sites had ≥ 49% dissimilarity from each other, suggesting greatly different community compositions between sites.

From the Orders identified (Coleoptera, Diptera, and Hymenoptera), eighty taxa within these Orders were further identified and analysed (Fig. 3a, c, e).

Coleoptera

There was a general trend of increasing Coleopteran abundance from pasture to old regen, with mature planting having the highest abundance (Fig. 3a). Of the 40 taxa identified, only six taxa were found at all sites, nine at three sites, and four at two sites. Over half of the taxa (21) were only found at one site, and are hereby referred to as unique taxa. Old regen had the highest number of coleopteran taxa (21), and of unique taxa (8), whilst pasture had the lowest number of taxa (17), and pasture and young planting had the lowest numbers of unique taxa (4) (Fig. 3a).

Old regen had the highest Coleopteran Bray-Curtis dissimilarity value when compared with all other sites, but in particular when compared to pasture (71% dissimilarity) (Fig. 3b). This indicates a highly different Coleopteran community composition between old regen and pasture. The sites that had the most similar Coleopteran community composition were young planting and mature planting (36% dissimilarity), and there was high dissimilarity between mature/ young planting and old regen (62–63%; Fig. 3b).

Diptera

There was lower total abundance of Diptera at young and mature planting sites (132–148 individual species) compared to pasture and old regen, which had similar abundance (221–237 individuals; Fig. 3c). There was an increasing abundance of Dolichopodidae across sites that corresponded with increasing age of mānuka trees, and the opposite trend was seen with Ephydridae. Of the 34 Diptera taxa identified, ten were collected at all sites, seven at three sites, and six at two sites. There were 11 unique taxa (Fig. 3c). Mature planting and old regen had the highest number of taxa (23), young planting had the lowest (17), while young planting also had the lowest number of unique Dipteran taxa (1), and old regen had the highest (4). As with Coleoptera, young planting had the most similar community composition for Diptera to mature planting, and old regen was most dissimilar to pasture (Fig. 3d).

Hymenoptera

There was a high abundance of honey bees (*A. mellifera*) at the young planting site, at which honey bees accounted for over half the total abundance (Fig. 3e). Hymenoptera community composition differed the most between young and mature planting (78% dissimilarity) and young planting and old regen (68% dissimilarity), while the most similar were the mature planting and old regen sites (23% dissimilarity) (Fig. 3f).

With honey bees removed from the Hymenoptera samples, trends in other Hymenoptera can be seen more easily (Fig. 3g). Abundance of Hymenopterans decreased from pasture to mature planting and old regen. Pasture and young planting had the highest abundance of *Bombus* sp., *Lasioglossum* sp., and *Leioproctus* sp., with *Lasioglossum* sp. being considerably more abundant in pasture than the other sites. The sites with the biggest between-site difference in Hymenopteran community composition changed from mature planting and young planting, when including *A. mellifera*, to pasture and mature planting/ old regen (60% and 51% dissimilarity, respectively) with the removal of *A. mellifera* (Fig. 3h). While old regen and mature planting still had the most similar Hymenoptera community composition with *A. mellifera* removed, there was a large decrease in the dissimilarity between community composition of old regen and young planting, going from 68% to 34% dissimilarity when *A. mellifera* were removed (Fig. 3f, h).

Indicator bird activity

Old regen and mature planting sites had the highest indicator bird activity, with 68 and 157 calls, respectively. The grey warbler was the most active species, accounting for over half of the calls heard at each site. Young planting had the lowest species richness, with three out of five of the target species heard, whereas pasture and old regen had four and mature planting had five target species (Fig. 4a). Of the target species, grey warbler and New Zealand falcon were the only species found across all of the sites. While the falcon was identified at all sites, it was only heard during one 5-minute period for all sites except for old regen (nine calls). Grey warbler, bellbird/ tui, and North Island robin call numbers were substantially higher at mature planting and old regen sites compared to pasture and young planting. Fantail call rates were low (fewer than nine calls) across all sites, and at the old regen site there was no recorded fantail activity (Fig. 4a).

Young planting and pasture were highly similar in community composition based on the Bray-Curtis dissimilarity values (10%; Fig. 4b). In contrast, mature planting was highly dissimilar to both pasture and young planting (73–75% dissimilarity). Old regen was over 49% dissimilar to all other sites (Fig. 4b).

Bat activity

All sites had bat calls recorded during the austral summer assessment period. The only species detected was the long-tailed bat (*Chalinolobus tuberculatus*). The mean number of overall bat calls per night per site increased from pasture through to old regen, corresponding with age of mānuka at each site (Fig. 5), with a substantially higher mean number of calls being recorded at old regen compared with the pasture and young planting sites (Fig. 5). No bat calls were detected at pasture and young planting sites during the Austral autumn assessment period. Similar numbers of calls were recorded at the old regen and mature Planting sites (Fig. 5b). Overall there were a substantially higher number of calls in the summer assessment period compared to autumn (Fig. 5).

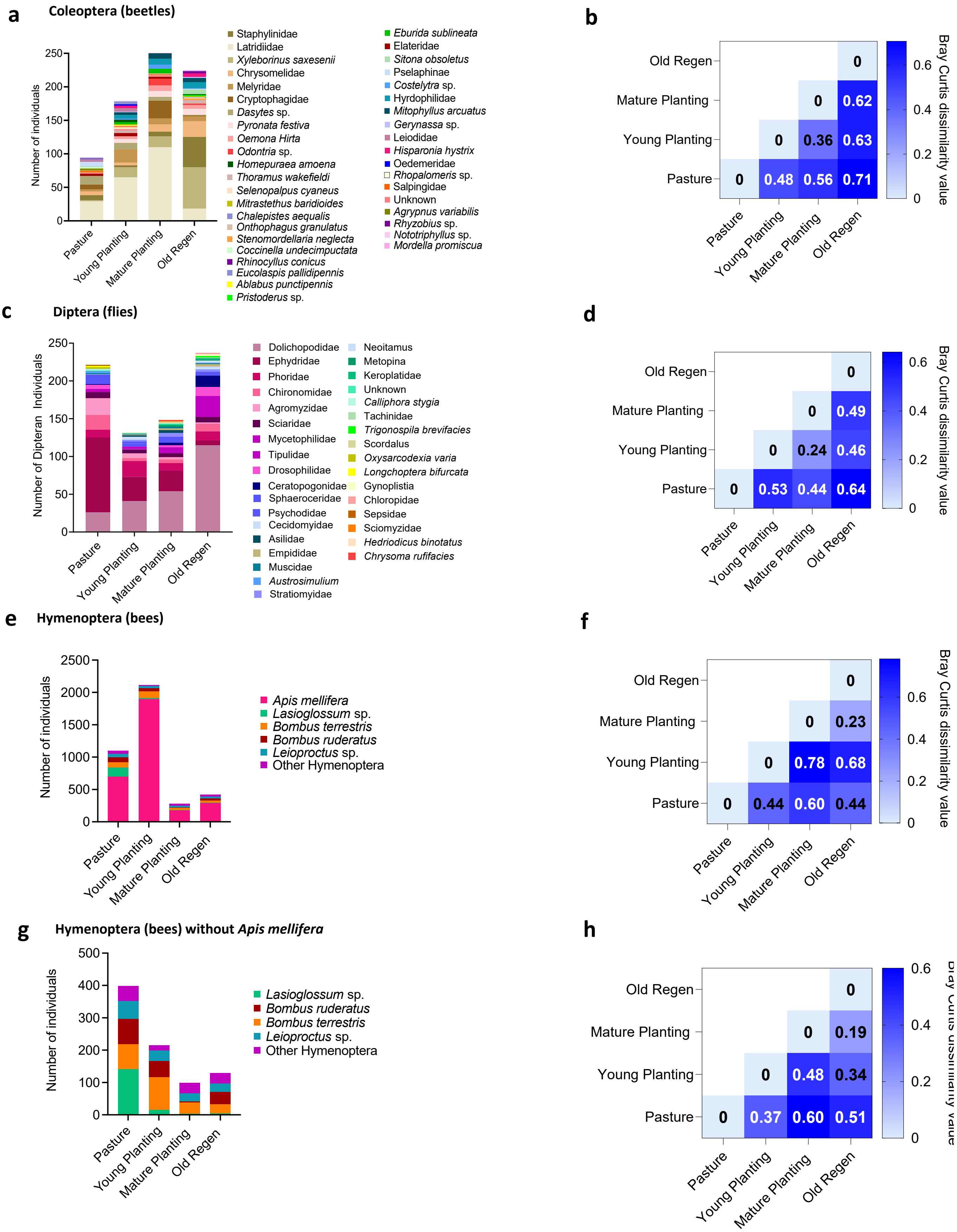

Figure 3. Terrestrial invertebrate order diversity. Coleoptera community composition (a), Coleoptera Bray-Curtis dissimilarity matrix (b), Diptera community composition (c), Diptera Bray-Curtis dissimilarity matrix (d), Hymenoptera community composition (e), Hymenoptera Bray-Curtis matrix (f), Hymenoptera with *A. mellifera* removed community composition (g), Hymenoptera with *A. mellifera* removed Bray-Curtis dissimilarity matrix (h).

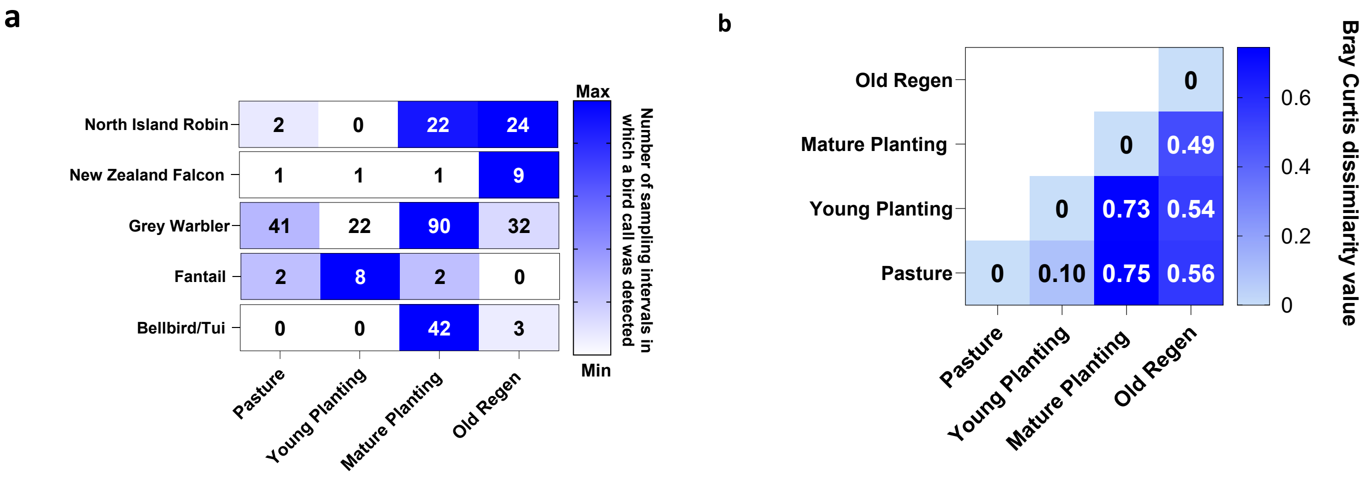

Figure 4. Bird diversity and relative abundance. Number of sampling intervals in which a bird call was detected per species (North Island robin, New Zealand falcon, grey warbler, fantail, bellbird/ tūi) (a), target bird species Bray-Curtis dissimilarity matrix (b).

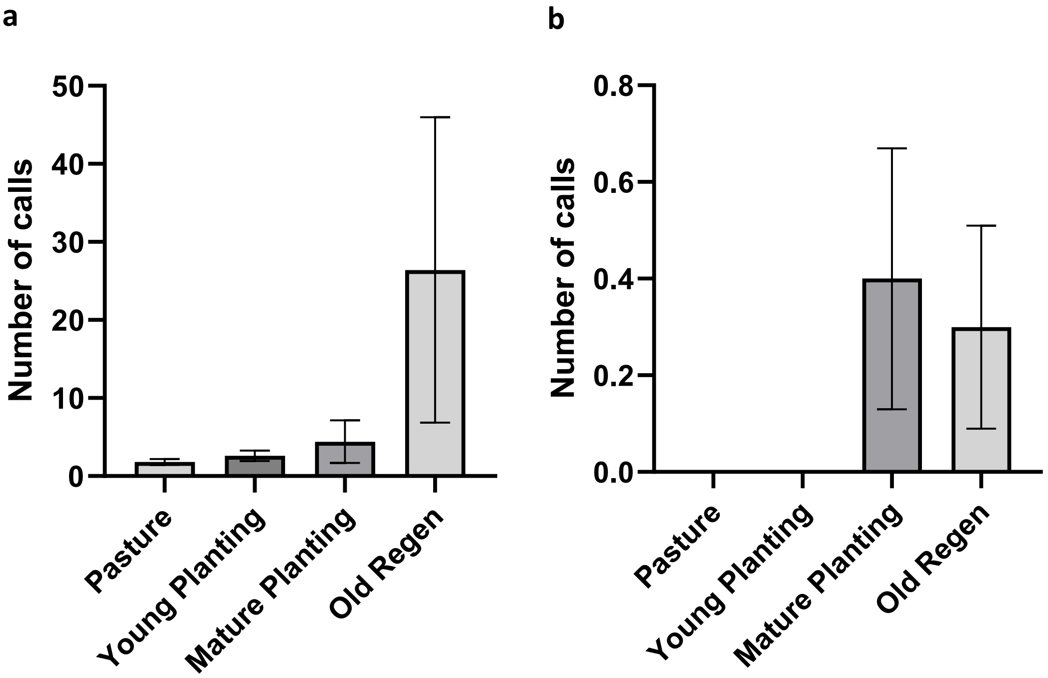

Figure 5. Bat abundance. Mean (± standard error of the mean) number of long-tailed bat (*Chalinolobus tuberculatus*) encounters recorded per night (using AR4 audio recorders) at the four study sites.

Freshwater stream health

The mānuka stream site had the highest macroinvertebrate count, taxon richness, and EPT richness compared to the pasture and native forest stream sites (Table 1). The pasture stream accounted for only 5.2% of the total macroinvertebrates collected across all stream sites, while the native forest stream accounted for 34%. Mānuka and native forest streams had similar MCI and ASPM scores, which were 35–40% higher than those in the pasture stream (Table 1). Native forest and mānuka stream EPT richness percentages were > 2 times that of the pasture stream (Table 1).

Both the mānuka and native forest stream sites had lower conductivity and nitrate-N scores and higher physical health ratings than the pasture stream site, with minimal differences between mānuka and native forest stream scores (Table 2). Water temperature was highest at the native forest stream site. Phosphate levels were below detection levels (Table 2).

Discussion

In Aotearoa | New Zealand there is a developing practice of converting marginal high hill-country grazing pasture to planted mānuka forests for the purpose of delivering a sustainable and secure resource for the production of high quality monofloral mānuka honey. The impact of converting exotic pasture to native planted mānuka forests on local ecosystems is largely unknown. Here, our assessment of ecosystem and biodiversity measures (indicator bird presence and invertebrate surveys) at a large high hill-country station generally indicates that in the continuum

Table 1. Biological stream health. Count of macroinvertebrates, taxon richness, EPT richness, percentage EPT, macroinvertebrate community index (MCI), and average score per metric (ASPM).

Site	Count	Taxon richness	EPT richness	% EPT	MCI	ASPM
Pasture	47	14	5	35.71	105.71	0.35
Mānuka	548	17	12	70.59	141.18	0.61
Native forest	308	13	11	84.62	144.62	0.65

Table 2. Physical and chemical stream health. Average stream width, max stream width, max stream depth, water temperature, conductivity, nitrate-N, phosphate, phosphate-P, physical health rating.

Site	Average stream width (m)	Max stream width (m)	Max stream depth (m)	Water temperature (°C)	Conductivity (µS cm^{-1})	Nitrate-N (mg L^{-1})	Phosphate (mg L^{-1})	Phosphate- P (mg L^{-1})	Physical health rating
Pasture	1.66	1.79	0.36	11.0	120	0.4	< 0.01	< 0.01	31 (Fair)
Mānuka	2.36	5.60	0.62	11.3	69	0.1	< 0.01	< 0.01	53 (Good)
Native forest	4.41	5.36	0.48	13.6	79	0.2	< 0.01	< 0.01	50 (Good)

from pasture to older mānuka forest, five-year-old planted mānuka forests are similar to 30-year-old naturally regenerating mānuka forest, whilst three-year-old mānuka is most similar to pasture and has a lower diversity of invertebrates, birds, and bats than the older forests. Furthermore, stream health metrics in the mānuka catchment showed improved health relative to the pasture catchment. As far as we are aware, this is the first multi-dimensional biodiversity survey that has been undertaken in mānuka forests, and because we were able to detect substantial differences in communities between pasture and different aged (from three years to regenerated for 30 + years) mānuka forest sites, we believe this demonstrates the viability of combined terrestrial invertebrate and bird and bat surveys for the assessment of the ecological changes of developing planted mānuka forests, in conjunction with other ecological and environmental measures.

Forest complexity and community composition changes with successional stage, which is reflected in the suite of species that were observed in the forest (Brockerhoff et al. 2008a). While mānuka forests are initially planted as a single-species seedling forest, more native plant species are likely to establish as the forest matures (Bergin et al. 1995). Due to the prior land use of these planted mānuka forests and the current surrounding agricultural land use, there is also a high chance of non-native, and invasive, plant species establishing (Rossignaud & Hulme 2023). It is unknown what management, if any, will be required to control the growth of other plant species if they start to outcompete mānuka or reduce flowering, and these future management needs may affect the longer-term ecological trajectory of these sites. However, as the mānuka canopy closes, a change in understorey plant species composition is likely to occur and this will lead to shifts in faunal communities, as seen in other studies where canopy closure suppressed non-native ground cover species and created stable low-light and higher-humidity environments for the growth of understorey native plants (Doroski et al. 2017; Wallace et al. 2018; Wallace & Clarkson 2019). The canopy had closed in some areas of both the mature planting and old regen sites, which may explain some of the similarities in faunal community composition in response to the shift away from non-native understorey plants to native plants and increased leaf litter.

In general, native species richness increases with time at restoration sites on degraded land (Atkinson et al. 2022). This paralleled results observed here, with the oldest planted mānuka site assessed showing community composition more similar to the much older naturally regenerated forest than the younger mānuka planting or the pasture site. While this was in line with our hypothesis, we did not anticipate such a substantial and detectable shift within just five years after planting. Rather than a continuum of change between the pasture site to the older naturally regenerated site, our results show that pasture and young planting sites are similar to each other, however, the mature planting that was just five years old already exhibited significant differences in faunal composition that were comparable to the old regen (> 30 years naturally regenerated) site. It should be noted that the naturally regenerated site is still regenerating, and its community composition does not resemble primary forest in the area. However, it is broadly characteristic of older naturally regenerated mānuka forest with a complete canopy cover and a diversity of understorey and new canopy species coming through, and thus acts as a control for the ecological values of the property if it were left to regenerate naturally rather than being deliberately planted in mānuka forest.

These major changes, which appear to be occurring within the first five to six years of plantings, need to be validated by studies at other sites. However, the design of this case-study is that all sites were within very close proximity (3–6 km apart), so we would expect limited geographical location-associated disparity in faunal communities or initial species assemblages.

The potential benefit of planted mānuka forests for the wider ecosystem is exemplified in our stream surveys. The physical habitat assessment scoring, such as amount of fine sediment, bank stability, erosion, and vegetation on the bank, were all improved in the mānuka-dominated catchments compared to the pasture stream site. The native and mānuka forest streams were also much higher in EPT richness, % EPT, MCI, and ASPM scores when compared to the pasture stream, indicating that these streams were healthier during this sample period, supporting a more diverse and pollution-sensitive macroinvertebrate community (Scarsbrook & Halliday 1999). The width of the streams was also consistent with this trend, with pastoral streams being narrower than native forest streams due to the infill of sediment from bank erosion caused by stock and by lack of vegetation on the stream bank (Davies-Colley 1997). The higher conductivity and nitrate levels at the pasture stream site were consistent with other studies of pasture streams due to increased stream bank erosion and effluent input from the stock, particularly from cattle (Wright 2013; Betteridge et al. 2017). However, conductivity and nitrate levels were all within the optimal range of < 150 µS cm^{-1} and < 1 mg L^{-1} according to Land, Air, Water Aotearoa set thresholds. Phosphate levels were below detection level at all of the streams: this result is uncommon and requires further testing to see if this is a regular occurrence, or if it was a one off, perhaps due to rain a few days before the time of sampling which may have diluted the samples. These results were from one point in time, and from one 20 m stretch of stream within a relatively small catchment area, and thus the true extent of change between these stream sites will only be revealed following repeated sampling.

Our data provide evidence that planted mānuka forests may have a positive impact on ecological communities even though this land use change had only been occurring over a five year period at the time of our study. In comparison, studies of riparian plantings on New Zealand farms indicate that 8–20 years are needed to observe significant changes in fish and invertebrate communities (Parkyn et al. 2003; Jowett et al. 2009). In many instances the riparian buffers in studies on the effects of riparian plantings (Greenwood et al. 2012; Baillie & Neary 2015) were less than the recommended 10 m per stream side (Parkyn et al. 2000), compared to the more catchment scale replanting in the current study. The increase in width to catchment scale is likely contributing to the rapid improvement. This is in line with a review on studies of afforestation of catchments with *Pinus radiata* which also shows freshwater quality improvement within five to six years of planting (Baillie & Neary 2015). A major difference between the catchment scale planting of mānuka forest and the more common scenario of riparian planting within still actively grazed farms is the initial preparation of the property for planting. This involves extensive removal of stock from the catchments and weed control to enable plant establishment, as well as the extent of planting (whole catchments vs. stream edge). Stock removal can have a large effect on the environment by lowering the nitrogen input and erosion levels, particularly when cattle are removed (Line et al. 2000). At the planted mānuka sites studied here, cattle were removed and sheep numbers were greatly reduced before planting, but feral deer are still abundant. Site preparation to remove vegetation such as grasses that could compete with the mānuka could have caused the observed initial reduction in terrestrial invertebrate diversity seen at the young planting site relative to pasture, however this reduction is reversed in the five year and older mānuka forests.

The numbers of calls recorded for our target bird species differed across the four sites. Grey warbler and fantail numbers did not appear to differ across sites, whilst North Island robin, tui/bellbird, and New Zealand falcon were found more often at the older mānuka sites. Some species are known to be more resilient to land-use change, in particular the smaller, more mobile species such as grey warbler and fantail (Walker et al. 2021). Grey warblers were the most active target species across all sites, but particularly at the five-year-old mature planted mānuka forest. They are widespread across New Zealand, found in most habitat types. Areas of thick mānuka are one of their preferred places to nest in (Gill 1983; Powlesland 2013), and the mature planting site had a high stem density of mānuka and a high level of canopy closure. It is possible that the high falcon activity at the naturally regenerating sites reflects a better foraging and nesting habitat for these large predatory birds. North Island robin and tui/bellbird are more associated with forest and areas of native scrub than with agricultural land, and thus are likely good indicators of the maturity and diversity of native forest ecosystems (Powlesland 2013; Robertson 2013; Sagar 2013). Overall, there was a substantially greater difference in bird community composition between the pasture and mature planting sites compared to the difference between the mature planting mānuka site and the old regen site, further suggesting that the largest biodiversity shift may occur within the first five years of mānuka planting. The mature planting and old regen sites had similarly high mānuka stem density and height, whilst the mānuka at the young planting site were much shorter and further apart. This change in habitat complexity, with associated changes in invertebrate communities, is likely a key driver in the change in bird species community (Ruffell & Didham 2017).

Critically endangered long-tailed bats were found at all sites during the summer sampling period, and the old regen site had a significantly higher level of activity recorded compared to the other sites. During the Autumn assessment period, the mature mānuka planting and old regen sites were the only sites to record bat activity. With the lower temperatures in autumn, it is likely that the long-tailed bats were in torpor and would have lower activity levels within a smaller home range to conserve energy (McNab & O'Donnell 2018). As the only bat calls were recorded at the two older mānuka sites, this environment might be a more suitable habitat (in terms of prey abundance), or be within their reduced home range. As long-tailed bats nest in larger tree trunks, cavities, splits, and peeling bark, it is unlikely that the bats are using the young mānuka plants for roosting; however, they may be roosting in nearby remnant native forest and using the mānuka forests as foraging grounds. Bats are thought to show similar patterns of movement in pine plantations (Borkin & Parsons 2010). The use of mānuka as a foraging site, particularly the two older mānuka sites, could provide valuable food resources to support populations based in the remnant native forest stands. The most common habitat for New Zealand bats is older native forest, however, long-tailed bats, have been found in pine plantations (Borkin & Parsons 2010), suggesting that they can exist in single-species planted forests. The restoration of pasture to mānuka forest decreases the fragmentation of the native bush cover, potentially improving foraging success at a landscape scape. Long-tailed bats prefer more closed-in habitat for roosting, but prefer forest edges for feeding. Further research is required to establish whether planted mānuka forests provide beneficial new habitat to support long-tailed bat populations (Borkin & Parsons 2010).

Invertebrates make up the largest biomass of animal taxa globally. With their short regeneration times and multitude of species characteristics and niches, some invertebrate taxa can be used as biological indicators of the health of the environment they are in. The planting of mānuka on pasture land changes the environment, increasing the number of habitats available for invertebrates via increased woody species, and leading to wood detritus, leaf litter, foliage, and flowers. Coleoptera can be used as a biological indicator taxon due to their life history traits and their response to environmental change (Niemelä 2001; Albert et al. 2021). Coleoptera appears to be the best taxon to demonstrate the increasing dissimilarity to pasture as mānuka forest matures, at least as regards community abundance and composition. This change in composition is particularly notable with the increase in species associated with mānuka and/or with forest species, such as saproxylic beetles, which are dependent on dead and dying wood or wood fungi to live (Parisi et al. 2020). In a study by Watts et al. (2008), where mānuka was used to help restore peat bogs on old mine sites, factors such as canopy density and height, total vegetation cover, and mānuka cover were important influences on beetle assemblages.

One limitation of this study was the cross-sectional (space-for-time substitution) case study design, meaning that there was not replication of assessments at independent locations, or over time. While it would be difficult to find another location/property with pasture and the range of different aged mānuka forest within close proximity, this study does set a baseline for future longitudinal monitoring studies. Whilst all assessment sites were in a similar geographical location, each site has its own history and confounding variables that may have contributed to some of the differences in ecological diversity seen independent of mānuka forest age. One such difference between sites was the density of mānuka planted. The mānuka at the mature planting site was planted at a higher density than the young planting site, which gives the canopy less open space to develop and hold flowers without encroachment. Furthermore, the distance to the nearest native woody forest was not measured for each site, and could have been slightly different. This is known to be an important factor in native regeneration as the species from the remnant native forests can disperse into regenerating sites (Mason et al. 2013; Pedley et al. 2023). Further assessments like this, repeated over subsequent years, will provide detailed insights into the ecological changes associated with this novel land-use change.

Other limitations of this study reflect the challenge with accessing some areas of the site due to terrain constraints, and the challenge of broadscale ecological monitoring across such large sites. For logistical reasons, we chose window traps as the primary invertebrate sampling method. These traps are best at collecting aerial invertebrates, and as such this study is likely biased towards these species. Future studies should also consider methods that target plant-dwelling and ground-dwelling invertebrates, such as tree-beating, artificial ground-cover objects, and pitfall traps. The species collected may have also been affected by the density of vegetation around each trap; at the pasture site there was no vegetation in the way, but in contrast, the mature planting and old regen sites had dense vegetation around each trap. It remains a challenge to design a consistent window trap method that can be used between years and between sites with different successional stages. This study did not measure distance to honey bee hive locations from each study site, as hive densities were high during the summer period throughout the site, and we assumed honey bee densities would thus be similar. However, we observed significant differences in the abundance of honey bees in the window traps at each site, and this may have had an impact on the number and diversity of non-honey bee insects collected in the window traps.

Planted mānuka forests may not only provide benefits for mānuka honey producers, beekeepers, landowners, and the rural New Zealand economy, but this novel land-use may also provide significant ecological benefits in comparison to grazed marginal hill-country pastures. If the current trend of land-use conversion continues, these changes may have a landscape scale benefit which could complement public conservation land by supporting native biodiversity. Although mānuka at these sites are planted as a single-species forest, natural successional processes are likely to occur, increasing the number of plant species and thus overall biodiversity over time.

Ongoing biodiversity monitoring is required to document these ecological changes, and to ensure any unintended negative consequences are detected early. The apparent significant shift in ecological communities and stream health within the short time frame of current planted mānuka forests suggests that this land-use change could have substantial benefit for freshwater quality and ecosystem health. Confirmation of these changes by replication at other sites could spur further retirement of grazing land to provide ecological values while also providing a new economic activity for landowners. In this cross-sectional (single location space-for-time substitution) case study, we found that a combination of monitoring measures, including invertebrates, birds, bats, and stream health metrics, provides a good indication of ecological changes in mānuka plantings. We recommend that longitudinal studies incorporating these elements should be implemented in order to gain further data to assess these changes, and to provide confidence that this novel land-use change is beneficial for the wider environment.

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Additional information and declarations

Conflicts of interest: T.L.M. and J.E. are employees of Comvita NZ Ltd. G.N.W. was an employee of Comvita NZ Ltd. after the study was conducted, and during manuscript preparation. All other authors declare no additional competing interests.

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Data availability: The datasets generated for this study are available on request to the corresponding author.

Author contributions: G.N.W., D.E.P., and J.E. conceived the idea and designed the study. G.N.W. carried out data collection. G.N.W., D.E.P. and T.L.M. carried out the analysis and wrote the manuscript, with editorial contributions from J.E.

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Supplementary material

Additional supporting information may be found in the online version of this article.

Appendix S1. Site aspects, slopes and altitudes.

The New Zealand Journal of Ecology provides online supporting information supplied by the authors where this may assist readers. Such materials are peer-reviewed but not copy-edited. Any issues relating to this information (other than missing files) should be addressed to the authors.