Ministry for Primary Industries
Manatū Ahu Matua

Indirect estimation of gorse and broom 'non-forest land' to 'forest land' transition

Final Report

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Executive Summary

The presence of forest species determines whether stands of gorse (*Ulex europaeus*) or broom (*Cytisus scoparius*) are classed as 'forest land' rather than 'non-forest land' within New Zealand's interpretation of Kyoto Protocol 'forest' definitions. 'Forest land' in the New Zealand Emissions Trading Scheme (ETS) has forest species (i.e. trees that reach at least 5 m height at maturity in situ) that are likely to achieve more than 30% crown cover on a per-hectare basis. The presence of sufficient seedlings of forest species within gorse or broom stands, coupled with human-induced action, is thought to indicate that these stands are capable of regenerating tall forests and could therefore be considered within the 'forest land' category. Use of remotely-sensed imagery to determine whether sufficient forest species are present in areas of gorse or broom to qualify the area as forest land is problematic, especially if they are obscured by the gorse or broom canopy. Therefore, this study aimed to determine national-scale indirect measures of 'likelihood', or ecological predictions on whether a given area of gorse or broom will become 'forest land' and the time taken for an area of gorse or broom, with regenerating forest species, to become 'forest land'. We found that areas of gorse or broom across New Zealand that experienced relatively mesic climates (i.e. warmer winter temperatures, reasonable rainfall) and were close to indigenous seed sources were more likely to proceed to 'forest land' status. Steeper places were also more likely to have emergent indigenous forest species. Across three intensively-studied gorse sites, indigenous forest species were present within the gorse within 6 years of gorse establishment, on average, although 30% crown cover of the 16-m^2 plots was not achieved for an additional 10 years. For broom, at least one lifecycle (c. 15 years maximum) was completed at two intensively-studied sites before forest species became established. On average, 12 more years elapsed before 30% crown cover was achieved for the 16-m^2 plots. Bioclimatic differences were not picked up through the small number of sample sites per species, despite their obvious importance in the nationwide survey. Finally, we developed a new allometric relationship between biomass and gorse height that incorporates indigenous forest species coming through the gorse canopy. We found that gorse of 30 years of age is generally around 4 m of height and contains approximately 125 t C/ha (= 458 t CO_2e/ha).

To develop measures of 'likelihood' we conducted a national survey of gorse and broom along broad temperature and rainfall gradients, including stands both regenerating into tall forest and those where broadleaved species are apparently absent. We investigated factors that may determine the likelihood of a given area of gorse or broom becoming or remaining 'forest land'. These factors included biological influences, namely distance to nearest seed source, gorse or broom canopy height (presumably a surrogate for gorse or broom age), as well as abiotic factors of aspect, slope and climate (e.g. minimum temperature, rainfall).

Based on occurrences of forest species within survey plots, we then modelled the occurrence of emergent (i.e. above the gorse or broom canopy) indigenous forest species for the whole of New Zealand using boosted regression tree (BRT) modelling. Boosted regression tree modelling was adopted because it can model interactions between predictors very effectively, without requiring explicit interaction terms to be defined before conducting the analysis. The most important predictor of 'likelihood' of emergent forest species occurrence in gorse or broom is their age – this is not surprising as transition to 'forest land' becomes increasingly likely as time elapses. In addition, older stands (and those on steeper slopes) tend to be more open, therefore receiving the optimum light for forest species' establishment. Because we do not have spatially explicit national-scale estimates of gorse or broom age, a national probability distribution for emergent indigenous forest species was fitted using factors that we do have national spatial coverage for, namely slope, minimum temperature, mean annual

rainfall, mean annual temperature, mean October wind speed, soil water deficit and distance to nearest forest. These factors were all more important than any distinction between gorse and broom. Forest species' emergence becomes increasingly likely as all factors increase, except for mean temperature and soil water deficit, for which there are 'optimum' values for transition, and distance to seed source, which has a negative influence with increasing distance. In order to use our mapped distribution of 'transition likelihood', we would recommend that areas with a less than 0.5 probability of transitioning to 'forest land' (i.e. those mapped red or orange in Fig. 13 of the report) can be considered 'non-forest land' unless sufficient supplementary evidence is provided to the contrary. There will definitely be some exceptions within these areas.

A similar approach was used to develop a surface of 'likelihood' of occurrence of indigenous forest species within non-forested land (in this instance this was predominantly grassland), irrespective of the presence of gorse or broom. A database of functional traits of indigenous woody species was also used so as to more accurately determine the biological influences of succession in non-forested landscapes. Functional traits are those that define species in terms of their ecological roles, i.e. how they interact with the environment and with other species. This analysis showed that mean annual temperature and amount of local woody cover were the most important determinants of indigenous forest species' occurrence; that is, cold places and very open land are less likely to turn into 'forest land'. The functional trait analysis revealed that as temperature and woody cover decrease and distance from seed source increases, trait filtering intensifies for ecological strategy traits but not dispersal traits; that is, species that did not require an animal to disperse them or had several modes of dispersal (e.g. wind dispersal plus sticking to animals plus floating in water) were favoured when remote from existing indigenous forests.

In order to determine the average length of time for forest species to establish within a gorse or broom canopy and then achieve 30% crown cover we studied five sites spread over three bioclimatic zones of regenerating gorse (3 sites) or broom (2 sites). The achievement of 30% crown cover is significant as there is no longer any doubt that the land is 'forest land' once species capable of 5-m height achieve that crown cover per hectare. The bioclimatic zones were warm-temperate, cool-temperate-wet and cool-temperate-dry. Destructive harvests were carried out on at least 14 stands at each of the five sites in order to (a) age all forest species present in the plot within the gorse or broom canopy, (b) age the tallest gorse or broom, and (c) measure the biomass of the stands and test whether a previously-determined height–biomass allometry for gorse could be scaled with age and gave reliable estimates of biomass when indigenous forest species were included. We also used existing allometric relationships between crown cover and tree growth to model changes in canopy area during natural regeneration. This enabled backcasting of the number of years between gorse or broom establishment and forest species occurrence as well as the number of years for forest species to achieve 30% crown cover in the 16-m^2 plots. At sites that already had emergent forest species in selected 16-m^2 plots within the gorse or broom, the average time elapsed since gorse or broom establishment and forest species establishment was 3.6 ± 1.5 years. However, broom and gorse differed significantly, with an average gorse age of 6.18 ± 1.21 years when first trees established, compared with a mean broom age of −0.22 ± 0.88 years. For the two broom sites studied intensively, forest species are generally not establishing into the first cohort of broom, and approximately one cohort (c. 15-year maximum lifespan) completes its lifecycle before they enter. An absolute estimate of time elapsed since the beginning of the broom woody succession to forest species establishment is therefore not possible. However, across all sites an additional 11.1 ± 0.9 years is required until definitive 'forest land' status is

achieved, i.e. 30% crown cover, on the 16-m^2 plots. Given that individual 16-m^2 plots were selected on the basis of forest species' emergence, a few additional years will be required for a whole hectare to achieve 30% crown cover. There was a significant difference between species with gorse sites (mean time to 30% cover from initial establishment of forest species = 10.2 ± 0.53 years) tending to transition quicker than broom sites (mean = 12.2 ± 0.75 years). In this respect, only one between-site difference was significant, largely reflecting the different species involved. The average number of forest species stems at the point of 30% cover was estimated as 13.21 ± 1.62 stems/plot or 8256 ± 1013 stems/ha across all sites. Among the sites intensively sampled we find no evidence for a bioclimatic effect, but this should be investigated across a greater range of sites. However, the bioclimatic factors highlighted in our nationwide survey using supplementary spatial layers are dependable.

In general, total above-ground biomass (TAGB) in gorse was found to be reasonably strongly correlated with both shrub height and age. On this basis, several existing allometric models for predicting TAGB from gorse height were compared with sampled estimates and, where possible, new or improved relationships developed. The current model for predicting TAGB in gorse based on height worked reasonably well in sampled gorse stands, but potentially is better for gorse with continuous cover and low or no tree species cover. A better model for predicting gorse TAGB where biomass includes both gorse and trees – using gorse height – was produced from data for the three intensively-studied sites. This is considered better for taller/older gorse stands especially. Here, we show that 30-year-old gorse stands are generally around 4 m in height and contain approximately 125 t C/ha (= 458 t CO_2e/ha).

Finally, management to maximise the probability of gorse or broom becoming 'forest land' should include the prevention of disturbance, especially early in the succession when indigenous forest species are not yet established. Extensive disturbance, especially by fire, favours resprouting and germination of long-lived seeds of gorse or broom, rather than facilitating the transition to indigenous forest. Domestic or pest animals should be excluded from, or controlled within, the gorse or broom as they will preferentially browse the indigenous forest species.

1 Introduction

There are approximately 1.1 million hectares of ‘Grassland with woody biomass’, a vegetation category within New Zealand’s Land Use Map (further details in Ministry for the Environment 2012). Much of this vegetation type is potentially eligible as post-1989 regenerating forest land if sufficient evidence exists that per hectare it can support 30% crown cover of ‘forest species’ (i.e. trees ≥ 5 m in height) at maturity in situ. Within the Land Cover Database version 3 (LCDB3), there are c. 205 000 ha (19% of ‘Grassland with woody biomass’) of ‘gorse and/or broom shrublands’ (D. Pairman, Landcare Research, pers. comm.). Therefore, the distinction between ‘forest land’ and ‘non-forest land’ in shrublands determines both eligibility and liability for a given piece of land. Both the New Zealand Emissions Trading Scheme for Forestry (ETS) and the Permanent Forest Sink Initiative (PFSI) require human-induced action (i.e. management) to create ‘forest land’. The Ministry for Primary Industries uses the following definition (C. Trotter, pers. comm.): ‘Forest land comprises an area of at least 1 hectare that has, or is likely to have, tree crown cover from forest species of more than 30% in each hectare. Forest species are tree species capable of reaching at least 5 m in height at maturity in the place they are located. The phrase ‘likely to have’ in the definition of forest land is interpreted as there being a sufficient stocking of forest species that are expected to survive and grow such that >30% crown cover will likely be reached in the future. Although the time for >30% cover to be reached is not formally defined, 20–30 years is considered reasonable for slower growing indigenous species.’

Within the ETS and PFSI, a change in management activity is generally required to prove that this forest land will achieve its potential of 30% crown cover of forest species. This may involve stock exclusion, pest control or cessation of manual clearance/burning. Sporadic grazing or clearance of shrubland by burning commonly recurs where shrublands occupy a portion of land primarily managed for agriculture. This is especially true for gorse (*Ulex europeaus*) or broom (*Cytisus scoparius*), which are considered pest plants and therefore subject to regulatory control. In some instances, the gorse or broom is reverting to indigenous forest without management changes – in these cases, the land is deemed to be ‘forest land’ already. The presence of indigenous broadleaved species within gorse or broom, coupled with human-induced action, is thought to indicate that the stands are capable of regenerating indigenous forest species and are therefore ‘forest land’. ‘Direct detection’, or the use of remotely-sensed imagery to classify trees as forest species that can achieve 30% crown cover per hectare, can be problematic, especially if the trees are included within a gorse or broom canopy. Therefore, the purpose of this study was to use ecological modelling to provide an indirect indication (at national-scale) of the likelihood that a given area of gorse or broom will become forest land.

In order to develop measures of ‘likelihood’ we conducted a national survey of gorse and broom at 249 sites along broad temperature and rainfall gradients, noting both stands regenerating into tall forest and those where broadleaved species were apparently absent. A recent review suggested that both gorse and broom can act as a nurse crop for indigenous woody seedlings in the absence of repeated disturbance and where seed sources are plentiful (Williams 2011). The review also suggests that cooler sites (for gorse) and drier sites (for broom) may encourage self-perpetuation, although Williams (2011) concedes that these more extreme sites also had significantly fewer seed sources, and another author suggests that exclusion of browsing animals can make a significant difference to the rate of establishment of indigenous species (Wilson 1994). An earlier study, by Lee et al. (1986), suggested that warmer climates may favour faster replacement of gorse by indigenous plants because the seedlings grow faster and there is less gorse litter because of faster decomposition rates. The fastest recorded time interval until indigenous species dominate preceding gorse stands is around 30 years (Wotton & McAlpine 2013), although anecdotal evidence suggests that the most favourable areas within sites can succeed within 20 years (H. Wilson, Hinewai Reserve, pers. comm.). Some evidence suggests that the shorter lifespan of broom (c. 15 years) may result in a more rapid succession, perhaps facilitated by an intermediate canopy of an exotic species such as elderberry (*Sambucus nigra*) to ameliorate the conditions and promote regeneration of indigenous seedlings (e.g. Williams 1983).

In this report we investigate a range of biotic and abiotic variables that may impact on the likelihood of gorse or broom transitioning to forest land. Factors investigated include both biological influences – distance to nearest seed source, gorse or broom canopy height (presumably a surrogate for gorse/broom age) – and abiotic factors of aspect, slope and climate (minimum temperature, rainfall).

In addition to determining the likelihood of gorse or broom becoming forest land we also investigated the factors determining the likelihood of non-forested vegetation (predominantly grasslands) becoming ‘forest land’, irrespective of gorse or broom. The establishment of woody indigenous vegetation within grasslands depends on abiotic conditions, propagule availability, and histories of fire and grazing (Wardle 1991). Successful woody establishment, therefore, appears to depend on the contemporaneous removal of a set of multiple filters (sensu Dickie et al. 2007) such as limited seed source, appropriate moisture during seedling establishment and initial growth phase, and the absence of grazing animals that preferentially browse the indigenous species. In this study we tested which types of variables – land cover, proximity to seed source or abiotic stress – had a stronger influence on the likelihood of indigenous tree establishment. A database of functional traits of indigenous tree species was used to test how important these traits are in determining successful establishment within grasslands.

Functional traits are thought to determine species’ occurrences in response to both biotic and abiotic constraints (Mason et al. 2005, 2012). Functional traits are those that define species in terms of their ecological roles - how they interact with the environment and with other species (Díaz & Cabido 2001). Declining functional diversity indicates intensified filtering of environmental traits (i.e. exclusion of species with traits that are poorly adapted to local conditions) (Villeger et al. 2008; Mouchet et al. 2010). We can use indices of functional diversity to assess the relative importance of trait filtering for ecological-strategy traits versus dispersal traits in limiting tree establishment. Here we used these indices to determine the likelihood of grassland colonisation by indigenous trees. We would expect functional diversity for ecological-strategy traits to decrease as abiotic constraints on tree establishment increase (since only stress-tolerant species will be able to colonise). Functional diversity for height and seed size should decrease with decreasing woodiness in the landscape and increasing distance from forest (since only early-successional species will be able to colonise). Functional diversity should decrease for dispersal traits as dispersal limitation increases (since only species with high dispersal ability will be able to colonise).

In order to calculate the rate of tree crown area achieving 30% cover per hectare, we harvested trees in plots that would be assessed as ‘forest land’. These trees had already emerged through gorse or broom canopy and we determined their age in order to estimate the time between seedling establishment and qualification of plots as ‘forest land’. Harvest sites spanned three bioclimatic zones (warm-temperate, cool-temperate-wet, cool-temperate-dry). We then used existing allometric relationships between crown cover and tree growth to model changes in canopy area during natural regeneration to forest to estimate the number of years to 30% crown cover.

As carbon stock estimates are directly derived from biomass (the oven-dry weight of all organic material present), quantifying the biomass in these regenerating shrublands is an essential component of determining how much carbon is sequestered during natural regeneration. In this instance, the biomass component of interest is the total above-ground biomass (TAGB), typically expressed in kilograms per square metre (kg/m^2). The TAGB in gorse regenerating to indigenous forest comprises the biomass of the gorse component, any indigenous trees present, plus understorey and surface vegetation (such as grasses), as well as the biomass associated with the litter layer. Total carbon would also normally include the proportion of biomass associated with roots and soil organic matter, but these are not considered here as our focus is above-ground biomass.

Several allometric models are currently available for predicting TAGB in shrubland vegetation, including gorse (Fogarty et al. 1998; Fogarty & Pearce 2000; Pearce et al. 2010) and broom (Carswell et al. 2001), as well as other biomass components including understorey species (S.J. Richardson, Landcare Research, unpubl.), grasses (Fogarty et al. 1998; Fogarty & Pearce 2000; Pearce et al. in press) and litter (Fogarty et al. 1998). As Scion’s Rural Fire Research Group was involved in the development of many of these models, its expertise was sought on sampling and modelling comparisons. In addition to testing the fit of the Pearce et al. (2010) allometry, we also investigated whether an age–height relationship can be fitted to these data.

Finally, we summarise information from the literature on the factors most likely to promote the successful establishment of indigenous successions within gorse or broom where management can be applied.

2 Methods

2.1 Field survey of indigenous regeneration through gorse or broom

We contacted the Department of Conservation and regional councils to enquire about the presence/absence of patches of gorse or broom of at least 10 ha in their areas (i.e. at national scale). The Department's national weed database (presence/absence on a 10-km grid and degree of confidence – i.e. herbarium specimens or dataset records or 'expert opinion') was particularly comprehensive (C. Howell, pers. comm.). Using this information, we surveyed gorse and broom sites throughout the country. In total, 249 sites were visited of which 64 were associated with broom cover, 139 were predominantly in gorse cover and 47 were co-dominated by gorse and broom. Sites visited are shown in Fig. 1 and included the extremes of the distribution ranges for both gorse and broom, and spanned the major resource gradients known to influence the successional process (temperature, moisture, soil fertility, seed availability, herbivory).

The criteria used for sampling location were that

1) canopy cover was predominantly gorse or broom for young stands,
2) a gorse/broom component was still present if the stand was in later successional stages,
3) the patch size was a minimum of 1 ha, and preferably >10 ha,
4) the patch was not a ribbon of gorse or broom along the edge of other vegetation,
5) it was located within relatively uniform topography (i.e. aspect, slope) and
6) access was achievable to enable the identification and estimation of cover of any forest species emerging through the gorse/broom.

Each site was identified by dominant shrub cover (i.e. gorse or broom or a mixture) and GPS location at the centre of the gorse or broom stand. In addition, the size of the gorse or broom stand was estimated to the nearest hectare and physiography (ridge, face, gully, terrace), aspect, slope, drainage and current land use were recorded.

For each stand, we estimated the height of the gorse or broom to the nearest 0.5 m and crown cover (%) to the nearest 10%. We then recorded the forest species present that were emergent over the gorse/broom canopy and the approximate crown area (%) for each species along with total forest species cover (%), which could total more than 100%. Distance and direction to the nearest conspecific seed source of each indigenous forest species were recorded. Where forest species were fruiting within the site we also recorded the distance and direction to the nearest seed source outside the site. We also noted the abundance of other indigenous or exotic species that provide >10% cover within the stand.

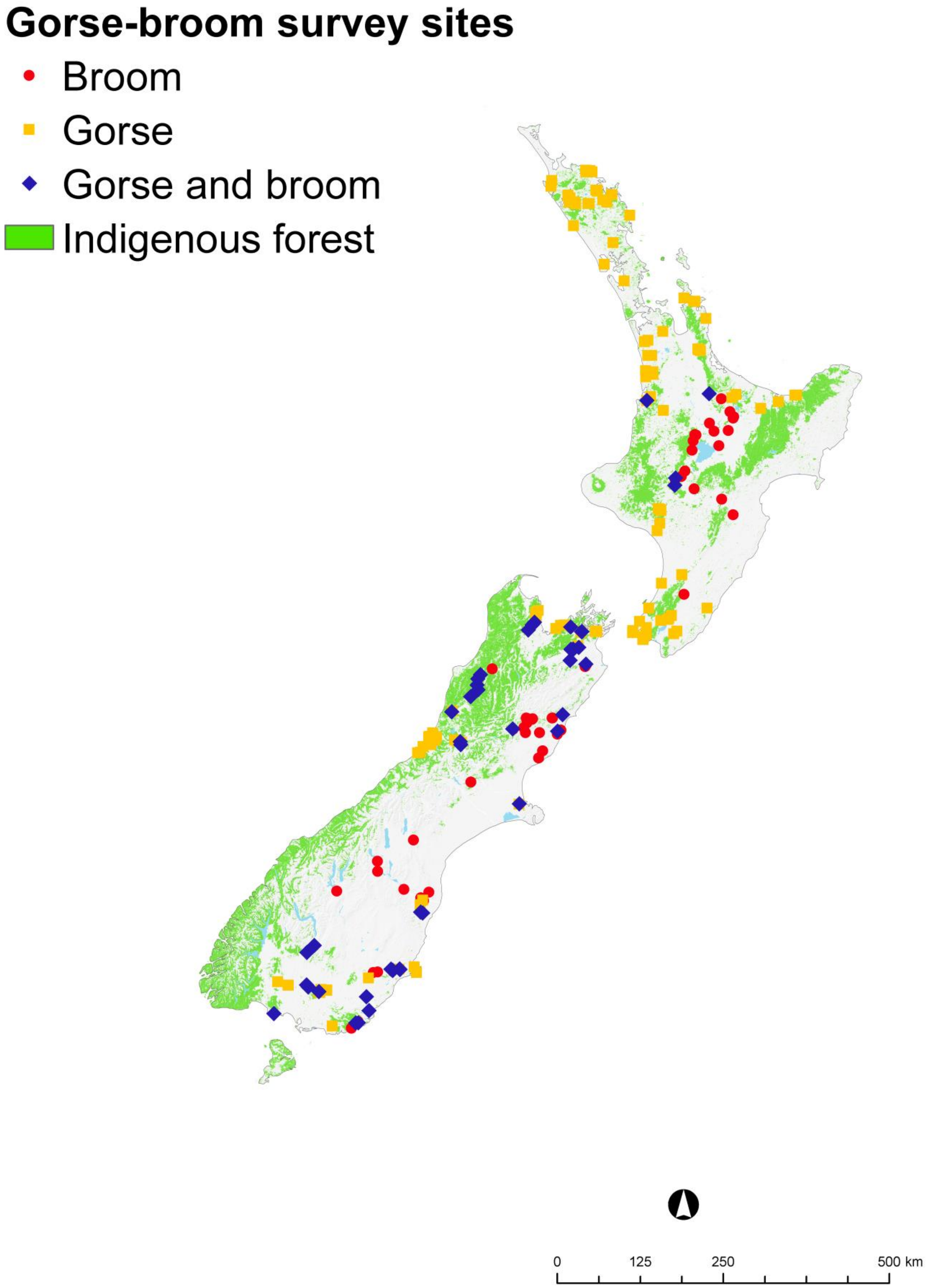


Figure 1: Sampling locations for gorse and broom plots across representative environmental gradients known to influence succession to indigenous forest.

2.2 Predicting likelihood of emergent tree occurrence within gorse and broom

2.2.1 Boosted regression tree modelling

In order to understand the processes influencing indigenous afforestation through gorse and broom we modelled the occurrence of emergent (i.e. above the gorse or broom canopy) indigenous trees. Trees have been defined (according to McGlone et al. 2010) as those reaching at least 5 m in height at maturity, therefore rendering the term ‘tree’ interchangeable with ‘forest species’, except only indigenous trees have been included in order to provide evidence that natural succession is actually occurring. The emergence of trees was used to indicate qualification of a given hectare as ‘forest land’ as they need to be c. 5 m in stature to be clearly visible above the gorse or broom canopy. We used boosted regression tree (BRT) modelling (Elith et al. 2008) to predict the probability of at least one emergent indigenous forest species occurring in the gorse and broom stands we surveyed (‘tree occurrence probability’ henceforth). We chose BRTs as they perform well compared with competing methods (Elith et al. 2008) and model interactions between predictors very effectively, without requiring explicit interaction terms to be defined a priori. In all BRT models we used a tree complexity of 5 and learning rate of 0.001. The maximum number of trees was set at 20 000, though this limit was never reached. Model goodness of fit was assessed using the cross-validated area under the relative operating characteristic (ROC) curve (Hanley & McNeil 1982), which is a standard measure of how well a model discriminates between presences and absences. Cross validation was performed by randomly removing 10% of the plots from the dataset, fitting the BRT model on the remaining plots, and then assessing the goodness of fit on the removed plots. In machine learning methods like BRT, the use of cross validation in assessing goodness of fit is essential to prevent over-fitting. To further reduce the chance of over-fitting, we conducted a step-wise process of model simplification that sequentially removes the predictor variable that causes the smallest decrease in goodness of fit. We performed this model simplification using the gbm.simplify function implemented in R (R Development Core Team 2009) provided in the supplementary material of Elith et al. (2008). Following the simplification procedure we fitted a new model containing only the variables whose removal caused a non-negligible decrease in goodness of fit.

We constructed two types of BRT model. First we included data on site characteristics that were collected during our surveys (aspect; gorse or broom canopy height; land use type; physiography – gully, hill, face or terrace; and drainage – poor, moderate or good; and whether gorse, broom or both were present) as well as climate data, slope and distance to nearest indigenous forest drawn from GIS layers (all at 100-m resolution, except for slope which was at 25-m). The second type of model included only GIS variables with the addition of latitude, since this enables us to produce a map of predicted tree occurrence probability (by using the BRT model to predict tree occurrence probability onto maps for GIS variables).

Some areas were excluded from the analysis on the basis of being unlikely to ever support forest species of at least 5 m in height except on an ephemeral basis. These areas are those:

1) with a mean annual temperature of less than 5.9°C as this is the basis of New Zealand treelines (Cieraad & McGlone 2014),
2) identified as wetland (Ausseil et al. 2011),
3) identified as belonging to Land Resource Inventory classes 7w, 8w or 8s, and
4) that cannot support trees due to edaphic factors M2-M4 (wetlands, sand dunes, pakihi heathlands; Newsome 1987).

In addition, areas that were mapped into any of the ‘forest’ classes by EcoSat5 have been masked out as these will be ineligible as post-1989 forests.

2.2.2 Differences in indigenous tree occurrence and site variables between gorse and broom

We used chi-square tests to determine whether there were significant differences in indigenous emergent tree occurrence between sites containing either gorse or broom, or both. We also used chi-square tests to test for differences in categorical site variables between sites containing either gorse or broom, or both. Finally, we used one-way analysis of variance (ANOVA) to test for differences in quantitative site variables.

2.2.3 BRT modelling of afforestation of currently non-forested land

In addition to predicting forest species emergence through gorse and broom shrublands we also calculated the probability of natural afforestation occurring in currently non-forested land. Non-forested land was defined as areas of New Zealand that are not currently mapped into any of the EcoSat5 (a digital map of indigenous forest classes in New Zealand) forest classes. This differs from ‘non-forest’ land as there is no way of remotely determining whether any given hectare of New Zealand contains sufficient forest species seedlings to be likely to form at least 30% crown cover in the future, but the BRT modelling was used to indicate the likelihood that a given hectare of non-forested land is actually forest land. ‘Non-forest land’ will therefore be a subset of the total ‘non-forested’ land area that we examined. We attempted to bring the two types of land area closer together by excluding data from plots with woody cover of more than 30% or where woody species made up more than 30% of the frequency scores. ‘Woody species’ are not the same as ‘forest species’ as they may include shrubland species such as bracken (*Pteridium esculentum*) or shorter statured *Coprosma* species but their mere presence increases the likelihood that forest species will also be present. To construct the BRTs we used 10 061 survey plots from the National Vegetation Survey Databank (NVS) to identify the primary environmental and land cover variables influencing tree occurrence probability in non-forested vegetation. As with the gorse and broom BRTs the term ‘tree’ is synonymous with indigenous ‘forest species’ only.

We also examined how environmental and landcover variables influenced diversity of functional traits linked to plant ecological strategy and dispersal ability as a way of assessing the probability of ‘forest land’ establishing. Environmental variables were drawn from existing GIS data layers held by Landcare Research. Landcover variables were derived as follows. We derived four variables from EcoSat: distance to nearest forest, percentage forest cover within a radius of 100 m, percentage local woody cover (either shrubland or forest) within a radius of 25 m, and the dominant vegetation type within a radius of 100 m. Distance to nearest forest gives an indication of proximity to forest species’ seed sources (Fig. 2), while percentage forest cover within a 100-m radius gives a more fine scale (< 100 m) measure of seed source availability in the locality. Percentage woody cover within a 25-m radius quantifies woodiness in the immediate vicinity of the plot, while the dominant EcoSat vegetation type within 100 m indicates neighbourhood vegetation type, and provides more detail on the variation in forest composition than using the Land Cover Database version 2 (LCDB2) alone.

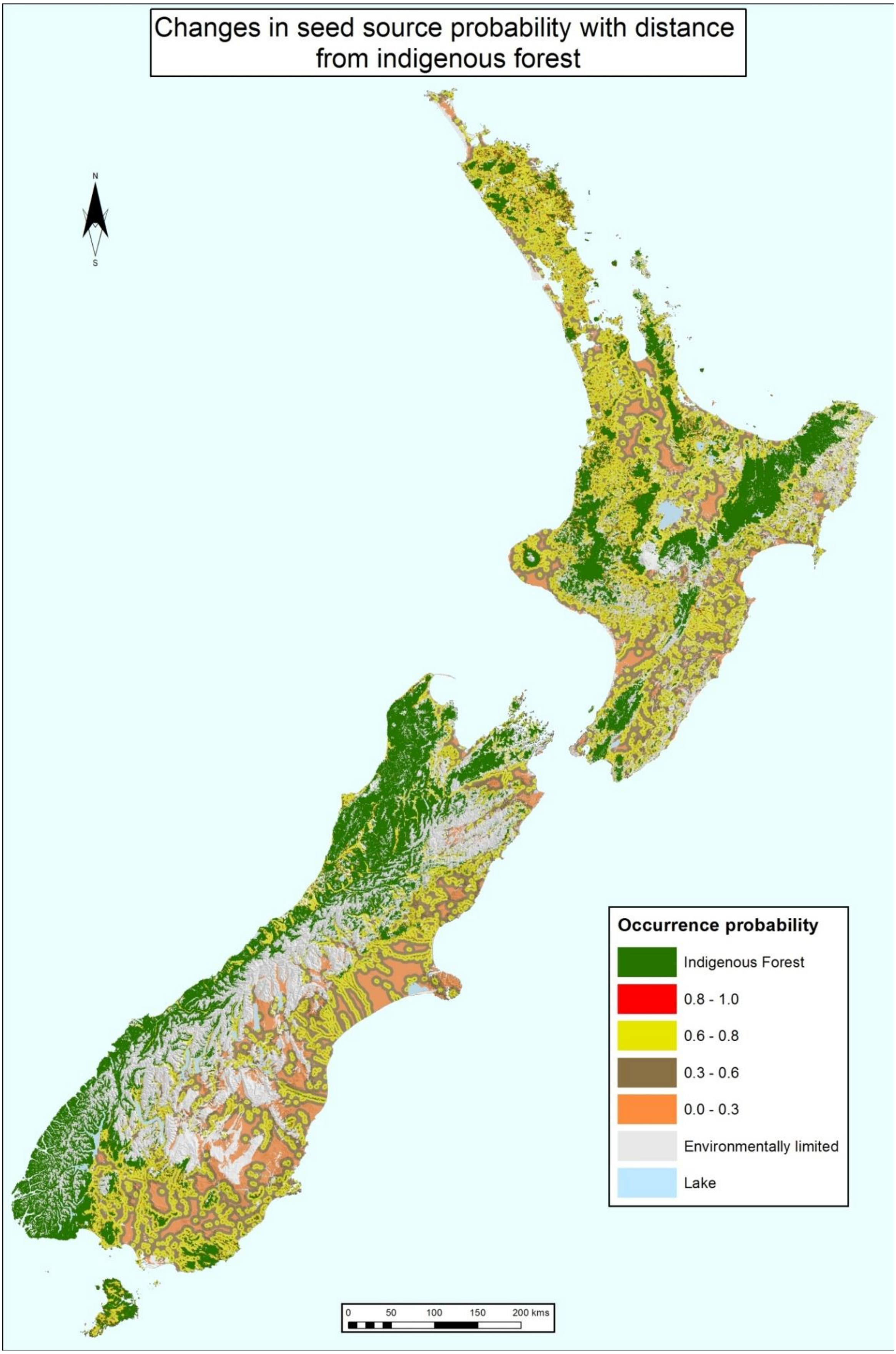


Figure 2: Change in seed source probability of indigenous tree species as distance from indigenous forest increases, derived from EcoSat, a digital map of indigenous forest classes.

In order to determine tree functional traits we measured seed mass for a range of species and also sourced data on tree height (m) and leaf size (mm^2) from McGlone et al. (2010). Dispersal trait data were taken from Thorsen et al. (2009). From this dataset we used four traits to describe dispersal: (1) nature of morphological dispersal adaption (none, aril, fleshy, hooked, minute, pappate, viscid or winged); (2) principal dispersal mode (ballistic, 'mobile' – i.e. epizoochorous, by frugivory, water or wind); (3) number of dispersal modes; and (4) whether or not the species is biotically dispersed. In all BRT models we used a tree complexity of 5 and learning rate of 0.001. The maximum number of trees was set at 20 000, though this limit was never reached. Model goodness of fit and cross validation were performed in the same way as for the gorse and broom BRT model fitting. We used the fourth-corner approach (Dray & Legendre 2008; ter Braak et al. 2012) to test the significance of relationships between functional traits and the best predictor variable from each of the three main types of predictor: land cover, proximity to seed source and abiotic stress.

Using the same criteria as in 2.2.1, areas where forest land cannot establish were excluded from the analysis, and additionally, areas shown in LCDB2 as planted forest have also been excluded.

2.3 Site and plot locations for intensively studied/harvested sites

Intensive sampling and destructive harvest was carried out for a minimum of 14 stands through the succession from gorse or broom through to emergent indigenous forest species at five locations (74 plots in total) around New Zealand (Fig. 3). At each location we targeted five plots in each of three vegetation states:

- Gorse or broom only, with no emergent forest species present above the scrub canopy (0% trees)
- Gorse or broom with less than 30% cover of emergent trees (0–30% trees)
- Gorse or broom with greater than 30% cover of emergent trees (>30% trees).

It should be noted that only data from plots with at least 30% cover of emergent trees were used for the back-casting to determine times to transition whereas all plot data were used to construct the allometric relationships between gorse cover and total above-ground biomass.

All five sites were located on private land and sampled with the approval of the owners, who kindly provided information on site history.

The areas selected for intensive harvest spanned three bioclimatic zones for gorse, namely warm-temperate, cool-temperate-wet and cool-temperate dry. These zones were characterised as follows (Wilson 2012; NIWA 2013):

- Warm-temperate-wet (mean annual temperature above 14°C, and at least 1400 mm rainfall per annum)
- Cool-temperate-wet (mean annual temperature below 14°C, and at least 1200 mm rainfall per annum)
- Cool-temperate-dry (mean annual temperature below 14°C, and less than 1000 mm rainfall per annum).

The three sites selected for gorse harvest were Tautoro (warm-temperate-wet), Featherston (cool-temperate-wet), and Turnbull Bay (cool-temperate-dry).

Selective harvest of broom spanned only the cool-temperate-wet (Glenthorne) and cool-temperate-dry (Hawkswood) zones due to a lack of suitable sites at sufficient density for measurement in the warm-temperate-wet zone, as broom seldom occurs there.

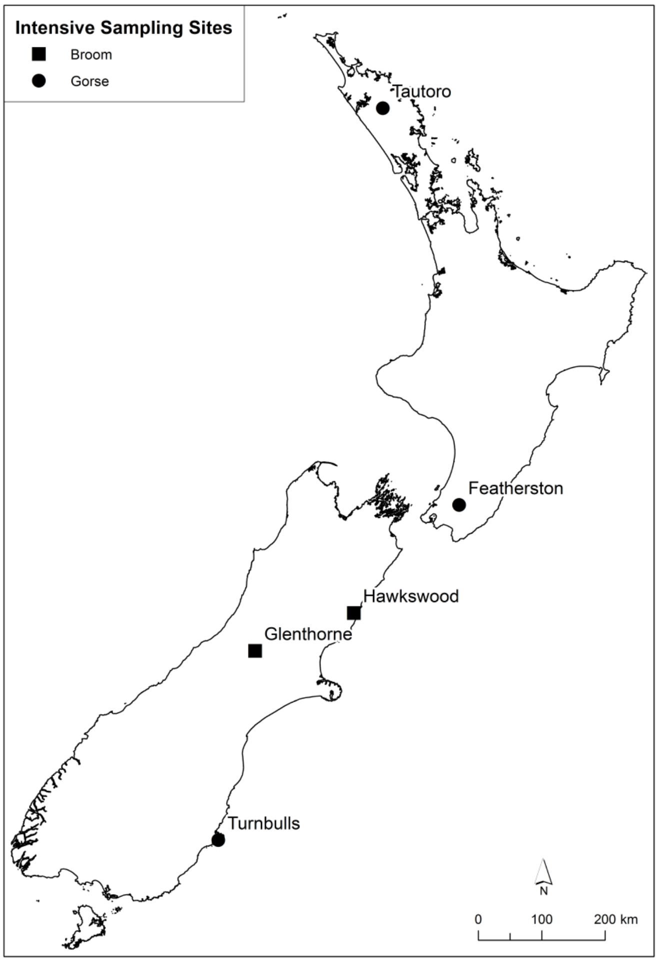


Figure 3: Sites of intensive harvest of gorse (3 sites) and broom (2 sites).

2.3.1 Tautoro (130 m a.s.l., mean annual temperature (MAT) ~16°C, ~1600 mm annual rainfall)

The Tautoro site in central Northland occupies approximately100 ha dominated by gorse. It has been subjected to repeated effort to control the gorse but c. 12 ha of hilly land has been allowed to regenerate since it was last burned in the 1970s. Initially the block was stock grazed, but because machine clearing operations were not possible, gorse has re-established dominance. Therefore, near-continuous cover of gorse has prevailed here for c. 30 years. The local district includes naturally regenerating indigenous forest species and seed sources. In particular, the bird-dispersed totara (*Podocarpus totara*), kahikatea (*Dacrycarpus dacrydiodes*), red mapou (*Myrsine australis*), and mahoe (*Melicytus ramiflorus*) are well-represented nearby and are readily regenerating within the gorse. As is typical in Northland, many exotic species (e.g. privet *Ligustrum* spp.) and woolly nightshade (*Solanum mauritianum*) were well established within the gorse and exceeded 5 m in height. The silvery crowns of the woolly nightshade are highly conspicuous in aerial photographs.

2.3.2 Featherston (180 m a.s.l., MAT ~12–14°C, ~1600 mm annual rainfall)

The Featherston site is situated on the lower south-eastern slopes of the Tararua Range in the Wairarapa. Gorse is widespread throughout the adjacent hill country and occupies many hundreds of hectares. Numerous attempts have been made to clear gorse using goats, chemicals, scrub-cutting, roller-crushing and fire. The block selected for sampling can be seen on aerial photos to be mostly free of gorse in 1989 following roller-crushing and burning of gorse and grazing in the mid-1980s until the early 1990s when grazing ceased. Predominant gorse cover in the area sampled is therefore assumed to be about 20 years old. The species most commonly regenerating in the gorse was mahoe although rangiora (*Brachyglottis repanda*) and hangehange (*Geniostoma rupestre* var. *ligustrifolium*) were also present. Nearby forest remnants include kanuka (*Kunzea ericoides*), black beech (*Nothofagus solandri*) and totara.

2.3.3 Turnbulls Bay (60 m a.s.l., MAT ~10–12°C, ~900 mm annual rainfall)

Turnbulls Bay on the Otago Peninsula is a N–NE aspect hill block of c.10 ha of dense gorse extending across two properties (Fig. 4). The senior occupier remembers burning the gorse on the whole block in about the mid-1970s. A few years later goats were introduced in an attempt to control gorse reinvasion but were removed around 1990. Since then, low numbers of sheep have had access to one part of the block while the fenced section has been grazer-free and managed for natural regeneration for over 20 years. Establishment of trees has occurred readily in the lower gully bottom where trees (mahoe, ngaio (*Myoporum laetum*), kohuhu (*Pittosporum tenuifolium*), fuchsia (*Fuchsia excorticata*) and wineberry (*Aristotelia serrata*)) are now over 6 m tall. Some gorse reversion here may be up to 35 years old, but due to the variable effects of grazing we have assumed that a continuous cover of gorse has been present for c. 25 years. The sampled site is adjacent to a Reserve containing the same broadleaved species as are regenerating within the gorse as well as narrow-leaved lacebark (*Hoheria angustifolia*).

Figure 4: Tall (3 m) gorse at Turnbulls Bay, Otago Peninsula, has mahoe, ngaio, kohuhu, fuchsia and wineberry establishing and growing through the solid gorse canopy.

2.3.4 Glenthorne (750 m a.s.l., MAT ~10–12°C, ~1400 mm annual rainfall)

Broom is widespread and spreading in the upper Rakaia and Wilberforce catchments in mid-Canterbury and occupies many hundreds of hectares. Although common on braided riverbeds, broom is also found at elevations above 1000 m on fine screes and tussock grasslands on hill slopes. We selected a block of approximately 18 ha of continuous broom cover, surrounded by mountain beech (*Nothofagus solandri* var. *cliffortioides*) forest, on Glenthorne Station (Fig. 5). Part of the sample area had been planted with Douglas-fir (*Pseudotsuga menziesii*) in the early 1990s. Past attempts to control broom at this site have included tillage, herbicide spray and goats. Plantation trees had been planted into short broom at the time and part of the area, where we sampled, only had scattered broom bushes and was originally grazed by sheep. Our estimate of broom cover at Glenthorne is that some individual broom bushes may be c. 20 years old but the majority of the now continuous cover is likely to be 10–15 years in age. Mountain beech is the main forest species in the vicinity and it was regenerating freely through broom within the limited seed dispersal range exhibited by beech.

Figure 5: Broom at c. 750 m elevation at Glenthorne, Mid-Canterbury, being invaded by mountain beech.

2.3.5 Hawkswood (400 m a.s.l., MAT ~12–14°C, ~900 mm annual rainfall)

Broom has been a widespread weed on the Hawkswood Range in North Canterbury for half a century or more and is still invading pasture-covered hill country there. Various treatments have been attempted historically to remove broom but large extents of hundreds of hectares are now being managed for regeneration to indigenous forest by several landowners (Fig. 6). Remnant forests nearby contain kanuka, manuka (*Leptospermum scoparium*), mountain beech, marbleleaf (*Carpodetus serratus*) and kaikomako (*Pennantia corymbosa*).Sample sites were situated on hill slopes with continuous broom cover that has invaded pasture over the past c. 20 years although individual plots may have had broom for less time. Mahoe, five-finger (*Pseudopanax arboreus*), cabbage trees (*Cordyline australis*), kohuhu, lancewood (*Pseudopanax crassifolius*) and kowhai (*Sophora microphylla*) are all readily invading broom. Wild pig (*Sus scrofa*) control operations are implicated in the low abundance of tree seedlings and saplings here.

Figure 6: Broom is extensive over many hundreds of hectares of hill country and is still invading pasture on the Hawkswood Range, North Canterbury. Mahoe, five-finger, cabbage trees and others are readily invading the broom.

At each of the five sites, areas were identified that were predominantly in either gorse or broom cover and had forest species emerging through the canopy. Plots of 4 × 4 m were intentionally located within areas where forest species were seen to be regenerating through gorse or broom. However, specific plot locations were chosen to include 5 plots where there was no crown cover of forest species above the shrub canopy, 5 plots with 0–30% crown cover of forest species, and 5 plots with >30% crown cover of forest species. Note that for plots with at least 30% crown cover the cover could be a mixture of above and below the canopy. Plot centres were placed within these areas, avoiding patches of old ‘remnant’ vegetation. Plots were established perpendicular to the slope. An additional 4 × 1 m strip immediately adjacent to these plots was harvested for above-ground biomass determination.

2.4 Plot-scale measurements for intensively studied/harvested sites

For each plot, total crown cover (circular projected area) was recorded for all forest species and separately for both indigenous and exotic woody species in general. These species were identified within standard height tiers and cover classes (Hurst & Allen 2007). Aspect, physiography (horizons), slope and GPS location were recorded. Ground cover <1.35 m (%)

was recorded for moss, vegetation, litter, rock and bare soil, and the percentage grass cover and average grass height were recorded to assess weed competition. Shrubland carbon measurements (% continuous cover and average heights) were made following Carswell et al. (2009).

Within each plot, for all trees present, the following items were noted/measured: diameter at breast height (if >2.5-cm diameter at 1.35 m), basal diameter (at 10 cm above ground avoiding swelling around root), natural tree height, height to base of crown, orthogonal crown widths (N–S, E–W), and crown exposure class using the Kraft classification scale (Kodrik 1994; 0 = understorey, 1 = supressed, 2 = subdominant, 3 = dominant, 4 = emergent) for all woody stems >30 cm tall. The X–Y grid coordinates of each stem were recorded to the nearest 10 cm.

Once the non-destructive measurements were complete, all stems >30 cm tall were cut at ground level (using handsaws or chainsaws, depending on the size of the trees) and basal disk samples taken for age determination. Ages were determined in the field using fresh ring counts. The disks were also dried in the lab and a subsample sanded and re-counted to check on field determinations.

Additional disk samples were taken from three gorse or broom individuals within the plot (or immediately adjacent to the plot) that represented the tallest and/or biggest individuals, to indicate stand age of the nurse species (three disks per plot). The ages of these individuals were also determined from fresh ring counts in the field plus lab verification from dried disks. Gorse/broom crown measurements of the harvested individuals were made in the same way as the tree measurements.

2.4.1 Crown position classes (refer to Fig. 7)

0 = Fully-shaded

- No direct sunlight strikes at any portion of the crown when sun directly overhead
- Crown entirely overtopped by other crowns

1 = Suppressed

- Up to ¼ of the horizontal crown area receives full sunlight
- Crown poorly developed and clearly suppressed from all sides

2 = Partially co-dominant

- More than ¼ and up to ½ of the horizontal crown area receives full sunlight
- Crown moderately well developed and suppressed from 2–3 sides

3 = Co-dominant

- More than ½ and up to ¾ of the horizontal crown area receives full sunlight
- Crown within and helping to form the main crown canopy for the stand
- Crown well developed, but still somewhat suppressed

4 = Dominant

- More than ¾ of the horizontal crown area receives full sunlight
- Large well-developed crown extends above the general canopy layer for the stand
- More than ¾ of the horizontal crown area receives full sunlight
- Large well-developed crown extends above the general canopy layer for the stand

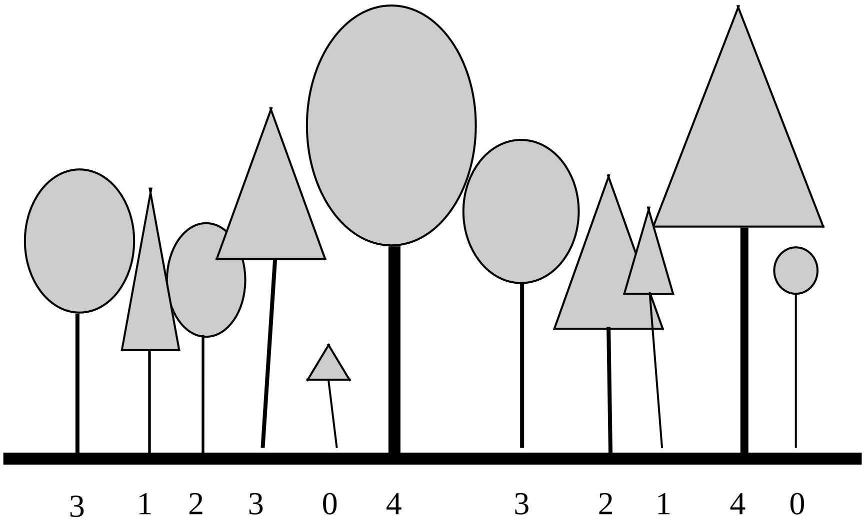


Figure 7: Schematic illustration of crown position classes (0–4).

2.4.2 Height and width measurements

The height to base of crown is the distance from the ground to the lowest live foliage of the ‘obvious live crown’ (Fig. 8). The ‘obvious live crown’ was described as the zone on the tree where most live branches/twigs are clustered. This excluded epicormic shoots and straggler branches that usually do not contribute much to the tree's growth. The live crown base was determined by the zone of live foliage and not by the point where a branch intersects with the main bole.

Crown-top-height was the height of the highest foliage. This was the same as tree height for all non-damaged stems.

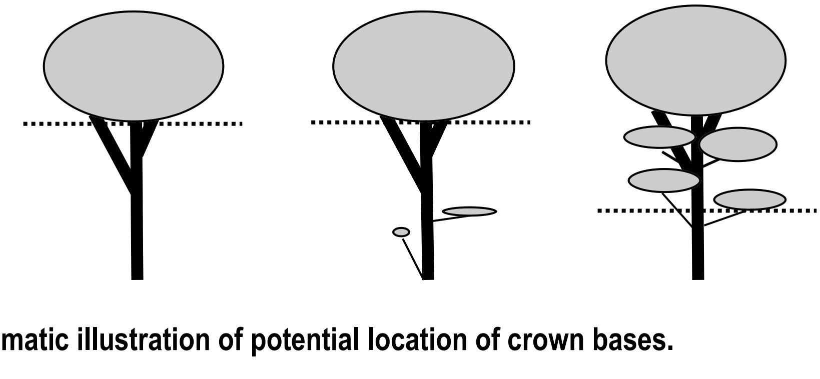


Figure 8: Schematic illustration of potential location of crown bases.

Examples of crown width measurements are given in Fig. 9. Orthogonal widths were measured in both the N–S and E–W planes at the widest point of the crown and not necessarily directly past the trunk of the tree to the nearest 0.1 m.

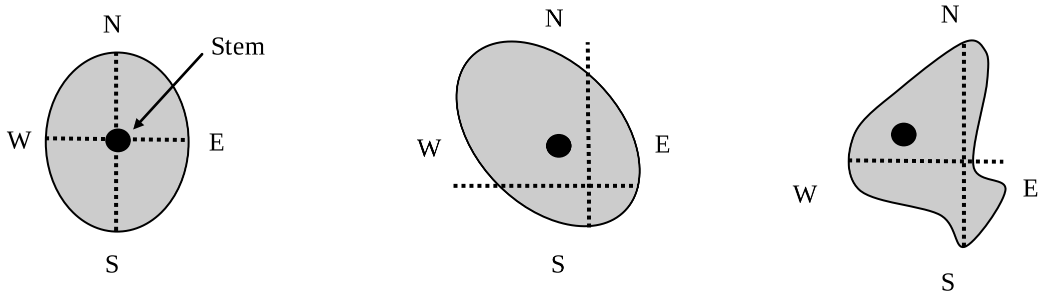


Figure 9: Schematic illustration of examples of orthogonal widths from top-down perspective of tree/shrub crown.

2.5 Above-ground biomass estimation

Adjacent to each of the 4 × 4 m plots located across the gradient of ‘transition’ at each field site, we placed a 4 × 1 m biomass quadrat. Quadrats were located either immediately adjacent to the 4 × 4 m plot (along one edge, but outside) or within the 4 × 4 m plot itself (along one edge, but inside), in order to best replicate the same cover as in the 4 × 4 m plot (Fig. 10).

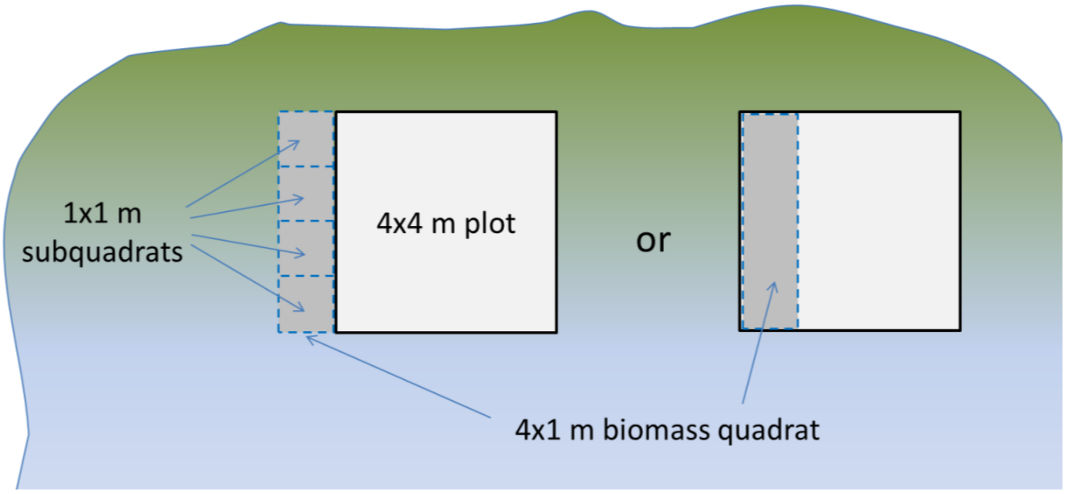


Figure 10: Options used for the location of biomass sampling quadrats relative to vegetation assessment plots, i.e. either outside (left) or inside (right). The division of the 4 x 1 m biomass quadrat into four 1 x 1 m ‘subquadrats’ for shrub structure measurement is also shown.

2.5.1 Gorse structure measurements

Use of four adjoining 1 × 1 m ‘subquadrats’ (Fig. 10) has previously been shown to adequately describe shrub structure and above-ground biomass in continuous-cover shrubland where structure is still influenced by the presence of individual shrub plants (Pearce et al. 2010). Gorse structure measurements were obtained using the methods described by NZ FRI Fire Research (1994), Fogarty et al. (1998) and Pearce et al. (2010). Gorse canopy height and height of any trees present were determined by averaging estimates of canopy height within each of the four 1 × 1 m subquadrats. Continuous cover for the gorse and tree components were assessed separately as net proportion for each subquadrat, clumping areas of canopy foliage together (i.e. ignoring small gaps between leaves) but accounting for larger gaps in the canopy. Similar estimates of height and cover were also made for understorey vegetation (e.g. other woody shrubs, ferns, grasses) and of surface litter depth and cover, with measurements from each 1 × 1 m subquadrat averaged to provide an estimate for the entire 4 × 1 m quadrat.

2.5.2 Destructive biomass assessment

The biomass or weight of vegetation material present within each tree-cover regeneration stage was assessed by destructive sampling, where all vegetation was clipped from the 4 × 1 m quadrat, sorted into vegetation and species components, and weighed.

The continuous-cover biomass estimates obtained represent the weight of all material contained within the 4 × 1 m quadrat volume, so prior to commencing sample collection the edges of the quadrat were first 'squared off' to account for any vegetation hanging over the edges of the quadrat (Fig. 11). Any vegetation from plants rooted inside the quadrat but hanging outside it was trimmed along the line of the quadrat edge and discarded. Any vegetation from plants rooted outside the quadrat but hanging inside was clipped off along the line of the quadrat edge and included in the sampled material to be weighed.

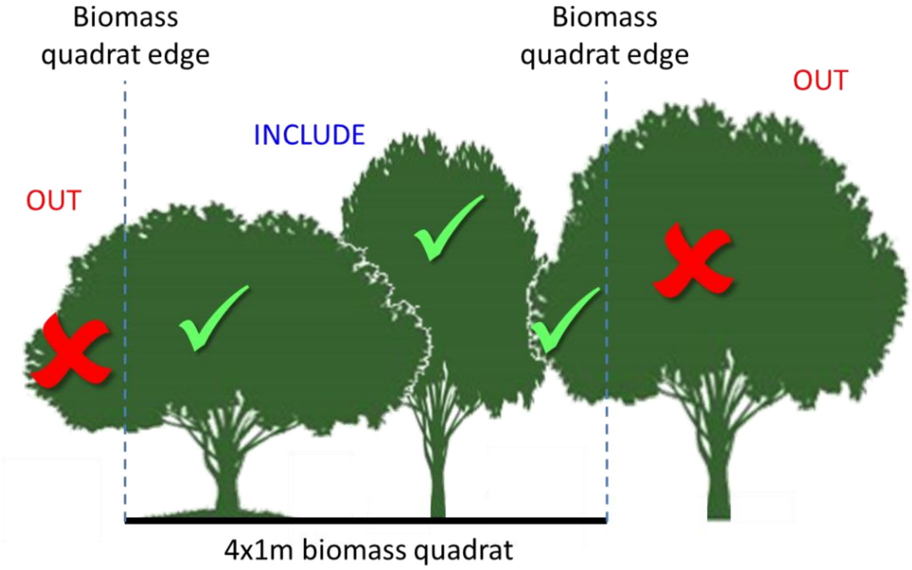


Figure 11: Illustration of the process for determining whether vegetation components are included or excluded from the biomass sampling volume.

Within each quadrat, the elevated gorse layer was usually destructively sampled first, followed by the forest species' component by species, then any woody understorey component if present. Due to expected differences in field moisture contents of different plant elements (foliage, twigs, branches and stems), sampled material within each vegetation component or forest species was further sorted into size classes (<1-cm diameter, 1–3 cm, and >3 cm) for weighing and/or oven-drying.

Biomass weights are expressed on an oven-dry weight basis, so were obtained by either weighing bulk samples in the field (using fishing/hunting scales), and collecting subsamples of each component for oven-drying and moisture content correction; or collecting the entire sample for oven-drying and weighing. Where moisture-content-correction samples were used, samples were collected in ziplock bags and stored in a cool shaded location prior to being weighed as soon as possible after collection (typically within 4–5 hours) to minimise the chance of moisture loss. Where biomass components were present in large amounts (e.g. gorse and tree foliage), material was collected in a woolsack for weighing, and weights adjusted for the tare weight of the woolsack.

Samples for moisture content correction, as well as those for the entire biomass component where present, were oven-dried at 105°C. This higher temperature, which is the standard for determining fuel moisture contents for fire research purposes, was used to maximise the drying of large woody material and provide a consistent drying temperature across all materials. It is recognised that there may be volatilisation losses at this higher temperature, especially for live foliage (and soils/duff), but these are considered to be minor compared with the variability in moisture content for these components. Samples were dried to consistent weight, with foliage and twigs typically taking 48–72 hours, medium branches/stems 5–7 days, and larger woody material 7–14 days.

Due to the amount of material and time to sample, vegetative ground cover (grasses, ferns, herbs/forbs, moss) and litter were not collected and weighed. Instead vegetation structure (height/depth and cover) was measured and used to estimate biomass for these components where present, based on existing regression relationships (Fogarty et al. 1998; Pearce et al. in press; Bray 1991; S.J. Richardson, Landcare Research, unpubl.).

2.6 Rapid gorse age and height/cover assessments

In an effort to better understand the relationship between shrub age and structure across a wider range of sites, assessments of gorse age and structure only (without associated destructive biomass sampling) were also undertaken at sites across Canterbury and the Kapiti Coast (Table 1).

At each site, three assessment locations were randomly selected and the 4 × 1 m quadrat frame used to identify the area to be measured. Average height and cover measurements were made for each of the four 1 × 1 m subquadrats as for the destructive biomass sampling, which were later averaged for the quadrat as a whole. However, only the three largest gorse stems either within the quadrat area or immediately adjacent to it were cut down and disks collected for subsequent aging. The slope, aspect, rough stem density and other species present were also recorded.

Table 1: Rapid gorse age/structure sampling sites

Site	Northing	Easting	Age (yrs)	Description
Pines Beach, NE of Kaiapoi (Canterbury)	5198585	1575225	4–8	Large area of gorse with cattle tracks, wide range of ages, sandy dune soil, has been burnt in the past
McLeans Golf Course (Canterbury)	5187315	1558115	5–7	Small spread-out patches of gorse, uneven-aged, rocky soil
McLeans Island 2 (Canterbury)	5187315	1558115	1–4	Similar to McLeans Golf Course
Summit Road, Port Hills (Canterbury)	5167100	1569670	5–6	Relatively young, even-aged stand, burnt a number of years ago
Gerkins Road, Port Hills near Tai Tapu (Canterbury)	5163755	1566300	5–14	Lower north-west-facing slopes, large range of ages/sizes, some indigenous trees growing through, some gorse sprayed
Torlesse Station, Springfield (Canterbury)	5206250	1511875	4–9	East-facing slope Very large gorse patches Relatively high altitude. Some grazing/animal damage on outer gorse
Waikanae River estuary (Kapiti)	5472940	1769385	5–7	Swampy soil, some indigenous regeneration, vines and grass intertwined with gorse, old gorse stumps with new growth made aging difficult
Paraparaumu Business Park (Kapiti)	5469795	1768290	4–5	Urban gorse patch, land being developed, gorse mixed with some blackberry and broom
Kapiti/Mazengarb roads (Kapiti)	5469795	1768805	7–8	Large area of urban gorse patches, mixed with some blackberry (*Rubus* spp.)
Beehives, Otaihanga Road (Kapiti)	5471785	1770490	4–6	A number of large patches of gorse, lots of grass
Path to QE2, Paraparaumu (Kapiti)	5466070	1767300	5–7	Large paddocks of gorse, segmented with fences and creeks, wide range of ages.

2.7 Analysis to determine time to transition

To model time to transition, only plots with at least 30% crown cover of forest species were used (58 plots across all five sites). This is because these are the only plots for which it can be definitively stated that ‘forest land’ has been achieved. To model change in tree-canopy crown area through time, we developed relationships to predict individual tree crown area from tree age, basal diameter (D10), species, and the maximum age of trees present on the plot. Total crown area was calculated by overlaying the measured crown areas (assuming circular geometry) within a spatially explicit framework based on the grid coordinates of each stem. Individual tree crowns were merged to remove overlapping crown area, and then summed to give the total projected canopy area. Canopy areas outside of the plot boundaries for individuals rooted within the plot were included in the total crown area estimate on the assumption that these likely equalled overhanging crowns of trees rooted outside the plot. The spatial analysis was done using the *spatstat* package in R (R Development Core Team 2009, http://www.r-project.org/).

Within this spatially explicit framework, back-simulations were made to predict changes in crown area from the time of establishment of the oldest tree on each plot to its current state. This was done on a yearly time-step by reducing the age of all stems by 1 year, and reducing the basal diameter of each stem using a power-law relationship between basal diameter and age:

$$Basal\ diameter\ (cm) = aAGE^{b}\ .$$

The parameter α was estimated as 0.167 by fitting the above relationship to the entire dataset. To account for tree-level variability in the diameter–age relationship, the parameter b was fitted for each stem using the observed basal diameter and age values and assuming that α was constant. This allowed for the non-linear relationship that generally exists between age and basal diameter. Allometric relationships were then used to estimate individual tree crown areas for each time step on the basis of the new data on basal diameter and age. Total plot-crown area for each year was calculated using the spatially explicit methods described above. This simulation progressed back through time until no trees remained on the plot. The total number of tree stems was also tracked though time.

The establishment time of gorse or broom (t_0) was estimated from data on tree ring age of gorse or broom stems collected for each plot, based on the oldest observed age. For some sites this was younger than the oldest tree on the plot, indicating either that the trees established before the gorse or broom, or that multiple cohorts of gorse or broom existed prior to the establishment of the forest species.

3 Results

3.1 Field observations of gorse, broom and emergent species occurrence

A relatively small number of indigenous shrub and forest species typically invade grasslands that are no longer grazed or where grazing intensity is low. Manuka and kanuka are the primary woody colonisers of grasslands over much of New Zealand. Gorse- and/or broom-dominated successions are widespread across both the North and South Island, but are rare or absent in some landscapes. Woolly nightshade is a prominent early coloniser in the northern half of the North Island (Fig. 12a). Further south, barberry (*Berberis glaucophylla*) and hawthorn (*Crataegus monogyna*) are locally important (Fig. 12b). In coastal areas gorse and/or broom typically give way to tauhinu (*Ozothamnus leptophyllus*) and grey scrub, largely indigenous divaricate species. The other primary woody colonisers of grasslands are the exotic conifers, especially the pines (*Pinus* spp.), larch (*Larix decidua*), and Douglas-fir. Early-stage successions may be dominated by a single woody species such as gorse or broom, but are equally likely to consist of a mix of these species.

In addition to the primary woody colonisers, a total of 30 forest species were recorded as emergent over gorse and/or broom. Of these mahoe was by far the most frequent (43% of sites), with cabbage trees (17%), five-finger (10%), elderberry (10%) and wineberry (*Aristotelia serratus*) (9.3%) (Fig. 12c) also present on more than 5% of sites.

3.1.1 Broom still appears to be expanding its range

In South Canterbury, for example, between Fairlie and Burkes Pass broom is the dominant early woody coloniser of hill country grasslands, harvested forestry blocks and creek beds (Fig. 12d). From Burkes Pass to Twizel it is virtually absent, except as occasional plants along the roadside and in creek beds. On the outskirts of Twizel and Omarama it has colonised disturbed areas. Near the turnoff to Lake Ohau it is beginning to invade low producing grasslands (Fig. 12e). Otherwise broom is conspicuously absent from that part of the Mackenzie Country visible from SH8. Broom spread appears to be associated with disturbed sites such as road and river edges, and with infrastructure development.

In the Clarence Valley north of Hanmer Springs broom and wilding pines dominate the hill slopes on the south side of the valley between St James Station and a point several kilometres south of the Jollies Pass turnoff. Elsewhere in the upper Clarence Valley broom is sparse or absent.

Similar situations occur on the central North Island volcanic plateau, Taranaki and East Cape.

3.1.2 Gorse and broom communities are rare or absent in some parts of the country

Broom patches are hard to find in Auckland and Northland. A small patch (<50 m^2) adjacent to SH1 just north of Okaihau was the largest occurrence observed during the current scan of gorse and broom communities. Farmers and weed spraying contractors we spoke to were unable to point us to larger areas of broom in Northland. A similar situation exists with gorse in parts of Otago.

In some cases the apparent absence of gorse and broom results from more intensified land use. In other cases it results from local and regional government policies to eradicate existing populations of gorse or broom and to prevent the reinvasion of these woody weeds (S. Fowler, pers. comm.).

3.1.3 Broom is a relatively short lived species

As part of this study we cut and aged the largest broom stems from a range of sites across the national survey (in addition to those measured at intensively studied sites). None were over 15 years old, and most were less than 10 years. Published estimates of broom age from 20 South Island sites (Paynter et al. 2003) are similar with a maximum age of c. 15 years. Individual broom ages sampled during our nationwide survey ranged from 7 to 21 years with an average age of 13 years.

This was true for sites where broom has been present for many decades as well as areas that were newly invaded. The presence of a previous cohort in the form of broom logs on the ground was noted on many plots at the intensive sample sites. The conclusion we draw from this is that broom succeeds broom until such time as it is overtopped by taller woody species.

Gorse is a longer-lived species, but may also require more than one generation before tree species overtop and supress it (Lee et al. 1986). Maximum recorded ages for gorse in New Zealand are 39 (Kelly 1965) and 29 years (Lee et al. 1986) respectively, and the oldest stems sampled in this study were c. 30 years old.

3.1.4 Broom appears able to tolerate swampy conditions

In south-east Otago we were told by a landowner who had farmed the area for over 40 years that an area of swampy ground on his property had been in broom for as long as he had known it. The swampy ground was devoid of any forest species (Fig. 12f). On adjacent slopes broom was part of the regenerating forest succession. This suggests that broom communities on waterlogged soils may not progress to forest land, at least in the short to medium term.

3.1.5 Broom pushes further into higher altitude areas than gorse

In the Clarence Valley, north of Hanmer Springs, broom is invading subalpine shrublands and tussock grasslands well above the treeline (Fig. 12g). On Mt Oakden and the Birdwood Range to the west of Lake Coleridge in inland Canterbury, broom is now found in patches of subalpine scrub that dot high altitude scree slopes (Fig. 12h).

Figure 12a: Woolly nightshade and gorse invading grassland, Northland.

Figure 12b: Hawthorn, elderberry and gorse establishing in a gully fenced off from grazing, South Otago.

Figure 12c: Cabbage trees and wineberry emerging through gorse, Otago.

Figure 12d: Broom and larch invading hill country grassland near Burkes Pass, South Canterbury.

Figure 12e: Broom invading low producing grasslands near Lake Ohau, South Canterbury.

Figure 12f: Long-established broom community on swampy ground, south-east Otago.

Figure 12g: Broom invading subalpine shrublands and grasslands above Jacks Pass,

Figure 12h: Broom at 1150 m on Mt Oakden to the west of Lake Coleridge in inland

Hanmer Springs, North Canterbury. **Canterbury.**

3.1.6 Biological control effects on broom

Biological control efforts to reduce broom are being implemented with variable success. Currently, several vectors can reduce stand health and cover (Jarvis et al. 2006). The effect of biocontrol on indigenous forest establishment and growth under weakened broom is currently unknown. In this study we did not include estimates of biocontrol impacts at sample sites. In general it could be expected that a thinned canopy could allow for more light, which could improve the growth of indigenous trees. However, it has been observed that where light levels get too high or the broom stand dies, exotic pasture grasses can dominate and competitively retard establishment and growth of indigenous forest species.

3.2 BRT models of emergent indigenous tree occurrence

The final BRT model, including variables from both site surveys and GIS layers, accurately predicted occurrence of emergent indigenous trees (cross-validated ROC = 0.823, Fig. 13). In this model, gorse/broom canopy height had the greatest influence on occurrence probability of emergent trees (Table 2, Fig. 13). This suggests that successional stage (or age since establishment) is a key factor in indigenous tree growth, with taller, older gorse/broom stands being more likely to contain emergent indigenous trees. The second most important predictor was aspect, with flat sites (having undefined aspect) being much less likely to have emergent indigenous trees. These two factors are thought to be associated with more open canopies that favour emergent forest species – i.e. a critical amount of light is present. We suggest that there is a narrow light range for succession to forest, beyond which, at higher light levels, exotic pasture grasses dominate and effectively out-compete indigenous woody species. There was also evidence that south-east-facing slopes were more likely to have emergent indigenous trees than other aspects. Slope was also included in the final model, with moderate and steeply-sloped sites being more likely to have emergent indigenous trees than flat or shallow-sloped sites. Of the climatic variables, minimum temperature was the most important, with emergent indigenous trees being more likely to occur at sites with higher minimum temperatures. Emergent indigenous trees were also more likely to occur at windy sites with high rainfall, and to decrease with distance from established indigenous forest.

The BRT model including only variables for which we have national spatial layers also predicted emergent tree occurrence accurately (cross-validated ROC = 0.819, Fig. 14). In this model, slope and minimum temperature were the most important predictors (Table 3, Fig. 14). The relationships between occurrence of emergent indigenous trees for each of the variables in this model were similar to those observed in the model including both GIS variables and site observation variables. Overall, the model revealed that moderate and steeply-sloped sites with high minimum temperature, high rainfall, and in windy areas were more likely to contain emergent indigenous trees than sites with alternative attributes.

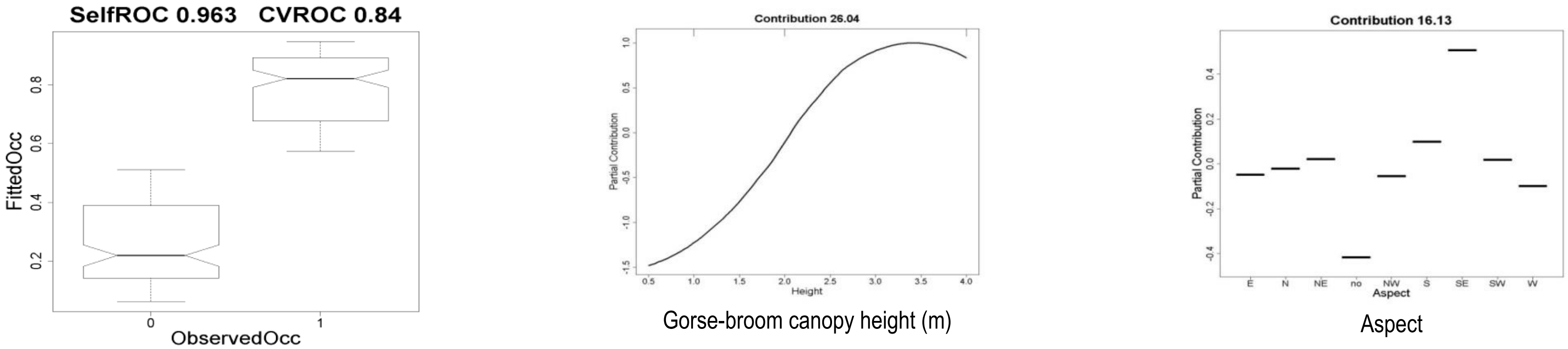
Spatial predictions of the probability of indigenous tree emergence above a gorse or broom canopy are given in Fig. 15. Predicted occurrence probability for 99% of plots where indigenous emergent trees were present was >0.5 while more than 97% of plots where no emergent indigenous trees occurred had a predicted probability of <0.5 (Fig. 14). Consequently, it can be assumed that areas with a predicted occurrence probability of 0.5 or higher are very likely to regenerate from gorse or broom into tall forest.

Table 2: Relative influence of predictor variables in emergent indigenous tree occurrence in gorse or broom plots in the final BRT model including variables obtained both from site surveys and GIS layers

Variable	Contribution
Gorse/broom height	26.04
Aspect	16.13
Minimum temperature	14.85
Slope	13.32
Mean annual temperature	9.09
Mean October wind speed	7.61
Mean annual rainfall	7.30
Distance to nearest forest	5.67

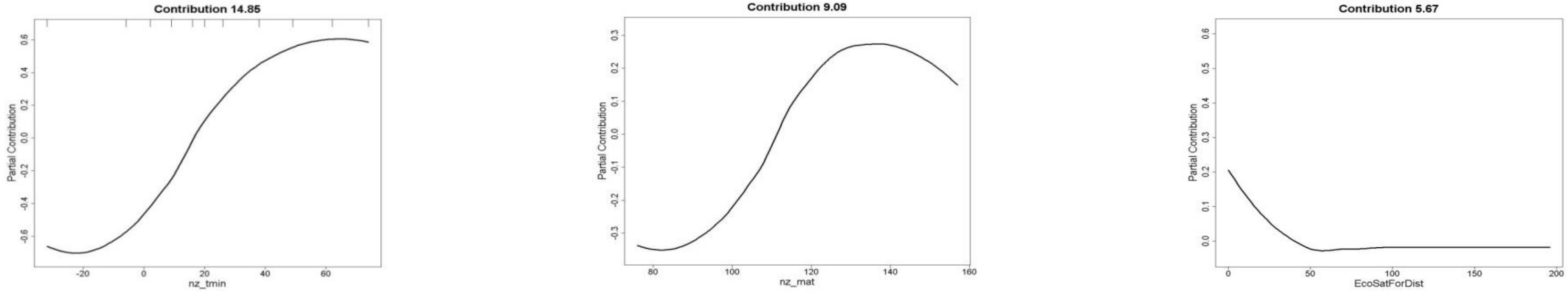
Table 3: Relative influence of predictor variables in emergent indigenous tree occurrence in the gorse or broom plots in final BRT model including only variables obtained from GIS layers

Variable	Contribution
Slope	22.92
Minimum temperature	16.22
Mean annual rainfall	14.91
Mean annual temperature	13.55
Mean October wind speed	13.27
Soil water deficit	12.95
Distance to nearest forest	6.17



Gorse-broom canopy height (m)

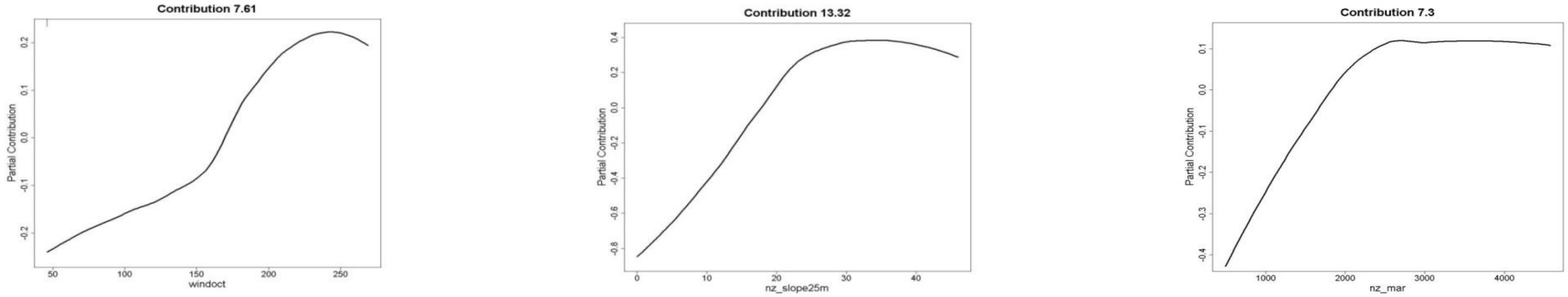
Aspect



Min temperature (˚C x 10)

Mean temperature (˚C x 10)

Distance to forest (m x 100)

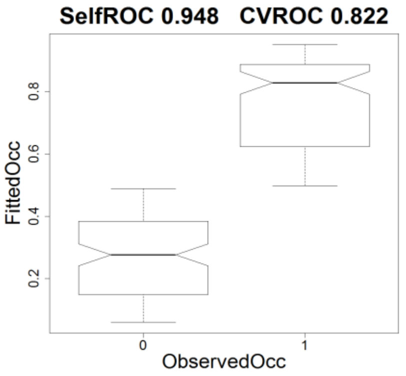


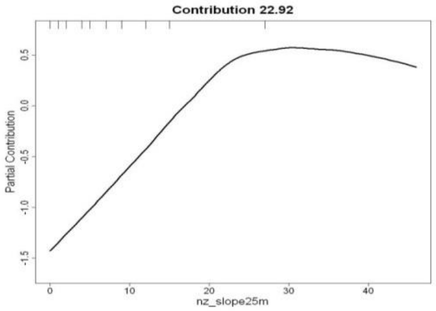
October wind speed (km h^{-1} x 10)

Slope

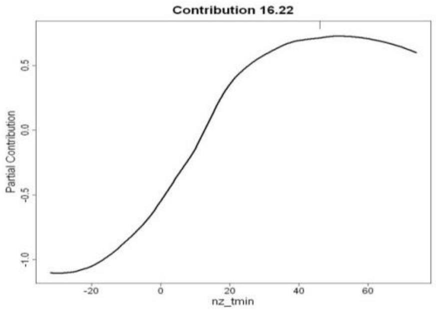
Mean annual rainfall (mm)

Figure 13: Results for BRT model of emergent indigenous tree occurrence in gorse or broom plots including variables obtained from site surveys and spatial variables obtained from GIS layers. The graphs show observed vs fitted forest species occurrences plus 'Contribution' as the percentage of regression tree branches involving each variable. CVROC is the cross-validated receiver operator curve (ROC) for the final boosted regression tree model. ROC is a measure of discrimination accuracy when predicting a binary response. The smoothed partial-contribution plots reflect the influence of a predictor variable when all other variables are held constant.

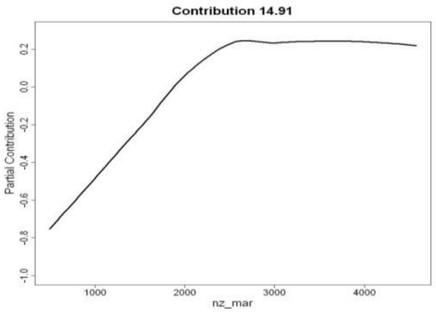




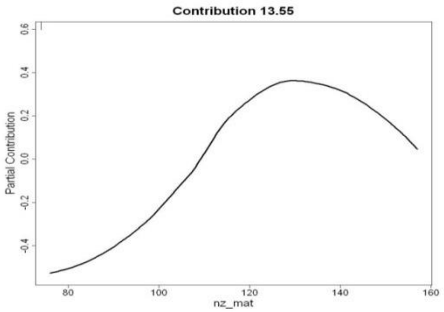
Slope (degrees)



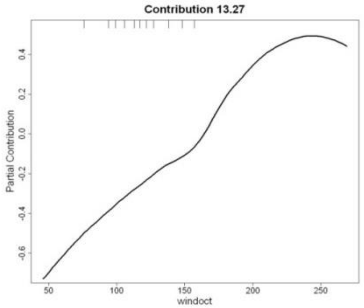
Min temperature (˚C x 10)



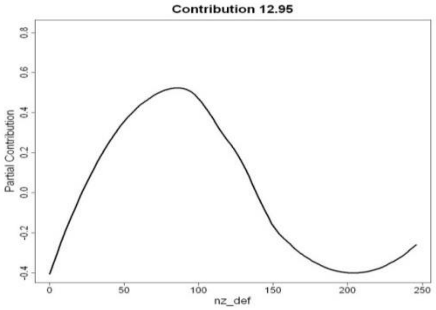
Mean annual rainfall (mm)



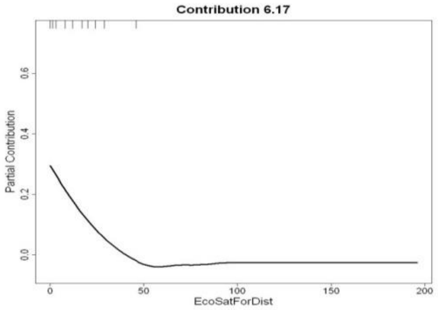
Mean temperature (˚C x 10)



October wind speed (km h r^{-1} x 10)



Water deficit (mm)



Distance to forest (m x 100)

Figure 14: Results for BRT model of emergent indigenous tree occurrence in gorse or broom plots including only spatial variables obtained from GIS layers. The graphs show observed vs fitted forest species occurrences plus 'Contribution' as the percentage of regression tree branches involving each variable. CVROC is the cross-validated receiver operator curve (ROC) for the final boosted regression tree model. ROC is a measure of discrimination accuracy when predicting a binary response. The smoothed partial-contribution plots reflect the influence of a predictor variable when all other variables are held constant.

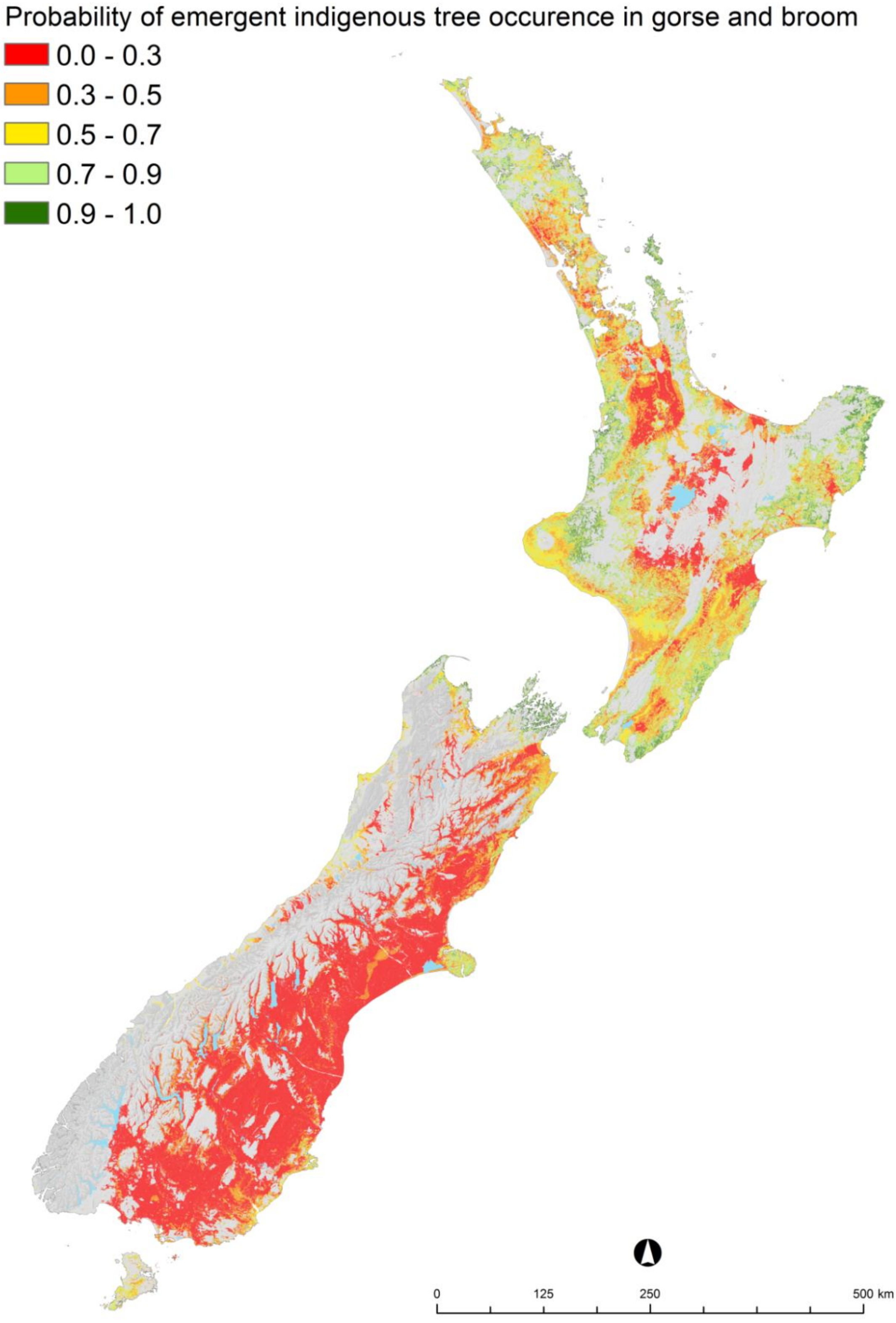


Figure 15: Spatial predictions of the probability of forest species emerging above a gorse or broom canopy, based on GIS layers. Grey areas are those that are already classed as forest in EcoSat (a digital forest map) or are very unlikely to regenerate tall forest due to edaphic factors.

3.2.1 Differences in indigenous tree occurrence and site variables between gorse and broom

Sites containing only broom had significantly lower probability of emergent-indigenous-tree occurrence (0.21) than sites containing only gorse (0.67), or both gorse and broom (0.61). Broom stands we sampled were much less likely to contain emergent indigenous trees partly because they had much lower minimum temperature and mean annual rainfall. There were no significant differences in slope, aspect, land use type, canopy height or physiography. This suggests that differences in occurrence probability between broom-only and other sites may reflect more extreme conditions limiting indigenous tree colonisation rather than an intrinsic tendency for broom to inhibit succession to indigenous forest.

Our study design does not allow us to distinguish the effect of gorse and broom on indigenous tree colonisation from that of climate. However, the fact that composition of the canopy species (i.e. gorse only, broom only or both) was not retained in the simplified model containing both site and GIS variables suggests that we should assume the observed differences in tree occurrence are probably due to differences in climate between broom-only and other sites. It is acknowledged that colder, drier areas may also co-vary with areas with less indigenous seed source.

3.2.2 BRT models for forest species occurrence within non-forested land

The BRT model of indigenous tree occurrence probability predicted tree occurrences remarkably accurately (cross-validated ROC = 0.924, Fig. 16). Predicted occurrence probability for all plots where indigenous forest species were present was >0.6 while more than 95% of plots where no indigenous forest species occurred had a predicted probability of <0.6 (Fig. 16a). Consequently, it can be assumed that areas with a predicted occurrence probability of 0.6 or higher are likely to qualify as forest land.

Mean annual temperature and local woody cover were the two most influential predictors in the simplified BRT model of tree occurrence (Fig. 16, Table 4). The relationship for mean annual temperature was asymptotic, with minimal change in predicted tree occurrence probability above 9°C. The relationship for local woody cover was positive and monotonic, with predicted tree occurrence probability increasing with increasing woody cover. The influence of almost all other variables in the model was an order of magnitude lower than these factors (Table 4). Notable relationships were a monotonic increase in predicted occurrence with minimum temperature and rainfall and a monotonic decline with distance from nearest forest.

Table 4: Relative contribution of predictor variables in the simplified BRT model

Variable	Contribution
Mean annual temperature	44.9
% woody cover within 25 m	34.1
Mean annual rainfall	5.0
Mean annual solar radiation	3.9
Minimum annual temperature	3.5
Distance to forest	2.5
October soil water deficit	2.3
Mean October wind speed	2.2

The two most important interactions were between mean annual temperature and local woody cover (contribution 12.5%), and between minimum temperature and distance to nearest indigenous forest (contribution 5%). Occurrence probability at low mean annual temperature was higher at high levels of local woody cover, reflecting the ameliorating effect for indigenous tree regeneration of extra woody cover on low mean annual temperature. For the minimum temperature – distance to forest interaction, occurrence probability at low minimum temperature was higher when distance to forest was lower and vice versa. This suggests that higher minimum temperatures also reduced the effects of dispersal limitation (by increasing establishment probability of seeds dispersing to sites distant from forest, and hence compensating for lower propagule pressure). It also indicates that proximity to seed source reduced the negative effects of low temperature on tree occurrence probability (by increasing propagule pressure, and hence compensating for the lower per-individual establishment probability). The probability of indigenous tree presence nationally is given in Fig. 17.

There is strong evidence for relationships between trait values, functional trait diversity and key predictors of forest species occurrence. These revealed that abiotic stress (particularly low temperatures), local indigenous woody cover and proximity to seed sources interacted with traits to influence tree occurrence probability (Table 5). Specifically, they demonstrate that as temperature and woody cover decrease, and distance from seed source increases, trait filtering intensifies for ecological-strategy traits but not dispersal traits.

Table 5: Correlations between functional diversity and key land cover and environmental variables. All results are significant at $P < 0.05$

	Mean temperature	Distance to forest	% woody 25 m
Tree height (m)	0.332	−0.303	0.247
Leaf size (mm²)	0.087	−0.158	0.187
Seed mass (mg)	0.330	−0.148	0.145
Dispersal adaptation	−0.414	0.223	−0.261

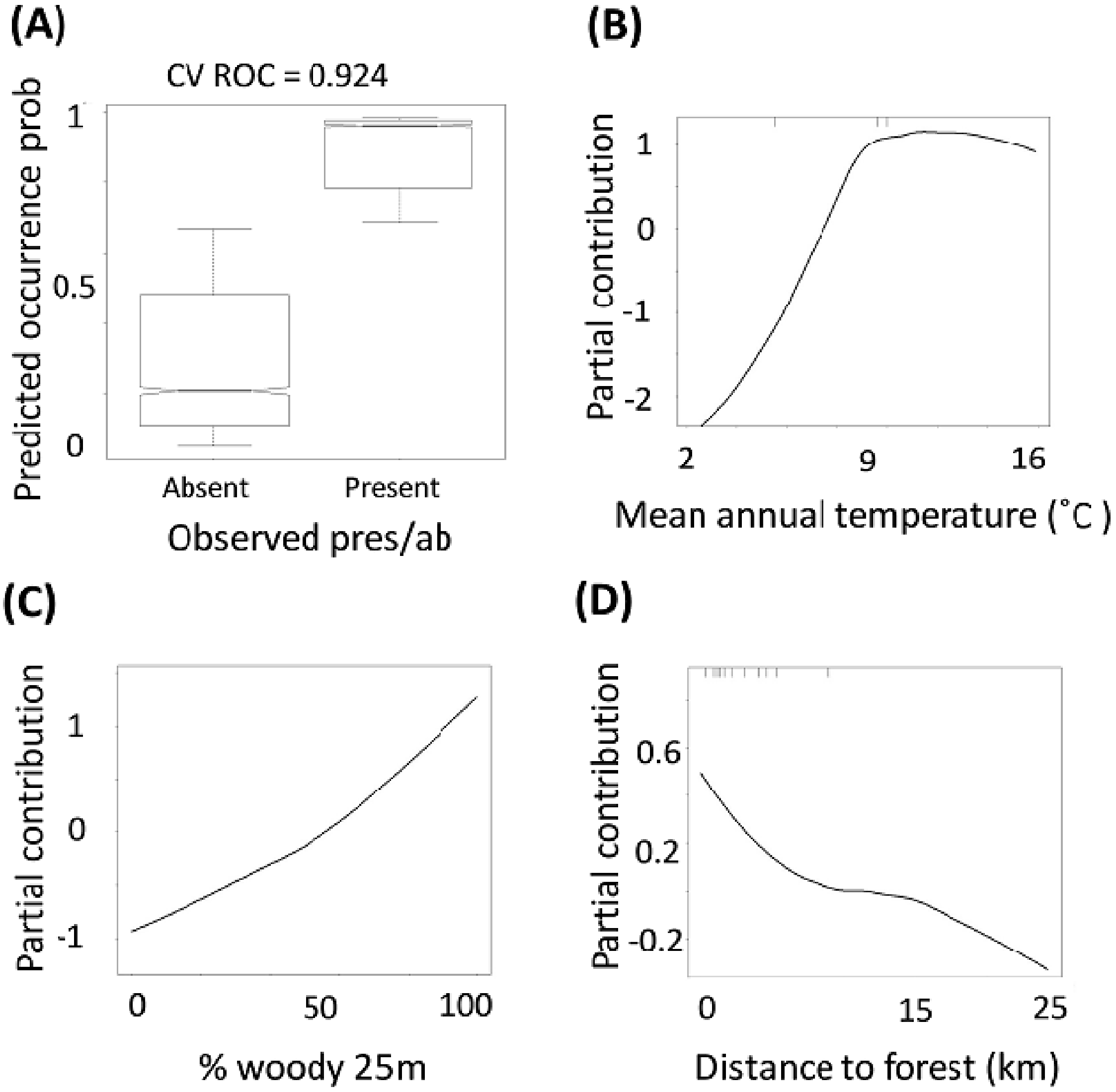


Figure 16: Partial contributions to observed vs predicted indigenous tree occurrences in non-forested land within the simplified BRT model. The graphs show observed vs predicted tree occurrences (A) and smoothed partial contributions within the simplified BRT model for (B) mean annual temperature, (C) percentage woody cover in a 25-m radius and (D) distance to nearest forest. The smoothed partial-contribution plots reflect the influence of a predictor variable when all other variables are held constant.

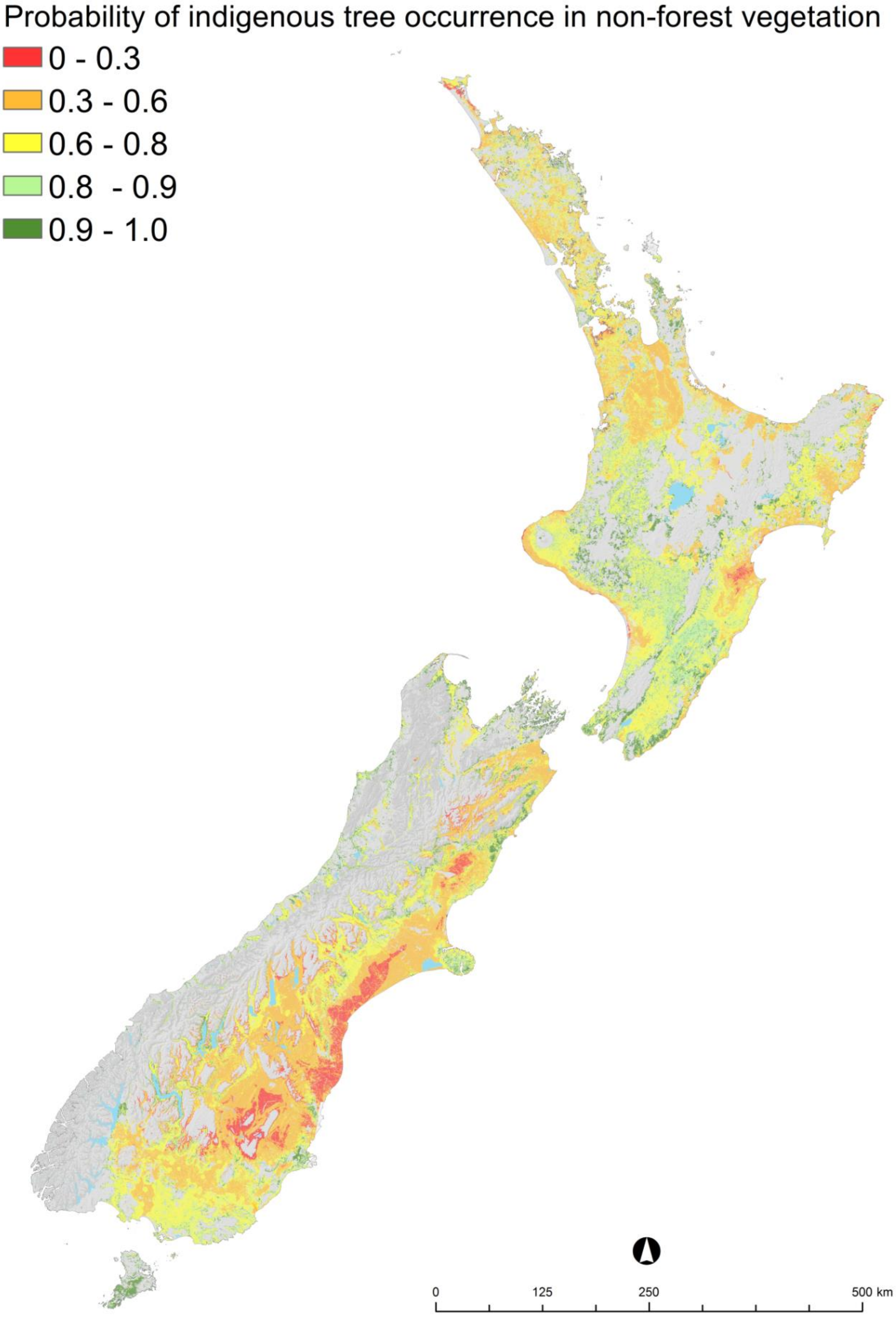


Figure 17: Map of predicted occurrence probability of indigenous forest species in non-forested land across New Zealand. Grey areas are those already in indigenous or planted forests (LCDB2) or where forests will not occur because of edaphic conditions.

Plant dispersal is a highly stochastic process, and makes it difficult to detect patterns. The fact that we obtained such clear evidence for the influence of dispersal limitation on traits emphasises that large datasets, such as the one used in this study, may overcome the variability caused by random dispersal events, such as long-distance dispersal, to reveal dispersal-driven patterns (Higgins et al. 2003). The relative frequency of biotically dispersed species, particularly those dispersed primarily by frugivory, increased with local indigenous woodiness and decreased with distance to indigenous forest. By contrast, the frequency of abiotically-dispersed species, species with no obvious dispersal adaptations and more than one mode of dispersal decreased with local woodiness and increased with distance to forest.

Further detail on this analysis can be found in Mason et al. (2013).

3.3 Time to transition

The maximum age of indigenous and exotic trees within the stands ranged from 3 to 20 years, and measured total crown area above the nurse canopy ranged from 0 to 80%. The best model for individual crown area based on Akaike's Information Criteria (AIC) (Burnham & Anderson 2002) was a log-log relationship with tree age, with basal diameter (D10), maximum tree age on plot (MaxAge) and species as linear covariates (Table 6):

$$\ln(CrownArea) = a(\log(\mathrm{TreeAge})) + b\mathrm{D10} + c(\mathrm{MaxAge}) + d(\mathrm{Species}) \quad .(1)$$

Table 6: Crown-area model selection results

Model	Parameters	AICc	Delta AICc	AICc Weights	Log-likelihood
log(TreeAge) +D10 +MaxAge +Species	25	1996.33	0	1	−972.32
log(TreeAge) +D10 +MaxAge	4	2681.55	685.22	0	−1336.75
log(TreeAge) +D10	3	2964.18	967.86	0	−1479.08
log(TreeAge)	2	3408.93	1412.6	0	−1702.46

This best-fitting model (r^2 of 0.88) had little bias in the predicted vs fitted values, except for very large canopies (Table 6, Fig. 18). Diameter and tree age were the most important predictors, followed by maximum age and then species identity (*F*-values given in Table 7, all terms significant $P < 0.001$). Forest species found across all plots are listed in Table 7.

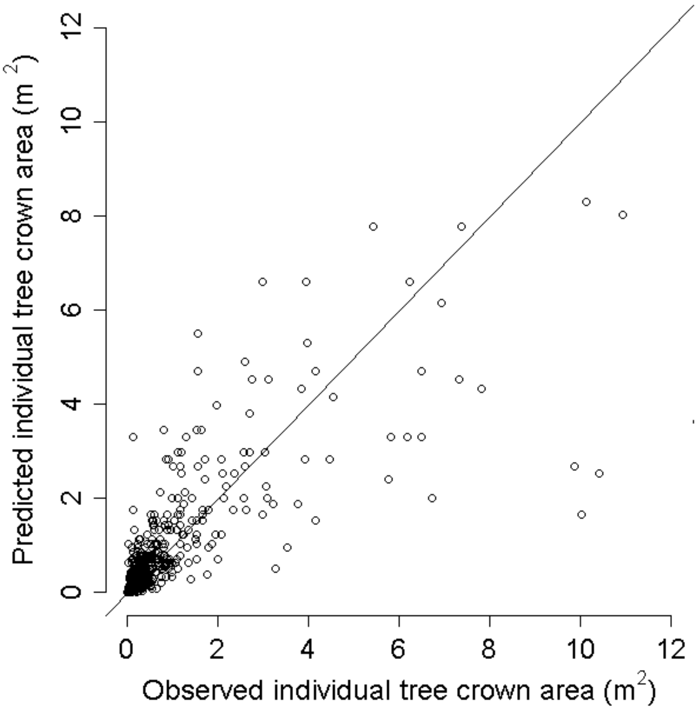


Figure 18: Predicted vs observed individual tree crown areas based on the predictions of Eqn 1. Line is 1:1 derived from 74 plots in gorse or broom communities representing major bioclimatic zones in New Zealand.

Table 7: Coefficients for the best-fitting model (Eqn. 1) used to predict individual crown area derived from 74 plots in gorse or broom communities representing major bioclimatic zones in New Zealand.

Predictor	Coefficient	F-value
Log(age)	1.784	1641
Basal diameter	0.187	2076
Max age	−0.00991	829
Species		56
Carpodetus serratus	−4.518	
Coprosma grandifolia	−4.455	
Cordyline australis	4.168	
Crataegus monogyna	−3.966	
Dacrycarpus dacrydioides	−5.053	
Fuchsia excorticata	−5.625	
Hedycarya arborea	−5.157	
Hoheria angustifolia	−5.671	
Kunzea ericoides	−3.229	
Leptospermum scoparium	−4.443	
Melicytus ramiflorus	−4.836	
Myrsine australis	−4.232	
Myoporum laetum	−4.520	
Nothofagus solandri var. *cliffortioides*	−5.453	
Phyllocladus trichomanoides	−4.216	
Pittosporum tenuifolium	−4.997	
Podocarpus totara	−4.853	
Pseudopanax arboreus	−5.010	
Pseudopanax crassifolius	−4.721	
Schefflera digitata	−4.109	
Solanum mauritianum	−5.545	

The spatially explicit predictions of total tree crown area showed a strong relationship with field estimates of total crown area of forest species above nurse crop (Fig. 19). Field-based estimates were lower than modelled total crown area of forest species for plots with low total crown area, presumably because these plots had a significant portion of the trees occurring below the nurse-crop canopy. For plots with high crown area, the field-based estimate was more closely related to the modelled value (solid vs dashed lines in Fig. 19).

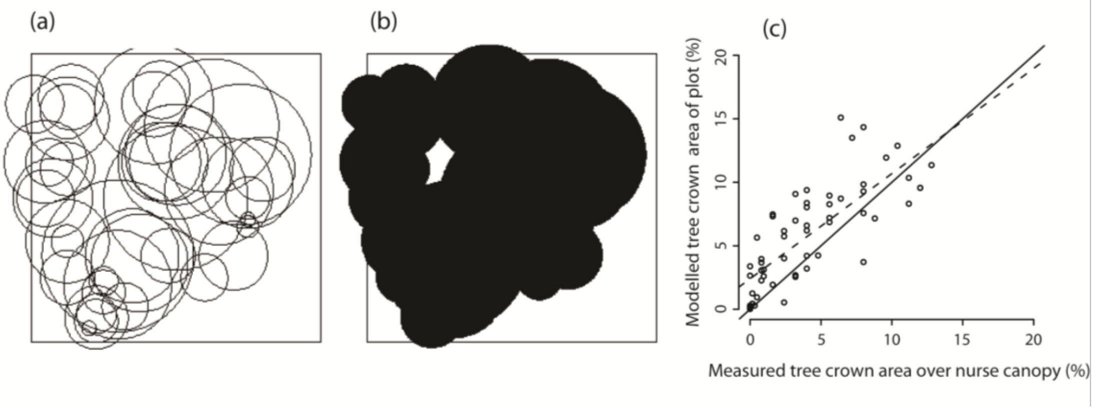


Figure 19: Spatially explicit predictions of total crown area of indigenous and exotic trees in gorse or broom plots showing an example of overlapping crowns (a), total crown area (b), and the relationship between measured and modelled plot-level crown area across all plots (c), with a solid 1:1 line and a dashed fitted line shown.

The individual crown area model (Eqn 1) was used to back-simulate changes in tree crown area and number of stems over time within plots. Simulation results revealed that across all five sites the average time taken for the first forest species to establish within the gorse or broom nurse canopy was 3.6 years ± standard error of 1.5. It took a further 11.1 ± 0.9 years (range 3–18) for the stands to reach 30% tree crown cover (Fig. 20).

The broom stands (Hawkswood and Glenthorne) showed strong evidence that the current cohort of broom was not the first to establish at the site. This is shown in plot records of dead broom logs, present on 70% of plots at Hawkswood and 42% at Glenthorne. Further evidence is shown in the way that the age of the oldest trees exceeds the age of oldest stems of broom on the plot. The time taken for the first tree to establish within gorse or broom significantly differs between species. Average gorse age at establishment of forest species was 6.18 ± 1.21 years while broom mean age when first trees establish was −0.22 ± 0.88 years (ANOVA $F(1,55) = 19.1$, $P < 0.001$). The most likely explanation for this discrepancy is that the oldest broom currently on the plot considerably predates the time that the oldest trees established, i.e. they established under a previous cohort. With these data we are therefore unable to predict the true length of time elapsed between first invasion of grasslands by broom and establishment of first tree seedlings, but it appears to be around 15 years (one lifespan) according to landowner estimates of how long the sites have been in broom. In contrast, the time elapsed between tree establishment in broom and achievement of 30% crown cover can be established as it is within the current cohort’s lifespan.

While there was a lot of within-site variability in the time taken to reach 30% cover from the establishment of the first forest species, there was a significant site effect (ANOVA $F(4,52) = 3.35$, $P = 0.016$). Tukey HSD tests showed that this was driven by the difference between Hawkswood (broom) and Featherston (gorse). All other site pairings within climate zones were non-significant. These sites represent the cool-temperate-dry and cool-temperate-wet climatic zones, respectively, but they contain different shrubland species. There was a significant difference in time to transition between gorse and broom (ANOVA $F(1,55) = 4.86$, $P = 0.03$), with gorse sites tending to transition faster (mean = 10.2 ± 0.53 years) than broom sites (mean time to 30% cover = 12.2 ± 0.75 years). From this dataset we cannot determine if there is a bioclimatic effect on time to transition, although the nationwide survey clearly indicates that relatively mesic climates (i.e. warmer winter temperatures, reasonable rainfall) and sites close to indigenous seed sources were more likely to proceed to ‘forest land’ status. We would expect a shorter time to transition to correlate with a higher likelihood of achieving forest land status but require a larger dataset to test this.

The average number of tree stems at the point of 30% cover was estimated as 13.21 ± 1.62 stems/plot or 8256 ± 1013 stems/ha across all sites (Table 8). Hawkswood is notable for the small number of stems (2187) in a plot, perhaps reflecting episodic bursts of slow regeneration with subsequent reduction in seedling numbers through pig browsing. Because five-finger is the predominant forest species at this site, and it often produces large spreading crowns, 30% crown cover per 4 × 4 m plot could be achieved with only a couple of large stems.

Table 8: Modelled density of tree stems (stems/ha) at point of 30% crown cover (mean ± standard error) using data from 74 gorse or broom sites at five locations across New Zealand.

Site	Modelled number of tree stems per hectare at the time 30% crown cover was first achieved (mean ± standard error)
All sites	8256 ± 1013
Tautoro	9432 ± 1295
Featherston	9261± 1803
Turnbull Bay	6719 ± 1963
Glenthorne	12500 ± 3127
Hawkswood	2187 ± 686

Because we used 4 × 4 m plots with at least 30% crown cover for modelling time to transition, it is likely the crown cover and density of tree seedlings on a per-hectare basis could be considerably smaller than measured within individual plots. Therefore our estimates of time taken to achieve ‘forest land’ transition are likely to be underestimates of the time taken for whole hillsides to transform.

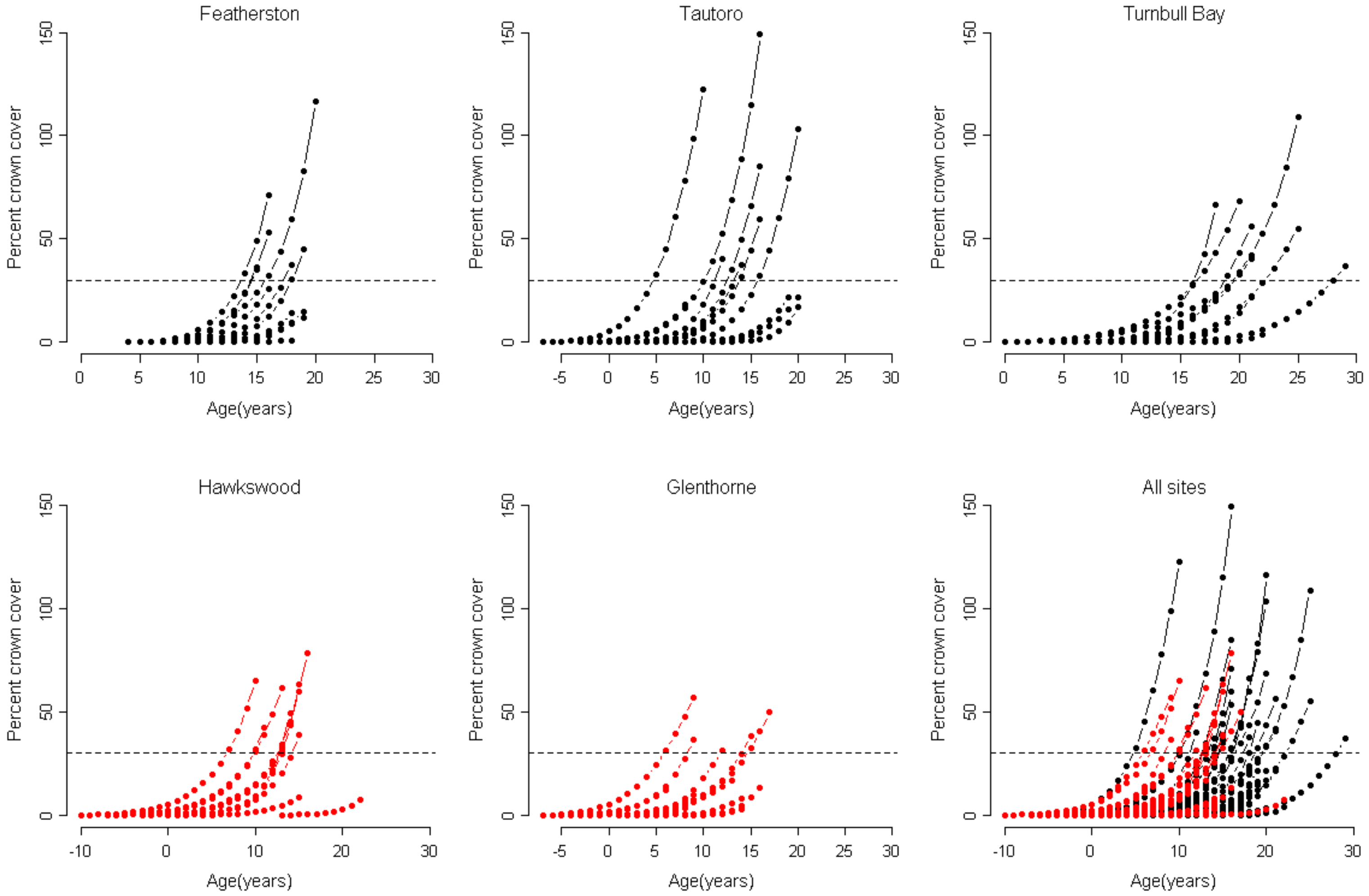


Figure 20: Simulated crown cover establishment for 58 stands of tree regeneration through broom (shown in red – Hawkswood and Glenthorne) and gorse (shown in black – Featherston, Tautoro, Turnbull Bay). Age (years) is relative to the age of the oldest gorse/broom stem for each plot (i.e. age = 0 when current cohort of gorse/broom first established). Dashed line is 30% crown cover.

3.4 Biomass estimation

3.4.1 Biomass components

Vegetation structure and composition within the biomass sampling quadrats at all five intensively-studied sites is shown in Table 9. Of the gorse sites, the shrub (=gorse) cover was tallest and most continuous at the Turnbulls Bay site, followed by Featherston and Tautoro. This was in line with the average age of the shrub cover at these sites, which varied from 17 years old at Turnbulls Bay, to 14 years at Featherston and 11 years at Tautoro. Tree height and total cover followed a similar trend. For the biomass sampling only, forest species could include both indigenous and exotic trees that achieve 5 m height on site (referred to collectively as ‘trees’ in this section).

At the Turnbulls Bay site on Otago Peninsula, the main indigenous tree regenerating through the gorse in the biomass quadrats was mahoe, although black matipo (*Pittosporum tenuifolium*), fuchsia and narrow-leaved lacebark were also present in a few quadrats. The only understorey (non-tree) plants included a low cover (1–2%) of ferns on two of the 14 biomass quadrats, which were not sampled separately but included with grasses. However, almost all quadrats had a cover of litter from the gorse and/or indigenous forest species. All five quadrats with no tree cover included pasture grasses, but other quadrats did not.

At Featherston, the main forest species was again mahoe, followed by rangiora and fuchsia. Understorey shrubs were only present in two quadrats, and included small-leaved *Coprosma* species, and hangehange respectively. Litter was present in most quadrats but not sampled, and there was no grass present.

The Tautoro site in Northland contained the greatest number of indigenous forest species present in the biomass quadrats. These included, in general order of dominance, manuka, red matipo or mapou, mahoe, seven-finger or pate (*Schefflera digitata*), tree ferns (*Cyathea dealbata* and *Dicksonia squarrosa*), totara and in a few instances, kahikatea (*Dacrycarpus dacrydioides*). The only understorey species present was hangehange. Most quadrats had litter, but only two included grass.

For the broom sites, shrub (=broom) height was tallest at the Hawkswood site, but shrub cover and age were slightly higher at Glenthorne. Maximum tree height and total tree cover were also highest at the Glenthorne site.

At Hawkswood, the indigenous forest species present in the biomass plots were kanuka, black matipo, five-finger and lancewood. Many of the biomass quadrats included an understorey of bracken, and a few also included small-leaved coprosma (*Coprosma rhamnoides*) and/or matagouri (*Discaria toumatou*). Only three quadrats included significant litter, but almost all included grass.

At the Glenthorne broom site, the indigenous forest species present was exclusively mountain beech, and no other forest species were present in any of the biomass quadrats. Snowberry (*Gaultheria antipoda*) and tauhinu were present as understorey species in two of the 15 quadrats. All biomass quadrats included litter of broom and/or beech, and all but four included grasses.

Table 9: Vegetation structure and composition at the five intensively-sampled gorse and broom sites. Vegetation structure characteristics are expressed as a mean ± standard error and range

Site	Turnbulls	Featherston	Tautoro	Hawkswood	Glenthorne
Shrub type	Gorse	Gorse	Gorse	Broom	Broom
No. samples	14*	15	15	15	15
Shrub height (m)	2.5 ± 0.98 0.8–4.0	1.8 ± 0.60 1.0–3.1	1.6 ± 0.70 0.1–2.4	2.0 ± 0.45 1.2–3.0	1.7 ± 0.68 0.8–3.5
Shrub cover (%)	59 ± 23.9 29–96	57 ± 29.1 8–100	37 ± 19.6 12–73	54 ± 25.5 13–98	67 ± 17.5 39–89
Shrub age (yr)	17 ± 7.0 6–32	14 ± 3.3 7–18	11 ± 4.9 3–20	11 ± 3.7 3–15	12 ± 3.6 5–19
Max tree height (m)[†]	2.5 ± 1.94 0–5.2	1.9 ± 1.47 0–3.9	1.9 ± 1.38 0–4.0	1.2 ± 1.13 0–3.0	1.9 ± 1.25 0–4.2
Total tree cover (%)[‡]	31 ± 29.6 0–89	35 ± 30.1 0–85	18 ± 16.0 0–49	8 ± 15.3 0–58	14 ± 15.1 0–45
Forest species present	Mahoe, black matipo, fuchsia, hoheria	Mahoe, rangiora, fuchsia	Manuka, red matipo, mahoe, seven-finger, totara, kahikatea, tree ferns	Kanuka, black matipo, five-finger, lancewood	Mountain beech
Understorey (non-tree) height (m)	0	n/s	0.18 ± 0.33 0–1.05	0.20 ± 0.33 0–0.80	0.02 ± 0.08 0–0.30
Understorey (non-tree) cover (%)	0.2 ± 0.6 0–2	n/s	2.5 ± 6.2 0–23	6.5 ± 10.2 0–30	0.7 ± 2.6 0–10
Understorey (non-tree) species	Fern	*coprosma*, hangehange	Privet	Bracken, *coprosma*, matagouri	Snowberry, tauhinu
Grass height (cm)	5.7 ± 11.0 0–36	0	1.6 ± 4.0 0–15	20.5 ± 7.9 0–50	5.2 ± 5.0 0–18
Grass cover (%)	12 ± 24 0–68	0	9 ± 25 0–91	50 ± 25 0–98	23 ± 25 0–81
Litter depth (cm)	5.7 ± 2.4 0–9	n/s	1.2 ± 0.7 0–3	0.7 ± 1.5 0–5	1.7 ± 1.3 0–3
Litter cover (%)	85 ± 30 0–100	n/s	52 ± 28 0–91	7 ± 16 0–50	59 ± 44 0–100

* Only 14 quadrats were sampled at the Turnbulls Bay site due to a change of study site from the original intended study location (at Taioma Road, Mosgiel), which had insufficient tree cover.

[†] Maximum tree height is the site average of the maximum tree heights recorded in each of the 15 biomass quadrats.

[‡] Total tree cover is the site average of the sum of the percent covers for all forest species present within each of the 15 biomass quadrats. As such, it includes possible overlap of tree crowns from species present within different vertical strata.

Total above-ground biomass (TAGB) represents the oven-dry weight of all material above the soil within the sampling quadrat area (expressed as kg/m^2). It includes the gorse and tree canopies, and branch and stem wood, as well as understorey, surface vegetation (such as grass) and litter, if present. The TAGB biomass, as well as the biomass of the contributing components, for the three intensively-studied gorse sampling sites is shown in Table 10 (also see Fig. 21).

Average TAGB across all gorse sites was 5.92 kg/m^2. TAGB was highest at Turnbulls Bay, where it averaged 10.20 kg/m^2 and in one instance exceeded 31 kg/m^2. This was followed by Featherston (5.25 kg/m^2), then Tautoro (2.59 kg/m^2). In all instances, the gorse component provided the majority of the TAGB, averaging 70% (or 3.65 kg/m^2) across all quadrats. Gorse biomass was greatest at Turnbulls Bay (79% or 8.37 kg/m^2), followed by Featherston (68% or 3.26 kg/m^2), then Tautoro (59% or 1.61 kg/m^2). Gorse biomass was lower for quadrats containing trees (see Fig. 21).

Table 10: Biomass associated with the various vegetation strata present at the harvested gorse sites. Values are expressed as mean ± standard error, and range

Site	Turnbulls	Featherston	Tautoro
Gorse (kg/m^2)	8.37 ± 6.93	3.26 ± 1.20	1.61 ± 1.21
	1.23–30.08	1.49–5.58	0.25–3.84
Trees (kg/m^2)	0.85 ± 0.94	1.94 ± 1.96	0.58 ± 0.86
	0–2.80	0–7.04	0–2.74
Understorey (kg/m^2)	0	0.05 ± 0.18	0.03 ± 0.06
		0–0.71	0–0.18
Grass (kg/m^2)	0.07 ± 0.13	0	0.03 ± 0.09
	0–0.32		0–0.28
Litter (kg/m^2)	0.91 ± 0.34	n/s	0.33 ± 0.20
	0–1.24		0–0.73
AGB excl. trees (kg/m^2)	9.35 ± 7.00	3.31 ± 1.27	2.01 ± 1.25
	1.51–31.09	1.49–5.58	0.53–4.57
TAGB (kg/m^2)	10.20 ± 7.06	5.25 ± 1.94	2.59 ± 1.38
	1.51–31.09	2.70–9.15	0.53–4.63
Tree biomass by volume (kg/m^3)*	0.56 ± 0.50	2.25 ± 1.85	1.65 ± 3.05
	0–1.29	0–4.98	0–12.18
AGB by volume excl. trees (kg/m^3)†	3.53 ± 1.46	2.03 ± 0.85	2.05 ± 3.19
	1.80–7.77	0.49–4.06	0.51–13.41
TAGB by volume (kg/m^3)‡	3.70 ± 1.53	3.28 ± 1.17	1.61 ± 0.63
	1.80–7.77	1.63–5.75	0.39–2.71

* Biomass by volume of trees only, based on the cover-weighted average height of trees present.

† Biomass (excluding trees present) by volume expressed based on average shrub canopy height.

‡ Total biomass (including trees present) by volume expressed based on the cover-weighted average canopy height for shrubs and trees if present.

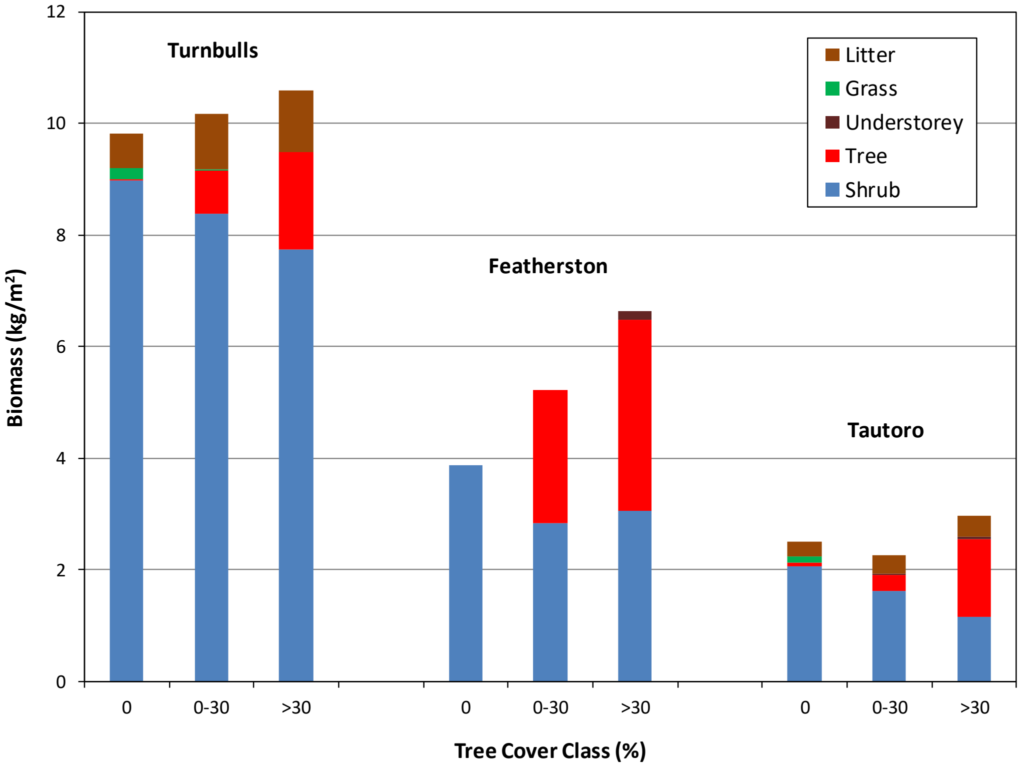


Figure 21: Biomass of vegetation components for each of the intensive gorse sampling sites by tree cover class.

The indigenous tree layer was the next most significant component, averaging 17% of TAGB (or 0.86 kg/m^2). Tree biomass in gorse was greatest at Featherston (32% or 1.94 kg/m^2), followed by Tautoro (20 % or 0.58 kg/m^2) and Turnbulls Bay (8% or 0.85 kg/m^2). As would be expected, tree biomass increased by tree cover class at all sites (Fig. 21).

Litter was a significant contributor to biomass at several sites, comprising on average c. 8% of TAGB (0.4 kg/m^2). The litter contribution may have been greater if litter at the Featherston site had been measured (where depths and biomass were at least comparable with Tautoro). Litter biomass weight was greatest at Turnbulls Bay (0.91 kg/m^2) but it only comprised 10% of TAGB, whereas at Tautoro it made up 14% but averaged only 0.33 kg/m^2. Litter biomass increased with tree cover class at all sites, and especially those with no tree cover present.

Grass was only a relatively minor component of the TAGB, averaging 5% (0.09 kg/m^2) across all sites. Grass biomass was highest at Turnbulls Bay (0.07 kg/m^2 or 3%) then Tautoro (0.03 kg/m^2 or 5%). Grass biomass decreased with increasing tree cover class at all sites (Fig. 21), with the exception of the Featherston gorse site where there was no grass present.

The understorey component was also only a minor contributor to the overall TAGB. For Featherston it was only 0.05 kg/m^2 (0.7%) and Tautoro 0.03 kg/m^2 (1.4%), where privet or *Coprosma* species were present in just a few quadrats. There were no understorey species present in any of the biomass quadrats at the Turnbulls Bay gorse site. Due to the relatively low occurrence of understorey vegetation across sites, and quadrats within sites, there was no clear pattern of understorey presence with tree cover class (Fig. 21).

Data collected for the ETS using the Forest Measurement Approach (FMA) is separated for some specific shrubland types from others as well as for forest species for regenerating indigenous forests. Therefore the above-ground biomass (AGB) excluding the biomass associated with forest species is also shown in Table 10. This includes biomass contributed by all other components present. However, this should be interpreted with caution, as it was not possible to exclude all influences associated with the tree cover. For example, the litter component includes some tree leaf litter for plots containing larger trees and where tree cover was higher. Similarly, the presence of tree cover can also influence shrub cover, and potentially also the presence of understorey species, thereby affecting the biomass associated with these components.

To aid in quality control when processing FMA data, biomass values associated with the key vegetation components in Table 10 have also been expressed by volume (kg/m^3). These were obtained by converting biomass weights using canopy heights. In the case of trees, the cover-weighted average height of tress was used; for AGB (i.e. excluding trees) the average shrub canopy height was used; and for TAGB the cover-weighted average canopy height of both shrubs and trees was used.

3.4.2 Gorse height–biomass allometry

Correlations between biomass components and measured variables for gorse are shown in Appendix 1. This shows a relatively strong correlation between TAGB and gorse height ($r = 0.70$), and between TAGB and gorse age ($r = 0.75$). These results agree with those from many previous studies in a wide range of vegetation types, including shrublands.

These previous studies have led to the development of allometric relationships predicting biomass from vegetation height or age, sometimes with other structural characteristics. For example, biomass in gorse has previously been related to shrub height and/or shrub cover (Fogarty et al. 1998; Fogarty & Pearce 2000; Pearce & Anderson 2008). Most recently, Pearce et al. (2010) produced a log-log model using linear mixed-effects modelling for predicting TAGB in gorse from shrub height. This model was based on data obtained from 55 quadrats at 10 sites across the country. The model for TAGB (in kg/m^2) and gorse height in metres (m) is:

$$\ln(\text{TAGB for gorse}) = 1.440 + 0.872 \times \ln(\text{height}). \tag{2a}$$

During rearrangement of this equation to the allometric form of the model,

$$\text{TAGB (gorse)} = 4.221 \times (\text{height})^{0.872}, \tag{2b}$$

the right-hand side of equation (2a) needs to be multiplied by the bias correction factor, which in this case was 1.02. The modelled regression line, 95% confidence bands for individual quadrat estimates, and 95% predictions intervals for site averages for this Pearce et al. (2010) model are shown in Fig. 22. The resulting regression line had a relatively strong fit to the original data from which it was developed, as illustrated by low root mean square error (RMSE = 2.00) and mean absolute percentage error (MAPE = 17%).

The TAGB estimates obtained for the three intensively-sampled gorse sites from the current study compare relatively well with this model from Pearce et al. (2010). Estimates from the Turnbulls Bay site fit well with the model prediction intervals, on the basis of both individual quadrat data (Fig. 22a) and the site mean (Fig. 22b). Estimates for Featherston fall slightly below the regression model (potentially due to the lack of litter biomass), while the Tautoro values are lower than would be predicted by the model.

It is important to note that the Pearce et al. (2010) model was developed from continuous-cover quadrats, where there was relatively high shrub cover (average 77%, standard deviation 16%) and little or no tree cover present. As such, it would not be expected to predict as well as where shrub cover was low or where tree cover and/or understorey or grass cover was high. This may explain why the model overpredicts for some Tautoro and Featherston quadrats.

From Fig. 22a, it is apparent that there is some effect of forest species cover class on the fit of the Pearce et al. (2010) gorse model. TAGB estimates including tree biomass for the higher tree cover classes (>30% represented by the squares, and 0–30% represented by the diamonds) are generally within or just above the prediction intervals for these quadrats at Turnbulls Bay, and within or just below for Featherston. All quadrats for Tautoro (with the exception of one) are well below the Pearce et al. (2010) model prediction intervals. Unexpectedly though, the majority of quadrats with no tree cover (0%, represented by circles) for both Turnbulls Bay and Featherston fall below the prediction interval, when these are the plots for which the model should predict best (i.e. continuous shrub cover, with no trees). A possible explanation for this would be that shrub cover here was lower than in the dataset used to produce the Pearce et al. (2010) model, but this is not the case, as this averaged 81%, 92% (and 48%), respectively, at the Turnbulls Bay, Featherston (and Tautoro) sites. This would tend to suggest that the Pearce et al. (2010) model is simply overestimating TAGB for these sites.

The one Tautoro plot where measured biomass (gorse height 0.09 m, TAGB 3.95 kg/m^2) was well above that predicted by the model may be explained by the very high manuka tree cover present in this quadrat (height 2.55 m, cover 43%), as when some or all of the biomass associated with this (2.55 kg/m^2) is removed, the estimate lies much closer to the model prediction. This data point should therefore be considered as an outlier.

The uppermost Turnbulls Bay data point with >30% tree cover with the highest recorded average gorse height (4.0 m) and TAGB (31.09 kg/m^2) should be considered less of an outlier, however. This quadrat contained the oldest gorse plants recorded at any of the sites (mean age 30 years for the three plants sampled), and large stemwood comprised the majority of the gorse shrub (82%) and total (79%) biomass respectively. This plot was representative of a number of older gorse bushes observed at this site and at other sites in Otago (Lee et al. 1986) where senescent gorse begins to collapse. Similar biomass and structure is likely to occur elsewhere where gorse reaches maturity without disturbance (e.g. being burnt).

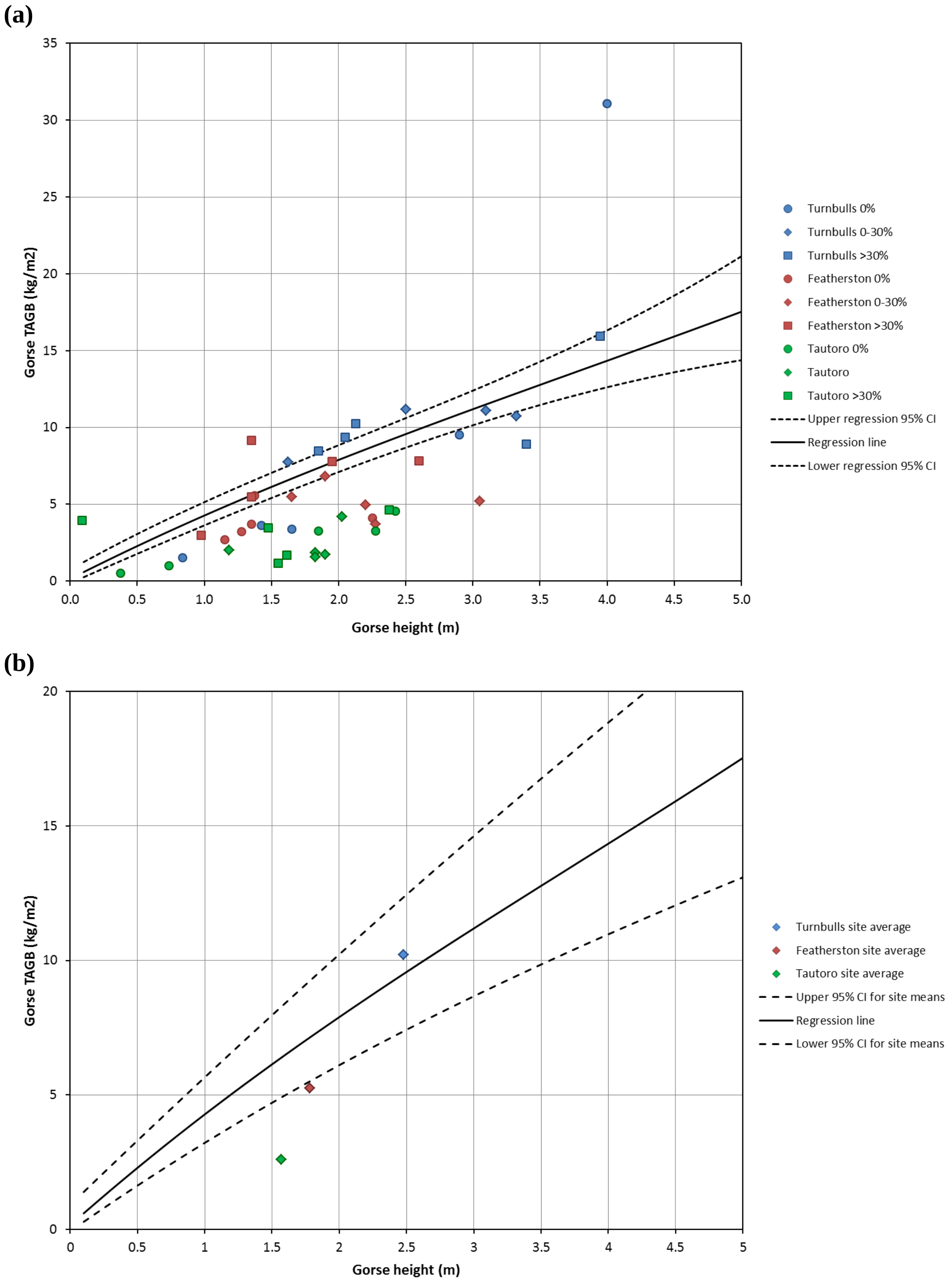


Figure 22: Comparisons of measured total above-ground biomass (TAGB) in gorse with model predictions from Pearce et al. (2010), based on (a) individual quadrats by site and tree cover class, and (b) site means.

A plot of above-ground biomass (AGB) (Fig. 23) shows that excluding the biomass associated with the tree cover does not improve the fit with the Pearce et al. (2010) model. In fact, the fit is worse (cf. Fig. 22a), with many more of the sampled quadrats falling below the confidence interval of the model. While the Tautoro >30% tree cover data-point that was previously an outlier (see Fig. 22a) now falls within the confidence interval, many of the other quadrats that included tree cover at Tautoro, and at Featherston in particular, now fall further below the model estimates. The Turnbulls Bay outlier also remains, as this did not include any trees.

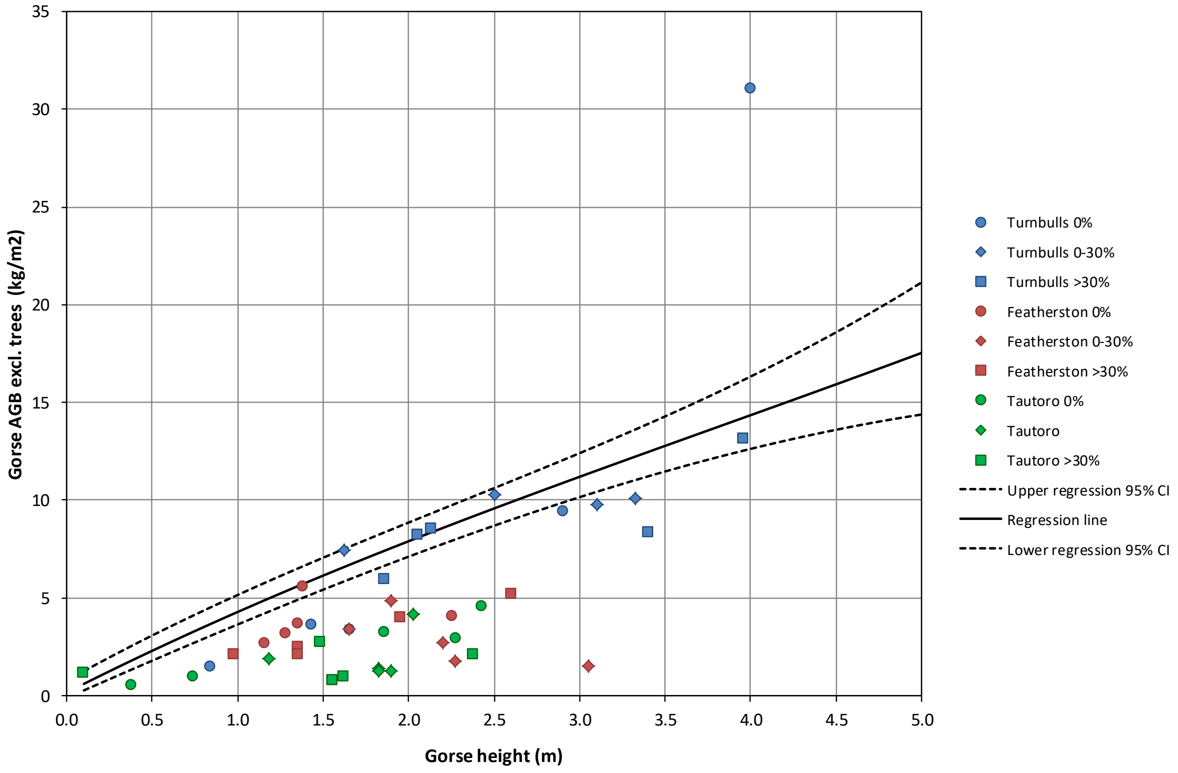


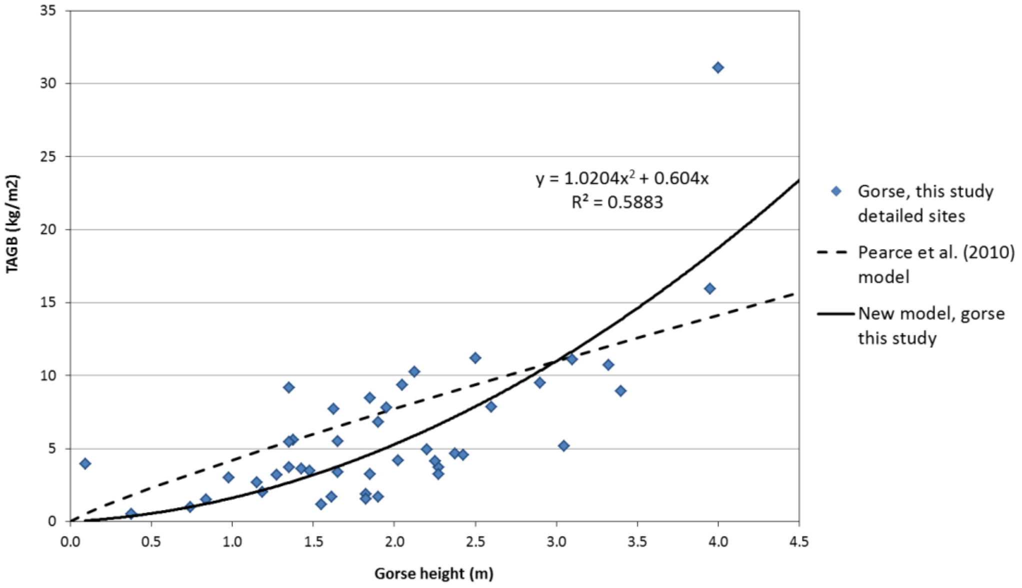
Figure 23: Comparisons of measured above-ground biomass (AGB) in gorse excluding indigenous tree biomass with model predictions from Pearce et al. (2010), for individual quadrats by site and tree cover class.

Gorse TAGB from the three intensive sampling sites is relatively strongly correlated with gorse shrub height (r = 0.75), which suggests that a new TAGB prediction model could be developed using these data. However, a simple power function model of the same form as the underlying Pearce et al. (2010) model results in only a very low model fit ($R^2 = 0.30$). Exponential ($R^2 = 0.49$) and quadratic polynomial ($R^2 = 0.59$) models produce better fits, especially for the higher TAGB values observed at taller gorse heights. The best of these models, with a quadratic form (Fig. 24a) forced through the origin, produces lower TAGB values than the Pearce et al. (2010) model for low-height gorse, and similar values for medium heights. But as noted above, it provides better estimates for taller (and, likely older) gorse, where large gorse stemwood or tree cover makes up significant components of the total biomass. However, it is based on data from only three sites, so this model should be used with caution until data from further sites are obtained to validate it.

An alternative model, using the data from the three intensively sampled gorse sites in this study combined with the underlying data used to produce the Pearce et al. (2010) model, is also presented (Fig. 24b). Also a quadratic model forced through the origin, this has an even better fit for the combined dataset ($R^2 = 0.65$), predicting better for low to intermediate heights, but less well for tall gorse. The equation for this model is:

$$\text{TAGB (gorse)} = 0.440 \times (\text{height})^2 + 2.32 \times (\text{height}). \tag{3}$$

(a)



(b)

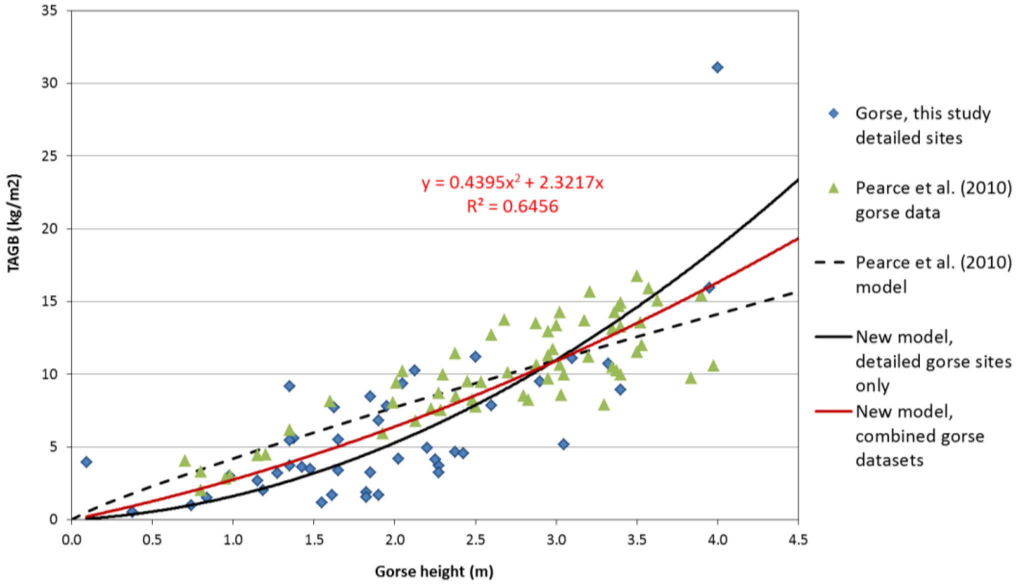


Figure 24: Comparison of model predictions for TAGB in gorse from height for: (a) a new quadratic model based on study data, versus the model from Pearce et al. (2010); and (b) an alternative quadratic model based on the data from this study and Pearce et al. (2010) combined, versus the Pearce et al. (2010) relationship and new model proposed in (a).

3.4.3 Gorse age–biomass allometry

TAGB in gorse was also strongly correlated with shrub age (r = 0.75). No models are known to exist for predicting biomass from age for New Zealand gorse, although some data do exist for different gorse age classes (e.g. Egunjobi 1969, 1971; Lee et al. 1986). However, models have been developed for *Ulex* species overseas (e.g. Beaza et al. 1998; Navarro Cerrillo & Blanco Oyonarte 2006; Augusto et al. 2009).

Therefore it was necessary to derive a new allometric model for predicting gorse TAGB from gorse age. Again a quadratic polynomial model produced the greatest predictive capability ($R^2 = 0.73$):

$$\text{TAGB (gorse)} = 0.031\times (\text{height})^2 - 0.095\ (\text{height}) \qquad (4)$$

This model (Fig. 25) shows a good fit across the full range of gorse ages observed.

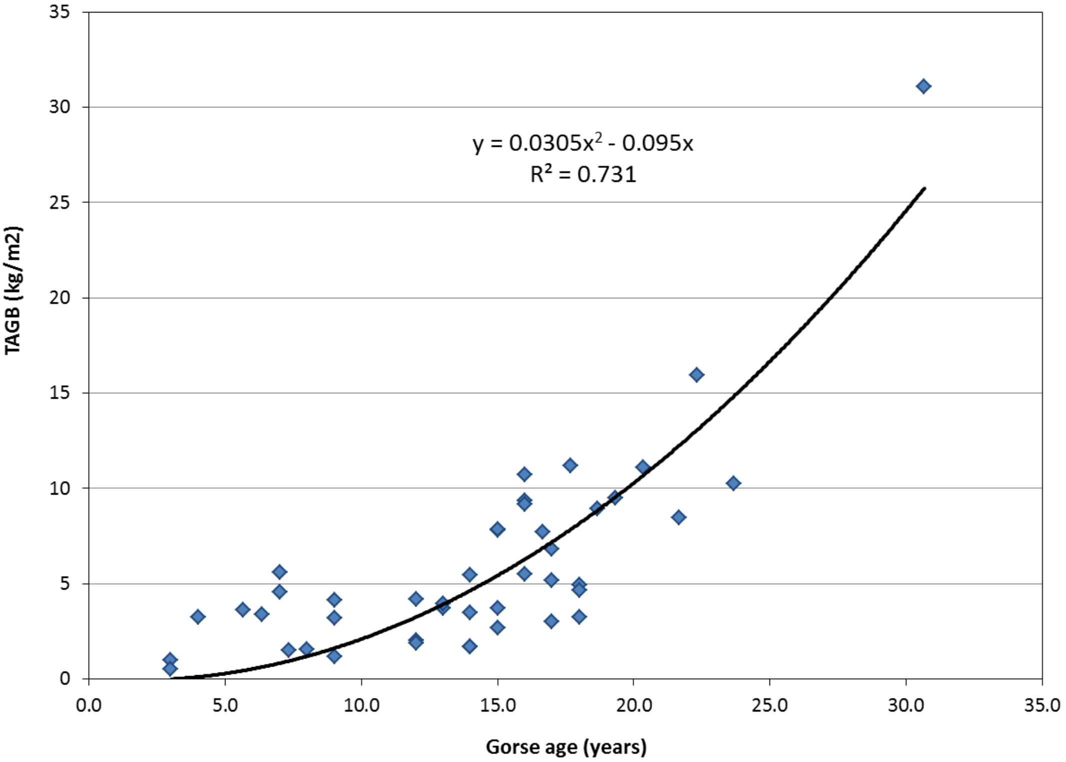


Figure 25: New allometric model for predicting TAGB in gorse from gorse age.

Age and height of gorse from the sampled quadrats were also relatively strongly correlated with each other (r = 0.62) (see Appendix 1). There are at least two known unpublished New Zealand studies that report relationships for predicting between height and age in gorse vegetation (Richardson et al. 1998, 2001). These studies tested a range of sigmoidal growth curves (e.g. Chapman-Richards, Schumacher and Weibell) as well as exponential relationships for predicting gorse height from age at a range of forest sites in the central North Island (Richardson et al. 2001), and Southland, Otago and Canterbury (Richardson et al. 1998) where gorse and other woody species such as broom and buddleia (*Buddleja davidii*) are present and compete with plantation tree growth. Some of the published data sources already mentioned for New Zealand gorse (e.g. Egunjobi, 1969, 1971; Lee et al.

1986) may also provide more information that can be used to test or further refine these models, as well as those developed here. There are also international studies containing relationships for predicting height from age for gorse, especially from the Mediterranean (e.g. Beaza et al. 1998; Puentes & Basanta 2002), although many of these are for natural populations of *Ulex* species other than *U. europaeus* so they are unlikely to be applicable to New Zealand gorse.

Despite the relatively strong correlation for gorse, the age–height relationship (Fig. 26) was more variable than for the previous gorse TAGB–height and TAGB–age relationships (Figs 24 and 25). This results in lower model R^2 values. In this instance, a quadratic polynomial model (forced through the origin) again proved to have the greatest predictive capacity (R^2 = 0.32), better capturing the height increase observed for older gorse (Fig. 26a):

$$\text{Height (gorse)} = -0.0015 \times (\text{Age})^2 + 0.158 \times (\text{Age}) + 1.123. \tag{5}$$

This and the next best logarithmic model (R^2 = 0.31) do show a general flattening off of height with age for older stands as they reach a maximum canopy height (of c. 4 m). However, there is no evidence from the sampled gorse data of a decrease in canopy height for older stems as the canopy collapses. The inverse relationship for predicting gorse age from gorse height (R^2 = 0.29) is:

$$\text{Age (gorse)} = -0.805 \times (\text{Height})^2 + 8.987 \times (\text{Height}). \tag{6}$$

Additional data from the gorse rapid height–age assessments undertaken at a range of sites in Canterbury and the Kapiti Coast also were used to determine whether a better model could be produced. However, despite being collected from gorse stands growing under a range of environment conditions, these additional data were obtained for predominantly young gorse less than 10 years old (Fig. 26b). As a result, the combined model has lower prediction capability (R^2 = 0.25), particularly for older gorse, where it significantly under-predicts maximum canopy height. It is recommended that this combined model not be used. Further information, especially for gorse stands older than 20 years, is required to develop a more robust model for predicting gorse canopy height from gorse age.

Correlations also exist between shrub TAGB and other structural characteristics, including indigenous tree cover characteristics, and also between the various structural components themselves (Appendix 1). It may therefore be possible to develop even better models by including additional factors. For example, in gorse, TAGB is correlated with gorse age and height, but not with shrub cover, tree height or cover, or litter depth or cover. However, it is correlated with shrub layer biomass, which in turn is also correlated with gorse height and age. Gorse age is also correlated with tree height and cover (for Tree1, the most significant forest species present), which are also both correlated with Tree1 biomass as well as total tree cover and biomass. Total tree cover in turn is inversely correlated with shrub cover. Gorse age is also inversely correlated, albeit only weakly, with grass cover and grass biomass. Similarly, gorse height is also weakly correlated with litter biomass. Interestingly, altitude is also correlated relatively strongly with litter depth, cover and biomass.

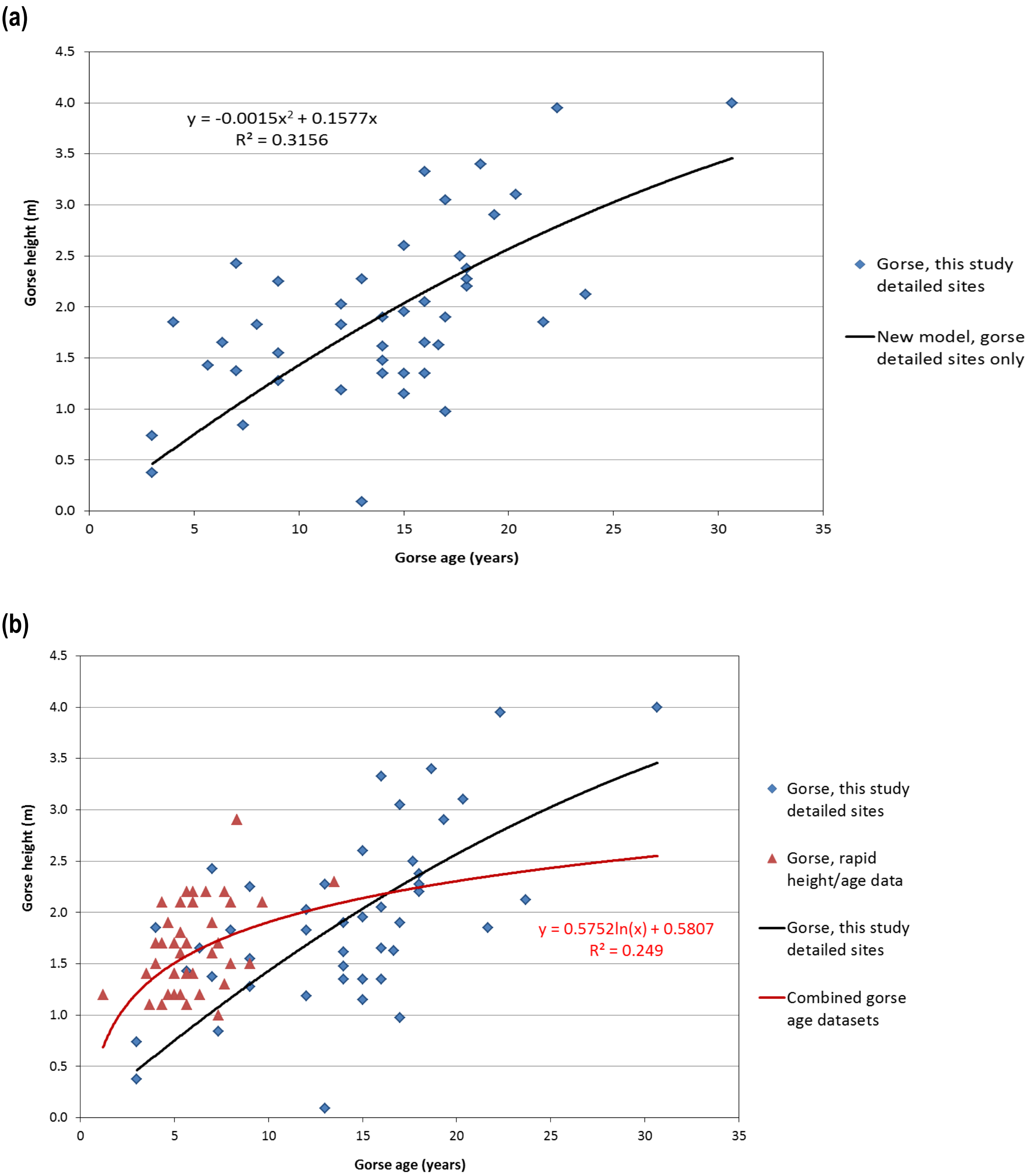


Figure 26: New allometric models for predicting gorse height from gorse age based on data collected from: (a) the three detailed destructive sampling sites only; and (b) detailed sites and rapid non-destructive gorse height–age assessment sites from Canterbury and the Kapiti Coast.

However, to achieve the intended purpose of providing simple models for use in determining carbon stocks for regenerating shrubland, any additional factors included need to be variables that are easy to measure or estimate (e.g. by landowners or from remote sensing). One area where potential might exist for including additional factors to improve model accuracy is through the inclusion of tree cover or cover class, or tree height, which should all be relatively simple to estimate.

In the above allometry, models have been fitted to total above-ground biomass (TAGB) data that include the non-gorse biomass components (i.e. trees, as well as understorey and litter). This is due to the effect of tree cover on reducing the shrub cover (and biomass) in continuous-cover shrublands, as the trees grow up through and eventually overtop and shade out the shrub cover. In effect, a proportion of the shrub cover is being replaced by trees, but the additional tree cover and biomass offset the reduction in shrub cover and biomass. This is evident in data from the current study, where shrub cover is negatively correlated with tree cover (r = −0.44) and tree biomass (r = −0.47). Comparisons based on TAGB including the non-gorse component were also necessary to enable comparison with the model of Pearce et al. (2010), where the TAGB included the biomass associated with understorey vegetation (including any indigenous seedlings if present) and litter.

Therefore, unlike discrete shrubs, biomass in continuous-cover gorse shrublands shows little correlation with shrub cover. Pearce et al. (2010) found that addition of cover to predictive biomass models produced very marginal gains in model accuracy and predictive capability. These minimal gains were, in their view, outweighed by errors in cover estimation, and the greater time and expense associated with collecting these additional data. The correlations between shrub cover and gorse biomass for the three sites in this study were very low (r = 0.13 for TAGB, and r = 0.28 for shrub biomass), and there were no observable trends with shrub cover for any of the biomass components (tree biomass did show a moderate relationship with tree cover, R2 = 0.56) (Fig. 27).

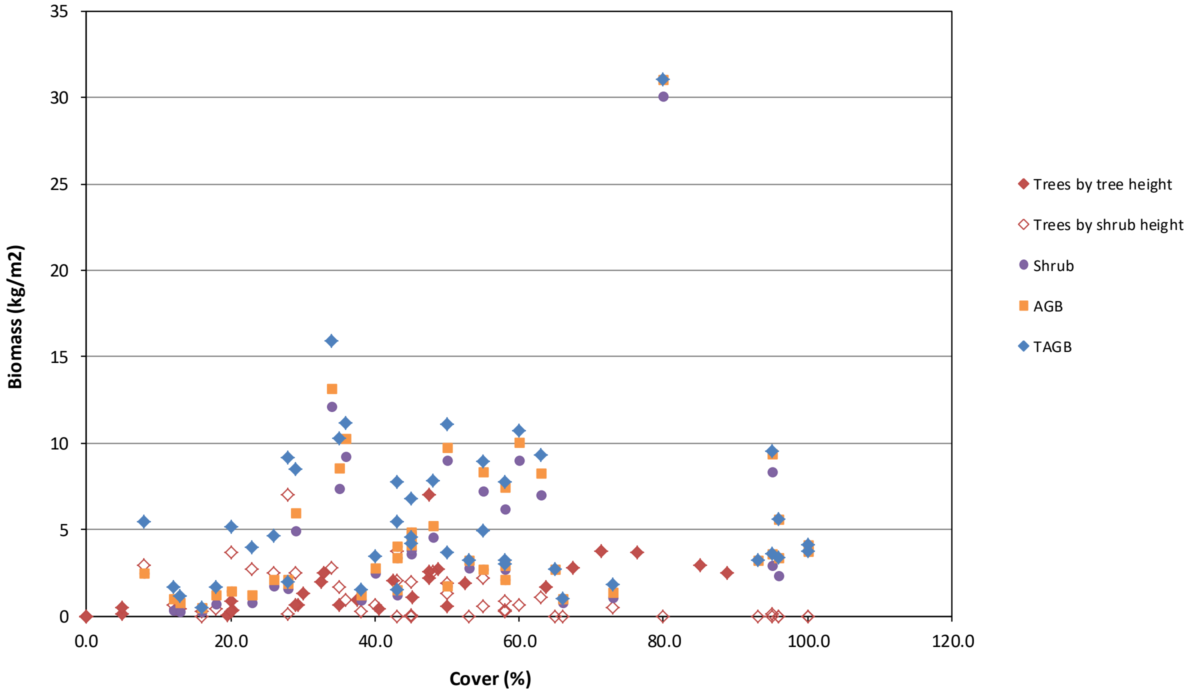


Figure 27: Relationships between vegetation cover and biomass in gorse. Biomass associated with trees present is shown plotted against both tree cover (solid diamonds) and shrub cover (open diamonds); all other components are plotted against shrub cover.

Scaling of biomass based on crown cover therefore offered no advantage over use of the raw biomass data. Plotted data were more scattered so that relationships with height (Figs 28 and 29) and also age were weaker, and fitted models for biomass normalised by cover showed less predictive capability (lower R2) than for non-normalised biomass. This was the case for TAGB for both data from the present study and those of Pearce et al. (2010) (Fig. 28 cf. Fig. 24), and for various biomass components where only very weak relationships (R2 < 0.45) were apparent (Fig. 29).

The modelling analyses undertaken here also represent an initial investigation only, involving simple correlation studies and trend fitting. Future studies should include more comprehensive analyses aimed at developing more robust models where the statistical significance of the variables included and resulting model fit are more accurately summarised. This should also include consideration of the use of more advanced statistical modelling procedures, such as multiple regression and linear mixed effects modelling where more than one predictor variable is considered/used.

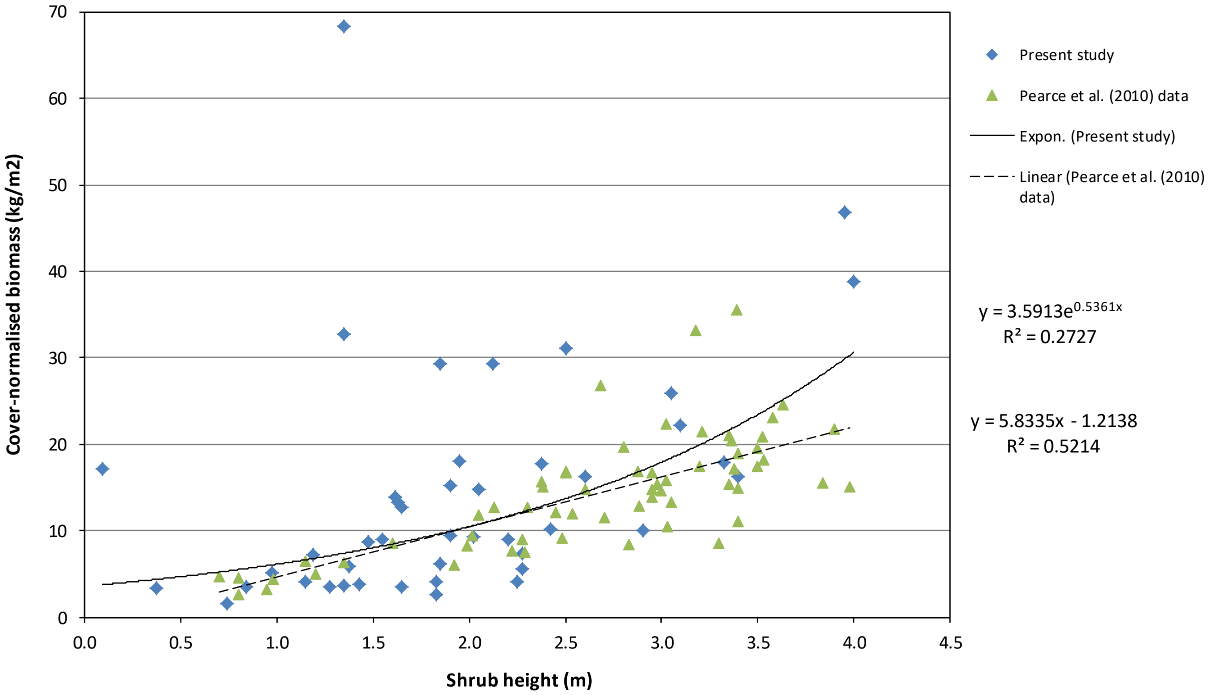


Figure 28: Gorse biomass normalised by shrub cover. Best-fit trend relationships are shown for the data from the present study, plus the data of Pearce et al. (2010).

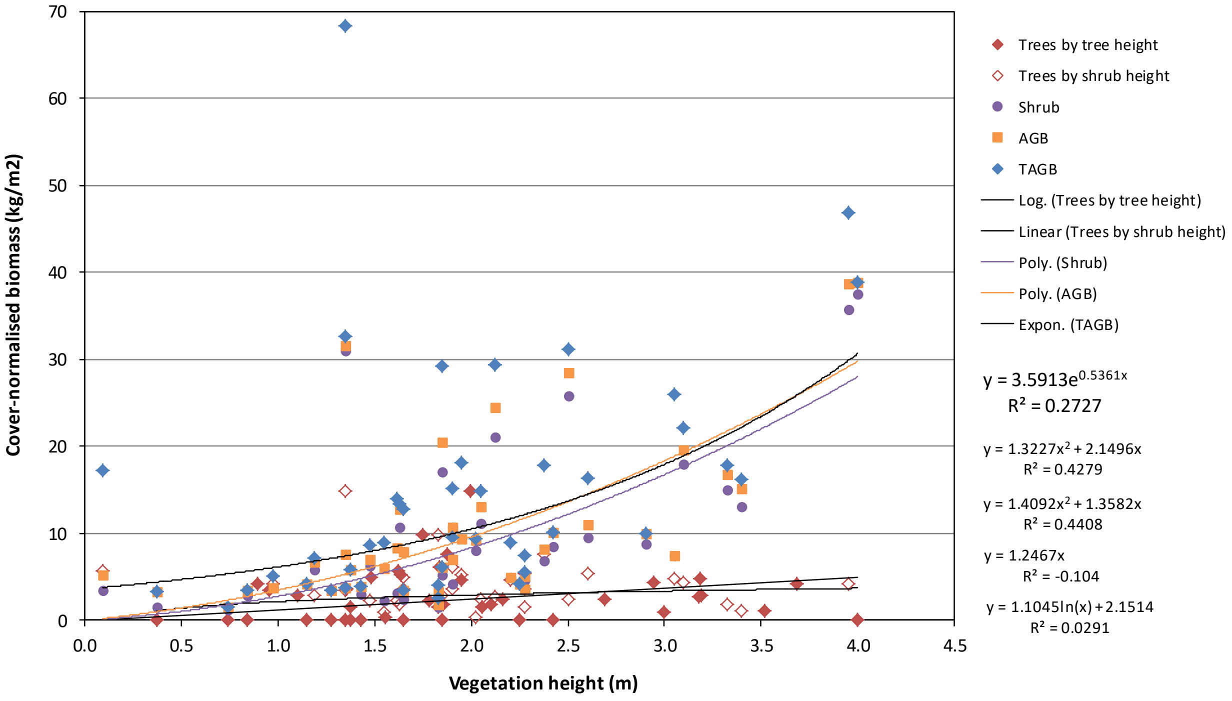


Figure 29: Biomass in gorse for various components normalised by cover. Biomass associated with trees present is shown plotted against both tree height (solid diamonds) and shrub height (open diamonds); all other components are plotted against shrub height.

However, the relatively simple models developed or tested here do provide a means for estimating shrub TAGB, and therefore carbon, either directly from shrub height or age, or from height via age to TAGB or vice versa. The new models developed do a better job of accounting for the presence of indigenous tree cover when estimating TAGB, and therefore offer some significant improvements over existing models. However, caution still needs to be taken when applying these new models, as they are based on sampling conducted at just a few (three) sites, and further validation is required using data from a wider range of shrub sites and environmental conditions.

As part of this validation and extension, it may also be possible to reanalyse old data from existing studies as one way of achieving this. This could include, for example, information on gorse from Lee et al. (1986), where good data are available on biomass as well as structural characteristics and stand age. Egunjobi (1969, 1971) might provide another suitable data source for gorse. In the case of broom, Williams (1983, 2011) might provide useful data, and there are likely to be other sources, including unpublished datasets, as well. As part of this, there is also a need to improve on models for estimating other biomass components, such as understorey species (e.g. bracken) and litter, as well as tree cover.

3.5 Management to enhance likelihood of gorse or broom becoming 'forest land'

In their practical guide to managing successions through shrub weeds Davis and Meurk (2001) suggest that gorse and broom can be facilitated to transition to tall forest, even in dry sites distant from seed sources (such as the eastern South Island), but that all domestic and wild grazing and browsing animals must be excluded and fire prevented. In addition enrichment planting may be required where seed sources are particularly scarce. They also recommend control of gorse or broom at treeline, as they consider its replacement by forest trees is unlikely. Successions through gorse may be diverse, with different successional trajectories across environments and from that occurring through kanuka, one of the species most likely to be replaced by gorse (Sullivan et al. 2007). Creating ‘forest land’ may therefore also necessitate additional enrichment planting, and podocarps could be used to provide maximum carbon gain (Hall 2001).

Introduced browsing mammals have a significant impact on the establishment of indigenous forest seedlings (Wilson et al. 2003). Gorse can protect seedlings to some extent from browsing hares and rabbits (e.g. Williams & Karl 2002) but consistent pest control does appear to be a key component for indigenous tree regeneration and for maximising its success (e.g. Wilson 1994). On, the other hand, domestic stock (sheep) can be used to keep pasture free of broom (e.g. Bellingham & Coomes 2003) if the desire is to prevent natural succession from establishing.

4 Conclusions

The most important predictor of ‘likelihood’ of gorse or broom transition to ‘forest land’ is gorse or broom age – transition becomes increasing likely as time elapses. Among the spatially explicit variables tested, the national probability of emergent indigenous trees was associated with higher slopes, increasing minimum temperature, mean annual rainfall, and mean October windspeed. In addition ‘optimum’ mean annual temperature is c. 13.5°C with a soil water deficit of c. 90 mm. The probability of transition decreases with increasing distance to indigenous seed source. Areas with a probability of < 0.5 are unlikely to have emergent forest species seedlings present.

For three measured sites in New Zealand covered with gorse with a high probability of achieving ‘forest land’ status, there was evidence of forest species presence within gorse within c. 6 years of gorse canopy establishment. Seedlings of forest species will not be visible above the gorse or broom canopy at this time. Broom passed through at least one ‘treeless’ cohort (c. 15 years maximum) before forest species established. Across all gorse and broom sites an additional 11–12 years are required until ‘forest land’ status is achieved, i.e. 30% crown cover. These figures were derived from selected16-m^2 plots with a relatively high density of trees present so hectare-scale qualification will likely take a few extra years.

In general, total above-ground biomass (TAGB) in gorse was found to be reasonably strongly correlated with both shrub height and age. On this basis, an existing allometric model for predicting TAGB from gorse height was compared with sampled estimates, and new models developed for predicting TAGB and gorse height based on gorse age. The existing Pearce et al. (2010) model for predicting TAGB from gorse height worked reasonably well in sampled gorse stands, but is potentially more suited to continuous-cover gorse with low or no indigenous tree cover. A better model for predicting gorse TAGB, including tree biomass where present, was produced using gorse height data for the three gorse sampling sites. This is considered better for taller/older gorse stands especially. A relatively good model was also produced relating gorse TAGB to gorse age. Here, we show that gorse of 30 years’ age is generally around 4 m height and contains approximately 125 t C/ha (= 458 t CO_2e/ha). In most instances the new models are based on a small number of samples from just a few sites, so further validation of these new models is required for a wider range of regenerating gorse sites and environmental conditions. In the short term, it may be possible to reanalyse old data from existing studies as one way of achieving this.

5 Recommendations

In order to maximise the probability of gorse or broom transitioning to 'forest land' status, we recommend:

- Minimal disturbance of the maturing gorse/broom canopy. Following disturbance these species will invariably re-establish from a long-lived seed bank (up to 80 years; Zouhar 2005) and are superior invaders of pasture than most indigenous species. Gorse, and broom to a lesser extent, are fire-adapted and will resprout from stem bases if adult plants are weakened or killed. Fire will destroy any established trees, therefore consolidating site dominance by gorse or broom.
- Reduced domestic or pest animal grazing or browsing within the gorse or broom. Indigenous broadleaved species that naturally establish in gorse or broom are more palatable than their nurse canopy and will be preferentially browsed. Indigenous broadleaved species need to become well established in small light gaps within the gorse or broom canopy so that when natural senescence, snow-damage or widespread insect attack (even deliberate biological control) occurs they can overtop and supplant the woody weed species.

Conversely, if the desired effect is to retain pasture and prevent achievement of 'forest land' status, then clearance of gorse or broom will need to be repeated to exhaust the germinating seed bank, supplemented with domestic grazing in order to reduce the success of post-disturbance germinants.

6 Acknowledgements

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Appendix 1. Correlations between measured variables for gorse

Correlations greater than 0.5 are shown in bold; cells with correlations greater than 0.7 are highlighted.

Gorse	Altitude	Aspect	Slope	CovCls	SAgeX	SAgeM	ShrHt	ShrCov	T1 Ht	T1 Cov	T2 Ht	T2 Cov	T3 Ht	T3 Cov	TotT Cov	US Ht	US Cov	Lit Dpt	Lit Cov	Gr Ht	Gr Cov	Shr bio	T1 bio	T2 bio	T3 bio	Tree Bio	US bio	Lit bio	Gr bio	TAGB
Altitude	1.00	-0	0.01	0.04	-0.30	-0.31	-0.35	-0.18	-0.25	-0.12	0.30	0.29	0.17	0.19	0.03	0.08	0.05	**-0.85**	**-0.72**	-0.39	-0.23	-0.49	0.20	0.22	0.19	0.25	0.16	**-0.80**	-0.34	-0.46
Aspect		1.00	-0.14	-0.03	-0.12	-0.17	-0.05	-0.10	-0.08	-0.15	0.31	0.11	0.16	0.03	-0.14	0.10	0.09	-0.11	0.08	-0.12	0.04	-0.10	-0.18	0.10	0.16	-0.13	0.01	0.01	-0.05	-0.13
Slope			1.00	-0.19	0.19	0.19	0.12	0.23	-0.18	-0.01	-0.12	0.10	-0.11	0.17	0.01	-0.18	-0.17	-0.04	-0.19	-0.05	0.01	0.30	0.11	-0.01	-0.02	0.10	0.03	-0.09	-0.01	0.30
Cover Class				1.00	0.42	0.41	0.09	**-0.63**	**0.76**	**0.76**	0.35	0.21	0.40	0.23	**0.78**	0.09	0.18	0.06	0.21	-0.40	-0.41	-0.08	**0.57**	0.29	0.34	**0.62**	0.23	0.19	-0.44	0.11
Shrub Age (mean)					1.00	**0.99**	**0.62**	-0.12	**0.58**	**0.55**	0.15	0.13	0.01	0.00	0.49	-0.17	-0.16	0.30	0.39	-0.45	**-0.51**	**0.65**	0.35	0.18	0.00	0.36	-0.02	0.45	**-0.52**	**0.75**
Shrub Age (median)						1.00	**0.59**	-0.09	**0.57**	**0.53**	0.13	0.12	0.03	0.00	0.46	-0.17	-0.17	0.30	0.37	-0.45	**-0.51**	**0.64**	0.36	0.17	0.01	0.37	-0.03	0.44	**-0.52**	**0.73**
Shrub Ht							1.00	0.10	0.38	0.34	0.03	0.13	-0.23	-0.13	0.26	-0.12	-0.04	0.39	0.48	-0.25	-0.40	**0.68**	0.07	0.19	-0.16	0.09	0.09	**0.53**	-0.35	**0.70**
Shrub Cov								1.00	**-0.53**	**-0.52**	-0.32	-0.19	-0.41	-0.29	**-0.58**	-0.26	-0.24	0.23	-0.10	0.30	0.05	0.28	-0.44	-0.14	-0.34	-0.47	-0.10	0.02	0.19	0.13
Tree1 Ht									1.00	**0.87**	0.22	0.07	0.15	-0.01	**0.79**	-0.02	0.19	0.33	0.46	-0.35	-0.38	0.11	**0.54**	0.14	0.06	**0.54**	0.11	0.49	-0.39	0.29
Tree1 Cov										1.00	0.18	0.13	0.07	0.11	**0.95**	-0.10	0.06	0.16	0.22	-0.34	-0.36	0.09	**0.72**	0.20	0.07	**0.72**	0.15	0.29	-0.37	0.31
Tree2 Ht											1.00	**0.76**	**0.53**	0.08	0.34	-0.07	-0.07	-0.30	-0.09	-0.22	-0.22	-0.24	0.13	0.69	0.33	0.30	-0.04	-0.21	-0.24	-0.16
Tree2 Cov												1.00	0.14	0.13	0.37	0.00	-0.01	-0.28	-0.23	-0.17	-0.18	-0.15	0.13	**0.84**	0.09	0.31	0.10	-0.26	-0.19	-0.08
Tree3 Ht													1.00	**0.51**	0.21	-0.07	-0.08	-0.15	0.10	-0.12	-0.12	-0.20	0.05	0.06	**0.81**	0.13	-0.05	-0.03	-0.12	-0.16
Tree3 Cov														1.00	0.34	-0.04	-0.06	-0.14	-0.15	-0.07	-0.07	-0.08	0.14	-0.05	**0.81**	0.18	0.02	-0.14	-0.07	-0.04
Total Tree Cov															1.00	-0.10	0.05	0.03	0.09	-0.34	-0.36	0.00	**0.71**	0.35	0.25	**0.75**	0.14	0.13	-0.38	0.22
Understorey Ht																1.00	**0.87**	-0.10	0.01	-0.11	-0.11	-0.16	-0.17	-0.12	-0.02	-0.19	0.65	-0.08	-0.12	-0.20
Understorey Cov																	1.00	-0.08	0.06	-0.09	-0.09	-0.14	-0.12	-0.09	-0.06	-0.14	**0.92**	-0.03	-0.10	-0.17
Litter Depth																		1.00	**0.78**	0.26	0.02	0.47	-0.15	-0.23	-0.15	-0.20	-0.15	**0.89**	0.16	0.46
Litter Cov																			1.00	-0.05	-0.22	0.45	-0.16	-0.20	-0.04	-0.19	-0.15	**0.96**	-0.15	0.44
Grass Ht																				1.00	**0.71**	-0.12	-0.24	-0.15	-0.10	-0.26	-0.08	-0.02	**0.92**	-0.17
Grass Cov																					1.00	-0.17	-0.25	-0.15	-0.10	-0.27	-0.08	-0.18	**0.93**	-0.24
Shrub bio																						1.00	-0.05	-0.09	-0.14	-0.07	-0.04	**0.54**	-0.16	**0.96**
Tree1 bio																							1.00	0.20	0.03	**0.98**	0.09	-0.12	-0.26	0.22
Tree2 bio																								1.00	-0.05	0.40	0.14	-0.22	-0.16	0.01
Tree3 bio																									1.00	0.10	-0.02	-0.10	-0.11	-0.12
Total Tree bio																										1.00	0.11	-0.16	-0.28	0.20
Understorey bio																											1.00	-0.16	-0.09	0.00
Litter bio																												1.00	-0.10	**0.54**
Grass bio																													1.00	-0.22
Total AGB																														1.00