Post-Quake Farming Project: Native Forestry Report

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Report Prepared for the Post-Quake Farming Project

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Cover photograph:

An example of regenerating native vegetation in the Post-Quake Farming Project Area.

EXECUTIVE SUMMARY

The Post-Quake Farming (PQF) project covered an area of 420 500 ha of hill country which was land struck by the 2016 Kaikoura earthquake. The PQF native forestry workstream investigated the nature and issues relevant to native forestry in the project area. Approximately 22% (91 650 ha) of the project area featured native forest or scrubland. The majority of these vegetation communities were secondary, having regenerated following much earlier primary forest clearance. A range of exotic vegetation types present (e.g., tree lucerne, radiata pine, exotic broom and gorse) also presented options for native forest establishment.

Possible benefits of native forestry for people in the project area were many, including income streams through the New Zealand Emissions Trading Scheme, continuous cover forestry, honeybee forage, or cultural products. A range of climate, soil, water and social ecosystem services would also be provided by the native vegetation present. Further, New Zealand's once-forested heritage means that terrestrial ecosystems have been severely impacted by forest loss and the forests that do remain hold heightened importance for supporting biodiversity.

Key issues identified were dispersal limitation arising from local/functional extinction of old-growth forest species across most of the project area and excessive herbivory by domestic stock, but more so, by feral herbivores (i.e., deer, goats, possums, also pigs which are omnivores). The existing secondary forests were missing old-growth species meaning their restoration potential is limited with regard to composition, structure and function – and specifically in terms of biodiversity and ecosystem services outcomes (e.g., long-lived, high volume tree species are missing).

Enrichment planting demonstration trials were established on nine farms in a variety of native and exotic vegetation types. Old-growth species of low palatability were planted into the shelter of existing vegetation cover to direct forest composition and structure towards that found in forests during pre-human times. Forest composition and structure were surveyed and it was found that those stands protected from domestic herbivores featured twice the number of woody species and a different community structure compared to unfenced forests. However, even forests fenced from domestic herbivores showed signs of severe damage by feral herbivores for which stock fencing presented no barrier.

Enhanced forest management is needed across the project area if the vegetation communities are to achieve truly permanent status with a composition, structure and function representative of healthy and intact forest communities. Enhanced forest management would involve mimicking disturbance (e.g., canopy interventions) and

reintroducing locally/functionally extinct forest species (enrichment planting of native conifer and angiosperm species representative of pre-human forests), reducing herbivore browse to levels where forest regeneration and succession can occur, and provision of funding and support for both existing and new forests at levels adequate to enable forest restoration.

Significant opportunities were identified across the project area for management of naturally occurring regeneration (i.e., passive restoration, where threats and shortcomings of forest ecosystems are managed, rather than focusing on intensive and costly tree planting) to restore forest cover at landscape scales. Addressing herbivory across the remote hill country of the project area would be best approached at landscape scales through landowner collaborations. Management of feral herbivore populations following approaches such as ongoing and professionally executed aerial or ground basing hunting (preferably generating an income stream) would be more feasible than attempting to protect forests with (deer) fencing which would be prohibitively expensive and difficult to maintain in this landscape at the scales required.

Overall, enhanced forest management needs to follow an ecosystem approach where lost propagules are reintroduced into suitable microclimatic conditions and threats such as excessive herbivory are comprehensively addressed. Other threats such as invasive vines and shade tolerant weeds may also form part of enhanced forest management. The native forest estate of the PQF area presents major opportunities for restoration of biodiversity and ecosystem services from which landowners, wider society, and nature would benefit.

The issues identified in this project area also relate to much of Aotearoa/New Zealand's lowland area and enhanced forest management, as described here, will be central to addressing our ongoing biodiversity and emerging climate change crises. Avenues of funding and support should be structured to deliver enhanced forest management of existing and new forests and recommendations are made accordingly in this report.

1.0 INTRODUCTION

1.1 The Post-Quake Farming Project

1.1.1 About the Post-Quake Farming Project

The Post-Quake Farming Project (the PQF project) is a work programme to assist the hill and high-country farming community to recover from the November 2016 earthquake event. It was designed to recognise the skills and understanding that farmers have to manage land and make complex land use decisions, and supported extension and actions to develop better information on the potential of the land resource.

In the scoping phase, it became apparent that a key stakeholder requirement related to the carbon and restoration potential of the native forest resource and reported lack of support for tools or resources to facilitate improved results for the environmental, economic, and social values of native forests.

The hill and high-country land in the project area features rough topography and is therefore challenging to manage, which means farming is of a low intensity and individual farm units are often large (i.e., many farms exceed 2 000 ha in area). Secondary native vegetation[1] is a significant feature of both grazed and un-grazed areas and covers a large proportion of public and private land (i.e., c. 91 650 ha; Belton, 2019). Despite this opportunity, there is limited support to undertake improved management to sequester carbon or advance other forest restoration goals. The project therefore commissioned a programme of work, alongside other workstreams, to better understand this opportunity and develop information to inform future decisions.

1.2 The Purpose of this Report

The purpose of this report is to describe work undertaken by the project to understand the restoration potential of the native forest resource in the project area and to recommend changes to afforestation incentives.

The report covers:

- Past and current state of native forest in the project area.
- Benefits of native forests.

[1] Secondary vegetation is vegetation that has reassembled following clearance of the original old-growth vegetation cover.

- Trials to pilot the method of enrichment planting across a range of environments and situations.
- Botanical assessments to investigate the implications for herbivory within the project area.
- Recommended improvements to the One Billion Trees Programme, which is the main afforestation incentive scheme available in New Zealand at the time of writing.

Work undertaken to explore the potential of the native forest resource includes assessment of the resource and its potential (mapping, enrichment planting trials and botanical investigations), as well as policy recommendations.

This report should be read along with the companion reports: Belton (2019) – which is a desktop assessment of the native forest resource contrasted with experience on the ground; and (Belton, in prep) – which provides specific recommendations to better recognise or reflect the carbon sequestration potential of the native forest resource.

The intended audience are individuals, academics, industry groups, and government organisations looking to better understand how they might support restoration and carbon sequestration outcomes for areas of existing or regenerating native forest.

2.0 STATE OF THE NATIVE FORESTRY RESOURCE

2.1 Vegetation, Past and Current

Prior to humans' arrival, the mild to cool, semi-arid lowland environments (Singers & Rogers, 2014) of the PQF project area supported diverse forests comprising dry conifer and conifer-broadleaved forests with pockets of beech forest (Leathwick, 2012). Widespread scrubland was not a feature of this pre-human vegetation cover (McWethy et al., 2010). The majority of native forest cover was eliminated by humans-lit fires (initially by Polynesians, followed by more intensive burning by Europeans; Fig. 1A). Following burning, at sites of low to middle elevations[2] and those with dry climates[3], typical of much of the PQF project area, there was little recovery of the pre-existing closed-canopy forest (McWethy et al., 2010).

While old-growth[4] forest remnants have been largely eliminated or are otherwise spatially scarce, with reduced forest heath and functionality (Forbes et al., 2020), large areas of secondary native vegetation has assembled on hill country (e.g., Fig. 1B) which presents a number of important restoration opportunities. Our investigation of these limitations and opportunities is described in this report and accompanying documents.

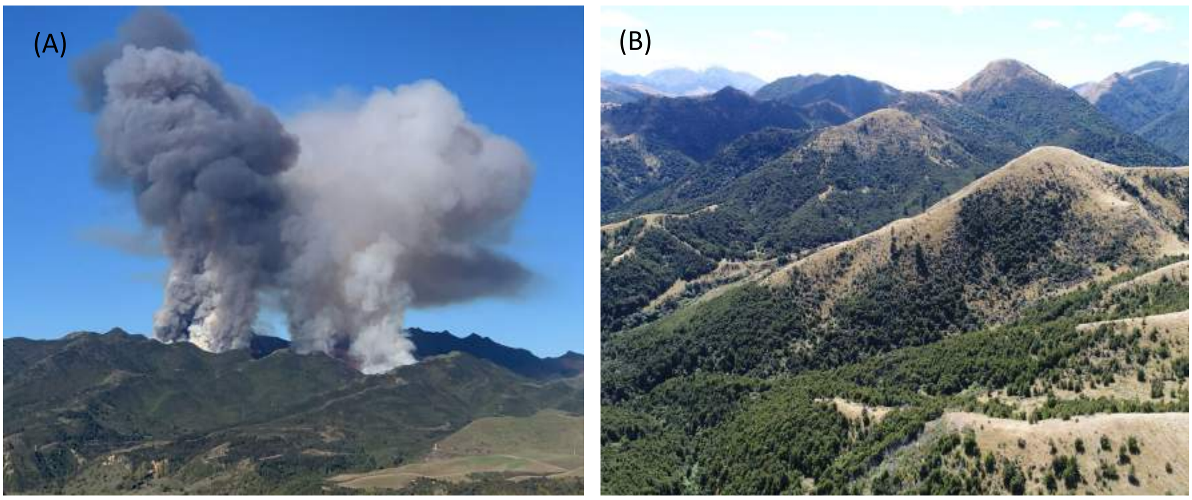

Figure 1(A&B). (A) Contemporary land clearance of exotic weeds by fire, reminiscent of past burns and (B) regenerating native vegetation on hill country in the PQF project area.

[2] c. 0-600 m a.s.l.
[3] c. rainfall of <1 600 mm year^{-1}.
[4] Old-growth refers to forest that have been growing for a very long time. In the New Zealand context old-growth stands often pre-date the time of European arrival. Old-growth forests usually have canopy compositions representative of pre-human times, although components may today be missing.

Today the project area features approximately 22% (91 650 ha) cover by native forest and scrubland (Belton, 2019) and much of this vegetation is embedded within pastoral farming systems. A similar distribution of native cover occurs at a national scale, with 24.5% (2.8 M ha) of native vegetation and 17% (1.4 M ha) of native forest estimated to be on Aotearoa/New Zealand's sheep and beef farms (Pannell et al., 2021). The integration of native forestry with working farms presents a range of important opportunities, most principally for both native forestry and farming (Norton et al., 2020).

2.2 Benefits of Native Forests

Economic, environmental, and social/cultural benefits can be obtained when native forest is retained and incorporated into farming landscapes. Examples of these benefits include:

- The New Zealand Emissions Trading Scheme (ETS) provides a potential long-term income stream in return for sequestration of atmospheric carbon in forest biomass.
- Where specific tree species occur, or can be introduced to regenerating forests, those trees present options for sustainable timber harvest following continuous cover forestry techniques (e.g., Barton, 2008).
- Many native tree species provide excellent honeybee forage, as both pollen and nectar sources (McPherson & Newstrom-Lloyd, 2019).
- Control of various ecologically harmful weed and pest species to support economic use. Examples include the nationwide programme to control possums which are a TB-vectors, feral pigs and ungulates which farmers often control because they damage or consume forage species, and wasps which predate bees.
- Avoid costs where control is exerted to prevent ongoing native tree species (such as kānuka) establishing in exotic pasture. Retirement and management of such land areas to assist reversion to native forest is beneficial as the costs of control (e.g., herbicide application) are no longer required.

Native forests provide a variety of ecosystem services, thereby yielding environment and social benefits (Ausseil et al., 2013; van den Belt & Blake, 2014; Brockerhoff et al., 2017; Maseyk et al., 2017). Recognised services include:

- Climate regulation,
- Control of soil erosion,
- Regulating water flows,
- Provision of clean water,
- Provision of natural habitats,
- Cultural heritage,
- Provision of taonga (treasured) species for whakairo (carving) and rongoā (medicine),

- Stock shelter,
- Recreation and ecotourism,
- Aesthetics and inspiration,
- Landowner wellbeing,
- Education,
- Sense of place,
- Soil formation,
- Nutrient cycling.

Furthermore, many aspects of Aotearoa/New Zealand’s native biodiversity are contingent on maintenance and enhancement of natural forest cover and therefore achieving functional levels of native cover amongst farmland is a cornerstone of conserving native biodiversity (Cieraad et al., 2015) in these areas and across the country as a whole.

3.0 KEY ISSUES FOR NATIVE FORESTRY IN THE PQF PROJECT AREA

3.1 Ecological Isolation and Dispersal Limitation

Due to past forest clearance in the PQF project area, old-growth forest remnants are today scarce, and this means forest tree seed sources are equally scarce. Infrequent or low densities of long-distance (landscape scale) dispersal of forest tree seeds means that the probability of dispersal decreases rapidly with increasing distance from seed source. For example, the predicted dispersal probability for the common forest tree fivefinger[5] by the forest pigeon, kererū[6], suggests that only 12% of seeds would be dispersed further than 100 m and only 0.39% of seeds further than 1 km from the parent tree (Fig. 2A; Wotton & Kelly, 2012). Seed rain estimates for rimu[7], silver, and mountain beech[8] indicate peak seed rain occurring within 3-6 m of the parent tree, and for thin-barked totara[9] an ongoing decline from <1 m from the parent tree (Fig. 2B; Canham et al., 2014).

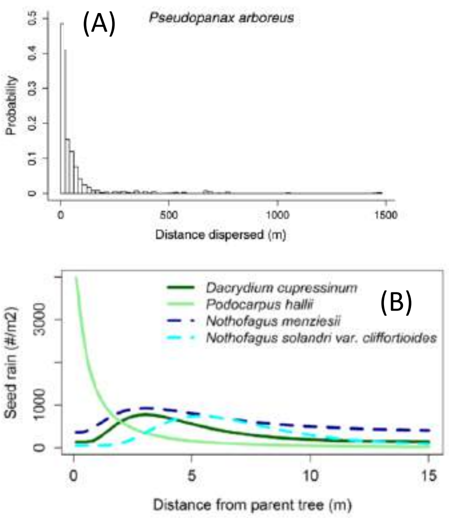

Figure 2(A & B). (A) Predicted dispersal probability for fivefinger. (B) Predicted seed rain for rimu, thin-barked tōtara, silver and mountain beech. Sources (A) Wotton and Kelly (2012), and Canham et al. (2014) respectively.

The absence of those species which represent intact mature natural forest limits the potential of forest succession (Fig. 3A). Old-growth tree species bring traits of high biomass, large stature, large fruit size, and in time, high levels of habitat complexity; tree holes for roosting, host opportunities for epiphyte communities (Weiher et al., 1999; Fig. 3B & C; Fig. 4A). Therefore, those secondary forests which are missing old-growth tree species (whether they have been eliminated or their distribution is strongly aggregated, for instance to gullies Fig. 4B & C) are limited in their ability to succeed to more advanced forest phases, and attributes of these secondary forests such as biomass (and carbon sequestration), biodiversity and

[5] *Pseudopanax arboreus.*
[6] *Hemiphaga novaeseelandiae.*
[7] *Dacrydium cupressinum.*
[8] *Lophozonia menziesii and Fuscospora cliffortioides.*
[9] *Podocarpus laetus.*

habitat will be profoundly limited (Forbes et al., 2020; see Appendix A). These limitations are particularly problematic where secondary forests are needed to support biodiversity and sequestration of atmospheric carbon through the accumulation of forest biomass, such is the case in Aotearoa/New Zealand and also in most parts of the developed world that would be naturally forested.

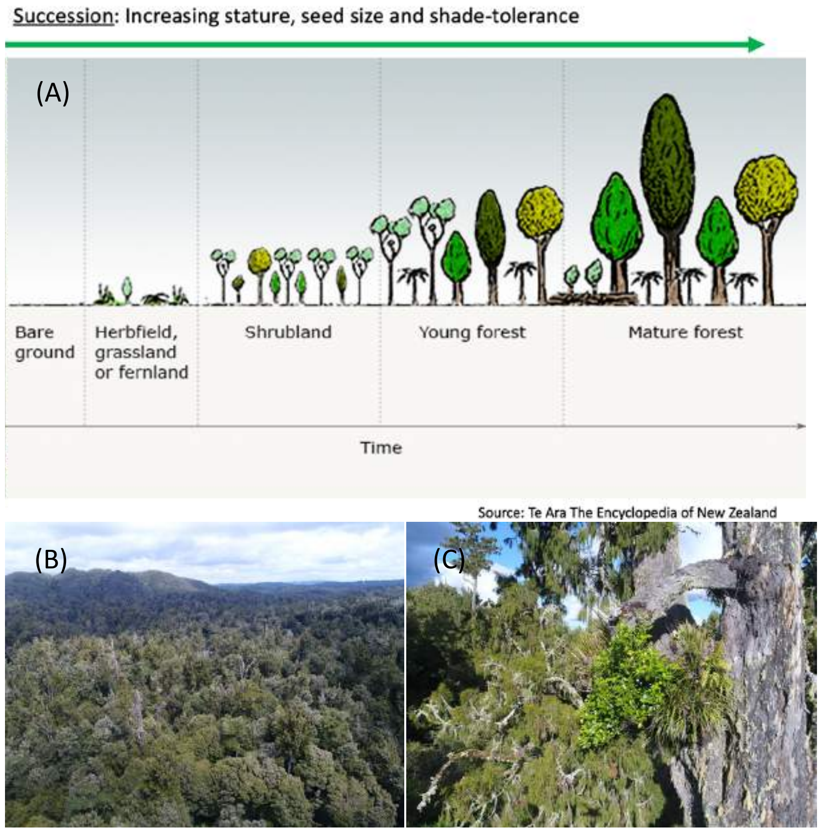

Figure 3(A–C). (A) Illustration of forest succession. (B & C) Landscape featuring old-growth forest as would have occurred prior to humans' arrival in Aotearoa/New Zealand.

3.1.1 Enrichment planting

An emerging restoration action, enrichment planting, is the planting of desirable species (in this case, old-growth species) into secondary, exotic or degraded forest to overcome the limitations of ecological isolation and dispersal limitation.

The seedlings of old-growth forest tree species have specific microclimate requirements (i.e., they need some shelter) and this means their planting needs to occur into the shelter of existing vegetation cover (e.g., Fig. 4D–F; Tulod & Norton, 2020). This approach mimics the shelter provided by a forest, where old-growth species would establish naturally. A challenge with planting seedlings into existing cover is to ensure levels of competition between the existing vegetation and the planted seedlings are sufficient to provide shelter but not too great that planted seedling growth rates are reduced.

It is also important to observe ecological principles, such as selecting species for planting that would be found naturally in local forests, and that species are used that represent advanced stages of forest development (meaning traits such as shade tolerance are well matched to the shaded planting site), and seedlings raised are from seeds collected close to where the planting will occur[10]. Planting seedlings that are of an advanced grade (e.g., >60 cm tall at planting; >18-24 months old) is important for enrichment planting, as the seedlings are planted into existing vegetation where taller seedlings will be more competitive and emerge from the semi-sheltered environment sooner. Species choice should also observe the levels of feral browsers present in the area and species of lower palatability can be chosen to address the risk of browsing.

Figure 4 (A–F). (A) An old-growth forest remnant in the PQF project area and (B) isolated old-growth species tōtara and mataī having survived in a gully position and now surrounded by secondary forest. (C) Secondary thinned-barked tōtara regenerating in gullies following clearance and (D) secondary forest comprising mahoe and kānuka regenerating from a cover of exotic gorse. (E) Kahikatea emerging from secondary broadleaved forest in Tairawhiti provides an example of what successful enrichment planting would look like and (F) extensive secondary forests of almost pure kānuka in the PQF area are prime examples of degraded forests that would benefit from enrichment planting.

[10] This principle is known as ecosourcing.

3.1.2 Enrichment planting in the PQF project area

As ecological isolation and dispersal limitation are significant native forestry issues for the PQF project area. For this reason, the project funded demonstration enrichment planting projects across nine farms in north Canterbury and south-eastern Marlborough (Fig. 5; Appendix B). The purpose of this was to show how enrichment planting can be applied in practice in an area subject to significant ecological isolation and dispersal limitations for forest regeneration. Enrichment planting sites featured a range of existing vegetation types, including mānuka and kānuka (Fig. 6A) forest and scrub, native broadleaved scrub, radiata pine, tree lucerne (Fig. 6 B & C), exotic broom and gorse, and small-leaved shrubland. Species were selected which represented pre-human mature forest compositions and, due to the current population sizes of feral browsers present across this area of Aotearoa/New Zealand, the species chosen for planting were those recognised as being avoided by ungulates (Table 1; Forsyth et al., 2002).

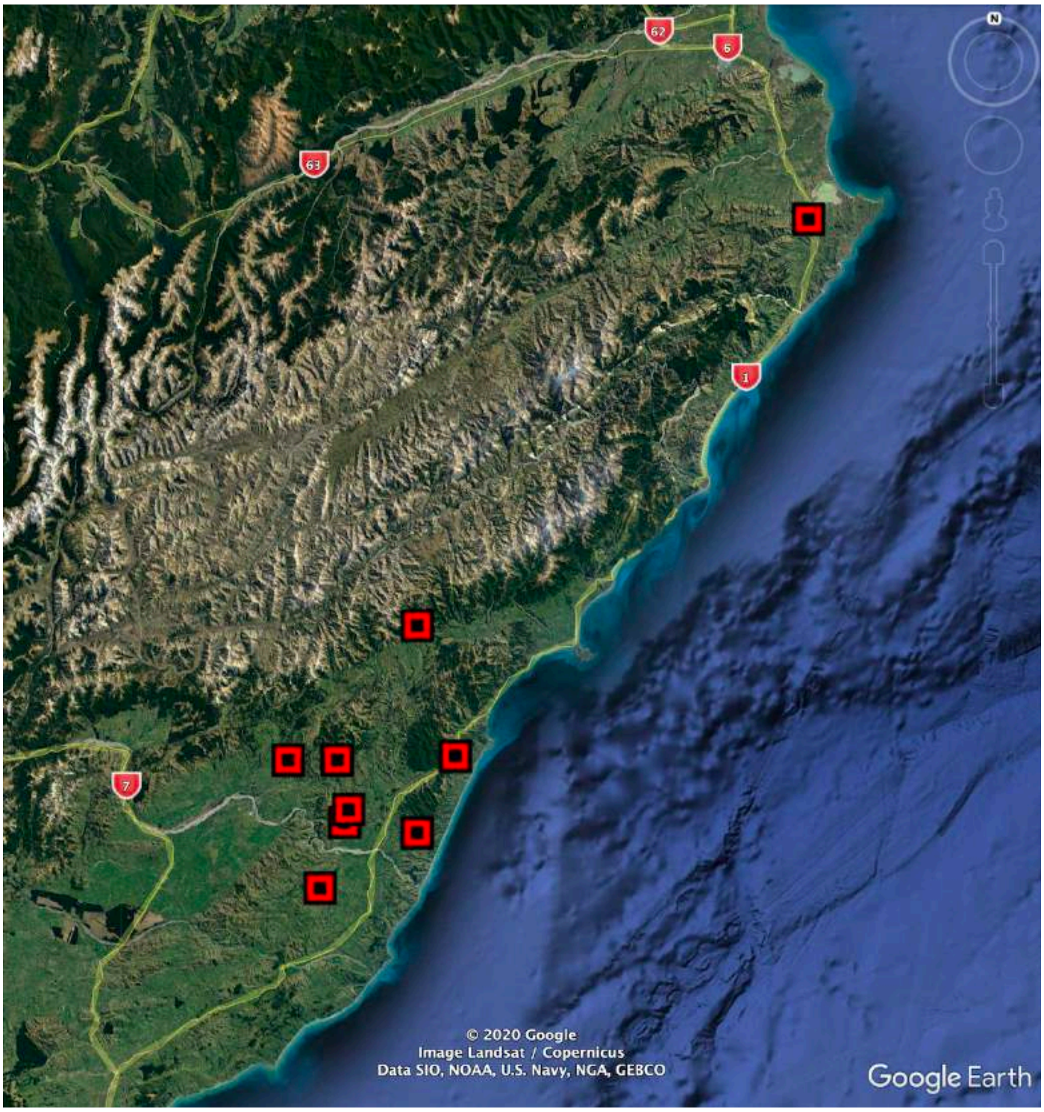

Figure 5. Locations (denoted by red squares) of the nine enrichment planting demonstration sites within the PQF project area.

Figure 6 (A–C). (A) Kānuka forest enriched on the Hundalee hill country, north Canterbury. (B & C) Enrichment planting configured as an experiment comparing seedlings planted into exotic pasture (B) versus the shelter of mature tree lucerne (C), with the experiment running over the 2020/2021 summer.

Table 1. Species chosen for inclusion in the PQF enrichment planting demonstration project

Scientific name	Common/Maori name	Palatability class[11]
Dacrydium cupressinum	Rimu	Avoided
Fuscospora fusca	Red beech	Avoided
Fuscospora solandri	Black beech	Avoided
Myoporum laetum	Ngaio	Avoided
Olearia paniculata	Golden akeake/ Akiraho	Not classified
Pittosporum eugenioides	Lemonwood/ Tarata	Avoided
Podocarpus laetus	Thin-barked tōtara/ totara-kiri-kotukutuku	Not selected
Podocarpus totara	Lowland totara	Avoided
Sophora microphylla	Small-leaved kowhai	Not selected

Notes: Palatability classes follow Forsyth et al. (2002). Classes are defined as: Avoided, Not Selected, or Preferred. No classification is available for Golden akeake (*O. paniculata*). When considering the palatability classes, it should be considered that ungulates will consume species classed as Avoided, but consumption is less than expected based on availability (Forsyth et al., 2002).

3.2 Herbivory by Introduced Domestic and Feral Herbivores

3.2.1 Herbivore populations and management

Since the latter stages of humans' arriving in Aotearoa/New Zealand, a range of mammalian species have been introduced and these today form significant domestic and feral populations (hereafter domestic or feral herbivores). Introduced herbivores can significantly alter forest community composition and structure by reducing the abundance of palatable species and promoting non-palatable species (Wilson et al., 2006; Wardle et al., 2001). Feral herbivores can, in addition, compete with or place domestic livestock at risk of disease and they can also damage other aspects of primary production (e.g., horticultural & sylvicultural systems; Latham et al., 2020). Although for a period around the 1980s national feral deer populations declined due to the effect of commercial hunting, deer numbers were determined to be increasing in the 2000s (Forsyth et al., 2011) and anecdotal evidence from interactions with farmers across mainland New Zealand suggests feral deer numbers are gradually increasing as of 2019/2020 (Adam Forbes, Personal Observation).

While herds of domestic herbivores tend to be well controlled through fencing, populations of feral herbivores such as possum, deer, goat and pig are subject to differing levels of control. The home ranges of the more mobile species can be large, meaning that population

[11] Following Forsyth et al. (2002).

management should be expected to cross property boundaries. For instance, red deer (*Cervus elaphus*) can range 100-2 074 ha and up to 11 000 ha (Nugent et al., 2001).

Due to their slow-growing nature, the recovery of our temperate forest ecosystems following herbivore control typically takes decades (Fig 7A; Tanentzap et al., 2009) and the recovery of floristic composition and structure is recognised to require an ecosystem approach to management rather than being achieved by just simply reducing herbivore abundance (Coomes et al., 2002; Wright et al., 2012). In this context, ecosystem management could include interventions such as mimicking disturbance[12] to optimise competitive interactions, re-introducing lost propagules (enrichment planting), or managing other pests such as invasive vines or shade tolerant weeds which may inhibit forest regeneration.

While fencing standards exist for feral herbivores, such as deer (Fig. 7B) or goats (Fig. 7C), fencing to protect forests from feral herbivores at large scales or on steep or difficult topography (Fig. 7D) is often logistically and economically unviable. Installation costs of >$30/m plus earthworks for tracks and fence lines puts have been reported by farmers in the PQF area, and cost of maintenance, essential to ongoing functionality, is also a significant factor. In addition to the barriers to installing the fence, ongoing maintenance is essential to effective fencing. Fences near forests are susceptible to damage from tree fall (Fig. 7E), fencing may be overgrown by pest plants such as blackberry allowing animals to climb, and over time fences lose their structural integrity, this can occur within several years where animals such as goats are pushing against and loosening stables and wires, soon rendering the relatively new fence ineffective. Even when built to standard, the configuration of fencing can lead to weak spots where spooked animals are concentrated/funnelled into and will eventually find their way through, or over, out of desperation (Adam Forbes, Personal Observation).

With fencing out of reach as a practical and cost-effective option to defend native forest from feral herbivores at large scales, or on difficult topography, the only viable approach is to actively manage feral animal populations. A range of non-fencing methods for feral mammal control exist, with the main options being poisoning, trapping (including capture and removal), ground-based shooting (professional or recreational, with or without dogs), aerial shooting, Judas animals, fertility control, mustering, and commercial harvest. Population management by its very nature needs to be carried out at landscape scales. Suitably resourced cooperative action at a community level therefore presents

[12] For instance, by creating canopy gaps (Forbes et al., 2016; Tulod & Norton, 2020).

opportunities for forest restoration at large scales which are practically unattainable through fencing alone.

Figure 7 (A–F). (A) Example of a recently retired native conifer forest where feral goats and domestic herbivores have been excluded to allow the forest understorey to regenerate so that tree species can be recruited to the canopy in due course, thus sustaining the forest in decades to come. (B) Example of a deer fence in the PQF project area installed to protect native forest. (C) Example of a goat proof fence installed to protect native forest in northern Hawke's Bay. (D) Example of steep hill country backed by extensive mountainous wildness area where deer fencing is impractical and managing herbivore populations across boundaries is a more achievable (yet still demanding) approach to addressing the effects of herbivory on forest health. (E) Example of a plantation pine tree toppling onto a remotely located fence that is in place to protect native forest, breaching the fence until the

breach is discovered and fixed. (F) Example of forest canopy collapse without canopy recruitment due to high numbers of deer and goats in hill country south of Nelson City.

Several examples exist in the PQF project area where neighbouring landowners have together commissioned aerial hunting operations which have been cost-positive due to the commercial meat salvage and sale, alongside reduced feed competition with livestock. This approach is beneficial in that the control is executed at landscape scales and at very little cost or risk to the landowner. Despite this, sustained, professionally led, and strategic approaches to guide control operations based on current and emerging best practice are needed with a focus on outcomes rather than animal population numbers per se (Goldson et al., 2014).

3.2.2 The important social dimension

Most feral herbivores are viewed collectively as both a pest and a resource (Hughey & Hickling, 2006). Hunting has recreational, economic and social benefits and maintaining feral mammal populations is desirable from these viewpoints. Proposals to control feral animals can conflict with public preferences and create strong negative perceptions and controversy (e.g., the relationship between red deer and New Zealanders, Figgins & Holland (2012); the 1080 debate, Parliamentary Commissioner for the Environment (2013)). Thus, the topic of feral mammal control is one with the potential to either unite or divide communities and is therefore an issue that requires careful investigation and engagement. A balanced and well-reasoned approach is required. Unless people are in agreement over types and levels of control, there will be ongoing discord and inefficiency in achieving desirable outcomes for both forests and people.

3.2.3 Herbivores and the carbon cycle

In addition to achieving sustainable management of native forests, feral herbivore populations raise implications for climate change management in relation to our native forest estate, as they can have significant (albeit context specific) effects on the forest carbon cycle (Holdaway et al., 2012). In addition, herbivore population control brings opportunities for methane avoidance as some species also contribute methane emissions.

In addition to direct consumption of biomass, feral herbivores selectively remove palatable species resulting in demographic changes in forests and a lack of recruitment to the forest canopy (Fig. 7F). Canopy trees can be killed through direct browsing or ringbarking, affecting the forest composition and structure. In addition, feral herbivores can act as both seed predators and dispersers (e.g., pigs spreading mataī fruit: O'Connor & Kelly, 2012). Greatest positive effects of feral mammal control are found in localised areas of highly palatable early-successional vegetation where animal numbers are high and where control results in rapid development of woody vegetation (Holdaway et al., 2012).

Where browsing prevents adequate levels of regeneration and succession, forests will gradually breakdown and collapse, and in the process release stored carbon back into the atmosphere (Fig. 7F). For this reason, feral mammal control is important not only for achieving carbon sequestration but also maintaining native forest carbon reservoirs.

Ruminant feral herbivores (e.g., deer & goats; but not marsupials/possums or pigs) emit methane from ingested carbon. Where population sizes are high, methane emissions can be reduced through control. For example, reducing and maintaining red deer numbers from 30–50 deer km^{-2} to 3–4 deer km^{-2} (a reduction of 26–47 deer km^{-2}) would result in methane savings of 20.8–37.6 Mg CO_2e km^{-2} $year^{-1}$ (Holdaway et al., 2012).

3.2.4 Herbivory and associated management needs in the PQF project area

To investigate the effects on forest composition and structure from differing levels of mammal access, we surveyed 18 10 × 10 m vegetation plots using the RECCE method (Hurst & Allen, 2007), in part. Plots were located randomly into forest protected (e.g., Fig. 8A) and unprotected (e.g., Fig. 8B) from domestic herbivores. Neither forest was protected from feral ungulates. Plots were on face landforms over an elevation range of 76–187 m above mean sea level, on two farms in the southern part of the project area.

A total of 25 woody species were surveyed across all plots (see Fig. 9 for species names), 24 (96% of all species) species in retired and 11 (44% of all species) in non-retired forest. In forests fenced/retired from domestic herbivores, woody species with meaningful levels of cover (Importance Value[13] (IV) >15) were evenly split in levels of cover between species that are preferred by ungulates (combined IV 163) and those species that are either not selected or preferred (combined IV 167). In contrast, of the species making up meaningful levels of cover in non-retired forests, only one species preferred by ungulates was present (i.e., fivefinger, IV 14), the remaining species all being not selected or avoided in the diets of ungulates (combined IV 123; Fig. 9A).

Analysis of tree and shrub community composition (i.e., mMDS ordination analysis) between retired and non-retired forests showed two distinctly separate community compositions (Fig. 9B) indicating that fencing is a method of enhancing forest community composition.

No sites were protected from feral ungulates and even the retired site showed signs of deer presence (e.g., Fig. 8C).

[13] Importance Value (IV) are the summed cover class scores across all forest tiers as measured in the vegetation survey plots. IV therefore represents a measure of cover with greater weighting given to vegetation occurring in higher elevation tiers.

Figure 8 (A–C). (A) Grazed forest and (B) forest protected from domestic herbivores. Neither (A) or (B) represent feral herbivore control. (C) Bark stripping of fivefinger by deer in the forest protected from domestic stock.

Table 2. Importance values of woody species in retired and not-retired forests of the PQF study area.

Retired			Not retired		
Palatability class	Species	IV	Palatability class	Species	IV
Preferred	PSEARB	98	Avoided	KUNROB	64
Avoided	LEUFAS	62	Not selected	LEPJUN	22
Preferred	MELRAM	51	Avoided	COPRHA	21
Avoided	COPRHA	47	Avoided	LEUFAS	16
Avoided	LEPSCO	32	Preferred	PSEARB	14
Avoided	PITTEN	26			
Preferred	COPLUC	14			

Notes. Palatability classes follow Forsyth et al., 2002 and A. Forbes personal observation for *Kunzea*. Species codes relate to the species names listed in Fig 9.

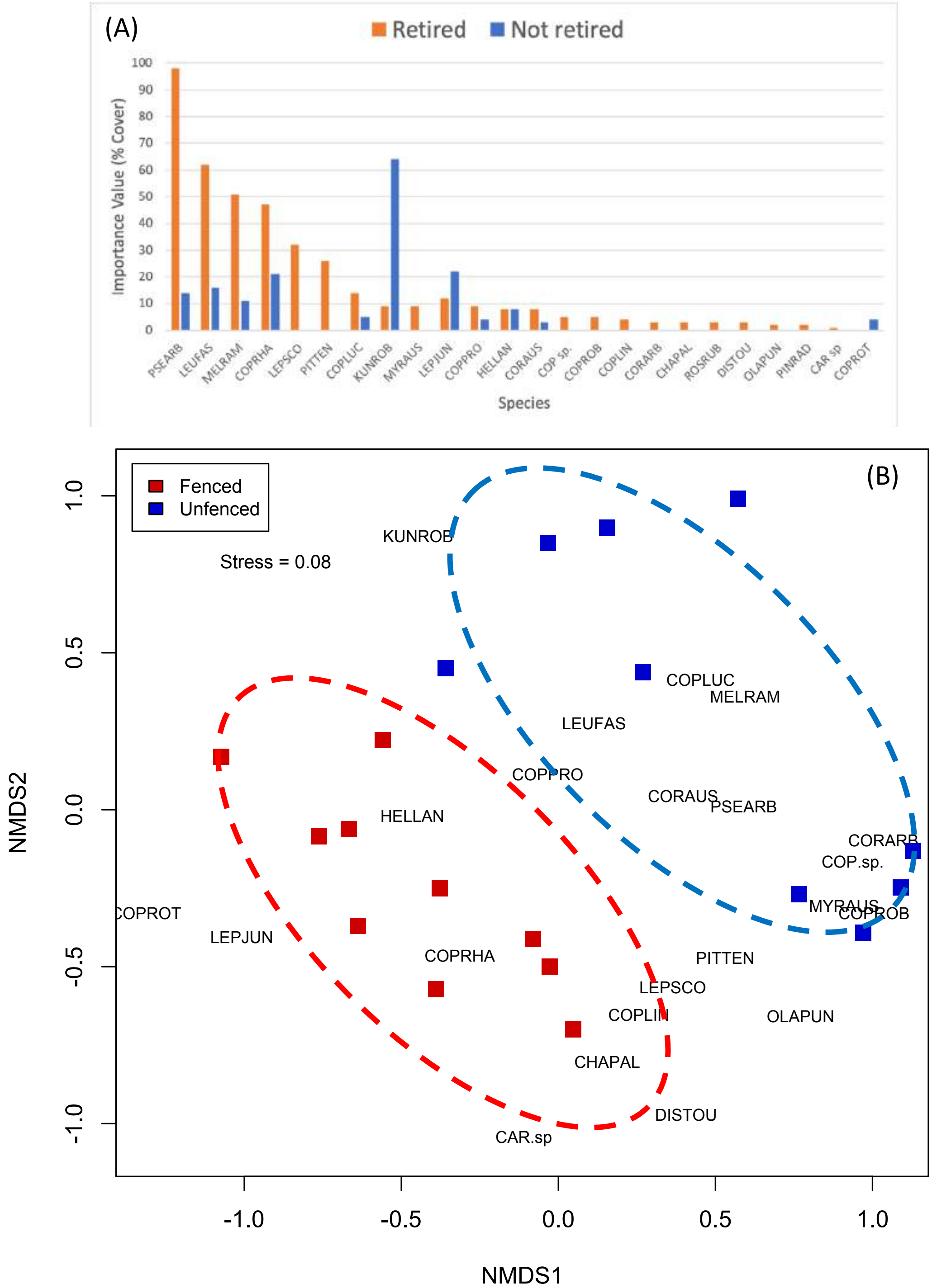

Figure 9 (A & B). (A) Species IVs in retired (orange columns) and grazed (blue columns) forest of the PQF project area. (B) nMDS ordination showing difference in community composition between grazed and ungrazed forest of the PQF project area.

Note. Species codes are: CARsp = *Carmichaelia* sp., CHAPAL = *Chamaecytisus palmensis*, COP sp. = small-leaved coprosma, COPLIN = *C. linariifolia*, COPLUC = *C. lucida*, COPPRO = *C. propinqua*, COPRHA = *C. rhamnoides*, COPROB = *C. robusta*, COPROT = *C. rotundifolia*, CORARB = *Coriaria arborea*, CORAUS = *Cordyline australis*, DISTOU = *Discaria toumatou*, HELLAN = *Helichrysum lanceolatum*, KUNROB = *Kunzea robusta*, LEPJUN = *Leptecophylla juniperina*, LEPSCO = *Leptospermum scoparium*, LEUFAS = *Leucopogon fasciculatus*, MELRAM = *Melicytus ramiflorus*, MYRAUS = *Myrsine australis*, OLEPAN = *Olearia paniculata*, PINRAD = *Pinus radiata*, PITTEN = *Pittosporum tenuifolium*, PSEARB = *Pseudopanax arboreus*, ROSRUB = *Rosa rubiginosa*.

3.2.5 Implications for management

The forests where all ungulates were uncontrolled (stock could access freely) had less than half the number of woody species compared to that found in fenced forests. Unfenced forests were missing species of a stature that could form part of the forest canopy in the future. Without recruitment to the forest canopy, as the existing trees senesce and die these forests will gradually thin and disintegrate. These data demonstrate that fencing domestic ungulates from native forests is essential for diverse and permanent forest cover and this conclusion has previously been reached in other areas of New Zealand (Smale et al., 2008). The data also show that in the PQF area, even when forest is fenced from stock, feral herbivores are still impacting forest health. In places this effect is severe, with bark stripping, ring barking and only a moderate cover of palatable tree species, together these factors provide strong indications of detrimental levels of feral ungulates in the PQF area.

This means that feral herbivores require control across the PQF area if the area is to support diverse, permanent native forest in the long term. In particular, there are anecdotal accounts and evidence from our surveys that feral deer populations are well above population sizes where native forest can regenerate adequately. Where control does not occur, or where feral herbivores are fostered for economic or recreational/cultural reasons, a profound trade-off occurs, where as a result native forest health and longevity is significantly compromised. Unless management addresses feral herbivores, the native forest estate is limited in its ability to support a diversity of biological life and factors such as biomass (carbon), biodiversity and ecosystem services will continue to be severely limited.

Achieving healthy native forest at a landscape scale will require an ecosystem management approach, where animal control is coupled with enrichment planting and mimicked disturbance to address local extinction of seed sources (Forbes et al., 2020) and control of other pests to attain conditions where regeneration and succession can proceed (Norton et al., 2018; Coomes et al., 2002). This will in turn require access to information and material support which we discuss in subsequent sections.

3.3 Funding and Technical Support

3.3.1 Management of existing forests

The large area of existing native forests in Aotearoa/New Zealand means native forest is central to our ability to address the ongoing biodiversity crisis and also assist with addressing the emerging climate crisis. Despite this, there is currently a profound lack of financial and technical support to assist owners' management of existing forest. For instance, funding offered to target erosion control is common among Regional Councils, yet as these sites are already forested, existing forests do not qualify. Funding for biodiversity projects is also offered by many Regional Councils, however, these funds are miniscule relative to what would be required to address existing forest management at landscape scales. Further, both the Emissions Trading Scheme (ETS) and the One Billion Tree Programme (discussed in detail below) focus on establishing additional forest area rather than supporting management of existing forests.

Existing forests have to be included in funding mechanisms if we are to secure the services forests provide, such as storing carbon, providing habitats and supporting biota, regulating soil and water quality and quantity, providing seed sources for natural diversification, and the rest – as outlined earlier in this report. The essential and critical physical management actions that need to be supported following an ecosystem management approach are:

- Fencing to exclude domestic stock,
- Management of feral herbivories, implemented at a community scale,
- Management of other pests, e.g., invasive vines and shade tolerant weeds,
- Enrichment planting to address stalled successions and local species extinctions.

3.3.2 Establishing additional forest area

Stemming the continued decline in the national extent of native forest cover is also essential. Across Aotearoa/New Zealand, 71% (14 M ha) of native forest cover had been lost (Ewers et al., 2006) and during 1996–2012 a net loss of 40 000 ha of native shrub and forest occurred (Ministry for the Environment & Stats NZ, 2018), signalling ongoing declines in native forest cover.

There are several possible approaches to restoring native forest cover. In locations and circumstances where forest species can regenerate, land areas can be reverted from the existing landcover type (normally retired exotic grassland with regenerating native scrub, but also woody species such as gorse (Sullivan et al., 2007) or radiata pine (Forbes et al., 2019) can facilitate native forest regeneration) and in this case management focuses on threats to regeneration and limitations on achieve a long-term succession. This style of restoration is less resource intensive (more passive) than planting to establish a native forest canopy and critically this method of forest establishment presents options to restore forest cover at scale, which is essential if we are to address our biodiversity and climate crises.

At the other end of the spectrum, active planting can be used at sites where natural regeneration is inadequate to form a forest canopy. This active approach is more resource intensive and costly. In most cases the area that can be planted is limited by resources or logistics meaning planting native forests is currently unlikely to be of a meaningful scale in terms of addressing our most pressing environmental concerns, for which more emphasis is needed on management of regeneration, following an ecosystem approach and passive restoration principles.

Having differentiated active from passive approaches to native forest establishment, there is a need for ready access to free/affordable, expert, independent advice regarding methods of forest establishment at a given site. One example of this exists, Te Uru Rākau have funded for 24 months a Restoration Ambassador role to support their 1BT programme, which has proven to be an extremely successful extension service throughout mainland New Zealand and Chatham Island. This model is now proven and should be scaled-up nationally.

3.3.3 Specific recommendations for enhancement of the One Billion Trees Programme

The design of funding models such as the One Billion Trees Programme should address identified constraints to management of existing forests and methods and challenges of establishing new forest. Regarding the 1BT programme, the following specific recommendations are made:

1. Better support for improved management of existing forests and forest land
   Specific measures to support improved forest management include:
   - i. Stock proof fencing, alternative water sources, reconfiguring existing fencing, or other amendments to farm infrastructure to enable forest retirement,
   - ii. Feral herbivore population management with preference to collaboration among communities to achieve landscape scale outcomes,
   - iii. Addressing other pest issues (e.g., invasive vines, shade tolerant species) that threaten the viability of existing forest,
   - iv. Enrichment planting as a recognised/funded treatment to promote forest succession,
   - v. Fund native forest establishment on ex-plantations <5-years-old,
   - vi. Fund native forest establishment on land where native vegetation makes up the existing forest (e.g., scattered native treelands of >30% crown cover).

2. Structure the fund in accordance with accepted ecological priorities
   Including ecological criteria for funding will help address our biodiversity and climate crises:
   - i. National Priorities of Conserving Biodiversity on Private Land[14]
   - ii. Threatened Environments[15]
   - iii. Induced and naturally rare ecosystems[16]

[14] See https://www.mfe.govt.nz/more/biodiversity/protecting-nzs-biodiversity/statement-national-priorities-biodiversity
[15] See https://www.landcareresearch.co.nz/tools-and-resources/mapping/threatened-environment-classification/
[16] See https://www.landcareresearch.co.nz/publications/naturally-uncommon-ecosystems/

iv. Application of ecological and landscape ecology principles for planting treatments with different minimum requirements (e.g., area, width, exceptions to the forest land exclusion)
    i. Funding buffer plantings and forest clearing plantings around and in existing remnants to ease microclimate effects and promote ecological integrity,
    ii. Considering landscape configuration (connectivity) and functionality,

3. Greater support for passive restoration approaches (reversion)
   Potential measures to better support establishing native forest through management of regeneration (reversion) include:
   i. Fund areas <5 ha (suggested 1 ha minimum) for reversion,
   ii. Supporting specific forest establishment approaches following the active to passive theory (e.g., Forbes et al., 2021; Crouzeilles et al., 2020). In other words: topping up on what nature can achieve naturally to optimise forest establishment outcomes. Why fund planting of a native canopy when a native canopy will establish naturally? Instead, fund interventions to make the naturally established canopy/forest better.
   iii. Feral herbivore management (either by fencing or population management), promoting collaboration among communities to achieve landscape scale outcomes,
   iv. Addressing other pest issues (e.g., invasive vines, shade tolerant species) that threaten the viability of existing forest,
   v. Enrichment planting as a recognised/funded treatment to promote forest succession,
   vi. Predator control to restore and maintain healthy seed disperser (e.g., kererū, tūī, korimako) populations.

4. Levels of funding and access to expert advice
   Establishing native forest through planting is currently a relatively expensive exercise[17]. Cost is a barrier for many people who wish to proceed with native forest establishment. The active-to-passive theory goes a long way to address this issue, however, at sites and in circumstances where native forest restoration planting is required, funding a greater proportion of the actual cost (of both planting and fencing) would enable greater levels of forest establishment.

[17] Actual costs vary depending on a range of factors (e.g., composition, spacing, accessibility, preparation & maintenance requirements) but estimates for planting and five years of maintence are \$15 250 ha^{-1} (see The Aotearoa Circle publication) and \$25 000–\$30 000 ha^{-1} (Douglas et al., 2007).

Access to expert advice has proven to be an enabler of native forest restoration. The Restoration Ambassador extension service should be scaled up to provide free, expert, independent advice over issues such as the active to passive theory, species choice, and management interventions required. There is also a need for training and support in the field of forest restoration for example scholarships, mentoring/internships, and other pathways to work for the next generation of forest advisors.

4.0 CONCLUSIONS AND RECOMMENDATIONS

4.1 Conclusions

4.1.1 Addressing dispersal limitation through enrichment planting

1. Due to past forest clearance, old-growth forest remnants (and therefore, forest tree seed sources) are today scarce in the PQF project area.
2. Without specific management, secondary forests in the PQF project area are limited in their ability to succeed to more advanced forest phases, and important attributes of these secondary forests such as biomass, biodiversity, and habitat are constrained.
3. Highly applicable to the project area is an emerging restoration action, enrichment planting, which involves the planting of desirable species into secondary, exotic or degraded forest to overcome the limitations of ecological isolation and dispersal limitation.
4. As ecological isolation and dispersal limitation are such significant native forestry issues for the PQF project area, enrichment planting demonstration sites were established across nine farms in north Canterbury and south-eastern Marlborough.

4.1.2 Addressing the effects of herbivory through ecological management

5. Introduced herbivores (both feral and domestic) can significantly degrade forest community composition and structure, can compete with domestic livestock, serve as disease vectors, and they can also damage other aspects of primary production.
6. Herbivore populations are significant in the PQF project area and feral populations sizes of at least some species have (and continue to) increased over recent decades.
7. Most feral herbivore populations are viewed collectively as both a pest and a resource. Hunting has recreational, economic and social benefits and maintaining feral mammal populations is desirable from these viewpoints, so a balanced approach to management is required.
8. Fencing feral herbivores in remote hill country, or at large scales, is often unviable. Home ranges of the more mobile species can be large, meaning that population management will need to cross property boundaries, ideally in community-led collaborations.
9. The recovery of our temperate forest ecosystems following herbivore control will take decades and an ecosystem approach to management is required: mimicking disturbance to optimise competitive interactions, re-introducing lost propagules (i.e., enrichment planting), or managing other pests (e.g., invasive vines or shade tolerant weeds).

10. The large area of existing native forests in the PQF area, and nationally, means improved forest management of native forest is central to our ability to address the ongoing biodiversity crisis and also to assist with addressing the emerging climate crisis.
11. Stronger avenues of advice and support are needed to support improved forest management. This should include extension services, training and education, and adequately sized and well-structured forest restoration funding.

4.2 Recommendations

4.2.1 Restoring secondary native forests, exotic vegetation and degraded remnants

1. In areas of the PQF project area (and nationally) where a forest canopy can establish itself, enrichment planting should be conducted at scale to direct successional development towards diverse, high-biomass forests representative of pre-human composition and structure.
2. Feral herbivore populations require greater management to enable the regeneration and succession of native forest species across the PQF project area (and nationally). Community collaborations will be important to achieve forest outcomes at scale especially given the home range sizes of feral deer. A balanced approach will be required to address the social values ascribed by many to feral herbivores while still reducing population sizes to levels where native forest species can regenerate.
3. Overall - improved forest management is needed, and this would comprise a bundle of complementary management approaches to enhance the forests' ecologies: such as mimicking disturbance to optimise competitive interactions, re-introducing lost propagules through enrichment planting, or managing pests such as feral herbivores, invasive vines, or shade tolerant weeds which may inhibit forest regeneration.

4.2.2 Restoration funding and support

4. Native afforestation grant programmes (such as the One Billion Trees Programme) should be structured to (1) provide greater support for improved management of existing forests and forest land, (2) follow a structure that incorporates accepted ecological priorities when allocating grants, (3) give greater support for passive restoration approaches so that restoration can be upscaled, (4) provide adequate levels of funding and ready access to expert advice.

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APPENDICIES

Appendix A: Forbes el at. 2020; Restoring mature-phase forest tree species through enrichment planting in New Zealand's lowland landscapes

Appendix B: Enrichment Planting Demonstration Site Photo Point Monitoring Data

Appendix A

Forbes el at. 2020; Restoring mature-phase forest tree species through enrichment planting in New Zealand’s lowland landscapes

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FORUM

Restoring mature-phase forest tree species through enrichment planting in New Zealand's lowland landscapes

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Abstract: New Zealand's formerly extensive lowland native forests have been comprehensively cleared or modified, and large areas of secondary-growth vegetation have subsequently established. These areas are comprised of native, exotic, and mixed tree and shrub species assemblages. The mature-phase canopy and emergent tree species representative of pre-human New Zealand forests are often rare or locally extinct in these forests, indicating negative ramifications for long-term biodiversity conservation and ecosystem service provision, especially such as carbon sequestration. The successful recruitment of mature-phase canopy and emergent tree species may be prevented by biotic and abiotic filters related to dispersal (e.g. lack of seed sources or lack of dispersal agents), environmental variation (e.g. unsuitable germination microclimate or light availability), and competition (e.g. exotic weed competition). Failure of mature-phase tree species to cross through these filters may halt forest succession and cause arrested development of the ecosystem. There are also social and cultural imperatives for restoring mature-phase tree species, such as reassembling desired forest habitat and landscapes and providing lost natural heritage and cultural resources. Therefore, to restore secondary forests, depauperate remnant forests and create new forests that have complex structure, high biomass, and natural canopy tree diversity, mature-phase canopy and emergent species should be reintroduced through human interventions (i.e. enrichment planting). Experiments demonstrate that mature-phase tree species establishment can be optimised through canopy manipulation to address competition for light. Such targeted management can determine successful recruitment of mature-phase tree species, as can weed maintenance post-enrichment planting and landscape-level pest animal control. Currently political focus is emphasising planting of new early-successional native forests. However, support from scientific research and policy development is essential to actively recruit mature-phase tree species where they are now poorly represented and hence forest succession may be arrested. Afforestation and emissions trading policies need to support the reinstatement of mature-phase tree species within existing regenerating and degraded forests and newly created forests to facilitate the substantial ecological and ecosystem service benefits they provide over the long-term.

Keywords: broadcast seeding, Emissions Trading Scheme, enrichment planting, forest canopy, forest restoration, mature-phase forest tree species, One Billion Trees, pre-human, restoration plantings, secondary forest

Introduction

Upon human arrival in New Zealand about 1230–1280 CE (Wilmshurst et al. 2008), lowland forests featured a diverse array of mature-phase forest tree species comprising a mix of native conifers (Podocarpaceae, Cupressaceae, Araucariaceae) and angiosperms, with the conifers often the structural dominants (Wardle 1991). These ancient forest assemblages evolved under the natural selection pressures of climate, physiography and disturbances (McGlone et al. 2001; Singers & Rogers 2014; McGlone et al. 2016; Wyse et al. 2018;). Subsequent anthropogenic pressures such as large-scale deforestation (14 M ha, 71% of original forest has been cleared; Ewers et al. 2006) have reshaped New Zealand's forests, resulting in starkly different composition, structure, and configuration of woody land-cover types. In particular, the original mature lowland forests have been extensively cleared, leaving small, ecologically isolated remnants that often lack large canopy and emergent tree species because they were selectively logged. Cieraad et al. (2015) have delineated and quantified the threatened environments of New Zealand – these are environments where indigenous cover is less than 10% of their former extent, thus being most threatened. This analysis provides a useful spatial framework for focusing on regions where active habitat restoration is required. Large-scale forest fragmentation typically creates impacts with long time lags,

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such as extinction debt of K-selected tree species (Vellend et al. 2006; Tilman et al. 1994). Large areas of secondary growth vegetation have established in some cleared areas, comprising completely native species, completely exotic, and sometimes mixed compositions. These secondary stands are often on fundamentally different successional trajectories to the forests of pre-human times (Sullivan et al. 2007).

Important in pre-settlement forests (Wardle 1991; McGlone et al. 2017) were the conifers that occurred as canopy or emergent trees, including: (Podocarpaceae) *Podocarpus totara* (tōtara), *Dacrycarpus dacrydioides* (kahikatea), *Prumnopitys taxifolia* (mataī), and *Dacrydium cupressinum* (rimu), and in northern North Island, New Zealand, *Agathis australis* (Araucariaceae; kauri). Associated angiosperm canopy trees included: *Beilschmiedia tawa* (Lauraceae; tawa), *Beilschmiedia tarairi* (Lauraceae; taraire), *Weinmannia racemosa* (Cunoniaceae; kāmahi), *Elaeocarpus dentatus* (Elaeocarpaceae; hīnau), *Dysoxylum spectabile* (Meliaceae; kohekohe), *Laurelia novae-zelandiae* (Atherospermataceae; pukatea), *Nestegis cunninghamii* (Oleaceae; black maire), although *Metrosideros robusta* (Myrtaceae; northern rata) often grew as an emergent (Wardle 1991). The southern hemisphere beech (Nothofagaceae) species *Fuscospora solandri* (black beech), *F. fusca* (red beech), and *F. truncata* (hard beech) extended into lowland environments, although these species typically formed less species-rich forests (Wardle 1984).

Today, the mature-phase tree species that characterised New Zealand's pre-settlement forests are typically poorly represented, which may result in the functional or local extinction of these tree species. Intervention to establish these mature-phase tree species is paramount to ensuring representative, diverse, resilient, long-lived forest communities. Maintaining the presence of such trees is also important for restoring social practices such as use of taonga species (Harmsworth & Awatere 2013) for whakairo (traditional Māori carving, e.g. *P. totara*; Timoti et al. 2017) and rongoā (medicine, Williams 2008; e.g. *D. dacrydioides* & *D. spectabile*).

Currently, heavily deforested countries such as New Zealand promote biodiversity conservation through a focus on active planting of new forests where none exist, or facilitating natural regeneration (Norton et al. 2018). When forest cover is below 5–10%, even small increases in cover may produce large benefits for native bird and other communities (Ruffell & Didham 2017). This focus also helps to provide ecosystem services such as carbon sinks for climate change mitigation (Bastin et al. 2019; One Billion Trees Fund, Te Uru Rākau 2019). In addition to planting new forests, restoration through enrichment of existing degraded forest remnants is critical for meeting key conservation and ecosystem services objectives, including climate change mitigation. Support from scientific research and policy development could bolster restoration efforts to target the restoration of landscapes where mature-phase forest species are poorly represented or where ecological succession is arrested in an alternative stable ecosystem state (*sensu* Connell & Slatyer 1977; Beisner et al. 2003). Such states require interventions to address filters limiting mature-phase tree species establishment such as seed dispersal (Kelly et al. 2010; Hansen & Traveset 2012), exotic weed competition (Wallace et al. 2017), herbivory (Bernardi et al. 2019) and seed predation (Daniel 1973; Overdyck & Clarkson 2012). It is likely that new forests planted from scratch under current government initiatives will face some of these barriers in the coming decades. We therefore suggest it is imperative that we enrich new forest restoration plantings, spontaneous natural regeneration, and depauperate forest remnants (of both native and exotic species). However, successful enrichment requires ecologically informed guidance and implementation and government policies and funding to enable long-term restoration goals.

Although restoring mature forest composition and structure is a long-term process (Crouzeilles et al. 2016), which comes with uncertain successional trajectories (Johnson & Handel 2016), it is a highly desirable restoration aim to achieve a range of ecological, social and cultural benefits. To successfully establish mature-phase forest trees, it is critical we simultaneously appreciate the change in understory conditions during forest succession as well as the changing habitat requirements of mature-phase tree species as they age and develop. We can then appropriately target enrichment planting and management action timing.

As secondary forests develop sheltering canopies that provide stable microclimates and suppression of light-demanding weeds, mature-phase tree species may germinate successfully (Wallace et al. 2017). The dominant canopy and emergent tree species typical of lowland forest remnants in New Zealand exhibit a range of variation in their light requirements, but generally seem to benefit from elevated light conditions associated with canopy gaps and similar disturbances (Knowles & Beveridge 1982; Lusk & Ogden 1992; Wyse et al. 2018). Therefore, as a forest matures into late succession, the initially planted early successional trees senesce, forming light gaps in which the saplings of mature-phase tree species can grow towards the forest canopy. This process may only occur if the mature-phase tree species somehow colonise or are introduced to the forest (e.g. via dispersal agents or human intervention), otherwise a bare understorey may persist for many decades (Fig. 1).

In addition to challenges in appropriate environmental conditions and timing of the regeneration of mature-phase canopy and emergent tree species, common impediments to mature-phase tree seedling establishment and recruitment include reduced pollinator abundance, which can lead to reduced pollination and seed set (Rathcke & Jules 1993), or local extinction of mature-phase species seed sources (Török et al. 2018; Fig. 2), or dispersal mechanism mutualisms (Kelly et al. 2010; Wotton & Kelly 2011), any of which can cause dispersal failure. Even where seed successfully sets, and is successfully dispersed, favourable establishment sites can be unavailable due to disturbance regimes arising from land use (e.g. as might occur in landscapes containing high numbers of herbivores). Any one of these issues can prevent recruitment of mature-phase tree species.

To restore secondary and depauperate remnant forests with attributes such as complex vertical and horizontal structure, high biomass, and representative canopy diversity, mature-phase canopy and emergent species will in many circumstances need to be reintroduced through human interventions such as planting seedlings and saplings (enrichment planting, also known as strip-, gap-, or under-planting) or direct seeding. There is, therefore, an urgent need to demonstrate the efficacy of interventions at management scales through experiments and to develop specific guidelines on incorporation of mature-phase tree species into existing native and exotic vegetation stands.

Here we review current practices for incorporating mature-phase canopy and emergent tree species through enrichment planting generally and discuss the role of enrichment planting in forest restoration and permanent carbon forestry

Figure 1. Heavily shaded understories of early-successional forests with full canopies, such as this *Melicytus ramiflorus* (māhoe) forest, may foster seedling establishment of mature-phase canopy species if they can reach the site. Later, canopy senescence and subsequent light gaps will be required for them to mature beyond the sapling stage and reach the canopy. Porirua, New Zealand (photo: AF).

Figure 2. Large tracts of secondary *Kunzea* (Myrtaceae; kānuka) forest form persistent monocultures due to an absence of the mature-phase canopy or emergent tree species that would have characterised the mature forests. Near Waiau, Canterbury, New Zealand (photo: AF).

in New Zealand. We also outline directions needed for future ecological research and policy support that will enable effective enrichment planting of mature-phase canopy and emergent tree species.

Current practices and future research needs for restoring mature-phase tree species

Enrichment planting and mature-phase forest tree regeneration in canopy gaps

Enrichment through planting and broadcast seeding to restore mature-phase tree species has been practiced globally (Ramos & del Amo 1992; Schulze 2008; Cole et al. 2011; Cunningham et al. 2015; Bertacchi et al. 2016). The performance of naturally established or human-introduced seedlings is typically assessed in contexts such as canopy gaps, cut lines, or beneath intact canopies. The structure of the vegetation surrounding the seedling is an index of available light and has been found universally to be the principal explanatory variable driving seedling growth (Paquette et al. 2006). In enrichment planting research, light environments have been quantified directly through surveys of percentage of available light (Magnoux et al. 2018), or indirectly through surveys of percentage of original stocking/biomass (Lu et al. 2018), percentage canopy cover or canopy openness (Gustafsson et al. 2016; Inada et al. 2017), or gap diameter to canopy height ratio (gap ratio, Zhu et al. 2019).

A global review (Paquette et al. 2006) of the survival and growth performance of canopy species beneath differing levels of forest cover concluded that growth followed a similar pattern in most biomes. In uncut stands, a sharp increase in seedling growth consistently resulted from canopy opening to create < 25% available light transmission, > 75% canopy cover, or a gap ratio of < 0.25. In temperate biomes, further canopy opening to levels of 25–50% available light transmission, 50–75% canopy cover, or gap ratios of 0.25–0.4, caused comparably more gradual increases in seedling growth, but beyond this level of canopy opening seedling growth tended to decline (Paquette et al. 2006). This review suggested that irrespective of biome, very open canopies (e.g. clear-cuts of c. 100% available light, c. 0% canopy cover, gap ratio > 2) are not advantageous for growth or survival of mature-phase tree seedlings (Paquette et al. 2006). The performance of New Zealand's mature-phase tree species should be tested empirically in enrichment planting applications using universal indices such as these to allow international comparisons for growth and survival.

The importance of canopy gaps for passive regeneration of mature-phase tree species in remnant forest stands and in plantations is also well established in the existing literature (Runkle 1982; Kern et al. 2017). The performance of seedling regeneration in gaps varies with forest type, gap characteristics, average rainfall and temperature, seedling shade tolerance and soil nutrient availability (Coomes et al. 2009; Zhu et al. 2014; Lusk 2019). Some of these attributes have been tested experimentally in New Zealand's forest ecosystems, although it appears that no New Zealand studies have replicated the effect of gap treatments across gradients of rainfall and temperature, but see Ogden et al. (1991). Gap closure rates have not been assessed in New Zealand conditions, and we expect the effect of gap closure would be a function of canopy height (and vertical height growth rate) and the rate of horizontal growth that would eventually lead to gap closure. We expect that gap ratio would be a variable useful for quantifying levels of above-ground competition between enrichment planted seedlings and the surrounding vegetation cover across a diverse range of existing vegetation types.

Recent New Zealand advances in enrichment planting

Reconstructed forests (e.g. restored anew by planting) in the Waikato region have been used to assess the effect of early successional canopy age and composition on enrichment seedling survival, in particular, of shade-tolerant native angiosperms planted under canopies comprised primarily of *Leptospermum scoparium* (Myrtaceae; mānuka) and *Kunzea* spp. (Myrtaceae; kānuka) or other broadleaved native species (Laughlin & Clarkson 2018). These reconstructed forests were initially planted at fairly high densities of up to 10 000 stems ha^{-1} with no subsequent canopy manipulation to alter light conditions. High mortality (c. 70%) of the enrichment species (i.e. the species planted through enrichment planting; *Melicytus ramiflorus* (Violaceae; māhoe), *Litsea calicaris* (Lauraceae; mangeao), *Alectryon excelsus* (Sapindaceae; tītoki)) occurred under the older *Leptospermum* and *Kunzea* canopies and it was speculated that this could have been due to low light availability and possibly allelopathy (Laughlin & Clarkson 2018). However, mortality of the same enrichment species was remarkably low (c. 30%) under the planted broadleaved canopies of the same age, which differed by allowing more light to reach the forest floor. In a similar vein, twenty years after planting at the coastal restoration site Tiritiri Mātangi Island (north-east of Auckland), high-density (i.e. 85% canopy cover) restoration plantings of *Metrosideros excelsa* (Myrtaceae; pohutukawa) limited seedling abundance and richness compared with thinned *M. excelsa* stands or mixed species stands (both 56% canopy cover; Forbes & Craig 2013).

A study in north Canterbury trialled artificial canopy gaps as a means of addressing light limitation in mature *Kunzea robusta* (kānuka) forest to assist the growth and restoration of the long-lived native conifer *P. totara* (Tulod et al. 2019). Seedling height growth of *P. totara* was significantly greater beneath canopy gaps than under a closed canopy, with seedling growth rates in the gaps nearly twice those under the closed canopy. Gaps of approximately 3 m radius and 0.6 gap ratio allowed 33% of available light to reach the understorey (gaps where four *Kunzea* trees were removed). These canopy gaps equated to a 76% increase in transmitted light compared to that measured beneath the intact forest canopy (Tulod et al. 2019).

Canopy gap trials have also been used in conjunction with restoration of the mature-phase species *B. tawa* and *P. totara* in an 18 year old 24 ± 0.5 m tall exotic *Pinus radiata* (Pinaceae; radiata pine) plantation in the eastern Marlborough Sounds (Forbes et al. 2016). Interspecific variation in life history traits was important for seedling growth and the species suitability between large (5.6 m radius; 84% light transmission; expanded gap ratio = 0.58) and small (2.3 m radius, 49% light transmission; expanded gap ratio = 0.4) canopy gaps. The relatively light-demanding species *P. totara* grew better in large gaps while the shade-tolerant *B. tawa* grew better in small gaps. The effect of herbivory is an important secondary consideration of gap creation, with Forbes et al. (2016) recording greater levels of seedlings damaged from herbivore browse in large gaps. These results suggest a balance is required between canopy species palatability, shade tolerance, growth rate and gap size for successful restoration of mature-phase species within canopy gaps.

In a large-scale forestry trial on the Kaingaroa Plateau (at 520 m above sea level) in the central North Island, three species of native conifer, *D. dacrydioides*, *D. cupressinum*, and *P. totara* were underplanted into a degraded *Pinus ponderosa* (Pinaceae; ponderosa pine) plantation of approximately 30% *Pinus* canopy cover, with *Pinus* canopy cover gradually declining to approximately 5% at 51 years after underplanting (Forbes et al. 2015). Fifty-one years following planting, the best-performing native conifer, *D. cupressinum*, had attained 11.5 ± 0.25 m height, 20.1 ± 0.5 cm diameter at breast height, basal area of 16.5 ± 2.1 m^2 ha^{-1}, and had stored 32.3 ± 3.9 t ha^{-1} of carbon (Fig. 3; Forbes et al. 2015). In addition, Forbes et al. (2015) found that underplanting *D. cupressinum* resulted in a significantly higher native species richness in the forest understorey compared to the two other underplanted native conifer stands, indicating that an optimal species choice can result in good structural performance and the natural regeneration of shade-tolerant native plants.

In addition to light availability and seedling predation, limitation of enrichment plant establishment success has also been evaluated in the context of exotic weed competition. Research in Hamilton of urban forest remnant understoreys dominated by herbaceous weeds demonstrated that enrichment was most successful when planting tall (> 1 m) *B. tawa* plants (Wallace 2017). *Beilschmiedia tawa* of this size typifies a mature-phase tree species seedling, exhibiting a slow growth rate, extreme shade-tolerance (Knowles & Beveridge 1982; Carswell et al. 2012), and requirement for a stable understorey microclimate (Clarkson & McQueen 2004). Despite the importance of a closed canopy to protect *B. tawa* from frosts and desiccation while young, growth rates increase for saplings if more indirect light is available from canopy gaps (Knowles & Beveridge 1982), which also favourably warms the microclimate. In this work, an initial planting height of > 1 m under a closed canopy limited suppression by the aggressive groundcover weed *Tradescantia fluminensis* (Commelinaceae; wandering Jew) (Standish et al. 2001). *Beilschmiedia tawa* growth rate was tested in conjunction with a factorial design including concurrent mulching and weeding, neither of which significantly increased growth rate over four years of establishment (Wallace 2017).

Broadcast seeding to introduce mature forest canopy species (*E. dentatus*; *L. calicaris*; *B. tawa*) was also trialled in Hamilton City urban forest restoration enrichment (Overdyck et al. 2013). This study determined to find best practice for limiting seed predation and improving seedling germination through a factorial design including a control and three factors: caging, removal of fleshy pericarp, and incorporation into fertiliser-enriched clay balls. Their results indicated that caging and clay balls significantly increased survival and establishment. Uncaged seeds were 58% predated compared with only 4% of caged seeds. Uncaged seeds with pericarp removal that were also in clay seed balls had a better outcome with an intermediate loss of 35%. Use of the clay ball doubled the seedling establishment rates after germination in *B. tawa* (6% vs 12%).

Management interventions required to support enrichment planting

Enriching existing vegetation with mature-phase canopy and emergent trees is likely to require supportive management by creating favourable planting sites, releasing (i.e. pruning or other forms of targeted vegetation removal) existing vegetation to address competition from surrounding vegetation, and also management at wider scales to address the effects of introduced herbivores and omnivores (Richardson et al. 2014).

Canopy gaps can be created through pruning, felling, ring barking (Tulod et al. 2019) or poisoning of existing

Figure 3. After 51 years, *Dacrydium cupressinum* (rimu) underplanted in a degraded *Pinus ponderosa* plantation has taken up structural dominance. Kaingaroa Plateau, central North Island, New Zealand. (photo: AF).

canopy vegetation. The gap creation approach should be used conservatively in contexts subject to intense exotic seed rain because exotic species (e.g. herbaceous vines) may take advantage of the newly available light resources and outcompete the saplings of mature-phase tree species. Gap size can be controlled by the amount of vegetation manipulated and ongoing releasing (e.g. weed removal) may be necessary to minimise competition until the planted seedlings have grown into or above the surrounding canopy level (Paquette et al. 2006).

We suggest practical methods for helping to address the threat of herbivores and omnivores with large home ranges, such as *Cervus elaphus* (red deer), *Capra hircus* (feral goat) and *Sus scrofa* (pig), include establishing landscape-scale collaborations, such as community pest control schemes, and selecting mature-phase tree species of lower palatability (Forsyth et al. 2002). Furthermore, planting taller seedlings that will rapidly grow vertically out of the browse tier, and striking an appropriate balance of light-demanding life history traits and growth rates to ensure rapid growth even in the presence of browsers (Forbes et al. 2016). Another approach could be to plant mature-phase tree seedlings into situations featuring physical barriers to herbivores, thus reducing, or preferably avoiding, visitation by introduced mammals (Whyte & Lusk 2019). Finally, government policies to remove herbivorous exotic mammals such as deer from public forests would help ensure the next generation of forest growth. Eradication of exotic mammalian herbivores and omnivores is the most desirable solution in the long term.

We see a need to explore novel opportunities to plant mature-phase tree species into the shelter of light-demanding exotic weeds, such as stands of the shrub *Ulex europaeus* (Fabaceae; gorse). This species forms dense monocultures and is less likely to be penetrated by introduced herbivores, thus providing safe sites for seedling growth. Methods that may be used to plant in such scenarios include the use of clay balls (Overdyck & Clarkson 2012) and drone technology (Elliott 2016) for seed dispersal. Planting mature-phase tree species into communities dominated by light-demanding species reduces the threat of halted succession because the cover of light-demanding species will ultimately be suppressed and outcompeted through canopy shading by the planted species (e.g. Sullivan et al. 2007).

There are also potential risks arising from enrichment planting that need to be carefully managed. For instance, it is important that seedlings are ecosourced to avoid genetic homogenisation or the introduction of genetic material from maladapted local ecotypes, and the scrupulous nursery practices are enforced to prevent the propagation and spread of disease (Norton et al., 2018). Further, to ensure natural patterns in species distributions are correctly observed and maintained when enriching existing vegetation, species choice requires expert input.

Enrichment planting in the context of restoration and climate change mitigation

In contemporary fragmented landscapes, in particular those comprising predominately secondary regeneration or degraded urban forests, there are a number of situations where mature-phase canopy species will not join secondary forest successions or where they may only establish gradually over multiple centuries (Kelly et al. 2010; Rozendaal et al. 2019). Reduced recruitment of mature-phase canopy species limits the restoration of forest biodiversity and the ability of forests to sequester atmospheric carbon (Lennox et al. 2018). Mature-phase species are important for restoration of forest biodiversity as they provide unique ecological resources, habitats and structural features in forests (Lindenmayer 2017), while contributing long-term carbon sequestration and storage services (Luyssaert et al. 2008). Restoring mature-phase canopy and emergent species is, where required, an important restoration intervention to direct secondary successions for biodiversity restoration and climate change mitigation purposes. The draft National Policy Statement on Indigenous Biodiversity (NPSIB) (Biodiversity Collaborative Group 2018) recognises the need for creation of new indigenous dominated forests in biodiversity depleted environments of New Zealand, most notably urban and peri-urban zones. Proposed guideline 19 recommends restoration and reconstruction objectives for establishing a minimum of 10% indigenous cover in such environments.

We also see opportunities to reflect this important aspect of forest restoration in climate change policy. For instance, in New Zealand, the Emissions Trading Scheme (ETS; Climate Change Response Act 2002) classes forests that occurred prior to 1990 as not being eligible for registration under the ETS. Yet, in many cases, for the reasons given herein, natural forests in this class are incapable of recruiting mature-phase forest tree species and would require intervention to reintroduce ecologically valuable, long-lived, high-biomass canopy and emergent species to progress successional development. In this context, enrichment planting provides a means of addressing arrested successions in pre-1990 forests so that those stands can perform much-needed carbon sequestration and storage services. Similar benefits could be added to afforestation grants that do not currently support enrichment planting of existing forest stands (e.g. One Billion Trees Fund, Te Uru Rākau, 2019). In their current form, these schemes prevent opportunities to restore mature-phase canopy tree species and could be restructured to instead provide for the restoration of these species and the critical ecological and ecosystem services they provide. A current anomaly in the ETS also needs to be addressed where the definition of ‘forest’ regards pre-1990 native woody vegetation such as kānuka stands as ineligible for carbon credits, yet there are no carbon liabilities associated with converting them to plantation species. This policy can result in the perverse outcome of landowners clearing *Kunzea* stands that have the potential to develop into forests containing mature-phase canopy and emergent forest tree species. Better alignment of the One Billion Trees Fund and the NPSIB could also lead to better outcomes for both policy goals.

Conclusions

The canopy and emergent tree species that characterise New Zealand’s lowland forest remnants are critical for supporting forest biodiversity and for ecosystem services such as atmospheric carbon sequestration and storage. Due to traits of mature-phase tree species and the limited establishment opportunities in contemporary landscapes, in many areas of New Zealand enrichment planting is required to ensure that these species join secondary successions in both native and exotic stands. Further work should be undertaken to determine where in New Zealand, for either social or ecological imperatives, enrichment planting or seeding is required. Management-scale experiments are required across rainfall and temperature gradients in order to develop clear restoration guidelines for planting into different statures and compositions of existing vegetation and to refine the requirements of establishment to ensure successful recruitment of planted mature-phase forest tree species. Additional research is also required to establish the optimum ways to include mature-phase canopy and emergent species in new native forest restoration plantings, and the potential role of exotic species such as pines and eucalypts as nurse species for their establishment. Climate change policy and governmental afforestation grants should recognise the importance of enrichment planting to enable the restoration of mature-phase forest tree species and the critical ecological and ecosystem services they provide. Anomalies, such as the ETS status of seral kānuka stands, should be removed. By investing in opportunities to enrich New Zealand's many forest types now, we will leave a valuable legacy for generations to come.

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Appendix B

Enrichment Planting Demonstration Site Photo Point Monitoring Data – Available by Request