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Manatū Ahu Matua

Best options for land use following radiata harvest in the Gisborne District under climate change: Literature review

SLMACC 405415

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Executive summary

The Gisborne District is highly erosion prone, and *Pinus radiata* forestry has been a major factor in reducing erosion and stimulating the economy in the region. Climate change is predicted to increase the intensity of storm events and alter rainfall amounts, which will affect erosion directly and indirectly: directly by influencing soil water content, and indirectly by affecting tree growth.

There is a ‘window of vulnerability’ between harvesting and re-establishment of canopy closure in pine plantations, when the landscape is particularly vulnerable to erosion during storm events. Potential future erosion under climate change has been comprehensively reviewed by Basher et al. (2012), who found that erosion in the Gisborne District will probably decrease due to decreasing annual rainfall. The effect of storm intensity and frequency will be a major factor affecting future erosion patterns, but this is currently inadequately predicted.

Increasingly, alternative land-use options for plantation forestry in erosion-prone areas are being assessed by forestry companies and land owners. We assessed the benefits and limitations of three land uses with respect to erosion control and future climate: remaining in plantation pine forest, shifting to native forest, or mānuka plantations for honey production (Table 1).

Table 1. Benefits and limitations of three land uses after pine harvest in the Gisborne District with respect to maintaining erosion mitigation services

Land use / vegetation type	Benefits	Limitations
Mānuka plantations	Effective for erosion mitigation at about 8 years old	Estimate based on limited information: more information is required
	Plantation mānuka productivity under current climate = 1.6 t DM/ha/yr	Estimate based on limited information: more information is required
	Climate-change-induced increases in temperature and CO_2 will probably increase mānuka growth	Mānuka plantation management strategies are not fully established so are difficult to interpret within this report. However, erosion mitigation potential may be reduced under a 15-year rotation clear-fell/replanting strategy.
		Effect of increased windiness on mānuka not known
	Potential for high profits from honey production	The reasons for the large variability in honey profits are not fully understood, and profits are unlikely to be larger than pine production.
		Under clear-fell/replant management, mānuka plantations will probably not be eligible for the PFSI, though may be eligible for the ETS if the manuka is allowed to reach at least 5 m in height. Expenses for conversion to mānuka plantations remain at the cost of the land owner.
Regenerated native forest (includes indigenous shrublands and forest)	Natural regenerated mānuka is effective for erosion mitigation at 16 years old.	There is insufficient information to determine the age at which effective erosion mitigation occurs in mixed-species native forest, and this will depend on species diversity and stem density.
	The productivity of naturally regenerated mānuka under current climate = 3.6–4.2 t DM/ha/yr. Native monocultures = 0.8–3.6 t DM/ha/yr.	There is a paucity of data on growth rates of native trees (other than mānuka and kānuka) on ex-pasture land, particularly in mixed-species stands
	Climate-change-induced increases in temperature and CO_2 will probably increase mānuka growth.	The establishment and growth of native trees may be hindered under climate-change-induced decreases in rainfall, with the exception of mānuka. The effects of elevated temperature and/or elevated CO_2 on native tree growth are not known, except those relating to mānuka.
	Retirement to regenerated native forest may be the best option for erosion mitigation on highly erosion-prone land	Wind-throw and stem snap are more likely under climate-change-induced increased windiness and storm intensity.
	Eligible for PFSI and ETS once regenerated forest established.	Expenses for conversion to native forest remain at the cost of the land owner.
Pinus radiata plantations	Effective for erosion mitigation at 8 years old.	Debris flows and other issues relating to the social licence to operate.
	Rapid-growing trees with productivity under current climate = 17.5 t DM/ha/yr	Regulatory pressure from the NES-PF and Gisborne District Council
	Climate-change-induced increases in temperature and CO_2 will probably increase pine tree growth when not restricted by lack of moisture or nutrients.	Wind-throw and stem snap are more likely under climate-change-induced increased windiness and storm intensity.
	An established industry with considerable research to support the industry.	
	Eligible for PFSI (with no or limited harvesting) and for ETS.	Usually subject to liabilities under the ETS if considered pre-1990 forest land and deforested before a new plantation is established

Notes: PFSI = Permanent Forest Sink Initiative; ETS = Emissions Trading Scheme; NES-PF = National Environmental Standards for Plantation Forestry

Many factors will remain constant for all of our land uses, including fire risk and weed and pest management under future climate scenarios. Climate change may exacerbate the effects of some pathogens on pine production, but it is unclear how climate change will interact with pathogens for New Zealand native trees.

Our assessment of the land-use change from pine plantation to mānuka plantation and natural regeneration was hindered by the lack of information on New Zealand native species, particularly relating to the time at which canopy closure will be reached. Further, plantation mānuka management will greatly affect long-term erosion mitigation potential and possible regulatory pressures, but could not be analysed in this report due to a lack of information. Some of the knowledge gaps identified in this report will be addressed by future work within the SLMACC 405415 project, of which this literature review is a component.

1 Introduction

The land in Gisborne District has a unique erosion profile, with 26% susceptible to severe erosion compared to the national average of 8% (Ministry for Primary Industries 2017c). The Gisborne District currently has approximately 156,000 ha under exotic forest management, principally *Pinus radiata*, and approximately 30% of these forests are currently potentially harvestable (Ministry for Primary Industries & New Zealand Forest Owners Association 2016).

Climate change is expected to result in more intense storm events (Ministry for the Environment 2016; Savage 2006), and therefore it is increasingly important that erosion-prone areas remain under a forested landscape. The rapid development of mānuka (*Leptospermum scoparium*) plantations for large-scale honey production, settlement of Waitangi Tribunal claims, slimmer margins on *P. radiata* wood production, and regulatory pressures have meant a reassessment by commercial forestry companies and land owners with respect to continued exotic plantation forestry.

There are currently three main potential drivers for the conversion of exotic pine plantations to other tree stocks.

1. *Economic:* This may include reduced profit margins due to increased costs and decreased wood demand or prices, and potentially greater profits from other wood/tree-based industries, including niche wood products (e.g. tōtara) or honey production from mānuka. Regulatory drivers and concessions within the Emissions Trading Scheme (ETS) may also affect profitability, and provisions by government funding may create financial incentives for land-use change.
2. *Social:* Many iwi groups will have their land returned in Gisborne District in the near future. This change in governance may bring a change in the management for some forestry blocks, with a possible shift from exotic forestry to native plantations or natural reversion.
3. *Environmental:* There are increased environmental concerns about the effects of forestry on terrestrial and aquatic environments, including slash deposition during storm events.

The identification of factors that will affect land-use decisions in the Gisborne District will be pivotal to understanding how land management of this erosion-prone region may need to adapt to climate change. The SLMACC 405415 project will assess the viability of three land-use options: (1) remaining in *P. radiata* plantation forestry; (2) land-use change to plantation mānuka for honey production; and (3) retiring land for natural reversion to native forest.

The project has several components, including identifying the most erosion-prone land under current production forestry, a literature review to assess the potential limitations and benefits of each land use with respect to erosion mitigation and climate change, economic and biophysical modelling within each land use, and engagement with the community to best incorporate their aspirations.

This report contains the literature review, and will focus on the possible benefits and limitations of land-use options on long-term erosion control under climate change. The review will summarise current national and international (where appropriate) literature, identify knowledge gaps, and recommend parameters for modelling.

Specifically, the review will assess information on:

- the establishment of mānuka plantations or native regenerated forest on recently harvested *P. radiata* land, including soil biochemical parameters, landform position, effects of management, and pathogens under climate change
- the potential implications of these land-uses change on erosion mitigation, particularly during the transitional phase from *P. radiata* to other forest types
- possible revenue sources for carbon sequestration
- the impacts of central and local government regulations on our three land-use scenarios.

2 Background

The Gisborne District is situated on the eastern side of the North Island of New Zealand (Figure 1a) and has a population of 48,500 people (Stats NZ 2017a), with most people (36,600) concentrated in Gisborne city (Stats NZ 2017b). The main industries in the Gisborne District are forestry, horticulture, agriculture and viticulture (New Zealand Trade and Enterprise 2016). The regional climate is warm, temperate maritime, with warm moist summers and cool wet winters (Chappell 2016). Rainfall increases from south to north and from the coast inland, and varies greatly over the region (Figure 1b). The region's climate is also strongly influenced by the El Niño/Southern Oscillation (ENSO), with an increase in major rainfall events during La Niña conditions and severe and prolonged droughts during El Niño years (Gordon 1985; Gomez et al. 2004).

Plantation forests in the Gisborne District make up approximately 9% (155,079 ha) of New Zealand's plantation forest and account for 14% of log exports from New Zealand (Ministry for Primary Industries & New Zealand Forest Owners Association 2016). The forestry industry in the Gisborne District contributes about NZ$96 million, or nearly 5.5% of the area's GDP, as well as 3.2% of employment (Nixon et al. 2017). About 90% of forestry stocks in New Zealand are *Pinus radiata*, with the remaining 10% consisting of Douglas fir, cypress, eucalypts and other exotic soft- and hardwoods (Ministry for Primary Industries & New Zealand Forest Owners Association 2016).

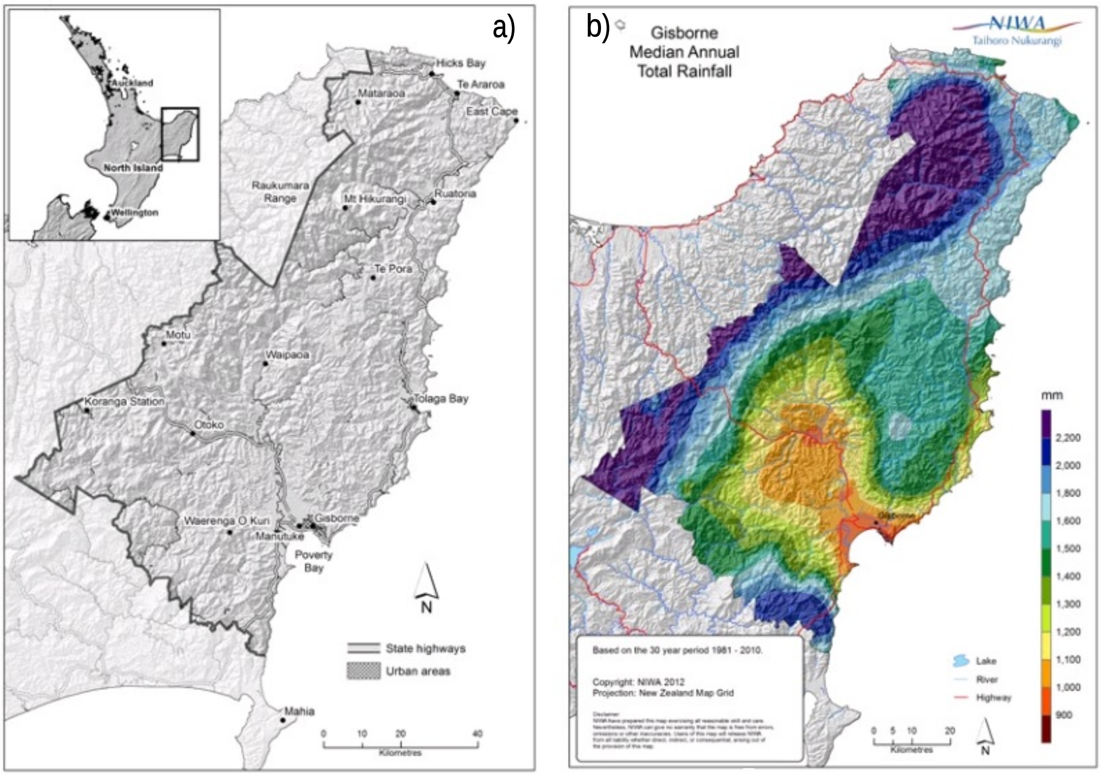


Figure 1. a) The Gisborne District of New Zealand, and b) rainfall variation in the Gisborne District (Chappell 2016).

The Gisborne District consists of three main catchments: Waipaoa (2,208 km²), Uawā (560 km²) and Waiapu (1,758 km²). The sediment content of the rivers in these catchments is among the highest in New Zealand, contributing 33% of New Zealand's sediment yield to the ocean (Hicks & Shankar 2003; Hicks et al. 2011). A combination of factors predisposes the Gisborne District to erosion, including tectonic influences, geological influences, climate, clearance of vegetation, and the high proportion of hilly and steep land.

The Gisborne District is on the circum-Pacific mobile belt at the boundary of the converging Pacific and Indian–Australian plates, which results in high rates of tectonic uplift (1–7 mm/year) and frequent large-magnitude earthquakes (Smith & Berryman 1986; Pondard 2012).

Hill country makes up 92% of Gisborne's land (Gisborne District Council 2004), where 'hill country' is defined as land over 15° (Cameron 2016). The region can be subdivided into two geological terrains: an inland Cretaceous terrain (29% of the region) of variably indurated, extensively sheared, alternating siliceous mudstone; and sandstone of late Cretaceous to Paleocene age comprising part of the East Coast Allochthon (Mazengarb & Speden 2000). Eastward of this, the autochthonous Tertiary terrain (61% of region) comprised of less-deformed bedded to massive sandstones and mudstones of early to middle Miocene age. In places the two terrains are separated by areas of mélange, consisting of Cretaceous-aged lithologies, which formed during the emplacement of the allochthon in the early Miocene; it has been included as part of the Cretaceous terrain.

The topography within the terrains is distinctly different, and is dominated by different erosion processes. The Cretaceous terrain is more prone to gullying, deep-seated earthflow, and slumping, so the topography tends to be subdued, with the majority of slopes between about 15 and 25° (Water and Soil Directorate 1987). Conversely, the Tertiary terrain is dominated by steep slopes and is more prone to storm-initiated shallow landsliding and gullying (Water and Soil Directorate 1987).

Climate is also a significant factor contributing to erosion in the Gisborne District, and the region has a history of severe storm events (Parkner et al. 2006; Chappell 2016), which drive erosion (Water and Soil Directorate 1987). Since the turn of the 20th century, 37 extreme rainfall events have occurred in the Gisborne District (Gisborne District Council hydrologists, pers. comm.), with the last notable storm event being Cyclone Bola in 1988, during which 300–900 mm of rain fell over 5 days, which is estimated to have caused around NZ$176 million in damage (Jones et al. 2008).

Land clearance has led to geomorphic slope adjustment (Hill 1895; Henderson & Ongley 1920) and the initiation of widespread gully erosion (Allsop 1973; Gage & Black 1979), which has become a significant form of erosion in the Gisborne District (Marden et al. 2008, 2011). Reforestation efforts by the New Zealand Government in the Gisborne District between 1961 and 1985 proved to be practical, inexpensive and successful for the treatment of erosion-prone land (Phillips et al. 1991; Marden & Rowan 1993; Marden 2004; Marden et al. 2005). However, much of these conservation plantings are currently ready for harvest.

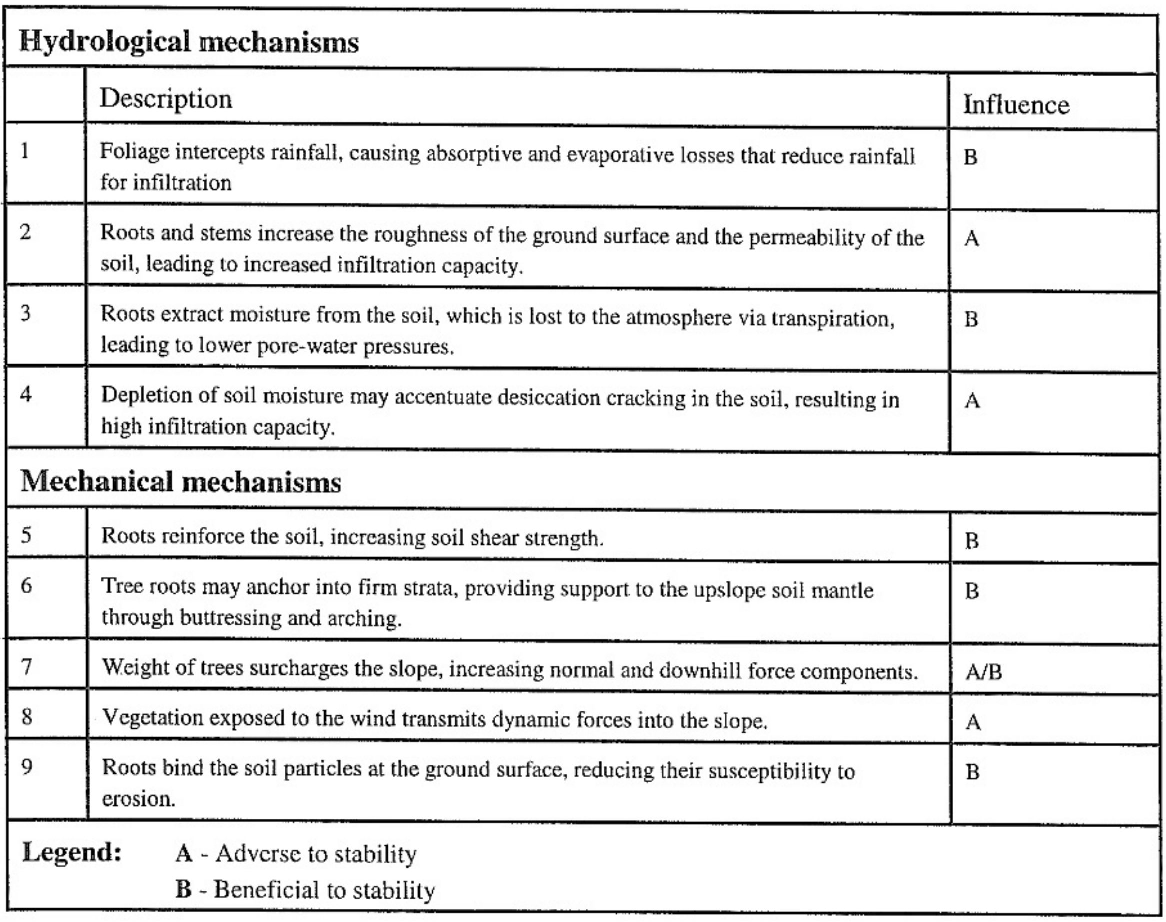
3 Erosion mitigation by vegetation

The type, density, age and extent of vegetative cover all contribute to determining the extent of erosion mitigation within a vegetation type. In the following section we discuss the erosion mitigation effectiveness of pine plantations, natural regeneration, and mānuka in both natural stands and plantation scenarios. During land-use change and forestry operations the act of clearcutting trees also affects erosion, and the effect of tree harvest on erosion will be further discussed.

Many authors have demonstrated that trees mitigate erosion (e.g. Marden & Rowan 1993; Marden et al. 1995; Phillips et al. 2015) through mechanical and hydrological means (Table 2), including by affecting soil strength via root interactions and influencing soil moisture content (Greenway 1987).

Tree roots interact with the soil, and the tree canopy tends to make the soil drier by interception and transpiration, leading to increased soil strength under forests (Stokes et al. 2009). Tree species, growth rates, planting density, and tree age influence the time after planting at which plantings afford effective erosion protection. Therefore, species that have fast growth rates and/or are planted at densities that enable root occupancy and canopy closure in the shortest time are likely to provide erosion mitigation earliest.

Table 2. Hydrological and mechanical effects of vegetation on slope stability (Greenway 1987).



Hydrological mechanisms		
	Description	Influence
1	Foliage intercepts rainfall, causing absorptive and evaporative losses that reduce rainfall for infiltration	B
2	Roots and stems increase the roughness of the ground surface and the permeability of the soil, leading to increased infiltration capacity.	A
3	Roots extract moisture from the soil, which is lost to the atmosphere via transpiration, leading to lower pore-water pressures.	B
4	Depletion of soil moisture may accentuate desiccation cracking in the soil, resulting in high infiltration capacity.	A
Mechanical mechanisms		
5	Roots reinforce the soil, increasing soil shear strength.	B
6	Tree roots may anchor into firm strata, providing support to the upslope soil mantle through buttressing and arching.	B
7	Weight of trees surcharges the slope, increasing normal and downhill force components.	A/B
8	Vegetation exposed to the wind transmits dynamic forces into the slope.	A
9	Roots bind the soil particles at the ground surface, reducing their susceptibility to erosion.	B
Legend:	**A** - Adverse to stability **B** - Beneficial to stability	

3.1 Erosion mitigation by *Pinus radiata*

The age of a stand can be a critical factor in establishing erosion control. Assessments of *P. radiata* during early growth periods found that the bulk of the root system is confined to the top 1 m of the soil (Watson & O'Loughlin 1990; Watson & Tombleson 2002; Marden et al. 2016), and most landslides in the Gisborne District typically fail at ≥1 m depth (Marden et al. 1991; Page et al. 1994). Marden et al. (2016) assessed the below-ground growth of *P. radiata* and found the stands would remain susceptible to shallow landsliding up to at least 4 years of age. However, Watson and O'Loughlin (1990) reported that root systems overlapped between rows at 8 years of tree age. Marden and Rowan (1993) and Marden (2004) showed that exotic forests effectively reduce the incidence of shallow landslides, and that the magnitude of erosion mitigation during storm events is a factor of stand age. Furthermore, Marden et al. (1991, 1995) and Marden and Rowan (1993) reported that exotic forests reach their maximum erosion mitigation potential once the trees are 8 years or older, and that plantations of this age are 16 times less susceptible to landslides than plantations less than 6 years old. Bergin et al. (1995) similarly reported that 10-year old *P. radiata* provides 75% erosion mitigation, but by the time stands are 20 years old they afford 100%. Therefore, while erosion mitigation of very young pine trees is minimal, at 8 years of age they reach sufficient size to enable erosion control services.

The time until canopy closure is an important factor affecting erosion mitigation by trees. Planting density affects the time until canopy closure is reached, and stands with greater planting density (i.e. where trees are closer together) can decrease the time until canopy closure (Halim & Normaniza 2015). In a temperate climate, such as the Gisborne District, the rotation length for *P. radiata* ranges between 30 and 35 years (Hockey & Page 1983), and the recommended planting density for the steepest of the eroding land classes is 1,250 stems per hectare (Ministry of Forestry 1994). More recently there has been a trend towards lower initial stocking rates (e.g. 833 stems per hectare, 4 × 3 m spacing) due to better-quality nursery stock and establishment practices (Shepherd et al. 2017; DairyNZ 2018). A reduction in the current recommended planting density for erosion-prone hill country would further increase the time until canopy closure and increase the risk of storm-initiated landslides in young stands (Marden et al. 2016).

P. radiata is also effective at reducing gully erosion, but the effectiveness of planting is influenced by gully size. Marden et al. (2005) found that gullies planted with pine trees had over 80% probability of stabilising if they were less than a 1 ha in size at the time of planting. Gullies between 1 and 5 ha had a 60% chance of stabilising, a 50% chance of stabilising for those greater than 5 ha, but stabilisation in larger (10 ha) gullies was unlikely. Further, Marden et al. (2011) reported a 17 to 33% reduction in sediment generation from catchments following stabilisation of gullies with trees. At some of the more difficult gully sites, where soil has been removed or where soil pH is low, forest plantings fail to establish (Phillips et al. 2000). Therefore, if gullies are planted while they are smaller there is a greater probability of stabilisation, but this may depend on the soil surrounding the gully.

Pine plantations may also exacerbate erosion due to the toppling of mature trees on the sides and perimeter of the gully, thereby increasing its size (Phillips et al. 2000). In the case of large, actively eroding gullies, there is some evidence that a combination of willow with *Acacia* or poplar (Dolman 1982) may be preferable to pine trees for erosion mitigation. The New Zealand Forest Service instigated staged plantings for gully stabilisation in the 1960s (Marden et al. 2011). A first planting of pine trees in the area surrounding the most actively eroding parts of a gully was initiated. Then, once they had stabilised the area, usually after canopy closure (about 8–10 years), a second planting was undertaken. This strategy proved to be highly successful (Marden 2004).

3.2 Erosion mitigation by mānuka

Native forest cover in the Gisborne region increased by 3,630 ha between 1986 and 2012 (Gisborne District Council 2015b), which reflects a trend of increasing natural reversion in this area. Natural reversion occurs via two methods: *passive*, with no human intervention (i.e. with no planting of native trees), and *active*, where native plants are planted into fenced land with pest control. In passive regeneration, mānuka/kānuka can colonise disturbed sites first and act as nursery plants for eventual secondary succession. Kānuka (*Kunzea ericoides* var. *ericoides*) is commonly associated with mānuka (*Leptospermum scoparium)* in naturally regenerating shrublands and eventually shades out mānuka after 20–30 years (Bergin et al. 1995). Kānuka remains the dominant species in Gisborne District shrublands for at least 80 years before transitioning to native broadleaf forest (Bergin et al. 1995).

Naturally regenerating mānuka/kānuka initially contains very high numbers of stems, which decrease with increasing age due to self-shading and self-thinning, leading to highly variable stand densities (Watson & O’Loughlin 1985). For example, Esler and Astridge (1974) reported that young mānuka stands contain around 80,000 stems per hectare, but as the stands age they approach a stem density of 1,200 stems per hectare. Bergin et al. (1995) measured stand densities of 20,000 stems per hectare in 10-year-old mānuka, which had decreased to 3,000 stems per hectare, with some kānuka, when 40 years old.

The effectiveness of mānuka for erosion control may depend on high stem density, a dense root biomass, a compact root system and high root strength (Marden & Phillips 2015). Watson and O'Loughlin (1985) found that mānuka have small, compact root systems, and this was supported by Marden and Phillips (2015), who found that 50-year-old mānuka had roots extending only up to 1 m from the stem, and only to 0.5 m depth. By comparison, the root systems of 30-year old kānuka tend to be larger, with a root diameter of 3.8 m and roots penetrating to a depth of 1.3 m (Marden, unpublished data).

Watson and Marden (2004) reported that the tensile strength of both mānuka and kānuka roots sampled from a naturally regenerated stand was greater than that of pines, and was a significant contributing factor in increasing soil shear strength. This increase in shear strength in shrublands was sufficient to reduce the initiation of shallow landslides by 54% in 8-year-old stands, and by 91% 16 years after the commencement of the regeneration. Similarly, Marden et al. (1995) reported naturally reverting stands of mānuka reached 90% protection from erosion after 14 years of growth. However, Bergin et al. (1995) also found that erosion mitigation effectiveness was reduced in mānuka stands with lower stand density.

3.2.1 Erosion mitigation by plantation mānuka

The conversion of marginal grazed land to plantation forestry was the previous predominant land-use change, but sites that are some distance from the Gisborne Port are looking at other land uses due to high transportation costs and diminishing forestry returns (Gisborne District Council 2015a). Recently there have been a large number of applications to the Erosion Control Funding Project for afforestation of marginal land with mānuka plantations, increasing from 40 ha in 2013 to 186 ha in 2014 (Gisborne District Council 2015b).

As the establishment of mānuka plantations is a relatively new land use in New Zealand and none of the plantations appear to have been affected by a major storm event, their erosion mitigation effectiveness is untested. Nonetheless, erosion mitigation by naturally regenerating shrublands is attributed to high stem density, so it has been suggested that mānuka plantations established at between 1,000 and 1,600 stems per hectare will be less effective at mitigating erosion than natural stands (Boffa Miskell Limited 2017). Further, Boffa Miskell Limited (2017) suggested that a planting spacing of plantation mānuka would need to be in the range of 1−1.5 m, compared to the industry standard of 2.5−3 m, for effective erosion control.

However, in another study of the growth performance of mānuka (1,100 stems per hectare) at Lake Tūtira, canopy closure was estimated on more fertile sites to occur 7−8 years after planting (Marden & Lambie 2015, 2016). This is about half the time required for a naturally regenerating stand of mānuka to reach maximum erosion mitigation (e.g. Marden et al. 1995; Watson & Marden 2004). Also, Marden and Lambie (2015, 2016) concluded that canopy closure and root occupancy would probably take longer on less fertile and more eroded sites, where plant losses tend to be greater.

The potential effectiveness of plantation mānuka in mitigating erosion will be affected by management decisions, including thinning, canopy trimming, felling and replanting. Boffa Miskell Limited (2017) reported that some plantation managers recommend felling and replanting of a mānuka plantation every 15 years to maintain maximum honey production.

This would increase the frequency at which mānuka plantations would become vulnerable to increased erosion activity, compared to every 30−35 years for plantation pine.

3.3 Erosion mitigation by regenerating native forest

The successional development of New Zealand's native forests generally occurs in two phases: re-colonisation by mānuka/kānuka or other shrubland, followed by secondary succession to broadleaf−podocarp species. In areas of *passive* regeneration, mānuka/kānuka shrubland is commonly the first – but often not the only – species to colonise disturbed sites. In areas of *active* regeneration, native broadleaf/conifer species are commonly interplanted with a mix of early colonising indigenous species (Bergin & Gea 2005), or plantings are delayed for a few years until the colonising species have become well established (Fred Lichtwark, Whāingaroa Harbour Care, pers. comm.). There are no research data to suggest that the erosion control effectiveness of passive or active regeneration differs, but factors that affect the age at which a regenerating stand will mitigate erosion would probably include the extent and type of erosion, soil type, altitude, slope, aspect, rainfall, proximity and abundance of local seed source, weeds and pests, planting density, and mortality.

There are few published accounts on the erosion mitigation effectiveness of New Zealand's indigenous colonising and successional species, with the exception of mānuka and kānuka. Most accounts of other species are of mature trees based on observations of partially exposed root systems along river banks (Cameron 1963) or of wind-thrown trees (Bergin & Steward 2004). Anecdotally, indigenous species are considered to be slow growing, and there is a perception that they are hard to establish compared to some exotic species like willows and poplars (Phillips et al. 2000, 2011).

Phillips et al. (2011) reported that the roots of indigenous forest species are predominantly restricted to the top 2 m of the profile and overall tend to have shallow but stronger roots than the more common exotic species. There are, however, exceptions. *Agathis australis* (kauri) roots, for example, have been recorded at depths in excess of 2−4 m (Ron Lloyd, pers. comm.; Bergin & Steward 2004). While many native trees may have a shallow root system, a dense, shallow, lateral network of woody roots often forms a membrane that stabilises shallow soils (Schmidt et al. 2001) and significantly reduces shallow (<1−2 m) landslide potential on steep slopes (Ekanayake et al. 1997). However, deeper soil profiles benefit little from shallow root networks (Stokes et al. 2009). Common barriers limiting the development of a deep rooting system include a permanent high water-table, bedrock depth or a cemented iron pan, stoniness, and depth of colluvium. Watson et al. (1995) reported that the rooting depth of New Zealand woody species correlated with soil stoniness and depth to colluvium rather than tree age.

Studies of root growth rates of some of the more common native riparian (Marden et al., accepted-b), conifer and broadleaved species (Marden et al. accepted-a) during the juvenile stage indicate there is a rapid decline in roots with increasing depth and distance from the stem. There are also interspecies differences in root distribution, in that each species has differing proportions of root biomass at different stages of growth. As also found by other authors (Watson et al. 1995, 1999; Abernethy & Rutherfurd 2001; Easson & Yarbrough 2002). These interspecies differences in root distribution have implications for the stand densities required to provide a root network sufficient for erosion mitigation of shallow landslides (Phillips et al. 2012).

Using canopy growth data for juvenile New Zealand native species, we estimated the time after planting when they may become effective for erosion mitigation. For successional conifer and broadleaved species we found that canopy closure, and therefore effectiveness for erosion control, at a planting spacing of 1 m (line indicated at 0.5 m) would occur between 2 to 3 years after planting (Figure 2) (Marden et al. 2018, accepted-a). Further, at a wider planting spacing of 3 m (line indicated at 1.5 m), canopy closure would occur after 4 years in the fastest-growing species (tōtara), but for the slowest-growing species (kauri) canopy closure was estimated at 9.7 years (data not shown).

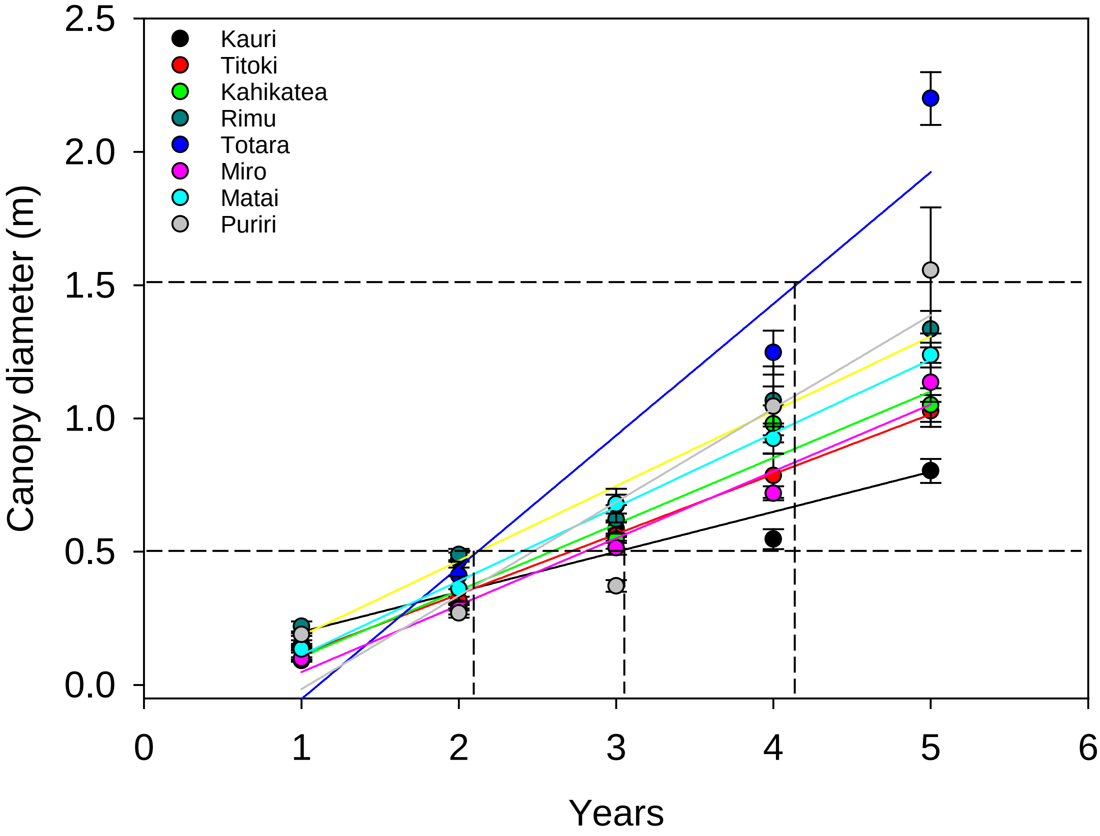


Figure 2. Canopy diameter of eight native conifer or broadleaf tree species measured for the first 5 years of growth. Lines at 0.5 m and 1.5 m canopy diameter represent canopy closure at planting spacings of 1.0 m and 3.0 m (Marden et al. accepted-a). Kauri (*Agathis australis*), tītoki (*Alectryon excelsus*), kahikatea (*Dacrycarpus dacrydioides*), rimu (*Dacrydium cupressinum*), tōtara (*Podocarpus totara*), miro (*Prumnopitus ferruginea*), mātai (*Prumnopitus taxifolia*) and pūriri (*Vitex lucens*) were included in the analysis.

Similarly, Marden et al. (accepted-b) measured the canopy growth annually of 12 juvenile early colonising species (Figure 3). Species with a denser and wider canopy (e.g. tutu and karamū) were more likely to reach canopy closure earliest and provide greater hydrological benefit than taller, more upright species with smaller-diameter canopies. At 1 m planting spacing all of the species reached canopy closure between 1 and 4 years. However, at 3 m planting spacing tutu reached canopy closure at about 3 years after planting, karamū at about 4.5 years, and the remaining species between 5.3 and 14.5 years (data not shown). Further, Marden et al. (accepted-b) suggested that a mixed-species planting of these species established at 4 × 2 m spacing is unlikely to attain canopy closure until 8−9 years after planting, until which time they will remain vulnerable to the initiation of shallow landslides during storm events.

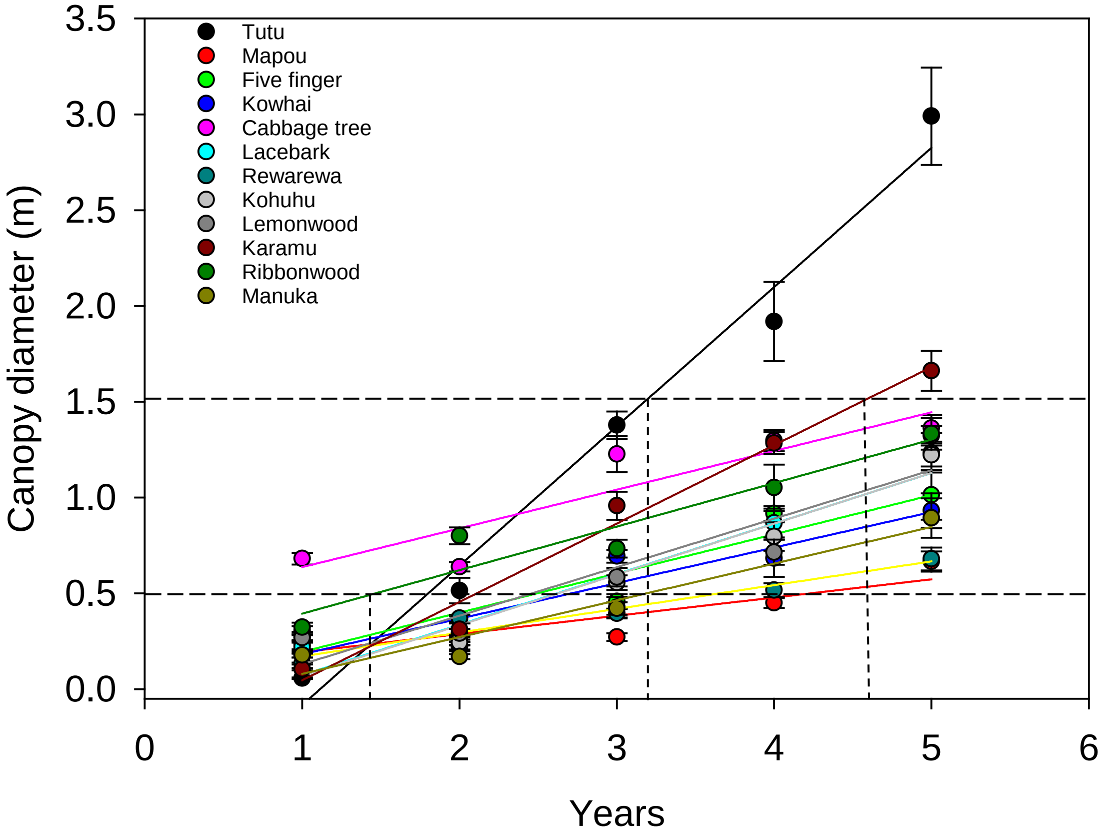


Figure 3. Canopy diameter of 12 native seral species measured for the first 5 years of growth. Lines at 0.5 m and 1.5 m canopy diameter represent canopy closure at planting spacings of 1.0 m and 3.0 m (Marden et al. accepted-b). Tutu (*Coriaria arborea*), māpou (*Myrsine australis*), five finger (*Pseudopanax arboreus*), **kōwhai (***Sophora tetraptera)* cabbage tree (*Cordyline australis*), lacebark (*Hoheria populnea*), rewarewa (*Knightia excelsa*), **kōhūhū** (*Pittosporum tenufolium*), lemonwood (*Pittosporum eugenioides*), karamū (*Coprosma robusta*), ribbonwood (*Plagianthus regius*), **and mānuka (***Leptospermum scoparium*) were included in the analysis.

The results of these two studies provide an indicative trend in canopy growth rates of juvenile plantings on which to assess the probable timing when planted indigenous monocultures and/or mixed species plantings may ultimately provide an effective erosion mitigation role. The data collected by Marden et al. (accepted-a & -b) were from a flat, fertile site, and the growth rate of plantings on eroded slopes in the Gisborne area are likely to be slower and the time until canopy closure longer than indicated here.

Although extensive plantings of indigenous species have not traditionally been established specifically for soil conservation, their successional role meets most of the requirements for long-term erosion control (Pollock 1986). It is also important to consider the additional ecosystems services provided by mixed-species native forest, which may have greater associated ecological, cultural, aesthetic and economic benefits than a mono-species stand.

In addition, most species are sufficiently diverse in rooting form to stabilise slopes prone to shallow landslides, but those with shallow roots will be less effective at stabilising the deeper forms of mass movements, such as earthflows and slumps. Plant density, species selection and species mix are also key considerations in accelerating canopy closure and root development at all levels of the soil profile, and in promoting slope stability (Phillips et al. 2001).

There have been few quantitative reports on the erosion control potential of mature indigenous forests. Hicks (1991) reported no incidents of severe erosion in indigenous forest in the Gisborne District after Cyclone Bola. However, Marden and Rowan (1993) reported that while landslide densities increased in mature indigenous forests following Cyclone Bola, indigenous forest was 4 times less susceptible to landsliding than areas of regenerating scrubland. Mature indigenous forests were still, nonetheless, susceptible to gully erosion, which increased in area 8-fold relative to the area affected by gullying in pre-Bola times (Marden et al. 2012).

In a comparison between pine plantations, shrubland (mānuka/kānuka) and native regenerated forests, Hicks (1991) found that differences in erosion mitigation were dependent on the level of erosion. They reported that 16% of land under pine, 27% under shrubland, and 33% under indigenous forest had no occurrence of erosion following Cyclone Bola. They also found 66% of land under pine, 35% under shrubland and 53% under indigenous forest experienced ‘slight erosion’ of between 1 and 4% of their area. Moderate erosion (4−9% of the area) was exhibited in 13% of the land under pines, 19% under shrubland, and 14% in indigenous forest.

However, when assessing severe erosion (>10% of area affected), 20% of land under pine was affected, 15% under shrubland and 0% under indigenous forest. So while native regenerated forest was susceptible to slight and moderate erosion during a storm event, it was not as susceptible to severe erosion compared to the other land uses. Phillips et al. (2000) also suggested that mature native forest is more effective than pine trees for erosion mitigation, while the data presented by Marden and Rowan (1993) showed that after Cyclone Bola areas of indigenous forest (age unknown) and mature pine plantations (>8 years old) provided the same degree of protection against shallow landslide initiation. However, contrary to previous authors, Marden and Rowan (2015) found that sediment generation in indigenous hardwood forest was greater (c. 115 t/ha) than in standing mature exotic forest (c. 10 t/ha).

3.4 Impact of pine harvesting on erosion

Harvesting negates the erosion control ecosystem services provided by trees. The forestry industry in New Zealand uses a clear-felling approach to tree harvesting, whereby forests are separated into ‘compartments’ of trees and upon harvest an entire compartment is logged (NZ Wood 2017). Clear felling leaves the post-harvest landscape vulnerable to heavy rainfall events until the new forest crop has established a closed canopy, which increases interception, and the new tree roots are well established, increasing soil strength. The duration of the period between completion of harvesting and re-establishment of a closed canopy is known as the ‘window or vulnerability’ (Figure 4), which generally lasts between 2 and 8 years (Phillips et al. 2012). The duration of the window of vulnerability varies and is contingent upon the type of previous vegetation cover, the management regime, the rate of loss in root-soil reinforcement following clearance, the length of time a site remains fallow, the planting density of the replacement vegetation, the species type and diversity of plantings, and other influences that have an impact on plant survival and growth rate.

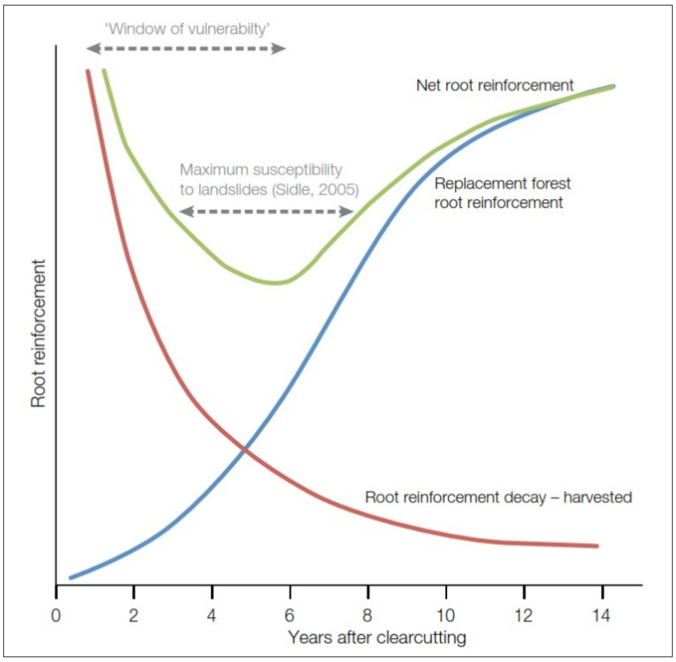


Figure 4. Window of vulnerability to erosion events after harvesting of pine trees (Phillips et al. 2012).

The duration of the window of vulnerability is influenced, in part, by the rate of decay of the roots and stumps of the harvested trees. O'Loughlin and Watson (1979) found that the strength of pine roots decreases to about half only 14 months after harvest, resulting in a loss in root reinforcement. This is countered by rapid root growth of the replacement crop to re-occupy and reinforce the soil. However, removal of forest cover and the consequent reduction of rainfall interception by the canopy and forest floor litter, as well as reduced evapotranspiration, increases soil moisture content and increases the risk of erosion (Pearce et al. 1987). Thus, for replanted forest cutover, factors that affect both above- and below-ground growth rates, including planting density and mortality, are important in determining the duration of the window of vulnerability.

Grant and Wolff (1991) found that cumulatively over a 30-year period post-harvest sediment yields in a logged Oregon catchment were more than twice those of an unlogged catchment. They also found that a single storm event was responsible for most of the sediment yield in their three study catchments. This was also found in the Coromandel area of New Zealand, where storm-related landsliding was the greatest sediment source post-harvest (Marden et al. 2006). The United States Department of Agriculture (2010) also found that sediment generation doubled immediately after harvest, and was still elevated by 60% above background levels 3 years after harvest. Imaizumi et al. (2008) determined that the contribution of sediment supply from landslides in central Japan from slopes harvested 1−10 years earlier was approximately 10 times greater than from unharvested sites. They concluded there was a relationship between the initiation of landslides and debris flows and changes in slope stability due to post-harvest root decay and root recovery of replanted trees.

In a paired-catchment study in Hawke's Bay, New Zealand, Fahey et al. (2003) compared sediment generation in paired pasture and adjacent mature pine forest catchments pre- and post-harvest. They found that during the pre-harvest period the pasture site generated significantly more sediment than the mature forest. However, during harvesting and 2 years after harvest there was greater sediment generation from the forest catchment. Subsequent over-sowing with short-term pasture species resulted in decreased sediment generation. They reported that over the 7-year assessment period, despite the short-term increase in erosion at the forestry site due to harvesting there was still greater generation of sediment from the pasture site. Forestry management practices therefore both contributed to and ameliorated erosion during the harvest and post-harvest period.

Marden and Rowan (2015) assessed the sources of sediment in post-harvest catchments in Coromandel, New Zealand, and found that a single storm in 1995 initiated more than 900 slope failures, of which >99% of the sediment generated was from landslides. Within the landslides, 71% of the sediment was in debris avalanches and about 28% in soil slip. Sediment generation rates were highest (about 254 t/ha) from areas of cutover clear-felled 3 years before the storm, 2.5 times higher than from cutover clear-felled just before the storm (about 94 t/ha). This is explained by the progressive loss of strength from decaying roots that had not yet been counteracted by the development of an effective root system under the new tree crop.

Slope wash and sediment generated from forestry roads and skid sites are considered to be very minor contributors to stream sediment loads (McMahon 1995; Marden & Rowan 1997, 2015; Marden et al. 2006, 2007; Phillips et al. 2005; Williams & Spencer 2013).

The risk of debris flows also increases on post-harvest sites that retain woody debris (Grant & Wolff 1991; Landcare Research & Scion 2017). Debris flows are generated as a result of storm events and are formed when large amounts of water containing high amounts of sediment and harvest debris are channelised (Landcare Research & Scion 2017). Debris is usually deposited in waterways, fans or floodplains (Landcare Research & Scion 2017), potentially affecting road security and having an environmental and visual impact in deposition zones (Landcare Research & Scion 2017; Neilson 2017b). Gisborne District Council (2015a) estimated that the risk of a debris flow across any 50 km^2 catchment in the Gisborne District is once every 5.5 years.

3.5 Erosion under future climates

Climate change may increase the frequency and duration of drought (particularly in El Niño years), temperatures and windiness, and the intensity of storm events in the Gisborne District (Ministry for the Environment 2016). While annual rainfall will decrease due to decreased winter precipitation, there will be an increase in summer rainfall (Ministry for the Environment 2016). Climate change has the potential to alter the extent of erosion by both direct and indirect means (Figure 5). Increased severity of storm events is likely to accelerate erosion via direct impacts on soil strength and water content. Indirect effects of climate change on erosion affect tree growth and establishment (Figure 5). Alteration of rainfall patterns, increased temperatures and elevated atmospheric carbon dioxide (CO_2) concentrations will also interact to influence tree growth (Kirschbaum et al. 2012). Climate change will influence pests, weeds, and fire prevalence in forestry, which can further affect the growth of tree crops.

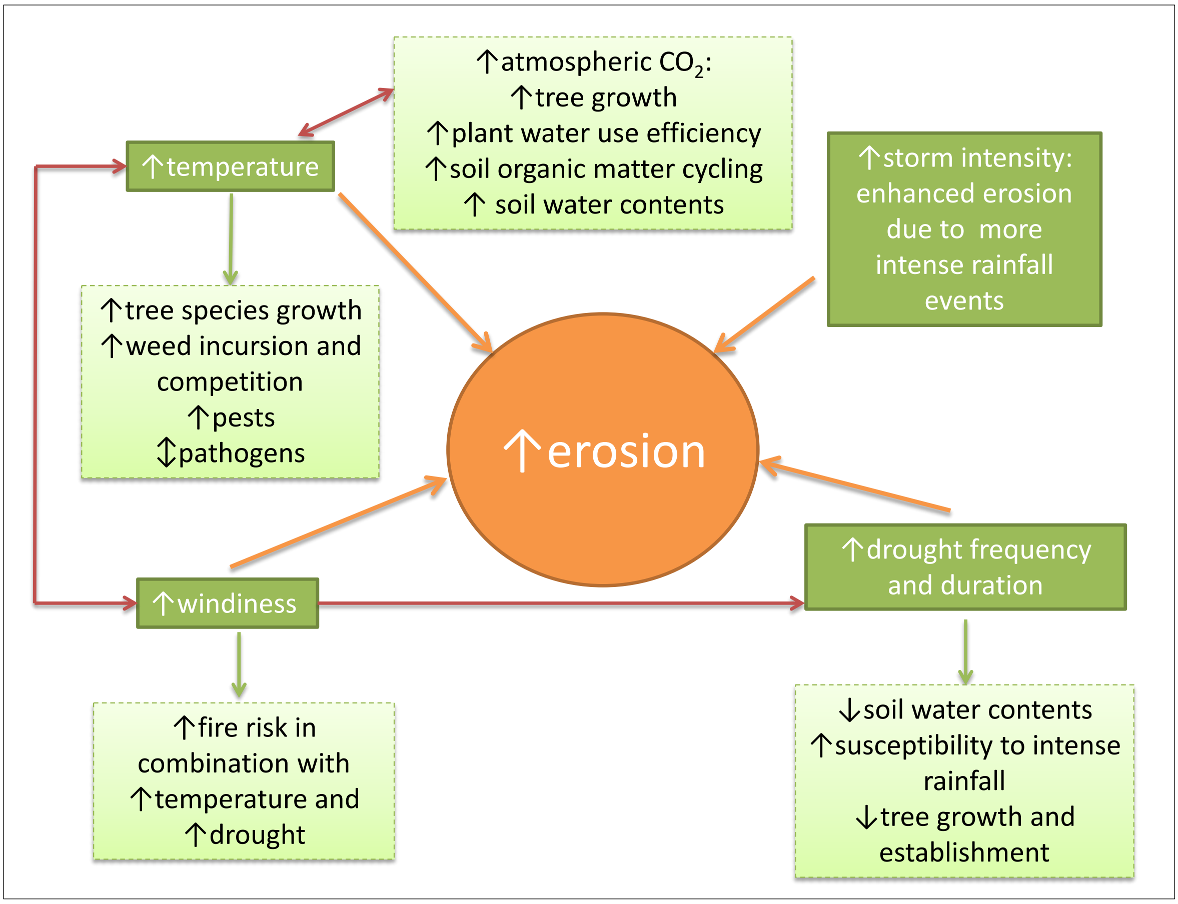
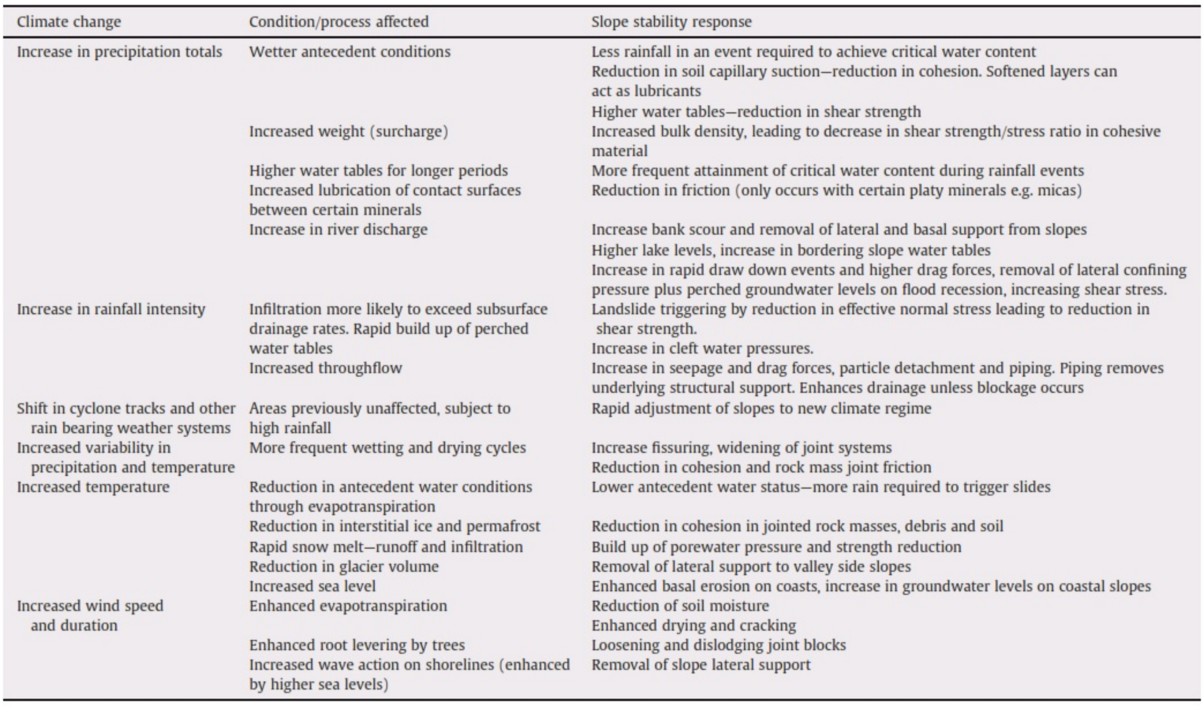


Figure 5. Direct (dark-green boxes) and indirect (light-green boxes) effects of climate change on erosion. Red arrows indicate interactions between climate change parameters.

Basher et al. (2012) comprehensively assessed the impacts of climate change on erosion processes in New Zealand. They suggested that as the annual rainfall in the Gisborne District is likely to be reduced under climate change, erosion will also decrease, but future erosion will be highly dependent upon the intensity and frequency of storm events. Marden et al. (1995) reported an increase in storm damage with increasing total rainfall for some areas of the Gisborne District, but Basher (2013) concluded there is insufficient published literature, models that include erosion processes, or reliable predictions of changes in storm events on the relationships between rainfall and erosion processes to be able to adequately predict erosion under climate change.

Crozier (2010) also assessed the potential impact of climate change on landsliding, and summarised the potential effects of climate change on slope stability (Table 3). While Crozier (2010) did not include a scenario of decreased rainfall that would be relevant to the Gisborne District, they did elucidate the potential changes in erosion processes with increased rainfall intensity, windiness and temperature.

Table 3. Effects of climate change on slope stability (Crozier 2010)



Climate change	Condition/process affected	Slope stability response
Increase in precipitation totals	Wetter antecedent conditions	Less rainfall in an event required to achieve critical water content Reduction in soil capillary suction—reduction in cohesion. Softened layers can act as lubricants Higher water tables—reduction in shear strength
	Increased weight (surcharge)	Increased bulk density, leading to decrease in shear strength/stress ratio in cohesive material
	Higher water tables for longer periods	More frequent attainment of critical water content during rainfall events
	Increased lubrication of contact surfaces between certain minerals	Reduction in friction (only occurs with certain platy minerals e.g. micas)
	Increase in river discharge	Increase bank scour and removal of lateral and basal support from slopes Higher lake levels, increase in bordering slope water tables Increase in rapid draw down events and higher drag forces, removal of lateral confining pressure plus perched groundwater levels on flood recession, increasing shear stress.
Increase in rainfall intensity	Infiltration more likely to exceed subsurface drainage rates. Rapid build up of perched water tables	Landslide triggering by reduction in effective normal stress leading to reduction in shear strength. Increase in cleft water pressures.
	Increased throughflow	Increase in seepage and drag forces, particle detachment and piping. Piping removes underlying structural support. Enhances drainage unless blockage occurs
Shift in cyclone tracks and other rain bearing weather systems	Areas previously unaffected, subject to high rainfall	Rapid adjustment of slopes to new climate regime
Increased variability in precipitation and temperature	More frequent wetting and drying cycles	Increase fissuring, widening of joint systems Reduction in cohesion and rock mass joint friction
Increased temperature	Reduction in antecedent water conditions through evapotranspiration	Lower antecedent water status—more rain required to trigger slides
	Reduction in interstitial ice and permafrost	Reduction in cohesion in jointed rock masses, debris and soil
	Rapid snow melt—runoff and infiltration	Build up of porewater pressure and strength reduction
	Reduction in glacier volume	Removal of lateral support to valley side slopes
	Increased sea level	Enhanced basal erosion on coasts, increase in groundwater levels on coastal slopes
Increased wind speed and duration	Enhanced evapotranspiration	Reduction of soil moisture Enhanced drying and cracking
	Enhanced root levering by trees	Loosening and dislodging joint blocks
	Increased wave action on shorelines (enhanced by higher sea levels)	Removal of slope lateral support

4 Effects of climate change on tree growth

Basher et al. (2012) and Crozier (2010) have established the direct effects of climate change on erosion, but further assessment of the indirect effects of altered tree growth on erosion is examined in the following section. Tree growth depends on the availability of adequate moisture and nutrition, as well as temperature, within the ideal range for a tree species (Yang et al. 2006; Boffa Miskell Limited 2017), which will alter the time until revegetation achieves erosion mitigation status. All of these environmental factors are affected by climate change, including alteration of precipitation, and increasing temperatures and atmospheric CO_2 concentrations. In isolation many of these factors can enhance tree growth, but in combination with other climate dependent factors, such as pests and weeds, tree growth can be inhibited, potentially affecting trees' ability to provide erosion control ecosystem services. In the following section we will discuss tree growth in our target vegetation types under current climatic conditions, as well as the potential effects of climate variables, including precipitation, temperature, CO_2, storms, wind, fire and tree nutrition on tree productivity. We will also assess the potential impacts of climate change on weed incursion and pest prevalence, and how this will affect our land-use scenarios.

4.1 Tree growth under the current climate

Modelling of *P. radiata* growth in New Zealand under current climate conditions has been undertaken by Kirschbaum and Watt (2011) using the process-based model CenW. They found that pine tree growth is greatly influenced by regional rainfall and temperature, and that the average growth rate of pine trees in the Gisborne District is 17.5 t DM/ha/yr.

Mānuka often occurs in combination with kānuka in New Zealand shrublands, and recent modelling of the growth of mānuka/kānuka shrubland under current climatic conditions was also undertaken using CenW (Watt et al. 2012b). Mānuka/kānuka biomass production was estimated to be in the range of 3.6–4.2 t DM/ha/yr in the Gisborne area, and mānuka/kānuka growth was dependent on temperature and rainfall. The work of Watt et al. (2012b) is representative of native regenerating mānuka/kānuka stands, but growth data for mānuka plantations for honey production is somewhat limited (section 3.2.1). Boffa Miskell Limited (2017) report a range of growth rates of plantation mānuka, based on different environmental and site conditions (Table 4). However, the data presented by Boffa Miskell Limited (2017) are not well described and must be treated with some caution.

Table 4. **Growth parameters for mānuka plantations (Boffa Miskell** Limited 2017)

Parameter	*Planting spacing (m)*	*Growth Year 1 (mm/year)*	*Growth Year 2 (mm/year)*	*Site/environment*
Height	1.5−3.0	600−700	500	Sheltered, high fertility
	1.5−3.0	400−500	Not stated	Exposed, low fertility
	Not stated	410 (470 maximum)	410	Warm temperate site
Canopy diameter	No stated	150−300	450−600	Not stated

More comprehensive above- and below-ground growth data for mānuka plantation trees in the Hawke’s Bay and Gisborne areas have been collected by Marden and Lambie (2015, 2016). They found that trees in their 4th year after planting had growth rates that ranged between 500 and 730 mm/year for height and 500−550 mm/year for canopy diameter (Marden & Lambie 2016). Mānuka measured 3 years after planting near Gisborne had a growth rate of between 210 and 310 mm/year for height and 120−180 mm/year for canopy diameter (Marden & Lambie 2016). Further, mānuka production was estimated in Hawke’s Bay to be 1.6 t DM/ha/yr (Marden & Lambie 2015, 2016), which is below half the rate of regenerating mānuka. This is likely to be due to the considerably lower stand density in plantation mānuka compared to naturally regenerating stands. There is a lack of growth data for mānuka plantations, however, and growth is likely to be site specific.

The Hawke’s Bay and Gisborne sites were formerly pasture with low-intensity extensive grazing, which was then afforested with mānuka for honey production (Marden & Lambie 2015). Mineral soils under pine plantations generally have less calcium, nitrogen and phosphorus than neighbouring pasture sites (section 4.2.5). Therefore, it is likely that mānuka planted into pasture rather than previous pine plantations will have more available nutrients, and growth rates may be higher under land-use change from pasture to mānuka than from pines to mānuka. Research assessing the differences in mānuka growth performance between plantations established in pasture or harvested pine blocks could not be located.

Kirschbaum et al. (2011) assessed the growth rates of 10 New Zealand native trees, including tawa (*Beilschmiedia tawa*), rimu (*Dacrydium cupressinum*), pōkākā (*Elaeocarpus hookerianus*), māpou (*Mysrine australis*), red beech (*Fuscospora fusca*), silver beech (*Lophozonia menziesii*), black beech (*Fuscospora solandri*), Hall’s tōtara (*Podocarpus hallii*), miro (*Prumnopitys ferriginea*), and kāmahi (*Weinmannia racemose*). They used over 5,000 plots extracted from the New Zealand National Vegetation Survey Database, which included repeated measurement of individual trees. From those data they extracted growth equations for each species, and then used those equations to predict the productivity of stands of equal-aged monocultures of these species. Using this approach, Kirschbaum et al. (2011) calculated growth rates between 0.8 and 3.6 t DM/ha/yr for the 10 different native trees species. It is not clear to what extent growth equations can be developed from competing mixed-species stands.

4.2 Tree growth under future climate

Under climate change scenarios the Gisborne District is projected to have decreased annual rainfall (but increased summer rainfall), increased temperature, as well as enhanced atmospheric CO_2. These factors will interact to affect tree growth, and will affect trees both positively and negatively. Increasing CO_2 will probably increase photosynthesis and stand growth, and increased temperature is likely to also increase growth, as well as canopy transpiration and organic matter cycling (Kirschbaum 2000; Watt et al. 2008). Increases in transpiration may be at least partially mitigated by increased water use efficiency under elevated CO_2 (Kirschbaum 2000; Watt et al. 2008). Kirschbaum (2000) has suggested that the overall effects of climate change on forest growth will be small compared to the ecological effects of potential changes in forest species distribution.

Watt et al. (2008) completed a comprehensive review on the effects of climate change on pine plantation forests, including an assessment of the physiological response of trees to climate change. They concluded that photosynthesis will probably increase under elevated CO_2, increasing tree productivity, with the magnitude of this increase dependent on water and nutrient availability. Greater responses to elevated CO_2 are expected under water-limited conditions, whereas responses are likely to be limited under nutrient-limited conditions. In summary, future tree productivity will be affected directly and indirectly by climate change, as outlined in Figure 6.

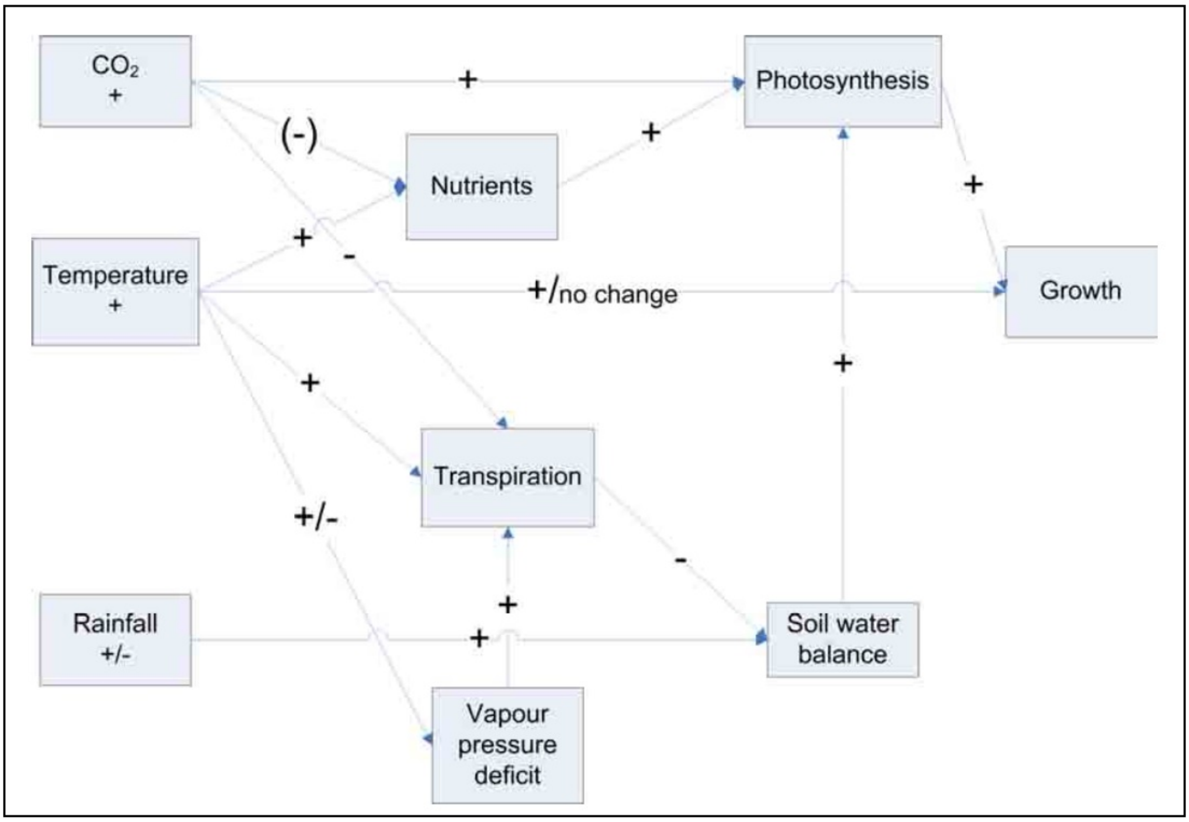


Figure 6. Possible effects of CO_2, temperature, rainfall and nutrients on tree parameters of photosynthesis, respiration, transpiration and growth, and their interaction. Positive and negative symbols indicate the direction of effects (Watt et al. 2008).

In this section we will outline the stand-alone effects of climate change variables (e.g. precipitation, temperature and elevated CO_2) as well as other interactive factors related to climate change, including storms and wind, fire, and tree nutrition on tree growth.

4.2.1 Precipitation

Gisborne is predicted to receive decreasing mean annual rainfall due to decreased winter and spring rainfall, despite a forecast increase in summer rainfall (Savage 2009; Ministry for the Environment 2016). The best growing conditions for *P. radiata* are between 1,000 and 2,000 mm/year (Grace et al. 1991), and the annual rainfall for the Gisborne area varies widely between <1,000 and 1,800 mm (Chappell 2016). Therefore, in areas in the Gisborne District that already experience rainfall less than 1,000 mm, pine tree production may be reduced. Grace et al. (1991) also reported that pine trees are not tolerant of extreme summer drought, but compared to the much more severe drought conditions in Australia (e.g. Kirschbaum 1999), conditions in the Gisborne District are unlikely to become severe enough to curtail the growth of pine stands in the region.

Ronghua et al. (1984) describe mānuka as the ‘most widely distributed, abundant, and environmentally-tolerant’ native shrub in New Zealand. Mānuka adapts well to both wet and dry soils, therefore changing climate may have a limited effect on this shrub once established. However, Watt et al. (2012b) found a linear relationship between the probability of mānuka/kānuka shrubland occurrence and annual rainfall up to about 1,000 mm/yr, although mānuka/kānuka occurrence decreased with rainfall above 2,000 mm/yr. Based on the annual rainfall range for the Gisborne District, it would seem to be well suited to the growth of mānuka/kanuka, regardless of possible future changes in annual rainfall.

Native seedlings are usually slower to establish and grow under dry conditions (Phillips et al. 2000; Wotton & McAlpine 2013), and therefore with the drier conditions predicted under climate change, native regeneration and growth may occur more slowly in the future, although the relative distribution of summer and winter rainfall may also affect that pattern. However, further research is required to assess the impact of climate change on native species, but this is complicated by the range of species that may be present in a natural system.

4.2.2 Temperature

Temperature affects the growth of *P. radiata* (Rook & Whitehead 1979), more so in conjunction with water deficit. Gisborne is predicted to have a mean annual increase in temperature of 0.9°C by 2040 and of about 2.1°C by 2090 (Savage 2009). Grace et al. (1991) reported the ideal annual temperature range for *P. radiata* is between 10 and 14°C in New Zealand, and Kirschbaum and Watt (2011) estimated optimal temperatures slightly higher, at between 12 and 15°C. As the Gisborne area currently has a mean annual temperature of 8°C (Chappell 2016), the predicted increases in temperature would result in an annual temperature of 10°C by 2090, which is still below the optimum for pine growth. Therefore, pine growth is expected to increase with climate change-induced increases in temperature (Kirschbaum & Watt 2011). Wood density is also expected to increase with increasing temperatures and may lead to improved wood quality (Grace et al. 1991).

The growth rate of mānuka can be restricted below 18 and above 22°C (Grant 1966). Gisborne currently has daily maximum temperatures that exceed 20°C between December and March, and that can exceed 30°C, therefore growth rates of mānuka are likely to slow in hotter months and possibly increase over the cooler months. While Grant (1966) has undertaken a comprehensive assessment of factors that affect germination of mānuka, there is surprisingly little information on the effects of temperature on growth rates for this shrub.

Watt et al. (2012b) found a linear relationship between air temperature and probability of occurrence of mānuka/kānuka shrubland, whereby shrublands were not present below annual mean temperatures of 5°C. They found that mānuka/kānuka productivity in response to climate change, without elevated CO_2, would increase by 6% by 2040 and by 9−11% by 2090, probably in response to increasing temperatures.

Lundquist et al. (2011) reviewed literature on the effect of climate change on New Zealand's biodiversity. They concluded that native forests will be altered in their distribution and composition, as determined by the temperature sensitivity (combined with moisture and wind sensitivity) of each tree species. In general, northern species will extend southward and to higher altitudes, and broad-leaf species will become more prevalent in upland forests, but this may be tempered and delayed by the availability of appropriate seed sources. There are limited data on the effect of increasing temperatures on native tree growth and establishment.

4.2.3 Elevated atmospheric carbon dioxide

Elevated CO_2 can increase tree growth and productivity. For example, Conroy et al. (1990) measured an increase in tree growth and wood density under elevated CO_2 (660 µmol/mol) compared to ambient CO_2 (340 µmol/mol), but only if there was sufficient soil phosphorus for plant growth. Elevated CO_2 increases net photosynthesis, possibly by as much as 70% under ideal conditions, and improves water-use efficiency (Grace et al. 1991). However, the response to elevated CO_2 will also be affected by nutrient provision (Conroy et al. 1990; Grace et al. 1991). Conroy et al. (1990) also found that tree genetics affect the growth response of pine trees to CO_2, and that identifying trees with suitable genetic traits would be beneficial to maintain wood production in the future.

Literature that assesses the combined changes in precipitation, temperatures and CO_2 on tree growth is rare. However, at a national scale, Kirschbaum et al. (2012) found an increase in pine productivity of 19% by 2040 and 37% by 2090 for all climate change factors, and they suggested that under the influence of elevated CO_2 the greatest increases in pine productivity will be in the drier areas on the eastern side of New Zealand, including the Gisborne District.

The productivity of mānuka/kānuka shrubland will increase with climate-change induced increases in temperature and alteration of rainfall patterns, but when projected under elevated CO_2 there were further increases in productivity. Watt et al. (2012b) found climate change will enhance mānuka/kānuka productivity by 15−18% by 2040 and by 21−37% by 2090. They also state that when assessing the North Island of New Zealand only, the drier East Coast will exhibit beneficial increases in mānuka/kānuka shrubland productivity in response to elevated CO_2.

As mānuka plantations are a recent land use, the response of this land cover to elevated CO_2 has not yet been estimated. Although space-planted mānuka has less dry matter production per year than naturally regenerating mānuka/kānuka stands, the time at which canopy closure is reached in plantation mānuka is considerably shorter than in natural mānuka. This indicates a greater growth rate in plantation mānuka, possibly due to decreased competition. Therefore it is possible that the growth of space-planted mānuka will also increase under elevated CO_2, but this requires confirmation.

No information on the possible effects of elevated CO_2 on native New Zealand tree species could be located.

4.2.4 Storms and wind

Extreme wind events are often associated with heavy rainfall during storms, but current predicting of the prevalence of future storms is inadequate. Under the current climate, analysis of current storm frequency in the Waipaoa catchment suggests there is a 29% chance of a major event every year, and a greater than 99% chance that one will occur every 10 years (Kelliher et al. 1995). The Waiapu catchment has a recurrence interval for storm events of between 2.6 years in the headwaters and 3.6 years near the coast (Hicks 1995). However, future storm events are less clear, although it is possible that the intensity of extra-tropical storms (such as Cyclone Bola) may increase (Ministry for the Environment 2016), as will the frequency of storm events travelling across the Tasman Sea during summer (Mullan et al. 2011).

The most damaging winds in New Zealand are during extra-tropical storms or ‘topographically enhanced westerly air flows’ (Mullan et al. 2011). It is projected that the frequency of extreme windiness will increase, as well as a slight increase in the maximum wind speed experienced during these wind events (Mullan et al. 2011). Windiness also contributes to increasing drought and fire risk, and may also decrease the radial growth of pine trees, potentially leading to an increase in rotation lengths (Grace et al. 1991). Further, the interaction between increased drought frequency and windiness may lead to increased wind erosion, although the incidence of this in the Gisborne District is likely to be minimal (Basher et al. 2012).

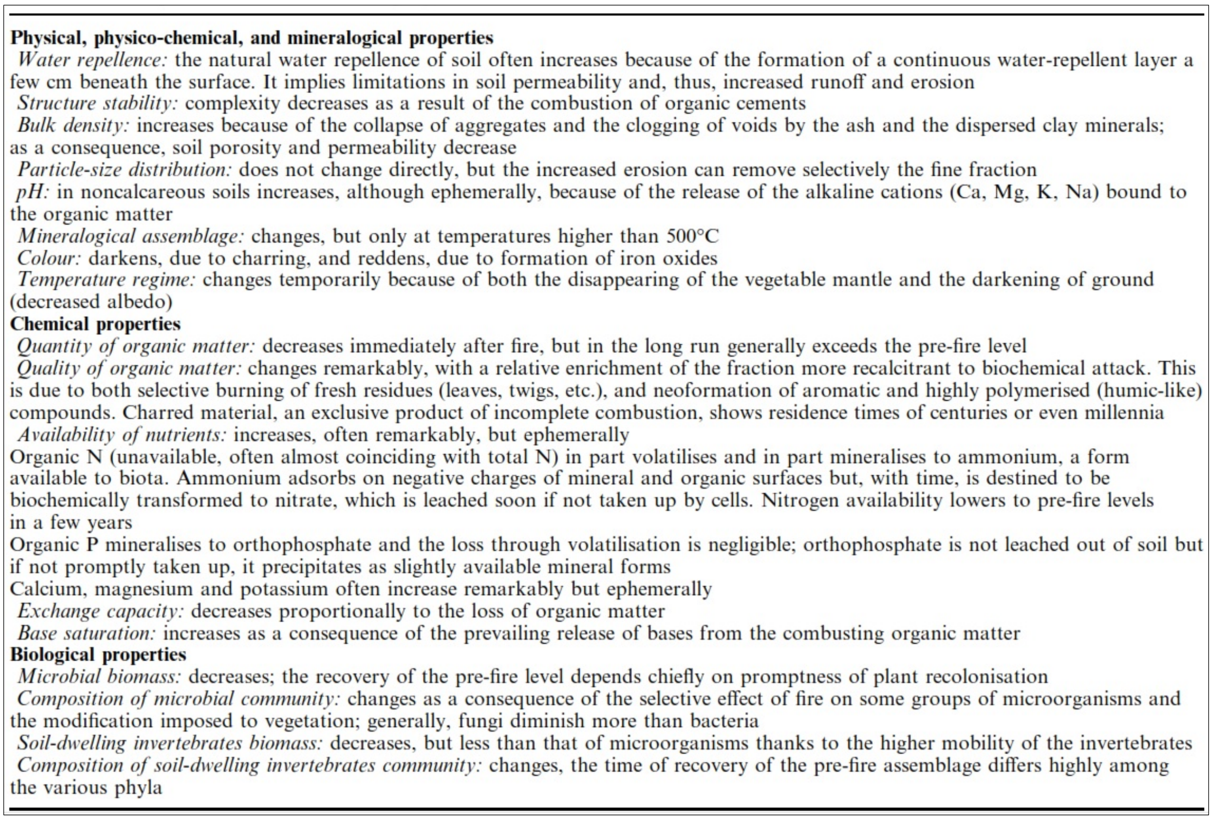
Increased windiness will increase damage to forests, predominantly by windthrow and stem snapping (Grace et al. 1991; Basher et al. 2012). Storm damage to pine trees generally increases as the trees get taller (Scion Research 2012). In 1988 Cyclone Bola caused damage to 27,000 ha of production forest in the Gisborne District due to windthrow and stem snap (Scion Research 2012). Due to the predicted increase in storm intensity in the Gisborne District, forests are likely to incur an increase in wind-related damage (Savage 2006; Watt et al. 2012b). Watt et al. (2012b) used growth models incorporating wind damage models to assess the impact of wind extremes on tree performance. They found that increased growth rates under elevated CO_2 increases the risk of wind damage in production forestry. The risk was further increased by factoring in any increase in extreme wind speeds. Therefore, shorter species of trees (with large root systems) may be more preferable to maintain wood production under increased storm intensity.

Mānuka is commonly found in windy areas (Grant 1966), but no information on the susceptibility of mānuka to windthrow or stem snap could be found in the literature. However, mānuka is a smaller stature, may be less affected by severe wind events. Windthrow is common in our native forests and is a normal part of forest regeneration (Jane 1986; Allen et al. 2013). As in pine trees, windthrow is generally restricted to mature trees over 18 m in height with large stem diameters (Jane 1986).

4.2.5 Fire

Forest fires can have devastating effects on tree production, destroying wood stocks, altering soil nutrient contents, organic matter cycling and microbial communities, as well as increasing erosion (Basher et al. 1990; Allen et al. 1996; González-Pérez et al. 2004; Certini

2005; Guild & Dudfield 2010). Forest fires can also have long-lasting effects on soil properties (Figure 7), which can have further implications for tree growth. The magnitude of the effects of fire on forest systems is associated with fire temperature, intensity, frequency and timing (Basher et al. 1990; Certini 2005). Any increased risk of fire will affect all of the vegetation types in our land-use scenarios.



Physical, physico-chemical, and mineralogical properties

Water repellence: the natural water repellence of soil often increases because of the formation of a continuous water-repellent layer a few cm beneath the surface. It implies limitations in soil permeability and, thus, increased runoff and erosion

Structure stability: complexity decreases as a result of the combustion of organic cements

Bulk density: increases because of the collapse of aggregates and the clogging of voids by the ash and the dispersed clay minerals; as a consequence, soil porosity and permeability decrease

Particle-size distribution: does not change directly, but the increased erosion can remove selectively the fine fraction

pH: in noncalcareous soils increases, although ephemerally, because of the release of the alkaline cations (Ca, Mg, K, Na) bound to the organic matter

Mineralogical assemblage: changes, but only at temperatures higher than 500°C

Colour: darkens, due to charring, and reddens, due to formation of iron oxides

Temperature regime: changes temporarily because of both the disappearing of the vegetable mantle and the darkening of ground (decreased albedo)

Chemical properties

Quantity of organic matter: decreases immediately after fire, but in the long run generally exceeds the pre-fire level

Quality of organic matter: changes remarkably, with a relative enrichment of the fraction more recalcitrant to biochemical attack. This is due to both selective burning of fresh residues (leaves, twigs, etc.), and neoformation of aromatic and highly polymerised (humic-like) compounds. Charred material, an exclusive product of incomplete combustion, shows residence times of centuries or even millennia

Availability of nutrients: increases, often remarkably, but ephemerally

Organic N (unavailable, often almost coinciding with total N) in part volatilises and in part mineralises to ammonium, a form available to biota. Ammonium adsorbs on negative charges of mineral and organic surfaces but, with time, is destined to be biochemically transformed to nitrate, which is leached soon if not taken up by cells. Nitrogen availability lowers to pre-fire levels in a few years

Organic P mineralises to orthophosphate and the loss through volatilisation is negligible; orthophosphate is not leached out of soil but if not promptly taken up, it precipitates as slightly available mineral forms

Calcium, magnesium and potassium often increase remarkably but ephemerally

Exchange capacity: decreases proportionally to the loss of organic matter

Base saturation: increases as a consequence of the prevailing release of bases from the combusting organic matter

Biological properties

Microbial biomass: decreases; the recovery of the pre-fire level depends chiefly on promptness of plant recolonisation

Composition of microbial community: changes as a consequence of the selective effect of fire on some groups of microorganisms and the modification imposed to vegetation; generally, fungi diminish more than bacteria

Soil-dwelling invertebrates biomass: decreases, but less than that of microorganisms thanks to the higher mobility of the invertebrates

Composition of soil-dwelling invertebrates community: changes, the time of recovery of the pre-fire assemblage differs highly among the various phyla

Figure 7. Effects of fire on soil properties (Certini 2005).

Fire contributes to accelerated erosion by increasing the extent of the bare soil surface that can be affected by falling or flowing water, and is further exacerbated by the development of soil hydrophobicity (Basher et al. 1990; McNabb & Swanson 1990; Sala & Rubio 1994). Erosion processes can also be affected by fire due to removal of above-ground biomass, which leads to the eventual loss of roots, contributing to erosion mitigation (Basher et al. 1990).

There are few New Zealand native species that have coping strategies for fire. Mānuka, *Hebe* and *Gonocarpus* have greater seeding and germination after fire, but it is likely that under intense fire events the seed bank for other species will be compromised (Allen et al. 1996). Other plants, including flaxes, *Celmisia* and *Aciphylla*, grasses and sedges, can re-sprout after fire, and natives with buried rhizomes may also survive fires (Allen et al. 1996).

Pearce et al. (2011) suggest that due to a combination of drier conditions, increased frequency of drought and increased windiness, the fire risk in New Zealand will increase, and fires will become more difficult to control. Despite the Gisborne District being predicted to have increased temperatures, increased drought and increased windiness (both frequency and strength), Pearce et al. (2011) suggest there will be no increase in fire risk in the Gisborne District. They argue that the unchanged fire risk in the Gisborne District may be due to decreased summer/autumn westerly winds, increasing rainfall. However, the Ministry for the

Environment (2016) suggested there will be an increase in summer rainfall, and this would mitigate an increase in fire risk during this part of the year. Watt et al. (2012b) also assessed the fire risk in Gisborne and suggested there was an increase in the number of days of very high and extreme fire risk between October and April.

4.2.6 Tree nutrition

Tree growth can be governed by the availability of nutrients within the soil. Pine production leads to a decrease in soil calcium and nitrogen concentrations due to the heavy demand for these nutrients by these fast-growing species (Giddens et al. 1997; Bouillet & Berhand-Reversat 2001; Garrett et al. 2015). This reduction in nutrients can compromise the growth of successive rotations if not corrected (Bouillet & Berhand-Reversat 2001; Garrett et al. 2015). Chirino-Valle et al. (2016) also found a decrease in soil phosphorus content under pine trees, although this contradicted the findings of Giddens et al. (1997) and Davis (2001). Pine tree litter and slash represent a source of nutrients, which is available to feed newly planted trees about 2−3 years after harvest and replanting (Will 1968). While fertiliser addition to plantation forestry is becoming more common, retention of litter and slash leads to better growth performance of pines than fertiliser addition (Smaill et al. 2008). However, it is not always possible to retain forest litter and slash on-slope, particularly in steepland forest plantations (Garrett et al. 2015; Visser 2016), where slash can be mobilised into debris flows during storm events.

The previous management approach for harvest debris was to burn it *in situ* after harvest and before re-planting (Bay of Plenty Regional Council 2011). However, due to regulatory restrictions on this practice, many forest managers now choose to retain harvest debris, with the added benefit of nutrient retention (Bay of Plenty Regional Council 2011; Visser 2016). Burning of thinning debris has also been shown to inhibit native forest regeneration, presumably through destruction of the native seed bank, infestation by exotic weeds, and reduction in mycorrhizal diversity and biomass (Owen et al. 2009). Retention of post-harvest debris can also increase browsing by pest species: Pellerin et al. (2010) found an increase in deer browsing of seedling trees during forest regeneration due to the retention of post-harvest debris.

Degradation of the root systems of harvested trees will also add a small amount to soil nutrient stocks for future rotations. Will (1968) indicated that root nutrient contents at tree harvest may be approximately 45 kg/ha nitrogen, 18 kg/ha phosphorus, 33 kg/ha potassium, and 33 kg/ha calcium. Zhou et al. (2016) found that coarse roots and below-ground woody debris of *Pinus massoniana* degraded faster than above-ground post-harvest stumps and other coarse woody debris, potentially supplying nutrients more quickly.

This was further supported by Garrett et al. (2012), who came to the same conclusion for *P. radiata* material in New Zealand. They also found that root decay varied across their eight study sites, but ranged between 10 and 86 years before 95% mass loss was reached. This indicates that at some sites harvested trees would still be releasing nutrients two rotations after initial harvests. Interestingly, they found that woody debris decay did not increase soil carbon content but did increase soil nitrogen concentrations. Root decay was increased at warmer sites, indicating that climate-change-induced increases in temperature may enhance debris degradation, although the retention of woody debris can reduce soil temperatures and may at least partially negate climate-change-induced increases in woody decay (Smethurst & Nambiar 1990).

Kirschbaum and Watt (2011) found that New Zealand's current pine plantations have sufficient nutrition provisions to maintain growth. However, under climate change, nutrition availability will likely be a balance between increased plant litter production as a result of increased temperature and atmospheric CO_2, and increased soil organic matter turnover in response to increased temperatures (Watt et al. 2012b). Whether this balance maintains sufficient nutrients for plant uptake to maintain the increases in tree growth under climate change will probably be site- and tree-crop specific.

Mycorrhizae

Tree nutrition can be greatly affected by mycorrhizal infection. Members of the *Pinaceae* family are Ectomycorrhizae (ECM), and many species of ECM have been identified in association with *P. radiata* in New Zealand (e.g. Walbert et al. 2010). ECM colonise tree roots and form a beneficial relationship by improving water and nutrient provision to the tree, as well as resilience to pathogen infection, in return for carbon substrate (Smith & Read 1997; Wiensczyk et al. 2002). Pine trees in New Zealand are commonly infected with ECM during the nursery stage to ensure inoculation once planted out. The services provided by ECM to trees can greatly affect tree growth. For example, Ortega et al. (2004) found that *P. radiata* growth was considerably greater in ECM-infected trees compared to non-infected trees, particularly at a drier site. However, Duñabeitia et al. (2004) found that pine growth response differed depending on the species of ECM; they also found that there was large variability to environmental tolerance between ECM species. Despite the species-dependent benefits of ECM colonisation in pine trees, the prevalence of ECM associations will probably be important for maintaining tree growth under the increasing drier conditions associated with climate change.

Only a few native tree species in New Zealand form an association with ECM fungi. They include mānuka, kānuka and beech trees (Orlovich & Cairney 2004). Mānuka and kānuka have the further ability to form associations with arbuscular mycorrhizae (endomycorrhizal fungi; AM), although ECM generally dominate over AM (Weijtmans et al. 2007). Inoculation with AM has been shown to improve mānuka growth and alter the composition of oil extracts compared to non-inoculated shrubs (Wicaksono et al. 2018), although no assessment of the effect of mycorrhizae colonisation and honey production could be located. Mycorrhizae increase the uptake of phosphorus in mānuka and can greatly improve the growth of these shrubs in highly competitive forest/shrublands (Stephens et al. 2005). There has been no assessment of ECM or AM colonisation of mānuka in post-harvest pine plantations, and it is not known whether infection should be undertaken at the nursery stage as for plantation pines.

As stated above, the only other native tree species to be colonised by ECM are beech trees (*Fuscospora* and *Lophozonia* spp.; Orlovich & Cairney 2004). Baylis (1980) suggests that the spread of beech in New Zealand may be inhibited by the lack of appropriate ECM fungus in our soils, but there has been little investigation of the growth response of beech to mycorrhizal colonisation. Further, whether there are appropriate ECM in post-harvest pine soils has not been investigated, although there is an indication from Orlovich and Cairney (2004) of some overlap in the occurrence of ECM genera between pine and beech trees.

The vast majority of plants (about 80%) form relationships with AM fungi (Smith & Read 1997). Johnson (1977) studied mycorrhizal diversity in a conifer–dicotyledonous forest in Dunedin, New Zealand, and found that the majority of the plants present had formed mycorrhizal associations. There was no difference in tree growth between the different mycorrhizae tested, but tree growth was not assessed compared to non-inoculated trees and therefore the findings were inconclusive.

AM also have positive effects on soil structure. Owen et al. (2009) reported that both plant cover and AM biomass contribute to aggregate stability in a thinned production forest, with 31% of the variation in soil stability explained by the abundance of AM. Chaudhary et al. (2009) found that AM abundance in soil contributes to 37% of the variation in surface soil stability. Demenois et al. (2017) found that inoculation of plants with AM or ECM improved plant growth, increased root length density and increased aggregate stability.

Mycorrhizae can be affected both directly and indirectly by climate change. Changes in tree species distribution and substrate availability will directly affect mycorrhizae, while alteration of plant species distributions and allocation of resources to root stores are the main indirect effects (Bellgard & Williams 2011). Compant et al. (2010) reported increased abundance of mycorrhizae, which enhanced plant growth; under elevated CO_2 the response of mycorrhizae to increased temperature was too variable to reach a conclusion.

In summary, the effect of climate change on mycorrhizae, and their associated benefits to trees, remains unclear.

4.3 Weed incursion under climate change

The New Zealand forestry industry spends about NZ$108 million every year on weed control (Watt et al. 2008), and climate change may exacerbate weed incursion (Savage 2006; Watt et al. 2011b; Basher et al. 2012). Weeds can have both positive and negative roles in forest systems: they compete with native or pine trees for light, space, water and nutrients, but they can also contribute to soil nutrition (e.g. nitrogen-fixing plants) and act as nursery plants (e.g. Wotton & McAlpine 2013). Pine forest harvesting usually leads to a dominance of exotic pioneer species, particularly in heavily disturbed soils (Atkinson & Cameron 1993; Brockerhoff et al. 2003; Wotton & McAlpine 2013). A range of weeds are prevalent through plantation forests or colonise after tree harvest. In this section we will discuss the effect of climate change on conifer wildings, gorse (*Ulex europaeus*), Scotch broom (*Cytisus scoparius*), wild ginger (*Hedychium* spp.), and buddleia *(Buddleja davidii*), and their potential impact on target tree crops in future climate scenarios. We will also summarise the current information on these weeds as nursery crops during native forest regeneration.

Conifer wildlings are common throughout New Zealand as a result of the introduction of exotic pines. The rate of spread of wildings has reached the extent that the New Zealand Government has established a management strategy to remove these weeds (Wotton & McAlpine 2013; Ministry for Primary Industries 2014). Further, the National Environmental Standard for Plantation Forestry (section 6.2.2) introduces a responsibility on landowners for the control of wilding conifers (Ministry for the Environment & Ministry for Primary Industries 2017).

After pine harvests, pine seedlings can compete with native seedlings under dry conditions due to the slow growth rate of native trees and high light conditions that favour pine seedlings (Wotton & McAlpine 2013). Therefore, under climate change, wilding conifers may present a greater threat to native tree growth and survival in the Gisborne District due to a forecast increase in drier weather and drought events. Growth and establishment of native forest regeneration can be enhanced by the use of a nursery crop (New Zealand Farm Forestry Association 2005), and pine trees could be used as a nursery crop for native regeneration (e.g.

Wotton & McAlpine 2013). Whether this would occur through no logging or selective logging of pines once regeneration is established would require more investigation, as there would be some practical considerations relating to the health and safety of selective logging on windy steep slopes (NZ Wood 2017), such as in the Gisborne District.

The predominant pioneer species of trees in New Zealand used to be kānuka and mānuka, but these native species have since been replaced by gorse (Sullivan et al. 2007). The Gisborne District Council has set control measures to reduce the prevalence of gorse in the region (Gisborne District Council 2016). An assessment of the effects of climate change on gorse distribution or growth could not be located, but Lee et al. (1986) indicate that gorse may prefer warmer climates. The Gisborne District is forecast to increase in temperature, and it is possible that the distribution or growth of gorse may increase. However, as this weed is already being actively controlled within the district its spread is not likely to change greatly.

Further, while gorse can act as a nursery plant for native trees, it can also impede native regeneration due to the persistence of its seed bank (Atkinson & Cameron 1993). The occurrence of deep litter under gorse can also affect native regeneration and can cause a bias in species distribution, as only species with bigger seed sizes or those that are bird distributed can establish (Wotton & McAlpine 2013). Gorse can form a self-perpetuating stand, particularly where native tree seed sources are limited, but where native seed sources are prevalent, native forest can be established within 30–60 years and gorse shaded out (Sullivan et al. 2007; Wotton & McAlpine 2013). Gorse can also inhibit the growth and decrease the diameter of pine trees (Richardson et al. 1996).

Broom has been shown to compete directly with pine seedlings for light (Richardson et al. 1996) and water and can reduce the above-ground biomass of pine trees by up to 25% in two years (Watt et al. 2003). Potter et al. (2007) assessed the distribution of Scotch broom in New Zealand and reported that despite the Gisborne District having ‘optimal’ environmental conditions for Scotch broom, there are few forestry sites in the region currently infested with this weed. While it seems likely that Scotch broom will remain a potential weed for this region, it is unlikely to be affected by climate change (Potter et al. 2008) and could be maintained at low infestation levels by maintaining Scotch-broom-free transport corridors (Potter et al 2007). Scotch broom may be a better nursery weed than gorse for native regeneration, mostly due to its shorter life span and shallower litter layer, and the fact that it allows more light through to the soil surface (Wotton & McAlpine 2013). However, it is unlikely to be used as a nursery plant in the Gisborne District due to its limited infestation and a community preference for nursery species that are not known pests.

There are two forms of wild ginger in New Zealand: *Hedychium gardnerianum* (kahili ginger) and *Hedychium flavescens* (yellow ginger). Both of these plants form dense rhizome mats which smother young plants and inhibit seedling establishment (Waikato Regional Council 2014). Savage (2006) reported that wild ginger is already prevalent at Te Araroa (Gisborne District), prefers warmer temperatures and a humid climate with regular rainfall, and does not establish well in drier parts of Gisborne District. Climate change will increase temperatures but also increase the occurrence of drought and dry windy conditions, and therefore it is possible that wild ginger may decrease under climate change.

Buddleia is a perennial, semi-deciduous plant which is common in plantation forests, where it establishes easily in disturbed soils under a range of climatic conditions (Watt et al. 2011b). Buddleia usually occurs in temperate, Mediterranean, subtropical and tropical climes, and at high densities can reduce the growth of radiata pine (Watt et al. 2011b). The North Island of

New Zealand is considered to range from ‘suitable’ to ‘optimal’ conditions for buddleia, and this is unlikely to change under future climate change due to the large environmental tolerance of this weed. The Gisborne District will experience windier conditions under climate change, and there is potential for greater wind throw as a result of stronger or more extreme wind events (Scion Research 2012), especially on the majority of the hill country where shallow soils prevail (Gibbs 1959). This increased disturbance under climate change may increase the prevalence of buddleia and other advantageous weeds due to increased disturbance in plantation forest systems in the Gisborne District.

Watt et al. (2011b) and Richardson et al. (1996) reported that buddleia inhibits pine tree growth by shading out young seedlings rather than by affecting moisture or nutrient availability. An advantage of buddleia for native successive species is that it concentrates phosphorus in the soil, possibly due to higher root uptake of phosphorus, translocation from deeper in the profile or mycorrhizal associations (Bellingham et al. 2005). Bellingham et al. (2005) also found that the presence of buddleia had no effect on native or exotic plant diversity in their plots, but they did not measure biomass of plants other than that of buddleia and *Coriaria arborea*.

While there is limited information on the effects of weed species on native regeneration and pine plantation growth, there is a dearth of knowledge on the competition between mānuka and the weed species discussed here. This is probably a result of the long-held belief that mānuka is itself a weed species (Grant 1967) and its use as a plantation crop is very recent. Grant (1967) found that establishment of mānuka is favoured by low, open cover and reduced by low light and competition from other plants. It is also recommended that regular weed control occur before and after planting mānuka (Millner et al. 2016). Watt et al. (2012b) estimated the effect of the abundance of weed species present during mānuka/kānuka stand establishment and found that as the abundance of weeds increased, the productivity of mānuka/kānuka decreased. Further assessment of the impact of weed prevalence on mānuka/kānuka growth performance is required.

Peter and Harrington (2012) found that in American post-harvest forests, when coverage of exotic weeds increased by 4%, native plant cover decreased by 10%, regardless of site. Therefore, a small increase in exotic weed abundance post-harvest may have a larger impact on native regeneration. However, they also found that plant species diversity post-harvest decreased at the rate of one species per 10% decrease in total plant cover. Therefore, in cases of passive natural regeneration, there will need to be a site assessment of the benefits of weed control versus tree biodiversity.

Watt et al. (2008) suggested that weeds are likely to have greater growth responses to climate change than target tree crops. The effect of weed competition on erosion mitigation by the target crop species will probably be dependent on the root architecture of the weed species, for which there is little information. The potential financial loss associated with a longer rotation length (pine), delayed flowering (mānuka) or native forest establishment (e.g. ETS income) may also be negated by reduced weed control costs, but further assessment is required.

4.4 Plant pathogens under climate change

Tree productivity can be hindered by pathogen infections, and the abundance and spread of pathogens can be affected by climate change, and could potentially effect erosion mitigation in future climates by hindering tree survival or growth. In the following section we will assess the most prevalent pathogens in each of our target vegetation scenarios of *P. radiata* plantation, mānuka plantation and native forest.

4.4.1 Pinus radiata

There are four main diseases affecting *P. radiata* in New Zealand: *Dothistroma* needle blight, red needle cast, physiological needle blight, and *Cyclaneusma* needle cast. Perhaps the most devastating of these diseases is *Dothistroma* needle blight, caused by the fungi *Dothistroma septosporum* and *Dothistroma pini* (Bulman et al. 2008; Sturrock et al. 2011; Watt et al. 2011a; Watt et al. 2011b). This disease causes red bands on the needles, leading to defoliation, which starts at the base of the tree and continues upwards, leading to death in extreme cases (Bulman et al. 2008; Watt et al. 2011a).

Radiata pine is particularly susceptible to *Dothistroma*, and the industry loses about NZ$19–24 million per year due to reduced growth performance and treatment costs (Watt et al. 2011a, 2011b). Watt et al. (2011a) indicated that *Dothistroma* needle blight is already present in the Gisborne District, but ranges between 0 and 12 for disease severity (where the maximum disease severity is >18). Watt et al. (2011b) found that *Dothistroma* needle blight has the potential to occur in all pine forests under future climate scenarios, but they stated that the need for finer-scale spatial modelling of this pathogen inhibited clarity of disease severity. The reproduction, spread, infection and survival of *Dothistroma* are directly affected by temperature and moisture (Sturrock et al. 2011), and Woods et al. (2005) found a clear link between climate-change-induced increases in summer rainfall and *Dothistroma* infection and expression in Canadian forests. Bulman et al. (2008) also found an increase in *Dothistroma* needle blight when summer rainfall was higher than usual. The Gisborne District is forecast to become warmer, with increased summer precipitation (Ministry for the Environment 2016), which may lead to an increase in the prevalence of *Dothistroma* needle blight in the future.

Red needle cast is caused by infection with *Phytophthora pluvialis* (Ganley et al. 2014) and is abundant in the Gisborne District despite being introduced to New Zealand relatively recently (Ganley et al. 2014; New Zealand Farm Forestry Association 2017). *Phytophthora* belongs to the kingdom *Chromista* and is not fungal in origin. It causes the premature loss of older needles, leaving the younger needles unaffected (Ganley et al. 2014; Scott & Williams 2014). A decrease in tree production occurs only when there is repeated defoliation of a tree, and defoliation is unlikely to occur annually, and more like every 2–3 years (Ganley et al. 2014). The spread and occurrence of red needle cast is likely to be affected by temperature, moisture and leaf wetness (Ganley et al. 2014). Dick et al. (2014) inferred that red needle cast is more prevalent in areas of high winter rainfall. Under climate change scenarios for the Gisborne District, a decrease in winter rainfall has been forecast (Ministry for the Environment 2016), and therefore the abundance of this disease may be reduced in pine forests in this area in the future.

Physiological needle blight causes localised but severe defoliation of pine trees, generally in trees over 15 years old (Bulman et al. 2008). However, this disorder does not appear to be a direct outcome of bacterial or fungal infection but is in response to climate factors that alter the regulation of needle water content (Gould et al. 2008). Rapid and large reductions in relative humidity leading to a decrease in needle water content in combination with a loss of hydraulic transport of water from roots to needles is the most likely cause of needle death (Gould et al. 2008). Physiological needle blight has been reported to occur in the Gisborne District (e.g. Bulman et al. 2008; Gould et al. 2008). The relative humidity in New Zealand will decrease country wide under climate change, with the exception of the West Coast during winter (Ministry for the Environment 2016). However, how this will affect sudden large changes in relative humidity (which contribute to physiological needle blight) has not been established.

Cyclaneusma needle cast is a fungal disease that leads to premature needle cast, which in turn leads to a decrease in wood production in pine forests. The Gisborne area was estimated to lose NZ$4.1 million in 2005 due to *Cyclaneusma* (Bulman & Dadgil 2001; Bulman 2009). In areas of higher autumn rainfall there is an increase in the expression of *Cyclaneusma* needle cast (Bulman & Dadgil 2001; Bulman 2009), and the Gisborne District and central North Island are the most affected regions under the current climate (Watt et al. 2012a). In future climate projections for the Gisborne District, *Cyclaneusma* needle cast appears to be relatively unaffected in its prevalence (Watt et al. 2012a).

The retention of harvest debris may increase the possibility of pathogens re-infecting a post-harvest tree crop. For example, Oblinger et al. (2011) found that harvest debris hosted *Diplodia* shoot blight in red pine (*Pinus resinosa*) up to 5 years after harvest. The blight was transferred to seedlings planted in the harvest debris, affecting their growth, survival and establishment. There was no information in the literature as to whether diseases of exotic pine trees would be transferable to New Zealand native conifers or other native trees.

4.4.2 **Mānuka**

There are two main diseases that currently affect mānuka in New Zealand: myrtle rust and mānuka blight. Recent infections of trees in the North Island with myrtle rust have led to concerns for the potential effects of this disease on mānuka and other members of the *Myrtaceae* family in New Zealand. Myrtle rust is the disease expression of infection with *Austropuccinia psidii* (formerly *Puccinia psidii*), which is a fungal infection that affects foliage, stems (and flowers and fruit in some species), leading to decreases in reproduction and regeneration and possible death in affected plants (Morin et al. 2012). Myrtle rust is a globally distributed pathogen, with infections reported in Hawaii, Florida, Mexico, Asia, Australia, New Caledonia and Africa (Hood 2016).

The Gisborne District has been identified under current climate regimes to be ‘suitable’ or ‘optimal’ for myrtle rust, and this area has been forecast to remain susceptible to myrtle rust under future climate scenarios (Ramsfield et al. 2010). However, little is known about how myrtle rust will affect mānuka in New Zealand (Ramsfield et al. 2010; Hood 2016). Media releases from Comvita have stated that the honey industry is under little threat in New Zealand because Australia has not found infection of their *Leptospermum* (Peters 2017); however, mānuka in the Kermadec Islands has shown susceptibility to the pathogen (Hood 2016).

However, evidence of pathogenic effects on species in other countries may not reflect what will happen on New Zealand native mānuka (Hood 2016). Simpson et al. (2006) found that myrtle rust had a greater effect on trees in areas that had not previously been exposed to the pathogen compared to areas that had a history of myrtle rust infections. New Zealand has not been infected with this pathogen previously, so our native trees may be more susceptible on first contact with the disease. Myrtle rust is spread by wind (Westaway 2016), and in the Gisborne District increased windiness is expected, so there is a possibility that the spread of this pathogen may be accelerated under climate change.

Mānuka blight is a disease that results from the combination of scale insect (*Eriococcus orariensis*) and fungal (*Capnodium walteri* Sacc.) infestation (Mulcock 1959). The scale insects feed on the sap within the shrub and excrete honeydew, which feeds sooty mould infestations (Mulcock 1959). The disease was first reported in New Zealand in the 1930s, after the accidental introduction of the scale, and contributed to widespread death of mānuka between 1950 and the 1960s (Zondag 1977; Johnson & Mayeux 1992; van Epenhuijsen et al. 2000). It has been suggested that scale insect infestation leads to nutrient deficiency and death of mānuka, rather than reduced photosynthesis from sooty mould infestation (van Epenhuijsen et al. 2000).

Mānuka blight was present to a limited extent in the Gisborne District in the 1950s, but by the late 1990s the disease was no longer present in this area (van Epenhuijsen et al. 2000). This decrease in the spread of mānuka blight may have been due to a decrease in the spread of *Eriococcus orariensis,* which has been replaced by other scale insects (mostly *Ericoccus leptospermii* Maskell). These still feed sooty mould, but do not lead to the death of the shrub (van Epenhuijsen et al. 2000). *Eriococcus orariensis* is predated on by the fungus *Myrangium thwaitessi,* which can also affect the occurrence of mānuka blight. The greatest reduction in scale numbers was in areas of high humidity in the North Island (Hoy 1961, in Grant 1966). The effect of *Myrangium thwaitessi* on *Eriococcus orariensis* was largely ineffectual in the drier areas of the North Island (Hoy 1961, in Johnson & Mayeux 1992). So while the spread of mānuka blight is being curtailed by the species shift to other scale insects, should mānuka blight return to Gisborne District shrublands in the future it may be unaffected by *Myrangium thwaitessi* predation due to the predicted drier conditions.

Kānuka has a greater resistance to mānuka blight compared to mānuka, and kānuka can dominate in infected mixed mānuka-kānuka shrublands as mānuka may die due to the blight (Burrell 1965). Mānuka blight was found to be common in stands surveyed in the Gisborne District, and, as in other areas, it is clear that this and other factors have a role in determining the relative abundance of mānuka and kānuka in the region (Bergin et al. 1995). No studies assessing the effect of climate change on mānuka blight could be located.

Naturally regenerated mānuka is highly variable in leaf size and flowering characteristics (Ronghua et al. 1984), and it is possible that this variation is a reflection of adaptability of the mānuka species (Gratani 2014). However, high-performance mānuka is bred to be more consistent between individuals and may not be as adaptable as naturally occurring mānuka due to a narrowing of phenotypic expression. Despite indications from members of the Primary Growth Partnership project High Performance Mānuka Plantations that disease resistance of mānuka would be assessed as part of the project (Burke 2013), it does not appear this was undertaken.

4.4.3 Native trees

There are two main pathogens that effect native forests: myrtle rust and *Phytophthora*. Myrtle rust is unusual in that it has a large range of potential hosts, many of which are New Zealand native species, and it presents a serious threat to our native forests (Morin et al. 2012; Hood 2016). Myrtle rust can affect the diversity of our native systems by causing the death of *Myrtaceae* species and inhibition of reproduction, which may have flow-on effects on food stocks for birds and insects (Hood 2016). Comprehensive lists of native species that may be susceptible to myrtle rust have been produced by Hood (2016) and Ramsfield et al. (2010). In this section we will focus on *Phytophthora,* as myrtle rust has been discussed in the previous section.

Phytophthora species are members of a broad group of *Chromista*, which occur in both exotic and native forests in New Zealand (Scott & Williams 2014). *Phytophthora* are a very successful group of organisms, which easily disperse, have a wide range of hosts, rapidly adapt to form new pathogens and diseases, and are difficult to identify (Scott & Williams 2014). While *Phytophthora* have been identified to be present in New Zealand, many are not associated with disease expression in plants; *Phytophthora* associated with disease in New Zealand have been comprehensively described by Scott and Williams (2014) and amended by Studholme et al. (2016). *Phytophthora pluvialis*, which causes red needle cast, has already been discussed, but there are several other *Phytophthora* that specifically attack New Zealand native species.

The *Phytophthora* species that has been most notorious in recent history is *Phytophthora agathidicida*, which is responsible for kauri (*Agathis australis*) dieback (Studholme et al. 2016). Kauri does not naturally occur in the Gisborne District (TerraNature Trust 2004), so kauri dieback is not a current issue for this area. However, with climate change and warming it is possible that kauri may spread into this region and would need to be assessed in the future.

Phytophthora cinnamomi is perhaps one of the most invasive *Phytophthora* and is widely spread in New Zealand, causing isolated outbreaks of root rot (Podger & Newhook 1971). *Phytophthora cinnamomi* is found in both exotic and native forests (Scott & Williams 2014), and Podger and Newhook (1971) reported severe damage in Auckland (kauri forest), Kaiteretere (mānuka-dominated shrubland), Nelson (*Coriaria sarmentora*) and Harihari (rimu *Dacrydium cupressinum*) forests. In these cases, acute damage may have been associated with eroded soils (which can have impaired root regeneration) and high summer rainfall (Podger & Newhook 1971). Further, Johnston et al. (2001) have stated that *Phytophthora cinnamomi* tends to be more prevalent when a warmer, drier winter is followed by conditions that stress trees, such as drought. Many areas within the Gisborne District have eroded soils, including under forestry established post-Bola, and this region is forecast to undergo increased summer rainfall. Therefore the prevalence of *Phytophthora* is likely to increase in the Gisborne District with climate change due to a combination of warmer, drier winters, increased risk of droughts, and eroded soils.

4.5 Pest management under climate change

One of the key issues that arose during assessments of the survival of newly planted trees was the control of pest species, in particular, possums, deer, pigs, rabbits, goats and black beetle (e.g. MacIntyre 2017). Pest grazing can also inhibit natural succession from mānuka/kānuka shrubland to broadleaf forest (Bergin et al. 1995). In this section we discuss the effects of common exotic pests on plantation forests, mānuka plantations and native forests.

4.5.1 Possums

The Australian brush-tail possum (*Trichosurus vulpecula* Kerr) is a serious threat to biodiversity in both native and exotic forests (McGlone & Walker 2011). In pine plantations possums strip the bark of young trees and browse newly planted seedlings (Clout 1977), and they also consume pine pollen cones in late winter and spring (Clout 1977; Warburton 1978). While literature has indicated that possums can cause substantial damage to young seedlings, with up to 50% loss, other authors have not found a significant effect of possums on pine trees (Warburton 1978; Jacometti et al. 1997). Interestingly, the main diet of possums in plantation forests actually consists of food sources from other species, predominantly broom foliage and flowers, and gorse (Clout 1977; Warburton 1978).

Possums contribute to the alteration of tree diversity and a shift to less palatable or browse-resistant plants in established native forests, although the extent of this effect is highly variable between sites and between years (McGlone & Walker 2011). Possums generally browse native seedlings greater than 10 cm in height, but the effect of possums on seedling survival is highly variable and depends on the density of possum population and tree species (Wilson et al. 2003). Possums may also enhance the spread of native trees within naturally regenerating forest by seed dispersal (Wilson et al. 2003). Within native forests, elevated CO_2 concentrations can alter the palatability of leaves, although the degree to which this occurs differs between species (McGlone & Walker 2011). However, whether the change in fodder quality increases or decreases possum grazing pressure remains to be determined. No published literature on the effects of possums on mānuka was found.

Possums are widespread throughout New Zealand (McGlone & Walker 2011). The Gisborne District is predicted to experience drier, warmer winters under climate change, which may extend the breeding season of possums, potentially increasing population size or allowing quicker recovery of numbers after an eradication programme (McGlone & Walker 2011). However, the effect of possums on forests will be confounded by other factors, including soil fertility and vegetation cover (McGlone & Walker 2011).

4.5.2 Deer, pigs, goats and livestock

Deer are common in pine plantations and contribute to tree damage by browsing new growth, bark stripping and fraying, and uprooting small seedlings (Gill 1992). Information on the potential distribution of deer under predicted climate change in New Zealand could not be located. However, in Canada there has been an increase in the abundance and range of white-tailed deer (*Odocoileus virginianus*) in response to climate change (Dawe & Boutin 2016). In Europe, the roe deer (*Capreolus capreolus)* expansion has been moderated by increased death of fawns due to temporal displacement between food supply and breeding season under climate change (Plard et al. 2014). Therefore, the effect of climate change on the species of deer present in New Zealand remains unknown and requires further investigation to determine their effect on forest performance within our land-use scenarios.

While the impact of deer on conifer forests has been studied overseas (e.g. Gill 1992), there is little information on their effect in New Zealand. Nugent et al. (2001) found that deer contribute to the near defoliation of their preferred fodder species in native forests, which can lead to a profound impact on tree diversity. The grazing pressure exerted by deer is affected by the provision of food sources other than foliage and therefore reflects an inherent variability between sites and over time (Nugent et al. 2001). Intensive removal of deer from native forest leads to much greater growth of beech seedlings, and the extent of the negative effect of deer on beech forest is dependent on the species of deer (Husheer & Robertson 2005; Husheer et al. 2006). There is little information on the effect of deer on mānuka, with the exception of that found by Smale et al. (1995), who suggested that deer browsing slows the progression from kānuka shrubland to secondary forest. As found for possum herbivory, the effect of deer browsing on native trees is highly site and time specific, and therefore it is difficult to determine any clear patterns (McGlone & Walker 2011).

Pigs were introduced into New Zealand in the late 1700s and inhabit mostly pastoral systems, but are also prevalent in native and exotic forests (Clark & Dzieciolowski 1991). Pigs damage trees by digging up seedlings and rubbing on larger trees (Greater Wellington Regional Council 2012; Marlborough District Council et al. 2016), possibly leading to decreased stem densities in forests. As found for possums, it is expected that warmer and drier winters will increase the breeding season for pigs (McGlone & Walker 2011), potentially increasing their populations in the Gisborne area under climate change.

Goats also alter understorey composition and prevalence and can damage seedlings in both exotic and native forests (McGlone & Walker 2011; Greater Wellington Regional Council 2012). As found for pigs and possums, the extension of the breeding season may be expanded for goats in the Gisborne District due to increased temperatures and decreased rainfall during winters (McGlone & Walker 2011). Therefore, regardless of the land-use scenario within this report, the control of feral goats during seedling establishment may require further investment. Although grazing of indigenous shrubland by cattle, sheep, or goats may favour the initial establishment of mānuka/kānuka shrubland, progression to secondary succession is likely to be slowed compared to sites with stock exclusion (Bergin et al. 1995).

The degradation of the remaining areas of indigenous forest vegetation through the sustained browsing of the canopy and understorey by recently introduced mammals (Wallis 1966; James 1969), and the consequent deterioration of forest canopy by wind damage in the early 1980s (Littlejohn 1984), has been significant. Areas of natural forest in Gisborne District are largely confined to the Raukūmara Range. The interaction between pest introduction, forest degradation, topography and lithology increases erosion in native forests in the Gisborne District (McKelvey 1960), particularly gullying and associated rotational slumping (Marden et al. 2012).

4.5.3 Hares and rabbits

Hares were introduced to New Zealand in the mid- to late 1800s for sport hunting and food provision (Hawke's Bay Regional Council 2003; National Pest Control Agencies 2015). Hares affect newly planted trees by browsing (National Pest Control Agencies 2015), which can inhibit the growth performance and survival of trees. In some cases the damage to seedling trees can be substantial, with a hare able to damage as many as 50 trees in one night (Hawke's Bay Regional Council 2003). In native forests, rabbits contribute (in combination with other mammalian browsers) to reducing the prevalence of palatable plant species, as do deer (McGlone & Walker 2011). Rabbits will be favoured by warmer, drier weather as predicted for the Gisborne District (McGlone & Walker 2011), and therefore it is likely that hares will become more prevalent in the Gisborne District and may require greater control measures in the future.

4.5.4 Black beetle

Black pine beetle (*Hylastes ater)* was first detected in New Zealand in 1929 and contributes to tree damage and death in production forests (Reay et al. 2012). The beetle breeds under the bark of harvest debris (including tree stumps), feeding on phloem and forming egg galleries in the wood (Reay et al. 2012). Upon adult emergence from the host debris, the beetles feed on seedlings and potentially ring-bark trees under the soil surface (Reay et al. 2012). Black beetle infection is not normally lethal, although it can kill seedlings and contribute to infection by pathogens due to open wounds underground (Reay & Walsh 2002; Reay et al. 2012). Because these beetles are prevalent in pine forests during establishment of second and third rotations, they pose a potential threat to pine trees should they be replanted. The prevalence of black pine beetle can be increased after storm events where windthrow and stem snap have occurred due to the large increase in woody debris (Albrecht et al. 2009).

There has been no specific assessment of black pine beetle under climate change. However, as the Gisborne District will undergo increased storm intensity and windiness, and higher temperatures, these factors are likely to contribute to increased prevalence of this pest. Increased wind and storms will increase tree debris on forest floors and provide habitat for black beetle. However, since the main impact occurs in the year after planting, this effect could be short lived, but has the potential to decrease seedling survival (Albrecht et al. 2009). Elevated temperatures in the USA have led to the increased spread of mountain pine beetle (*Dendroctonus ponderosae*) and caused a shift from one to two breeding seasons per year (Mitton & Ferrenberg 2012). However, Bentz et al. (2010) found that the effect of warming on pine bark beetles was highly variable, both within and between species. Therefore, the effects of climate change on black pine beetle and the flow-on effects on pine production in New Zealand remain unclear.

5 Revenue sources within land-use options

There are several government funds that support increasing forest cover in this country, including the Afforestation Grant Scheme, the Erosion Control Funding Programme, the Permanent Forest Sink Initiative, the New Zealand Emissions Trading Scheme, and the new Billion Trees initiative. However, not all of these funding options support the conversion of pine plantation forestry to other forest covers, and this land-use change would not be eligible for the Afforestation Grant Scheme or the Erosion Control Funding Programme. The Billion Trees initiative already includes the baseline planting by the forestry industry, but the main goal is to convert more land to trees (Ministry for Primary Industries 2018b), so it largely sits outside the scope of this report. The Afforestation Grant Scheme is only available for land-use change from unforested land to forested land. The Erosion Control Forestry Programme will not be an option for this land-use change either, as eligible land must have had no forest (either exotic or native) on it since 1989 (Ministry for Primary Industries 2017d). However, land-use change from pines to native forest or plantation mānuka would most likely be eligible for funding from the Permanent Forest Sink Initiative and the New Zealand Emissions Trading Scheme under certain management strategies.

5.1 The Permanent Forest Sink Initiative

Within the scope of this report, the Permanent Forest Sink Initiative (PFSI) is an option to continue an income from land after conversion from plantation pines to natural regenerated native forest. This initiative does not confer any actual funding for the conversion from pines to native regeneration, and these expenses remain at the cost of the land owner. The PFSI does, however, allow land owners to receive carbon credits for their permanent forest, which can be on-sold (Review Panel 2011; Orme 2015).

Eligibility for the PFSI is determined by the following.

1. Forests must be established after 1 January 1990.
2. The plantation must be induced by human action (e.g. planted, seeded or facilitation of natural regeneration).

3. The forest must not consist of 5 or more hectares of land that was cleared containing naturally occurring indigenous trees on or after 1 December 2007.
4. The covenant with the New Zealand Government must be entered for 99 years. The covenant can be ended after 50 years, but in that case all carbon credits obtained from the forest will have to be repaid to the government.
5. Evidence of land eligibility must be provided, such as aerial photos taken around 1990, planting records or contractor invoices.

The PFSI allows for the accumulation of ‘assigned amount units’ (AAUs), and one AAU is equivalent to 1 tonne of carbon stock increase (or decrease) once the regenerating forest reaches more than 30% of Crown cover per hectare (Ministry of Agriculture and Forestry 2011). After this time, AAUs earned by the forest will increase with time as the trees grow (Ministry for Primary Industries 2017a). However, after pine harvest, the carbon credits for new permanent forest blocks would be reduced for a period of time after conversion due to CO_2 emissions from the degradation of pine roots and litter during establishment of the new forest (Ministry for Primary Industries 2017a). There is also a fee for claiming the AAUs produced by the land, and the Ministry holds the AAUs until they are applied for by the land owner (Ministry of Agriculture and Forestry 2011). The PFSI allows for harvesting of some trees within these registered permanent forests, although at a small scale (Review Panel 2011; Orme 2015).

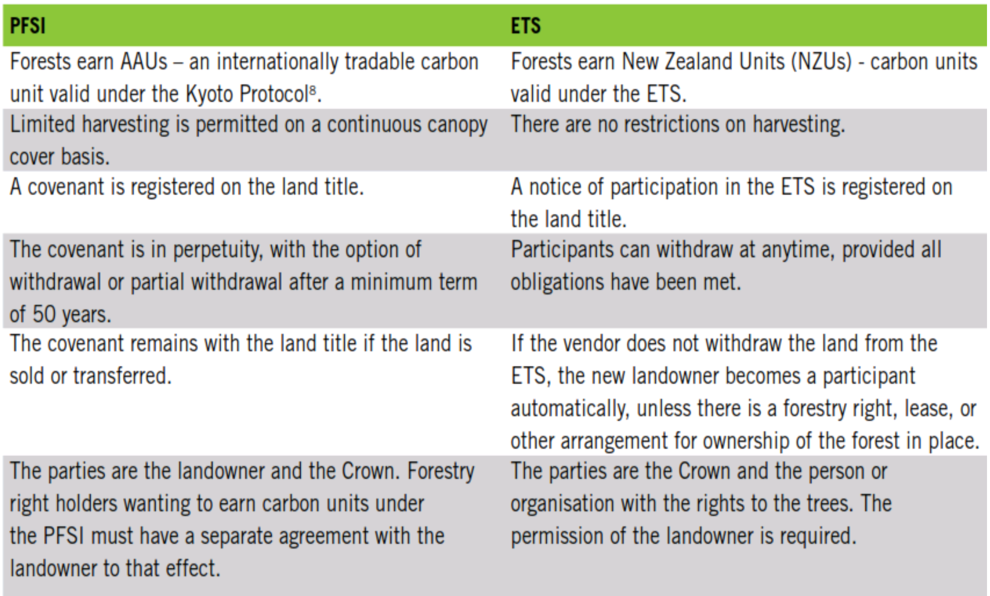
As mānuka plantations are likely to be harvested and replanted over time to ensure a steady supply of honey, it is unlikely that the PFSI will be a viable option after land-use change from plantation pine to plantation mānuka.

5.2 New Zealand Emissions Trading Scheme

The New Zealand Emissions Trading Scheme (ETS) may also be an option for income following conversion from pine plantations to natural reversion and remaining in plantation forestry. The ETS runs in parallel with the PFSI but is more flexible and can be entered or left at any time, and the eligibility of blocks is more clearly defined. The ETS has units of trade called New Zealand Units (NZUs), which are also knowns as carbon credits, where each credit is equivalent to 1 tonne of CO_2 (Ministry for Primary Industries 2015). Forest owners can voluntarily opt into the scheme if their forests were planted after 31 December 1989; when non-exempt forests that had been planted before 1 January 1990 are harvested, they mandatorily become part of the ETS. Importantly, carbon credits can only be claimed for the period for which the forest is registered with the ETS and cannot be backdated to when the forest was planted. Further, if forest cover is decreased due to harvesting or fire, then the land owner will be required to pay the government for any decrease in forest mass (Ministry for Primary Industries 2015). In some post-1990 forests, land owners may not be responsible for carbon credit costs associated with deforestation, particularly where they have no legal rights over whether the forest is removed (Ministry for Primary Industries 2015).

In situations where forested land is cleared but is likely to meet the criteria for forested land again, it can be considered ‘temporarily unstocked’. Pine forests harvested and left to naturally regenerate would fit the temporarily unstocked definition of forest. However, if regeneration is slow or inconsistent within a site, then this will delay the time until the site is considered a forest once more, and therefore when it is eligible to earn carbon credits (Ministry for Primary Industries 2015). As for the PFSI, the ETS considers forested land to have more than 30% of crown cover in each hectare or is likely to have an average crown cover width greater than 30 m (Ministry for Primary Industries 2015). There are some fundamental differences between the PFSI and the ETS, as outlined in Table 5.

Table 5. Differences between the Permanent Forest Sink Initiative and the New Zealand Emissions Trading Scheme (Ministry of Agriculture and Forestry 2011)



PFSI	**ETS**
Forests earn AAUs – an internationally tradable carbon unit valid under the Kyoto Protocol[8].	Forests earn New Zealand Units (NZUs) - carbon units valid under the ETS.
Limited harvesting is permitted on a continuous canopy cover basis.	There are no restrictions on harvesting.
A covenant is registered on the land title.	A notice of participation in the ETS is registered on the land title.
The covenant is in perpetuity, with the option of withdrawal or partial withdrawal after a minimum term of 50 years.	Participants can withdraw at anytime, provided all obligations have been met.
The covenant remains with the land title if the land is sold or transferred.	If the vendor does not withdraw the land from the ETS, the new landowner becomes a participant automatically, unless there is a forestry right, lease, or other arrangement for ownership of the forest in place.
The parties are the landowner and the Crown. Forestry right holders wanting to earn carbon units under the PFSI must have a separate agreement with the landowner to that effect.	The parties are the Crown and the person or organisation with the rights to the trees. The permission of the landowner is required.

The length of time needed for native regeneration to reach the point of income from the ETS or PFSI will probably be longer under native regeneration than for plantation forestry due to the slower growth of native species. Mason and Morgenroth (2017) modelled the different CO_2 sequestration rates for forestry crops in New Zealand, assuming 50,000 ha were planted for 26 years (Figure 8). The ‘Radiata pine plant and leave’ treatment was assumed to have a planted density of 1,000 stems per hectare, and the ‘Radiata pine pruned’ treatment was managed with three prune lifts and thinning to a final stocking rate of 300 stems per hectare. Their work indicated that pine trees planted and then left permanently may be the best option for maximising income from the ETS, followed by commercially managed normal rotation pine forestry, and then native forests. It is unclear from Mason and Morgenroth (2017) whether native forest is planted with thinning for commercial wood production or a permanent forest with no commercial management.

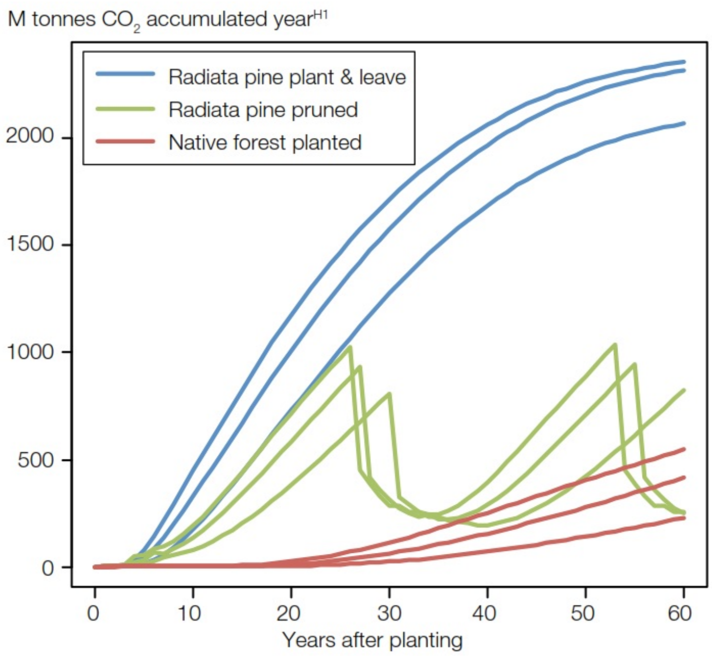


Figure 8. Carbon dioxide accumulated over 60 years in three land uses: radiata pine planted and left without harvest or pruning, radiata pine with harvest rotations and pruning, and planted native forest (Mason & Morgenroth 2017). The three sets of lines for each land-use option indicate low, medium, and high productivity.

For our land-use change option of converting pine plantations to mānuka plantations, the returns for carbon credits generated in mānuka plantations will be significantly less than those generated in a pine forest due to the different forms of the trees and a much greater amount of wood produced by pine trees (MacIntyre 2017). Although mānuka honey production is considered compatible with carbon farming, mānuka that is regularly cleared and replanted would not fit the definition of a 'forest' under the ETS and would not be eligible for carbon credits (Funk 2009). Further, the income from the ETS is provisional on mānuka plantations having a sufficiently long rotation to allow for the stands to reach a height of 5 m. Boffa Miskell Limited (2017) state that the mature height of mānuka varies between 4 and 8 m, and so if plantations are cleared and replanted on a regular basis (15 years) they may not reach the qualifying height and may generate little income from the ETS. However, the core income from plantation mānuka is likely to be honey production anyway.

5.3 Biodiversity protection subsidies

In the Gisborne District there may also be options for income after conversion of pine plantations to native forest via biodiversity protection subsidies. Biodiversity protection funds for moderate to high conservation value land include the Queen Elizabeth II Fund (QE2), Ngā Whenua Rāhui, the Biodiversity Condition Fund, and the National Heritage Fund, all of which are administered by the Department of Conservation (DoC). However, these funds would only be available once native regeneration has already taken place (Funk 2009), but they may offer future financial return following land-use change from pine plantations to native forest after about 50 years.

5.4 Honey production from plantation **mānuka**

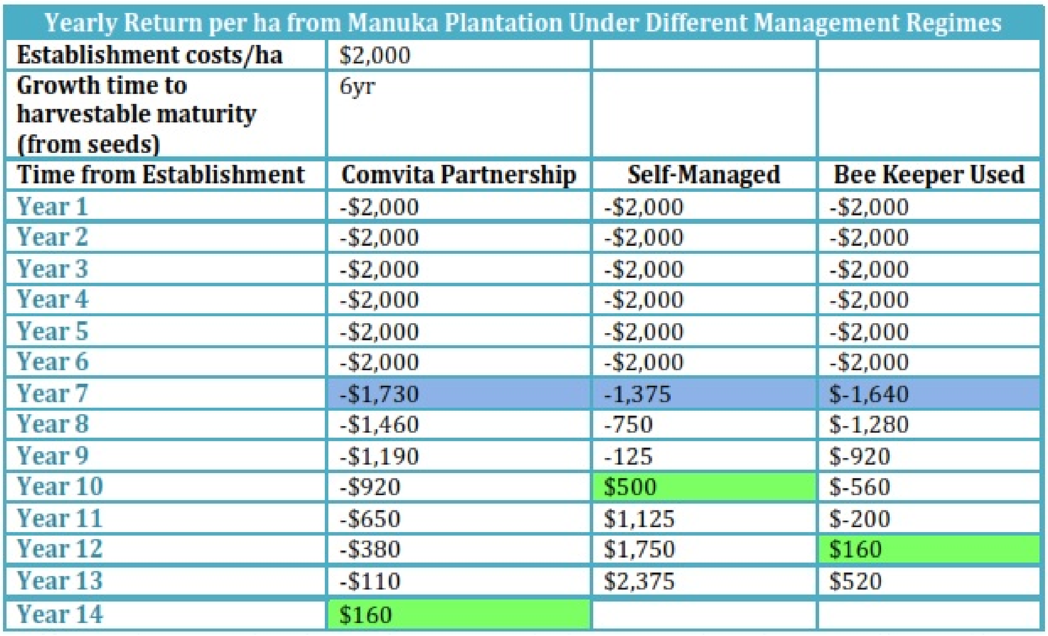
The mānuka honey industry in New Zealand is still undergoing development, and estimates of profits within the industry have varied over time and are an important factor in determining whether conversion of plantation pine to plantation mānuka for honey production is a viable option. In the following section we have identified the possible costs and benefits of establishing plantation mānuka, with particular reference to the partnership between land owners and Comvita.

Mānuka plantations are commonly established at 1,100 stems per hectare (MacIntyre 2017) at a cost of about NZ$2,000/ha using the ‘high performance’ mānuka trees available through the Comvita partnership (Burke 2013, 2015; Edlin & Duncan 2013). Comvita also offer technical, start-up, and honey harvesting assistance in return for 70% of the revenue for 10 years (Burke 2013, 2015). The advantage of this partnership is that any lack of knowledge or experience of the land owner about honey production is supplied by Comvita, but the revenue to the land owner is also greatly reduced over the 10-year period of commitment. High-performance mānuka seedlings offered by Comvita may or may not be eco-sourced (Burke 2013), and for those land owners (particularly iwi groups) that support eco-sourcing this would have to be established with Comvita if considering entering a partnership with them. MacIntyre (2017) suggests that eco-sourcing will be beneficial for the plantation mānuka industry because it will improve the survival of seedlings.

Plantation mānuka is likely to produce nectar 7–8 years after planting and may not reach its full potential for honey production until 13 years after plantation establishment (MacIntyre 2017). This differs from the view of Boffa Miskell Limited (2017), who state that maximum honey production occurs about 6 years after planting and flowering would naturally decrease between years 9 and 10 after planting. Canopy closure has been predicted to occur between 6 and 8 years after mānuka planting (Marden & Lambie 2015), but the effect of canopy closure on honey production has not been established. None the less, these figures indicate a limited window for honey profits unless plantations are managed to maintain greater flowering for longer periods (Boffa Miskell Limited 2017).

There are some indications that there are considerably greater profits to be gained from using high-performance trees developed by the Mānuka Research Partnership and Comvita within the Primary Growth Partnership project (Burke 2013). However, Edlin and Duncan (2013) indicate that there will be a delay in a return on investment, which is dependent on the management regime of mānuka plantations (Table 6).

Table 6. **Return on investment after mānuka plantation establish**ment under three different management regimes (Edlin & Duncan 2013). These regimes are estimated using $2,000/ha for high-performance trees. The blue cells indicate when revenue starts to be generated, and the green cells indicate when profit is generated. The authors estimated a return under a Comvita regime of $270/ha, under self-management of $625/ha, and under self-management using a bee-keeper of $360/ha



Yearly Return per ha from Manuka Plantation Under Different Management Regimes			
Establishment costs/ha	$2,000		
Growth time to harvestable maturity (from seeds)	6yr		
Time from Establishment	**Comvita Partnership**	**Self-Managed**	**Bee Keeper Used**
Year 1	-$2,000	-$2,000	-$2,000
Year 2	-$2,000	-$2,000	-$2,000
Year 3	-$2,000	-$2,000	-$2,000
Year 4	-$2,000	-$2,000	-$2,000
Year 5	-$2,000	-$2,000	-$2,000
Year 6	-$2,000	-$2,000	-$2,000
Year 7	-$1,730	-1,375	$-1,640
Year 8	-$1,460	-750	$-1,280
Year 9	-$1,190	-125	$-920
Year 10	-$920	$500	$-560
Year 11	-$650	$1,125	$-200
Year 12	-$380	$1,750	$160
Year 13	-$110	$2,375	$520
Year 14	$160		

MacIntyre (2017) indicated that conversion to mānuka plantation may only be more profitable than plantation forestry in a few cases. Daigneault et al. (2015) estimated that a mānuka plantation over a 25-year cycle could generate an average profit of $122/ha/yr, although some very productive stands with high unique mānuka factor (UMF) are capable of producing much higher profits (Wetere 2015). This equates to an internal rate of return (IRR[1]) of 9–12%, and other studies, such as Edlin & Duncan 2013, have estimated an IRR of 6.7–14.5% for mānuka plantation. In comparison, pine plantation forestry could generate a mean net return of $680/ha/year, assuming a 28-year rotation (Hock et al. 2014) with an IRR of 14% (Park 2011).

Although the price of honey fluctuates from year to year, the general trend is an increase in price, particularly for mānuka honey (Figure 9). Honey production values for the Gisborne District could not be sourced, but values for the combined areas of Coromandel, Bay of Plenty, Rotorua and Gisborne indicate increased production from 2011 to 2016 (Ministry for Primary Industries 2017b). The production for these combined areas was 3,094 tonnes in the year ending 30 June 2016 and contributed 15% to total honey production in that year for New Zealand (Ministry for Primary Industries 2017b).

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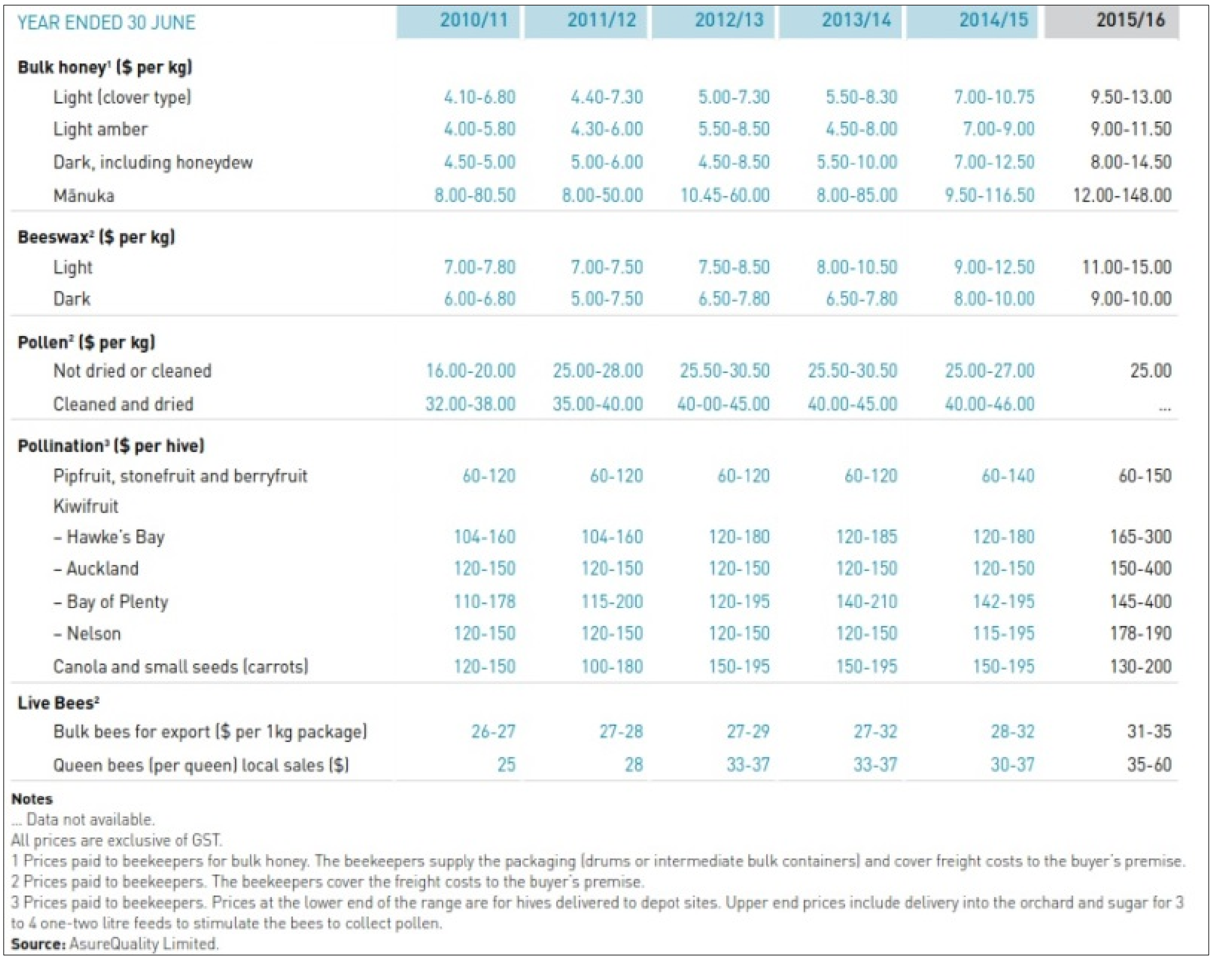
[1] The Internal Rate of Return calculates the interest rate received for an investment, consisting of costs and income that occur over 30 years. By examining the costs, and when they occur, compared with the revenues earned over time, the IRR calculation estimates the return from the project as an interest rate calculation. It is the rate of interest at which the present value of future cash flows is exactly equal to the initial capital investment.



YEAR ENDED 30 JUNE	2010/11	2011/12	2012/13	2013/14	2014/15	2015/16
Bulk honey[1] ($ per kg)						
Light (clover type)	4.10-6.80	4.40-7.30	5.00-7.30	5.50-8.30	7.00-10.75	9.50-13.00
Light amber	4.00-5.80	4.30-6.00	5.50-8.50	4.50-8.00	7.00-9.00	9.00-11.50
Dark, including honeydew	4.50-5.00	5.00-6.00	4.50-8.50	5.50-10.00	7.00-12.50	8.00-14.50
Mānuka	8.00-80.50	8.00-50.00	10.45-60.00	8.00-85.00	9.50-116.50	12.00-148.00
Beeswax[2] ($ per kg)						
Light	7.00-7.80	7.00-7.50	7.50-8.50	8.00-10.50	9.00-12.50	11.00-15.00
Dark	6.00-6.80	5.00-7.50	6.50-7.80	6.50-7.80	8.00-10.00	9.00-10.00
Pollen[2] ($ per kg)						
Not dried or cleaned	16.00-20.00	25.00-28.00	25.50-30.50	25.50-30.50	25.00-27.00	25.00
Cleaned and dried	32.00-38.00	35.00-40.00	40-00-45.00	40.00-45.00	40.00-46.00	...
Pollination[3] ($ per hive)						
Pipfruit, stonefruit and berryfruit	60-120	60-120	60-120	60-120	60-140	60-150
Kiwifruit						
– Hawke's Bay	104-160	104-160	120-180	120-185	120-180	165-300
– Auckland	120-150	120-150	120-150	120-150	120-150	150-400
– Bay of Plenty	110-178	115-200	120-195	140-210	142-195	145-400
– Nelson	120-150	120-150	120-150	120-150	115-195	178-190
Canola and small seeds (carrots)	120-150	100-180	150-195	150-195	150-195	130-200
Live Bees[2]						
Bulk bees for export ($ per 1kg package)	26-27	27-28	27-29	27-32	28-32	31-35
Queen bees (per queen) local sales ($)	25	28	33-37	33-37	30-37	35-60

Notes
... Data not available.
All prices are exclusive of GST.
1 Prices paid to beekeepers for bulk honey. The beekeepers supply the packaging (drums or intermediate bulk containers) and cover freight costs to the buyer's premise.
2 Prices paid to beekeepers. The beekeepers cover the freight costs to the buyer's premise.
3 Prices paid to beekeepers. Prices at the lower end of the range are for hives delivered to depot sites. Upper end prices include delivery into the orchard and sugar for 3 to 4 one-two litre feeds to stimulate the bees to collect pollen.
Source: AsureQuality Limited.

Figure 9. Returns for the honey industry 2011 to 2016 (Ministry for Primary Industries 2017b).

Profits on honey may be reduced by mortality of trees after planting. Marden and Lambie (2015) found that mānuka plantation mortality ranged between 9 and 51% of the planted trees, depending on which part of the landscape they were planted on (e.g. mortality of shrubs on earthflows and landslide scars was generally greater than on more stable landscapes). MacIntyre (2017) suggests that increasing the stand density of mānuka from 1,100 to 1,600 stems per hectare may compensate for tree mortality. Another option would be to replace dead shrubs with new plants, which would require further planting investment after initial planting has been completed.

Any profits from mānuka honey production would be tempered by the offset costs of pest and weed control, which are essential as these plantations become established (MacIntyre 2017). The rotation length of mānuka in natural stands is thought to be between 25 and 30 years, when mānuka is outcompeted for light by secondary forest species and kānuka (Bateson 2014). However, eradication of native seedlings within mānuka plantations could potentially lengthen plantation viability, but would incur labour costs.

Although the extent of flowering can affect honey production, so can other factors that affect the bees directly. For example, a New Zealand bee colony loss survey showed that hive loss rates during 2017 were 10%. These losses were mainly attributed to queen problems (43%), pests (24%) and starvation (10%) (Brown & Robertson 2017). Natural disasters, accidents, and theft/vandalism were significantly less common but also contributed to colony losses.

5.4.1 Site factors

The honey industry is significantly affected by the quality of honey produced, and the greater the UMF of the honey the greater the income (Waikato Regional Council 2016). High Performance Mānuka Plantations, a Primary Growth Partnership (PGP) project, has been investigating issues associated with increasing production of high UMF mānuka honey in New Zealand. The project has found that the combination of correct land, climate, plantation size, and management strategy for high-production mānuka is more difficult than initially thought and may not be suitable for many locations (MacIntyre 2017). In particular, the PGP project concluded that smaller blocks of land put into mānuka plantations may not be profitable (MacIntyre 2017), although the authors did not define what size a ‘smaller block’ was. However, Evision (2016) suggest that the minimum economic mānuka plantation size is 20 ha.

There is a long-held belief that mānuka thrives on low-fertility soils, but MacIntyre (2017) states that high-performance mānuka has better growth performance on higher fertility soils. The soils in the hill country of the Gisborne District are predominately ‘skeletal’, with shallow topsoil, but are moderately fertile (Gibbs 1959). Therefore, soil fertility may not be an issue in the Gisborne District.

Warmer temperatures increase mānuka nectar production, therefore increasing honey profits (MacIntyre 2017). Climate change is predicted to increase temperatures in the Gisborne District, and therefore mānuka plantations in this area may lead to an increase in honey production, assuming there are no limiting factors such as rainfall and nutrition.

Issues relating to the management of mānuka plantations are yet to be resolved, particularly around thinning and harvesting and replanting regimes to maintain honey production. This may be particularly relevant to erosion-prone land, where new legislation may affect mānuka plantation management and should be investigated with the Gisborne District Council and reviewed regularly.

5.5 *Pinus radiata* wood production

As outlined previously in this report, the plantation forestry industry contributes substantially to the economy of New Zealand and has been assessed by Nixon et al. (2017). Approximately 50% of trees harvested are exported as logs, and the remaining half are processed in New Zealand, with most logs and pulp exported to China (Clark 2018; Ministry for Primary Industries & New Zealand Forest Owners Association 2016). The bulk of exports from New Zealand are low-value softwood logs with little high-value processed timber, which is opposite to the trend in competing wood export countries (Clark 2018). The largest proportion of industry profits is from logs and wood chips (Figure 10), and though the domestic log price is relatively stable, the export price shows variability over time (Figure 11) and would contribute to some instability in profits. Further, the New Zealand forestry industry contributes 1.3% of global trade in wood products (Ministry for Primary Industries 2018ba).

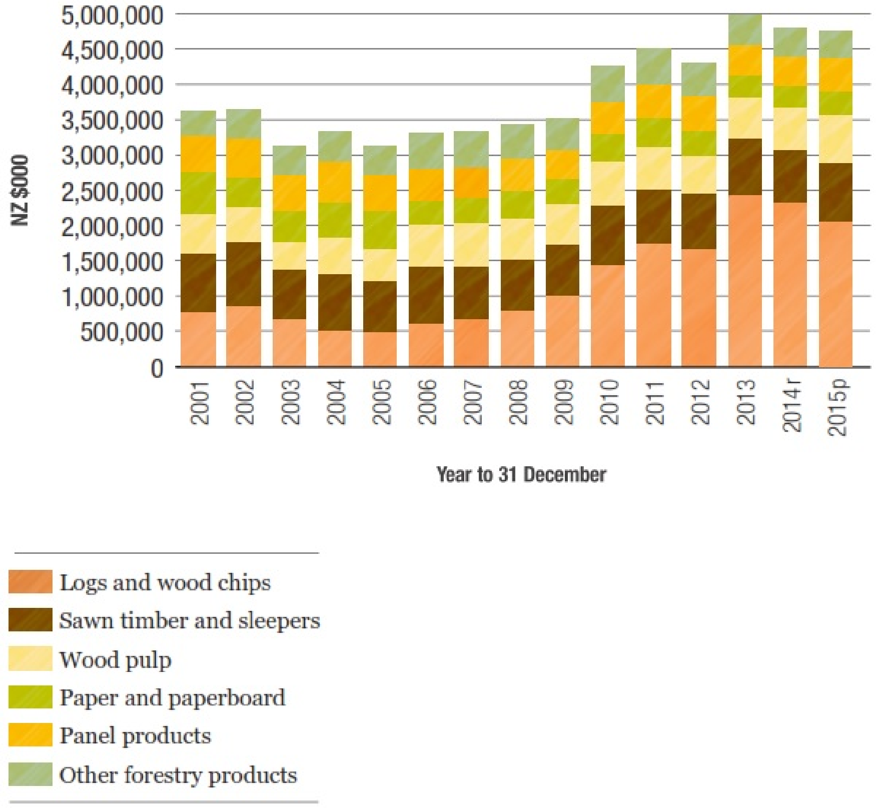


Figure 10. Major exports for the plantation forestry industry in 2016 (Ministry for Primary Industries & New Zealand Forest Owners Association 2016).

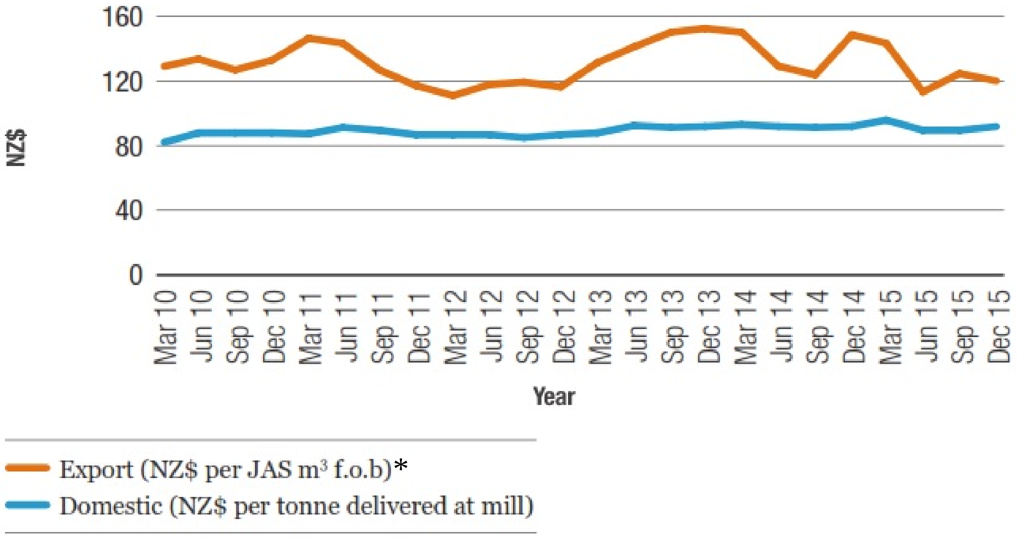


Figure 11. Export and domestic log pricing from 2010 to 2015 (Ministry for Primary Industries & New Zealand Forest Owners Association 2016).* Japanese Agricultural Standard (m³) free on board.

The domestic demand for processed wood is mostly the result of increasing demand due to the Auckland housing crisis and the Christchurch rebuild (Clark 2018). Wood pulp for paper production has suffered some negative impacts due to technological advances and the reduction in the demand for some types of paper, particularly newsprint and stationary (Clark 2018).

Because the wood prices are driven by international supply and demand, profitability within the forestry industry can be governed by production costs, which is reflected in industry innovation towards reducing waste and improving efficiency (Clark 2018). Generally, the New Zealand forestry industry responds to international log prices by increasing harvesting when log prices are high and decreasing harvesting when log prices are low (Clark 2018). However, this has led to over-harvesting in some parts of the country, with decreased future harvesting prospects (Clark 2018).

5.6 Non-market impacts

In addition to the changes in income mentioned above, several other ecosystem service impacts have been monetised in New Zealand. For instance, Jones et al. (2008) detail many of the costs and benefits of erosion, and in a recent analysis Walsh et al. (2017) analysed several afforestation options and monetised several outputs, including changes in carbon and water quality. That analysis had a slightly different focus, in modelling changes from non-forested areas to forest. However, many of the same ecosystem services would be affected by the land-use change scenarios considered in the present report.

Several other projects have explored changes in New Zealand’s ecosystem services as a result of forest-related programmes, including those by McAlpine & Wotton (2009), Ausseil et al. (2013), Greenhalgh & Hart (2015), and Daigneault et al. (2017). There are also many stated and revealed preference studies on non-market valuation of forest-related ecosystem services, as described in Yao et al. (2014). Land-use changes may also affect water quality, and changes in water quality have been valued in several New Zealand studies (e.g. Baskaran et al. 2009).

Changes in carbon represent one of the larger impacts of changes in forestry practices. There are several potential ways to value changes in carbon, with the social cost of carbon (SCC) representing one of the more widely internationally accepted methods (Marten et al. 2015). The SCC is a measure of the present value of the future stream of damages associated with marginal carbon emissions (note that most of SCC’s are global figures, which include international impacts). The ETS system mentioned above has a carbon market associated with it. The price of a tradeable unit of carbon might be used to represent the marginal value of emissions change in New Zealand. For more background on these different approaches, the New Zealand Treasury provides an overview here: http://www.treasury.govt.nz/publications/research-policy/wp/2005/05-02/06.htm.

6 Regulation of activities on erosion-prone land

The severe erosion in the Gisborne District has long been recognised as a serious issue by both local and central government. The Gisborne District Council has implemented rules within the Tairāwhiti Resource Management Plan to mitigate erosion on susceptible land. Further, central government has implemented the recently enacted National Environmental Standards for Production Forestry (NES-FP), which expands on the Resource Management Act 1991. In this section we will outline how these regulations may affect the management of vegetation, including clear-fell harvesting of exotic forest, and shrubland maintenance and clearance with reference to plantation mānuka management. Regulatory controls on the establishment and clearing of woody vegetation are likely to influence decision-making on the practicality and feasibility of land-use change from pine production to plantation mānuka or natural regeneration.

6.1 Resource Management Act 1991

The regulation umbrella under which forestry activities are regulated is the Resource Management Act 1991 (RMA), the purpose of which is ‘to promote the sustainable management of natural and physical resources’. Within the RMA there are two processes – the development of regional and district plans and application for resource consent (Clark 2018) – which have implications for the forestry industry (Table 7).

Table 7. Resource Management Act 1991 implications for the forestry industry in New Zealand (Clark 2018)

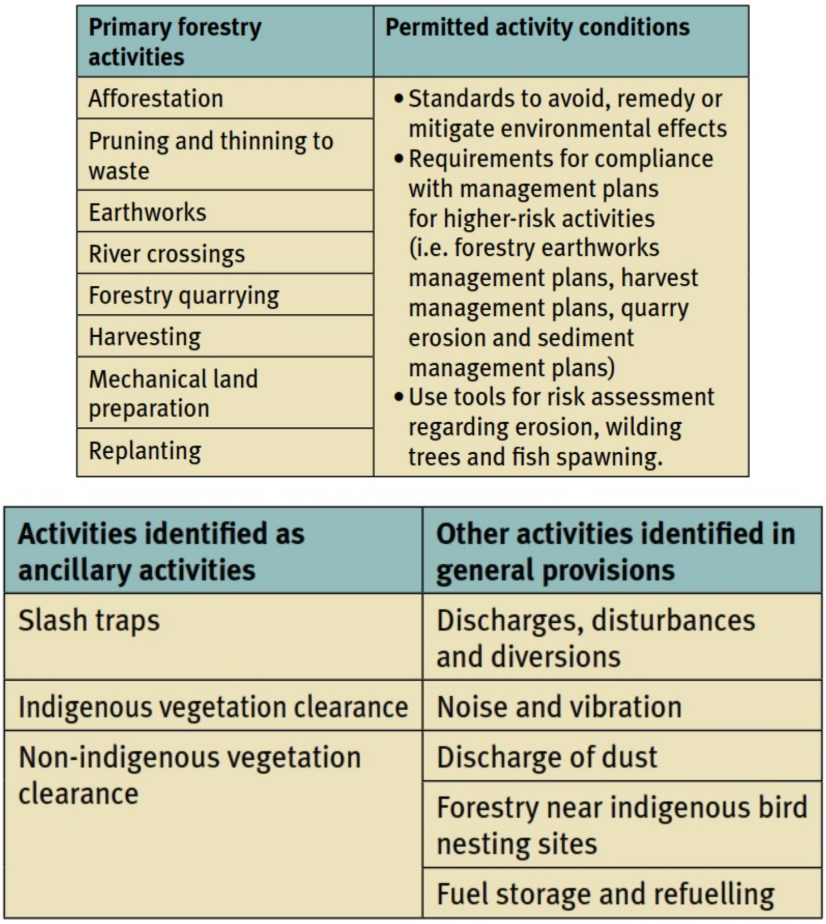
RMA Issues	Implications for forestry	
	1st Order	2nd Order
– Re-litigation of issues in regional and district planning documents – Requirement to interpret variable planning rules and standards	– Higher plan development, administrative and compliance costs for various parties, including costs associated with ongoing council/industry/stakeholder engagement – Uncertainty	– Inefficiency – the cost to NZ when producing wood is greater than is necessary
– Inconsistent treatment between districts & regions	– Operational costs for forestry companies (on-ground costs) are higher – Foreign investment is deterred	– Investment uncertainty
– Varying regulations in some districts & regions	– Uncertain environmental outcomes	– Variable forestry practice

Regional/district plans differ among local authorities dependent on the environmental issues within specific regions. The NES-FP was developed to provide regulatory consistency across New Zealand (Ministry for Primary Industries 2017e).

6.2 National Environmental Standards for Plantation Forestry

On 1 May 2018 the NES-FP came into law (Ministry for the Environment & Ministry for Primary Industries 2017). It defines plantation forestry as ‘forest deliberately established for commercial purposes being at least 1 ha of continuous forest cover of a forest species that has been planted and has or will be harvested or replanted’ (New Zealand Government 2018). Under this definition a mānuka plantation for honey production with a harvesting management strategy would fall under the NES-PF and have the same regulatory restrictions as pine plantations, including pruning and thinning. The NES-PF would not apply to native regeneration and mānuka planted with no harvesting/replanting strategy.

The NES-PF applies to several primary and ancillary activities in the forestry industry (Figure 12). The NES-PF also allows for restrictions of activities where they may affect environmentally sensitive areas and other national RMA instruments (e.g. the National Policy Statement for Freshwater Management), and it allows councils to offer further protection (Fowler 2017; New Zealand Government 2018). These environments include drinking-water supplies, and other natural areas of significance, including significant vegetation types, archaeological sites and habitat for endangered species. Fowler (2017) suggested the key features of the NES-PF are the application of the Erosion Susceptibility Classification (ESC), wilding tree calculator and earthworks/harvesting plans.



Primary forestry activities	Permitted activity conditions
Afforestation	• Standards to avoid, remedy or mitigate environmental effects • Requirements for compliance with management plans for higher-risk activities (i.e. forestry earthworks management plans, harvest management plans, quarry erosion and sediment management plans) • Use tools for risk assessment regarding erosion, wilding trees and fish spawning.
Pruning and thinning to waste	
Earthworks	
River crossings	
Forestry quarrying	
Harvesting	
Mechanical land preparation	
Replanting	

Activities identified as ancillary activities	Other activities identified in general provisions
Slash traps	Discharges, disturbances and diversions
Indigenous vegetation clearance	Noise and vibration
Non-indigenous vegetation clearance	Discharge of dust
	Forestry near indigenous bird nesting sites
	Fuel storage and refuelling

Figure 12. Primary and ancillary forestry activities addressed by the National Environmental Standards for Plantation Forestry (Fowler 2017).

6.2.1 Erosion Susceptibility Classification

The ESC is a simple system that classifies land into low (green), moderate (yellow), high (orange) or very high (red) erosion susceptibility risk. It is largely based on the New Zealand Land Resource Inventory and Land Use Capability database (Figure 13; Ministry for Primary Industries 2017e). Further refinement of the ESC included assignment of land to two primary erosion type categories: (1) gully or tunnel erosion, of severe erosion potential or greater; or (2) earthflow, where the geology is crushed argillite, Tertiary mudstone or sandstone

(Ministry for Primary Industries 2017e). Importantly, Basher & Barringer (2017) also identified Class 8e land, which is a sub-class of land assigned to the red zone and represents land at risk of severe to extreme erosion (Figure 14) which is deemed unsuitable for arable, pastoral, or commercial forestry use (New Zealand Government 2018).

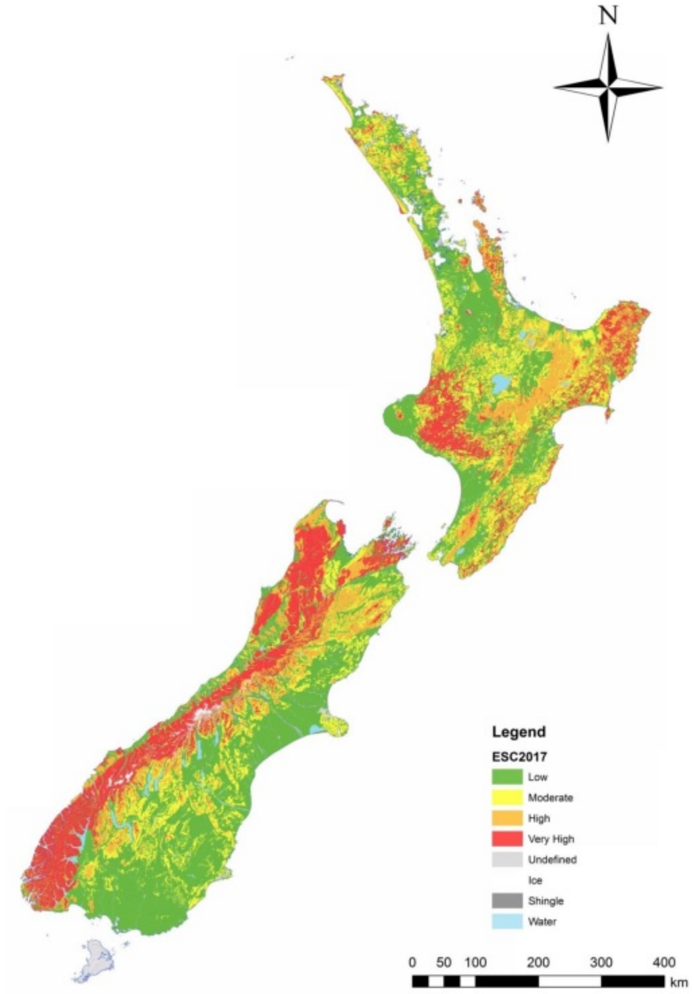


Figure 13. Erosion Susceptibility Classification map (Basher & Barringer 2017).

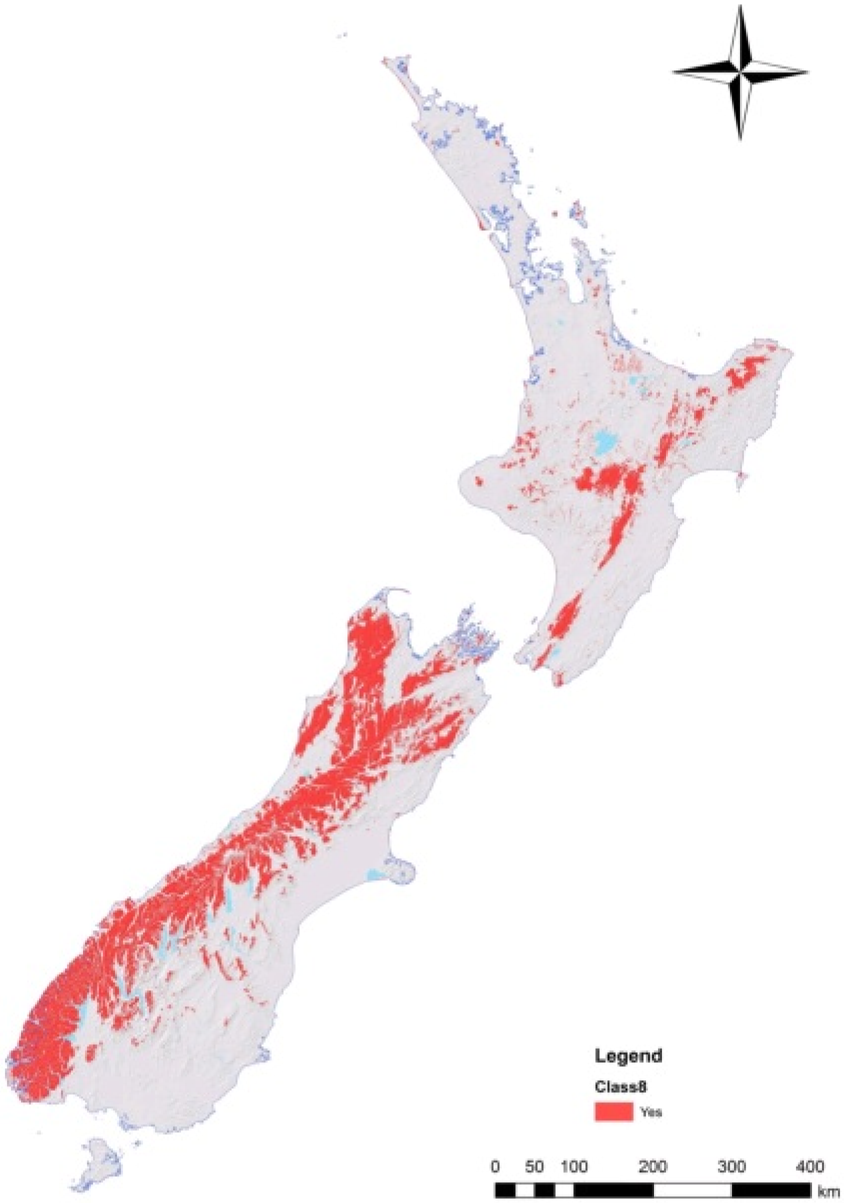


Figure 14. Distribution of class 8e land in New Zealand (Basher & Barringer 2017).

The combination of ESC classification and the amount of land to which an activity will be applied determines if the activity is permitted, controlled or restricted discretionary (New Zealand Government 2018). With the exception of class 8e for which many forestry operations, including harvesting, are restricted discretionary activities. The Gisborne District has land in all four of the ESC classes, with the largest amount in the very high class (Figure 15). The majority of class 8e land is in the South Island, although there is a small amount in the Gisborne District (Figure 14).

Existing resource consents for forestry activities will remain valid until they expire, and then any future consent will need to be undertaken within the NES-PF (Anderson Lloyd 2017; 4Sight Consulting Limited 2018). If an activity is being conducted as a permitted activity on a district plan, in general these activities ‘the existing use rights’ mean that compliance with the NES-PF is not required (Anderson Lloyd 2017). However, if the activity is permitted under a regional plan, then this will remain valid during an interim period, but following that compliance with the NES-PF is required (Anderson Lloyd 2017).

Slash management is an important factor for the forestry industry's licence to operate, and it is regulated under the NES-PF. Generally, if slash is not deposited into waterways, onto land that is susceptible to flooding, or into coastal waters, then pruning and thinning is a permitted activity (New Zealand Government 2018). If pruning and thinning material is deposited into these waterways or onto flooding-susceptible land, then it must be removed (unless doing so is unsafe) for this to remain as a permitted activity. If the activity does not adhere to these conditions, then it becomes a controlled activity.

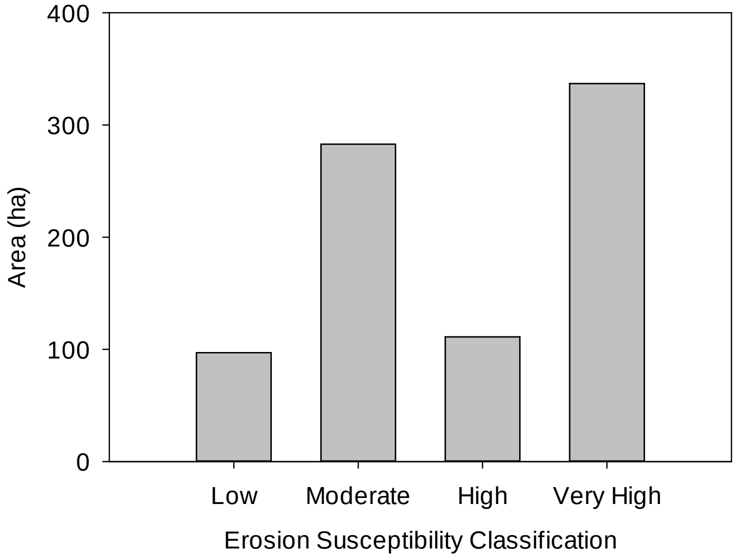


Figure 15. Area of land in each erosion susceptibility class in the Gisborne District (Basher & Barringer 2017).

Forestry activities in low, moderate, and high erosion risk areas are permitted (except for earthworks in the high zone if slope is >25°), but councils may apply further controls on high erosion risk areas (Fowler 2017), such as restrictions determined by the Gisborne District Council in the Tairāwhiti Resource Management (section 6.3).

6.2.2 Wilding tree calculator

The wilding calculator will also have an impact on pine plantation forestry because it initiates controls on replanting or afforesting land with conifer trees with a wilding risk of 12 or more (Paul 2015; Fowler 2017; New Zealand Government 2018). The wilding calculator does not apply to non-conifer species, so it would not apply after land-use change to mānuka or, presumably, natural regeneration. There are also provisions within the NES-PF to compel forestry operators to remove conifer wildings near their properties, which would be on a case-by-case basis and would depend on the nature of the areas surrounding the pine plantations.

6.3 Tairāwhiti Resource Management Plan

The Gisborne District Council has included provisions in the Tairāwhiti Resource Management Plan (TRMP) for soil conservation with respect to the land-use capability category of the land in the region (Gisborne District Council 2017). Land-use capability (LUC) units are assigned to one of three categories:

- land overlay 1: land-use capability (LUC) classes I–IV and class VI units e1, 2, 3, 5, 7 and 8
- land overlay 2: balance of LUC Class VI
- land overlay 3: LUC classes VII and VIII, including overlay 3A, which identifies the most erosion-prone land (Gisborne District Council 2017).

All of the high (orange) and very high (red) zones in the ESC are LUC class VII and VIII, with the exception of a small number of areas of land, which have dual LUC classifications and may contain some class VI land in combination with class VII or VIII (Basher & Barringer 2017). Therefore it is likely that the red zone is very similar to the land overlay 3 used by the Gisborne District Council, although clarification of the similarities and differences between the TRMP land overlays and the ESC in the Gisborne District is required.

The TRMP places greater restrictions on plantation forestry activities, which presumably would also cover mānuka plantations where they fit the NES-PF definition of a plantation forest and are outlined at http://www.gdc.govt.nz/tairawhiti-plan/. The regulations enforced by the Gisborne District Council are more complex and descriptive than those in the NES-FP, so forming a good working relationship with the council is the best way to ensure compliance with both the TRMP and NES-FP.

The soil conservation regulations contained within the TRMP are currently operative, with the exception of provisions proposed in the Gisborne Regional Freshwater Plan (GRFP, as found at http://www.gdc.govt.nz/freshwater-plan-proposed/). However, the GRFP has been appealed by several organisations in the Environment Court (Neilson 2017ba) and has not yet been finalised. The Māngatu Blocks (forestry company) and Wi Rere Trust (land owners) appealed on the basis of insufficient definition of 'intensive stock grazing' and the scope of farm environmental plans, and also the proposed limits for nutrients and sediment discharges in the Waipaoa catchment (Neilson 2017a). The Eastland Wood Council appealed the inclusion of schedule 7 on protected waterways, as well as proposed restrictions on cable hauling across streams and the implementation of non-regulatory projects within the Waipaoa Catchment Plan (Neilson 2017a). The appeals to the Environment Court were not resolved within the time span of this report, but further assessment by land owners/forestry managers of their compliance under the TRMP and NES-PF are recommended once this dispute is settled.

7 Summary and conclusions

The Gisborne District is highly erosion prone due to the interaction between geology and weather patterns. *Pinus radiata* forestry has been a major factor in reducing erosion and boosting the economy in the region. However, a window of vulnerability exists between harvesting and re-establishment of canopy closure in pine plantations, the duration of which is usually 6–8 years. During this time the landscape is susceptible to erosion from severe storm events.

Climate change is predicted to increase the intensity of storm events and alter rainfall amounts, and will affect erosion directly and indirectly: directly by influencing soil water content, and indirectly by affecting tree growth. Potential future erosion under climate change has been comprehensively reviewed by Basher et al. (2012), and their main findings were that erosion in the Gisborne District is likely to decrease due to decreasing annual rainfall because of reduced winter rainfall, despite an increase in summer rainfall. The effect of storm intensity and frequency will be a major factor affecting future erosion patterns, but it is currently inadequately predicted.

Increasingly, alternative land-use options for plantation forestry in erosion-prone areas are being assessed by forestry companies and landowners, and we assessed the benefits and limitations of three land uses with respect to erosion control and future climate: remaining in plantation pine, shifting to native forest, or shifting to mānuka plantations for honey production (Table 8).

Table 8. Benefits and limitations for three land uses after pine harvest in the Gisborne District with respect to maintaining erosion mitigation services

Land use / vegetation type	Benefits	Limitations
Mānuka plantations	Effective for erosion mitigation at about 8 years old	Estimate based on limited information: more information is required
	Plantation mānuka productivity under current climate = 1.6 t DM/ha/yr	Estimate based on limited information: more information is required
	Climate-change-induced increases in temperature and CO_2 will probably increase mānuka growth	Mānuka plantation management strategies are not fully established so are difficult to interpret within this report. However, erosion mitigation potential may be reduced under a 15-year rotation clear-fell/replanting strategy.
		Effect of increased windiness on mānuka not known
	Potential for high profits from honey production	The reasons for the large variability in honey profits are not fully understood, and profits are unlikely to be larger than pine production.
		Under a clear-fell/replant management strategy, mānuka plantations will probably be eligible for the PFSI and probably not for the ETS. Expenses for conversion to mānuka plantation remain at the cost of the land owner.
Regenerated native forest (includes indigenous shrublands and forest)	Natural regenerated mānuka is effective for erosion mitigation at 16 years old.	There is insufficient information to determine the age at which effective erosion mitigation occurs in mixed-species native forest, and this will depend on species diversity and stem density.
	The productivity of naturally regenerated mānuka under current climate = 3.6–4.2 t DM/ha/yr. Native monocultures = 0.8–3.6 t DM/ha/yr.	There is a paucity of data on growth rates of native trees, particularly in mixed-species stands
	Climate-change-induced increases in temperature and CO_2 will probably increase mānuka growth.	The establishment and growth of native trees may be hindered under climate-change-induced decreases in rainfall, with the exception of mānuka. The effects of elevated temperature and/or elevated CO_2 on native tree growth are not known, except those relating to mānuka.
	Retirement to regenerated native forest may be the best option for erosion mitigation on highly erosion-prone land	Wind-throw and stem snap are more likely under climate-change-induced increased windiness and storm intensity.
	Eligible for PFSI and ETS once regenerated forest established.	Expenses for conversion to native forest remain at the cost of the land owner.
Pinus radiata plantations	Effective for erosion mitigation at 8 years old.	Debris flows and other issues relating to the social licence to operate.
	Rapid-growing trees with productivity under current climate = 17.5 t DM/ha/yr	Regulatory pressure from the NES-PF and Gisborne District Council
	Climate-change-induced increases in temperature and CO_2 will probably increase pine tree growth when not restricted by lack of moisture or nutrients.	Wind-throw and stem snap are more likely under climate-change-induced increased windiness and storm intensity.
	An established industry with considerable research to support the industry.	
	Eligible for PFSI (with no or limited harvesting) and for ETS.	May have compulsory inclusion in the ETS.

Fire remains a risk for all land uses, and under climate change the fire risk may increase due to increasing frequency and intensity of drought events, although this may be partially mitigated by forecast increasing summer rainfall.

Pine production decreases soil calcium and nitrogen concentrations, but whether this is sufficient to reduce productivity in subsequent rotations will be site specific. Litter, tree root degradation, and harvest debris retention contribute to soil stocks, but retention of harvest debris on steepland sites is not always achievable and can contribute to debris flows during storms. Under climate change the adequacy of tree nutrition will be determined by the interaction between increased litter production and increased soil organic matter turnover, and will be site and crop specific. Mycorrhizae are currently used to enhance pine tree growth by increasing water and nutrient uptake, and mānuka is colonised by both arbuscular mycorrhizae and ectomycorrhizae. Little is known about how mycorrhizae affect native tree productivity. Climate change may affect mycorrhizae, but their response under the combination of harvesting and climate pressures is not clear.

Climate change is likely to maintain or increase current rates of weed incursion in plantation forests. However, this conclusion has been inferred from limited information and must be veiwed with caution. Under forecast drier conditions in the Gisborne District it is possible that competition from the weed species we assessed (conifer wildings, gorse, Scotch broom, ginger, and buddleia) may inhibit the establishment and growth of potentially slower-growing native species.

Pathogen prevalence under climate change is dependent on future rainfall patterns. Current inferences are that *Dothistroma* and *Cyclaneusma* needle diseases will increase or remain unchanged under climate change. There are insufficient data on the possible transfer of pine tree diseases to rotations of subsequent native tree species. Naturally regenerating and plantation mānuka will continue to be susceptible to myrtle rust and mānuka blight, but there is insufficent information to be able to determine the effect of climate change on these infections.

Myrtle rust will also remain a factor in the species distribution and survival of *Myrtaceae* species in New Zealand, as will *Phytophthora*. There is also little information on the potential effects of climate change on the prevalence and distribution of these pathogens in native forests.

Climate change is likely to increase the abundance of common pest species (including, possums, hares, rabbits, pigs, deer) in the Gisborne District by prolonging the breeding season and increasing forest disturbance events due to increasing storm intensity and windiness. However, it is not clear whether Black pine beetles would infest native species planted post-pine harvest.

There was no economic information on the profitability of changing from pine plantations to mānuka plantations or native regeneration. There is also significant uncertainty about the potential quality of honey produced from mānuka plantations, which can substantially affect profitability.

The Tairāwhiti Resource Management Plan and the National Environmental Standards for Plantation Forestry both affect the management of forests on erosion-prone land in the Gisborne District. Restrictions in these regulations may affect the viability of pine and mānuka plantations in some parts of the region. There will be minimal regulatory effects on land-use change from pine plantations to natural regeneration unless harvesting of native trees is part of the management of post-pine native stands, in which case this may be restricted in some areas.

In summary, climate change will affect erosion in the Gisborne District both directly and indirectly. All of our land-use options provided effective erosion mitigation between 8 and 16 years of growth − after the period of vulnerability to storm events. Pine and mānuka plantations, as well as naturally regenerating mānuka/kānuka, will probably have greater productivity under climate change, but whether this will also occur for naturally regenerated forest is not known. All land uses will be similarly affected by weeds and pests under climate change, and the management of these factors will play an integral part in all of the land-use options assessed in this report. Climate change may exacerbate the effects of some pathogens on pine production, but it is unclear how climate change will interact with pathogens of New Zealand native trees.

There are many benefits to remaining in pine production, but the introduction of the NES-PF and further restrictions on activities on erosion-prone land by the Gisborne District Council will greatly affect forestry in this region. However, these same regulatory pressures are also likely to affect mānuka plantations, depending on how the management of these plantations is developed in the future. The retirement of land to naturally regenerated forest will be relatively unaffected by legislation, and for highly erosion-prone land with minimal production forestry value this may be a viable option.

The SLMACC 405415 project, of which this literature review is a component, will address some of the issues raised in this report in the remaining 2 years of the project. Biophysical and economic modelling of the land-use options will be used to further inform land owners and industry.

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