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Effect of changes in forest water balance and inferred root reinforcement on landslide occurrence and sediment generation following *Pinus radiata* harvest on Tertiary terrain, eastern North Island, New Zealand

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Abstract

Background: The frequent occurrence of storm-initiated landslides following harvesting of *Pinus radiata* D.Don in steep, Tertiary terrain, East Coast region, North Island, New Zealand, is of increasing concern. This paper documents the influence of tree removal and of replacement plantings on the canopy water balance and soil moisture regime when slopes are at their most vulnerable to landslide occurrence.

Methods: At a previously established study site, rainfall, throughfall, and soil moisture data were collected before a mature stand of *P. radiata* was harvested. After harvesting, part of the study site was replanted with *P. radiata* at 1000 stems ha^{-1} and part with 500 stems ha^{-1}. Relationships between hydrological changes and landslide occurrence are discussed in relation to planting density, site factors, root system development, silvicultural regimes, and alternative land use options for mitigating erosion in highly erodible hill country.

Results: Following harvesting, soil moisture levels remained higher for longer than under a mature forest until rainfall interception and evapotranspiration returned to pre-harvest levels. This coincided with canopy closure, irrespective of planting density. After thinning, interception and evapotranspiration decreased, then regained the equivalent of a closed canopy 2 years later. Landslide occurrence was highest on slopes >25° and with a NE aspect. Sediment generation rates were highest in 2–4-year-old plantings, then decreased markedly with increasing tree age.

Conclusions: Irrespective of planting density, *P. radiata* had little influence on the soil-water regime until canopy interception, evapotranspiration rates, soil-drying and recharge cycles returned to pre-harvest levels, coinciding with canopy closure. During this period, pore-water pressures at times of heavy or prolonged rainfall likely result in soil saturation and an increase in landslides. The progressive loss of root strength of the harvested trees had a secondary influence. The duration of the post-harvest period of heightened slope vulnerability to landslide initiation is a function of the combined influences of site factors on rates of tree growth and survival, and of the planting density regime on the canopy water balance and soil water content until the development of an effective live soil-root reinforcement system. For areas identified as high risk, the targeting of high-value timber species with longer rotation length, including consideration of coppicing species, would minimise the risk of slope failure at harvest. Very high-risk areas unsuited to rotational harvesting will ultimately require transitioning to a permanent indigenous forest cover.

Keywords: forest removal; hydrological change; duration of heightened slope vulnerability, alternative species options.

Introduction

The East Coast region of the North Island (835,500 ha) has the greatest proportion of erosion-prone land in New Zealand (Ministry for Primary Industries 2017). Here, 68% of the region (Fig. 1) is underlain by unconsolidated sedimentary lithologies of Tertiary age, of which 35.9% is classed as moderately to very highly susceptible to shallow landslides and debris flows (Figs. 2 & 3). Conversely, within 32% of the region underlain by sedimentary lithologies of Cretaceous age, the dominant erosion processes include earthflows and gully erosion, and shallow slope failures are largely confined to steep-sided (>25°) slopes flanking incised stream channels and gullies (National Water and Soil Conservation Organisation 1975).

In this region, successive storms in 1980 and 1982 resulted in landslide failure across extensive areas of pastoral hill country. These storms were typically of short duration, and although rainfall intensities were highly variable, total rainfall proved sufficient to initiate landslides (Selby 1967; Crozier & Eyles 1980; Crozier et al. 1980; Salter et al. 1983; Harmsworth et al. 1987; Hicks 1995; Glade 1998; Rowe et al. 1999; Crozier 2005). Landslides are typically translational (Fig. 3), with failure occurring either along a basal shear plane coinciding with impermeable bedrock (Marden et al. 1991), or within skeletal, well-drained, cover-bed materials. These comprise weathered colluvium and vestiges of late Quaternary volcanic airfall pumice and ash (McLeod & Rijkse 1999) that are prone to saturation,

FIGURE 2: Aerial view of harvested *P. radiata* cutover at Wakaroa (top left of photo) and Mangarara (centre and bottom right of photo) Forests. (Photo flown on 06/10/2015, courtesy of Aratu Forests Ltd).

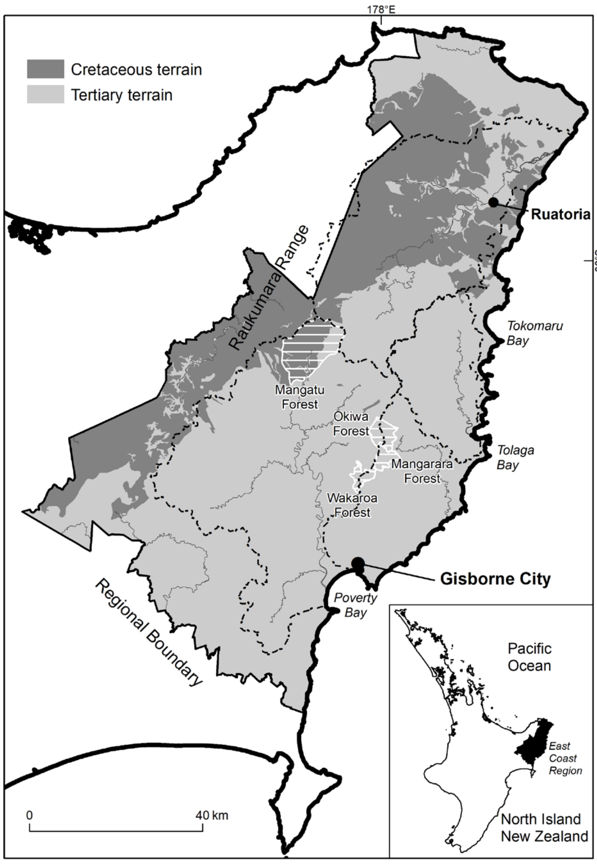


FIGURE 1: Location map of *P. radiata* forest study sites, East Coast region, North Island, New Zealand. *(Figure updated 20/04/2023)*

a condition conducive to the triggering of shallow translational landslides (Fourie 1996).

As a means of reducing both the on- and off-site impacts of erosion, particularly within and downstream of areas of Cretaceous terrain, ~35,000 ha of severely eroding pastoral hill country was progressively retired and planted (1962–1985) in exotic forest species as "protection forests" (Poole 1960). In 1988 a major storm (Cyclone Bola) caused further significant landslide-related damage within existing areas of planted exotic forest and across extensive areas of remaining pastoral hill country, predominantly within the Tertiary terrain (Phillips et al. 1990; Marden et al. 1991; Marden & Rowan 1993), contributing significantly to existing high rates of erosion (Water and Soil Directorate 1987). To protect against further soil loss within the Tertiary terrain, an additional ~100,000 ha of exotic forest was planted between 1992 and 1997, and 20,000 ha since 1997, bringing the regional total of planted exotic forest to ~155,000 ha (Forest Owners Association & Ministry for Primary Industries 2016).

Noted for its rapid growth rate and tolerance of a wide range of site and environmental conditions, *Pinus radiata* D.Don has remained the preferred tree species. With increasing economic potential, the earliest of the "protection forests" (Poole 1960) were later reclassified as "protection-production forests", raising concerns over the probability that their harvesting would reactivate erosion (Kelliher et al. 1995). These concerns have since been realised, with several reports confirming that during major storm events a significant source of

FIGURE 3: Shallow landslides initiated after the harvesting of *P. radiata* and typical of areas within the Tertiary terrain at Mangarara Forest. (Photo taken on 25/09/2015, courtesy of Aratu Forests Ltd).

landslide-derived sediment and logging slash (woody debris) originated from harvested *P. radiata* forests (e.g., Baillie 1999; Amishev et al. 2013).

It is well known that forest removal affects the forest water balance, including the loss of evapotranspiration (interception and transpiration), resulting in:

(i) increased soil water content and pore-water pressures that at times cause soils to fail when saturated (Terzaghi 1950);

(ii) accelerated changes in hillslope or catchment hydrology (Keim & Skaugset 2003; Dhakal & Sidle 2004);

(iii) enhanced surface erosion (Glade 2003; Imaizumi & Sidle 2012); and

(iv) changes to various hydrogeomorphic processes

(Croft & Adams 1950; Bishop & Stevens 1964; O'Loughlin 1972, 1974; Swanson 1981; O'Loughlin et al. 1982; Jakob 2000; Brardinoni et al. 2003; Gyenge et al. 2003; Sidle & Ochiai 2006).

The progressive loss of strength of decaying root systems also contributes to an increased occurrence of mass movement (O'Loughlin & Ziemer 1982; O'Loughlin 1985; Millard 1999; Jakob 2000; Guthrie 2002; Brardinoni et al. 2003; Sidle& Ochai 2006; Imaizumi et al. 2008; Imaizumi & Sidle 2012) and sediment production (Beschta 1978).

In New Zealand, previous studies of cause–effect relationships of storms following the harvesting of *P. radiata* forests have generally focused on establishing the relationships between landslide occurrence, site factors, and root decay rates (O'Loughlin & Watson 1979, O'Loughlin & Ziemer 1982; Watson et al. 1999), whereas data on sediment generation by landslides during the post-harvest period is less common (e.g., O'Loughlin 1972; O'Loughlin et al. 1982, 1978; Hicks & Harmsworth 1989; Basher et al. 2011; Marden & Rowan 2015). Similarly, analyses of soil-root reinforcement interactions have tended to focus on the measurement of growth parameters of the root systems of well-established trees at specific ages (O'Loughlin 1985; Watson & O'Loughlin 1990; Watson et al. 1995), and until relatively recently, there has been very little information available on the below-ground growth performance and/or root reinforcement of juvenile trees (Watson & Tombleson 2004; Marden et al. 2016).

The canopy water balance (e.g., interception, evapotranspiration) and its influence on soil water patterns have previously been documented in a mature stand of *P. radiata* forest (Pearce et al. 1987), both following thinning (Whitehead & Kelliher 1991; Kelliher et al. 1992b; Beets & Oliver 2007) and for a short period after harvesting (Jackson et al. 1987; Jackson 1987). There are, however, no known studies where changes in the canopy water balance and its influence on soil water patterns within juvenile replacement plantings established on forest cutover have been recorded over an extended post-harvest period. Used in combination with the mechanical soil-root reinforcement properties of trees of the same age, this information is essential to an assessment of the duration of the 'window of vulnerability' (Sidle & Ochiai 2006) before replacement plantings significantly reduce landslide occurrence and sediment generation on forest cutover.

Furthermore, on the assumption that genetically improved *P. radiata* seedlings and cuttings are sturdier, grow faster, and therefore attain hydrological and soil-root reinforcement benefits earlier, there has been a trend towards lowering the recommended planting density for *P. radiata* on erosion-prone East Coast hill country from 1250 stems ha^{-1} (Ministry of Forestry 1994) to around 1000 stems ha^{-1}. However, Kelliher et al. (1992a) cautioned that "Consideration of improved strains of *P. radiata* in formulating tree planting policy should not be done until comparative locally-based regional data on tree growth (i.e., root and canopy development) are obtained".

In this paper we present a ~9-year post-harvest (1996–2005) record of changes in the canopy water balance and soil water regime within immature plantings of a genetically improved seed lot (GF 27) of *P. radiata* established at 1000 and 500 stems ha^{-1} on harvested areas in Mangatū Forest (Fig. 1). We use the growth rates of the crowns of these trees to establish the influence of planting density on:

(i) canopy development;

(ii) the duration (years after planting) required before replacement plantings reattain canopy closure,

(iii) components of the canopy water balance (interception and transpiration), and

(iv) the soil-water regime until seasonal soil drying and recharge cycles return to pre-harvest levels.

To establish the approximate duration of the "window of vulnerability' (Sidle & Ochiai 2006) of harvested areas to storm-initiated landslides, we correlate the extent of damage and sediment generation rates derived from harvested areas after storms at Mangarara and Wakaroa forests (Fig. 1) with: (i) known root decay rates of harvested *P. radiata* (O'Loughlin & Watson 1979), and (ii) time series measurements of the growth rates of the lateral roots of immature *P. radiata* (Marden et al. 2016) established at Okiwa Forest (Fig. 1) and on the same terrain as at Mangarara and Wakaroa forests. Finally, we discuss the implications of this research for the management of existing and future plantings of exotic forest and the role of alternative forest species options to better alleviate the risk of landslide occurrence within a region subject to frequent high-intensity storms.

Methods

Study areas

In Mangatū Forest (177° 51′ E, 38° 16′ S) (Fig. 1) the type and distribution of hillslope instability is strongly influenced by slope physiography, which in turn reflects the structural complexities and composition of variably indurated, extensively sheared sedimentary lithologies of Cretaceous age (Mazengarb & Speden 2000).

In Mangarara and Wakaroa forests (178° 04′ E, 38° 23′ S) (Fig. 1), relatively well-bedded alternating sandstone and mudstone of late Tertiary age exert a strong influence on drainage patterns (Mazengarb & Speden 2000), and where stream channels are deeply dissected and flanked by steep-sided (mean slope of 31°), landslide-prone valley slopes.

Climate

The climate in the East Coast region of the North Island is warm temperate maritime, with warm, moist summers and cool, wet winters, strongly influenced by the El Niño / Southern Oscillation (ENSO) (Gomez et al. 2004). During La Niña conditions, there tends to be a higher frequency of high-intensity, localised 'storm cells' that have in the past caused widespread hillslope failure (Phillips et al. 1990; Marden et al. 1991; Marden & Rowan 1993; Kelliher et al. 1995) and extreme flooding. In the north of this region, landslide-generating storms in headwater reaches of the major catchments have a recurrence interval of between 2.6 years and 3.6 years near the coast (Hicks 1995). In contrast, return periods for cyclonic storms are longer, and based on flood level records, the return period (of storms such as Cyclone Bola in 1988) is estimated to be 70 years (pers. comm., Dave Peacock, Gisborne District Council). Monthly rainfall data collected at Mangatū Forest for the period 1983 to 2004 were recorded at a meteorological station located near the study site. Rainfall data for storms recorded at Wakaroa and Mangarara Forests are estimates based on readings from rain gauges located nearest to these forests and extrapolated for individual forests. Rainfall intensities (mm h^{-1}), totals (mm), and return periods (years) for storms cited in this paper are presented in Table 1.

Forest hydrology

In 1983, a study site was established under a mature stand of *P. radiata* located within Mangatū Forest. Over a 3-year period (1983–1986) rainfall interception, transpiration, and the soil-water content were recorded and subsequently presented in Pearce et al. (1987). The study site is located on a north–northeast-facing slope at an elevation of ~230 m above sea level (Marden et al. 2008), a portion of which was harvested in 1986 and the soil-water response recorded (Jackson et al. 1987). Before the remaining area of the then 30-year-old forest was harvested, the existing array of throughfall (rainfall not intercepted by the canopy and reaching the ground) troughs was decommissioned. Throughout the harvest period the neutron probe access tubes remained in place and were protected against damage.

After harvesting in 1991, the study site was initially left fallow before being replanted in 1992 with 1-year-old genetically improved GF 27 *P. radiata* seedlings. A ~0.5 ha area surrounding one group of neutron probe tubes was replanted at 1000 stems ha^{-1} (4 × 2.5 m spacing), and at 500 stems ha^{-1} (5 × 4 m spacing) around another group of tubes (Fig. 4). The distance between the neutron probe plots is ~200 m. Following

TABLE 1: Rainfall intensities, total rainfall, and return periods for major storms in the East Coast Region, New Zealand.

Forest	**Rainfall characteristics**	**Date**			
		March 1988 (Cyclone Bola)	**June 2009**	**June 2014**	**September 2015**
Coastal areas of Tertiary terrain, East Coast Region	Max. 24 h intensity (mm)	346			
	Hourly intensity (mm)	23			
	Total rainfall for period (mm)	917			
	Return period (yrs)	70			
Mangarara Forest	Max. 24 h intensity (mm)		20	152	152
	Hourly intensity (mm)		4	11	20
	Total rainfall for period (mm)		33	197	183
Wakaroa Forest	Max. 24 h intensity (mm)		21	138	172
	Hourly intensity (mm)		5	11	18
	Total rainfall for period (mm)		36	177	204

the same soil-water measurement protocols outlined by Jackson et al. (1987) and Jackson (1987), data collection resumed through to 2005.

In 1995, as part of normal forest practice, trees within both stands were pruned to a height of 2 m. A new array of three PVC (polyvinyl chloride) troughs and water collection drums were then installed, as described by Rowe (1979, 1983) within each stand (Fig. 4). Throughfall was collected at approximately monthly intervals between 1996 and 2005. Gross annual rainfall during the entire pre-and post-harvest period (1983–2005) was collected in a Belfort rain gauge located outside the forested area and within 800 m of the study plots (Fig. 4). For the post-harvest period (1991–2005), mean annual rainfall was 1325 mm (cf. the long-term average of 1350 mm yr^{-1} for the period 1946–1983).

We present a ~9-year (1996–2005) record of the soil-water response to changes in the canopy water balance in stands replanted with *P. radiata* seedlings at 1000 stems ha^{-1} and 500 stems ha^{-1}.

Tree growth data

Annual measurements of the crown diameters of individual trees (average of diameters measured in the upslope-downslope and across-slope directions) are used as an approximation of the proportion of ground area covered by the canopy until the canopies of adjacent trees come into contact (i.e., a canopy occupancy index of 1) (Kelliher et al. 1992a). For the stand planted at 500 stems ha^{-1}, canopy occupancy indices are based on annual measurements of the mean diameter of the crowns of ≥40 individual trees; for the stand planted at 1000 stems ha^{-1}, 77 trees were measured. To assess the influence of stand density on the growth performance of other tree metrics, we measured tree height (obtained using a Forestor vertex hypsometer), root collar diameter (RCD, over bark at ground level), and diameter at breast height (DBH, over bark at 1.4 m above ground level on the upslope side of each tree). Measured trees included those located within a 30 m distance of each of the instrumented plots and considered likely to have an influence on the hydrology within each plot.

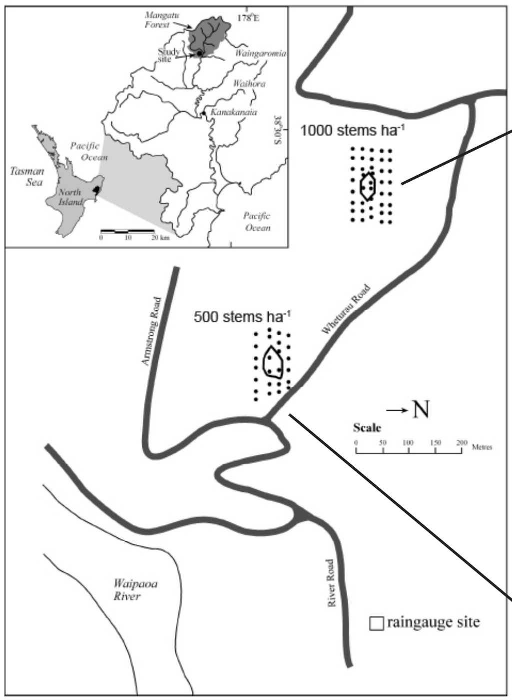
Similarly, the proportion of ground area covered by the canopy for stands of *P. radiata* established at the recommended planting density of 1250 stems ha^{-1} is based on annual measurements of the tree crowns of fifty 1-year-old trees, fifty 2-year-old trees, twenty-five 3-year-old trees, and fifty 4-year-old trees from an earlier study site located at Okiwa Forest (Fig. 1) (Marden et al. 2016). From this study, we also drew on annual measurements of the mean maximum lateral spread of the root systems of fifty 1-year-old trees, fifty 2-year-old trees, twenty-five 3-year-old trees, and ten 4-year-old trees to calculate the proportion of ground occupied until the roots of adjacent trees come into contact (i.e., a root occupancy index of 1) (Kelliher et al. 1992a).

Tree growth regression analyses

Above-ground tree growth metrics were modelled using a linear regression, with the annual means of each metric as the response and year as the predictor. Separate regressions were used for each planting density.
Growth relationships in canopy diameter and root spread over time were modelled using linear regression. Individual tree identity was included in the model as a blocking term to account for variation among individual trees. Linear relationships and confidence intervals were converted to occupancy indices using the following equation:

$$Occupancy = (diameter/2)^2 . \pi . (stand\ density/10{,}000)$$

All analyses were carried out in R (R Core Team 2022), and regression lines and confidence intervals were extracted for the growth metric and occupancy index relationships using the effects package (Fox & Weisberg 2019).



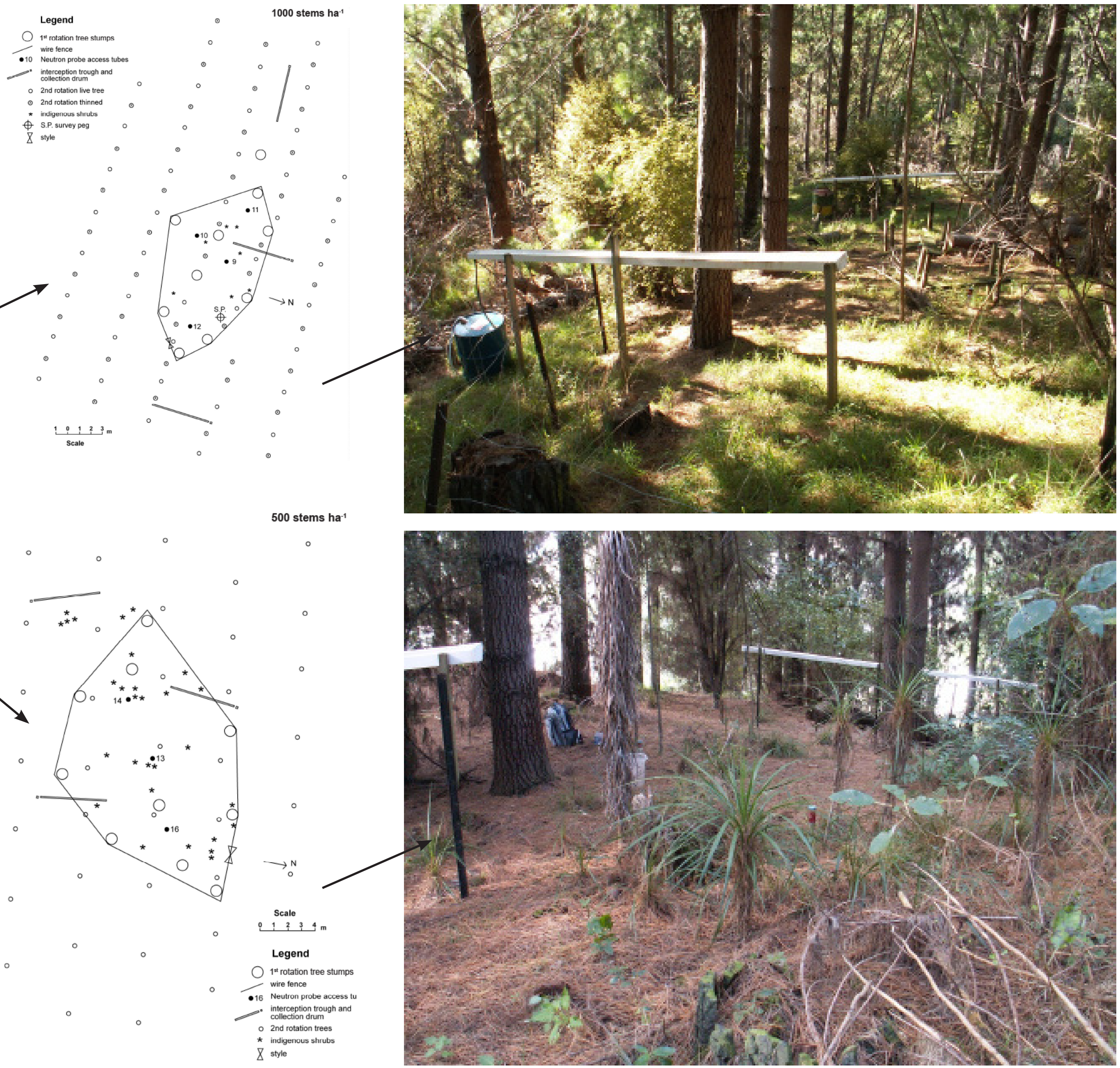


FIGURE 4: Mangatū Forest hydrology study site. Photographs and schematic diagrams show the spacing of *P. radiata* trees, and the location of interception troughs and neutron probe access tubes within plots replanted on forest cutover at 1000 and 500 stems ha^{-1} in 1992.

Modelling the influence of planting density and thinning on hydrology

The influence of planting density on the canopy water balance is based on canopy occupancy estimates derived from tree-growth regression analyses described above.

Annual interception and evapotranspiration (interception + transpiration) for stands established at different densities were based on the long-term average annual rainfall of 1350 mm yr^{-1}. An annual interception rate under a closed canopied stand of *P. radiata* (i.e. canopy occupancy equal to 1) is ~35% of gross annual rainfall (Pearce et al. 1987). Using this value as the maximum potential interception, we can model interception as a function of occupancy as follows:

$$\text{Interception (\%)} = \max(35, \text{Occupancy} \times 35)$$

An annual tree transpiration rate for a closed canopied stand of *P. radiata* when soil water deficit is not an issue is ~50% of rainfall (Whitehead & Kelliher 1991). Annual evaporation from pasture or bare ground in this region is ~900 mm (New Zealand Meteorological Service 1986). For closed-canopy forests located at ~200 m elevation in this region, a typical annual total evapotranspiration rate is ~1150 mm yr^{-1} (Kelliher et al. 1992a; that is, 85% of gross annual rainfall. Therefore, the maximum additional effect of trees on transpiration is ~250 mm yr^{-1}. We can use this value to model transpiration as a function of occupancy as follows:

$$\text{Transpiration (mm year}^{-1}) = \max(1150, 900 + \text{Occupancy} \times 250)$$

Landslide measurements and sediment generation

At Mangarara and Wakaroa forests combined, 1346 individual landslide scars (sediment source areas) distributed across 3481 ha of harvested forest (Fig. 2) were digitised from orthorectified sequential aerial photography flown after storms in 2009, 2014, and 2015. Data were captured in ArcMap at 1:5000 scale. Forest compartment maps were used to establish the year in which each harvested area was replanted. This provided an opportunity to relate landslide occurrence to site factors such as slope steepness and aspect, and to calculate sediment generation rates by age class of the replacement trees in relation to documented rates of root decay of harvested trees (O'Loughlin & Watson 1979).

To derive mass (t ha^{-1}), the volume of eroded sediment derived from measured landslide scars (ha) was multiplied by a bulk density of 1.6 ± 0.2 (Wilde & Ross 1996), assuming a typical landslide depth of ~1 m (Marden et al. 1991; Page et al. 1999).

The proportion of landslides connected/not connected to stream channels was documented relative to slope steepness. There was no attempt to correlate landslide occurrence with forest silvicultural (pruning and thinning) practices.

Results

Soil moisture

By the first winter following summer harvesting, and in the absence of forest cover, there was a rapid rewetting of the soil profile, with the water table rising to within 0.5 m of the ground surface. For the first 5 years after replanting the soil-water content remained at levels typical of the winter months under a closed-canopy forest cover, although for longer periods. There was little or no sign of summer drying, irrespective of differences in planting density (i.e., 1000 stems ha^{-1} versus 500 stems ha^{-1}) (Fig. 5).

Thereafter, with increasing tree age, canopy growth, and rates of evapotranspiration, soil-water deficits in

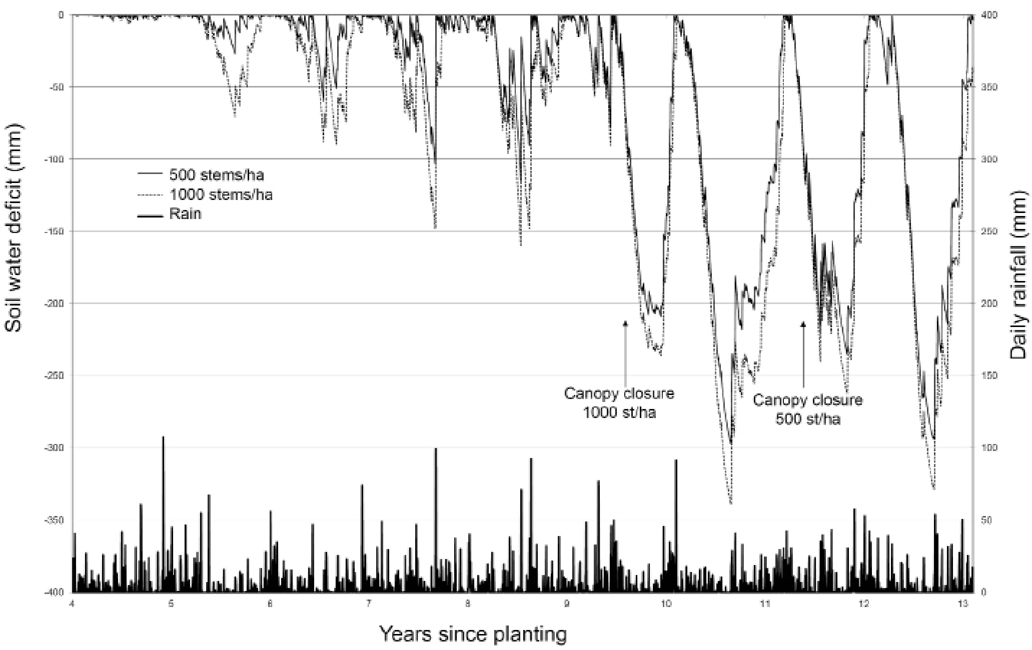


FIGURE 5: Changes in soil water deficit and daily rainfall in *P. radiata* stands between ages 4 to 13 years (1996–2005) replanted on forest cutover at 500 and 1000 stems ha^{-1}, Mangatū Forest.

both stands began to exhibit an increasingly stronger seasonal influence, exemplified by a gradual shortening in the duration of periods of high soil-water content (winter), and increasing levels of soil moisture deficit during the drier summer months, although more so in the denser of the two stands (Fig. 5). Despite differences in stand density, this seasonal soil-moisture deficit pattern became particularly pronounced in both stands 9 years after planting. This coincided with canopy closure in the denser of the two stands, when interception (35% of gross annual rainfall) and evapotranspiration (85% of gross annual rainfall) returned to levels recorded at the same site before the original forest was harvested. At the same age, canopy closure in the stand planted at 500 stems ha^{-1} was 64%, with interception at 22% of a potential 35% of annual rainfall, and evapotranspiration at 79% (1067 mm yr^{-1}) of that before the original forest was harvested. Throughout the post-harvest period, soil water deficits were consistently greater in the more densely planted stand, while during the wetter winter period soil moisture in both stands returned to field capacity.

During the 2-year monitoring period following thinning, with evapotranspiration loss decreasing by only 6% of annual rainfall, the 83 mm of added water appears to have had no discernible effect on soil-moisture storage (Fig. 5).

Growth metrics

For *P. radiata* seedlings of a single genetically improved seed lot established on harvested slopes at Mangatū Forest, individual trees planted at 500 stems ha^{-1} year-on-year were consistently taller, and both RCD and DBH exceeded that of trees planted at 1000 stems ha^{-1}. The crown diameter of individual trees planted at 500 stems ha^{-1} increased at a rate of 0.6 m year^{-1} and by year 9 were ~0.5 m greater than that of trees planted at 1000 stems ha^{-1}. Growth rates in both stands were essentially linear, with r^2 values of between 0.96 and 0.99 (Fig. 6).

Throughfall, rainfall interception, and evapotranspiration

Throughfall was initially highly variable and greater in the less densely planted stand, where gaps in the canopy were largest. During the first 3 years after planting, throughfall in both stands approximated annual rainfall. Thereafter, interception (Fig. 7) and evapotranspiration (Fig. 8) equivalent to that of a mature forest coincided with the timing of canopy closure (canopy occupancy index of 1), occurring earliest (year 9) in the denser of these two stands, ~2 years earlier than in the less densely planted stand (Figs. 7, 8 & 9A).

Before thinning from 1000 stems ha^{-1} to 400 stems ha^{-1} in year 11, canopies overlapped significantly (canopy closure index of 1.68), so thinning resulted in only a 36% reduction in canopy cover. Interception of annual rainfall decreased by 12% to ~23% of annual rainfall (Fig. 7) and evapotranspiration decreased by ~6% to 1067 mm year^{-1} (Fig. 8), that is, ~79% of annual rainfall (cf. 85% of annual rainfall before forest removal). Thereafter, with increasing canopy growth, canopy closure (and hence maximum interception and transpiration) was regained 2 years after thinning (Figs. 7 & 8).

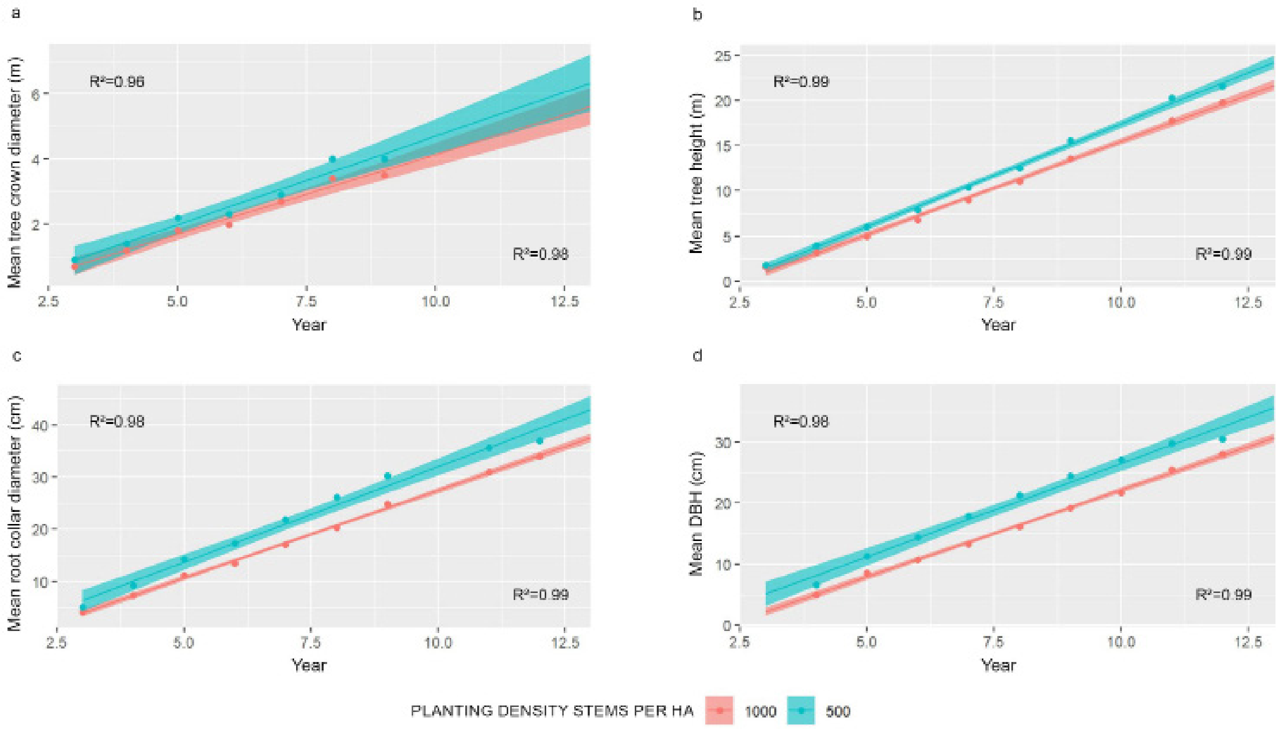


FIGURE 6: Linear regressions and 95% confidence intervals for mean tree crown diameter (a), mean tree height (b), mean RCD (c), and mean DBH (d) for *P. radiata* replanted on areas harvested areas at 1000 (blue) and 500 (red) stems ha^{-1}, Mangatū Forest.

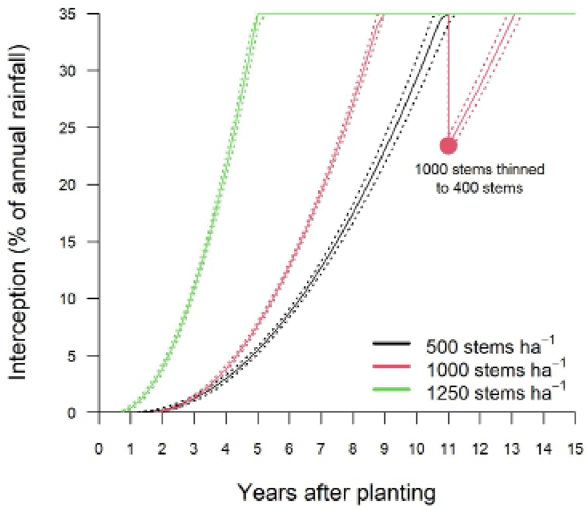


FIGURE 7: Relationship between stand density and rainfall interception as a percentage of annual rainfall and effect of thinning *P. radiata* from 1000 to 400 stems ha^{-1}, Mangatū Forest. Solid lines are the fitted regression line, and error bands are the 95% confidence intervals.

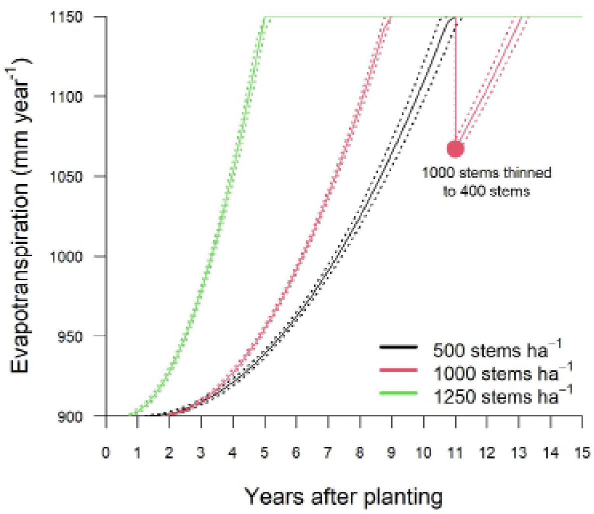


FIGURE 8: Relationship between stand density and evapotranspiration rate (mm yr^{-1}), and effect of thinning *P. radiata* from 1000 to 400 stems ha^{-1}, Mangatū Forest. Calculations assume an annual evaporation rate of 900 mm by groundcover vegetation during the early post-harvest period and an increase in evaporation in accordance with tree canopy development and rainfall interception. Solid lines are the fitted regression line, and error bands are the 95% confidence intervals.

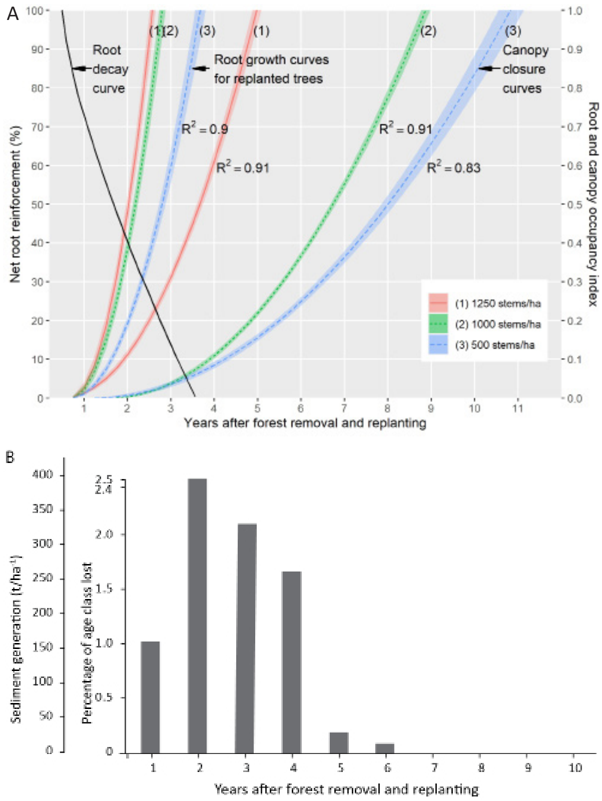


FIGURE 9: A. Loss of net tree root reinforcement of *P. radiata* after harvest (after O'Loughlin & Watson 1979) and influence of planting density on root and canopy recovery, expressed as occupancy indices. Solid lines are the fitted regression line and error bands are the 95% confidence intervals. B. Corresponding sediment generation rates (t ha^{-1}) and percentage of trees lost on cutover areas replanted 1 to 6 years before storms in 2009, 2014, and 2015, Mangarara and Wakaroa Forests.

Landslide occurrence, sediment generation rates, and connectivity with streams

Of the 1364 landslides triggered on replanted cutover located within Mangarara and Wakaroa forests, 71% occurred on slopes >25° (Figs. 3 & 10) and 54% on north- to east-facing slopes (Fig. 11). Landslide occurrence within 1-year-old stands resulted in a loss of 1% of the net stocked area and produced 164 t ha^{-1} of sediment. The highest concentration of landslides occurred in 2-year-old stands, where 2.4% of net stocked area was lost and where the corresponding rate of sediment generation more than doubled to 391 t ha^{-1}. Thereafter the extent of landslide damage and sediment generation decreased with increasing tree age, the most significant reduction occurring in plantings established 5 and 6 years before these storms (Fig. 9B, Table 2).

Two-thirds of all the landslides initiated during these events failed to connect with or deliver sediment to a stream channel. Of the 'connected' landslides, equal

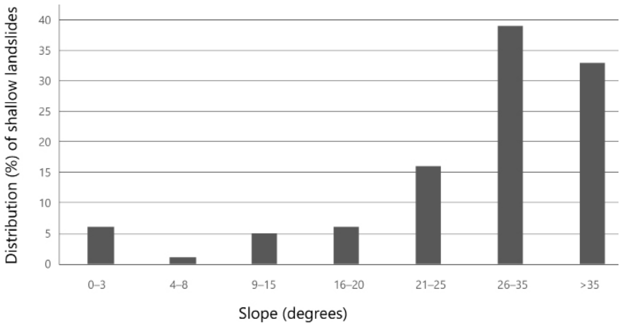


FIGURE 10: Distribution (%) of shallow landslides initiated on forest cutover replanted between 1 to 6 years before storms in 2009, 2014, and 2015, by slope group, Mangarara and Wakaroa Forests.

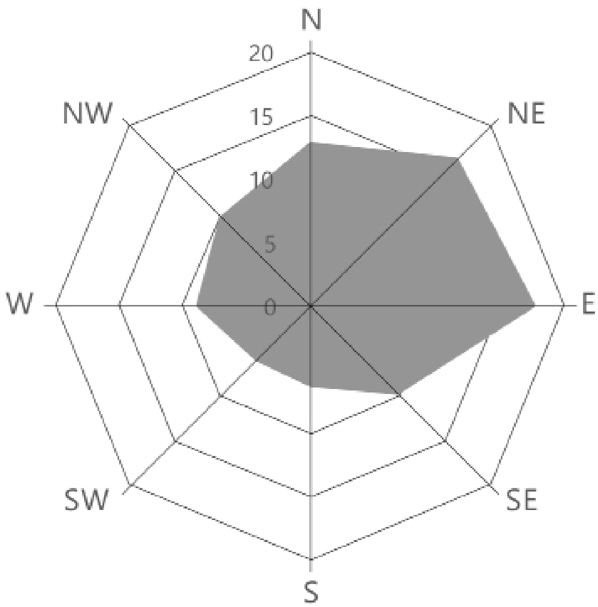


FIGURE 11: Distribution (%) of the area of shallow landslides initiated in 1─6-year-old plantings of *P. radiata* during storms in 2009, 2014, and 2015, by slope aspect, Mangarara and Wakaroa Forests.

numbers were triggered on slopes between 26 and 35° and on slopes >35°. Landslide connectivity decreased with increasing age of the replacement plantings.

Discussion

Globally it is well known there is a cause–effect relationship between forest removal and mass slope failure, especially shallow, rapid, mass movements originating on steep slopes (Croft & Adams 1950; Bishop & Stevens 1964; O'Loughlin 1972; Swanson 1981; O'Loughlin et al. 1982; Sidle et al. 1985; Jakob 2000; Brardinoni et al. 2003; Gyenge et al. 2003; Sidle & Ochiai 2006; Imaizumi & Sidle 2012).

In New Zealand, and largely on account of the predominance of steep slopes, the harvesting of exotic forests originally planted as 'conservation forests' for erosion-control purposes, has become problematic. Here, the presence of thin soils with high available water capacities (McLeod & Rijkse 1999) in sharp contact with impermeable bedrock, increases the susceptibility of forests to landslide-initiation during major storms (Kelliher et al. 1995).

Following forest removal there was a rapid re-wetting of the soil profile. For the first 5 years after the completion of harvesting, and irrespective of the density of replacement plantings, soil moisture levels remained at or near field capacity. Thereafter, interception and evapotranspiration had an increasingly stronger influence on soil-water storage patterns, exemplified by a gradual shortening in the duration of periods of high soil-water content (winter), and consistently increased

TABLE 2: Landslide-related damage and sediment generation rates (t ha^{-1}) on areas of *P. radiata* forest harvested and subsequently replanted at Mangarara and Wakaroa forests.

Forest age class at time of storms	**Harvested area assessed (ha)**	**Landslide scars (ha)**	**Percent of age class lost**	**Sediment generation rate (t ha^{-1})**
1	332.5	3.4	1.0	164
2	169.3	4.1	2.4	391
3	863.9	18.1	2.1	339
4	1164.0	19.4	1.7	267
5	453.0	1.4	0.3	49
6	498.5	0.7	0.1	16
Totals	3481.2	47.1		

levels of soil-water deficit for longer periods during the drier summer months (Fig. 5). Soil-water deficits were consistently greater in the more densely planted stand, suggesting that planting density had a greater influence on the soil-water deficit during the drier summer months than did differences in tree growth performance, such as canopy diameter and tree height. Conversely, there was no discernible difference in soil-water deficit between stands during the wetter winter period, or following thinning, times when evapotranspiration losses decrease and the net addition of water to soil storage is likely to result in a return to field capacity.

The significance of the hydrological record presented in this paper is that the components of the forest water balance were recorded at the same site as for a mature stand of *P. radiata* before harvesting (Pearce et al. 1987), and that repeated sampling in the same neutron probe access tubes is preferable for tracking changes in water content over time because it avoids errors that arise from soil variability (Jackson 1987).

As noted in an earlier study by Jackson et al. (1987), with little or no sign of pronounced seasonal fluctuations in soil-water content below 0.5 m depth, increased levels of soil moisture within the deeper parts of the soil profile are likely to produce high pore-water pressures, and the extraction of increasing amounts of water stored deeper in the soil profile was important in keeping the period of wet soil conditions and associated problems of slope instability to a minimum (Ekanayake & Phillips 1999).

Landslides initiated during storms in 2009, 2014, and 2015 affected just 1% of areas harvested 1 year earlier. This suggests that for the period that soils remain at field capacity, and until the onset of significant root decay, residual root reinforcement is sufficient to minimise landslide initiation and sediment generation rates for up to 1 year after the completion of harvesting (Fig. 9B, Table 2). Thereafter, the decline in root reinforcement of a soft-wood species such as *P. radiata* is particularly rapid, with structural roots (>5 cm diameter) losing half their tensile strength within 15 months of harvest (O'Loughlin & Watson 1979), and once in an advanced state of decay the root systems lose 70 to 80% of their root reinforcement effectiveness (Watson & O'Loughlin 1990). Furthermore, with ~70% of tree-root length and mass located at depths shallower than the landslide failure plane (Watson & O'Loughlin 1990), landslide occurrence and sediment generation rates tended to be highest on areas harvested and replanted 2 years before each storm (Fig. 9B, Table 2). Thereafter the extent of landslide damage and rate of sediment generation slowed with increasing age of the replacement plantings, with the greatest reductions occurring in stands harvested and replanted 5 and 6 years before each storm event (Fig. 9B, Table 2). This reflects the increasing influence of the combined hydrological and mechanical contributions of the replacement plantings to reducing the vulnerability of soils that would otherwise saturate and fail at times when groundwater fluctuations and pore-water pressures were high (Terzaghi 1950; Imaizumi et al. 2008; Imaizumi & Sidle 2012).

Previous studies on root growth rates of *P. radiata* seedlings have tended to assume that net root reinforcement attains a maximum value when lateral root systems of individual trees overlap (Watson et al. 1999; Phillips et al. 2000); that is, a root occupancy index of 1 (Kelliher et al. 1992a). Based on annual root growth rates for seedlings established at 1250 stems ha^{-1} (the recommended planting density for hill-country areas prone to shallow landslides) (Ministry of Forestry 1994), full root occupancy by lateral roots for the range of planting densities considered in this paper probably occurs within 3–4 years of planting (Fig. 9A). However, the roots of juvenile *P. radiata* are generally asymmetrically distributed around the stem (Marden et al. 2016), and because there are often several 'extra-long' roots present, the contribution of the lateral roots of the replacement plantings to net soil-root reinforcement at the time of overlapping with adjacent trees (i.e., site occupancy index = 1) is likely to have been overestimated. Furthermore, while the root systems of 4-year-old trees exhibit strong vertical tap- and sinker-root development, only ~4% of root biomass, 16% of the total root length, and 4.5% of the total root cross-section area occur below the depth of landslide failure planes (Marden et al. 2016), where they would probably have a greater influence on slope stability (O'Loughlin & Ziemer 1982; Watson & O'Loughlin 1990; Watson et al. 1995, 1999; Ekanayake et al. 1997). Thus, for juvenile replacement plantings <5 years old, insufficient root development below the landslide failure plane probably accounts for the continuing occurrence of landslides and elevated sediment generation rates on slopes harvested 3 and 4 years before storms (Table 2). Not until slopes had been replanted for between 5 and 6 years was there clear evidence of a significant reduction in landslide occurrence and sediment generation (Fig. 9B, Table 2).

The finding that landslide occurrence and sediment generation during the first year after harvesting are: (i) initially inversely proportional to the time since harvesting (Fig. 9B); and (ii) are highest when root systems are in an advanced stage of decay before decreasing with increasing age of the replacement plantings, mirrors that recorded by O'Loughlin (1974). In a more recent study, Marden and Rowan (2015) confirmed a similar order of magnitude of sediment generation following a storm event with similar hourly rainfall intensities and return interval as the 2014/15 storms.

The conclusion drawn is that during the first 5-year period after planting and irrespective of planting density, the root systems of *P. radiata* replacement plantings alone do not afford sufficient soil-root reinforcement to fully restore slope stability above the threshold required to mitigate the initiation of shallow landslides (O'Loughlin & Ziemer 1982; Watson & O'Loughlin 1990; Watson et al. 1994, 1995, 1999; Marden & Rowan 1993; Ekanayake et al. 1997). Of concern, particularly for the East Coast forestry sector, is that there is an estimated 82% chance of an extreme weather event coinciding with forest plantings within 5 years of their establishment (Kelliher et al. 1995). It is during this period that increased pore-water pressures associated with soils at or near field capacity probably become the primary factor in initiating landslides (Taylor 1948; Terzaghi 1950), both in juvenile replacement plantings following forest removal and in newly afforested areas.

Furthermore, it might be expected that evapotranspiration from a wet plant canopy, which in turn is influenced by the vegetation's characteristics, would increase with the closing of the canopy, and be influenced by stand density and especially by tree height (Kelliher & Scotter 1992). For the range of planting densities discussed, however, neither the influence of planting density on canopy interception (Fahey 1964; Jeffrey 1970; Pearce et al. 1987) in dampening and smoothing rainfall intensities and throughfall (Schellekens et al. 1999; Xiao et al. 2000), nor the effect of increasing evapotranspiration effects in delaying saturation of the soil mantle (Hallin 1967), proved effective in mitigating landslide initiation in plantings <5 years old. Indeed, in an earlier assessment of the extent of storm-initiated landslides in exotic plantation forests during Cyclone Bola (1988), the incidence of storm-initiated landslides was shown to be highest in forest stands <5 years old. Older stands planted at the recommended 1250 stems ha^{-1} provided a significant level of protection, coinciding with the timing of canopy closure at around 7.5–8 years after planting, while forest stands 6–8 years old established at the same density provided a moderate level of protection (Marden & Rowan 1993). Had plantings at that time been established at densities less than 1250 stems ha^{-1}, and more typical of present-day planting regimes, the level of protection would probably have been considerably less and proportional to planting density.

Early establishment regimes for *P. radiata* forests planted for the purpose of controlling mass movement erosion required stands to be planted at 1250 stems ha^{-1}, which were then thinned to a final stand density of ~450 stems ha^{-1} at around year 11, ~6 years after canopy closure. Evapotranspiration would be expected to decrease following the opening of the canopy at the time of thinning, increasing the susceptibility of slopes to landslides, and be influenced by stand density. As demonstrated earlier, canopies overlapped significantly (canopy closure index of 1.68) before the stand planted at ~1000 stems ha^{-1} was thinned to ~400 stems ha^{-1} in year 11, resulting in a ~36% reduction in canopy cover and a 12% decrease in rainfall interception. This result is not too dissimilar to the 19% decrease in rainfall interception recorded following the thinning of a shorter (9 m tall) and younger (7-year-old) stand of *P. radiata* from 1070 to 450 stems ha^{-1} (Kelliher et al. 1992b).

With evapotranspiration rates decreasing by only 6%, the 83 mm (6% of annual rainfall) of added water had no discernible effect on soil moisture storage during the 2-year monitoring period after thinning. This suggests that the risk of pore-water pressures reaching thresholds considered conducive to the initiation of shallow landslides before stands regain canopy closure and evapotranspiration rates return to pre-thinning levels, would probably be minimal (Pearce et al. 1987). Also, although thinning would result in a significant decrease of >50% of the total live root biomass and a reduction in effective root reinforcement by 33% yr^{-1} (O'Loughlin & Watson 1979), the remaining trees – each with a 3.5 m wide network of lateral roots (Watson & O'Loughlin 1990) and root production of ≥7 to 8 t yr^{-1} (Watson et al. 1995) – would probably provide sufficient root reinforcement to maintain a slope safety factor above the threshold at which slopes would otherwise fail.

As demonstrated here, trees planted at a more equitable 5 × 5 m spacing (500 stems ha^{-1}) develop larger canopies sooner than do trees planted at higher densities (Fig. 6). Given that root spread generally exceeds that of the canopy diameter, a likely outcome is a more equitable and wider distribution of roots of larger diameter, and a greater volume of root biomass per area of soil between trees, such that a small reduction in planting density would not necessarily compromise their erosion control effectiveness. However, given similar site and climate conditions, it would also be expected that the higher the planting density, the sooner would root reinforcement influence a return to pre-harvest stability thresholds. Indeed, the success of the early exotic forest plantings in restoring slope stability has been attributed to high root mass production and an efficient utilisation of available soil water (Pearce et al. 1987) of plantings established at between 2000 and 2200 stems ha^{-1}, thereby producing the stronger and drier soils required for maintaining long-term slope stability (Zhang et al. 1991, 1993; Marden et al. 1992).

The preferred present-day planting regime is to establish high-quality seedlings at a lesser density of 800 stems ha^{-1} (4 × 3 m) and to thin stands to 400–500 stems ha^{-1} earlier; that is, in years 7 or 8, before they attain canopy closure. Thinning early would further delay a return to evapotranspiration rates typical of a closed-canopy stand by up to 6 years, but projected root growth rates and a concomitant increase in root reinforcement during this period could conceivably prove sufficient to ameliorate landslide occurrence on sites with low inherent erosion susceptibility. Conversely, for sites with a high erosion susceptibility rating, where, due to harsh physical site conditions (e.g. thin skeletal soils with limited capacity for soil moisture storage), tree survival and root and canopy growth rates are likely to be poor, thinning would further delay the dampening effect of the canopy on throughfall for longer, thereby increasing the likelihood of soil saturation occurring. It is during this interval (the window of vulnerability) before a closed canopy re-forms, and when root reinforcement is incomplete, that the risk of landslide occurrence is greatest (Watson et al. 1994).

Although the initial planting density has little effect on reducing the occurrence of shallow landslides until after canopy closure, a higher initial planting density in areas most susceptible to the initiation of landslides would allow for greater flexibility in the selection of trees to be removed/retained during thinning. Furthermore, depending on the final thinning regime, the higher the density of the remaining trees, the quicker canopy closure and an effective soil-root reinforcement network would be expected to recur, thereby minimising the risk of soil saturation and, ultimately, of landslide occurrence.

While landslides in well-established stands of mature exotic forest are not uncommon (Phillips et al. 1990; Marden et al. 1991; Marden & Rowan 1993; Fransen 1998), the conclusion drawn is that with increasing stand density and age, and as evapotranspiration rates return to near pre-harvest levels and soil water patterns exhibit increasingly stronger seasonal fluctuations (Fig. 5), combined with the increasing influence of root reinforcement, the probability of landslide initiation is effectively minimised (Sidle et al. 1985; Phillips 1988; Hicks 1989; Phillips et al. 1990; Marden et al. 1991, 1995; Marden & Rowan 1993; Hancox & Wright 2005; Dymond et al. 2006; Marden 2004, 2012). Thus, any forest management strategy that aims to maintain a near-closed canopy and an effective live soil-root reinforcement system by delaying thinning until after the canopy has closed, and at which time evapotranspiration rates reach a maximum of 35% of annual rainfall, has the potential to reduce the risk of soils reaching field capacity sufficiently to minimise the occurrence of storm-initiated failures on slopes identified as high risk. That said, estimations of the period until net root reinforcement is restored are complicated by the naturally high spatial variability in root characteristics and architecture (Marden et al. 2016). These are influenced by factors including heterogeneities in soil type and texture (Watson & O'Loughlin 1990), impenetrable bedrock at shallow depths, high water tables that limit vertical root development (Ray & Nicoll 1998), slope aspect and steepness (Chiatante et al. 2003), and the prevailing wind (Nicoll & Duncan 1996; Marden et al. 2016).

Slope steepness and aspect also have a significant influence on landslide frequency, sediment production, and landslide connectivity to stream channels following harvesting: each increasing with increasing slope steepness (Fig. 10) and predominantly on north- to north-east-facing slopes (Fig. 11). As with previous landslide-triggering storms, including Cyclone Bola (Marden et al. 1991, 1995, 2016; Marden & Rowan 1993), these relationships are both a reflection of the vulnerability of the Tertiary terrain to the initiation of shallow landslides and the approach direction of storms.

Alternative forest management options on erosion-prone land

Exotic forests can and do play a significant role in environmental improvement, not only in terms of reduced on-site erosion (Marden 2012) and rates of sediment generation (Marden et al. 1995, 2008, 2011, 2012), but also through improvements in water quality during the rotation (O'Loughlin 1995; Parkyn et al. 2006; Quinn & Phillips 2016), and increased rates of carbon sequestration (McLaren 1996). However, when forests are harvested the potential for rainstorms to trigger shallow landslides and debris flows that entrain logging residues (slash), and cause impacts beyond the forest boundary, increases (Phillips et al. 2012; Fuller et al. 2016).

Long before the harvesting of forested areas originally established in *P. radiata* over extensive areas of severely eroding hill country farmland, concerns had been expressed over their long-term future as "production forests". Approximately 30 years later, forest removal has exposed significant areas of this vulnerable terrain to multiple and localised storms, resulting in an increase in the supply of sediment and woody debris to stream channels and its transportation beyond forest boundaries. In many respects the impacts of these storms on both the forest sector and downstream communities were a legacy of decisions to blanket-plant extensive areas with predominantly exotic species as a quick and cheap means of mitigating further damage to the worst of the areas affected by erosion. Although seldom used, the staged harvesting of a percentage of a large catchment area of *P. radiata* forest seems a logical way to spread the risk of an erosion-triggering storm event occurring during the window of vulnerability while maintaining the overall protective effect of a forest cover at a medium-size catchment level (O'Loughlin 2005). The difficulty here is that what are referred to as 'localised storm cells' affect areas much larger than a single catchment. Furthermore, and as previously mentioned, a significant contributing factor to the extent and severity of damage sustained to forested areas within the East Coast region is that, as a legacy of planting one dominant tree species within a short time frame, the harvesting of these forests has occurred simultaneously over consecutive years and across contiguous areas of hill country, thereby compounding the magnitude of storm-related damage incurred both on-site and off-site.

Over recent years the influence of the Resource Management Act (RMA), the development of an improved National Erosion Susceptibility Classification (ESC) specifically to support the National Environmental Standard for Plantation Forests (NES-PF), and terrain analysis to support improved hazard and risk assessment at detailed operational scales have combined to improve many aspects of exotic forest management (Phillips et al. 2017). For example, before the introduction of the NES-PF, planting was permitted to the edge of streams, and inevitably when harvested this resulted in increased deposition of sediment and slash in streams. More recently, planting set-back distances for commercially harvestable species have been introduced and together with the goal of replanting/reversion of a riparian 'buffer' comprising indigenous species, the intention is to reduce sediment and slash input into streams during future harvests. However, in debates about the sustainability of exotic forests in this region's steepland areas and what should be sustained, the prevention of the loss of the soil mantle depth and its water-holding capacity, and of carbon, nitrogen, and nutrients for future rotations, should be paramount (O'Loughlin 1995).

Both the Erosion Control Classification and NES-PF provide a coarse screening tool with which to assess 'risk' and further improve risk analysis at a forest operational scale. However, a fit-for-purpose landslide susceptibility methodology and an improved understanding of the magnitude and frequency of triggering events, together with morphometric connectivity models that predict the likelihood of sediment delivery to streams following landslide initiation (Spiekermann et al. 2022), are still needed (Basher et al. 2015; Marden et al. 2015). The adoption of such models is critical for identifying areas of high and very high risk, and to inform decisions on whether these areas remain in exotic production or transition to an indigenous and/or mixed species forest as options likely to provide longer-term mitigation against the initiation of shallow landslides, and further reduce sediment delivery to streams. For the forestry sector to carry on business as usual is likely to invite further regulation.

Solutions will require the separation of areas clearly designated as:

(i) land classes with ongoing economic value from timber and where future harvesting is unlikely to jeopardise the initial erosion-control function of the forest;

(ii) high-risk areas with potential economic value over a longer time frame, and where timber extraction would be possible while maintaining a near-continuous canopy cover is paramount;

(iii) very-high-risk areas justifiably designated for retirement and reversion to a permanent forest cover (Marden & Seymour 2022).

For many high-risk areas, and where site conditions prove suitable, there is a good case for establishing longer-rotation species, for example, Douglas-fir (*Psudotsuga menziesii*) and/or coppicing species including coastal redwood (*Sequoia sempervirens* D.Don). The advantage of coppicing species is that the soil-root reinforcement effect during the post-harvest period is probably sufficient to prevent residual slope failure, particularly at times when, due to the loss of transpiration, the added water to the soil profile is likely to reach or be close to field capacity. Alternatively, very-high-risk areas within existing exotic forests could be left standing as a permanent carbon forest (Climate Change Commission 2021).

Centre stage of the topics for debate on longer-term solutions for mitigating slope failure in high-risk areas is the role and effectiveness of indigenous shrubland reversion. Reversion can be achieved passively (i.e., without human intervention) as a low-cost way of creating a permanent vegetation cover, or actively managed by planting. For shrubland reversion to passively recolonise and successfully stabilise high-risk areas, the critical variables will include the size, density, and area affected by landslides, and the presence of a viable seed source of species typical of those that colonise bare slopes during the early stages of reversion (Marden et al. 2005, 2018). Secondary influences include the control of browsing animals, as sustained browsing of the canopy and understorey of the more palatable species increases the susceptibility of slopes to failure (Wallis 1966; James 1969). Shrubland reversion in this region is dominated by the early-colonising species mānuka (*Leptospermum scoparium* J.R.Forest & G.Forst) and kānuka (*Kunzea ericoides* (A.Rich) Joy Thomps.), and, as demonstrated by Marden and Rowan (1993) and by Bergin et al. (1995), the timing and degree of protection they provide is largely dependent on their age and density at the time of a major storm. For example, at the time of Cyclone Bola (1988), fully stocked stands of mānuka and kānuka provided a greater level of protection against the initiation of landslides than did 6–8-year-old stands of exotic forest but were less effective than exotic forest >8 years old (Marden & Rowan 1993). A significant drawback, however, is that passive reversion would take twice as long (~16 years) before the root systems would afford a comparable level of protection to that of *P. radiata* (Watson & O'Loughlin 1985; Ekanayake et al. 1997; Watson et al. 1994, 1999; Watson & Marden 2004) and require an average stand density of ~13,000 stems ha^{-1} (Bergin et al. 1995).

Alternatively, options for managed reversion include either:

(i) the planting of expansive areas with nursery-raised seedlings (Marden et al. 2018) including mānuka (Marden et al. 2020), or

(ii) using existing areas of shrubland reversion as a 'nurse-crop' into which alternative exotic and/or indigenous tree species are planted for timber production.

Option (i) generally involves the planting of mānuka at densities of ~1100 stems ha^{-1}. At this density, evapotranspiration rates and root-reinforcement contributions towards improving slope stability take longer than would be the case in fully stocked stands of passively reverting shrubland, where mean stand densities often exceed 20,000 stems ha^{-1} at age 10 years (Bergin et al. 1995). Increasing the planting density and reducing early seedling mortality by better management of weed competition and/or by replacing dead plants (blanking), would significantly improve the erosion mitigation effectiveness of this treatment option. The underplanting of shrubland areas (option ii) with exotic species, including blackwood (*Acacia melanoxylon* R.Br.) and cypresses (*Cupressus macrocarpa* Gordon and *C. lusitanica* Miller), and indigenous tree species including tōtara (*Podocarpus totara* D.Don) and puriri (*Vitex lucens* Kirk), in lanes cut through shrubland or in gaps, would have the advantage of maintaining critical environmental values that would otherwise be lost if the shrubland were to be cleared and replanted (Bergin et al. 1995).

In the absence of stricter regulatory controls on forest establishment and removal, progress towards implementing change will be reliant to a large degree on the will of forest companies to adopt alternative management options, with or without financial incentives in the form of government subsidies, to better target erosion mitigation. Change will, however, have implications for the financial viability of many of the exotic forests located in the East Coast region (Lambie et al. 2021), where issues of sustainability will, in the longer term, lead to significant areas of high-risk production forestry land transitioning to a permanent forest cover.

With climate change predictions for the East Coast region of the North Island suggesting that extreme storm events will become more common, it is inevitable that there will be a need for additional areas of landslide-scarred pastoral hill country within the Tertiary terrain to be afforested. For these areas, and key to avoiding past mistakes, is the adoption of improved models and tools for identifying areas of high risk to the initiation of landslides, and their connectivity to stream channels during all stages of a rotation, but particularly after harvest. Forearmed with this knowledge, for areas deemed to be very high risk within terrain designated for future afforestation, a more environmentally and cost-effective option would be to select species, whether exotic or indigenous, that are better adapted to survive and provide long-term/permanent ground and canopy cover, than simply blanket-plant with *P. radiata* and risk destabilising slopes during successive harvests.

Conclusions

In the East Coast region of the North Island of New Zealand the harvesting of production forests on inherently unstable terrain predisposes significant areas to shallow landsliding. Landslides are primarily a response to rainfall-induced increases in pore-water pressures, elevated water-table levels, and higher soil-water content within the rooting zone for longer periods before returning to pre-harvest levels of soil drying and recharge. The progressive loss of root strength has a secondary influence until a new and effective soil-root reinforcement system is established. Planting density and the timing of silvicultural regimes can influence the duration of the window of vulnerability and susceptibility of harvested areas to landslides until replacement plantings attain an equivalent rate of evapotranspiration to that of a closed canopy forest.

The challenge ahead is to ensure the sustainability of rotational exotic production forests by better understanding the cause–effect relationships following forest removal on slopes considered to be susceptible to the initiation of shallow landslides. Equally important is the influence of planting density regimes and silvicultural practices in altering the hydrology and mechanical reinforcement properties during the post-harvest replanting period. Also, the adoption of improved erosion susceptibility and risk-assessment tools together with morphometric connectivity models will become critical for underpinning management strategies and policies aimed at mitigating storm-related mass movement and sediment production, particularly following harvesting.

Furthermore, for existing areas of exotic production forest and for areas designated for future afforestation within the Tertiary terrain, the adoption of alternative management options will be paramount in alleviating the risk of slope failure and resultant soil loss. The planting of high-value timber species with a longer rotation length, including consideration of coppicing species, and specifically targeted to areas identified as high risk to landslide initiation, would greatly minimise the risk of soil saturation resulting in slope failure after harvest. Areas considered very high risk and unsuited to rotational harvesting will ultimately require transitioning to a permanent indigenous forest cover.

Postscript

Since the storms documented in this paper, there have been four major weather-related events: ex-tropical Cyclones Debbie and Cook (April 2017), ex-Cyclone Hale (January 2023) and Cyclone Gabrielle (February 2023). Each storm has caused further significant on-site landslide damage within recently harvested areas in the exotic forest estate and on hillside pastoral land. The result has been widespread flooding and deposition of silt and woody debris on areas of pastoral floodplain, significant damage to the regions roading infrastructure, and the degradation of waterways and ecological habitats. An inquiry of all land-use management practices of planted-forest and pastoral land uses in the worst of the affected areas has been confirmed.

https://www.1news.co.nz/2022/03/23/live-gisborne-hit-by-3-months-worth-of-rain-in-24-hours/

https://www.rnz.co.nz/national/programmes/checkpoint/audio/2018878138/napier-mp-forestry-mininster-confirms-inquiry-into-east-coast-soil-erosion

Competing interests

The authors declare that they have no competing interests.

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Abbreviations

DBH: diameter at breast height
ENSO: El Niño / Southern Oscillation
GF: growth and form of known parent trees.
NES-PF: National Environmental Standard for Plantation Forestry.
PVC: polyvinyl chloride
RCD: root collar diameter

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Author contributions

MM is the principal author; DR assisted with data collection; AW contributed to data analysis and the writing of this paper.

Availability of data and materials

Please contact the corresponding author for data requests.

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