EROSION PROCESSES AND THEIR CONTROL IN NEW ZEALAND

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ABSTRACT: Erosion control is an important regulating ecosystem service in New Zealand because of the diversity and extent of erosion processes. Erosion rates in New Zealand are naturally very high by world standards as a consequence of the dominance of steep slopes, erodible rocks, generally high rainfall and common high-intensity rainstorms. Recent and extensive deforestation and the introduction of large numbers of grazing animals have increased erosion rates. The most widespread and active type of erosion is rainfall-triggered shallow landslides, but other mass movements (earthflows and slumps), gully, surface (sheet, rill, wind) and streambank erosion are locally significant. Temporal trends in erosion in New Zealand are poorly known because there is no comprehensive monitoring programme and many of the widespread types of erosion, such as shallow landslides, are triggered by storm events that have high temporal variability. Biological methods of erosion control are by far the most widely used, with structural methods used locally for earthworks and bank erosion. Although a large range of vegetation types and species have been used to control erosion, space-planted poplars and willows have been the most widely used soil conservation plants in New Zealand since they can be established as poles in the presence of grazing animals, and are appropriate for the control of landslide, earthflow, gully and streambank erosion. Afforestation is also used for erosion control in the worst eroding areas. Closed-canopy tall woody vegetation typically reduces landsliding in large storms by 70–90%. Space-planted trees can reduce landsliding by a similar amount so long as the tree establishment/survival is adequate. Trees have also been shown to reduce rates of earthflow movement and gully erosion. Maintaining good ground cover is the key to reducing rates of sheet, rill and wind erosion. Few studies have assessed the value of erosion control in an ecosystem services context, and most of the available research focuses on the value of afforestation with little information on other erosion control technologies. Forests have significant benefits for erosion control and carbon storage, but reduction in water yield neutralises these benefits in water-short areas. Afforestation affects other ecosystem services by improving water quality and aquatic habitat, reducing greenhouse gases by sequestering carbon and nitrogen, protecting biodiversity, and contributing to soil and nutrient retention.

Key words: afforestation, biological erosion control, carbon, ecosystem services, landslides.

INTRODUCTION

The benefits provided by erosion control are a key ecosystem service in New Zealand because of the widespread occurrence of many different forms of erosion. Most typologies for describing ecosystem services include erosion control as a regulating service (e.g. de Groot et al. 2002; Dominati et al. 2010), with some of the techniques used for controlling erosion also impacting on other ecosystem services such as climate regulation, flood mitigation, and water purification.

Erosion rates in New Zealand are very high by world standards, with about 200 megatonnes of soil delivered to the ocean each year (Hicks et al. 2011). While New Zealand makes up ~0.1% of the global land mass, it discharges 1–2% of annual average yields of sediment to the ocean (Hicks et al. 1996). The country has a very-high-energy geomorphic environment as a consequence of its location on an active plate boundary in the mid-latitude zone of strong westerly winds. Steep slopes, high rates of tectonic activity and volcanism, generally high rainfall and common high-intensity rainstorms all contribute to naturally high rates of erosion (Soons and Selby 1992; Hicks et al. 2011). In addition, deforestation of much of the country over the last millennium, the introduction of large numbers of grazing animals, and intensive land use in some areas have accelerated rates of erosion (e.g. Page et al. 2000; Glade 2003). A wide diversity of erosion processes occur with strong regional patterns in the types and activity of erosion related to climate, geological setting and land use (Cumberland 1944; Eyles 1983, 1985; Glade 2003; Basher et al. 2010).

National awareness of hill country soil erosion was polarised by storm events in the 1930s and 1940s, mostly in the Esk Valley, Wanganui and Waipaoa catchments, and Marlborough (Committee of Inquiry 1939; Roche 1994; Hicks and Anthony 2001). These and subsequent storms initiated severe soil erosion on recently developed pastoral hill country. In addition, the state of the South Island high country was ascribed to severe erosion caused by decades of frequent burning and overgrazing by sheep and rabbits (Gibbs and Raeside 1945). By 1941 concerns about soil erosion resulted in the passing of the Soil Conservation and Rivers Control Act and the establishment of catchment boards who were given responsibility for undertaking experimental, preventative and remedial soil conservation works. Since that time a substantial effort has gone into establishing practices suitable for erosion control in New Zealand, mainly using plant materials, but also targeted use of structural erosion control methods (summarised in Hicks and Anthony (2001)). Structural methods of erosion control, most widely used for managing the effects of earthworks and for river control, are used locally and are not discussed further.

In 2001 it was estimated that the annual expenditure on preventing erosion was approximately $24 million whereas it was (conservatively) estimated that the damage caused by erosion costs $103 million (Krausse et al. 2001). In addition, the authors highlight the implications of the demise of centralised funding for soil and water conservation through the National Water and Soil Conservation Organisation and the marked decline in direct government expenditure in this area.

This paper reviews the characteristic types and distribution of erosion in New Zealand, temporal trends in erosion and the influence of land cover on erosion before summarising recent research on erosion control in an ecosystem services context.

EROSION PROCESSES

Much of New Zealand is hilly or mountainous, with 60% of the land being above 300 m elevation and 70% hilly (12–25º) or

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steep (>25º). New Zealand lies at the boundary of the Pacific and Australian tectonic plates, resulting in high rates of uplift, frequent earthquakes, and common crushed and weakly lithified rocks that are prone to erosion (Soons and Selby 1992). Rainfall ranges from <500 mm yr^{-1} to >10 000 mm yr^{-1} with strong east–west and topographic gradients. The climate is characterised by frontal storms and extra-tropical cyclones that commonly bring high rainfalls and are the trigger for much of the erosion (Glade 1998). High winds are common in the east of the country, in the rain shadow of the main mountain ranges, and can cause severe wind erosion especially in the South Island (Basher and Painter 1997).

The erosion problems in New Zealand are exacerbated by recent and extensive deforestation. Polynesian settlers were the first humans to reach New Zealand about 800 years ago and caused widespread deforestation (of about 50% of the forest area), especially in the east of the South Island (e.g. McGlone 1983; McWethey et al. 2009). After the arrival of European settlers in the early 19th century extensive areas (a further 30% of the country) were cleared for farming and timber, and large numbers of grazing animals were introduced to the transformed landscapes. Within a few decades a serious erosion problem became evident, particularly in the soft rock hill country of both islands and in the hard rock greywacke terrain of the eastern South Island high country (Roche 1994). As late as the 1980s farmers were being offered subsidies, through Land Development Encouragement Loans, to convert 'unproductive' steep erosion-prone hill country, under scrub and forest cover, to pastoral farming despite the known erosion problems of this land.

All the main types of erosion occur in New Zealand:

- Surface erosion (sheet, rill and wind)
- Gully erosion
- Mass-movement erosion (shallow and deep landslides, slumps, earthflows)
- Streambank erosion

The regional distribution of erosion was first described by Cumberland (1944) and more recently has been comprehensively mapped (Eyles 1983, 1985) as part of the New Zealand Land Resource Inventory (NZLRI).

Because of the dominance of hilly and mountainous terrain the most widespread type of erosion is mass movement. A wide variety of landslide types occur in the New Zealand landscape,

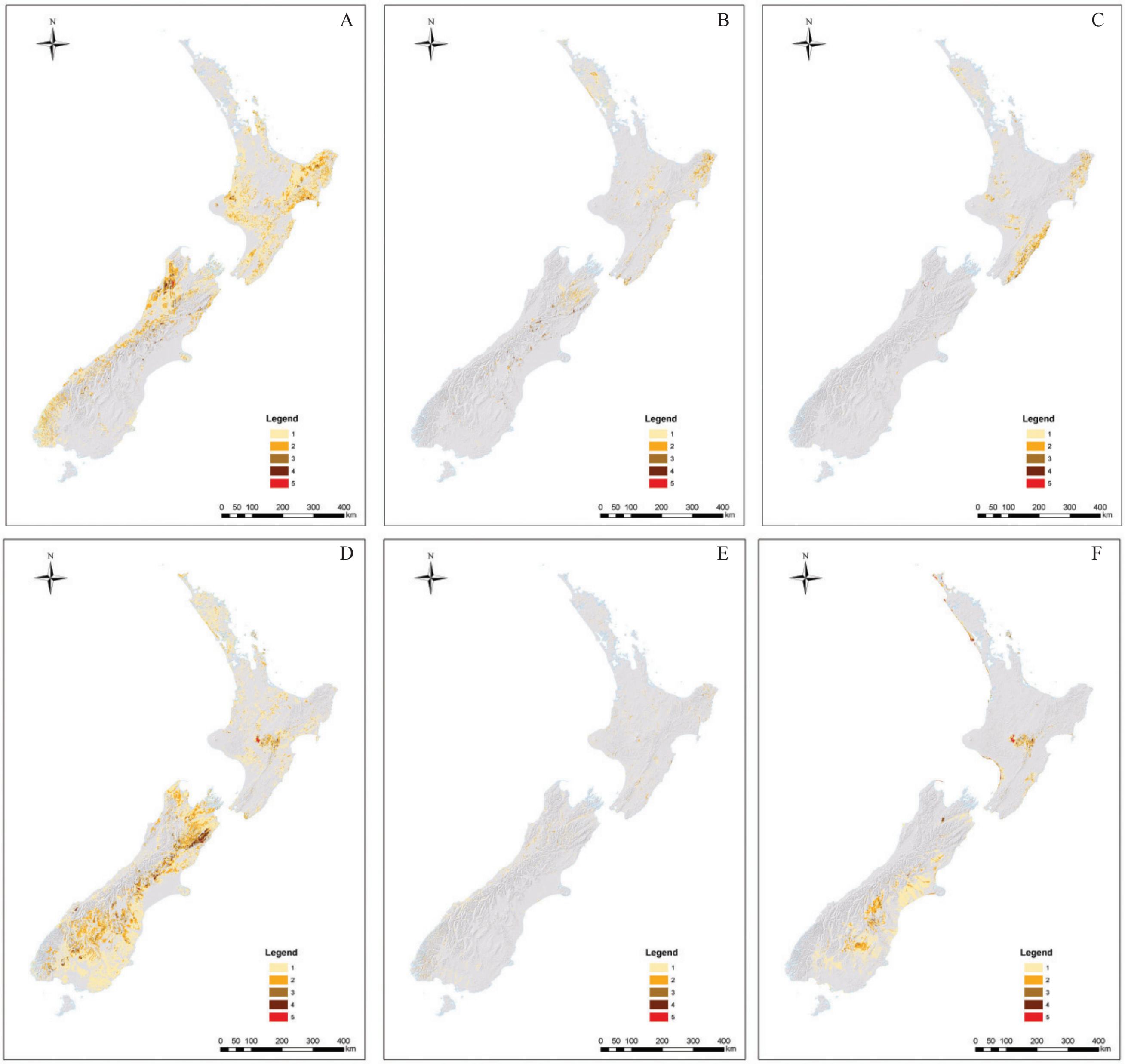

FIGURE 1 Distribution and severity of the main forms of erosion in New Zealand derived from the New Zealand Land Resource Inventory: (a) shallow landslides, (b) gully erosion, (c) deep mass movement erosion, (d) sheet and rill erosion, (e) bank erosion, (f) wind erosion.

ranging from small, shallow rapid failures to large, deep, creeping rock failures. The most common types are shallow, rapid slides and flows involving soil and regolith, which occur during rainstorms (Glade 1998; Crozier 2005). They are typically characterised by small scars and long narrow debris tails where much of the landslide debris is redeposited downslope. This type of landslide can be triggered by small rainfall events after prolonged wet periods leading to high antecedent soil moisture conditions or by individual, high-intensity-rainfall, storm cells. These landslides have been referred to as earthflows by Crozier (1996) and mapped as soil slips in the NZLRI (Eyles 1983, 1985). They are widespread throughout most of New Zealand on slopes over 15° (Figure 1a), and are particularly extensive in the Tertiary soft rock hill country of the North Island (Gisborne–East Coast, inland Whanganui–Taranaki–Manawatu, southern Hawke's Bay, Wairarapa), and the South Island mountainlands and hill country. Debris avalanches are similar types of failures with longer run-out zones and a deep narrow scar. They are common on steep forested slopes in the mountains of the North and South Island and also occur in steep grasslands in the South Island. These types of shallow, rapid failures are referred to collectively by the generic term shallow landslides throughout the remainder of this paper. Slumps and earthslips are deeper failures that have also been recognised in New Zealand (Eyles 1983, 1985) but have a very restricted distribution. Large-scale failures in bedrock are also common in the New Zealand landscape (e.g. Crozier et al. 1995; Hancox and Perrin 2009).

Gully erosion occurs as linear features cut by channellised runoff and as large, complex mass-movement–fluvial-erosion features that are typically amphitheatre-shaped (Marden et al. 2012). It is most common in the soft rock hill country of the East Coast North Island, on crushed argillite and mudstone, and in the North and South Island mountainlands (Figure 1b). It also occurs in Northland and the Volcanic Plateau (Eyles 1983, 1985). An additional form is tunnel gully erosion, where water moves down through the soil until it reaches a less permeable layer where it concentrates to form an underground channel. As this widens, the roof can collapse forming a surface gully. This form of erosion is common in the loess-mantled hill country of the South Island and the loess- and tephra-mantled hill country and hill country on deeply weathered sandstone in the North Island (Lynn and Eyles 1984).

Earthflow erosion is the slow movement of soil and associated regolith, along basal and marginal shear planes, and with internal deformation of the moving mass (Eyles 1983, 1985; Lynn et al. 2009). Earthflows may be shallow (<1–2 m) to deep-seated (>10 m, and typically 3–5 m). Deep-seated earthflows typically occur on slopes between 10° and 20° and can cover large areas of a hillslope, while shallow earthflows are more common on slopes >20°, and are smaller in area (Lynn et al. 2009). Earthflow erosion occurs mostly in the North Island, and is most extensive on crushed mudstone and argillite in the Gisborne – East Coast area, Wairarapa and in southern Hawke's Bay (Figure 1c). It also occurs in Northland, the soft rock hill country of inland Taranaki and the southern Waikato. Small areas occur on mudstone in North Canterbury, South Canterbury and coastal Otago.

Sheet erosion is the detachment of soil particles by raindrop impact and their removal downslope by water flowing overland as a sheet instead of in defined channels or rills. Two processes contribute: (a) rainsplash detaches soil particles and is strongly influenced by rainfall intensity; (b) the loosened particles are transported by overland flow, which is influenced by storm characteristics (infiltration-excess overland flow) and antecedent moisture conditions (saturation overland flow). Frost lift can also contribute to loosening surface soil particles in the South Island high country. Rill erosion (in small, ephemeral channellised flow) is commonly associated with sheet erosion and has similar controlling factors. It has not been widely studied in New Zealand nor is it widely mapped (Eyles 1983, 1985).

Sheet erosion is widely distributed in New Zealand (Figure 1d), particularly in the South Island, based on the presence of bare ground assumed to be eroding. In the South Island it is common in the dry hill country and mountainlands of inland Marlborough, Canterbury and Central Otago, while in the North Island the most affected areas are tephra-covered slopes of the Volcanic Plateau. Typically sheet erosion occurs on areas of bare ground, such as cultivated slopes (Basher and Ross 2002; Basher et al. 2004), forestry cutovers (Marden and Rowan 1997; Phillips et al. 2005; Marden et al. 2006, 2007), unsealed roads and tracks (Fahey and Coker 1989, 1992), stock tracks (Rosser 2006), earth-works associated with farming, forestry or other land uses (Hicks 1994), and on erosion features such as landslide scars, debris tails, and gullies. Sheet erosion also occurs in diffuse areas of bare ground within pasture that is heavily grazed or affected by drought. In addition to the presence of bare ground, factors that influence surface erosion include slope angle, length and aspect, soil texture, compaction, and rainfall, especially intensity and duration.

Streambank erosion is one of the least understood erosion processes in New Zealand. There are few published studies of bank erosion in New Zealand (Basher et al. 2012). A wide variety of fluvial and mass movement processes contribute to bank erosion (see review by Watson and Basher (2006)) and result in a wide range of styles of bank erosion. While bank erosion was mapped in the NZLRI (Figure 1e) it is undoubtedly more wide-spread than shown in this database. It is common along rivers and streams throughout New Zealand and has been one of the most common processes mitigated by both biological and structural erosion control.

Wind erosion has long been a concern in New Zealand with dust clouds commonly observed blowing off cultivated paddocks. The extent and significance of wind erosion was reviewed by Basher and Painter (1997). The NZLRI shows wind erosion affecting 13% of New Zealand, with quite different distribution patterns in the North Island and South Island (Figure 1f). The most severe wind erosion is mapped on small areas of coastal sand dunes of both islands and the Volcanic Plateau in the central North Island. Slight wind erosion is mapped over large areas of the eastern South Island. Salter (1984) suggests that 27% of New Zealand is susceptible to moderate to extreme wind erosion.

TRENDS IN EROSION

Temporal trends in erosion in New Zealand are poorly known because there is no comprehensive monitoring programme for erosion. In addition, many of the widespread types of erosion, such as shallow landslides, are triggered by storm events that have high temporal variability.

State of the environment reporting of erosion at national level has been limited to reporting 'soil intactness of erosion-prone land' (Ministry for the Environment 2007). This is derived by character-ising trends in the vegetation cover (derived from the Land Cover Database, LCDB) of erosion-prone land (defined as land with a slope >21º, with severe to extreme potential for erosion and under pasture). Table 1 shows the change in erosion-prone area between

1997 and 2002. The percentage change from pasture is small, with results showing a reduction of just over 36 000 hectares nationally between the two periods of land-cover monitoring (3% of the total area of erosion-prone land). Just over half of this total was in the Gisborne, Hawke's Bay, and Manawatu–Wanganui regions (17 481 hectares in total). In the South Island, the Marlborough and Tasman regions experienced a combined pastoral land cover change of 4119 hectares. LCDB analysis shows that of the 36 400-hectare reduction in pasture on erosion-prone hill country, 36 300 hectares were converted to exotic forestry or retired and left to revert to scrub. This indicator only provides trends relevant to shallow landslides, gullies and earthflows.

TABLE 1. Summary of area of hill country erosion-prone land under pasture in 1997 (LCDB1) and 2002 (LCDB2) (from Ministry for the Environment 2007). Negative values indicate a land use change to forestry or reversion

	Erosion-prone area (ha) in pasture		**Change in area 1997–2002**	
Region	**1997**	**2002**	**(ha)**	**(%)**
Northland	67 723	65 832	−1691	−2.50
Auckland	13 101	12 988	−53	−0.40
Bay of Plenty	27 000	25 855	−1104	−4.09
Waikato	116 049	112 315	−3680	−3.17
Gisborne	167 141	158 382	−8151	−4.88
Hawke's Bay	113 128	110 416	−2537	−2.24
Manawatu	230 585	223 535	−6793	−2.95
Taranaki	40 580	38 444	−2136	−5.26
Wellington	54 281	51 387	−2794	−5.15
Nelson	1612	1535	−76	−4.74
Tasman	24 249	22 697	−1012	−4.17
Marlborough	75 042	71 946	−3107	−4.14
Canterbury	113 995	113 770	−220	−0.19
West Coast	4623	4592	−16	−0.35
Otago	101 531	101 236	−294	−0.29
Southland	26 083	25 437	−646	−2.48
North Island	829 587	799 154	−30 433	−3.67
South Island	347 134	341 213	−5921	−1.71
Total	1 176 721	1 140 367	−36 354	−3.09

Dymond et al. (2010) use a modelling approach to estimate national trends in erosion associated with agriculture. LCDB2 was used to identify agricultural land in 2002 and the New Zealand Empirical Erosion model (NZeem®) used to calculate the mean erosion rate from that land. A time sequence of annual sediment yields from agriculture (Figure 2) was calculated by assuming a constant rate of erosion through time, constant rainfall through time, and using trends in the total area of agricultural land reported by the Department of Statistics. This analysis suggests a reduction in erosion since the early 1980s caused by an increase in plantation forestry and scrubland. The downward trend in total sediment yield only reflects changes in land use and does not represent the actual change in sediment yield because it ignores any effects resulting from climatic variation through this period, which may have had a greater effect than land use (Dymond et al. 2010).

More recently, regional councils have developed a protocol for assessing land stability (Burton et al. 2009). This procedure is based on point analysis of aerial photos and characterises

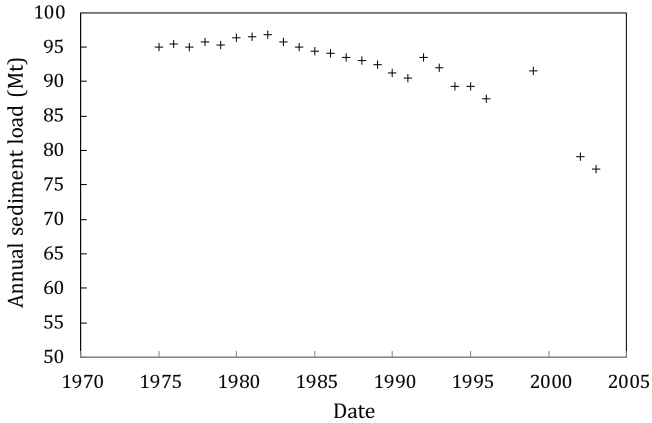

FIGURE 2 Total annual sediment yields from agriculture in New Zealand over the past 30 years (after Dymond et al. 2010).

whether soil is stable, unstable but inactive (erosion-prone), recently eroded (now revegetating), or freshly eroding (bare) and is essentially a survey of the extent of bare ground. The nature of disturbance is also recorded (natural or land-use-related erosion, type of erosion). This technique is now being used by many regional councils including Auckland (Hicks 2000), Waikato (Thompson and Hicks 2009), Horizons (Manawatu-Wanganui) (Crippen 1999), Wellington (Crippen and Hicks 2011), Gisborne (Crippen and Scholes 2001), and Tasman (Burton 2002). Some regions have completed repeat surveys that establish temporal trends. In the Waikato Thompson and Hicks (2009) found that the area of bare ground exposed by all forms of disturbance increased significantly between 2002 and 2007, doubling from 1.4% to 2.8% of the region's area. The major changes occurred to cultivated areas and tracks. On rural land in the Auckland Region between 1999 and 2007 (Hicks and Thompson 2009) the amount of erosion-prone surfaces decreased (from 37.6% to 33.8%) and eroded surfaces increased (from 9.1% to 13.2%). In the Wellington Region between 2002 and 2010 the area disturbed by land use activities increased from 11% to 14%, mainly from cultivation and tracking (Crippen and Hicks 2011). In time this type of data will provide a far better picture of trends in erosion, and in efforts to control erosion, than is presently available.

Much of the worst erosion in New Zealand is in the Gisborne – East Coast region and attempts have been underway since at least the 1960s to reduce erosion in this area, primarily by afforestation (see Taylor 1970; Bayfield and Meister 1998, 2005). In 1992 the East Coast Forestry Project (ECFP) was established by the Ministry of Agriculture and Forestry (MAF) to try and reduce the erosion problem by subsidising targeted afforestation on the most erosion-prone land. By 2011, 35 522 hectares of target land (out of a total of 60 000 hectares identified as requiring erosion control) had been treated (MAF 2011). Additional areas of erosion-prone land have also been targeted under Gisborne District Council's sustainable hill country project. Both programmes, along with earlier afforestation, have made significant progress in afforesting erosion-prone land and reducing erosion. Marden et al. (2005) illustrate the changes in area of gullies in the Waipaoa catchment between 1939 and 1988 (Figure 3) with a large reduction in gully erosion associated with afforestation in the 1960s. Sediment production from gullies during the pre-afforestation period (1939–1960) was ~27 000 t km^{-2} yr^{-1}, increasing to ~30 000 t km^{-2} yr^{-1} during the 1960–1970 period, before decreasing to ~11 000 t km^{-2} yr^{-1} during the 1970–1988 period, by which time most of the reforested area had reached

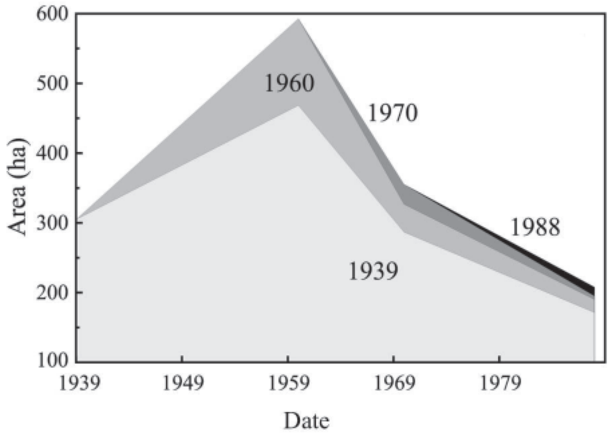

FIGURE 3 Change in total area of gully erosion in the Waipaoa catchment for periods pre-reforestation (1939, 1960) and the reforestation period (1970, 1988) (after Marden et al. 2005). The different grey shading indicates when gullies first appeared.

maturity. Herzig et al. (2011) model the impact of past afforestation and predict the effect of current erosion-control programmes on trends in gully erosion in three catchments in the Gisborne – East Coast region. They suggest sediment yield from gullies is currently 22% less than if there had been no afforestation.

A similar programme of targeted erosion control has been initiated in the Manawatu-Wanganui Region following a severe storm in February 2004 (Hancox and Wright 2005; Dymond et al. 2006). This programme targets 450 000 hectares of highly erodible land, with farm plans now having been completed on 280 000 hectares of this land (see http://www.mpi.govt.nz/environment-natural-resources/funding-programmes/slm-hill-country-erosion-programme/slmhce-project-sustainable-land-use-initiative). When fully implemented this will in time have a significant effect on erosion trends within the Manawatu-Wanganui Region.

LAND COVER AND EROSION CONTROL

Although a wide range of methods are used for erosion control in New Zealand (Table 2), biological methods are by far the most widely used. A large range of vegetation types and species have been used to control erosion throughout New Zealand. These include herbaceous, shrub and tree species, mainly of exotic species with more limited use of indigenous species. There are numerous publications on the use of plants in erosion control programmes, their establishment and management, and their effectiveness in reducing the occurrence and severity of erosion (Lambrechtsen 1986a, b; Pollock 1986; van Kraayenoord and Hathaway 1986a, b; Hawley and Dymond 1988; Phillips et al. 1990, 2008, 2011; Hicks 1991a, b, 1995; Marden and Rowan 1993; Quilter et al. 1993; Thompson and Luckman 1993; Bergin et al. 1995; Douglas et al. 1998, 2009, 2011; Anthony 2001;

TABLE 2 Erosion control techniques used for different types of erosion in New Zealand (after Hicks and Anthony 2001)

Erosion type	Soil conservation principle	Erosion control practices
Sheet and rill	Maintain ground cover Maintain soil structure and health	Water control Improving drainage Conservation tillage (contour cultivation, minimum tillage, direct drilling, herbicides) Wheel track ripping Stubble mulching Rotational cropping Strip cropping Use of low-ground-pressure machinery Cover crops Timing cultivation to avoid risk Adjusting grazing pressure to avoid risk Matching crop and pasture species to site conditions
Shallow mass movement (landslides, debris avalanche, earthflow)	Maintain root strength contribution to slope stability Reduce soil water	Space-planted trees Reversion to scrub Afforestation Adjusting grazing pressure and fencing Drainage control
Deep-seated mass movement (landslides, slumps, earth and rock flow)	Maintain root strength/contribution to slope stability Reduce soil water	Space-planted trees Reversion to scrub Afforestation Adjusting grazing pressure and fencing Drainage control Debris dams
Gully	Control runoff Avoid exposure of bare ground in overland flow paths Reduce peak flood flows Stabilise margins	Water control (diversions, flumes, pipes, drop structures) Space-planted trees Reversion to scrub Afforestation Debris dams Ground recontouring
Tunnel gully	Control runoff Manage ground cover	Water control Manage ground cover in overland flow paths Space-planted trees Ground recontouring
Wind	Maintain ground cover Maintain soil structure and health to reduce erodibility Maintain surface soil moisture	Maintain ground cover Maintain soil structure and health to reduce erodibility Maintain surface soil moisture
Streambank	Maintain riparian vegetation Reduce bank undercutting and lateral migration	Tree planting of banks and riparian buffers Structural control (rock riprap, gabions, groynes, geotextiles) River diversion Bank regrading Reseeding stream banks Control stock access by fencing Subsurface drainage at seepage sites

Hicks and Anthony 2001; Hicks and Crippen 2004; Marden 2004; Phillips and Marden 2005; McIvor et al. 2011; Basher et al. 2008; Davis et al. 2009). Space-planted poplars and willows have been the most widely used soil conservation plants in New Zealand, since they can be established as poles in the presence of grazing animals, and are appropriate for the control of landslide, earthflow, gully and streambank erosion.

Surface erosion (sheet, rill, wind) can be prevented or reduced through establishing and maintaining a persistent, healthy, complete ground cover. The effectiveness of the cover depends on both above- and below-ground plant components (Hicks 1995; Hicks and Anthony 2001). Herbaceous species used for erosion control are often recognised as important forages for livestock, and in a number of erosion-prone farmland situations there must be a balance between providing ground protection for fragile soils and adequate quantity and quality of forage.

Aspects of the effect of vegetation on erosion have been reviewed by several authors (e.g. O'Loughlin 1995, 2005; Glade 2003; Marden 2004, 2012; Blaschke et al. 2008; Phillips et al. 2012), including the performance of biological erosion control methods (e.g. Thompson and Luckman 1993; Douglas et al. 2011; McIvor et al. 2011). These include process-based studies documenting the mechanisms underlying the impact of trees on slope stability as well as data comparing erosion rates under different vegetation communities.

Most of the data available on the impact of vegetation cover on erosion is derived from the study of landslides during large storm events such as Cyclone Bola and the February 2004 Manawatu-Wanganui storm, with far less multiple-event and time-averaged data available and very limited data for other erosion processes. During these large storms woody vegetation has a profound impact in reducing landsliding, with results suggesting that the presence of tall, closed-canopy, woody vegetation typically leads to a 70–90% reduction in the amount of landsliding (e.g. Phillips et al. 1990; Marden et al. 1991; Marden and Rowan 1993; Bergin et al. 1995; Fransen and Brownlie 1995; Reid and Page 2002; Hancox and Wright 2005; Dymond et al. 2006). These results are generalised in the NZeem® model (Dymond et al. 2010) as a long-term order-of-magnitude reduction in erosion where tall woody vegetation is present. However, this factor is likely to be spatially variable depending on landscape characteristics such as rock type, slope steepness and rainfall.

The relationship between probability of landsliding and slope angle shown in Dymond et al. (2006) shows clearly that slope had

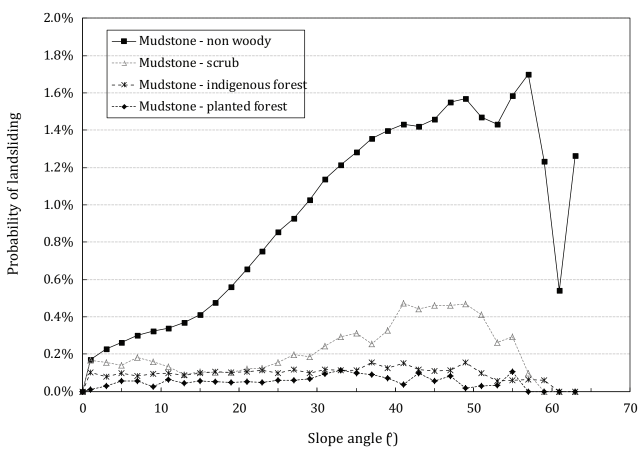

FIGURE 4 Effect of variation in slope angle and vegetation cover on probability of landsliding in the February 2004 storm (after Dymond et al. 2006)

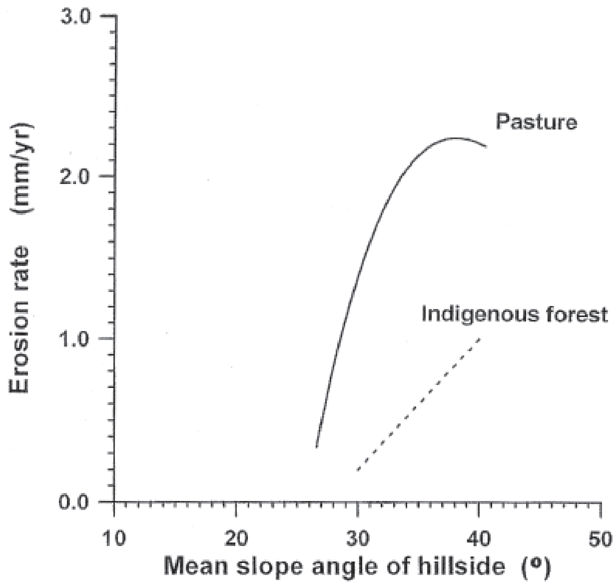

FIGURE 5 Effect of variation in slope angle and vegetation cover on erosion rate (from DeRose (1996), courtesy of *Zeitschrift für Geomorphologie*).

a significant effect on the magnitude of reduction in landsliding in the February 2004 storm (Figure 4). Similarly in the Taranaki hill country (DeRose 1996), the difference between erosion rates under pasture and forest increases with slope angle (Figure 5). The impact of tall woody vegetation in reducing landsliding is likely to be less in smaller storms. Reid and Page (2002) found that there was a 25 times increase in areal landslide density under pasture (compared with tall woody vegetation) for a 600-mm rainfall but only a 5 times increase for a 260-mm rainfall. Similarly, Barton et al. (1988) found landslide density and area increased with storm rainfall. In a number of studies comparing landslide densities under pasture and tall woody vegetation before and after Cyclone Bola, the differences in landslide density were always smaller before Bola (Phillips et al. 1990; Marden and Rowan 1993). A number of studies describe considerable spatial variation in the effect of vegetation cover on landsliding or sediment generation. In four areas of the Manawatu-Wanganui hill country in a large storm in February 2004, landslide area under pasture ranged from 3 to 11 times higher than under forest (Hancox and Wright 2005). The same authors also note the lack of landsliding in the greywacke of the Tararua and Ruahine ranges in this storm despite very high rainfalls. Hicks and Crippen (2004) also reported considerable spatial variation in the effect of vegetation on landsliding in this storm. DL Hicks (1990) comments that there was generally less landslide damage in Taranaki during Cyclone Hilda than in the Gisborne – East Coast area during Cyclone Bola, as a result of differences in underlying rock types. Reid and Page (2002) compiled sediment generation rates for six different land systems in the Waipaoa catchment during Cyclone Bola and found they ranged from 50% to 90% less under forest than pasture depending on the land system. It is likely that the magnitude of landslide reduction would be greatest on the most erodible terrain.

Despite the widespread use of space-planted trees for erosion control in New Zealand there has been surprisingly little experimental or quantitative work to establish the effectiveness of space-planted trees in reducing erosion. The published studies emphasise the importance of both initial establishment of the trees and subsequent maintenance to ensure their effectiveness. Most of the empirical data on performance are based on individual or small groups of trees rather than hillslope-scale performance. Hawley and Dymond (1988) back-calculated what the reduction in landslide damage would have been (70%) with 10-m tree spacing and 100% establishment and survival, although the actual performance was considerably lower (14% reduction in landsliding, tree spacing of 20 m and 66% survival). Smaller reductions in landsliding were documented by Varvaliu (1997) and Hicks et al. (1993) in storms in 1992 in the Manawatu-Wanganui hill country. Using a similar approach to Hawley and Dymond (1988), small groups of space-planted trees were found by Douglas et al. (2009, 2011) to locally reduce landsliding by 95%. The authors do not comment on the performance of these small groups of trees in a broader whole-hillslope context. However, a number of studies have shown that space-planted trees have performed poorly due to inadequacy of establishment and maintenance of plantings (e.g. Hicks 1989, 1992; Cameron 1991; Thompson and Luckman 1993; Hicks et al. 1993).

The data on closed-canopy and space-planted tree have been used to model the effect of vegetation change on erosion (e.g. Dymond et al. 2010; McIvor et al. 2011) by assuming closed-canopy trees reduce erosion by 90% over 20 years, space-planted trees reduce erosion by 70% over 15 years, and scrub or native forest reversion reduces erosion by between 10% (early-stage incomplete canopy closure) and 90% (complete canopy closure). Using this approach Dymond et al. (2010) calculated that by targeting the 500 farms with the highest priority for soil conservation the total sediment load of the Manawatu River could be reduced by ~50% by the time the trees matured.

There are far less data on the influence of vegetation on suspended sediment yield, especially for space-planted trees, and there is clear evidence of scale effects. At small-catchment and storm-event scales comparisons of sediment yield under different vegetation cover, and studies of the impact of deforestation, show that forested catchments yield significantly less sediment than pasture catchments (e.g. Dons 1987; DM Hicks 1990; Fahey and Marden 2000; Fahey et al. 2003). Forested catchments can have a mean annual sediment yield up to 95% less than pasture catchments (DM Hicks 1990). In much of the published data forested catchments yield 50–80% less sediment than pasture catchments, whether it is pine or indigenous forest. There appears to be regional variation in the magnitude of this difference that may be due to catchment characteristics or different record periods. In some comparative studies other factors override the vegetation difference, and pasture catchments have similar or lower sediment yield than forested catchments (e.g. Dons 1987; DM Hicks 1990 (East Otago catchments); McKergow et al. 2010). Most of the data showing pasture catchments have a higher sediment yield than forested catchments come from very small catchments (<1–10 km^2). There are no published studies where the effect of space-planted trees on sediment yield has been measured at this scale.

At large-catchment to national scale, vegetation appears to be a secondary influence, with rainfall, geology and topography having more influence on sediment yield (e.g. Hicks et al. 1996, 2011; Elliott et al. 2008). Regional analyses from the Auckland area (Hicks et al. 2009) suggested yields from pasture catchments were ~30% higher than those from forested catchments while at national scale Elliott et al. (2008) found trees or scrub produced on average 80% less sediment than pasture (all other catchment characteristics being similar).

There is a very limited amount of data, all from a single set of studies in the Gisborne area, on the influence of vegetation on earthflow movement. O'Loughlin and Zhang (1986) describe early work on the mechanisms by which trees influence earthflow movement rates and compare wet-winter movement rates under pasture (1.5–2 m $month^{-1}$) and pine trees (0.05 m $month^{-1}$). Using similar data Pearce et al. (1987) summarise 4 years of data collection and suggest movement rates are an order of magnitude lower under pine trees (0.05 m $month^{-1}$ in winter and annual movement of 0.2–0.5 m) than pasture (0.5 m $month^{-1}$ in winter and annual movement of 3–5 m). With a longer period of record (up to 6 years) the differences between grassed earthflows (~1 m $month^{-1}$) and forested earthflows (0.005–0.001 m $month^{-1}$) were far larger (Phillips et al. 1990; Marden et al. 1992; Zhang et al. 1993). Thompson and Luckman (1993) also comment on the performance of biological erosion control on earthflows, suggesting treatment was 'successful' at 63% of sites when trees were closely (<5–8 m) and extensively (>60% of earthflow surface) planted.

There is also limited information on the influence of vegetation on gully erosion. In the Gisborne – East Coast region gullies are characteristic of both forested and grassed landscapes (Parkner et al. 2006, 2007); however, gullies under forest have a higher topographic threshold (a combination of slope and area) than pasture. Gully erosion in this area is closely associated with deforestation, and reforestation has been extensively used to control gully erosion (Marden et al. 2005, 2012). The ability to stabilise gullies with trees is highly dependent on gully size and shape, with an 80% chance of success for gullies of less than one hectare and little success once gullies exceed 10 hectares (Marden et al. 2005). Herzig et al. (2011) model the effect of reforesting gullies on sediment yield in the Gisborne region suggesting past afforestation has reduced sediment yield in the Waipaoa catchment by 33% and the Waiapu by 16%, and that targeted future afforestation could reduce sediment yield by 50%. Even less data are available on the influence of space-planted trees on gully erosion. Thompson and Luckman (1993) found that treatment of gully erosion was successful at only 42% of sites and it required very closely spaced trees to be highly effective. Where gullies were >5 m deep, space-planting was ineffective.

Ground cover is known to be highly effective in reducing rates of sheet and rill erosion although there is little New Zealand data. In a plot study at Pukekohe, Basher et al. (1997) found the short-term rate of soil loss from grass plots (38 t km^{-2} yr^{-1}) was two orders of magnitude less than that from bare soil (4400 t km^{-2} yr^{-1}). Very high rates of erosion have been measured under intensive cropping, where there is a high proportion of bare ground for long periods of time, at both Pukekohe (Basher and Ross 2002) and Ohakune (Basher et al. 2004). Studies of sediment yield in the Auckland area (Hicks 1994) showed that yield from an urbanising catchment (with a high proportion of bare ground eroding by sheet and rill erosion) was more than an order of magnitude higher than any other land use (Table 3). This study also showed that the sediment yield from a market gardening catchment was no different to a pasture catchment because, despite the high within-field erosion rates, much of the sediment was deposited locally and not transported downstream.

Similarly rates of wind erosion are strongly influenced by ground cover and by shelter. In the Mackenzie Basin Basher and Webb (1997) found that bare ground had lost ~4 cm of topsoil over a 40-year period, compared with no soil loss on vegetated sites. In the same area wind erosion under irrigated pasture was ~60% of that under dryland pasture (McDowell and Walker 2010). High rates of wind erosion in single storm events have been documented from several bare cultivated sites in Canterbury (Painter

1978; Hunter and Lynn 1988; McGuigan 1989; Basher 1990). Planting windbreaks for field shelter has historically been widely used on both cropland and pastoral farmland in New Zealand to reduce the wind erosion hazard (Sturrock 1984).

TABLE 3. Sediment yields from catchments in Auckland with different land uses (from Hicks 1994)

Site	Land use	Average annual sediment yield ($t\ km^{-2}\ yr^{-1}$)
Alexandra	Urbanising	2370
Wairau	Mature urban	100
Pakuranga	Mature urban	24
Manukau	Pasture	46
Whangapouri	Market gardening	52

EROSION CONTROL AND ECOSYSTEM SERVICES

Few studies have assessed the value of erosion control in an ecosystem services context, and most of the available research focuses on the value of afforestation with little information on other erosion control technologies.

Barry et al. (2011) outline a method to use scenarios for possible future afforestation of erosion-prone land (Watt et al. 2010), along with an erosion model (NZeem®), to predict the reduction in erosion from conversion of grassland to forest and value the economic benefits of avoided soil erosion. Included in their analysis are:

- Private costs: establishment and harvesting of forest, opportunity cost of land use change
- Private benefits: avoided farm infrastructure damage and private property damage
- Public costs: construction to reduce soil erosion damage, policy mechanism costs
- Public benefits: avoided public infrastructure and flood damage, avoided damage to consumptive water quality, avoided damage to soil regulating facilities

They suggest the separation into public and private benefits and costs avoids double-counting and would also help identify appropriate policy instruments to avoid soil erosion damage using the private and public net benefit framework. The analysis methodology was applied to marginal lands in the Gisborne area to assess the value of different policy options (Barry et al. 2012). This suggested in some cases forestry was not viable and thus there would be no public benefit from avoided erosion and that afforestation of these would require positive incentives or improvements in forest and farm systems and technologies. They suggest the former would be very expensive and the best policy mechanism is technology improvement. The authors do acknowledge that incorporation of other ecosystem services resulting from afforestation may change the policy options.

Little work has been done on the value of erosion control on arable farm land apart from Cullen et al. (2004) suggesting frequent cultivation of arable soils may diminish the level of this ecosystem service.

The first comprehensive national-scale attempt to characterise and map ecosystem services in New Zealand as a basis for exploring the impact of future land use change scenarios on ecosystem services is described by Rutledge et al. (2010). The aim of this work is to develop a multiple land use change model that can more accurately model the full range of ecosystem services spatially and temporally. Preliminary work by Ausseil and Dymond (2010) assesses the effect of land use change on erosion-prone land in the Manawatu catchment on five ecosystem services (regulation of climate, protection of soil, maintenance of clean water, water-flow regulation, provision of natural habitat). Sediment yield was used as an indicator of soil protection and two afforestation scenarios (conversion to planted forest, reversion to indigenous shrubland) were assessed using several models to predict the effects of land use change on the ecosystem service indicators. The ecosystem services were valued in dollar terms to allow summation of net benefits in economic terms. In both afforestation scenarios, the main environmental benefit was a large (50%) sediment yield reduction from the catchment. Rutledge et al. (2010) and Dymond et al. (2012) apply a similar approach nationally to investigate the trade-offs between regulation of soil erosion (change in erosion rate), provision of fresh water (water yield) and climate regulation (carbon storage) associated with afforestation. New *Pinus radiata* forests (once mature) have significant benefits for erosion control (reducing erosion by 10 times) and carbon storage (storing 8.5 t C $ha^{-1}\ yr^{-1}$), but the reduction in water yield neutralises these benefits in water-short catchments.

Blaschke et al. (2008) examine the impact of afforestation on water yield and erosion to demonstrate the potential effects of mitigating climate change via afforestation. While the primary benefit of afforestation is in reducing erosion and sediment yield (by at least 50% in small catchments and by a smaller amount in large catchments) there are additional benefits for other ecosystem services including improved water quality, water regulation, improved aquatic habitat, greenhouse gas reduction, biodiversity protection, soil and nutrient retention (Blaschke et al. 2008). The benefits of afforestation for aquatic habitat and freshwater biodiversity protection have been extensively studied (e.g. Death and Death 2006; Parkyn et al. 2006). Wilcock et al. (2008) summarise these benefits as reduced input of nutrients and contaminants, improved habitat and food supply by addition of wood and leaf litter, and a reduction in water temperature from shade provided by trees.

There is a positive effect on water regulation by reducing flood flows (at least in smaller catchments with a large proportion afforested). However, there can be a negative effect on water regulation by reducing low flows. The reduction in flood peaks depends on the proportion of the catchment afforested and the size of the flood (compared to pasture a reduction of 30–90% during small (up to annual) floods, 20–50% in large floods, and negligible in extreme floods) – see Rowe et al. (2003). Afforestation reduces flood peaks but not flood volumes as floodwaters are delivered over longer time periods. Measured reductions in low flow range from 0 to 50% (Rowe et al. 2003). Large changes in flood peaks, water yields or low flows have only been observed in small catchments where most of the catchment has been planted, with the few published studies of partial afforestation of large catchments showing much smaller changes in flow.

Erosion control also has a positive benefit for climate regulation through storing carbon and nitrogen both in the plants and soil, and for maintaining soil fertility. The impact of erosion on soils in the soft rock hill country has been characterised in a number of studies (Lambert et al. 1984; Douglas et al. 1986; Smale et al. 1997; Sparling et al. 2003; Basher et al. 2011; Rosser and Ross 2011; DeRose 2012). All show that shallow landslide erosion causes a reduction in soil depth (and water holding capacity), and a loss of carbon, nitrogen and nutrients. In loess-mantled hill country in the Wairarapa topsoil depths on landslide scars are about one-third those in uneroded soils (Rosser and Ross 2011) and soil depth to bedrock is about 9.5% less (DeRose 2012). By reducing rates of landsliding, erosion control contributes to the maintenance of soil carbon and nitrogen, soil fertility and water holding capacity. It is worth noting that most of the studies of the impact of landslide erosion have only characterised the landslide scars and have ignored the debris tails associated with the landslide scars. Basher et al. (2011) mapped both scars and tails and showed that the debris tails occupied 50–100% more area than the landslide scars. Much of the soil carbon removed from scars was redeposited in the debris tails rather than being completely lost. This redistribution has not been incorporated into analyses of the net effect of erosion on soil depth and organic matter.

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