Economic assessment of options to mitigate sediment loss from New Zealand agriculture – in the context of managing freshwater quality

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Executive Summary

The purpose of this literature review is to provide an accessible summary of relevant research carried out in New Zealand on methods to manage sediment and their economic assessment. It also includes a basic explanation of the processes that lead to erosion and sedimentation. This review is aimed primarily at regional councils and other parties involved in freshwater management in New Zealand. Sediment is a major freshwater contaminant throughout New Zealand and it is expected that sediment-related attributes will shortly be included within the *National Policy Statement for Freshwater Management 2014*. It is anticipated this would make the setting in regional plans of objectives, limits and methods for sediment related attributes compulsory.

The description of the physical processes that drive erosion and sedimentation is an introductory summary aimed at readers who may not have been exposed to sediment science before. Readers who would like (or need) to dive deeper may wish to refer to the references section.

Erosion and sedimentation are natural processes driven by climate and geology. However, these processes have been accelerated by human activities. Human settlement, forest burning and deforestation, mainly to make way for the development of pastoral farming, have resulted in increased levels of hillslope and surface erosion. Land use intensification, draining wetlands and inappropriate land management practices have further increased these forms of erosion, as well as streambank erosion. The suspended and deposited sediment generated results in a range of adverse economic, cultural and environmental effects on land and floodplains (e.g. reduced fertility, increased impacts from flooding, infrastructure damage, diminished aesthetic values), fresh water (e.g. algal growth from increased phosphorus, turbidity, smothering of beds), estuarine and coastal environments. All these impacts are further exacerbated by the loss of natural floodplains due to river straightening and the significant reduction in wetlands.

Sediment mitigation practices can be broadly divided into in-field mitigations and edge-of-field mitigations. The first ones aim to bind soil, prevent its mobilisation and reduce rainfall impact. These include:

- increased pasture density,
- spaced tree planting on pasture, and
- conversion to forestry or mānuka.

The second ones aim to reduce run-off velocity and trap suspended sediment. These include:

- riparian fencing and planting,
- sediment retention infrastructure (traps and bunds), and
- wetlands.

The practices reviewed here are those that could be considered at a catchment level. Identification of farm level mitigation strategies would of course involve a detailed on site assessment of risk and critical source areas which is beyond the scope of this review.

A particular challenge when managing diffuse freshwater pollution, including sediment, is that mitigation costs are often private while benefits are wider. Likewise, mitigation costs can generally be determined by market values while benefits often involve non-market values, which are more difficult to assess. A significant reason for this is the importance and variability of catchment contexts across New Zealand, which generally prevents the application of benefit transfer and 'one-size-fits-all' solutions. Nonetheless, good practice economic analysis, legislation and direction from the Courts1 tell us that all relevant costs and benefits should be taken into account when considering mitigation strategies and requirements.

Co-benefits of sediment mitigation that should be considered concurrently will generally include mitigation of nitrogen, phosphorus, *E. coli* and greenhouse gas emissions, maintaining soil health and fertility, amongst various other ecosystem services on site and off site. This is likely to involve the integration environmental or bio-physical assessments and economic assessments.

Recent economic assessments reviewed and discussed in this document include:

- static optimisation (e.g. cost effectiveness analysis),
- input, output models, and
- benefit-cost analysis.

The social and political context in which mitigation must occur should also not be overlooked. The adoption of credible solutions requires the input and participation of landowners, tangata whenua, communities and stakeholders. Likewise, the distribution of costs and benefits amongst these groups, and their perception of fairness and equity are also critical for durable solutions.

[1] *For example, Federated Farmers versus MacKenzie District Council, 2017*

Part 1

Introduction

The purpose of this literature review is to provide an accessible summary of relevant research carried out in New Zealand on methods to manage sediment and their economic assessment. It also includes a basic explanation of the processes that lead to erosion and sedimentation for those that may be relatively new to them.

This review is aimed primarily at Regional Council planning and land management staff, and other parties involved in freshwater management in New Zealand. Sediment is a major freshwater contaminant throughout New Zealand and it is expected that sediment related attributes will shortly be included within the *National Policy Statement for Freshwater Management 2014* (NPS-FM) (Parker, 2018). It is anticipated this would make the setting of objectives, limits and methods for sediment-related attributes compulsory by regional councils. Although, until now, the management of sediment has not been compulsory under the NPS-FM, regional councils (and their predecessor organisations) have been managing sediment for decades and several are considering sediment in current freshwater planning processes (e.g. Carter et al., 2017).

Agriculture supports employment, food security, and financial livelihoods (FAO, 2016). However, it has a significant impact on the natural environment. Deforestation and agriculture are together responsible for a quarter of global greenhouse gas emissions (Edenhofer et al., 2014), with livestock responsible for around 15% (Rojas-Downing et al., 2017). Further, the implications for soil and water resources is marked, with environmental degradation arising from elevated inputs of nutrients (nitrogen and phosphorus), sediment, and faecal microbes to fresh water (McDowell et al., 2008; FAO, 2017). Public policy introduces leverage to improve social outcomes. However, this can impair production and/or profit in farming systems that have evolved under different regimes. An example is that despite there being opportunities to improve profit through increasing efficiency (Ma et al., 2018), requirements for significant reductions in nitrogen loss from New Zealand dairy farms can impose significant cost (Doole, 2012; Doole and Kingwell, 2015).

The indigenous forest and wetland landscapes of New Zealand have been extensively modified over the last 700 years, largely replaced with exotic grasses (e.g. *Trifolium* spp.) (Brooking and Pawson, 2011). Extensive burning by Māori reduced the extent of native vegetation cover from 85% to 57% of New Zealand's land area, while further clearance since 1840 has decreased it further to 25% (McGlone, 1989). Intensive agriculture now covers around 14 million hectares, with around 85% used for grazing livestock. Agriculture is important to the New Zealand economy, especially at the regional level, with this sector responsible for nearly half of merchandise exports by value (StatsNZ, 2017). However, extensive deforestation has encouraged significant sediment loss. Mass movement processes are chiefly responsible, mainly through gully erosion and landslides (Soons and Selby, 1992; Glade, 1998).

New Zealand produces around 2% of suspended sediment in the world's fresh water, yet only makes up 0.2% of global land area (Hicks et al., 2011). It is estimated that 192 million tonnes of soil are lost in New Zealand every year, 44% of this from areas in pasture (MfE and StatsNZ, 2018). Many New Zealand soils are inherently susceptible to erosion with high rainfall, soft lithology, and steep topography (Soons and Selby, 1992; Hicks et al., 2011; Heaphy et al., 2014). Nonetheless, the impact of management is also significant, with forest clearance and its substitution with exotic grasslands increasing sediment loss by around five to ten times (Hawley and Dymond, 1988; Page and Trustrum, 1997; Hancox and Wright, 2005). However, agriculture is not solely responsible for high levels of sediment input to fresh water across New Zealand. Urban development, mineral extraction, and earthworks also contribute to the problem (Moores et al., 2016).

Public policy has targeted sediment loss for many years in New Zealand. Adverse climatic events, large rabbit populations, and increased farming intensity during the Great Depression (Brooking and Pawson, 2011) promoted erosion rates in the 1930s, motivating the establishment of the *Soil Conservation and Rivers Control Act 1941*. This was the first national legislation focused on an environmental issue, recognising the significance of sediment loss and its off site impacts in New Zealand. It also initiated a long history of local environmental bodies providing targeted soil conservation advice, particularly once reinforced through the *Water and Soil Conservation Act 1967*. This legislation also created institutions like catchment boards that, along with Central Government and private interests, undertook extensive reforestation between the 1950s and 1980s. These efforts concentrated on the mid and upper reaches of many catchments and were complemented by removal of stock, pest control, and soil conservation. A lot of effort was also placed into constructing stop banks, straightening and deepening river channels, and planting river banks to reduce the impact of floods, exacerbated by erosion. In the 1980s and 1990s, afforestation was mainly done by private landowners in the form of farm woodlots and some areas of pasture were reverted into scrub. However, despite these efforts, as late as the early 1980s, there were still Government incentives available to clear erosion-prone land. These mainly involved the provision of economic subsidies that encouraged increases in livestock numbers, production, and the development of marginal land (MacLeod and Moller, 2006). Most recently, there have been conversions of large areas from forestry into dairy in response to price signals (Blaschke et al., 2008). This has certainly happened in the Upper Waikato region, where a lot of the conversion was driven by the treatment of forestry in the Emissions Trading Scheme (ETS), and the accessibility of cheap emissions units from overseas. These combined effects of market shifts and Government policies on land use have not helped to reduce soil erosion.

The modern legal framework for soil conservation is defined through the *Resource Management Act 1991* (RMA) that devolves environmental management to regional councils and unitary authorities. Under the RMA, soils are managed alongside other natural resources in an integrated fashion through district plans, regional plans, and resource consents. The NPS-FM provides a national direction for regional decisions around fresh water. It has large implications for land use and soil conservation (McDowell et al., 2013), but even more so, once individual catchment limits for suspended sediment are introduced (MfE, 2013). A wealth of education, funding, and partnership programmes also exist to help complement the regulatory framework (MPI, 2015). Public policy has stimulated the broad-scale implementation of mitigation activities for sediment loss (Basher, 2013). However, it has also driven land use change, with the removal of agricultural subsidies in the mid-1980s motivating the afforestation and regeneration of marginal hill country (MacLeod and Moller, 2006). Together, these activities drove a 20% decline in sediment loss in New Zealand, from 95 to 77 million tonnes per year, from 1975 to 2005 (Dymond et al., 2010). This contributed to widespread improvement in visual clarity and phosphorus levels in New Zealand fresh water across 1989-2013 (Larned et al., 2016). Nonetheless, suspended sediment levels are still well above those that existed prior to extensive deforestation and fine sediment deposition is increasing (MfE and StatsNZ, 2017). This reinforces the need for sustained efforts to increase the uptake of mitigation actions for sediment loss across New Zealand landscapes.

The goal of this research is to review the current state of sediment and its management in New Zealand. The survey is broad, focusing on sediment science, the mitigation of soil loss and its effect on fresh water, and the economic assessment of management options to address sediment loss. The focus of the review is from an economic perspective, given the: broad-scale nature of the problem; limited assessment of sediment issues compared with nitrogen and phosphorus; national focus towards implementing the NPS-FM; ongoing development of novel techniques for sediment loss mitigation; scarcity of attention paid to this issue by social scientists; and the wide-scale focus on soil conservation by environmental authorities across New Zealand.

Part 2

Sediment science: the basics

Streambank erosion on the Waiari Stream

An introductory glossary

Sediment includes particles of soil and rock eroded from land and washed down or blown by the wind into rivers and lakes. Generally, mineral and some inorganic material, it can also include plant fragments. It can range in size from fine particles to boulders.

The smallest particles of silt, clay, fine sand or organic matter float in water as **suspended sediment.** Suspended sediment is measured in mass of sediment in suspension within a given volume of water (e.g. g/L).

Turbidity describes the haziness of water due to the presence of many small particles that are not visible to the naked eye.

Euphotic depth is the depth at which 1% of photosynthetically active radiation is present. A higher level denotes greater clarity.

Coarse sediment and fine sediment in calm waters, such as slow flowing sections of rivers, settle on the beds of water bodies as **deposited sediment** or **bed load**.

Sedimentation is the process of transportation, settling, or depositing of eroded sediment within waterbodies.

Hillslope erosion includes mass movement erosion influenced by gravity, earthquakes and water, such as landslips and earthflow. Mass movement erosion does not generally commence until storm rainfall and soil moisture conditions exceed certain thresholds. Most hill slopes steeper than 15 degrees are susceptible to mass movement, and those steeper than 28 degrees generally have severe potential. Storms are the primary triggers (Cairns et al., 2001).

Surface erosion includes sheet, rill and gully erosion and is also driven by rainfall and occurs when the soil's infiltration capacity is exceeded. Sheet erosion can be almost imperceptible and involves the removal of a few millimetres of top soil. Rills are long narrow miniature channels where 10-50 cms of top soil is removed. Repeated erosion of rills can lead to gully erosion when deep channels are scoured out by concentrated runoffs. This type of erosion can occur on any type of land (Cairns et al., 2001).

Streambank erosion is caused by waterway flows gouging waterway banks, as rivers migrate across the flood plain; this is exacerbated by livestock grazing on the banks of waterbodies. It delivers sediment straight into waterways.

Wind erosion involves the detachment and transportation of soil particles by wind when the airstream passing over a surface generates sufficient lift and drag to overcome the forces of gravity, friction and cohesion. Wind erosion may also carry material into waterways, although generally, this is less of an issue in most parts of New Zealand (Cairns et al., 2001).

Sediment load is the sum of all terrestrial erosion delivered to waterways, i.e. streambank erosion, plus all land based sources (surface, landslide, earthflow and gully); it is often measured in tonnes per year.

Sediment yield is the sediment load by area, for instance tonnes per hectare per year. It varies mainly by rainfall and geology and less by land cover or land use, but as described below, the latter are key drivers.

The deposition, and subsequent accumulation, of sediment is known as **aggradation**.

A natural process exacerbated by human activity

Erosion and sedimentation are natural geological processes. Natural rates of erosion are relatively high in New Zealand, as it is a geologically young country with steep topography, active tectonics and a fair amount of inclement weather. The processes of erosion and sedimentation slowly re-distribute material across the landscape and ecosystems are naturally adapted to them. Erosion and sedimentation are in fact essential to the formation of flood plains, river and stream channels and lake and estuary deltas and even to prevent coastal erosion (Hicks et al., 2004). Sediment is a natural feature of rivers.

Waves, wind, and floods can stir up deposited sediment, turning it again into suspended sediment (PCE, 2012). Similarly, benthic feeders, such as koi carp, can increase suspended sediment loads as they feed on material in the deposited sediment. High flows can flush accumulated fine sediment. However, this would not occur during low flows, like generally during the summer, or in spring-fed creeks with limited flow variability.

Deposited sediment is measured in terms of the percentage change in silt and clay accumulation, the elevation and shape of beds and degree of embeddedness. Deposited sediment in estuaries and harbours is often measured in terms of the annual average sedimentation rate or millimetres of aggradation or accretion per year, which is indicative of various adverse ecological effects. Sediment related freshwater attributes can be converted into estuary attributes (White et al., 2016; Green and Daigneault, 2018).

Modelling estimates that sediment would cover 8% of riverbeds in natural conditions without any human activity, while as of 2011, it was estimated to cover 29% of riverbeds (MfE and Stats NZ, 2017). A range of problems arise when these processes are accelerated, due to human activities that create sediment losses above those observed under natural conditions (Figure 1). The historical legacy of erosion and sedimentation which started with human settlement of New Zealand has irreversibly changed most catchments and we are still dealing with these challenges today (PCE, 2012).

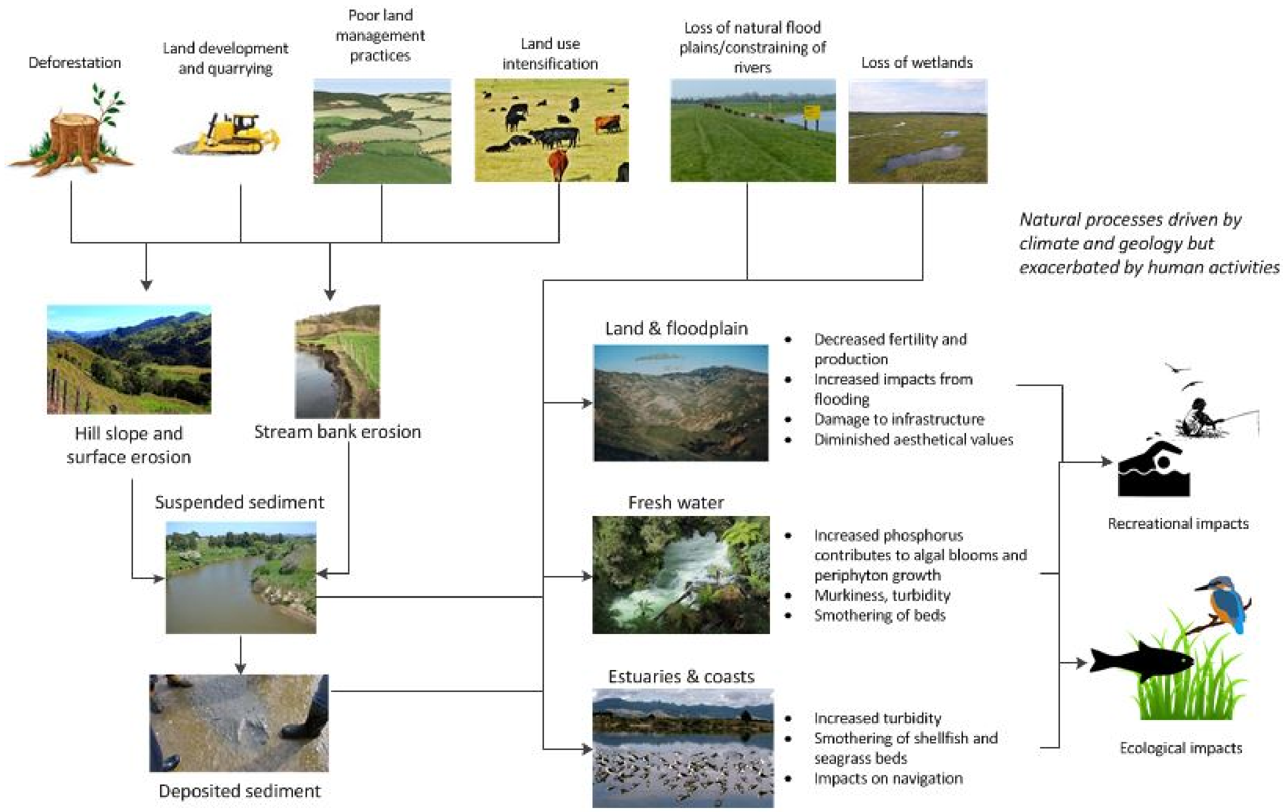

Figure 1 Causes and impacts of erosion and sedimentation.

Causes

Deforestation exacerbates erosion. As forests are cleared, as has occurred since Polynesian settlement of New Zealand, rain erodes fertile soils from hillsides directly down into valleys and eventually into waterways.

Forests store water for gradual distribution and protect soil from direct raindrop impact. Trees have deeper and stronger root systems and return large inputs of organic materials that improve soil structure, promote infiltration, and impede overland flow and sediment runoff, while maintaining a lower water table (Jones et al., 2008). Consequently, water yields, sediment yields and flood peaks from forested areas are much lower than from equivalent areas under different land covers. For example, a 34 year study of upland east Otago found a 33% lower water yield from a catchment in forestry, compared to a catchment in tussock grassland (Fahey and Payne, 2017). Other estimates have ranged from 30-80% (Davie and Fahey, n.d.). As a result, headwater streams in pastoral catchments are narrower than in equivalent forested catchments (Hughes, 2015). In some cases, streams from previously forested areas may disappear altogether. Likewise, sediment coring in the Kaipara Harbour shows that pre-settlement sediment accumulation from forest covered catchments was 1 mm per year while the current rate is 5 mm (D Kervell, personal communication).

Nonetheless, tree cover does not offer total protection; erosion and sedimentation will continue to occur in forested areas, particularly on certain slopes and geologies.

The condition of the canopy and understorey vegetation can also influence the amount of erosion that happens under forested areas. For instance, by reducing canopy cover and understorey vegetation, pests that feed on plants may also indirectly influence how much erosion takes place in forested areas.

Sediment losses from production forests are lower than from pasture for most of the forest rotation. Pasture produces between two and five times more sediment than equivalent areas of forest (Hicks et al., 2004). However, during harvesting and replanting, these losses can increase by up to five times, declining to pre-harvest levels within two and three years after replanting (Quinn and Phillips, 2016). Earthworks and clear-felled areas of forests create large amounts of sediment by surface erosion and, in large weather events, mass movement resulting in logging slash and severe off site effects (as observed in June 2018 in Tolaga Bay).

Open cast mining, earthworks, land development, livestock farming, quarrying, urban development, inappropriate land uses for the terrain, poor cultivation and soil management practices all increase erosion and sedimentation of waterways. Most land developments, land use intensification and land use changes, with the exception of pasture to forestry, will increase land disturbance until a new cover or crop is established and matures. Compacted wheel tracks in horticulture increase runoff. Livestock with direct access to the banks of waterbodies pressure the river edges and shape and push soil directly into the water. Overgrazing exposes soil, and sheep tracks can form channels; both increase runoff.

Techniques now exist for sediment finger printing and sediment source tracking (Quinn and Phillips, 2016). However, these do not form the basis of any management schemes throughout New Zealand, currently due to the difficulty of dealing with legacy loads, expense, and variability. However, these techniques can improve understanding of catchment sediment sources at a point in time, and therefore can be used to monitor change in response to interventions.

However, as noted above, geology and weather are the main determinants. A catchment with exposed steep slopes, silty or sandy soils and subject to sudden downpours of rain will be highly vulnerable to erosion. The Waipa and Whanganui Rivers for example, flow through soft erodible land and rock, making them particularly susceptible to sediment pollution. Likewise, many catchments on the North Island’s East Coast are characterised by high rainfall, erodible rock and tectonic uplift, which combined with widespread removal of native forest, results in high erosion. Inappropriate land use and land use practices result in the risk being realised more than under natural conditions.

Most sediment is transported at flow rates between mean flow and mean annual flood flow, in other words during average rather than extreme events. However, extreme events (i.e. high magnitude low frequency) can result in large amounts of erosion that have long lasting consequences, such as during Cyclone Bola, the 2004 Manawatu-Whanganui floods or the 2017 Bay of Plenty floods. Magnitude frequency characteristics of sediment yields during discrete runoff events can convey information on sediment supply within a basin and also the characteristics of erosion processes (Hicks et al., 2004).

Before modern land development, drainage schemes and flood control works, rivers regularly overflowed across floodplains during storm events. These flood waters are now often directed into engineered channels and confined rivers within stopbanks. These floodplains were a natural destination for the stormwater load of suspended sediment. As the floodplains have been developed for productive agricultural enterprise and the landscape occupied by infrastructure of roads, bridges and settlements, the natural process of widespread flooding and sediment deposition on floodplains is not tolerated by society. Sediment is forced to remain in river systems until a more acceptable destination in lowland areas, estuaries or the sea is reached. A natural compromise for the loss of the floodplain destination for sediment deposition is to artificially recreate still water interception areas within the landscape, by the installation of stormwater detainment structures.

Evolving technology and applied research is currently investigating the viability and opportunity for the wide uptake of ‘detainment bunds’ throughout the agricultural landscape. More on this in the mitigation practices section below.

Impacts

Land and floodplain

On site effects of erosion include reduced productivity and fertility due to the loss of organic matter, nutrients and biota, reduced infiltration rates and water holding capacity, all of which affect soil health. This impact may be masked by the increased use of fertiliser, pesticides, irrigation, and improved cultivars but would otherwise be reflected in reduced production, crop and livestock loss. The cost of this impact was estimated nationally at $46.5 million per annum (in 2008 dollars) (Jones et al., 2008). Cairns et al. (2001) report that surface erosion may result in a reduction of between one and three fifths in crop yield, two and four fifths in pasture growth and 90% of tussock biomass.

Erosion also results in the loss of soil carbon, an indicator of organic matter and fertility. This may also result in increased greenhouse gas emissions from soil. However, these processes are poorly understood (MfE and StatsNZ, 2018).

By altering water flows and reducing waterway capacity, sedimentation exacerbates the impacts of downstream flooding. This includes damage to settlements, farmland and facilities (bridges, roads, rail, utilities), livestock and crop losses. It is expected that the frequency and severity of extreme weather events and associated flooding will increase under a changing climate (MfE, 2016). The effects of sediment on downstream flooding can be limited through the establishment of flood protection infrastructure. However, this option is typically expensive, does not mitigate the amount of sediment in waterways but merely shifts it elsewhere, and also increases the potential for infrastructure itself to interfere with ecosystem function.

Property damage and associated social implications of flooding may occur both on and off site. These can be measured in terms of the cost of labour and materials to clean up and repair infrastructure. Although difficult to quantify, it has been estimated at $48.7 million per annum (in 2008 dollars) (Jones et al., 2008). A less evident but potentially significant impact is undermining farmer and grower confidence.

Erosion and flooding also result in visible impacts on the aesthetics of the landscape.

Fresh water and fresh water environments

Children measuring water clarity

The direct impact of sediment on freshwater environments can be broadly divided into chemical and physical, which in turn result in ecological and biological impacts. Fine sediment is an unusually complex contaminant. Its effects on ecology and recreational use are subject to concentration, grain size, composition and particle shape and duration of exposure (Davies-Colley et al., 2015).

The chemical impacts relate mainly to phosphate, which clings to soil particles in sediment. Sediment from erosion is therefore a major source of phosphorus in fresh water. If farmers apply excessive phosphorus fertiliser and maintain soil Olsen-P test levels above optimal for pasture uptake, phosphorus loss from farmland is exacerbated. Between 2014 and 2017, 33% of soil testing sites across 11 regions of New Zealand had phosphorus levels above targets (MfE and StatsNZ, 2018). Phosphorus fertiliser can also be lost excessively if stormwater runoff events occur soon after fertiliser application and particularly if soluble forms of phosphorus are used. Phosphorus, in combination with nitrogen, can lead to algal blooms and the settlement of conspicuous amounts of periphyton during low flows. Dissolved Reactive Phosphorus (DRP) in fresh water is readily available for plant growth. Total phosphorus is DRP plus phosphorus that is stuck to sediment and known as Particulate Phosphorus (PP).

The bond strength of the phosphorus attachment to sediment can be variable depending on the form of the cation coating on the soil particles. In some situations, adsorption and desorption of phosphorus (DRP to PP and back to DRP) can occur readily during stream flows from source to destination e.g. pasture to lake (Peryer-Fursdon et al., 2014).

Over time, phosphorus in deposited sediment can be released and become available for plant growth (PCE, 2012). Phormidium, a type of cyanobacteria (algae) that is prevalent in poor quality waterbodies, can access phosphorus directly from suspended sediment (Gluckman, 2017).

Another chemical impact related to sediment is the presence of heavy metals, mainly from urban environments, which can accumulate in sediment and then bio-accumulate in the food chain.

Physical impacts refer to suspended sediment making clear water murky and turbid, and deposited sediment blanketing stony riverbeds with mud and silt.

The ecological effects of fine suspended sediment in fresh water are less extensively researched than the effects of deposited sediment. Fine suspended sediment less than 2 mm can severely degrade stream habitats. The reduction in visibility reduces fish and birds' foraging ability. Reduced light penetration or euphotic depth affects aquatic plants' ability to photosynthesise (Clapcott et al., 2011 cited by Green and Daigneault, 2018). Suspended sediment acts as an abrasive for aquatic plants, fish and invertebrates. Certain migratory fish species avoid highly turbid waters and suspended sediment may clog gills of some freshwater fish species.

Excess deposited sediment fills up spaces between cobbles and gravel used by fish and invertebrates and can alter food resources or make them difficult to consume (Davies-Colley et al., 2015). It smothers aquatic life and the beds of rivers, streams and lakes, reducing and degrading habitat for shelter, feeding and spawning. Deposited sediment can also provide a foothold for exotic weeds. All of the effects from suspended and deposited sediment in fresh water, result in impacts on biodiversity and ecosystem health.

Lakes are particularly vulnerable as they trap sediment. Impoundments on rivers, such as dams, can give sections of rivers lake-like characteristics, also acting as sediment traps. Sedimentation in lakes and impoundments can cause excessive algal growth, affecting power generation and recreational values, particularly when the ability to flush dammed rivers is restricted. For example, the Waikato (Hughes, 2015) and Rangitāiki Rivers hydro schemes trap virtually all the suspended sediment, estimated at about 0.4 million ton per year (Hicks et al., 2004). The rate of sediment entrapment determines the life of the reservoir, in the absence of flushing or maintenance, and cuts sediment supply downstream. This can result in downstream riverbed degradation, streambank erosion, failure of bridge piers and coastal erosion. Logjams in rivers can act as temporary sediment traps as well (Quinn and Phillips, 2016).

Recreational values such as fishing, boating and swimming are also affected by suspended sediment decreasing the aesthetic appeal of waterbodies and concealing submerged hazards like rocks and logs. For instance, Batstone et al. (1994, cited by Green and Daigneault, 2018) found that people prefer hard sandy substrates and are put off by having to walk or stand on muddy substrate, which is the result of deposited sediment, while bathing.

Suspended sediment may also result in increased filtration, treatment and maintenance costs for water users. Deposited sediment will reduce the capacity of river channels (with associated impacts from flooding), hydro-dams and reservoirs, restrict navigation and conveyance for irrigation. To mitigate these impacts, more frequent dredging or flushing will be required.

The impacts of sediment on ecosystem services can occur many years after it is first displaced from the soil profile. This lag arises from its gradual movement down a catchment, in cycles of re-suspension and re-deposition (Blaschke et al., 2008). This is similar to lags that exist with nitrogen loss to groundwater; for example, nitrogen may take hundreds of years to move from the site where it leached to Lake Rotorua (Anastasiadis et al., 2014). Like with nutrients, there are also lag effects with sediment as well as it is stored in and near channels and is transported downstream over time (Blaschke et al., 2008). As a result, "interventions to reduce sediment loss may not generate positive ecological [and broader] effects in the short term; the legacy of sediment may impinge on the ecology for decades after management interventions are initiated" (Green and Daigneault, 2018, p. 6).

Estuaries and coasts

Except for lake catchments, estuaries and coasts are the ultimate receiving environments. Several of the chemical, physical, biological and ecological impacts that occur in fresh waters also occur in these environments. Seawater and marine sediment washed in from the coast and dispersed by waves and currents can compound sedimentation issues.

Turbidity and nutrient concentration is higher in estuaries than in other coastal environments (MfE and StatsNZ, 2015). Suspended sediment blocks feeding and breathing structures of animals including shellfish and fish. Reduced visual clarity reduces birds' ability to feed. Reduced light penetration also affects seagrass. Likewise, heavy metals from urban environments and some agrichemicals can also accumulate in sediment, and bio-accumulate in the food chain.

Muddying of the seabed changes the properties of the benthic environment and its suitability as habitat for benthic organisms. Smothering of seabed by mud kills plants and animals, including shellfish beds. By providing habitat for cordgrass and mangrove, deposited sediment can be further trapped. It can also smother horse mussel beds and seagrass beds which are important nursery grounds for significant species like snapper (Green and Daigneault, 2018). This is occurring in the Kaipara Harbour, which could have significant consequences for recreational and commercial fishing (MfE and StatsNZ, 2015) and importantly to local hapū and iwi customary harvesting. Fine sediment is believed to have been responsible, at least in part, for the demise of the Tasman Bay scallop fishery. Ninety per cent of fine sediment at the mouth of Tasman Bay's Moutere River was estimated to come from pine forestry plantations (Sivignon, 2018), using the finger printing techniques described above.

Areas of the Kaipara Harbour are now muddier and shallower due to sedimentation, seagrass and shellfish beds have decreased, or been completely lost from certain areas and the range of snapper and trevally within the harbour has retracted (Green and Daigneault, 2018). Likewise, silt from urban development, pastoral land, commercial forestry harvesting and quarries has accumulated in Tauranga Harbour, with an obvious impact on navigation, fish and shellfish habitats and recreational values (PCE, 2012).

Part 3

Mitigation practices to reduce sediment loss

A range of different mitigation options exist for sediment loss. This section reviews mitigation options, with a focus on those that are implemented *in the field* or at the *edge of field*. Practices to reduce soil erosion in the field chiefly focus on methods to bind the soil, prevent it from being mobilised in the first place and reduce direct rainfall impact. In contrast, those at the edge of the field chiefly seek to reduce water run-off velocity and trap suspended sediment.

In addition to field and edge of field mitigation, there are strategies to limit the impact of sediment on environmental quality within the receiving environment. One example is sediment caps that can work to prevent the release of phosphorus from lake bed sediment (e.g. as in Lakes Okaro and Okareka in the Bay of Plenty). Another example is removing sediment from waterways through the use of excavating equipment or dredges. This survey focuses just on strategies for sediment mitigation in the field or at the edge of field, given their wider relevance to policy.

The goal of this section is to review the farm level strategies available for soil conservation in New Zealand at a broad level and identify means to support their uptake on New Zealand farms. The strategies reviewed here could be considered in the context of a freshwater planning process under the NPS-FM.

The targeting of mitigation strategies at a farm level, on the other hand, requires a detailed on site assessment of erosion risks, their individual characteristics and critical source areas for sediment. Sector organisations and regional councils have, over decades, provided a lot of general guidance on good management practices for erosion and sediment control, aimed at minimising soil disturbance and exposure. However, there is limited evidence available for mitigation efficacy (Basher et al., 2016b) and even less present for cost. Furthermore, uptake of these practices is still variable around the country and between different sectors. This is to be expected because cost effective mitigation requires abatement strategies that vary across both space and time, according to the characteristics of a given watershed (McConnell, 1983; Doole and Pannell, 2012; Daigneault and Samarasinghe, 2015). However, there is evidence that water quality does not reach societal expectations in various places throughout the country (MfE and StatsNZ, 2017), suggesting that the uptake of such practices is beneath what is optimal from a social perspective.

Mitigation in the field

Increased pasture density

Increased pasture density can reduce soil erosion, as pasture roots and organic material serve to bind soil aggregates (Selby and Hosking, 1973). This strategy requires adequate fertiliser management, pasture improvement, and limited periods of overgrazing. A wealth of research has indicated that overgrazing leads to decreased pasture density, reduced rainfall infiltration, and greater rates of soil loss (Rauzi and Hanson, 1966; Warren et al., 1986). A major technical innovation that helped to reduce sediment loss through this source was the broad-scale adoption of aerial topdressing of superphosphate after World War 2 (Burdon, 1960). Dramatic improvements in pasture growth and quality were observed in response to the extensive application of artificial fertiliser from aircraft (Gillingham et al., 2008). This helped to increase the productivity and profitability of extensive-grazing systems, but also served to reduce soil loss through the development of increased pasture density (Lambert and Grant, 1980). Interestingly, early investigations into aerial topdressing were financed from funds targeted towards soil conservation and this remains a key strategy available to reduce sediment loss from steep hillslopes. Nevertheless, the rising cost of aerial topdressing (Morton et al., 2016), and the deleterious impacts of extensive forest clearance on erosion rates (Dymond et al., 2014) highlight the importance of other mitigation practices. Further, higher levels of fertilisation may lead to higher stocking rates on fragile hill soils, potentially leading to higher rates of soil loss. Mismanagement of fertiliser application on steep land that is prone to erosion can also lead to increased nutrient enrichment of waterways, particularly involving phosphorus and nitrogen.

Spaced tree planting on pasture

The principal focus of soil conservation in hill country for the last half century has been the spaced planting of exotic species, such as poplars (Populus genus) and willows *(Salix genus),* to anchor hill sides (Wilkinson, 1999). Common densities are 30, 50, and 100 stems per hectare, which correspond to spacings of 18 m, 14 m, and 10 m, respectively. Vegetative cuttings of 3 m length are usually planted by hand on steep slopes, being protected with a plastic cover of around 2 m length to protect the cutting from sheep and possum grazing. A broad literature outlines the benefits of these exotic species for drought fodder, shade, shelter, soil conservation, and timber production (van Kraayenoord and Hathaway, 1986; HBRC, 1998; McIvor et al., 2008; Phillips et al., 2012; Schwarz et al., 2016). The benefits for soil conservation are significant. Moderate tree densities have been shown to reduce erosion loss relative to open pasture by up to 70% (Hawley and Dymond, 1988; Dymond et al., 2010, 2014), 78% (McIvor et al., 2015), and 95% (Douglas et al., 2009, 2013), with variation arising from climate, planting density, plant species, tree age, and slope. Further, tree planting densities of 18, 36, and 54 stems per hectare were recently estimated, using expert opinion, to reduce sediment loss by 35, 75, and 90% (Fernandez and Daigneault, 2017).

However, this form of silvo-pastoral farming system has remained stubbornly uneconomic when evaluated in terms of tangible private returns, especially as often located on marginal hill country, unless values for timber production are considered (Parminter et al., 2001; Hill and Blair, 2005; McIvor, 2016). A major driver here is significant reductions in pasture production, which are typically around 20-40% (McElwee and Knowles, 2000; Guevara-Escobar et al., 2007). A complicating factor is that many on site (e.g. animal welfare, soil conservation) and off site benefits (e.g. offset flood damage, carbon sequestration) are difficult to value in economic terms (Jones et al., 2008; McIvor, 2016). The high cost of establishment has a significant impact on the economics of poplar adoption, even if their suppression of pasture production is not considered. For example, average farm profit on East Coast hard hill country has been around $105 per hectare for the last decade, but the annualised cost of poplar establishment is $85 per hectare, $470 per hectare, and $940 per hectare for 30, 50, and 100 stems per hectare, respectively. (These costings assume a base rate of $30 per stem, a 7% interest rate, and a 20 year payback period.)

The marginal economics of spaced planting has encouraged substantial investment in non-regulatory responses. The Hill Country Erosion Fund (HCEF) is a Ministry for Primary Industries (MPI) and Regional Council partnership, focused on improving the quantity and quality of land management advice available to landholders. It comprises around $2 million annually and is available through contestable funding. It began in 2007 and has been used to fund several flagship programmes, including the Sustainable Land Use Initiative (SLUI), Wellington Regional Erosion Control Initiative, the Kaipara Hill Country Erosion Project, and Waipaoa River Catchment Works facilitation activity. Further, poplar planting has been encouraged within the East Coast Forestry Project (ECFP), which ran from 1992 to 2014, and the Erosion Control Funding Programme that replaced it in 2014. Nonetheless, the major focus of these programmes has been full afforestation of land at severe erosion risk on the East Coast, especially in the Waiapu Catchment, given that widely spaced poplar trees are not effective at stabilising active gully erosion.

SLUI is a partnership between the Horizons Regional Council, ratepayers, and MPI that began in 2006. Over the last decade, it has received funding of $26 million from ratepayers, $15 million from the HCEF, $4 million from Central Government for science support and from the Afforestation Grant Scheme (see below), and $20 million from landholders (Cooper and Roygard, 2017). This $65 million funding has achieved 660 farm plans, land use mapping of 500,000 ha of land over 670 farms, and 30,221 ha of strategies to reduce erosion risk (ibid.). Recent evidence points to broad-scale improvement in trends for water clarity and E. coli concentration across the Manawatu-Whanganui region, since the SLUI began (Snelder, 2018). This provides confidence that public investment has led to an improvement in environmental outcomes, while recognising it is difficult to establish direct causality in these complex systems (Yang et al., 2014).

Conversion to forestry or manuka

Rangitāiki River hydro dam and exotic forestry

Another important mitigation strategy is full afforestation. This can broadly involve the establishment of mānuka (*Leptospermum scoparium*) for honey production, native forest with no utilisation, native forest for timber production, and exotic species (mainly *Pinus radiata*) for timber production. Manuka honey production in New Zealand is a growing industry, with positive economic outcomes experienced across the nation (Hock et al., 2014; Burke, 2015), but there are growing concerns around the deleterious impacts of oversupply. Manuka has the capacity to decrease soil erosion by around 65%, relative to adjacent pasture (Marden and Phillips, 2015).

Kanuka (*Kunzea ericoides*) has the potential to decrease erosion by up to 90%, given its larger leaf area and root system compared with mānuka. The regeneration of native forest, especially on marginal land, can provide a broad range of ecosystem services (McAlpine and Wotton, 2009). It also decreases soil erosion by around 90%, relative to open pasture (Dymond et al., 2010, 2014). However, it provides no income for landholders, which decreases its attractiveness to some.

National carbon policy has interesting implications here. On one hand, when emissions units are cheap, it can create incentives for deforestation and the subsequent establishment of pasture. (This pattern has been observed over the last decade in the Upper Waikato region.) Conversely, if emission units are more expensive, then it incentivises the afforestation of marginal land. This is important, given that carbon stores are more secure under native forest. In comparison, plantation forest builds up carbon, but this is lost if trees are harvested and not replanted (PCE, 2016). Another exception is where land retirement allows the intensification of other areas of a farm, through decreasing the inefficient use of inputs (such as fertiliser) on more marginal hill country.

There is also growing interest in native-timber production, with evaluations for kauri (*Agathis australis*) (Hock et al., 2014), silver beech (*Nothofagus menziesii*) (Evison et al., 2012), and totara (*Podocarpus totara*) (Cown and Bergin, 2009; Palmer and Bergin, 2017). These options diversify the national timber portfolio and also enhance the indigenous values of marginal hill country. Nonetheless, their economic proposition is challenged by (slow) growth rates that compare unfavourably with appropriate discount rates used in the investment appraisal. This aligns with standard theory in forest economics, whereby higher discount rates erode the value of slow-growing trees such that they appear uneconomic propositions for management (Buongiorno and Zhou, 2011).

The most common option for afforestation of erosion-prone land is the establishment of pine trees. These have been identified as being very effective for reducing soil erosion, in line with indigenous vegetation. Full afforestation reduces soil erosion by around 90%, relative to open pasture (Dymond et al., 2010, 2014). Accordingly, this strategy has been widely implemented. One example is that the ECFP has afforested over 40,000 ha of land (MPI, 2014), while around 15,000 ha of land has been afforested within the SLUI programme. Further, the Afforestation Grant Scheme aims to increase forest planting between 2015 and 2020 by 15,000 ha, to help build carbon storage, reduce soil erosion, and increase regional development. Also, there are now ambitious national plans to plant one billion trees before 2027, many of which will be pine trees for timber production.

It is fair to assume that most landholders highly value the opportunity to maximise economic opportunities (AgResearch, 2016; Elliott and Wakeline, 2016). Economic assessments of water quality policy have broadly identified a high-value proposition for plantation forestry, relative to extensive sheep and beef production (Doole, 2013; Parsons et al., 2015; Fernandez and Daigneault, 2017). However, these economic assessments of the superior value of forest are at odds with reality, where minor rates of afforestation occur on pastoral farms in the absence of subsidies. In part, this is due to major constraints accruing to the broad-scale adoption of plantation forest for reducing soil erosion. Barriers to the adoption of agricultural enterprises are reviewed at length by Pannell et al. (2006). The barriers listed in this review are used to determine constraints facing the extensive use of plantation forestry on sheep and beef farms. Key constraints are identified as:

1 There are limited benefits of the forest plantation to other enterprises on the farm. For example, plantation forests are seldom used as shelter or feed sources to support grazing animals.
2 Gully erosion can lead to dramatic losses in soil and hence pasture production. Poplar and willow trees are more successful at anchoring unstable gully sites, compared with pine trees.
3 There is high uncertainty around the returns that farmers will receive for pine products at harvest, in around thirty year’s time. This increases the uncertainty around the financial reward accruing to plantation forest.
4 High up front and maintenance costs are required for forest plantations, but most of the revenue is received at harvest that occurs around thirty years after planting. This is a significant impediment to landholders that require ongoing income to service debt and fund personal expenditures. In extreme cases, landholders may require bridging finance to help cover costs during the establishment phase, further increasing the cost and complexity of afforestation (MPI, 2014).
5 Smaller plantations involving a restricted set of age classes reduce the capacity to earn annual returns in the absence of carbon payments, relative to a larger forest consisting of all possible age classes. This arises from an inability to practice a profitable and sustainable (annual) harvest rotation in these circumstances.
6 Limited existing forest areas reduce the economies of scale that accrue to the harvest of larger plantations (Scion, 2016).
7 There is a risk that plantation forests can suffer damage from extreme weather and pests/disease.
8 Maintenance of significant forest areas on sheep and beef farms may conflict with the existing values and beliefs of landholders (AgResearch, 2016).
9 Maintenance of significant forest areas on sheep and beef farms may conflict with the existing skills of landholders, who generally possess more experience in the management of pastoral, rather than silvicultural, systems.

10 Broad uptake of forestry within a catchment will greatly impact community vitality and could reduce the amount of agricultural production from this region (TLG, 2016).

11 There is a strong inertia in established management plans given a strong drive to repeat learned actions, even in the presence of new opportunities or constraints (Gonzalez and Dutt, 2011).

12 Access to subsidy schemes may be more difficult on Māori land due to the presence of multiple owners (MPI, 2014).

13 Not all areas of a farm are generally suited to the establishment, maintenance, and harvest of plantation forests; for example, because of soil and slope constraints (Palmer et al., 2010).

Furthermore, the costs of establishing pine forests and silvo-pastoral systems based on poplar trees is quite similar (D. Kervell, Northland Regional Council, pers. comm.)[2]. An additional advantage of the poplar system is that grazing area is not lost under these trees, unlike for pine forestry.

These factors highlight the complexity of adoption, especially for forestry systems that take many years to yield an economic return. It also demonstrates the many attributes that impact the suitability of a new practice for a given farming system.

Further, the harvest of plantation forest can have adverse off site impacts, such as the transport and delivery of sediment and slash downstream. Plantation forest is effective at reducing erosion once fully established. However, land is vulnerable to soil erosion for five or six years during the harvest and re-establishment phase (Doole, 2015b). Also, certain harvest practices can accelerate soil loss. Examples are inadequate landing and track design, together with narrow stream buffers (Basher et al., 2016b). These factors have increased regulatory focus on the off site effects of harvest - for example, through the *National Environmental Standards for Plantation Forest 2018* - that increases the need for appropriate mitigation of both on site and off site effects.

Mitigation at the edge of the field

There are several strategies available at the edge of the field. These are focused on the capture and filtering of sediment moving in overland flow.

[2] For example, assuming 1,100 pine trees are planted per hectare at $2.50 per stem, then the total cost is $2,750 per ha. This is very close to the total cost of establishing 100 poplar trees per ha at $25 per stem, which is $2,500 per ha.

Riparian fencing and planting

Riparian fencing and planting on the Wairoa River, October 2013 (and January 2015 inset)

The benefits of riparian fencing for reducing streambank erosion arise from livestock exclusion and stabilisation of the streambank by the roots of more vibrant riparian plant communities. There is also the potential for riparian plants to filter sediment moving across the soil surface in overland flow; however, these benefits are often minor due to the high velocity of water during moderate- to high-rainfall events. Past work shows that the exclusion of cattle will reduce sediment loss (Hughes and Quinn, 2014), with reported levels of 30-90% (McKergow et al., 2007), 35% (Fernandez and Daigneault, 2017), 40% (Monaghan and Quinn, 2010), or 60% (Semadini-Davies and Elliott, 2012). Further, previous research indicates that the exclusion of all livestock will reduce sediment loss by: 20-25% (McDowell et al., 2013), 24% (Semadini-Davies and Elliott, 2012), 35% (Fernandez and Daigneault, 2017), 50% (Monaghan and Quinn, 2010), 60% (Daigneault and Samarasinghe, 2015), or 80% (Palmer et al., 2013). Riparian planting on fenced streambanks can help stabilise streambanks with their root systems. This can lift mitigation efficacy by 10-20% (McKergow et al., 2007), 15% (Monaghan and Quinn, 2010), or 15-25% (Sweeney and Newbold, 2014).

Riparian margins wider than 3-5 m may have limited benefit for sediment mitigation because the main filtering that is available has been done by this stage (Basher et al., 2016b). However, that may not particularly be the case for other contaminants, like nutrients or E. coli. The efficacy of stream fencing in reducing sediment loss to waterways varies according to the importance of streambank erosion in a given catchment. This can vary greatly, according to the catchment size, climate, soil type(s), stream density, legacy effect of previous erosion and topography in the locality. For example, the proportion of total erosion coming from a streambank has been reported as around 1% for the Waipaoa River (Poverty Bay) (De Rose and Basher, 2011), 60% for the Mangaotama Stream (Waikato) (De Rose, 1999), and more than 90% in the Kopurererua Stream (Bay of Plenty) (Hughes and Hoyle, 2014).

It is important to distinguish between the origins of sediment lost from streambanks and beds versus the suspended sediment originating from adjoining farmland and lost in runoff during storm events. Sediment mobilised during storms and being carried by surface water flow across the surface of pastures or fields most often is already amalgamated in rivulets of water or larger ephemeral flows when it reaches streams and is intercepted by a fence and riparian area. Such runoff flows generally rush through and over the 'filter' of riparian areas, with virtually no sediment capture achieved. Peryer-Fursden et al. (2015) also found that this sediment load in stormwater runoff from fields (rather than from streambanks) has a higher concentration of more weakly attached particulate phosphorus, which has a propensity to become biologically available downstream.

Riparian fencing provides an opportunity to decrease stream bank sediment loss (McKergow et al., 2007; Sweeney and Newbold, 2014), while not requiring landholders to change their farming system to a large extent (Doole et al., 2018). This significant leverage has contributed to high adoption rates - and consequently water-quality improvement - across the nation, though these benefits have principally been concentrated in the dairy sector (Semadini-Davies and Elliott, 2017). This leverage has also promoted the popularity of this strategy as part of integrated packages aimed at improving water quality across New Zealand, both regulatory and non-regulatory. These include the Auckland Unitary Plan, proposed Regional Plan Change 1 in the Waikato, Northland's proposed Regional Plan, Tukituki River Catchment Plan, proposed Gisborne Regional Freshwater Plan, Taranaki Regional Fresh Water Plan, Greater Wellington Proposed Natural Resources Plan, Canterbury Natural Resources Regional Plan, and the proposed Marlborough Environment Plan. Additionally, it is expected that national stock exclusion regulations will eventually be promulgated.

A key non-regulatory response has been the establishment of the Sustainable Dairying - Water Accord, which establishes a set of national targets around the environmental performance of dairy farms. The Accord represents a partnership between DairyNZ, Federated Farmers, fertiliser companies, Ministry for the Environment (MfE), Ministry for Primary Industries (MPI), and regional councils, among others. It is broadly focused on efficient water use, effluent management, nutrient management, riparian management, and stream fencing. Recent reports indicate substantial progress including, but not limited to: fencing to exclude stock from 26,000 km of streams; 99% of regular stock crossings do not require livestock to enter waterways; $10 million spent on environmental stewardship over 2013-2015; 9,500 nutrient budgets were processed; and only 5% of farmers being assessed as non-compliant with respect to effluent disposal (DCANZ and DairyNZ, 2016). However, the fencing and stock crossing provisions of the Dairy Accord only apply to larger streams (i.e. over 1 m wide and 30 cm deep).

Nonetheless, stream fencing as a mitigation option has a number of limitations in practice. These include:

1. While clearly effective for minimising 'streambank' erosion, it is less effective for sediment entering streams laterally in overland flow via rivulets and ephemerals.
2. The fencing of streams can mean that large proportions of a farm are excluded from grazing. This is most problematic on lifestyle blocks with small farm sizes or in hill country where stream density is high.
3. Nitrogen loss from intensive agriculture is a key issue in the context of water-quality degradation in New Zealand (Larned et al., 2016). However, stream fencing does little to abate this (McDowell, 2017).
4. Stream fencing is prone to flood damage, especially where riparian buffers are limited in their extent.
5. Weed control can be an issue, with riparian margins acting as a seed source for weeds that are potentially difficult to control in surrounding pasture. Pest species such as gorse (*Ulex europacus*) or blackberry (*Rubus fruticosus*) are examples.
6. The cost of stream fencing may exceed the private benefits of fencing for landholders, that chiefly accrue through reduced adverse animal health risks [e.g. liver fluke (*Fasciola hepatica*)] and animal losses from misadventure. These benefits are particularly profound in steep hill country, where access to quality water sources is often more challenging and stream banks are more incised.
7. Streams can be difficult to fence on steep slopes, due to low soil depths and undulation.
8. Stream fencing may require a high maintenance cost in steep country, due to high erosion risk.

9 Stream fencing necessitates water reticulation. This can be expensive in hill country, in most cases, doubling the cost of stock exclusion (Journeaux and van Reenan, 2016; Daigneault and Samarasinghe, 2015).

10 The difficulties of fencing in hill country means that a high proportion of the stream density within a watershed may not be subject to stock exclusion. McDowell et al. (2017) evaluated a proposed policy that excludes cattle, deer, and sheep from all streams wider than 1 m and more than 30 cm deep in catchments with an average slope of less than 15°. They showed that 84% of total sediment input to fresh water occurs outside of fenced areas, demonstrating the importance of small, headwater streams to contaminant delivery (ibid.).

Another option to improve riparian areas is the retention, protection, or restoration of headwater seeps. Headwater seeps are characterised by the presence of shallow groundwater springs, low slope gradients, poor drainage, and strong populations of rushes (*Juncus spp.*) and sedges (*Carex spp.*) in valley basins. They are not located in areas of high water flow, given that this prevents the establishment of the requisite plant species. Seeps act as important filters for many contaminants on New Zealand farms; moreover, they help to reduce nitrogen input to waterways through converting nitrate to nitrogen gas through the process of denitrification (Seitzinger, 1994). Their contribution to attenuation is encouraged by their high incidence in the headwaters of many ephemeral streams across New Zealand hill country (Merot et al., 2006; Hughes and Quinn, 2014). While important from an environmental perspective, many seeps have been damaged through drainage or the development of farm ponds (Tanner et al., 2014). If drained for pasture establishment, economic incentives for retention and restoration accrue because of the poor growth of pasture in these areas due to waterlogging. For example, McKergow et al. (2007) estimate that seeps can reduce sediment loads by up to 60%. However, it can be expensive to restore them, due to the cost of fencing and revegetation. Further, the temptation to graze these areas during periods of scarcity can increase sediment loss by between four and thirty six times (Black et al., 2015). Flood events can also lead to remobilisation of captured sediment, working to decrease the mitigation potential of headwater seeps.

Sediment retention infrastructure – sediment traps and detainment bunds

Sediment retention on farmland

Sediment traps

Another method to capture sediment along a flow path is the use of sediment traps. Sediment traps are ponds that slow the velocity of sedimentladen water down a water course, causing particles to settle. They resemble farm ponds but are more effective for sediment capture because they are usually built adjacent to, rather than on, a water course and their water level is often not at full capacity. Capacity is important because if full, then suspended sediments will simply flow through the structure rather than being deposited. Like farm ponds constructed in a stream channel, an issue with sediment traps is that they aggrade over time. Thus, annual maintenance is required to prevent sediment buildup and a subsequent decline in efficacy. Up to 90% of coarse material is captured (Hudson, 2002), with reported rates of removal of 70% of total sediment load if combined with a wetland (Doole, 2015b; Daigneault and Samarasinghe, 2015). If used by itself, mitigation will achieve a 20% reduction of sediment if the trap is situated in a stream channel or a 50-60% reduction in sediment if the trap is situated adjacent to the primary water course (C. Tanner, NIWA, pers. comm., 17/3/2015). Effectiveness of sediment removal is much higher for wetlands as, unlike sediment traps, they can capture fine sediments that are mobilised during storm events (Johannesson et al., 2015). Nonetheless, this does greatly depend on soil type, sediment source, and particle size. For example, colloidal clays that are broadly distributed in some regions - for example, weathered volcanic hill country in Northland - can stay in suspension for weeks.

Detainment bunds

Unlike the coarse sediment traps discussed above, a novel technology specifically targeting fine particle sediment capture is the ongoing development of the concept of ‘detainment bunds’ occupying high-value pasture areas without compromising pastoral production. A feature of these structures is their low cost, particularly relative to constructed wetlands (see below). For example, a detention bund is costed at around $25 per year for each hectare serviced by the structure (Doole, 2015b).

The concept of a detainment bund was initiated by Bay of Plenty Regional Council’s “P-Project” in 2010 and continued since 2017 by the independent Phosphorus Mitigation Project (PMP). This applied research project on detention bunds is funded by MPI in collaboration with the farming sector industry (seven co-funders). These detention bund structures involve the construction of a bund, an earth wall up to 2.5 m in height (the permitted activity threshold for dams) with a decanting outlet control built to intercept storm across ephemeral water courses. They create large areas of temporary ponding over pasture allowing fine particle settlement and DRP adsorption to occur.

The ability of detention bunds to capture phosphorus and sediment was qualitatively proven in 2012 (Clarke et al., 2013). The primary focus of these structures has been the capture of phosphorus and sediment during storm flows. The systems are designed such that the detained water does not unduly compromise pasture health and for this reason ponding is limited to a maximum of three days. Detention bunds easily fit within a farm system since they do not require fencing and planting; also, grazing land remains unaffected for much of the year.

Existing literature shows that bunds can capture sediment with reasonable efficacy, with recent estimates of around 70% (Doole, 2015b; Daigneault and Samarasinghe, 2015). Current research by PMP has some initial results indicating the sediment capture rates may be around 90%. PMP will also be assessing the effectiveness of detention bunds to capture nitrogen and bacterial contamination.

Such stormwater management structures also improve the effectiveness of downstream wetlands by moderating peak flows and minimising bypass by slowing the velocity of water through the catchment. Areas immediately behind a bund also benefit from the additional phosphorus that arises from deposited sediment. However, it is important to design these structures carefully, such that storm flows do not degrade performance through destruction of the bund. The PMP detainment bunds (22 now constructed) are built to achieve a ponding storage to catchment ratio of more than 120 m^3 per ha. Applying this design criteria, a minimum ratio (120:1), ensures an adequate detainment bund pond volume to enable settlement to occur. Settlement of particles needs a large body of relatively still water. If ponding areas are too small, incoming stormwater will remain in motion and turbid with little opportunity for settlement or absorption.

This careful matching of ponding volume to the size of the contributing catchment is essential to ensure success with both sediment and phosphorus capture, and minimising risk of destruction during large events. PMP uses 1 m contour maps generated from LiDAR data to scope the ephemeral flow paths of farming catchments for appropriate sites, with 'appropriate' being defined by the achievement of the minimum storage to catchment ratio (120 m^3 per ha). This process has been done manually for some years using LiDAR contours and ArcHydro; however, PMP, in conjunction with DairyNZ, is currently engaged with GIS specialists to attempt to automate the process (Paterson et al., 2018).

It is expected that this GIS scoping model for suitable sites will confirm that not all landscape types are suitable for detainment bund installation and achieving adequate storage to catchment size requirements. For example, it is expected that opportunities for detainment bund construction on particularly flat or steep land will be limited.

While initially targeting sediment and particulate phosphorus (PP), PMP is also researching the capture of DRP both naturally, with DRP adhering to particles whilst standing in the detainment bund pond, and also with a proposed trial (2019) to treat the inflow to the ponded water in the detainment bunds with a non-toxic flocculant.

Wetlands

A core activity for sediment management is the protection, restoration, and/or development of wetlands. Wetlands are shallow ponds in which rushes and sedges are prevalent. A broad array of wetland options exist, with the relative suitability of each form dependent on flow path, primary contaminant type, size, slope, and soil type. These structures provide an integrated means to mitigate nutrients and sediment arising from agricultural production (Tanner et al., 2013). Their suitability is encouraged by the capacity of wetlands to capture and attenuate contaminants with little implication for management within their catchment. Aggradation in a wetland is promoted by high vegetation density and detained water (McKergow et al., 2007); thus, it is important that wetlands are developed such that they do not intercept water travelling at high velocities. Significant mitigation efficacy is achieved for sediment with the use of wetlands, with estimates of around 60-80% removal prevalent in the recent literature (McKergow et al., 2007; Tanner et al., 2013; Doole, 2015b; Daigneault and Samarasinghe, 2017).

However, they are not effective in areas with a lot of free-draining soils because they cannot intercept sufficient amounts of water. Wetlands can also act as a source of faecal microbes, especially at low flows, due to inputs arising from bird populations. Moreover, it is critical to establish mechanisms that protect a wetland from high flows, to avoid failure during extreme weather events. This is particularly an issue with sediment management, as high flows can remobilise sediment captured by the wetland back into the water column. Also, wetlands are expensive to develop if they are not already present or are severely degraded. For example, small and medium sized wetlands are costed at $96 per year and $240 per year, respectively, for each hectare serviced by each structure in Doole (2015b). (These estimates do not account for the loss of grazing land with the establishment of the wetlands). This is around four to ten times greater than the cost of a detention bund in this study (ibid.), reflecting the substantial costs associated with fencing and planting edge-of-field structures. Additionally, Tanner et al. (2013) identify an average construction cost of $1.9 million each for thirty proposed wetlands in the Southland region. The most cost-effective option for development is restoring and improving degraded wetlands, given that this avoids or reduces the need for expensive earth works in the establishment phase. Moreover, earthworks are less costly if wetlands are established in rolling country, as opposed to on flat areas.

Detainment bunds coupled with constructed wetlands

Detainment bunds can be integrated with constructed wetlands. Bunds can be placed before wetlands, to capture sediment. Or, alternatively, on poorly-drained soils, fenced wetlands can be placed immediately upstream from the bund, to provide mitigation of nitrogen.

Constructed wetlands are generally built to target nitrogen abatement, rather than that of phosphorus or sediment (McKergow et al., 2007; Tanner et al., 2013; Doole, 2015b). Sediment aggradation in a constructed wetland can significantly shorten its lifecycle, as sedimentation fills up the carefully engineered flow paths designed to increase the residency time of water within it. At Lake Okaro in the Bay of Plenty, two large detainment bunds were built upstream of a large constructed wetland to maximise its effective working life. These detainment bunds modify peak storm flows, so there is less bypass of the wetland during high-intensity storm events, and remove most of the suspended sediment that would otherwise aggrade within the constructed wetland.

A major downside of including a wetland is cost. The use of a wetland increases total cost by 20-25% due to the additional need for fencing and planting, together with the opportunity cost of lost grazing land (Doole, 2015b; Daigneault and Samarasinghe, 2015). If the capture of sediment is the primary target for improving water quality in a catchment and nitrogen loads are not an issue, the construction of detainment bunds without wetlands in tributary ephemeral areas will be a more cost effective and sustainable option, but again depends on the particle size of the source sediment or eroded soil.

Discussion

There is extensive evidence that broad-scale deforestation of the New Zealand landscape and its replacement with exotic pastures has led to widespread soil erosion (McDowell and Wilcock, 2008; Brooking and Pawson, 2011), especially on steep hillsides with a sedimentary lithology (Dymond et al., 2010). A wide range of strategies for soil conservation exist and most have been available for many years. Moreover, many of them significantly reduce rates of sediment loss as outlined above. Nevertheless, high rates of soil erosion persist across New Zealand, with estimates of yearly losses ranging from 80 million to 192 million tonnes (Basher, 2013; MfE and StatsNZ, 2018).

Private decisions around sediment loss have both on and off site impacts that vary across time (Doole et al., 2018). Section 5(2) of the RMA requires “safeguarding [of] the life-supporting capacity of air, water, soil, and ecosystems”. However, some contend that public agencies have placed too little focus on the on site impacts of soil erosion, thereby downplaying the importance of this legislation (Bloomberg and Davies, 2012).

Rational farmers should consider the financial implications of soil loss, both in terms of annual returns and resale value (McConnell, 1983; Doole and Hertzler, 2011). However, current rates of soil erosion worldwide, including in New Zealand (Hicks et al., 2011), suggest these incentives for conservation are weak (Nearing et al., 2017). There are many causes for this, including: complex relationships between soil depth, annual profit, and resale value (Grepperud, 2000); high output prices earned in the current period (Doole and Hertzler, 2011); high uncertainty about future returns (Kingwell and Xayavong, 2017); low profit to interest ratios in agriculture (Oltmans, 1995); preferences to retain existing management plans (Gonzalez and Dutt, 2011); prices of land do not reflect soil depth (Goetz, 1997); short planning horizons (e.g. due to leasehold arrangements) (McConnell, 1983); and use of inputs to offset losses in production due to elevated soil loss (Burt, 1981). Further, there is a disparity between the magnitude of benefits that accrue to landholders, versus those earned by broader society. This disparity has three main, interdependent causes. First, low private gains drive underinvestment in mitigation technologies, resulting in perverse social outcomes (Pannell et al., 2006, 2014). Second, landholders generally have little to no incentive to consider the off site or external costs of their actions across time (Jaffe et al., 2004; Pannell et al., 2006). Last, a farmer is likely to preserve soil resources at a rate well below that which is preferred by society. This arises because of market distortions from taxation (Brealey et al., 1997), imperfect land markets (McConnell, 1983), risk attitudes (Teklewold and Kohlin, 2011), short business tenure (Lovo, 2016), and since society must ethically consider the needs of future generations, unlike private agents (Jaffe et al., 2004; Arrow et al., 2014). These effects work to reduce the diffusion of mitigation practices across a landscape.

Establishing the investment case for policy programmes targeted at sediment is difficult, given the complexity of the relationship between mitigation at the field scale and water quality outcomes across a catchment. This is even more complex where there is significant sediment loads present within a stream network, as investment in further mitigation will seldom be able to affect their capacity to influence instream suspended sediment (MPI, 2014). The relationship is such that it is statistically challenging to identify causality of performing mitigation and an improvement in environmental factors. A recent examination indicates that there is a correlation between broad investment in abatement options and water-quality measurements in a ten year review of the Horizons Regional Council's Sustainable Land Use Initiative (SLUI) (Snelder, 2018). However, it is challenging to establish causality, especially when the impact of interventions is masked by some natural recovery and changes in climate, land use, and land management.

Improving the adoption of mitigations for sediment loss is difficult, because changing agricultural practices entails the consideration of unfamiliar strategies, techniques, and technologies. The process of adoption for agricultural practices can be complex and time consuming, given that practices have multiple attributes that help diverse landholders satisfy their management goals. The following overview of these attributes draws from the seminal work of Rogers (2003) and Pannell et al. (2006). This list highlights the different characteristics of an innovation that influence its diffusion across a landscape. It also emphasises the complexity that landholders face in the assessment of new conservation practices.

There are two broad characteristics of a technology that drives its uptake: its relative advantage (Rogers 2003), and its trialability (Pannell et al., 2006). Relative advantage outlines the additional benefits and costs that are perceived to accrue to a landholder upon adoption, while trialability denotes the ease with which a landholder can test a technology. Relative advantage can be broadly perceived as how profitable a practice is, compared to current practice. The main implication of relative advantage is that conservation practices will not be broadly adopted unilaterally if they are unprofitable or otherwise fail to offer substantial additional benefits to an existing system. Trialability concerns how straightforward it is for a landholder to test a technology, to provide more information with regards to the specific magnitude of the relative advantage that the innovation offers in their management context. A key component of trialling is reducing the overall level of risk and/or uncertainty pertaining to a given technology. The higher a method's trialability, the less risk is involved in its adoption given that a user can discontinue its use rather rapidly. Successful initiatives have paid close attention to these factors. For example, the Living Water Programme - a joint initiative between the Department of Conservation (DOC) and Fonterra - focuses strongly on actions that are both game changing and scalable (Living Water, 2016). Also, relative advantage and trialability are influenced by contextual factors, such as social networks, interpersonal differences (e.g. in terms of innovativeness and risk preference), and ongoing changes in institutions, markets and technology.

The principal drivers of relative advantage for sediment conservation practices on farm are:

1 The short, medium, and long term input prices, output prices, and level of production associated with the practice (Pannell et al., 2014). These components are the principal determinants of short term profit for an innovation, so are highly relevant for landholders who place significance on economic outcomes (AgResearch, 2016). In the context of sediment loss, the adoption of plantation forest is constrained because it provides no short term income for landholders, though carbon payments (PCE, 2016) and the potential inclusion of the livestock sector into the Emissions Trading Scheme may change this.

2 Co-benefits of the new practice, for other parts of the agricultural system. In the context of sediment loss, the adoption of wetlands is constrained because they provide little benefit for agricultural production.

3 Transition costs associated with the new practice; in particular, this includes high up front costs for the establishment of new capital assets. In the context of sediment loss, the adoption of spaced planted poplars is constrained because of their high establishment cost.

4 The riskiness of the new practice. The adoption of plantation forest is challenged by uncertainty around its long term profitability, particularly with long lived species [e.g. kauri or Douglas fir (*Pseudotsuga menziesii*)].

5 The compatibility of the technology with a landholder's farming system. The new practice must be integrated into the existing mix of technologies, practices, and resources that exist on the farm (Crouch, 1972). There may be a degree of uncertainty about the compatibility of the practice with the farm system that simply cannot be resolved prior to implementing the practice. This means that the likely outcomes of changing practice can be quite difficult to predict. Consequently, the decision to change an agricultural practice is often perceived to be financially risky. An example in the context of sediment loss is that it may be difficult to introduce spaced planted poplars onto a farm because the maintenance of pasture quality requires rotational grazing of each paddock with cattle, which may damage the young trees.

6 The profitability of the practice that the innovation would replace. An example is that the adoption of agroforestry on dairy farms is low, given the high returns to intensive dairy production in New Zealand (Doole and Romera, 2015).

7 The timing of income received from the new practice. An example is that mature pine forest decreases sediment loss by around 90% (see previous section), but timber income is only received 28-30 years after establishment.

8 Compatibility with existing values and beliefs. Landholders often base their management decisions on their existing values and beliefs; thus, new practices are only likely to be adopted if consistent with these. An example is that pastoral farmers may find it difficult to maintain an agroforestry system, given their existing set of values and identity related to historical linkage with intensive pastoralism (Jay, 2005).

9 The environmental credibility of the practice. Some farmers have increased their use of riparian fencing and planting, as this helps to convey their care for the environment (Maseyk et al., 2017). However, this 'credibility' is under threat by criticism that riparian protection, while important, is not addressing the main losses of sediment, bacteria, and nutrients arising from increased intensification.

10 The major benefits of the action are borne off site. Fencing of large streams helps to reduce riparian erosion, improve animal health, and decrease livestock mortality through misadventure. In contrast, a large constructed wetland will often yield low direct financial benefits for a farming system. Nonetheless, edge-of-field options are more favourable where they are built in wet areas where the amount of pasture production that is lost is much lower (Tanner et al., 2013; Doole, 2015b).

The principal drivers of trialability for conservation practices on farm are:

1 The ability to cost effectively test the innovation on a small scale. This is favourable for many options for managing sediment loss, such as spaced planting and plantation forestry.

2 The observability of the effects of an innovation. The use of artificial fertilisers on pastoral farms in New Zealand diffused rather rapidly, given the significant impacts that they had on pasture yields, livestock performance, and farm income (White and Hodgson, 1999). In contrast, it can be difficult to appreciate the value of many mitigation activities (e.g. a wetland) given that their impact is off site and their implementation may not affect agricultural production.

3 Long time scales. It can be hard to discern the benefit of a new practice if it takes many years for it to have an impact. An example is that it may take many years for afforestation to arrest soil loss (Dymond et al., 2014), especially if there is established erosion activity (MPI, 2014).

4 Interdependencies between elements of the farming system. It may be difficult to separate out the benefits of new practices, relative to current practices, given climate variability and the complex nature of agricultural systems arising from the interdependency of many different elements. An example is trying to isolate the impact of spaced planted poplars on the profitability of an extensive sheep and beef farm.

5 Variability in the production and profitability of the farming system. It may be difficult to separate out the advantage offered by a new practice, as compared to current management, given that management, production, and profit varies every year. An example is trying to determine how stream fencing has impacted the production and profit of intensive dairy production across time.

6 A trial may fail or provide biased results because of exogenous factors. An example is that young poplars on a hillside may die due to low rainfall and high wind throw. These outcomes can create a legacy effect, reducing the likelihood of current and future adoption of this mitigation (and potentially others) on a particular farm.

7 A trial may fail or provide biased results because of poor management. An example is that young poplars on a hillside may die due to shallow planting or damage arising from livestock. As with point #6 above, this can also create an enduring legacy of reluctance to further investment in mitigation.

Together, these factors isolated as key determinants of relative advantage and trialability highlight the complexity of adoption, and the many attributes that impact the suitability of a new practice for a given farming system. Adoption can have weighty consequences for the future financial performance of the farm enterprise. Moreover, the decision can also entail social and psychological risks for the landholder in that the outcome could affect their feelings of achievement, self-fulfilment and self-esteem; the wellbeing of family members; and how the landholder is perceived by peers. Consequently, the process of influencing landholders to increase their adoption of mitigation practices can be challenging. Nonetheless, there are benefits to further diffusion, which helps to promote momentum. These returns will typically arise from the accumulation of knowledge and experience, band wagon effects, and capacity for co-learning (Park, 1994; Costanza, 2014).

The management of sediment loss is further complicated on New Zealand farms because there are potentially many sources of it. Sediment loss can occur from flatter land, hillsides, streambanks, and streambeds on a typical hill country farm. Moreover, the primary form of erosion that could occur on one part of the farm may be different from that occurring somewhere else on the same farm. For example, gully erosion may be prevalent on one part of the farm, while landslides are common elsewhere. These factors mean that an integrated approach to mitigation is required. This increases the complexity of the adoption decision by lifting the amount of information, processing, and decision making required, making it harder to identify the attractiveness of each component and the overall package itself (Schroder et al., 1967; Feder and Umali, 1993; Rogers, 2003).

Public policy aims to increase the adoption of practices to mitigate soil loss across New Zealand. There is a general reluctance to regulate land use and/or land management, given the subsequent need for expensive monitoring and enforcement actions. Rather, the focus has been towards the use of subsidies to encourage adoption, given the multi component nature of integrated strategies to mitigate sediment loss and the inability of many options to increase farm profit. Additionally, the multi component aspect of mitigation has driven a strong policy focus towards land management advisory services within regional and unitary councils. Such advisory services are readily taken up by the industry sectors rolling out their respective Environment Management Systems such as the Sustainable Milk Plan (DairyNZ), Fonterra farm environment plans, and the Land and Environment Planning toolkit (Beef + Lamb New Zealand and Deer Industry NZ).

Mitigation options for sediment have many benefits for the environment that are not valued in markets. For example, the spaced planting of poplar and willow trees secures and stabilises hillsides, filters nutrients and contaminants, improves biological control of diseases and pests, provides additional shade and shelter for grazing livestock, reduces soil compaction through encouraging transpiration, and sequesters carbon (Jones et al., 2008; McIvor, 2016). Capturing the value of these ecosystem services and passing them on to landholders is one means of helping to improve the value proposition of mitigation practices for sediment on New Zealand farms.

In the absence of these instruments, public funding has been important to help drive the uptake of measures to reduce soil loss. Examples are the establishment of erosion control measures on over 55,000 ha of land within the ECFP (MPI, 2014), and 30,221 ha of erosion protection under the SLUI (Cooper and Roygard, 2017). However, evidence with respect to the impact of these actions on rates of soil loss is mixed. The ECFP has involved broad-scale planting of susceptible hill country. This reduced the number of gullies exhibiting erosion by 54% over 1957-2008; however, over this period, total soil loss increased by 78% and total gully area increased by 17% (Scion, 2012). Further, while some headwater streams are stabilising given these actions, larger rivers in the catchment are still aggrading, reflecting continued sediment delivery (Scion, 2012). In contrast, SLUI investment has likely led to improvements in water quality trends (Snelder, 2018).

These differences are disparate, but reflect the practical difficulty associated with trying to stabilise or reverse turbidity issues in waterways when significant legacy loads are present in a watershed. These difficulties are pronounced in regions where soft sedimentary lithologies, steep slopes, and a long history of extreme changes in vegetation cover, increase the amount of sediment entering water courses, as on the East Coast (Marden, 2011).

The adage that “prevention is better than cure” is particularly relevant when dealing with sediment issues. An important first step is the adoption of ‘Good Management Practice’ (GMP). While in many cases this will not be sufficient to meet societal goals for improved water quality, this is important to help build early engagement, reinforce the value of the soil resource, and reduce uncertainty about the off site impacts of sediment loss. Local Government agencies have traditionally been a driver for soil conservation programmes in New Zealand, with various funded programmes available to assist landowners in specific areas with specific erosion issues. A more holistic approach to achieving greater environmental sustainability through promotion of GMPs has led to the development of ‘farmer owned’ environment programmes. This requires a receptive attitude and the adoption of a management framework that enables ‘continuous improvement’ of day to day practice to minimise the potential for soil loss from the farming/forestry landscape.

Undertaking a process of ‘continuous improvement’ is the core fundamental prescribed in International Standards for Environment Management Systems (ISO 14001). Since 2005 when the first industry promoted Environment Management System (EMS) was published in the New Zealand Deer Industry, there has been significant uptake of the key principles of ISO 14001 by the other New Zealand farming sector organisations. For example, both Beef + Lamb NZ and DairyNZ now offer Farm Environment Management templates to their farmer members, through the Land and Environment Planning tool kit (LEP) (Synge, 2013) and the Sustainable Milk Plan (SMP) (Brocksopp et al., 2015) frameworks, respectively. (Nonetheless, the uptake of some or all of these programmes is still limited in some regions throughout New Zealand).

Common GMPs that can be written into a farms’ Farm Environment Plan include practices that minimise soil disturbance and exposure of bare soil to raindrop impact; identifying Critical Source Areas (CSAs) for sediment (and phosphorus loss), maintaining adequate pasture covers, minimising cultivation and adoption of no-till or low-till methods, prevention of pugging particularly in areas frequently washed over by stormwater (ephemeral flow paths), and lower stock intensity on steep ground. A land manager engaged with an EMS will enter such practice improvements into the farms EMS plan where they are scheduled with achievable timeframes and followed up with a review or audit to check progress. Monitoring and reporting of farm practice change and outcomes are a significant component of the recently released national *Good Farming Practice Action Plan for Water Quality 2018* (Good Farming Practice Governance Group, 2018).

Non-regulatory programmes have much potential for gaining momentum around better environmental management on New Zealand farms. Providing advice, effort, and funds to farmers provides a foundation for scalable change, especially when the community is involved as well. Investing in this synergy between multiple stakeholders helps establish impetus, which can then build upon biophysical and economic research that helps to guide the placement of the right mitigation practice being established in the right place at the right time.

Nevertheless, a number of concerns have also been raised with respect to the impact of non-regulatory programmes. First, land management works within non-regulatory programmes are often targeted according to who is willing to work with the Regional Council and their partners. An example is that uptake of the farm planning process within the SLUI programme is voluntary and not determined based on the risk of sediment loss. Some people doubt the capacity of this approach to deliver practical outcomes, given the absence of compulsion (e.g. McBride, 2016) and the aversion of some landholders to work with regulators (AgResearch, 2016). Nonetheless, the eventual scale of the programme likely means that such a barrier will not affect the total amount of mitigation achieved once fully implemented (Dymond et al., 2014). Moreover, all of the farm-planning work within this programme is targeted at highly-erodible land, so erosion on those farms with lower relative levels of sediment loss is still problematic from an environmental perspective.

Second, non-regulatory work has focused extensively on building capacity and partnerships between stakeholders. This reflects the importance of strong relationships between environmental authorities and landholders whose management decisions impact the environment (Blackmore and Doole, 2014). Nonetheless, this approach is intensive and requires significant resourcing. The uptake of soil conservation work by landholders is improved if it fits well with the farm system at a holistic level (Pannell et al., 2006), reinforcing the need for strong relationships across the sector. However, this also increases the amount of resource required to effectively maintain a programme with strong environmental outcomes across a large area.

Third, the need for strong relationships and holistic farm planning also requires a unique set of skills among land managers - encompassing emotional intelligence and strong practical experience in farm systems and the place of soil-conservation works within them.

Fourth, the targeting of practical soil conservation activity has not encompassed cost-effectiveness as a goal, despite the presence of analytical frameworks that can investigate these issues in rich detail (Doole et al., 2015; Fernandez and Daigneault, 2017). This raises the possibility that environmental gains could be gained at less cost, if planners drew on information from economic frameworks (Pannell, 2008). Nonetheless, the static optimisation models used broadly throughout New Zealand for the economic assessment of water quality improvement contain many strong assumptions that abstract from the problem facing regulators (see previous Section). Thus, if economics, and the social sciences more broadly, are to better inform soil conservation policy, it is imperative that they better reflect the problem facing the regulatory body.

Fifth, non-regulatory approaches can have negative impacts at the aggregate level. Maintaining support for an industry can reduce incentives for innovation, reduce flexibility, increase land value, and make farmers less competitive at the international level. Further, they can impair aggregate efficiency by distorting the entry and exit decisions of firms (Devadoss et al., 2016).

Last, some of these programmes rely on public investment to help motivate the implementation of mitigation strategies (MPI, 2015). These funds are important to motivate landholders, who generally base their decision to be involved on the cost of environmental works and its impact on the economic viability of their farm (AgResearch, 2016). However, it reflects a need for sustained funding if implementation is to continue to grow (Doole et al., 2014), though the building of momentum among farmers may help to provide inertia to future progress if new social norms are developed. It also has motivated some to question why public funds should be used to subsidise private actors that profit from management actions that have negative social outcomes (Foote et al., 2015). This aligns with issues related to rights related to discharges to water, which is now reasonably clear following more than 25 years of case law associated with the *Resource Management Act 1991*.

The implementation of non-regulatory programmes has its challenges when broad-scale soil conservation is the primary goal of policy effort. Nonetheless, it recognises that achieving increased uptake of mitigation practices at scale through regulatory efforts is difficult because: it is antagonistic towards the very population whose behaviour you are trying to change; not everyone requires rules and/or enforcement to exhibit behaviour change; the cost to communities can be high, especially where farming is the backbone of the regional economy; legal arguments can increase conflict and stall implementation; and regulation can lead to a loss of political support, at both regional and national levels. For these reasons, non regulatory methods remain arguably the most successful strategy used to motivate soil conservation on New Zealand farms across the last century.

Part 4

Economic assessment of sediment mitigation

Introduction

This section is a review of economic evaluation studies that include sediment mitigation, particularly at the farm-level or the catchment-level in New Zealand. The main focus of this section is the last decade of work. This has the most relevance for managers, given that this body of work has arisen in response to the emerging policy regime and trends in economic research. The aim is to outline the economic modelling approaches used and to compare and contrast them. Key lessons, themes and gaps in the recent literature will be identified, alongside areas for improvement to incorporate into future work.

First, some background is provided on key terms and concepts used throughout this section. In general, economic assessments are undertaken to better understand the costs and benefits of sediment mitigation. Some studies focus on just the costs, some on just the benefits, and some on both. Much of the work considered by this review sit within the costs side. Studies of this type can quantify the costs of mitigation or analyse the costs of different policies to meet a given target.

The value of costs and benefits are associated with two types of goods and services - market and non-market. Goods and services that are traded within a market have a market value and therefore have a monetary price from that market. In terms of sediment mitigation, the costs of mitigation actions have market values as they have a market cost, for example when a farmer invests in riparian fencing. Non-market goods and services have no market in which they are currently traded (Tietenberg and Lewis, 2016). For example, flood protection from healthy soils is a non-market good (Dominati and Mackay, 2013). For non-market goods, a monetary value can be estimated, but the value of non-market goods can also be expressed in non-monetary terms. Generally, “non-market valuation” is defined as the valuation of a good or service in economic (monetary) terms, given the ubiquity of the practice (Tietenberg and Lewis, 2016). Values are considered in monetary terms in this section unless otherwise stated.

Overall, the costs of sediment mitigation tend to have market values, and the benefits of sediment mitigation tend to be non-market. Examples of non-market benefits from erosion and sediment mitigation include ecosystem services, such as maintaining the productive capacity of soil, increasing flood mitigation (Dominati and Mackay, 2013) and improving the freshwater habitat for native fish (Davies-Colley et al., 2015). Without policy intervention, the cost of sediment mitigation tends to fall on land holders, whereas benefits are more widely distributed among the community. Therefore, benefits are mostly external to land holders.

Approaches to assessing sediment mitigation

A number of studies assess the costs of mitigation scenarios, particularly at a catchment or regional level. The most common type is static optimisation modelling. Some studies have also included an assessment of the benefits of mitigation as well. Generally, methods to mitigate sediment pollution also have an impact on other diffuse contaminants (e.g. nitrogen [N], phosphorus [P] and E. coli). Thus, the economic assessments of water quality improvements reviewed here, which include sediment, would often result in wider benefits.

Static optimisation

The majority of recent studies in New Zealand that conduct an economic assessment of sediment mitigation are spatially explicit, static optimisation models at a catchment level. These models, described in more detail below, are essentially an economic model of farms within the catchment integrated with an environmental model. The approach is to model scenarios and determine the optimal set of mitigation actions and land use change, to achieve specific objectives under each scenario. Scenarios may be defined as a level of water quality, or outcome based, with the required level of mitigation of sediment losses determined by the environmental model and optimised by the economic model. Alternatively, scenarios can be practice based, where the environmental model determines the water quality improvement associated with each scenario, and the economic model determines the cost (Daigneault and Samarasinghe, 2015). No dynamics are considered within the models - they shift from a baseline allocation of land use and production to a new equilibrium under a given scenario and generally modelling production and environmental impacts in annual values.

The two optimisation models that have considered sediment are the New Zealand Forest and Agriculture Regional Model (NZFARM) (Daigneault et al., 2017a; Daigneault and Morgan, 2016; Daigneault and Samarasinghe, 2015; Fernandez and Daigneault, 2017) and the model based on the Land Allocation Management (LAM) framework developed for the Waikato and Waipa catchments as part of the Wai Ora Healthy Rivers Policy process (Doole, 2015a, 2015b, 2016, Doole et al., 2015a, 2015b, 2016). These models are very similar, only differing in the way that the model structure is manipulated so that output better resembles those land use patterns observed in reality. Predictions of current land use arising from a model typically differ from actual land use, given the broad determinants of land use change that a model cannot represent in explicit terms. Thus, calibration methods are used by modellers to improve the way that model output considers the impact of these omitted components. The NZFARM model fixes land use to baseline (Daigneault et al., 2017a; Daigneault and Morgan, 2016; Daigneault and Samarasinghe, 2015) or manipulates the profit of alternative land uses to ensure that the current pattern is observed (Fernandez and Daigneault, 2017). With the LAM model, land use is also fixed (Doole, 2015a) or assumed to lie within the bounds observed historically (Doole et al., 2015a, 2015b, 2016).

The NZFARM model is a comparative static, partial equilibrium, non-linear, mathematical programming model. In order to estimate sediment loads, the relevant variables implied by land use type and crop management practices, along with soil type and climatic factors, are fed into soil erosion models. These input variables are determined in part by the catchment, and partly by the outputs of NZFARM. Daigneault et al. (2017a), Daigneault and Morgan (2016) and Daigneault and Samarasinghe (2015) use SedNetNZ to model sediment losses. In the case of Fernandez and Daigneault (2017), the Universal Soil Loss Equation (USLE) is used to estimate surface erosion, and the New Zealand Empirical Erosion Model (NZEEM) predicts likely level of mass movement erosion. Practice based scenarios can be modelled, where the change in sediment load from baseline is estimated for a given bundle of mitigation practices on farms (Daigneault and Morgan, 2016; Daigneault and Samarasinghe, 2015). Alternatively, outcome based scenarios can be modelled, for example, a 20% reduction in sediment loads across the catchment (Daigneault and Morgan, 2016; Daigneault and Samarasinghe, 2015; Fernandez and Daigneault, 2017). In this latter case, sediment loads become an additional constraint on the NZFARM objective function, and the model provides the least cost means of reaching the outcome.

The Wai Ora Healthy Rivers process is focused on the development of policy for the catchments of the Waikato and Waipa rivers. The LAM model constructed for use within this process allows land use change up to a limit, which is based on historically observed land use patterns. Historical land use data from 1972 to 2012 was used to both constrain the extent of land use change possible, and to calibrate the model to implicit factors that determine the costs of land use change (Doole et al., 2016). This model was integrated with a version of the Catchment Land Use for Environmental Sustainability (CLUES) framework, to deal with the attenuation of sediment and how sediment loads determine visual clarity. Within the Wai Ora Healthy Rivers Programme, the LAM framework was used to estimate the impacts of different actions on sediment loss and visual clarity in waterways (Doole et al., 2016). Also, it was employed to assess which mitigations for sediment loss were required to achieve community goals for fresh water across a catchment (Doole et al., 2015a, 2015b, 2016).

Input-output

Another model type that has been used to economically model sediment mitigation is input-output (Market Economics, 2016; McDonald and Doole, 2016; NIWA, 2010). Input-output models provide a measure of net regional economic costs of a given mitigation scenario. These net economic costs are related to the gross regional or gross domestic product and therefore are related to the financial income within an area, and the studies reviewed here do not include any changes in environmental quality within the model. It is possible to account for environmental changes too, and such models have been extensively applied internationally, particularly for climate change (Wiedmann, 2009), but also water use and pollution (Okadera et al., 2006).

The basis of input-output models is to represent the regional economy through the size of, and relationship between, each sector within the economy. The relationship between each sector is estimated using data on the flows of inputs and outputs (sales and purchases) from one sector to another, which also provides an estimate of the value added by each sector. Any expected change in output of the farming sector can be used as an input to the model, and the effect of the mitigation scenario on each sector, and thus the regional economy as a whole, is calculated. Hence, the change in output in the farming sector due to sediment mitigation must be estimated somehow to feed into the model. The results of a static optimisation model can thus be used as input data, as was done for the Waikato and Waipa catchments (Market Economics, 2016; McDonald and Doole, 2016). Thus, input-output modelling is more of a complement rather than a substitute for optimisation modelling.

Input-output modelling on sediment mitigation has been undertaken with sources of input data other than optimisation models. Using an earlier version of the input-output model for Waikato, NIWA (2010) construct several mitigation scenarios to meet water quality goals using environmental models, data on sediment (and other pollution) mitigation options and expert knowledge. Their estimated aggregate cost of the actions for each sector is used as the input data for the input-output model.

Benefit-cost analysis

A benefit-cost analysis allows the costs of mitigation to be weighed against the benefits. When considering net benefits to society as a whole, both private and social benefits and costs are included. As covered in the introduction above, benefit-cost is the general framework within which economic assessments take place. Therefore, static optimisation and input-output modelling can fit within a benefit-cost approach. However, benefit-cost analysis is considered separately here given the research covered above largely only considers the cost side. This section describes benefit-cost analysis as it has been used for sediment mitigation in New Zealand, as well as studies that focus on the benefits (or avoided costs) of sediment mitigation.

As with static optimisation and input-output methods covered above, a benefit-cost analysis requires modelling of mitigation scenarios relative to a baseline. The costs of mitigation need to be quantified and summed, as do the benefits, appropriately discounting future values to calculate the net present value (NPV). To conduct this analysis, ideally all values should be monetised (Tietenberg and Lewis, 2016). If values cannot or are not monetised, a benefit-cost framework can still be used with as much information as is available. Non-monetised benefits can be qualitatively described, which can help determine whether their inclusion would change the overall conclusion of the benefit-cost analysis or not (Treasury, 2015; US Environmental Protection Agency, 2014). Even without monetary value available for all benefits, it is important to consider all benefits and costs involved with a decision as far as is practicable (see *Federated Farmers v MacKenzie District Council,* 2017). Private benefits at a farm-level include the market value of the production from the soil not lost to erosion (Dominati and Mackay, 2013; Parminter et al., 2001). As a significant proportion of the benefits are environmental, these social benefits must be estimated through some method of non-market valuation. A benefit-cost methodology is recommended by Jones et al. (2008) to assess the net economic benefits of erosion control in New Zealand.

Presently, studies that include sediment mitigation and use the benefit-cost approach generally provide a benefit-cost analysis for a single type of mitigation, rather than a combination. Examples include riparian planting (Daigneault et al., 2017b), tree planting around pasture (Dominati and Mackay, 2013; Parminter et al., 2001) and land use change to forestry (Barry et al., 2014; Dymond et al., 2012). The modelling of these mitigation scenarios does not constitute an overall optimisation of NPV for sediment mitigation. While a benefit-cost analysis could theoretically be undertaken to find an optimal level of mitigation, in practice environmental benefit-cost studies usually generate a point estimate for a benefit-cost ratio for one intervention (or policy) relative to the status quo, and usually without taking into account, all potential benefits. Sometimes several options may be ranked. Usually a functional relationship between benefits and costs from the level of mitigation is not estimated, as would be required for full optimisation. The distribution of costs and benefits is not necessarily estimated either, but is useful information for policymakers if it can be estimated (Atkinson and Mourato, 2015).

On the benefit side, some form of non-market valuation is required to inform the estimation of NPV. There are limited data for non-market values in the freshwater space in New Zealand, particularly for recreational and Māori values (Marsh and Mkwara, 2013). Additionally, the direct calculation of dollar benefits for reductions in sediment losses (such as increases in water clarity) may be difficult or impossible if they are not included as specific attributes in the study. Several studies in their review do evaluate water clarity for specific New Zealand catchments. SCION's (2012) study of sediment in the Waiapu Catchment is an example of a wider consideration of social and cultural values, which are not monetised. They use a bi-cultural aspirational framework, working with the local iwi to define their aspirations for the Waiapu. They then assess the extent to which these aspirations have been realised historically, and under mitigation scenarios. Iwi can provide their assessment of the extent to which aspirations are being met, with key indicators including levels of and trends in the availability of mahinga kai (freshwater food and resources) (Warmenhoven et al., 2014).

Two recent studies help fill some of the gaps for valuation of sediment mitigation. First, Phillips (2014) conducts a non-market valuation study to quantify, in monetary values, the benefits of water quality improvements in the Waikato River Catchment. Two methods are used - revealed preference, and stated preference - and the results are combined. The revealed preference method is aimed at understanding real world cultural and recreational use values of the Waikato River. For this method, a survey was undertaken to estimate the travel cost borne by those who visit freshwater bodies within the Waikato Catchment. The travel cost method allows the estimation of willingness to pay to utilise sites along the river, given the level of clarity and other quality measures of those sites. The stated preference method is also undertaken through a survey in which a choice task is presented to participants, who indicate their willingness to pay for certain attributes such as level of clarity, based on the choices they make and the hypothetical cost of those choices. The data collected from the study allow the monetary quantification of benefits of water clarity along the Waikato River, alongside other water quality measures, such as risk of infection and ecosystem health. Second, Tait et al. (2016) conduct a national non-market valuation study, using a stated preferences discrete choice experiment (Carson and Louviere, 2011), with a representative survey sample, regarding the value of water quality, including clarity, health risk and ecological quality.

In order to utilise the data from such non-market valuation studies, benefits from these studies can be used to estimate benefits in other catchments using a method called benefit transfer (Marsh and Mkwara, 2013). It is important to note the limitations of such a method, as there is limited ability to assume benefit transfer for sediment as timeframe and geographical context are critical. Sediment loads and their impacts on receiving environments vary greatly between catchments due to factors such as land use, soil type, slope and climate. Marsh and Mkwara (2013) provide a detailed review of criteria for the successful application of benefit transfer for those requiring more information, noting the limited scope for benefit transfer given current levels of data.

Krausse et al. (2001) was the first study to estimate the national economic cost of soil erosion and sedimentation, and their numbers are still used in recent studies. They estimated the national cost of sediment to be $27.4 million per annum in 1998 dollars, for categories where data was available, such as insured losses from increased flood severity and reduced water quality for consumption. The overall costs of soil erosion and sediment were estimated to be $126.7 million per annum, where data was available.

Daigneault et al.'s (2017b) benefit-cost analysis of nationwide riparian planting includes estimated non-market benefits from reduced soil erosion, using estimates of costs per tonne of sediment from Forgie and McDonald (2013), based in turn on Krausse et al. (2001). Daigneault et al. (2017b) use a range of sediment cost scenarios, ranging from $1.50 to $6.00 per tonne. The market costs of mitigation are estimated using the NZFARM model, demonstrating how static optimisation modelling can be undertaken as part of a benefit-cost analysis (Daigneault et al., 2017b).

An overarching conceptual framework that is commonly used to assess environmental goods and services is the natural capital and ecosystem services approach. The value of natural capital provides an assessment of the stocks of environmental assets at a given point in time, and value of ecosystem services a quantification of the flow of value from such stocks. The approach identifies non-market goods and services provided by ecosystems, which allows the assessment of their value to society (see Dominati et al. 2010). Several recent studies have used the ecosystem services approach to help conduct a benefit-cost analysis of sediment mitigations.

Dymond et al. (2012) use the ecosystem services framework to quantify the benefits of establishing Pinus radiata forests around New Zealand. For the benefits of avoided soil erosion, they use Krausse et al.'s (2001) estimated national costs of erosion. Extrapolating Krausse et al.'s values to 2010 gives Dymond et al. a value of $1 of benefit for each tonne of soil erosion avoided. Barry et al. (2014) conduct a similar study, estimating the net private and social benefits of afforestation across the New Zealand landscape. They estimate the benefits of avoided sedimentation using avoided expenditure costs, estimated through discussions with councils. Avoided expenditure is a method for estimating the value of ecosystem services by estimating the cost of providing a similar benefit through human-made capital investment. This method does not necessarily represent the value to society of the ecosystem service as it is a measure of cost, rather than benefit (see more discussion on this point below) (Bennett, 2011). Nevertheless, Barry et al. (2014) use the method and estimate the value of each tonne of avoided sediment losses at $0.90 for avoided flood damage, plus $5.60 for avoided water treatment costs for consumption.

Farm scale modelling can help provide more detailed information about one or more case studies and the effects of sediment mitigation on their profitability. Parminter et al. (2001) conduct a benefit-cost analysis of the erosion control measure of planting poplars on a typical hectare of a steep North Island hill country farm. They only consider on-farm effects, so their analysis is applicable to a farm manager in absence of any policy and does not consider non-market benefits of sediment mitigation. Costs include planting costs, reduction of productive land and reduced grass growth from shading. Benefits include increased production from lack of erosion and reduction in erosion repair costs. In general, only the most favourable of assumptions leads to a net present benefit. Hence, they conclude promotion of non-market benefits and perhaps even financial incentives are required to stimulate poplar planting. Site measurement studies and the latest environmental models can feed into such studies (e.g. Guevara-Escobar et al., 2007; see also Phillips et al., 2012). Dominati and Mackay (2013) use an ecosystem services approach to conduct a benefit-cost analysis of soil conservation from space planting poplars on erosion prone pasture in the Hawkes Bay. They find net private costs to the system, unless the trees are harvested, but significant net benefits are estimated when accounting for the benefits of ecosystem services from soil conservation.

Summary of findings from recent modelling

Overall, the recent modelling that has been undertaken has shown that there are reasonable reductions in sediment loss available at relatively low to moderate costs. An extreme example is in Kaipara, where 15% mitigation of sediment flows is modelled to cost only 0.2% of catchment revenue (Daigneault et al., 2017a). On farm mitigation bundles in the Ruamāhanga Catchment can achieve reductions up to 25% for sediment, along with decreases in other diffuse freshwater pollutants, at a cost of 14% net revenue (Daigneault and Morgan, 2016). Once the lower cost mitigation options are exhausted, however, deeper reductions in sediment are significantly costlier. In the Kaipara and Ruamāhanga catchments, once land use change becomes necessary to achieve deeper cuts in diffuse pollution, economic costs become significantly higher (Daigneault et al., 2017a; Daigneault and Morgan, 2016). Similar results are found in the Waikato-Waipa Catchment. Catchment level profitability loss is 25 to 51% for scenarios with moderate to substantial improvement in water quality (Doole et al., 2015a).

The importance of different farm types and locations for mitigation costs is observed in the Waikato by Fernandez and Daigneault (2017). A 50% reduction of mass-movement erosion is significantly less costly than the same target for surface erosion (3% versus 19% of net aggregate annual revenue). Mitigation costs differ due to surface erosion being concentrated on dairy farms with already high levels of mitigation adoption in the baseline, whereas mass erosion is concentrated on sheep and beef farms with low baseline mitigation. Therefore, there are more low cost mitigation options available to sheep and beef farms. Mitigation costs and options also vary spatially, with a number of areas in the catchment having a high degree of cost effective mitigation actions available. Efficiency from a catchment perspective may lead to heterogeneous costs being borne by individual farms in the region, unless there are policies to address the spread of costs.

Daigneault et al. (2017a) and Fernandez and Daigneault (2017) focus on sediment, whereas all of the other recent catchment scale modelling reviewed here consider a range of pollutants alongside sediment. There are several likely reasons for this trend. First, the NPS-FM recognises that water-quality degradation is associated with a broad range of contaminants, and it does not presently include sediment related attributes. Second, the significance of N and P to water quality degradation in New Zealand is now well established (Larned et al., 2016); thus, it is appropriate to consider these important pollutants when broadening the focus to sediment. Third, it is important to study P and sediment together, given that some of the P load in storm water binds to sediment (as does E. coli). Last, mitigation actions often have multiple benefits across the various sources of diffuse freshwater pollution. Indeed, both N and climate change mitigation policies have been shown to have co-benefits for sediment losses (Daigneault et al., 2017c).

Recent work has also highlighted the need to target the primary forms of sediment loss within a catchment. The proportion of the total sediment load arising from sources such as hillslopes and streambanks varies greatly across different environments (Dymond et al., 2014). Thus, it is critical to vary the types of mitigation practice used, depending on the primary sources that exist in a given set of circumstances (White et al., 2016; Green and Daigneault, 2018).

Tait et al.'s (2016) representative survey for non-market valuation of water quality shows the average respondent is willing to pay $4.13 for each 1% increase in the proportion of water bodies nationwide that are rated with a moderate level of clarity. They use these results to estimate the national value of various stock exclusion scenarios, to feed into a national benefit-cost analysis. In this benefit-cost analysis, benefit-cost ratios from 2.3 to 8.1 are estimated, particularly for fencing relating to dairy cattle, but also for the most extreme scenario excluding all cattle and deer from waterways in slopes up to 28 degrees (MfE and MPI, 2016).

Discussion

Strengths and limitations of the approaches to assess sediment mitigation

Optimisation models have been widely applied, demonstrating the versatility and feasibility of the approach. Strengths of static optimisation models include:

1. They model optimisation of mitigation strategies on farms across a catchment scale, thus providing a measure of economic impact that allows land managers to respond to the policy by optimising their farm practices or changing land use.
2. They are set up to include available data on mitigation options and usually also the potential for land use change. There are sufficient data nationwide to make the modelling exercise viable across New Zealand (e.g. Daigneault and Elliot, 2017; Daigneault et al., 2017b; Daigneault et al., 2017c).
3. They are designed to interact with an environmental model and are flexible enough to utilise whatever environmental model is most suitable for sediment in each catchment.
4. There are well established calibration methods.
5. The models can assess the net cost of mitigations potentially required under policy scenarios that are a bundle of mitigation actions (practice based) or aimed at meeting an environment target (outcome based). Therefore, the models can provide regulatory decision makers information on the magnitude and incidence of costs of and constraints on policy.

Weaknesses of the method and how it has been applied include:

1 Given the complexity of the models and precision of figures they provide, it is easy to overstate the certainty and reliability of the results. Simulating the future is inherently subject to large uncertainties. The complexity of the models also means the results can be challenged.[3]

2 Behaviours that may be empirically true, but do not fit within the profit maximisation framework, are not represented, with the exception of elasticities between land uses and any non-profit maximising on-farm practices that are present in the baseline.

3 No dynamics are represented in the static optimisation models used for sediment mitigation. Therefore, adjustment costs are poorly represented or not represented at all. Dynamic factors that land managers take into account when deciding on mitigation responses to adopt, such as uncertainty and option value, are also unaccounted for.

4 Generally, knife-edge adoption is simulated, whereby a management option is employed if it is, on average, more profitable than alternative options. Given that there will be substantial sub-optimisation by landholders in practice, static optimisation models will provide an optimistic view of the effects of policy.

5 By calibrating the model to a single baseline year, it is assumed that the baseline year is a static equilibrium, rather than a snapshot of a dynamic, constantly changing system.

6 The behaviour of the models away from equilibrium are seldom, if ever, validated against independent data (Doole and Marsh, 2014a, 2014b; Doole and Pannell, 2013). This generally reflects a scarcity of suitable data, but also fails to recognise the strong impact that a calibration method has on model behaviour outside of the baseline scenario.

7 The coarse representation of farms across catchment does not take into account heterogeneity present amongst farmers, such as life stage (for example proximity to retirement) and human capital. Hence, the models do not take into account likelihood of adoption of mitigation practices and the distribution of costs and benefits.

8 Land values and debt are not present in the models, even though these are major features of farm businesses.

9 The rational, profit maximising behaviour built into the models will always mean market or price based policies are least cost, whether or not that is empirically true.

The NZFARM and LAM frameworks are the two static optimisation models that have been applied to sediment mitigation in New Zealand. In studies that include sediment, the NZFARM model has been applied a number of times in specific catchments and nationwide in New Zealand, whereas the LAM framework has just been applied to the Waikato-Waipa Catchment.

Input output models provide a framework within which to consider the regional and national implications of environmental policy. The strengths of input output models include:

1 The net regional economic cost (excluding any environmental changes) is calculated, rather than just for the sector directly affected. This gives a picture of the extent to which the direct costs of whatever policy is implemented might flow through to the wider economy.

2 Input output models can complement optimisation models or use other data sources of the direct costs of a mitigation scenario.

[3] *See for example the conclusion of the Independent Hearing Commissioners in Environment Court Hearings for Lake Rotorua Plan Change 10 that: "The criticism levelled at the economic analysis of the Regional Council witnesses was of a general nature and reflected, to some extent, the reality of future economic analysis which has so many variables. The extent of such variables leaves open room for criticism. Such is the lot of economics." (Bay of Plenty Regional Council, 2017, para. 358).*

3 There are not major requirements for data to set up the model and thus they are relatively inexpensive. Indeed, regional councils may already have their own models, which can be adapted to model the sediment mitigation scenarios (McDonald and Doole, 2016).

Weaknesses of input output models include:

1 The models are relatively fixed and therefore do not allow for much adaptation of the regional economy to the policy, thus they may overstate the economic costs.

2 As with static optimisation models, there are no dynamics and thus they also assume the baseline year is the equilibrium of the system.

Optimisation and input output modelling provide estimates of net direct financial costs of mitigation. They can be used as part of a benefit-cost analysis; for example, as done by Daigneault et al. (2017b), who integrated the NZFARM optimisation model with monetary estimates of the benefits arising from a riparian restoration regime.

A benefit-cost analysis is a tool to assess the net benefit of a scenario, ideally including non-market values, and therefore can theoretically identify when a policy with higher gross financial costs is preferred on the basis of superior net benefits. However, many non-market valuation techniques require survey data within the context of freshwater quality, whether through revealed preference methods (for example, travel cost surveys) or stated preference methods (discrete choice experiment surveys). Collection of survey data can be expensive and time consuming. As mentioned previously, benefit transfer can be used to estimate values for catchments outside of the original non-market valuation studies, as long as certain criteria are met (Marsh and Mkwara, 2013).

Non-market valuation methods that do not require a survey include the avoided expenditures method (e.g. Barry et al., 2014). This method requires the estimation of the value of replicating the particular ecosystem service, such as water purification, through human made infrastructure, but has theoretical shortcomings. It estimates the marginal costs rather than the marginal benefits of ecosystem services, which will only be the same when society is at the optimum level of the ecosystem service. Furthermore, human made substitutes are often of lower quality compared with those provided by natural capital (Bennett, 2011).

Non-market valuation studies require a high level of expertise, given the potential for survey design to affect the results (Treasury, 2015). Stated preference methods have been refined over recent decades and as such they have been widely applied to benefit-cost analyses, particularly in the US as part of the regulatory regime (Arrow et al., 1993; Sunstein, 2005; Tietenberg and Lewis, 2016; U.S. Environmental Protection Agency, 2014). These methods are therefore widely accepted, but it is important to be aware that stated preference methods continue to be subject to criticism by some (e.g. Hausman, 2012; McFadden and Train, 2017). Others still continue to debate the appropriateness of monetising environmental goods and services more generally (Tietenberg and Lewis, 2016; Bertram and Terry, 2013). Therefore, there could be nervousness over using non-market values in RMA processes that might be subject to court challenges (e.g. Bertram and Terry, 2013). However, it is important to take into account all costs and benefits, whether valued in monetary terms or not. It is not good practice to ignore benefits that are difficult to monetise (Treasury, 2015; U.S. Environmental Protection Agency, 2014) and a reasonable attempt should be made to identify non-market values and measure them in some form (see Federated Farmers v Mackenzie District Council, 2017 for a ruling on the legal requirement to consider all costs and benefits under the RMA).

Overall, the importance of including all values in benefit-cost analysis is seen as a strong reason for undertaking non-market valuation (and the monetisation of these values), and best practice can be followed for estimating these values (Tietenberg and Lewis, 2016; Treasury, 2015). Indeed, it is also important to ensure valuations are up to date, given the recent use of Krausse et al. (2001) to value the cost of erosion and sedimentation, even though their numbers are based on the best available data at the time of their study and thus only consider selected costs.

Methodological gaps and areas for improvement

The modelling of sediment policy to date has been undertaken within a wider context of evidence based or informed policy (Bowen and Zwi, 2005; Gluckman, 2013; Gluckman, 2017). The generation of appropriate evidence has driven demand for economic modelling of policy scenarios that is feasible and portable to catchments around New Zealand. Hence, a significant amount of static optimisation modelling has been undertaken around New Zealand on sediment policy, and diffuse pollution of fresh water more generally. It is important to reflect on how to best direct future research so that it adds value, rather than just repeating similar exercises and providing little new insight.

Purpose of modelling and the wider context

The purpose of conducting this type of economic modelling is to improve and ultimately optimise society's welfare. The modelling supports decision makers manage resource use by estimating the costs and benefits involved. This management can involve understanding what levels of resource use are appropriate and what interventions and regulations might best achieve society's goals.

In terms of the wider context, freshwater quality in New Zealand is a wicked problem, which is intractable from the point of view of public policy. Complexity, unknowability and subjectivity are key features of wicked problems, which, in the case of freshwater quality in New Zealand, arises from the dynamic interaction of social and biophysical systems (Gluckman, 2017; Rittel and Webber, 1973). On the biophysical side, freshwater systems and their links with land use and landscapes are consistent with the behaviour of a complex, adaptive system (CAS) (Zellner and Campbell, 2015). CASs possess complex, dynamic, and emergent behaviour based on the nonlinear, spatial, and temporal interaction of a large number of individual elements (Wulun, 2007). There is complexity in the social system's interactions with the biophysical system too. Society places a high value on freshwater systems, for a range of cultural, recreational and economic reasons, including as a means of absorbing non-point source pollution from farms (Gluckman, 2017). Competing values amongst stakeholders means there is little consensus on defining the problem or solution, also known as 'low value consensus' (Basco-Carrera et al., 2017; Head and Alford, 2015; Pielke, 2017).

Economic modelling of sediment mitigation or freshwater policy is consistent with a philosophy of evidence based or evidence informed policy making (Gluckman, 2013). This approach to policy making is an important philosophy within the New Zealand public sector. It also underpins section 32 of the RMA, which requires an evaluation report for new regional policies, with a focus on economic benefits and costs, alongside environmental, social and cultural effects. The NPS-FM states objectives and policies for freshwater management under the RMA (MfE, 2017b). Under the NPS-FM, regional councils are required to establish freshwater objectives and set limits on resource use to achieve them. Additionally, many Treaty of Waitangi settlements include a role for iwi and hapū within an integrated, catchment based management regime for fresh water. Regional councils may have further requirements due to Treaty Settlements. In the case of the Waikato and Waipa rivers, the Waikato River Authority's Vision and Strategy prevails when there are inconsistencies between the NPS-FM and Treaty Settlement legislation (MfE, 2017a). Iwi rights and interests therefore may add an additional layer of considerations to analysis requirements for sediment policy.

The NPS-FM provides a framework for enhanced involvement with community through providing opportunities for collaborative determination of local values, standards, and management interventions (MfE, 2017a). This trend towards increased collaboration is motivated by a desire to understand and recognise a multiplicity of values and views, and make progress towards a consensus on actions to take (Alford and Head, 2017; Sarewitz, 2004). Clearly, there is a role for evidence to inform such collaborative processes, which is important to keep in mind when producing such evidence. On the other hand, the knowledge and viewpoints that arise from collaborative processes can equally have a role in helping shape evidence being produced (Connolly, 2017).

Potential improvements to current models

It is important to be mindful that these are complex economic and biological systems that are being modelled in any assessment of sediment mitigation policies. As such, the dynamics of the systems and their interactions are difficult to understand and predict. Any modelling exercise will be wrong, therefore there should not be too much weight placed on the exact results of a static optimisation or input-output model. Good models do provide useful insights, and therefore the focus should be on undertaking the most useful modelling possible and spending time on understanding what such a modelling exercise is showing. Furthermore, while the rest of this section considers improvements, extensions and new modelling approaches, it is important to remember that parsimony is an important attribute of good models as it helps ensure the important mechanisms are clear (Box, 1976).

There are two key features that are missing from static optimisation models that may be important to include, to provide a more complete economic assessment of the costs of policy. These features are related and can be summarised as dynamics of adoption and heterogeneity of farm managers.

In terms of how heterogeneity of farm managers is currently modelled, representative farms are distributed across space (Doole, 2010; Doole and Pannell, 2012). This approach is pragmatic, reducing model complexity and data needs. However, farms are diverse, and the impact of agri-environmental policy can depend on farm equity, farm intensity, landholder objectives, life stage of the business, management ability, natural resources, risk aversion, and social networks (Doole, 2010, 2012; Nuthall, 2001; O'Sullivan et al., 2016; Pannell et al., 2006; Pannell et al., 2000). Enhanced modelling of heterogeneity could provide insight into the distribution of costs and benefits of policy, shedding light on impacts and any need to consider adjustment or augmentation of such policy. Some of these sources of heterogeneity may also impact the likelihood of adoption of mitigation actions.

Knife-edge adoption is the standard assumption in static optimisation models, whereby a management option is employed if it is, on average, more profitable than alternative options. In reality, diffusion of technology and practices across a population is dynamic and does not depend solely on average profitability. Factors that affect diffusion include the benefits of a new practice over an existing practice (Rogers, 2003), the trialability of the innovation (Pannell et al., 2006; Pannell et al., 2014), the depreciation rate of the mitigation capital (for example the reduction in the rate of sediment removals from a wetland over time) (Doole et al., 2018), social networks (Iyer et al., 2015), uncertain, volatile trends in the decision making environment (Lee et al., 2016), and the interactions between multiple technologies that may be involved with many mitigation options (Feder and Umali, 1993). Additionally, insights from psychology and behavioural economics show how humans have limited cognitive abilities and use certain heuristics to make decisions. Such behaviours may prove inconsistent with profit maximisation as modelled by static optimisation models. Furthermore, uncertainty complicates a decision maker's assessment of relative options (Kahneman, 2003; Pindyck, 2007). The limitation of static optimisation models on all these counts is a motivation for enriching the models with respect to heterogeneity and dynamics, or developing new approaches to understand these important elements for policy.

More empirical work could be undertaken to better understand the process of and barriers to adoption of mitigation practices to understand how policies might maximise their cost effectiveness by leveraging more than just price or regulation to increase adoption rates (Doole et al., 2018). Fernandez (2017) provides some data in this area. He conducts one study that looks at adoption of erosion management practices, using data from the Survey of Rural Decision Makers, and finds adoption is correlated with climatic variables, sociodemographic variables of farmers and adoption of other erosion or nutrient management practices. Kaine and Wright (2017) look at a range of water quality practices and their relative advantage for dairy farmers, by conducting a survey within the Waikato and Waipa regions. They have a range of findings related to likely uptake of practices and what policy options could increase adoption rates, but find little correlation between likely adoption and farmer demographics.

Given the setup of optimisation models assumes profit maximising behaviour, they will always favour some form of market or price based policy as the most cost effective. In addition to more realistic adoption behaviour, frictions in market based policies could be added to enhance the models, as discussed above. The costs of setting up a market or price based policy, or imperfection in the market, may mean a simpler minimum standard approach is more cost effective, particularly when it is cost effective for the minimum standard to be widely or universally adopted (Doole and Paragahawewa, 2012).

Clearly validation is another important exercise to undertake (Daigneault et al., 2014; Doole and Marsh, 2014b, 2014a). There is a role for ex post analysis of how well a model predicted the outcomes of a policy, or using current models to predict the response to historical policies, to see how well the models replicate reality. Such an exercise can help with calibration of models. Additionally, the exercise may present areas for improvement in widely applied models that are unexpected. Snelder (2018) correlates the SLUI programme with improvements in water quality in the Manawatu-Whanganui region, but clearly there is more work that could be done in this area.

A benefit-cost approach has its strengths, particularly that it ensures that a more stringent policy does not just seem more expensive, but that the benefits of such a policy is also measured and thus the best policy can be identified. Given this approach has such strengths, perhaps more effort could be expended on understanding benefits from mitigation of sediment flows, to allow the approach to be applied more widely. Further research into benefits could include monetary and non-monetary valuation.

A cost effectiveness approach is an alternative to a benefit-cost analysis, whereby a community or Government, through an appropriate democratic or expert led process, sets the sediment load goals to be met. The modelling exercise then looks at the least cost means of meeting the goals (Tietenberg and Lewis, 2016). This idea has been used for water policy, and for other areas of environmental policy such as climate change. Cost effectiveness analysis avoids having to measure the benefits of environmental policy using monetary value, instead finding the most cost effective means of reaching an agreed goal (Ackerman and Finlayson, 2006; Balana et al., 2011). Such an approach does require a means of agreeing on a goal, for example a collaborative or other governance process, or a determination by appropriate experts. In the case of a wicked problem, agreeing to a goal is non-trivial. Estimating the economic cost of meeting different goals is an important part of such a process (e.g. Doole et al., 2015a). Once a goal is agreed, optimisation modelling, or economic modelling more generally, can help compare various policies in terms of meeting the goals. Hence, static optimisation modelling is consistent with a cost effectiveness approach, a fact that could be important to recognise to help contextualise such modelling.

Potential for new modelling approaches

System dynamics models provide one approach for modelling sediment policy that is a significant departure from what has been covered so far in this section. They are a means of describing a complex system in terms of its constituent elements and the dynamic relationships between them (Sterman, 2000). These relationships are represented graphically, which improves accessibility for people with lesser technical training. An example is causal loop maps that provide a qualitative description of cause and effect within a system (Senge, 1990). The graphical set up, and ability to describe relationships either qualitatively or quantitatively, means system dynamic models can be developed in stages with a group, through group based model building (Luna-Reyes et al., 2006; Vennix, 1996). Such a process can help with collaborative decision making, by supporting constructive discussion and improving group understanding and dynamics (Luna-Reyes et al., 2006). This has been noted in several applications carried out in New Zealand in the context of collaborative decision making for freshwater resources (Connolly, 2017). Further, the Southland Economic Project has been focused on working collaboratively with stakeholders to apply a quantitative system dynamic model to water quality issues in this region.

Simple numerical modelling for the purposes of supporting informed collaborative processes, where stakeholders may struggle to understand more complex models, is one option. Experts can develop such model where key relationships are represented (Harris and Snelder, 2014). However, such models would generally have to represent even less real world complexity of the types discussed above, and thus the impacts of policy may be even more poorly represented. Hence, such models may play a useful role in terms of working with stakeholders, particularly through collaborative processes, but more complex economic modelling is still required.

Part 5
Conclusion

Extensive land clearance for the development of agriculture has dramatically altered the evolution of New Zealand landscapes. A reliance on grazing livestock has historically provided a firm foundation for the economic and social fabric of rural and urban communities. However, it has also increased the loss of sediment, nutrients and faecal microbes to water courses. Further, deforestation and its replacement with exotic grasses has encouraged soil loss, with erosion rates in New Zealand among the worst in the world. Development of drainage and river control structures to enable productive use of land, coupled with the infrastructure modern society requires (settlements, roads and bridges), has restrained the courses of rivers and effectively removed their ability to freely flood and deposit sediment naturally across vast areas of natural flood plains.

New Zealand has a long history of non-regulatory and regulatory programmes aimed at reducing soil loss and the effects of sediment on public assets. Indeed, soil loss was the first environmental issue to receive attention through national government policy. The legacy of this legislation is a strong non-regulatory programme of extension and subsidisation for on-ground works, within regional councils. The NPS-FM charges regional authorities to extend and reinvigorate their conversations around improving freshwater quality. Nonetheless, in recent years, sediment has notably received lesser focus, relative to nutrients and faecal microbes, given limited data around the effects that different levels of sedimentation have on use and non-use values.

There is increasing national attention on sediment, as its significance in New Zealand is once again being recognised. The national water quality conversation has focused on nutrient loss from agriculture across the last decade, particularly nitrogen. More recently, there has been an increasing focus on faecal microbes. This broader discussion is helpful to ensure that the manifold effects of agriculture, especially that involving grazing livestock, are considered in environmental policy. Nonetheless, sediment remains a major pollutant of New Zealand waterways, arguably being more important than nitrogen when considering the impacts of agricultural contaminants on the values that water provides to communities.

The sediment problem reflects many of the key characteristics of the policy issues experienced with other contaminants of water arising from agricultural activity. These focus around the existence of many mitigation strategies for the contaminant at hand, but the insufficient adoption of them across our agricultural landscapes due to the low relative advantage and trialability of the technologies. This framing of the problem suggests a number of key lessons that need to be considered in the management of sediment.

First, view the value proposition offered by sediment mitigations in a broader context than just average profit. This ideally needs to be holistic and based around the advantage that each technology offers a farming system and the social benefits of reduction of sediment loads in a given catchment.

Second, design non-regulatory and regulatory interventions around this broader interpretation of the value proposition of sediment mitigations.

Third, recognise that agricultural sectors in New Zealand are making a significant effort and progress to provide (non-regulatory) environmental management system frameworks for their respective memberships with the provision of farm plan templates and staff dedicated to their roll out and uptake.

Fourth, leverage off the strong lessons for sediment management that New Zealand has gained from our experience with soil conservation across the last 70 years. This includes the importance of personal relationships between environmental authorities and producers, farm specific tailoring, and the significance of farm system context. These lessons have much to offer regional councils as they work to introduce regulation emerging from the NPS-FM.

Fifth, economic and social assessments should be structured around the nexus between mitigation adoption and water quality, thinking broadly about how to deal proactively with farm heterogeneity.

Sixth, put more work in validation of the models to identify weaknesses and improve their performance.

Seventh, there is an argument for increased research into measuring or better understanding non-market benefits of sediment policy.

Eighth, consider how best to inform and be informed by collaborative processes.

Last, be mindful that the models we are applying are trying to make sense of dynamic and complex systems. This general point implies that we need to focus on what can be learnt from the modelling process, rather than the precise numbers the models are providing.

Part 6

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