environment SOUTHLAND

Te Taiao Tonga

Environmental Effects of Activities within the Riparian Zone

Technical Review

prepared by

Ryder Consulting

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Technical Review

prepared by

Ruth Goldsmith, PhD.

Dean Olsen, PhD.

Greg Ryder, PhD.

Ryder Consulting

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Cover photo: Riparian management in the Waituna catchment (supplied by Katrina Robertson, Environment Southland)

Ryder Consulting Limited
PO Box 1023
Dunedin, 9054
New Zealand

Phone: 03 477 2119
Web: www.ryderconsulting.co.nz

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1. Introduction

1.1 What is riparian management?

The riparian zone is the area immediately adjacent to the stream or river channel (Figure 1). It forms the interface between land (the terrestrial system) and water (the aquatic system) and is a transition between truly terrestrial ecosystems and aquatic ecosystems (Figure 1). However, for the purposes of this review, we have used a broad definition of riparian management that includes any activity that is adjacent to a waterway. Such activities include the construction and maintenance of structures (e.g., roads, tracks, bridges, culverts, fords), stock grazing within the riparian zone (and fencing), vegetation management, drainage activities (construction of drains, channel modification) and adjacent land uses. The proximity of these activities to surface water increases their likely impacts on those waterways.

1.2 Factors affected by riparian management

The 'health' of waterways is greatly affected by the management of the riparian zone and riparian management is a key ingredient in the achievement of management objectives for freshwater in Southland (Environment Southland Regional Water Plan 2010).

1.2.1 Water quality

All water and contaminants that flow from the land and into surface waters must pass through the riparian zone, with the exception of some groundwater and artificial drainage flows which bypass the riparian zone completely to enter surface waters directly. Hence, management practices within and immediately adjacent to the riparian zone can affect the effectiveness of riparian processes at filtering, storing and transforming these contaminants, which can affect water quality.

The main water quality variables likely to be affected by activities in Southland within the riparian zone, and their potential effects on freshwater ecosystems, are:

- **Nutrients (nitrogen and phosphorus);**
  Nutrients promote algae (called periphyton when it is attached to the bed) and plant (aquatic weeds) growth in streams, lakes and wetlands. The types of periphyton and plant communities that develop on a stream bed influences

the type of macroinvertebrate[1] community that develops, in terms of both abundance and diversity of species. The importance of this relationship is that macroinvertebrates are a primary food item for most of our freshwater fish species (and a number of river bird species). Their ability to transfer primary production (e.g., algae biomass) into a food source for fish and birds is a fundamental aspect of healthy river ecosystems.

In stony bed rivers and streams that dominate much of the Southland landscape, 'healthy' macroinvertebrate communities are dominated by larvae of insect taxa such as mayflies, caddisflies and stoneflies, which generally require good water quality and a relatively thin periphyton layer to graze upon. In contrast, worms, snails and some dipterans (trueflies like chironomid larvae) are more tolerant of lower water quality, soft sediment materials like silt and mud (see below), and thicker periphyton cover such as filamentous and tufted algae (which are generally less palatable to many mayfly, caddisfly and stonefly taxa). These more tolerant macroinvertebrate taxa are typically less common in the diets of our freshwater fish.

Abundant algae and plant growths also influence water quality by influencing dissolved oxygen and pH levels in the water column. During the day, algae and plants photosynthesise (and respire) and, in doing so, produce oxygen. Conversely, at night, algae only respire and so consume oxygen. Thus, high abundance of algae and plants can result in large daily swings in dissolved oxygen levels, with low levels occurring early in the morning, potentially compromising sensitive fish and macroinvertebrate species. Hence, this potential effect, coupled with a less palatable food source for macroinvertebrates, smothering the habitat of more desirable macroinvertebrates, and an unsightly appearance for recreational users, means that excessive algae and plant growths are generally regarded as something to avoid through management of nutrient inputs and stream shading (see below). It should be acknowledged, however, that such growths can occur on occasions even in relatively unmodified catchments.

[1] Freshwater benthic macroinvertebrates are bottom dwelling invertebrates and include crustaceans (e.g., amphipods), insect larvae (e.g., beetles, caddisflies, mayflies stoneflies and trueflies), snails and worms.

- **Temperature;**

  Water temperatures are principally influenced by the input of heat from solar and atmospheric radiation. Riparian vegetation that intercepts and therefore reduces the amount of radiation reaching the water surface can consequently reduce water temperature. However, the magnitude of this reduction is dependent on the degree and length of channel shading, and on channel size.

  Aquatic biota are known to be strongly influenced by water temperature in terms of their growth, reproduction, and survival. Temperature has been identified as an important factor in determining algae and plant biomass and community structure. Higher temperatures favour high biomass accrual and the dominance of erect, stalked and filamentous algae (often synonymous with nuisance algal growths). In general, algae and plants are much more resilient to high temperatures than macroinvertebrates.

  Water temperature can affect abundance, growth, metabolism, reproduction, and activity levels of aquatic insects. A detailed analysis of 88 New Zealand rivers (Quinn and Hickey 1990) identified water temperature as one of the important variables affecting species distribution. Stoneflies were largely confined to rivers between 13 and 19°C, and mayflies were less common in rivers with maximum temperatures of >21.5°C (Quinn and Hickey 1990). Other common taxa found in South Island rivers have much higher LT_{50}'s[2], ranging from 25.9-34°C and therefore are less important from a temperature management perspective than mayflies.

  Fish are often strongly affected by temperature, with effects of temperature on mortality, growth and reproductive behaviour all described from New Zealand or elsewhere. Fish are highly mobile, and are therefore able to seek out 'thermal refuges' to avoid temperature extremes. However, there will be an energy cost in having to seek refuge in terms of reduced foraging time and increased stress. These factors may contribute to poor fish condition and growth. The effects of water temperature on New Zealand native fish have been summarised by Richardson *et al.* (1994). In general the tolerances for native fish species are much higher than for trout and lethal temperatures are unlikely to ever be achieved in flowing sections of most rivers.

[2] LT_{50} – lethal tolerance at which 50% of the test sample die.

If trout are protected against temperature increases, this will generally result in protection of other freshwater fish species. Lethal effects occur at between 24-30°C for adults and 17.5°C for juveniles, although this is a questionable level and appears overly low given observation of juvenile fish throughout rivers in mid summer.

- **Sediments and water clarity (suspended solids and deposited sediment);**

  Typically, the greatest ecological impacts to lakes and rivers caused by sediment disturbance are due to inputs of the finer fractions such as silts and clays. Coarser fractions (e.g., fine gravels and sands) usually deposit on to the bed a short distance from where they enter. Thus, their impacts are localised, although they can be transported throughout a river system by movement along the river bed during high flows. Silts and clays can remain in suspension in the water column for considerable periods of time, even in lakes. However, even proportions of these very fine fractions can eventually settle out or become "trapped" on the bed.

  Fine sediments can affect freshwater macroinvertebrates and fish by five primary mechanisms: reduction of light penetration (see below), abrasion, adsorbed toxicants, changes in substrate character and reduction in food quality (already noted under nutrients and their effect on algae growth).

  Water clarity relates to the transmission of light through water and is affected by sediment suspension and as such by losses of sediment from the land to water. Water clarity is important because of its influence on river ecology, angling and aesthetic value. The effects of clarity on river ecosystems have been well documented in New Zealand although many of the relationships described are general ones only. Effects of reduced clarity and increased suspended sediments on river ecosystems and recreational use vary depending on the species or activity.

- **Microbial indicators (micro-organisms such as faecal bacteria).**

  Faecal microorganism contamination is most relevant to humans and stock that come into contact with river water, rather than having adverse effects on aquatic biota. New Zealand guidelines have been developed to provide various levels of protection based on allowable concentrations of faecal indicator bacteria (e.g., faecal coliforms and *E. coli*).

The potential effects of the various activities on water quality are reviewed in each respective section.

1.2.2 Habitat quality

Activities in the riparian zone can also affect the quality of habitat in streams through effects on:

- **Riparian vegetation including;**
    - Shading (water temperature, algal growth),
    - Leaf litter (food for invertebrates),
    - Wood (cover for fish, structure habitat),
    - Bank stabilisation (erosion prevention, sediment reduction).

  Riparian vegetation may have a variety of direct and indirect influences on instream habitat. Indirect effects include water temperature reductions through shading (as discussed in Section 1.2.1) and decreases in sediment inputs through bank stabilisation by vegetation. The input of organic matter (e.g., leaf litter and wood) from riparian vegetation can have the direct benefit of providing a food resource for invertebrates that feed on detritus (e.g., crayfish) and increasing habitat complexity through retaining substrate and providing stable cover (Parkyn *et. al.* 2009).

- **Geomorphology (undercut banks, slumping);**

  Riparian zone activities can affect the shape, sinuosity and substrate of a stream channel, which in turn influences instream habitat diversity. High habitat diversity provides for a diverse aquatic community through the presence of meso-habitats that cater for different habitat requirements. For example, deep pools formed on bends and undercut banks provide habitat for eels, while shallow, gravelly riffles provide trout spawning habitat.

- **Sediment (fish habitat, invertebrate habitat).**

  Excessive sediment input can negatively impact invertebrates and fish both directly (e.g. smothering, abrasion) and indirectly, through effects on the quality and quantity of resources (e.g. food, habitat). The response of the aquatic community to sediment inputs is influenced by factors such as sediment grain size, the duration and frequency of the input event, the presence of pollution, and features of the existing habitat and community (e.g. substrate, sensitivity of biota to sediment input) (Crowe and Hay 2004)

1.3 Report objectives

Environment Southland commissioned this technical review of the environmental effects of activities within the riparian zone. The review includes the following components:

1. A review of available information on the potential environmental effects of riparian zone activities, including:

    a. Adjacent land use;
    b. Subsurface drainage;
    c. Runoff from hard surfaces;
    d. Watercourse crossings;
    e. Stock access and vegetation management;
    f. Drainage maintenance;
    g. Damming and water abstraction;
    h. Gravel extraction.

2. Examples of case studies, from Southland where possible, demonstrating the effects of these activities.
3. A qualitative examination of the impact of climate change on the potential effects of these activities.
4. A summary of the relative environmental effects of activities within the riparian zone identifying the key issues and priorities.

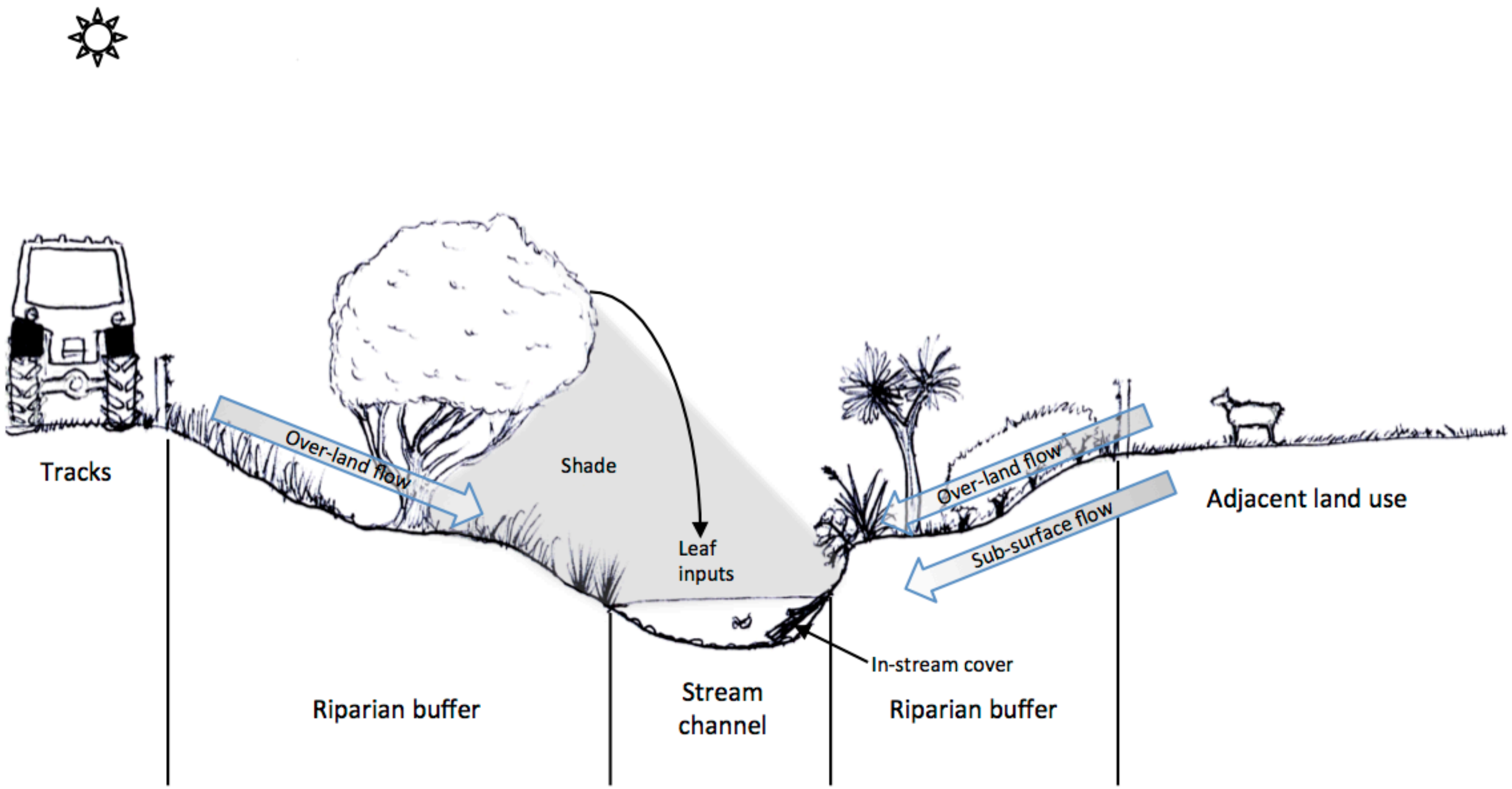

Figure 1 *Diagram showing the riparian zone of a small stream with some adjacent land activities.*

2. Environmental effects of activities within riparian zones

2.1 Adjacent land use

2.1.1 Background

While the effects of land use activities adjacent to riparian zones are technically outside the scope of this review, there will always be some debate over where the riparian zone ends and adjacent land and associated land use begins[3]. Consequently, we have provided some commentary on adjacent land uses in this report while acknowledging that more detailed assessments associated with land use effects on surface waters are provided elsewhere (separate reviews will provide more specific information on adjacent land use effects by considering hill country development (Ledgard and Hughes 2012) overland flow paths and wintering effects, Ryder Consulting 2013 and Monaghan *in prep*.).

There are a wide range of land uses that may take place adjacent to watercourses in Southland, including beef, dairy, deer and sheep farming, cropping, horticulture, flood control structures and production forestry. The potential of each of these land uses to impact on watercourses through losses of contaminant depends on a variety of complex factors relating to management practices, such as the timing and quantity of fertilizer application, effluent management, winter grazing, cultivation and harvest timing, drainage structures, and climate and landscape factors, such as rainfall, land slope and soil type. The main land use in developed parts of Southland has changed rapidly in recent years with a substantial increase in dairy farming (the number of dairy cows in the region has increased from approximately 50,000 in 1992 to 615,000 in June 2011, Statistics New Zealand), and a corresponding decrease in beef and sheep farming (the number of sheep has decreased from approximately 9 million in 1985 to 4.1 million in June 2011). Contaminant loss varies among land uses and therefore land use changes alter the risk of contaminants impacting on watercourses.

2.1.2 Reference material

There is a large body of literature investigating the impact of contaminant losses from different land uses on watercourses, including some studies from Southland

[3] Similarly, where the riparian zone ends and the aquatic environment begins is something that will vary in space and with time, and be affected by physical variables such as discharge, bank profile and plant tolerances to inundation and desiccation.

or studies that are directly relevant to Southland conditions (e.g., diary cows – Monaghan *et al.* 2005, deer – McDowell and Stevens 2008). Overall research on cattle farming predominates, with fewer studies on national coverage of sheep farming systems, and even fewer studies on deer and arable farming. Nitrogen loss has received the most attention with research on phosphorus and sediment losses less common, and few studies on faecal micro-organism losses (e.g., *Escherichia coli* or *E. coli*[4]). As indicated, this is a complex area of research and it is outside the scope of this review to quantify the potential impacts from all land uses, however we provide an overview of the relative potential impact of the main land uses in Southland.

Monaghan *et al.* (2010) recently undertook a comprehensive review of the relative risks of different agricultural land uses practices on water quality in the Southland region, drawing where necessary from research in other parts of the country. Four main potential contaminants were investigated; nitrogen, phosphorus, sediment and faecal micro-organisms. Based on their review and comparative assessments of different land uses, Monaghan *et al.* (2010) were able to derive a simple index showing the effects that different land uses and land use practices have on contaminant losses from farms (Table 4.1 in Monaghan *et al.* 2010). Some of the main conclusions that can be drawn from this index are:

- Converting from lower intensity land uses (e.g., sheep, forestry) to higher intensity land uses (e.g., dairy) will result in increased losses of nitrogen to water.
- If these conversions occur on heavy soils, and/or management practices are poor, then increased losses of phosphorus, sediment and faecal bacteria are also likely.
- Converting to deer farming is likely to increase losses of phosphorus and sediment, unless riparian and wet areas are well protected (due to deer wallowing and pacing).
- Converting from sheep farming to mixed cropping is likely to lead to increased losses of some contaminants, depending on how cropping is managed.

[4] *E. coli* is a type of bacteria that is usually found the lower intestine of warm-blooded organisms. It is used as an indicator of faecal contamination of waterways and as an indicator of the risk of exposure to disease-causing micro-organisms.

2.1.3 Case studies

Monaghan *et al.* (2010) took a modelling approaching to determining the relative risks of different land uses in Southland. This approach allowed the authors to account for some of the resource (e.g., soil, slope, rainfall) and management (e.g., nutrient inputs, farm dairy effluent management, riparian protection) risks that are known to contribute to the variability observed in nutrient and microbial losses from different farming systems (Monaghan *et al.* 2010). Nitrogen and phosphorus losses were modelled using OVERSEER® to make predictions for different land use scenarios based on the Bog Burn catchment (Figure 2), the lower part of which is dominated by poorly drained soil (Monaghan *et al.* 2010). To provide a contrast, nitrogen losses were also modelled for the Oteramika Creek catchment, which has a more feely draining soil type (Figure 3). If nitrogen inputs and cattle stocking rates were similar between the catchments, then nitrogen losses are expected to be lower from the Bog Burn catchment, as poorly drained soils have a greater soil denitrification[5] rate (Monaghan *et al.* 2010).

Both catchments showed a similar pattern of nitrogen loss for the different farm types, with dairy and beef farms having a greater nitrogen loss than sheep and deer farms (which were similar to each other) (Figures 2 and 3) (Monaghan *et al.* 2010). Monaghan *et al.* (2010) also observed that nitrogen losses were lower for less intensively farmed hill country sheep farms than lowland finishing farms (Figures 2 and 3). Phosphorus losses were greater for dairy farms than beef and sheep farms and did not vary greatly between hill and lowland sheep farms (Figure 2) (Monaghan *et al.* 2010). Relatively high nitrogen losses from mixed cropping are related to management factors, including having a large proportion of the farm in crop, pre-winter tillage, high rates of nitrogen fertilizer application, and the use of cultivation rather than direct drilling (Figure 2) (Monaghan *et al.* 2010). Forestry losses of nitrogen were relatively low compared to the other farm types (Figure 2).

Monaghan *et al.* (2010) concluded that the variation in nitrogen and phosphorus losses from the different farm types related to differences in the size and nitrogen loading of urine patches (higher for dairy cows), stocking rates (also higher for dairy cows than sheep), and the management of farm diary effluent. Ultimately

[5] The conversion of nitrate to gaseous nitrogen or nitrous oxides by bacteria.

whether or not the loss of these potential contaminants impacts on water quality in the catchment depends on various landscape and management factors, including how riparian zones are managed.

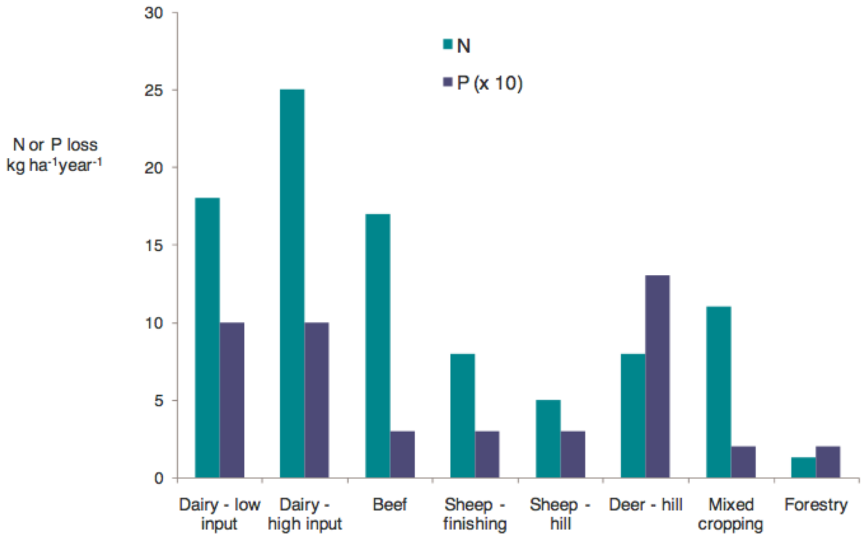

Figure 2 *Modelled estimates of nitrogen and phosphorus (x 10) losses for contrasting model farm types set within the Bog Burn catchment, Southland (Figure 4.1 from Monaghan et al. 2010).*

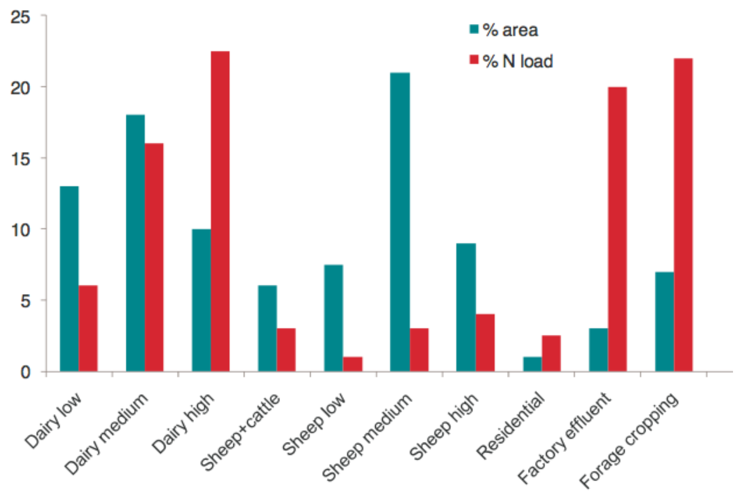

Figure 3 *Modelled estimates of nitrogen losses from a range of sources within the Oteramika catchment study, Southland (Thorrold et al. 1998). Taller red bars compared to green indicate sources that make a proportional greater contribution to catchment nitrogen loads than suggested by their areal extent i.e., they have a high nitrogen loss per unit land area (Figure 4.2 from Monaghan et al. 2010).*

2.2 Subsurface drainage

2.2.1 Background

The installation and use of artificial subsurface drainage (e.g., tile, mole, Novaflow) is a common practice in some parts of Southland where heavy soil conditions result in slow natural land drainage. Artificial drains transport contaminants (e.g., nutrients, sediment, faecal micro-organisms) directly to watercourses, therefore bypassing the natural processes of contaminant filtering that occurs in groundwater and riparian zones, and potentially leading to negative water quality impacts. Nutrient and microbial losses have been shown to be greatest when effluent is irrigated directly onto artificially drained land when the soil is saturated or during wet conditions (see below).

2.2.2 Reference material

The potential impact of subsurface drainage on surface water quality is well understood and there have been several studies carried out in different regions of New Zealand that quantify the input of contaminants to watercourses through artificial drains. Contaminants that are generally less mobile, such as phosphorus, sediment and faecal micro-organisms, are thought to enter subsurface drainage through cavities and fissures in the soil (macropores), which allows water carrying contaminants to infiltrate and drain through quickly (Monaghan *et al.* 2010). In this situation the losses of phosphorus have been shown to be the same as the losses in overland flow from undrained land (Monaghan *et al.* 2010) (overland flow paths will be the subject of a separate review, Ryder Consulting 2013). Attention has also focussed on methods of altering farm practices to reduce contaminant losses to drains (Houlbrooke 2005), and the use of wetlands to intercept drains and thereby reduce nutrient inputs to watercourses (Tanner *et al.* 2010).

2.2.3 Case studies

The impact of tile-and-mole drain installation on water quality in dairy and mixed sheep and beef land units has been monitored in the lower Pomahaka River catchment by the Otago Regional Council (ORC 2010). Chemical, physical and biological variables were measured in 21 tile drains (in 12 dairy and 9 sheep paddocks) within the catchment over one year (ORC 2010). Tile drain concentrations of five contaminants (nitrate-nitrite nitrogen, dissolved reactive phosphorus, total phosphorus, total nitrogen and suspended solids) were much higher in dairy pasture than sheep pasture (ORC 2010). *E. coli* concentrations in tile drains were high in both dairy and sheep pasture after rainfall, and also high in dairy pasture during dry periods following effluent application.

Water quality in tile drains has been measured in the Bog Burn catchment in Southland since 2003 as part of a study into the use of wetlands to reduce nutrient input from subsurface drains (Sukias and Tanner 2011). Annually, tile drains were found to export between 14-34 kg N per hectare of total nitrogen from the intensively grazed dairy pasture, the majority of which was soluble nitrate. Total phosphorus losses were 0.54-1.72 kg P per hectare, mostly dominated by soluble (dissolved reactive) phosphorus.

The Bog Burn study was part of a larger research project titled "*Best Practice Dairying Catchments for Sustainable Growth*". Wilcock (2006) reported *E. coli* concentrations in open and tile drains in three of the "*Best Practice*" study farms in Waikato, Taranaki and the Bog Burn, and calculated that if a drain flowed one day a week over a year, and there were 50 drain inflows to a stream, the total load of *E. coli* contributed to the receiving stream would be approximately 3 x 10^{10} *E. coli* per hectare of pasture per year. This is similar to the amount of faecal material sourced from dairy cow crossings or oxidation ponds (with typical effluent strength) (Figure 4) (Wilcock 2006).

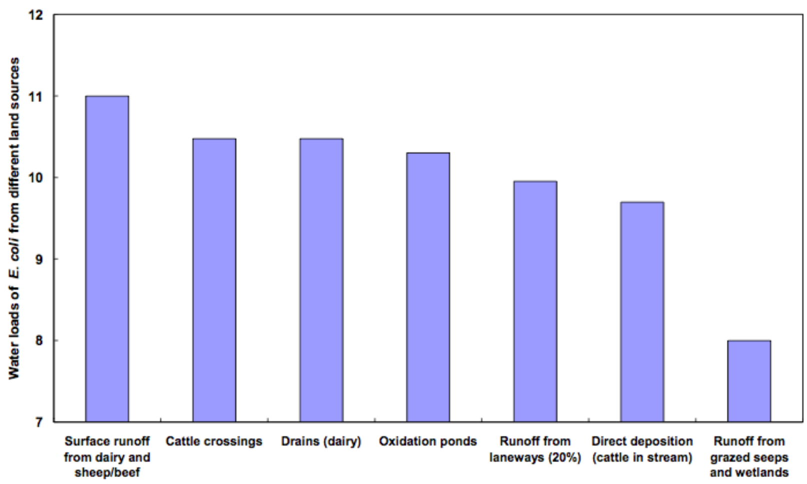

Figure 4 *Waterway loadings, $\log_{10}$ (E. coli/ha-pasture/year), for major sources of faecal material in the Waikato region (Figure 3.2 from Wilcock 2006). The calculated loads take into account the magnitude of the inputs and the duration of each type of loading.*

2.3 Runoff from hard surfaces

2.3.1 Background

Tracks, lanes, formed crossings, yards, and other hard surfaces on farms that stock use regularly can act as sources of contaminants to watercourses. Runoff from these compacted surfaces drains rapidly and, if located nearby to a watercourse, may directly deliver nutrients, sediments and faecal material. The potential for significant impacts is greater on dairy farms than other land uses as, depending on the distance required to travel, cows moving to and from the milking shed may spend an hour or more per day on a laneway, providing sufficient time to deposit large amounts of urine and faecal material (Wilcock 2006).

2.3.2 Reference material

The role of hard surface areas as sources of surface water pollutants has received little attention in New Zealand or overseas, with only a few published studies that have measured contaminant losses in farm lane runoff (Monaghan and Smith 2012). These have found elevated levels of nitrogen, phosphorus, sediment and faecal pollution in yard and lane runoff, indicating the importance of properly siting laneways and other hard surfaces in relation to watercourses and diverting runoff (Monaghan and Smith 2012 and references therein).

2.3.3 Case studies

Monaghan and Smith (2012) measured the concentrations and loads of contaminants transferred to watercourses in runoff discharging directly from lanes on dairy farms in the Bog Burn catchment (Southland). The concentration of nutrients, sediment and *E. coli* in the runoff was very high compared to that measured in catchment or paddock discharges, and was similar to that found in farm dairy effluent. Concentrations were higher at sites closer to the milking parlour, with exports of nitrogen, phosphorus, total suspended solids and *E. coli* between three to six times higher adjacent to the milking parlour than 500 m away. This demonstrates the importance of locating dairy lanes near the milking parlour away from watercourses (Monaghan and Smith 2012). Although the authors found that laneway contaminants made a relatively minor contribution to the overall annual pollutant load in the Bog Burn, during summer when stream flows are lower and temperatures warmer, laneway runoff was predicted to be an important contaminant source (Monaghan and Smith 2012).

2.4 Watercourse crossings

2.4.1 Background

Managed stock movement within farms generally involves stock following a formed track, lane or route. Where these intersect with a watercourse, if there is no culvert or bridge present, the stock must wade (ford) directly through the channel. Stock access to watercourse crossings may negatively impact water quality through the direct input of faecal contaminants and sediment, or other contaminant mobilization due to disturbance to the stream bed and banks. Sheep, deer and beef cattle generally only make infrequent crossings, as they move between grazing areas or yards, however dairy cows may cross the same watercourse multiple times on a single day while making their way to and from a milking shed. The potential impact on water quality is high as cows are more likely to defecate in water than on the adjacent track, contaminants from their bodies may wash off as they walk through, and their hooves may cause disturbance to the stream bed and banks (Davies-Colley *et al.* 2004). Bridge or culvert crossings are therefore preferred to fords as, when well-designed, any contaminants that may have directly entered the water at a ford crossing are diverted away from the watercourse.

2.4.2 Reference material

There is limited quantitative information on the water quality effects of stock crossing watercourses (which is distinct from the unrestricted access of stock to unfenced watercourses), although visual impacts on water clarity downstream have often been observed anecdotally. Guidelines for managing stock movement on farms to minimize impacts on water quality have been developed and include the installation of culvert or bridge crossings with raised edges (to prevent direct surface runoff to the watercourse) and dispersing runoff from crossings to land, ideally to a vegetated buffer area, rather than water (Environment Southland 2004, MfE 2001).

2.4.3 Case studies

Davies-Colley *et al.* (2004) measured the effect of a herd of 246 cows crossing the Sherry River (near Motueka) on their way to and from a milking shed. Total increases over the daily background level of 50% for suspended sediment, 400% for *E. coli*, and 10% for total nitrogen were observed. When the cows moved to the shed they did so in a tight herd, resulting in a sharp spike in contaminant levels, however following milking the cows crossed in smaller groups and contaminant levels were less elevated, although elevated for a longer period (Davies-Colley *et al.* 2004).

Based on the increases in *E. coli* concentrations observed in the Sherry River, Wilcock (2006) calculated that if four crossings were made per day then this would be equivalent of the total daily faecal output of nine cows entering the river, or looked at another way, at a stocking rate of three cows per ha, a 100 ha dairy farm would contribute 1 x 10^{10} *E. coli* per day to the river (3 x 10^{12} *E. coli* annually if 300 milking days per year). Wilcock (2006) noted that the faecal *E. coli* concentrations observed by Davies-Colley *et al.* (2004) were high and that it would be useful to have other corroborating studies. However, the Davies-Colley *et al.* (2004) study does demonstrate that dairy cow ford crossings can quadruple the concentration of *E. coli* in a river, and therefore the installation of a bridge or culvert crossing as an alternative should achieve an equivalent reduction (i.e., for a 246 cow herd crossing a medium size stream four times per day) (Wilcock 2006).

2.5 Stock access and vegetation management

2.5.1 Background

Stock access to waterways is known to exacerbate bank erosion and increase stream sedimentation and associated turbidity-related issues. For example, one study at Whatawhata (Stassar and Kemperman 1997 cited in Ritchie 2011) found that cattle entering streams over several days increased the number of bank erosion sites by 66% and increased turbidity six fold above background levels.

One of the best-known options for riparian management is the use of buffer strips, fenced-off strips of ungrazed vegetation on the banks of the waterway. Riparian buffer strips can fulfil several functions depending on the management goal (e.g., enhancing terrestrial biodiversity, controlling downstream flood potential), here we focus on their role in reducing nutrient and sediment inputs from adjacent land use to streams and improving aquatic habitat quality. There has been considerable scientific investigation of the effectiveness of buffer strips for management of non-point source pollution. Much of this work has been targeted at mitigation, but the results also provide insights into the likely effects of stock access to waterways, grazing of stream banks and vegetation removal.

Nutrient management

Riparian vegetation can reduce nutrient inputs by intercepting, retaining and transforming nutrients in overland and sub-surface flows. Fencing riparian buffer strips are a way of maintaining riparian vegetation by preventing grazing, which also serves to prevent direct inputs of animal waste to water (by preventing stock access). The effectiveness of buffer strips at removing nitrogen and phosphorus (the main nutrients of concern for nuisance periphyton growth) and trapping sediment can vary depending on the width and make-up of the buffer. For example, 5 m wide grass filter strips have been shown to be very effective in trapping large sediment particles, but up to 10 m wide strips are needed to remove finer particles (Gharabaghi *et al.* 2002 cited in Parkyn 2004). Buffer effectiveness is influenced by site-specific characteristics of the buffer zone and/or surrounding land, but in general to maintain effectiveness, buffer widths should increase as the slope length, angle and clay content of the adjacent land increase and as soil drainage decreases (Parkyn 2004).

The primary way that nitrate is removed by the riparian zone is by denitrification[6] (Howard-Williams *et al*. 1986, Cooper 1990), especially where favourable conditions exist (water-logged, anoxic soils). Nitrate can also be removed via uptake by growing plants (Prosser *et al.* undated). Larger woody plants (trees and shrubs) in particular can remove nitrate from deeper sub-surface flows. Ammonium concentration can also be reduced by infiltration and retention in the soil matrix of the buffer (Triska *et al.* 1994).

Phosphorus occurs in both soluble and particulate forms. In the soluble form, it can be removed by adsorption to the surface of soil particles, absorption into the soil matrix or by uptake by plants (Cooper *et al.* 1995). Particulate forms of phosphorus are bound to (or within) sediment or organic matter and are retained by infiltration into soils, deposition of the particles on the surface of the

[6] The conversion of nitrate to gaseous nitrogen or nitrous oxides by bacteria.

soil and filtration by the soil matrix. Unless there are significant erosion losses of phosphorus in a catchment, the majority of phosphorus lost from pastures is in the soluble form and will pass through vegetated buffer strips without being taken up (Monaghan *et al.* 2007). Therefore riparian management is likely to be more effective at reducing particulate than soluble phosphorus inputs, and as phosphorus adsorption is finite, a buffer may become saturated.

Habitat quality

Riparian vegetation can affect water and habitat quality by providing shade, which can reduce the growth rate and cover of algae, and reduce water temperatures, although such effects are dependent on channel size (Davies-Colley and Quinn 1998). Quinn *et al.* (1992) concluded that the greatest effect of stream bank grazing on streams was reduced shading, which affected water temperatures and periphyton growth (through increased light penetration). The shading provided by riparian vegetation can also indirectly affect water quality by suppressing the growth of algae and macrophytes, which can, in turn, lead to an increase in nutrient concentrations in the water column (due to reduced nutrient uptake) (Parkyn *et al*. 2003). Careful implementation of any riparian management that increases shading is therefore required to ensure that nutrient concentrations downstream are not increased as an unintended result (Parkyn 2004).

Riparian vegetation loses leaf and woody matter to surface waters, and in doing so can provide habitat for invertebrates and fish, and a food source for aquatic invertebrates. Some native fish species, such as inanga, spawn amongst streamside vegetation (Mitchell 1990, Mitchell 1991, Ravenscroft 2012, Taylor *et al.* 1992). Removal, alteration or grazing of riparian vegetation can negatively affect the suitability of such areas for spawning. Some invasive plant species (e.g., *Glyceria maxima* and crack willow) appear unsuitable as spawning habitat for inanga (Ravenscroft 2012). In addition, some fish species utilise in-stream structure as cover, particularly during the day. Such in-stream structure can include bank undercuts, boulders, macrophytes and woody debris. Activities in areas adjacent to the stream channel can affect the amount and quality of in-stream structure. Riparian vegetation is a primary source of woody debris present in the channel, although riparian fencing can reduce the cover provided by macrophytes by shading the channel. Sedimentation may reduce the quality of in-channel cover, such as interstitial spaces under boulders and cobbles and woody debris and damage to bank.

2.5.2 Reference material

Removal of riparian vegetation

Given the effort that has been undertaken to educate landowners on the benefits of riparian strips, and industry initiatives, such as the Dairying and Clean Streams Accord (2003, the accord expired in December 2012 but a new 'Sustainable Dairying: Water Accord' will become effective in 2013), the value of riparian margins are generally understood. However, there may be circumstances where riparian vegetation needs to be removed (e.g., for weed control). In such circumstances, it is likely that without efforts to exclude stock (by fencing) and re-vegetate the riparian zone, this may have detrimental effects on water quality and aquatic communities (e.g., Jowett *et al.* 2009).

Fencing

Fencing of streams creates a simple buffer that prevents direct stock access to the waterway, trampling of the banks and grazing of riparian vegetation that can trap sediments. Direct inputs of faecal matter and urine can be a significant source of nutrients and pathogens (bacteria, viruses and protozoa) to waterways where stock have access. Stock access can also cause bank instability, pugging and trampling damage, which can lead to significant sediment inputs to the waterway (Trimble and Mendel 1995). The creation of buffer strips by constructing fences on both sides of the channel is the most effective way to reduce such direct inputs. However, such buffers are prone to invasion by weeds (such as broom and gorse) without active management. While this is unlikely to reduce their effectiveness at improving water quality, it may present issues for farm management.

A study of riparian characteristics for the Waikato region (Storey 2010 cited in Ritchie 2011) noted that riparian fencing was strongly correlated with stream bank condition. Whereas fencing on both banks was strongly correlated with reduced pugging and total erosion, fencing on only one bank was not significantly correlated, suggesting that fencing both sides of the stream is important for bank protection.

Planted riparian buffers

Buffers can also be planted with woody plants in effort to suppress weed invasion and to stabilise the banks (sometimes in combination with artificial stabilisation works such as battering). Such planted buffers may also remove nitrogen from deeper subsurface flows and shade the channel to a greater degree. Woody plants can also have the benefits of providing leaf litter and wood inputs to the stream channel, which can provide food and habitat for aquatic species. It should be kept in mind, however, that it might take some time for substantial amounts of wood material to enter the stream.

Riparian buffers in forestry

The purpose of riparian buffers in forestry is mainly to reduce the disturbance of the streambed caused by harvesting operations and to provide resources (leaf litter, woody debris) for stream food webs after harvest. Generally, one of the main concerns post-harvest is erosion (of hill slopes and stream banks) and resulting sedimentation of waterways downstream. The riparian buffers are intended to stabilise the stream banks and traps any material eroded from hill slopes.

2.5.3 Case studies

Various studies have considered the degree to which buffer strips affect water quality, although the conditions in each study vary, with some considering very large forested buffers (>30 m), whilst others are much smaller (<8 m). However, these studies do provide some indication of the potential effectiveness of riparian buffers at reducing nutrient, sediment and microbial loads.

Fencing

Monaghan *et al.* (2010) investigated the effect of various mitigation scenarios (including stock exclusion from waterways) on nitrogen, phosphorus and *E. coli* in the Oreti catchment using a catchment land-use model. Based on this, they estimated that current stock exclusion rates (75% for dairy, 35% for sheep and beef for all farms in Land Use Class 1-3) would reduce the potential loss of nitrogen and the faecal indicator *Escherichia coli*[7] from dairy farms by 15% and by 3.5% for sheep and beef farms. Total stock exclusion was predicted to further reduce the loss of nitrogen and *E. coli* from dairy farms (20%) and sheep and beef farms (10%). This modelling also predicted that current stock exclusion rates would reduce phosphorus loss from dairy farms by 30% and 10.5% from sheep and beef farms, while total exclusion for all farms in Land Use Class 1-3 was predicted to reduce phosphorus loss from dairy farms by 40% and by 30% from sheep and beef farms.

Riparian buffers have been shown to be effective at reducing sediment and nutrient concentrations in a wide range of systems. Smith (1989) compared sediment and nutrient loadings between sites with retired and grazed riparian pasture adjacent to waterways. Sediment, phosphorus and particulate and nitrate-nitrogen concentrations at retired riparian sites were substantially lower than at grazed sites. This study estimated that, in the short-term (<2 years), the ungrazed riparian strips reduced loads of suspended sediment (<87%), particulate phosphorus and nitrogen (<84%), dissolved reactive phosphorus (<55%) and nitrate-nitrogen (<67%).

Haygarth *et al.* (undated) reported that riparian fencing and provision of stock drinking water could reduce ammonium loss to waterways through overland flow paths by 25-75%, mainly through the prevention of direct inputs of animal waste (faeces, urine). Exclusion of stock from near waterways will also dramatically reduce the risk of faecal matter entering water, with a resulting reduction in the risk of microbiological contamination.

Preventing stock access also reduces sediment inputs through reduced erosion. It will also reduce soil compaction, which can reduce the effectiveness of riparian zones at capturing and storing runoff, by reducing the rate of infiltration and capacity to absorb run-off.

Planting riparian buffer strips

The potential for riparian buffer planting to reduce instream water temperatures

[7] *E. coli* is a type of bacteria that is usually found the lower intestine of warm-blooded organisms. It is used as an indicator of faecal contamination of waterways and as an indicator of the risk of exposure to disease-causing micro-organisms.

is dependent on the development of shade, buffer age, and buffer length (Parkyn *et al.* 2003). In their assessment of nine riparian buffer zone schemes (fenced and planted) in the North Island (age range 2-24 years, length 100-4200 m) Parkyn (*et al.* 2003) found that there were only slight differences in water temperatures (measured continuously for up to three weeks) between buffered and control reaches at most sites. Within some buffered reaches median and maximum water temperatures were in fact slightly higher than at control sites. The degree of canopy closure over the stream appeared to be an important factor in achieving water temperature reduction. This agreed with the outcome of modelling studies by Rutherford (*et al.* 1999), which indicated the deeper and wider the stream the longer the buffer needed to achieve water temperature reductions (Parkyn *et al.* 2003). Rutherford (*et al.* 1999) predicted that for buffer planting to achieve a 5°C temperature reduction its length would need to be 1-5 km for first order streams versus 10-20 km for fifth order streams, and 75% shade of the channel would need to be achieved (Parkyn *et al.* 2003). These findings illustrate that the use of riparian buffer planting to achieve water temperature reductions through shading effects may take decades, and is dependent on canopy closure, long buffer lengths, and protection of small tributaries and headwaters (Parkyn *et al.* 2003).

Parkyn *et al.* (2003) found that riparian buffers lead to a rapid improvement in visual clarity and channel stability, but that the responses in nutrient and faecal contamination were variable. Macroinvertebrate densities were lower in streams with buffers, but that the macroinvertebrate community in stream reaches with riparian buffers were more indicative of "clean water" than nearby stream reaches without riparian buffers. They emphasised the need to consider the length of the reach planted and the position of the planted reach relative to sources of recolonizing macroinvertebrates. Greenwood (*et al.*, 2012) recommended that to maximise the effectiveness of riparian buffers for macroinvertebrates management should target small or headwater streams and those that retain relatively un-impacted habitat. In habitats that are already degraded more intensive measures (i.e. channel and instream habitat modification) may also be needed to achieve large local benefits, although riparian planting and fencing in degraded areas may reduce downstream effects (Greenwood *et al.*. 2012). Identifying and focusing on the most vulnerable locations and the factors that limit macroinvertebrate communities (e.g. sedimentation, suitable water velocities) will increase the effectiveness of riparian management (Greenwood *et al.*. 2012).

The choice of tree species for riparian buffer planting needs to consider their long term effectiveness, some species may be useful when young and in an active growth phase but become less so as they mature (Mander *et al.* 1997 cited in Parkyn 2004).

Restoration of two small pastoral streams (North Island) by planting riparian vegetation and preventing stock access was found after 10 years to have increased numbers of several native fish species relative to before restoration (Jowett *et al.* 2009). Numbers of giant kokopu, and redfin bullies were doubled, and banded kokopu numbers slightly increased. Shortfin eel numbers decreased, however this was expected as eels are typically less abundant in forested than pastoral streams. Macroinvertebrate communities also changed to become more similar to native forest sites upstream.

Buffer strip width

A review by Ritchie (2011) for Environment Waikato, presented in the following three paragraphs, concluded that buffer width required for sediment removal depends on the steepness of slope. A steep slope requires a wider buffer to slow the velocity of surface run-off (Collier *et al.* 1995). Dillaha *et al.* (1989) measured removal rates between 70 and 54% on slopes between 11-16° with 4.6 m filter strips under shallow, uniform flow draining 18m strips of bare cropland. For a longer (9.1 m) buffer width, they found effectiveness was significantly higher (84-73%) over the same range of slope. They cautioned these reductions might not occur in field conditions, due to longer upslope distances and flow concentration effects, although the rainfall they applied was heavy (equivalent to a 100- year return period event). A similar magnitude of removal was reported from a New Zealand field study on gley soils, where Smith (1989) found a sediment load reduction of 87% for a retired pasture buffer of 10-13 m over 22 months of rainfall events, when compared with unretired riparian pasture areas nearby.

In steep, hilly terrain, buffers may be less effective because overland flow tends to concentrate in channelised natural drainage ways with high flow velocities (Parkyn 2004). Dillaha *et al.* (1989) formed this conclusion after surveying 10% of Virginia's filter strips established to reduce sediment from cropland. They did, however, find these strips had some benefits in protecting banks from erosion. Smith's (1989) study at Tauwhare sampled channelised drainage ways and did detect a substantial reduction in sediment loads where riparian areas were retired from grazing. Dillaha *et al.*'s (1989) experiment found that the filter strips were less effective in their later runs, partly because the soils were inundated and produced more run-off, and partly due to sediment accumulation in the filter strip. Parkyn (2004) also reports that over time, pore spaces in riparian buffer soils may clog with sediments, reducing effectiveness.

The Waikato River Independent Scoping Study (NIWA 2010) estimated that on sheep and beef farms, 5 m buffers could reduce suspended sediment by 55-60% and this could increase to 65% with 15 m buffers. These estimates were based on a number of the studies cited above.

Mander *et al.* (1999) reported that large (30-50 m wide) riparian buffer strips reduced nitrogen by 67-94% and phosphorus by 81-97%. Dillaha *et al.* (1989) considered the effectiveness of buffers of varying widths. They found that a 4.6 m strip reduced nitrogen by 54%, phosphorus by 61% and suspended sediments by 74%. In comparison, 9.1 m strips reduced nitrogen by 73%, phosphorus by 79% and suspended sediments by 84%. Etterna *et al.* (1999) and Schoonover and Willard (2003) reported that nitrogen concentrations were reduced by 70% by 10 m wide buffer strips, which are important to maintain conditions for denitrification as well as noting that larger trees provide nutrient uptake to greater depths than short vegetation (such as grasses) and can reduce concentrations of nitrate-nitrite nitrogen in lateral sub-surface flows.

Management of riparian strips

Riparian zones may become nutrient saturated, resulting in reduced ability to capture and retain nutrients (e.g., Cooper *et al.* 1995). Harvesting of vegetation (e.g., selective wood harvesting, managed light sheep grazing) and sediment removal are ways of removing captured and stored sediment and nutrients, and can increase the life span and effectiveness of buffer strips (Howard-Williams and Pickmere 1989, Barling and Moore 1994).

Howard-Williams and Pickmere (2004) report on the variable success of riparian protection in assisting with nutrient attenuation, ranging from no apparent attenuation in large rivers to high attenuation in small streams and wetlands. Nutrient attenuation (uptake) in the Whangamata Stream (Lake Taupo) over a 32-year study period following the establishment of riparian strips in 1976 was greatest in the 1980s when there was reduced flow, little channel shading and dense macrophyte growth. Attenuation decreased during periods of higher flow, increased overhead shading and a reduction in macrophytes. Howard-Williams and Pickmere (2004) concluded that if nutrient attenuation was the objective of the riparian strip planting then active management of stream bank vegetation that minimises shade over the stream channel and maximises fast-growing stream plants is required. Harvesting of instream vegetation may also need to be considered.

Use of riparian buffer strips in forestry

Research in the Coromandel Peninsula has found that post-harvest, streams without riparian buffers had wider channels, increased bank erosion, higher light intensity, higher maximum air temperatures, benthic invertebrate communities dominated by taxa that are tolerant of organic pollution and reduced fish abundance in comparison to channels with an average buffer width of 18 m (reviewed in Quinn 2005). Buffers may also reduce the amount of slash entering streams (Fahey *et al.* 2004).

2.6 Drainage maintenance

2.6.1 Background

Watercourses provide drainage to the surrounding land, channelling surface runoff, reducing flooding and lowering water tables. The ability of a watercourse to provide these essential functions can be compromised though by the excessive growth of aquatic plants and the accumulation of sediment and debris within the channel (which can be related to nutrient and sediment inputs from surrounding land). As a consequence, regular channel engineering (i.e., realignment, straightening) and removal of plants, sediment and/or instream debris using herbicides, mechanical excavation and other less common methods (e.g., deploying plant eating fish) is carried out by regional and unitary councils throughout the country (e.g., Southland, Marlborough, Waikato). The intensity and frequency of drainage maintenance varies, but can involve excavation of the bed and banks along the entire channel length (Hudson and Harding 2004). Aside from drainage, watercourses also have many other functions including providing habitat for aquatic communities, and no matter their size, or whether they are considered to be artificial or modified, are linked to larger freshwater and ultimately coastal systems. Depending on the method employed and the location, drainage management activities may therefore have significant adverse effects on aquatic communities within the watercourse and downstream, including direct removal of fish and invertebrates with excavation, habitat loss (including spawning habitat) through aquatic plant and instream debris removal and disturbance to the channel bed and banks, reductions in water quality due to the decay of plants and the mobilization of sediments and other contaminants within the channel, and the spread of nuisance aquatic plant species.

2.6.2 Reference material

A comprehensive review of drainage management in New Zealand (and overseas) was prepared by Hudson and Harding (2004), including an assessment of the environmental and economic costs associated with the commonly used methods. Local studies that have attempted to quantify some of the potential impacts of drainage management have encountered difficulties of varying magnitude related to differences in sampling efficiency between before and after treatments and/or in achieving adequate sample sizes and replication (e.g., Ryder 1997, Goldsmith 2000, Young *et al.* 2004a, Greer *et al.* 2012). While the Hudson and Harding (2004) review identified that there were gaps in the New Zealand knowledge of the impact on aquatic communities and how to best manage watercourses to maintain drainage, the authors were able to recommend several alternative management practices to reduce adverse impacts (Hudson and Harding 2004). Guidelines for managing waterways on farms have been developed that include alternative drainage maintenance practices to reduce adverse impacts (e.g., no spraying during spawning, avoid excavating to a depth that effects connectivity to tributaries), and also to reduce the need for maintenance (MfE 2001, Hudson 2005, Gibbs 2007).

2.6.3 Case studies

Young *et al.* (2004a) investigated the impacts of two drainage maintenance activities, herbicide (diquat) application and mechanical excavation, on water levels, plant cover, water quality and macroinvertebrate community composition in three spring-fed drains (two treatment and one control) in Marlborough. Sites were monitored on three occasions prior to treatment then again one week, one month, three months and six months after treatment. The direct removal of eels and invertebrates was observed with mechanical excavation, and there was a short-term (one month) decline in macroinvertebrate density. Short-term declines in downstream water clarity were also observed due to the suspension of sediment and organic material, and there were short term increases in ammoniacal nitrogen and dissolved reactive phosphorus concentrations. Herbicide application had no effect on oxygen concentrations and no direct toxic effects on invertebrates, although taxonomic richness decreased, probably due to an increase in detritus. Young *et al.* (2004a) recommended that manual return to the water be considered for any fish removed, and the use of weed rakes rather than excavator buckets to reduce the removal of eels, although this may have negative effects on water quality by causing sediment disturbance but not effective sediment removal.

A recent study in Southland investigated how fish populations were impacted by mechanical plant (macrophyte) removal and if the magnitude of any impact was altered by limiting removal to alternate 50 m lengths of the channel (Greer *et al.* 2012). Twenty-three 350 m long reaches within four streams in the Waituna Lagoon catchment were completely cleared of macrophytes, cleared in 50 m alternate lengths or not cleared at all (control). Macrophyte removal was not found to impact on the number of fish species present, although this may have been related to the history of regular macrophyte removal. The number of individuals of each species caught was not sufficient to detect changes in the abundance (catch per unit effort) of each species, however, considering all species together, there was a 51% decrease in fish abundance in the completely cleared reaches and a 39% decrease in the alternately cleared reaches approximately 18 to 33 days after removal, relative to abundance 4 to 19 days prior to removal. There was no evidence that alternately clearing reaches minimised impacts on fish abundance at the 350 m reach scale. The movements of 11 large giant kokopu were also monitored throughout the study using radiotelemetry. Four of the fish were radio-tagged prior to macrophyte removal and the remaining seven during removal, having been lifted from the stream by the excavator. Four of the five fish from completely cleared treatments moved after macrophyte removal (another three were not re-located). Giant kokopu from alternately cleared reaches remained and used uncleared sections of the reach during the day then moved into cleared sections during the night. The authors concluded that the use of uncleared areas as refuges by large giant kokopu indicates alternately clearing reaches can reduce the negative impacts of macrophyte removal on this species, however further research would be needed to determine if other fish species would also benefit.

Ballantine and Hughes (2012) reviewed sediment and nutrient data from Waituna Stream collected prior to, during and following mechanical drain cleaning operations. They concluded that suspended solids and total phosphorus concentrations increased sharply through drain cleaning events, while concentrations of nitrate and total nitrogen differed minimally from long term concentrations. Small increases in flow after the drain cleaning gave rise to sharp increases in turbidity (implying a decrease in clarity), suggesting a shift in the turbidity regime.

2.7 Damming and water abstraction

2.7.1 Background

The magnitude of potential impacts of dams and water abstractions on aquatic communities is largely determined by their size, which can range from small dams or weirs and stock water supply abstractions to large irrigation schemes and hydroelectric generation projects. This review considers the potential impact of dams and water abstractions of the smallest size, of the type that would provide water to a single farm for irrigation and stock use. Small stock water storage dams are often built 'on-line' by the placement of an earth bund across the stream channel (as opposed to a water abstraction to an 'off-line' storage area) and are generally located near the head of the catchment where the watercourse flow is intermittent. There is no record of the exact number of small dams in New Zealand, as not all regional councils require consent for their construction (e.g., Environment Southland only require consent if a dam is greater than 3 m high or the catchment area greater than 500 ha) (Thompson 2012). Auckland Regional Council has estimated that there are approximately 4,500 in its region, however higher numbers are expected in seasonally drier regions (Maxted *et al.* 2005). In Southland the number of small dams is expected to be greatest in drier inland areas where they are required to ensure stock water security over the summer period.

Although the magnitude may vary, the impacts of all dams and water abstractions are potentially the same. The congregation of stock around an 'on-line' water supply dam may result in increased nutrient and faecal inputs to the watercourse downstream and increased erosion through trampling, although sediment may be retained within the dam. Impacts will be more significant for dairy cows rather than sheep, and if there are multiple dams within the same catchment the effects can be cumulative (Thompson 2012). The timing and volume of the inflow and outflow to the dam, and the magnitude of nutrient inputs, have a major impact on the effect on downstream water quality. In larger watercourses with permanent flow and where fish communities are present the absence of adequate screening at a water abstraction may result in fish being drawn into the intake. If flow decreases substantially as a result of the intake, reductions in habitat and water quality may occur downstream, including water temperature increases, and nuisance algae growths. Construction of the system may require disturbance of the bed by machinery (presenting a risk of introducing contaminants such as lubricants and/or invasive weeds) resulting in a temporary increase in sediment levels downstream, which if not timed correctly may impact on fish spawning. Poorly constructed dams may also present a barrier to fish passage.

2.7.2 Reference material

There has been considerable research on the impact of large dams and water abstractions in New Zealand (reviewed in Young *et al.* 2004b) and methods and guidelines have been developed to assess and minimize the potential effects on aquatic communities (e.g., Jowett and Hayes 2004, Jamieson *et al.* 2007). In contrast, the effects of small dams have not been studied extensively in New Zealand or overseas. To our knowledge there have been only two studies on small farm dams, in Hawke's Bay (Thompson 2012) and Auckland (Maxted *et al.* 2005).

2.7.3 Case studies

Maxted *et al.* (2005) found that water quality was degraded within and downstream of six small 'on-line' ponds within pasture and native forest in the Auckland region. Water quality was the poorest in rural ponds with large surface areas and long retention times. In some cases water temperatures were elevated for several hundred metres downstream of a pond, and approached temperatures that may be harmful for some fish species. Macroinvertebrate communities downstream of the ponds were also negatively affected, with reductions in the abundance of 'pollution sensitive' taxa. Maxted *et al.* (2005) recommended that where possible unused ponds are removed and alternative 'off-line' sources of stock water are considered to reduce effects.

2.8 Gravel abstraction

2.8.1 Background

Gravel abstraction is a common activity in many regions of New Zealand, especially in those areas with abundant and accessible sources of alluvial gravels. However, this activity can have a number of effects on channel geomorphology, water quality and in-stream habitat.

Geomorphological effects

Rinaldi *et al.* (2005) reviewed the effects of alluvial gravel extraction and identified the following effects:

Upstream incision – Excavation steepens the gradient of the channel bed upstream of the extraction area. This may also lead to incision of tributaries.

Downstream incision – Excavation can also create a trap for sediments, interrupting downstream sediment transport.

Lateral channel instability – Incision can lead to channel instability, changes in channel width, bank erosion and channel migration.

Bed armouring – Reduced sediment supply can lead to selective removal of finer bed material and development of an armour layer.

Gravel bar skimming – can have the above effects and can favour removal of larger particles at the surface of the bed, increasing bed erosion and bedload transport.

Off-channel pits and reactivation of inactive channels – excavation lateral to the active channel can result in channel migration into the pit during high flows.

Coastal effects – reduced sediment loads from rivers to estuaries and coastal environments can have geomorphological effects in these environments.

Such geomorphological changes caused by gravel extraction can result in structures (e.g., bridges) being undermined and other structures, such as water intakes, being exposed.

Environmental effects

Gravel abstraction has been found to increase the proportion of fine sediments within and downstream of the extraction area and can lead to reduced water clarity. The physical changes outlined above (channel form, substrate size, flow velocity) can directly affect habitat availability and suitability for invertebrates and fish. Changes in the form and composition of river channels can also affect the level of the water table and surface-groundwater interaction, which can indirectly affect water and habitat quality (e.g., water temperature, nutrient exchange).

Changes in habitat quality and the abundance of invertebrates resulting from gravel extraction can affect the number and/or biomass of fish able to be supported by the stream.

2.8.2 Reference material

The most comprehensive review of the geomorphological effects of gravel extraction is provided by Rinaldi *et al.* (2005), which presents five case studies from Italy and Poland as well as reviewing studies from California (USA), Washington (USA), Italy, Spain, France, England, Poland, Australia and New Zealand (Lower Manawatu River). This study also summarises some ecological effects of gravel abstraction.

Kelly *et al.* (2005) looked at the sedimentary and ecological effects of gravel extraction in the Kakanui River in North Otago and the Wairoa River near Nelson.

2.8.3 Case studies

Kelly *et al.* (2005) looked at the effect of gravel extraction in the Kakanui River in North Otago and the Wairoa River near Nelson. Much more gravel was extracted from the Kakanui River (32,000 m^3 per annum) than from the Wairoa River (7,000 m^3 per annum). This study found that gravel extraction from the Kakanui resulted in the channel becoming narrower and more incised, a reduction in the size of bed particles (with more silt and sand), more exposed bedrock and more uniform and higher near-bed water velocities, while some of the same changes were observed in the Wairoa, but to a lesser degree.

In addition to these changes in the Kakanui River, Kelly *et al*. (2005) also observed lower numbers of invertebrates, lower taxon richness (the number of different types of invertebrate) and a smaller proportion of clean-water taxa (mayflies, stoneflies and caddis flies). In the Wairoa, the number of invertebrates in and downstream of the gravel extraction site were only slightly lower than upstream, but the taxon richness was much lower downstream. The percentage of clean-water taxa was higher in the extraction site than upstream, although the site downstream of the extraction site was lower than both the other sites.

3. Potential climate change effects in Southland – why include it?

The Ministry for the Environment provides climate change predictions for each region of New Zealand[8]. These predictions are dependent on future greenhouse gas emissions, which are uncertain, and the models that are used to make the predictions also vary in their sensitivity to the emissions. Predictions of future climate therefore provide an indication of the range of conditions that may be experienced, rather than absolute predictions, however they provide a useful basis for understanding how climate change could affect future land use in Southland.

By the end of the century, Southland is predicted to be warmer, with air temperatures expected to be approximately 2°C higher and the number of days that maximum temperatures exceed 25°C increasing by 30 days. Southland will also be wetter, particularly in winter and spring, with very heavy rainfall events becoming more frequent. The intensity of storms will decrease in summer and increase in winter, with an increase in the frequency of extreme winds, particularly westerly winds.

The potential effect of these changes on land use within Southland will vary depending on the activity and the location. Parts of Southland that are already dry may become drier, however in other areas water security may be improved with increases in average annual rainfall. With warmer temperatures and milder winters there may be opportunities for new crops to be grown and farmers may benefit from faster crop and pasture growth. However, warmer temperatures may increase the spread of pests and weeds, and increases in storm intensity are also expected to increase the risk of flooding, landslides and erosion in Southland. As a result climate change could have a significant impact on land use in Southland and as a consequence of this also on the environmental effects of activities within riparian zones. A qualitative assessment of climate change effects has therefore been presented below.

Adjacent land use

Climate change may provide opportunities for new crops to be grown in some parts of Southland and farmers may benefit from faster crop and pasture growth,

[8] http://www.mfe.govt.nz/issues/climate/about/climate-change-affect-regions/southland.html

potentially leading to intensification. As illustrated in Section 2.1.3, different land use activities present different contaminant risks and, in general, increasing intensification is likely to increase contaminant loss to land and therefore ultimately to watercourses.

Subsurface drainage

Increases in storm intensity are also expected to increase the risk of flooding, and erosion. While more rainfall may provide increased dilution of contaminants, in combination with land use intensification, losses of faecal micro-organisms and nutrients to subsurface drainage are likely to increase.

Runoff from hard surfaces

If land use intensity does not change in the interim, then, over time, more rainfall may provide increased dilution of contaminants within hard surface runoff, potentially reducing effects on watercourses. However, if land use intensifies, the quantity (load) of a contaminant may also increase, potentially increasing environmental effects on watercourses.

Watercourse crossings

Climate change may result in land use intensification in some areas, potentially increasing the quantity of a contaminant being deposited at a watercourse crossing and therefore increasing environmental effects.

Stock access and vegetation management

One of the main effects of riparian vegetation, particularly in small streams, is the shading of the stream channel, which reduces the potential for algal growth as well as reducing peak water temperatures. This is likely to be an effective way of mitigating the effects of warmer air temperatures on stream systems.

Apart from increased air temperatures, climate change is predicted to increase the frequency and severity of extreme weather events. As discussed above, riparian buffers can capture and store contaminants in run-off during rainfall events, offering a potential opportunity to manage increased contaminant loads that may occur as a result of climate change.

Drainage maintenance

Climate change is predicted to produce a warmer and wetter climate in Southland. Increases in storm intensity are also expected to increase the risk of flooding, landslides and erosion. In this event drainage maintenance will become even more important. The frequency of maintenance may need to increase as sediment inputs increase through erosion, and plant growth within the channel increases due to improved growing conditions. Any increase in the frequency of drainage maintenance will increase any effects on aquatic communities.

Damming and water abstraction

Although climate change is predicted to result in Southland becoming wetter in general, parts of the region that are already dry may become drier. This could increase the reliance on water abstraction and dams to provide stock water supply security. Any increase in stock water dams or water abstraction in a catchment is likely to result in a cumulative increase in environmental effects on aquatic communities downstream.

Gravel abstraction

Potential increases in storm intensity are expected to increase the risk of flooding, landslides and erosion. Such changes may affect the consequences of gravel extraction. Channel degradation resulting from gravel extraction will increase the flood capacity of some channels (which is often used as a justification for gravel extraction), but in some cases it may also pose a risk to infrastructure (e.g., through undermining bridge abutments or pilings). Increased water temperatures resulting from climate change may be exacerbated if surface water-groundwater exchange is reduced. Demand for gravel may change with changing landuse.

4. Environmental effects matrices

The detailed information presented in Section 3 on the potential environmental effects of activities within the riparian zone has been summarized in a series of tables that allow the relative effects of each activity to be visually compared (Tables 1 to 4). Each table presents a simplified interpretation and therefore is not able to account for the variety of climate, landscape and management factors that will influence the environmental effect of a particular activity. However, the presentation of the information in this way does allow some key issues and priorities for the management of riparian zone activities in Southland to be identified.

Adjacent land use activities have perhaps the greatest potential to significantly impact water quality, by increasing nutrient, sediment and faecal micro-organisms inputs, and, through these contaminant losses, negatively affecting aquatic communities (Table 1). Any proposed change in land use must therefore be carefully considered against the potential for an increase in contaminant loss. Management practices can then be put in place to ensure that the conversion of land use does not have an increased effect on the local riparian and surface water environment.

The installation of subsurface drainage (e.g., tile, mole, Novaflow) causes surface runoff to bypass the natural processes of contaminant filtering and can result in a significant increase in nutrient, sediment and faecal micro-organisms inputs to a watercourse (Table 2). For dairy farms, faecal micro-organisms inputs to subsurface drainage can be as great as the amount derived from oxidation ponds. Consideration should be given to restoring the natural filtering process by intersecting subsurface drainage and capturing contaminants using wetlands.

The input of faecal micro-organisms from ford crossings is similar to that from subsurface drainage, with a major contributor being the direct deposition of faecal matter as dairy cows cross watercourses on their way to and from the milking parlour (Table 2). Considerable reductions in faecal matter and other contaminants can be achieved simply by the replacement of a ford with a culvert or bridge. Contaminant runoff from these and other hard surfaces may also contribute contaminants to watercourses but in much smaller amounts than occurs at ford crossings.

Unlike many of the other activities outlined, riparian fencing and planting have the potential for many environmental benefits. Conversely, the removal of riparian vegetation and stock access can have many detrimental effects on water quality and ecology of waterways. The exclusion of stock can improve the condition of the channel and banks by reducing erosion, slumping and trampling, while riparian planting can further stabilise the stream channel and banks (Table 3). Stock exclusion will reduce the direct input of animal waste into waterways as well as reducing the run-off of some nutrients, sediment and microbes from the riparian area (Table 3). Vegetation within the riparian zone, if ungrazed, can reduce nutrient, sediment and microbial loads by filtering overland flow and can reduce nutrient loads in sub-surface water by nutrient uptake, as well as maintaining or promoting conditions suitable for denitrification (Table 3). One of the greatest effects of riparian vegetation is shading of the channel, with the magnitude of this effect dependant on the size of the stream channel. Shading can reduce algal growth, which can reduce the risk and severity of “blooms” as well as reducing water temperatures (Table 3). The provision of a riparian buffer in plantation forests can help reduce the effects of harvesting operations and post-harvest disturbance, by stabilising stream banks, trapping eroded sediment and slash and by providing shade, which will reduce algal growth and promote cooler water temperatures (Table 3). Overall, riparian vegetation can improve habitat conditions for fish and invertebrates (Table 3).

Riparian fencing and planting does come with some ‘negative’ effects. For example, there are costs to the land owner associated with fencing, planting, loss of grazing, weed and pest control and alternative water supply. Flooding issues are an important consideration for fencing riparian areas (Bewsell *et al.* 2005 cited in Ritchie 2011). Fences may need to be placed further from the waterway in flood-prone areas, meaning loss of more grazing area.

Drainage maintenance activities that involve excavation of the channel can have significant negative effects on macroinvertebrates and fish through their direct removal, and also habitat modification and loss, and lesser indirect effects through water quality reductions (Table 4). Plant removal through the use of herbicides has a relatively smaller impact, although it may not be as effective and therefore require more frequent maintenance. Drainage maintenance is primarily focused on increasing the hydraulic capacity of the watercourse and depending on the method employed any environmental benefits (e.g., removing sediment accumulations) are likely to be outweighed by the potential negative effects. Also, current drainage maintenance conditions in some parts of Southland require stream access to diggers, which can restrict substantial riparian planting and full fencing.

The damming of a watercourse to create a water supply for stock can have negative effects on all aspects of water quality that may persist further downstream in the catchment for some distance (Table 4). The effects of water abstraction for 'off line' water storage or direct applications (i.e., without storage) are relatively smaller, depending on the proportion of the flow abstracted, and therefore 'off-line' storage or direct application is preferred over 'on-line' storage dams (Table 4). With any water abstraction adequate screening is required to exclude fish, and if flow decreases substantially as a result of the intake, reductions in habitat and water quality may occur downstream.

Table 1 *Relative potential environmental effects of different land use activities adjacent to the riparian zone. Greater number of crosses = greater effect.*

Adjacent land use	Type	Geomorphology	Water quality					Aquatic biodiversity		
			Temperature	Nutrients		Sediment	Microbial	Periphyton	Benthic invertebrates	Fish
				N	P					
Farming [a] (typical stocking rates, unrestricted riparian access)	Sheep	x	x	xx	xx	x	xx	x	x	x
	Dairy	xx	x	xxx	xxx	xx	xxx	xx	xx	xx
	Deer	xxx	x	xx	xxx	xxx	xxx	xx	xx	xx
	Beef	xx	x	xxx	xxx	xx	xxx	xx	xx	xx
Cropping [b]		-	-	xxx	x	xx	x	x	x	-
Forestry [c]		x	-	x	x	xx	-	x	x	x
Horticulture [d]	Winter vegetables	-	-	xx	xx	xx	x	x	x	-

a For all farming land uses effects depend largely on soil type, climate and management factors.
b Cropping effects depend on soil type and fertilizer and cultivation management.
c Sediment loss varies depending on harvesting management practises. Adverse effects on aquatic biota are more cyclical and coincide with harvesting events.
d Nutrient and sediment effects vary depending on management practices such as fertilization and cultivation.

Table 2 *Relative potential environmental effects of contaminant runoff from subsurface drainage, hard surfaces and stock crossings adjacent to the riparian zone. Greater number of crosses = greater effect.*

Type	Geomorphology	Water quality					Aquatic biodiversity		
		Temperature	Nutrients		Sediment	Microbial	Periphyton	Benthic invertebrates	Fish
			N	P					
Subsurface drainage [a]	-	-	xx	xx	xx	xxx	xx	x	-
Hard surfaces [b]	-	-	x	x	x	xx	x	-	-
Ford [c]	x	-	xx	xx	x	xxx	xx	x	x
Culvert [d]	x	-	-	-	-	-	-	-	-
Bridge [e]	x	-	-	-	-	-	-	-	-

a Potential effects of increased contaminant inputs due to bypassing the contaminant filtering that occurs in groundwater and riparian zones.
b Applied to dairy cows, increased potential for effects where lanes near the milking parlour are close to a watercourse.
c Applied to dairy cows.
d Temporary potential effect of increased sediment during construction. Some habitat loss for macroinvertebrates and fish along culvert length. No effect on fish passage if installed correctly. As with hard surfaces there is a potential for contaminant runoff, however balanced out by benefits of keeping stock out of the watercourse.
e Temporary potential effect of increased sediment during construction. As with hard surfaces a potential for contaminant runoff, however balanced out by benefits of keeping stock out of watercourse (assuming runoff from bridge is managed appropriately).

Table 3 *Relative potential environmental effects of different stock access and vegetation management activities adjacent to the riparian zone. Greater number of crosses = greater negative effect. Greater number of ticks = greater positive effect.*

Type/activity	Geomorphology	Water quality					Aquatic biodiversity		
		Temperature	Nutrients		Sediment	Microbial	Periphyton	Benthic invertebrates	Fish
			N	P					
Removal of existing riparian vegetation	xxx	xxx	x	xxx	xxx	xx	xx	xx	x
Riparian planting [a]	✓✓	✓✓✓	✓	✓	✓✓	✓✓	✓✓	✓	✓✓
Dual buffers [b]	✓✓	✓✓✓	✓✓	✓✓	✓✓	✓✓	✓✓✓	✓✓✓	✓✓
Fencing of riparian zone [c]	✓✓	-	✓	✓	✓✓	✓✓	✓	✓	✓✓
Fencing + planting	✓✓✓	✓✓✓	✓	✓✓	✓✓	✓✓✓	✓✓	✓	✓✓✓
Forestry –use of riparian buffers	✓✓✓	✓✓✓	-	✓	✓✓✓	-	✓✓✓	✓✓✓	✓✓✓

a Long-term effect. For temperature, main positive effect is a potential reduction in daily peak temperature.
b Outer buffer of grass to trap and store P and sediment, inner buffer to promote denitrification.
c Potential temporary increase in sediments during installation. Primary effect to restrict stock access to waterway. Need to provide for alternative drinking supplies.

Table 4 *Relative potential environmental effects of drainage maintenance activities, damming, water abstraction and gravel extraction. Greater number of crosses = greater effect. Tick = positive effect. ? = unknown.*

Type/activity	**Geomorphology**	**Water quality**					**Aquatic biodiversity**		
		Temperature	**Nutrients**		**Sediment**	**Microbial**	**Periphyton**	**Benthic invertebrates**	**Fish**
			N	**P**					
Channel engineering [a]	xxx	-	x	x	x	-	x	xx	xx
Plant and sediment removal – excavation [b]	xx	x	x	x	x	-	x	xx	xx
Plant removal – herbicide [c]	x	x	-	-	-	-	x	x	xx
'On-line' stock water supply dam [d]	xx	x	x	x	x	x	x	x	x
Water abstraction [e]	x	x	-	-	-	-	x	x	x
Gravel extraction	xxx	x/√	?	?	xxx	-	-	xx	xx

a Depending on the time of year potential effects could be significant for fish spawning, especially if in a coastal area (e.g., whitebait) or a trout stream.
b Depending on the time of year potential effects could be significant for fish spawning, especially if in a coastal area (e.g., whitebait) or a trout stream. Otherwise the main effect on fish is the risk of direct removal and habitat loss. Potential water quality effects are generally expected to be short-term, although as a result of the channel becoming more open water temperatures and periphyton growth may increase.
c Potential water quality effects are generally expected to be short-term, although as a result of the channel becoming more open water temperatures and periphyton growth may increase. Possible indirect effect on the likes of whitebait spawning through habitat loss (inanga spawn on spring high tides amongst bankside vegetation).
d The timing and volume of the inflow and outflow to the dam and the magnitude of nutrient inputs has a major impact on the effect on downstream water quality.
e Potential effects depend on the magnitude of the abstraction in relation to instream flow. Note that abstraction can affect contaminant dilution but again depends on proportion of flow removed.

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