Review of Riparian Buffer Zone Effectiveness

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Prepared for MAF Policy by:

Stephanie Parkyn
NIWA

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1. Introduction

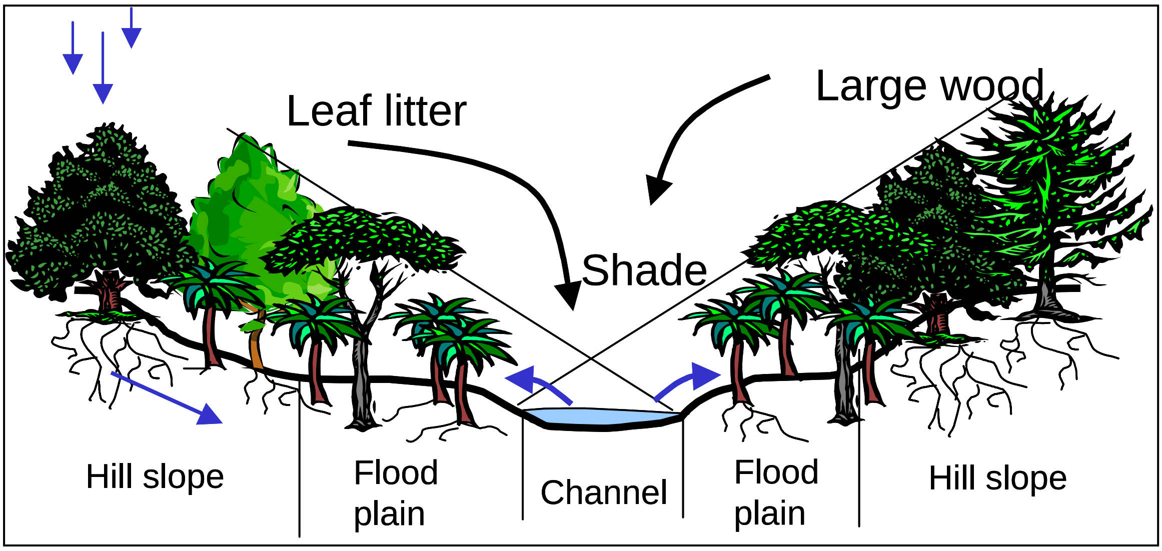
1.1 PURPOSE OF THE REVIEW

The purpose of this report is to review and summarise published research on the efficiency and management of riparian buffer zones (RBZ) with respect to the attenuation of sediment and nutrients, and biodiversity enhancement. While there have been numerous studies on the efficiency of RBZ with respect to sediment and nutrients, many of these studies have been small-scale and site-specific. Therefore, a review of these studies needs to consider an assessment of the catchment scale factors that influence the effectiveness of RBZ in attenuating catchment loads.

1.2 WHAT ARE RIPARIAN ZONES?

The riparian zone generally encompasses the vegetated strip of land that extends along streams and rivers and is therefore the interface between terrestrial and aquatic ecosystems (Gregory et al. 1991, Martin et al. 1999; Fig. 1). In addition to streams and rivers, the definition of riparian zones in the literature often includes the banks of lakes, reservoirs and wetlands.

Figure 1: The riparian zone is the land beside the stream that interacts with (1) runoff from hillslopes and (2) streamwater when this overflows into the floodplain. The vegetated riparian zone can affect the stream by intercepting runoff, and thereby improving water quality, by providing shade, leaf matter and wood, and stabilising stream banks.



1.3 RIPARIAN BUFFER ZONE FUNCTIONS

Riparian buffer zones are often advocated as environmental management tools for reducing impacts of land use activities on aquatic resources. The buffer zone, area, or strip is generally regarded as the strip of land that separates an upland or hillslope area from streams, lakes or wetlands. Land use activity is modified in this zone to prevent adverse effects on the water quality, biota and habitat within the watercourse. Buffer zones or strips have also been variously labelled as Stream Protection Zones (SPZ), Streamside Management Zones (SMZ), or Riparian Management Zones (RMZ). In agricultural landscapes, buffer zones often consist of a fenced area alongside streams that stock are excluded from and this may be left as a grassy sward, or planted with woody vegetation. In forestry systems, a buffer zone is generally one of production trees left beside the stream when the surrounding area is harvested.

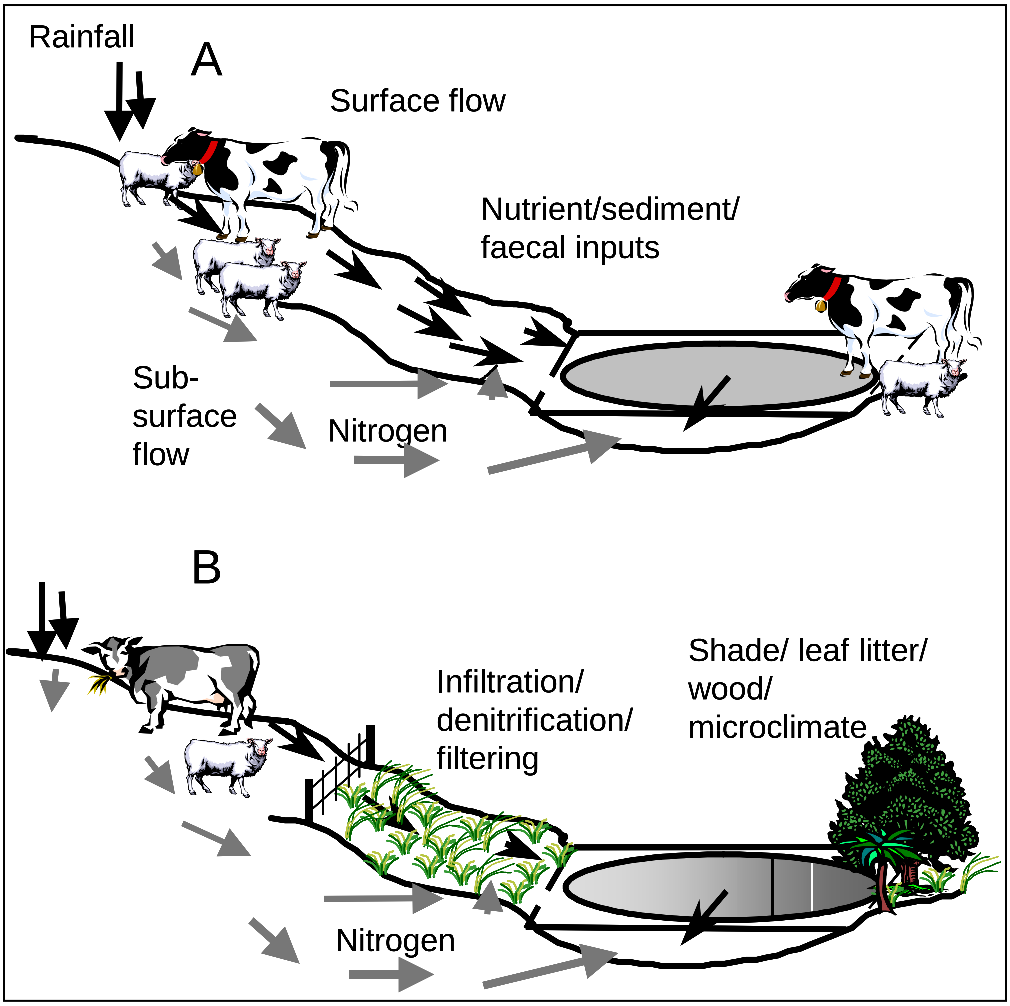
Riparian management can take various forms, some of which are summarised below:

- **Grass Filter Strips**: Fenced strip of rank paddock grasses to filter nutrients and sediment.
- **Headwater or riparian wetlands**: Fenced wetlands as hotspots for nutrient removal
- **Rotational Grazing**: Filter strips with varied stock grazing practices, such as occasional light grazing by sheep.
- **Forested or planted native trees**: a buffer of native trees to return ecological function to the stream and provide water quality benefits.
- **Production trees or plants**: a buffer of forestry trees left unharvested along stream banks, or production trees that are planted in riparian zones for selective harvesting with minimal disturbance (e.g., Tasmanian blackwoods) Plants such as flax for weaving, or fruit and nut trees, or high value native tree species that can be selectively harvested may also provide ecological function and a mechanism to remove nutrients such as phosphorus from the riparian zone.
- **Multi-tier system**: a combination of buffers where native forest trees may be used beside the stream to enhance ecological function and biodiversity, a buffer of production trees may occur outside of that and at the outer edge beside agricultural land a grass filter strip may be used.

Riparian buffer zones are used as a management tool to perform many functions (Appendix 1, Fig. 2) including stabilising channels, preventing stock access to waterways, filtering sediment and other particulates (including nutrients and microbes attached to sediment), removing soluble nutrients, and providing terrestrial and aquatic habitat. In addition, wet riparian soils, generally rich in carbon and low in oxygen, can promote a significant loss of N through denitrification. Riparian vegetation can also provide corridors for the movement of native fauna and flora between geographically separate areas, although the spread of weed species can also be facilitated in this way.

Riparian zones are commonly areas with heterogeneous vegetation and soils and therefore provide a diverse habitat for terrestrial and semi-aquatic organisms (birds, insects, amphibians and plants) (Boothroyd & Langer 1999). Vegetation in the riparian zone can influence water flow, both surface and subsurface (through root systems) and has direct effects on stream functioning. Forest vegetation in particular can shade streams and lower stream temperatures. High light levels from deforestation around streams leads to increases in algae and in-stream primary production, and changes to invertebrate community composition. Stream temperature has a direct impact on aquatic species as most metabolic processes are accelerated with increasing temperature and many fish and invertebrate species have thermal tolerances that can be exceeded in unshaded streams (Quinn et al. 1994, Martin et al. 1999). Trees provide organic matter inputs in the form of leaves and woody debris, creating a diversity of food resources and habitats for in-stream fauna. Terrestrial insects may also be attracted to vegetated riparian zones and become a valuable food source for fish when they fall into the stream (Barling & Moore 1994).

Figure 2: (A) Inputs of direct and diffuse sources of contaminants to pasture streams; (B) implementation of riparian management through fencing allows infiltration, denitrification and filtering of contaminants from flows (except for deep sub-surface flow), and planting provides additional ecological benefits



2. Nutrients and Sediments

Agricultural development has led to widespread increases in the levels of nitrogen (N) and phosphorus (P) in lowland watercourses and subsequent nuisance growths of algae and other aquatic plants. In New Zealand the introduction of nitrogen-fixing clovers, use of nitrogen fertilisers, including the practice of spreading animal wastes on pastures, and direct addition of stock urine and faeces in pastures have increased the amounts of nitrate leaching from pastoral catchments.

Suspended sediments affect stream habitat and water quality by reducing water clarity for sighted organisms, and reducing light penetration for plant growth. Aesthetic appeal is also reduced by high suspended sediments. Sediments that settle on the substrate fill interstitial spaces and affect the habitat available for invertebrates (Ryan 1991). In agricultural areas, hill slope (sheetwash) erosion, mass movement, stock damage to stream banks, and erosion of tracks and raceways are the key factors that introduce sediment to streams. Increases in sediment delivery to streams in production forestry catchments occur mainly after harvesting, and mass movements and erosion of roads and tracks can be major sources of sediment.

2.1 MODES OF PARTICULATE AND DISSOLVED POLLUTANT TRANSPORT

The ability of buffer zones to attenuate pollutants will depend upon the mechanisms by which these pollutants reach surface waters. Three transport processes can occur:

- direct pollution (e.g., stock access to streams, bank erosion);
- surface runoff;
- subsurface flow and drainage.

Surface runoff

Surface runoff can occur through several mechanisms. It may result when the surface soil becomes saturated (saturation excess) which is common where flow pathways converge as a result of topography. It may also occur when rainfall intensity exceeds the infiltration capacity of the soil (infiltration-excess) a process that is common in poorly drained clay-rich soils (Muscutt et al. 1993). Surface runoff can be a major transport mechanism for soluble pollutants, particularly when land beside a stream has been grazed, or fertiliser or livestock waste applied to the land during or prior to rain events. Surface runoff can also be a conduit for sediment and particulate pollutants. Sediment transport can occur through sheet erosion in spatially uniform flows over hillslopes, but is most likely to result from areas where flow is concentrated (Fig. 3), and from bare soils (e.g., stock tracks, slips, cultivated soils). Sediment in surface runoff can also carry particulate forms of phosphorus, and a high proportion of total P loss has been found to occur during periods of high flow (Culley & Bolton, 1983, Smith 1987).

Smith (1987) studied the runoff generated from hillslopes in Waikato, New Zealand after rain events between July 1983 and March 1985. This study reported large runoff volumes, and demonstrated that nutrients and sediment were carried downslope, often becoming concentrated into preferred flowpaths. Consequently, nutrients and sediments were not uniformly discharged along stream banks. Almost all of the total phosphorus (TP) and total nitrogen (TN) transport occurred in mid-late winter and early spring. TP and TN were predominantly in particulate forms, suggesting that riparian management that trapped sediment and particulate nutrients would be effective in improving stream water quality at that site.

Figure 3: Water flow concentrated into an ephemeral stream. Photo T. Wilding.

Subsurface flow and drainage

Subsurface flow is frequently the major pathway of N transport in catchment runoff and high concentrations commonly occur in artificial subsurface drains (Muscutt et al. 1993). Intensive agriculture is often accompanied by subsurface drainage especially in clay-based soils. These drains provide routes for the rapid transport of water and pollutants from the soil during high water table conditions and, in many cases, can bypass the riparian zone by directly discharging to the stream.

Subsurface flow paths are influenced by the surrounding topography and soil drainage characteristics. For instance, on land that is free draining, water and associated pollutants may bypass the riparian zone, whereas on poorly drained soils or where the water table is high, pollutants in subsurface water may be carried into the soils of the buffer zone. On occasion, subsurface flows may re-emerge, and discharge downslope as surface runoff.

2.2 MODES OF SEDIMENT AND NUTRIENT REMOVAL

Buffer zones can be effective at removing nutrient and sediment inputs to streams by restricting the direct use of land beside the stream and by processing water that has been transported into the riparian zone. The mechanisms of contaminant removal in buffer zones differ according to characteristics of the hydrology, soils, and vegetation as well as the mode of transport to streams.

Direct effects

The removal of stock from streams and riparian areas has obvious benefits for water quality. Belsky et al. (1999) reviewed published information on the effects of livestock grazing in the western US and found that livestock negatively affected water quality, stream channel morphology, hydrology, riparian zone soils, instream and streambank vegetation, and aquatic and riparian wildlife. Livestock contribute nutrients directly to streams and riparian areas in their dung and urine. Faecal material deposited in the riparian zone on soils that have been damaged by treading may be readily washed overland into the stream with little opportunity for filtration of contaminants by vegetation (often reduced, damaged or absent), or infiltration into compacted soil (Nguyen et al. 1998; Trimble and Mendel 1995).

Livestock damage to riparian vegetation and soils destabilises the banks and leads to mobilisation of fine sediment (Trimble 1994), that in turn causes sedimentation in the channel and reduced stream clarity (e.g., Waters 1995). In addition, more runoff of sediment occurs from soils disturbed and compacted by livestock trampling (Nguyen et al. 1998). The resulting increased sediment load is accompanied by particulate nutrients that may contribute to stream enrichment as well as eutrophication of lakes and estuaries downstream (e.g., Williamson et al. 1996). Culley & Bolton (1983) estimated that bank erosion contributed 32% of sediment discharge and 10% to the export of P from an agricultural catchment in Canada.

Bank destabilisation can occur as a result of deforestation and conversion to agriculture, as well as from the direct effects of livestock. Planting trees and shrubs in the riparian buffer zone can stabilise stream banks, as long as the rooting depth of the plants are appropriate for the size of the bank.

Line et al. (2000) demonstrated that stock exclusion by fencing reduced TKN, TP and suspended sediment loads in a stream running through a dairy farm in North Carolina, USA by 78, 76, and 82%, respectively. Weekly loading rates of nutrients and sediments were monitored for 81 weeks prior to fencing and 137 weeks after fencing. The effects of installing alternate watering systems only were also evaluated, but these were not as effective as fencing stock from the streams. Nitrate loads were not significantly reduced in the time span of the study and the authors suggested that these were likely to decrease in the future when trees had established in the buffer and mechanisms of nutrient uptake and denitrification developed. This study clearly shows that rapid improvements in water quality can be seen after exclusion of stock (i.e., within 2.5 years), particularly reductions in particulate nutrients and sediment.

Surface pollutant transport

Buffer zones where stock have been excluded and long grass or natural vegetation has been allowed to develop or been planted can reduce diffuse pollutant transport from agricultural land by:

- infiltration within the buffer zone which reduces surface runoff;
- reduction of surface flow velocities from increased hydraulic roughness of the vegetation in the buffer;
- physical filtering effect of dense vegetation.

Much of the research into the effectiveness of buffer zones for removing contaminants from surface runoff has focussed on vegetated filter strips (VFS), usually consisting of rank paddock grasses. Researchers including Young et al. (1980), Dillaha et al. (1989), Magette et al. (1989), and Daniels and Gilliam (1996) have studied the effectiveness of grass filter strips in trapping sediment and nutrient through laboratory or field experiments (see also Table 1). They reported trapping efficiencies exceeding 50% for sediment and nutrients adsorbed to sediments (such as phosphorus), while trapping of dissolved nutrients was less efficient.

The main function of vegetated filter strips for sediment removal is to provide flow resistance (through enhanced hydraulic roughness) that reduces the flow velocity and sediment transport capacity of surface runoff. This leads to an enhanced deposition of particulates (Neibling and Alberts 1979, Gharabaghi et al. 2002). Ponding can also occur at the upslope edge of the buffer zone causing sediment accumulation (Muscutt et al. 1993). Some removal of soluble pollutants also occurs, but infiltration, not deposition is the primary mechanism for removal of soluble pollutants from overland flow (Gharabaghi et al. 2002).

Increased infiltration may occur in buffer zones as a result of the change in soil structure associated with the change in vegetation type (Muscutt et al. 1993). Vegetation can provide root channels for improving infiltration of water into the soil (Collier et al. 1995). Removal of stock grazing increases the infiltration capacity of the soil, because trampling can compact soils.

In an experimental study of grass filter strip (perennial rye grass) efficiencies, Gharabaghi et al. (2002) found that the first 5 m of the filter strip were critical for sediment removal. Almost all of the easily removable particles (larger than 40 microns in diameter) were captured with the first few meters of the filter strip. However, the remaining small size particles were very difficult to remove by filtering as they stayed in suspension. The only mechanism that helped in removal of small size sediments was infiltration. During experimental runs with low to moderate flow rates on longer plot lengths (20 m wide filter strips), 90% removal efficiencies of sediment could be achieved because fine sediments were able to infiltrate into the soil. Gharabaghi et al. also concluded that sediment removal efficiency did not increase much beyond 10 m filter strip widths, although the potential for the buffer to become clogged with fine sediment over time should be considered when establishing optimum buffer widths.

Nutrients that are sediment-bound can also be effectively removed in VFS. Dillaha et al. (1989) applied manure and fertiliser to bare fallow plots with different filter strip widths of 0, 4.6 and 9.1 m. The 9.1 and 4.6 m filter strips with shallow uniform flow (11% and 16% slopes) removed an average of 84 and 74% of the incoming solids, 79 and 61 % of the incoming P, and 73 and 54% of N. The removal of P and N from the runoff was nearly as effective as sediment removal because a large proportion of the nutrients were sediment-bound.

In a New Zealand study, Smith (1989) found that retired pasture buffers of 10-13 m were capable of reducing suspended sediment and particulate nutrients in channelised surface runoff by over 80%. Dissolved N and P removal was less (67, 55%; Table 2).

The proportion of surface runoff that infiltrates into the buffer soil is likely to reduce the load of soluble pollutants that are transported through the buffer, at least in the short term (Muscutt et al. 1993). In other words, improving the infiltration capacity of the buffer will also improve the efficiency of buffers for soluble nutrient removal.

Subsurface pollutant transport

Reduction in soluble nutrients, largely N, from buffer zones and headwater and riparian wetlands has been demonstrated in a number of studies and the two main mechanisms for nutrient removal are:

- uptake by vegetation;
- denitrification.

The relative importance of these two processes may differ between buffer zone types, however, most researchers agree that riparian zones can be highly effective for soluble nitrate removal. Many studies have shown >90% reductions in nitrate concentrations in subsurface flows as water passes through riparian areas or wetlands (Gilliam 1994, Fennesey & Cronk 1997, Table 3.). Buffers are consistently reported to reduce nitrate to below 2 mg/L, often throughout the year and even when nitrate inputs are extremely high (Muscutt et al. 1993). Biological denitrification is the most desirable means of nitrate attenuation as the microbial conversion of NO_3 removes nitrate from the system in the form of N gases (Martin et al. 1999). Plant uptake can eventually return the nitrogen to the system as the plants die and decompose. Denitrification is a microbially mediated process whereby bacteria convert nitrate to N_2 gas when there is a plentiful carbon source, such as wetland or riparian soils rich in organic matter, and when conditions are anoxic or low in oxygen.

Wetlands

Wetland areas and seeps that intercept drainage before the flow enters streams have been clearly identified as areas causing loss of NO_3, with denitrification being the most important mechanism. Cooper (1990) found that the majority of nitrate loss occurred in riparian organic soils, despite these soils occupying only 12% of the border of a small pasture stream in New Zealand. He attributed this to characteristics of the catchment hydrology, as a disproportionately high percentage of groundwater flowed through these small wetlands in the base of hollows, and also to their high capacity for denitrification (anoxic, high in denitrifying enzymes and available carbon). Stream channel nitrate removal was largely through plant uptake (watercress) and was much more variable. Cooper (1990) also identified that the capacity for denitrification in these soils was under-utilised.

Nguyen et al. (1999a) found 27% removal of phosphorus and 54% removal of nitrogen over a 6-month period in a wetland at the head of a small stream at Whatawhata, Waikato. Measurements and modelling in two contrasting wetlands showed the importance of hydrology and contact time in determining the effectiveness of riparian wetlands in removal of nitrate (Nguyen et al. 1999b); the longer the contact time, the greater the removal.

Wetlands can also be effective at phosphorus removal depending on the physical-chemical-microbiological processes that affect P uptake (McDowell et al. 2004). These methods include sorption-precipitation of dissolved phosphorus by wetland substrate, sedimentation-deposition of particulate P, and P assimilation by microbial and plant biomass. Plant P, unless it is removed by harvesting, can be released back to wetlands via decomposition of plant litter. Thus, the most important processes are sorption and sedimentation (Cooke et al. 1992, Nguyen 2000). However, P removal by wetlands generally declines after a period of years or decades depending on loading rates, hydraulic retention time, wastewater characteristics, wetland substratum, and accumulation of organic solids (McDowell et al. 2004).

Gilliam (1994) called for an effort to protect ephemeral and intermittent stream channels as well as wetlands, as these are areas that initially receive surface runoff and where shallow groundwater seeps into surface water, and thus may be some of the most important areas for preserving water quality.

2.3 BUFFER STRIP DESIGN AND EFFICIENCY FOR SEDIMENT AND NUTRIENT REMOVAL

Much of the variability in studies of nutrient or sediment removal efficiencies can be explained by site specific differences in characteristics of the buffer zone or in characteristics of the surrounding land. Some of these factors are outlined below.

Width

Data from studies comparing multiple width buffers in the same location (Young et al. 1980, Dillaha et al. 1988, Dillaha et al. 1989, Magette et al. 1989, Peterjohn and Correll 1984, Vought et al. 1994) have shown that sediment and total phosphorus removal rates (between 53 and 98%) increase with increasing buffer width (4.6 m to 27m). Where a grass buffer strip has been designed sensibly to treat sheet rather than channelised flow, many researchers report substantial sediment removal within a few metres of the upslope boundary (Barling & Moore 1994, Fennessy & Cronk 1997). Grass filter strips in particular have been shown to be very effective at trapping sediment particles. Neibling & Alberts (1979) found that 91% of incoming sediment to a grass filter strip was deposited in the first 0.6 m. Much of the larger particles of sediment may be removed in 5 m of grass buffer, but finer particles may require 10 m (Gharabaghi et al. (2002)

The width required to optimise nutrient removal has been debated with little systematic study of the issue. Fennessy and Cronk (1997) reviewed studies of RBZ effectiveness for the removal of contaminants, particularly soluble nitrate. Almost 100% of nitrate can be removed by buffers 20-30 m wide, while examples of forested buffers of 10 m achieved over 70% retention of N. Table 3 lists a range of buffer widths that have been assessed for nitrate removal (from Fennessy & Cronk 1997). Many of the buffers were forested, and N uptake by plants and denitrification were believed to have been an important factor in removing soluble N. However, one problem in assessing minimum widths is that many studies have had to use existing buffer widths, rather than deriving it experimentally.

Because of the different modes of particulate and dissolved contaminant transport, multi-tier or combination buffers are often advocated. A narrow combination buffer consisting of 5 m of grass filter strip and a 1m wide row of deciduous trees significantly reduced nitrate in subsurface flows beneath cropland in Italy (Borin & Bigon 2002). A substantial reduction in nitrate (average 81%) was observed at the field/grass buffer boundary and the authors concluded that the roots of the trees were extending beyond the combined 6 m buffer so that the zone of influence was larger than the land that was retired from use. Further reductions in nitrate were measured through the buffer and discharge to the stream had concentrations that were less than 2 ppm.

The width required for nutrient and sediment removal can be quite variable and the Auckland Regional Council suggested another method of determining optimal buffer width, which was based on the width needed to develop a self-sustaining buffer of native vegetation. Parkyn et al. (2000) recommended a buffer width of 10-20 m as the minimum necessary for the development of sustainable indigenous vegetation with minimal weed control, and to achieve many aquatic functions.

Vegetation

The consensus in the literature is that grass buffer strips are effective at filtering sediment and sediment-associated pollutants (particulate P and N) from surface runoff. However they are less effective in removing soluble nutrients such as nitrate, ammonia, and dissolved P. Nitrate removal from subsurface flows is considered to be greater in forested buffers, partly through uptake by plants (Fennessy & Cronk 1997, Martin et al. 1999). However, the main mechanism by which nitrate is removed from groundwater is thought to be biological denitrification. Wetlands and soils in riparian zones have been shown to have a high capacity for denitrification compared to terrestrial and aquatic soils (Cooper 1990). Vegetation in the riparian zone can contribute to denitrification through root exudates and plant decomposition in some ecosystems, and organic matter status of the soil may have a major effect on N removal efficiency (Cooper 1990, Muscutt et al. 1993)

Riparian carbon inputs to streams (i.e., leaf litter and wood) can also increase the potential for stream bed denitrification. This may be particularly important for systems where groundwater inflows bypass the riparian zone or where there are tile drains. In a study comparing buffer effectiveness in well-drained and poorly drained settings in North Carolina, USA, Spruill (2004) showed that buffers were effective at reducing nitrate in the groundwater of both. Most of the nitrate removal occurred through denitrification in the buffer soils and streambeds. Thus, even though nitrate in ground water passed beneath the buffer, the relatively high organic carbon in the discharge zone of those sites (derived largely from riparian vegetation) provided an environment conducive to denitrification (Spruill 2004).

The type of vegetation planted in buffer zones can also influence nutrient removal. For instance, James et al. (1990) (cited in Gilliam 1994) noted the failure of a riparian buffer to reduce NO3 in Maryland, USA, was due to leguminous trees that actually increased the NO3 in groundwater.

Nutrient removal efficiencies in buffers may also be affected by the age of the vegetation. Mander et al. (1997) studied N and P budgets in four riparian forests of varying age in Estonia and USA. While the buffers were able to remove both nitrogen and phosphorus, even when the input concentrations were very high, young forest stands, bushes, and wet grasslands showed the most intensive nutrient removal. This was due to intensive nutrient uptake by plants as they were in an active growth phase, and high microbiological activity and adsorption capacity of the soils.

Phosphorus accumulates in riparian soils and can be taken up by plants but there is no process similar to denitrification that removes P to the atmosphere. Therefore, buffer zones could potentially become saturated and their ability to trap P may decline with age unless sediments or organic matter are removed from the buffer zone (Barling & Moore 1994).

Harvesting production trees or plants, or fruit and nuts from trees in riparian zones can provide a mechanism where P can be removed from the riparian zone. Examples of this include indigenous systems of tropical agroforestry where non-timber products (fruits, nuts and ornamentals) can be harvested (Robles-Diaz-de-León & Kangas 1999). There may be scope in New Zealand to use riparian buffers as zones for flax harvesting, medicinal plant growth, manuka honey, etc.

Combination buffer systems in the USA often consist of an upslope grass buffer, a managed forest zone and an undisturbed forest zone next to the stream. Because no studies had assessed the impact of forest harvesting and management on these riparian systems, Hubbard & Lowrance (1997) studied the nitrate removal from shallow groundwater where the forest zone was either mature forest, clear cut, or selectively thinned. All three forest management treatments were effective in assimilating nitrate and there were no differences between treatments. Concentrations of nitrate in shallow groundwater were highest at the field – grass buffer interface and dropped most dramatically (by factor of 10) within the managed forest zone.

Slope

Slope angle is a key factor in determining sediment entrapment within RBZ (Young et al. 1980, Peterjohn and Correll 1984, Dillaha et al. 1989, Magette et al. 1989, Phillips 1989). Dillaha et al. (1988) compared sediment removal under differing slopes with all other factors constant, deriving an inverse relation between slope angle (6º-9º) and sediment entrapment (50-90 %). In general, many review articles of buffer zone studies conclude that buffers need to be wider when the slope is steep, generally to give more time for the velocity of surface runoff to decrease (Barling & Moore 1994, Collier et al. 1995).

Soils and drainage

Soil drainage properties influence RBZ performance. Free draining soils minimise the generation of surface runoff, both on the hillside and within a buffer. Better paddock management may increase buffer effectiveness. For example, in a grazing system, a reduction in stocking rates and longer times between paddock rotations may be sufficient to alleviate surface compaction problems and enhance infiltration, while establishment of a good groundcover will slow incoming water to the buffer (Herron & Hairsine 1998).

Lowrance et al. (1997) use existing information and their best professional judgement to provide expected levels of pollutant control by riparian forest buffer systems (RFBS) in 9 different physiographic provinces of Chesapeake Bay, USA. They stress the importance of the hydrologic connection between pollution sources, the buffer zone and the stream, stating that water quality improvements will be most likely in areas where most of the excess precipitation moves across, in, or near the root zone of riparian forest buffers. Several studies in this region had shown that high rates of nitrogen removal occurred in areas with high water table conditions and shallow groundwater movement near the root zone. In regions with deeper soils (i.e., aquiclude or bedrock 10-30 m below surface) or where water drained into aquifers or large rivers, the removal potential of RFBS were expected to be much less.

Topography

The effectiveness of grass buffer strips as filters for nutrients and sediment is less in steep hilly terrain than rolling land, as overland flow is concentrated in channelised natural drainage-ways giving rise to high flow velocities. As a result buffer effectiveness is minimal, or at best, patchy along the stream length.

Dosskey et al. (2002) studied four farms in Nebraska, USA, to develop a method for assessing the extent of concentrated flow in riparian buffers and for evaluating the impact that this has on sediment trapping efficiency. Riparian buffers averaged 9-35 m wide and 1.5-7.2 ha in area, but the effective buffer area that actually contacted runoff water was only 0.2-1.3 ha due to the patterns of topography preventing uniform distribution of runoff. Using mathematical relationships, it was estimated that between the four farms, buffers could theoretically remove 41-99% of sediment, but because of non-uniform distribution it was estimated that only 15-43% would actually be removed.

Grass buffers may need to extend further inland following a drainage way, resulting in a non-uniform buffer width along the length of the stream.

Longevity

There are indications that buffer zones may have a limited life span where they are effective. For example they may become saturated with P, pore spaces in soils may clog with sediments, or dissolved nutrient uptake by plants may be greatest during early growth phases and decline as vegetation matures. Some evidence of P saturation of a riparian zone was shown by Cooper et al. (1995) who studied riparian soils in native scrub (manuka), grazed pasture, and 12yr old retired pasture (tussock dominated) near Taupo, New Zealand. Retired pasture soils had extremely high hydraulic conductivity indicating that surface runoff water transported into the zone would infiltrate, fill soil pores and emerge as subsurface flow at the stream edge. The runoff that emerged from the buffer was depleted in sediment-bound nutrients and dissolved N, but enriched in dissolved P.

To optimise the long term value of riparian zones as nutrient filters, a number of strategies could be employed: (1) riparian retirement needs to be accompanied by improved land use practices over the broader landscape so that nutrient influx to the riparian zone is reduced. (2)

Periodic harvesting of plant material could ensure plant uptake remains a continued net nutrient removal mechanism (3) Buffer widths should be established on the basis that a sustainable nutrient removal can be achieved with regard to the nutrient influx it would receive.

Methods to remove P could include selective harvesting for wood or fruits as mentioned earlier, or in the case of grass buffers, light grazing with sheep for a short time during summer may be acceptable providing that temporary fences are used immediately beside the stream to keep stock out. Alternatively, the strip could be mown for haymaking.

Predictive tools for designing RBZ

Phillips (1989) employed mathematical models to estimate the relative importance of soil hydrologic properties, topography, and surface roughness in determining the effectiveness of water quality buffers in North Carolina, USA. He found that slope gradient was the most critical factor for effective removal of sediment or particulate pollutants transported in surface runoff. However, buffer width was by far the most important factor for effective removal of dissolved pollutants in surface or subsurface flow.

The effectiveness of buffers can be greatly affected by its design and site-specific factors such as slope, clay content of the soil, drainage patterns, etc. In 1995, DoC and NIWA published a set of guidelines (Collier et al. 1995) that provided practical measures to improve the design of RBZ to manage bank stability, light climate, water temperature, carbon supply, habitat diversity, flood flows, and contaminants. For contaminants, the guidelines can be used to calculate the optimal filter strip width for attenuating overland flow. These calculations were based on the modified CREAMS model (Chemical, Runoff, and Erosion from Agricultural Management Systems), and they require information on topography, slope, soil types for drainage and clay content categories, and hillslope length. Generally, buffer widths will need to widen as the slope length, angle and clay content of the adjacent land increase and as soil drainage decreases. For nitrate removal in subsurface flow, the guidelines recommend protection of existing riparian wetlands, based on their proven effectiveness for nitrate removal.

Herron & Hairsine (1998) used time independent equations to assess the effectiveness of riparian buffer zones in reducing overland flow to streams. Buffer widths, expressed as a proportion of total hillslope length, were calculated based on a variety of Australian rainfall environments and varying topographic convergence. From these scenario results, the authors proposed a riparian width not exceeding 20% of the hillslope length as a practical management option, although larger buffer widths may be required where riparian areas or slopes are degraded.

High rates of denitrification are known to occur in water-logged soils, anoxic conditions, and with a source of organic carbon. Recent research in the USA has used soil map data to identify area with wet soils as a planning tool for riparian management to enhance denitrification (Gold et al. 2001).

Table 1: Contaminant removal efficiencies from references within Castelle et al. (1994) review of U.S. vegetated buffers. VFS = vegetated filter strip.

Contaminant	Buffer width	Removal (%)	Slope (%)	Farm type	Buffer type	Reference
Sediment	30.5	90	2			Wong & McCuen (1982)
Sediment	61	95	2			Wong & McCuen (1982)
Sediment	24.4	92			Veg.	Young et al. (1980)
Sediment	22.9	33		dairy	Filter strip	Schellinger & Clausen (1992)
Sediment	61	80			Grassy swale	Horner & Mar (1982)
Sediment	30	75-80		Logging activity		Lynch et al. (1985)
Sediment	9.1	85	7 and 12		Grass VFS	Ghaffarzadeh et al. (1992)
NO3-N, NH4-N, PO4-P	4.6	90%			Grass VFS	Madison et al. (1992)
NO3-N, NH4-N, PO4-P	9.1	96-99.9			Grass VFS	Madison et al. (1992)
Sediment, N, P	9.1	84, 79, 73	11-16		Grass VFS	Dillaha et al. (1989)
Sediment, N, P	4.6	70, 61, 54	11-16		Grass VFS	Dillaha et al. (1989)
NO3-N	10	99.9%			forested	Xu et al. (1992)
N, P	19	89, 80			forested	Shisler et al. (1987)

Table 2: Some New Zealand studies of efficiency.

Contaminant	Buffer width	Removal (%)	Farm type	Buffer type	Reference
Nitrate	c. 3-4m	88-97	pasture	Riparian organic soils - wetland	Cooper 1990
Nitrate	c. 3-4m	0-62	pasture	Riparian mineral soils - wetland	Cooper 1990
Nitrate		–140-91	pasture	streambed	Cooper 1990
Nitrate		32-100%	Waste water treatment	wetland	Cooper 1994
Nitrate	10-13m	67	pasture	Retired pasture	Smith 1989
Dissolved P	10-13m	55	pasture	Retired pasture	Smith 1989
Particulate P, N	10-13m	80, 85	pasture	Retired pasture	Smith 1989
Total Suspended solids	10-13m	87	pasture	Retired pasture	Smith 1989

Table 3: Experimental studies of buffer widths required for sub-surface and surface nitrate removal (from Fennessy & Cronk 1997).

Flow type	Buffer type	Buffer width (m)	N retention (%)	N inflow (mg/l)
Subsurface	Forest	9	61-97	180
	Forest	10	70-90	13.5
	Forest	10	Up to 77	0.6-2.5
	Forest	19	93	7.4
	Forest	>20	90	7.4
	Forest	>20	99	6.8
	Forest	20	Up to 87	0.6-2.5
	Forest	25	68	~2-6
	Forest	26	~100	2-9
	Forest	30	~100	5
	Forest	>10-50	94	1.8
	Forest	50	95	8
	Forest	60	~100	10
	Herbaceous	22	84	2-12
Sub- and surface	Forest	16	90	10
Surface	Forest	19	60	4.5
	Forest	>20	79	4.5
	Herbaceous	5	54	-
	Herbaceous	8	20	20
	Herbaceous	9	73	-
	Herbaceous	16	50	20
	Herbaceous	27	84	-
	Herbaceous	30	11	20

3. Biodiversity

The key to improving biodiversity in streams and riparian zones is habitat diversity and connectivity with other habitats. The greatest improvements in habitat diversity are likely to occur when riparian management involves planted trees or remnant forest.

Riparian planting effects on stream habitat for aquatic biota include:

- provision of woody debris as trees fall into streams over the long term, providing habitat diversity and cover for aquatic invertebrates and fish;
- increased shade and provision of terrestrial food sources (fallen leaves etc.) as riparian plants grow so that levels of instream productivity and trophic pathways resemble the natural state;
- reduced erosion and inputs of fine sediment from (1) exclusion of livestock, leading to an improvement in streambed and bank habitat and (2) interception of hillslope sediment over the long term, and (3) tree roots that stabilise the stream banks and provide habitat;
- reduced water temperatures if sufficient lengths of upstream shade exist, and lower air temperatures and humidities, and less wind exposure, in the riparian zone where the adult stages of some aquatic insects spend part of their lives and some native fish lay their eggs (banded kokopu, short-jawed kokopu).

Lack of stream shade appeared to be the most important factor affecting invertebrate populations in Waikato hill-country streams (Quinn et al. 1997). Quinn et al. (1997) concluded that shade effects on algal biomass were a major cause of the lower abundance of some invertebrate groups, notably midge larvae, in some Waikato forested streams. Reduced water temperatures can also be expected with riparian planting, particularly if the planted buffer zones extend over several hundred metres of shallow stream systems. Many New Zealand stream invertebrates (e.g., mayflies, stoneflies) are sensitive to water temperatures >20ºC, temperatures that are commonly exceeded in open pasture streams. Rutherford et al. (1999, 2000) used computer models to show how high water temperatures can release periphyton from control by temperature-sensitive invertebrates, like mayflies, resulting in algal proliferations.

Quinn et al. (1997) found that ‘stream health’, as indicated by invertebrate communities, was similar in pine plantation streams to that in native streams (and very different from the pasture streams) in the Hakarimata Range – despite the sedimentation and turbidity in the pine plantation streams from bank erosion. This suggests that shading benefits outweigh the sedimentation side-effects associated with channel widening. The reduced inputs of fine suspended sediment expected over the long term following bank stabilisation may also improve conditions for migrating fish such as banded kokopu whose juvenile migrations are adversely affected when turbidity increases above 25 NTU (Richardson et al. 2001).

Riparian trees add leaf litter and wood that are an important source of habitat diversity for invertebrates and fish, particularly in silt-bed streams. Recent work has demonstrated that stable bank habitat and the presence of riparian tree roots penetrating into those banks creates habitat for freshwater crayfish (Parkyn & Collier in press). Field investigations of Auckland stream plantings aged from 10-30 years showed that woody debris from fallen branches, wind damage to plants, and unsuccessful plantings had begun to accumulate in small stream channels (pers. obs.).

Furthermore there has been increasing recognition recently of the role of riparian vegetation in creating suitable microclimate conditions for the adult stages of some stream insects. Collier & Smith (2000) reported that 50% of female stonefly adults died within 4 days at constant air temperatures of 22-23ºC. These temperatures were exceeded 25% of the time in January next to a Waikato pasture stream. Davies-Colley et al. (2000) found that at least 40 m of forest habitat next to pasture was required before air temperatures became comparable to those in a large block of native forest in the Waikato. However, narrower buffer zones can give significant temperature control. Air temperatures measured in a clear cut pine plantation within a 5 m buffer of well-established native vegetation on one side of a stream were similar to those in a 30 m buffer on the other side of a stream (John Quinn, pers. comm.). Daily maximum temperatures during summer were reduced from about 30ºC in the clear cut area to 25ºC in the buffer zones.

3.1 EFFECTIVENESS OF RIPARIAN MANAGEMENT FOR HABITAT AND BIOLOGICAL DIVERSITY

Parkyn et al. (2003) studied a number of riparian restoration schemes in the Waikato region to determine whether riparian management was achieving improvements in stream health. The sites were grouped according to the stream substrate or land topography, e.g., cobble/gravel substrate, lowland (silty substrate), pumice substrate. The buffer zones had been fenced to exclude stock and tree species had been planted (or remnant vegetation was present). The age of planting ranged from 'recent' (c. 2 years) to "mature" (>20 years) within each substrate/hydrological grouping. Each buffer zone was compared to an unfenced and actively grazed stream section upstream of the buffer zone or in a neighbouring stream when no upstream control was available. In general, streams in buffer zones showed rapid improvements in clarity, bank stability, and nutrient contamination. Often channel widths decreased in buffered reaches where the plantings were young, presumably from a reduction in trampling by stock.

However, significant changes to macroinvertebrate communities towards "clean water" or "native" communities did not occur at most of the sites over the time-scales that were measured in this study. The lack of improvement in QMCI scores and taxa richness may indicate (1) a lack of source areas of colonists, (2) lack of suitable microclimate for adult invertebrates, (3) time-scales of recovery are large, or (4) that buffers were not achieving habitat goals. However, one stream with a wide buffer of >50 m, 25 year old plantings, and the whole stream length planted did show significant improvement in invertebrate communities compared to a nearby pasture stream. Improvement in invertebrate communities appeared to be most strongly linked to decreases in temperature suggesting that restoration of in-stream communities would only occur after canopy closure and after protection of headwater tributaries. This was particularly evident in lowland streams where catchment influences had a greater impact than local riparian influences.

Quinn et al. (2004) studied the effect of native forest buffers within plantation forestry on stream invertebrate communities in the Coromandel Peninsula, New Zealand. Clearcut reaches had the lowest diversity and taxon richness of 28 stream sites, while sites that had been logged leaving continuous buffers did not differ from those in intact native or mature plantation forest, indicating that buffers greatly reduced disturbance associated with logging. Logging impacts were strongly related to increases in periphyton biomass, water temperature, fine sediment, and channel instability.

In North American streams, Weigel et al. (2000) found that the macroinvertebrate community response suggested higher organic pollution in continuously grazed sections compared to woody buffered sections. However, they also found that catchment differences produced greater overall differences in the invertebrate communities than between different grazing treatments along the same stream. This variability between streams is a common problem with interpretation of riparian buffer zone studies, and can mean that the same management technique can have variable outcomes in different stream systems (Belsky et al. 1999). Sovell et al. (2000) found that faecal coliforms and turbidity were greater at continuously grazed stream sections than at rotationally grazed sites. However, they were unable to show associated changes to the macroinvertebrate or fish communities.

Biodiversity in streams with riparian plantings may be affected by being in a "transitional" state. As plantings mature and shade is introduced and water quality changes occur, some species characteristic of pasture streams may be lost, while there is a time lag before the buffer zone matures or until connectivity of riparian patches allows recolonisation of native forest species to occur. Scarsbrook & Halliday (1999) found that aquatic invertebrate community composition had recovered fully within native forest patches that were about 60 years old, in hill-country pastoral catchments in the Waikato.

Collier et al. (2000) found that streams draining catchments entirely in pasture or native forest had similar percentages of total taxa. Biodiversity of open, pasture streams can equal that of shaded, forest streams at the reach scale. However, if you stand back and look at a larger catchment scale, biodiversity of the entire stream system would have been reduced. Deforestation of headwater streams has enabled species characteristic of more open conditions, which would have been present in the lower, wider reaches of stream systems where canopy closure was not possible, to establish themselves further upstream. Therefore the reduction in habitat diversity over the whole stream system has led to a homogenising of species diversity.

3.2 HABITAT IN LOWLAND STREAMS

A potential problem associated with riparian plantings shading out macrophytes in soft-bottomed lowland streams is that these macrophytes (particularly submergent species) can provide important stable substrates for invertebrate colonisation at certain times of the year (Collier 1995, Collier et al. 1999) and increase habitat heterogeneity through their influence on water velocities (Champion & Tanner 2000). The highest number of invertebrate taxa in a lowland stream south of Auckland was found in macrophyte patches with intermediate biomass leading to the recommendation that patchy shade conditions should be maintained in soft-bottomed streams to enable moderate quantities of submerged macrophytes to grow (Collier et al. 1999; Champion & Tanner 2000). In many lowland streams submerged wood can also provide an important stable habitat for invertebrates (Collier et al. 1998), but riparian plantings would not be expected to contribute considerable amounts of woody debris to streams for many years after planting. However, growth of trees large enough to shade lowland streams will also take some time resulting in low levels of shading for many years, and fallen branches and failed plantings or even plantings lost once channel widening has begun will accumulate in the streams, particularly once early successional trees become mature (e.g., manuka).

3.3 BUFFER WIDTH

In Australia, Davies & Nelson (1994) found that small buffers (<10 m wide), retained after forest harvesting, did not significantly protect streams from changes in algal, macroinvertebrate and fish biomass and diversity. Buffer widths of >30 m appeared to provide protection from short-term impacts in a variety of forest types and geomorphology. However stream temperatures were only increased when buffer widths were below 10 m. The buffer width required to decrease stream temperatures may be less than that required to provide a microclimate similar to forested conditions. A single line of trees can provide about 80% shade to streams when the trees have grown tall enough to achieve canopy closure (Collier et al. 1995). Five and 30m wide riparian buffers of native forest reduced the median daily maximum air temperatures by 3.25 and 3.42ºC, respectively compared with a clearcut area downstream of the site (Meleason & Quinn 2004), indicating that narrow buffers can maintain cool riparian air temperatures. The buffer widths of Coromandel forestry sites studied by Quinn et al. (2004) ranged from 8-27 m and supported stream invertebrate communities similar to those in native or mature plantation forest.

In a review of buffer width requirements for wildlife species distribution and diversity in the U.S., Castelle et al. (1994) found that many studies showed improvement in salmon, trout and benthic invertebrate communities with buffers of >30m. However, a number of habitat suitability models in the U.S. found that buffer widths could range between 3 and 107 m depending on the particular resource needs of individual species (Castelle et al. 1994). Brosofske et al. (1997) concluded that a buffer of at least 45 m was necessary to maintain a natural riparian microclimate after harvesting of Douglas fir and western hemlock. International studies of buffer width requirements for biodiversity may therefore be of limited value to New Zealand, as individual species will most likely differ in their requirements.

4. The need for an integrated catchment-wide perspective on riparian management

4.1 HYDROLOGIC PATHWAYS

An understanding of hydrological pathways is critical to determining buffer strip effectiveness. For example, the key nitrate pathway upon porous pumice soils in the central North Island of New Zealand is vertical, down to groundwater, which may take several decades to emerge into surface waters. In this case, the nitrate predominantly bypasses riparian vegetation and is difficult to mitigate using conventional riparian management (Howard-Williams and Pickmere 1999). Determining overall buffer effectiveness, therefore, not only requires an understanding of the attenuation efficiency with respect to nutrients washed into the buffer, but also quantification of the nutrient load that bypasses the buffer. Both pieces of information are required across the catchment to fully evaluate buffer effectiveness.

A further example of the importance of hydrological pathways is found on flat or gently sloping dairy land. Typically, these areas are underlain by artificial drains that feed a network of open drains discharging directly to streams. The subsurface drains reduce surface ponding and runoff by aiding infiltration. Studies within the Toenepi catchment, Waikato have shown that the N-load leaving the catchment outlet can be accounted for by the sum of all drainage inputs to the stream network (R. Wilcock-NIWA personal communication). In other words, the presence of riparian buffers at Toenepi would be ineffectual (other than preventing direct access to streams) at attenuating Nitrogen delivery to waterways.

Williamson et al. (1996) predicted that total phosphorus loads of the Lake Rotorua catchment had been reduced by 20% after implementation of the Upper Kaituna Catchment Control Scheme, in one of the few studies that has extrapolated specific yield estimations of nutrient and sediment reductions from retired riparian margins to a whole catchment estimation of whether riparian management was effective in improving water quality. The measures included tree plantings on erosion-prone hillslopes, preservation of wetlands and lake margins, and retirement and planting of stream riparian zones. The study took data from the Ngongotaha stream catchment , which showed reductions of 85% for sediment and 26-40% for nutrients and scaled up the findings to the whole catchment. The reduction in phosphorous loads was expected to reduce the chlorophyll a concentration by enough to shift the lake's trophic status from eutrophic to mesotrophic. The reduction in phosphorus was achieved by the improved land management, despite hydrological characteristics of the Ngongotaha catchment where springs from deep ground water contain naturally high levels of soluble P. However, dissolved N concentrations were higher after control measures were implemented, partly due to an increase in nitrate concentrations in the deep groundwater bypassing riparian zones.

4.2 CHANGING FUNCTIONS WITH STREAM SIZE

In a catchment context, it has been suggested that maximum water quality benefits will occur if buffer strips are located along headwater reaches, partly because most of the water in a catchment originates in the headwaters (Fennessy & Cronk 1997). Many small wetlands are distributed throughout the upper reaches of catchments, providing denitrification and uptake of soluble pollutants. Small streams are intimately linked to their riparian zones and riparian buffers in these systems will achieve many of the benefits of shading and nutrient filtering. As streams get larger, e.g., rivers, their main interaction with the riparian zone are when flood waters over top the banks. Riparian vegetation in buffers is then more useful for slowing flood flows, rather than shading or filtering functions. The spatial pattern of riparian planting has, therefore, a clear influence upon overall buffer effectiveness.

4.3 SHADE IMPACTS ON NUTRIENT REMOVAL AND SEDIMENT

Streams convert inorganic nutrients (nitrogen and phosphorus) to instream plant biomass under stable flow conditions. Given the same nutrient inputs, a shaded stream can be expected to retain less nutrient as plant biomass than an unshaded stream. Thus, as noted by Rutherford et al. (1999), restoration of shade through riparian planting can change the way streams transform and process nutrients, and lead to increased transport of inorganic nutrients downstream. In streams dominated by macrophytes (i.e., where their biomass is much greater than that of algae) it would be reasonable to infer that they will have a much greater influence on nutrient removal than algae (B. Wilcock, NIWA, pers. comm.). Typically, plant uptake of N varies with stream size; headwater streams are better processes of N than larger channels. Consequently, the impact of shading will vary with stream size, supporting the need for a catchment wide holistic approach to riparian management.

Pasture streams in New Zealand typically display marked seasonal changes in dissolved nutrient concentrations that reflect seasonal growth of the streambank and aquatic vegetation (Howard-Williams et al. 1986). A long-term study of Whangamata Stream draining into Lake Taupo has clearly demonstrated how dissolved nutrient levels can fluctuate in response to changes in instream plant biomass as riparian plantings grow (Howard-Williams & Pickmere 1994, 1999). This study recognised 3 phases in changes to water quality over 24 years following riparian planting:

Years 1-5 – an initial moderate decline in dissolved nutrients (30-50% for NO3-N and 10-60% for DRP) for 1-2 months in summer as channel vegetation increased (mainly watercress).

Years 5-13 – very high dissolved nutrient removal (up to 100% for NO3-N and DRP) for 4-5 months of the year due to the proliferation of plants that do not die back in winter (mainly monkey musk).

Years 13-24 – decreasing nutrient removal capacity as increased levels of shade limited the biomass of light-requiring plants.

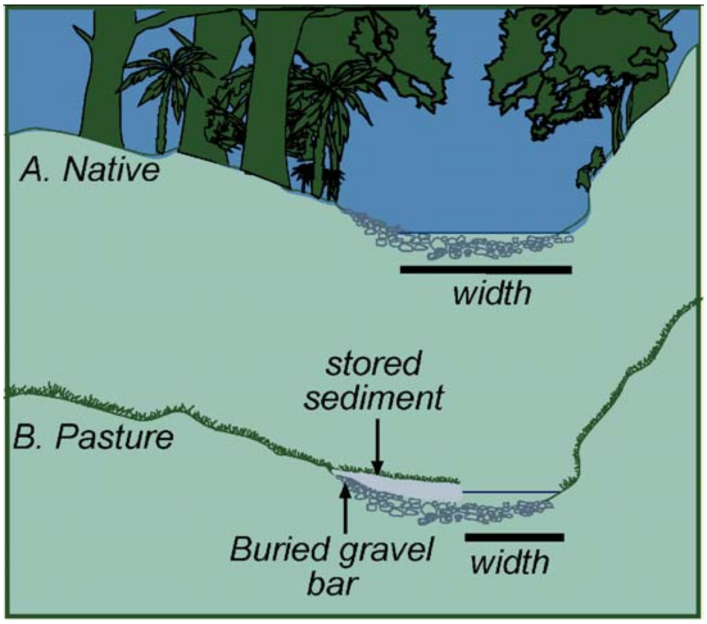
Davies-Colley (1997) found that 2nd order Waikato streams were wider in native forest than in pasture catchments, and that the streams formerly in pasture catchments that were now covered in mature pine plantations had actively eroding streambanks (Fig. 5). This observation raised the concern that riparian planting along pasture streams could lead to the mobilisation of stored sediment if stabilising streamside grasses are shaded out, and that if this occurs there could be a period of increased water turbidity, streambed sedimentation and sediment export until the channels reach a new equilibrium. However, it is expected that the ultimate (forest) channel width will be much more stable than under pasture where appreciable bank erosion occurs during floods.

The findings of Davies-Colley & Quinn (1998), who compared stream widths and light climates for a range of streams throughout the northern North Island, provide wider-scale qualified corroboration of the phenomenon of stream narrowing in pasture catchments. Davies-Colley (1997) reviewed several studies from overseas which suggest that the phenomenon of channel narrowing in pasture is a general feature of formerly forested stream channels.

Collier et al. (2001) estimated that the total mass of sediment stored in streambanks in the 250 ha Mangaotama catchment near Hamilton is about 13,000 tonnes, equivalent to around 21 years of current annual sediment yield (assumed to be from hillslope sources). Forecasts of the mass of sediment exported following riparian planting in this catchment (assuming eventual doubling in channel cross-sectional areas consistent with Davies-Colley 1997) suggest that, over a 25-year timescale, there would be an increase in sediment yield compared to the status quo as stream channels widened in response to shaded conditions. Over the longer term, however, once this stored bank sediment has been exported, banks are expected to stabilise as channels reach new steady-state ‘forest’ morphology and sediment yield will eventually decline to a lower level than currently experienced.

Figure 5: Change in stream channel width from native forest (A) to pasture (B), where pasture grasses trap sediment resulting in narrow and incised channels. (see Davies-Colley 1997).

The significance of these studies for riparian management are that if a whole catchment



perspective is not considered when implementing forested buffers, then shade may cause problems downstream if there are estuaries or lakes sensitive to sediment and nutrients.
In a conceptual modelling exercise, Parkyn et al. (unpubl. data) investigated concerns that riparian management could lead to increased yields of nutrients and sediments. A simple model of the trade-off between interception of nutrients in runoff by forest buffers versus reduction of in-stream uptake due to shade, predicted that a buffer strip alongside a small headwater stream would reduce nutrient export, while a buffer strip instigated as an isolated patch alongside a larger stream (> c. 2.5 km^2 upstream catchment size) would increase nutrient export, as the relative amount of nutrients trapped by the buffer decreases as the nutrient load present in the stream water increases. However, in these larger streams with width exceeding approx. 6 m, sufficient light may reach the streambed for plant and algal growth, which in turn would promote instream nutrient processing. At the peak of streambank erosion after planting, predicted total sediment yield (hillslope plus bank sources) was appreciably higher than the hillslope pasture yield, but sediment yield stabilised c. 35-40 years after planting. When planting was extended over 40 years in the model, the sediment yield never exceeded that in pasture before planting. This conceptual modelling exercise highlighted the problems associated with implementing riparian tree planting programmes in a piecemeal fashion and concluded that planting should commence in the headwaters and progress downstream to avoid nutrient yield increases. To avoid or reduce peak losses of sediment downstream, riparian planting would need to be implemented slowly or some riparian management options such as grass filter strips or spaced plantings may need to be investigated if downstream environments are sensitive to sedimentation.

5. Summary

Riparian management can be viewed as a last line of defence for attenuating contaminants before entering the stream. Fencing stock out of streams and retiring riparian margins from agricultural land use are also particularly important practices to improve stream water quality. Buffer zones can filter contaminants and sediments from overland flow by increasing the infiltration into soil, intercepting particulates, and removing soluble nutrients by plant uptake and denitrification.

Riparian management can take various forms, summarised below:

- **Grass Filter Strips:** Fenced strip of rank paddock grasses to filter nutrients and sediment.
- **Headwater or riparian wetlands**: Fenced wetlands as hotspots for nutrient removal.
- **Rotational grazing**: Filter strips with varied stock grazing practices, such as occasional light grazing by sheep.
- **Forested or planted native trees**: a buffer of native trees to return ecological function to the stream and provide water quality benefits.
- **Production trees or plants**: a buffer of forestry trees left unharvested along stream banks, or production trees that are planted in riparian zones for selective harvesting with minimal disturbance (e.g., Tasmanian blackwoods). Plants such as flax for weaving, or fruit and nut trees, or high value native tree species that can be selectively harvested may also provide ecological function and a mechanism to remove nutrients such as phosphorus from the riparian zone.
- **Multi-tier system**: a combination of buffers where native forest trees may be used beside the stream to enhance ecological function and biodiversity, a buffer of production trees may occur at outside of that and the outer edge beside agricultural land would be a grass filter strip.

The ability of buffer zones to attenuate pollutants will depend upon the mechanisms by which these pollutants reach surface waters. Three main transport processes can occur:

- direct pollution (e.g., stock access to streams, bank erosion);
- surface runoff;
- subsurface flow and drainage.

Buffer zones can be effective at removing nutrient and sediment inputs to streams by restricting the direct use of land beside the stream and by processing water that has been transported into the riparian zone. The mechanisms of contaminant removal in buffer zones differ according to characteristics of the hydrology, soils, and vegetation as well as the mode of transport to streams.

Buffer zones where stock have been excluded and where long grass or natural vegetation has been allowed to develop, or been planted, can reduce diffuse pollutant transport from agricultural land by:

- direct removal of stock trampling and faecal inputs;
- enhanced infiltration by riparian soils which reduces surface runoff thereby aiding the deposition of particulates (sediment and particulate nutrients);
- reduction of surface flow velocities from increased hydraulic roughness of the vegetation in the buffer (sediment and particulate nutrients);
- physical filtering effect of dense vegetation (sediment and particulate nutrients).
- denitrification (dissolved N);
- plant uptake (dissolved nutrients).

The consensus in the literature is that grass buffer strips are effective at filtering sediment and sediment-associated pollutants (particulate P and N) from surface runoff. However they are less effective in removing soluble nutrients such as nitrate, ammonia, and dissolved P. Nitrate removal from subsurface flows is considered to be greater in forested buffers, partly through uptake by plants (Fennessy & Cronk 1997, Martin et al. 1999). However, the main mechanism by which nitrate is removed from groundwater is thought to be biological denitrification (a microbial process whereby nitrate is converted to gaseous forms of nitrogen and returned to the atmosphere).

What type of buffer and how wide for optimal nutrient and sediment attenuation?

Studies comparing multiple width buffers in the same location have shown that sediment and total phosphorus removal rates (between 53 and 98%) increase with increasing buffer width (4.6 m to 27m). Many researchers report substantial sediment removal within a few metres of the upslope boundary (Barling & Moore 1994, Fennessy & Cronk 1997). Grass filter strips in particular have been shown to be very effective at trapping sediment particles. Much of the larger particles of sediment may be removed in 5 m of grass buffer, but finer particles may require up to 10 m (Gharabaghi et al. 2002).

The width required to optimise nutrient removal has been debated with little systematic study of the issue. Fennessy and Cronk (1997) reviewed studies of RBZ effectiveness for the removal of contaminants, particularly soluble nitrate: Nitrate removal rates of almost 100% were measured in buffers 20-30 m wide, while buffers of 10 m width achieved over 70% retention of N. Many of the buffers in this study were forested, and N uptake by plants and denitrification were believed to have been an important factor in removing soluble N. Saturated riparian wetlands have been shown to be highly effective at attenuating (denitrifying) N, although this process is strongly dependent upon hydrological residence time.

Because of the different modes of particulate and dissolved contaminant transport, multi-tier or combination buffers are often advocated. For water quality benefits, a narrow combination buffer consisting of 5 m of grass filter strip and a 1 m wide row of deciduous trees has been shown to reduce nitrate in subsurface flows beneath cropland in Italy (Borin & Bigon 2002). The single row of trees may also provide some shade to the stream, but is unlikely to achieve terrestrial or aquatic habitat benefits.

Combination buffer systems in the USA often consist of an upslope grass buffer, a managed forest zone and an undisturbed forest zone next to the stream. Hubbard & Lowrance (1997) studied the nitrate removal from shallow groundwater where the forest zone was either mature forest, clear cut, or selectively thinned. All three forest management treatments were effective in assimilating nitrate and there were no differences between treatments.

Harvesting production trees or plants, or fruit and nuts from trees in riparian zones can provide a mechanism where P can be removed from the riparian zone. Phosphorus accumulates in riparian soils and can be taken up by plants but there is no process similar to denitrification that removes P to the atmosphere. Therefore, buffer zones could potentially become saturated and their ability to trap P may decline with age unless sediments or organic matter are removed from the buffer zone (Barling & Moore 1994). Examples in addition to production forestry include indigenous systems of tropical agroforestry where non-timber products (fruits, nuts and ornamentals) can be harvested (Robles-Diaz-de-León & Kangas 1999). There may be scope in New Zealand to use riparian buffers as zones for flax harvesting, medicinal plant growth, manuka honey, etc.

The optimal width required for nutrient and sediment removal can be highly variable and Auckland Regional Council have suggested an alternative approach based on the width needed to develop a self-sustaining buffer of native vegetation. Parkyn et al. (2000) recommended a buffer width of 10-20 m as the minimum necessary for the development of sustainable indigenous vegetation with minimal weed control, and to achieve many aquatic functions.

Other factors affecting buffer zone effectiveness for nutrient and sediment attenuation

The effectiveness of grass buffer strips as filters for nutrients and sediment is less in steep hilly terrain than rolling land, as overland flow is concentrated in channelised natural drainage-ways giving rise to high flow velocities. As a result buffer effectiveness is minimal, or at best, patchy along the stream length. Grass buffers may need to extend further inland following a drainage way, resulting in a non-uniform buffer width along the length of the stream. Similarly, many review articles of buffer zone studies conclude that buffers need to be wider when the slope is steep, generally to give more time for the velocity of surface runoff to decrease (Barling & Moore 1994, Collier et al. 1995).

Soil drainage properties can influence RBZ performance. Free draining soils minimise the generation of surface runoff, both on the hillside and within a buffer, thus reducing sediment and particulate nutrient delivery to the buffer. In regions with deeper soils (i.e., aquiclude or bedrock 10-30 m below surface) or where water drains into aquifers or large rivers, the removal potential of RBZ is expected to be low for soluble nutrients, as the subsurface hydrological pathways may bypass the root zone of buffers (i.e., zone of uptake and denitrification). Artificial subsurface drainage can also bypass the riparian zone and deliver nutrients directly to streams.

Buffer zones may have a limited life span where they can continue to be effective for contaminant removal. For example they may become saturated with P, pore spaces in soils may clog with sediments, or dissolved nutrient uptake by plants may be greatest during early growth phases and decline as vegetation matures. Some researchers suggest that these factors need to be taken into account and widths may need to be larger than early stage studies suggest. Methods to remove P could include selective harvesting for wood or fruits as mentioned earlier, or in the case of grass buffers, light grazing with sheep for a short time during summer may be acceptable providing that temporary fences are used to keep stock out of the stream. Alternatively, the strip could be mown for haymaking.

Because the effectiveness of buffers can be greatly affected by design and site-specific factors such as slope, clay content of the soil, drainage patterns, etc. DoC and NIWA published a set of guidelines (Collier et al. 1995) that provided practical measures to improve the design of RBZ to manage bank stability, light climate, water temperature, carbon supply, habitat diversity, flood flows, and contaminants. For contaminants, the guidelines can be used to calculate the optimal filter strip width for attenuating overland flow. These calculations were based on the modified CREAMS model (Chemical, Runoff, and Erosion from Agricultural Management Systems), and they require information on topography, slope, soil types for drainage and clay content categories, and hillslope length. Generally, buffer widths will need to widen as the slope length, angle and clay content of the adjacent land increase and as soil drainage decreases. For nitrate removal in subsurface flow, the guidelines recommend protection of existing riparian wetlands, based on their proven effectiveness for nitrate removal.

Biodiversity

The key to improving biodiversity in streams and riparian zones is habitat diversity and connectivity. The greatest improvements in habitat diversity are likely to occur when riparian management involves planted trees or remnant forest.

Riparian planting effects on stream habitat for aquatic biota include:

- provision of woody debris as trees fall into streams over the long term, providing habitat diversity and cover for aquatic invertebrates and fish;
- increased shade and provision of terrestrial food sources (fallen leaves etc.) as riparian plants grow so that levels of instream productivity and trophic pathways resemble the natural state;
- reduced erosion and inputs of fine sediment from (1) exclusion of livestock, leading to an improvement in streambed and bank habitat and (2) interception of hillslope sediment over the long term, and (3) tree roots that stabilise the stream banks and provide habitat;
- reduced water temperatures if sufficient lengths of upstream shade exist, and lower air temperatures and humidities, and less wind exposure, in the riparian zone where the adult stages of some aquatic insects spend part of their lives and some native fish lay their eggs (banded kokopu, short-jawed kokopu).

The buffer width required to achieve improvements in aquatic biodiversity is uncertain and variable between studies. Few studies have the luxury of experimentally testing mature buffer widths (i.e., with replication and under similar physical conditions), rather it is a case of looking at whatever existing buffers are available.

In Australia, Davies & Nelson (1994) found that small buffers (<10 m wide), retained after forest harvesting, did not significantly protect streams from changes in algal, macroinvertebrate and fish biomass and diversity. Buffer widths of >30 m appeared to provide protection from short-term impacts in a variety of forest types and geomorphology. However stream temperatures were only increased when buffer widths were below 10 m. The buffer width required to decrease stream temperatures may be less than that required to provide a microclimate similar to forested conditions. A single line of trees can provide about 80% shade to streams when the trees have grown tall enough to achieve canopy closure (Collier et al. 1995). Five and 30m wide riparian buffers of native forest reduced the median daily maximum air temperatures by 3.25 and 3.42ºC, respectively compared with a clearcut area downstream of the site (Meleason & Quinn 2004), indicating that narrow buffers can maintain cool riparian air temperatures. The buffer widths of Coromandel forestry sites studied by Quinn et al. 2004) ranged from 8-27m and supported stream invertebrate communities similar to those in native or mature plantation forest.

Parkyn et al. (2003) studied a number of riparian restoration schemes in the Waikato region to determine whether riparian management was achieving improvements in stream health. Significant changes to macroinvertebrate communities towards “clean water” or “native” communities did not occur at most of the sites over the time-scales that were measured in this study. The lack of improvement in QMCI scores and taxa richness may indicate (1) a lack of source areas of colonists, (2) lack of suitable microclimate for adult invertebrates, (3) time-scales of recovery are large, or (4) that buffers were not achieving habitat goals. However, one stream with a wide buffer of > 50 m, 25 year old plantings, and the whole stream length planted did show significant improvement in invertebrate communities compared to a nearby pasture stream. Improvement in invertebrate communities appeared to be most strongly linked to decreases in temperature suggesting that restoration of in-stream communities would only occur after canopy closure and after protection of headwater tributaries. This was particularly evident in lowland streams where catchment influences had a greater impact than local riparian influences.

Biodiversity in streams with riparian plantings may be affected by being in a “transitional” state. As plantings mature and shade is introduced and water quality changes occur, some species characteristic of pasture streams may be lost, while there is a time lag before the buffer zone matures or until connectivity of riparian patches allows recolonisation of native forest species to occur.

Catchment scale issues

Plantings, especially through provision of shade, aim to restore the ecological function of streams. However, shade can result in a widening of stream channels with a subsequent loss of sediment downstream, and also reduce nutrient attenuation within a given stream reach as instream plants are shaded out. These issues may become a problem when riparian management is implemented in a piecemeal fashion and where there are sensitive lakes or estuaries downstream. It is therefore important that the linkages within the whole catchment are considered when designing riparian management schemes and best management practice in most cases would be to begin planting from the headwaters and continue downstream.

The success of riparian management in terms of biodiversity can also be linked to the sources of recolonists within the wider catchment. Often remnant blocks of native forest exist in headwaters, so planting from the headwaters down the catchment would also increase the chances of recolonisation and improved biodiversity. An understanding of the connection between patches and dispersal potential of biota would also aid predictions of biodiversity improvements and help avoid unrealistic expectations.

An understanding of hydrological pathways is critical to determining buffer strip effectiveness. For example, the key nitrate pathway upon porous pumice soils in the central North Island of New Zealand is vertically down to groundwater that may take several decades to emerge into surface waters. In this example, the nitrate predominantly bypasses riparian vegetation and is difficult to mitigate using conventional riparian management (Howard-Williams and Pickmere 1999).

Because of the link between streams and their catchments, improved land management together with riparian management are required to achieve improvements in water quality and stream habitat. Examples of improved land management include: avoiding overstocking and pugging of soils, retiring steep and erosion-prone land, protecting wetlands which are sites of denitrification, diverting road and track runoff which can be a concentrated source of effluent and sediments, ploughing in directions parallel to the stream, and avoiding fertiliser application directly to streams or when the water table is high or heavy rain is likely. There will also be many other land use specific practices that will be important to consider in conjunction with riparian management.

Research for the future will be most effective if it addresses catchment scale issues such as these. Riparian management options will need to be designed with the hydrological pathways, soil drainage, and topography of the catchment in mind and targeted to areas where the most benefit can be achieved. Assessment of catchment and regional hydrology, such as soil drainage profiles and mapping of wetlands as hotspots for denitrification, as well as grouping streams and riparian zones into classes according to their potential effectiveness for water quality and biodiversity goals, are approaches that could assist resource managers with these issues.

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Appendix 1

SUMMARY OF RIPARIAN ZONE FUNCTIONS

Key riparian zone functions	Explanatory notes
Stream bank stability	The root systems of trees and grasses strengthen streambanks and groundcover reduces surface erosion – provides habitat stability in the form of refuges during floods.
Filtering overland flow	Surface roughness provided by grassy vegetation, or litter, reduces the velocity of overland flow, enhancing settling of particles. High infiltration of uncompacted soils encourages subsurface flowpaths, with resulting particulate filtering and nutrient uptake by plants and microbes.
Fish spawning habitat and fish cover	Inanga spawn amongst herbs and grasses near the upper edge of the salt wedge (usually Jan-May). Tree roots, overhanging branches and woody debris provide key habitat (hiding & resting places) for a wide variety of fish and for crayfish.
Suitable habitat for adult phases of stream insects	Some stream insects spend extended periods (weeks – months) as adults in the terrestrial area. Riparian vegetation may be a key element of these species ability to persist in pastoral streams. (e.g., humidity, temperature, food resources)
Shade for stream temperature	Removal of shade can result in summer temperatures that can be lethal to some invertebrates and fish, or winter temperatures that are too warm for successful trout spawning.
Shade for instream plant control	Shade removal provides light for instream plant growth, sometimes resulting in streams becoming choked and/or variations in dissolved oxygen and pH that stress invertebrates and fish.
Woody debris and leaf litter input	Riparian trees add leaf litter and wood that are an important source of habitat diversity for invertebrates and fish, particularly in silt-bed streams. Leaf litter is also a food resource for stream invertebrates.
Plant nutrient uptake from groundwater	Roots of riparian plants intercept groundwater reducing nutrient input to streams.
Denitrification N Control	Denitrifying bacteria can remove substantial quantities of nitrate from groundwater passing through riparian wetlands, venting this to the atmosphere as nitrogen gases.
Control of direct animal waste input	Preventing direct access of stock to waterways prevents hoof-damage to streambanks and direct input of nutrients, organic matter and pathogens in dung and urine.
Downstream flood control	Well-developed riparian vegetation increases the roughness of stream margins, slowing down flood-flows. This reduces the peak flows downstream but may result in some local flooding. Riparian wetlands provide temporary storage of water during rain events.
Terrestrial biodiversity	Riparian zones contain a high diversity of soil and water conditions, resulting in correspondingly diverse terrestrial plant and animal communities