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REVIEW ARTICLE

Riparian management and stream bank erosion in New Zealand

AO Hughes

National Institute of Water and Atmospheric Research, Hamilton, New Zealand

ABSTRACT
Improvements in riparian management, such as shrub/tree planting and livestock exclusion, are often assumed to result in reduced stream bank erosion and associated catchment sediment yield. Studies that quantify the effectiveness of riparian interventions aimed at reducing bank erosion and river sediment yields are, however, rare. This paper discusses how bank erosion processes can vary throughout catchments (with particular reference to their scale dependence) and hence how the effectiveness of different riparian interventions can be variable. The findings of known published accounts of the effectiveness of riparian management interventions for reducing stream bank erosion in New Zealand are also summarised. Only nine relevant studies were identified and most used qualitative or semi-quantitative analysis methods. Most studies compared stream banks in pasture catchments (with unlimited livestock access) with stream banks where livestock were excluded and riparian shrubs/trees were present. Many studies reported that managed stream banks were in better condition than unmanaged banks. The exclusion of livestock from riparian areas was generally reported as the principal factor in the measured improvements or differences. Only two studies specifically attributed reduced stream bank erosion to the presence of riparian vegetation. The dearth of research identified here highlights the need for further quantitative studies to determine the effectiveness of riparian management measures for reducing bank erosion.

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Introduction

Stream bank erosion can be an important source of sediment with some international studies estimating that up to 90% of a catchment's sediment yield is derived from channel sources (Caitcheon et al. 2012; Kronvang et al. 2013; Olley et al. 2013). Stream bank erosion has also been identified as an important process in New Zealand (Basher et al. 2012), although there has been very little quantitative research carried out to date (Basher 2013).

Stream bank erosion is generally portrayed as a negative characteristic of rivers, however, it is a natural process that may occur as a result of changes in flow regime and sediment supply, driven by processes such as climate change or natural catchment

CONTACT AO Hughes andrew.hughes@niwa.co.nz

disturbances. Indeed, bank erosion is seen as an important and necessary process as it may encourage riparian vegetation succession and create dynamic habitats that are critical for aquatic and riparian flora and fauna (Florsheim et al. 2008). In New Zealand, recent human-induced catchment disturbance has probably resulted in increased rates of catchment erosion (including stream bank erosion), deposition and sediment yield (Page & Trustrum 1997; Glade 2003). Stream bank erosion rates may have increased in response to: (1) changes in catchment hydrology due to the clearance of natural land cover and resultant response of channels to new, more flashy, regimes; (2) removal of natural riparian vegetation; (3) direct channel modification (e.g. channel straightening) and resulting increased stream power (due to increased channel slope); and (4) introduction of livestock (e.g. cattle, sheep, deer) to catchments and their unrestricted access to streams.

In catchments where stream bank erosion is perceived to be a problem, intervention measures focusing on riparian areas are often implemented by catchment managers. Such intervention measures can be categorised as 'hard' (e.g. rock gabion, rip rap) or 'soft' (e.g. livestock exclusion, riparian planting) management solutions. Most regional councils in New Zealand use a combination of hard and soft riparian management solutions (Phillips & Daly 2008). Hard solutions are likely to be very effective when designed well and are not undermined by channel incision. Due to the replacement of readily erodible surfaces with engineered hard surfaces, sediment derived from bank erosion in engineered reaches can be reduced to zero or negligible amounts. Such solutions do, however, tend to have relatively high costs and are both unsuitable and undesirable as a catchment-wide solution.

The rehabilitation objectives of soft riparian management solutions are often broader than just stream bank stabilisation and reduced sediment yield. The impacts of riparian planting and livestock exclusion on factors such as water quality and ecosystem health have been assessed in a number of environments (e.g. Lee et al. 2003; McKergow et al. 2003; Parkyn et al. 2003; Anbumozhi et al. 2005). While such riparian interventions have been shown to be beneficial tools, their effectiveness can be site dependent (McKergow et al. 2003, 2004; Parkyn et al. 2003).

With soft riparian solutions there is often an assumption that planting riparian areas and/or excluding grazing animals will result in lower rates of bank erosion. This assumption is based on observations that: (1) livestock trampling and foraging of riparian vegetation increase erosion susceptibility (e.g. Trimble 1994; Evans 1998); and (2) bank erosion rates are lower in forested streams than in comparable grazed pasture streams (e.g. Beeson & Doyle 1995; Stott 1997; Collier & Quinn 2003; Micheli et al. 2004). In the case of riparian vegetation, higher bank erosion rates of pasture streams may be the case for streams that are in a state of geomorphic equilibrium (i.e. are relatively stable and are not responding to a large-scale disturbance). However, the change in physical conditions that result from the planting of riparian areas could also be considered a form of 'disturbance', to which the fluvial system may respond.

This paper summarises the findings of the New Zealand-based studies that have attempted to measure the differences in erosion rates between sites that have had some form of riparian management intervention implemented and those that have not. The importance of understanding stream bank processes if planned riparian management interventions are to effectively reduce erosion rates and/or sediment yield are also discussed.

Scale dependence of stream bank erosion processes and riparian management

Within catchments a number of different erosion processes act on stream banks (e.g. mass failure, fluvial entrainment and preparatory processes). For riparian management interventions (particularly riparian planting) to be effective within a particular reach there needs to be an appreciation of the processes that act upon that reach. Previous international research has shown that there tends to be a scale dependence of bank erosion processes in catchments (e.g. Lawler 1995; Abernethy & Rutherfurd 1998) as follows:

- In headwater streams where stream power and banks heights are low, bank 'preparatory' processes (also known as sub-aerial processes; e.g. drying of banks, freeze–thaw, micro-rill development and livestock trampling) dominate.
- In the mid-reaches of a catchment, stream power is higher and fluvial entrainment (e.g. scour) dominates.
- In the lower reaches of catchments where bank heights are greatest, mass-failure mechanisms (e.g. bank slumping) dominate.

While all three processes (i.e. preparatory, fluvial entrainment and mass failure) can occur throughout a catchment (or indeed, at a specific site), it is their relative importance as an erosion source that is argued to differ with catchment scale (Figure 1). This scale-dependence concept provides a useful framework to consider the potential effectiveness of riparian management interventions. For example, in the lower reaches of a catchment (where mass-failure mechanisms are likely to be of most importance) removal of livestock from riparian areas and/or the planting of small shrubs and trees are unlikely to have a significant effect on reducing the contribution of sediment from these reaches. This does not mean that such interventions will have no effect, but rather they are unlikely to have much impact on reducing sediment derived from the dominant erosion

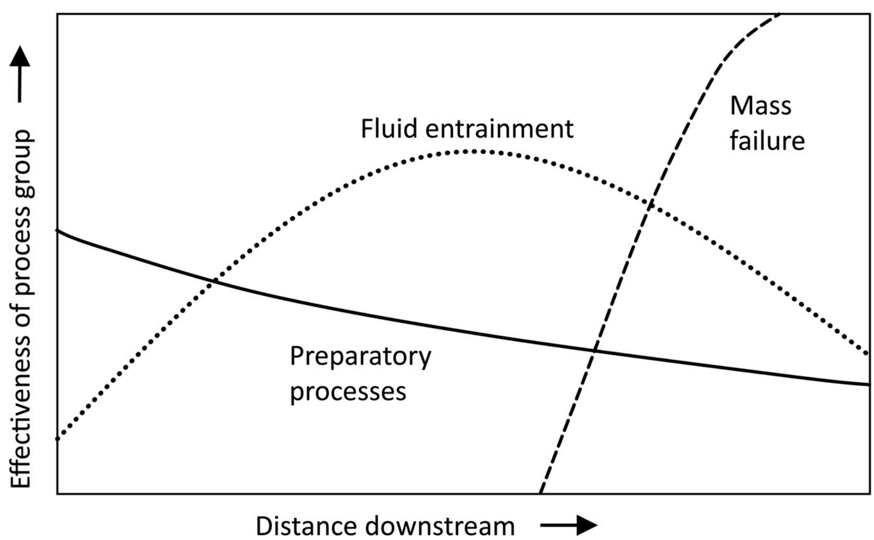

Figure 1. Conceptual model of downstream change in bank erosion process groups (from Lawler 1995).

process. Where mass-failure mechanisms dominate, an intervention measure that provides some structural support to the banks (such as the planting of large-growing tree species) would be more appropriate. However, even when large-growing tree species are used it is unlikely that such an intervention will have a measurable protective effect on stream banks until the trees develop deep root systems (Abernethy & Rutherfurd 1998).

Also related to this issue of catchment scale and process dominance is the potential impact of planting riparian vegetation within headwater catchments that have been used for pastoral-based land uses. Research from both New Zealand (e.g. Davies-Colley 1997; Chappell & Brierley 2013) and overseas (e.g. Murgatroyd & Ternan 1983; Trimble 1997; McBride et al. 2010) indicate that in headwater streams, channels (for a given catchment area) are narrower in pasture catchments than forested catchments. This phenomenon was effectively illustrated by Davies-Colley (1997), who measured the mean widths of 20 paired pasture reaches and 20 forested reaches within headwater streams in the Waikato region (Figure 2). These studies attributed the occurrence of wider channels in forested catchments to: (1) the suppression of protective groundcover vegetation within the riparian areas of stream channels with a closed canopy (Murgatroyd & Ternan 1983; Davies-Colley 1997); and (2) local stream bank erosion resulting from flow deflection by large woody debris (LWD) (Murgatroyd & Ternan 1983; Trimble 1997). Although both of these factors may play a role, when riparian areas are planted with shrubs/tree there is likely to be a time lag effect as: (1) it takes some years, perhaps decades, depending on stream size (e.g. Davies-Colley & Quinn 1998) before riparian vegetation is large enough to effectively shade out the groundcover vegetation; and (2) a significant volume of LWD of effective size will only be supplied to streams after many decades or even centuries of growth (Meleason & Hall 2005; Davies-Colley et al. 2009). The suppression of groundcover vegetation is particularly important in

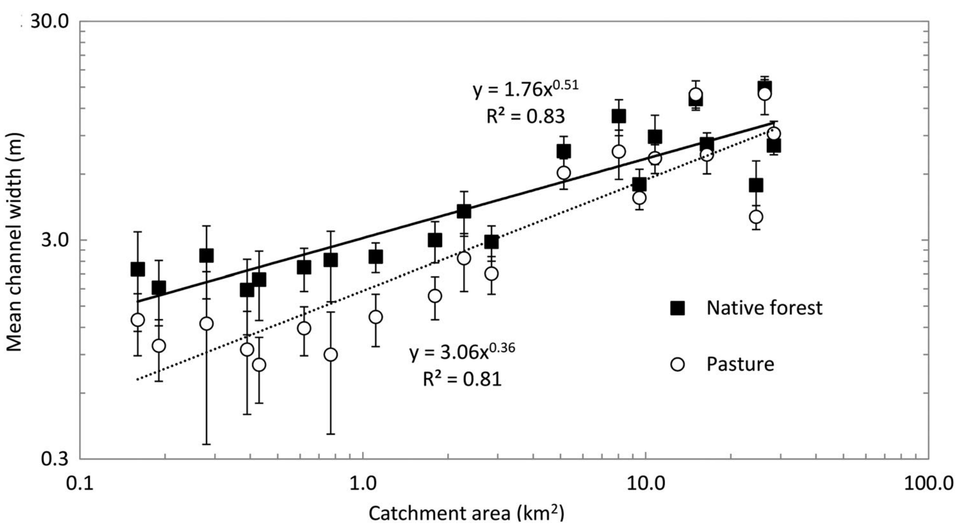

Figure 2. Stream channel width versus catchment area for paired native forest and pasture reaches for 20 headwater streams within the Hakarimata Range, Waikato region (from Davies-Colley 1997). Solid and dashed lines are the power law regression fits for the native forest and pasture sites, respectively. The vertical bars are standard deviations of stream width ($n = 20$ cross-sections).

many headwater streams as it is likely to result in an increased susceptibility to the preparatory processes described above (which may be the most effective erosion process). In a long-term monitoring study at the Whatawhata Research Station in the Waikato region, Hughes et al. (2012) attributed the continued dominance of near-channel sources to preparatory processes 8 years after extensive riparian vegetation planting. The implication of this is that the establishment of riparian vegetation in pastoral catchments may in some circumstances result in channel widening and hence increased sediment input, at least during the phase (of perhaps a few decades) needed for 'adjustment' to a new forest-shaded morphology.

A study by Parkyn et al. (2005) used knowledge of the difference in widths of forested and pasture headwater streams to estimate that an increase in channel widths in response to riparian planting of streams in the Auckland region could result in the release of 940 tonnes of sediment per km of river length. They predicted that, due to the increasing importance of groundcover suppression by canopy shading and the introduction of LWD to the river system, bank erosion would peak c. 25 years after planting. They also argued that bank erosion rates would eventually decline (below pre-riparian planting levels) as the river channel approached a new forest-shaded morphology. Working at the Whatawhata Research Station in the Waikato region, Collier et al. (2001) also proposed that stream bank shading by riparian planting of headwater pastoral channels would result in increased sediment yield over a 25 year period, with yields peaking about 15 years after planting. The difference in the timing of the predicted peak sediment yields by these two similar studies reflects different assumptions regarding how long it takes for full riparian canopy shading to occur.

Although there are many factors (e.g. bank material, presence of large woody debris, and land use) that play a role in determining channel widths, the potential influence of riparian vegetation on the width of channels in headwater streams is well appreciated. However, as catchment size increases, the trend appears to reverse with channels in forested catchments being narrower than those in pasture catchments (Anderson et al. 2004). Some evidence of this can be observed in the data of Davies-Colley (1997) where the ratio of channel width in pasture to that in forest becomes negligible in catchments greater than c. 10 km^2 (Figure 2). Again, this is likely to reflect the change in relative importance of difference erosion processes as catchment and channel size increase. As channels get wider, riparian vegetation is less likely to provide a closed canopy over the stream channel (Davies-Colley & Quinn 1998) and therefore the suppression of groundcover by shading will be less important. Furthermore, as catchment size increases, shrubs and trees on forested stream banks are likely to provide better protection against the dominant processes (e.g. fluvial entrainment and mass failure) than they do in headwater streams.

Effectiveness of riparian interventions at reducing bank erosion in New Zealand

As previously noted, bank erosion studies in New Zealand are rare and there is little information on the contributions of bank erosion to river sediment yields. This scarcity of quantitative research also includes studies that quantify the effects of catchment intervention measures on bank erosion. The dearth of such studies probably reflects the necessarily, but inconveniently, long duration of bank erosion studies if they are to be meaningful. The required study length would be even greater if a stream reach of interest was to be assessed for erosion rates before any intervention measures were put in place (i.e. a 'before and after' research design). Another issue with measuring bank erosion rates is the problem of accurately representing the nature of erosion for a whole river reach, let alone for a whole river system, from measurements at comparatively few channel locations.

A study by Parkyn et al. (2003) sought to assess the effectiveness of riparian buffers on, among other factors, stream bank stability. Nine streams in the Waikato region were assessed, each having a fenced and planted riparian reach downstream of an unfenced and actively cattle grazed reach. No information on catchment area or stream order of the nine streams was presented. The fenced sites had been retired from grazing for between 2 and 24 years and the median buffer width was 12.7 m (range = 3.5–75 m). The species used to rehabilitate the riparian areas varied between sites, but most sites were planted with native shrubs/trees and/or varieties of willow and poplar. The authors used the Pfankuch method to assess stream bank stability. The Pfankuch method is a qualitative assessment approach that evaluates mass wasting potential, erodibility of bank and bed materials, channel capacity and evidence of excessive erosion and/or deposition (Pfankuch 1975). Scores are assigned to individual assessment components and depending on the total score (Pfankuch index) for a reach, a rating of 'good', 'fair' or 'poor' is assigned. Parkyn et al. (2003) found that only three of the nine study streams had better ratings of bank stability in the buffered reaches than the non-buffered reaches. The authors did not, however, present any specific explanations for this limited improvement in bank stability.

In an earlier study, Williamson et al. (1992) used the semi-quantitative approach of Platts et al. (1983) to determine the difference in channel form and bank stability between grazed (by both sheep and cattle) and retired reaches along five streams in Southland. The streams varied in width from 0.74 to 20.7 m and the widths of the retired riparian areas varied from 20 m to >50 m. Unlike the Parkyn et al. (2003) study, the reaches that were retired from grazing were not planted or fenced. Channel characteristics, such as width, degree of undercut, bank angle, vegetation overhang and erosion extent were measured on two occasions within the grazed and retired reaches. The authors found no evidence that grazed stream banks were more susceptible to erosion than retired stream banks, except when intensive grazing occurred on wet riparian soils adjacent to narrow (<2 m), low-order streams. Williamson et al. (1992) observed that that the dominant erosion process in the wider, higher-order streams was channel undercut (by fluvial scour/entrainment), which was largely unaffected by grazing of stream banks. Williamson et al. (1992) suggested that stream bank erosion in the reaches experiencing fluvial scour may have been reduced had the grazing retirement been complemented by riparian tree planting.

In the only known 'before and after' study from New Zealand that specifically addresses bank erosion, Williamson et al. (1996) measured (on two occasions) the proportion of eroding stream bank from a 14.3 km reach of mid-to-lower reaches of the 73 km^2 Ngongataha Stream in the central North Island. Areas of active erosion (as evidenced by fluvial scour, mass failure and recently exposed bare soil) were visually estimated in 1976 and again in 1989 after widespread riparian planting (with native shrubs/trees and large-growing exotic species) and fencing in the catchment (between 1982 and 1988). The riparian buffers ranged from 2–100 m but were mostly <40 m. The study design was slightly compromised by the fact that the growth of riparian vegetation (in particular blackberry) made it difficult to assess the stream banks in the same way on both occasions. The authors did, however, account for the different assessment methods by only looking for large differences in stream bank stability in the comparison of the results of the two surveys. The 1975 survey showed that c. 30% of the stream banks in the study reach were actively eroding while this had reduced to c. 4% in 1989. Using historical flow records they also estimated that the decrease in bank erosion resulted in an approximately 85% reduction in sediment yield from the catchment. The authors attributed this reduction in stream bank erosion and sediment yield to the removal of livestock (sheep, cattle and deer) from riparian areas and the ability of riparian buffers to trap hillslope-derived sediment.

A study by Boothroyd et al. (2004) measured channel width and percentage of eroding banks from stream reaches with differing forest and riparian covers in *Pinus radiata* plantations within the Whangapoua area of the Coromandel Peninsula. The measured reaches were mainly within small headwater catchments (median catchment area = 93 ha). Five forest/riparian cover groupings were identified: (1) post-harvest pine plantation with all trees clearcut to stream edge; (2) post-harvest pine plantation with the retention of a vegetated riparian buffer (buffers were mostly <20 m wide); (3) mature pre-harvest pine plantation with a riparian vegetation dominated by indigenous species; (4) mature pre-harvest pine plantation with riparian vegetation dominated by pine (with an indigenous species understorey); and (5) mature indigenous forest reference sites. Results of parametric statistical tests of relatively small sample sizes ($n = 4$ for three groups and $n = 8$ for two groups) suggested that stream channels at clearcut sites were significantly wider than pine forest sites with indigenous vegetation buffers (at both pre- and post-harvest sites). The percentage of eroding stream bank was highest at clearcut sites (mean of 30.8% versus a maximum mean of 9.2% for all other groups) and significant differences were detected between pre- and post-harvest sites regardless of whether a riparian buffer was retained.

Also of relevance is a short technical report (i.e. Hicks 1992) that summarised the effects of tree planting on stream bank erosion from a number of locations throughout the North Island. This report visually compared bank erosion at planted sites with non-planted sites, generally after large-flow events. No information was presented on the size or catchment areas of the measured reaches; however, the author concluded that the planting of riparian vegetation can reduce bank erosion, but it does not always do so. The major findings of Hicks (1992) were: (1) some tree species (e.g. poplars, osier willows and native shrubs/trees) were more effective than others (e.g. tree willows and pines) at protecting banks from scour and bank collapse; (2) interlocking roots formed by several years' growth of dense tree plantings provided superior protection from bank erosion; and (3) removal of dead trees or trees that had fallen into the river reduced the occurrence of bank erosion. Hicks (1992) also suggested that the exclusion of grazing animals (type not specified), without tree plantings, reduced the occurrence of bank erosion.

Given the small number of studies that directly measure the effects of riparian interventions on bank erosion, it is also worth considering the findings of some New Zealand studies that infer changes in rates of bank erosion (e.g. changes in suspended sediment yields) after catchment interventions. For example, Hughes et al. (2012) estimated the annual suspended sediment loads (from a long-term continuous turbidity record) for a 12 year period from a 268 ha headwater catchment within the Whatawhata Research Station. For the first 2 years of this study, the catchment was used exclusively for unrestricted livestock (sheep and cattle) grazing. During the third year, an integrated catchment management plan was implemented whereby a number of catchment interventions were implemented, most significant of which were the exclusion of cattle (but not sheep) from riparian areas, planting eroding headwater streams with poplar trees and the planting of some riparian areas with indigenous vegetation. Despite these catchment interventions, this study failed to show any difference between sediment yields in the pre- and post-intervention periods. The authors attributed this to the limited pre-intervention data set (2 years) and the high natural inter-annual variability in sediment yields. Hughes et al. (2012) also hypothesised that, while the exclusion of cattle from riparian areas may have reduced sediment from livestock trampling of stream banks, the channels may be contributing more sediment due to the loss of protective groundcover caused by riparian shading.

Wilcock et al. (2013) examined the results of applying best management practices (BMPs) in five dairying catchments (for periods of between 7 and 16 years) throughout New Zealand. The BMPs included riparian planting and livestock exclusion from streams. No information was presented on what type of vegetation was planted or the widths of the buffers formed by fencing. The Dairying and Clean Streams Accord agreement between New Zealand's largest dairy company (Fonterra), regional councils, the Ministry for the Environment and the Ministry of Agriculture and Forestry does, however, only require that livestock be 'excluded' from streams (over a specified size) and hence buffer widths are usually minimal. The authors attributed reductions in measured total suspended solids concentrations (of 4%–11% a year) to reduced bank erosion after the removal of cattle from streams and riparian areas. However, no qualitative or quantitative stream bank erosion data were presented to support this claim.

Another study (Collins et al. 2013) looked at a number of water quality variables in four stream reaches that had riparian buffers directly downstream of non-buffered reaches within the Lake Ellesmere catchment. Livestock (type not stated) were excluded from both the buffered (<10 m wide) and non-buffered (control) sites. The riparian buffers were planted mainly with native shrubs and flax. They found that turbidity, measured during fortnightly visits, was marginally (but significantly) lower in buffered reaches (mean of 16.0 nephelometric turbidity units [NTU]) than non-buffered (17.6 NTU) reaches. The authors attributed this to the interception of hillslope-derived sediment by the buffers, but did not consider the potential of the riparian vegetation to reduce sediment supply (and hence a source of increased turbidity) directly from stream banks.

Finally, a riparian pasture retirement study carried out in the Waikato region (Smith 1989) provides some further insights into the overall impact of riparian management on sediment delivery to streams. In this study, runoff collectors were positioned within both grazed (sheep and cattle) and retired riparian areas (near stream banks) so that runoff from the adjacent hillslopes was collected. The buffers were vegetated with perennial ryegrass and white clover and varied between 10 and 13 m wide. Suspended sediment concentrations of runoff through the retired riparian sites were c. 90% lower than from grazed sites. Hence, as also noted later by Williamson et al. (1996), riparian buffers have the potential to reduce sediment delivery from hillslopes as well as from stream banks.

Discussion

Riparian management interventions aimed at reducing bank erosion are sometimes applied in the same way throughout catchments in the expectation that they will be equally effective everywhere. This paper argues that different erosion processes may act upon stream banks in different parts of a catchment and therefore catchment managers need to consider this when implementing riparian management interventions. The previously identified concept of scale-dependence of stream bank erosion processes is presented as a framework for identifying the riparian intervention measures that may be most effective in different parts of a catchment.

The catchment rehabilitation literature was searched for New Zealand-based studies that evaluated the effect of riparian management interventions on bank erosion, and few were found. Consequently, it is not feasible to make a detailed assessment of the applicability of the concept of scale-dependence of stream bank erosion processes in the New Zealand setting. The study of Williamson et al. (1992) is perhaps the only study to demonstrate scale-dependence of stream bank erosion processes. Williamson et al. (1992) found that livestock grazing of the riparian areas had a greater effect on narrow (<2 m), low-order channels than on wider, higher-order channels. They attributed this to the fact that the dominant erosion process on higher-order channels was channel undercut (by fluvial scour/entrainment) and this is unaffected by grazing of stream banks. Within the low-order streams (where stream power is low and bank heights are low), preparatory processes (of which livestock trampling damage can be considered one) were probably the dominant form of bank erosion, hence livestock access had the greatest relative impact in these reaches.

The dearth of studies that have quantified the effectiveness of riparian interventions in reducing rates of stream bank erosion is mirrored internationally. Researchers from North America have noted the disparity between the widespread use of riparian interventions for the control of bank erosion and the scarcity of quantitative assessment of their effectiveness for such purposes (e.g. Bernhardt et al. 2005; Shields 2009; Miller et al. 2014).

In regard to the studies summarised here, the intervention measures appear to have been mostly effective (in terms of an observable or inferred reduction in bank erosion or decreased suspended sediment concentration/yield) (Table 1). A caveat to this is that most studies used only qualitative or semi-quantitative measurement methods. Furthermore, one of the few quantitative studies (i.e. Hughes et al. 2012) reviewed found no change in measured sediment loads (no direct measurements of bank erosion were made) over a 12 year period after extensive riparian management within a pastoral headwater stream.

The studies considered in this paper were carried out in a variety of geographical locations where factors such as geology, soil type, climate, and land use history are likely to vary considerably. Furthermore, the methods used to quantify (or infer) changes or differences in erosion rates and suspended sediment yields varied between studies. Therefore an assessment of which specific approach is most effective is not feasible. Importantly, however, most of the studies attribute the measured improvements or differences in bank erosion to the exclusion of livestock from riparian areas. The importance of livestock (in particular cattle) as an erosive mechanism in catchments is well documented (e.g. Trimble & Mendel 1995) and their role in stream bank degradation

Table 1. A summary of the findings of the identified bank erosion studies.

Study	Measurement method	Intervention measure/Monitored effectiveness
Smith (1989)	Runoff plots	Fenced grass riparian buffers. Space for time study. Bank erosion not measured but TSS concentrations in hillslope runoff were ~90% lower at treated sites.
Williamson et al. (1992)	Semi-quantitative assessment	Riparian zone retired from grazing. Space-for-time substitution study. No evidence that grazed banks >2m wide were more susceptible to erosion than retired banks. For streams >2m wide grazing on wet riparian soils resulted in increased erosion.
Hicks (1992)	Visual assessment	Visual comparison of bank erosion at planted sites with non-planted sites, generally after large flow events. Found that i) tree species used was important; ii) interlocking roots formed by several years growth of dense tree plantings provided superior protection from bank erosion; and iii) removal of dead trees or trees that had fallen into the river reduced the occurrence of bank erosion.
Williamson et al. (1996)	Visual assessment	Planted and fenced riparian buffers. Before and after study indicated a decrease of actively eroding banks from 30% to ~4%
Boothroyd et al. (2004)	Channel width measurements and visual assessment	Retention of riparian buffers during pine harvesting. Stream channels from clearcut sites were significantly wider than pine forest sites with indigenous vegetation buffers (for both pre- and post-harvest sites).
Parkyn et al. (2003)	Qualitative assessment (Pfankuch method)	Planted and fenced riparian buffers. Space-for-time substitution study indicated that 3 out of 9 assessment sites (where riparian buffers were established) scored better.
Hughes et al. (2012)	Suspended sediment load estimates	Planted and fenced riparian buffers.* Before and after study. No measured change in sediment loads. No bank measurements made.
Wilcock et al. (2013)	Monthly water quality sampling	Planted and fenced riparian buffers.* Before and after study. Reductions in TSS concentration of between 4 and 11%. Improvements attributed to livestock exclusion. No bank measurements made.
Collins et al. (2013)	Turbidity measurements	Planted and fenced riparian buffers. Space for time study. Marginal difference (1.6 NTU) in mean nephelometric turbidity at treated sites. No bank measurements made.

*Other catchment rehabilitation measures also implemented

and soil compaction in riparian areas has been reported (e.g. Trimble 1994; Tufekcioglu et al. 2012). Furthermore, because of the high connectivity between stream banks and streams, the prevention of physical damage to the banks is likely to be effective.

While the benefits of livestock removal from riparian areas may be clear, the effects of riparian planting (another popular riparian intervention measure) on stream bank erosion are more equivocal. Only two studies (Hicks 1992; Boothroyd et al. 2004) specifically reported that the establishment of riparian vegetation actually resulted in reduced bank erosion. The results of these two studies were, however, largely based on qualitative visual assessments of the extent of bank erosion. This does not mean that the role of riparian vegetation is not important, but rather its effect is probably not so easily observed as the trampling damage of stream banks caused by livestock. The benefits of riparian plantings on stream bank stability (particularly where mass wasting dominates) are likely to be observable only in the long term, after the root systems of large trees have developed and start providing structural support. The response of river channels to the establishment of riparian vegetation is, however, complex, with some previous research arguing that the planting of riparian vegetation in headwater streams (and the subsequent shading of stream banks) can in fact result in channel widening (and hence a release of sediment). Accordingly, it is feasible that the retention of grassed riparian areas (management to avoid weed growth may be required) may be an effective approach in some circumstances due the ability of grass to both protect banks of low-order streams from preparatory processes and provide an effective buffer from sediment derived from adjacent hillslopes (Lyons et al. 2000). The implications of this are (and as previously noted by Montgomery [1997]) that a one-size-fits-all approach to the application of riparian intervention measures to reduce stream bank erosion is unlikely to be appropriate or effective.

Conclusions

The reduction of stream bank erosion is a goal of many riparian management interventions. Studies that document the effectiveness of such interventions within New Zealand catchments are, however, rare. The few studies carried out to date suggest that riparian management interventions (in particular, livestock exclusion) can be effective in reducing stream bank erosion. Most New Zealand research to date has been qualitative or semi-quantitative in nature. Clearly, further research is required to improve our understanding of the influence of riparian interventions on stream bank erosion and sediment yield. Future researchers should consider quantitative methods for measuring bank erosion (e.g. erosion pins, bank surveys, Lidar data analysis). Furthermore, because the effects of establishing tree-based riparian vegetation are likely to be observable only over the long term (decades to centuries), future studies need to be carried out over long periods. Alternatively, more space-for-time substitution studies should be conducted that examine the role of riparian interventions on channel morphology.

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ORCID

AO Hughes http://orcid.org/0000-0001-8839-8524

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