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Issues for Riparian Management in the Auckland Region: Analysis of trade-offs between benefits and impacts

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Issues for Riparian Management in the Auckland Region: Analysis of trade-offs between benefits and impacts

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1 Executive Summary

Riparian zone management involving tree planting is widely recognised as a means to improve stream habitat. However, research has shown that shading of pasture grasses on the banks of streams can lead to channel widening when the 'armouring' (resistance to fluvial scour) of pasture turf declines and sediment accumulated on banks erodes. This has led to concerns that there will be increased sediment yields from catchments, and deposition in downstream environments. Furthermore, increased riparian shade may limit the growth of macrophytes and algae that assimilate dissolved nutrients from the water column, leading to increased downstream export of dissolved plant nutrients.

ARC have asked NIWA to assess the likelihood and implications of these consequences of riparian zone management occurring in the Auckland region, and to place any anticipated impacts in the context of overall changes from improved riparian management. Specifically we address the concerns that:

- shading will result in decreased in-stream uptake of nutrients by instream primary production (macrophytes and algae) and lead to increased yields of dissolved nutrients and eutrophication in estuaries;
- shading of grasses on stream banks will cause erosion and channel widening, leading to the loss of riparian plantings and to significant sedimentation and associated ecological effects in estuaries;
- tree planting will not result in a net improvement of stream habitat – either because any benefits are over-ridden by deposition of sediment from eroding streambanks or because the expected ecological enhancement of habitat for fish and other stream life does not eventuate.

There are two counteracting effects of a planted riparian strip on nutrient exports. Firstly, a reduction due to the interception of nutrient runoff from the land. Secondly, an increase due to riparian shade reducing in-stream nutrient uptake rates by macrophytes and benthic algae. From simple 'modelling' of these trade-offs, we predicted that a patch of riparian strip installed beside a small headwater stream would reduce nutrient export while a patch of riparian strip installed beside a larger stream further down the catchment would increase nutrient export. Allowing some degree for error, the 'cross-over' point (i.e., from reduction to increase) lies somewhere between 50 to 400 hectares of upstream catchment. We can conclude from this 'modelling' that:

- maximum benefits are achieved with a contiguous riparian strip from headwaters to the mouth;

- where implementation occurs in a 'patchwork' fashion then it should be a managed process, initially restricting riparian patches with full planting density to streams with catchments less than 200 hectares. Options to manage light conditions could be considered for larger streams;
- where stream channel widths exceed > c. 6 m, sufficient incident light is likely to reach the stream to sustain patchy macrophyte growth even with forest riparian planting.

In some situations heavy shade may not totally exclude macrophytes. Several native macrophyte species are shade tolerant and can grow well under moderate to high levels of shade, for example, species like *Nitella hookeri* and *Potamogeton ochreatus.* Also, to some extent, the input and retention of leaf litter may counteract the loss of instream plant biomass in terms of nutrient retention, particularly 'soft' leaves that decay quickly are more likely to take up nutrients quickly. However, it is not known whether these changes to the nutrient assimilation mechanisms in streams would be as effective as benthic algae and introduced species of macrophytes that are present in open stream sections.

Field surveys of streams in the Auckland region that had riparian plantings or had regenerated native forest in riparian areas compared to pastoral downstream sections, showed that channel widening does occur with increasing levels of shade. Furthermore, comparison of channel widths in pasture with widths in native forest showed similar increases as found in Waikato streams. Small pasture streams (catchment area < 1-2 km^2) were found to be about 50% narrower than the *same* streams in forest. In large streams (catchment area > 10 km^2), width of pasture reaches approached that of forest reaches. This result indicates that bank erosion is likely to occur primarily along small streams that are planted with riparian trees.

The amount of sediment estimated to be stored in stream banks was 940 tonnes km^{-1} of stream based on data from the Waikato. We modelled the dynamics of this loss of this sediment over time. About 10-15 years after planting the shade of growing native plants starts to reduce the vigour of riparian grasses leading to streambank erosion. Bank erosion, and therefore sediment yield, is predicted to peak about 25 years after planting, and at the peak (about 140 tonnes y^{-1} from each km of permanent stream), the total sediment yield (hillslope plus bank sources) would be about twice the hillslope pasture yield. Bank erosion and sediment yield can then be expected to decline, reaching low levels when the stream is assumed to have widened to 'forest' morphology by about 35-40 years after planting.

Actually the whole stream length in a sizeable catchment (say draining into an estuary) would not be planted simultaneously, rather in a patchwork fashion over a number of decades. We modelled the loss of sediment expected over time when streams are planted over 5, 10, 20 and 40 years. The peak sediment yield declined systematically with duration of planting, and with planting over 40 years, the maximum sediment yield did not exceed ambient catchment sediment yield.

To establish the effect of this sediment on estuaries, we have looked at modelling work for the Okura estuary. NIWA have produced critical catchment sediment loads for the Okura estuary based on (1) a critical sediment level of 2 - 3 cm thickness, above which benthic organisms would die, developed using field and laboratory investigations and (2) computer models of currents and sediment transport in the estuary. Under a worst case scenario where there is instantaneous planting of the whole stream length, sediment deposition could exceed 2 cm thickness 2.5 times per year for 5 years.

We expect that riparian planting programmes will be protracted and may take more than 20 years for whole catchments to be planted, based on the considerable time required to plant large numbers of trees (7,500-13,300 per km of stream). Thus, the sediment yields in flood events are likely to result in only thin layers of deposited sediments in estuaries. However, it is recognised that even thin sediment deposits can have detrimental ecological effects on some biota and there may be long term chronic effects.

Potential options for riparian management that controls shade and may avoid problems of sedimentation, are presented. However, evidence from buffer zone surveys and studies of landuse on stream communities suggests that shade and lowered stream temperatures, which can only be achieved by planting in buffer zones, will improve stream habitat and enable invertebrate communities to recover over long time-scales.

Therefore, riparian tree planting programmes should ensure that planting begins in headwaters, and is avoided in riparian and catchment wetlands so that their denitrification function is uncompromised. This progression downstream is likely to negate the need for macrophyte nutrient attenuation downstream and provide improvements in stream habitat and terrestrial biodiversity, although channel widening and loss of plantings close to stream banks is likely to occur. The impact of bank erosion on sediment yield will be slight if whole-catchment planting is extended over 20 to 40 years.

2 Introduction

Riparian zone management involving tree planting is widely recognised as a means to improve stream habitat (DOC-NIWA guidelines - Collier et al. 1995, MfE 1999). However, recent research has shown that shading of pasture grasses on the banks of streams can lead to channel widening when the 'armouring' (resistance to fluvial scour) of pasture turf declines and sediment accumulated on banks erodes (Davies-Colley 1997). This will, in turn, result in increased sediment yields from catchments, deposits of fine sediments on stream substrates, and higher turbidity in streamwaters during floods until the stream channel stabilises. Furthermore, increased riparian shade may limit instream primary production, growth of macrophytes and algae that assimilate dissolved nutrients from the water column, leading to increased downstream export of dissolved plant nutrients (e.g. Howard-Williams & Pickmere 1999). Much of the nutrient is expected to be exported eventually as particulate matter and dissolved organic nutrients that are much less active in stimulating eutrophication downstream.

In the draft ARC riparian management guidelines, the desired riparian zone is one that is fenced and planted with successional species of native plants that will be self-sustaining and require little maintenance. Even if a riparian zone is fenced to exclude livestock and no plantings are initiated, successional processes from the release of grazing pressure are likely to lead to woody plants developing in the riparian zone (Davies-Colley & Parkyn 2001). The shade from these plantings could potentially lead to increased sediment export to downstream estuaries during a transitional stage of channel widening. Additionally, the shade could lead to higher nutrient export to estuaries. ARC have asked NIWA to assess the likelihood and implications of these consequences of riparian zone management occurring in the Auckland region, and to place any anticipated impacts in the context of overall changes from improved riparian management. They have specifically asked us to address four scenarios, namely that:

1. Shading will result in decreased in-stream uptake of nutrients by macrophytes and lead to increased nutrient yields and eutrophication in estuaries
2. Shading of grasses on stream banks will cause erosion and channel widening and lead to significant sedimentation and associated ecological effects in estuaries
3. Tree planting will not result in a net improvement of stream habitat – either because any benefits are over-ridden by deposition of sediment from eroding streambanks or because the expected ecological enhancement of habitat for fish and other stream life does not eventuate.
4. Channel widening will result in loss of riparian plantings

3 Nutrient Assimilation and Export

3.1 Background

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 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.).

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 NO_3-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 NO_3-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.

A study conducted by Davies-Colley & Quinn (1998) in a range of streams throughout the northern North Island reported that filamentous algal blooms did not occur when shade levels in small streams exceeded 88% of incident light. However, when channel width of forested streams exceeded c. 4 m, stream lighting increased abruptly and continued to increase with stream size. These findings suggest that loss of plant biomass due to shading, and consequent increased downstream nutrient transport, is likely to be an issue only in small streams. Although sufficient light levels for macrophyte growth may be more realistically achieved at channel widths of > c. 6 m, when riparian vegetation consists of tall forest species.

In some situations heavy shade may not totally exclude macrophytes. Recent experimental work in a small Waikato stream draining a lowland pastoral (dairying) catchment found that 90% shade, provided by shade cloth placed over a 100m reach, led to marked reductions in the biomass of the adventive emergent *Polygonum hydropiper* and increased biomass of shade-tolerant submerged native species such as *Nitella hookeri* (M. Scarsbrook, NIWA, unpubl. data). Although conducted at a small scale at one site, this study suggests that riparian plantings may not necessarily lead to the exclusion of macrophytes, but rather to a change in the structure of macrophyte communities. Furthermore, there was no evidence that shade-driven reductions in nutrient uptake rates occurred in the 100 m shaded reach, although it is not clear how this can be extrapolated to larger scales.

Several native macrophyte species are shade tolerant and can grow well under moderate to high levels of shade (P. Champion, NIWA, pers. comm.). For example, species like *Nitella hookeri* and *Potamogeton ochreatus* are often restricted to shady sites whereas weedy species like *P. crispus* and the oxygenweeds are dominant in open areas. The nutrient uptake rates of these shade-tolerant macrophytes compared to other species are not known, however, and would be influenced by a wide range of environmental factors. Macrophytes typically scenesce in late summer-autumn when water temperatures and day length decline (Champion & Tanner 2000). The annual die-off and decomposition of scenescent macrophyte material can be expected to lead to the release and downstream transport of stored nutrients. Thus, the net result of reduced macrophyte biomass caused by shading from riparian plants may be minimal in terms of annual nutrient export from catchments. However, about a third of the nutrients released from decomposing macrophytes are retained in the sediments, and only half of the remaining nutrients in the water column may be bioavailable.

To some extent, the input and retention of leaf litter can counteract the loss of instream plant biomass in terms of nutrient retention, because leaf litter is typically low in nitrogen and phosphorus and these nutrients are taken up from the water column by microbes during decomposition. Quinn et al. (2000a) found that leaf up-take of dissolved N and P was greatest for soft, low-nitrogen litter. However, where dense riparian plantings are implemented, the subsequent uptake of instream nitrogen by fallen leaves was not expected to compensate for the reduced uptake of N due to lower algal biomass.

Removal of nutrients carried on overland and subsurface flow is possible before they reach the stream through the use of good catchment management practices such as (i) restricting stock access to the stream, (ii) developing grass filter strips, and (iii) protecting catchment and riparian wetlands which play major roles in controlling inputs of nitrogen and phosphorus. 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). Riparian plantings should ensure that important riparian wetland function is maintained.

3.2 Calculating the effect of Riparian Management on nutrient export.

There are two counteracting effects of a planted riparian strip on nutrient exports. Firstly, a reducing effect due to the interception of nutrient runoff from the land. Secondly, an increasing effect due to riparian shade reducing in-stream dissolved nutrient uptake rates by macrophytes and benthic algae. Computer models such as BNZ and WAM simulate the detailed spatial and temporal dynamics of these two effects within a catchment (Cooper & Bottcher 1992; Elliott & Sorrell 2001) but using such complex models was beyond the scope of this project. Instead we used some of the mathematical relationships of these models to create a simple spreadsheet model to illustrate the balance between these two effects when installing a riparian strip along a stream reach.

For a stream reach without a riparian zone the equation used was:

$$M_{DN} = \left[\left(E_C A_U \right) e^{-a_N \left(Q_{sp} A_u \right)^{-b_N} d} \right] + \left[E_C A_{sp} d \right]$$

For a stream reach with a riparian zone the equation used was:

$$M_{DR} = \left[\left(E_C A_U \right) e^{-a_R \left(Q_{sp} A_u \right)^{-b_R} d} \right] + \left[E_C A_{sp} d R_E \right]$$

Where,

M_{DN} = mass of nutrient delivered from reach with no riparian planting (g year-1)

M_{DR} = mass of nutrient delivered from reach with riparian planting (g year-1)

E_C = nutrient export coefficient (g m-2 year-1)

A_U = upstream catchment area (m2)

Q_{sp} = specific flow (m3 s-1 m-2)

d = reach length (m)

a_N , a_R = empirical nutrient decay coefficients

b_N , b_R = empirical coefficients for dependence of decay coefficient on flow

A_{sp} = catchment area per unit length of stream (m2 m-1)

R_E = removal efficiency (fraction remaining)

We ran the 'model' using the following initial input values and variations around these (in parentheses):

E_C = 10 kg N ha-1 year-1 (5 – 15 kg N ha-1 year-1)

= 1 kg P ha-1 year-1 (0.5 – 1.5 kg P ha-1 year-1)

These values are based on export coefficients from rural pasture reported in the updated Lake Managers' Handbook (Elliott & Sorrell 2001).

A_U = 0 – 20 km2

These values were arbitrarily chosen to cover a range of upstream catchment areas.

Q_{sp} = 20 L s-1 km-2

This value is an estimate of specific mean flow for the Auckland region based on an analysis of flow records from 25 catchments in the region (pers comm. Charles Pearson, NIWA Christchurch)

d = 1 km (0.5 – 10 km)

Values arbitrarily chosen to represent a range of reach lengths over which riparian zones may be created.

a_N , a_R = 0.0001 (0.00005 – 0.0002), and 0.00002 (0 – 0.00004), respectively

These values are derived from the summary of stream attenuation data provided by Rutherford et al. (1987) and cover the range of values that have been used to calibrate catchment models to nutrient data. In streams this value has been found to vary over these ranges depending upon plant type and season (i.e., seasonal senescence affecting nutrient uptake). The values for a_R are very arbitrary as little data exists for nutrient uptake in streams with riparian shading.

b_N , b_R = 0.7 (0.3 – 0.8), for both

These values are derived from the summary of stream attenuation data on dissolved nutrients provided by Rutherford et al. (1987) and cover the range of values that have been used to calibrate catchment models to nutrient data. In streams this value has been found to vary over this range depending upon plant type and season (i.e., seasonal senescence affecting nutrient uptake).

A_{sp} = 0.18 km2 km-1

This value is derived from an analysis of relationships between catchment area and stream length for three catchments in the Auckland region. (but see Section 4.3 where figures are based on ARC data and differ slightly to the value used in this calculation – this discrepancy will be addressed)

R_E = 0.25 (0.1 – 0.75)

These values are arbitrary but not unreasonable. Reviews of literature indicate that riparian buffers can lower nutrient input to streams significantly but that performance is highly variable, being dependent upon such things as the form of the nutrient, slopes, soils and storm size. These dependencies have been captured in mathematical form in computer models (such as BNZ and WAM). The output from a large number of computer simulations of the effectiveness of riparian buffers for sediment retention was reported in Collier et al. (1995) and indicates that a removal efficiency of 75% is reasonable at least for particulate nutrients associated with sediment.

An example of output from the 'model' is shown in Fig. 3.1. The essential shape of this plot remained the same throughout the sensitivity analyses, even if the numerical values differed. The key point from this 'modelling' is the prediction that a patch of riparian strip installed beside a small headwater stream will reduce nutrient export while a patch of riparian strip installed beside a larger stream further down the catchment will increase nutrient export. Allowing some degree for error, the 'cross-over' point (i.e., from reduction to increase) lies somewhere between 50 to 400 hectares of upstream catchment. At riparian patch lengths of 1 km or less, the increase in nutrient exports in larger streams is small, but at patch lengths of 2 km or greater the increase could exceed 30%. The longer the riparian patch the greater the reductions if they are installed next to smaller streams and the greater the increases if patches are installed only next to larger streams (upstream catchment area greater than 200 ha).

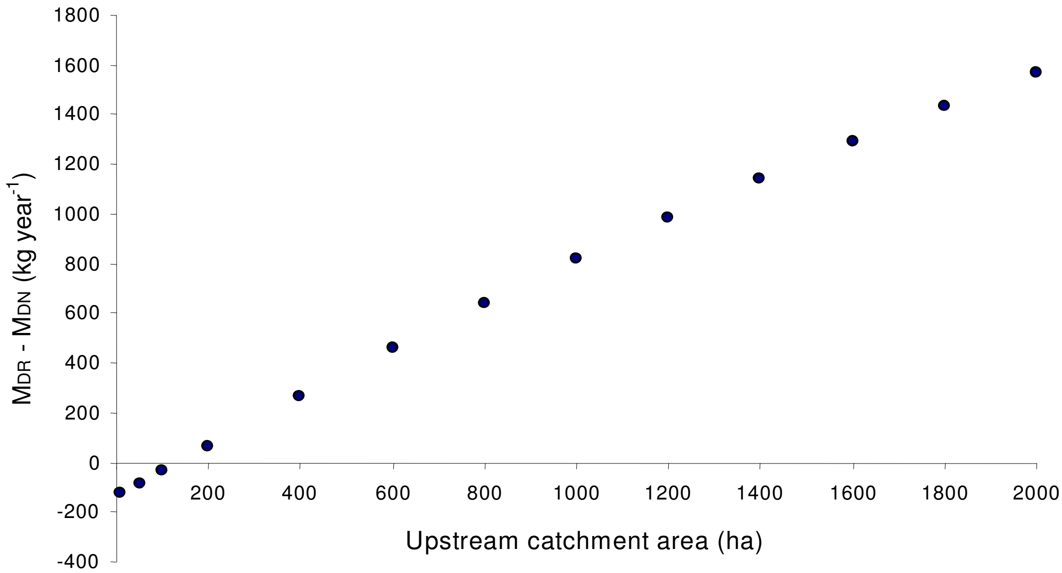

Figure 3.1: Net effect of installing a riparian patch at different locations in a stream network (defined by catchment area) on downstream transport of nutrients.

We expect that instream plant uptake of dissolved nutrient will be linked to light via photosynthesis and will decline with increasing shading (Quinn et al. 1997). We cannot yet quantify the effect of a shift to shade tolerant macrophyte species, should this occur. Our 'modelling' to date leads to the following *tentative* recommendations from the sole viewpoint of nutrient loading to downstream receiving environments:

- Maximum benefits are achieved with a contiguous riparian strip from headwaters to the mouth.
- Where implementation occurs in a 'patchwork' fashion then it should be a managed process, initially restricting riparian patches with full planting density to streams with catchments less than 200 hectares.
- Where the opportunity arises for riparian protection in the lower reaches of a large catchment then this opportunity should be taken to reduce nutrient inputs from the land. However, riparian planting design should be aimed at preserving sufficient light inputs to the stream to maintain macrophyte beds for in-stream nutrient uptake. This may involve reducing planting densities, selected low growing plants, and greater use of deciduous species.
- Where stream channel widths exceed c. 6 m or greater, sufficient incident light is likely to reach the stream to sustain patchy macrophyte growth even with forest riparian planting.

4 Sediment Trapping and Channel Widening

4.1 Background

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. 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 (e.g., Trimble 1997) which suggest that the phenomenon of channel narrowing in pasture is a general feature of formerly forested stream channels.

Collier et al. (in press) 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 a new steady-state 'forest' morphology and sediment yield will eventually decline to a lower level than currently experienced.

Scarsbrook & Halliday (1999) suggested that riparian planted stream reaches with low channel gradient may act as storage zones for sediment eroded from upstream. This finding suggests that some of the sediment mobilised from channel widening may accumulate in localised areas along stream reaches rather than being exported downstream.

4.2 Field Survey

We considered that it would be valuable to inspect some streams in the Auckland Region to confirm that stream channels in forest are indeed wider than in pasture in this region, and by implication test the hypothesis that channel widening is likely to occur there following riparian fencing and planting. With the assistance of ARC staff we identified a range of streams over a broad sweep of the Auckland Region where there had been riparian planting or shade trees were present, and there were 'control' reaches in pasture nearby or on the same stream systems. We also chose sites for inspection on the fringes of the Waitakere Ranges, west of Auckland, where recent land-cover maps were available from the Waitakere City Council and stream channel morphology could be studied in relation to riparian vegetation, including regenerating native forest, in a limited area of supposedly similar climate/hydrology.

4.2.1 Methods

Field visits were made on 9 and 10 May 2001 and characteristics of the field sites are listed in Table 1. At each site, notes on vegetation (including an estimate of ages of the largest trees in planted or regenerating forest) were made. Measurements of stream channel cross-sections were made at a number (usually 10) of equally-spaced cross-sections along a reach of up to 50 m. Channel width (w) was measured (by tape measure or survey pole) from bank crest to crest (crest being defined as the point of maximum rate of change of slope). Perennial vegetation was used as a guide where there was any doubt about position of channel bank edge. At some sites, for reasons of limited time with failing light, only quick subjective estimates of stream width could be made.

In order to provide an estimate of channel cross-sectional area without resorting to detailed survey techniques, depth (d) from bank level to thalweg (deepest point on the section) was measured with a survey pole, and the channel shape was classified as closest to a U-shape, V-shape or 'box'-shape (rectangular section). This is an adaptation of Robison & Beschta's (1989) method for estimating stream water cross-sectional area using wetted width and thalweg (water) depth. Cross-sectional area of box-shaped sections was estimated as wd, cross-sectional area of V-shaped (triangular) sections was estimated as 0.5wd, and the cross-sectional area of (most common) U-shaped sections (assumed to be intermediate in area) was estimated as 0.75wd.

4.2.2 Results

Site 1. Whangapouri Stream off Glenbrook Road

Along this 'lowland' (low gradient), meandering stream, school children have planted natives inside fenced riparian set-aside areas for the past 10 years. The plantings are up to about 4 m high, but their shade over this rather large stream (Catchment area 15 km2) is still minor (Plate 1). Nevertheless we noticed that the minimal level of shade was apparently sufficient to reduce the density of Glyceria maxima (sweetgrass) growing in the channel by comparison with an upstream pasture reach.

Upstream of the oldest plantings a control reach in pasture was surveyed. Here the (meandering) channel was appreciably wider (6.76 m) than in the planted reach, possibly because of slumping of the bank crest by cattle accessing the channel for water or grazing (of the dense sweetgrass stands). However, we cannot be sure that the contrast in stream channel size does not reflect the contrast in planform of the stream (meandering in the pasture reach versus fairly straight in the planted reach) because meander bends are likely to be generally wider than straight reaches.

It would appear that, at least at an early stage of riparian plantings, some stream channels may actually narrow, rather than widen, owing to livestock exclusion and re-building of the bank crest by high (bank-full), sediment-laden flows. It is also possible that lowland streams may differ from steeper gradient streams as regards their response to shade of riparian plantings.

Plate 1: Whangapouri Stream

Table 1: Characteristics of field sites visited on 9 and 10 May 2001 (nm = not measured)

Map ref.	Name	Location	Type	Catchment area (km^2)	Age of planting	Bank height (m)	Channel width (m)		Cross-sectional area (m^2)		Dominant spp.	Notes
							vegetated	pasture	vegetated	pasture		
R12 765-489	Whangapouri Stream	Glenbrook Rd	Lowland, rolling hills	15	Up to 10 yrs	2.5	4.85	6.76	6.43	10.47	sweetgrass water celery flax karo manuka akeake cabbage tree karamu kohuhu	• Channel narrower in vegetated reach, no shading out of grasses yet • Narrower channel could be lack of cattle trampling and trapping of silt in grasses or because planted reach was straighter than pasture reach measured (wider channels on bends).
R12 565-553	Golfcourse Stream	Awhitu Regional Park	Head-water valley wetland	0.05	25-30 yrs	c. 0.5	c. 0.8	No channel	nm	nm	totara kauri rimu kahikatea mamaku puriri kanuka flax tarata	• In pasture, wetland c. 0.7 – 1m deep, 4m wide, no channel • In buffer, channel formation begins immediately with shade but is square shaped (0.5 x 0.5) initially and widens out to a V shape c. 20 m into the buffer
R10 729-092	Entrance Rd Stream	Shakes-pear Regional Park	Small valley stream	0.05	25 yrs	c. 0.5	c. 0.4 - 1	No comparison	nm	nm	manuka tarata ponga rimu karo mamaku cabbage tree kauri	• Plantings old, but appear to be wind/salt damaged. Canopy gaps and abundant grasses (native *Microlaena stipoides*) • Channel erosion may be occurring in high

Map ref.	Name	Location	Type	Catch-ment area (km^2)	Age of planting	Bank height (m)	Channel width (m)		Cross-sectional area (m^2)		Dominant spp.	Notes
							vegetated	pasture	vegetated	pasture		
											broadleaf mapou	shade e.g. under dense tree fern growth, although channel appears to have cut down and become deeper rather than wider
Q09 472-224	Tributary of Araparera Stm No. 1	Ahuroa	Valley Stm	0.6	Remnant totara, animal access on one side	nm	c. 3 – 4	1 – 1.5	nm	nm	totara mamaku mahoe *Blechnum* spp. parataniwha hangehange five finger	• Definite narrowing in pasture reach, grasses and wetland plants encroaching on channel • Dense shade in remnant reach, little vegetation on banks, although some *Microlaena stipoides*
Q09 472-225	Tributary of Araparera Stm No. 2.	Ahuroa	Valley Stm	1.2	c. 30 yr old Eucalypts and Conifers, under-grazed	nm	c. 3	c. 1.5	nm	nm	eucalyptus exotic conifers kahikatea *Microlaena stipoides*	• Channel widening occuring • Transitional stage, tall stand of trees, open underneath and grazed but with c. 60% shade
Q09 480-230	Tributary of Araparera Stm No. 2.	Ahuroa	Valley Stm	1.2	Native forest (original or old regrowth)	nm	c. 4	c. 1.5	nm	nm	kahikatea mamaku ponga	• Native forest reach above in transitional stage and open pasture was densely shaded and wider still.
Q11 439-800	Koropotiki Stm	Bethells Rd, Waitakare	Valley Stm	0.8	Forest regrowth with animals excluded c. 20 yrs?	1.2	2.76	2.18	2.65	2.24	kanuka wheki *Blechnum* spp. mahoe hangehange blackberry	• Pasture reach has some remnant trees but extensive grass and weed growth on banks • Stream within regenerating bush was

Map ref.	Name	Location	Type	Catch-ment area (km^2)	Age of planting	Bank height (m)	Channel width (m)		Cross-sectional area (m^2)		Dominant spp.	Notes
							vegetated	pasture	vegetated	pasture		
												wider, but erosion appeared to be continuing as banks were largely bare and shrubby native species commonly found on riparian margins were absent.
Q11 461-782	Cascade Stm	Cascade Car park, Falls Rd, Waitakere	Valley Stm	2.6	Remnant native forest	nm	c. 5 - 7	No comparison	nm	nm	rimu kahikatea parataniwha *Blechnum* spp. mamaku ponga rewarewa	• wide native forest stream for catchment area, which suggests high rainfall gradient. Extent of channel widening with replanting will be linked to local rainfall and flood power.

Site 2 'Golfcourse' Stream, Awhitu Regional Park

Just outside the park boundary, a small (catchment area only 0.05 km^2) stream rises in neighbouring farmland. A valley bottom wetland of approximately 4 m width and up to 1 m depth (average about 70 cm - probing to underlying soil using a survey pole) occupies the former stream channel (Plate 2a). Downstream the stream flows perpendicular to a fence at the park boundary and into a 20 m wide riparian buffer alongside the golf course consisting of 25-30 year old planted natives (including kahikatea, kawaka, titoki, tarata, broadleaf, puriri, cabbage tree and flax). The shade of these plantings has suppressed the high-light wetland plants in what was (presumably) formerly a continuation of the valley bottom wetland, resulting in erosion of the wetland sediment and re-establishment of a single-thread stream channel (average width about 0.8 m; Plate 2b). At a few points along the buffered reach, where there is a gap in the canopy cover, wetland still persists to some extent alongside the channel.

While travelling round the Auckland Region on this trip we observed many small valley bottom wetlands in low order stream channels in pasture. Our observations and interpretations at the Awhitu site suggest that these wetlands will be destabilised by the shade of tall riparian plantings, leading to a decline of the wetland plants and eventual erosion of the wetland sediments. Moreover, this decline of the wetlands will reduce or eliminate their sediment and nutrient trapping and processing functions.

Site 3 'Entrance Road' Stream, Shakespeare Regional Park

This site lacked any useful pasture control reaches because essentially all stream channels have been fenced and planted. However valuable observations were made along a small stream near the road to the Park Headquarters. Plantings here appeared to be about 25 years old, but some plants were in rather poor condition with 'burnt off' tops suggesting damage by salt-laden winds at this (very exposed) site. Some gaps in the canopy along the stream where plantings have died provide a contrast with more shady reaches under vigorously growing plantings. The small stream (about 0.4-1m width) appears to have a systematically greater cross-section (sometimes deeper as well as, or instead of, wider) at more shady sections than at sections under canopy gaps (where grasses persisted). However, we were not able to properly stratify our measurements by shade level so as to verify this impression.

Plate 2a: Golfcourse stream at Awhitu regional park above riparian buffer

Plate 2b: Golfcourse stream at Awhitu regional park within riparian buffer

Site 4 Tributaries of the Araparera Stream, near Ahuroa

Two tributaries of the Araparera Stream were inspected on Lynvale Farm owned by Bruce and Shirley Jenkins (Dairy No 9476, 900 W Coast Road). At the first of these (No 1 tributary – Table 1) there is an obvious and marked contrast in channel width between a shady reach with remnant totara trees (channel about 3-4 m wide; Plate 3a) and a pasture reach downstream (channel about 1-1.5 m wide; Plate 3b). At a second tributary (No 2 tributary – Table 1) there is a similar marked change in channel width between a pasture reach (channel about 1.5 m wide) and a reach about 50 m downstream shaded by large (> 30 years old) eucalyptus and conifer trees (channel about 3 m wide). A transitional reach with intermediate lighting (perhaps 30% lighting) was also intermediate in channel dimensions. Further downstream of the large exotic trees, in an area of remnant native forest, the stream appeared even wider (channel about 4 m wide).

The marked contrasts in channel width between the shaded (by trees) and unshaded (grassy) reaches of the same streams is good evidence that the streams are significantly narrower in pasture.

Site 5 Koropotiki Stream, Bethells Road, Waitakere

A reasonably good vegetation 'transition' over which to study the contrast in stream channel dimensions between forest and pasture was found at this site. The pasture reach is grazed by cattle and has a few remnant manuka, mahoe and other native trees. Mean stream channel width is 2.18 m (Table 1; Plate 4a). The forested reach upstream of a fenceline appears to be recovering from a former time of grazing – perhaps over 20 years ago, consistent with the condition/construction style of the fence. The channel (at 2.76 m width; Plate 4b) is wider than in pasture and appears to be in an active phase of erosion with considerable areas of bare clay banks and some obvious recent bank slumps. The undergrowth is dominated by hangehange and other shrubby plants indicative of 'disturbed' sites. The riparian area and banks lack large areas of exposed roots, ferns and herbs that are common along undisturbed native streams in the Waitakere ranges.

Our interpretation is that the forested reach at this site may have originally been similar to the downstream pasture reach (grazed, with a few remnant trees), but that fencing perhaps 20 years ago has permitted extensive regeneration of native forest. The shading of the pasture in this upstream reach appeared to have caused bank destabilisation and the stream channel is now actively widening in an attempt to re-establish a 'forest morphology'. Based on Davies-Colley's (1997) demonstration of approximately 2-fold wider channels of small streams in forest compared to pasture, this stream may eventually widen to about 4 m.

Plate 3a: Tributary of the Araparera Stream

Plate 3b: Tributary of the Araparera Stream downstream of forested section

Plate 4a: Koropotiki Stream in pasture section

Plate 4b: Koropotiki Stream in regenerating bush

Site 6	Cascade Creek, Waitakere Regional Park

A quick visit to the Regional Park was made to observe stream morphology in relation to riparian plant associations in an 'undisturbed' native forest system. Cascade stream, a tributary of the Waitakere River, has a rather wide (5-7 m) channel despite its small (2.6 km^2) catchment. The channel dimensions of this stream are probably consistent with its undisturbed forest riparian zone.

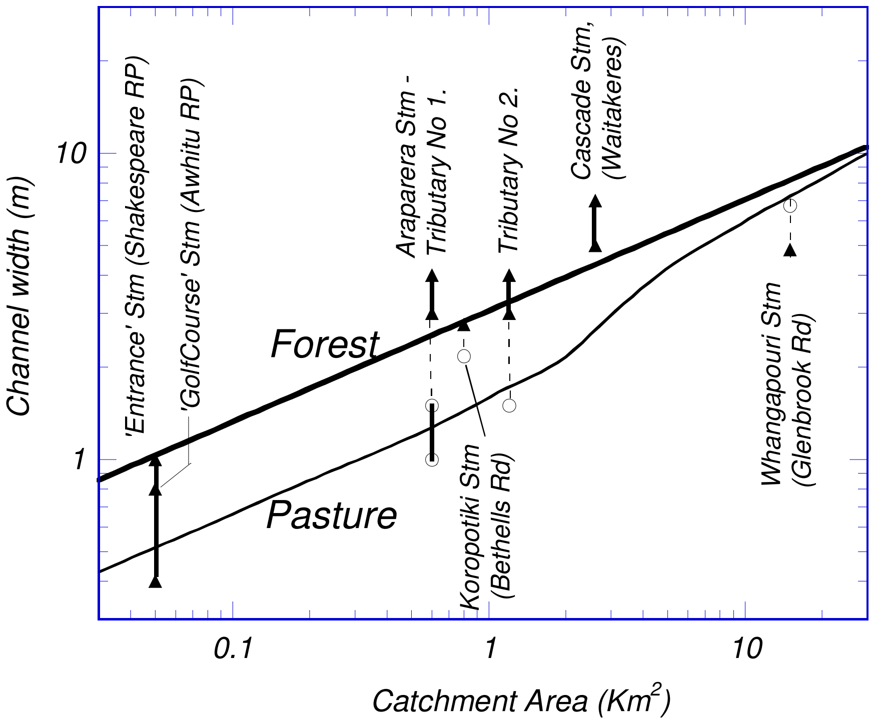

Figure 4.1: Stream channel width plotted versus catchment area for streams in the Auckland Region. Data are compared with the regression line for forested stream reaches and an indicative curve for pasture reaches of streams in the Waikato Region (adapted from Figure 1 of Davies-Colley 1997). Solid triangles are for forested reaches and open circles are for pasture reaches. Estimated ranges of widths are indicated by solid lines joining symbols. Forest-pasture 'transitions' are joined by dotted lines.

4.2.3 Summary

Figure 4.1 summarises the stream width measurements and estimates in terms of catchment area as an independent index of stream size. The widths of widely spaced forest streams (or forested reaches) in the Auckland Region are in fairly good agreement with the regression line of Davies-Colley (1997) – equation of line: $w = 3.06A^{0.363}$. Cascade Stream in Waitakere Regional Park appears to be rather wider than expected, probably because of the high rainfall (with large channel-forming floods) in this area. Our impression is that Koropotiki Stream, also in the Waitakere ranges, is actively eroding along a regenerating forest reach and may eventually widen to around 4 m.

Figure 4.1 also shows data for pasture reaches in relation to a trend-line for pasture calculated from the data of Davies-Colley (1997). This trend-line shows small pasture streams (catchment area < 1-2 km^2) to be about 50% narrower than the *same* streams in forest, while in large streams (catchment area > 10 km^2), width of pasture reaches approaches that of forest reaches. This result indicates that bank erosion is likely to primarily occur along small streams that are planted with riparian trees.

Taken together, our observations and measurements on widely spaced stream sites in the Auckland Region suggest:

- Pasture streams will widen following shading of pasture-covered streambanks by riparian forest plantings, by a factor of 2 in small streams (say < 4 m width in forest). This widening will be accompanied by ('temporary' - decades) stream water turbidity and streambed sedimentation.
- Valley bottom wetlands also seem likely to be reduced or eliminated by the shade of riparian forest plantings, so reducing their sediment and nutrient mitigating functions.
- The stream channel erosion and wetland suppression might only be controllable, long-term, by large amounts of management intervention so as to indefinitely maintain lighting levels required by grasses and wetland plants. We do not currently know what those lighting levels are, but they are appreciably above the lighting levels under dense native forest (1-2% light).

4.3 Amount of Sediment Loss from Stream Banks

Calculation of the amount of sediment stored in the streambanks and wetlands of pasture areas in the Auckland Region would require a detailed survey of stream channel dimensions, preferably at sites stratified by climate as well as riparian vegetation type. Distributions of stream length with catchment area are required for this kind of calculation (see Collier et al., in press, for a description of these calculations for the Mangaotama Basin using DEM-generated data). This is beyond the scope of the current report, but would be a worthwhile task for the future in the catchments of any particularly 'sediment-sensitive' estuaries.

However, a rough indicative calculation of sediment storage in pasture streambanks of the Auckland Region, and sediment yield from erosion of these streambanks, may be made by analogy with the Mangaotama Catchment - for which sediment yields under different planting scenarios have been predicted (Collier et al. in press). The 2.59 km^2 Mangaotama Catchment (in the Hakarimata Range, west of Hamilton) is estimated to have about 13,000 tonnes of sediment stored in the pasture banks of its 20 km network of stream channels and valley bottom wetlands – that is, 5000 tonnes/km^2 catchment area. Drainage density of this (hill-land) catchment is high at 20km/2.59 km^2 = 7.7 km^{-1}.

According to ARC riparian guidelines (in prep.) the Auckland Region has about 10,000 km of *permanent* stream length in a total land area of 4500 km^2, of which area about 2240 km^2 (49%) is classified as pasture. Assuming these measurements have been made at the same scale, the average drainage density of permanent streams in the Auckland Region is therefore 10,000/4500 = 2.2 km^{-1}, appreciably lower than in the Mangaotama Catchment. If we assume (somewhat arbitrarily) that the 1st order streams (8.1 km total length) in the Mangaotama are all ephemeral or intermittently flowing, the comparable drainage density of this catchment reduces to 4.6 km^{-1} (for 12 km of permanent streams), still over twice as high as in Auckland's streams. Drainage density is generally lower in lowland (low gradient) land, so these drainage densities seem 'reasonable' considering the appreciable lowland area that is probably present in the Auckland Region and that the Mangaotama is entirely 'upland' above the flow recorder.

If we were to assume that all the pasture area in the Auckland Region has the same bank sediment storage as the Mangaotama at 5000 tonnes/km^2 we get 11.2 Mega-tonnes. This is probably an over-estimate given that the drainage density in the Auckland Region is lower on average than in the Mangaotama. A more realistic approach is to assume that *bank sediment storage scales with stream length*. The 2nd order and greater (assumed permanent) stream channels in the Mangaotama Basin store about 11,000 tonnes of sediment, i.e. 940 tonnes km^{-1}. If the 49% of the Auckland Region that is pasture also has 49% of the total stream length, we have 4900 km of permanent pasture streams. These streams may therefore be storing 4900 km x 940 tonnes km^{-1} = 4.6 Mega-tonnes of sediment in their banks.

Typical sediment yield of pasture catchments in the Auckland Region is of the order of 150 tonnes km^{-2} y^{-1} (= the *geometri*c mean of data in Table 2 – so as not to overweight by the high value for the Mahurangi which may itself reflect some channel widening within pine plantations), cf. about 220 tonnes km^{-2} y^{-1} in the Mangaotama. This sediment yield can be expressed relative to the 2.2 km of (permanent) stream length per km^2 of catchment, giving about 68 tonnes y^{-1} km^{-1} of permanent stream channel. Therefore, the pasture streambank-stored sediment in the region is equivalent to approximately (940 tonnes km^{-1}/68 tonnes km^{-1} y^{-1}) = 14 years of average (hillslope in pasture) sediment yield, compared with 25 years at Whatawhata.

In the Mangaotama Catchment we estimated that bank-stored sediment would be mobilised with a peak at about 15 years after proposed pine planting, based on observations of streambank erosion under 15 year old pines elsewhere in the Hakarimata Range (Collier et al. in press). Our observations in the approximately 20 year old regenerating native forest along the Koropotiki Stream off Bethells Road (widening to a mean of 2.76 m compared to 2.18 m in pasture) suggests that the widening under planted or regenerating native forest will be delayed compared to faster growing pines - possibly peaking at about 25 years after fencing/planting.

Table 2: Current sediment yields to several Auckland estuaries

Catchment	Primary land use	Catchment sediment yield (tonnes/km^2/year)	Source
Mahurangi	Pasture	450	Stroud & Cooper 1997
Upper Waitemata Harbour	Pasture	220 - 350	Van Roon 1981
Pakuranga	Urbanising	< 60	Williamson et al. 1998
Wairoa	70% pasture	78	Luckman et al. 1999
Okura	Pasture	117	Stroud et al. 1999

We have modelled the general trend of sediment from channel widening as a Gaussian function peaking at 25 years and with an 'error' value of 4 yrs, and set the area under this curve equal to the total 940 tonnes to be eroded **along each km of permanent stream channel to be planted**. We have assumed that there is a benefit from riparian fencing in terms of reduced hillslope sediment yield from (a) the reduced area in grazing (we assumed a 15 m wide riparian set-aside either side of the stream which implies about a 6.7% reduction in grazed land area – refer ARC riparian guidelines in prep.) and (b) the sediment trapping in the riparian zone (we assume 70% trapping using trapping efficiency figures from the DoC-NIWA Riparian Guidelines for moderate slopes, low drainage category, and medium clay content). This trapping figure only applies to the approximately 50% of the catchment runoff that enters the riparian set-aside area; the other 50% flows in the channels of ephemeral and intermittently flowing streams so bypassing the (planted, fenced) riparian zones of the permanent streams.

Figure 4.2 shows the possible widening response of a representative pasture stream in the Auckland Region with a (hillslope erosion-dominated) sediment yield of 150 tonnes km^{-2} y^{-1} assuming peak erosion (peak rate of widening) at 30 years. The general trend represented in Figure 4.2 can be summarised as follows. Immediately after animal exclusion by fencing there is a reduction in sediment yield by 40% (70% trapping of about 50% of the hillslope sediment runoff – giving 35% reduction – plus a further reduction due to 'retirement' of about 6.7% of the catchment). This benefit (= 40%) is assumed to be 'instantaneous'; in practice it may take a matter of years, but we do not know how to represent the time-response in detail. About 10-15 years after planting the shade of growing native plants starts to reduce the vigour of riparian grasses leading to streambank erosion. Bank erosion, and therefore sediment yield, peaks at 25 years after planting, and at the peak (about 140 tonnes y^{-1} from each km of permanent stream), the total sediment yield (hillslope plus bank sources) is about twice the hillslope pasture yield. Bank erosion and sediment yield then declines, reaching low levels when the stream is assumed to have widened to 'forest' morphology by about 35-40 years after planting. Sediment yield is greater than originally for about 12 years from about 19 to 31 years after riparian fencing/planting (Figure 4.2).

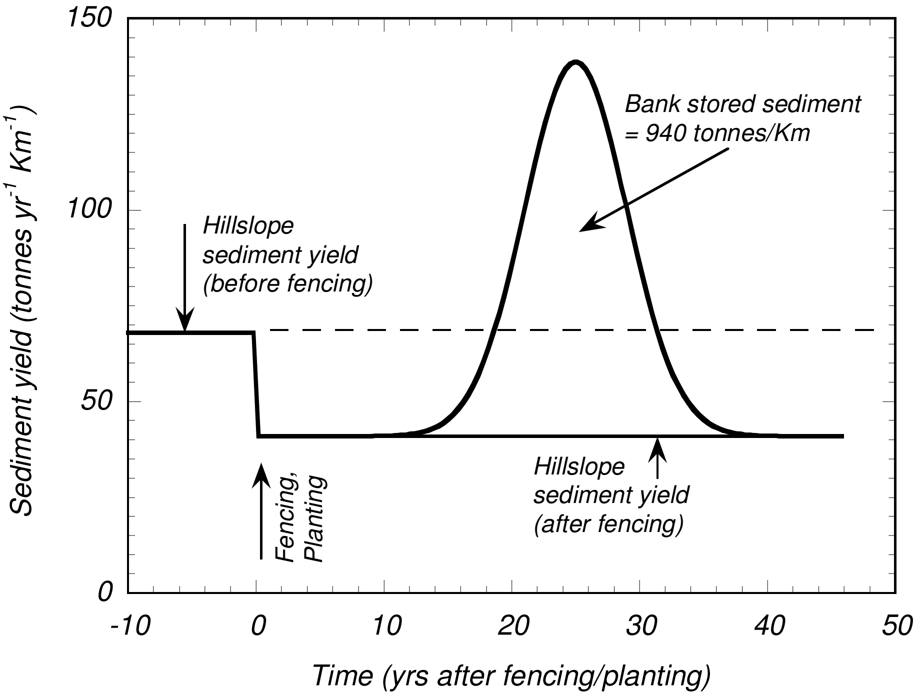

Figure 4.2: Predicted sediment yield per km of representative (permanent) pasture stream channel in the Auckland Region. Prior to riparian management action, sediment yield is assumed to derive only from hillslope erosion at the rate of 68 tonnes km-2 yr-1. Immediately after fencing and planting of a 15 m buffer zone either side of the stream channel, hillslope sediment yield is assumed to fall to 41 tonnes km-2 yr-1 owing to retirement of 6.7% of the incremental catchment of the 1 km of stream and entrapment of about 35% of sediment runoff in the riparian buffer. For a period of time, peaking 25 years after planting, erosion of 940 tonnes of sediment stored in the 1 km of streambanks is assumed to contribute to total sediment yield following a Gaussian trend.

It is interesting to note that about 36 years after planting, the sediment yield 'benefit' from a 40% reduction in hillslope sediment is just balanced by the 'side effect' of the increased sediment from bank erosion. Thereafter it is 'all downhill', with a net benefit to sediment yield from the riparian fencing and planting.

The Gaussian trend line in Figure 4.2 is just that, a *trend*. In practice, most of the geomorphic work, and most of the sediment yield, will occur in very large floods that produce bank-full or greater flows (that is, we expect the sediment rating curve to steepen during the bank erosion phase). The instantaneous sediment yield will be a very variable quantity, depending mainly on flow, but also on bank strength as it is affected by pasture turf and therefore by riparian shade.

We have deliberately presented the sediment yield trends in Figure 4.2 in terms of km of representative permanent stream length in the Auckland Region. The peak sediment yield, coincident with peak bank erosion, corresponds to perhaps doubling of pasture, hillslope sediment yield. However, this is for a hypothetical km of stream that is 'instantly' planted (say in one season). This planting effort at 1 m^{-2} plant spacing would require many thousands of plants to be 'instantaneously' planted. In practice, the planting will be likely spread out over a fairly long time, perhaps decades.

To explore this concept we modelled the sediment yield in a whole catchment (on a km^2 area basis), assuming that planting occurs on 20% of the catchment's permanent stream length on each of 5 successive years, 10% on each of 10 years, and 5% on each of 20 successive years (Figure 4.3). It can be seen that protracted planting 'spreads out' the sediment peak from bank erosion, such that, with planting over about 20 years or more, the sediment yield at the (much reduced, albeit widened) peak is reduced to about half its height above the hillslope 'background', and with planting over perhaps 40 years the sediment yield never exceeds the sediment yield before management intervention. However, protracted planting also means that the 'balance point' of net benefits in terms of cumulative sediment yield is delayed. We expect that riparian planting of sizeable catchments, such as those of most estuaries in the Auckland Region, will inevitably be protracted (if only limited by the supply rate of suitable plantings), such that the sediment yield trend curve may be more like the '20 year' curve in Figure 4.3 than the single planting season curve (bold).

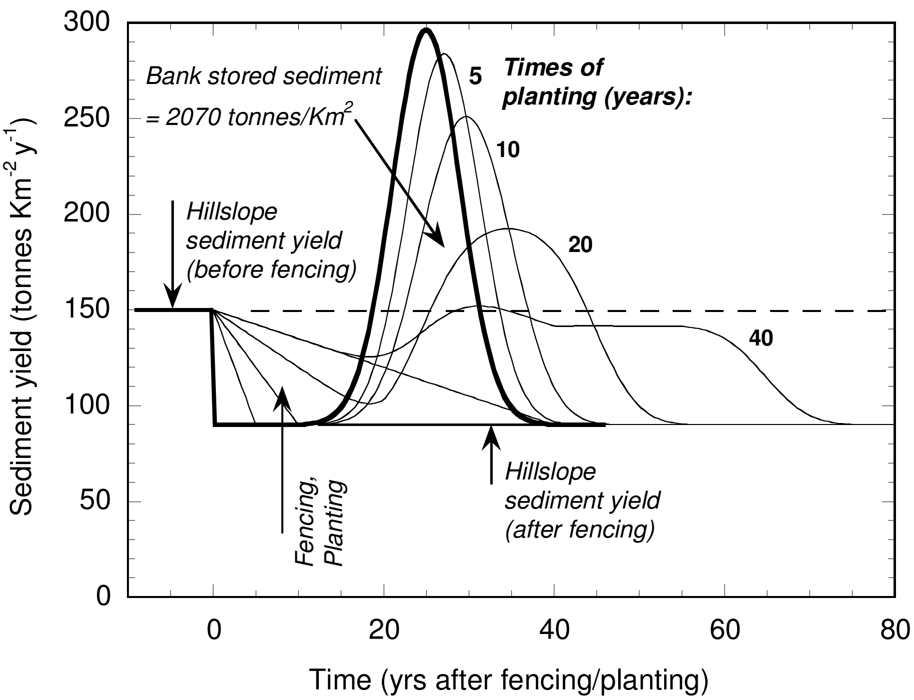

Figure 4.3: Predicted sediment yield per km2 of representative pasture catchment in the Auckland Region. (Assumptions as for Figure 4.2). Planting in just one season (bold curve – same shape as in Figure 4.2) is compared to protracted planting – spread out over 5, 10, 20 and 40 year periods

5 Effects of Estuary Sedimentation

With the current riparian management proposed for the Auckland region of a 10 – 20 m buffer of native vegetation, we can expect that 100% channel widening will occur in streams < c. 4 m wide diminishing to < 10% widening at streams of 9 m width in pasture (Davies-Colley 1997). This option is most likely to achieve improvements in stream habitat, so if this remains the preferred management option, we need to assess whether sediment loss from channel widening will have a significant impact on downstream estuaries.

5.1 Background – Factors that influence estuary sedimentation

In New Zealand, catchment deforestation from the mid-1800's, and more recently urban development, have permanently altered sediment inputs to estuaries. Catchment development has not only accelerated estuary infilling but also altered the characteristics of estuarine sediments such as sedimentation rates and particle size and introduced contaminants (e.g., heavy metals). These physical changes can result in major shifts in estuarine ecosystems, such as the displacement of the pre-existing benthic communities and, in upper North Island estuaries, rapid mangrove colonisation of intertidal flats.

In Auckland and the Coromandel Peninsula, where a number of estuarine systems have been studied, a general trend of accelerated sedimentation, following catchment deforestation is apparent. Background net sedimentation rates (<1 mm yr^{-1}) have greatly accelerated following catchment deforestation and conversion to agricultural, production forestry and urban land uses (i.e., up to 20+ mm yr^{-1}) (Hume 1983, Hume & McGlone 1986, Hume & Dahm 1992, Oldman & Swales 1999, Swales & Hume 1994, 1995, Swales et al. 1997, Vant et al. 1993, Williamson et al. 1998).

Estimates of estuary sedimentation rates, usually expressed as average annual values, give an impression of a constant rate of soil erosion. In reality, individual catchment flood events may supply a large proportion of the annual sediment load. For example, Swales et al. (1997) found that a single flood in May 1985 supplied 75% of the annual average sediment load delivered to the Mahurangi Estuary. Similarly, in the Pakuranga catchment 45% of the annual sediment load was delivered by the same May 1985 flood (Williamson et al. 1998).

Environmental factors influencing estuarine sedimentation primarily relate to the supply and deposition of sediments. Following human settlement, a large proportion of sediments delivered to estuaries has been catchment derived. Whether or not these catchment inputs are trapped in an estuary is determined by the physical and biological processes that characterise estuaries. These factors are briefly reviewed below.

5.1.1 Catchment Factors

The two primary factors determining the supply of catchment sediments are:

- soil erosion rate
- relative catchment size

The soil erosion rate integrates a number of catchment characteristics; landuse, climate soil type (i.e., particle size), and slope. Land use effects on soil erosion are well documented, and pastoral land use yields more sediment than forested catchments (Stroud & Cooper 1997, Van Roon 1981, Williamson et al., 1998). The estuary/catchment size ratio is another key factor influencing estuarine sedimentation. Generally, the rate of estuary infilling varies in proportion to the relative catchment size. For example, the ratio for the 25 km^2 Mahurangi Estuary is 0.2 (catchment 122 km^2) whereas the ratio for the 2 km^2 Wharekawa Estuary (Coromandel Peninsula), is only 0.02% (catchment 100 km^2).

In the Mahurangi Estuary, sedimentation has increased substantially following catchment deforestation, however today it still retains a large sub-tidal area (35%) and the fine catchment-derived sediments are episodically reworked by waves and currents. There is some evidence that the sediment trapping efficiency of the estuary has also declined as it has infilled (Swales et al. 1997). The Wharekawa Estuary is an extreme example of an infilled estuary where the rate of catchment soil erosion has exceeded the ability of physical processes to sort and flush sediments (Swales & Hume 1995). Today the estuary is almost entirely intertidal (90%), having rapidly infilled with poorly sorted sand and gravel (up to 20 mm yr^{-1} since 1945) derived from a steepland catchment. Coincident with infilling over the last 50 years has been the rapid expansion of mangrove on the intertidal flats. There is no mechanism to flush catchment sediments - tidal currents are weak and of limited duration and waves are ineffective. Rapid infilling of the Wharekawa Estuary is a direct consequence of catchment deforestation and the large catchment size.

5.1.2 Estuary Factors

The fate of eroded soils, once delivered to an estuary, will depend on the physical and biological factors that characterise estuaries. These include:

- tidal currents and waves
- salinity regime
- sediment particle size
- biological factors

Tidal currents and waves are important physical mechanisms influencing estuarine sedimentation. The flood tide may occupy only 3-4 hours of the 12.4 hour tidal cycle and consequently current velocities are higher. Because of the resistance to erosion and low settling rates of muds (diameter < 62.5 μm), flood tide dominance favours progressive fine sediment transport towards the head of the estuary. However, in estuaries that have substantially infilled (e.g., Wairoa) tidal currents alone may not be sufficient to mobilise muddy sediments, particularly away from the tidal channels.

In the long-term, estuary infilling reduces the volume of tidal water exchanged with the sea (i.e the tidal prism), reducing tidal current speeds and therefore sediment reworking. The tidal prism is also an important factor influencing estuarine morphology. An increase or decrease in the tidal prism implies compensatory erosion or deposition on the channel bed and estuaries infill as do the tidal channels.

In estuaries, sedimentation of particles < 20 μm diameter is enhanced by flocculation in salt water and can occur at salinities as low as 1-2 parts per thousand (‰ – the salinity of sea water is 35‰). The aggregation of individual particles to form flocs significantly enhances their fall speeds and hence flocculation is an important process influencing estuarine sedimentation. Rapid sedimentation rates measured at the head of many estuaries are attributable in part to flocculation.

Estuarine sedimentation is also influenced by density differences (i.e., stratification) between saline tidal water and freshwater discharged from the catchment. In partially mixed estuaries, (i.e., Auckland estuaries) this density difference drives a landward-directed residual flow of salt water near the bed. As a result, a zone of high suspended sediment concentration, the turbidity maximum, occurs at the head of such estuaries (Dyer 1986). This is a zone of net deposition. Flood events episodically interrupt the partially mixed regime and stormwater may displace seawater entirely from some estuaries. In this situation a proportion of the flood sediment load may be discharged directly into the sea. In estuaries that are largely infilled, the likelihood of sediment bypassing during floods is high because the estuaries are generally short and have low tidal volumes such that even moderate sized catchment floods (e.g., annual return period) displace seawater and discharge their sediment load outside the estuary.

Biological factors also have a significant influence on estuarine sedimentation. Plants and animals can modify the physical and chemical characteristics of estuarine sediments and in doing so influencing estuarine sediment dynamics. In many estuaries, salt marshes and mangroves trap large quantities of fine suspended sediment that may not have otherwise been deposited because of their low settling rates. Sedimentation in these environments is enhanced by direct adhesion of particles to stems and leaves and by reducing tidal current speeds and attenuating waves, creating quiescent conditions where particle settling can occur. Swales & Morrisey (1998) have reviewed the effects of saltmarsh on estuarine sedimentation processes.

In Auckland and the Coromandel, colonisation and/or expansion of mangrove has been documented in many estuaries, which indicates significant environmental changes. Mangrove colonisation follows sedimentation, and in Auckland, the expansion of mangrove has been particularly rapid in the last 20-30 years, coinciding with catchment urbanisation (e.g., Swales 1989).

In summary, infilling of estuaries is likely to be influenced by: (1) increased inputs of sediment from soil erosion in catchments (2) tidal asymmetry (i.e., progressive sediment transport towards head of estuary); (3) flocculation of fine suspended sediments; (4) residual circulation, due to vertical density stratification, transporting suspended sediments to the upper estuary; (5) biological factors, including binding of sediment deposits by plants and animals – increasing sediment resistence to erosion and trapping of fine suspended sediments by saltmarshes and mangroves.

5.2 Ecological Effects of Sediment on Estuaries

NIWA have produced critical catchment sediment loads for the Okura estuary in an attempt to link the risks of sediment runoff from development (i.e. earthworks) to a potential ecological effect in the estuary (Cooper et al. 1999, Green & Oldman 1999, Norkko et al. 1999, Stroud et al. 1999, Nicholls et al. 2000). A critical sediment level of 2 - 3 cm thickness, above which benthic organisms would die, was established using field and laboratory investigations. Polychaetes and bivalves (cockles, wedge shells, and pipis) were shown to be very sensitive to sediment deposition and crabs were the only animals able to survive sediment levels of 3 cm thickness by climbing through the deposited layer. Recovery from deposition can also take a considerable length of time and depends upon the rates of mixing and reworking of sediment by currents.

Using computer models of currents and sediment transport in the estuary, NIWA predicted the minimum sediment load ("critical catchment load") from a single flood event that could result in a 2 cm thick deposition over an area of 100 m^2 for each of the different sub-environments of the Okura estuary. Critical catchment loads ranged from 460 – 36 419 tonnes per event depending on bed sediments, exposure to currents, and distance from the head of the estuary. Deposition is most likely to occur near where streams enter the estuary (often the sediments are already muddy) and least likely to occur where currents are swift. There is a wide variation in the critical loads within the different areas of the estuary making an evaluation of a general critical value difficult. To avoid significant ecological damage to any part of the estuary, critical loads would have to be below 460 tonnes in any one event.

The average number of times the critical load is exceeded in a year under existing land use for any of the sub-environments of the Okura Estuary is up to 1.3 times. The Okura estuary has an annual average sediment load of 117 tonnes km^{-2} yr^{-1}. Three scenarios of land use development (Stroud et al. 1999) were predicted to increase the catchment load to 238, 514, and 1145 tonnes km^{-2} yr^{-1} and this would produce exceedences of up to 2.5, 5.3 and 9.1 times per year.

Is the level of sediment loss from channel widening significant?

In Section 4.3, we estimated that the amount of sediment stored in stream banks would double the average sediment yield of a representative pasture catchment in the Auckland region at the peak of erosion (Fig. 4.3). For the Okura estuary, this might produce sediment levels similar to that of Scenario 1, and exceed critical catchment loads in the estuary approximately 2.5 times per year for about 5 years at the peak of erosion. However, this worst case scenario assumes that all perennial streams in a catchment will be planted instantaneously. In reality this will not be the case and it may take 20 years or more for the whole catchment to be planted. In this case, the peak amounts are greatly reduced and the amount of sediment produced (that exceeds background hillslope sediment yields) is roughly similar to the amount reduced by initial retirement and exclusion of stock (see Fig. 4.3). Thus, peak sediment yields are estimated to be increased by a third over a 5 – 10 year period. It may take even longer, say 40 years for the whole catchment to be planted, which may therefore produce erosion levels that are less than the current sediment yields under pastoral land use.

This report gives an indication of the sediment yields likely to be produced by channel widening under planted riparian management and some idea of the ecological impact of this sediment on estuaries (based on critical catchment loads for one estuary). This report does not attempt to give a specific account of the impact on biota in estuaries from the sediment released from stream banks. It is recognised that estuary environments are complex and effects of sediment deposition are variable within estuaries due to a variety of physical and biological factors.

We expect that riparian planting programmes will be protracted and may take up to 20 years for whole catchments to be planted. Thus, the amounts of sediment released in flood events are likely to produce thin layers of deposited sediments in estuaries (worst case scenario for instantaneous whole catchment planting of exceeding 2 cm thickness = 2.5 times per year for 5 years). However, it is recognised that even thin layers of sediment deposition can have detrimental ecological effects on some biota and there may be long term chronic effects of sediment deposition (Norkko et al. 1999).

6 Planted Riparian Buffer Zones – Are they worth it?

6.1 Background

The Auckland Regional Council would like to promote riparian buffer zones that are planted zones of sustainable regenerating vegetation that will suppress weed growth and require minimal maintenance. Parkyn et al. (2000a) suggested that a buffer of 10 – 20 m would be sustainable in the Auckland Region and would achieve most aquatic functions required of a buffer zone. However, certain limitations were identified, in particular that grass buffers may be better than tree species as a filter for sediment and nutrients. Grass buffers are known to be more effective at filtering nutrients and sediment than forested buffers, and often multi-tiered buffers of forested riparian zones with upslope grass buffers are recommended as riparian management.

This limitation recognises that planted buffer zones may not achieve all aims of riparian management. In fact, water quality and stream habitat goals should almost be considered as separate parts to riparian management. Water quality improvements will benefit downstream environments (such as lakes and estuaries) as well as streams themselves, whereas stream habitat improvements will generally be localised or primarily improve stream systems. In addition, we have found that planted riparian zones can increase nutrients delivered to estuaries if whole catchments are not planted from the headwaters down (Section 3.2) and that sediment yields will increase while stream channels are widening to forest widths, particularly in the headwaters (Section 4).

6.2 What Improvements to Stream Habitat can be Expected with Planted Riparian Zones?

Riparian planting effects on stream habitat for aquatic biota include:

- 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, once channel conditions have stabilised and (3) tree roots that stabilise the new (wider) stream banks.
- 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)

- provision of woody debris as trees fall into streams over the long term, providing habitat diversity and cover for aquatic invertebrates and fish.

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 > 20°C, temperatures that are commonly exceeded in open pasture streams. Rutherford et al. (1999, 2000) used computer models to show how high 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 2000). 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.

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 22-23°C. 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 30°C in the clear cut area to 25°C in the buffer zones.

6.3 What Evidence do we have that Riparian Zones are effective at achieving Habitat Goals?

Parkyn et al. (2000b) 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 are 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 communites 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.

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, but were unable to show associated changes to the macroinvertebrate or fish communities.

Therefore, it is likely that the time-scale of habitat recovery and subsequent recovery of invertebrates and fish is longer than most riparian planting efforts in this country. However, evidence from buffer zone surveys and studies of landuse on stream communities (Quinn et al. 1997, Quinn et al. 1992) suggests that shade and lowered stream temperatures, only achieved by planting in buffer zones, will enable invertebrate communities to recover over long time-scales.

6.4 Difficulties with 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 a long time 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).

Therefore, if riparian trees do shade out a significant proportion of macrophytes in soft-bottomed lowland sites, invertebrate biodiversity may potentially decline for a period until sufficient amounts of woody debris fall into streams and provide an alternative stable substrate. However, as noted above, recent evidence from a Waikato lowland stream indicates that heavy shade might lead to a shift in community dominance of macrophytes rather than total exclusion.

7 Options for Riparian Management

This report has shown that buffer zones that are retired from grazing and planted with (or allowed to revert to) native tree species, have the potential to impact on instream nutrient processing, and result in channel widening. Options to mitigate some of these effects are outlined below.

7.1 Nutrient attenuation

The impacts of increased dissolved nutrient export distances brought about by shading out of instream plants will vary depending on the type of environment that is downstream (e.g., an oligotrophic estuary will be more sensitive than a turbid lowland river). With sensitive downstream environments:

1) ensure that planting programmes begin in the headwaters (not valley wetlands) and progress downwards, so that downstream macrophytes can assimilate any nutrients transported from upstream.
2) promote the protection of riparian and catchment wetlands as these are areas of denitrification
3) provide for a grassy buffer zone outside of the planted area to enhance the effectiveness of the buffer for intercepting contaminants in overland flow
4) encourage the use of suitable tree species whose fallen leaves promote instream uptake of nutrients (e.g. soft-leaved species that decay quickly)
5) where plantings do not begin in the headwaters and progress downstream, sparse plantings of deciduous trees can be used to maintain some level of instream primary productivity. Introduced tree species are able to provide at least some of the instream functions, like shade, that native trees provide, and can be preferred over some native species as food for leaf-shredding stream invertebrates (Parkyn & Winterbourn 1997). Alternatively, space plantings of native trees could be attempted.

7.2 Channel Widening

To prevent channel widening from occurring with riparian management there are a number of potential options:

1) Retire stream sections from grazing and maintain as a grassy sward for sediment and nutrient trapping. (Provides no improvement in stream or terrestrial habitat).

2) Attempt to control the level of shade in riparian areas. To maintain stream bank stability, Rutherford et al. (1999) recommended a maximum bank side shade level of about 70%. One strategy to achieve this could be to fence along the stream and space plant deciduous trees, which provide dense shade for the streamwater during critical summer periods, but which allow bank side groundcover vegetation to develop during spring and autumn. (Provides some improvement in stream habitat through temperature reduction and carbon inputs, but management through pruning and weed control would be required).

3) Maintain high light environment alongside small streams. Densely plant with native grasses immediately adjacent to the stream, which will provide some shade to small streams. (Once riparian areas are retired from grazing, successional processes are likely to eventually lead to taller woody species establishing in the riparian areas unless active management prevents this. Therefore, channel widening will ultimately occur, but may be delayed for many decades).

7.3 Potential loss of Plantings

The widening of re-vegetated pasture streams poses the issue of whether some riparian planting beside pasture streams will eventually be lost. As stated elsewhere in the report, this will depend on whether a sufficiently dense ground cover can be maintained over the first 2-3 m adjacent to the stream channel.

The draft ARC planting guidelines (Julian, 2001) recommend planting sedges within the area that is likely to widen (i.e. the stream bank and flood area planting units). These sedges are likely to form a dense enough ground cover (particularly the sward forming *Carex lessioniana*) to prevent widening from occurring. The drawback of these low-growing sedges is that they will only provide desirable levels of stream shading in small streams (1-1.5m in width). It is likely that other taller growing native species will eventually establish in these areas particularly if these species (e.g. kahikatea, putaputaweta, cabbage tree) are planted adjacent to the sedges as recommended in the draft guidelines. It may however take many decades before these taller plants establish and form a dense enough canopy to reduce ground cover. Some of the shade tolerant sedges are likely to persist (e.g. *Carex dissita*) but not in sufficient density to retard fluvial scouring.

The amount of riparian planting that could be lost will depend on the expected increase in stream width and the density of plantings. The Christchurch City Council's (1998) Streamside Planting Guidelines recommend planting densities of 0.3-0.5 m spacings for grasses and sedges and 1 m spacings for shrubs and trees. Assuming a doubling in stream width with channel widening (Davies-Colley 1997), streams that widen by 3 m as a worst case scenario would result in a 5-20% loss of plants in 10-20 m riparian margins at the above densities.

There are several possible options for avoiding the loss of riparian plantings:

1) All planting could be set-back to allow for the approximate doubling in channel width of streams less than would ultimately be about 4-5 m wide in forest (2-2.5 m wide in pasture). A lesser degree of setback will be required along wider streams (the curves in Fig 2 of Davies-Colley (1997) may be used to predict the likely degree of widening).

   In practice, it will not be necessary to setback 50% on both sides of the existing (pasture) stream channel as the expected bank erosion will occur mostly in the small flood plains that form in pasture. These flood plains occur one side or the other of the valley bottom as the stream channel meanders in pasture. These mini-floodplains should be recognised on site and be avoided for planting. Similarly, where valley-bottom wetlands occur, these are expected to erode and could also be avoided for plantings. This will also help retain their nutrient removal functions.

   One consequence of these attempts to avoid erodible features of the riparian morphology in pasture is that the canopy gap over the stream channel/wetland will be larger than otherwise and canopy closure will take rather longer than otherwise. This will delay stream shading and the accompanying benefits to in-stream life but may help reduce the severity of sedimentation/turbidity problems during the stream channel adjustment phase.

2) As mentioned elsewhere in the report, attempts could be made to maintain the 70% maximum shade level suggested by Rutherford et al. (1999) for maintaining sufficient ground cover. An alternative, but with the same outcome, would be to only plant sedges and native grasses within 5-7m of the stream edge to maintain a dense ground cover. Both of these options are likely to be only short term solutions unless there is considerable management effort to prevent these areas being colonised by taller growing natives.

3) Another option is to accept the eventual loss of some riparian plantings as the stream widens. This has several benefits in that it will help achieve stream shading much sooner, provide additional terrestrial habitat and increase the amount of detritus and woody debris to the stream as the stream widens. Remember that channel widening will only begin to occur 15 – 20 years from planting.

8 Conclusions

This work has shown that increases in nutrient export can be expected when streams are shaded (limiting macrophyte and algal growth) and when there are no buffer zones in upstream areas to limit nutrient inputs. While changes towards shade-tolerant macrophyte communities may occur, it is not known whether the amount of nutrient assimilation will be equivalent to open stream sections.

Similarly, shade will affect the stability of stream banks and is likely to cause channel widening. A large amount of bank-stored sediment will be lost downstream, but the magnitude of impact will depend on the length of time that a whole stream system is planted. If planting takes 20 to 40 years, the amount of sediment lost may be similar to current catchment sediment yields, assuming an initial decrease in catchment sediment through the stabilisation and filtering actions of the buffer.

The options for riparian management that relate to these issues can be summarised as follows:

1) Retire from grazing, no planting, active management to control plant growth (e.g. mowing for hay, weed control)
   - instream macrophyte nutrient attenuation possible, filters overland flow
   - no channel widening expected
   - small improvements in stream habitat expected – (e.g. rank grass in small streams may shade channels and retirement from grazing may improve stream habitat)
2) Space plant deciduous trees, possibly with controlled grazing –
   - may provide enough light for channel stability and to allow macrophyte growth for associated instream dissolved nutrient attenuation;
   - some improvements in stream habitat can be expected
3) Attempt to create low growing, dense, native grass community (requires high maintenance)
   - open light allows macrophyte growth and prevents channel widening from shading
   - filters overland flow
   - shade benefits to stream habitat only in small streams;
   - long-term success unlikely without ongoing active management to maintain herb community.

4) Ensure dense planting of native tree species in buffer zones begins in headwaters; leave riparian and catchment wetlands intact

- negates the need for macrophyte nutrient attenuation downstream - assumes most runoff carrying nutrients passes through wetlands
- channel widening and loss of plantings close to stream banks is likely to occur leading to changes in sediment yield depending on rate of planting
- improvements in stream habitat and terrestrial biodiversity expected

We have established that the expected benefits to stream habitat, particularly lower water temperatures, will occur where appropriate levels of riparian tree planting occur (i.e. depends on the length of reach that is shaded and stream depth etc), but over long time scales. Planting back from the stream channel to allow for the eventual stream width will prevent loss of plantings and slow the process of channel widening, but also slow the recovery of stream habitat (e.g. shade and lower water temperature).

Therefore, we conclude that tree planting in riparian areas will be beneficial to stream habitat. To lessen the impacts on downstream environments, planting should begin in the headwaters and progress downstream, and whole catchment planting should be extended over several decades.

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