

Figure 41 Maximum daily water temperatures predicted in uniform stream channels under partial shade of 75% (squares), 50% (triangles) and 25% (circles). Temperatures at 0 km are equilibrium temperatures predicted under heavy (95%) shade. Channel parameters are given in Table 13.

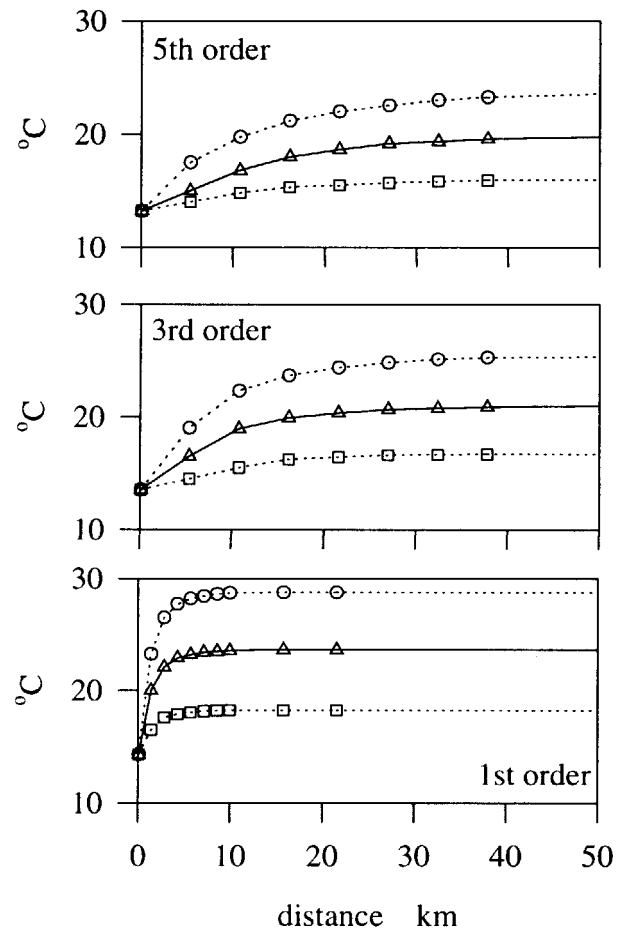


Figure 42 Maximum daily water temperatures predicted in uniform stream channels under heavy (95%) shade. Temperatures at 0 km are equilibrium temperatures predicted under shade of 75% (squares), 50% (triangles) and 25% (circles). Channel parameters are given in Table 13.

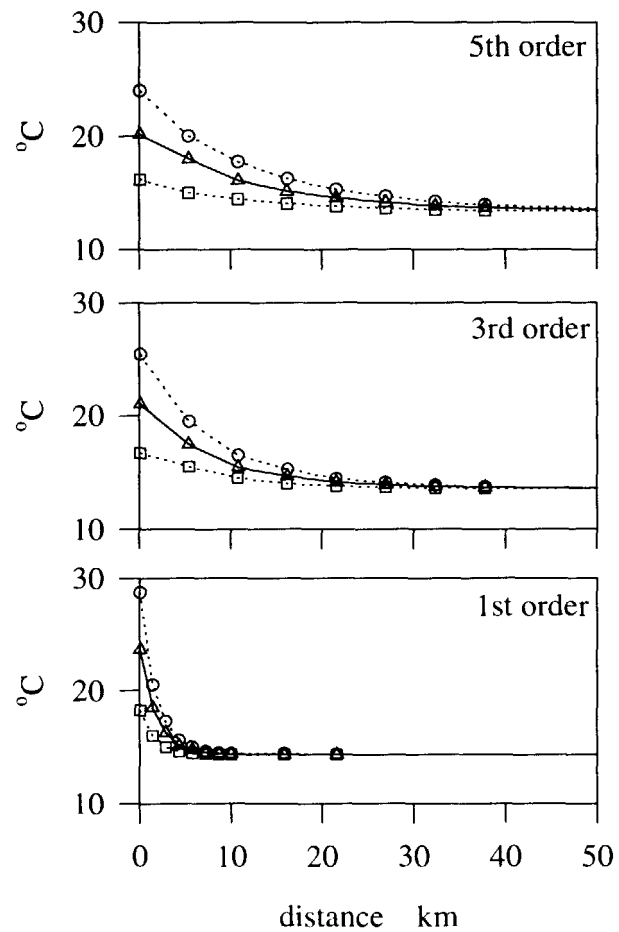
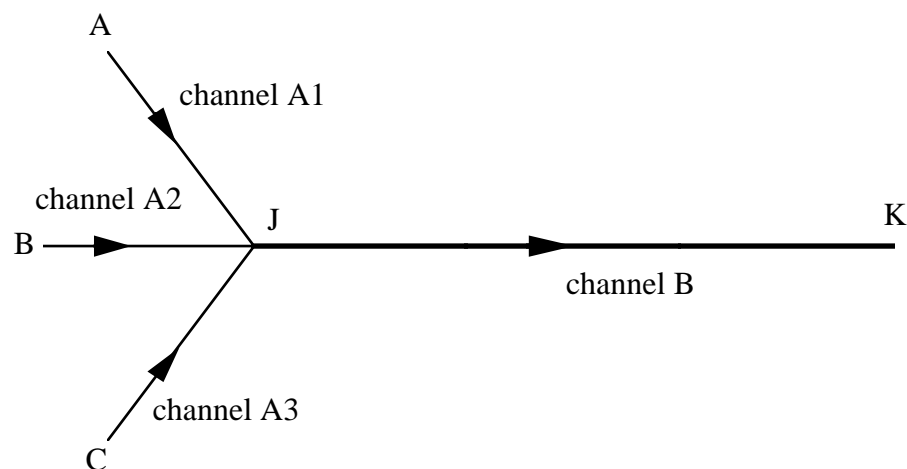


Figure 43 Sketch of three identical first-order channels (A1-A3) each 500 m long which meet at point J to form a 1500 m third-order channel (B).



Consider the stream network shown in Fig. 43. Three identical first-order streams leave the bush (where they are assumed to be at equilibrium under 95% shade) at points A, B and C. They then flow for 500 m through open (25% shade) pasture (channels A1, A2, A3) before meeting at point J to form a third-order channel (channel B). Channel B then flows for another 1500 m through open (25% shade) pasture to point K. First consider the likely daily maximum stream temperatures. From Fig. 41 the equilibrium bush temperature in a first-order stream at points A-C is 14.2°C. After 500 m of 25% shade this temperature can be expected to increase to 18°C. For a third-order stream, Fig. 41 indicates that a temperature of 18°C occurs at 4.5 km in a channel with 25% shade. Thus in Fig. 41, the starting point for assessing temperature changes in channel B is at a distance of 4.5 km. Figure 41 predicts that a further 1.5 km downstream the temperature is 19.5°C. Let us examine two extreme options for reducing stream temperatures. In both cases 1500 m of stream planting is required. The question arises whether the two options achieve the same degree of temperature reduction at point K. Option I: restore 95% shade to reaches A1, A2 and A3 while leaving reach B unshaded, Option II: leave reaches A1, A2 and A3 unshaded while restoring 95% shade along reach B.

Option I: Because channels A1, A2 and A3 are fully shaded, their temperature at point J remains at 14.2°C. From Fig. 41 a temperature of 14.2°C occurs at 700 m in a third-order channel with 25% shade. This is the starting point for assessing the temperature increase in channel B. Figure 41 predicts that heating along a further 1500 m of a third-order channel with 25% shade increases the temperature to 16.5°C. Option II: After 500 m of 25% shade the temperature of channels A1, A2 and A3 is predicted by Fig. 41 to increase from 14.2°C to 18°C. This is then the initial temperature of the third-order stream at point J. From Fig. 42 a temperature of 18°C occurs at 8000 m in a third-order channel with 95% shade. Then Fig. 42 predicts that cooling along a further

1500 m of heavily shaded (95%) channel reduces the temperature to 17.5°C. In this example, shading 1500 m of first-order stream channel (Option I) gains an additional temperature reduction at point J of the order 1°C over shading the same length of third-order stream (Option II).

If there is important habitat (e.g., for fish or invertebrates) along the *entire length* of each of channels A1-A3 and B, then it may not be sensible to compare the predicted temperatures for Options I and II at just a single point (e.g., point K). Rather, for each option it may be important to integrate the water temperature along each channel and then compare the total degree-kilometres between Options I and II. This is illustrated as follows. *Option I:* The average temperature in channel B is approximately $(14.2+16.5)/2 = 15.4^{\circ}\text{C}$, giving a total of 23.1°C-km in channel B. In channels A1-A3 the temperature is constant at 14.2°C giving a total of 21.3°C-km in channels A1-A3. The total for Option I is then 44.4°C-km. *Option II:* The average temperature in channels A1-A3 is $(14.2+18)/2 = 16.1^{\circ}\text{C}$, giving 24.2°C-km. The average temperature in channel B is $(18+17.5)/2 = 17.75^{\circ}\text{C}$, giving 26.6°C-km. The total for Option II is 50.8°C -km, which is 6.5°C-km (14%) higher than for Option I for the same total length of riparian shading. This comparison again shows that it is better to shade 1500 m of first-order stream channel than the same length of third-order stream.

4.8 Microclimate effects

The most obvious effect of removing riparian vegetation is to increase the amount of solar radiation which reaches the stream water surface. Model predictions described above (e.g., Figs 40–42) assume that wind speed, humidity and air temperature are the same in native bush and pasture and so the predicted temperature increases. Riparian vegetation has the potential to change the microclimate surrounding the stream. Riparian vegetation (especially trees) increases friction near the stream channel and decreases wind speed across the water surface. This reduces the rate of exchange between the air mass above the stream channel and that above the surrounding land, which reduces the rate of evaporation. Because water vapour which evaporates from the stream is dispersed more slowly from above a vegetated channel, water vapour pressure (and hence humidity) tend to increase. This further reduces the rate of evaporation because it reduces the vapour pressure deficit above the water surface which drives evaporation. The stream cools when water evaporates because the latent heat of evaporation must come from the water. Thus the effect of riparian vegetation is to reduce the rate of evaporation and hence to increase the water temperature. Within the riparian canopy, air, canopy and topography temperatures tend to decrease because of a reduction of incoming solar radiation. This reduces the longwave radiation emitted within the canopy, some of which reaches the water surface. The magnitude and direction of the net temperature change depends on the type, height and density of riparian vegetation and the width of the riparian zone. Thus we would expect that a single line of trees planted along the stream bank is unlikely to affect the microclimate

as much as reforestation of the entire catchment. We used the STREAMLINE model to predict the effects on equilibrium stream temperature of successively changing: wind speed, air temperature and humidity while flow, depth, solar radiation, topography and canopy shade were assumed constant. Note that changing the air temperature changes the incoming longwave radiation from the atmosphere, canopy and topography.

Figure 44 shows that if riparian vegetation increases the humidity (without altering any other parameter) then the stream water temperature increases uniformly throughout the day. Predictions indicate that in a second-order stream, humidity changes from 50 to 100% cause equilibrium water temperatures to increase by about 3°C because of decreased evaporative cooling and sensible heat loss.

Figure 45 indicates that as the wind speed increases, the rate of evaporation increases (because water vapour is carried away from the water surface more quickly) thereby accelerating cooling of the stream. Model predictions indicate that a change in average wind speed from 0 to 5 m s⁻¹ has the potential to reduce equilibrium stream water temperatures by about 5°C. This prediction is critically dependent on the accuracy of the wind speed function in the evaporation formula used in the STREAMLINE model: see Brocard & Harlemann (1976). This formula was derived in open waters (e.g., cooling ponds), but has been tested in sheltered streams and found to be satisfactory (Gulliver & Stefan 1986).

Figure 46 shows that air temperature also has a major impact on water temperature. The model predicts that, as the air temperature decreases (without any other parameter changing), so too do the longwave radiation fluxes emitted by the sky, canopy and topography. Counteracting this is the fact that, as the air temperature decreases, the saturated vapour pressure decreases, which in turn reduces the rate of evaporative cooling. The model predicts that the latter effect is smaller than the former and that, if the air temperature decreases by 5°C uniformly throughout the day, then the equilibrium water temperature decreases by about 4°C.

Figure 44 Effects of humidity on predicted temperature in a second-order stream (similar to PKL): shade factor 0.30, wind speed 1 m s⁻¹, air temperature 10-20°C.

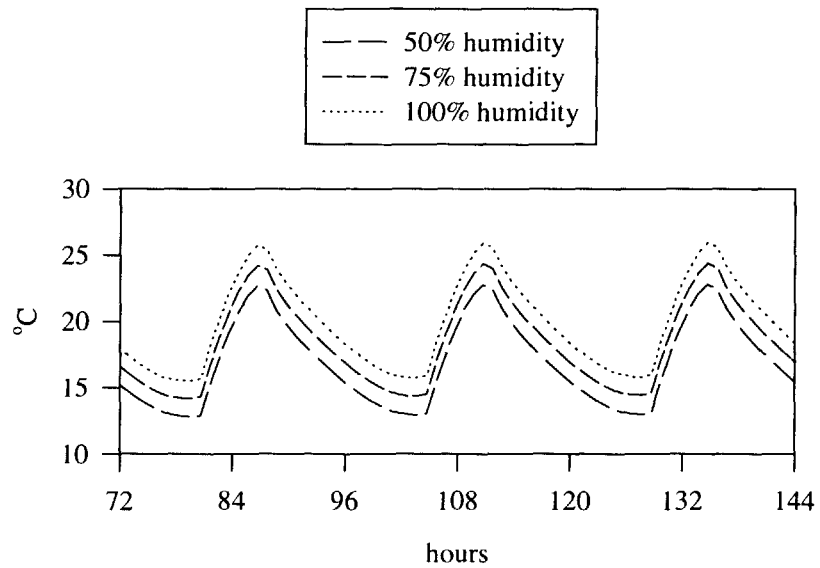


Figure 45 Effects of wind speed on predicted temperature in a second-order stream (similar to PKL): shade factor 0.30, humidity 90%, air temperature 10-20°C.

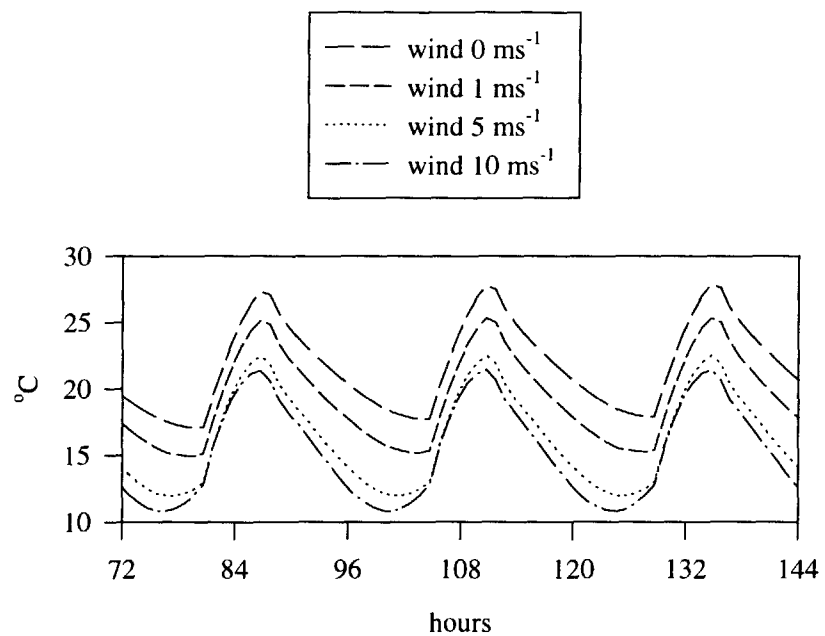
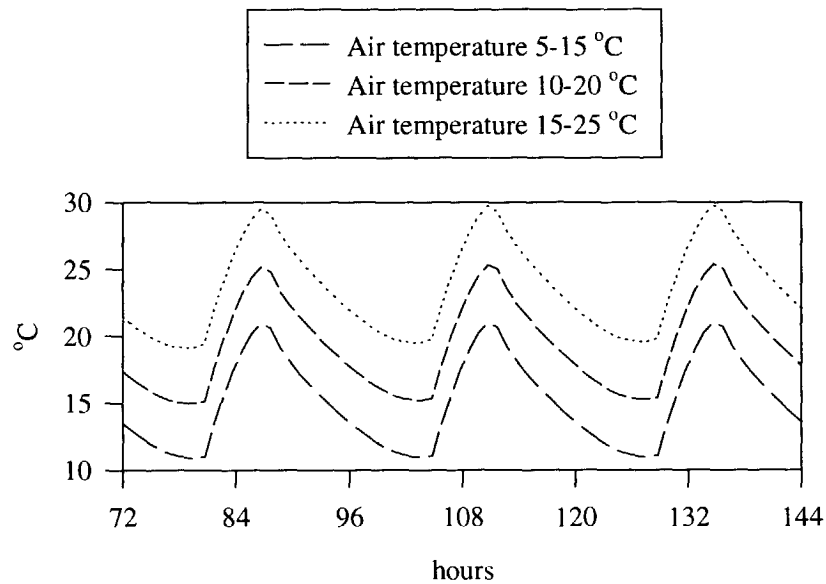


Figure 46 Effects of air temperature on predicted temperature in a second-order stream (similar to PKL): shade factor 0.30, humidity 90%, wind speed 1 m s⁻¹.



In practice the removal or restoration of riparian vegetation will change several parameters simultaneously. Thus, while Figs 44-46 give useful insights into the effects of each microclimate parameter separately, they are difficult to use for predicting actual stream water temperature changes. Unfortunately, there is very limited information available about the microclimate of stream channels and riparian buffer zones.

Young & Mitchell (1994) measured changes in air temperature, humidity and light along transects across the boundary between a mixed podocarp-broadleaf forest and pasture at a site north of Auckland. Figure 47 shows average light, humidity and air temperature profiles in March (late summer) which we estimated from Young & Mitchell (1994, fig. 4.2). The forest reduced light levels by 95-98% from those measured at open pasture sites. Strong gradients in air temperature and humidity were measured over distances of about 50 m. No measurements were reported of wind speed variations along the transects. There appears to be a fairly close inverse relationship between air temperature and humidity: between forest and pasture, air temperature increases by 5°C while relative humidity decreases by about 25%.

As a first approximation we can assume that the effects of solar radiation and microclimate on stream water temperature are independent and additive. This enables us to estimate, very roughly, the marginal effects of increased solar radiation, increased air temperature and decreased humidity on water temperature in a second-order stream. Table 14 summarises the marginal effects shown in Figs 40, 44 and 46. A rough estimate is that reducing shade by 65%, increasing air temperature by 5°C,

and decreasing humidity by 25% result in an increase in daily maximum temperature of the order 12.5°C . About 80% of this increase is attributable to increased solar radiation and 20% to microclimate changes.

Figure 47 Variation of microclimate at the edge of a podocarp-broadleaf forest. Average values for March (late summer) estimated from data in Young & Mitchell (1994).

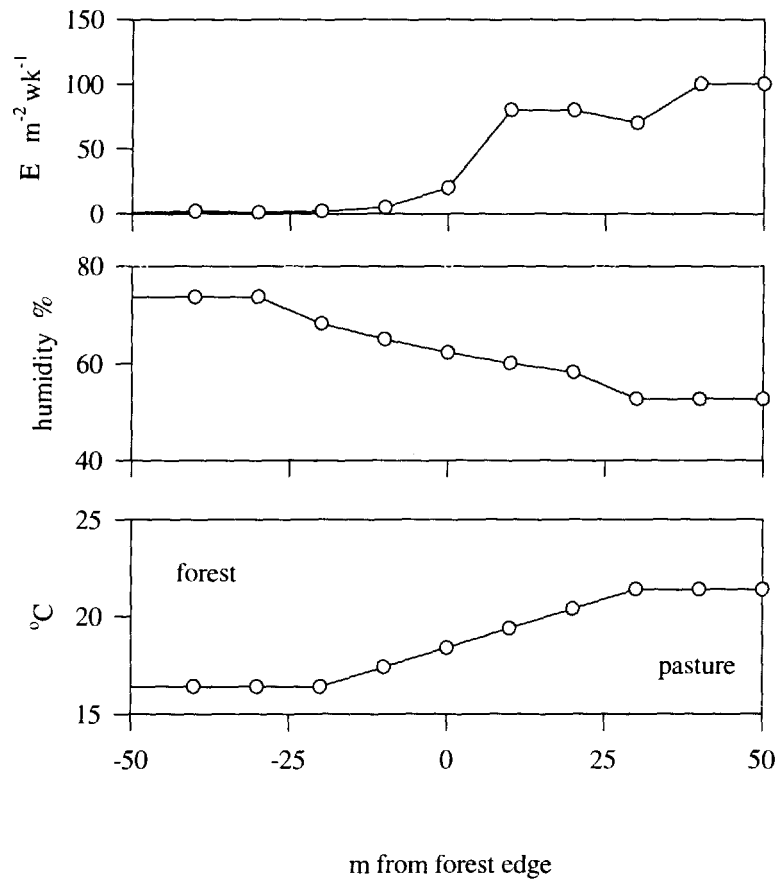


Table 14 Marginal effects on equilibrium water temperature of changing shade, humidity and air temperature

parameter	change	water temperature change	
shade	95% to 30%	+10°C (daily max)	Figure 41
humidity	75% to 50%	-1.5°C (mean)	Figure 45
air temperature	+5°C	+4°C (mean)	Figure 47

There is an obvious implication for the restoration of riparian vegetation. By far the most important effect of riparian vegetation is to reduce solar radiation inputs. If, however, in addition to decreasing the solar radiation input, riparian vegetation can be restored to the point where the microclimate is altered (notably air temperatures decreased), then stream water temperatures may be further reduced. Increases of humidity and reductions of wind speed resulting from restoration of dense riparian shade counteract the effects of decreased air temperature.

The question then arises: how wide does a riparian buffer strip need to be in order to change the microclimate sufficiently to affect stream water temperatures? Young & Mitchell's study indicates that the air temperature decreased by 5°C and the humidity increased by 25% over a distance of about 50 m. It is tempting to infer that a buffer strip 10 m wide would decrease the air temperature by 1°C and increase the relative humidity by 5%. This assumes a linear relationship between the width of the buffer strip and the microclimate parameters: something which is not yet established and, indeed, would not be expected. It probably requires only limited riparian vegetation to reduce wind speeds near stream channels: perhaps a single row of trees may be sufficient. It seems unlikely, however, that a single row of trees (especially if widely spaced and/or pruned for timber production) would result in substantial changes of air temperature and/or humidity. Sparse tree plantings may, however, increase shade sufficiently to reduce solar radiation inputs. Further review and experimental work is currently under way aimed at quantifying the minimum width of buffer strips required to alter the microclimate around streams of various sizes. These studies also aim to quantify, more precisely than can be done with the available information, the effects of microclimate on stream thermal response.

4.9 Summary

1. A simplified sub-model for topographic and canopy shading has been incorporated into a heat budget model of stream temperature. The shade sub-model requires only three coefficients: topography angle, canopy angle, and canopy shade factor. Topography and canopy angles can be defined in several azimuthal directions, while angles and shade factors can vary between sub-reaches.
2. There are two main simplifications in the shade sub-model. Firstly, radiation fluxes are assumed uniform across the width and equal to the mid-channel value. Secondly, the fraction of incoming radiation absorbed by the canopy is constant regardless of solar elevation. These simplifications lead to opposite (compensating) biases.
3. Despite the simplicity of the shade sub-model, a successful calibration was achieved for a small pasture stream at Whatawhata, near Hamilton. Predicted and observed water temperatures matched closely under two different sets of shade, flow, and bed conditions. In addition a close match was achieved between observations and predictions for bed temperature, and incoming and outgoing longwave radiation. Model predictions of shade matched reasonably well direct measurements of shade made using canopy analysers (Spier & van Veen 1994).
4. Further model refinements can be justified to calculate the average radiation flux across the channel rather than just at mid-channel and to allow the percentage shade to decrease with increasing solar elevation. In addition, further work is desirable to permit direct input of canopy analyser shade measurements into the model. This requires refining the way the canopy analyser is used and the results analysed so that measured shade values can be corrected for differences between visible light and NIR transmission through the canopy and so that canopy shade can be separated from topographic shade.
5. We now have reasonable confidence in the STREAMLINE model and have used it to simulate the effects of removing and restoring riparian shade in small streams (see for example Collier *et al.* 1995).
6. Measurements at Whatawhata indicate that there is a substantial amount of shading in small pasture streams (average 50%) arising from topography (hillsides and stream banks) and overhanging vegetation (grass, fern and sedges). Thus, in the South Auckland region 25% shade represents a likely lower bound for pasture streams, 50% is more typical and 75% may be typical of streams with sparse tree plantings along the banks.

7. Model predictions and field measurements indicate that in small streams (mean depth about 10 cm) which flow out of the native bush into pasture, the daily maximum temperature typically increases by 5–6°C over a distance of 600 m. This is largely the result of increased solar radiation inputs. The daily minimum temperature increases by a smaller amount, typically 1.0–1.5°C, over the same distance. It is not clear whether this is the result of the thermal inertia of the stream and its bed, or the altered microclimate (air temperature and humidity).
8. Nomographs have been developed from computer model predictions to help quantify the rates of heating and cooling in first-, third- and fifth-order streams when riparian shade is either decreased or increased. These nomographs are also discussed by Collier *et al.* (1995). Regional differences in meteorological parameters (notably air temperature and solar radiation) undoubtedly affect the *absolute values* of water temperature. Nevertheless, it is likely that the predicted *changes* in water temperature summarised in these nomographs will be similar throughout New Zealand in streams with comparable channel parameters.
9. An important conclusion from our field and modelling studies is that in shallow streams (e.g., first- and second-order), temperature changes of 5°C can occur over distances of the order 1–5 km, while in deeper streams (e.g., third- and fifth-order) not only are maximum temperature changes smaller but they typically occur over distances of the order 10–20 km.
10. The fact that the thermal inertia of small streams is small means that if low stream temperatures need to be maintained throughout a stream network (e.g., to maintain suitable fish or invertebrate habitat) then it is more important to maintain dense shade along the small streams (first- and second-order) than along the larger streams (third-, fourth- and fifth-order). Similarly, when attempting to reduce stream temperatures in a catchment (comprising a network of streams of different orders) it is more efficient to restore riparian shading on the shallow first- and second-order streams than on the deeper third-, fourth- and fifth-order streams. The nomographs and/or computer model can be used to help optimise riparian shade management in order to meet specified water temperature targets.
11. A preliminary estimate is that under extreme meteorological conditions a second-order stream flowing from native bush (95% shade) into pasture (30% shade) will eventually experience a temperature rise of about 12.5°C of which about 10°C is attributable to the increase in solar radiation and the remaining 2.5°C is attributable to changes in microclimate. This indicates that the major impact on water temperature of removing shade arises from the increase in solar radiation input. By inference the major impact of restoring riparian shade arises from the reduction in solar radiation input. If during stream restoration, in addition to providing shade, the microclimate can be altered (notably air temperatures reduced), then water temperatures can be further reduced.

5. SYNTHESIS: RESTORATION OF STREAM SHADE

5.1 Introduction

The restoration of stream shade may be the most effective means of improving water quality and habitat in pasture streams. Such restoration measures range between the two extremes of reforestation of the whole catchment, and providing a narrow riparian buffer strip. In our landuse comparisons (Section 2) we studied the difference between streams whose entire catchment had a single predominant landuse. Consequently our findings can be used with greatest confidence to forecast the effects of changing the predominant landuse in the whole catchment. Where possible, however, we have sought to understand the processes operating, and this improves our ability to forecast the effects of restoring only a narrow riparian buffer strip. The aim of this section is to assess the potential for riparian strips to buffer streams against the adverse effects of pastoral agriculture.

5.2 Effects of conversion to pasture

Table 15 summarises the effects of replacing native vegetation by pasture. This summary is based on our observations of streams at Whatawhata, near Hamilton, but many of the processes have been observed by us and others elsewhere in New Zealand.

High light levels in pasture streams result in high water temperatures. The landuse comparisons (Section 3) and the thermal budget and computer model studies (Section 4) indicate that daily maximum temperatures in summer are typically 5–7°C (occasionally 10°C) higher in pasture than forest streams. We believe that temperature increases of this order: reduce slightly dissolved oxygen concentrations in stream water, the benthic boundary layer (i.e., the thin layer at the interface between stream water and streambed) and the hyporheic zone (i.e., the streambed); increase respiration and periphyton growth rates significantly; and adversely affect sensitive invertebrates and fish.

High light levels stimulate primary production in pasture streams. This does not necessarily lead to high periphyton biomass because other loss processes may increase (notably grazing). Our observations indicate that in unshaded channels there is an increased likelihood of high periphyton biomass (occasionally approaching ‘nuisance’ levels) but that periphyton biomass is highly variable both in space and time.

Pasture streams are often severely impacted by fine sediment. Those studied at Whatawhata had high suspendable sediment loads in the streambed, high water-column suspended solids concentration, high turbidity and moderately low

clarity (Section 2). One obvious source of sediment was bank erosion: the pasture streams studied were narrow with steep banks, large parts of which were actively eroding. In pasture catchments, rainfall interception is low, runoff is rapid, and flood flows are high. This results in the streams having high energy to erode the bed and banks. Sediment also finds its way into streams in overland flow after being eroded from pasture within the catchment. Erosion tends to be higher in pasture than forest catchments. The hillcountry at Whatawhata is characterised by occasional landslides which significantly increase sediment loads in the streams.

Differences in water temperature, primary production, periphyton biomass, hydrology and sediment between pasture and forest streams contribute to differences in their invertebrate communities. Compared with forest streams, pasture streams have more grazers (notably snails and chironomids); an increased abundance of organisms 'tolerant' of high temperature, organic enrichment and/or fine sediment (notably oligochaetes and chironomids); and a decreased abundance of those organisms which are sensitive to high water temperature, low benthic boundary-layer dissolved-oxygen concentration, and/or fine sediment (e.g., filter feeders, stoneflies, and mayflies).

The conversion from forest to pasture has caused stream channels at Whatawhata to become narrower and more deeply incised (Section 2). Channel narrowing in pasture streams has been observed elsewhere in both hillcountry and lowland streams (Davies-Colley 1997). The probable reason is that high light levels allow pasture grasses to become firmly established on those parts of the streambed which are inundated only at high flows. These grassy areas then build up by trapping alluvial sediment (i.e., streambed sediment mobilised during floods) and colluvial sediment (i.e., sediment eroded from the catchment). In hillcountry catchments, the removal of forest increases the rates at which soil creep, and at which slumping and landslides transport soil towards the valley floor, thereby contributing to channel narrowing.

One effect of high stream banks and narrow channels is to increase topographic shading, but in the pasture streams studied, this was insufficient to compensate fully for the loss of dense canopy shade. Pasture stream narrowing also results in greater mean water depth which may degrade habitat by, for example, inundating sites for emergence of adult aquatic insects.

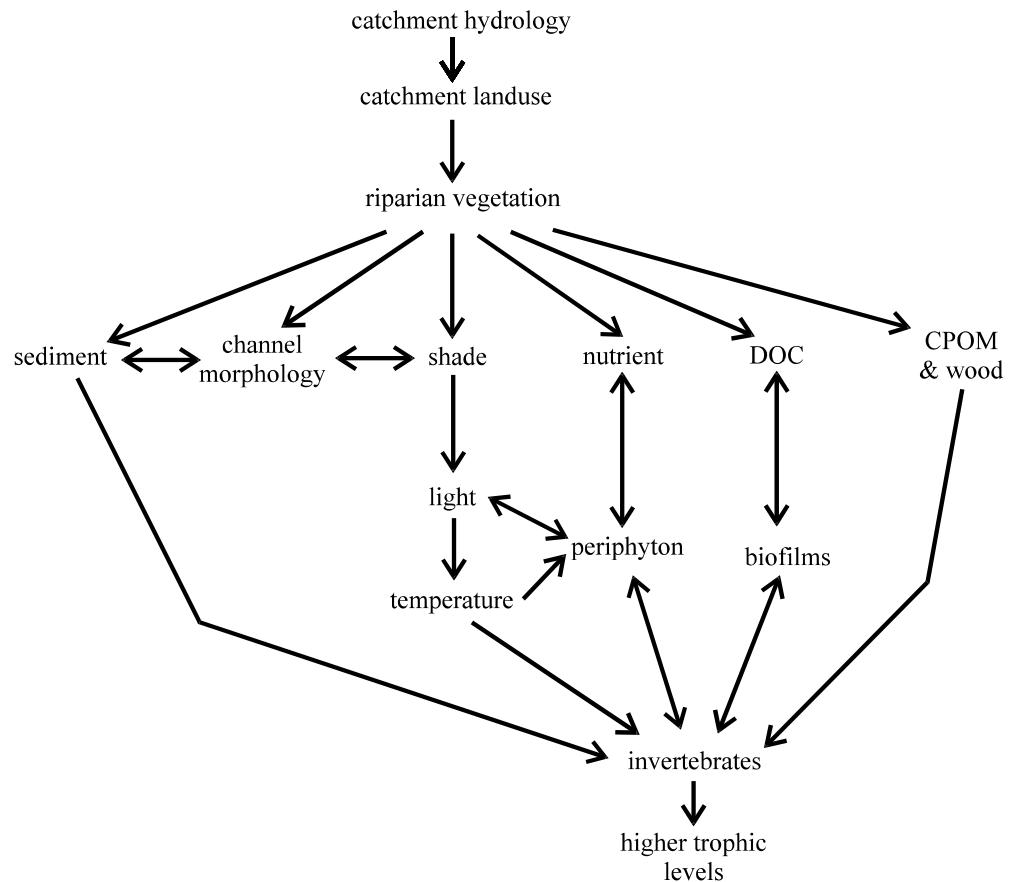
We conclude that the conversion from native forest to pasture has caused significant changes in the streams studied. Many of these changes result directly from the loss of riparian vegetation and shade while others arise from the change in catchment landuse. The former are more amenable to restoration using narrow buffer strips.

Table 15 Effects of replacing native vegetation by pasture on characteristics of hill-country streams at Whatawhata, near Hamilton.

Variable	Change	Mechanism	Effect
shade	reduced	reduced riparian vegetation	increased temperature, increased primary production, groundcover helps stabilise banks
runoff	increased	decreased interception, increased runoff rate	increased peak flows, increased catchment erosion & delivery
peak flow width	Increased	increased runoff	increased channel erosion
	decreased	increased pasture encroachment, mass movement of soils	reduced streambed habitat
bank height	increased		increased water depth, increased bankside shading ^a
stock damage	increased	increased soil compaction, decreased groundcover, increased bank damage, voiding directly into stream	increased catchment erosion & delivery of sediment
temperature	increased	increased solar radiation	increased nutrient supply loss of sensitive species, reduced dissolved oxygen saturation, increased respiration rates, increased plant growth rates
dissolved nitrogen	increased	increased catchment supply, reduced riparian removal, stock voiding into streams	increased potential primary production
fine sediment	increased	increased catchment erosion & delivery, increased channel erosion, increased bank damage by stock	reduced boundary layer flows loss of fish & invertebrate habitat, reduced clarity
clarity	reduced	increased fine sediment	degraded aesthetics, reduced visibility for animals
woody debris	reduced	reduced supply	reduced habitat diversity, reduced CPOM retention
coarse particulate organic matter	reduced	reduced litterfall, reduced retention	reduced food supply for shredders
primary production	increased	increased solar radiation, temperature & nutrient supply	increased food quantity for invertebrates, increased periphyton biomass ^c
periphyton biomass	increased ^b	increased primary production ^c	possible blooms, reduced boundary layer flows, reduced habitat diversity
invertebrates	altered	increased temperature & periphyton biomass, reduced habitat <i>quality</i> (increased fine sediment & reduced boundary layer flows, woody debris & CPOM retention), reduced habitat <i>quantity</i> (increased depth) of habitat	loss of sensitive species, increased snails, chironomids & oligochaetes, increased abundance & diversity

^a but insufficient to compensate for the loss of shade vegetation^b biomass occasionally high but variable in space and time^c sometimes counteracted by increased grazing and sloughing

Figure 48 Diagram showing the major linkages between riparian vegetation and stream ecosystems. DOC = dissolved organic carbon, CPOM = coarse particulate organic matter.



5.3 Effects of shade restoration

Figure 48 shows the main linkages between riparian vegetation and the stream ecosystem, and Table 16 summarises the changes which we believe are likely to occur when stream shade is increased by restoring riparian vegetation. The changes are inferred not only from the differences which we measured between streams but also from our understanding of the important processes operating, deduced from process studies. The latter allow us to estimate the likely timescales over which the changes will occur, although these timescale estimates are tentative and will be investigated further during planned studies to monitor the effects of experimental restoration at Whatawhata.

We have assessed the changes as being either positive or negative in terms of restoring streams towards conditions prevailing in native forest. We do not attempt to assess whether changes are 'beneficial' or 'desirable' from a particular management perspective. Rather the effect of a change is deemed to be positive if it makes the stream more like streams found in native forest. For example, increased primary

production in pasture streams could be argued to be beneficial because it increases the food supply for invertebrates and fish. By our definition, however, increased primary production is deemed to be a negative effect because native forest streams have low primary production.

The majority of the effects of restoring riparian vegetation are positive in terms of restoration. The major exception is channel widening, with its associated temporary increase in sediment supply.

Table 16 Effects on pasture streams of excluding stock and restoring dense riparian vegetation. Effects are assessed to be positive (+) or negative (-) in terms of restoration towards native forest characteristics (as explained in the text).

Variables	Change	Mechanism	Years	Effect	Reason
channel width	increased	transition: banks destabilised;	10–20	-ve	increased sediment supply
		equilibrium: low, stable banks	>20	+ve	increased habitat area
runoff	decreased ^a	increased interception, decreased runoff	10–20	+ve	decreased catchment erosion & delivery
peak flow stock access	decreased	decreased bank damage, increased soil permeability, groundcover & wetlands, reduced voiding to stream	1–10	+ve	reduced channel erosion, decreased sediment supply, decreased runoff, increased retention, decreased nutrient input
woody debris	increased	increased supply	>10	+ve	increased habitat diversity, increased CPOM retention
temperature	decreased	decreased solar radiation	1–20	+ve	recolonisation by sensitive species
dissolved nitrogen	decreased	reduced stock access, increased riparian retention, increased denitrification ^b	1–10	+ve	decreased primary production
fine sediment	increased	transition: unstable banks	10–20	-ve	degraded habitat
	decreased	equilibrium: stable banks, decreased, runoff & increased retention	>20	+ve	improved habitat more filter feeders
cpom	increased	increased terrestrial litterfall, increased retention	10–20	+ve	increased carbon supply for shredders ^c
primary production	decreased	decreased solar radiation, temperature & nutrient input	1–20	+ve	reduced carbon supply ^d
periphyton biomass	decreased	decreased primary production	1–20	+ve	no 'nuisance' blooms, improved habitat
invertebrates	altered	decreased temperature, primary production & biomass, decreased fine sediment, increased CPOM	1–20	+ve	conservation of sensitive species

^a effect large if catchment landuse changes, small for a narrow riparian buffer strip

^b if wetlands are protected

^c which compensates for reduced primary production

^d compensated by increased terrestrial litterfall

Channel widening

Following the restoration of dense riparian shade by trees, we predict a period of transition during which the channel widens. Once the canopy closes over the streams, groundcover (e.g., grasses) is shaded out, leaving the banks susceptible to erosion by flood flows. In hill country, tree roots are also likely to reduce the mass movement of soil towards the channel (e.g., soil creep, slumping and landslides). The shading effect on stream width declines with increasing stream size and becomes unimportant once stream width exceeds a certain critical value, estimated to be of the order 10 m (Davies-Colley 1997). At the end of the transition period (i.e., after about 20 years) we predict that the sediment supply from the catchment will balance the sediment transport rate of the stream, and the average stream width will no longer increase.

Channel widening increases the sediment supply to the stream and thereby delays stream restoration. In the pine streams studied, invertebrate communities were similar to those in native forest streams, suggesting that sediment effects were minor. It is important to remember that in these streams the entire catchment was reafforested. Peak flows, which are the primary agent of channel formation, will have been greatly reduced and this may have partly offset any shade-induced bank destabilisation. Channel widening may have a significant adverse effect on the stream ecosystem if dense shade decreases the strength of the stream banks by shading out groundcover and if channel-forming flood flows remain high because the underlying landuse remains pasture. This highlights one potential danger of restoring vegetation in a narrow riparian strip. There may, however, be an intermediate level of riparian shade which enables restoration objectives to be met (e.g., for water temperature and primary production) without significantly reducing the groundcover and strength of the banks. We attempt to quantify the optimal level of shading at the end of this section. Where channel widening is likely to occur, care must be taken to ensure that trees are planted far enough back from the banks to avoid being undermined, particularly as it may take several years before tree root systems develop to the stage where they make a significant contribution to stability.

Notwithstanding the potential for channel widening and bank erosion, the ultimate source of sediment is catchment erosion followed by delivery to the stream via overland flow. The question arises whether restoration can reduce sediment inputs from the catchment. Clearly a change in catchment landuse from pasture to forest is likely to effect the largest reduction. Sediment is transported into the stream by overland flow and by flow in ephemeral flood channels. Restoration should aim to reduce these flows and/or to decrease flow velocities below the threshold for sediment transport. Overland flows can be reduced if soil permeability is increased. One obvious place to target is the riparian zone along the edge of the stream. Instead of flowing directly into the stream, overland flow is encouraged to infiltrate the soil, eventually reaching the stream below the ground surface. If this happens, most

sediment and particulates (e.g., particulate bound nutrients) will be trapped in the soil. Soil permeability in the riparian zone is expected to increase naturally if stock are excluded and organic matter is encouraged to accumulate (e.g., leaf litter, dead branches, dead grass etc). In terms of our stated restoration strategy, the accumulation of organic matter in the riparian zone and an increase in soil permeability will make streams more like those in native forest.

Dense groundcover (e.g., grasses, herbs, reeds, shrubs) is likely to increase friction and allow particulates to deposit. Three areas merit targeting. Firstly, soils underneath shade trees (especially near the stream banks) will benefit from dense groundcover to protect them from erosion by overland flow and/or stream flow. Secondly, wetland areas along the edges of streams merit protection (e.g., stock exclusion) to encourage plant growth and the accumulation of organic matter, because these areas have substantial capacity for denitrification and sediment trapping. Thirdly, ephemeral channels merit protection to encourage dense groundcover, increase friction, reduce overland flow velocities, and hence trap particulates. We are unable at this stage to quantify precisely the flow, sediment and nutrient reductions which can be achieved by these measures, but there are strong indications that a combination of these measures can be effective.

Shade measurement

Lighting in stream channels under a riparian canopy is extremely variable both in space and time. Shade tends to be high near the stream banks and measurements at mid-channel underestimate the cross-sectional average shade, typically by 10–20% (Mason 1983). Longitudinal variations in shade arise from variations of bank height, channel width, channel orientation, and riparian vegetation. Consequently, accurate estimation of reach-averaged shade in streams requires measurements at a large number of points (e.g., 20) both along and across the channel.

Our studies (Section 2) have shown that a canopy analyser is a very useful instrument for measuring shade in stream channels with only a moderate amount of fieldwork. The canopy analyser measures ‘visible shade’ (i.e., shade to visible solar radiation) and tends to overestimate ‘total shade’ (i.e., shade to visible plus near infra-red (NIR) solar radiation). Total shade is required when predicting water temperatures (see Section 4), but our measurements show that this distinction only becomes significant under very heavy shade (> 90%). In situations of lesser shade, visible shade closely approximates total shade, whilst under heavy shade an empirical relationship can be used to ‘unbias’ visible shade measurements.

The canopy analyser is likely to remain a research instrument, but we have shown that shade can be measured reliably using more commonly available PAR (photosynthetically available radiation) sensors. Qualitative assessments of stream shade can be made by photography with a fish-eye lens. We cannot recommend visual assessment of shade as a substitute for objective measurements.

Temperature

Our study reinforces the finding of several previous studies that increased shading reduces water temperature and hence aids stream restoration. Arguably the most important effect of reduced temperatures is on sensitive stream organisms such as stoneflies, mayflies, and fish. As discussed in Section 2, field surveys showed that stoneflies and mayflies are absent from streams in which maximum temperatures exceed 19°C. Laboratory studies indicate lethal temperatures for sensitive invertebrates between 22.6 and 25.0°C. Allowing a safety margin of 3°C below the measured lethal limit (Simons 1986), the upper thermal tolerance for these sensitive invertebrates appears to be about 19–20°C.

In pasture streams during spring (November), the measured daily mean and maximum temperatures were below this limit being typically 15 and 20°C. Model predictions, however, suggest daily mean and maximum temperatures can approach 20 and 25°C respectively under summer low flow conditions (Fig. 40), which may explain the absence of sensitive organisms from pasture streams.

An important question is how the upper thermal tolerance for sensitive organisms (viz., 19–20°C) should be applied in natural streams where temperature varies diurnally. Should management aim to keep maximum temperatures below 19–20°C or will it suffice to keep the daily mean below this limit? If we take the precautionary approach then we would apply the upper thermal tolerance measured using constant temperature experiments to the daily maximum temperature. Recent laboratory studies on two important New Zealand invertebrates (the mayfly *Deleatidium* and the snail *Potamopyrgus*) suggest, however, that this approach may be unduly conservative. Mortality was comparable in experiments at constant and diurnally varying ($\pm 5^\circ\text{C}$) temperatures when the constant temperature was mid way between the daily mean and the daily maximum (Cox & Rutherford 2000). It is desirable to test that other stream organisms show the same temperature response but these results suggest that stream invertebrates can tolerate short periods of exposure (viz., for a few hours in the afternoon) to temperatures somewhat higher than the constant temperature at which they can survive long term. Stoneflies and mayflies are used here as examples of organisms likely to respond positively to shade restoration. There may be other organisms (e.g., fish) whose thermal tolerances will need to be considered in a particular restoration project.

Given shade and channel measurements, a computer model such as STREAMLINE can be used to predict water temperature (Section 4). The model has fairly high data and operating requirements and its use can only be justified for detailed studies (e.g., to help 'optimise' shade regimes in a particular catchment in order to achieve desired temperature changes at minimum cost). Nomographs have been developed using the computer model which can be used to make a more rapid estimate of possible temperature changes (Figs 41 and 42). These nomographs only require an estimate of stream size (stream order or mean depth), together with an estimate of the percentage shade, and are ideally suited to making an initial assessment of the likely cost-effectiveness of stream restoration. The major practical difficulty is to estimate the 'percentage shade'.

The nomographs indicate that for third- to fifth-order streams (mean depth 0.3–0.5 m) in most of the North Island, average shade levels of about 50% are sufficient to prevent daily maximum equilibrium temperatures exceeding 20°C. In first-order streams (mean depth 0.1 m) higher shade levels are required (50–75%) because shallow streams heat up rapidly during the day. Predicted temperatures are sensitive to variations in meteorology and we would expect daily maximum temperatures to vary with latitude (i.e., to be lower in South Island streams and, possibly, higher in Northland streams). Predicted temperature *changes* are more robust and are unlikely to vary significantly from those presented.

Field and modelling investigations indicate that small streams (first- and second-order) have a lower thermal inertia (i.e., heat up and cool down faster) than large streams (> third-order). When shade is significantly increased or reduced (> 50% change), temperature changes in first- and second-order streams are ecologically significant (typically of the order 5°C) and occur over short distances (typically 1–5 km). By comparison, temperature changes in larger streams are smaller and occur over much longer distances (10–20 km). Using the computer model, we estimated the marginal effect on water temperature of restoring shade at any point in a stream network (°C reduction per metre of streambank shaded). These calculations show that in order to achieve a given lowering of water temperature, it is more effective to shade small headwater streams than larger lowland streams because the former have a low thermal inertia.

Dense riparian vegetation not only reduces exposure to solar radiation, but also has the potential to 'improve' the microclimate. The main effect of riparian vegetation clearly is to reduce solar radiation input, but model simulations suggest that additional water temperature reductions may be achieved if the riparian buffer strip is wide enough, and the vegetation dense enough, to reduce air temperature. We are currently conducting experimental work which will help decide how wide a riparian buffer strip needs to be in order to protect the stream's microclimate, but published work on the edge of native forest suggests widths of the order 50 m (Young & Mitchell 1994).

There may be additional benefits to adult insects and native birds from microclimate changes (notably reduced windspeed, increased humidity, and less changeable conditions) associated with dense riparian vegetation.

Periphyton

Increased shading reduces the input of photosynthetically available radiation (PAR) and hence reduces primary production in the stream channel. It can be argued that decreased primary production decreases food resources for invertebrates and fish, and hence is not a beneficial effect. In terms of our stated restoration goals, however, increased shade, in reducing primary production, makes the stream more like those in native forest, and hence is deemed a positive effect.

Increased primary production need not necessarily result in higher periphyton biomass since biomass also depends on factors other than light, such as grazing and scour. While there is a large body of information on the relationship between photosynthesis rate and light level (e.g., Section 3, Fig. 22), it is not straightforward to predict the effect of shade restoration on grazing, sloughing, scour and abrasion loss. Our landuse comparisons (notably Figs 11 and 12) and channel experiments (Fig. 21) provide an indication of the periphyton biomass differences between shaded and unshaded channels. These studies may not enable definitive, quantitative statements to be made about the effects of shade on primary production and biomass, but the general trends are clear and are summarised in Table 17.

Our channel studies (Section 3) showed that grazing pressure exerted strong 'top-down' control on periphyton biomass especially in channels with low shade, high grazer numbers, and high primary production. We also found that in channels with low shade, periphyton biomass was very patchy spatially and fluctuated greatly over time. Apparently, in unshaded channels top-down control by grazers is effective for much of the time, but occasionally rates of carbon fixation exceed the ability of grazers to consume the periphyton produced. Existing knowledge of factors controlling invertebrate grazing rates is poor. We noted that periphyton blooms did not occur until daily mean stream temperatures exceeded about 16–17°C and daily maximum temperatures exceeded about 20°C. This approaches our recommended thermal tolerance for sensitive invertebrates (19–20°C) which suggests that, in the streams studied, high temperature periodically releases periphyton from top-down control by grazers.

In our landuse comparison (Section 2) and channel studies (Section 3), an increase in shade was accompanied not only by a decrease in primary production, but also by reduced grazer numbers (notably snails and chironomids). The reduction in grazing pressure, however, was insufficient to compensate for the reduction in primary production. Consequently, periphyton biomass was always low in the heavily shaded

channels. We interpret the lack of significant biomass accrual in the heavily shaded channels to the ability of grazing invertebrates to always effectively crop the carbon fixed at these low light levels. Not only was periphyton biomass always low in the shaded channels, but it was also spatially uniform (in marked contrast to the patchy distribution in open channels). This suggests that when food resources are scarce, grazers forage widely and crop periphyton uniformly.

We found that algal community composition was very similar between the 60%, 90% and 98% shade treatments, being dominated by diatoms. Filamentous green algae (FGA) were only present in the open channels (0% shade) during late summer. We have observed prolific FGA growths in pasture streams during late summer. Periphyton can attain 'nuisance' biomass levels in the pasture streams studied, but such proliferations are sporadic, and even moderate levels of shade appear sufficient to prevent FGA from attaining 'nuisance' biomass levels.

Overall we conclude that the restoration of shade will result in a periphyton community which is spatially uniform and which typically comprises a thin layer of diatoms. Filamentous green algae (FGA) require high light levels and are likely to disappear from shaded streams. Low light levels and grazing pressure are likely to prevent periphyton blooms from occurring in shaded streams.

Table 17 Summary of periphyton characteristics in shaded and unshaded streams.

	Shaded (>90%)	Unshaded (<60%)
periphyton biomass	always low	fluctuates over time sometimes high
biomass distribution	spatially uniform	patchy
species diversity	low	higher
	predominantly diatoms	mostly diatoms, some FGA
algal pigments		no significant difference
I_k	low	high
(see note 1)	(see note 2)	
P_{max}		no significant difference
(see note 3)		(see note 4)
gross photosynthesis	low	high
g(carbon) m ⁻² d ⁻¹	(see note 5)	
nitrogen uptake rate	low	high
mg(nitrogen) m ⁻² d ⁻¹	(see note 5)	

1 I_k = half-saturation coefficient for light.

2 periphyton in shaded streams are adapted to low light levels.

3 P_{max} = maximum photosynthesis rate per unit biomass

4 as periphyton mats become thicker, diffusion limits their specific carbon fixation rate and they become self-shading

5 nitrogen uptake and photosynthesis rate are correlated.

FGA = filamentous green algae

Nutrients

The clear signal from our landuse comparison is that phosphorus rarely limits plant growth in the streams studied. Volcanic soils of the central North Island and Waikato Basin have a high natural phosphorus content (Timperley, in Viner 1987) and in addition many pasture catchments are topdressed with superphosphate. Phosphorus is likely to be present in excess in similar catchments elsewhere. In some other parts of the country, periphyton appears to be limited by low phosphorus concentrations at certain times of the year (e.g., Freeman 1986).

In our studies, nitrate concentrations were usually markedly higher in pasture than native streams. This indicates that nitrogen inputs are higher from pasture catchments than native forest. Nitrate concentrations in the pine catchments were intermediate between pasture and native streams, suggesting that nitrate losses decline with time following reforestation of pasture land.

Measured nutrient concentrations reflect the balance between inputs and instream processing and consequently care must be exercised when interpreting concentration measurements. Occasionally during summer low flows, nitrate concentration in pasture streams dropped to levels likely to reduce plant growth rate. During summer, lateral inputs of nitrogen are low because pasture soils are dry and groundwater enters the stream through wetland areas, where denitrification occurs. In addition, nitrogen uptake rate in the stream is high because of high periphyton biomass, high

photosynthesis rates, and shallow water. Instream uptake by periphyton was almost certainly responsible for the marked reduction of nitrate concentration observed during summer low flows in the pasture stream PW2 (see Section 2, Fig. 14). By comparison, high nitrate concentrations in forest streams throughout the year reflect the fact that periphyton growth rate, and hence nutrient demand, is low because of light limitation.

A corollary of high nitrate uptake rate by periphyton in pasture streams is that the restoration of riparian shade is likely to result in reduced uptake by periphyton and hence higher nitrate concentration. Given the same nitrate input from a predominantly pasture catchment, a shaded stream will act primarily as a downstream ‘transporter’ of nitrate. By comparison an unshaded stream acts primarily as a ‘processor’, storing nitrogen as biomass under stable flow and flushing it out during storm events. One ramification is that if the headwater streams are shaded, the region of maximum periphyton production will move downstream into the unshaded reaches. If the downstream unshaded reaches are unsuitable for aquatic plants (e.g., because of unstable sediments) the stream will export nitrate. Therefore the restoration of riparian shade could aggravate eutrophication in a downstream river reach, lake, or estuary. There are two relevant factors. Firstly, instream uptake by plants may represent only a temporary loss from the stream since particulate organic nitrogen from the periphyton communities can be recycled, often after being transported downstream: the process of ‘nutrient spiralling’. Secondly, riparian management may reduce nitrogen inputs to the headwater streams.

It is not clear whether replanting a narrow riparian strip will substantially reduce the input of nitrogen from an otherwise pastoral catchment. Restricting stock access, encouraging groundcover, and increasing soil permeability in the riparian zone are likely to have beneficial effects, especially on particulate nitrogen. On the other hand, if grasses are eliminated by heavy shade, stock are allowed to use the riparian zone frequently, and surface runoff is unaffected, then nitrate inputs may increase following riparian revegetation (Smith 1992). Riparian wetlands strongly influence nitrate inputs because they are sites for denitrification, and if these wetlands can be protected, restoration is likely to reduce nitrogen inputs. It is not yet possible to make definitive quantitative statements about the effects of shade restoration on stream nitrogen inputs and instream nitrogen dynamics.

Particulate carbon

Differences in vegetation (particularly riparian vegetation) between landuses have the potential to influence ecosystem function by altering the supply and type of coarse particulate organic matter (CPOM) entering, and retained within, the stream ecosystem. CPOM provides food directly for shredders and indirectly (after shredding, mechanical grinding, and microbial conditioning) for collector-browsers

(Cummins 1992). In our landuse comparison (Section 2), CPOM levels were only slightly higher in pine and native forest streams than in pasture streams, although a flood just prior to our surveys may have affected benthic CPOM levels. A similar study in small North Canterbury streams (Harding & Winterbourn 1995) found higher CPOM levels in forest than pasture streams. Flow disturbance has been identified as an important factor influencing both CPOM retention and invertebrate shredder abundance in New Zealand streams (Rounick & Winterbourn 1983b, Quinn et al. 1994b). We can expect riparian shade to result in an increase in woody debris, increased benthic CPOM inputs and retention, and increased shredder abundance.

Invertebrates

There is clear evidence that the effect of shade on periphyton has flow-on effects for benthic invertebrates. The main differences between shaded and unshaded streams observed during our studies are summarised in Table 18. Generally, pasture streams have fewer filter feeders, mayflies, and stoneflies (often regarded as ‘desirable’), more snails, chironomids, and oligochaetes (often regarded as ‘tolerant of sediment or organic enrichment’), slightly greater total biomass, and slightly greater taxonomic richness than forest streams. Differences between pine and native forest stream communities are minor. We have not attempted to identify the ramifications of these differences for higher predators (e.g., koura, eels, or native fish).

In both the landuse comparisons (Section 2) and the channel studies (Section 3), grazer densities (notably for chironomids and snails) decreased with increasing shade. This is almost certainly the result of reduced periphyton production (McIntire 1973). Some grazers can switch to alternative food sources such as heterotrophic biofilms when periphyton become scarce (Rounick & Winterbourn 1983a), but our findings indicate that grazers are either unable to utilise, or cannot find enough, alternative food to maintain high populations in shaded streams.

The abundance of many collector-browsers was unaffected by shade. This implies that they are able to find sufficient food (e.g., diatoms, heterotrophic biofilms, or detritus) in both shaded and unshaded streams. It also implies that they are unable to increase their density in open streams despite the increased primary production, possibly because of high water temperatures and/or fine sediment loads. For example, the density of the important collector-browsing mayfly *Deleatidium* was unaffected by shade treatment in the experimental channels where temperature was constant at pasture stream values, but was absent from pasture streams which were significantly warmer than forest streams. *Deleatidium* has a low thermal tolerance (Quinn et al. 1994b), which may explain its absence from pasture streams. However, we cannot rule out the possibility that *Deleatidium* is adversely affected by fine sediment, rather than by high temperatures, in pasture streams.

Oligochaetes were more abundant in pasture than forest streams. It is not clear whether these detritus feeders are stimulated by autochthonous carbon production (originating from the periphyton communities) or by allochthonous carbon (which collects passively in the periphyton mats). Oligochaetes, snails, and chironomids are known to be tolerant of organic enrichment and of low dissolved oxygen concentrations which may occur near biofilms (Stark 1985).

We found that shredding invertebrates were relatively scarce in our study streams. This may be an artefact from streambed CPOM levels being low because of a flood just prior to our surveys. Shredders depend on the supply of CPOM from the catchment and might be expected to benefit from increased litterfall. Filter feeders were more abundant in forest streams, possibly because attachment sites were more abundant, or because the water contained a lower proportion of mineral silt and a higher proportion of FPOM (their principal carbon source).

The increase in snail, chironomid, and oligochaete numbers results in pasture streams having a slightly higher biomass, and significantly higher total numbers, of invertebrates than forest streams. This might be regarded as beneficial because it contributes to a greater food supply for predators (e.g., koura, eels, and native fish), but we regard the change as negative in terms of restoration towards conditions found in native streams.

We can infer that the restoration of riparian shade will reduce periphyton production and biomass which, in turn, will reduce snail, chironomid, and oligochaete abundance. In shaded streams, heterotrophic biofilms are an important food source for the remaining grazers. Restoration of shade will reduce water temperatures, which may increase the biomass of sensitive invertebrates (e.g., mayflies and stoneflies), although the reduction of sediment inputs and concentrations may also be required before these organisms will recolonise the streams. Reduction of sediment input would seem to be a prerequisite for recolonisation by sensitive filter feeders. As discussed above, the restoration of dense shade may result in channel widening, which is likely to delay recolonisation by those invertebrates which are sensitive to sediment inputs. Restoration is likely to increase leaf litter and woody debris inputs, thereby increasing CPOM input and retention, increasing the diversity of invertebrate habitat, and increasing the abundance of shredders.

Table 18 Summary of the effects of increasing shade on invertebrate numbers inferred from measured differences between shaded and unshaded stream and experimental channels.

Variable	Increasing shade
total biomass	slight decrease
taxa richness	slight decrease
total numbers	slight decrease
chironomids	Decrease
snails	Decrease
oligochaetes	decrease ¹
filter feeders	slight increase
mayflies & stone flies	increase ²
collector-browsers	no change
shredders	no change

¹ significant decrease in landuse comparison but not in channel experiments

² probably a temperature effect.

Heterotrophic biofilms

An important gap in our understanding of stream restoration is the likely response of heterotrophic biofilms ('biofilms' for short). Some grazers and collector-browsers utilise biofilms, either in addition to, or in preference to, periphyton (Rounick & Winterbourn 1983a). We can infer from biomass measurements, measured respiration rates, and P/R ratios that biofilm metabolism is significant in forested streams (Section 2). The relatively small differences in invertebrate biomass between forested and open streams suggests that biofilms are an important food source for invertebrates.

Heterotrophic biofilms are not affected directly by shade, although in pasture streams they must compete for space with periphyton and may be subject to increased grazing pressure if increased periphyton production leads to increased grazer numbers. Biofilms depend on the supply of dissolved organic carbon to the stream which may increase following the restoration of riparian vegetation. Thus we expect restoration to stimulate biofilm production and biomass. If so, then those invertebrate grazers and collector-browsers able to utilise biofilms will prosper.

Spectral composition of light

One unresolved question is the ecological significance of the green enrichment of visible light when it passes through leaves. In our studies the changes in spectral quality of shadelight were small (i.e., there was only a slight green shift in visible shadelight measured under trees). We are of the opinion that changes in the spectral quality of shadelight are less important ecologically than changes in light quantity.

6. PRACTICAL CONSIDERATIONS IN RESTORING STREAM SHADING

Natural regeneration or replanting

When planning to restore riparian vegetation, one question is whether to allow natural regeneration or to replant. A consideration is the width of the stream. Narrow headwater streams (less than 2–3 metres wide) will often become heavily shaded by the unassisted regrowth of grasses, herbs, ferns and shrubs. Wider streams (>5 metres) require tall trees in the riparian zone to significantly reduce radiation inputs, which may regenerate naturally but may require enhancement by judicious replanting. The time-scales for shade restoration differ significantly: natural regrowth of grasses and herbs may shade a narrow channel within one year; exotic trees (e.g., pines, eucalyptus) may require 10 or more years to shade a channel; and slow-growing native trees may take 20–50 years.

For many parameters we studied at Whatawhata there were large differences between pasture and forest streams but similarities between native forest and pine forest streams. This was most noticeable for shade, water temperature, and benthic invertebrate communities. This suggests that reforestation in pines may go a long way towards restoring pasture streams although there may be subtle, but important, differences between pine and native forest streams which our studies did not detect.

How much shade?

It is apparent that in streams like those studied at Whatawhata, a compromise must be made between providing enough shade to reduce water temperature and primary production without destabilising the stream banks. We attempt here to estimate this 'optimal' level of shade.

If we assume a target daily maximum water temperature of 20°C for the protection of sensitive invertebrates, computer modelling (Section 4) indicates required shade levels of about 50% in third- to fifth-order streams (mean depth 0.3–0.5 m) and of 50–75% in first-order streams (mean depth 0.1 m). In South Island streams these shade levels are likely to result in water temperatures significantly lower than 20°C, suggesting a higher level of protection for sensitive invertebrates. Conversely, in Northland they may not be sufficient to keep temperature below 20°C, suggesting lower protection. To some extent organisms are able to acclimatise to local temperatures and tend to respond to temperature *change* rather than to absolute temperature (Simons 1986). We believe that model predictions of temperature change are robust to variations in meteorology and that the above targets for shade can be used widely throughout the country for preliminary assessment. Given more detailed information about

temperature tolerances and local meteorological conditions, it is possible to refine these preliminary estimates.

Our mesocosm experiments showed consistently low periphyton biomass at 90% and 98% shade. This suggests that in order to reduce periphyton biomass to the low levels characteristic of native forest streams it is necessary to restore shade levels to about 90%. At 0% and 60% shade the potential existed for periphyton biomass to reach 'bloom' levels, albeit spasmodically, which indicates that the minimum shade level required to control blooms lies somewhere in the range 60–90%.

These mesocosm studies also indicated that over 60% shade was required to produce marked reductions in chironomid abundance and increases in *Austrosimulium*, whereas more than 90% shade was needed to produce significant reductions in taxonomic richness and *Pycnocentroides* abundance, and increases in *Aoteapsyche* abundance.

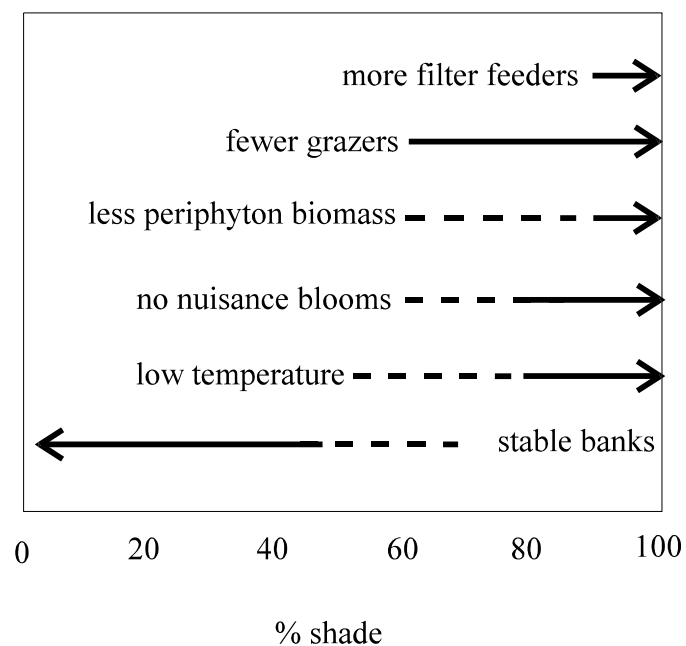
Observations of riparian vegetation at Purukohukohu indicate that a pine density of 180 stems ha⁻¹ provides sufficient light for dense riparian vegetation. From Fig. 4 this stem density corresponds to a DIFN value of 30% (viz., 70% shade) measured at bank height. The bankside vegetation at Purukohukohu comprised shade tolerant ferns and shrubs, and we might expect pasture grasses to decline in vigour at lower shade levels. We are not able to make firm recommendations about the maximum shade levels required to maintain bank stability. We suspect a value somewhere in the range 50–70%.

These data are summarised in Fig. 49. It is clear that there is no one shade level which meets all the restoration targets. The main conflict is that about 90% shade is required to restore periphyton biomass to the low levels typical of forest streams and to enable sensitive invertebrates (e.g., filter feeders) to recolonise, whereas shade of 50–70% may adversely affect bank stability. If the nature of the stream being restored is such that it is important to ensure bank stability, then our preliminary recommendation is for a maximum shade level of 50–70% at bank height. Deciduous trees could provide summer shade levels in this range but still allow sufficient groundcover to develop during autumn-winter to stabilise the banks. It is unclear whether native trees can be used for this purpose.

In wide shallow streams, even 70% shade from trees is likely to mean that periphyton and invertebrate communities are intermediate between those characteristic of pasture and forest streams. In narrow streams, however, there may be additional stream shade (i.e., shade at the water surface) arising from the banks and bankside groundcover.

If channel widening is unlikely (e.g., because of the geology of the catchment) or can be tolerated during the transition phase, then higher shade and denser tree plantings levels can be contemplated. It appears that most of the restoration targets can be attained with shade levels of about 90% and that it is not essential to restore shade levels to the 98% values typical of native forest streams.

Figure 49 Comparison of the tentative limits for the amount of shade required to meet various restoration targets. Solid lines indicate that targets are met, dashed lines that targets are only partly met or that the limits are uncertain.



Selection of trees

Having decided on the level of shade required, the next step is to decide what density of trees is required to achieve this shade level. The information currently available is rather sparse. Figure 4 does, however, show light levels measured under plantation forest stands of pines (ranging in age from 15 to 23 years) eucalypts, and blackwoods. To achieve a shade level of 70% (a DIFN of 30%), densities would be required of about 300 stems ha^{-1} (eucalypts) and 50–200 stems ha^{-1} (pines). This is equivalent to a tree spacing of between 6 m (300 stems ha^{-1}) and 14 m (50 stems ha^{-1}). According to Fig. 4 it is not possible to achieve shade greater than about 80% under 17 year old plantation eucalypts. To achieve 90% shade, pine trees would need to be spaced at intervals between 3.5 and 10 m depending on their vigour.

These figures provide some guidance about the density of tall trees required to achieve a certain level of shading, but small streams are often shaded by their banks and by groundcover growing along the edge of the stream. Bankside shade can be substantial, as indicated by the levels measured in pasture streams at Whatawhata (53–96% shade), and is additional to any shade by trees in the riparian zone. In small streams where the canopy closes over the channel, the effects on lighting are multiplicative. Thus if trees and streambanks both pass 50% of the incident light then in combination they pass 25% light (i.e., provide 75% shade). If the stream is wide, the effects of tree shade may not be fully felt in mid-channel. In theory, the amount of shading of a stream channel can be calculated knowing the geometry of the stream banks, riparian vegetation, and surrounding hills, together with the attenuation coefficient of the vegetation (Mason 1983). At present we have insufficient information about canopy attenuation coefficients for different tree species, planting regimes, and soil types to allow these calculations to be made accurately.

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8. APPENDICES

Appendix 1: Shade assessment using the Lai-2000 canopy analyser

The LAI-2000 instrument was developed primarily (as its name suggests) for measuring indices of canopy architecture under plant or forest stands, particularly the *leaf area index* (LAI, the area of leaf surface per unit level ground area) and the *mean tilt angle* of the leaves. The instrument calculates these quantities from the measurement of canopy gaps in five ranges of zenith angle. Welles & Norman (1991) should be consulted for a description of the LAI-2000 instrument in relation to the theoretical basis of these quantities.

Figure A1 (from Welles & Norman 1991) gives a schematic of the LAI-2000 sensor. Detector ring 1 ‘sees’ zenith angles 0–13°, ring 2 angles 16–28°, ring 3 angles 32–43°, ring 4 angles 47–58°, and ring 5 angles 61–74°. The signals from these five detectors, referenced to the signals recorded at an open site (ideally under identical sky lighting), provide estimates of the *gaps* in the canopy (symbol $T(\theta)$ which is a function of zenith angle, θ). $T(\theta)$ is a quantity analogous to transmittance. The values of $T(\theta)$ in turn provide the basic data for calculations of ‘leaf area index’ (LAI) and other indices of canopy structure. Small view caps can be placed over the fish-eye lens of the sensor to screen out certain ranges of azimuthal angle for particular purposes (in particular the sensor is routinely prevented from ‘seeing’ in the direction of the operator and in the direction of the sun, if clear). The sensor head must be levelled for each point reading, using a small bubble indicator fixed to the head.

Figure A2 gives typical transmittance and reflectance curves for foliage. Chlorophyll *a* and other pigments absorb blue and red light more strongly than green light, leaving green peaks of transmission and reflectance centred about 550 nm. Absorption of light by foliage is very weak in the near infra-red (>700 nm) where reflectance and transmission are both high.

A blue filter (restricting light transmission to the 420–490 nm range) is used in the LAI-2000 instrument so that (green) foliage is ‘seen’ as black (indicated by the ring in Fig. A2). This means that it slightly underestimates the total amount of light in the PAR range (400–700 nm) which is transmitted through the canopy. (The peak at around 550 nm in the transmission spectrum shown in Fig. A2 is the cause of the greenness of shade light under plant canopies.) As far as possible, readings with the LAI-2000 should be taken with low sun angles near dawn or dusk, or under stable overcast conditions: it does not function well in direct sunlight owing to light scattering by foliage (glint off leaves).

Figure A1 Left: Schematic cross-sectional view of the LAI-2000 canopy analyser optical sensor showing lenses, blue filter, mirror and ring-shaped detectors. Right: Angular range of rays from the upper hemisphere received by each of the five ring-shaped detectors.

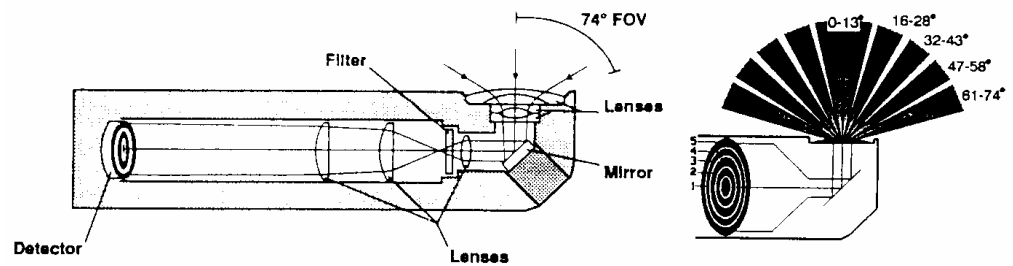
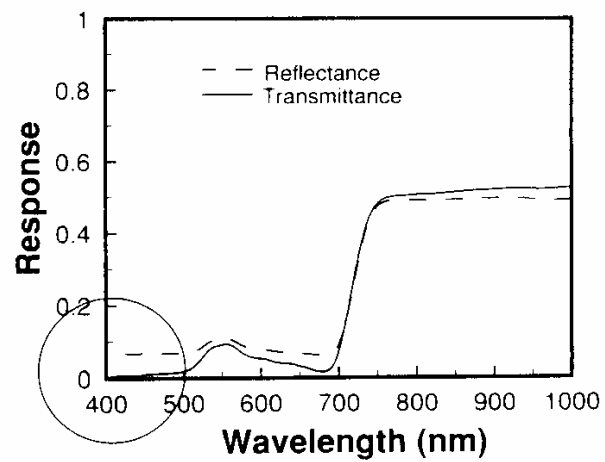


Figure A2 Typical reflectance and transmittance curves for foliage.



Stream shade measurement

The leaf area index is very useful as an index of canopy architecture, particularly as a non-destructive biomass estimate, in crop science and forestry. However, it is not a very appropriate index for quantifying stream shade because measured values include topographic shading. What we require is an index of the ‘amount’ of sky visible as a proportion of the upper hemisphere (i.e., an average of the canopy and topography ‘gaps’). A quantity termed the diffuse non-interceptance (DIFN, symbol τ) is a convenient index of reach-averaged stream shade, as appears to have first been recognised by DeNicola *et al.* (1992). DIFN is a measure of the light (radiant flux) received on a horizontal plane (such as a stream water surface) as a proportion of that from a perfectly overcast sky. The formal definition of DIFN can be stated:

$$\tau = \frac{\int_0^{\pi/2} T(\theta) \sin \theta \cos \theta d\theta}{\int_0^{\pi/2} \sin \theta \cos \theta d\theta} \quad \text{A1}$$

where $T(\theta)$ = the gap fraction (ratio of light measured at the shaded and open sites) which is a function of zenith angle, θ ; τ = diffuse non-interceptance (DIFN) = the average gap fraction (over the full range of zenith angles) weighted by the ‘cosine correction’ factor, $\sin \theta \cos \theta$. The DIFN is calculated by numerical integration using data from the five channels as follows:

$$\tau = \sum_{i=1}^5 T_i W_i \quad \text{A2}$$

where T_i = measured gap fractions and W_i = normalised values of $\sin \theta \cos \theta \Delta \theta$ for the five channels of the LAI-2000 sensor (0.066, 0.189, 0.247, 0.249 and 0.249 for rings 1 to 5 respectively, Li-Cor 1992). Rather than using the ‘hard-wired’ LAI-2000 calculation of DIFN (which uses the *geometric* mean of the individual gaps) we developed our own spreadsheet programme (in Excel) which accepts raw data from the LAI-2000 and calculates the required *arithmetic* mean gaps (and standard deviation of gaps) for use in Eq. A2. The spreadsheet also calculates the distribution of gaps in the five zenith angle fields of view, which may be useful when modelling stream thermal response.

Field protocol

In order to calculate gap fractions accurately, the shaded site (B) readings must be made as close as possible in time and space to the reference (A) readings (i.e., ambient lighting should be identical). Early field work was done with a single LAI-2000 sensor and logger which had to be moved between shaded and reference sites. The measurements were frequently confounded by lighting changes during the time taken to travel (often over steep or rough ground) between reference and stream sites.

Purchase of a second sensor and logger permitted us to obtain much better-quality data. One unit (unit A) was set up (using laboratory retort clamps fitted to steel pins driven into the ground) at an open site (usually a hill top) and configured to log ambient lighting from the upper hemisphere in five channels at intervals as short as 15 seconds (Plate 1). The second unit (unit B) was used to take readings along the stream channel (Plate 2) after suitable cross-calibration. At the end of each experiment the two loggers were interconnected to allow the B unit to calculate gaps using the closest (in time) reference readings stored by the A unit.

To measure shade levels at the stream water surface, a small plastic bottle was attached under the sensor head (see Plate 2) which reduced the risk of submersion (the sensor is weather-proof but not water-proof). The sensor could be levelled fairly easily with a small degree of force applied to push the bottle down against buoyancy. Light levels were also measured at bank height to give an indication of the light environment for the growth of riparian herbs and shrubs.

Larger gaps are 'seen' by the canopy analyser in mid-channel than near the stream bank (i.e., topographic shade is higher near the banks). To account for this spatial heterogeneity, we adapted the protocol suggested by Li-Cor (1992) for use in row crops and averaged measurements made at 10%, 30%, 50%, 70% and 90% of the stream width so as to weight all parts of the stream surface approximately equally.

Shade may vary with azimuthal direction, depending on stream channel orientation and anisotropy in distribution of riparian vegetation and topography. For example, near noon an east-west oriented stream would tend to have more shade from the direct solar beam than a stream flowing north-south. We were most concerned with light from north of the east-west line: the direction of the sun when near its zenith. The majority of measurements were made under uniform heavy overcast. A 90° viewcap was used routinely to restrict the azimuthal field of view to 45° either side of north.

Table A1: Protocol for use of a pair of LAI-2000 canopy analysers for estimating stream shade

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1. The two sensors are cross-calibrated, so that the corresponding rings of both sensors give the same readings, when viewing the same portion of the sky.
 2. A view cap, with the same field of view is placed on both sensor "heads" with the uncovered segment positioned away from the sensor handle, so the operator will be excluded from the sensor's field of view. (A 90° view cap was used to collect the data reported herein.)
 3. The reference sensor and control unit are installed at a site as near as possible to the stream reach to be surveyed with a clear view of the sky (usually a hilltop). The reference console is set up to log at frequent intervals (usually every 15 seconds) during the time taken to do the B readings in the stream channel.
 4. While the reference sensor is logging, readings are taken along the stream channel with the other sensor (B). A plastic bottle is attached to the B sensor, to permit readings to be taken at water level without submersion of the sensor. The readings are always done in the same pattern, at 10%, 30%, 50%, 70%, and 90% distance across the stream width while maintaining orientation to the north (in the southern hemisphere). Usually 20 readings are taken to characterise the spatial variation of shade at the bank height (above low-growing herbs) as well as at water level.
 5. After the data have been measured along the stream channel the reference sensor and control unit are collected. The two control units are connected to permit the readings taken at stream level and bank level to be matched to the nearest reference reading in time for the calculation of gap fractions.
-

Care was taken to maintain the same sensor orientation for all measurements, including the reference sensor, since the sky lighting may vary with azimuthal direction. A magnetic compass was used to indicate direction for each reading, taking into account the declination (see Plate 1). Viewing to the north is not advisable if the sun is visible, because reflection off glossy leaves causes problems and direct solar beams bias the DIFN value strongly in the direction of the sun. If the overcast was not sufficiently dense and stable to eliminate direct sunlight, measurements were made by pointing the LAI-2000 away from the sun even though this may introduce bias due to anisotropy in the distribution of shade elements. Under clear skies the LAI-2000 was only deployed early or late in the day when the sun was very low and there was little reflection off glossy foliage. The standard field protocol is given in Table A1.

Appendix 2: Model Equations for the STREAMLINE Model

The heat balance for a parcel of water is:

$$\frac{dT}{dt} = \frac{\Phi_{net}}{\rho C_p H} \quad B1$$

where t = time (s); T = water temperature ($^{\circ}\text{C}$); Φ = net surface heat flux (W m^{-2}); ρ = density (t m^{-3}); C_p = heat capacity ($\text{J t}^{-1} \text{K}^{-1}$) and H = mean water depth (m). The net surface heat flux comprises several components

$$\Phi_{net} = \Phi_{sol} + \Phi_{atm} + \Phi_{can} + \Phi_{top} - \Phi_{evap} - \Phi_{sens} - \Phi_{bed} \quad B2$$

where the subscripts refer to the net heat gains from: short-wave solar radiation (*sol*), long-wave atmospheric radiation (*atm*), long-wave canopy radiation (*can*), and long-wave topography radiation (*top*); and the net heat losses from evaporation (*evap*), sensible heat loss (conduction and convection) to the air (*sens*), and streambed conduction (*bed*). The incident solar radiation (before shading) is assumed to be 80% of the measured short-wave radiation flux at an unshaded site using a hemispherical pyranometer. The solar altitude is (Tennessee Valley Authority 1972):

$$\sin \alpha = \sin \phi \sin \delta + \cos \phi \cos \delta \cos \tau \quad B3$$

where α = solar altitude ($^{\circ}$ above the horizon); ϕ = latitude ($^{\circ}$) taken as negative in the southern hemisphere; δ = solar declination ($^{\circ}$) given by:

$$\delta = 23.45 \frac{\pi}{180} \cos\left(\frac{2\pi(172 - D)}{365}\right) \quad B4$$

D = Julian day number and the hour angle of the sun (radians) is given by:

$$\tau = \frac{\pi(t - t_{noon})}{12} \quad B5$$

where t = clock time (h); and t_{noon} = clock time at solar noon (h). Solar azimuth is:

$$\sin \psi_{sun} = -\frac{\cos \delta \sin \tau}{\cos \alpha} \quad B6$$

where ψ_{sun} = azimuth angle of the sun measured from north. Topography and canopy angles are measured from the centreline of the stream to the top of surrounding

topography (i.e., hillsides or streambanks) or riparian vegetation. Both vary with azimuth angle. When the solar altitude is less than the topography angle at the sun's azimuth (i.e., the sun is below the level of the hills or stream banks) then no direct solar radiation enters the stream. When the solar altitude (symbol α) is greater than the canopy angle (i.e., the sun is above the riparian vegetation) the stream is in full sunlight. At intermediate altitudes the sun shines through the canopy and a fixed fraction of the incoming radiation is absorbed regardless of solar altitude. Thus

$$\Phi_{sol} = 0 \quad \text{if } \alpha < \theta_{top}(\psi_{sun}(t)) \quad \text{B7}$$

$$\Phi_{sol} = (1 - SF)\Phi_{msd} \quad \text{if } \theta_{top}(\psi_{sun}(t)) < \alpha < \theta_{can}(\psi_{sun}(t)) \quad \text{B8}$$

$$\Phi_{sol} = \Phi_{msd} \quad \text{if } \alpha > \theta_{can}(\psi_{sun}(t)) \quad \text{B9}$$

where Φ_{msd} = short-wave solar radiation flux measured at an unshaded site; SF = canopy shade factor ($0 < SF < 1$); θ_{top} and θ_{can} = elevation angles of the surrounding hills or streambanks (topography) and vegetation (canopy) respectively, measured from the stream centreline in the azimuth angle of the sun. A fraction of the incoming solar radiation is reflected (Beschta & Weatheredd 1984):

$$R_{sol} = \frac{0.091}{\cos(\frac{\pi}{2} - \alpha)} - 0.0386 \quad \text{if } \frac{\pi}{2} < \alpha < \frac{\pi}{18} \quad \text{B10}$$

$$R_{sol} = 0.0515(\frac{\pi}{2} - \alpha) - 3.635 \quad \text{if } \frac{\pi}{18} < \alpha < 0 \quad \text{B11}$$

Diffuse short-wave solar radiation is assumed uniform from the entire hemisphere and totals 20% of the total measured short-wave radiation. The fraction of the total diffuse radiation which reaches the stream is (Mason 1983):

$$f_s = 1 - \frac{1}{\pi} \int_0^{2\pi} \left[\int_0^{\theta_{top}(\psi)} \sin \alpha \cos \alpha \, d\alpha + SF \int_{\theta_{top}(\psi)}^{\theta_{can}(\psi)} \sin \alpha \cos \alpha \, d\alpha \right] d\psi \quad \text{B12}$$

where SF = canopy shade factor (i.e., the fraction of the incident radiation intercepted by the canopy vegetation); and ψ = azimuth angle of the sun. We assume that 3% of diffuse solar radiation is reflected.

The amount of long-wave radiation emitted by any solid body can be estimated using the Stefan-Boltzmann equation:

$$\Phi = e\sigma T^4 \quad \text{B13}$$

where σ = Stefan-Boltzmann constant ($2.0411 \times 10^{-7} \text{ kJ m}^{-2} \text{ h}^{-1}$); T = temperature (K) and e = emissivity. For long-wave atmospheric radiation, the emissivity of the atmosphere is given by Swinbank's formula:

$$e_{atm} = 0.937 \times 10^{-5} (1 + 0.17C^2) \quad \text{B14}$$

where C = cloud cover (fraction); and T is approximated by the air temperature measured 2 m above the ground (Tennessee Valley Authority 1972). Atmospheric radiation is assumed to be emitted uniformly from the entire hemisphere and the fraction of the total which reaches the stream is estimated from Eq. B12 (i.e., the canopy is assumed to intercept the same proportion of the incident short-wave solar and long-wave atmospheric radiation). The incoming atmospheric radiation is:

$$\Phi_{atm} = f_s 0.937 \times 10^{-5} (1 + 0.17C^2) \sigma T_{air}^6 \quad \text{B15}$$

The emissivity of water is set to 0.97 (Tennessee Valley Authority 1972) and the long-wave 'back' radiation emitted by the stream is:

$$\Phi_{wat} = 0.97\sigma T_{wat}^4 \quad \text{B16}$$

Canopy vegetation, streambanks and surrounding hills emit long-wave radiation which affects stream temperature and can be estimated in either of two different ways. First, the model user may choose to estimate canopy and topography long-wave radiation using Eq. B13. The emissivity of vegetation is then set to 0.95, canopy temperature is approximated by air temperature, and the canopy flux normal to the water surface is:

$$\Phi_{can} = SF I_{can} \int_0^{2\pi} \left[\int_{\theta_{top}(\psi)}^{\theta_{can}(\psi)} \sin \alpha \cos \alpha d\alpha \right] d\psi \quad \text{B17}$$

$$I_{can} = 0.95\sigma T_{air}^4 \quad \text{B18}$$

where SF = shade factor (see B12 above) which is also used to quantify the fraction of the canopy which emits long-wave radiation. The emissivity of the banks and hillsides is set to 0.90, topography temperature is again approximated by air temperature, and the topography flux is:

$$\Phi_{top} = I_{top} \int_0^{2\pi} \left[\int_0^{\theta_{top}(\psi)} \sin \alpha \cos \alpha d\alpha \right] d\psi \quad \text{B19}$$

$$I_{top} = 0.90\sigma T_{air}^4 \quad \text{B20}$$

Second, the user may assume that all long-wave ‘back’ radiation emitted by the stream water which is intercepted by the canopy or topography is absorbed and re-emitted (i.e., the net long-wave radiation in the solid angle occupied by topography or vegetation is zero: see text for a discussion). ‘Back’ radiation estimated from Eq. B16 is reduced by the factor $(1-f_s)$ where f_s is defined by Eq. B12 and Eq. B17-B20 are omitted.

Evaporation is given by the Brocard & Harlemann (1976) formula:

$$\Phi_{evap} = (e_w - e_a) \rho L_v F \quad \text{B21}$$

where e_w and e_a = saturated vapour pressures at the temperature of water and air respectively; ρ = density of water; L_v = latent heat of vaporization (J kg^{-1}) estimated from Tennessee Valley Authority (1972):

$$L_v = 3152.9 - 2.39T_{wat} \quad \text{B22}$$

T_{wat} = water temperature (K), F = wind function (Brocard & Harlemann 1976):

$$F = 1.3 \times 10^{-9} W + 1.1 \times 10^{-9} |T_{air}^* - T_{wat}^*|^{\frac{1}{3}} \quad \text{B23}$$

W = wind speed (m s^{-1}); $T_{air}^* - T_{wat}^*$ = virtual temperature difference between air and water given by:

$$|T_{air}^* - T_{wat}^*| = \left| \frac{T_{air} P}{P - 0.378e_{air}} - \frac{T_{wat} P}{P - 0.378e_{wat}} \right| \quad \text{B24}$$

P = atmospheric pressure (mbar); e_{air} = water vapour pressure in air (mbar)

$$e_{air} = 1.01725h \exp\left(21.381 - \frac{5347.5}{T_{air}}\right) \quad \text{B25}$$

h = relative humidity (fraction); T_{air} = air temperature (K); e_{wat} = water surface vapour pressure (mbar):

$$e_{wat} = 1.01725 \exp\left(21.381 - \frac{5347.5}{T_{wat}}\right) \quad \text{B26}$$

Convective (sensible) heat transfer is given by:

$$\Phi_{sens} = B \Phi_{evap} \quad \text{B27}$$

where B = Bowen's ratio

$$B = 0.613 \times 10^{-3} P \frac{T_{wat} - T_{air}}{e_{wat} - e_{air}} \quad \text{B28}$$

The heat flux across the sediment-water interface is:

$$\Phi_{bed} = -\frac{E_{bed}}{\Delta h} (T_{bed} - T_{wat}) \quad \text{B29}$$

where T_{bed} = sediment temperature in the surface layer of the stream bed; E_{bed} = thermal diffusivity of the bed sediments ($\text{W m}^{-1} \text{K}^{-1}$); and Δh = thickness of the surface sediment layer. T_{bed} is predicted by assuming the bed comprises n sediment layers of thickness Δh and solving the heat conduction equation numerically. The thermal diffusivity and heat capacity are assumed constant. The model user can assume either that there is no heat flux through the bottom boundary of the stream bed, or can specify the temperature at the bottom boundary of the stream bed (e.g., from groundwater temperature measurements).

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