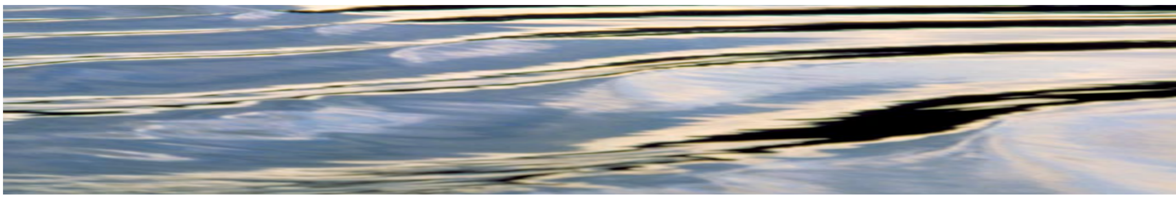
COOPERATIVE RESEARCH CENTRE FOR **CATCHMENT HYDROLOGY**

THE ROLE OF RIPARIAN VEGETATION IN CONTROLLING STREAM TEMPERATURE IN A SOUTHEAST QUEENSLAND STREAM

TECHNICAL REPORT
Report 05/3

April 2005

Nick Marsh / Christopher Rutherford / Stuart Bunn



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Bibliography

ISBN 1 920813 22 5

1. Riparian ecology - Queensland - Echidna Creek. 2. Water temperature - Effect of riparian plants on - Queensland - Echidna Creek. 3. Riparian plants - Queensland – Echidna Creek. I. Rutherford, J. C. (James Christopher), 1949- . II. Bunn, Stuart E. III. Cooperative Research Centre for Catchment Hydrology. IV. Title. (Series: Report (Cooperative Research Centre for Catchment Hydrology); 05/3).

333.9162153099432

Keywords

Water temperature
Revegetation
River management
Rivers
Riparian vegetation
Temperature profile
Seasonal variation
Rehabilitation
Modelling
Aquatic environment
Solar radiation
Shading
Channels

The Role of Riparian Vegetation in Controlling Stream Temperature in a Southeast Queensland Stream

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Preface

Despite the considerable government and private resources invested in the rehabilitation of damaged environments, little is known about the success of such projects. The Cooperative Research Centre (CRC) for Catchment Hydrology conducted a project (2000-2003) in collaboration with the CRC for Freshwater Ecology and the Moreton Bay and Catchments Healthy Waterways Partnership to assess the impact of stream rehabilitation on a few key elements of stream health. The project aimed to quantify the effects of a commonly adopted stream rehabilitation strategy on a small stream in southeast Queensland. The stream rehabilitation strategy was to exclude stock by fencing the stream, provide off-stream stock watering and to revegetate the riparian zone using endemic native species for a 1.5 km^2 catchment (Echidna Creek) near Nambour in southeast Queensland. Four key elements were monitored through the life of the project:

1. Suspended sediment load;
2. Channel morphology;
3. Water temperature;
4. Aquatic macrophyte growth.

The results of the water temperature response to revegetation are presented in this report. The other key research areas are presented in separate CRC for Catchment Hydrology technical reports.

Mike Stewardson
Program Leader – River Restoration
CRC for Catchment Hydrology

Executive Summary

This report presents the results of water temperature change following the revegetation of a small stream in southeast Queensland. Water temperature was measured every half hour from December 2000 until March 2004 in the treatment stream (that was subject to revegetation in early 2001), a control stream that remained cleared of trees and a fully forested reference stream. All streams had a catchment area of approximately 1.5 km^2 and were located within 3 km of each other.

There was up to 11ºC difference in daily maximum summer stream temperatures between vegetated and unvegetated streams. A similar magnitude (10ºC) was also shown for the daily temperature range in unvegetated streams compared to a single maximum daily temperature range of 2.6ºC in the reference stream. The mean of summer daily maximum temperatures was around 7-10ºC greater in the control stream than the reference stream and the mean of summer daily temperature ranges were from 2 - 5.5ºC greater in the unvegetated control stream than the reference stream.

We found that both the maximum daily summer temperatures and range in daily summer temperature increased in the summer following revegetation, followed by a continuous decrease over subsequent summers. The initial increase in stream temperatures following revegetation was due to the removal of woody weeds (blackberry, lantana) and tall grasses prior to revegetation which temporarily reduced the shading of the stream. An equilibrium summer temperature regime still had not been reached at the completion of the study, as the riparian vegetation had not yet achieved full canopy cover. At the current rate of recovery, we would not expect full temperature restoration until at least eight years after revegetation.

By considering downstream changes in channel geometry we presented and tested a model for predicting where revegetation projects are likely to have the greatest effect on stream temperature. We found that for vegetation of 10-15 m in height, revegetation is likely to have the greatest effect on streams with a bankfull width less than 20 m.

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1. Introduction

The replanting of riparian vegetation is a common stream rehabilitation activity largely because of the relatively low cost, limited expertise required and large range of potential benefits. These benefits include reducing channel erosion (Anderson, 1985; Beeson and Doyle, 1996; Gurnell, 1995; Henderson, 1986; Shields *et al.,* 1995; Stott, 1997), reducing stream temperatures (Johnson and Jones, 2000; Rutherford *et al.,* 1999); increasing carbon supplies (Bilby and Likens, 1980); and enhancing nutrient cycling. This report focuses on the quantification of temperature "recovery" following riparian revegetation.

Water temperature is a critical habitat element of freshwater ecosystems. Water temperature controls the solubility of gases, viscosity of water, rates of biota metabolism, and biota growth and decomposition rates (Islam *et al.,* 1986). For fish, the temperature regime can influence migration cues, egg maturation, spawning, incubation success, growth, and general stress which relates to intra-specific competition and susceptibility to parasites and diseases (Armour, 1991). Ectothermic organisms (such as fish and macroinvertebrates) are so called because their body temperature is controlled by the temperature of the surrounding environment. Such organisms are heavily influenced by the temperature regime of a stream, whereby otherwise suitable habitat niches can become uninhabitable due to a change in the temperature regime. An example would be the Goulburn River (Victoria, Australia) below Lake Eildon. The low-level cold water release from Lake Eildon has resulted in a drop in the January median water temperature from 19.5ºC to 12.5ºC (Gippel and Finlayson, 1993). The reduction in water temperature has removed Australian native warm-water fish from the reach downstream of the dam because their spawning temperature cues are likely to range from 16-23ºC through the summer months (Gippel and Finlayson, 1993). Similar results have been found by surveys of fish species abundance taken downstream of ten small dams in Michigan, USA (Lessard and Hayes, 2003). The results revealed a decreasing downstream response according to the temperature preferences of the fish under consideration. For warm-water fish, their abundances increased downstream from the cold-water dam releases, and cold-water fish had largely unchanged population densities (Lessard and Hayes, 2003).

The effect of water temperature on North American cold-water fish such as Coho Salmon (Welsh *et al.,* 2001) and Chinook Salmon (Armour, 1991) has been well researched, with the specific effects of a change in the temperature regime documented by Johnson and Kelsch (1998). For example, the recommended temperature range for the migration of adult Chinook Salmon is 2.0-16.0ºC , and for spawning is 5.0-14.0ºC (Wilson *et al.,* 1987). The prediction of temperature-preference relationships for fish has been hypothesised to relate to the combination of short cycle (daily range), (Beitinger *et al.,* 2000) long cycle (annual range) (Harris *et al.,* 2000; Johnson and Kelsch, 1998) as well as extreme thermal tolerances (Beitinger and Bennett, 2000; Currie *et al.,* 1998).

Quinn and McFarlane (1989) found that high water temperatures increased the metabolic rates of epilithon communities and decreased the saturation Dissolved Oxygen (DO) concentration in the Manawatu River, New Zealand. During summer low flows, the combination of high nutrient input (from farmland) and organic inputs (i.e. from sewage, milk and meat processing wastes) caused low night-time DO and occasional fish kills. Hopkins (1971) found that several species of Trichoptera were less abundant at downstream pastures sites in New Zealand streams than at headwater native bush sites, and attributed their decline to high temperature. Quinn and Hickey (1990) found that stonefly abundance in New Zealand streams declined markedly once maximum summer temperatures exceeded 19ºC and postulated that forest clearance for pastoral agriculture has reduced invertebrate species diversity. Quinn *et al.,* (1994) found that the lethal temperature for invertebrates from New Zealand streams (i.e. the constant temperature at which 50% of test animals acclimated at 15ºC died during a 96 hour test) varied from 22.6-22.6ºC (for the mayfly *Deleatidium* spp, the most sensitive organism tested) to 32.4ºC (for the snail *Potamopyrgus antipodarium* and the caddis *Pycnocentrodes aureola,* the two least sensitive species tested). Cox and Rutherford (2000b) extended Quinn's work by measuring the effect of diurnally varying temperature on *Deleatidium* and

Potamopyrgus mortality and from their findings developed a simple method for setting 'safe' limits for diurnally varying stream water temperature based on laboratory experiments conducted at constant temperature (Cox and Rutherford, 2000a).

Our knowledge of the specific temperature cues for Australian native fish and macroinvertebrates is, in comparison, extremely limited. Our ability to predict the impact of altering the temperature regime of a stream is curtailed by the absence of quantitative data. In Australia investigations of the effect of altering in-stream habitat on biotic response have in general adopted a 'reference stream approach'. The objective of this approach is to describe the condition of the habitat in relation to an undisturbed or reference stream rather than pursuing a process-based line of enquiry more commonly employed in North America and New Zealand (Rutherfurd *et al.,* 2000). The results of a reference stream based approach are based on differences in habitat conditions and population dynamics between reference and disturbed streams which are used to infer the presence and importance of any ecological impacts. This approach is particularly useful in the field of stream restoration as the restoration goal is often to recreate habitat conditions that mimic those of a reference stream. With such an approach there is no explicit consideration of the specific requirements of individual species, rather it is assumed that all physical habitat elements are important either for direct interaction with the target species or indirectly such as the creation of habitat for the food source of the target species. In this way, the dearth of information on the detailed habitat requirements of Australian native species is not an impediment to rehabilitation design.

A considerable downside to the reference stream approach to stream rehabilitation is the reality that many sites cannot be fully restored to reference condition because of some limiting constraint such as altered hydrology. Therefore, to maximise the probability of successful rehabilitation, restoration design should strive to meet both species-specific habitat requirements while concurrently restoring streams to near-natural (i.e. reference) condition.

In this paper, we compare the temperature regimes of reference and disturbed streams in subtropical southeast Queensland, Australia. The results illustrated the influence of riparian vegetation on stream temperature in small streams that are subjected to high ambient air temperatures. The results can be used as a guide to possible change in stream temperature caused by revegetation.

Poorly-vegetated streams with high summer maximum water temperatures may be unsuitable for colonisation by some temperature sensitive biota. Large ranges in daily temperature could also have a similar effect of making poorly vegetated streams uninhabitable. One of the effects of having poor riparian vegetation is direct solar heating of the water surface. To date, the potential difference in stream temperatures for vegetated and unvegetated streams has not been presented for subtropical streams. The purpose of this report is to firstly quantify the difference in daily maximum temperatures and daily temperature ranges between vegetated and unvegetated streams, and secondly to estimate how long it takes to restore stream temperatures following a riparian replanting project.

1.1 Conceptual Model

The effect of riparian shading on stream temperature can be modelled using conventional energy flux and flow routing algorithms (Armour, 1991; Rutherford *et al.,* 1999). Stream temperature predictive models are available, although the data needed to set up, calibrated and validate such models is often not available and is often onerous to collect. In association with this project, a water temperature model (Rutherford *et al.,* 1999) has been applied and calibrated for the treatment stream (Echidna Creek), see (Rutherford *et al.,* in review) for details. The contribution of this report to the study of instream temperature is through the provision of regionally specific stream temperature data, and the quantification of the response of stream temperature to a revegetation project.

The water temperature at any given point in the stream is a function of diffuse solar radiation, direct solar radiation, heat flux from upstream and heat transmission with the channel bed (Figure 1). The principal effect of revegetation is the influence of shading on direct solar radiation.

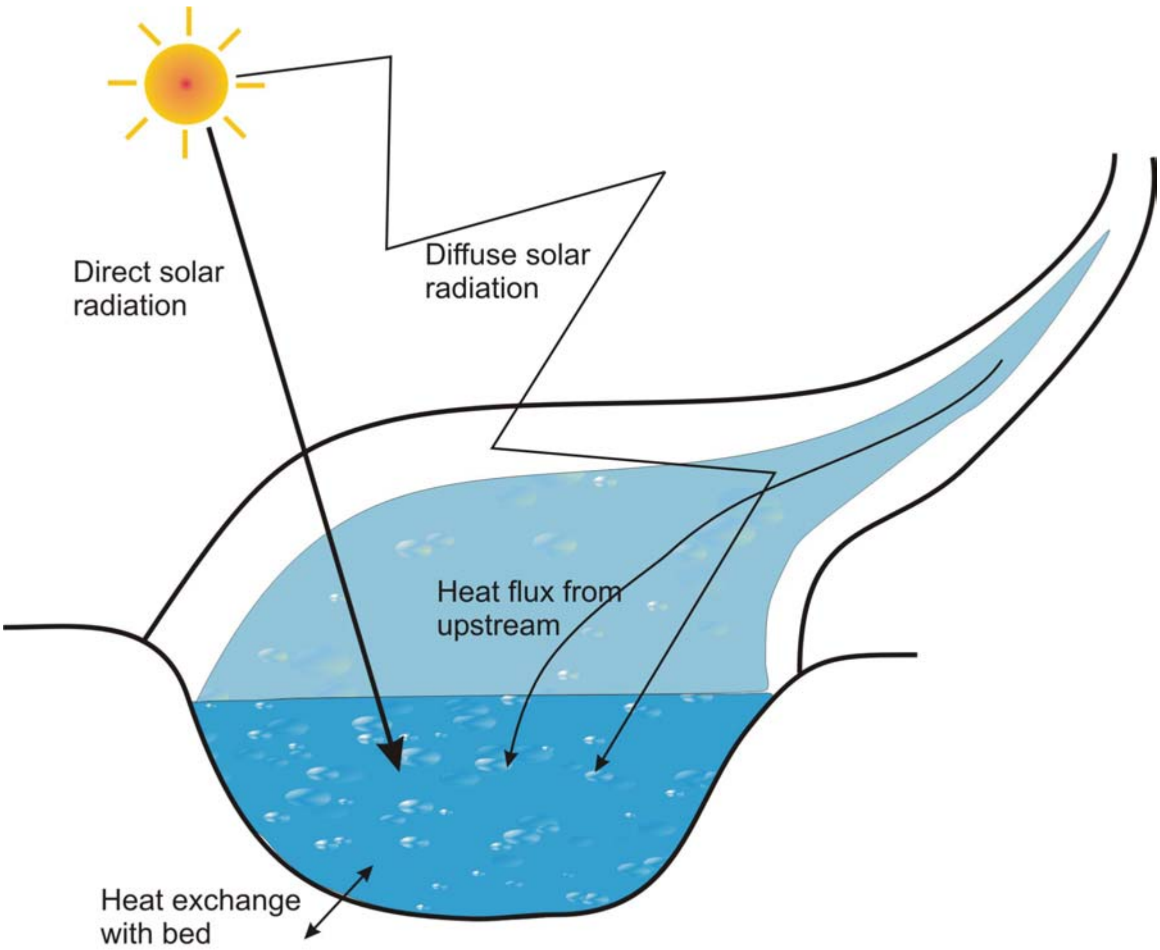

Figure 1. Key Controls on Water Temperature.

It was expected that prior to revegetation, the maximum daily instream temperature and temperature range in our treatment stream (Echidna Creek) would be similar to that measured in a nearby, degraded or control stream (Figure 2). During the rehabilitation process, it was expected that the instream temperature of Echidna Creek would move from that of the control stream to that of a fully forested reference stream. We expected an initial increase in maximum daily summer temperature in the treatment stream due to the removal of woody weeds and tall grass prior to planting seedlings. The maximum daily water temperature in the treatment stream were not expected to decrease below that of the control stream until the planted vegetation began to shade the stream.

We expected a similar model for the daily range in summer temperature such that the control and treatment streams would initially have a higher daily temperature range than the reference stream. The process of revegetation would initially result in an increase in the daily temperature range in the treatment stream due to clearing of woody weeds that shade the channel. This would be followed by a decrease in temperature range as the planted vegetation became established and began to shade the stream. These expectations are represented in our conceptual model (Figure 2) as variations in stream temperature over time.

1. 2 Methods

The experimental design was based on the 'BACI' approach (Before, After, Control, Impact). Water temperature was monitored at the treatment site (Echidna Creek), an unshaded control site (Dulong Creek) and a forested reference site (Piccabeen Creek) for one summer prior to revegetation, followed by three summers of monitoring after the revegetation of the test site, Echidna Creek (Figure 3).

The three streams are all of a similar elevation (200-300 m above sea level), and similar catchment area (1.5 km^2) and geological characteristics. The catchment vegetative cover varies dramatically

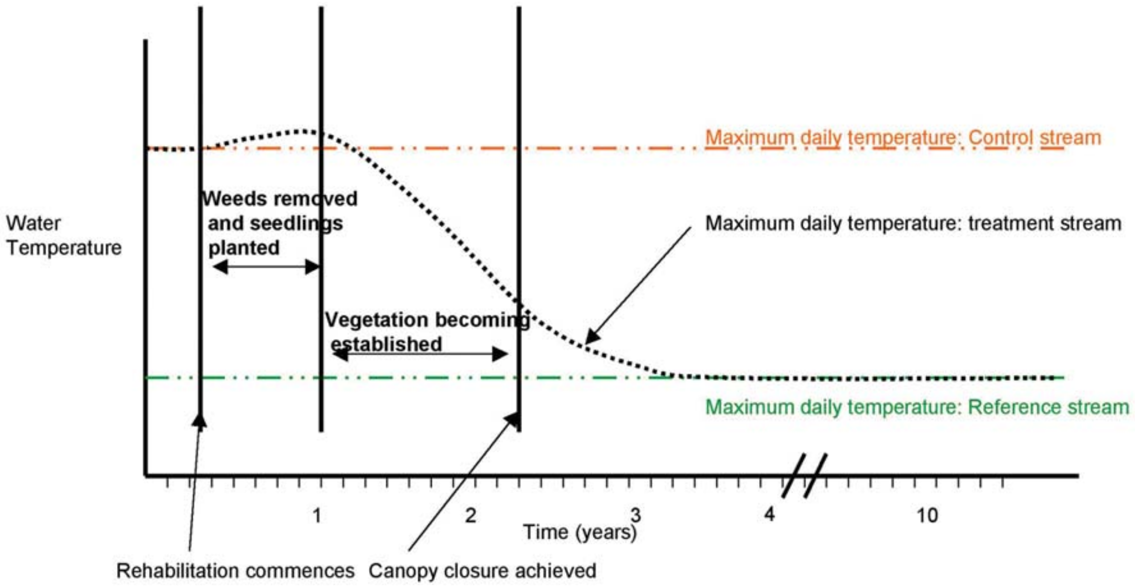

Figure 2. Conceptual Model of Temperature Response to Stream Rehabilitation.

between sites, with 100% forest for the reference site, approximately 50% forested for Echidna Creek and less than 10% forested for the negative control site. The amount of forest cover influences catchment hydrology, with dense, mature forests producing an increased proportion of rainfall infiltration and potentially lower evapotranspiration resulting in an increased baseflow supply compared to a catchment of immature forest (Cornish and Vertessy, 2001). The rate of delivery of baseflow is likely to vary between the catchments due to the alternate landuses. Baseflow delivery rates may be important when measuring stream temperature because a constant inflow of low temperature groundwater can buffer

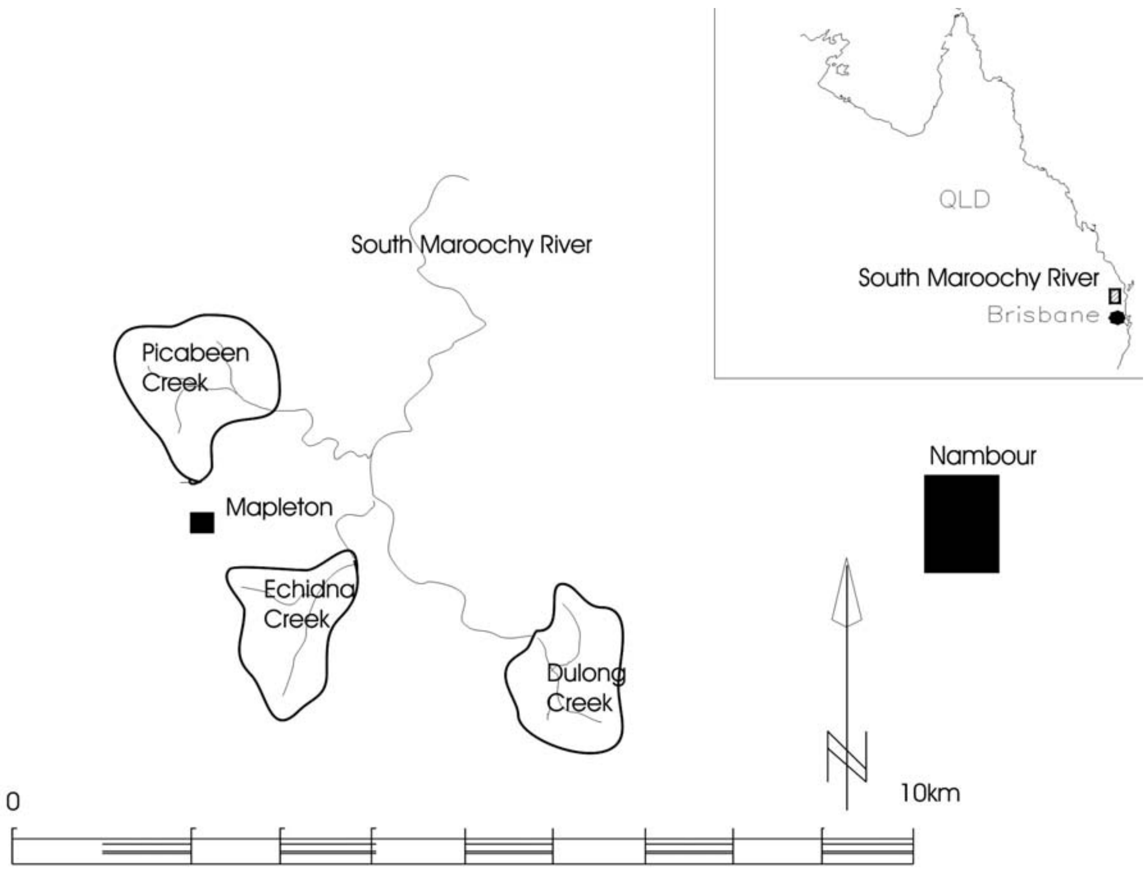

Figure 3. Field Site Locations (Temperature is recorded at the most downstream point of each catchment).

Figure 4. Echidna Creek, Downstream Monitoring Site April 2001 (left), and March 2004 (right).

against high water temperatures. We have gauged the three streams for the duration of the project to detect any significant differences in discharge regime (see Marsh and Rutherfurd 2004 for details of stream gauging). Stream temperature was logged at the three sites from December 2000 until April 2004 at 30-minute intervals.

The difference between the treatment site, the reference and the control sites can be tracked through time from before the revegetation commenced. This comparison of the difference between control and treatment sites removes ambiguity due to inter-annual climatic differences when considering simply a before and after type design.

1. 3 Site Descriptions

1.3.1 Treatment Site (Echidna Creek)

The treatment stream, Echidna Creek, is small with a bankfull width of around 1 m to 5 m and a catchment area of 1.5 km^2. Echidna Creek is a tributary of the South Maroochy River near Nambour in Southeast Queensland. The catchments near Echidna Creek were previously used for intensive dairy farming, although the present land use is largely hobby farming and low intensity cattle grazing. The area has fertile volcanic soils, and is about 250 m above sea level. The climate in Echidna Creek is subtropical, with a late summer dominated rainfall (average annual rainfall since 1952, 1732.6 mm/yr). Echidna Creek has two large farm dams in the upper 0.5km^2 of the catchment. The channel bed is cobbled with bedrock outcrops. The riparian vegetation for this length of stream is patchy and varies from dense over-storey to pasture grass.

Stock exclusion was achieved using a four-strand barbed wire fence with solar-powered off-stream stock watering points. At points where stock or vehicles traverse the stream low level concrete fords were constructed. Rehabilitation commenced in February 2001, with most rehabilitation complete by May 2001. The riparian revegetation consists of species grown from locally collected riparian seed stock. The revegetation process began by poisoning existing grass and weeds in the riparian zone using an organophosphate pesticide, followed immediately by planting tube-stock by digging a single hole for each tube stock. Each tube stock was watered-in and the whole riparian zone (approximately 5 m either side of stream was covered in a thick layer of mulch hay. A small section of fencing and revegetation was conducted in November 2001. Further secondary planting occurred in February-March 2002 to replace non-viable plants. Figure 4 compares the downstream sampling site at the beginning and end of the project.

1.3.2 Control Site (Dulong Creek)

Dulong Creek is located 2 km from Echidna Creek, and has a catchment area of 1.53 km^2. The land use is improved pasture (mostly Kikuyu grass (*Pennisetum clandestinum*)) for dairy and beef cattle grazing. There are two similar sized tributaries to Dulong Creek that join approximately 200 m upstream of the sampling site. A large dam is located on one of these tributaries.

Figure 5. Dulong Creek (control site).

1.3.3 Reference Site (Piccabeen Creek)

Piccabeen Creek is located approximately 2 km from Echidna Creek and has a similar catchment area (1.55 km^2). Piccabeen Creek is located within the Mapleton State Forest and has a fully forested catchment. The catchment has been subject to logging in the past, but regrowth within the catchment appears to comprise mature trees older than 30 years.

1.4 Results and Discussion

The temperature loggers were calibrated before being placed in the field and checked periodically with additional loggers installed at each site.

The data loggers recorded temperature every half hour on the half hour, giving around 17,500 individual temperature values per year for each site. The mean difference in maximum daily summer (Dec-Feb) temperatures was 5.3ºC and maximum winter (June-Aug) daily temperatures was 1.9ºC (treatment versus reference) (Figure 7). The difference in the mean summer daily range was 4.5ºC and 3.0ºC for winter (treatment versus reference). Given the largest divergence occurred during the summer months, our analysis concentrated on differences in the maximum daily temperatures and daily temperature ranges during this period.

Figure 6. Piccabeen Creek (Reference Site).

Stream water temperatures during rainfall periods tend to be less dependent on riparian shading than on hot days because incident solar radiation is reduced due to cloud cover, and discharge is increased as upstream surface runoff is increased, leading to a reduction in the residence time of open channel flow and an increase in the available thermal mass (greater volume of water) by comparison with non-rainfall periods. In effect, during rainfall periods the relative importance of shading on stream temperature is reduced by comparison with the buffering associated with increased thermal mass per unit time. Therefore, days where stream temperature is most likely to pose a threat to stream biota corresponds to ‘hot’ days, where the shading provided by riparian vegetation is likely to be a significant factor. Vegetation shading is critical on hot days, hence an algorithm was developed to distinguish between ‘hot’ days and ‘cool’ days, and the subsequent analysis is focussed on temperature differences between the various sites on ‘hot’ days. Data was excluded from the analysis (i.e. a cold day was designated) when the discharge in any two of the three streams exceeded a specified discharge threshold. (see Marsh and Rutherfurd, 2004 - How does riparian revegetation influence suspended sediment in a southeast Queensland stream?)

1.4.1 Maximum Daily Summer Temperatures

There was little inter-annual variation in the mean daily summer maximums of the air temperature (Figure 8).

The mean of daily maximum summer temperatures for the treatment site (Echidna Creek) is midway between the control and reference sites for the 2000-2001 summer (before treatment) (Table 1, Figure 8). For the first summer following treatment in April 2001, the water temperature in Echidna Creek increased

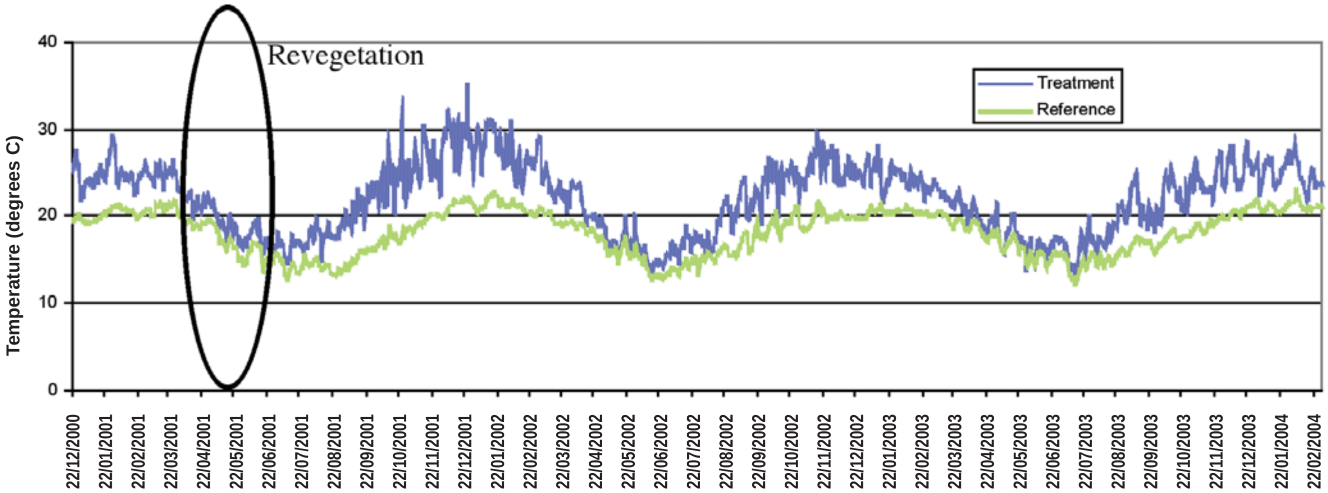

Figure 7. Daily Maximum Temperatures for the Treatment and Reference Stream for the Duration of the Study Period.

relative to the control and reference sites, as predicted by our conceptual model. This increase in water temperature is thought to be due to the removal of scrubby weeds along the channel. For the second and third summers following revegetation the maximum daily water temperature in Echidna Creek decreased to be less like that of the control site (Dulong Creek) and more like the reference site (Piccabeen Creek).

We explored the difference between the treatment and unshaded control streams graphically, then statistically. Firstly, considering just the treatment stream (Echidna Creek), in Figure 8, the first summer treatment in 2001-2002 appears to show a dramatic increase in stream temperature followed by a steady decrease in temperature over the subsequent two summers. The standard deviation of the daily maximum temperature at the treatment site increases from 1.6ºC to 2.5ºC following revegetation and then reduces again to 1.7ºC and 1.6ºC for the subsequent two summers. This increase in maximum temperature (Figure 8), and in the variability of daily maximums, was likely to be because of the removal of scrubby vegetation at the treatment site allowed more direct solar heating.

To remove the effect of inter-annual climatic variation we consider the response of the treatment stream relative to the temperature of the other streams and the ambient air temperatures. The inter-annual differences between the unshaded control and treatment stream (Figure 9) shows that the heating response of the treatment stream is more similar in shape to the heating response of the control stream in the first year after treatment than to the reference site. Three years after the treatment, the relative difference between the treatment and control sites is similar to before the treatment. The same comparison between

Table 1. Mean Daily Summer Maximum Temperatures for Baseflow Periods (mean, ±standard deviation (n)).

	2000-2001	**2001-2002**	**2002-2003**	**2003-2004**
Treatment (Echidna Creek)	24.97 ±1.60 (48)	28.14 ±2.46(88)	25.74 ±1.75 (56)	25.67 ±1.68 (41)
Control (Dulong Creek)	29.23 ±1.46 (48)	28.64 ±1.64(88)	26.42 ±1.16 (56)	30.85 ±2.05 (20)
Reference (Piccabeen Creek)	19.94 ±0.56 (48)	21.04 ±0.85 (88)	20.18 ±0.59 (56)	20.56 ±1.02 (41)
Air at Nambour DPI	28.60 ±2.22 (48)	29.41 ±4.75 (88)	28.33 ±2.71 (56)	28.42 ±3.50 (41)

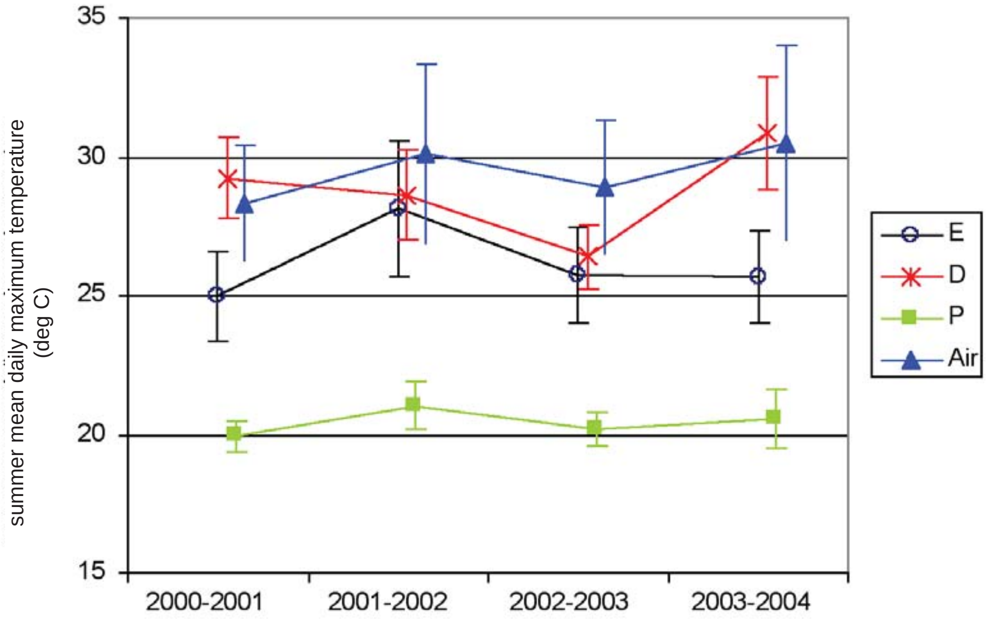

Figure 8. Mean Daily Maximum Summer Temperatures for Three Study Sites. (E- Echidna Creek - Treatment Site, D- Dulong Creek (Control Site), P-Piccabeen Creek (reference site), Air is air temperature at the Nambour Department of Primary Industries site). Intervention at the treatment site was after the first summer (bars show standard deviation).

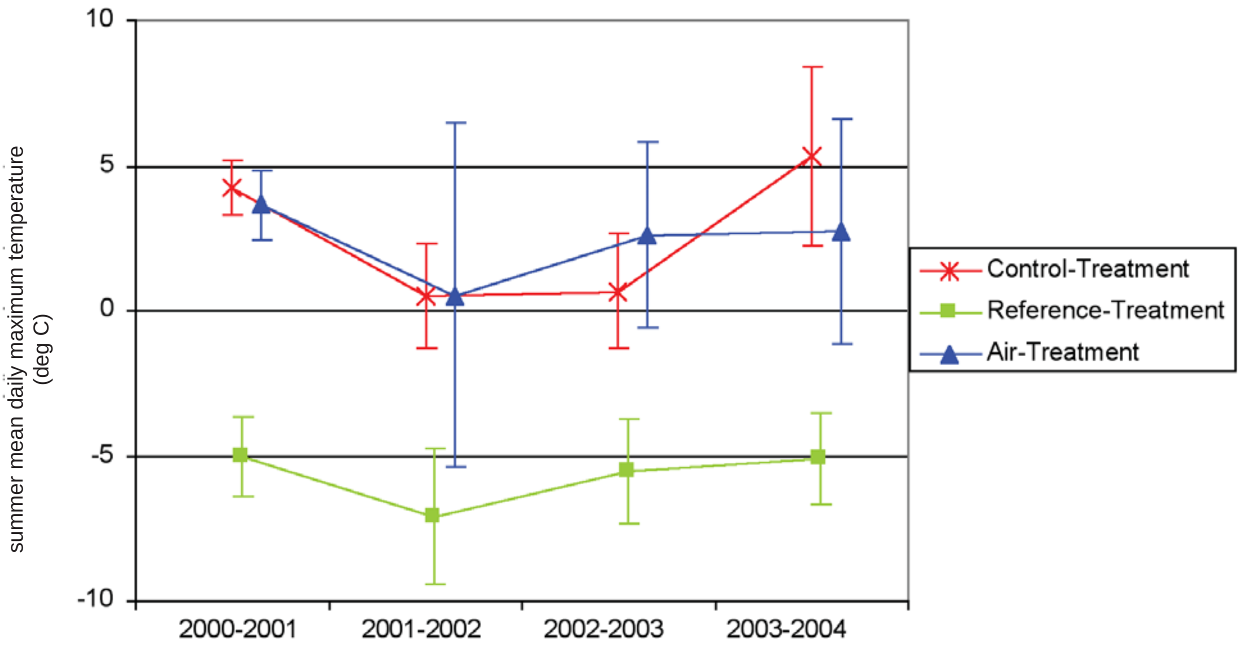

Figure 9. Mean and Standard Deviation of Control-Treatment for Daily Maximum Summer Temperatures (bars show standard deviation).

the treatment and forested reference stream shows a similar pattern (Figure 9).

The difference in summer mean daily maximum temperatures between the fully forested reference stream and unforested control stream varied from 6.23 to 10.9ºC over the four summers monitored. However by focussing only on summer means we may be missing the most ecologically-important aspect of the effect of water temperature, it may be that once the water temperature exceeds a critical threshold (e.g. lethal dose for 50% mortality value), the biological response may be dramatic. In the absence of critical threshold values to use as a guide for potential biotic impact we have compared the single hottest day each summer. Where biota are able to seek temperature

Table 2. Differences Between Mean Daily Summer Maximum Temperatures for Baseflow Periods (mean, ±standard deviation (n)). Summers that are not significantly different from each other have the same superscript (significance test: ANOVA, Tukey HSD $p < 0.05$).

	2000-2001	**2001-2002**	**2002-2003**	**2003-2004**
Control-Treatment	4.27 ±0.93 (48)^{a}	0.50 ±1.80 (88)^{b}	0.68 ±1.97 (56)^{b}	5.31 ±3.11 (41)^{a}
Reference-Treatment	-5.02 ±1.35 (48)^{c}	-7.10 ±2.34 (88)	-5.55 ±1.78 (56)^{c}	-5.11 ±1.56 (20)^{c}
Air-Treatment	3.63 ±1.19 (48)^{e}	0.54 ±5.94 (88)^{f}	2.60 ±3.20 (56)^{ef}	2.74 ±3.89 (41)^{ef}

refuges then this single worst case day is the most likely one to exceed any critical threshold for survival and could greatly affect the biota.

The single greatest difference in maximum temperature between the forested (reference) and unforested (control) stream is around 10ºC in each summer.

There is evidence that an acceptable temperature regime will be defined by more than just the 'hottest day' criterion. A series of temperature threshold experiments for macroinvertebrates were conducted in a study concerned with thermal pollution from sugar mills in North Queensland (Pearson and Penridge, 1979). For animals acclimated to 30ºC before the test, the authors found lethal thresholds 33-34ºC for Macrobrachium australiense, but make the point that the acclimation temperature was critical to determining the maximum survivable temperature. For example, goldfish which are considered a thermally tolerant species, has a thermal death point of 30.8ºC when acclimated to 10ºC, however when acclimated to 20ºC the lethal temperature increases to 34.8ºC, and further increases to 38.6 when acclimated to 30ºC (Pearson and Penridge, 1979). Hence it is not simply maximum stream temperatures that are critical, but the rate of change of temperatures. Pearson and Penridge's (1979) data suggests that for the streams considered herein, which have daily summer minimum temperatures of around 20-22ºC, the lethal thresholds for Macrobrachium australiense are likely to be lower than 33-34ºC reported. Given the sensitivity of organisms to rates of temperature change, these are examined in the following section by considering the maximum daily temperature range (i.e. rate = daily range divided by a maximum heating period of 12 hours).

Three summers after revegetation, the summer mean daily maximum temperature at the Echidna Creek treatment stream was still around 5ºC higher than at the reference stream, and absolute maximum daily temperatures were still around 6ºC higher. Unfortunately, the study did not continue long enough to determine exactly how long this continued recovery would take. If recovery continued at the same rate as observed over the first three years (approximately 1ºC improvement per year, then we would not expect the treatment stream to achieve 'temperature restoration' for at least another five years (2008-2009 summer). That is, a full eight years following treatment. This coarse estimation assumes a linear model and that the treatment stream will eventually have the same

Table 3. Maximum Temperatures for Each Forested (Reference) and Un-forested (Control Streams)

Summer	**Reference**	**Control**	**Maximum Difference Between Reference and Control Streams**	**Treatment**	**Air**
2000-2001	20.99	32.11	11.98	29.13	34.60
2001-2002	22.64	33.11	11.61	34.91	39.9
2002-2003	21.52	29.08	8.97	29.54	36.6
2003-2004	22.9	33.38	13.28	29.03	35.7

temperature regime as the reference stream. There may be other controls that limit the recovery ability of the treatment stream, and the recovery curve is unlikely to be linear, but rather the rate of recovery would decrease as the temperature regimes of the treatment and reference streams become more similar. Hence a total recovery time of eight years would be an absolute minimum estimate.

1.4.2 Daily Temperature Range

Riparian vegetation reduced the maximum water temperature by reducing the direct heating of the water surface through solar radiation. The presence of a dense canopy cover can also reduce overnight cooling (slightly) by reducing the radiant heat loss from the water surface. Hence, heavily vegetated streams should have a lower summer temperature range than open streams due to the decreased maxima and slightly higher minima. The biological implications for large temperature ranges are similar to those for extreme maximum temperatures. A rapid rate of heating or cooling may apply a similar stress on an organism as a single high water temperature event. We can compare the control, treatment and reference streams for temperature range in the same way that we considered maximum daily temperatures above.

Again we will only consider 'hot' days, where baseflow discharge prevailed in all streams (see Marsh and Rutherfurd (2004) for a detailed description of flow event definition).

The forested reference site has a very low summer daily temperature range throughout each summer, and a low variation within each summer (Figure 10). The treatment stream was similar to the control stream in the year prior to treatment (2000-2001) but had a much higher mean daily range for the first and second summer following treatment (2001-2002) after the shrubby riparian vegetation had been removed. Thereafter the mean daily temperature range for the treatment site has decreased steadily.

To remove the effect of inter-annual climatic variation we considered the response of the treatment stream relative to the other streams and to air temperature. Figure 11 shows the difference between the mean summer daily ranges for the reference and control streams and the air temperature with the mean summer daily ranges for the treatment site subtracted, these values are summarised in Table 5. If the lines on Figure 11 were horizontal, this would indicate that the treatment site was not changing through time relative to the other sites. However, the summer following treatment (2001-2002) the temperature range increased. Thereafter, a steady decline occurred in mean summer daily temperature ranges for the subsequent two summers. Thus, the inter-annual comparison between the treatment and other sites corroborates the story discussed previously for the maximum temperature analysis.

In the same way that we considered the single hottest day when considering maximum daily temperatures above, we can consider the day of single greatest temperature range. The pattern is the same as discussed previously with a dramatic difference between reference and control sites which is relatively consistent through time, and a dramatic decline at the treatment site after treatment, followed by a slow recovery.

Table 4. Mean Daily Summer Temperature Ranges for Non-event Periods (mean, ±standard deviation (n)).

	2000-2001	**2001-2002**	**2002-2003**	**2003-2004**
Treatment (Echidna Creek)	3.87 ±1.12 (48)	6.39 ±2.46(88)	5.48 ±2.03 (56)	3.42 ±1.37 (41)
Control (Dulong Creek)	4.78 ±1.47 (48)	3.06 ±1.08(88)	2.34 ±0.87 (56)	5.27 ±2.20 (20)
Reference (Piccabeen Creek)	0.27 ±0.34 (48)	0.34 ±0.19 (88)	0.45 ±0.51 (56)	0.26 ±0.2 (41)
Air at Nambour DPI	9.83 ±2.29 (48)	9.81 ±3.39 (88)	10.71 ±3.26 (56)	10.1 ±2.47 (41)

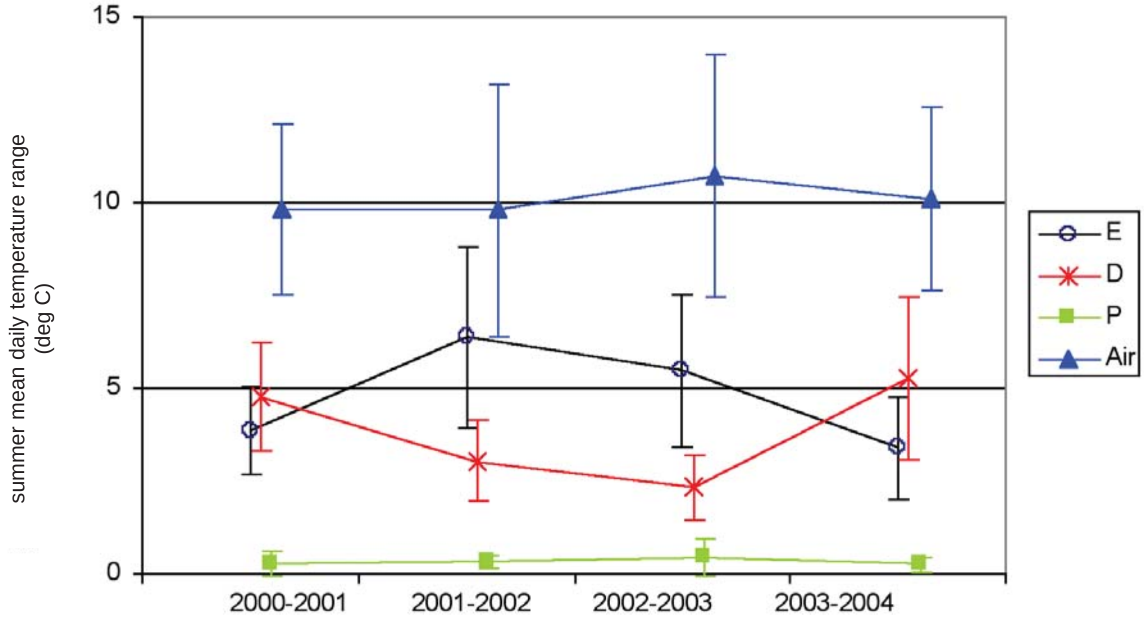

Figure 10. Mean and Standard Deviation of Daily Summer Temperature Range for Three Study Sites. (E- Echidna Creek - Treatment site, D- Dulong Creek (control sites), P-Piccabeen Creek (reference site)). Intervention at the treatment site was after the first summer.

Three summers after treatment the mean summer daily water temperature range is still about 3ºC above the reference site (Figure 11). In the three summers since treatment the mean summer temperature range has dropped from within 6ºC of the reference site to within 3ºC of the reference site. At this rate, the mean of the summer daily temperature range should be close to that of the reference site within another two years (five years from project commencement). However, the single greatest daily range is likely to take much longer to reduce to that of the reference site, if ever.

The above analysis has shown that:

1. The forested reference stream was cooler and had a lower daily temperature range than the unshaded control stream;
2. The maximum daily temperature and daily temperature range increased immediately following treatment, presumably because shrubby riparian vegetation removed to facilitate tree planting eliminated the modicum of shade that this had provided;

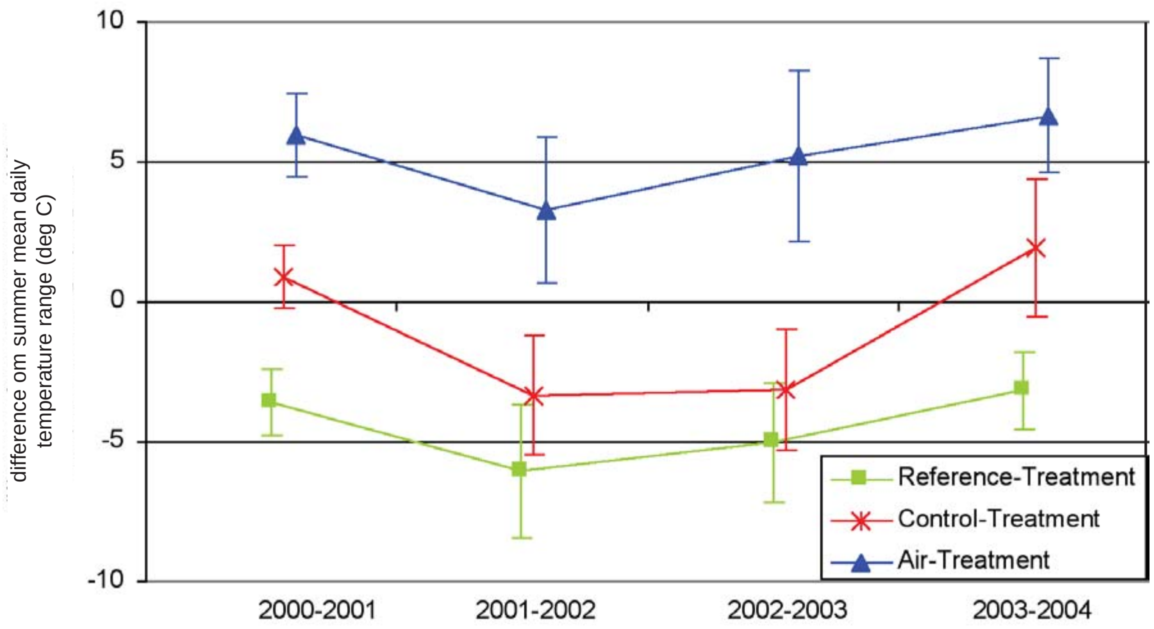

Figure 11. Mean and Standard Deviation for each Summer of the Daily Differences in Temperature Range between the Treatment and Reference, Treatment and Control and Treatment and Air Temperatures.

3. The maximum daily temperature and daily temperature range in the revegetated stream had returned to pre-revegetation levels in the third summer after revegetation, however;
4. We do not know how long it will take to achieve a fully restored temperature regime, but rough extrapolation indicates that a minimum of eight years will be required before the mean summer daily maximums are similar to those in the reference stream, and five years before the mean daily summer temperature range is comparable to the reference stream range.

The above analysis is very site specific. To investigate the likely response of revegetating streams of different size we conducted a subsequent experiment to investigate relationship between shade effect and channel size.

Table 5. Difference Between Mean Daily Summer Temperature Range for Baseflow Periods (mean, ±standard deviation (n)). Summers that are not significantly different from each other have the same superscript (significance test: ANOVA, Tukey HSD $p < 0.05$).

	2000-2001	**2001-2002**	**2002-2003**	**2003-2004**
Control-Treatment	4.27 ±0.93 (48)^{a}	0.50 ±1.80 (88)^{b}	0.68 ±1.97 (56)^{b}	5.31 ±3.11 (41)^{a}
Reference-Treatment	-5.02 ±1.35 (48)^{c}	-7.10 ±2.34 (88)	-5.55 ±1.78 (56)	-5.11 ±1.56 (20)^{c}
Air-Treatment	3.63 ±1.19 (48)^{e}	0.54 ±5.94 (88)	2.60 ±3.20 (56)^{e}	2.74 ±3.89 (41)^{e}

Table 6. Maximum Temperature Ranges for Forested (Reference), Un-forested (Control) Streams, Treatment, Air and an Upstream Site on the Treatment Stream.

	Reference	**Control**	**Maximum Difference Between Reference and Control Streams**	**Treatment**	**Forested site Upstream of Treatment**	**Air**
2000-2001	2.37	7.85	7.73	6.59	6.06	14.9
2001-2002	0.82	5.80	5.65	11.38	5.80	18.0
2002-2003	2.84	4.76	4.51	9.36	5.39	21.1
2003-2004	0.99	10.79	10.66	6.21	6.42	15.4

2. Downstream Trends in Water Temperature

The above case is likely to represent an extremely successful stream rehabilitation project (in terms of temperature restoration) because of the stream size. The channels were small (1-5 m wide) and shallow (<0.3 m) and had summer discharge regimes dominated by extended periods of low flow. These elements combined to provide shallow, low velocity water that has a high residence time in open sections of the channel, providing a large potential for heating due to direct solar radiation. Hence the provision of shade through revegetation is likely to have a large influence on water temperature. To put our results into a spatial context and to demonstrate that the success in mediating water temperature through vegetation demonstrated above may not be directly applicable to larger streams, we have investigated the downstream trends in water temperature in the upper 50 km of the Mary River in southeast Queensland. The intention of this part of the study was to look at the effects of gross channel dimensions on stream temperature and to present a simple model to illustrate where stream revegetation is likely to have the greatest effect on water temperatures.

We hypothesise that there are optimal locations within the catchment to place revegetation for maximising temperature restoration. We first present a conceptual model of downstream trends in water temperature for predicting the location of maximum benefit of vegetation placement (in terms of temperature reduction) based on the concepts in Bunn *et al.,* (1999) then test this using water temperature data from the Mary River. The focus of the model is exploration of the sensitivity of in-stream temperature to the ratio of

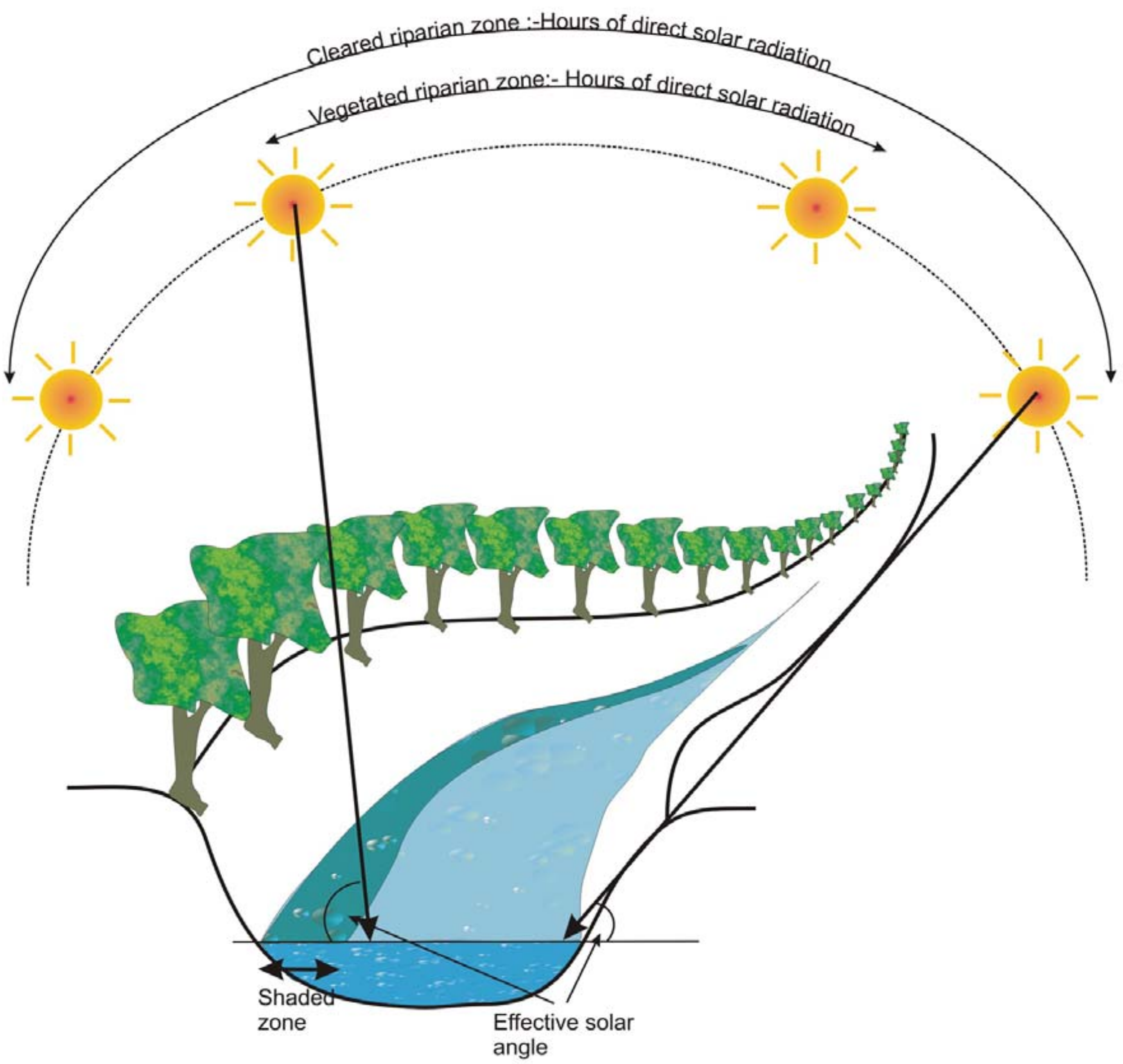

Figure 12. Shading Reduces the Number of Hours of Direct Solar Radiation.

vegetation height: stream width. We will focus the following sections on the effect that geometry has on channel shading, and assume other controls over stream temperature are constant across the sites considered.

2.1 Conceptual Model

If we consider the effectiveness of stream shading on altering stream temperature we can construct a simple conceptual model to quantify the impact shade (Figure 12). Shading the stream effectively reduces the number of hours each day that direct solar radiation reaches the water's surface (for a detailed discussion see Rutherford *et al.,* (1999)). The relative effect of revegetation is therefore affected by scale because trees have a limited maximum size, hence for a small channel the relative decrease in direct solar radiation hours will be greater than for wide channels. This is a simple model for predicting where bank revegetation is likely to have the greatest effect. A key assumption here is that the stream orientation is north-south. For other orientations reach length is also of central importance.

2.2 Methods

To test the model we installed water temperature loggers at seven sites in the Upper Mary River (Figure 13) on 17th February 2002 until 24th April 2002. The water temperature loggers recorded every half an hour. Key environmental variables that we collected were riparian cover (spherical densiometer readings) at, and upstream from, the temperature recording sites, wetted width, bankfull width and bankfull depth. The sites sampled had a reasonably continuous stand of fringing riparian vegetation on the bank top (but poor floodplain vegetation). The vegetation height varied slightly according to the dominant riparian vegetation type, but was mostly around 10-15 m in height.

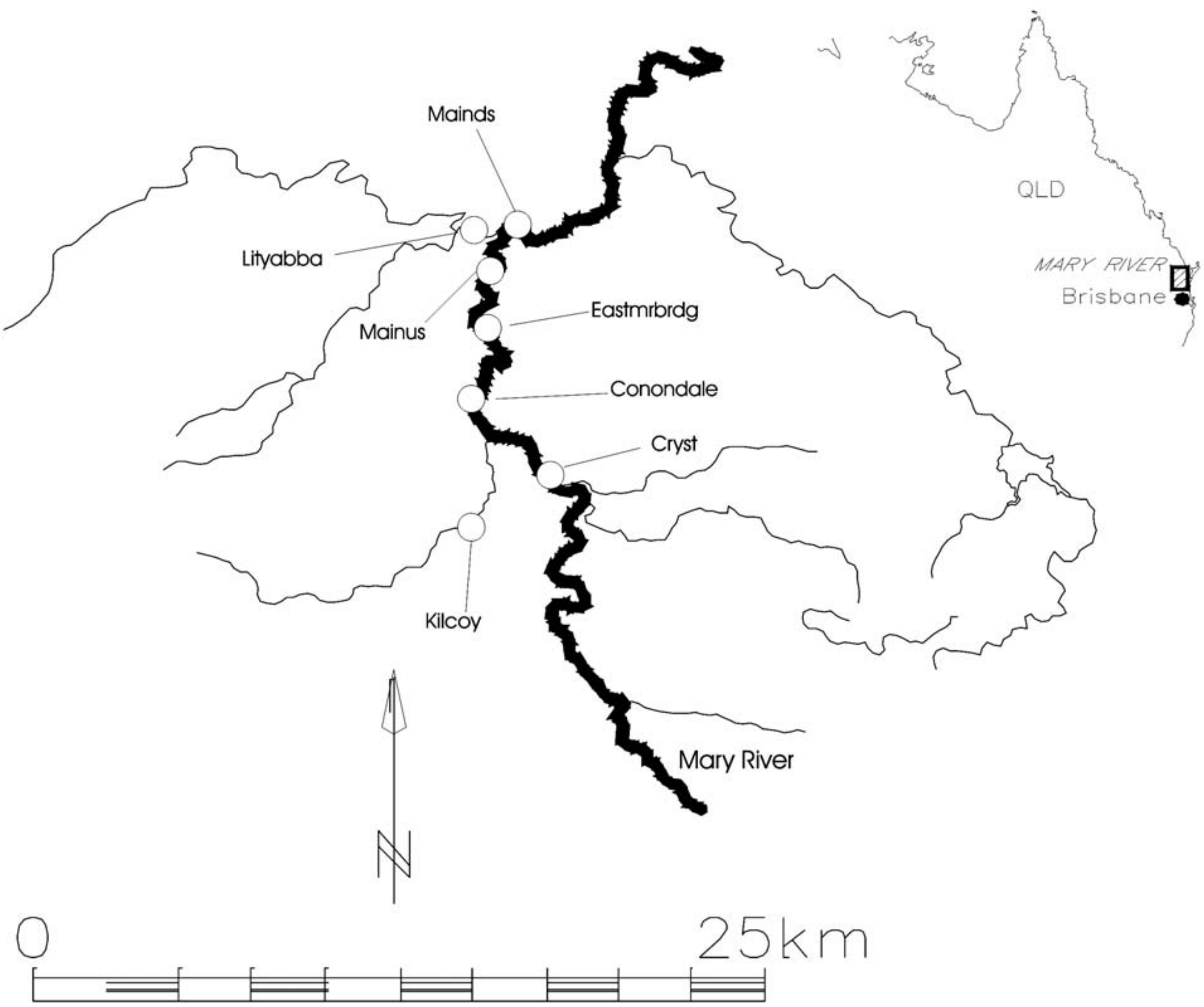

Figure 13. Location of Water Temperature Loggers on the Mary River.

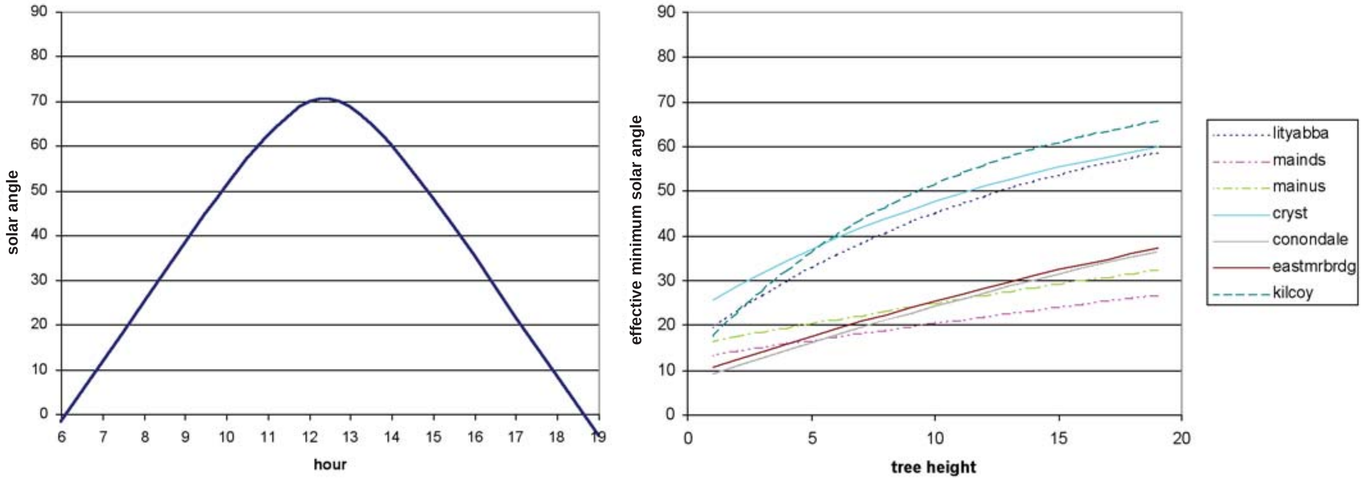

Figure 14. Solar Angle for Each Hour of the Day (1/3/2003) (left) Compared to the Minimum Solar Angle to Hit the Water Surface for Different Tree Heights (right).

2.3 Analysis

We considered the measured channel geometry for each reach and assumed that the low flow channel was centred within the bankfull channel. We determined the elevation angle of the banks from the stream centreline and assumed that, in the absence of trees, this was the minimum solar angle to hit the water surface. We also determined the minimum solar angle to hit the water surface for alternate tree heights (Figure 14). To explain Figure 14, further, consider the right hand side plot and the line representing the site "lityabba". For a tree height of 10 m on this site the minimum effective solar angle is 50º C, i.e. the angle between the sun and the water surface must be 50ºC or more before there is direct solar heating of the water surface. Looking across to the left hand plot of solar angle and hour for the 1st of March, we can see that the 50ºC solar angle is exceeded between 10 am and 3 pm (a total of five effective solar hours for the lityabba site). If you replaced the trees at the lityabba site with 1 m high shrubs or weeds, the solar angle is reduced to around 180 and the duration of direct solar heating is doubled to ten hours per day.

We then calculated the elevation angle of the sun every hour on the 1st March 2003, compared this with the elevation angle of the banks or trees and calculated the proportion of the wetted channel under direct solar radiation for each hour of the day from 6 am to 6 pm. We assumed 10 m high trees on all of the reaches. We then totalled these values to give an effective number of hours that the channel was under full solar radiation. We accept that this is a coarse assessment because it does not consider the reduced heating effect of a low azimuth sun early and late in the day, but the intention is to provide a rapid assessment of where trees are likely to have the greatest effect rather than a detailed predictive model of absolute temperatures.

There was a strong, positive correlation between the observed daily maximum water temperature at the seven sites on the Mary River and the calculated hours of solar radiation (Figure 15). To investigate scenarios of altered riparian vegetation we fitted a model to the measured data of Figure 15. Considerations in model fitting in this case are that the total range of hours of full solar radiation for the 1st March is about 12.5 hrs, and where the number of hours of solar radiation is small the relationship is likely to be poor because there would be reduced time for the solar radiation to affect temperature change. We have chosen a sigmoidal (S-shaped) model (shown on Figure 15) because it tends to flatten beyond the extents of the input data, making estimations conducted by extending the curve beyond the available data conservative (R^2=0.90). (Note: We do not present the equation for the curve because it is site and day specific, therefore of little use in generalising the results.)

Given the relationship between the mean of maximum daily water temperatures and the calculated index (hours of solar radiation) we then investigated the effect of vegetation in influencing water temperatures by calculating the hours of full solar radiation for a scenario where trees had been removed. With the trees

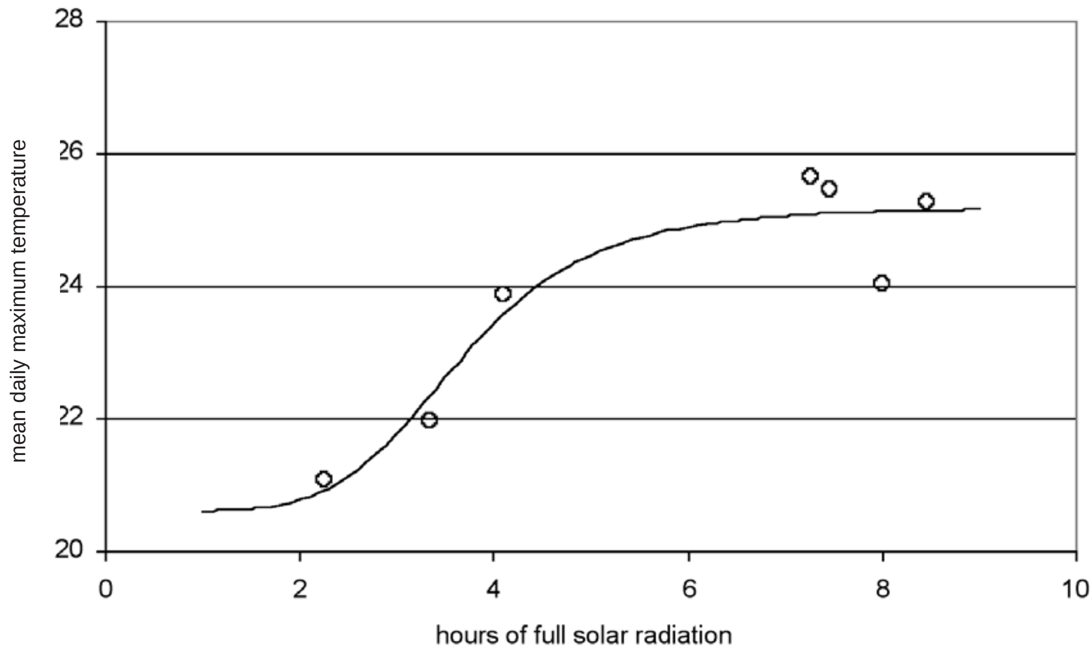

Figure 15. Computed Hours of Full Solar Radiation Versus Measured Mean of Daily Maximum Water Temperatures.

removed, the minimum effective solar angle reduces and the hours of full solar radiation increases. We applied the sigmoidal model in Figure 15 to the calculated hours of full solar radiation under the case where no trees were present on the banks to predict the response in maximum daily temperatures (Figure 16). We predicted that in the absence of trees the maximum daily temperature (in March) would be around 25ºC in all sites (25.1-25.2ºC). This represents a maximum temperature increase of 4ºC for Kilcoy Creek (bankfull width =12 m) and no real change in maximum temperatures for reaches with bankfull widths greater than around 40 m.

Whilst the absolute values are not transferable to other locations in Australia, the modelling approach could be used to predict where revegetation would be most effective. There was a dramatic increase in the number of solar hours predicted for channels with a bankfull width less than around 22 m (Figure 17), downstream from this point (i.e. larger channel size) the effect of vegetation was limited in controlling the number of solar hours. Davies-Colley and Quinn (1998) found that in New Zealand streams, the amount of incident light reaching the stream increased dramatically when the stream width was above 3.5m, regardless of vegetative cover. The wetted width and bankfull width that Davies-Colley and Quinn (1998) report are very similar. The wetted width appears to be within 90% of the bankfull width in most cases (Figure 5 of (Davies-Colley and Quinn, 1998)). The Mary River is an entrenched stream with low channel width to depth ratios (width to depth ratio = 5-12), such that the wetted width of the low flow channel is inset within a much deeper channel. For the seven sites of the Mary River considered here, the bankfull width was 2.8 times the wetted width (R^2=0.94), and the bankfull height was on average 0.4 times the wetted width (at baseflow discharge). For the Mary River, the top of the bankfull channel provides considerable shade to the low flow channel, hence the effect of riparian vegetation contributing to shade is effective for larger channels than streams with high bankfull width to depth ratios.

A general conclusion from this study is that there appears to be a positive relationship between solar hours and water temperature (near linear in this case), and that the relationship between the presence of trees and solar hours is strong for small streams that run north-south. Hence, where the height of riparian vegetation reaches 10-15 m, stream revegetation for temperature control would be most advantageous on reaches less than around 22 m bankfull width. This effective width is heavily influenced by the channel geometry, with vegetation having a relatively lesser effect in deeply incised streams than wide shallow channels.

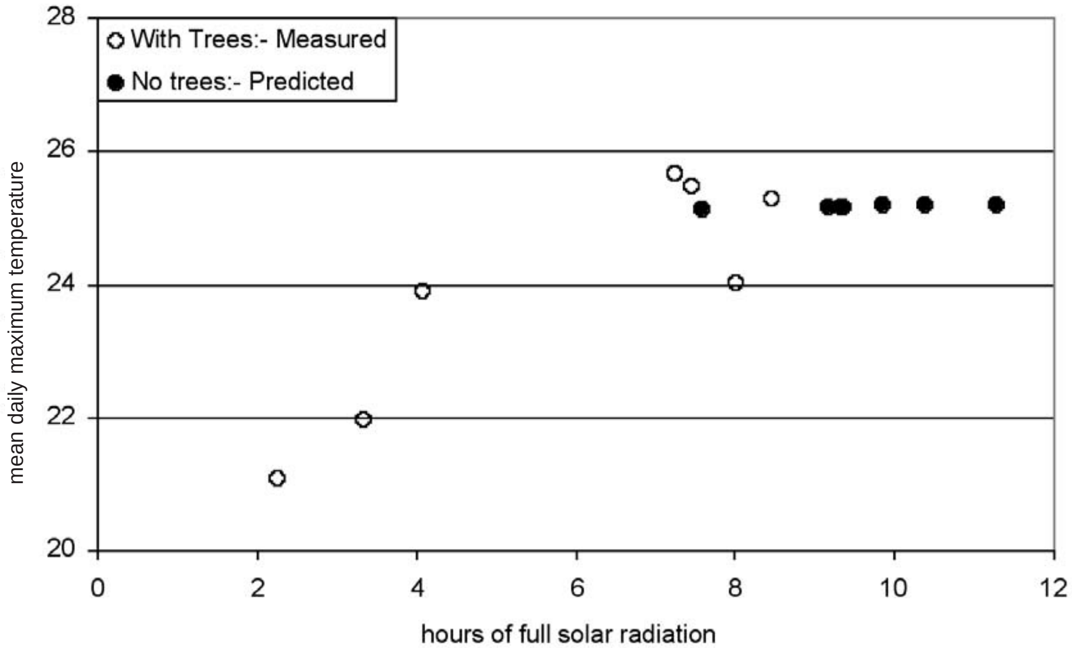

Figure 16. Predicted Stream Temperatures Without Trees.

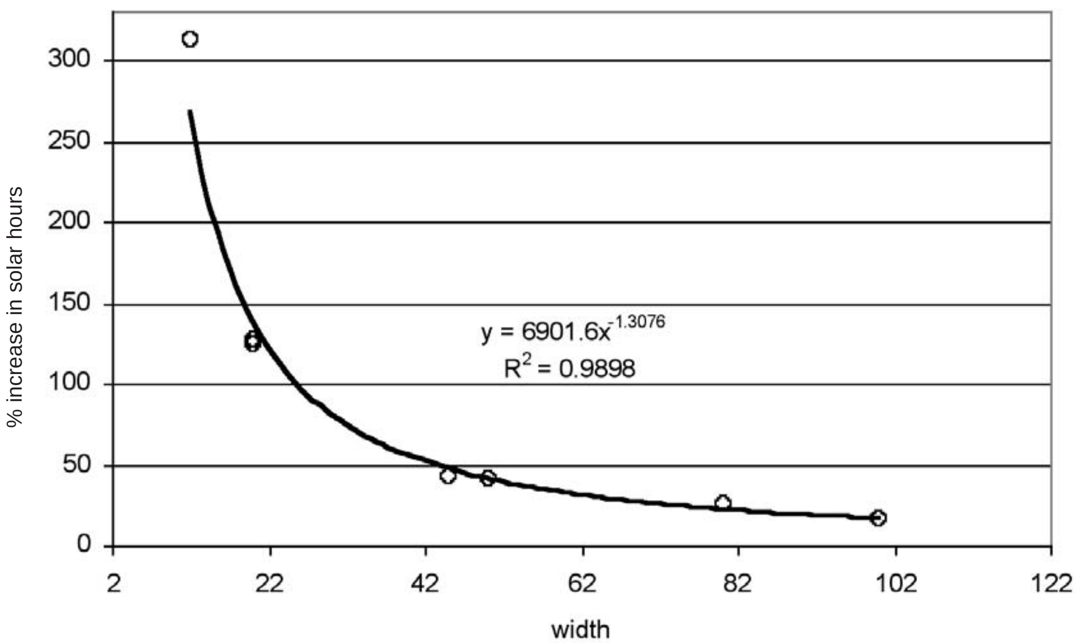

Figure 17. Increase in Solar Hours (Hours of Full Sun for the Full Low Flow Wetted Width) with Increasing Channel Size if Trees are Removed.

3. Management Implications

Riparian vegetation is clearly a major controlling influence on stream water temperatures. We have shown differences in maximum daily water temperatures and maximum daily temperature ranges of around 10ºC by comparing adjacent forested and un-forested streams. This large increase in temperature may cause the un-forested streams to be uninhabitable for some native biota.

The revegetation of Echidna Creek resulted in an initial increase in stream temperature in the summer following the treatment. This increase in stream temperatures was ascribed to the removal of scrubby vegetation that was providing some shade to the stream. Three summers after revegetation, the water temperature reduced to below the levels recorded prior to revegetation, but still remains warmer than conditions that prevail in a nearby forested stream. The warmer conditions limit the suitability of Echidna Creek for some biota. Based on the current trends in the temperature recovery curves we would expect temperature restoration to take a minimum of eight years from the commencement of planting. In addition, the restoration timeframe would vary across Australia, where vegetation is slower growing or the stream channel is larger than Echidna Creek.

The effectiveness of vegetation in controlling stream temperature is heavily influenced by channel width. Vegetation is likely to have the most dramatic effect on water temperature where the bankfull width is less than around 20 m for southeast Queensland streams with a low width to depth ratio. For streams with a width to depth ratio greater than around 12-15, the vegetation will only be effective on smaller channels. This does not mean that we cannot provide temperature refuges in large main channels. For instance, the confluence of small tributaries with the main channel could be the focus for revegetation activities for providing a suitable temperature regime in large channels. The cooler water in the small shaded tributary could provide a localised temperature refuge at the mouth of the tributary. Such a temperature refuge may only be important for a few hours each year during extremely hot days, but without a refuge from high temperatures, this few hours per year could limit the suitability of a whole reach for temperature sensitive biota.

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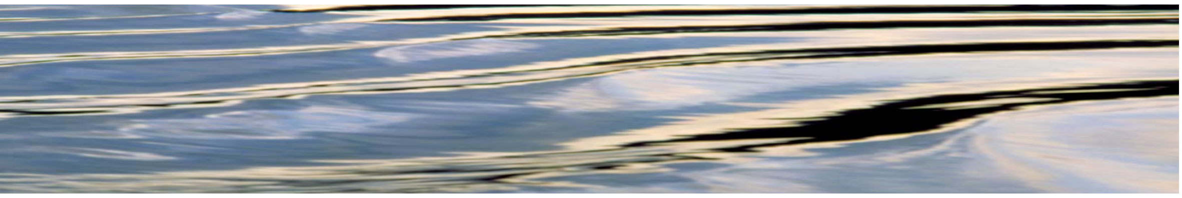

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