Brown trout redd hydraulics in three lowland streams

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A brown trout alevin from the Avon River

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Contents

1 Executive Summary

In our opinion, the state of the lowland brown trout fisheries is a cause of concern with falling angler patronage, and anecdotally low catch rates. In addition, monitoring of trout spawning activity in urban and rural streams in Canterbury has depicted significant shifts and declines in trout spawning activity, with the apparent loss of trout spawning grounds in the lower reaches of urban, and semi-rural streams.

The objectives of this investigation were to firstly measure the physical properties and water quality of trout redds within selected spawning reaches. Secondly, using the available literature, assess whether these habitat parameters meet the requirements for egg and juvenile trout survival. Thirdly, and a key objective of this study, was to highlight any parameters which are at, or approaching, intolerable levels for egg and juvenile trout development, and to recommend follow-up measures.

Three spawning reaches were chosen, two were along rural streams near Christchurch, and a third reach was on an urban waterway. For each of five redds along each reach, monitored physical parameters were the rate of water flow through the redd, the magnitude of upwelling or downwelling currents, and the redd's gravel composition. Water samples were drawn from the redds, via small probes, and the samples were assayed for dissolved oxygen, ammonia, biological oxygen demand, and combined nitrate/nitrite levels. Water quality counterparts were also obtained from the overlying surface waters.

Our results indicated statistically significant differences in redd oxygen levels (IGDO) within redds, between spawning reaches; and over the egg development period. IGDO's levels were always less than oxygen-saturated surface water, and sometimes markedly less. Redds were associated with both upwelling and downwelling water pressures, as measured at the centre of the redd, and there was a significant correlation between IGDO and vertical hydraulic gradient, with higher IGDOs tending to be associated with stronger downwelling pressures.

Late-term IGDO's in each of the 15 monitored redds, were negatively co-related to the proportion of fine particulates within the redd, that is the proportion of fines (by weight) less than 2 mm in diameter. The relationship between IGDO and fines became more sensitive when the proportion of fines less than 1 mm were considered. Superimposition of redds tended to confound relationships between IGDO and substrate composition.

Combining the physical factors into a mixed-effects predictive model indicated that IGDO is predicted by longitudinal location within the redd, the age of the redd since the date of construction, and the proportion of accumulated fines in the substrate. The age of the redd, and the proportion of accumulated fines were negative factors. These factors, combined, explained most of the significant differences in IGDO between spawning reaches. The downwelling effect on IGDO, mentioned above, while initially added to the model, but was rejected as being relatively insignificant compared to levels of fine sediment and redd age.

Water quality assays indicated that, across all spawning reaches, BOD was high in the redd gravels compared to surface waters, and this was also the pattern for total ammonia levels. Nitrate/Nitrite levels in the rural spawning reaches were also high for both the redd gravels and surface waters, but relatively low for the urban spawning reach. Based on North American trials, the nitrate/nitrite levels in the rural spawning reach were significantly higher than recently-released chronic threshold nitrate level for rainbow trout fry. We are concerned that if these threshold levels are applicable to New Zealand salmonids, that egg and fry health could be compromised, including that of brown trout. Further we thought it possible that low IGDO levels, at least in some redd environments, may acerbate higher levels of the more toxic nitrite form of dissolved nitrogen because of the effect of de-nitrifying bacteria in hypoxic groundwater.

Therefore, we consider it a priority that North American threshold levels for nitrate be placed into the context of the environmental conditions in New Zealand's lowland salmonid redds.

Accordingly, we recommended that an investigation be undertaken which monitors water quality within trout redds to the survivability of brown trout eggs. Currently a field investigation (by NCFG) is already underway, that, with minor modification, could provide more information on the association between nitrate levels and egg survivability in one rural stream. However, further studies across different catchment landuses would be required. Other recommendations are made in respect to managing and monitoring lowland brown trout fisheries for trout spawning.

2 Introduction

The New Zealand brown trout fishery, along with other salmonid fisheries, are a highly valued resource and national asset (McDowall 1990). The international quality of the trout fishery, at least in pristine habitats, attracts external revenue from overseas anglers, whereas fisheries with a lower international profile, but still productive, attract the interests of keen local anglers. Such is the strong link between New Zealanders and the accessibility of good fishing water, that fishing for many forms part of our national identify and social fabric.

Despite the undoubted social and financial value of the resource, there is little data on important and sensitive aspects of the brown trout's habitat in New Zealand, even though habitat quality must unpin the value of the fishery. Habitat quality is becoming recognised as a potentially limiting factor, because it is apparent to many anglers and scientists that the lowland trout fishery in particular has significantly declined over the past few decades, along with water quality and the surrounding environment. This is perhaps one of the reasons why there has been a decline in angler use of the more accessible lowland brown trout fishery, whereas inland trout fisheries continue to be well patronised (Fig. 1).

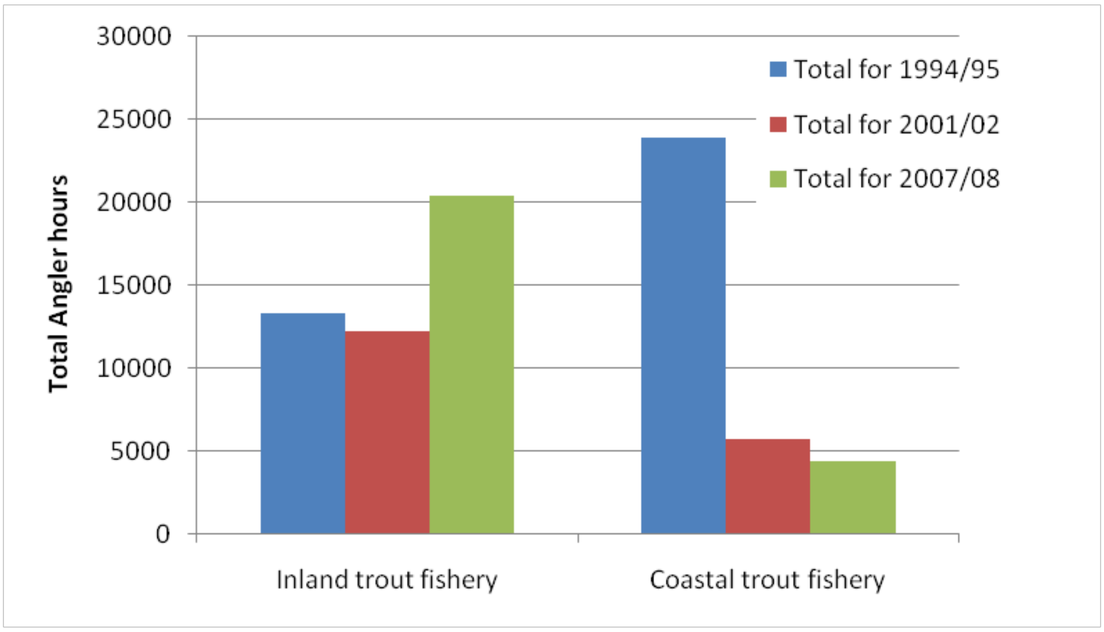


Figure 1. Decline in total angler hours for 18 popular inland trout fisheries and 11 coastal fisheries in the North Canterbury Fish and Game Region. Data obtained from Unwin (2009). Dataset excludes primarily salmon fishing waters.

We regard this as disadvantageous to the many anglers that lack the resources to fish the high-country lakes and rivers, and yearn for the convenience and enjoyment of a productive lowland fishery close to their home or workplace. From a fishery management perspective, a more geographically concentrated fishery is also not ideal. An intensification of fishing pressure on an increasingly small section of the national resource may cause over-fishing for some smaller waters. Even if the majority of the high-country anglers practice catch-and-release, increased fishing pressure could facilitate the risk of the transfer of the algae *didymo* between pristine waterways. Over time, even if not exactly crowded, the inland fisheries could begin to lack the solitude which attracts anglers in the first instance. We consider then, for social reasons, and from a fishery management perspective, a focus on the management of an accessible valued, lowland trout fishery is both timely and important.

While the ecological factors causing the decline in the lowland fishery are likely to be numerous and varied, habitat and water quality, are regarded as important factors, and there are good indications these qualities are declining in lowland streams. One indicator of a declining fishery is that trout catch rates for anglers are reducing especially those that have undergone significant changes in landuse and possibly landuse intensity, which has been linked to a decrease in river health (Harding *et al.* 1999). Declining water quality was reported from a six-year (1995-2001) Southland study (which included noted trout fisheries) where nitrate levels had increased, and dissolved oxygen levels decreased as farming intensified (i.e. livestock numbers) (Hamill & McBride 2003). Of interest was that, in addition to the nitrate and oxygen changes, dairying intensification as a sub-group, was associated with increased concentrations of dissolved reactive phosphorous.

A recent nationwide analysis of water quality in low-elevation streams demonstrated over-all temporal (1996-2002) decreases in flow, and increases in temperature, and clarity in streams draining pastoral catchments (Larned *et al.* 2004) although the definition of pastoral was broad, and included both agricultural and horticultural industries (e.g. dairying, row cropping). The pastoral land-cover class, as a whole, had the highest nitrate/nitrite levels compared to other lowland land cover types (i.e. pastoral, native forest, exotic forest, urban). The median nitrate/nitrite levels of climate class ‘cool dry’ pastoral streams (ca. 1.0 g/m^3), in particular, well-exceeded the ANZECC guideline for nitrate/nitrite (i.e. 0.44 g/m^3).

In Canterbury, over the last few decades, lowland trout river catchments near population centres have been dominated by two major changes. Firstly, the incursion of peri-urban residential developments in the vicinity of cities, and the development of large-scale satellite towns during the height of the residential development boom. The spawning grounds of brown trout are often close to towns and cities, and are exposed to these perturbations through the discharge of stormwater and its associated, often sediment-bound, contaminants. There is some indication that redd numbers along spawning reaches in urban and suburban rivers in mid-Canterbury indicate significant shifts in the location of trout spawning reaches, and often a reduction in spawning fish numbers (Taylor 2005).

The most juvenile stages of trout development, the developing embryo within the egg, and the yolk-sac fry which emerge, lack the ability to evade exposure to compromised water quality or habitat, and are physiologically the most vulnerable to poor water quality. Therefore, in an environment of compromised stream health, natural trout recruitment into a fishery could be potentially jeopardised the most, or at least the earliest.

3 Background

3.1.1 Background to trout redd hydraulics

Typical of others in the salmonid family of fishes, brown trout spawn in the gravels of the upper and middle reaches of flow-stable rivers (McDowall 1990). The oval-shaped gravel constructions in which a female trout (the hen) deposits her eggs is called a redd. Trout spawning has been well-described in New Zealand, with pioneering studies undertaken by (Hobbs 1948), and later on the morphology of redds by (Hardy 1963; Hawke 1978).

In the streams which form the basis of the study trout spawn in gravels immediately upstream of riffle beds. This zone is sometimes called the riffle, and was characterised as a zone of smooth, accelerating water, with decreasing depth. At the riffle crest, the water flow is swift and laminar, and often the flow is forced downwards into the gravels, owing to the abrupt increase in down gradient caused by the riffle downstream. The change in water gradient forces oxygenated surface water into the redd (Geist & Dauble 1998; Kondolf 2000). In more powerful rivers, which can scour pools into the gravel substrate, trout also spawn at the tail of pools, where a similar downwelling of water occurs (Fig. 2).

The stream gravels of the spawning ground are visible from the surface, although in lowland streams may be embedded into surrounding silt to a certain extent. In the process of creating a redd, the hen vigorously disturbs the gravels with her tail and flanks, and much of the surrounding silt is initially removed. After the hen deposits her eggs, which are then fertilised by the attendant male trout (the jack), the gravels cover the eggs, and the trout leave the redd. Within the redd, the flow of water to the embedded trout eggs, and therefore the influx of oxygen, and the efflux of metabolites, is driven by a hydrodynamic forces within and around the redd.

The rate in which oxygen and metabolites are transferred in the redd is determined by Darcy's Law:

$$V = -K\frac{dh}{dl}$$

Where: V = intragravel water velocity
K = hydraulic conductivity
dh/dl = hydraulic gradient

Clearly, these parameters are important properties in respect to maintaining water quality in the trout redd. However, the hydraulic conductivity of the bed is a rather difficult physical property to measure, and available techniques, while trialled, were heavy and impractical for transport by foot (e.g. double-ring infiltrometer). However, measuring the hydraulic gradient is relatively easy, and is achieved with a steel standpipe or peizometer driven into the redd gravels, and the hydraulic head can be measured directly. This method is detailed in the methods.

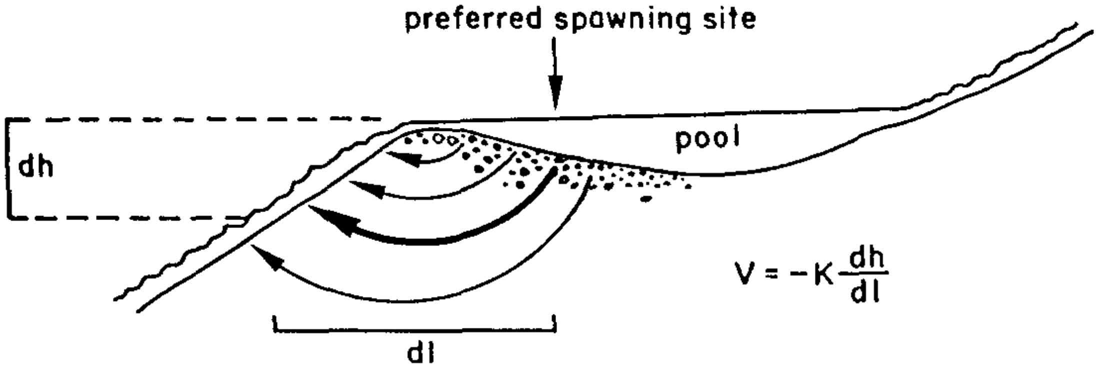


Figure 2. Water flow through a salmonid (trout) redd from Kondolf 2000. Also illustrated is Darcy's Law, which relates the water velocity through the redd (V) to the product of the hydraulic gradient (dh/dl), and the gravel permeability (or hydraulic conductivity, K). The vertical scale is grossly exaggerated. In our study, the flat glide at the crest of a riffle served as suitable habitat for trout redds.

3.1.2 Study Background

Overseas research, largely in North America, indicated that the quality of the physico-chemical environment of brown trout redds can have a major effect on egg development and survival. We considered it reasonable to consider that New Zealand brown trout redds require similar physico-chemical conditions, but it was highly apparent there was little knowledge on whether the quality of lowland trout redds met those conditions. Obtaining this knowledge became the primary objective of this survey.

A number of water quality parameters have been implicated in trout egg mortality in overseas studies (BioAnalysts 2003; Maret *et al.* 1993). This is because salmonid eggs, especially the very early stages, are sensitive to low levels of dissolved oxygen within the waters flowing through the redds. This parameter is called the intra-gravel dissolved oxygen (IGDO). IGDO had been found to be negatively associated with the levels of substrate fines within the redd gravels, although a high level of biological oxygen demand (BOD) has also been implicated as a possible cause of low IGDO (Maret *et al.* 1993).

While information on these redd characteristics (i.e. IGDO, BOD, and VHG) are scarce for New Zealand trout redds; other water quality parameters in lowland streams are of concern as catchments undergo rapid landuse change. In this context, the toxic effects of nitrates have come under renewed scrutiny both in New Zealand and overseas (Hickey 2009), but ammonia is also known to be quite toxic at low concentrations, and is common in both rural and industrial discharges (Hickey 2000).

In this study, we evaluate the properties of trout redds in spawning reaches on three lowland rivers, with differing catchment landuse, and attempt to establish estimates of some water quality parameters. Aquatic Ecology Limited (AEL) has undertaken recent trout spawning surveys, and therefore has good general knowledge of the whereabouts of spawning grounds in lowland rivers around Christchurch and mid-Canterbury (Taylor & Good, 2006; Taylor & Bray, 2008; Taylor & Burrell, 2002).

In this study, one spawning reach drained a low-elevation catchment with dairy and mixed-farming landuse. This particular reach had been subject to a riparian enhancement scheme administered by ECan and the North Canterbury Fish and Game Council (NCFGC) which is detailed further below. The second spawning reach is on the upper reaches of a lowland river fed by shallow groundwater from a major glacially-fed braided river. The third reach, on the Avon River, in the city of Christchurch, is within a well-established urbanised catchment with untreated stormwater inputs. This river rises from springs fed by aquifers derived ultimately from Waimakariri River groundwater.

4 Objectives

Our principal objective is to provide information and techniques to resource managers so that the water quality of the lowland fishery can be accurately determined, and therefore better managed. The following objectives form the focus for this study:

- To develop and trial practical ways of assessing trout redd water quality and health, and which can be applied to lowland streams.
- Comparing our results to those from overseas and New Zealand.
- Formulate recommendations in which the methods could be improved, and data obtained from other lowland environments.

5 Catchment and reach descriptions

The general location of the three lowland trout spawning reaches are depicted in Fig. 3, which portrays the urban catchment of the Avon River in central Christchurch, compared to the rural settings of Boggy Creek and Silverstream. Boggy Creek discharges directly into Lake Ellesmere, whereas Silverstream discharges into the Waimakariri River.

Figure 3. The location of the three trout spawning reaches (yellow pins and labels) monitored during the 2009 winter.

5.1 Boggy Creek

The Boggy Creek catchment has been converted from sheep grazing into dairy production over the last decade. Historical Fish and Game records indicate that the reach of Boggy Creek between Volckman and Rushbrooks Roads was subject to stock damage and little trout spawning was reported (Ross 1984). In the summer of 2002, this reach was fenced and planted under a restoration programme under ECan's Living Streams Programme. Following restoration, an AEL spawning survey reported that the number of redds had increased greatly compared to those reported in 1984 (Taylor & Good 2006). There are no known previous data in respect to trout redd hydraulics.

Boggy Creek is a normally perennial spring-fed feeding tributary to Lake Ellesmere/Te Waihora. The following catchment description of Boggy Creek is based on excerpts from a detailed physical description from Taylor & Good (2006).

".. Boggy Creek rose from a straight farm drain in Killinchy Swamp near the homestead of Glencairn, 840 m north-west of Beltons Road. It flows generally south-east for 16 km before discharging into Lake Ellesmere/Te Waihora."

There were clearly upstream problems with channel widening and clearing at the time of the survey.

"Downstream of Beltons Road to Irwell Rakaia Road, Boggy Creek had been widened. Immediately downstream of Beltons Road the width was approximately 3.5 m; with a mean mid-channel depth of 0.4 m. However, about 140 m downstream of Beltons Road, the channel had been excavated further to a width of 7 m, and a depth of 0.2 The substrate was silted, with a patchy distribution of aquatic macrophyte beds (i.e. Curly pondweed., monkey musk, starwort etc). Over this 750 m reach, the flow was laminar and sluggish, with a thick layer of silt overlying gravel."

This section relates directly to the study reach in 2006.

".. Between Rushbrooks and Volkman Roads, a distance of 600 m, both banks of Boggy Creek were fenced, and the riparian strips were vegetated in purei (*Carex secta*) (App. II, Fig. dj). Gorse had been recently removed from most of the reach, and replanted with native vegetation. This restoration was the result of an ECan "Living Streams" project. The flow character alternated between riffle and run, with a channel width of approximately 3 m, and a mid-channel depth of 0.2 m. Beds of monkey musk lined the bank in some reaches. The substrate was predominantly coarse gravel, with some smaller cobbles." It became part of ECan Living Streams programme and the reach between and was fenced from stock along both banks, and planted with native saplings and shrubs in the summer of 2003.

This description still approximates the physical appearance of the study reach in the winter of 2009. Six years after establishment of the buffer strip, the 4 m buffer strip (on each bank) is patchy and thinly planted in places. The native trees have now reached a maximum height of approximately 3m. In terms of flowtype, along the study reach, the flow is composed of short riffles, interspersed by smooth glides of smooth, laminar flow (Figs. 4a, b). No pools of any significant depth were recorded over the study reach.

Figure 4a. Boggy Creek looking downstream from Rushbrooks Road (Taylor & Good 2006).

Figure 4b. Boggy Creek, looking upstream from near Volkmann Road.

5.1.1 Boggy Creek trout fishery

Boggy Creek does not appear as a fishery in Fish and Game's recent or older Angler Survey reports (Teirney *et al.* 1987; Unwin 2009). However, it is still likely to be fished to a minor extent in the lower reaches and its mouth (J. Holland, pers. comm.).

5.2 Avon River

The following catchment description is abridged from (Eldon & Kelly 1992):

"The Avon River catchment extends over an area of approximately 84 km^2. The terrain is flat and does not rise above 30 m a.s.l. The mainstem rises as a number of springs to the west of the city, and follows a course of 26 km to empty into the northern apex of the Avon-Heathcote Estuary. At the point of discharge onto the estuary, the Avon "carries an estimated low flow of 2.7 cumecs"

The monitored spawning reach flows as a laminar glide upstream of Wood Lane, in the suburb of Fendalton. The reach gravels possessed patches of aquatic macrophytes interspersed across the bed. The channel width of approximately 6 m, with a depth varying between 0.3-0.5 m (Fig. 5).

5.2.1 Avon River trout fishery

In the 1920s, the Avon River supported a good brown trout fishery, with a major decline from 1962 to 1968 (Eldon & Kelly 1992). Based on an angler response surveys in 1980, the Avon River was rated 3 out of 5, with 5 being high in terms of angler appeal. This would suggest that at least then the river was still productive (Teirney *et al.* 1987). In 1992, in the inaugural general fisheries report, the Avon River was considered a remarkable trout fishery, in the context of an urban river, with an appeal for junior anglers (Eldon & Kelly 1992).

In the present day, it is apparent that while trout are still present, it is not regarded as possessing a fishery of any importance to adult anglers. Fishing pressure is light, with angler days on the Avon River have steadily declined from 1020 ± 450 in 1994/95, to 730 ± 250 in 2001/2002, to 550 ± 240 in 2007/08 (Unwin 2009). Fishing pressure is primarily over the summer months. A recent review of trout spawning activity indicated that the formerly productive trout spawning reaches in the central business district and the inner suburbs now have low or no utilisation by spawning trout (Taylor & Bray 2008). However, in the upper tributaries, trout spawning gravels are still well-utilised.

A recent comparison of trout numbers from reaches repeat electric-fished indicated a marked reduction in juvenile trout numbers in the upper reaches of the Avon River compared with the same reaches in 1992 (Main & Taylor 2010).

Figure 5. A trout spawning reach on the Avon River, downstream of the Mona Vale Weir. Monitoring probes can be seen inserted in a redd in this photo.

5.3 Silverstream

Silverstream is the colloquial name for the headwaters of the Kaiapoi River, which is a lowland tributary of the Waimakariri River. Silverstream is fed from shallow groundwater rising beside the stopbanks of the Waimakariri River. A close examination of the Google Earth aerial photographs of the Silverstream headwaters (dated June 2009) showed that Silverstream was once an old Waimakariri River braid, and this old braid is still discernable and now covered in grazed pasture grass. Silverstream continued to meander through rural land for a distance of 15.4 km to its confluence with the lower Waimakariri River.

A reach upstream of Hayward Road was selected, which is approximately 1700 m downstream of its groundwater source. This reach is also upstream of the former NIWA salmon hatchery and research station. The station has been sold to private interests, and still operates a fish trap across the river. However, the trap is monitored, and spawning trout are netted, and allowed to migrate further upstream to the spawning reach upstream of Haywards Road.

Along the spawning reach, the waterway flows as a flat glide, with a clean gravel substrate interspersed with small patches of macrophytes (Fig. 6). The water is normally very clear, except when the Waimakariri is high and carrying turbid flow. At these times, Silverstream rises, and a degree of turbidity is discernable, but the headwaters would not appear to become turbid at high flow.

5.3.1 Fishery

The lower river, the Kaiapoi River, is a well-utilised salmon fishery, with a fishing pressure which tends to follow the erratic fortunes of the salmon run. However, the upper reaches of the Kaiapoi River, Silverstream, tends to be more of a trout fishery. Silverstream has evidently suffered a decline in use by anglers, because, based on the angler surveys, angler effort has declined from 1400 ± 620 angler days (1994/95), to 320 ± 150 angler days (2001/02), to just 20 ± 20 annual angler days over the (2007/08) summer period (Unwin 2009). The decline in fishing pressure may be at least partially due to a reduction in the induced salmon run from ocean ranching that was employed experimentally by the former NIWA salmon hatchery.

Figure 6. The trout spawning reach on "Silverstream", the upper reaches of the Kaiapoi River. The orange marker stakes (arrowed), indicate the location of monitored redds.

6 Methods

6.1 Survey Design

6.1.1 Rationale for Study Reach selection

The spawning reach on Boggy Creek was selected because it represented a habitat which has been subject to stock (sheep) access in the past, but one which has also been protected by fencing, and the development of a narrow (ca. 2-3m) buffer strip of native vegetation. The 2006 trout spawning survey had indicated a marked increase in utilisation of this reach by spawning trout since the 1980s (Ross 1984).

The Avon River spawning reach, like most reaches flowing thorough the older suburbs, receives direct untreated stormwater inputs, and thus contrasts with Boggy Creek. Its location downstream of a substantial waterfall (Mona Vale weir) ensures that surface water oxygen levels are close to saturation (AEL raw data). However, the site is downstream of the Mona Vale ponds, which are a favoured habitat for ducks, and possible source of ammonia and high BOD (biological oxygen demand). Similarly, a significant positive relationship was drawn between numbers of birds on a Canterbury River (Ashburton River) and local faecal coliform levels (Main 1999). Significantly high faecal coliform levels are reported from the groynes picnic area on the Otukaikino River system, which is partially attributed to waterfowl in the ponds (Environment Canterbury 2004). An overview of water quality in Canterbury waterways attributes birdlife (along with agriculture) as principal sources of high faecal coliform counts in lowland rivers (Meredith & Hayward 2002).

For this study, the Silverstream spawning reach served as a 'reference' site, a spawning habitat close to the headwaters, and with only minimal stock access. Silverstream is fed by the Waimakariri surface waters which have low conductivity and nutrient levels. For the purposes of comparison, there are previous data on IGDO levels using a different collection method, and VHG data.

6.1.2 Redd selection

Foot surveys were conducted along the three spawning reaches commencing 18-19 June 2009 through the spawning season with the objective of locating either fresh redds or redds under construction. Thus, if required, surveys for fresh redds were conducted during the monitoring visits for the older redds. For Boggy Creek, five very fresh completed redds were already present on the 18th June, as evidenced by clean gravel without periphyton and the lack of attendant trout. At the time, further redds were still under construction by attendant trout, and these gravels appeared just as clean as the completed redds. We assumed that the five completed redds were fresh, no more than two days old, based on the rapid rate of periphyton accrual on the redds during subsequent visits.

Several partially completed redds were located on the Avon River on the 19th June, with three completed redds monitored after 24th June. A local landowner had seen trout on these redds up to that morning. Two more redds were subsequently constructed and monitored after this date, with later completed redds monitored after 29th June, and 10th July.

For Silverstream on the 19th June, two fresh completed redds were located at the time of the first visit, and there were monitored after that date. Two more redds were under construction at the time, and water quality monitoring of these two redds commenced after the trout had left these redds. A late fresh redd was monitored in early July.

6.1.3 Redd superimposition – monitoring rationale

After the insertion of the probes in the trout redds, one of the redds at Silverstream (replicate 5) and the Avon River (replicates 1, 2, and 4), were partially re-excavated by later spawning fish. At the time, it was decided to continue monitoring the superimposed redds given that it was uncertain that if we abandon these redds whether we would find sufficient single redds for superimposition. It was also possible that the study may yield information on how the hydraulics of superimposed redds differs from single redds.

Replicate 1 on the Avon River appeared to have been superimposed by at least two more redd excavations, upstream and to the side of the monitored redd. Redd superimposition is not an uncommon phenomenon in spawning reaches. Photographs of all monitored redds are provided in the Appendix (App. I).

6.2 Monitored water quality parameters

6.2.1 Probe placement

From May 2009 onwards, known trout spawning reaches along the three rivers were surveyed from the banks for spawning activity. When spawning trout were no longer observed patrolling the redd, this redd was assumed to be completed, and fresh. That is, for the purpose of the study and analysis, the age of the redd was assigned as zero days.

For each freshly excavated redd, three IGDO probes were inserted equidistantly along the mid-line of the redd, between the excavation pit and the end of the tail spill. These probe locations are referred to in the results as A, B, and C, in an upstream to downstream order. In addition a steel VHG tube was inserted at the highest point on the redd (Fig. 7). Prepared in this fashion, a total of 5 redds were monitored in each of the three study rivers. A full description of the inserted probes, and their placement, are described below.

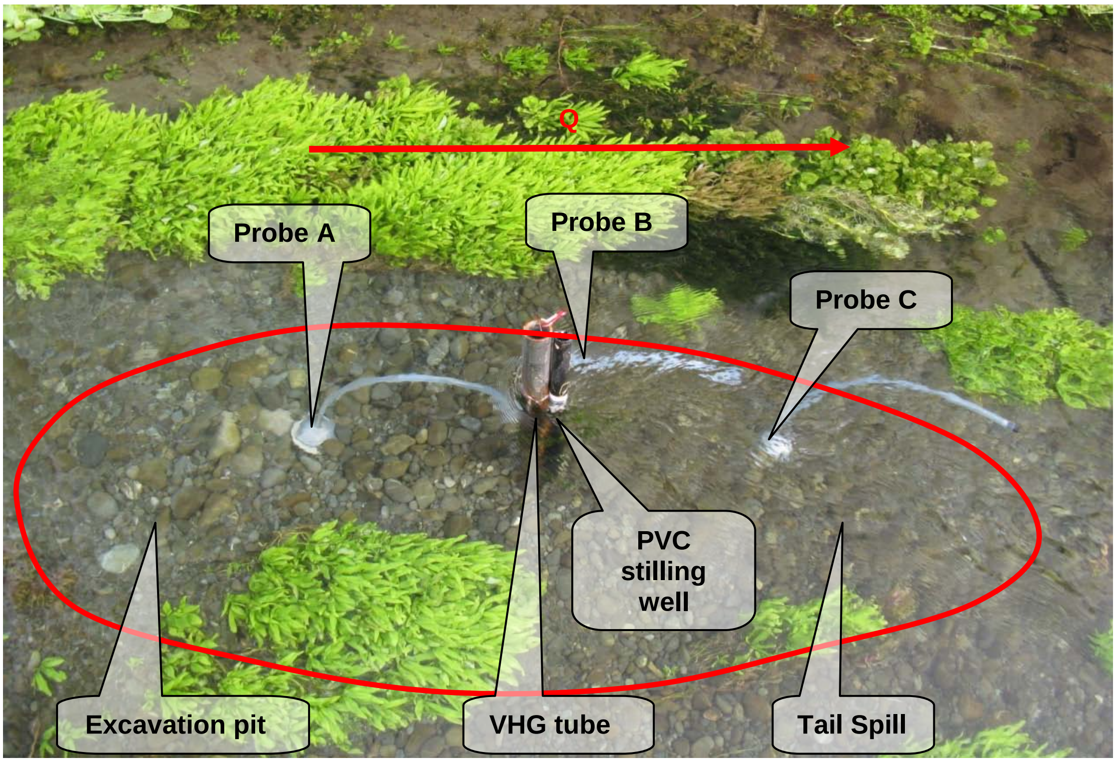


Figure 7. An excavated brown trout redd at Silverstream, showing the placement of the three IGDO probes, the VHG tube, and stilling well. The red line indicates the approximate size of the redd, and the redd arrow the current direction.

6.2.2 Intragravel Dissolved Oxygen (IGDO)

Our monitoring of IGDO was based on the insertion of aluminium water sample probes inserted *in-situ* into each monitored redd (Fig. 8a). Probes were inserted along the redd mid-line at 25%, 50%, and 75% of the redd's length, with the redd length being the distance between the deepest part of the excavation pit and the end of the tail spill. Water samples could then be periodically drawn from each probe, as explained below.

The design of the probes were adapted from those deployed in a North American study on large Chinook salmon (BioAnalysts 2003). Modifications from the North American probe included a reduction in probe length from 30 cm to 16 cm, to match the shallower depth at which brown trout deposit their eggs. A study of dewatered brown trout redds in the Selwyn River (Hardy 1963) showed that the median depth between the redd surface and the eggs was 16 cm. Intragravel water entered the probe through 8 small holes (3 mm dia.) drilled in the distal end. The narrow diameter of the probe, and its short length, meant that they could be pushed into the redds by hand, using the welded flange plate for purchase. Once fully inserted, the welded flange plate sat close to the substrate and stabilised the probe's vertical orientation. However, field trials demonstrated that water drag during high flows tended to dislodge the probes, therefore hemispherical lead weights were added to each probe to keep them in place.

Another modification was the elimination of the underwater bung and copper tubing assembly used in the North American study, to a simpler arrangement with an attached PVC tube with a terminal bung (Fig. 8b). The terminal bung is necessary to prevent contamination of the intragravel water samples from surface water. The arch and flex in the PVC tube was orientated downstream, which facilitated the shedding of flotsam (e.g. floating weed fragments) from the apparatus.

Figure 8a. An aluminium IGDO probe used for this study, but without the stabilising weight. The probes were inserted to a depth of 16 cm in each redd. A syringe and catheter was used to obtain water samples from each probe.

Figure 8b. An IGDO probe in place in Silverstream. Hemi-spherical lead weights were added to weight the probes as a precaution against the probes washing out of the redds during freshes. The terminal bung can be seen which prevents surface water contaminating the intragravel water.

Water samples, with of volume of approximately 20-30 cm^3, were slowly drawn from the IGDO tube and injected into a narrow plastic vial using a syringe assembly (Fig. 5a). DO readings were obtained immediately, with the narrow vial just sufficiently wide to accommodate the dissolved oxygen sensor. The water was slowly stirred with the sensor as the IGDO reading equilibrates to the minimum level which was considered to be the low IGDO level. Field trials had shown that it takes about a minute before the sensor to equilibrate from the surface DO to the sample's DO level, after which the sample reading is influenced by the diffusion of air (Fig. 9).

Samples were obtained from the most downstream probe first (Probe C), then the upstream probes, Probe B, then Probe A, with the sensor allowed to equilibrate to saturated surface water between measurements. The sampling process was then repeated, so that two replicate IGDO readings were obtained from each of the three probes.

After the redd IGDO monitoring was completed, background IGDO levels were obtained from the undisturbed substrate adjacent to selected redds in November. For Boggy Ck, and the Avon River the median value were 1.89 mg/L (next to Replicate 1), and 2.67 mg/L (next to replicate 5). Unfortunately background IGDO levels for Silverstream were not available as the probe was dislodged from the substrate.

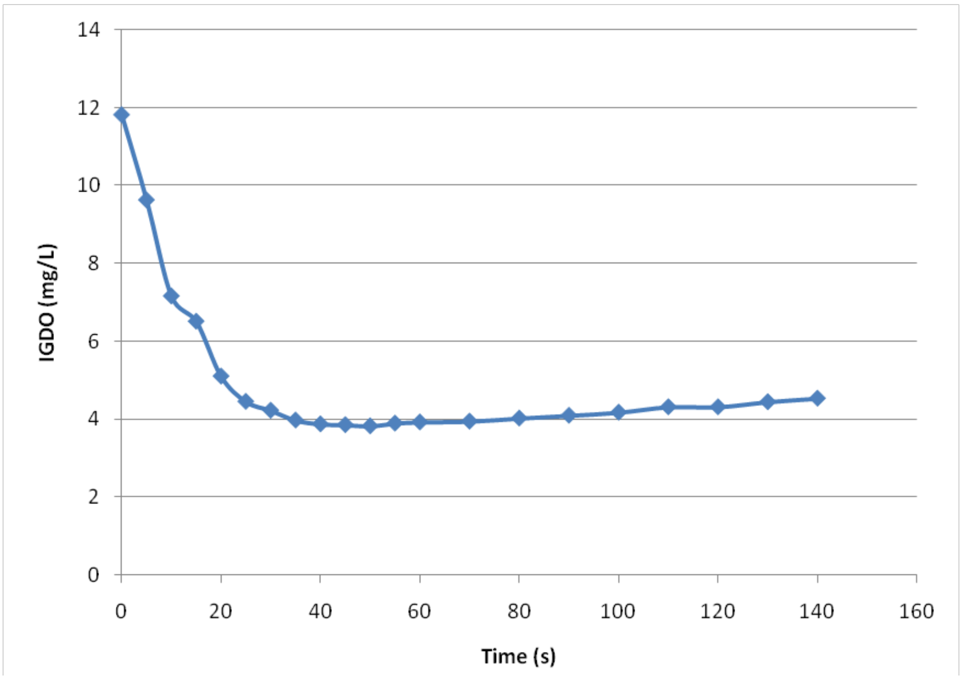


Figure 9. A measured response curve from the DO meter (equilibrated to surface water) when placed in a sample of water drawn from an Avon River redd (AEL raw data, winter 2008).

6.2.3 Vertical Hydraulic Gradient

The vertical hydraulic gradient (VHG) was measured using a steel standpipe (50 mm internal diameter) inserted into the centre of each monitored redd. The centre of each redd was arbitrarily determined as the location where the redd gravels have the highest elevation. The standpipe was hammered into the substrate (using a fitted steel drift) to the selected insertion depth (16 cm) which was determined from an etching mark onto the side of the standpipe.

The standpipe was left for a period of 24 hrs for levels to equilibrate, after which the vertical head in the standpipe (ΔH) was directly measured. The standpipe tube was open-ended and the walls were not perforated, so water would freely enter the open end of the tube. A modification was the fixing (with plastic ties) of a plastic 'stilling well' tube along the length of the VHG tube. The stilling well was a short tube which filled with surface water, and therefore provided an accurate indication of the surface water level, without the bow wave effect that forms around the VHG tube (Baxter *et al.* 2003)(Fig. 10). A wooden box ruler, with the distal end coated in flour, was used to provide an accurate indication of water height inside the pipes. However, even then, some surging was apparent in some habitats; and several replicate measurements were required to obtain an accurate estimate.

VHG is defined thus:

$VHG = \frac{\Delta H}{\Delta L}$ where:

ΔH = head difference between the outside and inside of the standpipe.
ΔL = insertion depth of the standpipe into the stream bed (i.e. 160 mm)

Because we were interested in comparing the relative differences in the VHG, and the insertion depth was the same on all redds, we undertook the analysis of the results simply on the ΔH component, the difference in water level (in mm) between the surface water elevation, and the elevation or depression caused by the upwelling or downwelling current.

Figure 10. A deeply-placed redd at Silverstream, where the ΔH component could be directly observed. The parameter ΔH is the difference in water level between the upwelling in the steel standpipe and the water surface in the black PVC stilling well.

6.2.4 Intra-gravel flow rate

Rather than use a Tehune tube (Terhune 1958), we opted to attempt to detect the horizontal water flow through the redd by directly measuring a non-toxic marker and timing the period before the marker and its detection with a down-current sensor. This was a similar approach taken in a recent North American study (BioAnalysts 2003), although the Americans used an array of four 'Mini Sonde' sensors.

Our technique was similar to that used in the BioAnalysts study, in that a saline solution is injected into the upstream IGDO probe, with a Schott logging conductivity meter used to monitor detection further downstream. The density of the saline solution was adjusted with a small amount of isopropyl alcohol, so that it was the same density of freshwater at around 12 °C.

In the field, the conductivity sensor was placed down the steel standpipe (Fig. 11), or a specialised stainless steel detection probe. The horizontal intragravel flow was then calculated by simply dividing the distance between the IGDO probe and the detection probe by the time taken for the conductivity probe to log a rise in conductivity. The logging interval was set at 5 minutes, and the experiment ran for about 12 hours. We conducted 13 tracer experiments within redds from the three study reaches, and with replicates in a number of redds.

Figure 11. Use of the intragravel flow-monitoring technique. The logging conductivity meter is wired onto a waratah, with the probe inserted in the standpipe. The coloured saline solution is injected into the upstream IGDO probe.

6.2.5 Redd substrate composition

At about the time when the eggs in each redd were considered to be developed, two substrate samples were obtained from the centre of the tailspill. These were obtained by manually pushing a tin can into the centre of the redd as far as possible (Fig. 12). The median depth of insertion was 105 mm, which was calculated by subtracting the distance the base of the can protrudes above the bed from the height of the can.

A hole was drilled in the base of the can to prevent the possible driving of clean surface water through the intragravel space as the can was pushed into the redd gravels. A close-fitting plastic lid was then placed on the top end of the can, while the stones adjacent to the can's sides were carefully removed to allow a second sealing lid to be placed over the open end of the can. The can, and the enclosed substrate and intragravel water was then carefully removed from the redd, and placed in a plastic bag for transport to the laboratory.

Substrate particle size analysis was undertaken at the University of Canterbury's Geology Department laboratory. The samples were chemically digested to remove organic material, and then mechanically sieved for 15 minutes in six nested sieves using the following Udden-Wentworth size fractions:

(1) -5 phi (31.5 mm)
(2) -4 phi (16mm)
(3) -3 phi (8mm)
(4) -2 phi (4mm)
(5) -1 phi (2mm)
(6) 0 phi (1mm)

Fine particulates less than 1mm in size were analysed with a technical instrument called a 'laser sizer', capable of resolving the particle size frequency distribution from several microns to 1000 microns.

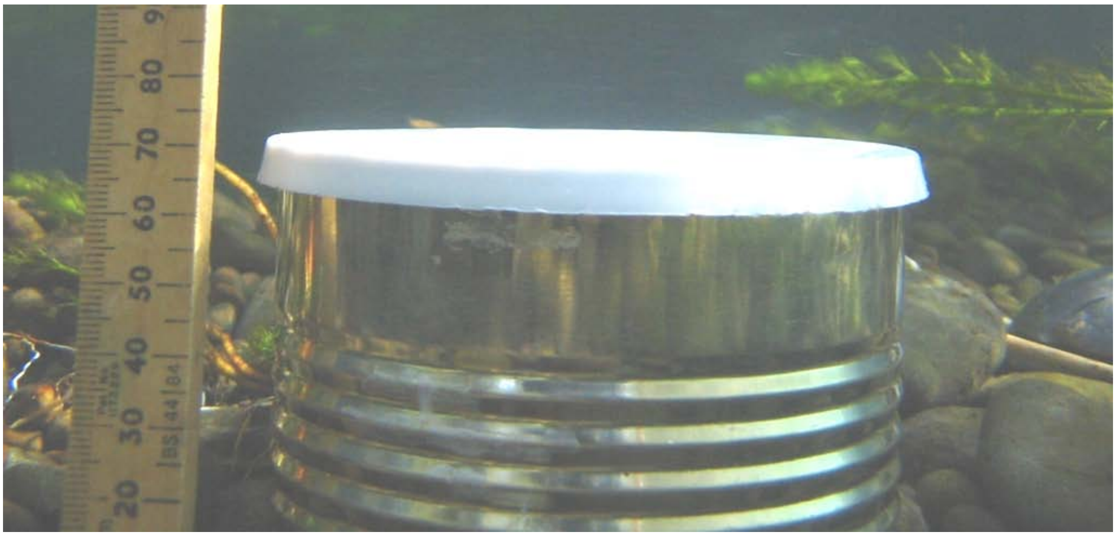


Figure 12. A cylindrical tin-plate can was manually pushed into the substrate, to a measured insertion depth of approximately 15 cm. A second plastic cap was placed on bottom end to entrap the substrate sample and interstitial water.

6.2.6 Timing of redd substrate sampling

We wished to obtain substrate data from mature redds, between the time the eggs hatched, but before the alevins emerged from the gravels. We used accumulated thermal units to estimate hatching dates for each of the monitored redds, using intragravel water temperature data for each of the three study reaches to obtain an estimate of development temperature (Table 1).

Using this information we could estimate the hatching date, and the date in which redd substrate should be sampled (Table 2). The redd Silverstream, reps 2, and 5 was sampled a day earlier than the hatch date, and replicate 4 were 2 weeks later. The Avon River redd, replicate 5 were also sampled later.

Table 1. Estimated egg development for the three studied reaches.

Reach	Winter intragravel diurnal temperature range	Mean temperature (C^o)	Development time (days)$_1$
Boggy Creek	6.2-8.5	7.4	51
Avon River	10.7-11.9	11.3	33
Silverstream	9.5-11.0	10.3	37

1 = Based on thermal unit accumulation of 378 Thermal Units (Maret *et al.* 1993).

6.2.7 Water quality assays

Water quality samples (Ammonia Nitrogen, Biological Oxygen Demand, and Nitrate and Nitrite Nitrogen) were obtained from each redd by drawing water samples from the IGDO probes. Surface water counterparts were also obtained. Samples were analysed by the laboratory at Environment Canterbury, but IGDO samples were pooled for each river to obtain a sufficiently large volume to make analysis possible.

Table 2. Estimated hatching dates, substrate core-sample dates, and redd excavation dates.

River	Replicate Number	Known or probable Redd completion date	Date probes inserted	Egg Hatching date	Substrate sample date	Redd excavation date
Boggy	1	16/06/2009	18/06/2009	6/08/2009	10/08/2009	17/8/09
Boggy	2	16/06/2009	18/06/2009	6/08/2009	8/08/2009	17/8/09
Boggy	3	16/06/2009	18/06/2009	6/08/2009	8/08/2009	17/8/09
Boggy	4	16/06/2009	18/06/2009	6/08/2009	10/08/2009	17/8/09
Boggy	5	16/06/2009	18/06/2009	6/08/2009	10/08/2009	17/8/09
Silverstream	1	17/06/2009	19/06/2009	24/07/2009	28/07/2009	n/a
Silverstream	2	22/06/2009	24/06/2009	29/07/2009	28/07/2009	31/7/09
Silverstream	3	17/06/2009	19/06/2009	24/07/2009	28/07/2009	n/a
Silverstream	4	8/07/2009	10/07/2009	14/08/2009	31/08/2009	n/a
Silverstream	5	22/06/2009	24/06/2009	29/07/2009	28/07/2009	n/a
Avon	1	22/06/2009	24/06/2009	25/07/2009	29/07/2009	n/a
Avon	2	22/06/2009	24/06/2009	25/07/2009	29/07/2009	31/7/09
Avon	3	22/06/2009	24/06/2009	25/07/2009	29/07/2009	n/a
Avon	4	27/06/2009	29/06/2009	30/07/2009	30/07/2009	31/7/09
Avon	5	9/07/2009	10/07/2009	11/08/2009	30/08/2009	n/a

6.3 Redd Excavations

After the predicted hatching date, a number of redds were excavated with a shovel and downstream stop-net (Table 2). For Silverstream and Avon spawning reaches, excavation was undertaken within 6 days of their predicted hatching date. For Boggy Creek, excavation was about 11 days after their predicted hatching date.

6.4 Statistical Analysis

Statistical analyses were conducted using the `nlme` library in R (v. 2.11.1) for mixed effects models and Akaike Information Criterion (AIC) comparisons. The response variable IGDO was first tested with a mixed-effects model using a maximum likelihood method. River and Probe (i.e. the relative position of the IGDO probe in a redd) were specified as fixed effects whilst Redd and Age (i.e. approximate age of redd in days) were specified as a random effect (Pinheiro & Bates 2000; Zuur et al. 2009). The model took the following form:

$$IGDO_{ij} = \alpha + \beta_1 \times River_{ij} + \beta_2 \times Probe_{ij} + \beta_3 \times River_{ij} \times Probe_{ij} + a_i + \varepsilon_{ij}$$

$$a_i \sim N(0, d^2_{Redd,Age})$$

A second mixed-effects model tested the importance of various parameters as predictors of IGDO. In this model Probe was specified as a fixed factor with VHG, % Fines (sediment <2mm) and Age as covariates. Because of the nested nature of the data and the violation of independence, River and Redd were specified as random factors. Residual plots of the initial model indicated that there was heterogeneity of variance. This was attributed to the high variability of the IGDO values at the Avon river site and between the different probes across all sites. To account for this effect the `varIdent` function in the `nlme` library was used allowing for different residual variances depending on River and Probe (Zuur *et al.* 2009). The initial model was fitted to a subset of the data where VHG recordings were collected, and when this parameter was rejected as a non-significant covariate, the same model excluding VHG was fitted to the entire data set.

The model was calibrated by removing each term and comparing the AIC values, and had the following form:

$$IGDO_{ij} = \alpha + \beta_1 \times VHG_{ij} + \beta_2 \times Fines_{ij} + \beta_3 \times Age_{ij} + \beta_4 \times Probe_{ij} + a_i + \varepsilon_{ij}$$
$$a_i \sim N(0, d^2_{River,Redd})$$
$$\varepsilon_{ij} \sim N(0, \sigma^2_{Probe,River})$$

Linear regression was used to analyse Age as a dependent variable for IGDO at each site, whereas Pearson Product Moment Coefficients were used in correlation analysis of IGDO and its relationship to VHG and % Fines. One-way ANOVA, and its non-parametric equivalent (i.e. Kruskal-Wallis), were used to compare average values for these same physical parameter across sites.

7 Results

7.1 Intragravel Dissolved Oxygen (IGDO)

7.1.1 Spatial effects between rivers

There were significant differences in mean IGDO values between all three lowland rivers surveyed (indicated in Fig. 13 by the median-based Boxplots). Boggy Creek had the lowest mean value (4.409 mg/L ± 0.192 SEM), followed by that of the Avon (6.488 mg/L ± 0.389 SEM), whereas the reference site Silverstream had the highest mean value (8.285 mg/L ± 0.224 SEM). The results from the mixed effects model analysing the effects of River and Probe on IGDO indicated that the difference was highly significant ($F_{2,169} = 52.4$, $P < 0.001$), with a post-hoc inspection of mean standard errors indicating that all three rivers had significantly different mean IGDO values. This result accounted for the effects of repeated measures on the same redds through time (for a further explanation of this process see Methods Section under the sub-header Statistical Analysis).

Similarly, the same model indicated that there were significant differences ($F_{2,169} = 21.9$, $P < 0.001$) in the mean IGDO values between the three probes used per redd (Fig. 14). Probe C had the lowest mean value (5.075 mg/L ± 0.288 SEM), followed by that of the Probe A (6.377 mg/L ± 0.336 SEM), whereas Probe B had the highest (7.044 mg/L ± 0.315 SEM). The post-hoc inspection of mean standard errors indicates that Probe A and B had significantly higher IGDO values than Probe C, but that there was no significant difference between the two more upstream probes.

The mixed model testing the effect of River and Probe on IGDO also incorporated an interaction term between these two factors. This interaction term indicated that there was a significant difference ($F_{2,169} = 6.39$, $P < 0.001$) in the mean IGDO values between the three probes used per redd and the river surveyed. Comparing mean standard errors indicated that this result was the consequence of the low values recorded from all three probes used at Boggy Creek in addition to the low value for Probe C in the Avon River (Fig. 15).

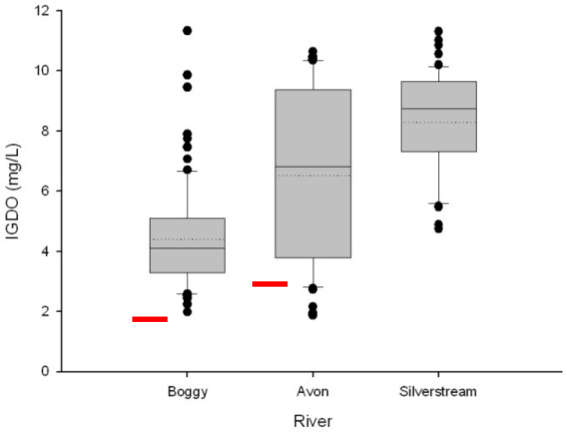


Figure 13. Boxplots indicating the median (and mean as dotted line) intragravel dissolved oxygen (IGDO) values (mg/L) for brown trout redds in three lowland Canterbury rivers. The red bars equate to IGDO background levels for Boggy Creek and Avon River local substrate.

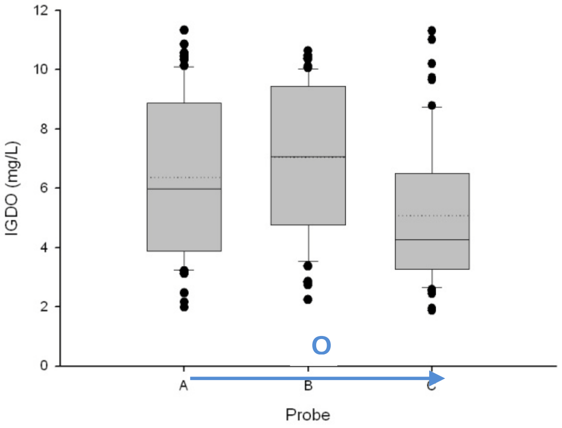
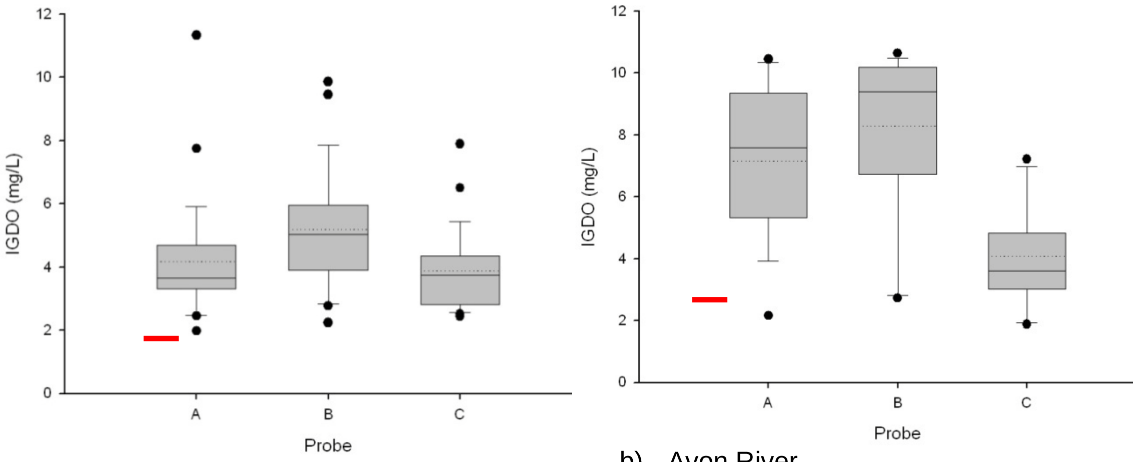
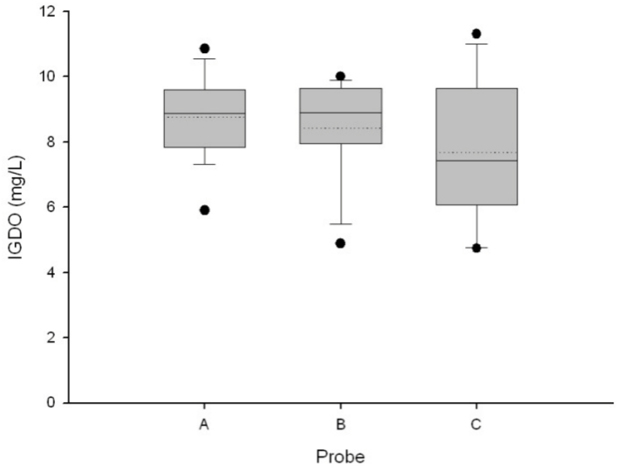


Figure 14. Boxplots indicating the median (and mean as dotted line) intragravel dissolved oxygen (IGDO) values (mg/L) for the three probes inserted into brown trout redds in three lowland Canterbury rivers. River flow direction is indicated.



a) Boggy Creek

b) Avon River



c) Silverstream

Figure 15. Boxplot showing median (and mean as dotted line) intragravel dissolved oxygen (IGDO) values (mg/L) for brown trout redds in three lowland Canterbury rivers. The red bars equate to the IGDO levels in the substrate for Boggy Creek and Avon River. The background level for Silverstream was not measured.

7.1.2 Redd superimposition

At two of the sites sampled (Avon and Silverstream), some of the redds were superimposed (i.e. brown trout spawned on or in a close vicinity to monitored redds) during the survey. At the Avon river, Redds 2 and 4 had single superimposition whereas Redd 1 had a double superimposition (i.e. two additional redds built directly upstream). At Silverstream, Redd 5 was partially superimposed, with a single redd excavated upstream. A one-way ANOVA for redds at these sites using a mixed-effects model was used to investigate the nature of this potentially confounding factor, and post-hoc differences were identified using standard errors of the mean. However, while there were significant differences between IGDO values for redds at the Avon River ($F_{4,41}$ = 3.78, $P < 0.05$), it was equivocal as to whether these differences were the result of superimposition, with no consistent effect apparent (Fig. 16). Similarly, although Redd 5 at Silverstream had a higher mean IGDO value compared to two of the other four redds ($F_{4,44}$ = 8.20, $P < 0.001$), there was still one redd (Redd 4) greater (Fig. 17). Similarly, regression analysis of individual redds from each site over time (results not shown) indicated that there was no consistent effect of superimposition on IGDO levels.

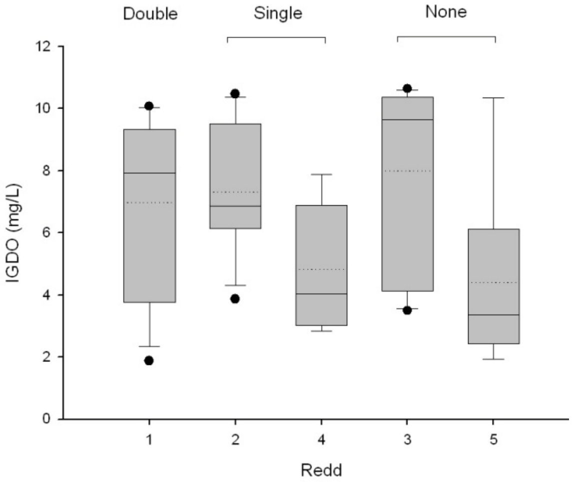


Figure 16. Boxplots indicating the median (and mean as dotted line) intragravel dissolved oxygen (IGDO) values (mg/L) for brown trout redds at the Avon River site. (Double = two redds superimposed, single = one red superimposed, none = no superimposition on monitored redd.)

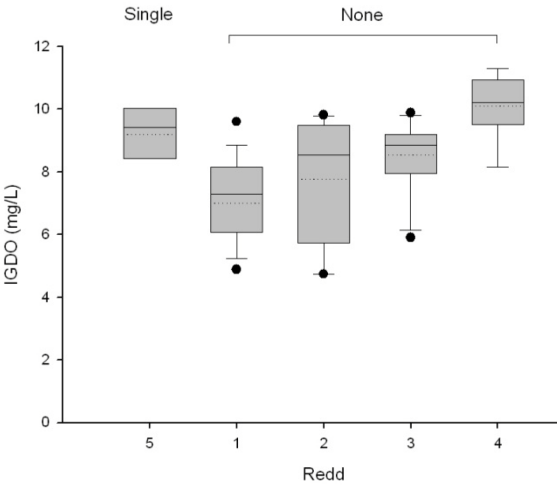


Figure 17. Boxplots indicating the median (and mean as dotted line) intragravel dissolved oxygen (IGDO) values (mg/L) for brown trout redds at the reference site Silverstream. (Single = one redd superimposed, none = no superimposition on monitored redd.)

7.1.3 Temporal effects

Mean IGDO within brown trout redds from the three lowland streams surveyed showed a general trend of decreasing levels over time. The results from Silverstream and Boggy Creek both clearly demonstrated a linear decline in mean IGDO with increasing age expressed as development days (Fig. 18). Regression analysis showed that both relationships were significant, with Silverstream ($F_{1,\ 8}$ = 9.19, $P < 0.05$, R^2 = 0.568) having a larger y-axis intercept (9.48 mg/L) and slope (-0.0525) than that of Boggy Creek ($F_{1,\ 5}$ = 8.92, $P < 0.05$, R^2 = 0.641), which had a lower intercept (5.33 mg/L) and slope (-0.0343).

In contrast to these two relationships, mean IGDO in the Avon River showed no significant difference over time (Fig. 18; $F_{1,\ 7}$ = 1.69, P = 0.235, R^2 = 0.194). However, a closer inspection of this result indicated a divergent pattern between three of the 5 monitored redds (Redds 1, 2, 3) and the remaining two (Redds 4, 5) at this site (Fig. 19). Probe C had a strong negative influence on mean IGDO (Fig. 20) which was uniformly low across all redds irrespective of time sampled ($F_{1,7}$ = 0.456, P = 0.521, R^2 = 0.0611).

Accordingly, regression analysis was performed on the raw IGDO values from Avon redds 1, 2, and 3 excluding the records from Probe C. This revealed that there was no significant effect of time on IGDO values for these three replicates ($F_{1,\,2}$ = 3.578, P = 0.199, R^2 = 0.641). In contrast, removing the Probe C records and testing the linear regression of raw IGDO values from Redds 4 and 5 did demonstrate a significant decline over time ($F_{1,\,4}$ = 17.4, P < 0.05, R^2 = 0.813). Using the same model structure employed for the analysis of redd superimposition, the one-way ANOVA showed that there was a significant difference in mean IGDO values between these two groups of replicates if Probe C was excluded (Fig. 21; $F_{1,41}$ = 14.6, P > 0.001).

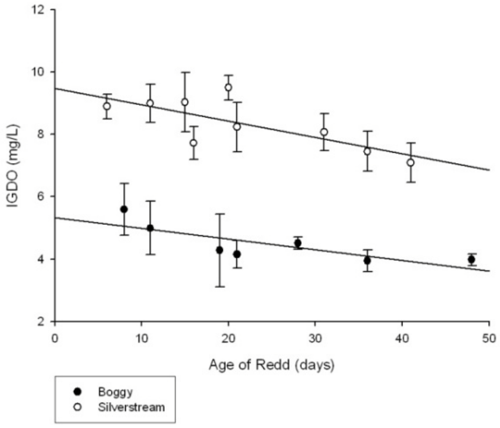


Figure 18. Mean intragravel dissolved oxygen values (mg/L ± 1 SEM) against age since construction of brown trout redds in two lowland Canterbury rivers (Boggy Creek and Silverstream). The substrate IGDO level for Boggy Creek was 1.89 mg/L in November 2009.

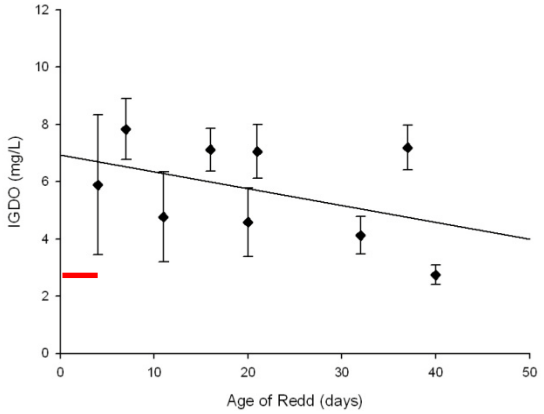


Figure 19. Mean intragravel dissolved oxygen values (mg/L ± 1 SEM) against age (days) since construction of brown trout redds in the Avon River. The red bar represents the substrate IGDO level measured in November 2009.

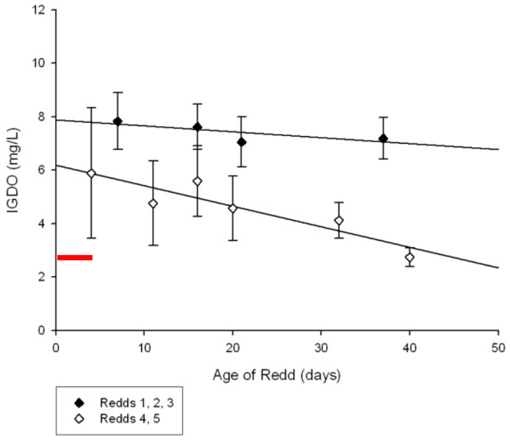


Figure 20. Mean intragravel dissolved oxygen values (mg/L ± 1 SEM) against age (days) since construction of brown trout redds in the Avon River demonstrating the differences among replicates. The red bar represents the substrate IGDO level measured in November 2009.

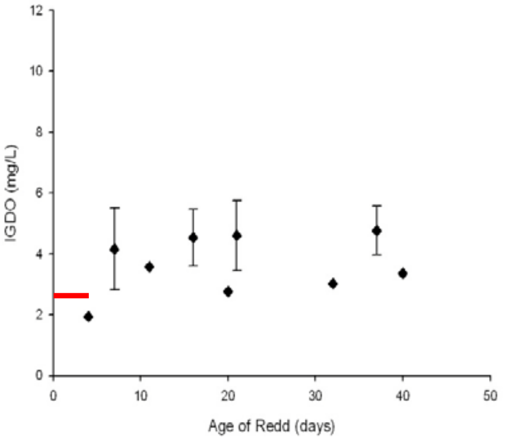


Figure 21. Mean intragravel dissolved oxygen values (mg/L ± 1 SEM) from Probe C against age (days) since construction of brown trout redds in the Avon River. The red bar represents the substrate IGDO level measured in November 2009.

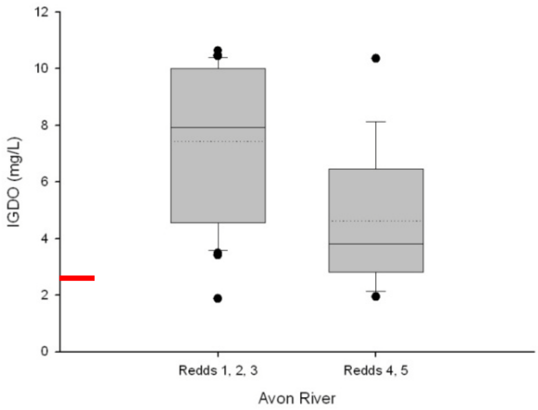


Figure 22. Boxpot showing median (and mean as dotted line) intragravel dissolved oxygen values (mg/L ± 1 SEM) of Brown Trout redds in the Avon River demonstrating the differences among replicates.

7.2 Physical factors

7.2.1 Vertical hydraulic gradient (expressed as ΔH)

The ΔH values recorded from the three rivers surveyed showed no significant differences in the mean value between sites (Fig. 23) with a Kruskal-Wallis ANOVA on ranks demonstrating this statistically ($H_{2\ d.f.} = 4.50$, $P = 0.105$). Most redds had a weakly positive (upwelling) ΔH component with an overall median of 8 mm. However, some redds had negative (downwelling) currents. For example, the superimposed Redd 5 at Silverstream had strongly negative ΔH values which increased the variation about the mean at this site and reduced the normality of the data. Removing this outlier showed that the differences between rivers were more non-significant than the above test ($F_{2,\ 46} = 1.23$, $P < 0.301$). Correlation analysis of the relationship between ΔH and IGDO indicated a significant negative association ($r = -0.304$, $P < 0.001$, $n = 148$), although this result was strongly influenced by Redd 5 at Silverstream. Removal of this outlier reduced the strength of the correlation, but did not materially alter the significance of the association ($r = -0.183$, $P < 0.05$, $n = 142$)(Fig. 24).

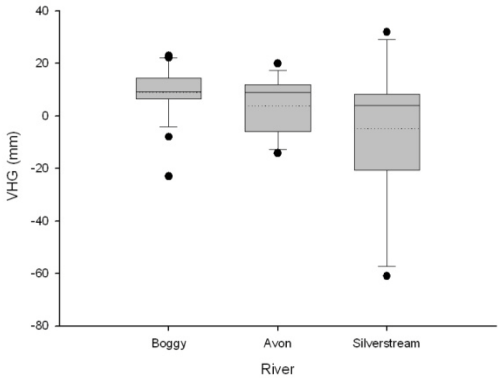


Figure 23. Boxplot showing median (and mean as dotted line) vertical hydraulic gradient (VHG) values (mm) adjacent to Brown Trout redds in three lowland Canterbury rivers.

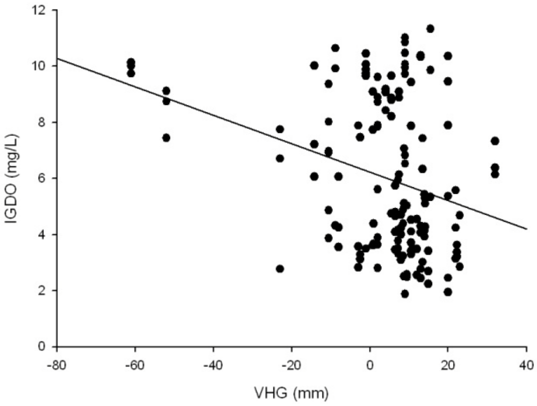


Figure 24. Mean intragravel dissolved oxygen values (mg/L) from Brown Trout redds against vertical hydraulic gradient (VHG) values (mm) in three lowland Canterbury rivers.

7.2.2 Redd substrate composition

The substrate fraction less than 2 mm considered to be the most important in influencing trout redd permeability (Maret *et al.* 1993). The variation in the 2mm fine fraction across the monitored sites is illustrated in Fig. 25, with the superimposed redds in the Avon indicated with red arrows. It would appear that the superimposed redds have lower amounts of fine sediment, especially the fraction below 1 mm.

There was marked differences in IGDO between rivers, although it was not a significant difference at the 95% confidence level (One-way ANOVA, $F_{2,\ 12} = 2.90$, $P = 0.094$). However the power of this test ($B = 0.310$) meant that the likelihood of incurring a Type II error (not detecting a difference when one actually exists) was increased (assuming a desired power of 0.800 when alpha = 0.05). It was apparent that Boggy Creek and the Avon River had higher levels of fine sediment than that of the reference site Silverstream (Figs. 25, 26).

When analysing the relationship between IGDO and fine sediment it became clear that inclusion of the redds on the Avon River increased variation significantly. This was possibly because 3 of the 5 redds were superimposed to various degrees, and this affected the relationship between IGDO and % fines. Therefore for this analysis, we opted to restrict the dataset to the data from the 10 Boggy Creek and Silverstream redds.

For Silverstream and Boggy Creek redds, correlation analysis of the relationship between % fines (< 2 mm) and the IGDO reading (obtained at the time of substrate sampling) indicated a significant negative association between these two variables (Fig. 26; $r = -0.726$, $n = 10$, $p < 0.05$). The influence of very fine sediment (< 1 mm) to IGDO was even stronger in terms of influence on IGDO, as indicated by the steeper slope of the least-squares best fit line (Fig. 28). The correlation of also significant, and approximately the same as for the coarse size fraction (i.e. r=-0.71, n=10, p < 0.05).

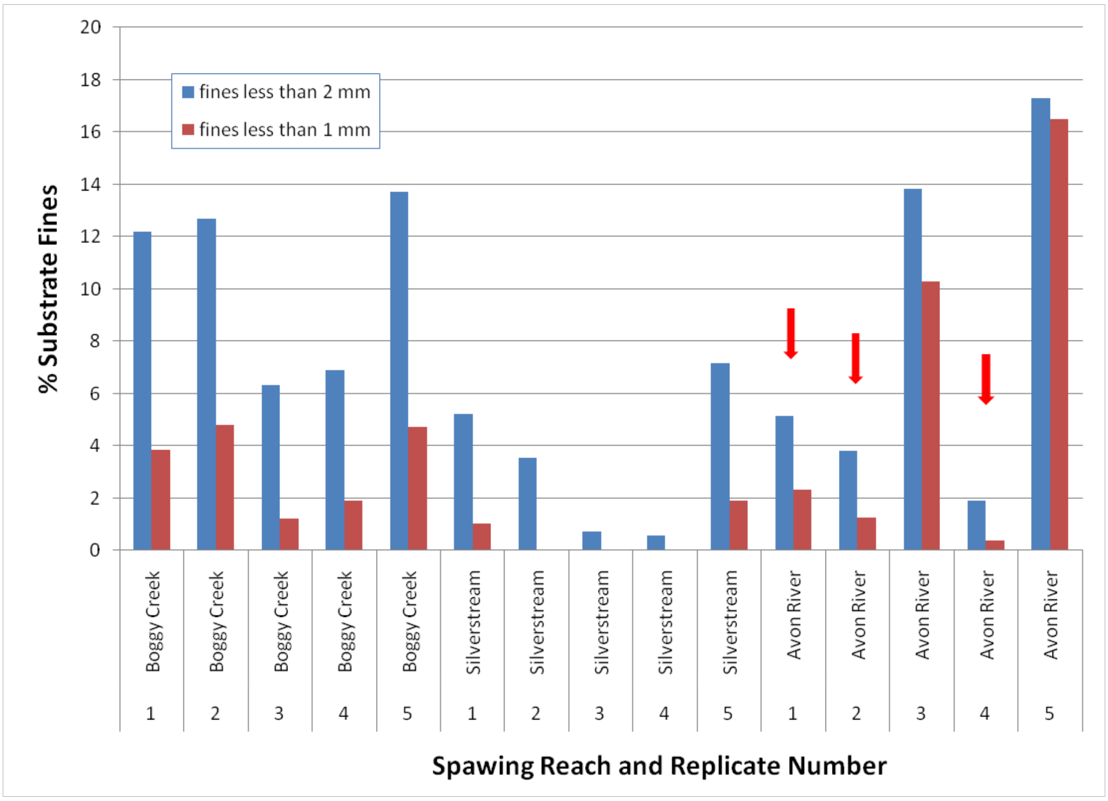


Figure 25. Distribution of substrate fines across the 15 monitored redds towards the end of the monitoring period. The red arrows indicate the superimposed redds in the Avon River.

With the inclusion of the Avon River redds, the over-all relationships between IGDO and fines were still negative, but not statistically significant at the 95% confidence level. In summary, the addition of the 5 Avon River redds, added variation to the relationship between IGDO and % fines. Significant outliers was Avon River redd (Replicate 3) which had a much higher IGDO than predicted from its high level of fine sediment.

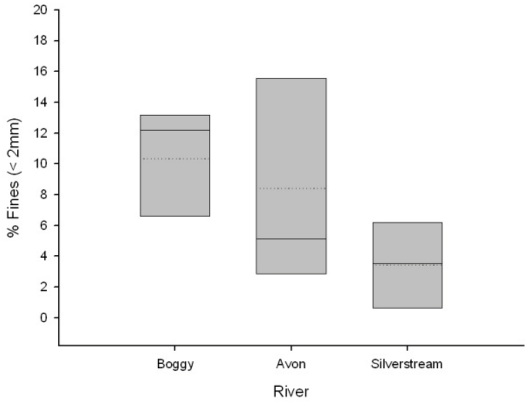


Figure 26. Boxplot showing median levels (and mean as dotted line) of % fine sediment (< 2mm) in three lowland Canterbury rivers.

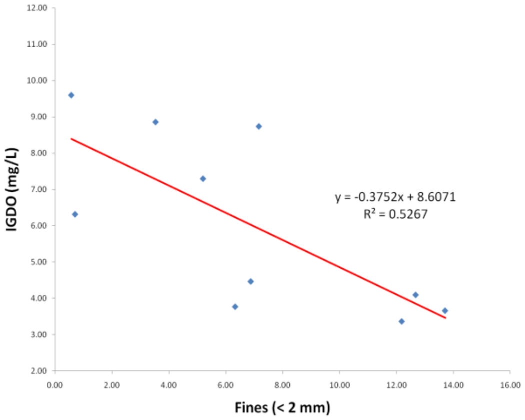


Figure 27. Mean intragravel dissolved oxygen values (mg/L) from brown trout redds against % fine sediment (< 2mm) in two lowland Canterbury rivers. This data excludes that from the Avon River.

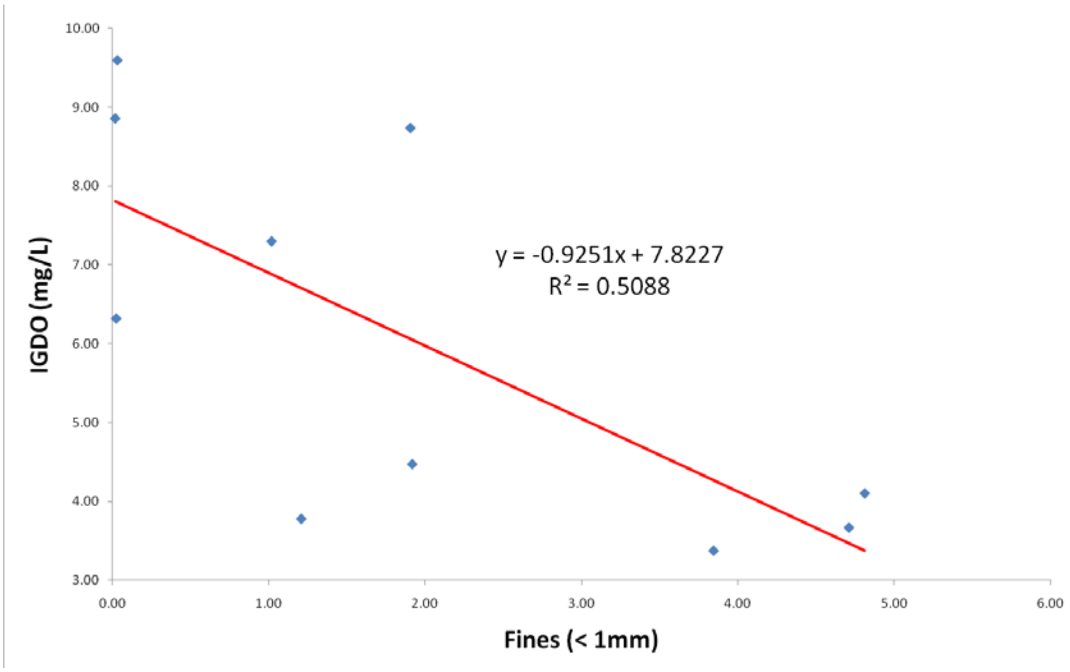


Figure 28. Mean intragravel dissolved oxygen values (mg/L) from brown trout redds against % fine sediment (< 1mm) in two lowland Canterbury rivers. This data excludes the data from the Avon River.

7.2.3 Intra-gravel flow

For 8 of the 13 trials, there did not appear to be a detectable response in terms of a conductivity spike on the logger trace. For those redds, we considered that possibly the salinity plume was upwelling or downwelling beyond the detection probe. However, there were 5 trials which appeared to be produce a distinct downstream response signal on the conductivity logger, these conductivity traces appear in App. II.

For Boggy Creek and Silverstream, replicate trials on the same redd were achieved, with a single estimate on one (non-superimposed) redd on the Avon River. Trials were run on different river stages, and illustrated considerable variation in intra-gravel flow (Fig. 29). Also depicted, for the purpose of comparison, are the median intra- gravel flow rates obtained from several brown trout redds from the Clutha trout catchment (Bickel & Closs 2007).

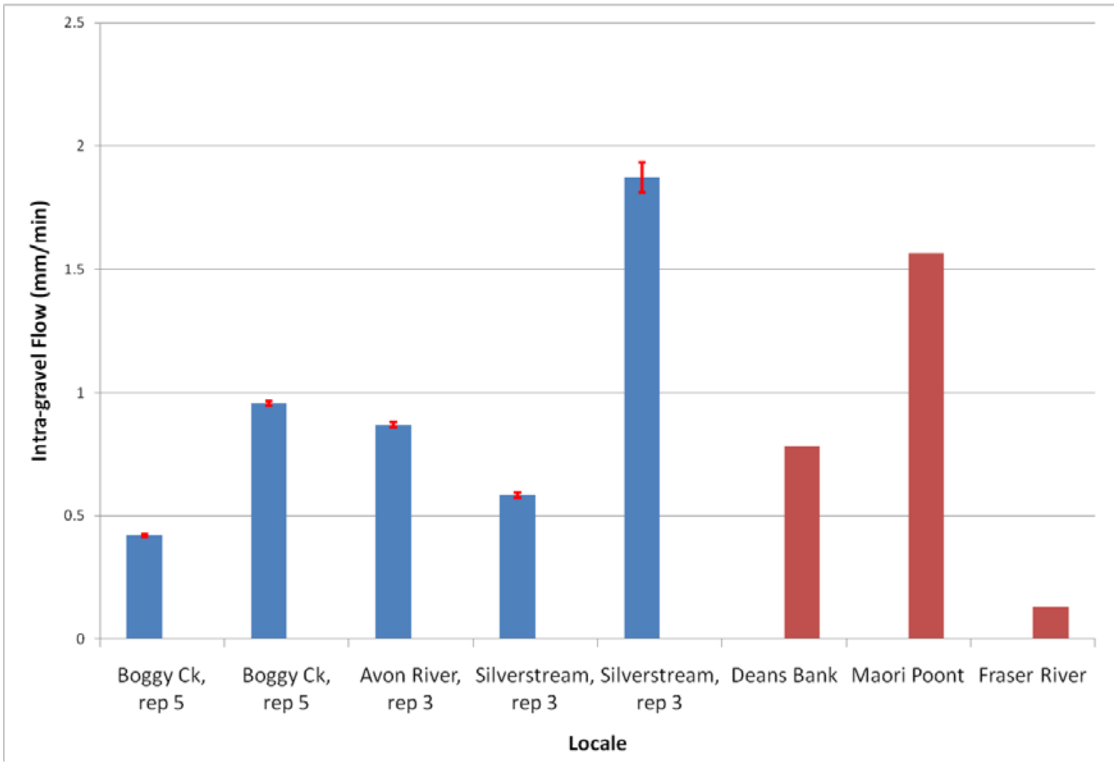


Figure 29. Estimated intra-gravel flow for three redds from this study (blue bars), and median values from an Otago study (red bars)(Bickel & Closs 2007). Error bars are based on an estimated error of 10 minutes in determining the response time. The logging interval was 5 minutes.

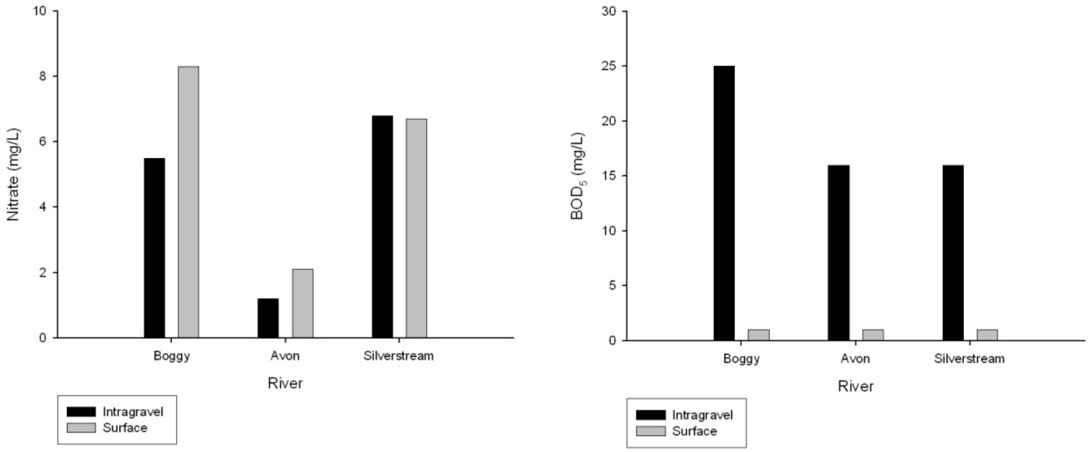
7.2.4 Combined effects of spatial, temporal and physical parameters

A mixed-effects model was used to assess the importance of selected parameters (Probe, Age of redd, VHG, and % Fines) as factors and covariates to predict IGDO values, whilst accounting for the spatial effects of repeated measures and heterogeneity of variance (for a further description of this process see Methods section under the sub-header Statistical Analysis). VHG was rejected as a non-significant covariate at the model calibration stage (Log-likelihood ratio = 0.228, $P = 0.633$), but the remaining variables all had significant effects on IGDO. Both % fines ($F_{1, 13} = 10.5$, $P < 0.01$) and age of redd ($F_{1, 177} = 23.6$, $P < 0.001$) had negative effects on IGDO, whereas Probe C was more likely to have lower values than Probe A and B ($F_{1, 177} = 14.2$, $P < 0.001$).

7.3 Water chemistry

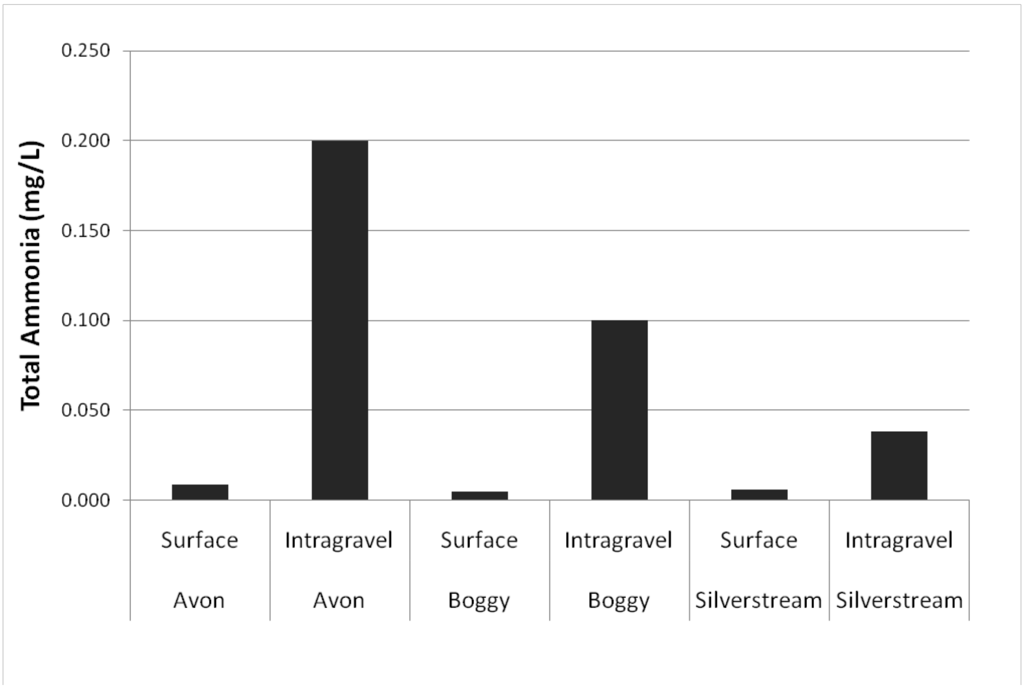
The analysis of pooled samples from water drawn out of the IGDO probes demonstrated some patterns consistent with the trends shown above. However, the process of pooling the samples, necessary to obtain sufficient water for testing, also prevented statistical analysis. Concentrations of Nitrate (mg/L) were high in surface and intragravel water collected from both Boggy Creek and Silverstream, whereas the samples from the Avon River had markedly lower values (Fig. 30a). Both Boggy Creek and the Avon River had lower concentrations of

Nitrate in the intragravel water than that of the surface water at the same site, whereas levels were comparable at Silverstream. BOD_5 values (mg/L) were over an order of magnitude greater in all intra-gravel samples compared to surface water samples, and surface water BOD_5 values were predictably low (Fig. 30b). The intragravel water from Boggy Creek had a higher biological demand for oxygen (BOD_5) than water collected from the same habitat in the Avon and at Silverstream. Levels of interstitial total Ammonia (mg/L) were highest in the Avon River, intermediate in Boggy Creek, and lowest in Silverstream gravels (Fig. 30c). Like BOD_5, Ammonia levels were considerably higher in intragravel water than that recorded at the surface.



a) Nitrate levels

b) Biological Oxygen Demand



c) Total Ammonia Levels

Figure 30. Bar-graphs showing measured water chemistry parameters for brown trout redds in three lowland Canterbury rivers. A) Nitrate, B) BOD_5, C) Ammonia.

7.4 Catch from redd excavations

All of the Boggy Creek redds were excavated about 11 days after their expected hatching date. Only a few dead eggs were found, and no alevins.

At Silverstream, only replicate 2 was excavated, about 2 days after their expected hatching date. A total of 14 eggs were extracted, two were eyed, and appeared viable. Two empty chorions were also indentified. The remaining 12 eggs appeared dead, and having done so at an early stage of development.

On the Avon River, redd replicates 2 and 4 were excavated, 6 days and 1 day after their respective and expected hatching dates. In redd replicate 2, numerous (about 80) viable alevins were recorded (Fig. 31), along with two fungus-infected eggs. For replicate 4, only seven dead eggs were found in the redd.

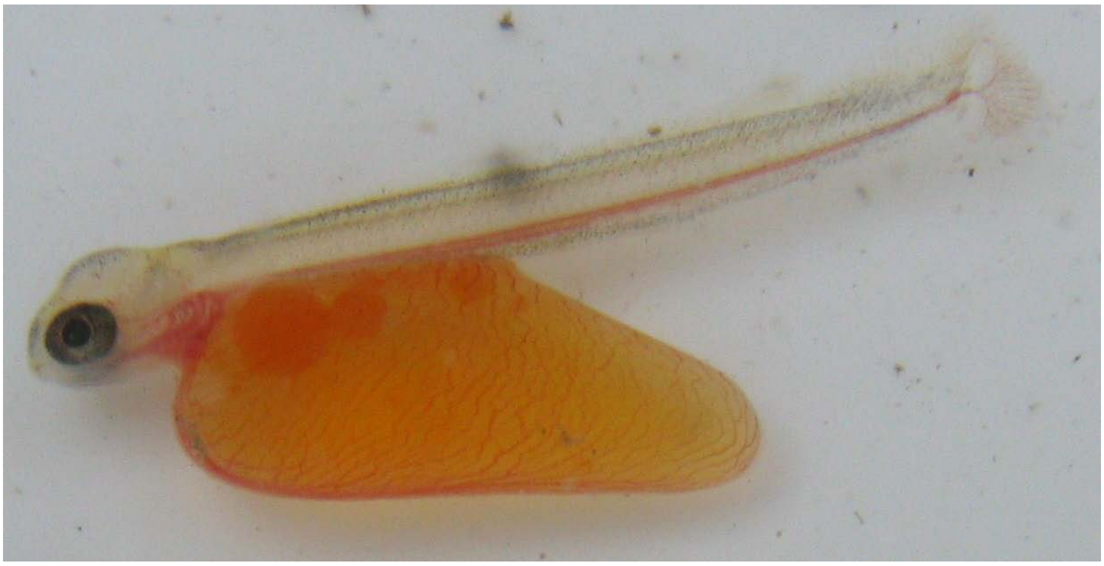


Figure 31. A developing alevin from a brown trout redd in the Avon River (31/7/09).

8 Discussion

8.1 IGDO - Spatial effects between rivers, and the influence of fine sediment

The statistically significant differences in IGDO concentrations across streams indicated that the lower Boggy Creek reach was the most impacted site and the reference study reach on Silverstream as the most pristine. The Avon River was at an intermediate level of degradation. IGDO values in Boggy Creek were considerably lower than the levels recommended in North American studies. For example, based on egg basket experiments in artificial redds that were exposed to agricultural contaminants, egg and alevin survival was negligible when mean IGDO was less than 8.0 mg/L, and less than 70% saturation (Maret *et al.* 1993). This prompted the authors to suggest that a pre-study proposed standard of 6 mg/L, based on their results, proved to be conservative in terms of protecting trout redd quality. Other researchers set a higher minimum of approximately 76% of saturation between 0^{o}C-10^{o}C (i.e. min. of 8.3 mg/L) for protection of a salmonid fishery in the Northern Rockies of the U.S. (Chapman & McLeod 1987). However, various salmonid species may have differing tolerances to depressed IGDO, and what might be intolerable to some salmon species may be quite acceptable for brown trout, which are the dominant species in New Zealand's lowland streams. A level of 8 mg/L is also markedly higher than the USEPA set for the early lifestages of coldwater biota (5 mg/L)(USEPA, 1986), although the USEPA level may not be appropriate for the specific requirements for trout ova.

The link between egg and alevin mortality and IGDO suggests that oxygen limitation in Boggy Creek could be an important stressor limiting the recruitment of trout fry, because its mean IGDO was much less than the overseas guidelines discussed above. However, as also mentioned, there is some uncertainty about the applicability of these standards to those for New Zealand brown trout. Certainly, there is a need to conduct field experiments, and possibly those under controlled laboratory conditions, to determine if the low IGDOs reported here actually manifest into increased trout ova or alevin mortality.

The key factors influencing the flux of water through the egg incubation environment includes surface roughness of the substrate particles, bed topography and permeability. Bed permeability is particularly affected by the presence of fine sediment (i.e. inorganic particles < 2 mm dia.) which can clog interstices and reduce the exchange of surface and groundwater (Brunke & Gonser 1997; Vervier *et al.* 1992). This was verified in the Maret *et. al* (1993) study, which demonstrated statistically significant positive associations between % survival and IGDO, a significant negative association between IGDO and % fines (i.e. % fines < 2 mm), and between % survival and % fines. With these links, and based on their data, Maret *et. al* (1993) suggested that % fines should be less than 15% to maintain an IGDO level of 8 mg/L or greater.

Other researchers have recommended threshold values for finer sediment particulates. Jensen *et al.* (2009) showed in a meta-analysis that egg-fry survival of salmonids dropped rapidly when the percent fines less than 0.85mm were greater than 10%. Sear et al. (2008) considered the effects of sediment over both egg and alevin survival and hatching. He suggested that in addition to sediment infiltration and accumulation within the gravels affecting embryonic survival, sedimentation (or capping) could affect the emergence of the alevins from the interstitial space once hatched, effectively “entombing” then within the redd.

How does the North American sediment analyses compare with our results? Both the Maret *et al.* (1993) study, and our work, depict negative relationships between IGDO and percent fines, although our data indicates a steeper decline in IGDO with increasing quantities of fine sediment. The table below summarises the relationships between IGDO and % fine fractions using our data from all three spawning reaches, and also the Silverstream and Boggy Creek data subset (Table 3).

This greater sensitivity of IGDO decline with sediment is more apparent, and more statistically significant with the data subset from Silverstream and Boggy Creek, where the complications of redd superimposition and urban inputs cloud the relationship. The inclusion of the Avon River data results introduces more variation, to a point that is was not significant at the conventional 95% level. However, similar to the 2mm fraction, our locally-derived data suggests low levels of very fine sediment (i.e. < 1 mm) will cause depleted oxygen levels (see Fig. 28).

Table 3. Comparison of regression equations between IGDO and % fines between our study and an earlier North American study.

Study	Particulate size fraction	Regression equation	R^2 , confidence level
Maret et. al (1993)	< 2mm	$IGDO = -0.15\, fines + 10.0_1$	0.22, p < 0.05
This study, Silverstream, Boggy Creek, and Avon River.	< 2mm	$IGDO = -0.18\, fines + 7.4$	0.16, p < 0.1
This study-Silverstream and Boggy Creek only,	< 2mm	$IGDO = -0.38\, fines + 8.6\ _2$	0.53, p < 0.05
This study-Silverstream and Boggy Creek only.	< 1 mm	$IGDO = -0.93\, fines + 7.8$	0.51, p < 0.05
This study – Silverstream, Boggy Creek, and Avon River	< 1 mm	$IGDO = -0.1\, fines + 6.4$	0.04, p > 0.1

1 = Note the best-fit equation quoted in Fig .9 *in* Maret et al (1993) is incorrect. We have derived the above equation graphically from the best-fit line shown in Fig. 9 of the Maret (1993) study.

Based on the reference habitat Silverstream and rural Boggy Creek dataset, our data indicates a greater sensitivity of IGDO to % fines than the U.S. study (i.e. comparing the equation gradients on rows 1 and 3 in Table 1). Threshold % fines levels to achieve a minimum IGDO are much lower for our dataset, than in the U.S. study. To illustrate, using a compromise interim value that IGDO should not be below 7 mg/L for trout egg survival, then sediment levels would have to remain below 2.2% (using the formula for all reaches) or 4.2% (excluding the Avon River dataset). These sediment thresholds are much lower (more stringent) than in the United States study, with suggesting maximum substrate fines around 15%, even though it was based on a more conservative IGDO threshold of 8 mg/L. However, a scan of our data indicated that high IGDO levels were maintained in some redds with % fine levels well in excess of 5%, indicating extraneous factors may influence IGDO, including heterogeneous substrate composition in the redds, and high BOD.

Another possibility is that the difference in the relationship between substrate composition and IGDO is dependent on the method to obtain the substrate sample. In the Snake Basin Study (Maret *et al.* 1993), a shovel was used to transfer gravels from the redd into a underwater container. The shovel method, while basic, was found to produce comparable results as excavated-core sampling, although frozen-core sampling was found to be biased against larger substrate particles (Grost *et al.* 1991). Our core samples were to a core depth of 13 cm, but if redd sediment increases with gravel depth, it is possible that the samples under-estimated the fraction of fine sediment present at the expected median depth of brown trout eggs (i.e. 16 cm).

We don’t know if our substrate core sample technique biased our results in respect to the American study, although at the time we did not want to destroy the redds with a shovel for the sake of obtaining substrate samples. We were aware the shovel method was considered to produce comparable results as core sampling, so considered core sampling as the most effective, less obtrusive approach.

From a broader ecological perspective, Suttle *et al.* (2001) showed that shifts in invertebrate prey species from vulnerable (mayflies, stoneflies, caddisflies) to armoured and burrowing forms (e.g. snails and worms, midge larvae) induced by fine sediment deposition had a negative effect on the survival and growth of juvenile salmonids. These studies suggest that the management and reduction of sediment sources and impacts is of concern regarding the maintenance of salmonid populations in lowland rivers.

8.2 IGDO – longitudinal variation within redds

Our results indicate a significant differences in IGDO between the most downstream probe (Probe C) in the Avon River, and the two more upstream probes (Probe A and B). IGDO levels in Silverstream were more even, and relatively high, compared to Boggy Creek, which were also fairly uniform, but low. The reasons for these patterns are unclear to us, although one possibility is that redd gravels near the end of the redd may accumulate more fines over time, or simply are not excavated as thoroughly by the hen fish at the time of redd construction. The consequent heterogeneity of the substrate may then be reflected in the IGDO variation within the redd.

While tends are evident, it is clear that redd habitats are quite variable, and some variation could occur by chance, or the influence of an outlying redd. It is also possible that the redds intrinsic microbial BOD, or the eggs themselves, produce such a drain on IGDO, that a decrease in IGDO is detectable along the direction of the ventilating water current. IGDO variation also be related to the distribution of VHG over the redd, and a more detailed picture of how VHG varies along the length of the redd may provide valuable information.

8.3 IGDO – Short-term temporal trends during egg development

For Boggy Creek and Silverstream redds, our study demonstrated a significant, and linear reduction in IGDO during the course of redd maturation, and differing rates of IGDO reduction occurring in different rivers. For the Avon River, IGDO approached measured substrate levels over about 40 days, for Boggy Creek, the initial IGDO level was lower, but the rate in decline of IGDO was also less. In both rivers, the decline in IGDO is likely to be a function of the river-specific build-up of fines in the redds, which are expected to occur over time, and which could trap organic material and the associated BOD.

Within artificial redds exposed to non-point agricultural pollution, IGDO declined over time compared to upstream controls which showed little net effect (Maret *et al.* 1993). The author suggested that BOD may explain this decline, or that the death of the eggs themselves may have created their own BOD. We obtained water samples for BOD late in the egg development phase, which was uniformly high across study reaches, but have no information of how BOD varies during the course of egg development.

However, the Avon River redds exhibited a dichotomy in the relationship between IGDO and redd age, possibly reflecting the impact of redd superimposition, which affected three of the monitored redds. Certainly, superimposed redds on the Avon River had lower levels of fine sediment (Fig. 25), but this did not translate into necessarily higher IGDOs at this site that was statistically detectable from single redds. The microstructure of individual redds, including VHG are also likely to be important in determining IGDO.

Overseas behavioural experiments have demonstrated that brook trout (*Salvelinus fontinalis*) prefer to spawn on existing redds, and that redd superimposition does not necessarily imply that spawning habitat is limited, but that this behaviour is triggered because it is easier for the later spawning fish to excavate redd gravels from a habitat which has already been recently excavated (Essington *et al.* 1998). In a New Zealand study, superimposition of later spawning rainbow trout caused almost total loss of brown trout eggs (Hayes 1987), and it is easy to see how such a behaviour would confer a reproductive advantage for the later spawning fish. In the context of this study, redd superimposition behaviour allows the trout population to clean trout redd gravels twice, and the inclination to superimpose redds may be increased where redd gravels are more embedded into fines in the first instance. This would be the case in the lower spawning reaches of the Avon River. If the above hypothesis is correct, we would expect to see a higher frequency of superimposition of redds where the level of sediment is becoming excessive. As a rule of thumb, early researchers indicated that once spawning gravels become more than 50% embedded in surrounding fines (“waist-deep”) then trout cannot lift the gravels (Hobbs 1948).

In a study monitoring silt build-up in redds along a steam draining a Scottish cultivated catchment, levels of interstitial fines in redds quickly rose during freshes over the winter months (Soulsby *et al.* 2001). While we don’t have data on temporal changes in redd sediment, and there appears to be a gradual fairly linear decline in IGDO, the variation in IGDO along the line may relate to sharp influxes of sediment during freshes. Egg mortalities following spawning were variable, but can be high as 86%, with sediment loads entering the stream from intensively managed land. However, as the authors noted, some redds with high levels of fines exhibit high egg survival. This variation may be explained by ecologically significant variations in physical habitat (redd composition) operating at a microhabitat level within the individual redds, and more research has been recommended to evaluate this micro-habitat variation, at its relationship with trout recruitment (Soulsby *et al.* 2001).

In the context of high flows, Boggy Creek, in particular, was subject to protracted periods of high turbid flows during the winter monitoring period (Pers. Obs. M. Taylor), although flow and water clarity were not quantified. Silverstream remains remarkably clear during rain events, owing undoubtedly to its close groundwater source, and the Avon River would appear to be intermediate in terms of turbidity increases at high flow.

8.4 IGDO – Long-term temporal changes in the Avon and Silverstream Rivers

There is useful background IGDO data on Avon River redds from previous years, as preliminary studies were conducted on IGDO on the Avon River downstream from our study reach eight years ago (Taylor & Burrell 2002). In that study, a different water extraction technique was used, with samples obtained from 15 cm and 25 cm depths, compared with a uniform depth of 16 cm in this study. In the year 2000, when between river contrasts were undertaken, IGDOs differed significantly between the monitored rivers (Avon, Styx, and Silverstream)(ANOVA, $p < 0.001$) with IGDO levels highest from Silverstream redds, intermediate in the Avon, and lowest in a spawning reach in the Styx River. The Styx River trout spawning reach was semi-rural in 2000, and at the time the monitored spawning reach was exposed to stock trampling and access, with largely rural landuse upstream of that point.

Taylor & Burrell (2002) reported follow-up monitoring undertaken on the Avon River during the winter of 2002, from the same location as in 2000, which was adjacent to the Hagley Park car park. This comparison was based on 2 replicates from 3 redds in 2000, and replicates from 2 redds in 2002. Compared to 2000 data, IGDO at a gravel depth of 15 cm strata exhibited an insignificant change from 7.8 mg/L to 7.41 mg/L. However, a significant decline in mean deep-gravel (25 cm depth) IGDO was reported over the same period, from an initial 5.4 mg/L in 2000 to 0.96 mg/L 2002. While brown trout eggs are unlikely to be deposited at this depth, the change in deep IGDO is concerning if trout spawn in upwelling zone. Our recent study, indicated a mean IGDO of 6.5 mg/L at 16 cm depth, which although slightly deeper than in the Taylor & Burrell study, is consistent with declining, and thus worsening, IGDO in the Avon River.
Long-term temporal comparisons are also possible for Silverstream. The mean IGDO in 2000, at 15 cm gravel depth was approximately 8.65 mg/L (AEL, raw data, n=3 redds, 2 reps/redd), compared to our 8.3 mg/L, at 16 cm depth. These results, while sparse, indicate that there is no evidence of a decline in redd IGDO, compared to their counterparts in the Avon River.

8.5 Vertical Hydraulic Gradient, expressed as ΔH

We reported a significant negative association between the magnitude of water upwelling and downwelling (ΔH) and IGDO, although the spawning reaches, as statistical treatment groups, did not differ significantly between each other in mean VHG. Our data indicated that most redds had weakly positive (upwelling) currents, although those with strong downwelling currents, tended to have the highest IGDO levels. The scatter in IGDO as a function of VHG indicates that other factors contribute to IGDO but were not considered in this single-factor analysis. These other factors which will be discussed in the relevant sections below but include levels of fine sediment, and the age of the redd.

We initially assumed that water would consistently upwell through the redd gravels, produce a positive ΔH component, and with a magnitude related positively to the IGDO. This was not the case, with many redds recording significant downwelling (-ve VHG) with relatively high IGDOs compared to those with weak to moderate upwellings.

For the purposes of comparison, we re-evaluated the relationship between IGDO and VHG obtained from the Styx, Avon, and Silverstream Rivers during the winter of 2000. At the time the Styx River spawning reach was vulnerable to stock access. These data are plotted in Fig. 32, and portray a steeper decline in IGDO with increased up-welling, than the relationship in Fig. 24. Silverstream IGDOs are higher for a given VHG, and there is a general indication that IGDOs are less sensitive to the magnitude of the water current flux than for the Styx and Avon River.

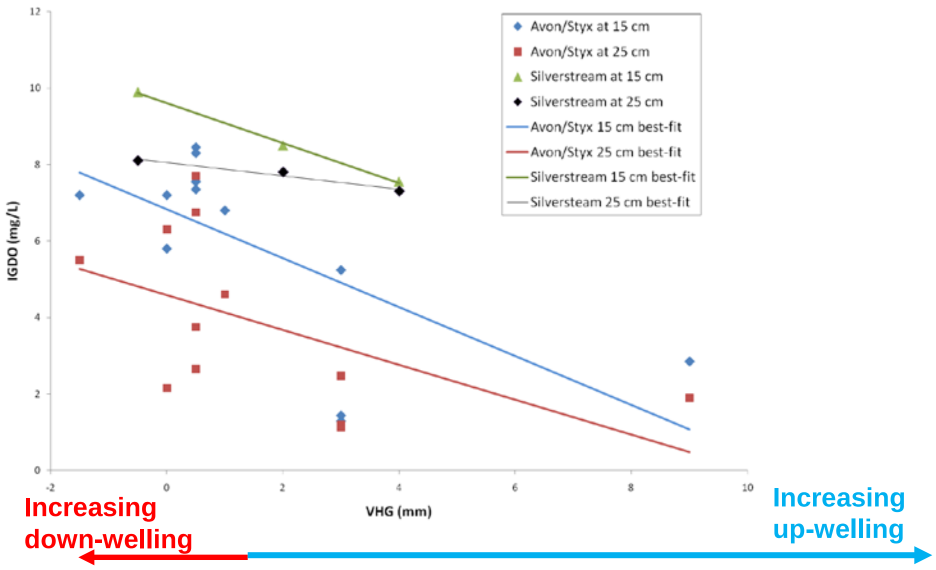


Figure 32. IGDO vs redd VHG from Styx River, Avon, and Silverstream redds in 2000 (unpub. data from Taylor & Burrell 2002). N = 6 redds Styx River, n = 3 redds Silverstream, n=3 redds Avon River.

Our interpretation of these patterns is that groundwater upwellings are often oxygen-depleted, and may be even more so in rural catchments due to microbial activity in the shallow groundwater under the stream bed. Even if the upwelling current is derived from diverted oxygen-saturated surface water, the passing of the water through gravels with significant BOD may deplete the intragravel oxygen level. We recorded very low IGDO levels in the bed of the Avon River at a depth of 25 cm below the redd surface (1 mg/L)(Taylor & Burrell 2002), and interaction with upwellings, even if derived with oxygen-saturated surface water, may deplete the upwelling current.

In the literature, it would appear that salmonid redds are associated with reaches with either downwellings or upwellings. Kondolf (2000) reports of Chinook salmon preferring reaches where there was downwelling, whereas other salmonids prefer, or were associated with, upwelling currents. Conceivably, both water flux directions are potentially effective at ventilating eggs with oxygen, but downwellings may be relatively more effective than upwellings at maintaining high IGDO in reaches with oxygen-depleted groundwater. This is because the redd surface gravels are (due to being excavated by the hen fish) more open and permeable than the deeper gravels which the up-welling current must pass through, and the diffusion path is shorter.

8.6 Rate of intragravel flow

It was evident from our results that our estimated intragravel flows were very low, so we were surprised to find that intragravel flow rates determined from a Clutha River study were broadly similar to ours (Fig. 29) (Bickel & Gloss, 2007), despite these researchers using a totally method based on a falling hydrostatic head in a peizometer (Baxter & Hauer, 2003).

In contrast, in a study of Chinook salmon redds, intragravel flow estimates were much higher, and on average, 794 cm/hr (BioAnalysts, 2003), or 132 mm/min, in what was claimed as ‘very porous redds’. These flow rates were two orders of magnitude higher than our estimates, and although these redds were described as possessing low levels of fines, the substrate composition was not quantified.

While our data are sparse, there is a tentative indication that the intragravel flow in Silverstream was faster than the selected Boggy Creek and Avon River redds. The proportion of fine sediment (< 2 mm) in the tested Boggy Creek redd (replicate 5) was high (in the context of this study) and similar to the tested Avon River redd (replicate 3). In contrast, the fines in the Silverstream redd (replicate 3) was very low in fines. It would be expected that the level of fines would have a major bearing on the intragravel flow, and the hydraulic gradient, by reducing the gravels permeability (i.e. hydraulic conductivity), and the application of Darcys Law outlined in Section 3.0.

The tracer-detection technique used in this study has the advantage of simplicity, it provides a result over a long physical distance, and should therefore be less sensitive to micro-scale physical variations within the redd. Further, no mathematical assumptions are required, which underpin the piezometer technique, although those assumptions could well be valid. However, our replicate flow estimates still possessed significant variation, although the river stage may cause some variation between trials. Another problem was that plume detection was problematic, probably because we just possessed the one detection sensor. The North American study was based on an array of probes, and much faster intra-gravel velocities.

8.7 Water Chemistry within Redds

We were surprised by the relatively high interstitial BOD of 16 mg/L from the reference site Silverstream, and this was the same value as that recorded from the Avon River. While the groundwater source of Silverstream, derived from Waimakariri River water, is likely to have low BOD, there was a significant stock presence near its headwaters during the winter of 2009, and it is clear from the Google Earth aerial map (June, 2009) that old Waimakariri River braids underlie the grazed paddocks, and connect to the surface waters Silverstream (Fig. 33). We surmise these old channels may function as conduits of contaminants from the stocked paddocks to the upper Silverstream surface water.

Figure 33. Old Waimakariri River braids can be discerned (ringed) under grazed paddocks. These braids connect to the surface waters of the upper Silverstream.

The Boggy Creek levels were higher than the Avon or Silverstream, but that was of little surprise, given the stock access upstream of the spawning reach. Interstitial ammonia levels were highest in the Avon River (0.2 mg/L), but were half this in Boggy Creek, and approximately half again in Silverstream. These levels are not likely to be toxic for aquatic life tested in New Zealand, with EC_{50} levels for sensitive biota around 100 times the level recorded in the Avon River.

The overseas literature is dominated by the eco-toxicity of unionised Ammonia on rainbow trout, and it is clear unionised Ammonia is quite toxic at low concentrations. For example, chronic exposure (42 days) of rainbow trout eggs and alevins to unionised ammonia caused damage to gill lamellae (0.19 mg/L), with some detectable growth retardation at levels as low as 0.05 mg/L. The LC_{50} for rainbow trout alevins was of 0.25 mg/L (Burkhalter & Kaya 1997). However, these tests are somewhat misleading because at circum-neutral pH, which all other study reaches will certainly be, ammonia is almost completely ionised to the less toxic NH_4^+ form.

In contrast, New Zealand tests on ammonia toxicity have been standardised to a more realistic, and slightly conservative pH 8, where ammonia exists in the unionised and ionised form. These indicate that two New Zealand species, an amphipod, and a snail are more sensitive to Ammonia than rainbow trout, which places the EC_{50} for total ammonia for this species at between 20 to 30 mg/L for total ammonia at pH 8. No ecotoxicity data was available for brown trout, but they are considered to have a slightly lower sensitivity that rainbow trout. Bird activity in the upstream Mona Vale ponds may be the cause of this high ammonia level.

There has been renewed focus on the eco-toxicity of nitrates in New Zealand waters (Hickey 2009). This follows from increased concern at the rising level of nitrates in our waters, and the earlier misleading perception that nitrate was largely non-toxic. This is still true for acute levels of nitrate (i.e. causing mortalities in 48hr-96hr exposure) as the latest data indicates that some amphipods and snails (i.e. Lymnaea) are more sensitive to nitrate than fish, including rainbow trout. The acute exposure LC_{50} level for the most sensitive organism resident in Canterbury waters, the common mud snail (*Potamopyrgus antipodarum*) was 1042 mg/L. The LC_{50} of juvenile rainbow trout was 1084 mg/L, and brown trout will be around at this level.

However, chronic levels in respect to nitrate toxicity are very much lower, although most of the tests do not relate to New Zealand fauna. Of note, is that rainbow trout (and Chinook salmon) fry had a 30 day NOEC level of just 2.2-2.3 mg/L for nitrate, with egg mortality increasing above these concentrations. We note that nitrate levels within the redd waters were well above these levels in Boggy Creek and Silverstream, and that brown trout alevins and fry would be in the gravels for much longer than 30 days, allowing for both egg and alevin development. However, work is required on the relative sensitivity of brown trout to nitrate exposure in comparison to that of rainbow trout, Another salmonid, Chinook salmon, which also excavates redds in Silverstream, has tested sensitivity to nitrate similar to that of rainbow trout, suggesting that these levels are a useful indicator of harm. Based on this knowledge, this new chronic threshold level data for nitrate indicates that there may be a potential problem with toxicity effects of nitrate on salmonid eggs. Further, while not measured separately, the level of nitrite could be higher than expected if nitrate is converted to nitrate by denitrifying bacteria in hypoxic groundwater. Nitrite is regarded to be toxic to fish, and possibly even more so, than nitrate. We note that the Avon River’s surface and intragravel nitrate levels were below the chronic level, possibly because of active uptake by macrophytes beds in the shallow large ponds a short distance upstream.

8.8 Trout redd excavations

The discovery of so few eggs from 5 redds on Boggy Creek suggests that the alevins may have emerged from the gravels prior our excavation, or alternatively the eggs had died and decomposed at an early stage of development. The redd excavation from Silverstream is interesting, in that most of the eggs were dead, and evidently had perished at a relatively early stage in their development. However, the presence of a few viable late-stage eggs, and empty shells, suggests some survival to the alevin stage.

The results from the Avon River were more encouraging, with the presence of numerous viable alevins suggesting a successful environment for developing trout fry in one of the two excavated redds. Similar to Boggy Creek, one of the redds (replicate 4) may have been sampled too late, explaining why only a few dead eggs were recovered from that redd.

In conclusion, owing to a paucity of eggs and alevins, our results from this exercise were inconclusive, and may have been biased if fry emergence had already taken place, or dead eggs had decomposed. Egg survival trials, designed around a rigorous scientific design, are recommended to evaluate egg survival. This is discussed further below.

8.9 Implications for management of fisheries

Our study suggests that trout recruitment in lowland streams is potentially being compromised by poor water quality, and that this has significant implications for managing the fishery. Our results indicate that low water quality may compromise redd health in at least some rural streams where nitrate levels are high relative to chronic tolerance levels for juvenile trout.

The use of inexpensive portable probes (i.e. transportable in a back-pack), combined with scientifically rigorous study design, has the potential to yield good information on the quality of trout redds in lowland streams. Resulting data will highlight trout fisheries which are prone to water quality issues in respect to spawning, so that resources towards mitigation can be utilised effectively.

The use of riparian buffer strips is well recognised as a technique to reduce the input of non-source pollutants in agricultural catchments. Buffer strips offer only some mitigation towards reducing nitrate levels in streams, but they must be of a width, density, and root penetration sufficient to intercept and uptake nitrate approaching the shallow groundwater of an affected waterway. Another important process is the conversion of nitrate to gaseous nitrogen by anaerobic bacteria in the soil and riparian strip, although this process is most effective in riparian wetlands (Collier *et al.* 1995).

In the case of Boggy Creek, a narrow riparian buffer strip has already been established along the spawning reach, which would assist by intercepting some shallow groundwater nitrate, but this strip may be insufficient to uptake all the nitrate entering from the surrounding dairy pasture. Further, the reach, and the trout redds along it, are subject to nitrate sources entering the channel from upstream sources. Thus, the ecological benefits of this work can be enhanced if riparian management is improved further upstream. In addition to buffer strips, eliminating, or at least managing sediment transfer into the system during high flows is likely to be conducive to trout redd health, as phosphates and bacteria are bound to sediment. Upstream sources of sediment should be identified, including feeding pads and stream fords, and further efforts made to inform and educate landowners on their effects and potential mitigation techniques. Regrettably, the loss of the historical boggy nature of Boggy Creek through land drainage may also have decreased its ability to remove leached nitrate from its associated groundwater. This is especially so in the face of increased nitrate inputs from dairying in the catchment. The management (i.e fencing from stock, native plantings) of natural boggy areas, and organic seeps in the headwaters of the Boggy Creek catchment may facilitate nitrate removal in the groundwater. Nitrate management in agricultural catchments is becoming better understood in New Zealand, and guidelines are available which specifically address this issue, for example Collier *et al.* (1995).

Ultimately, water quality must be maintained along the entire length of the water course, so that the trout spawning grounds, rearing areas and foraging habitat are all protected. For small catchments, these management challenges are not insurmountable, but require the goodwill and co-operation of landowners and resource managers.

9 Conclusion

We have data which indicates that lowered IGDO, and elevated nitrate and fine sediment are reaching levels which are documented as causing problems to salmonid egg survival and development. Further study is required to link egg development and survival to measured water quality parameters.

It would appear there is little knowledge of the relative sensitivity of brown trout eggs to chronic nitrate levels compared to the nitrate sensitivity of rainbow trout. Further, there is no knowledge of the synergistic effects of both low IGDO and elevated BOD to nitrate/nitrite eco-toxicity. Therefore we recommend that further water quality samples be obtained from brown trout redds in lowland streams, and related directly to egg survivability. Our observations from trout redd excavations only offer a preliminary indication of effects, and require further corroboration and a stronger experimental framework.

At the time of writing, a factorial designed experiment is being conducted in Waikuku Stream under the auspices of North Canterbury Fish and Game Council. Waikuku Stream is a rural spring-fed tributary of the Ashley River north of Christchurch, and the trout spawning reaches passes through dairying land, possessing varying degrees of riparian fencing. The experiment involves monitoring water quality in mostly artificially constructed redds, which incorporate baskets containing hatchery-fertilised brown trout eggs. Aluminium IGDO probes and VHG tubes have been inserted for monitoring oxygen levels, and VHG, but it would be constructive to draw intra-gravel water samples from these probes, to assess the nitrate/nitrite levels and relate these to the resultant egg survival.

10 Recommendations

AEL presents the following recommendations to Fish and Game New Zealand:

- Extend redd monitoring to cover lowland brown trout fisheries with contrasting background levels of nitrates or other contaminants e.g. Otukaikino River (South Branch, Waimakariri River), and the Waikuku Stream or Cam River. AEL has good knowledge on the trout spawning distribution in these rivers. This work would benefit through using egg basket trials to better demonstrate cause and effect. Some initial studies are already underway, and could be modified slightly to improve knowledge about nitrate levels in particular.

- Intragravel water flow rates are an important contributor to redd quality, and it is recommended that further data be obtained on this parameter, but with technique modification to improve reliability. The vertical hydraulic gradient (VHG) is also an important hydraulic property for redd ventilation which should be measured, and is easy to do so.

- Catchments upstream of trout spawning grounds, where possible, need to be managed as a whole, to control water quality in trout redds. Small catchments will obviously be better to manage. In this context, an investigation into upstream sediment, nitrate, and other contaminant sources in Boggy Creek would enhance the investment in community good will and resources which have been undertaken in this catchment. Once problem sources are determined, and mitigated (e.g. riparian fencing and planting), the downstream water quality in trout redds can be re-assessed.

- Managing entire small lowland catchments for trout redd habitat quality may be more effective than isolated reach enhancement work spread over a number of catchments. This is because of the intrinsic dependence between the water quality in trout redds and what is occurring upstream both at winter baseflow and high flow events.

- Improving and standardising substrate measures for fine sediment to assist assessing thresholds of harm for trout eggs. The relationship between IGDO and fine sediment is clearly negative, but quantifying fine sediment levels using various collection methods is method specific, and therefore influences the relationship between fines and IGDO. We used a simple portable manually-operated substrate coring method which produced meaningful results, but which could be improved upon through further field trials.

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12 References

Baxter, C.; Hauer, R. F.; Woessner, W. W. 2003: Measuring Groundwater-Stream Water Exchange: New Techniques for installing minipiezometers and estimating hydraulic conductivity. *Transactions of the American Fisheries Society 132*: 493-502.

Bickel, T. O.; Closs, G. P. 2007: *Didymosphenia geminata*: a possible threat to trout spawning success? *Report prepared for Clutha Fisheries Trust*: 10.

BioAnalysts, I. 2003. Effects of powerhouse operations on intragravel flows and water quality within Chinook Redds; draft. BioAnalysts, Inc, Wenatchee. *No*. 41 p.

Brunke, M.; Gonser, T. 1997: The ecological significance of exchange processes between rivers and groundwater. *Freshwater Biology 37* (*1*): 1-33.

Burkhalter, D. E.; Kaya, C. M. 1997: Effects of prolonged exposure to Ammonia on fertilized eggs and sac fry of rainbow trout (Salmo gairdeneri). *Transactions of the American Fisheries Society 106*: 470-475.

Chapman, D. W.; McLeod, K. P. 1987. Development of criteria for fine sediment in the Northern Rockies Ecoregion. U.S. Environmental Protection Agency, *No. 910/9-87-162*.p.

Collier, K. J.; Cooper, A. B.; Davies-Colley, R. J.; Rutherford, J. C.; Smith, C. M.; Williamson, B. R. 1995. Managing Riparian Zones: A contribution to protecting New Zealand's rivers and streams. . Department of Conservation, Wellington. *No*. p.

Eldon, G. A.; Kelly, G. R. 1992. Fisheries survey of the Avon River, 1991-1992. Freshwater Fisheries Centre, Christchurch. *New Zealand Freshwater Fisheries Report No. 137*. 33 p.

Environment Canterbury 2004. Waimakariri Regional Plan. Christchurch. *No. R04/7*. 149 p.

Essington, T. E.; Sorensen, P. W.; Paron, D. G. 1998: High rate of redd superimposition by brook trout (Salvelinus fontinalis) and brown trout (Salmo trutta) in a Minnesota stream cannot be explained by habitat availability alone. *Canadian Journal of Fisheries and Aquatic Science 55*: 2310-2316.

Geist, D. R.; Dauble, D. D. 1998: Redd Site selection and spawning habitat use by fall Chinook Salmon: The importance of geomorphic features in large rivers. *Enviromental Management 22* (*5*): 655-669.

Grost, R. T.; Hubert, W. A.; Wesche, T. A. 1991: Field comparison of three devices used to sample substrate in small streams. *North American Journal of Fisheries Management 11*: 347-351.

Hamill, K. D.; McBride, G. 2003: River water quality trends and increased dairying in Southland, New Zealand. *New Zealand Journal of Marine and Freshwater Research 37*: 323-332.

Harding, J. S.; Young, R. G.; Hayes, J. W.; Shearer, K. A.; Stark, J. D. 1999: Changes in agricultural intensity and river health along a river continuum. *Freshwater Biology 42*: 345-357.

Hardy, C. J. 1963: An examination of eleven stranded redds of brown trout (Salmo trutta), excavated in the Selwyn River during July and August, 1960. *New Zealand Journal of Science 6*: 107-119.

Hawke, S. P. 1978: Stranded redds of quinnat salmon in the Mathias River, South Island, New Zealand. *New Zealand Journal of Marine and Freshwater Research 12* (*2*): 167-171.

Hayes, J. W. 1987: Competition for spawning space between brown trout (Salmo trutta) and rainbow trout (S. gairdeneri) in a lake inlet tributary, New Zealand. *Canadian Journal of Fisheries and Aquatic Science 44* (*1*): 40-47.

Hickey, C. W. 2000: Ecotoxicology: laboratory and field approaches.415 *In:* Collier, K. J.; Winterbourn, M., J. *eds*: New Zealand Stream Invertebrates: Ecology and Implications for Management. New Zealand Limnological Society. Christchurch.

Hickey, C. W. M., M.L. 2009. A review of nitrate toxicity to freshwater aquatic species. Environment Canterbury, Christchurch *Environment Canterbury Technical Report No*. 46 p.

Hobbs, D. F. 1948. Trout fisheries in New Zealand. New Zealand Marine Department, Christchurch. *N. Z. Marine Department Fisheries Bulletin No*. *9*. 175 p.

Jensen, J. W.; Steel, E. A.; Fullerton, A. H.; Pess, G. R. 2009: Impact of fine sediment on egg-to-fry survival of Pacific Salmon: A meta-analysis of published studies. *Reviews in Fisheries Science 17* (*3*): 348-359.

Kondolf 2000: Assessing Salmonid Spawning Gravel Quality. *Transactions of the American Fisheries Society 129*: 262-281.

Larned, S. T.; Scarsbrook, M. R.; Snelder, T. H.; Norton, N. J.; Biggs, B. J. F. 2004: Water quality in low-elevation streams and rivers of New Zealand: recent state and trends in contrasting land-cover classes. *New Zealand Journal of Marine and Freshwater Research 38*: 347-366.

Main, M. R. 1999. Sources of faecal contamination in the Ashburton River. Environment Canterbury, Christchurch. *Canterbury Regional Council Technical Report No*. *U99/88*. 24 p.

Main, M. R.; Taylor, M. J. 2010. Draft: Monitoring of fish values in the City Rivers: 2nd round; Avon River. Aquatic Ecology Limited, Christchurch. *AEL Report No*. *59*. 17 p.

Maret, T. R.; Burton, T. A.; Harvey, G. W.; Clark, W. H. 1993: Field testing of new monitoring protocols to assess brown trout spawning habitat in an Idaho stream. *North American Journal of Fisheries Management 13*: 567-580.

McDowall, R. M. 1990: New Zealand Freshwater Fishes: A Natural History and Guide. Auckland, Heinemann Reed. 553 p.

Meredith, A. M.; Hayward, S. A. 2002. An overview of the water quality of the rivers and streams of the Canterbury region. Environment Canterbury, Christchurch. *Environment Canterbury Technical Report No. R02/25*. 32 p.

Pinheiro, J. C.; Bates, D. M. 2000: Mixed-Effects Models in S and S-PLUS. New York, Springer. p.

Ross, B. J. 1984. Fisheries investigation: Spawning potential of streams and drains flowing into Lake Ellesmere. North Canterbury Acclimatisation Society, Christchurch. *Unpublished report No.* 2 p.

Soulsby, C.; Youngson, A. F.; Moir, H. J.; Malcolm, I. A. 2001: Fine sediment influence on salmonid spawning habitat in a lowland agricultrual stream: a preliminary assessment. *The Science of the Total Environment 265*: 295-307.

Suttle, K. B.; Power, M. E.; Levine, J. M.; McNeely, C. 2001: How fine sediment in riverbeds impairs growth and survival of juvenile salmonids. *Ecological Applications 14* (*4*): 969-974.

Taylor, M. J. 2005. (Draft) Brown trout spawning in the Lake Ellesmere (Te Waihora) tributaries, and some surrounding catchments Aquatic Ecology Limited, Christchurch. *AEL Report No. 31*. 116 p.

Taylor, M. J.; Bray, J. 2008. Brown trout spawning in the Avon River, and comparisons with previous surveys. Aquatic Ecology Limited, Christchurch. *AEL Report No. 61*. 21 p.

Taylor, M. J.; Burrell, G., P. 2002. Brown trout redd distribution in the Avon River, and a comparison with the 1991 trout redd survey. Aquatic Ecology Limited, Christchurch. *AEL Report No. 14*. 24 p.

Taylor, M. J.; Good, M. 2006. Brown trout spawning in the Lake Ellesmere (Te Waihora) tributaries, and some surrounding catchments Aquatic Ecology Limited, Christchurch. *AEL Report No. 31*. 116 p.

Teirney, L. D.; Richardson, J.; Unwin, M. J. 1987. The relative value of North Canterbury rivers to New Zealand anglers. Freshwater Fisheries Centre, MAFFish, Wellington. *New Zealand Freshwater Fisheries Report No. 89*. 113 p.

Terhune, L. D. B. 1958: The Mark VI groundwater standpipe for measuring seepage through salmon spawning gravel. *Journal of the Fisheries Research Board of Canada 15* (*5*): 1027-1063.

Unwin, M. J. 2009. Angler usage of lake and river fisheries managed by Fish & Game New Zealand:results from the 2007/08 National Angling Survey. National Institute of Water and Atmospheric Research, Christchurch. *NIWA Client Report No. CHC2009-046*. 110 p.

Vervier, P.; Gibert, J.; Marmonier, P.; Dole-Olivier, M. J. 1992: A perspective on the permeability of the surface freshwater-groundwater ecotone. *Journal of the North American Benthological Society 11* (*1*): 93-102.

Zuur, A. F.; Leno, E. N.; Walker, N. J.; Saveliev, A. A.; Smith, G. M. 2009: Mixed effects models and extensions in ecology with R. New York, Springer. 574 p.

13 Appendix I. Photographs of monitored trout redds

Monitored brown trout redds on Boggy Creek

A) Boggy Creek, Replicate 1.

B) Boggy Creek, Replicate 2.

C) Boggy Creek, Replicate 3.

D) Boggy Creek, Replicate 4.

E) Boggy Creek, Replicate 5.

Monitored brown trout redds on Silverstream

E) Silverstream, Replicate 1,

F) Silverstream, Replicate 2

G) Silverstream, Replicate 3.

H) Silverstream, Replicate 4.

I) Silverstream, Replicate 5. Compound redd.

Monitored brown trout redds on the Avon River

J) Avon River, Replicate 1, multiple compound redd.

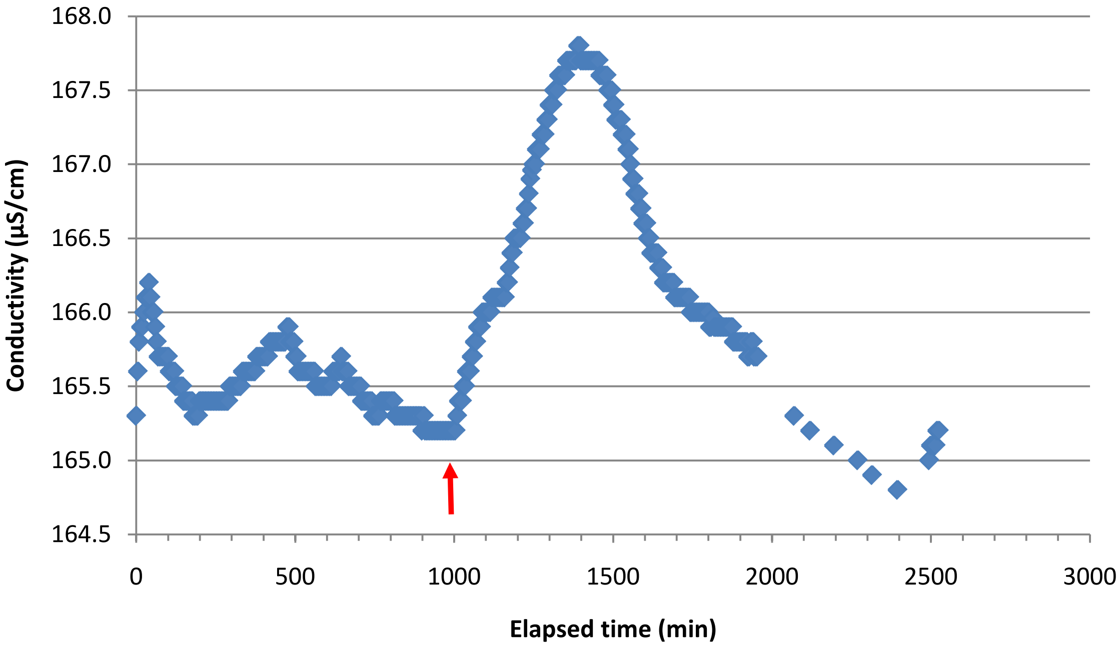
K) Avon River, Replicate 2. Compound redd.

L) Avon River, Replicate 3.

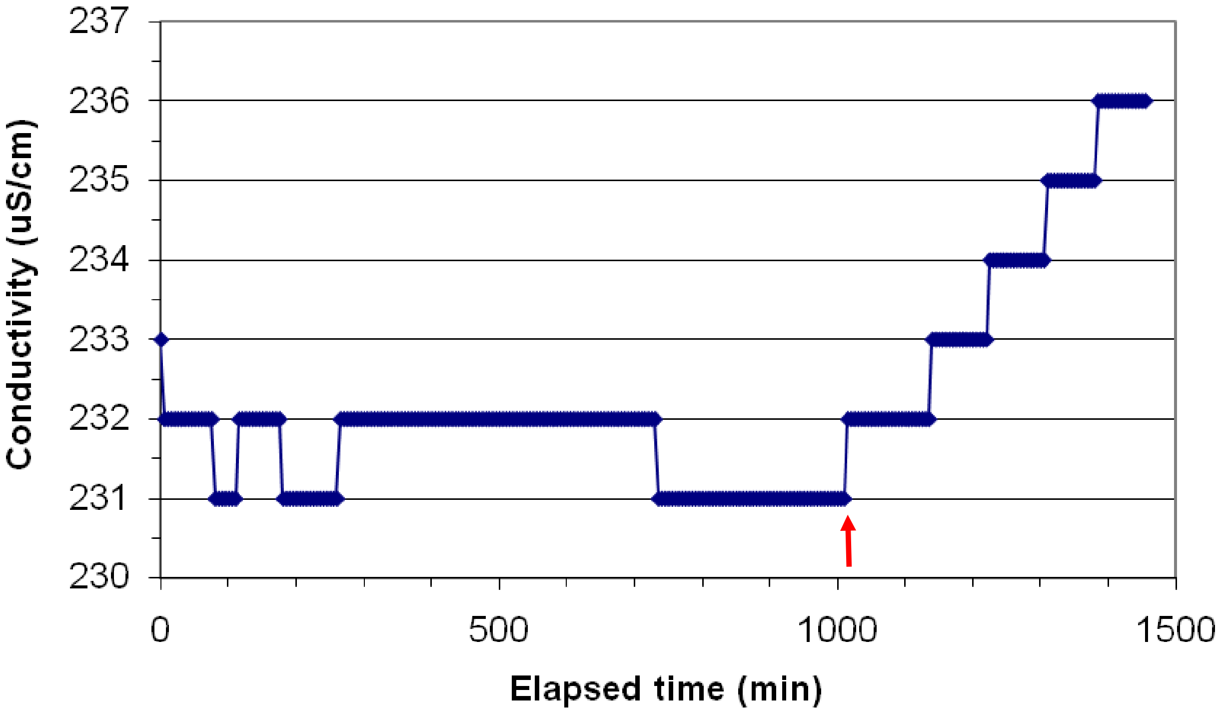
M) Avon River, Replicate 4, compound redd.

N) Avon River, Replicate 5.

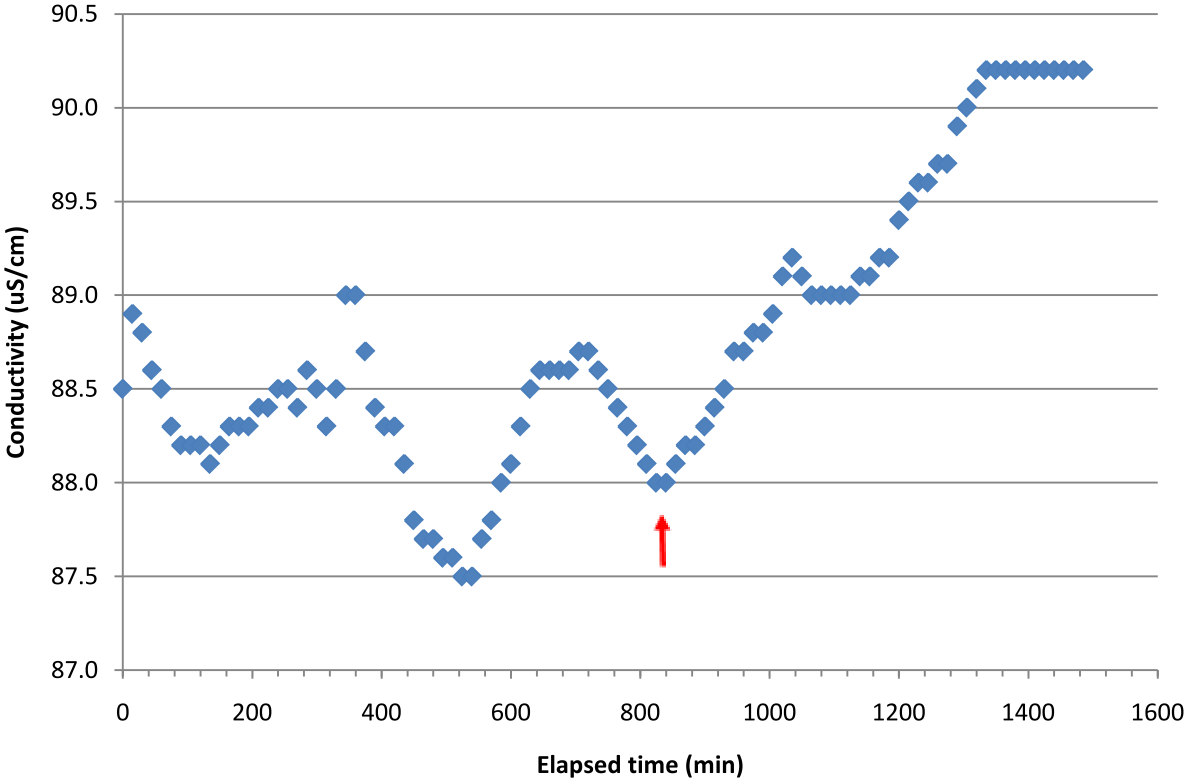
14 Appendix II. Intra-gravel flow conductivity traces



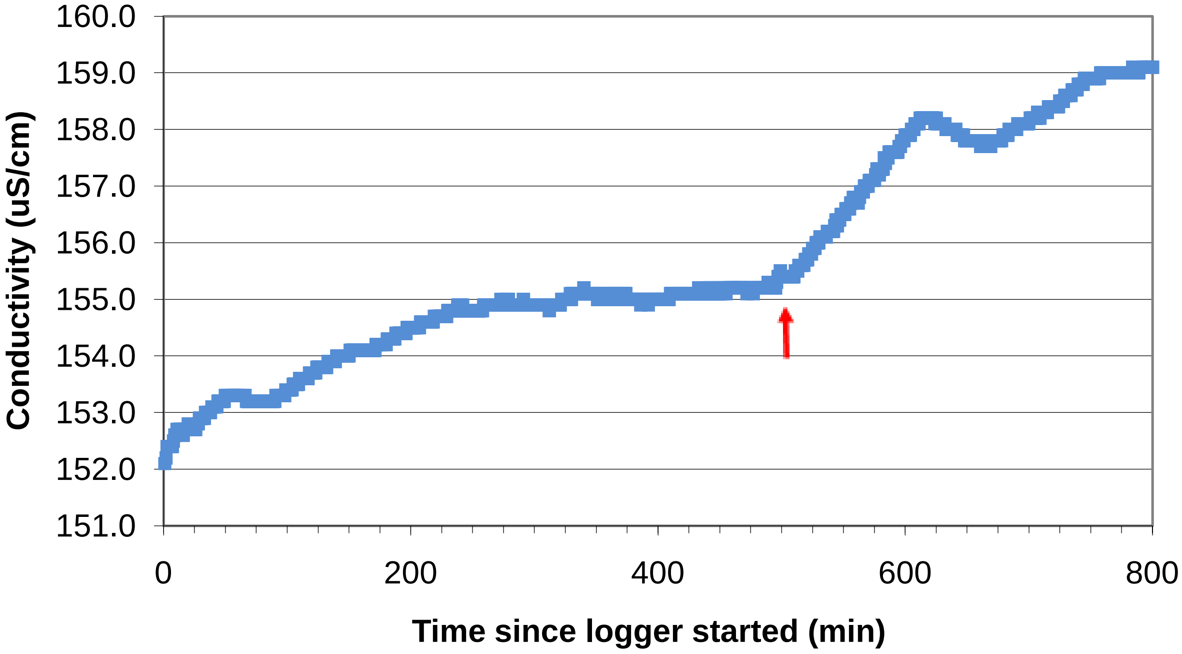
A) Boggy Creek, Redd Replicate 5, trial 3. Assigned response is arrowed.



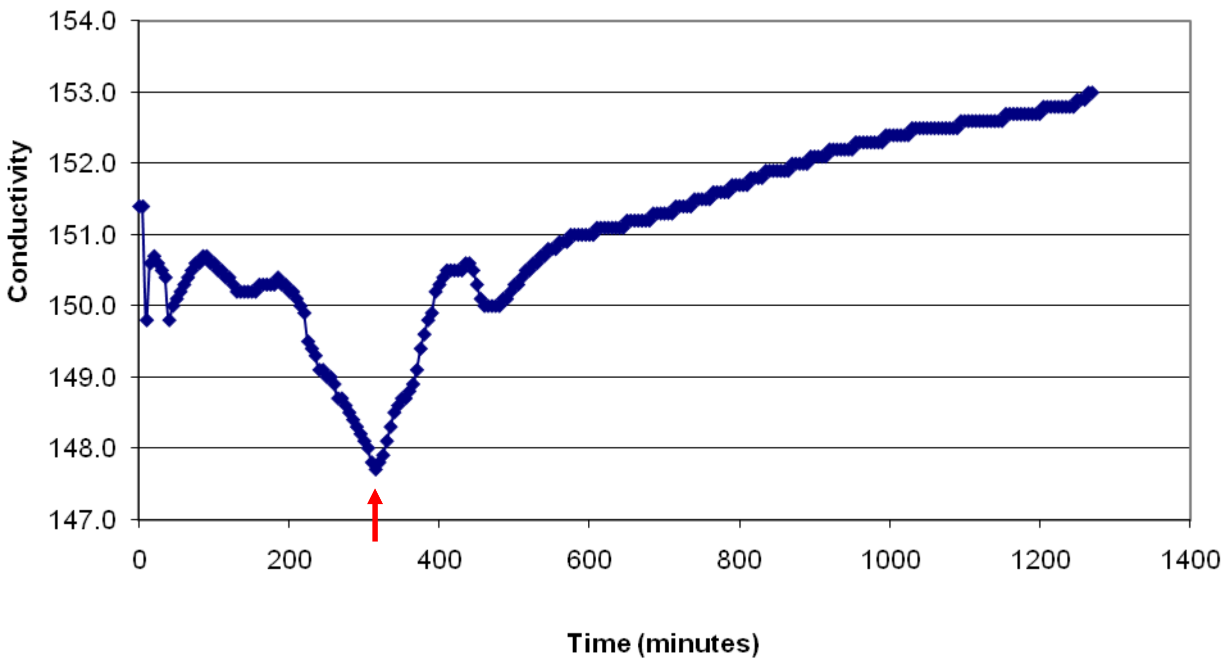
B) Boggy Creek, Redd Replicate 5, trial 1. Assigned detection response is arrowed.



C) Avon River, Redd Replicate No. 3, trial 3. Assigned detection response is arrowed.



D) Silverstream, Redd Replicate 3, trial 1, Assigned detection response is arrowed.



E) Silverstream, Redd Replicate 3, trial 2. Assigned detection response is arrowed.