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Complete versus partial macrophyte removal: the impacts of two drain management strategies on freshwater fish in lowland New Zealand streams

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Abstract – Complete macrophyte removal to maintain drainage performance in lowland streams can have a negative effect on resident fish communities, but few studies have quantified this impact. Moreover, limited research has been carried out exploring alternative approaches for macrophyte removal that minimise the impact on the resident fish community. The aims of this study were (i) to determine how the current practice of removing almost 100% of available macrophyte cover affects native fish populations in lowland New Zealand streams and (ii) to see whether this impact can be reduced by limiting macrophyte removal to alternating 50-m sections of the waterway. Native fish populations were surveyed before and after experimental macrophyte removal for the following three treatments: (i) complete macrophyte removal, (ii) macrophyte removal from alternating 50-m reaches and (iii) control with no macrophyte removal. Radiotelemetry was used to monitor the behavioural response of individual giant kokopu (*Galaxias argenteus*) to the different treatments. The results of this study suggest that current drain management practices reduce CPUE of fish by 60%. Although limiting macrophyte removal to alternating 50-m sections did not minimise the community impacts of drain clearing, large giant kokopu did benefit from this strategy. All tagged giant kokopu remained in stream reaches partially cleared of macrophytes, while in completely cleared reaches all individuals were displaced. These results demonstrate the threat current drain management practices pose to New Zealand native fish and highlight the value of trialling alternative methods of macrophyte removal.

Key words: catchment management; common bully; plant management; aquatic weeds; anthropogenic disturbance; giant kokopu

Introduction

Anthropogenic disturbance of streams draining agricultural and industrial land has reduced both the abundance and the range of native fish species globally (Maitland 1995). The negative effects of dams and chemical pollution are well documented (e.g., Alabaster & Lloyd 1982; Maitland 1995; Santos et al. 2006; Zhai et al. 2010), but relatively little is known about the effects of many other human disturbances on freshwater ecosystems. For effective conservation management of lotic ecosystems it is essential that anthropogenic threats to native fish are understood, as they often play key roles in aquatic communities (Maitland 1995). Thus, changes in fish abundance can have a disproportionately large effect on community structure (Maitland 1995).

A little understood source of disturbance in low-altitude rivers draining pastoral land is the regular removal of aquatic macrophytes (Swales 1982; Young et al. 2004). For successful drainage of agricultural run-off, streams must remove water from the pasture quickly and efficiently while promoting physical and chemical conditions in the soil that are favourable for agricultural production (Hudson & Harding 2004). Accelerated macrophyte growth associated with increased nutrient input can limit drainage outfall (Armitage et al. 1994; Kaenel et al. 1998).

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High densities of these plants can increase sediment deposition, reduce flows and potentially flood the surrounding pastoral land (Kaenel & Uehlinger 1988; Hearne & Armitage 1993). To prevent this, it is necessary to regularly clear macrophytes from the streams that drain agricultural land, using herbicides, mechanical or manual extraction of plant material and, occasionally, plant-eating fish (Pieterse & Murphy 1990; Wells et al. 2003; Hudson & Harding 2004).

Although macrophytes limit the drainage efficiency of pastoral land, they play a key role in maintaining desirable physical and chemical conditions in freshwater systems (Fox 1992). It is therefore likely that the absence of macrophytes following drain clearing reduces the functioning of aquatic ecosystems. In addition, the physical process of removing macrophytes can be a source of disturbance in itself. Herbicides may adversely affect non-target organisms and alter the chemical properties of the water column via the decay of aquatic plant material (Murphy & Barrett 1990). Mechanical excavation is the most disruptive method of macrophyte control and can result in the immediate loss of a high proportion of the available plant cover (Kaenel & Uehlinger 1988). This can reduce the heterogeneity of the stream bed, thereby limiting the number of species it can support (Hicks & Reeves 1994). Despite a widespread understanding that macrophyte removal probably has a detrimental impact on community structure in streams (Swales 1982), its impact on freshwater fish populations has been the focus of relatively few studies. The limited research, however, has suggested that mechanical removal of macrophyte cover reduces growth rates and increases the predation of juvenile fish (Mortensen 1977; Garner et al. 1996), leads to the removal of fish during mechanical clearing (Dawson et al. 1991; Serafy et al. 1994), and alters the behavioural patterns of many freshwater fish species (Swales 1982).

To date, research on the effects of drain management on New Zealand's native fish populations has produced conflicting results. Goldsmith (2000) reported that there is anecdotal evidence that macrophyte removal increases the abundance of some fish species like inanga (*Galaxias maculatus)* and smelt (*Retropinna* spp.), but leads to decreases in the abundance of other species like common bully (*Gobiomorphus cotidianus)* and eels (*Anguilla* spp.). Unfortunately, this information is not available in the primary literature, and these unpublished findings have since been questioned because of a lack of analytical power (Young et al. 2004). As a result of the limited research on this topic, it is unclear how the widespread removal of frequently abundant macrophyte species like oxygen weed (*Egeria* spp.), starwort (*Callitriche* sp.), swamp willowweed (*Polygonum* sp.) and pondweed (*Potamogeton* spp.) affects New Zealand's native fish (Hudson & Harding 2004).

One species of fish that may be particularly susceptible to macrophyte removal is the drift-feeding giant kokopu (*Galaxias argenteus*). Macrophyte removal not only reduces the availability of diurnal cover for this fish, but also results in the removal of a large quantity of invertebrate prey from the water column (Hudson & Harding 2004). Therefore, we expect reduced food availability to increase the size of giant kokopu home ranges, increase levels of aggression and alter diel activity patterns (David & Closs 2003; Hansen & Closs 2005, 2009). Understanding how macrophyte removal impacts this species is particularly crucial given its 'declining' status (Allibone et al. 2010).

The present study aimed to examine the community level impacts of mechanical drain clearing on native fishes and the possibility of minimising these effects with an alternative macrophyte clearing approach. Specifically, we aimed to quantify the impacts of complete drain clearing on native fish communities and contrast these changes with those from areas where macrophyte removal was limited to alternating 50-m sections. We predict that fish abundance will be reduced by macrophyte clearing and that community composition will change. We also predict that these changes will be less severe when macrophyte removal is limited to alternating sections and that large giant kokopu will be less likely to be displaced from streams cleared in this manner.

Study area

Waituna lagoon is a largely unmodified coastal lake located approximately 40 km south-east of Invercargill in the South Island of New Zealand (Fig. 1). The lagoon's catchment consists of about 20,000 hectares of farmland, native forest and the internationally important Awarua wetlands. Drainage input for the catchment is provided by three major streams that flow directly into the lagoon. Of these streams, the Waituna Creek drains the largest area (12,500 ha), followed by Carrans Creek (5700 ha) and Moffat Creek (1700 ha). The majority of waterways in the region are extensively modified as a result of agricultural development, and regular mechanical excavation is needed to maintain adequate drainage of the surrounding low-lying pastoral land (Riddell et al. 1988). Despite regular macrophyte removal, the fish community in the area is relatively diverse, and the catchment has a large population of the declining giant kokopu (Riddell et al. 1988; Thompson & Ryder 2002; Allibone et al. 2010).

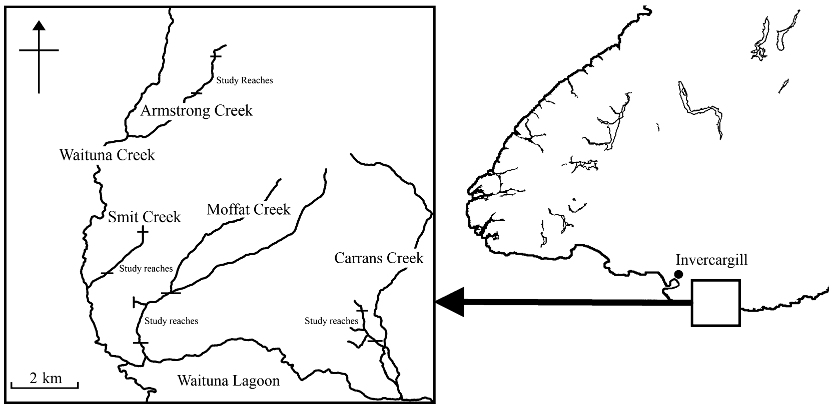

Fig. 1. Study sites located in the reaches and tributaries of Armstrong Creek, Smit Creek, Moffat Creek and Carrans Creek in the catchment of Waituna Lagoon in the South Island of New Zealand.

Methods

Study sites and treatments

The impacts of macrophyte removal were examined in twenty-three 350-m treatment reaches located across four streams (Armstrong Creek, Carrans Creek, Smit Creek and Moffat Creek). Eight 350-m reaches were available along Moffat Creek, while five reaches were available in each of the remaining three streams. These 23 reaches were then randomly allocated to one of the following three treatments: (i) complete macrophyte removal (cleared), (ii) macrophyte removal from alternating 50-m reaches – leaving half of the overall habitat intact (staggered) and (iii) no macrophyte removal (control) (Fig.2). All of these reaches had >50% macrophyte coverage and were all separated by at least 50 m of undisturbed waterway. This resulted in seven replicates of the cleared treatment (three in Moffat Creek, two in Armstrong Creek, one in Smit Creek and one in Carrans Creek), eight replicates of the staggered treatment (three in Moffat Creek, two in Armstrong Creek, two in Smit Creek and one in Carrans Creeks) and eight control replicates (two in Moffat Creek, two in Armstrong Creek, one in Smit Creek and three in Carrans Creek). Unfortunately, the different number of replicates of each treatment in each stream was unavoidable because of the wishes of various landowners and local government agencies. Experimental drain clearing was carried out between 22nd and 25th of March 2011 using a mechanical excavator.

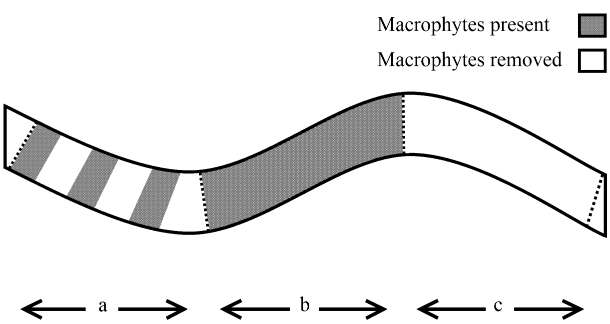

Fig. 2. Illustration of the drain clearing treatments employed in this study. Section a = macrophyte removal limited to alternating 50-m sections (staggered); section b = all macrophytes left undisturbed (control); section c = complete removal of all aquatic macrophytes (cleared).

Physico-chemical properties and habitat structure

To determine the impact of the different drain clearing techniques on fish habitat structure, the physico-chemical characteristics of each treatment reach were analysed twice, once before macrophyte removal (between the 5th and 20th of March 2011) and once after (between the 10th and 25th of April 2011). Water temperature and relative conductivity were measured at the most upstream point of each reach using a YSI probe (YSI Inc. Model 85). Eight transects were then placed along the length of the reach at 50-m intervals. At each transect, the percentage of the stream width covered by key plant groups (macrophytes, bryophytes, mat algae and filamentous algae) was estimated and coverage was recorded on a scale of 1–3 (1 = rare, i.e., <20% coverage; 2 = common, i.e., 20–60% coverage; 3 = abundant, i.e., >60% coverage). Stream width was then measured at water surface level, and water depth was measured at 0.5-m intervals across the transect. Transect data were compiled and used to estimate the maximum depth, mean depth and mean width of each reach as well as the mean abundance scores of key aquatic plant groups.

The percentage of the stream width covered by the following substrate types – mud (<1 mm), sand (1–2 mm), fine gravel (3–20 mm), coarse gravel (21–60 mm) and cobble (61–260 mm) – was visually assessed at each transect. A total of 20 particles within each substrate category were collected from four points picked haphazardly along each transect and measured across the intermediate axis, excluding mud and sand that were assumed to be 0.5 mm and 1.5 mm, respectively. From these values, the mean size of particles belonging to each substrate category was estimated and used in the following formula to calculate overall mean particle size at each transect.

$$\text{Mean particle size} = \frac{\sum_{n=i}^{n} (S_t \times P_t)}{100}$$

Here, S represents the mean particle size of substrate type t, and P represents the percentage of the width of the stream covered by t. Substrate size was averaged across the eight transects to calculate a mean value for each treatment reach.

Fish population surveys

The impacts of the different drain clearing methods on fish abundance and diversity were measured by surveying the fish populations of each study reach twice, once before drain clearing (between the 5th and 20th of March 2011) and once after (between the 10th and 25th of April 2011). Treatment reaches were separated into four individual 50-m sampling sites, and the resident fish populations in each site sampled separately by overnight netting. To avoid catching non-resident fish, the most downstream 50 m and the most upstream 100 m in each 350-m treatment reach were excluded from sampling. Large fish (>130 mm in length) were captured using two plastic fish traps per 50-m sampling site. The traps were constructed from cylinders of plastic netting (mesh size, 20 mm) and measured 1.2 × 0.40 m. Funnelled vertical entrances constructed from plastic netting allowed fish to enter the trap from either end, even in water too shallow for the use of conventional fyke nets. The traps were set approximately 20 m apart, positioned parallel to the bank and secured to the stream bed using metal pegs. Smaller fish (<130 mm) were caught using G-minnow traps that measured 420 × 230 mm, had 25-mm entrance apertures and a mesh size of 2 mm (Swales 1987). Three G-minnows, secured to one another with a length of nylon rope, were set in the centre of each 50-m sampling site and positioned so the long axis ran parallel to the bank.

All fish traps were deployed 1-3 h before dusk, and collection began 1 h after dawn. Captured fish were identified to the species level and released at the point of capture immediately after examination (except large giant kokopu kept for radio tagging). The abundance of individual species in each 50-m sampling site was recorded in CPUE (catch per unit effort), that is, the total number of fish caught in all nets per hour of fishing (fish per hour). As the catch rates of individual species were low, the CPUE data from all species present at a site were pooled to produce a satisfactory estimate of total fish abundance (Swales 1982).

Radiotelemetry

To measure the response of individual giant kokopu to the different drain clearing techniques, radio transmitters (ATS model F1030) were surgically implanted into the abdominal cavities of 11 large adult giant kokopu (three from Armstrong Creek, four from Moffat Creek and four from a small drainage network in the catchment of Carrans Creek). Four fish were captured in the first sampling period between the 8th and 21st of March, and the remaining seven fish were collected between the 22nd and 25th of March 2011 after being removed from the waterways by the excavator during macrophyte removal. Eight fish were tagged in reaches from the cleared treatment group, two from the staggered and one from the control. The limited numbers of fish caught from within the stream (i.e., not with the excavator) resulted in low numbers of fish being tagged in staggered and control reaches (Table 1). Further surveys were carried out to increase the numbers of fish in the control reaches, but no fish were caught.

Radio implants were carried out following the methodology of David & Closs (2001). Briefly, fish were anaesthetised with a dilute solution of AQUI-S (20 $\mu l \cdot l^{-1}$) and placed ventral side up in a fish-shaped relief cut into a wet sheet of foam rubber. A 10-mm incision was made approximately 15 mm posterior to the base of the pelvic fins and between three and five mm to the right of the ventral midline. A radio transmitter was inserted into the fish's abdominal cavity through this incision. The fish was kept unconscious during surgery by aerating the gills with a dilute solution of AQUI-S (20 $\mu l \cdot l^{-1}$). The incision was flushed with saline solution and closed with three interrupted stitches. To reduce the risk of infection and decrease healing time, the wound was also coated with a povidone-iodine topical antiseptic (Betadine, Purdue Pharma).

The transmitters weighed 2.1 g (in air), had an internal loop transmitter and an estimated battery life of 94 days (30 pulses min^{-1}, pulse width 15 ms). Based on Winter's (1983) recommendation, the transmitters were no more than 2% of the body weight of the fish into which they were implanted. The mean weights and standard lengths (SL = snout to base of caudal fin) of tagged fish were 186.3 mm and 142.7 g (Armstrong Creek), 214.9 mm and 195.9 g (Moffat Creek) and 214.4 mm and 195.1 g (Carrans Creek) (Table 1).

Following surgery, the fish were left to recover for between 10 and 20 min and then released at the point

Table 1. Weights and standard lengths of individual fish, whether it was located in the study area after macrophyte removal, the total number of day and night locations and the dates between which location data were collected.

Stream	Fish	Located	L (mm)	W (g)	Treatment	Day	Night	Track Period
Armstrong	A176	No	190.5	118.9	Cleared	–	–	–
	A296	Yes	180.9	142.6	Cleared	1	–	26/5–27/5
	A127	No	187.5	166.5	Cleared	–	–	–
	Mean	–	186.3	142.7	–	–	–	–
Moffat	M176	Yes	231.0	244.8	Control	2	2	13/5–26/5
	M299	Yes	198.1	135.9	Staggered	5	7	13/5–12/6
	M127	Yes	195.6	162.8	Cleared	2	2	13/5–26/5
	M266	Yes	235.0	240.0	Staggered	5	6	13/5–26/5
	Mean	–	214.9	195.9	–	–	–	–
Carrans	C156	Yes	221.4	234.8	Cleared	2	–	14/5–16/6
	C286	No	209.1	158.5	Cleared	–	–	–
	C226	Yes	198.9	156.7	Cleared	7	6	14/5–11/6
	C136	Yes	228.1	230.3	Cleared	1	–	14/5–15/5
	Mean	–	214.4	195.1	–	–	–	–

of capture once equilibrium had been regained. Fish were located twice in the 48 h following surgery to ensure the tags were not expelled. As in David & Closs (2003), location data were not recorded for a minimum of 2 weeks after tagging to ensure behavioural patterns had returned to normal.

Telemetry data collection commenced on the 13th of May 2011 and ended on the 11th of June 2011. When weather conditions permitted, each fish was tracked twice every 48 h, once during the day (09.0017.30 h) and once at night (18.30–03.00 h). Fish were located using a scanning receiver (Falcon five; Wildlife Materials International Inc., Carbondale, IL, USA) and a hand-held directional three-element Yagi antennae. The positions of located fish were recorded as the distance and direction (either upstream or downstream) of their current location in relation to where they were first located and released following tagging. If a fish was located in a study reach, the treatment group of the reach was recorded. If the reach belonged to the staggered treatment group, it was noted whether the fish was in the cleared or undisturbed areas. Retrieval of concealed transmitters prior to data collection indicated that estimated positions were accurate to ± 0.5 m. Fish that could not be located within 1500 m of where they had been tagged over 4 days of attempted tracking were considered to have left the study area. Additional checks to locate them were then undertaken once each week after this 4-day period.

Analyses

Fish catch data collected from the same 50-m sampling site were not independent through time (Zar 1984); consequently, paired *t*-tests were used to compare total CPUE and changes in the total number of species collected before and after the experimental drain clearing for each of the different treatment groups. The same tests were used to analyse changes in the combined and individual CPUE of common bullies and giant kokopu. Paired *t*-tests were also used to compare CPUE before and after experimental drain clearing in uncleared and cleared sampling sites in the staggered treatment. To achieve this, it was necessary to treat data from different 50-m sampling sites in the same treatment site as independent. Catch data were not normally distributed, and both CPUE and species data were log-transformed ($\log_{x+1}$) to approximate normality. Paired *t*-tests were also used to compare the physical characteristics of each 350-m reach before and after macrophyte removal in each treatment group. All statistical analyses were carried out using SPSS Statistical Software version 20.0.0 (International Business Machines Corporation, Armonk, NY, USA).

Statistical comparisons of the movement patterns of radio-tagged individuals from the different treatment groups were not possible because of small sample sizes and low levels of individual replication. Instead, data are presented descriptively as in David & Closs (2003). The locations of radio-tagged fish after drain clearing in relation to where they were originally tagged were used to make inferences about the impacts of the different macrophyte removal techniques. The recorded positions of individual fish in relation to available macrophyte cover were plotted on a drawn-to-scale map of the study sites and used for inferences about habitat use in staggered sites (David & Closs 2003), and the potential benefits of this technique for giant kokopu management.

Results

Physico-chemical properties and habitat structure

After experimental drain clearing, mean water temperature decreased from 13.58 °C to 11.25 °C in

Table 2. Physico-chemical parameters measured in cleared, staggered and control sites before and after experimental macrophyte removal.

Parameters	Cleared			
	Before Mean ± SD	After Mean ± SD	t (df = 6)	P
Temperature (°C)	13.58 ± 0.46	11.25 ± 0.58	7.51	<0.001*
Relative conductivity (μS cm^{-1})	193.29 ± 14.70	211.74 ± 11.89	−1.19	0.278
Mean width (m)	1.36 ± 0.18	3.26 ± 1.51	1.84	0.261
Mean depth (cm)	24.25 ± 4.98	20.08 ± 3.37	8.80	0.07
Maximum depth (cm)	32.35 ± 4.36	27.54 ± 3.91	8.30	0.015*
Mean particle size (mm)	19.87 ± 10.92	11.9 ± 4.81	32.08	0.449
Macrophyte abundance	2.43 ± 0.21	1.14 ± 0.14	2.00	0.005*
Bryophyte abundance	1.32 ± 0.25	1.14 ± 0.14	0.52	0.253
Mat algae abundance	1.32 ± 0.25	1.05 ± 0.03	0.81	0.231
Filamentous algae	1.32 ± 0.21	1.07 ± 0.05	0.74	0.267
	Staggered			
	Before Mean ± SD	After Mean ± SD	t (df = 7)	P
Temperature (°C)	14.06 ± 0.56	11.35 ± 0.51	4.52	0.003*
Relative conductivity (μS cm^{-1})	198.67 ± 12.54	206.73 ± 10.61	−0.46	0.657
Mean width (m)	1.41 ± 0.10	1.60 ± 0.15	−2.09	0.075
Mean depth (cm)	26.51 ± 3.83	26.01 ± 2.05	0.19	0.850
Maximum depth (cm)	36.92 ± 4.07	33.81 ± 2.32	1.38	0.210
Mean particle size (mm)	17.64 ± 7.07	15.91 ± 5.41	0.52	0.618
Macrophyte abundance	2.26 ± 0.16	1.75 ± 0.09	4.49	0.003*
Bryophyte abundance	1.15 ± 0.16	1.06 ± 0.06	1.00	0.351
Mat algae abundance	1.03 ± 0.03	1.03 ± 0.03	–	–
Filamentous algae	1.34 ± 0.23	1.00 ± 0.00	1.51	0.173
	Control			
	Before Mean ± SD	After Mean ± SD	t (df = 7)	P
Temperature (°C)	11.72 ± 1.56	10.98 ± 0.56	0.42	0.682
Relative conductivity (μS cm^{-1})	208.04 ± 17.12	212.00 ± 12.03	−0.15	0.881
Mean width (m)	1.67 ± 0.21	2.45 ± 0.75	−0.99	0.351
Mean depth (cm)	29.40 ± 3.97	33.95 ± 4.41	−0.83	0.429
Maximum depth (cm)	38.82 ± 3.51	43.50 ± 5.39	−0.78	0.461
Mean particle size (mm)	26.75 ± 13.54	20.42 ± 7.81	0.50	0.605
Macrophyte abundance	2.71 ± 0.17	2.25 ± 0.25	3.41	0.011*
Bryophyte abundance	1.18 ± 0.10	1.12 ± 0.08	1.52	0.170
Mat algae abundance	1.09 ± 0.07	1.25 ± 0.14	−0.95	0.370
Filamentous algae	1.06 ± 0.06	1.00 ± 0.00	1.00	0.351

*Represents a significant difference ($P < 0.05$) in values observed before and after treatment.

cleared sites, from 14.06 °C to 11.35 °C in staggered sites and from 11.72 °C to 10.98 °C in control sites (Table 2). The observed difference in temperature before and after macrophyte removal was statistically significant in the cleared and staggered treatment groups but not in the control (paired t-tests, cleared $t_6 = 7.513$, $P < 0.001$; staggered $t_7 = 4.524$, $P = 0.003$; control $t_7 = 0.428$, $P = 0.682$). In all three treatments, relative conductivity, mean width, mean depth and mean substrate size did not differ significantly before and after drain clearing. Mean maximum depth was significantly shallower for cleared treatments, decreasing from 32.35 to 27.54 centimetres (paired t-test, $t_6 = 8.304$, $P = 0.015$), but depth did not change in staggered or control treatments (paired t-tests, staggered $t_7 = 1.38$, $P = 0.21$; control $t_7 = -0.78$, $P = 0.46$) (Table 2).

Mean macrophyte coverage was significantly reduced in both the treatment groups and the control group after drain clearing (paired t-tests, cleared $t_6 = 2.00$, $P = 0.005$; staggered $t_7 = 4.492$, $P = 0.003$; control $t_7 = 3.416$, $P = 0.\ 01$). The mean coverage score dropped from 2.43 to 1.14 at cleared sites, from 2.26 to 1.75 at staggered sites and from 2.71 to 2.25 at control sites (Table 2). There were no significant differences in the abundance of bryophytes, filamentous algae or mat algae before and after experimental drain clearing for any of the treatments.

Fish population surveys

In total, 1250 native fish were collected from the 23 sites, 682 prior to experimental drain clearing and 568 after. Common bully made up the majority of the total fish catch (N = 1023). Giant kokopu (N = 140) and longfin eel (*Anguilla dieffenbachii*) (N = 69) were also relatively common, while inanga (N = 14), banded kokopu (*Galaxias fasciatus*) (N = 2), redfin bully (*Gobiomorphus huttoni*) (N = 1) and shortfin eel (*Anguilla australis*) (N = 1) were only found occasionally. The number of species caught from each replicate ranged from zero to four and was not found to differ significantly between the before and after drain clearing sampling periods for any treatments (paired t-tests, cleared t_{27} = 1.148, P = 0.261; staggered t_{31} = 0.504, P = 0.618; control t_{31} = 1.082, P = 0.288).

After macrophyte removal, mean CPUE significantly decreased in cleared treatments from 0.45 to 0.22 fish per hour (t_{27} = 2.159, P = 0.040). Similarly, CPUE was significantly reduced in staggered treatments from 0.52 to 0.32 fish per hour (staggered t_{31} = 2.088, P = 0.045) (Fig. 3). CPUE did not differ before and after macrophyte removal for the control treatments (t_{31} = −0.747, P = 0.461) (Fig. 3). In staggered sites, mean total CPUE significantly decreased in cleared areas from 0.51 to 0.206 (t_{15} = 2.060, P = 0.049). Mean CPUE in uncleared areas did not differ before and after macrophyte removal (t_{15} = 0.760, P = 0.461) (Fig. 4).

The two most abundant species, common bully and giant kokopu, significantly decreased as a result of macrophyte removal, with a combined CPUE in cleared treatments decreasing from 0.48 to 0.21 fish per hour (t_{27} = 2.286, P = 0.030). In contrast, combined CPUE did not differ significantly in either the staggered treatment or the control (paired t-tests, staggered t_{31} = 1.915, P = 0.065; control t_{31} = −1.383, P = 0.176) (Table 3). After drain clearing, the CPUE of common bully only differed significantly in the cleared treatment, decreasing from 0.43 to 0.18 fish per hour (paired t-tests, cleared t_{27} = 2.104, P = 0.045; staggered t_{31} = 1.907, P = 0.066; control t_{31} = −1.764, P = 0.88) (Table 3). CPUE of giant kokopu did not differ significantly before and after drain clearing in either the treatments or the control (paired t-tests, cleared t_{27} = 1.311, P = 0.201; staggered t_{31} = 0.086, P = 0.932; control t_{31} = 1.014, P = 0.319) (Table 3).

Radiotelemetry

Of the 11 fish originally tagged, only three could not be located within the time frame of the study, all of which were from cleared treatments. Of the eight fish located at least once after drain clearing, five were from the cleared treatments, two were from staggered treatments and one was from a control treatment. Fish were tracked between the 10th of May and the 12th of June, with earliest contact lost on the 14th of May (Table 1). Four of the five fish that were originally tagged in the cleared treatments left after drain clearing. M127 and C156 moved into control sites, C136 moved into a staggered site and A296 moved upstream into an undisturbed area. In contrast, fish C226 was found to regularly use a completely cleared section of waterway at night (100% of night-time locations), but consistently moved upstream into a control site before dawn (100% of daytime locations) (Fig. 5a). This pattern was also seen in the movements of the two fish tagged in staggered sites. Both M299 and M266 used cleared sections of staggered sites at night (100% of night-time locations) and returned to uncleared sections during the day (100% of daytime locations) (Fig. 5b & c). The single fish tagged in a control site, M176, remained in that site following macrophyte removal at the treatment sites.

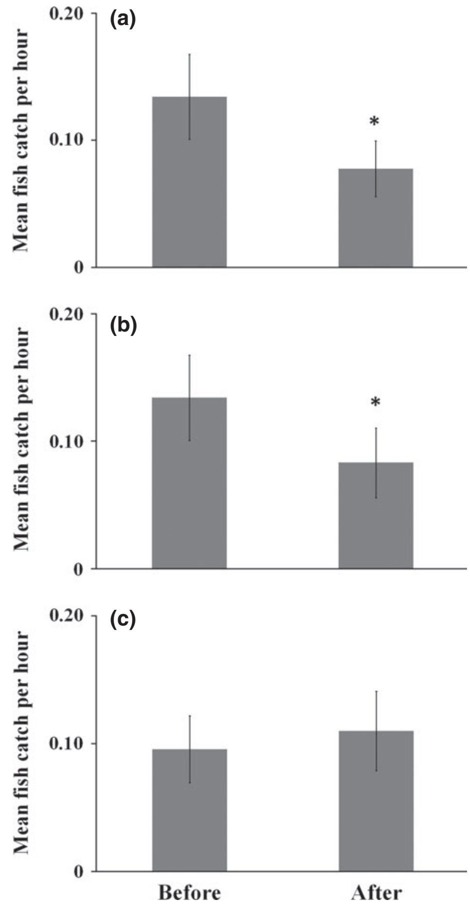

Fig. 3. Mean CPUE ($\log_{x+1}$) per 50 m before and after macrophyte removal in the three treatment groups (a = cleared; b = staggered; c = control). Error bars represent standard error from the mean. Statistically significant differences (P < 0.05) in CPUE before and after macrophyte removal are illustrated with an *.

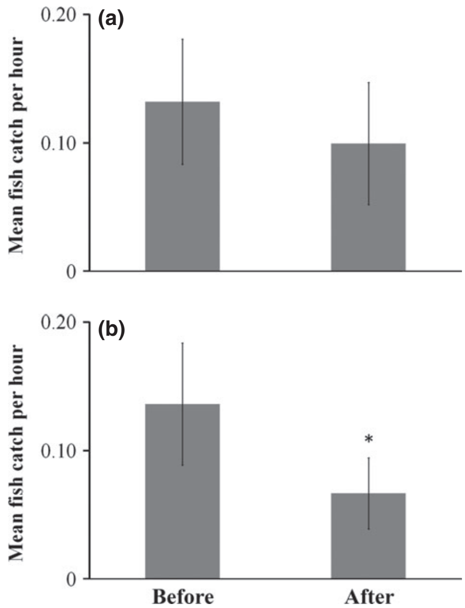

Fig. 4. Mean CPUE ($\log_{x+1}$) per 50 m before and after macrophyte removal in the staggered treatment group (a = uncleared sections; b = cleared sections). Error bars represent standard error from the mean. Statistically significant differences ($P < 0.05$) in CPUE before and after macrophyte removal are illustrated with an *.

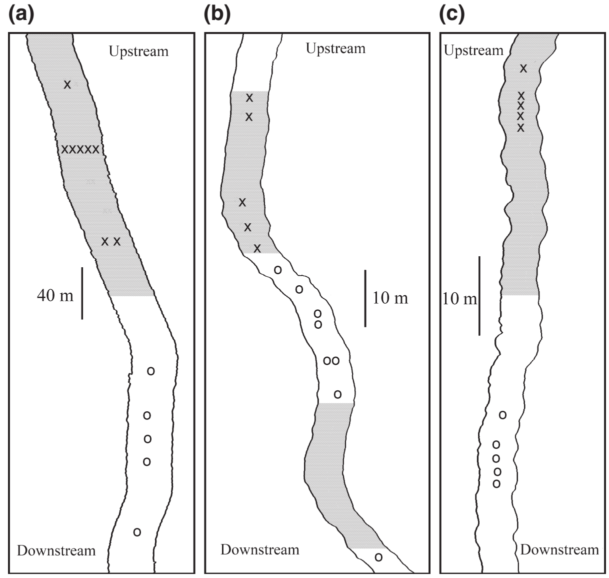

Fig. 5. Point in time locations for three individual fish (a = fish C226; b = fish M299; c = fish M266). X = day cover positions and O = night positions. Shaded areas represent sections of waterway from which macrophytes were not removed. For ease of interpretation, channel widths are not drawn to scale.

Discussion

The results of this study suggest that complete macrophyte removal can significantly reduce total CPUE of native fish in lowland New Zealand streams. The reduction in combined CPUE of common bully and giant kokopu after macrophyte removal was statistically more significant than the decreases observed in either species individually. This indicates that drain clearing has the potential to reduce the abundance of both species and that the lack of a significant change in giant kokopu abundance after macrophyte removal may be due to low statistical power associated with

Table 3. CPUE of common bully, giant kokopu and both species combined in cleared, staggered and control sites before and after experimental macrophyte removal.

Species	CPUE Before Mean ± SD	CPUE After Mean ± SD	t	P
	Cleared			
	CPUE Before Mean ± SD	CPUE After Mean ± SD	t(df = 27)	P
Common bully	0.4281 ± 0.132	0.181 ± 0.062	2.104	0.045*
Giant kokopu	0.050 ± 0.014	0.029 ± 0.012	1.311	0.201
Combined	0.478 ± 0.134	0.2104 ± 0.067	2.286	0.030*
	Staggered			
	CPUE Before Mean ± SD	CPUE After Mean ± SD	t(df = 31)	P
Common bully	0.463 ± 0.149	0.279 ± 0.147	1.907	0.066
Giant kokopu	0.031 ± 0.010	0.023 ± 0.009	0.086	0.932
Combined	0.494 ± 0.149	0.308 ± 0.148	1.915	0.065
	Control			
	CPUE Before Mean ± SD	CPUE After Mean ± SD	t(df = 31)	P
Common bully	0.150 ± 0.063	0.305 ± 0.115	−1.764	0.088
Giant kokopu	0.071 ± 0.018	0.054 ± 0.013	1.014	0.319
Combined	0.222 ± 0.063	0.360 ± 0.113	−1.383	0.176

*Represents a significant difference ($P < 0.05$) in values observed before and after treatment.

large variation within a small sample. Although there was no evidence to suggest that staggered macrophyte removal minimised the impacts on native fish abundance, our data suggest that uncleared areas act as refuges for large giant kokopu that may be less likely to leave streams cleared in this manner. These findings support our predictions that the complete clearing of streams negatively impacts freshwater fish populations and that these impacts can be minimised by limiting macrophyte removal to alternating sections of a waterway.

Our findings are of particular importance as they demonstrate the threat posed by current drain management practices to native fish communities throughout New Zealand's lowland waterways, and suggest that the implementation of the staggered drain clearing strategy in areas where the giant kokopu are found may prevent further losses of this already-threatened fish. Mechanical removal of macrophytes has been shown to lead to short-term decreases in fish abundance in both North American and British streams (Swales 1982; Serafy et al. 1994), with limited impact on the species diversity of the resident fish communities (Serafy et al. 1994). It is commonly accepted that mechanical excavation of macrophytes in New Zealand streams most likely has a negative influence on native fish (Hudson & Harding 2004), but the present study is the first to quantify this knowledge gap (Young et al. 2004).

A common problem encountered with past macrophyte removal studies is the difficulty isolating seasonal changes in abundance from treatment effects (Swales 1982; Armitage et al. 1994). In this case, however, CPUE did not differ between sampling periods in the undisturbed control reaches despite seasonal decreases in temperature and significant reductions in macrophyte cover. This suggests that decreased fish abundance in cleared and staggered reaches was caused by the physical process of macrophyte removal, the resulting changes in habitat structure or a combination of both these factors. The total loss of available plant cover is, most likely, a contributing factor in the reductions in fish abundance seen in the two treatment groups. Removing large quantities of aquatic vegetation increases predation of smaller fish (Mortensen 1977), reduces cover for adult fish (Swales 1982) and decreases food availability for both predatory and herbivorous fish (Swales 1982; Garner et al. 1996). Although staggered clearing still led to a significant decrease in total fish abundance, maintaining half of the available plant cover did mitigate at least some of the impacts of drain clearing. Undisturbed macrophyte beds acted as 'refuges' for giant kokopu and decreases in fish abundance were limited to the areas from which all available plant cover had been removed. Significant reductions in macrophyte coverage, however, had no impact on CPUE in the control reaches. Subsequently, it is likely that changes in fish abundance in cleared and staggered reaches were influenced by a number of factors associated with drain clearing other than reduced plant biomass.

Disturbance caused by the physical process of macrophyte removal may also be partly responsible for the results observed in this study. Mechanical macrophyte removal causes physical and chemical changes in the water column that can negatively impact the resident fauna (Brookes 1988; Hudson & Harding 2004). Drain clearing has been reported to temporarily increase sediment suspension (Hudson & Harding 2004; Young et al. 2004), which interferes with normal respiration in fish by clogging the gills (Bruton 1985), and limits the feeding success of predatory species by increasing turbidity and invertebrate drift (Ryan 1991; Wood & Armitage 1997). Although these factors were not measured in the current study owing to logistical constraints, high levels of suspended sediment were observed during experimental drain clearing and it is possible that this led to the short-term decreases in CPUE observed in the treated reaches. From the physico-chemical variables examined, only water temperature changed significantly in both treatment groups, but this was unlikely to explain the observed changes in fish community. Decreases in temperature were probably the result of seasonal changes in climate and were unlikely to be associated with macrophyte removal. Temperatures in the treatment groups and the control were relatively similar after drain clearing, and the larger decreases in temperature seen in the cleared and staggered areas were a reflection of the relatively low temperatures recorded in the undisturbed reaches prior to macrophyte removal. Determining the physico-chemical drivers behind changes in the fish community following drain clearing is vital, and future research in this area is required to further develop drain management strategies.

The limited change in species richness we observed following drain clearing may be a reflection on the long-term removal history of the study area rather than the effects of short-term macrophyte removal. Regular excavation of waterways has been carried out in the Waituna catchment since at least the 1960s (Johnson & Partridge 1998), which may have generated localised fish communities that are resistant to macrophyte removal in areas that are frequently cleared. If so, fish assemblages in developed waterways, such as those where the current study was carried out, may only consist of species that are at least partially resistant to this form of disturbance (Orrego et al. 2009). Determining how the distribution of native fish species relates to current and historical drain clearing activities in catchments like the Waituna may provide a better understanding of the community level impacts of macrophyte removal.

The design of this study did not allow measurement of the long-term impacts of drain clearing on the fish community. The experimentally cleared reaches included in this study were bordered by undisturbed macrophyte beds. Recovery time after disturbance has been shown to decrease when undisturbed areas from which displaced animals can recolonise are close (Niemi et al. 1990; Reice et al. 1990). Therefore, the fish populations in our study sites would be expected to recover much more rapidly than they would under real-world conditions where macrophyte removal is typically carried out on a much larger scale (Resh et al. 1988). Future research focussed on monitoring resident fish populations following routine large-scale drain clearing operations is needed to measure long-term community responses to macrophyte removal accurately.

Although staggered clearing still resulted in significant reductions in native fish abundance, we found that this technique may reduce the effects of drain clearing on adult giant kokopu. Large giant kokopu are generally nocturnal and seek cover when not active (Whitehead et al. 2002; David & Closs 2003). The radiotelemetry data collected in this study suggest that the response of giant kokopu to drain clearing is heavily dependent on the availability of macrophytes, and large individuals will leave completely cleared areas when diurnal concealment is no longer possible. In contrast, staggered macrophyte removal may preserve enough cover and eliminate the need for large giant kokopu to leave treated waterways. Furthermore, we found this technique may actually benefit giant kokopu by increasing the availability of desirable nocturnal feeding habitat (David & Closs 2003). Three individuals tracked in this study regularly moved from the densely vegetated areas in which they sheltered during the day to cleared areas at night. Giant kokopu prefer to feed in open habitats (David & Closs 2003; Hansen & Closs 2009), and by creating this habitat through staggered clearing, the numbers and condition of resident fish may remain unchanged despite reductions in food availability as a result of macrophyte removal (Garner et al. 1996). Although the idea of leaving undisturbed refuges when clearing macrophytes from streams has been suggested in the past (Swales 1982; Armitage et al. 1994; Garner et al. 1996; Kaenel et al. 1998; Aldridge 2000; Hudson & Harding 2004), the actual benefits of this technique for individual fish species have not been quantified until now. Further trials should be carried out in other systems, both in New Zealand and overseas, to determine whether this approach can benefit other threatened fish species found in regularly cleared lowland streams.

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