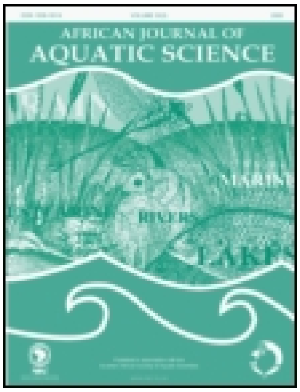

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OLF Weyl [a b], BR Ellender [c], DJ Woodford [a b] & MS Jordaan [d]

[a] South African Institute for Aquatic Biodiversity (SAIAB), Private Bag 1015, Grahamstown, 6140, South Africa

[b] Centre for Invasion Biology, SAIAB, Private Bag 1015, Grahamstown, 6140, South Africa

[c] Department of Ichthyology and Fisheries Science, Rhodes University, PO Box 94, Grahamstown, 6140, South Africa

[d] CapeNature Scientific Services, Private Bag X5014, Stellenbosch, 7599, South Africa

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Fish distributions in the Rondegat River, Cape Floristic Region, South Africa, and the immediate impact of rotenone treatment in an invaded reach

OLF Weyl[1,2]*, BR Ellender[3], DJ Woodford[1,2] and MS Jordaan[4]

[1] *South African Institute for Aquatic Biodiversity (SAIAB), Private Bag 1015, Grahamstown 6140, South Africa*
[2] *Centre for Invasion Biology, SAIAB, Private Bag 1015, Grahamstown 6140, South Africa*
[3] *Department of Ichthyology and Fisheries Science, PO Box 94, Rhodes University, Grahamstown 6140, South Africa*
[4] *CapeNature Scientific Services, Private Bag X5014, Stellenbosch 7599, South Africa*
* *Corresponding author, e-mail: o.weyl@saiab.ac.za*

Alien fishes are considered the most serious threat to native headwater stream fishes in South Africa. A 4 km reach of the Rondegat River is the first section of a South African river to be rehabilitated through the attempted removal of alien fish by using the piscicide rotenone. The objectives of the current study were to establish the distribution and relative abundance of native and alien fish prior to treatment, and to assess the immediate impact of the treatment on the fish population. Forty-three sites were sampled using backpack electrofishing, snorkel transects and underwater video analysis. In the invaded lower reaches, native *Labeobarbus capensis* was detected only at very low densities, while three other native fish species were not detected. Alien fish were not detected above a barrier waterfall 5 km upstream of the river's confluence with a reservoir. The fish density of 97 fish per 100 m^2 in non-invaded reaches was more than an order of magnitude higher than that of 7 fish per 100 m^2 in the invaded reach. A total of 470 *Micropterus dolomieu* and 139 *L. capensis* were removed from a 4 km treatment zone during the rotenone operation. No fish were detected in this area after the rotenone treatment.

Keywords: *Austroglanis gilli*, *Barbus calidus*, Clanwilliam Dam, *Galaxias zebratus*, invasion, *Lepomis macrochirus*, non-native, smallmouth bass, *Tilapia sparrmanii*

Introduction

Native fish populations in the Cape Floristic Region (CFR) are characterised by high diversity, endemism and geographic isolation (Linder et al. 2010). This makes them vulnerable to human actions such as water abstraction, canalisation, erosion, siltation and the introduction of alien fishes (Tweddle et al. 2009). The combined effects of these impacts have resulted in the extirpation of native fishes from invaded reaches of several rivers (Woodford et al. 2005, Ellender et al. 2011, Swartz and Tweddle 2011), resulting in decreased distributional ranges and genetic isolation (Swartz et al. 2004). Many CFR fishes are now IUCN red-listed as Critically Endangered, Endangered or Vulnerable (Tweddle et al. 2009). As in other countries (Abell et al. 2007), headwaters are considered of high conservation priority in the CFR (Marr et al. 2012). The main threat to native fishes in these streams is predation by alien fishes (Ellender et al. 2011, van Rensburg et al. 2011).

The Rondegat River, a small perennial tributary of the Olifants River which flows into Clanwilliam Dam (reservoir), is a good example of a river where alien fish predation has impacted on native fishes. Historically, the Rondegat River contained six native species, including Clanwilliam sawfin *Barbus serra* Peters 1864, Clanwilliam yellowfish *Labeobarbus capensis* (A. Smith 1841), Clanwilliam sandfish *Labeo seeberi* Gilchrist and Thompson 1911, fiery redfin minnow *Pseudobarbus phlegethon* (Barnard 1938), Clanwilliam redfin minnow *Barbus calidus* Barnard 1938, Clanwilliam rock catfish *Austroglanis gilli* (Barnard 1943) and Cape galaxias *Galaxias zebratus* Castelnau 1861 (van Rensburg 1966, Woodford et al. 2005). As a result of formal stocking programmes in its catchment, the dam contains a variety of alien fishes including largemouth bass *Micropterus salmoides* (Lacepède 1802), spotted bass *Micropterus punctulatus* (Rafinesque 1819) and smallmouth bass *Micropterus dolomieu* (Lacepède 1802). Fish surveys conducted in 1998 and 2004 showed that *M. dolomieu* had invaded the lower sections of the Rondegat River up to a small waterfall c. 5 km from its inflow into Clanwilliam Dam (Woodford et al. 2005). In the invaded section, predation by *M. dolomieu* had extirpated both the native minnow species and had altered the invertebrate community structure (Lowe et al. 2008).

In 2005, CapeNature, the provincial conservation authority for the Western Cape, initiated a project to rehabilitate selected rivers by removing alien fishes using the piscicide rotenone (Marr et al. 2012). The Rondegat River was considered an ideal site for rehabilitation because a water abstraction weir, located c. 1 km upstream of the dam, had effectively isolated the *M. dolomieu* population in the section of river between the waterfall and the weir (Marr et al. 2012). The primary goal of the Rondegat project was to increase the amount of habitat available to native fishes by removing *M. dolomieu*, and therefore on 29 February 2012 CapeNature treated the section of river between the waterfall and the weir with rotenone.

The long-term success of the Rondegat fish eradication project will depend firstly on the ability of the piscicide to eradicate the alien fish, and secondly on the ability of native fish to re-colonise the river after treatment. Quantitative monitoring of the immediate and long-term effects of fish eradication is critical, so that the effectiveness of the treatment method can be ascertained. The objectives of the current study were therefore to establish a baseline of native and alien fish distribution and relative abundance in both invaded and non-invaded zones, against which recovery can be measured, and to determine the immediate impact of the rotenone treatment.

Methods

Study area

The Rondegat River, a 5 m wide, clear, perennial river, flows 25 km from its source into the 1 043 ha Clanwilliam Dam (Figure 1). Lowe et al. (2008) provided a detailed description of the physical characteristics and invertebrate fauna of this river, which flows through relatively pristine fynbos vegetation in its upper reaches and through agricultural pastures in its lower reaches. The dense alien riparian vegetation that dominated the middle reaches of the river in 2004 (Woodford et al. 2005, Lowe et al. 2008) was cleared during catchment rehabilitation. Three potential barriers to fish movement are present in the system. The first, a small 1 m high waterfall followed by a long bedrock cascade (32°15.365' S, 18°57.135' E) is located 625 m above the dam, and a 2 m high water abstraction weir (32°15.536' S, 18°57.812' E) is located 365 m further upstream. The upper limit of smallmouth bass distribution in the Rondegat River is the 1.3 m high Rooidraai waterfall, located 4 km upstream of the weir (Figure 1).

Field methods

Forty-three sites were sampled, 17 in the non-invaded reach upstream of Rooidraai waterfall and 26 sites in the invaded reach downstream of the waterfall. Within the invaded reach, 17 sites were sampled in the 4 km 'treatment area' between Rooidraai waterfall and the weir. Sampled habitats included riffles, runs and pools. At each site, temperature, conductivity and pH were measured using a Hanna HI98129 Combo pH and electrical conductivity meter and turbidity (NTU) was measured using a Hanna HI 98703 turbidimeter (HANNA Instruments Inc., Woonsocket, USA). To determine the area and water volume sampled, the length (±0.1 m) of each habitat was measured, followed, depending on habitat, by between three and five equally-spaced width measurements (±0.1 m). On each width transect, three depths (±0.1 m)

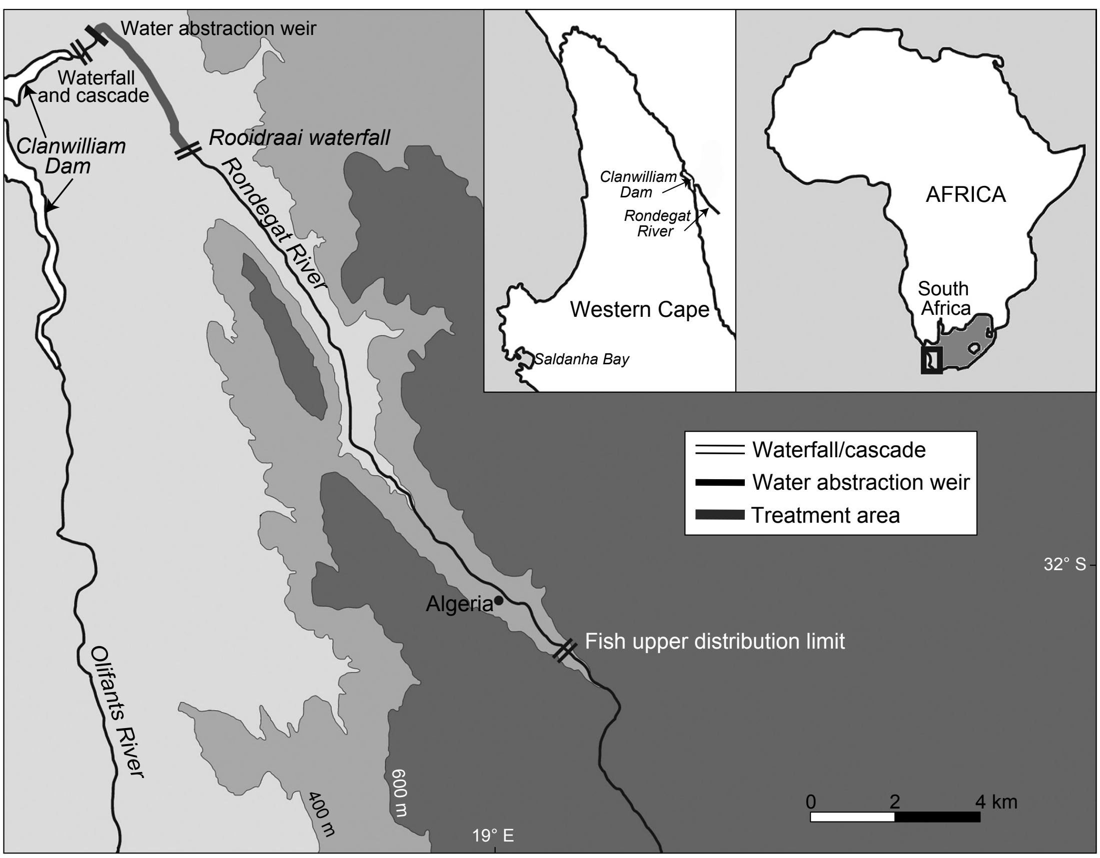

Figure 1: Map of the study area, showing treatment area on the Rondegat River in relation to natural and artificial barriers

were measured, the outer two each being 0.2 m from the left- and right-hand river banks and the third measurement taken midstream.

Pre-treatment fish surveys were conducted from 15 to 17 February 2011 and from 24 to 27 February 2012. The timing of these surveys at the end of summer fell within a low-flow period, during which sampling was considered most effective, allowing for better replicability on subsequent surveys. In the treatment area, a post-treatment survey was conducted, 24 h after the rotenone application, on 01 March 2012. Three sampling methods, including backpack electrofishing, snorkelling transects and underwater video analysis, were used to assess the fish community for species composition, population structure and relative abundance. Habitat type and site characteristics determined the sampling method employed at each site. Whilst electrofishing at 30 sites was limited to shallower sites, <1 m deep, snorkelling at 40 sites, and underwater video analysis at 37 sites, were used in a wide range of habitats. The locations, dimensions and sampling method used at each site during each survey are given in the Appendix.

Snorkelling transects were conducted following the method described by Ellender et al. (2011), whereby fish were counted during two consecutive passes and averaged to give an estimate for the number of fish present in the pool. During these fish counts the lengths of *L. capensis* were also recorded in the categories <15, 15–30 and >30 cm. Length was estimated only on the first pass, to avoid measuring the same fish twice.

A Samus© 725G backpack electrofisher was used, with settings standardised at 0.3 ms duration and 90 Hz frequency. As a result of the low conductivities of 11–70 µS s^{-2}, electrofishing was conducted downstream into a fine-meshed block-net. At each sample site, three passes were conducted with the electrofisher. Fish captured during each pass were placed in separate buckets and later identified to species level, counted, measured to the nearest 1 mm fork length (FL) and released. The total number of fish sampled during three passes was taken as being representative of the number of fish in that sampling site.

Underwater videoing was conducted using a GoPro® HD Hero® high-definition camera fitted with a corrective lens for full use underwater. Camera settings were standardised: field of view = 127°, resolution (Full HD) = 1 080 p (1 920 × 1 080), frames per second = 30 NTSC, 25 PAL. Methods for camera placement, time of observation and analysis followed those recommended by Ellender et al. (2012). The camera was deployed at each site for 30 minutes, with the first five minutes regarded as an acclimation period for conditions to return to normal in the sample pool following camera deployment, and therefore excluded from the subsequent analysis.

Collection of fishes during eradication exercise

The fish eradication was implemented by CapeNature in two phases: (1) a fish rescue operation conducted from 04 to 07 February 2012, when fish in the treatment area were caught using fyke nets and by angling; and (2) the rotenone treatment on 29 February. The rotenone treatment was conducted according to standard operating procedures (Finlayson et al. 2000). Rotenone was applied to the river using a series of seven drip stations, spaced at approximately 1-hour water-travel time intervals, to maintain the recommended treatment concentration of 1 mg l^{-1} CFT Legumine® (i.e. 5% rotenone) for a 6-hour treatment period (as recommended by BJ Finlayson, California Department of Fish and Game, pers. comm.). During the rotenone treatment, all dead fish were collected by 15 volunteers who patrolled the entire 4 km treated reach of the river. All fish caught or collected during both phases of the process were identified, enumerated, measured and weighed to the nearest 0.1 g.

Analysis

In all analyses, each site sampled was treated as a replicate and all tests were conducted at a significance level of $p < 0.05$. To compare the efficacy of different sampling methods, a detection rate (i.e. sample sites with positive records/all sample sites) was calculated for each species. Differences between methods were assessed by 2 × 3 methods χ^2 contingency analysis. To test for differences in fish abundance between surveys and between invaded and non-invaded sites for *L. capensis*, the only species that occurred in both reaches, snorkel survey and electrofishing fish counts were converted to densities of fish per m^2 of habitat sampled and compared using the non-parametric Mann-Whitney *U*-test. Underwater videoing lacks a spatial dimension and therefore the MaxN index, which is the maximum number of individuals for each species visible in the field of view simultaneously during a 25-minute videoing session, was used as a measure of relative abundance (Ellender et al. 2012). Estimates of fish density obtained in the treatment area using snorkel transects were compared to the fish densities estimated from the fish rescue and rotenone treatments using a 2-tailed *t*-test.

Results

Fish distribution and abundance

A summary of the morphological and edaphic characteristics of the pools sampled is presented in Table 1. Site-specific presence and absence data for all fish species are provided in Table 2. Detection rates and minimum fish densities estimated from electrofishing and snorkelling, and relative abundance estimated using underwater video analysis for the eight fish species sampled during the surveys, are summarised in Table 3.

Table 1: Summary of morphological and edaphic characteristics of Rondegat River sites sampled during 2011 and 2012 fish surveys. SD = standard deviation

Character	Min.	Max.	Average	SD	*n*
Length (m)	3.70	38.00	14.17	7.06	43
Width (m)	2.00	10.81	4.77	1.85	43
Depth (m)	0.19	0.91	0.44	0.17	43
Surface area (m^2)	10.00	369.85	73.72	68.79	43
Volume (m^3)	2.66	337.80	37.31	55.66	43
pH	6.13	8.23	7.57	0.39	51
Temperature (°C)	19.5	32.0	25.1	3.3	87
Turbidity (NTU)	0.54	4.76	1.78	0.68	66
Conductivity (µS cm^{-1})	14	79	45.20	16.06	88

Detection rates were significantly dependent on reach (invaded and non-invaded), species and methods. The native *A. gilli*, *B. calidus* and *P. phlegethon* were never detected in the invaded zone. *Labeobarbus capensis* were detected in all zones, but at very low detection rates in the invaded zone. Alien *M. dolomieu* were never detected above Rooidraai waterfall, and *Lepomis macrochirus* and *Tilapia sparrmanii* were detected only in the two lowest pools sampled in the invaded reach. In the non-invaded zone, *A. gilli* were detected at all sites by using electrofishing, but were detected only incidentally by using visual survey methods. For the minnows, the detection rates of 0.83–0.93 for *B. calidus* were high for all methods but, for the less abundant *P. phlegethon*, detection rates were highest using electrofishing (0.5 vs 0.21–0.36), due to difficulties in their identification underwater. Detection rates for *L. capensis* differed between invaded and non-invaded zones. In the non-invaded zone, detection rates using electrofishing were comparable to those from snorkel transects and from underwater video analysis, but in the invaded zone estimates from visual methods were consistently higher than those obtained from electrofishing. Detection rates for alien *M. dolomieu* were lowest (0.28) using electrofishing and highest (0.85) using snorkel surveys.

Native *A. gilli*, *B. calidus* and *P. phlegethon* occurred only in the non-invaded zone (Table 2). *Labeobarbus capensis* density was significantly lower in the invaded zone. Minimum fish densities estimated using snorkelling (97 fish per 100 m^2) and electrofishing (82 fish per 100 m^2) in the

Table 2: Presence (1) and absence (0) of seven fish species at 43 sites on the Rondegat River. Presence/absence was determined using electrofishing, snorkel surveys and underwater video surveys conducted on 15–17/02/2011 and 24–27/02/2012. Z = Zone; N = not invaded by *M. dolomieu*; T = invaded sites treated with rotenone; B = invaded sites below the rotenone treatment zone. Fish species: AG = *Austroglanis gilli*; BC = *Barbus calidus*; PP = *Pseudobarbus phlegethon*; LC = *Labeobarbus capensis*; MD = *Micropterus dolomieu*; TS = *Tilapia sparrmanii*; LM = *Lepomis macrochirus*

Site #	Z	Coordinates: Latitude	Coordinates: Longitude	AG	BC	PP	LC	MD	TS	LM
1	N	32°22.536' S	19°03.890' E	1	1	1	0	0	0	0
2	N	32°22.534' S	19°03.842' E	1	1	1	1	0	0	0
3	N	32°22.525' S	19°03.789' E	1	1	1	1	0	0	0
4	N	32°22.321' S	19°03.444' E	0	1	0	1	0	0	0
5	N	32°22.301' S	19°03.411' E	1	1	1	1	0	0	0
6	N	32°22.237' S	19°03.258' E	0	1	0	1	0	0	0
7	N	32°22.219' S	19°03.191' E	0	1	1	1	0	0	0
8	N	32°17.653' S	18°59.749' E	0	1	0	1	0	0	0
9	N	32°17.628' S	18°59.731' E	1	1	0	1	0	0	0
10	N	32°17.340' S	18°59.477' E	1	0	0	1	0	0	0
11	N	32°17.327' S	18°59.470' E	1	1	1	1	0	0	0
12	N	32°17.316' S	18°59.459' E	1	1	1	1	0	0	0
13	N	32°17.311' S	18°59.448' E	1	1	1	1	0	0	0
14	N	32°17.080' S	18°59.246' E	1	1	1	1	0	0	0
15	N	32°17.067' S	18°59.244' E	1	1	1	1	0	0	0
16	N	32°16.657' S	18°58.596' E	1	1	0	1	0	0	0
17	N	32°16.657' S	18°58.596' E	0	0	0	1	0	0	0
18	T	32°16.645' S	18°58.580' E	0	0	0	1	1	0	0
19	T	32°16.645' S	18°58.580' E	0	0	0	1	1	0	0
20	T	32°16.632' S	18°58.563' E	0	0	0	0	1	0	0
21	T	32°16.623' S	18°58.558' E	0	0	0	1	0	0	0
22	T	32°16.587' S	18°58.505' E	0	0	0	1	1	0	0
23	T	32°16.567' S	18°58.479' E	0	0	0	1	1	0	0
24	T	32°16.560' S	18°58.475' E	0	0	0	0	1	0	0
25	T	32°16.526' S	18°58.467' E	0	0	0	1	1	0	0
26	T	32°16.507' S	18°58.459' E	0	0	0	1	1	0	0
27	T	32°16.476' S	18°58.427' E	0	0	0	0	1	0	0
28	T	32°15.672' S	18°57.936' E	0	0	0	0	1	0	0
29	T	32°15.615' S	18°57.907' E	0	0	0	0	1	0	0
30	T	32°15.556' S	18°57.843' E	0	0	0	0	1	0	0
31	T	32°15.549' S	18°57.833' E	0	0	0	0	1	0	0
32	T	32°15.541' S	18°57.823' E	0	0	0	0	1	0	0
33	T	32°15.533' S	18°57.814' E	0	0	0	0	1	0	0
34	T	32°15.536' S	18°57.812' E	0	0	0	0	1	0	0
35	T	32°15.192' S	18°57.203' E	0	0	0	0	1	0	0
36	B	32°15.201' S	18°57.186' E	0	0	0	1	1	0	0
37	B	32°15.327' S	18°57.191' E	0	0	0	0	1	0	0
38	B	32°15.332' S	18°57.194' E	0	0	0	0	1	0	0
39	B	32°15.353' S	18°57.161' E	0	0	0	0	1	0	0
40	B	32°15.369' S	18°57.116' E	0	0	0	0	0	0	0
41	B	32°15.357' S	18°57.150' E	0	0	0	0	1	0	0
42	B	32°15.461' S	18°57.010' E	0	0	0	0	1	1	1
43	B	32°15.516' S	18°56.886' E	0	0	0	0	1	1	1

Table 3: Abundance estimates for native *Austroglanis gilli* (AG), *Barbus calidus* (BC), *Pseudobarbus phlegethon* (PP) and *Labeobarbus capensis* (LC), and for non-native *Micropterus dolomieu* (MD), *Lepomis macrochirus* (LM) and *Tilapia sparrmanii* (TS), in invaded and non-invaded reaches of the Rondegat River, derived from backpack electrofishing, snorkel surveys and underwater video surveys in 2011 and 2012. SE = standard error, DR = detection rate (sites with species present ⁄ all sites), *n* = sample size

Species	Non-invaded: Density (fish 100 m^{-2}) Mean	SE	DR	*n*	Invaded: Density (fish 100 m^{-2}) Mean	SE	DR	*n*
Backpack electrofishing								
AG	19.26	4.34	1.00	12	0	0	0	18
BC	42.16	26.34	0.83	12	0	0	0	18
PP	5.42	2.01	0.50	12	0	0	0	18
LC	15.63	5.31	0.75	12	0	0	0	18
MD	0	0	0	12	0.41	0.20	0.28	18
LM	0	0	0	12	0	0	0	18
TS	0	0	0	12	0	0	0	18
Snorkel transect								
AG	0.08	0.08	0.07	14	0	0	0	26
BC	58.52	14.45	0.86	14	0	0	0	26
PP	3.11	2.38	0.21	14	0	0	0	26
LC	34.93	9.59	0.71	14	0.46	0.23	0.19	26
MD	0	0	0	14	2.17	0.40	0.85	26
LM	0	0	0	14	0.07	0.06	0.08	26
TS	0	0	0	14	4.72	4.40	0.08	26
	MaxN Mean	SE	DR	*n*	MaxN Mean	SE	DR	*n*
Underwater video								
AG	0.14	0.12	0.14	14	0	0	0	22
BC	6.91	1.39	0.93	14	0	0	0	22
PP	1.09	0.74	0.36	14	0	0	0	22
LC	6.09	1.39	0.93	14	0.76	0.46	0.27	22
MD	0	0	0	14	0.94	0.18	0.73	22
LM	0	0	0	14	3.0	1.40	0.09	22
TS	0	0	0	14	6.0	1.20	0.09	22

non-invaded zone were more than an order of magnitude higher than that of 7 fish per 100 m^2 from the snorkel survey for the whole invaded zone (Table 3).

The population size structures of alien and native fishes sampled using electrofishing in the non-invaded zone and those collected during fish removals in the invaded zone are shown in Figure 2. The invasive *M. dolomieu* population was dominated by fish smaller than 20 cm FL. The *L. capensis* population differed between invaded and non-invaded zones, with that in the non-invaded reaches comprising both juvenile and adult fish (5–35 cm FL), whereas in the invaded zone the population comprised almost entirely adults larger than 20 cm FL (Figure 2).

Immediate impact of rotenone treatment

In January 2012, prior to the rotenone treatment, 45 *L. capensis* and 85 *M. dolomieu* were removed from the treatment zone by fyke netting, electrofishing and angling.

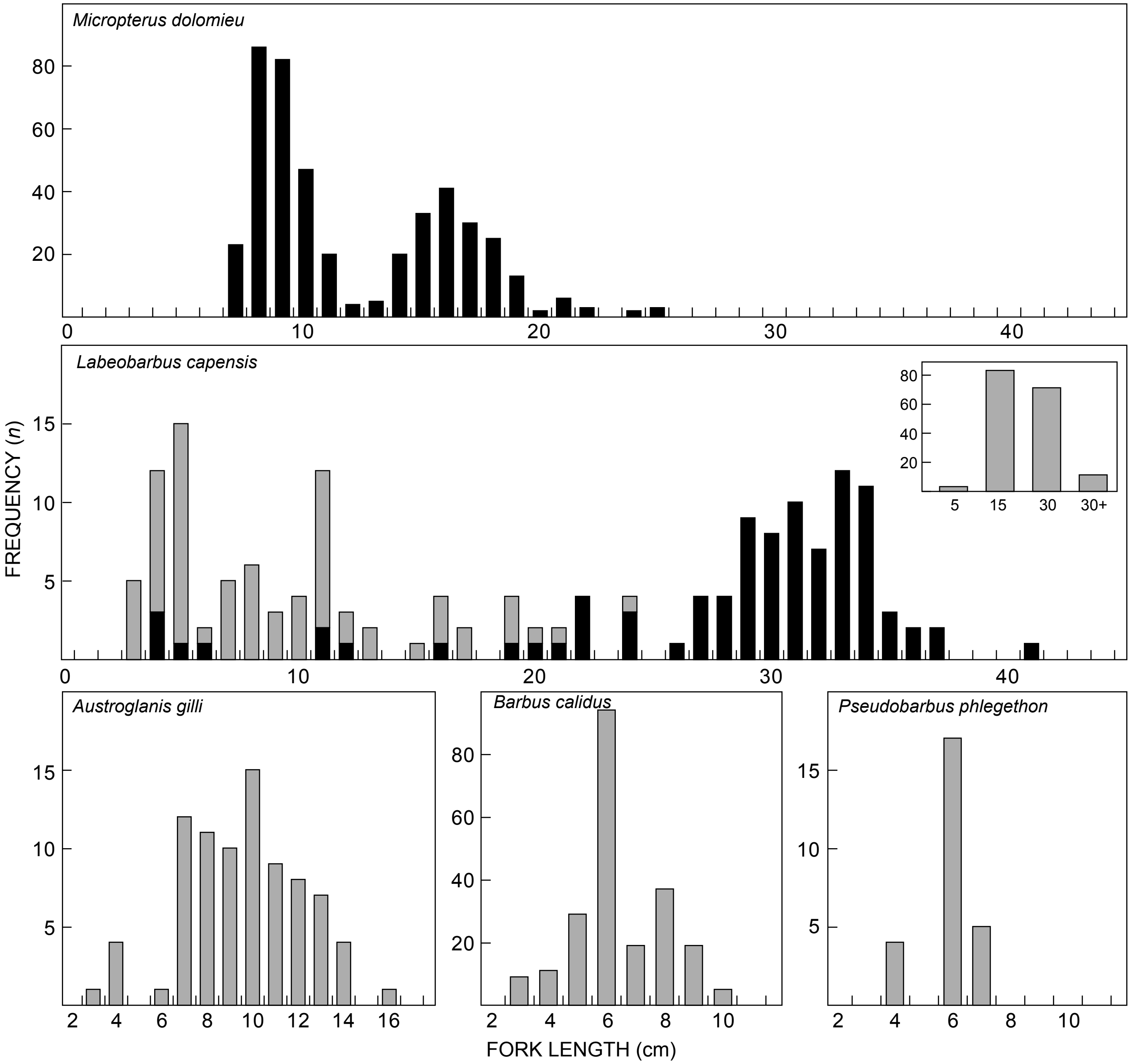

Figure 2: Fork length (cm) frequency of non-native *Micropterus dolomieu* and native *Labeobarbus capensis*, *Austroglanis gilli*, *Barbus calidus* and *Pseudobarbus phlegethon* in invaded (black bars) and non-invaded (grey bars) zones in the Rondegat River. In the invaded zone, length frequencies are based on the total population of fish removed during rotenone treatment in 2012. In the non-invaded zone, length structure was estimated from fish measured during electrofishing surveys in 2011 and 2012. Insert in the *L. capensis* length frequency distribution shows length structure estimates from 2011/2012 snorkel surveys, demonstrating that fish larger than the sizes sampled by electrofishing were present in this region

During the rotenone treatment in February 2012, 94 *L. capensis* and 385 *M. dolomieu* were collected here. The total biomass of fish removed from the 4 km treatment section was 63 kg, of which 27.2% (17.175 kg) was *M. dolomieu* and 72.8% (45.824 kg) was *L. capensis*. At a mean river width of 4.77 m in the 4 km long treatment area, fish density here was 3.0 fish per 100 m^2, and biomass was 330 g per 100 m^2.

In the treatment zone, snorkel survey estimates of fish density and underwater video relative abundance estimates decreased between 2011 and 2012 and, after the rotenone treatment, no fish were detected in the treatment area (Figure 3). Densities of *M. dolomieu* did not differ significantly between the 2011 and 2012 pre-treatment surveys, but those for *L. capensis* differed significantly (Mann-Whitney U, $p < 0.05$).

Snorkel survey estimates of *M. dolomieu* density did not differ from those estimated by fish removal, including both rescue and rotenone treatment, during either the 2011 (t-test, $t = 0.149$, df = 14, $p = 0.88$) or 2012 ($t = 1.01$, df = 16, $p = 0.38$) pre-treatment survey. *Labeobarbus capensis* density estimates for 2011 did not differ from fish removal estimates (t-test, $t = 0.399$, df = 14, $p = 0.69$), but the 2012 pre-treatment estimates of were significantly lower ($t = 10.4$, df = 16, $p < 0.001$).

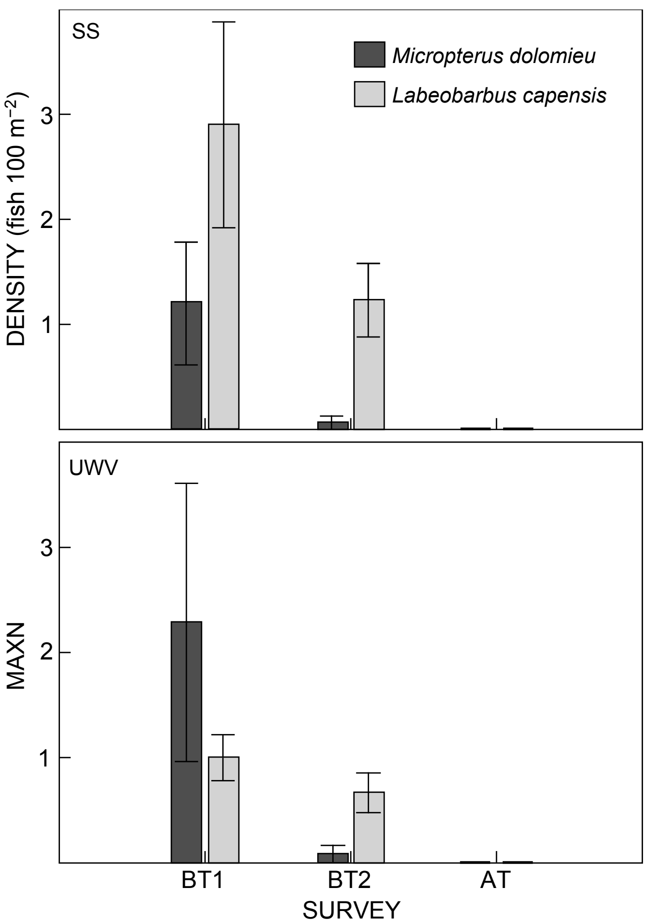

Figure 3: Estimates of mean fish density from snorkel surveys (SS) in the rotenone-treated area of the Rondegat River, and of mean relative abundance (MaxN) from underwater video analysis (UWV), during two before-treatment surveys and one after-treatment survey. BT1 = before-treatment survey on 15/02/2011–17/02/2011; BT2 = before-treatment survey on 24/02/2012–27/02/2012; AT = after-treatment survey on 01/03/2012. Error bars denote 1 SE

Discussion

Monitoring fish communities in small, clear rivers such as the Rondegat needs to take into consideration the influence of habitat, target species behaviour and the conservation status of the fishes that are to be monitored. For example, backpack electrofishing can only be used in shallower sections of the river and is often ineffective at low conductivities (Zalewski 1986). Snorkelling and underwater videoing, while useful in a variety of habitats, are more likely to detect diurnally active mid-water fishes such as minnows and yellowfish than nocturnal bottom-dwelling catfishes that hide in rocky crevices during the day. In the CFR, monitoring programmes also need to take into consideration the conservation status of the fishes that are to be monitored. Of the four native species recorded in the Rondegat River, three are classed as Vulnerable and *P. phlegethon* as Endangered (Tweddle et al. 2009). As stress, injury and mortalities are considered unavoidable consequences of electrofishing (Snyder 2003), less destructive techniques have been advocated for rare and endangered species (Gray et al. 2002, Hickey and Closs 2006, Ellender et al. 2012). Underwater video analysis has already been demonstrated to be a viable alternative to electrofishing for estimating relative abundance of other CFR fishes (Ellender et al. 2012).

To account for differences in sampling efficacy, we used three different sampling methods, each of which was useful for a specific purpose. Backpack electrofishing, for example, was used in the shallower riffle and run sections of the river. This method, included primarily to test whether non-destructive sampling methods produced similar abundance estimates, was particularly effective at determining the presence and relative abundance of the nocturnal catfish *A. gilli*. This catfish hides among rocks during the day and was therefore not adequately detected using snorkel transects and underwater video. Electrofishing was also considered an adequate method for detecting the presence of native minnows and *L. capensis* in the non-invaded reach of the river, where these fish were abundant. In the invaded reach, this method did not adequately sample *M. dolomieu* and *L. capensis*. Avoidance behaviour was observed in both species, and the large adult *L. capensis* that inhabited this zone generally occurred in pools that were too deep to electrofish effectively. With the exception of their inability to detect catfishes, snorkelling surveys and underwater videoing were both considered good methods for assessing the presence of non-catfish species. Underwater videoing, however, had the disadvantage of lacking a spatial component and thus abundance estimates from this technique are useful only for estimating relative fish abundance.

Despite differences in detection rates between methods, all methods demonstrated that alien *M. dolomieu* were present only below the Rooidraai waterfall barrier, but that other alien fish were limited to sites below the first waterfall barrier, situated close to Clanwilliam Dam. These results were consistent with observations from earlier surveys (Woodford et al. 2005) and indicate that the lower waterfall has been a barrier to *L. macrochirus* for at least eight years, and that Rooidraai waterfall continues to be the upper invasion barrier for *M. dolomieu* in this river. Our survey findings also support earlier observations that native *B. calidus*, *P. phlegethon* and *A. gilli* have been extirpated from the invaded reaches of the river. This hypothesis was supported by the total absence of any native fish other than *L. capensis* in the collections made during the rotenone application. Interestingly, *G. zebratus* was not detected using any methods, despite our sampling many of the sites where this species had previously been recorded by Woodford et al. (2005). Although this species was never abundant, its absence during the present study is cause for concern, and therefore research into its present status is recommended.

Tilapia sparrmanii have not previously been reported from the Rondegat, but have been present in the system since before 1961 (de Moor and Bruton 1988). Their presence, in conjunction with African sharptooth catfish *Clarias gariepinus*, collected from below the first waterfall barrier by MJ using an electrofisher in January 2012, suggests that there is a risk of further invasions. As the upper limit of the distribution of these two alien species, and of alien *L. macrochirus*, was the first waterfall barrier close to the dam, this waterfall and cascade are likely to be a second invasion barrier in this system. It is not clear why only *M. dolomieu* have been able to pass this barrier, but an introduction pathway via intentional stocking above the barrier cannot be excluded. This finding does, however, indicate that future rehabilitation efforts should extend the treatment zone down to this lower waterfall barrier, which would create another potential buffer against reinvasion of the treatment area.

The rotenone treatment exercise demonstrated two important factors. First, it showed that this study's monitoring results provide a good estimate of fish density and, second, that the non-detection of native fishes in the invaded zone was not an artefact of the sampling method. With respect to the immediate impact of the rotenone treatment, snorkelling surveys and underwater videoing failed to detect the presence of fish following the treatment. The removal of 470 *M. dolomieu* and 139 *L. capensis* from the system during the rotenone operation therefore reduced fish abundance in the treated section of river to below detectable levels. Longer-term monitoring will, however, be necessary to determine whether the treatment resulted in the eradication of alien fish. If the treatment proves to have been successful, the lower section of the Rondegat River presents a unique opportunity to better understand the response of native fishes to river rehabilitation. This could be achieved by monitoring the rates of native fish recruitment into the treatment area and by undertaking assessments of fish community structure in the rehabilitated environment.

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Appendix: Location, dimensions and sampling techniques used at 43 sites on the Rondegat River sampled for fish abundance. Three surveys were conducted: BT1 = before-treatment survey on 15/02/2011–17/02/2011; BT2 = before-treatment survey on 24/02/2012–27/02/2012; AT = after-treatment survey on 01/03/2012. Zones: N = not invaded by *Micropterus dolomieu*, T = invaded by *M. dolomieu* and falls in the rotenone-treatment area; B = invaded by *M. dolomieu* but below the treatment area. Sampling methods: EF = electrofishing; SN = snorkel survey; UWV = underwater video

Site #	Zone	Coordinates		Habitat dimensions (m)			Survey							
							BT1			BT2			AT	
		Latitude	Longitude	L	W	D	EF	SN	UWV	EF	SN	UWV	SN	UWV
1	N	32°22.536' S	19°03.890' E	8.0	4.68	0.45	•	•	•		•	•		
2	N	32°22.534' S	19°03.842' E	9.3	3.32	0.26	•		•		•	•		
3	N	32°22.525' S	19°03.789' E	7.9	3.46	0.32	•		•		•	•		
4	N	32°22.321' S	19°03.444' E	15.7	6.08	0.69		•	•		•	•		
5	N	32°22.301' S	19°03.411' E	9.0	4.22	0.19	•				•	•		
6	N	32°22.237' S	19°03.258' E	9.9	4.98	0.51					•	•		
7	N	32°22.219' S	19°03.191' E	10.0	5.00	0.50					•	•		
8	N	32°17.653' S	18°59.749' E	3.7	3.17	0.23		•	•		•	•		
9	N	32°17.628' S	18°59.731' E	10.8	5.28	0.41				•	•	•		
10	N	32°17.340' S	18°59.477' E	11.7	3.12	0.42				•				
11	N	32°17.327' S	18°59.470' E	12.7	3.31	0.47		•		•	•	•		
12	N	32°17.316' S	18°59.459' E	11.2	5.48	0.25	•		•	•		•		
13	N	32°17.311' S	18°59.448' E	9.5	5.30	0.72				•	•	•		
14	N	32°17.080' S	18°59.246' E	8.5	3.78	0.25	•	•	•					
15	N	32°17.067' S	18°59.244' E	12.7	5.52	0.25	•							
16	N	32°16.657' S	18°58.596' E	9.0	3.70	0.37	•		•		•	•		
17	N	32°16.657' S	18°58.596' E	5.0	2.00	0.50					•	•	•	
18	T	32°16.645' S	18°58.580' E	21.0	7.03	0.59		•				•		•
19	T	32°16.645' S	18°58.580' E	18.0	5.85	0.35		•	•		•	•	•	•
20	T	32°16.632' S	18°58.563' E	13.9	2.58	0.47	•	•			•	•	•	•
21	T	32°16.623' S	18°58.558' E	9.7	7.48	0.64	•	•	•		•	•	•	•
22	T	32°16.587' S	18°58.505' E	34.2	10.81	0.91		•	•		•		•	
23	T	32°16.567' S	18°58.479' E	10.3	5.32	0.41	•	•	•		•	•	•	•
24	T	32°16.560' S	18°58.475' E	15.3	3.69	0.36	•	•			•	•	•	•
25	T	32°16.526' S	18°58.467' E	12.3	3.87	0.64	•	•	•		•		•	
26	T	32°16.507' S	18°58.459' E	10.3	3.12	0.89	•	•	•		•	•	•	•
27	T	32°16.476' S	18°58.427' E	27.0	4.37	0.52	•	•	•		•	•	•	•
28	T	32°15.672' S	18°57.936' E	12.4	3.24	0.58				•	•		•	
29	T	32°15.615' S	18°57.907' E	16.4	4.42	0.52				•	•		•	
30	T	32°15.556' S	18°57.843' E	21.5	3.47	0.38		•		•	•	•	•	•
31	T	32°15.549' S	18°57.833' E	17.5	5.75	0.33	•	•		•	•	•	•	•
32	T	32°15.541' S	18°57.823' E	17.7	3.06	0.22	•			•	•	•	•	•
33	T	32°15.533' S	18°57.814' E	13.5	6.20	0.26	•			•	•	•	•	•
34	T	32°15.536' S	18°57.812' E	22.3	3.96	0.37	•			•	•		•	
35	T	32°15.192' S	18°57.203' E	30.0	6.82	1.06		•	•		•	•	•	•
36	B	32°15.201' S	18°57.186' E	22.0	6.04	0.47	•	•	•		•	•		•
37	B	32°15.327' S	18°57.191' E	10.0	3.63	0.31	•	•				•		
38	B	32°15.332' S	18°57.194' E	16.0	3.84	0.45	•				•			
39	B	32°15.353' S	18°57.161' E	14.2	4.03	0.39		•	•		•	•		
40	B	32°15.369' S	18°57.116' E	8.0	9.50	0.31	•	•			•	•		
41	B	32°15.357' S	18°57.150' E	15.0	4.26	0.53		•			•	•		
42	B	32°15.461' S	18°57.010' E	38.0	8.24	0.53		•	•		•	•	•	
43	B	32°15.516' S	18°56.886' E	11.6	2.62	0.35		•			•	•	•	