Abundance of native fishes, wild-introduced salmonids and escaped farmed rainbow trout in a Patagonian reservoir

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Abstract

The introduction of salmonids in Patagonia has resulted in significant impacts on its lakes, as well as a major impact on streams, in which native fishes seem to have been displaced almost completely by rainbow trout (*Oncorhynchus mykiss*). Another perspective is that the introduced salmonid species have resulted in wild fish populations that sustain an economically important sport fishery. The wide distribution and high abundance of escaped farmed rainbow trout, and a clear decrease in the abundance of native and successfully introduced salmonid species in Alicura Reservoir were all observed, based on comparison of recent data and data from 1993 to 1995 corresponding to littoral gillnet captures. Thus, both native fish and introduced salmonid populations seem to have been drastically reduced in the presence of farmed fish escapees. The results of the present study regarding fish escapes deserve major consideration when making decisions about fish cage culture activities for other Patagonian reservoirs.

Key words

cage culture, fish escapes, native fishes, patagonia, rainbow trout.

INTRODUCTION

Cage culture of salmonids in fresh water has been recognized as a significant source of nutrients and organic waste, resulting in increased levels of nutrients and associated benthic and planktonic algae in the water column, localized enrichment of lake sediments and changes in lake fish populations and fisheries (Davies 2000). Thus, fish cages, unconsumed artificial food and salmon farm escapees may have significant effects on self-sustaining salmonid populations, as well as the general lake community (Phillips *et al.* 1985; Alpaslan & Pulatsü 2008).

The risk of escaped fish from fish farms and their consequent potential impacts on wild fish populations, including genetic, ecological and social impacts, have been related to an open-net cage-based culture system (Tacon & Halwart 2007; see Jensen *et al.* 2010 for a thorough review). When economic considerations prevail, however, highly valued fish species are cultivated in these sometimes rudimentary systems as a means of enhancing the economy in developing regions. In such cases, the selection of long-term domesticated individuals has been proposed as a conservation strategy, as their capacity for naturalization, and therefore invasiveness, is likely lower than that for recently domesticated farm stocks (Valiente *et al.* 2010). Although released or escaped hatchery-selected fish may be more susceptible to predation than wild fish (Jonsson 1997; Einum & Fleming 2001), farmed fish escapees are commonly observed entering rivers and breeding with their wild conspecifics (e.g. Hansen *et al.* 1991; Youngson *et al.* 1997; Utter 2000; AquaWild 2002). This interbreeding may threaten the fitness of wild fish populations already adapted to the local environment, as was the case with salmonids introduced in Patagonia almost a century ago (Pascual *et al.* 2007; Macchi *et al.* 2008). To mitigate this impact, the use of triploid fish has been suggested in aquaculture (Cotter *et al.* 2000). Furthermore, acoustic conditioning for recalling and

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recapturing escaped individuals (Tlusty *et al.* 2008) might help to minimize escape effects (Skilbrei & Jørgensen 2010; Chittenden *et al.* 2011).

In Argentina, salmonid aquaculture consists exclusively of rainbow trout (*Oncorhynchus mykiss* (Walbaum)), primarily developed in the Limay River basin (http://www.aic.gov.ar/aic/Imag01.aspx, Wicki & Luchini 2002). The potential carrying capacity of the region has been assessed by Wicki and Luchini (1996) to be approximately 17000 ton.yr^{-1} in reservoirs and rivers (Vigliano & Alonso 2007): particularly, 3600 ton.yr^{-1} for the Alicura Reservoir, 6800 ton.yr^{-1} for Piedra del Aguila Reservoir and 4500 ton.yr^{-1} for Ezequiel Ramos Mexía Reservoir (Wicki & Luchini 2002). Rainbow trout cage culture currently is carried out in the Alicura Reservoir, with its fish production ranging from 6 to 600 ton.yr^{-1} until 2004 (Dirección de Acuicultura 2004), and subsequently reaching 1800 ton.yr^{-1} in 2009 (Zeller *et al.* 2009).

Fish captures in the Limay River before construction of the Alicura dam were dominated by perca (*Percichthys trucha* (Valenciennes)) and the successfully introduced rainbow trout (HIDRONOR 1981). After the filling of the reservoir in 1985, perca, rainbow trout and Patagonian pejerrey (*Odontesthes hatcheri* (Eigenmann)) predominated (HIDRONOR 1989; Freyre *et al.* 1991). After farming activities commenced in 1992 (Temporetti *et al.* 2001), rainbow trout became the most abundant species in Alicura Reservoir, comprising 47.5% of littoral gillnet captures between 1993 and 1995 (Cussac *et al.* 1998).

The Limay, Neuquén and Negro river-basin Inter-jurisdictional Authority (http://www.aic.gov.ar, Autoridad Interjurisdiccional de la Cuenca de los Ríos Limay, Neuquén y Negro, AIC) is a governmental agency whose mission is to administer the use and secure the preservation, of the basin of the previously mentioned rivers. Information regarding fish escapes is not collected by the AIC, as fish farmers have no legal obligation to report them. Since 1996, however, the AIC has performed annual sampling in Alicura Reservoir and has been able to distinguish escaped rainbow trout (ERT) from wild rainbow trout on the basis of eroded dorsal, pectoral or caudal fins (Temporetti *et al.* 2001). The use of discriminant analysis (DA) of scale morphology proved to be complex for ERT identification, as ERT differed significantly not only from wild rainbow trout, but also from hatchery-reared rainbow trout, likely attributable to complex effects of farming and natural conditions throughout their life history (Alonso 2003).

Published data about escapes in Alicura Reservoir are limited to Temporetti *et al.* (2001), who noted that ERT comprised more than 60% of the total rainbow trout capture between 1996 and 1998, a few years after the beginning of rainbow trout cage culture. Accordingly, the aim of this work was to analyse the AIC database for Alicura Reservoir in order to: (a) obtain an updated view of the abundance of ERT; (b) evaluate changes in the abundances of native fishes and wild salmonids and biomass on the basis of fishing captures; and (c) test the association of ERT with native and introduced salmonid species, in order to contribute to informed fish management decisions.

MATERIALS AND METHODS

Study sites and species composition

Alicura Reservoir is a large (6750 ha), man-made reservoir located on the Limay River in southwestern Argentina (Fig. 1). Inaugurated in 1985, it is located 87 km downstream of Lake Nahuel Huapi, in the transition zone between Andean forest and steppe, being used primarily for hydropower generation (http://www.aic.gov.ar/aic/Imag01.aspx). The native fishes, *Galaxias maculatus* (Jenyns), *Hatcheria macraei* (Girard), *O. hatcheri, Diplomystes viedmensis* MacDonagh (Muñoz-Ramírez *et al.* 2014) and *P. trucha* (including, according to Ruzzante *et al.* 2006, 2011; other nominal forms considered by López-Albarello 2004), and the exotic salmonids *O. mykiss,* Atlantic salmon *Salmo salar* Linnaeus, brown trout *Salmo trutta* Linnaeus and brook trout *Salvelinus fontinalis* (Mitchill) have been reported in the reservoir (Cussac *et al.* 1998; Macchi *et al.* 1999; Aigo *et al.* 2008). *O. hatcheri,*

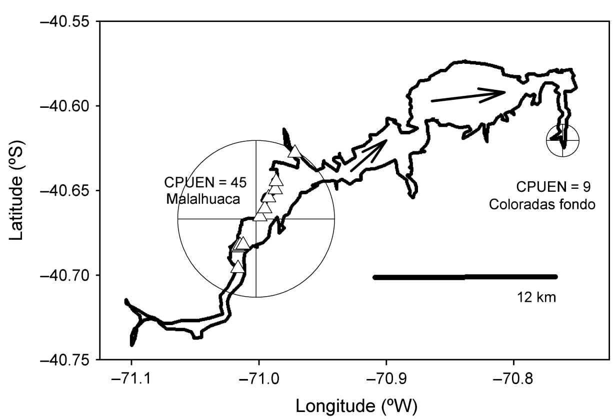

Fig. 1. Map of Alicura Reservoir (arrows = water flow direction; crosses = sampling sites; triangles = position of fish farms circle diameters proportional to mean capture per unit effort in number (CPUEN, number of fish per 100 m^{-2} of net and 15 h of overnight operation) of escaped rainbow trout (ERT) in 2000–2011 sampling period).

P. trucha, O. mykiss and *S. trutta* are the most common captured species.

Data collection and analysis

Data for analysis were extracted from the AIC database, comprising a compilation of an annual monitoring program carried out by AIC since 1996. Fish were caught using two sets of floating gillnets per summer at each of two locations within the reservoir, namely Malalhuaca (next to the cages) and Coloradas Fondo (located 25 km distance from the farm site). Each set of gillnets from 1996 to 1999 consisted of six stretched mesh sizes (42, 50, 60, 70, 76 and 105 mm, with each mesh being 25 m long) worked overnight in the littoral zone between surface and 4 m depth. AIC added a smaller mesh (stretched mesh size 30 mm, 25 m long) to each gillnet set (seven mesh sizes in total) beginning in 2000, as it was suspected the small-sized fish were not being captured (Garcia Asorey 2001). Comparison of the species distributions before and after the addition of the 30 mm mesh size nets yielded differences for all the species, including pejerrey (Kolmogorov–Smirnov $Z = 2.543$, $n = 346$, $P < 0.0001$), brown trout ($Z = 2.172$, $n = 149$, $P < 0.0001$) and wild rainbow trout ($Z = 3.275$, $n = 743$, $P < 0.001$), except for perca ($Z = 0.962$, $n = 188$, $P > 0.962$). Only pejerrey of smaller sizes, however, were captured with the subsequent use of the 30 mm mesh size. As a result of different fishing efforts and selectivity, the analysis was performed on two separated datasets, namely 1996 to 1999 and 2000 to 2011 (Table 1). The captured fish were counted and weighted (± 1 g). The catch per unit effort (CPUE) was referred to 100 m^2 of net and 15 h of overnight operation. The fish catches were expressed in terms of numbers (CPUEN) and kilograms (CPUEW). The CPUEW * $CPUEN^{-1}$ ratio was used to estimate mean body mass (kg). The geographic location of the fish farms was obtained using Google^{tm} Earth software.

The AIC records data on ERT caught by gillnets on the basis of eroded fins (Temporetti *et al.* 2001). As fins can regenerate quickly in the wild, however, not all escaped fish exhibited eroded fins, particularly if they had escaped at a young age. Two recent studies of rainbow trout escaping from fish farms in Chilean Patagonia indicate escapees can exhibit no obvious phenotypic differences, and many can go undetected (Consuegra *et al.* 2011; Schröder & García de Leaniz 2011). Thus, estimated abundance of escaped fish should be considered an underestimation in the present study.

Fish captures in Alicura Reservoir (%CPUEN of pejerrey, perca and salmonid species) were compared with data from other Patagonian lakes and reservoirs not exhibiting fish-farming activities, based on present data (Table 1) and the databases of Quirós (1991) and Aigo *et al.* (2008). Percentages were transformed as arcsin $(0.01 \text{ X } \%\text{CPUEN})^{0.5}$ and treated with DA (SPSS®, Armonk, NY, USA).

Comparisons of CPUEN and CPUEW between species were performed using one-way ANOVA or, when assumptions were not satisfied, the nonparametric Kruskal–Wallis test and Tukey pairwise comparisons. Paired *t*-tests, or the Wilcoxon signed-rank test when assumptions failed, were used to test the abundance of each species between capture sites. Furthermore, as ERT has a unidirectional influx from the farming area because of the escapes, the abundances of ERT were compared to the combination of the other species at each sampling sites separately, using the Mann–Whitney rank-sum test. Correlations of CPUEN and mean body mass (CPUEW * $CPUEN^{-1}$) of native fish, wild salmonids and ERT were tested with Spearman's rank-order correlation. Tests and plots were carried out with SIGMAPLOT® (Systat Software, Inc., San Jose, CA, USA). The possibility of a trend in the time series data was analysed with the Mann–Kendall test, and the correlation between time series data and time was assessed with Spearman's rho test, both using TREND v 1.0.2 (F. Chiew & L. Siriwardena, TREND User Guide, www.toolkit.net.au/trend).

RESULTS

Alicura Reservoir and other Patagonian lakes and reservoirs that do not have ongoing fish farming

Alicura Reservoir was discriminated significantly (DA, $n = 97$, Wilkes′ lambda = 0.715, $P < 0.001$) from other Patagonian lakes and reservoirs without fish farming, with 61.9% of original grouped cases correctly classified, on the basis of a lower abundance of perca (*P. trucha*) and a slightly higher abundance of pejerrey (*O. hatcheri* and *O. bonariensis)* in the reservoir (Fig. 2).

Years 1996–1999

In spite of this short period of time, some significant differences were observed between species. Both, CPUEN (Kruskal–Wallis, $H = 29.834$, $n = 48$, $P < 0.001$) and CPUEW (Kruskal–Wallis, $H = 27.383$, $n = 48$, $P < 0.001$) of ERT were greater (Tukey test, $P < 0.05$) than those of pejerrey, perca, Atlantic salmon and brown trout (the latter only for CPUEN). CPUEN and CPUEW of wild rainbow trout were greater than those of Atlantic salmon (Tukey Test, $P < 0.05$, Fig. 3).

No significant differences were observed between capture sites when considering the reservoir as a whole. How-

Table 1. Alicura Reservoir fish catches (catches per unit effort expressed in number (CPUEN) and kilograms (CPUEW) by species and capture site; CPUEN and CPUEW were referred to 100 m^2 of net and 15 h of overnight operation

		Wild *O. mykiss*		Escaped *O. mykiss*		*S. trutta*		*S. salar*		*O. hatcheri*		*P. trucha*	
Sampling Site	Year	CPUEN	CPUEW	CPUEN	CPUEW	CPUEN	CPUEW	CPUEN	CPUEW	CPUEN	CPUEW	CPUEN	CPUEW
Malalhuaca	1996	19.30	4.50	158.30	33.27	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	1997	9.70	2.50	59.70	21.69	7.80	5.36	0.00	0.00	0.00	0.00	3.80	1.29
	1998	14.60	6.77	218.40	96.78	5.30	3.58	0.00	0.00	0.00	0.00	14.40	4.82
	1999	7.90	1.73	185.60	60.26	7.50	9.01	0.00	0.00	0.00	0.00	4.00	3.76
	2000	14.60	5.48	109.60	75.79	2.40	1.18	0.00	0.00	0.00	0.00	0.00	0.00
	2001	5.90	2.46	18.60	10.85	3.20	8.10	0.00	0.00	0.00	0.00	0.00	0.00
	2002	14.20	5.03	9.37	4.92	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	2003	9.73	3.88	4.43	1.97	3.45	1.74	0.00	0.00	0.00	0.00	0.00	0.00
	2004	23.70	11.69	3.70	2.08	8.20	6.08	0.00	0.00	0.80	0.58	0.00	0.00
	2005	8.98	8.46	5.12	2.85	2.40	3.04	0.00	0.00	0.00	0.00	0.00	0.00
	2006	8.30	4.39	29.60	21.22	0.00	0.00	0.90	0.42	0.00	0.00	0.00	0.00
	2007	9.40	3.27	23.67	9.32	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	2008	8.00	2.83	43.20	16.77	0.90	0.65	0.00	0.00	0.00	0.00	0.00	0.00
	2009	6.90	4.62	222.40	127.44	2.30	8.15	0.00	0.00	0.00	0.00	0.00	0.00
	2010	13.90	6.77	60.60	32.09	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
	2011	4.40	1.72	6.40	2.45	1.00	0.38	0.00	0.00	0.00	0.00	0.00	0.00
Coloradas	1996	33.00	11.14	17.80	6.03	5.30	1.17	2.20	0.29	10.30	3.95	4.30	2.82
Fondo	1997	7.50	2.31	40.80	13.61	7.50	2.01	1.60	0.17	34.50	8.68	0.00	0.00
	1998	23.20	8.11	78.00	27.18	4.10	1.07	0.00	0.00	34.90	14.45	39.30	14.57
	1999	6.10	1.85	59.60	19.41	6.90	2.19	0.00	0.00	4.30	0.89	1.80	0.77
	2000	10.80	3.63	23.70	7.56	2.00	0.90	0.00	0.00	17.00	2.61	0.00	0.00
	2001	7.70	3.09	9.10	5.46	1.60	0.79	0.00	0.00	58.60	19.51	10.80	4.22
	2002	3.20	1.63	0.00	0.00	2.53	2	0.00	0.00	6.01	1.35	2.11	0.03
	2003	20.99	8.17	4.05	1.48	0.00	0.00	0.00	0.00	29.98	7.51	0.82	0.41
	2004	13.90	5.93	0.80	0.30	2.90	0.98	0.00	0.00	41.60	19.56	2.30	1.87
	2005	18.80	8.15	5.10	2.33	3.20	2.11	0.00	0.00	10.40	4.34	0.00	0.00
	2006	13.90	6.49	6.80	3.28	2.10	1.64	1.20	0.44	7.00	1.26	0.00	0.00
	2007	3.38	2.55	3.40	1.48	2.90	5.22	0.00	0.00	11.40	2.16	0.00	0.00
	2008	7.33	2.56	30.03	12.09	3.80	3.01	0.90	0.40	2.60	0.65	0.00	0.00
	2009	4.00	1.89	19.40	8.75	1.60	0.35	0.00	0.00	3.80	1.63	0.00	0.00
	2010	2.13	0.95	2.05	1.25	1.90	0.61	0.00	0.00	0.00	0.00	1.07	0.78
	2011	2.10	1.06	8.30	4.34	6.40	3.16	11.90	1.69	0.00	0.00	0.00	0.00

ever, when looking at the abundance of ERT versus the combination of the other species captured in Malalhuaca, we found that ERT were more abundant in number (CPUEN, Mann–Whitney rank-sum test, $U = 0.00$, $P < 0.029$) and in mass (CPUEW, $U = 0.00$, $P < 0.029$). On the contrary, no differences were detected in Coloradas Fondo.

Mean body mass (CPUEW * CPUEN^{-1}) showed significant and positive correlation between ERT and wild rainbow trout (Spearman's rank-order correlation coefficient = 0.738, $P < 0.029$, $n = 8$) and negative correlations between ERT and perca (Spearman's coefficient = −0.943, $P < 0.017$, $n = 6$) (Fig. 4).

Years 2000–2011

CPUEN (Kruskal–Wallis, $H = 75{,}052$ $n = 144$, $P < 0.001$) of both ERT and wild rainbow trout were greater than those of Atlantic salmon, perca, pejerrey and brown trout (Tukey test, $P < 0.05$). CPUEW (Kruskal–Wallis, $H = 77.144$ $n = 144$, $P < 0.001$) of ERT and wild rainbow

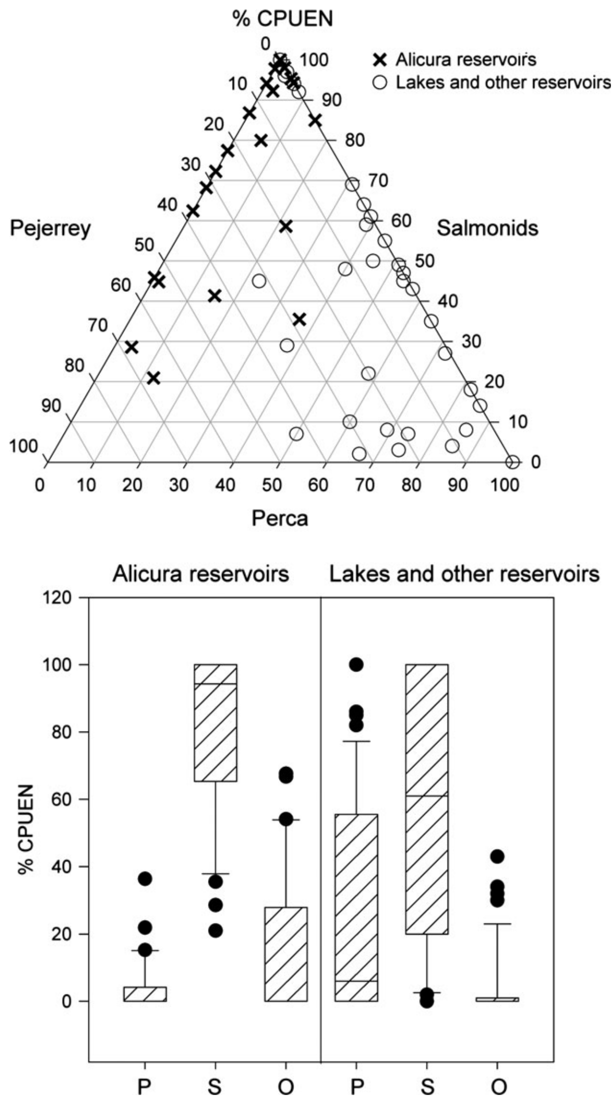

Fig. 2. Top panel: relative abundance (%CPUEN) of *P. trucha* (Perca, P), salmonids (S) and *O. hatcheri* (pejerrey, O) in Alicura Reservoir (n = 33) and other lakes and reservoirs without fish farming (n = 65) of Argentinean Patagonia (Aigo *et al.* 2008); bottom panel: boxplots of relative abundance (%CPUEN), indicating median, quartiles and data outside 5 and 95 percentiles.

trout were greater than those of Atlantic salmon, perca and pejerrey (Tukey test, $P < 0.05$).

Between capture sites, ERT were more abundant, in number (CPUEN, Wilcoxon signed-rank test, $n = 12$, $Z = 2.824$, $P < 0.002$) and in mass (CPUEW, $Z = 2.746$, $P < 0.003$), in the trout farm area (Malalhuaca) than in Coloradas Fondo. Notwithstanding this significant difference, the capture of ERT in Coloradas Fondo (far from the farming area) suggests a wide dispersion for ERT throughout the reservoir (Table 1, Fig. 1 and 3). As a counterpart, pejerrey were nearly absent close to trout farms (Malalhuaca) and were more abundant in Coloradas Fondo (CPUEN, Wilcoxon signed-rank test, $n = 12$, $Z = -2.803$, $P < 0.002$, CPUEW, $Z = -2.803$, $P < 0.002$). In fact, CPUEN of ERT showed negative correlations with those of pejerrey (Spearman's rank-order correlation coefficient = −0.404, $P < 0.044$, $n = 24$) and perca (Spearman's coefficient = −0.535, $P < 0.007$, $n = 24$), while pejerrey and perca showed themselves a positive correlation for CPUEN (Spearman = −0.515, $P < 0.010$, $n = 24$).

The abundance of ERT in Coloradas Fondo was lower in number (CPUEN, Mann–Whitney rank-sum test, $U = 121.5$, $P < 0.005$) and weight (CPUEN, *t*-test, $t = -2.702$, $P < 0.013$) than the combination of the other species captured, whereas no differences were detected in the farming area (i.e. Malalhuaca).

Mean body mass (CPUEW * CPUEN^{-1}) showed a positive correlation between ERT and wild rainbow trout (Spearman's coefficient = 0.477, $P < 0.021$, $n = 23$) (Fig. 5). Mean body mass of ERT showed decreasing trends in Malalhuaca (Spearman's rho = −0.373, $P < 0.05$, Mann–Kendall $Z = -1.246$, $P < 0.05$). However, mean body mass of both ERT (Spearman's rho = 0.455, $P < 0.01$, Mann–Kendall $Z = 1.401$, $P < 0.01$) and wild rainbow trout (Spearman's rho = 0.364, $P < 0.05$, Mann–Kendall $Z = 1.44$, $P < 0.01$) showed increasing trends in Coloradas Fondo.

No significant trend of CPUEN or CPUEW could be observed in Malalhuaca. However, in Coloradas Fondo, the time series analyses showed decreasing trends for the abundance of pejerrey (CPUEN, Mann–Kendall $Z = -2.743$, $P < 0.01$, Spearman's rho = −0.797, $P < 0.01$, CPUEW, Mann–Kendall $Z = -2.4$, $P < 0.05$) and wild rainbow trout (CPUEN, Mann–Kendall $Z = -1.851$, $P < 0.05$), whereas Atlantic salmon showed a significant increase (CPUEN, Mann–Kendall $Z = 1.371$, $P < 0.05$, CPUEW Mann–Kendall $Z = 0.686$, $P < 0.01$, Spearman's rho= 3.317, $P < 0.05$, Fig. 6). However significant, the latter trend should be corroborated in the following years.

DISCUSSION

The comparison between Alicura Reservoir and other Patagonian lakes and reservoirs indicated Alicura Reservoir had a lower abundance of perca (*P. trucha*) and a slightly higher abundance of pejerrey (*O. hatcheri* and *O. bonariensis)*. The results for the reservoir indicated a high, fluctuating abundance of ERT and a low abundance of pejerrey, perca and brown trout, with an almost null abundance in the past years, compared to the 1993–1995 dataset (with *O. hatcheri* 19.5%, *P. trucha* 21% and *S. trutta* 11% of the total CPUEN; see Cussac *et al.* 1998). The absence of several native species previously reported for the reservoir, including *G. maculatus*, *D. viedmensis* and

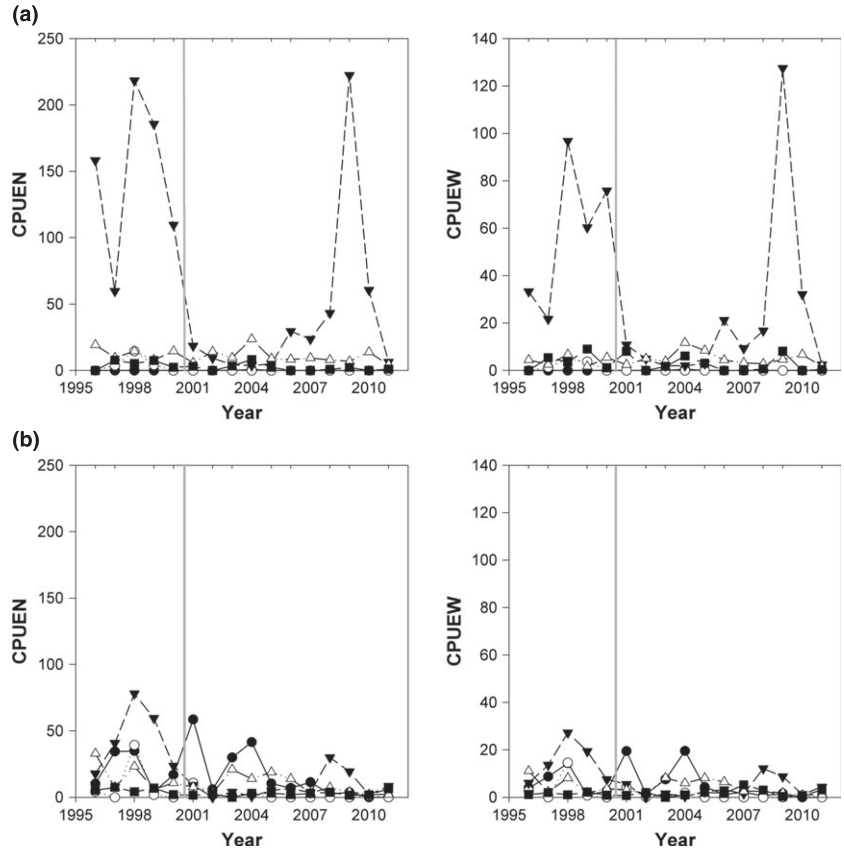

Fig. 3. Abundance (CPUEN and CPUEW, number and kilograms, respectively, of fish per 100 m^{-2} of net and 15 h of overnight operation) of *O. hatcheri* (black circles and continuous lines), wild *O. mykiss* (white triangles and dot/dashed lines), escaped rainbow trout (ERT) (black triangles and short dashed lines), *S. trutta* (black squares and long dashed lines) and *P. trucha* (white circles and dotted lines) of littoral captures in Alicura Reservoir ((a) Malalhuaca sampling site; (b) Coloradas Fondo sampling site; vertical grey lines indicate incorporation of 30 mm stretched mesh size to gillnet sets).

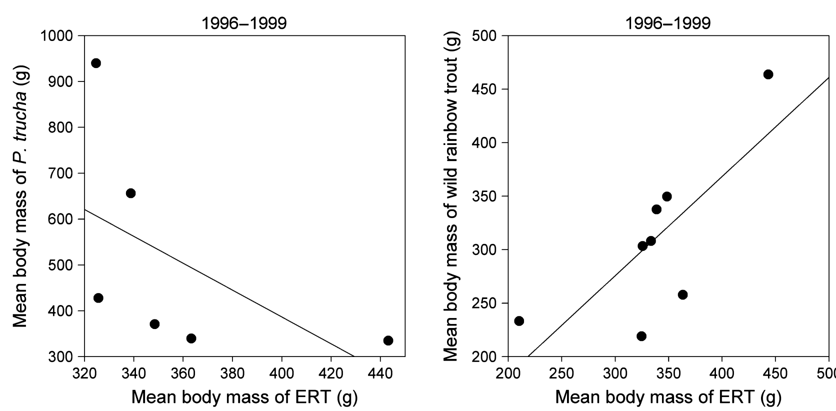

Fig. 4. Between-species significant correlations (lines illustrate tendencies) for mean body mass (CPUEW. $CPUEN^{-1}$) for the 1996–1999 time period and both Malalhuaca and Coloradas Fondo sampling sites.

H. macraei (Aigo *et al.* 2008) in the set of gillnets used, is likely a consequence of their small size (i.e. *G. maculatus* and *H. macraei;* Ringuelet *et al.* 1967) and a different habitat use *(D. viedmensis*), which are much smaller than adult rainbow trout, brown trout, Atlantic salmon, pejerrey and perca.

The fish captures analysed here corresponded to a maximum depth of 4 m in the littoral zone. Recent observations reported a high abundance of *P. trucha* and the presence of *Galaxias platei* Steindachner at depths of 10 m or more in Alicura Reservoir (Nabaes Jodar *et al.* 2012). This suggests a spatial segregation meriting further study, as the depth distribution of fish assemblages can affect the interpretation of fish abundance. Furthermore, the observed increase of Atlantic salmon captures at far distances from the trout farms (Coloradas Fondo), and the negative correlations between the CPUE of ERT with pejerrey and perca, may also be reflecting spatial segregations.

The positive correlation between the mean body mass of ERT and the wild rainbow trout captured in the same place and time suggests similar patterns of habitat use,

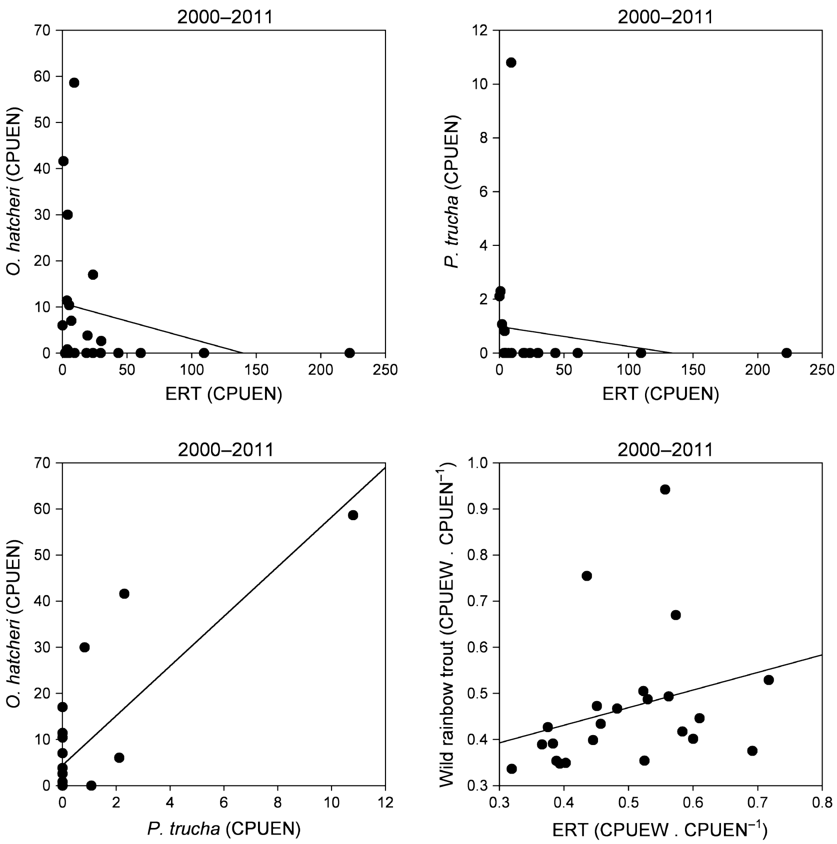

Fig. 5. Between-species significant correlations (regression lines indicated only to show tendencies) for CPUENs and mean body mass (CPUEW. CPUEN^{-1}) for the 2000–2011 time period.

as the diet of rainbow trout changes with its size (Macchi *et al.* 1999). Jacobsen and Hansen (2001) and Rikardsen and Sandring (2006) previously observed that escaped Atlantic salmon and postsmolt rainbow trout, respectively, can readily adapt to feeding on natural marine prey, exhibiting similar feeding intensities as other wild anadromous salmonids. Although similar patterns of habitat use are observed for Alicura Reservoir, preliminary data from Nabaes Jodar *et al.* (2012) indicated ERT are less piscivorous than wild rainbow trout. Further studies are being conducted to understand the ability of escaped fish to adapt to natural prey in this freshwater system.

Spatial differences were notorious in Alicura Reservoir, with the results of the present study consistent with those of Skilbrei (2012) who proposed that as ERT prefers to stay very close to the surface, their recapture might be increased in shallow waters in the vicinity of the fish farms. Furthermore, in agreement with the results obtained with experimental releases of farmed rainbow trout (Jonsson *et al.* 1993; Lindberg *et al.* 2009), higher ERT abundances were observed at the farming site (Malalhuaca) than at the farthest sampling site (Coloradas Fondo). The impact of ERT abundance in areas close to the farms seems to be so high that no temporal trends could be observed for any species, with all fishes other than ERT exhibiting a notably low abundance. Furthermore, comparing the ERT to the combination of the other fish species indicated the closer the ERT was to the farming site, the more the ERT was found in distant areas. Moreover, the decreasing trend in the abundance of pejerrey and wild rainbow trout far from the cages suggests a cumulative deleterious effect over the years for these species. Thus, the more farmed fish escaped from the cages, the biggest the impact observed at distant sites.

It is untenable to treat exotic salmonids as ‘best’ or ‘pest’, based solely on whether or not they accidentally escaped from fish farms or were deliberately introduced by anglers, or because they provide short-term revenues (Garcia De Leaniz *et al.* 2010). Salmonid introductions in Patagonia resulted in a highly valued sport fishery (Arismendi & Nahuelhual 2007; Vigliano & Alonso 2007; Macchi *et al.* 2008). As has happened elsewhere, however, (Cambray 2003; Fausch 2007), introduced salmonids (rainbow, brown, brook, and lake trout and Atlantic and Chinook salmon; Pascual *et al.* 2013) had a significant negative impact on Patagonian lakes (Aigo *et al.* 2008; Habit *et al.* 2010; Elgueta *et al.* 2013), which was partly ameliorated by the availability of littoral refuges where native fish could breed (Cussac *et al.* 1992; Barriga *et al.*

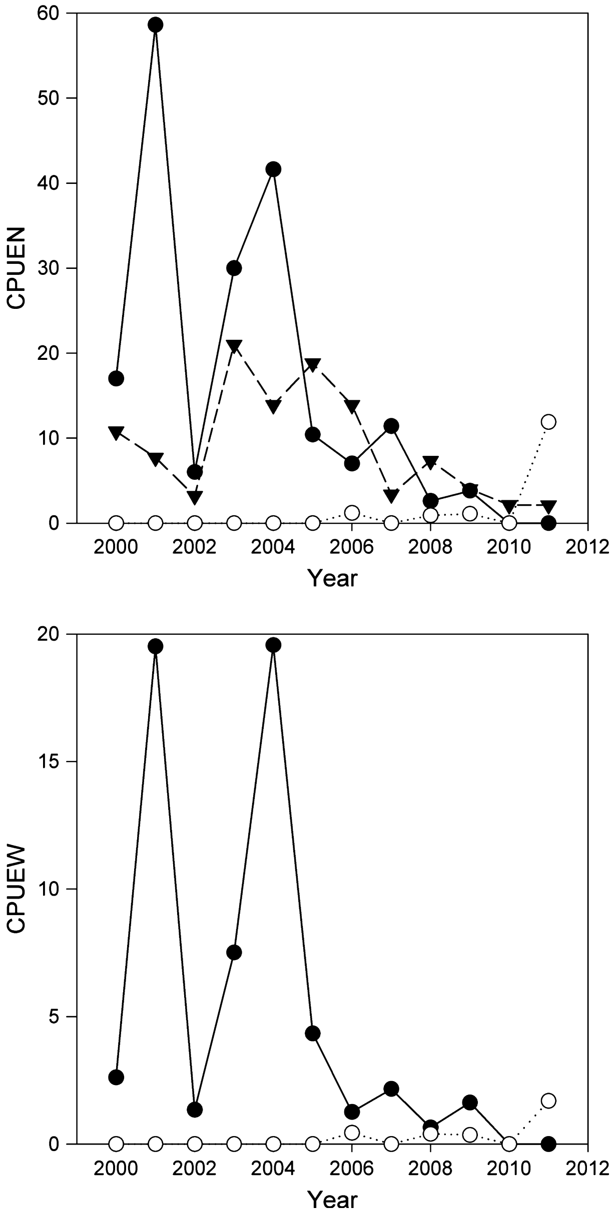

Fig. 6. Significant decreasing trends in CPUEN and CPUEW for pejerrey (black circles and continuous lines) and wild rainbow trout (black triangles and short dashed lines) (increasing trend observed for Atlantic salmon (white circles and dotted lines); Coloradas Fondo sampling site).

2002; Buria *et al.* 2007; Lattuca *et al.* 2008a,b). Moreover, major impacts seem to have occurred for streams where salmonids, particularly *O. mykiss,* seem to have almost completely displaced native fishes (Aigo *et al.* 2008; Habit *et al.* 2010; Young *et al.* 2010). Furthermore, long-time naturalized populations of rainbow trout appeared to exhibit early life-survival advantages over more recently introduced hatchery-selected populations, for which cross-mating resulted in outbreeding depression (Miller *et al.* 2004). Wild-origin lake trout, *Salvelinus namaycush* (Walbaum), also exhibited better situations, compared to hatchery-origin fish (McDermid *et al.* 2010). Similarly, wild Atlantic salmon populations have experienced gene pool changes attributed to hybridization with farm fish escapees (Skaala *et al.* 2006). Thus, both the native fish populations and the salmonid sport fishery appear to be threatened by fish culture escapes, although for different reasons (Soto *et al.* 2001). The headwater lakes of Alicura Reservoir (i.e. Lake Traful; Lake Nahuel Huapi; Lake Moreno) have all supported a salmonid sport fishery for the past 60 years, which has attracted high-income level fishers from abroad (Vigliano *et al.* 2008). The variable presence and abundance of wild salmonids and native fishes in Patagonian lakes (Aigo *et al.* 2008; Habit *et al.* 2010) contrasts with the results of Vigliano *et al.* (2008) and Juncos *et al.* (2011) for these three headwater lakes, which exhibited a decreasing abundance ratio between rainbow trout and the native *P. trucha,* related to their distance from Alicura Reservoir (i.e. 25.41 for Lake Traful, located 20 km upstream of the reservoir; 1.09 for Lake Nahuel Huapi, located 45 km from the reservoir; 0.80 for Lake Moreno, located 90 km from the reservoir). These findings agree with the high abundance and wide distribution range of ERT observed within Alicura Reservoir and with other studies regarding abundance (Skilbrei & Wennevik 2006) and capacity for long-distance dispersal among escapees from aquaculture facilities (Hansen & Youngson 2010).

The importance and consequences of farmed fish escapes are an open discussion topic (Boyd *et al.* 2005; Sepúlveda *et al.* 2009) that is just beginning to be seriously considered in regard to Argentine aquaculture regulations (Somoza & Núñez 2010). It would be wise, for example, to consider the already-recommended and applied actions in Norway (Jensen *et al.* 2010). In spite of relevant environmental issues, lacustrine cage culture is well established in Chilean Patagonia region under the 2001 Environmental Regulation for Aquaculture (RAMA) and the control of the National Fisheries Service SERNAPESCA (Rojas & Wadsworth 2007). Although recent data on thermal constraints for salmonid reproduction (Pankhurst & King 2010) has triggered alerts in regard to its feasibility for Alicura Reservoir (Báez *et al.* 2011), economic expectations could lead to installation of salmonid cage culture facilities in the other reservoirs along the Limay River basin, at least for the near future.

It is noted that the methods employed in the present study to identify ERT likely impeded recognition of all escapees, particularly as fin regeneration is common for teleosts (Morgan 1900). Thus, ERT abundance in Alicura

Reservoir should be considered as significantly underestimated, and with a wild rainbow trout population that would actually be more impoverished than observed in the present study. Schröder and García de Leaniz (2011) recently successfully discriminated between escaped and wild rainbow trout on the basis of scale growth profiles and stable isotope analysis. In fact, future use of these methods will doubtless help improve the situation regarding the underestimation of escapees in Alicura Reservoir, thereby providing a stronger basis for interpreting the extent of the impacts of escapees in the reservoir. Although obtained on a correlative basis, the results obtained thus far suggest mitigating actions such as the potential use of homing by olfaction (Bridger *et al.* 2001), acoustic conditioning (Tlusty *et al.* 2008) and other effective recapture strategies (Skilbrei & Jørgensen 2010; Chittenden *et al.* 2011) should be considered when making decisions about new cage culture locations in the Argentinean Patagonia region.

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