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PRIMARY RESEARCH PAPER

Age, growth and population dynamics of two congeneric and invasive gobies, *Rhinogobius giurinus* and *R. cliffordpopei* (Actinopterygii, Gobiidae) in a plateau lake, southwestern China

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Abstract Although many goby species have become widely established outside of their native ranges, little is known yet about invasive population biology. The age, growth and population dynamics of two introduced goby species (*Rhinogobius giurinus* and *Rhinogobius cliffordpopei*) were quantified in Lake Erhai (southwest China) with the general aim to improve empirical knowledge and management strategies. The results revealed that their lifespan ranges 2 years. The size and time at sexual maturity for *R. cliffordpopei* was lower than for *R. giurinus*. *R. cliffordpopei* displayed higher total mortality, fishing mortality and exploitation rates than *R. giurinus*, whereas the natural mortality of *R. cliffordpopei* was lower than that of *R. giurinus*. The catch per unit effort of benthic fyke nets was low from April to June in *R. cliffordpopei* and from July to September in *R. giurinus*, whereas it was high from July to September in *R. cliffordpopei* and from April to June in *R. giurinus*. Thus, several specific strategies are suggested to improve the cost-efficiency of the current management practices related to the mesh size of benthic fyke nets used for physical removals, the timing of deployment of control actions and the use of synergic control methods.

Keywords Gobiidae · Biological invasion · Lifespan · Recruitments · Invasive fish · Invasive species management · China

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Introduction

Invasive species are one of the greatest threats to global biodiversity, socio-economic development and human health (Butchart et al., 2010) with an estimated annual cost of 1.4 trillion dollars (5% of the global economy, Ricciardi et al., 2011). Fish are among the most widely introduced animals in fresh waters and the overall number of introduced species is more than 624 species, with a rate of introduction that has doubled in the last 30 years (Gozlan et al., 2010). Invasive fish can have significant ecological and evolutionary impacts on the colonized habitats (Cucherousset & Olden, 2011) and management actions have been implemented to mitigate these impacts (Britton et al., 2011). Simberloff (2003) argued that the detailed study of population biology of introduced species is very likely to fine-tune the current management protocols, and may also facilitate the establishment of new control methods. Therefore, determining population biology and quantifying life-history traits (e.g. age, growth, survival, reproduction and population dynamics) of invasive fish are the crucial prerequisite for cost-effective management (Ludgate & Closs, 2003; Britton et al., 2011; Houston et al., 2014).

Gobiidae is one of the largest fish families with more than 2000 species belonging to more than 200 genera. Many goby species have been introduced outside of their native range and become established worldwide (e.g. Europe, Copp et al., 2005; Asia, Xie et al., 2001; Yuan et al., 2010; North America, Kornis et al., 2012). The rapid proliferation of introduced goby species has raised serious concerns over their impacts on the colonized habitats (Copp et al., 2005; Kornis et al., 2012). Round goby *Apollonia melanostoma* (Pallas, 1814), for instance, has been demonstrated to induce diet shift of co-occurring species, change in food web structure, alteration of nutrient and energy flow, change in contaminant pathways and bioaccumulation of toxic substances (see details in Kornis et al., 2012). There is, however, scarce information about specific control methods for invasive goby species.

Species of genus *Rhinogobius* (Gill 1859) are common benthic fish species in most East Asian countries (Sone et al., 2001; Wu & Zhong, 2008). *R. giurinus* (Rutter, 1897) and *R. cliffordpopei* (Nichols, 1925) are native to Japan, middle and southeastern of China (e.g. the middle and lower reaches of the Yangtze River, Wu & Zhong, 2008) and have been inadvertently introduced into several lakes in Yunnan-Guizhou Plateau (southwest of China) in the 1950–60s (Du & Li, 2001; Xie et al., 2001). The two species have then spread to most water bodies across Yunnan-Guizhou Plateau and become dominant in many lakes (Yuan et al., 2010; Tang et al., 2013). Invasive fish species are considered as one of the major causes of the local decline and/or extirpation of native fish (Du & Li, 2001; Xie et al., 2001; Yuan et al., 2010) and local control actions have been implemented. However, they have limited success (Du & Li, 2001; Yuan et al., 2010; Tang et al., 2013), probably because of a knowledge gap in the biology of the two targeted species.

In Lake Erhai, previous investigations have demonstrated that the two invasive gobies displayed strong differences in some important biological and ecological features such as habitat preference (Guo et al., 2012), reproductive biology (Guo et al., 2013) and diet composition (Guo et al., 2014). Currently, the principal management action is based on physical removal by the local fishery using benthic nets, without considering population biology (e.g. age, growth and population dynamics) and the potential differences between the two species. These life-history traits are, however, potentially important for the improvement of the current management programs in the lake as they can influence individual capturability and could therefore be used to optimize removal efforts. In the present study, we investigated and compared the age structure, growth rates and population dynamics of the two invasive gobies. Specifically, age structures were determined using scale reading and recruitment structure analysis, while growth rates were determined by monthly changes in body length-frequency and von Bertalanffy growth function. Population dynamics were determined by quantifying mortality, mean size and mean age at maturity, maximum expected life-time and monthly changes of catch per unit effort of the fyke nets. Based on this knowledge on the two invasive species, we then provide several specific suggestions to improve the cost-efficiency of the management practices.

Materials and methods

Study sites

This study was carried out in Lake Erhai (105°5-17′E, 23°35-58′N), a shallow freshwater lake in Yunnan-

Guizhou plateau, southwest China (further details about environment characteristics of the lake in Guo et al., 2012). The dominant fish in the lake are small-bodied and non-native species, including freshwater gobies (*R. giurinus* and *R. cliffordpopei*), *Neosalanx taihuensis* (Chen, 1956), *Pseudorasbora parva* (Temminck & Schlegel, 1846), *Hypseleotris swinhonis* (Günther, 1873) and *Hemiculter leucisculus* (Basilewsky, 1855) (Guo et al., 2012; Tang et al., 2013). *Channa argus* (Cantor, 1842) is the only recorded piscivorous fish species and it is a rare species in the lake (Tang et al., 2013). *R. giurinus* and *R. cliffordpopei* are the most abundant benthic fish species with annual yields representing about 48% of total fish yields (kg) in the lake (Guo et al., 2012). The two species display strong habitat segregation with *R. cliffordpopei* being highly abundant in the littoral habitat (water depth < 6 m, high abundance of submerged macrophytes, substrate composed of submersed-macrophyte detritus) and with *R. giurinus* being highly abundant in the profundal habitat (water depth ranges from 12 to 20 m, no submerged macrophytes, substrate composed of silt and coarse sand, Guo et al., 2012). In the present study, sampling was performed in the preferred habitat of each species (i.e. *R. cliffordpopei* sampled in the littoral habitat and *R. giurinus* sampled in the profundal habitat) to ensure a representative assessment of the biological characteristics of each species.

Data collection

The two species were sampled monthly from October 2009 to October 2011 using benthic fyke nets with stretched mesh size of 0.4 cm. Total length of the net was 15 m, including 12 m of traps, 2 m of end traps and 1 m of end pockets. In the first week of each month, four nets were deployed per sampling site at 8:00–12:00 with a stone in the each end of nets to fix them at bottom. After 24 h, all fish in the end pockets were identified and batch-weighted for each species. The water temperature was measured at each site in each sampling occasion. For each month, we pooled all individuals of *R. cliffordpopei* from the littoral habitat and all *R. giurinus* from the profundal habitat. Then, random sub-samples (about 400–800 individuals for each species in each month) were collected and frozen at −20°C prior to analyses. Sub-sampled individuals were measured for total length (L_T) and body mass (M_T) to the nearest mm and 0.01 g, respectively. For each month, scales were collected randomly above lateral line on 30 individuals for age determination. The scales were immersed in 10% NaOH for three hours, cleaned with running water and read under an optical microscope using the criteria of Steinmetz & Müller (1991). Only scales with clearly identified rings were used for the analysis (72.1% for *R. cliffordpopei* and 75.8% for *R. giurinus*). Finally, sex was determined from another sub-sample (about 200–300 individuals each month for each species) based on the macroscopic appearances of gonads (see details in Guo et al., 2013).

Statistical analyses

Monthly body length-frequency was used to analyse the recruitment structure. Kolmogorov–Smirnov tests (K–S tests) were used to test for differences in body length-frequency of the entire population between the two species in each month. Growth patterns were first estimated using monthly changes in mean total length (L_T, average body length of individuals) and von Bertalanffy growth function (VBGF, Von Bertalanffy, 1938) using the equation $L_t = L_\infty \left[-e^{-K(t-t_0)}\right]$, where L_t is L_T at time t, L_∞ is the asymptotic length, K is the von Bertalanffy growth coefficient, t_0 is the theoretical age at $L_T = 0$. t_0 was calculated as $\log_{10}(-t_0) = -0.392 - 0.275 \log_{10} L_\infty - 1.038 \log_{10} K$ (Pauly, 1979). The function parameters were estimated using Electronic Length Frequency Analysis (ELEFAN) in the FiSAT II (FAO–ICLARM Fish Stock Assessment Tools, Version 1.2.2, Gayanilo et al., 2005). ELEFAN estimates the function parameters using sequential length-frequency data in the software. The K-value was estimated by a plot of the fit index (i.e. $Rn = 10^{\mathrm{ESP}/\mathrm{ASP}}/10$, where ESP is "explained sum of peaks" and ASP is "available sum of peaks") versus K (see details in Gayanilo et al., 2005; Scalici & Gibertini, 2009). The curvature parameters of the population were estimated when Rn was highest in the diagram (Scalici & Gibertini, 2009).

Population dynamics of the two species were described using mortality, mean size at first sexual maturity (L_{50}), mean age at first sexual maturity (T_{50}), expected life-time (T_m) and catch per unit effort (*CPUE*) of the fyke nets. Specifically, mortality was calculated using the FiSAT II (Gayanilo et al., 2005). Total mortality (Z, i.e. the sum of natural and fishing mortality) was estimated using length-converted catch curves based on Powell–Wetherall Plot equation (Powell, 1979; Wetherall, 1986). The asymptotic length and the ratio between the mortality coefficient and the curvature parameter (Z/K) were calculated using length-frequency data in the FiSAT (Scalici & Gibertini, 2009). This method pools a long series of samples that represent a steady-state population and generate a single frequency distribution accounting for their relative importance. Z is then calculated on the descending part of this single global distribution using regression analysis (Pauly, 1983). The data points on the descending part were selected based on the best consequences of regression analysis (Gayanilo et al., 2005). Natural mortality (M) is correlated with asymptotic length (L_∞), the von Bertalanffy growth coefficient (K) and mean environmental temperature (T) by the Pauly's M equation (Pauly, 1987): $\ln(M) = -0.015 - 0.279\ \ln(L_\infty) + 0.654\ \ln(K) + 0.463\ \ln(T)$. The mean annual temperature used for M estimation was 18.7°C. It was averaged from the monthly temperature of the lake measured from October 2009 to October 2011. Fishing mortality (F) was then obtained by subtracting M from Z. The exploitation ratio (E) was calculated as $E = F/Z$ (Pauly, 1983). L_{50} was defined as the L_T at which 50% of the individuals were at an advanced maturation stage (i.e. gonads at stage IV and V; Guo et al., 2013). It was quantified using logistic regression models with binomial maturity data (immature 0, mature 1) at any L_T of the entire population. T_{50} was determined as $T_{50} = \{-\ln[1-(L_{50}L_\infty^{-1})]\,K^{-1}\} + t_0$ (García Vásquez et al., 2009). The maximum expected life-time (T_m) was calculated as $\log_{10}(T_m) = 0.549 + 0.957\ \log_{10}(T_{50})$ (García Vásquez et al., 2009). Finally, monthly catch per unit effort ($CPUE$, ind. $\text{net}^{-1}\ \text{day}^{-1}$) from fyke nets was calculated. Differences in $CPUE$ (log-transformed) between species and months were tested using repeated-measures ANOVA (RM-ANOVA) with species as a fixed factor and month as a random factor. Within a month, Mann–Whitney tests were subsequently used to test for differences in $CPUE$ between species if species had a significant effect on $CPUE$ in the RM-ANOVA. The same approaches were used to test the differences of $CPUE$ between sexes (fixed factors) within species and months (random factors). The normality and variances homogeneity of data were tested using K–S test and Levene's test. Statistical analyses were performed using R version 2.14.0 (R Development Core Team, 2011).

Results

Sample size

Over the study period, a total of 14,462 individuals of *R. cliffordpopei* and 16,793 individuals of *R. giurinus* were measured from October 2009 to October 2011 (Fig. 1). From November to May 2010 and 2011, 3935 *R. cliffordpopei* were individually determined for sex (2296 males and 1639 females). From January to July 2010 and 2011, 3297 *R. giurinus* individuals were determined for sex (1690 male and 1607 females).

Age and recruitment structure

Based on age determination from scale reading, the maximal age of the two species was 1^+. Based on body length-frequency, the three recruitment periods were detected for each species. In *R. cliffordpopei*, the three recruitments born in 2009, 2010 and 2011 were observed from October 2009 to May 2010, April 2010 to June 2011 and April 2011 to October 2011, respectively (Fig. 1a). In *R. giurinus*, the individuals born in 2009, 2010 and 2011 were observed from October 2009 to July 2010, July 2010 to August 2011 and July 2011 to October 2011, respectively (Fig. 1b). Body length-frequency analyses clearly revealed that adults die after the spawning season and their lifespan ranges 1–2 years. Body length-frequency between the two species was significantly different from December 2009 to October 2010 and from April to October 2011 (K–S tests, $P < 0.001$). There was no significant difference in the other months (K–S tests, $P > 0.05$). Moreover, based on the identified males and females of *R. cliffordpopei*, we found a single recruitment from November 2009 to May 2010 (the recruitment was born in 2009) and another single one in next cycle (the recruitment was born in 2010). Males and females displayed significantly different body length-frequency in December 2009, January and April 2010 and November and March 2011 (K–S test, $P < 0.05$). Similarly, based on the identified males and females of *R. giurinus*, we found a single recruitment from January to July in 2010 and another one in 2011). Except in April 2010, males and females displayed similar body length-frequency (K–S test, $P > 0.05$).

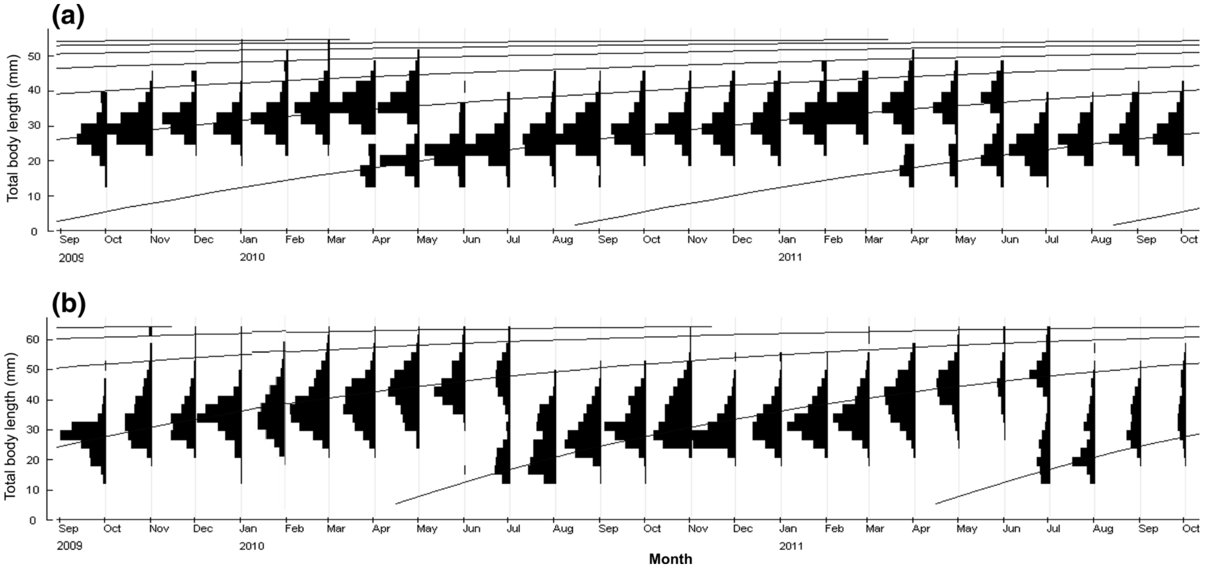


Fig. 1 Growth lines of **a** *R. cliffordpopei* and **b** *R. giurinus* obtained by Electronic Length Frequency Analysis (ELEFAN)

Body length and growth parameters

Mean total body length (T_L) of males and females *R. cliffordpopei* steadily increased from November to February but there was almost no further increase for females from March to May (Fig. 2a). With the exception of February, November and December 2010, males showed significantly higher T_L than females (Mann–Whitney test, $P < 0.05$). In *R. giurinus*, both sexes showed a similar increase in T_L from January to July (Mann–Whitney test, $P > 0.05$, Fig. 2b). For the entire population, T_L of *R. cliffordpopei* peaked in March and dropped to the lowest values in July, while *R. giurinus* showed the highest T_L June and the lowest T_L in August (Fig. 2c). The T_L differed significantly between the two species in November 2009, April to July 2010, and April to October 2011 (Mann–Whitney test, $P < 0.05$), except in August 2011. The asymptotic length (L_∞) was 55.65 mm for *R. cliffordpopei* and 68.25 mm for *R. giurinus*. The growth coefficient (K) was 0.58 for *R. cliffordpopei* and 0.99 for *R. giurinus*. The von Bertalanffy growth function was $L_t = 55.65\ [-e^{-0.58\ (t\,+\,1.48)}]$ for *R. cliffordpopei* and $L_t = 68.25\ [-e^{-0.99\ (t\,+\,0.78)}]$, respectively (Fig. 1).

Mortality, sexual maturity, expected life-time and abundance dynamics

R. cliffordpopei displayed higher total mortality ($Z = 2.12$), fishing mortality ($F = 1.25$) and exploitation rates ($E = 0.59$) than *R. giurinus* ($Z = 1.73$, $F = 0.56$, $E = 0.32$, respectively), whereas the natural mortality of *R. cliffordpopei* ($M = 0.85$) was lower than *R. giurinus* ($M = 1.17$, Fig. 3). The mean size at first sexual maturity (L_{50}) and the mean age at first sexual maturity (T_{50}) were 35.4 mm and 0.33 years for *R. cliffordpopei*, which were lower than 46.9 mm and 0.39 years for *R. giurinus*. The maximum expected life-time (T_m) was 1.23 and 1.45 years for *R. cliffordpopei* and *R. giurinus*, respectively.

At the population level, catch per unit effort (*CPUE*) differed significantly between species and months (RM-ANOVAs, $P < 0.001$, Table 1). *CPUE* was the lowest in winter (December to February, Fig. 4b), corresponding to the periods of the lowest water temperature (Fig. 4a). In addition, *CPUE* was low from April to June for *R. cliffordpopei* and from July to September for *R. giurinus*. The highest *CPUE* was observed from July to September for *R. cliffordpopei* and from April to June and from September to November for *R. giurinus* (Fig. 4b). Moreover, *CPUE* differed significantly

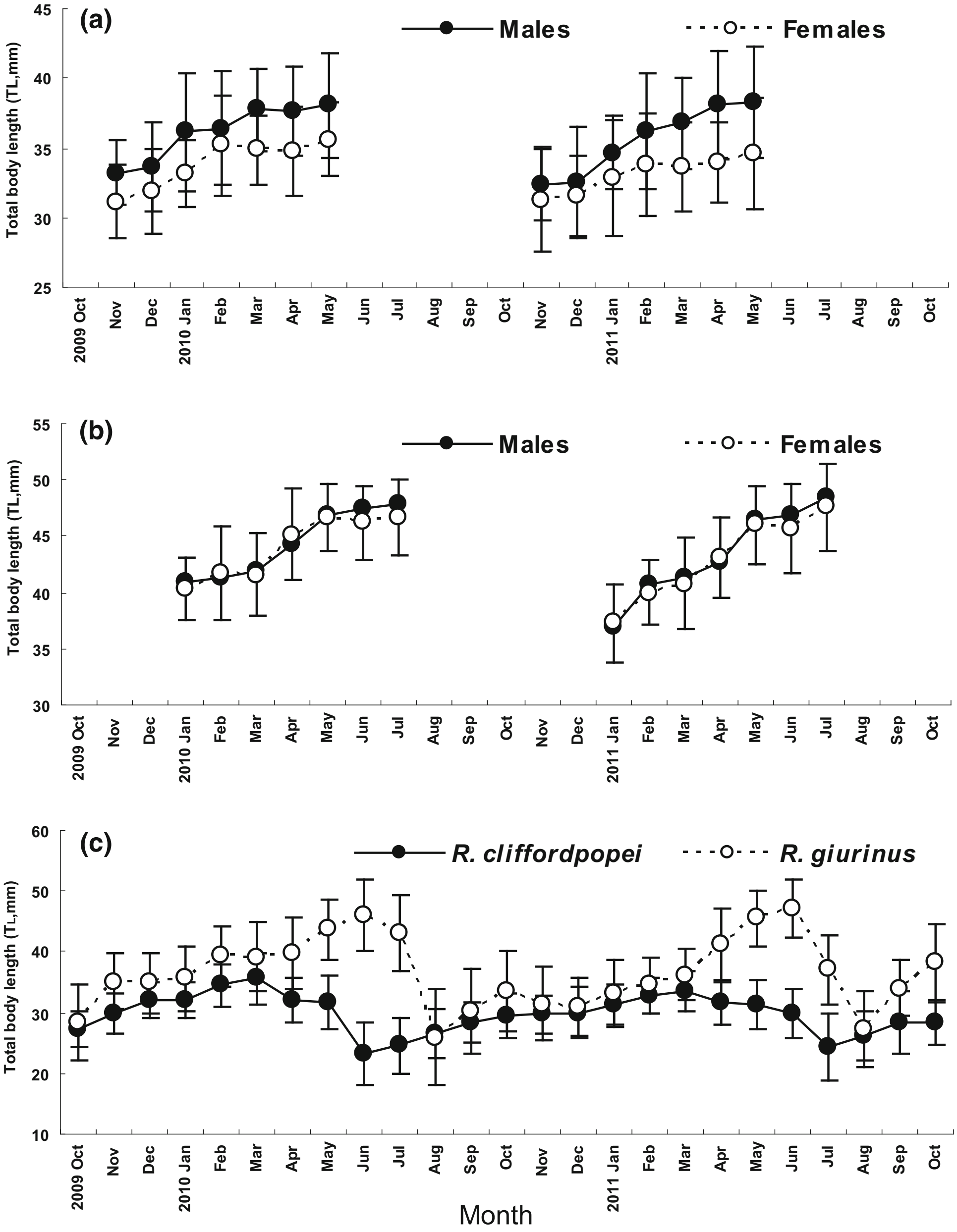


Fig. 2 Total body length for males and females of **a** *R. cliffordpopei* and **b** *R. giurinus*, and **c** all specimens of each species in Lake Erhai (China) from October 2009 to October 2011. The absence of data in (**a**) and (**b**) indicates that individuals were immature and not identifiable for sexes

between sexes for *R. cliffordpopei* (RM-ANOVAs, $P < 0.001$), while no significant differences between sexes were observed for *R. giurinus* (RM ANOVAs, $P = 0.122$, Table 1, Fig. 4c, d). The females of *R. cliffordpopei* displayed significantly higher *CPUE* than males in November 2009, February to May 2010 and 2011 (Mann–Whitney test, $P < 0.05$, Fig. 4c).

Discussion

The lifespan of gobies varies greatly between species (Azevedo & Simas, 2000; Scalici & Gibertini, 2009; Shemonaev & Kirilenko, 2009) and several species have been reported to display short-lived life cycle (e.g. about 1 year in tidewater goby *Eucyclogobius*

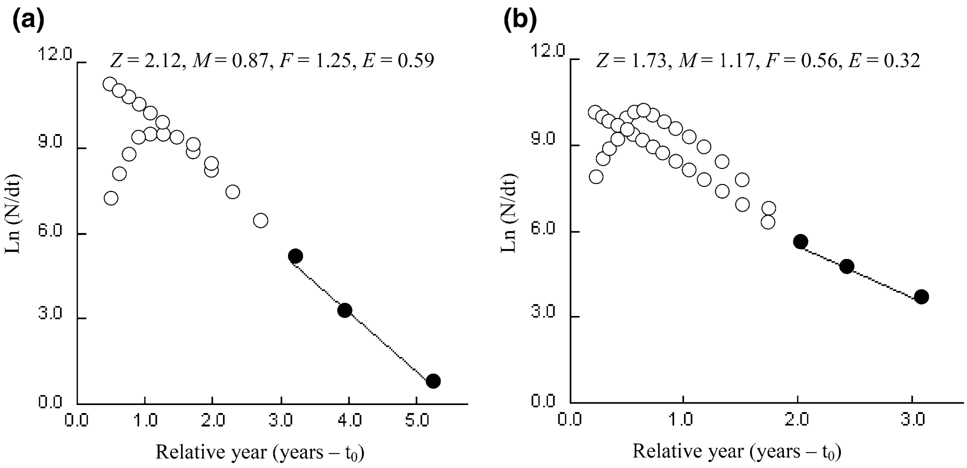


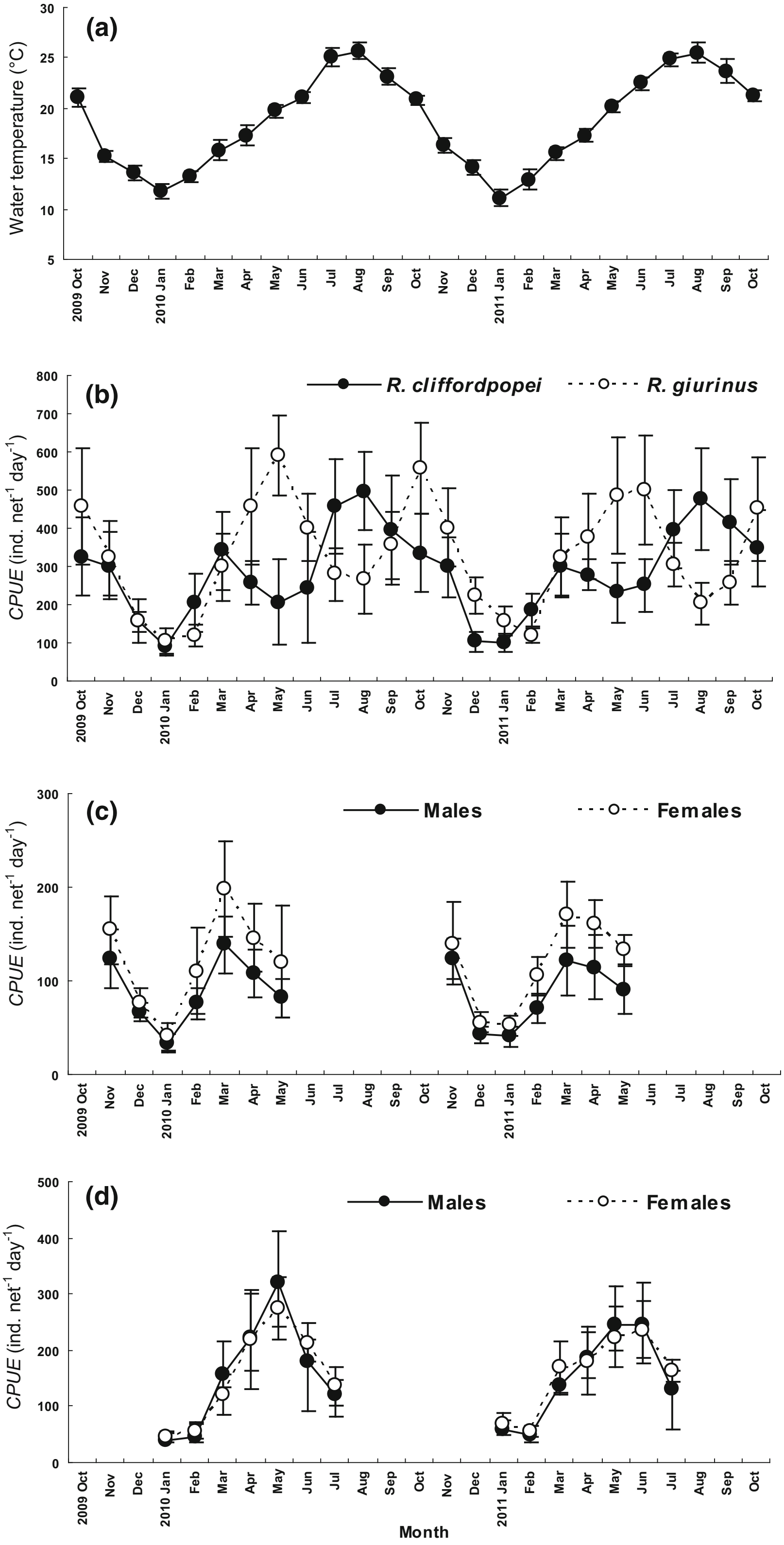
Fig. 3 Body length-converted catch curve and mortality estimates for **a** *R. cliffordpopei* and **b** *R. giurinus* in Lake Erhai (China) from October 2009 to October 2011. *Z*, total mortality; *M*, natural mortality; *F*, fishing mortality; *E*, exploitation rate. The regression was fitted using *filled circle* data points on the curve

Table 1 Repeated-measures ANOVAs (RM-ANOVA) used to test differences in catch per unit effort (*CPUE*, log-transformed) of *R. giurinus* and *R. cliffordpopei* between species (fixed factor) for the entire population, and between sexes (fixed factor) within species with sampling month used as the random factor in Lake Erhai (China) from October 2009 to October 2011

Sources	SS	d. f.	MS	*F*	*P*
Entire population					
Species	5.800	1	5.800	74.549	<0.001
Error	2.334	30	0.078		
Months	75.730	6.445	11.750	29.892	<0.001
Error	76.004	193.352	0.393		
Species × Months	83.056	8.671	9.579	32.784	<0.001
R. cliffordpopei					
Sexes	3.658	1	3.658	55.372	<0.001
Error	1.454	22	0.066		
Months	50.175	3.411	14.710	52.106	<0.001
Error	21.184	75.040	0.028		
Sexes × Months	5.282	3.549	1.488	3.406	0.014
R. giurinus					
Sexes	0.267	1	0.267	2.508	0.122
Error	4.043	38	0.106		
Month	170.028	3.549	47.913	109.653	<0.001
Error	58.923	134.849	0.437		
Sexes × Months	2.190	3.411	0.642	2.274	0.079

newberryi (Girard, 1856) (Swcnson, 1999) and decorated goby *Istigobius decoratus* (Herre, 1927) (Kritzer, 2002). In Lake Erhai, the maximal age of *R. giurinus* and *R. cliffordpopei* measured from scale reading was 1^+ year as it has been observed in native lakes along the middle and lower reaches of the Yangtze River (Zhang, 2005). The body length-frequency distribution of the two gobies was unimodal in most months and changed suddenly to bimodal when the recently hatched individuals were large enough to be sampled. The larger bodied adults subsequently die 2–3 months after spawning, indicating that the lifespan of the two species ranges from 1 to 2 years. Similarly, the maximal age was estimated at 1.23 years for *R. cliffordpopei* and 1.45 years for *R. giurinus*. Recently hatched individuals were captured

Fig. 4 Monthly values in **a** water temperature and catch per unit effort (*CPUE*) for **b** all individuals of each species, males and females of **c** *R. cliffordpopei* and **d** *R. giurinus* in Lake Erhai (China) from October 2009 to October 2011. The absence of data in (**b**) and (**c**) indicates that individuals were immature and not identifiable for sexes



at different periods between the two species, i.e. principally in April for *R. cliffordpopei* and in July for *R. giurinus*. *R. cliffordpopei* spawned from February to June with the main spawning activity from March to April, and *R. giurinus* spawned from April to August with the main spawning activity from May to June (Guo et al., 2013), which matched the spawning seasons of the two species. These biological traits and the differences between the two species are crucial for defining the mesh size of benthic fyke nets that could be used for physical removals, timing of deployment of control actions and control methods in different seasons (Ling, 2002; Britton et al., 2011). First, to remove small-sized individuals, the mesh size of benthic fyke nets should be smaller than 4 mm from March to June for *R. cliffordpopei* and from June to August for *R. giurinus*. *P. parva, H. swinhonis and H. leucisculus* are the other dominant fish species (relative abundance ≥10% of total catch) that were capture in fyke nets (Guo et al., 2012). These species are non-native in the lake and therefore fyke nets with small-sized mesh size are unlikely to impact substantially other fish species in the lake. In addition of fish, shrimp were captured in the fyke nets and further studies are needed to determine if fish removal practices might impact this taxa. Second, based on the characteristics of recruitment structure and spawning seasons, the current physical removals should be strengthened from September to February for *R. cliffordpopei* and from February to April for *R. giurinus* to maximize their efficiency because management operations are often most cost-efficient when implemented before spawning (Ludgate & Closs, 2003). Third, most small-sized individuals (i.e. larvae) are pelagic from February to June in *R. cliffordpopei* and from April to August in *R. giurinus* (personal observations). A similar pelagic stage of goby larvae has been observed in other species such as round goby, *A. melanostoma* (Hayden & Miner, 2009). Therefore, benthic fyke nets are less efficient for capturing larvae and we suggest that, during these periods, the efficiency of other control methods such as chemical treatments (e.g. rotenone, Ling, 2002), electro-fishing (Britton et al., 2011) and/or light trapping (Vilizzi et al., 2008) should be tested.

The von Bertalanffy growth model (von Bertalanffy, 1938) is widely used to determine fish growth, life-history traits and reference points for fishery management (Katsanevakis & Maravelias, 2008; Helidoniotis et al., 2011). However, growth patterns typically focused on long-lived demography from which the short-lived species are currently often overlooked (Helidoniotis et al., 2011). Few studies have estimated the growth parameters of freshwater gobies (e.g. Azevedo & Simas, 2000; Scalici & Gibertini, 2009) and, to the best of our knowledge, such information is extremely scarce for species of the genus *Rhinogobius* (Wu & Zhong, 2008). In the present study, the von Bertalanffy growth coefficient was 0.58 and 0.99 for *R. cliffordpopei* and for *R. giurinus*, falling within the range of values reported in the literature, e.g. 0.76–0.89 for rock goby *Gobius paganellus* (Linnaeus, 1758) (Azevedo & Simas, 2000) and 0.56–0.68 for Arno goby *Gobius nigricans* (Canestrini, 1867) (Scalici & Gibertini, 2009). In Lake Erhai, *R. giurinus* grew faster than *R. cliffordpopei* and displayed larger body size. The temporal fluctuations of mean body length of the two species were apparently different. Indeed, *R. cliffordpopei* showed the highest values in March and lowest values in June and July, while *R. giurinus* showed the highest values in June and the lowest in August. Combining the patterns of recruitment structures and monthly changes of mean body length should therefore be used as an important criterion for the selection of mesh size of benthic fyke nets for physical removals.

Significant variations in catch per unit effort (*CPUE*) of benthic fyke nets between months were mainly driven by changes in activity level and recruitment (Xie et al., 2000; Guo et al., 2013). In winter, *CPUE* of two species was extremely low which is not in accordance with patterns observed in Biandantang Lake, a shallow lake along the middle and lower reach of the Yangtze River (Xie et al., 2000). This discrepancy might have been caused by the sampling methods used. Indeed, benthic trap nets, like most of passive nets, are highly dependent on fish activity level and clearly differ from pop nets that were used by Xie et al. (2000). Thus, the observed lower *CPUE* of two species in winter was likely to result from a lower activity level at low water temperature. The low *CPUE* was also observed in April to June for *R. cliffordpopei* and in July to September for *R. giurinus*, coinciding with the death of adults after spawning. Afterwards, the *CPUE* sharply increased as a consequence of the increase in newly settled juveniles. These findings should therefore be incorporated into the management strategies and we suggest to decrease removal practices in winter and to transfer these efforts from July to September in the littoral habitats to control *R. cliffordpopei* and from April to June and September to November in the profundal habitats to control *R. giurinus*.

In conclusion, the present study provided new empirical knowledge on the biological characteristics of two invasive goby species outside of their native range and demonstrated that they are both short lived with lifespan ranging from 1 to 2 years but displayed different growth patterns and population dynamics. Consequently, specific management strategies could be designed. In the most lakes across Yunnan-Guizhou Plateau, current controls are only the physical removals by local fishery using benthic fyke nets. These techniques can be improved and synergistic management approaches should be developed and tested based on the population biology of the two gobies, such as chemical treatments, electricity fishing removals, light trapping and/or bio-controlling (Ling, 2002; Britton et al., 2011).

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