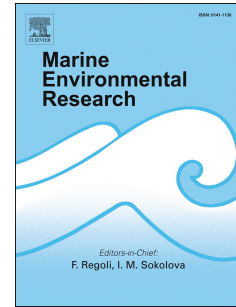


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Farming-up coastal fish assemblages through a massive aquaculture escape event.

Kilian Toledo-Guedes^{a,*}, Pablo Sanchez-Jerez^b, María E. Benjumea^a and Alberto Brito^a

^a BIOECOMAC. Universidad de La Laguna. Dpto. de Biología Animal. Ciencias Marinas. Facultad de Biología. Av. Astrofísico Francisco Sánchez s/n. CP 38206. San Cristóbal de La Laguna. Santa Cruz de Tenerife. Canary Islands, Spain. Phone: +34 922318387.

^b Department of Marine Sciences and Applied Biology. University of Alicante. Ap.C. 99. CP 03080. Alicante, Spain.

* ktoledo@ua.es

1 **Abstract**

2

3 We investigated the changes on the mean trophic level of fish assemblages across
4 different spatiotemporal scales, before and after a massive escape event occurred off La
5 Palma (Canary Islands), which resulted in the release of 1.5 million fish (mostly
6 *Dicentrarchus labrax*) into the wild. The presence of escaped fish altered significantly
7 the mean trophic level of fish assemblages in shallow coastal waters. This alteration was
8 exacerbated by the massive escape. A nearby marine protected area buffered the
9 changes in mean trophic level but exhibited the same temporal patterns as highly fished
10 areas. Moreover, escaped fish exploited natural resources according to their total length
11 and possibly, time since escapement. New concerns arise as a “farming up” process is
12 detected in shallow coastal fish assemblages where marine aquaculture is established.

13

14 **Keywords:** escaped fish, aquaculture, trophic level, diet, *Dicentrarchus labrax*, *Sparus*
15 *aurata*.

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35 1. Introduction

36

37 Aquaculture of high-trophic-level (HTL) fish species is growing, especially in
38 developed countries, as a result of a rising demand on these products and the highest
39 profit obtained from carnivorous species (Deutsch et al., 2007). This process has been
40 named “farming-up” (Pauly et al., 2001; Stergiou et al., 2008), and one of its major
41 concerns is the exploitation of wild fish stocks to feed high trophic level species, called
42 “tigers of the sea” by Naylor and Burke (2005). In addition, culturing non-native or
43 locally absent fish species is already a frequent practice (Casal, 2006; Arismendi et al.,
44 2009; Liao et al., 2010) that is predicted to grow in the next years (Shelton and
45 Rothbard, 2006). Thus, as a result of both mentioned trends, in some areas, HTL species
46 that were absent or with low abundances in natural habitats are being released into the
47 wild through escape events. Technical failures and sea storms provoke both recurrent-
48 small or punctual-massive escapes across the coasts where open-net cage aquaculture is
49 established (Jensen et al., 2010). This process could be comparable to continuous
50 restocking actions with non-indigenous or locally absent species (Lorenzen et al., 2012),
51 which benefits have been pointed by some authors (Briggs, 2008; Schlaepfer et al.,
52 2010) but are, in general, not recommended due to the unpredictable negative effects
53 they could have (Courtenay et al., 2009; Ricciardi and Simberloff, 2009).

54

55 Many studies have pointed out potential and detected consequences due to the release of
56 fish (exotic or not): genetic hybridisation (McGinnity et al., 2003); predation on native
57 species (Albins, 2013; Green et al., 2012); competition for trophic resources (Declerck
58 et al., 2002); introduction of parasites and diseases (Arechavala-Lopez et al., 2013);
59 changes in fisheries dynamics (Dimitriou et al., 2007), among others. Recently, it has
60 been demonstrated that marine ecosystems are much more susceptible to large-scale
61 invasion pressures than previously thought (Edelist et al., 2013). But even if escaped
62 fish do not establish self-reproducing populations, they may produce persistent impacts
63 due to the repeated supply of propagules through new escape events (Arismendi et al.,
64 2009; Jensen et al., 2010). Given the mobility of escapees (González-Lorenzo et al.,
65 2005; Arechavala-Lopez et al, 2011, 2012), they could affect particularly important
66 areas such as marine protected areas (MPAs). However, it has been suggested that
67 MPAs could show some resilience (*sensu* Holling, 1973, "the amount of disturbance
68 that an ecosystem could withstand without changing self-organized processes and

69 structures", but see Gunderson, 2000 for a review of the concept) to the effects caused
70 by different impacts, including species introduction, as assemblages within them are
71 expected to have a better conservation state (Stachowicz et al., 1999).

72

73 In the Canaries, where finfish production in open-net cages during 2009 was 7,910 tons
74 (APROMAR, 2012), European sea bass (*Dicentrarchus labrax*) and gilthead sea bream
75 (*Sparus aurata*) have been introduced in some of the islands where no natural
76 populations of these species existed (Brito et al., 2002; Toledo-Guedes et al., 2009).
77 That is the case of La Palma Island, where a massive escape event occurred between
78 December 2009 and January 2010. Repeated northwest sea storms generating waves up
79 to 6 meters height resulted in both lack of maintenance operations and increased
80 mechanical stress for aquaculture facilities (Ramírez et al., 2011; Puertos del Estado,
81 2012). As a result, around 1.5 million fish (90% sea bass and 10% sea bream) were
82 released into the wild during that period (Ramírez et al., 2011). A previous study
83 revealed that escaped fish entered a nearby (~15 km) MPA and their abundances within
84 were similar to those found in other areas of the island (Toledo-Guedes et al., 2014). As
85 far as we know, this is the largest sea bass escape event documented to date worldwide.

86

87 We capitalize on this event to examine the potentiality of escaped fish to alter the mean
88 trophic level (mTrL) of fish assemblages in shallow coastal waters and discuss the
89 potential consequences of these changes. In particular we studied i) if fish assemblages
90 mTrL was affected by the massive escape of HTL fish, ii) if the magnitude in mTrL
91 alteration was related to the presence of a MPA and iii) the trophic role of escaped sea
92 bass in coastal waters. For that we analyse the spatiotemporal variation of mTrL before
93 and after the massive escape event, using the estimation of fish abundances and size by
94 visual census in shallow coastal waters, and additionally we studied the diet of fugitive
95 sea bass, in relation to size, through stomach content analysis.

96

97 **2. Material and Methods**

98

99 *2.1. Study site and sampling effort*

100

101 Our study was carried out in La Palma (Fig. 1), one of the westernmost islands of the
102 Canarian archipelago, situated in the north-eastern part of the Central Atlantic (28°40'N,

103 17°52'W). Aquaculture facilities are in a single location off the western coast. A marine
104 protected area (MPA) is situated 15 kilometres to the south from fish farms.

105

106 A total of 6 localities (Fig. 1), and three sites (n=6) in each locality, were sampled by
107 means of visual census (see next section), at different distances from release point (0.8
108 to 30 km). Three of the localities were situated in La Palma MPA, the other three,
109 outside the MPA, were considered as highly fished areas (HFA) following Sangil et al.,
110 2013a. Each locality was sampled four times: March 2009, October 2009, March 2010
111 and October 2010. A total of 432 visual censuses were carried out through the study.

112

113 2.2. *Visual censuses*

114

115 Based on previous methodology (Toledo-Guedes et al., 2009), snorkelling visual
116 censuses of escapees were performed in transects of 100x5 m, between 1 and 5 m depth.
117 In the initial 25 m, all the fish species abundances and sizes were recorded, while across
118 the rest of the survey only escaped fish were counted. A second pass of the same
119 transects served to establish habitat heterogeneity and complexity, measuring the cover
120 % of different habitats and habitat features; sandy bottom, rocky platform, cliff and
121 boulders classified by the size of their major length (ML): small boulders-SB (ML≤50
122 cm), medium boulders-MB (50 cm<ML≤1 m), and large boulders-LB (ML>1 m)
123 (García-Charton et al., 2004).

124

125 2.3. *Mean trophic level calculation*

126

127 Length estimates of fish from surveys were converted to weight by using the allometric
128 length–weight conversion:

$$129 W = aTL^b,$$

130 where W is weight in grams (i.e. biomass), parameters *a* and *b* are constants obtained
131 from the literature (Froese and Pauly, 2012), and TL is total length in cm. When values
132 for *a* and *b* were unavailable, the parameters from a congeneric species with similar
133 shape and maximum total length were used.

134

135 Mean trophic level of the fish assemblage in each transect (mTrL_t) was then calculated
136 as follows:

137 $mTrL_t = \sum_i (TrL_{i-n} \cdot W_{i-n}) / \sum_i W_{i-n}$ (Pauly, 1998; CIESM, 2000),
138 where the summation of trophic level of each species (TrL_{i-n}) recorded in the transect,
139 multiplied by their weight (W_{i-n}), is divided by the total weight amounted in the same
140 transect. Trophic levels for each species were recorded from FishBase (Froese and
141 Pauly, 2012).

142

143 *2.4 Statistical analysis*

144

145 *2.4.1. Overall analysis*

146

147 Possible relations (i.e. direct trophic interactions) between the presence of escaped fish
148 and the abundance of other species were explored through Spearman's correlation
149 index.

150

151 To ascertain whether the mTrL of shallow coastal fish assemblages is altered by the
152 presence of escaped fish, we compared untransformed mTrL of transects with no
153 presence of escaped fish against those transects with presence of escapees across the
154 study. Due to the unbalanced nature of the analysis, a PERMANOVA (Anderson, 2001)
155 test was carried out over Euclidean distance matrix and 4999 permutations, using
156 distance to fish farm and $\arcsin(x+1)$ transformed environmental variables as covariates.
157 The latter allowed detecting differences in mTrL irrespective of the proven
158 environmental influence on fish assemblages (García-Charton et al., 2004).

159

160 Kolmogorov-Smirnov test was used to compare size frequency of both sea bass and sea
161 bream visual counts before and after the massive escape, aiming to test previous
162 hypothesis on the possible alteration of size frequency of escapees in the wild due to
163 punctual massive escape events (Toledo-Guedes et al., 2009).

164

165 *2.4.2. Spatiotemporal analysis*

166

167 Univariate PERMANOVA (Anderson, 2001) tests were performed over untransformed
168 mTrL to detect spatiotemporal patterns of change. Euclidean distances matrix and 4999
169 permutations were used. A five-factor design was constructed as follows:

170

171 *Year* – *Ye* – Fixed, two levels (2009, 2010). Test de influence of the massive escape
172 event over the analysed variables.

173 *Season* – *Se* – Fixed, two levels (March, October). Test possible cold versus warm
174 seasonal changes due to a higher winter release of farmed fish (Toledo-Guedes et al.
175 2014).

176 *Protection* – *Pr* – Fixed, two levels (marine protected area –MPA–, highly fished area –
177 HFA–). Test for differences in mTrL between MPA and HFA.

178 *Locality* – *Lo* – Random, nested in *Protection* (three levels).

179 *Site* – *Si* – Random, nested in *Locality* (three levels).

180

181 Again, environmental variables (arcsin+1 transformed), and distance to release point in
182 km, were added as covariates to remove their possible effect over mTrL. As the random
183 factor *Locality* remained not significant (p-value=0.657) in the first analysis, it was
184 pooled to gain power of analysis (Underwood, 1997). For the interpretation of the
185 results, significant interaction terms with random factors involved were not taken into
186 consideration, as the higher level fixed factor effect remains relevant regardless of the
187 outcome of the interaction with a random factor (Quinn and Keough, 2002).

188

189 2.4.3. Escapes vs. wild assemblages analysis

190

191 To assess the importance of escaped fish in the study area, their biomass ($\text{g } 100\text{m}^{-2}$) was
192 compared with that of other species with similar trophic level. Biomass of escaped sea
193 bass (trophic level 3.8 ± 0.6 ; Froese and Pauly, 2012) was compared with the biomass of
194 species whose trophic level is higher than 3.5 (i.e. medium-top predators). This group of
195 species was composed of potential sea bass predators and competitors: *Seriola* spp.,
196 *Pomatomus saltatrix*, *Mycteroperca fusca*, *Sphyrnaena viridensis*, *Aulostomus strigosus*,
197 *Scorpaena maderensis*, *Belone belone*, *Pseudocaranx dentex*, *Mustelus mustelus*,
198 *Pomadasys incisus*, *Epinephelus marginatus* and *Trachinotus ovatus*. This was also
199 done for sea bream (trophic level 3.3 ± 0.5 ; Froese and Pauly, 2012); in this case, we
200 compared against species with a trophic level between 3 and 3.5. This group was
201 composed of sparids: *Diplodus cervinus*, *Diplodus sargus*, *Oblada melanura* and
202 *Lithognathus mormyrus* and other species whose diet is composed mainly of small
203 crustaceans: *Thalassoma pavo*, *Canthigaster capistrata*, *Sphoeroides marmoratus* and
204 *Symphodus trutta*. Pair-wise comparisons were made for each area (MPA and HFA) and

205 time period; U-Mann Whitney test served to assess differences in the mean biomass of
206 the groups as normality was not met.

207

208 2.5. Stomach content analysis

209

210 Individuals of *D. labrax* (n=144) were caught by spearfishing. A total of 112 escaped
211 fish were captured during surveys in Tenerife and La Palma in 2008 and 2009. These
212 were not associated to any known massive escape event; therefore, this group of fish
213 was assigned to recurrent leaking escapees (leak group). On the other hand, 32 fish were
214 caught in June 2010 in La Palma Marine Protected Area and, thus, due to the recent
215 massive escape and their schooling behaviour, were assigned to that event (massive
216 group).

217

218 All fish were measured (total length TL) to the nearest mm and weighted (accuracy of
219 0.01 g). The stomach intestine was separated from the body and its contents removed.
220 Prey items were counted by number, fresh weighted and identified to the lowest
221 possible taxonomical level. Thus, for each prey, percentage by number (N%) and
222 weight (W%), frequency of occurrence (O%) and the alimentary coefficient ($Q = N\% \times$
223 $W\%$) were calculated (Hureau, 1970). The importance of prey groups was assessed
224 using the following categories (based on values of Q and O%; Rosecchi and Nouaze,
225 1987): main preferred prey ($Q > 100$, $O\% > 30\%$); main occasional prey ($Q > 100$,
226 $O\% < 30\%$); secondary common prey ($10 < Q < 100$, $O\% > 10\%$); secondary additional prey
227 ($10 < Q < 100$, $O\% < 10\%$); accidental prey ($Q < 10$). The index of relative importance (IRI)
228 (Pinkas et al., 1971) was also estimated, $IRI = (N\% + W\%) \times O\%$. Trophic level of both
229 groups of escaped sea bass were calculated as the mean weighted trophic level of the
230 food items plus one (Froese and Pauly, 2012).

231

232 With the aim of detecting possible differences in the diet of recent escapees, non-metric
233 Multidimensional Scaling (MDS) was performed over Bray Curtis similarity matrix of
234 the weights of the three main prey groups found in stomach contents (i.e. insects,
235 crustaceans and fish). Moreover, PERMANOVA test was carried out comparing the diet
236 of the two groups using total length (TL) as a covariate, given that the diet of sea bass
237 changes during its life cycle in the wild (Kelley, 1987; Rogdakis et al., 2010). For the
238 statistical analyses, SPSS 15.0 and PRIMER6 & PERMANOVA+ were used.

239

240 3. Results

241

242 3.1. Overall results

243

244 None of the pair-wise Spearman's correlations between abundances of escapees and the
245 rest of wild fish species resulted significant.

246

247 Mean trophic level of shallow coastal fish assemblages was significantly increased
248 ($p < 0.001$) in those surveys where the presence of escaped fish was recorded
249 ($mTrL \pm SE = 3.16 \pm 0.55$) in comparison to those transects with no escaped fish
250 ($mTrL \pm SE = 2.78 \pm 0.41$; Table 1 and Fig. 2).

251

252 Size frequency distributions in visual counts showed significant differences for sea bass
253 ($Z = 15.110$; $p < 0.001$) and sea bream ($Z = 6.948$; $p < 0.001$) before and after the massive
254 escape event. Mean TL of escaped sea bass (28.01 ± 5.51 cm) and sea bream (28.39 ± 8.08
255 cm) were clearly increased after the massive escape: sea bass and sea bream mean TL
256 was 40.35 ± 9.3 cm and 43.73 ± 3.52 cm, respectively (Fig 3a and b).

257

258 3.2. Spatiotemporal patterns

259

260 As the random factor *Locality* remained not significant (p -value=0.657), it was pooled
261 to gain power of analysis. In this way, PERMANOVA detected spatiotemporal patterns
262 of $mTrL$, these patterns consisted in higher $mTrL$ in 2010, during March and at HFA
263 localities when they are compared to 2009, October and MPA localities respectively
264 (Table 2; Fig. 4).

265

266 3.3. Escapees vs. wild assemblages

267

268 Pair-wise comparisons showed that biomass of escaped sea bass at the two sampled
269 areas was equal or higher than the sum of the other medium-high trophic level species
270 for all the sampled periods with the exception of October 2010 (Fig. 5a). Conversely,
271 biomass of escaped sea bream resulted always significantly lower than the biomass of
272 other species with similar trophic level (Fig. 5b).

273

274 *3.4. Use of trophic resources*

275

276 Table 3 shows the diet composition and importance of each prey for the two groups of
277 sea bass analysed. For the leak group (Table 3a), fish was the main prey group,
278 followed by crustaceans, being the rest of the prey groups classified as accidental. It is
279 remarkable that only one individual was found with pellets in the stomach. Lower prey
280 diversity is observed in sea bass associated with massive escape (Table 3b). In this case,
281 the main prey group was crustaceans; the most preferred being the decapod *Percnon*
282 *gibessi*. Regarding the vacuity index, leak group had a 33.9% of empty stomachs, while
283 the massive group showed a 12.5%. Trophic level of the massive escape group was
284 lower (3.2) than that of the leaking group (4.2).

285

286 MDS indicated that sea bass associated with massive escape in La Palma had a different
287 diet in comparison to the leak group (Fig. 6a.). The ordination responds to the
288 importance of the three prey groups in the diet of each individual (Fig 6bcd). The group
289 denoted as massive is situated in the area where crustaceans are the main item by weight
290 in the stomachs. PERMANOVA test confirms that sea bass of the leak and massive
291 groups had differing diets ($p < 0.001$), irrespective of their sizes (TL), but size resulted in
292 an important variable when explaining the diet of escaped sea bass ($p < 0.001$; Table 4).

293

294 **4. Discussion**

295

296 The input of HTL fish by aquaculture through escapes events generates a “farming up”
297 process over shallow wild fish assemblages, rising their mTrL. This alteration was
298 exacerbated by the massive escape that also changed the size frequency distribution of
299 escaped individuals in the wild. Although temporal patterns of mTrL were the same in
300 both HFA and MPA, the latter showed certain degree of resilience to alterations on
301 mTrL. Moreover, escaped fish exploited natural resources according to their total length
302 and, possibly, depending on the time at liberty.

303

304 A clear increase in mTrL is observed due to the escaped fish. However, these alterations
305 seem to be related to the direct presence of escapees. The cultured (and escaped) species
306 have a TrL well above the mTrL of native fish assemblages found in shallow coastal

307 waters in the area, thus, the presence of escapees in the wild inevitably provokes a rise
308 in this indicator. Consistent temporal trends revealed that mTrL was higher in 2010,
309 after the massive escape event, but also in March sampling period when compared to
310 October in both years. This is in concordance with previous studies that found the same
311 temporal trends for the abundance of escaped sea bass in the wild (Toledo-Guedes et al.,
312 2014). This would correspond to a higher release of farmed fish during winter storms,
313 reflected in March sampling (Toledo-Guedes et al., 2014). Our results show that,
314 although mTrL inside the MPA is altered, the magnitude of this alteration is not as large
315 as in HFA. In other words, MPA seems to exhibit certain resilience to changes in mTrL
316 if compared to HFA. The ability of MPA to buffer the rise in mTrL could be based on a
317 higher abundance of herbivorous (i.e. low trophic level) species. Owing to the
318 protection against fishing, an increment of predators of the barren-ground founder sea
319 urchin *Diadema africana* (Rodríguez et al., 2013) have been observed; subsequent
320 depletion of the latter and further recovery of erect algae together with herbivorous fish
321 species represents an already well-studied trophic cascade effect (Sangil et al., 2012).
322 Thus herbivorous biomass 'pulls down' the mTrL maintaining it at relative low levels
323 despite the MPA is equally affected by the presence of escaped fish (Toledo-Guedes et
324 al., 2014).

325

326 Although this study failed at detecting negative or positive correlations between number
327 of escapees and abundance of other fish species, direct or indirect effects (e.g. trophic
328 cascades) cannot be discarded in the mid and long term. Other studies have revealed
329 that the input of predators in a semi-closed systems can cause an alteration of trophic
330 interactions and cascade effects (Ojaveer et al., 2004; Casini et al., 2012). The same
331 scenario of relative isolation occurs in La Palma island as the absence of coastal shelf
332 (Acosta et al., 2003) promotes the concentration of shallow benthic-demersal fish
333 assemblages (including escapees) near the coast, limiting connectivity and dilution of
334 potential impacts among the islands (Sangil et al., 2013b).

335

336 Sea bass have become the most abundant predator in shallow coastal waters (between 1
337 and 5 m depth) in La Palma (this study) and probably in those islands where sea bass is
338 cultured (pers. obs.). Even in areas such as the MPA, where fish assemblages are
339 supposed to be best preserved and abundances of top predators reach maximum
340 (Newman et al., 2006), escaped sea bass outnumber in biomass the rest of medium-

341 high trophic level species. Actually, similar results have been indicated in other areas
342 where top predators have been introduced; as *Cephalopholis argus* and *Lutjanus*
343 *kasmira* in Hawaii (Friedlander et al., 2002; Dierking, 2007) or the red lionfish (*Pterois*
344 spp.) that is now established and in rapid expansion in the western North Atlantic
345 (Whitfield et al., 2002; Schofield, 2009). Nonetheless, the success of these invaders is
346 based upon their ability to close their life-cycle in natural habitats, while populations of
347 escaped fish are strongly dependant on new escapees in the Canaries (Toledo-Guedes et
348 al., 2009, 2012). It is necessary to remark that our results are valid in the depth strata we
349 have studied (i. e. very shallow coastal waters), where a big proportion of escaped fish
350 are found (Toledo-Guedes et al., 2009) and top predators are uncommon. In the case of
351 escaped sea bream, their abundances in the surveyed areas are not as high, and fish
352 species of their “trophic surroundings” are well represented by several species, mainly
353 sparids.

354

355 Functional diversity (sensu lato) has been proven to be an indicator of ecosystem health
356 and function (Clemente et al., 2010). It has been also suggested that it could prevent the
357 insertion of non-indigenous species in recipient ecosystems (Stachowicz et al., 1999).
358 Thus, the lack of top predators in the surveyed areas could favour escaped sea bass
359 (avoiding competition and predation), while abundance of sparids could avert the
360 success of escaped sea bream in the wild. Nonetheless, other reasons for a lower
361 prevalence of the latter seems to be the result of lower intensity of release, together with
362 a naive behaviour that could increase natural and fishing mortality in comparison to sea
363 bass (Arechavala-Lopez et al., 2012; Toledo-Guedes et al., 2014).

364

365 Our data on the diet of escaped sea bass supplement those previously published
366 (Toledo-Guedes et al., 2009). The results obtained for the individuals not associated to
367 any known massive escape (leak group) are in line with the mentioned study. On the
368 other hand, the diet of sea bass associated with the massive escape markedly differs to
369 the leak group in several aspects (e.g. lower percentage of vacuity, lower prey diversity
370 and lower trophic level). Even though the number of studied stomachs is limited,
371 vacuity percentage was very low for the massive group. This indicates that six months
372 after the massive escape event, surviving escapees were able to actively exploit
373 available resources in natural habitats as the MPA. Many studies have pointed to the
374 ontogenetic changes in diet suffered by sea bass in their natural range of distribution

375 (Kelley, 1987; Rogdakis et al., 2010 and references therein). In extensive cases, we can
376 say that trophic level of wild sea bass is positively related with the size (Rogdakis et al.,
377 2010). Our results support the importance of fish's total length in its diet, and highlight
378 that trophic interactions posed by escaped sea bass could largely depend on the size of
379 the fish that escape. Nonetheless, time at liberty seems to be another factor explaining
380 the observed diet, actually a "hunting learning" period has been already suggested for
381 escaped sea bream in the Mediterranean (Arechavala-Lopez et al., 2012). This
382 adaptation period would also account for the lower trophic level showed by recent
383 escapees (i.e. massive group), as they predated mainly over crustaceans that are less
384 mobile and thus, easier to catch than fish.

385

386 *4.1. Conclusions and final recommendations*

387

388 With this study we propose a further dimension of the concept 'farming up' related not
389 only to a rise in the trophic level of cultured species (as presented by Stergiou et al.,
390 2009) but also a rise of mTrL of wild fish assemblages in the coasts where aquaculture
391 facilities are present, due to the release of those farmed fish. If we scale up the issue, the
392 need for studies at wider spatial scales (e.g. Mediterranean) arises.

393

394 Taking into account the high mobility of the escapees, remediation through regular
395 eradication actions with highly selective fishing techniques (e.g. spearfishing) seems
396 unaffordable in terms of costs. However, as natural and fishing mortality apparently
397 control abundances of escaped fish (Toledo-Guedes et al., 2014), eradication efforts
398 should be centred in valuable areas where fishing pressure is low or absent as MPAs.
399 Adaptive management of these areas would prevent potential negative effects caused by
400 escaped fish. In any case, this would not be applicable to other regions (e.g.
401 Mediterranean), where wild counterparts cohabit with escapees because in most of the
402 cases, wild and escaped fish can only be differentiated after being caught (Arechavala-
403 Lopez et al., 2012).

404

405 A likely grow in aquaculture production worldwide, together with a concentration on a
406 few proven species (several of which would be exotic or locally absent; Shelton and
407 Rothbard, 2006) could lead to ethical and environmental issues. Thus, for a correct
408 development of aquaculture industry, the necessity of risk assessment plans grounded

409 on empirical data is clear. The implementation of mitigation, contingency and
410 restoration plans, has special relevance when aquaculture facilities are situated near
411 sensible areas (e.g. MPAs) and/or imply species introductions.

412

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631 **Captions**

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633 **Figure 1.** Study area. Black circle: aquaculture facilities / release point. White circles:
634 localities sampled outside La Palma MPA. White triangles: localities sampled at MPA.
635 Black line: limits of La Palma MPA.

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637 **Figure 2.** Box plot comparing mean trophic level of visual censuses with absence (grey)
638 and presence (black) of escaped fish.

639

640 **Figure 3.** Size frequency of escaped sea bass (a) and sea bream (b), before (grey) and
641 after (black) the massive escape event.

642

643 **Figure 4.** 3-dimensional plot of $mTrL_0$, $\Delta Troph$ and $mTrL$ at locality level through
644 sampling periods.

645

646 **Figure 5.** Mean biomass (\pm SE) of both escaped (a) sea bass and (b) sea bream (black
647 bars) and other fish species with similar trophic level (grey bars) at MPA and HFA
648 through sampling periods: n.s.: not significant, *: $p < 0.05$, **: $p < 0.01$, ***: $p < 0.001$.

649 Note the logarithmic scale in “y” axis.

650

651 **Figure 6.** MDS plot showing the ordination of sea bass individuals by diet according to
652 their (a) origin, and by weight of each group of preys: (b) insects, (c) crustaceans and
653 (d) fish.

1 **Appendix.** Mean biomass (grams 100 m⁻² ± SE) for each species recorded in the studied localities. Localities are numbered from North to South.
2

Species	March 2009						October 2009					
	1	2	3	4	5	6	1	2	3	4	5	6
<i>Dicentrarchus labrax</i>	289.8 ± 247.7	421.1 ± 403.3	42.9 ± 27.8	9.3 ± 8.9	1 ± 1.7	95.4 ± 78.2	88.4 ± 158.8	215,273.3 ± 152,105.3	5,483 ± 4,239.3	3,117.3 ± 3,166.8	4,716 ± 3,040	721 ± 502.7
<i>Sparus aurata</i>	9.3 ± 9.6	-	-	268.1 ± 319.6	1.3 ± 2.3	41.8 ± 60.6	32.1 ± 105.5	5,808.2 ± 10,060.1	-	-	489.4 ± 796.7	10.6 ± 18.3
<i>Abudefduf luridus</i>	58 ± 44.7	101.9 ± 89.3	192.4 ± 129.6	24 ± 14	189.4 ± 103.2	177.4 ± 74.1	167.4 ± 93.8	68.1 ± 78.5	104.7 ± 65.3	109.8 ± 83.9	7.6 ± 7.8	93.4 ± 71.2
<i>Atherina presbyter</i>	0.4 ± 0.3	0.1 ± 0.1	0.6 ± 0.7	0.1 ± 0.1	0.1 ± 0.2	-	0.5 ± 1.4	96.3 ± 166.9	5.1 ± 8.3	1.5 ± 2.5	-	1.4 ± 2.5
<i>Aulostomus strigosus</i>	-	-	-	-	-	-	3 ± 12.2	-	-	-	-	-
<i>Belone belone</i>	-	-	-	-	-	-	0.4 ± 2.1	-	-	-	-	-
<i>Boops boops</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Canthigaster capistrata</i>	-	0.4 ± 0.7	-	-	-	-	0.1 ± 0.4	-	-	-	-	-
<i>Diplodus cervinus</i>	-	-	-	1.8 ± 3.1	-	-	0.2 ± 1	-	-	-	77.8 ± 119.5	-
<i>Diplodus puntazzo</i>	-	-	-	-	-	-	-	-	-	-	96.8 ± 167.7	-
<i>Diplodus sargus</i>	1,024.5 ± 741.7	236.4 ± 314.4	1,875.7 ± 1,470.5	527.4 ± 279	384.2 ± 598	55.4 ± 51	556.5 ± 613.1	906.2 ± 497.1	5,848.4 ± 9,242.1	619.9 ± 342	2,401.6 ± 1,667.7	529.3 ± 546.7
<i>Epinephelus marginatus</i>	-	-	-	-	-	-	-	-	-	-	100.4 ± 130.7	-
<i>Gymnothorax unicolor</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Kyphosus saltatrix</i>	-	-	-	11.4 ± 19.7	-	-	15.3 ± 36.3	4 ± 7	34.2 ± 59.2	11.4 ± 19.7	23.3 ± 36.8	164.7 ± 285.2
<i>Liza aurata</i>	723.3 ± 1,120.6	-	600.4 ± 997	2,878.5 ± 2,350.1	2,304.8 ± 3,598.1	-	841.1 ± 1,571.9	7,052 ± 7,421.3	-	2,544.4 ± 4,407.1	4,316.1 ± 4,104.9	3,757.5 ± 3,989.4
<i>Lythognathus mormyrus</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Mugil cephalus</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Mustelus mustelus</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Mycteroperca fusca</i>	-	-	-	-	-	-	-	-	-	-	28.7 ± 49.6	-
<i>Oblada melanura</i>	1.4 ± 2.4	-	7.2 ± 8.6	-	-	-	1.3 ± 3.5	92.1 ± 159.5	-	-	6.9 ± 11.9	-
<i>Ophioblennius atlanticus</i>	240.7 ± 135.1	122 ± 79.4	275.5 ± 95.1	49.8 ± 42.1	201.1 ± 101.9	101.7 ± 42.3	321 ± 202.8	124.4 ± 105.8	123.1 ± 99.6	329.9 ± 162.2	23.5 ± 19.1	289.2 ± 181.1
<i>Pomadasys incisus</i>	-	-	-	-	-	-	32.6 ± 178.8	-	-	-	-	-
<i>Pomatomus saltatrix</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Pseudocaranx dentex</i>	-	2.6 ± 4.5	3 ± 5.2	-	-	-	1.1 ± 3.6	-	28.8 ± 49.9	-	-	-
<i>Sardinella aurita</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Sardinella maderensis</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Sarpa salpa</i>	13,944 ± 5,692.6	7,695.2 ± 3,141.6	13,269.6 ± 5,417.3	17,968.9 ± 7,335.8	2,247.8 ± 917.7	-	10,438.1 ± 4,261.3	3,765.2 ± 1,537.1	1,624.9 ± 663.4	11,221.3 ± 4,581.1	9,295.8 ± 3,795	16,404.2 ± 6,697
<i>Scorpaena maderensis</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Seriola spp.</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Sparisoma cretense</i>	29.6 ± 51.4	16.4 ± 25.9	403 ± 266.8	3,781.9 ± 1,946.1	178.9 ± 101.1	176.3 ± 127.6	525.3 ± 761.1	138.9 ± 110.9	48.1 ± 48.8	311.5 ± 195.1	3,900.9 ± 2,143.9	857.8 ± 736.7
<i>Sphoeroides marmoratus</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Sphyaena viridensis</i>	-	0.6 ± 1	-	-	0.3 ± 0.5	-	2.1 ± 7.2	46.8 ± 81.1	-	-	0.1 ± 0.1	-
<i>Symphodus trutta</i>	-	-	-	-	-	-	-	-	-	-	1.2 ± 2	-
<i>Thalassoma pavo</i>	72 ± 19.7	92.5 ± 56.5	107.8 ± 35.3	281.8 ± 122.2	224.6 ± 97	227.1 ± 99.6	214.6 ± 104.2	14.3 ± 13.2	324.7 ± 230.9	293.9 ± 127.1	155.8 ± 88.9	552.1 ± 495
<i>Trachinotus ovatus</i>	0.5 ± 0.9	-	59.6 ± 103.2	12.3 ± 21.3	-	-	10.6 ± 34.5	4 ± 6.9	11.4 ± 19.7	55.9 ± 72.7	83.3 ± 131	-
Unidentified larvae	-	1 ± 1.6	1.3 ± 1.7	-	-	-	0.5 ± 1	-	0.4 ± 0.4	0.3 ± 0.5	0.1 ± 0.2	0.1 ± 0.2

5 Appendix. Continued.

Species	March 2010						October 2010					
	1	2	3	4	5	6	1	2	3	4	5	6
<i>Dicentrarchus labrax</i>	5,676.7 ± 8,348.6	23,845 ± 53,765.9	11,966.7 ± 38,274.7	95.9 ± 76.1	117.5 ± 122.6	34.5 ± 35.4	12.7 ± 21.9	34.2 ± 50.3	17.8 ± 19.8	47.9 ± 59.8	1,262.8 ± 1,987.7	343.4 ± 390.2
<i>Sparus aurata</i>	-	641.9 ± 3,189.6	337 ± 2,256.9	7.7 ± 8	9 ± 11.7	-	-	-	-	3.3 ± 8.9	-	-
<i>Abudefduf luridus</i>	354.6 ± 104.8	141.3 ± 88.5	154.4 ± 91.2	95.2 ± 80.8	682.7 ± 388.4	786.5 ± 261.4	277.6 ± 120.3	253.3 ± 132.2	1,225.9 ± 417.4	471 ± 270.8	131.9 ± 56.2	499.7 ± 301.4
<i>Atherina presbyter</i>	3.6 ± 4.3	10.8 ± 52.8	5.7 ± 37.4	-	4.5 ± 4.7	4 ± 6.7	1 ± 1.7	9.1 ± 10.3	6.3 ± 6.2	3.4 ± 5.2	5.3 ± 8.3	0.1 ± 0.2
<i>Aulostomus strigosus</i>	-	3 ± 12.2	3 ± 12.1	-	-	19.7 ± 34.1	4.7 ± 8.2	-	-	2.4 ± 11.1	267 ± 308.3	40.4 ± 54.1
<i>Belone belone</i>	3.9 ± 6.8	0.4 ± 2.1	0.4 ± 2.1	-	-	-	-	-	-	-	-	78.1 ± 135.2
<i>Boops boops</i>	-	-	-	-	-	78.6 ± 94.1	17.2 ± 29.8	-	-	9.6 ± 31.9	987.2 ± 1,581.2	-
<i>Canthigaster capistrata</i>	-	0.1 ± 0.3	0.1 ± 0.3	-	2 ± 3.4	7 ± 11.3	-	-	-	1 ± 3.8	-	1.7 ± 2.3
<i>Diplodus cervinus</i>	-	7.8 ± 38	4 ± 26.9	0.9 ± 1.6	5.9 ± 7.4	-	2 ± 3.5	6.7 ± 11.6	-	1.6 ± 4.5	-	4.1 ± 4.8
<i>Diplodus puntazzo</i>	-	9.7 ± 53	4.8 ± 37.5	-	-	-	7.3 ± 12.6	24.1 ± 41.7	-	3.1 ± 13.8	-	-
<i>Diplodus sargus</i>	382 ± 235.2	1,180.6 ± 2,989.2	868.6 ± 2,158.4	941.9 ± 319.6	424.6 ± 252	428.5 ± 259.4	814.2 ± 603.3	217.7 ± 142.2	780 ± 589.2	682.2 ± 560.7	947.2 ± 720.7	315 ± 181.3
<i>Epinephelus marginatus</i>	-	10 ± 42.1	5 ± 29.8	-	-	-	-	-	-	-	-	-
<i>Gymnothorax unicolor</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Kyphosus saltatrix</i>	48.3 ± 62.7	37.9 ± 98.9	26.6 ± 74.5	-	4.9 ± 7.1	17.2 ± 29.8	-	-	79.7 ± 138.1	15.9 ± 51.3	22 ± 38	4.1 ± 7
<i>Liza aurata</i>	1,169.8 ± 1,800.6	1,957.4 ± 3,378.6	1,399.3 ± 2,641.1	-	3.6 ± 6.2	290.2 ± 441.4	307.8 ± 295.7	657.6 ± 637.8	835.4 ± 796.4	1,000.1 ± 2,036.3	4,509.8 ± 7,202.8	-
<i>Lythognathus mormyrus</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Mugil cephalus</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Mustelus mustelus</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Mycteroperca fusca</i>	-	2.9 ± 15.7	1.4 ± 11.1	-	-	-	-	-	-	2.2 ± 12.2	-	28.7 ± 49.6
<i>Oblada melanura</i>	-	10.4 ± 50.6	5.9 ± 35.8	6.9 ± 8	16 ± 19.1	-	4 ± 4.9	0.1 ± 0.2	-	9.2 ± 33.6	28.4 ± 48.5	40.3 ± 53.5
<i>Ophioblennius atlanticus</i>	394.1 ± 176.6	316.6 ± 213	318.8 ± 207.7	536 ± 223.9	448.2 ± 235.1	726.7 ± 179.1	550.9 ± 284.2	742.4 ± 311.7	472.8 ± 236	440.2 ± 214.8	213.4 ± 93.4	231.4 ± 153.9
<i>Pomadasys incisus</i>	-	32.6 ± 178.8	32.6 ± 178.6	-	-	-	97.9 ± 169.6	-	-	9.8 ± 53.6	-	-
<i>Pomatomus saltatrix</i>	-	-	-	525.2 ± 667.3	-	-	-	-	-	52.5 ± 215.5	168.7 ± 292.3	-
<i>Pseudocaranx dentex</i>	-	3.4 ± 16	2.2 ± 11.6	-	18.2 ± 31.3	-	-	-	5.5 ± 9.4	3.5 ± 11	1.5 ± 2.6	5.2 ± 9
<i>Sardinella aurita</i>	-	-	-	-	-	-	11.4 ± 19.7	-	-	1.1 ± 6.2	-	-
<i>Sardinella maderensis</i>	-	-	-	-	1.3 ± 2.3	-	-	-	-	6.5 ± 34.6	-	2.7 ± 4.6
<i>Sarpa salpa</i>	8,208.3 ± 3,351	10,310.9 ± 4,209.4	10,366.6 ± 4,232.1	7,352.2 ± 3,001.5	17,274.8 ± 7,052.4	11,664.7 ± 4,762.1	2,554.5 ± 1,042.9	5,366.9 ± 2,191	11,546.2 ± 4,713.7	11,654.1 ± 4,757.8	6,410.5 ± 2,617.1	4,113.9 ± 1,679.5
<i>Scorpaena maderensis</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Seriola spp.</i>	-	-	-	-	-	-	-	-	-	-	-	-
<i>Sparisoma cretense</i>	158 ± 165.5	595 ± 846.1	560.1 ± 803.7	153.5 ± 126.6	123.1 ± 131.2	978.1 ± 440.1	198.3 ± 93	248.5 ± 315.4	499.9 ± 289.7	600.3 ± 610.3	142.1 ± 116.9	204.6 ± 165.5
<i>Sphoeroides marmoratus</i>	-	-	-	-	0.6 ± 1.1	-	-	-	-	0.5 ± 2.3	-	-
<i>Sphyaena viridensis</i>	18.6 ± 22.1	6.7 ± 26.6	4.4 ± 19.5	66.6 ± 75.1	-	142 ± 245.9	185.8 ± 321.8	-	8 ± 10.6	41 ± 129.8	-	-
<i>Symphodus trutta</i>	-	0.1 ± 0.7	0.1 ± 0.4	-	-	-	-	-	-	0.8 ± 3.9	-	-
<i>Thalassoma pavo</i>	243.3 ± 105.1	259.7 ± 199.9	237.1 ± 159.4	182.3 ± 74.1	298.3 ± 118.1	326.4 ± 112.7	339.1 ± 155.6	117.9 ± 49.5	249 ± 122	246.6 ± 106.9	132.9 ± 62.6	168.7 ± 77.5
<i>Trachinotus ovatus</i>	10.8 ± 10.9	20.8 ± 48.9	15.7 ± 42.3	-	32.2 ± 51	-	-	-	-	13.5 ± 37.4	0.5 ± 0.9	2.5 ± 2.1
Unidentified larvae	0.9 ± 1.5	0.3 ± 0.7	0.4 ± 0.9	-	-	-	-	-	-	0 ± 0.1	-	-

Table 1. Univariate PERMANOVA, comparing mean trophic level of fish assemblages with presence of escaped fish versus those with no escaped fish. In bold those p-values < 0.05.

Source	df	MS	Pseudo-F	P(perm)
Presence vs. Absence	1	8.8648	39.33	0.0002
Res	423	0.2254		
Total	431			

Table 2. Univariate PERMANOVA, exploring spatiotemporal patterns of mTrL. In bold those p-values<0.05 for fixed factors and their relevant interactions. Ye: Year, Pe: Period, Pr: Protection, Si: Site.

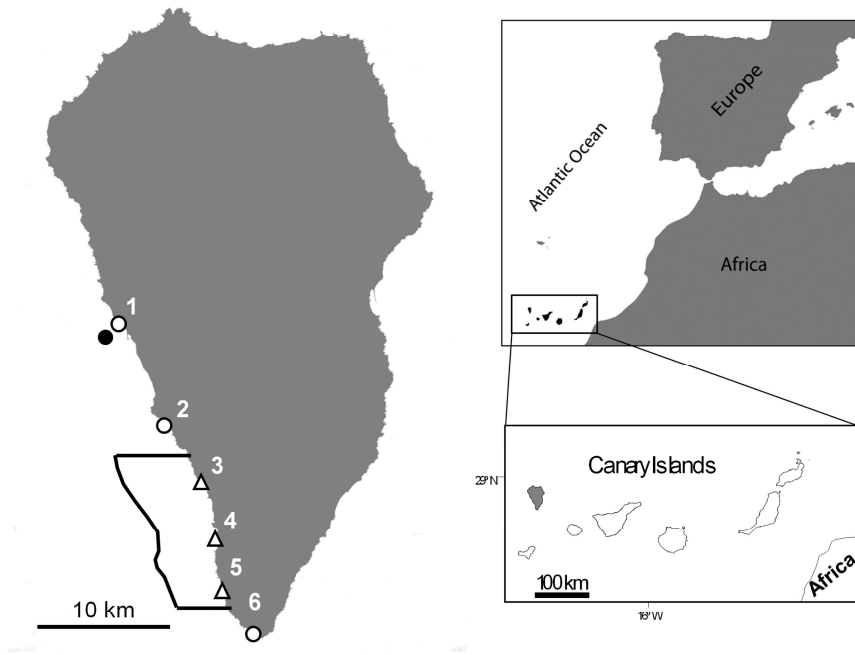
Source	df	MS	Pseudo-F	P(perm)
Year	1	6.5392	14.131	0.002
Period	1	3.6593	16.625	<0.001
Protection	1	5.1611	13.981	0.002
Site(Pr)	16	0.36111	2.0037	0.014
YexPe	1	0.81318	2.127	0.162
YexPr	1	1.1631	2.5015	0.132
PexPr	1	0.49561	2.2613	0.146
YexSi(Pr)	16	0.46161	2.5613	0.002
PexSi(Pr)	16	0.21772	1.2081	0.267
YexPexPr	1	6.1848E-2	0.16501	0.686
YexPexSi(Pr)	16	0.37805	2.0977	0.008
Res	353	0.18022		
Total	431			

Table 3. Diet composition of the “leak” and “massive” escaped *Dicentrarchus labrax*.

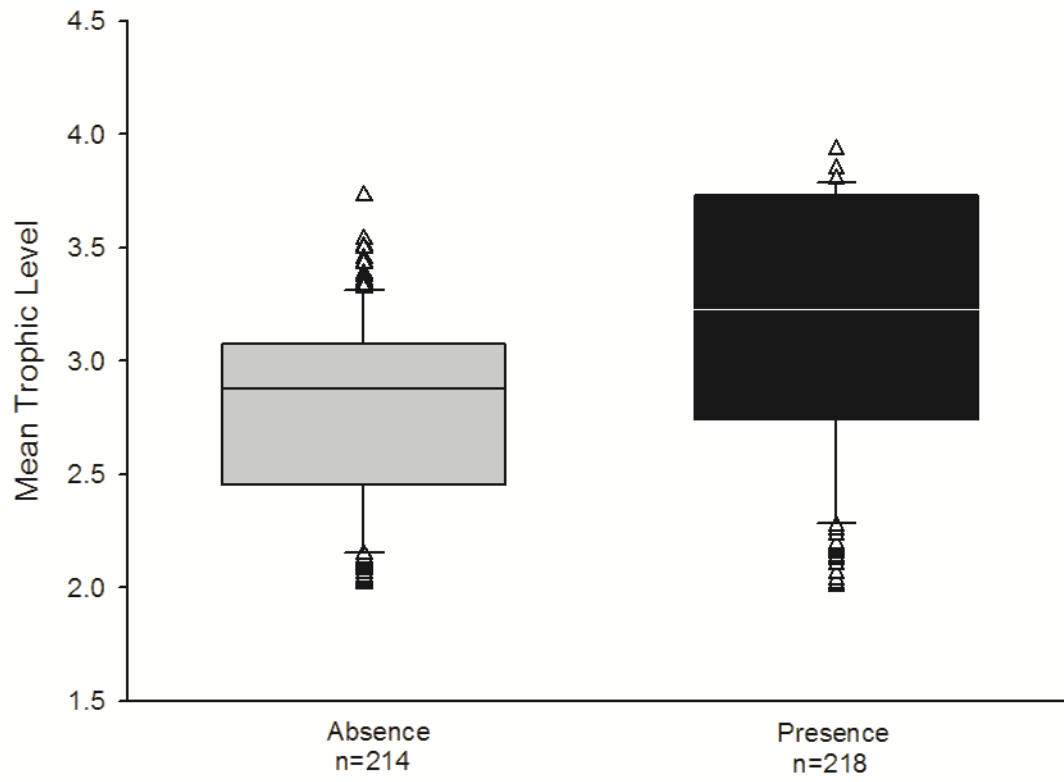
Leak group n=112		Vacuity=33.9% Trophic Level=4.2						a
Prey	N%	W%	O%	Q	IRI	Prey preferences		
Fish	62.9	76.4	54.1	4803.2	7528	Main preferred prey		
<i>Atherina presbyter</i>	2.4	0.4	4.1	1.0	11			
<i>Sparisoma cretense</i>	1.0	7.0	2.7	6.6	21			
<i>Thalassoma pavo</i>	1.4	10.3	2.7	14.7	32			
<i>Scorpaena sp.</i>	0.5	4.7	1.4	2.2	7			
<i>Trachinus draco</i>	0.5	5.7	1.4	2.7	8			
<i>Sardinella aurita</i>	33.3	23.9	5.4	795.7	309			
Unidentified fish	23.8	24.5	36.5	582.6	1762			
Crustaceans	14.3	5.1	32.4	73.4	630	Secondary common prey		
<i>Percnon gibbesi</i>	1.0	2.9	2.7	2.7	10			
<i>Plagusia depressa</i>	0.5	<0.01	1.4	<0.01	1			
<i>Pachygrapsus sp.</i>	2.9	0.2	2.7	0.6	8			
<i>Xantho sp.</i>	0.5	0.1	1.4	<0.01	1			
Unidentified crustaceans	9.5	1.9	27.0	18.5	310			
Mollusks	2.4	0.8	4.1	1.9	13	Accidental prey		
Cephalopoda	1.9	0.8	2.7	1.5	7			
Gastropoda	0.5	<0.01	1.4	<0.01	1			
Insects	10.5	0.7	28.4	7.4	317	Accidental prey		
Blattaria	0.5	0.3	1.4	0.2	1			
Hymenoptera	1.9	<0.01	5.4	<0.01	10			
Coleoptera	0.5	<0.01	1.4	<0.01	1			
Unidentified insects	7.6	0.4	21.6	2.7	173			
Plants								
<i>Cymodocea nodosa</i>	1.9	1.3	5.4	2.4	17	Accidental prey		
Mammals								
<i>Rattus norvegicus</i>	0.5	1.4	1.4	0.6	2	Accidental prey		
Pellets	1.0	10.3	2.7	9.8	30	Accidental prey		
Other Items	6.7	4.1	17.6	27.1	189	Secondary common prey		
Massive group n=32		Vacuity=12.5% Trophic Level=3.2						b
Prey	N%	W%	O%	Q	IRI	Prey preferences		
Fish	2.7	0.1	3.6	0.4	10	Accidental prey		
Unidentified fish	2.7	0.1	3.6	0.4	10			
Crustaceans	89.2	80.1	100.0	7142.7	16927	Main preferred prey		
<i>Percnon gibbesi</i>	45.9	57.3	42.9	2634.0	4426			
<i>Eriphia verrucosa</i>	2.7	10.0	3.6	27.0	45			
Unidentified crustaceans	40.5	12.8	53.6	518.1	2856			
Other Items	5.4	1.2	7.1	6.5	47	Accidental prey		

Table 4. PERMANOVA results comparing the diet of sea bass escaped in leak and massive events, including total length as a covariable. In bold those p-values<0.05.

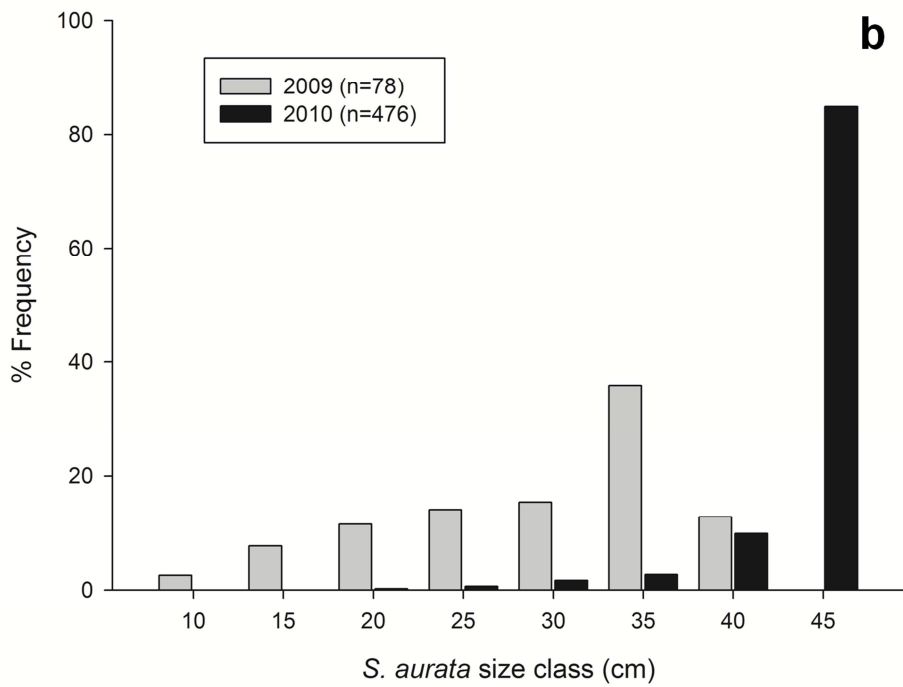
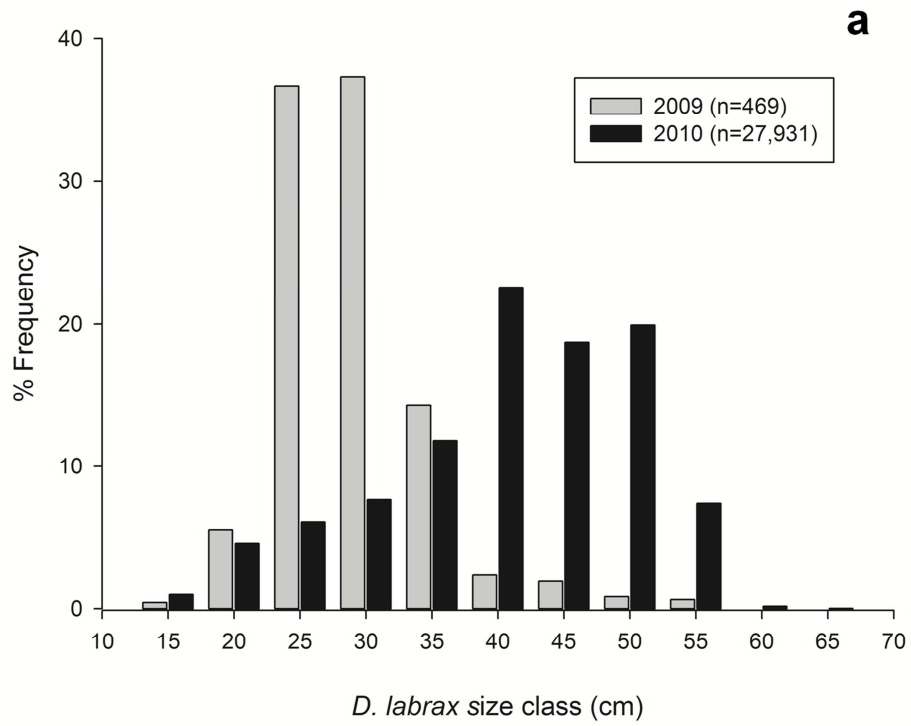
Source	df	MS	Pseudo-F	P(perm)
Total Length	1	16215	3.7611	0.0002
Leak vs. Massive	1	19094	4.429	0.0002
Total Length x Leak vs. Massive	1	6301.8	1.4618	0.0734
Res	94	4311.1		
Total	97			

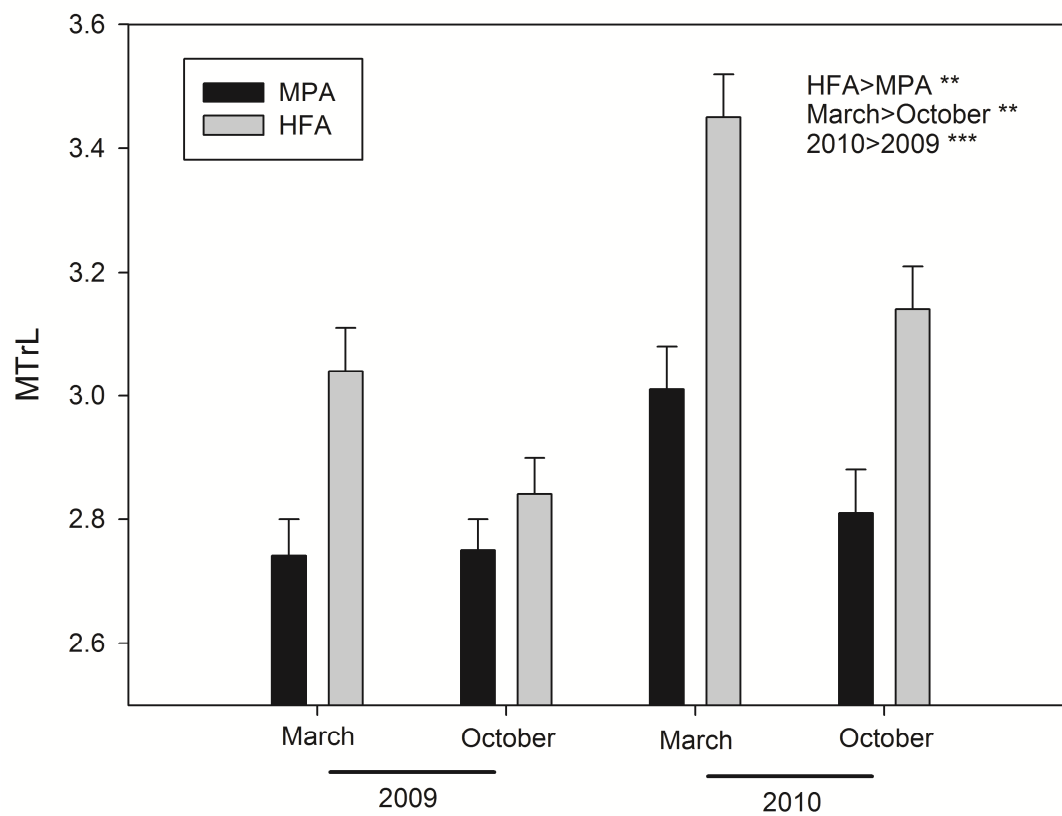


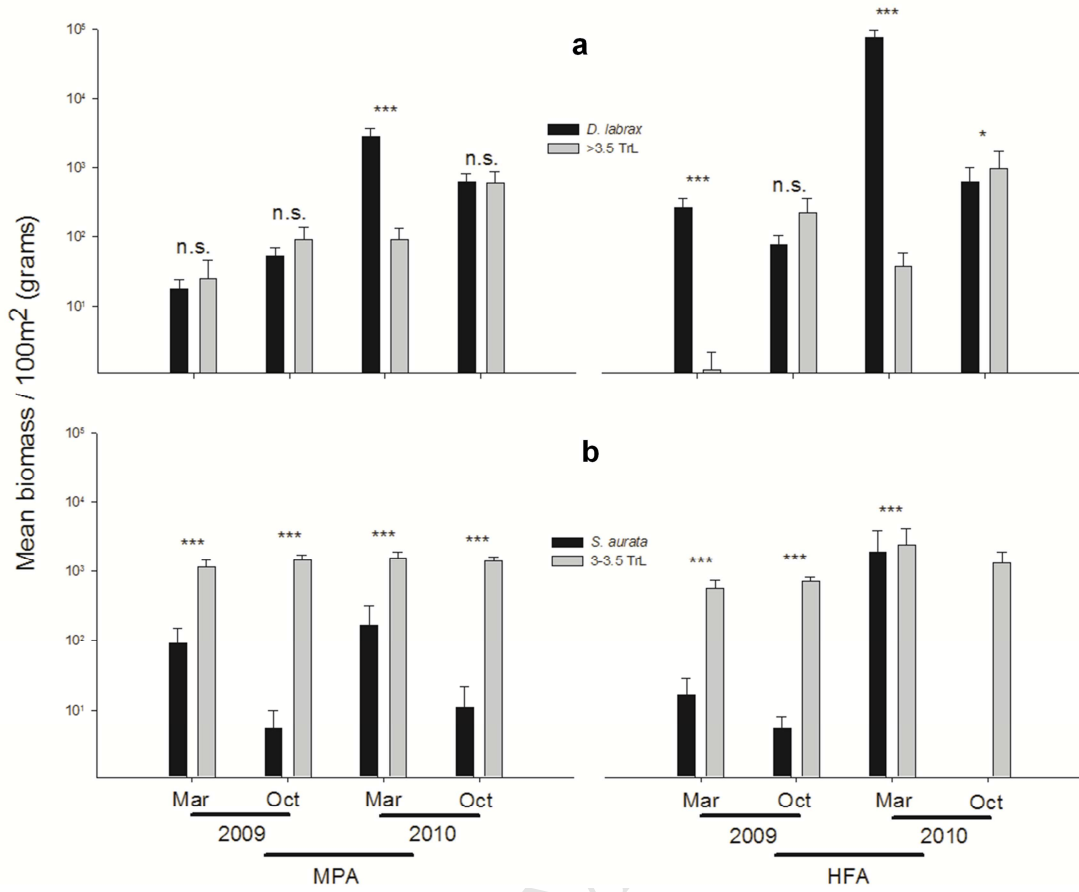
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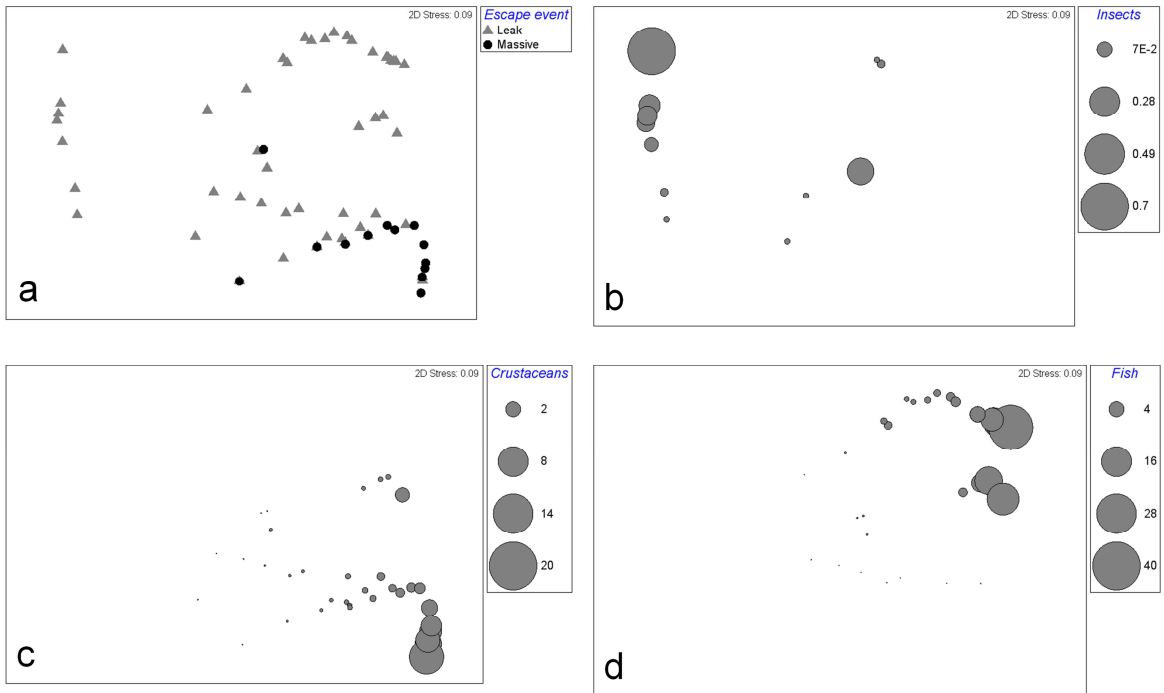


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Highlights:

- An aquaculture massive escape released 1.5 million fish into the wild in the Canaries.
- Mean trophic level of marine fish assemblages was investigated through visual census.
- Diet of escaped sea bass was studied regarding size and time in the wild.
- Mean trophic level was raised but a nearby marine protected area showed resilience.
- Escaped sea bass exploited natural resources according to size and time at liberty.