2006

Changes in demersal wild fish aggregations beneath a sea-cage fish farm after the cessation of farming

Fernando Tuya
Edith Cowan University

Pablo Sanchez-Jerez

Tim Dempster

Arturo Boyra

R Haroun

1 **Changes in demersal wild fish aggregations beneath a sea-cage fish**

2 **farm after the cessation of farming**

3

4 F. TUYA *[¥], P. SANCHEZ-JEREZ [§], T. DEMPSTER [¶], A. BOYRA * AND R. J.

5 HAROUN *

6

7

8 *Department of Biology, Faculty of Marine Sciences, University of Las Palmas de Gran

9 Canaria, 35017, Las Palmas, Spain, [§] Unidad de Biología Marina, Departamento de

10 Ciencias del Mar y Biología Aplicada, Universidad de Alicante, Ap.C. 99, 03080,

11 Alicante, Spain, [¶] SINTEF Fisheries and Aquaculture, NO-7465, Trondheim, Norway

12

13 Running headline: Changes in wild fish aggregations

14

15

16

17

18

19

20

21

22

23 [¥]Author to whom correspondence should be addressed. Tel.: + 34 928 457456; fax: + 34

24 928 457457; e-mail: ftuya@yahoo.es

ABSTRACT

Demersal, non-cryptic, wild fish were counted in replicate 100 m^2 transects beneath a floating sea-cage fish farm and two nearby sandy locations at Gran Canaria (Canary Islands, eastern Atlantic) four times before and after the cessation of farming. Cessation of farming involved the removal of farmed fish and ceasing of the daily feeding, although farm structures (cages and moorings) remained. A "beyond-BACI" sampling design provided the framework to detect the effect of the cessation of farming, which produced qualitative and quantitative changes in the composition and structure of the fish assemblages beneath the sea-cage fish farm compared with two nearby controls. The aggregative effect on wild fish due to the existence of the farm decreased from approximately 50 times compared to nearby controls when the farm was in full operation to <2 times when only the farm structures remained. Abundances of POM feeders (large-sized mugilids), large benthic chondrichthyid rays and *Pagellus* spp. declined markedly at the fish farm after the cessation of farming, suggesting that the removal of daily feeding was responsible for their disappearance. In contrast, abundances of herbivores, benthic macro- and meso-carnivores were similar beneath the fish farm both before and after the cessation of farming. Benthic macro-carnivores, however, were more abundant beneath the sea-cages compared to control locations, supporting the hypothesis that the increase in the physical structure beneath farms plays a role in aggregating these species. Sparids occurred beneath the sea-cages only after the cessation of farming, while the two natural control locations did not show differences from before to after the cessation of farming. Overall, the results show that the wild fish assemblage beneath the farm partially changed after the cessation of farming to a more natural state, approaching the assemblages observed at the control sites.

50   Key words: wild fish; aquaculture; impact; recovery; fish farm; Canary Islands.

51

52

53   **INTRODUCTION**

54   Since initial development of sea-cage aquaculture in the early 1980s, the number of sea-

55   cage fish farms has increased rapidly throughout coastal areas of the world (Ferlin &

56   LaCroix, 2000) to produce almost 2.5 million tons of fish each year (FAO, 2003). The

57   environmental impacts of operating sea-cage fish farms are well described and include

58   impacts on biogeochemical processes, seagrasses and benthic communities, and transfer

59   of antibiotics to the environment (Iwama, 1991; Rönenberg *et al*., 1992; Ruiz *et al*.,

60   2001). Numerous studies have also described the recovery of benthic habitats after the

61   cessation of farming (e.g. Karakassis *et al*., 1999; Brooks *et al*., 2004); however, in

62   comparison, no study has investigated the effects of the cessation of farming on

63   aggregations of wild fish closely associated with farms. While operating, fish farms

64   dramatically increase the presence, abundance and biomass of wild demersal and

65   pelagic fish in their immediate vicinity compared to control nearby areas (Carss, 1990;

66   Dempster *et al*., 2002; Boyra *et al*., 2004; Dempster *et al*., 2004; Dempster *et al*., 2005;

67   Tuya *et al*., 2005).

68   The persistent artificial input of food and possible chemical attraction due to the

69   presence of typically 100s of tons of farmed fish may influence which species of wild

70   fish associate with farms (Dempster *et al*., 2002). These factors, in addition to the

71   enhanced habitat complexity or 'artificial reef' effect, have been considered as

72   responsible for the increase in demersal wild fish abundances relative to nearby sandy

73   bottoms (Boyra *et al*., 2004; Tuya *et al*., 2005) with an overall effect size of 1 to 2

74   orders of magnitude. However, no study has empirically assessed the relative

importance of each of these factors involved in the attraction of different fish species around coastal aquaculture installations. In this context, we took advantage of the cessation of farming at a sea-cage fish farm at Gran Canaria (Canary Islands, eastern Atlantic Ocean) where the entire farm infrastructure was left intact. This allowed us to separate the attractive effect caused by the persistent artificial food input and the chemical attraction due to the presence of farmed fish, from the attractive effect caused by the increased structural complexity of the habitat, which has been shown to play a major role in structuring fish assemblages on shallow soft bottoms at temperate latitudes (Guidetti, 2000; Guidetti & Bussotti, 2002; Tuya *et al*., 2005). Boyra *et al*. (2004) demonstrated that this fish farm caused local aggregation of wild coastal fish through a "post-impact" sampling design (*sensu* Glasby, 1997), with differences in the abundances of some species compared to nearby sandy locations (controls).

We aimed to detect changes in the composition and abundance of sub-adult and adult demersal wild fish assemblages associated with a sea-cage fish farm 'before' and 'after' the cessation of farming, by establishing the temporal and spatial persistence of the differences between the 'impacted' location (the fish farm) and two nearby sandy locations as controls, following the appropriate criteria of a "beyond-BACI" design (*sensu* Underwood, 1992, 1993, 1994). More specifically, we hypothesized that (1) fish assemblages beneath the sea-cage fish farm would differ significantly before and after farming ended compared with control locations; whereas (2) assemblages at control locations before and after farming ended would be relatively consistent through time.

MATERIALS AND METHODS

AREA OF STUDY AND EXPERIMENTAL DESIGN

99   The study was conducted around the 'Arguineguín' farm located 300 m off the

100   southern coast of Gran Canaria (Canary Islands, 28ºN, eastern Atlantic, Fig. 1) in 9 to

101   11 m of water (Boyra *et al*., 2004). The farm cultured both gilthead sea bream (*Sparus*

102   *aurata* Linnaeus, 1758) and European seabass (*Dicentrarchus labrax* Linnaeus, 1758)

103   and operated from June 1993 to June 2003, when the aquaculture company went

104   bankrupt. The farm complex comprised 12 cages and produced an average of more than

105   180 t yr^{-1}. The entire farm infrastructure, including the majority of sea-cages and all

106   mooring devices, was left untouched after farming ceased. The cages were situated

107   above a sandy bottom covered by sparse patches of the seagrass *Cymodocea nodosa*

108   Ucria (Ascherson) and the green algae *Caulerpa* spp. The cessation of farming in June

109   2003 was considered as the 'impact', which consisted of the removal of all farmed fish

110   and termination of the daily input of food pellets. As a result, the term 'impact' used in

111   this paper differs to the classical 'disturbance' concept, as it describes a reduction of a

112   human-induced perturbation of the marine environment.

113   Selection of impact and control locations and sites within locations followed an

114   asymmetrical "beyond BACI" sampling design. The mechanisms and the logical

115   structure of these analyses, as well as the potential to detect both temporal and spatial

116   disturbances, have been widely discussed and demonstrated (Underwood, 1991, 1992,

117   1993, 1994; Kingsford & Battershill, 1998). Lack of nearby sea-cage fish farms

118   prevented the use of a more 'formal' set of controls, where farming should have been

119   monitored through time. We therefore were able to solely select nearby soft bottoms as

120   control locations; two controls (nearby sandy locations located between 600-1000 m

121   away from the sea-cage fish farm) and one impact location (the sea-cage farm) were

122   thus established. To increase the spatial replication, we randomly sampled two sites

123   within each of these locations approximately 40 to 80 m apart. Temporal replication

124 was included before and after the impact by randomly selecting 4 sampling times 2

125 years before (from May 2000 until June 2001), and 4 times 2 years after (from April

126 2005 to May 2005) the impact. As a result, our asymmetrical design involved sampling

127 two control and one perturbed location before and after the cessation of farming at

128 several nested temporal and spatial scales.

129

130 ESTIMATION OF FISH ASSEMBLAGES AND ECOLOGICAL FISH

131 CATEGORIES

132 Sub-adult and adult demersal, non-cryptic, fish populations (> 5 cm total length) were

133 sampled by visual counts (Boyra *et al*., 2004; Tuya *et al*., 2005). At each sampling time,

134 four replicate 100 m^2 transects were randomly censused during daylight hours at each

135 site within each location. Consequently, a total of 192 transects were made throughout

136 the study. The abundance of fish species was recorded by a SCUBA diver using a

137 modification of the method of Harmelin-Vivien *et al*. (1985). Therefore, when fishes

138 were grouped in schools larger than 20 individuals, their numbers were estimated

139 according to six abundance classes (20-40, 40-70, 70-150, 150-300, 300-700, >700).

140 Groups of fishes with similar ecological requirements were aggregated into six

141 categories (*sensu* Guidetti *et al*., 2003), defined on the basis of their feeding behaviour

142 and spatial organization around the farm: (1) POM feeders: particulate organic matter

143 feeders (large-sized fish within the family Mugilidae); (2) SPA: meso- and

144 macrocarnivorous sparids (the genera *Diplodus* and *Pagrus*, *Lithognathus mormyrus*,

145 *Spondyliosoma cantharus* Linnaeus, 1758, and *Oblada melanura* Linnaeus, 1758) – this

146 group did not include *Sparus aurata*; (3) HERB: herbivorous fish (*Sarpa salpa*

147 Linnaeus, 1758, and *Sparisoma cretense* Linnaeus, 1758); (4) BENMESO: bentho-

148 demersal meso-carnivores usually found as solitary individuals (*Bothus podas*

149 Delaroche, 1809, *Canthigaster rostrata* Bloch, 1786, *Chelidonichthys lastoviza*

150 Bonnaterre, 1788, *Mullus surmuletus* Linnaeus, 1758, *Sphoeroides marmoratus* Lowe,

151 1838, *Xyrichthys novacula* Linnaeus, 1758); (5) BENMACRO: bentho-demersal macro-

152 carnivores usually found as solitary individuals (the genera *Synodus* and *Trachinus*); (6)

153 RAYS: large-sized benthic chondrichthyes (*Dasyatis* spp., *Gymnura altavela* Linnaeus,

154 1758, *Myliobatis aquila* Linnaeus, 1758, *Squatina squatina* Linnaeus, 1758, *Taenuria*

155 *grabata* Linnaeus, 1758). In addition, *Pagellus* spp. was analysed individually, as it was

156 the most important taxa contributing to differences between controls and the sea-cage

157 fish farm in the previous "post-impact" study (Boyra *et al*., 2004). We also analysed

158 individually the gilthead sea bream *Sparus aurata*, since observed specimens probably

159 escaped from the cages. Occasional species (observed in less that 3% of the counts) and

160 pelagic schooling species with high spatial and temporal patchiness (e.g. *Sardina* spp.,

161 *Boop boops* Linnaeus, 1758) were not included in the analysis.

162

163 MULTIVARIATE STATISTICAL ANALYSIS

164 To visualize the extent of differences among assemblages before and after the impact

165 and among the farm and control locations, non-metric multidimensional scaling (MDS)

166 was selected as an ordination technique to generate a two-dimensional plot using the

167 PRIMER statistical package (Clarke, 1993). Prior to calculating the similarity matrix,

168 the data were pooled by summing the 4 counts at each site within each location and

169 sampling time. Data were then fourth root transformed to weight the contributions of

170 common and rare species in the similarity coefficient, and a triangular similarity matrix

171 was calculated using the Bray-Curtis similarity coefficient (Clarke, 1993; Clarke &

172 Warwick, 1994). Average dissimilarities between the fish farm and the two controls

173 were calculated by means of the SIMPER procedure (Clarke, 1993) before and after the

174 cessation of farming. A two-way crossed ANOSIM (Clarke, 1993) was used to examine

175 the significance of the differences on the overall fish dataset among the three locations

176 before and after the cessation of farming.

177

178 UNIVARIATE STATISTICAL ANALYSIS

179   Following the criteria appropriate for "beyond-BACI" designs, four–way

180 asymmetrical ANOVA models were used to compare mean abundances of each fish

181 category between the two periods ("before" and "after"), among times within each

182 period, among the three locations (including "impact vs. controls" and "between

183 controls") and between sites within locations (Underwood, 1993, 1994, 1997). First, we

184 analysed all data as though there were no asymmetries. Second, we re-analysed the data

185 while omitting the perturbed location. The asymmetrical components were further

186 calculated by subtractions and additions of components. The "impact vs. controls" term

187 was considered fixed, while the term "between controls" was random. Sites were

188 randomly nested within locations. The two analyses were carried out as a fully

189 orthogonal design of "before vs. after", "locations" and its nested "sites", and "times"

190 within "before" or "after". The interpretation of statistical interactions following

191 Underwood's indications (Underwood, 1993, 1994, 1997) was used to assess the effect

192 of the considered experimental factors. Prior to ANOVA, Cochran's test was used to

193 test for heterogeneity of variances. In all cases, data was transformed to ln (x + 1) to

194 achieve homogeneity of variances.

195

196   **RESULTS**

197   A total of 15204 fishes comprising 15 families and 23 fish taxa were recorded for the

198 studied sea-cage fish farm and the two adjacent natural control locations throughout the

199   study (Table I). Demersal fish were 45 – 52 times more abundant beneath the farm

200   compared with the two nearby control locations during the period when the farm was in

201   full operation. In contrast, abundances of fish were only 1.6 – 1.8 times higher beneath

202   the farm than the controls after farming ceased. *Pagellus* spp. (n = 11308 fish) and POM

203   feeders within the Mugilidae family (n = 1164) were the most abundant taxa at the sea-

204   cage fish farm before the cessation of farming, while *Synodus* spp. (n = 82) and

205   *Trachinus draco* Linnaeus, 1758 (n = 46) within the macro-carnivores were the most

206   abundant species after the cessation of farming. The sparids *Spondyliosoma cantharus*

207   (n = 343) and *Diplodus annularis* Linnaeus, 1758 (n = 336) dominated in terms of

208   abundance in the two controls before and after the cessation of farming. Table I lists the

209   overall abundance of each fish taxa recorded at the sea-cage fish farm and the two

210   controls before and after the cessation of farming.

211

212   MULTIVARIATE RESULTS

213   The two-dimensional MDS (Fig. 2) revealed a clear-cut separation of the sites

214   beneath the fish farm throughout the four sampling times before the cessation of

215   farming (left-side of the plot) from the sites within the two control locations throughout

216   the four sampling times before and after the cessation of farming (right-side of the plot).

217   ANOSIM permutational tests detected that these differences were significant ($P < 0.01$,

218   Table II). Sites beneath the fish farm throughout the four sampling times after the

219   cessation of farming lie in the middle of the two above-mentioned groups, indicating a

220   'transition' in the structure of the fish assemblage towards control locations. However,

221   differences in the structure of the fish assemblages between the fish farm and the two

222   controls were persistent after the cessation of farming, as indicated by pairwise

223   comparisons using the ANOSIM permutational test ($P < 0.01$, Table II). This change in

224   the structure of the demersal fish assemblage beneath the sea-cages of the fish farm can

225   be appreciated when comparing the average dissimilarities between the fish farm and

226   the two controls before and after the cessation of farming (Table II). For example,

227   average dissimilarity between the fish farm and control 1 was reduced from 98.8 to

228   86.1%. It is worth noting, moreover, how sites within the two control locations are

229   randomly spread throughout the right-side of the ordination space (Fig. 2), highlighting

230   the broad natural variation in the structure of fish assemblages at the control locations

231   before and after the considered 'impact'.

232

233   UNIVARIATE RESULTS

234   While the sea-cage fish farm was operating, abundances of the bentho-demersal

235   meso and macro-carnivorous fish categories and *Pagellus* spp. varied between the two

236   control locations [Figs. 3(d), 3(e) 3(g), Table III: "T(Bef) X C" was significant]. In the

237   same period, significant differences were observed between the sea-cage fish farm and

238   the average of the two controls for the mean abundances of the POM feeders, RAYS,

239   *Pagellus* spp. and *Sparus aurata* [Figs. 3(a), 3(f), 3(g), 3(h), Table III: "T(Bef) X I" was

240   significant]. Therefore, all of these taxa showed short-term temporal variability before

241   the cessation of farming.

242   POM feeders, RAYS, and the gilt-head sea bream, *Sparus aurata*, were more

243   abundant at the sea-cage fish farm before the cessation of farming compared to the two

244   controls [Figs. 3(a), 3(f), 3(h)], and almost completely disappeared after the cessation of

245   farming compared to controls [Figs. 3(a), 3(f), 3(h), Table III: "T(Bef) X I vs. T(Aft) X

246   I" were significant, whereas "T(Bef) X C vs. T(Aft) X C" were non-significant]. Mean

247   abundances of *Pagellus* spp. similarly decreased significantly after the cessation of

248   farming at the sea-cage fish farm compared to controls [Fig. 3(g), Table III: "T(Bef) X I

249   vs. T(Aft) X I" and "T(Bef) X I vs. T(Bef) X C" were significant, whereas "T(Bef) X C

250   vs. T(Aft) X C" was non-significant"], although this species showed short-term

251   temporal variability between controls before the cessation of farming (Table III, factor

252   "T(Bef) X C" was significant). As a result, we detected significant decreases in the

253   mean abundances of POM feeders, RAYS, *Sparus aurata*, and *Pagellus* spp. at the sea-

254   cage fish farm after the cessation of farming, while no differences existed between

255   before and after the cessation of farming at control locations for these taxa.

256   Short-term temporal variability for the sparids before the end of farming was

257   negligible (Table III: "T(Bef) X C", "T(Bef) X I" were non-significant). Abundances of

258   this group increased significantly after the cessation of farming at the sea-cage fish

259   farm, with no before/after interactions between controls [Fig. 3(b), Table III: "B X I"

260   was significant, while "B X C" was non-significant]. In fact, all sparids were observed

261   at the sea-cage fish farm after the finalization of farming [Fig. 3(b), Table I], while

262   control locations did not differ from before to after the cessation of farming (Table III:

263   "T(Bef) X C vs. T(Aft) X C" was non-significant).

264   Temporal trends in the mean abundances of the herbivorous fish was similar among

265   the three locations before the cessation of farming [Fig. 3(c), Table III: "T(Bef) X C"

266   and "T(Bef) X I were non-significant], despite the observation of a large group of *Sarpa*

267   *salpa* at the fourth sampling time [Fig. 3(c)]. Moreover, temporal trends in mean

268   abundances of the herbivores did not differ significantly between the two controls, as

269   well as between their average trends and that at the sea-cage fish farms from before to

270   after the cessation of farming (Table III: "B X C" and "B X I" were non-significant).

271   Consequently, no change was detected throughout the study for the mean abundances of

272   the herbivorous group.

273   Control locations showed different trends in the mean abundances of the bentho-
274   demersal meso and macro-carnivores from before to after the cessation of farming
275   [Figs. 3(d), 3(e), Table III: "T(Bef) X C vs. T(Aft) X C" was significant]. For the
276   bentho-demersal meso-carnivorous group, temporal variability, at both short and long-
277   scales, was largely attributable to the observation of large groups of *Mullus surmuletus*
278   in the first sampling time before the cessation of farming [Fig. 3(d)]. However, for both
279   the bentho-demersal meso and macro-carnivores, we observed similar abundances at the
280   sea-cage fish farm before and after the cessation of farming [Figs. 3(d), 3(e)], resulting
281   in the lack of significance of all the potentially affected terms in the ANOVA (Table III:
282   "T(Bef) X I", "T(Bef) X I vs. T(Aft) X I", "T(Bef) X I vs. T(Bef) X C", "B X I vs. B X
283   C" were non-significant"). We therefore did not register a significant change in the
284   mean abundances of the meso and macro-carnivorous categories at the sea-cage fish
285   farm throughout the study. However, mean abundances of the bentho-demersal macro-
286   carnivores were larger at the sea-cage fish farm compared to the two controls over the
287   entire study period [Fig. 3(e)].

288

289   **DISCUSSION**

290   The cessation of farming produced qualitative and quantitative changes in the
291   composition and structure of the demersal, non-cryptic, wild fish assemblages beneath
292   the sea-cage fish farm at Arguineguín (Gran Canaria Island) compared with nearby
293   control locations under the influence of natural variability. The significant differences
294   caused by the cessation of farming were clearly species or group-specific. The different
295   taxa and ecological categories of wild fish considered by our study responded
296   differentially to the finalization of farming. Abundances of POM feeders, RAYS,
297   *Pagellus* spp. and *Sparus aurata* declined significantly, whereas fish within the HERB,

298   The bentho-demersal meso and macro-carnivores were not significantly affected by the

299   cessation of farming.

300   In terms of overall abundance of wild fish, the aggregative effect due to the existence of

301   the farm decreased from approximately 50 times compared to nearby controls when the

302   farm was in full operation to <2 times when only the farm structures remained.

303   Therefore, the 'daily feeding and presence of caged fish' effect was far stronger than the

304   added structure due to cages and moorings or 'artificial reef' effect in aggregating wild

305   demersal fish at the farm. This result implies that aggregations of demersal wild fish

306   may decrease markedly if levels of food loss and POM output to the environment from

307   operating farms are reduced.

308

309   WILD FISH AFFECTED BY THE CESSATION OF FARMING AT THE FISH

310   FARM

311   The attraction and increase in the abundances of demersal wild fish populations to

312   soft bottoms beneath sea-cage fish farms in temperate latitudes seems to be mediated by

313   the persistent artificial food input and the chemical attraction due to the presence of

314   farmed fish, in addition to the increase in habitat complexity (Dempster *et al*., 2002;

315   Boyra *et al*., 2004; Tuya *et al*., 2005). Apparently, the great attraction of POM feeders,

316   large chondrichthyid rays, and *Pagellus* spp. is due to either the input of food

317   originating from the farm into the system or the presence of tons of farmed fish.

318   Food beneath fish farms is available to wild fish in the vicinity as large food pellets

319   lost through the cage, dead cultured fish, and as a 'soup' of POM of broken pellets and

320   faeces from caged fish (Dempster *et al*., 2002, 2005). We observed large mugilids

321   feeding directly upon this soup of POM at the farm while it was working. The same

322   pattern seemed to occur for *Pagellus* spp. This genus feeds on a wide variety of food

323   items in the Canarian Archipelago, including worms, molluscs, small crustaceans, small

324   fish, and algae (Fanlo *et al*., 1996). The disappearance of both taxa after the cessation of

325   farming could be indicative of attraction to the fish farm to feed on food pellets not

326   consumed by caged fish and lost from the cages towards the bottom. Nonetheless, an

327   important drawback of this approach is the lack of direct quantification of the

328   consumption of the soup of POM. Further research should focus on this point, for

329   example, using analyses of gut contents. In the Mediterranean Sea, several POM-

330   feeding fish species (*Trachurus mediterraneus* Necrasov, 1966, *Trachinotus ovatus*

331   Linnaeus, 1758, *Sardinella aurita* Valenciennes, 1847, *Boops boops*, large mugilids)

332   consume food available around farms (Dempster *et al*., 2002), and when they occur in

333   high abundances they may greatly influence the dynamics of nutrient flows (Vita *et al*.,

334   2004; Dempster *et al*., 2005).

335   Large-sized rays were also abundant beneath the sea-cage fish farm while it

336   operated; high concentrations of rays beneath farms are typical in the Canarian

337   Archipelago (Boyra *et al*., 2004; Dempster *et al*., 2005; Tuya *et al*., 2005). We do not

338   know if these rays fed upon lost feed, as well as on benthic infauna as they typically do

339   (Gray *et al*., 1997; Ebert & Cowley, 2003), when they foraged on the bottom. However,

340   we saw several specimens feeding on dead cultured fish during the study. Their

341   disappearance beneath the farm after fish culturing activities ceased may have been in

342   response to the loss of this source of large food items.

343   Small escapes of caged fish (10s to 100s of fish) are due to the periodic loss of fish

344   during harvesting, while mass escapes (1000s to 100000s of fish) are caused by

345   operational accidents that damage nets or sporadic storms. We observed 10s of gilthead

346   sea bream (*Sparus aurata*) in some counts at the fish farm before the finalization of

347   farming, indicating some degree of fidelity of escapees to the fish farm. The lack of

348 observations of individuals after the cessation of farming either resulted from heavy

349 fishing pressure in the study area or migrations to nearby rocky reefs, which is the

350 preferred habitat of wild *Sparus aurata* in the Canarian Islands (Brito *et al*., 2002).

351   Finally, sparids occurred beneath the sea-cages only after the cessation of farming,

352 while they appeared in similar abundances at the two natural control locations both

353 before and after the cessation of farming. This pattern may be related to the recovery of

354 a seagrass meadow of the marine phanerogam *Cymodocea nodosa* beneath the sea-cage

355 fish farm after the end of farming (F. Tuya, pers. obs.), which is typical natural habitat

356 for sparids such as *Diplodus annularis* and *Spondyliosoma cantharus* in the Canarian

357 Archipelago (Tuya *et al*., 2005).

358

359 WILD FISH NOT AFFECTED BY THE CESSATION OF FARMING AT THE FISH

360 FARM

361   Both before and after farming ceased, the bentho-demersal macro-carnivores

362 showed greater mean abundances beneath the sea-cages compared to nearby control

363 locations. To a lesser degree, a similar pattern was recorded for the bentho-demersal

364 meso-carnivores. This fact supports the hypothesis that the increase in the physical

365 structure of the system *per se* plays a partial role in structuring certain nearshore fish

366 assemblages associated with shallow soft bottoms in warm-temperate environments

367 (Jenkins & Wheatly, 1998; Guidetti, 2000; Guidetti & Bussotti, 2002; Pihl &

368 Wennhage, 2002; Tuya *et al*., 2005) with a relatively consistent effect through time

369 (lack of differences from before to after the cessation of farming). The increased habitat

370 complexity due to the sea-cage moorings creates many niches for fishes, providing a

371 fixed substrate as shelter, and favoring the establishment of epiphytic algae and sessile

372 invertebrates that are consumed by higher trophic levels such as carnivorous fish (Bell

373 & Pollard, 1989; Klumpp *et al*., 1989; Jenkins & Wheatly, 1998; Guidetti, 2000;

374 Guidetti & Bussotti, 2002). Moreover, sea-cage fish farms act as 'recruitment points'

375 for larvae of inshore fish (e.g. *Oblada melanura*; Valle, 2005), and even for decapod

376 crustaceans (M. Davis unpubl. data; www.hboi.edu/aqua). These larvae may be a source

377 of food for species within the bentho-demersal macrocarnivorous group; stomach

378 content analyses are needed to test this hypothesis.

379

380 IMPLICATIONS FOR MANAGEMENT OF COASTAL FISH FARMS

381   Impacts of sea-cage fish farms on the benthos have been shown to be reversible, to

382 some extent, after farms have been removed in other areas (e.g. Greece; Karakassis *et*

383 *al*., 1999). We show here that the large scale aggregation of wild fish at a coastal fish

384 farm was also partially reversible after the cessation of farming, with assemblages

385 returning towards the 'normal' state represented by the wild fish assemblages at sandy

386 sediment control locations after 2 years. An important implication for coastal

387 management arising from this study is that natural assemblages of wild fish can be

388 partially restored to particular areas if a fish farm ceases farming. We further

389 hypothesize that with complete removal of the fish farm cages and mooring structures,

390 demersal wild fish assemblages would shift further towards those observed at natural

391 control locations.

392   Marine aquaculture installations have been described as competing for space with

393 future potential Marine Protected Areas (MPAs) (www.wwf.org). While sea-cage fish

394 farms may be incompatible with MPAs designed to protect biodiversity as assemblages

395 shift away from those naturally observed, they may not be incompatible with MPAs

396 designed to enhance fisheries. An alternative solution to coastal managers adopting a

397 'one-or-the-other' approach to allowing fish farms or declaring MPAs in a given coastal

area may be to adopt a more sophisticated management framework that incorporates ecological knowledge of the wild fish that associate with farms (Dempster *et al.*, 2002, 2005). This study showed that a fish farm, once farming ended, did not greatly affect the demersal wild fish assemblage on a sandy bottom for more than a short period (< 2 yr). Further, while farms are in place, they concentrate large numbers of a variety of demersal fish species which constitute a portion of the available spawning stock (Dempster *et al.*, 2002; Boyra *et al.*, 2004; Dempster *et al.*, 2005; Tuya *et al.*, 2005; this study), which is the stated aim of MPAs designed for fisheries purposes. Rather than competing for space with MPAs designed for fisheries purposes, sea-cage fish farms should be designated as 'no-fishing zones' and incorporated into the management of coastal areas along with MPA zones. Nineteen farms operated in waters in the Canarian Islands in 2003 (www.apromar.es); however, no management measures to protect aggregations of wild fish in their vicinity from fishing are currently in place. Future management of the impacts of sea-cage aquaculture in the Canarian Archipelago should consider protection of wild fish aggregations as an option to ensure sustainable development of the industry.

References

Bell, J.D. & Pollard, D.A. (1989). Ecology of fish assemblages and fisheries associated with seagrasses. In *The biology of Seagrasses: An Australian Perspective* (Larkum, A.W.D., McComb, A.J. & Shepherd, S.A., eds.), pp 565-609. Amsterdam: Elsevier.

424 Boyra, A., Sanchez-Jerez, P., Tuya, F., Espino, F. & Haroun, R. (2004). Attraction of
425 wild coastal fishes to Atlantic subtropical cage fish farms, Gran Canaria, Canary
426 Islands. *Environmental Biology of Fish* **70**, 393-401.

427

428 Brito, A., Pascual, P.J, Falcón, J.M., Sancho, A. & González, G. (2002). *Peces de las*
429 *Islas Canarias. Catálogo comentado e ilustrado*. Francisco Lemus Editor, Islas
430 Canarias.

431

432 Brooks, K.M., Stierns, A.R. & Backman, C. (2004). Seven year remediation study at the
433 Carrie Bay Atlantic salmon (*Salmo salar*) farm in the Broughton Archipelago, British
434 Columbia, Canada. *Aquaculture* **239**, 81-123.

435

436 Carss, D.N. (1990). Concentrations of wild and escaped fishes immediately adjacent to
437 fish farm cages. *Aquaculture* **90**, 29-40.

438

439 Clarke, K.R. (1993). Non-parametric multivariate analyses of changes in community
440 structure. *Australian Journal of Ecology* **18**, 117-143.

441

442 Clarke, K.R. & Warwick, R.M. (1994). *Change in marine communities: an approach to*
443 *statistical analysis and interpretation*. Plymouth: Natural Environmental Research
444 Council.

445

446 Dempster, T., Sanchez-Jerez, P., Bayle-Sempere, J.T., Giménez-Casalduero, F. & Valle,
447 C. (2002). Attraction of wild fish to sea-cage fish farms in the south-western
448 Mediterranean Sea: spatial and short-term temporal variability. *Marine Ecology*
449 *Progress Series* **242**, 237-252.

450

Dempster, T., Sanchez-Jerez, P., Bayle-Sempere, J. & Kingsford, M. (2004). Extensive aggregations of wild fish at coastal sea-cage fish farms. *Hydrobiologia* **525**, 245-248.

Dempster, T., Fernandez-Jover, D., Sanchez-Jerez, P., Tuya, F., Bayle-Sempere, J., Boyra, A. & Haroun, R.J. (2005). Vertical variability of wild fish assemblages around sea-cage fish farms: implications for management. *Marine Ecology Progress Series* (in press).

Ebert, D.A. & Cowley, P.D. (2003). Diet, feeding behaviour and habitat utilization of the blue stingray *Dasyatis chrysonata* (Smith, 1828) in South African waters. *Marine and Freshwater Research* **54**, 957-965.

Fanlo, M., González, J.A. & Carrillo, J. (1996). Régimen alimentario de *Pagellus erythrinus* (L.) (Ostheichthyes, Sparidae) en las islas Canarias. In *Oceanografía y Recursos Marinos en el Atlántico Centro-Oriental* (Llinás, O., Gonzalez, J.A. & Rueda, M.J., eds.), pp 271-281. Las Palmas de Gran Canaria.

FAO. (2003). *Fishstat Plus. Aquaculture production: quantities 1950-2002*. FAO, Rome.

Ferlin, P. & La Croix, D. (2000). Current state and future development of aquaculture in the Mediterranean region. *World Aquaculture* **31**, 20 – 23.

474   Glasby, T. (1997). Analysing data from post-impact studies using asymmetrical

475   analyses of variance: a case study of epibiota on marinas. *Australian Journal of Ecology*

476   **22**, 448 – 459.

477

478   Gray, A.E., Mulligan, T.J. & Hannah, R.W. (1997). Food habits, occurrence, and

479   population structure of the bat ray, *Myliobatis californica*, in Humboldt Bay, California.

480   *Environmental Biology of Fish* **49**, 227-238.

481

482   Guidetti, P. (2000). Differences among fish assemblages associated with nearshore

483   *Posidonia oceanica* seagrass beds, rocky-reefs and unvegetated sand habitats in the

484   Adriatic Sea. *Estuarine Coastal and Shelf Science* **50**, 515-529.

485

486   Guidetti, P. & Bussotti, S. (2002). Effects of seagrass canopy removal on fish in shallow

487   Mediterranean seagrass (*Cymodocea nodosa* and *Zostera noltii*) meadows: a local-scale

488   approach. *Marine Biology* **140**, 445-453.

489

490   Guidetti, P., Terlizzi, A., Fraschetti, S. & Boero, F. (2003). Changes in Mediterranean

491   rocky reef fish assemblages exposed to sewage pollution. *Marine Ecology Progress*

492   *Series* **253**, 269-278.

493

494   Harmelin–Vivien, M.L., Harmelin–Vivien, J.G., Chauvet, C., Duval, C., Galzin, R.,

495   Lejeune, P., Barnabé, G., Blanc, F., Chevalier, J., Duclerc, J. & Lasserre, G. (1985).

496   Evaluation visuelle des peuplements et populations de poissons: méthodes et problèmes.

497   *Terre Vie* **40**, 467-539.

498

499   Iwama, G.I. (1991). Interactions between aquaculture and the environment. *Critical*

500   *Reviews in Environmental Control* **21**, 177-216.

501

502   Jenkins, G.P. & Wheatly, M.J. (1998). The influence of habitat structure on nearshore

503   fish assemblages in a southern Australian embayment: comparison of shallow seagrass,

504   reef algal and unvegetated sand habitats, with emphasis on their importance to

505   recruitment. *Journal of Experimental Marine Biology and Ecology* **221**, 147-172

506

507   Karakassis, I., Hatziyanni, E., Teacakes, M. & Plaiti, W. (1999). Benthic recovery

508   following cessation of fish farming: a series of successes and catastrophes. *Marine*

509   *Ecology Progress Series* **184**, 205-218.

510

511   Kingsford, M. & Battershill, C. (1998). *Studying Marine Temperate Environments: a*

512   *handbook for ecologists*. Canterbury University Press. Christchurch, New Zealand.

513

514   Klumpp, D.W., Howard, R.K. & Pollard, D.A. (1989). Trophodynamics and nutritional

515   ecology of seagrass communities. In *The biology of Seagrasses: An Australian*

516   *Perspective* (Larkum, A.W.D., McComb, A.J. & Shepherd, S.A., eds.), pp 394-457.

517   Amsterdam: Elsevier.

518

519   Pihl, L. & Wennhage, H. (2002). Structure and diversity of fish assemblages on rocky

520   and soft bottom shores on the Swedish west coast. *Journal of Fish Biology* **61**, 148-166.

521   doi: 10.1006/jfbj.2002.2074.

522

523 Rönenberg, O., Adjers, K., Ruokolahti, C. & Bondestam, M. (1992). Effect of fish

524 farming on growth, epiphytes and nutrient contents of *Fucus vesiculosus* L. in the Aland

525 Archipelago, Northern Baltic Sea. *Aquatic Botany* **42**: 109-120.

526

527 Ruiz, J.M., Pérez, M. & Romero, J. (2001). Effects of fish farm loadings on seagrass

528 (*Posidonia oceanica*) distribution, growth and photosynthesis. *Marine Pollution*

529 *Bulletin* **42**, 749-760.

530

531 Tuya, F., Sanchez Jerez, P., Boyra, A. & Haroun, R.J. (2005). Non-metric multivariate

532 analysis of the demersal ichthyofauna along soft bottoms of the Eastern Atlantic:

533 comparison between unvegetated substrates, seagrass meadows and sandy bottoms

534 under the influence of sea-cage fish farms. *Marine Biology* 147: 1229-1237.

535

536 Valle, C. (2005). *Impactos antrópicos sobre la ictiofauna litoral*. PhD thesis, University

537 of Alicante.

538

539 Vita, R., Marín, A., Madrid, J.A., Jiménez-Brinquis, B., Cesar, A., Marín-Guirao, L.

540 (2004). Effects of wild fishes on waste exportation from a Mediterranean fish farm.

541 *Marine Ecology Progress Series* **277**, 253-261.

542

543 Underwood, A.J. (1991). Beyond – BACI: experimental designs for detecting human

544 impacts on temporal variations in natural populations. *Journal of Marine and*

545 *Freshwater Research* **42**, 569-587.

546

547 Underwood, A.J. (1992). Beyond BACI: the detection of environmental impacts on

548 populations in the real, but variable world. *Journal of Experimental Marine Biology and*

549 *Ecology* **161**, 145-178.

550

551 Underwood, A.J. (1993). The mechanism of spatially replicated sampling programmes

552 to detect environmental impacts in a variable world. *Australian Journal of Ecology* **18**,

553 145-178.

554

555 Underwood, A.J. (1994). On beyond BACI: sampling designs that might reliably detect

556 environmental disturbances. *Ecological Applications* **4**, 3-15.

557

558 Underwood, A.J. (1997). *Experiments in Ecology: their logical design and*

559 *interpretation using Analysis of Variance*. Cambridge University Press, Cambridge.

560