

Changing status of three notothenioid fish at the South Shetland Islands (1983–2016) after impacts of the 1970–80s commercial fishery

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Abstract Owing to commercial fishing during the late 1970s/early 1980s, targeted notothenioid species had become depleted around the South Shetland Islands. Herein we report subsequent changes in the prevalence of three species, *Notothenia rossii*, *Gobionotothen gibberifrons* and *Notothenia coriiceps* in Potter Cove, King George Islands/Isla 25 de Mayo, in a 33-year effort to monitor recovery. *N. rossii* and *G. gibberifrons* had been severely impacted by industrial fishing but in offshore waters *N. coriiceps* had never been commercially fished; however, all three species exhibit similar nearshore habitats and life history. We examined composition in trammel net catches during 2012–2016, augmenting a time series started in 1983. Our inshore results were consistent with those from offshore bottom trawl sampling in 2007 and 2012 around the South Shetland Islands: (1) continued increase in the abundance of *N. rossii*; (2) further decline in *G. gibberifrons* recruitment evidenced by low proportions of juvenile fish; and (3) a high abundance of *N. coriiceps*. Reasons for lack of recovery in *G. gibberifrons* remain obscure but seemingly relate to the dramatically changing ecosystem of the region due in part to climate as well as recovery among previously depleted upper trophic level species. Our results are also consistent with trends reported in seabirds that feed on

juveniles of these notothenioids: decrease in the areas commercially fished. Under the regulation of CCAMLR, commercial fishing for finfish in the South Shetland Islands region (FAO Subarea 48.1) remains prohibited since 1991; results indicate that it cannot be reinstated.

Keywords Antarctic Nototheniidae · Fishery depletion · Slow recovery · CCAMLR

Introduction

In a classic example of “fishing down the food web”, with depletion first of seals and whales, then finfish, current fishing in the South Shetland Islands area of the Southern Ocean (FAO Statistical Subarea 48.1, Fig. 1a) is now confined to Antarctic krill, *Euphausia superba* Dana 1850 (Ainley and Pauly 2014). Finfishing in this region, beginning at the end of the 1970s, first targeted marbled notothenia, *Notothenia rossii* Fischer 1885, with reported catches of 41,000 tonnes, as well as mackerel icefish, *Champsocephalus gunnari* Lönnberg 1905, with catches of 19,000 tonnes. The humped rockcod, *Gobionotothen gibberifrons* Lönnberg 1905, was taken as by-catch (4200 tonnes) along with several other species. However, it is noteworthy that these reported catches (CCAMLR Statistical Bulletin) might be underestimates because of illegal and unreported catches as well as possible errors in species identification. The latter is relevant because miss-identification likely implies much higher catch levels in the targeted fisheries. Most of target and by-catch species became so depleted that by 1985 fishing for them was no longer economically feasible (CCAMLR 1986; Kock 1992). Shortly after the inception of the Convention for the Conservation of Antarctic Marine Living Resources

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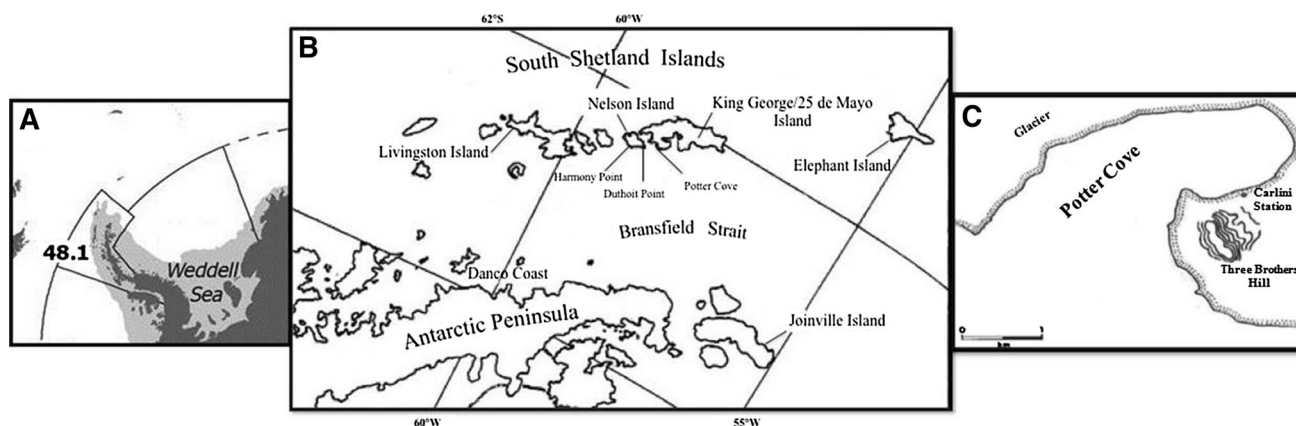


Fig. 1 **a** General view of FAO Statistical Subarea 48.1 in the southern Scotia Arc. **b** Enlargement of the western Antarctic Peninsula–South Shetland Islands region showing the main localities considered in this study. **c** The studied site, Potter Cove

(CCAMLR) in 1982, its Commission invoked a series of adopted conservation measures including the closing of Subarea 48.1 to finfishing in an attempt to promote recovery of these and other overexploited fish. Commercial fishing for these species in this region is still prohibited (CCAMLR 2015).

Fisheries in the CCAMLR region are managed under the proviso that harvesting should not lead to changes in the ecosystem or target species that cannot be reversed within two or three decades. Such a strategy relies strongly on the ability of the different populations to withstand harvest or to recover if left alone. Estimating recovery ability is difficult and has only been done on rare occasions when long time series have been available, especially in the face of climate change and natural variability of impacted food webs (CCAMLR 1982, text of the Convention).

After the commercial fishery in the South Shetland Islands area ended, to assess impacts on the inshore populations of the depleted nototheniids, a study was begun at Potter Cove, King George Island, initially from 1983 to 1990. Using trammel nets, the declining prevalence of *N. rossii* and *G. gibberifrons* relative to that of the unfished but co-occurring black rockcod, *Notothenia coriiceps* Richardson 1844 (previously known as *N. neglecta*), was noted (Barrera-Oro and Marschoff 1990). Data gathered before the heavy fishing in the late 1970s from nearby localities in the same Archipelago (Half Moon Bay, Livingston Island; Fildes Bay and Admiralty Bay, King George Island; and Discovery Bay, Greenwich Island) were included in the effort. Analysis showed that the relative proportions of *N. rossii* and *G. gibberifrons* at these locations were well above the starting point of the Potter Cove series in 1983, thus providing independent confirmation of these species' decline.

Subsequent monitoring in Potter Cove showed oscillations in the prevalence of juvenile *N. rossii*, which finally

reached its highest level in 2010. On the other hand, *G. gibberifrons* remained at very low levels and finally disappeared in 2007. For both species, the variations in relative abundances were accompanied by changes in length. Mean lengths increased simultaneously with decrease in relative abundance owing to the balance between the number of recruits entering the cove and the number of older fish moving offshore. During the whole period, 1983–2010, the stock of the unexploited *N. coriiceps* showed no discernible trends either in abundance, modal size or mean length (Marschoff et al. 2012). Results from Harmony Cove, over comparable periods, were consistent with those at Potter Cove (Barrera-Oro et al. 2000; Barrera-Oro and Marschoff 2007).

Marschoff et al. (2012) considered possible reasons, either environmental or biological, for the decrease in recruitment to the inshore sub-populations of *N. rossii* and *G. gibberifrons* in the South Shetland Islands. They attributed the decline to the brief but heavy fishery in the late 1970s, as shown by the size distribution in trammel net catches of *N. rossii* during 1983–87. Findings were consistent with information from scientific bottom trawl conducted around Elephant Island and the lower South Shetland Islands: a decline in the offshore populations of *N. rossii* and *G. gibberifrons* in the early 1980s, a slow recovery of *N. rossii* and further decline of *G. gibberifrons* by the end of the 1990s (Jones et al. 2001, 2003; Kock et al. 2002).

Monitoring efforts at Potter Cove continued through 2016, with new results presented herein, thus providing a continuous sampling over a 33-year period. With these new data, we show trends in relative abundance of the three nototheniids, aiming to evaluate: (1) whether the recovery of *N. rossii* observed from 1995 to 2010 has continued and if so, to what levels in comparison to those registered in the early 1980s; (2) whether the lack of recovery (and

disappearance) of *G. gibberifrons* in comparison to *N. rossii* has continued, and if so, to discuss the possible reasons; and (3) whether *N. coriiceps* remains as the most abundant species.

Materials and methods

Fish samples were collected at Potter Cove, King George Island/Isla 25 de Mayo, South Shetland Islands, close to the Scientific Station Carlini (ex Jubany, 62°14'S and 58°40'W) during the summer, 2011–2016 (Fig. 1a, b). Trammel nets (length 25, 35 and 50 m; height 1.5 m; inner mesh 2.5 cm; outer mesh 12 cm) were fixed to rocks for 6–96 h at 5–50 m depths in sites where the seabed is a uniform rocky bottom covered mainly with red and brown macroalgae. Detailed information of the biotic and abiotic features of these sites is given in Casaux et al. (1990) and Barrera-Oro et al. (2000). The fish were sexed and measured (Total Length, TL) to the nearest 0.1 cm. Gonadal stages were determined as described in Kock and Kellermann (1991).

In this study, we also include data obtained at the same location from 1983 to 2010, previously analysed (Barrera-Oro et al. 2000; Marschoff et al. 2012). Thus, we present here a single analysis of the total period from 1983 to 2016. Details of the methodology are described in Barrera-Oro and Marschoff (1990), but a portion of that description is reproduced here for clarification.

Since soak time and net size depend on weather conditions and experimental design, fishing effort per haul was highly variable between years and within the same season. Therefore, for analysis, the abundance of fish was enumerated both (1) as the total number of each fish species per haul and (2) as standardized numbers of *N. rossii* and *G. gibberifrons* relative to *N. coriiceps*. Trends were analysed by fitting a non-parametric regression model using smoothing techniques to obtain a continuous function.

The proportion of *N. rossii* and *G. gibberifrons* relative to *N. coriiceps* was obtained according to:

$$\text{Proportion } (b) = \frac{N_b}{N_c + N_b},$$

where N_b is the number of specimens of the species considered (*N. rossii* or *G. gibberifrons*), and N_c is the number of specimens of *N. coriiceps*.

The non-parametric regression model was fitted to the proportions and absolute abundances (Barrera-Oro et al. 2000) using a bandwidth of 0.15, equivalent to 1.65 years. The selection was made in order to reduce variance as much as possible without losing details of the process. Different bandwidths were tested in order to study whether the results are sensitive to bandwidth selection.

Results

Data from 3361 specimens of *N. coriiceps* (1871) and *N. rossii* (1490) collected in 336 trammel net deployments were acquired during 2012–2016. The size range of *N. coriiceps* was 14.5–53.9 cm TL, including immature and mature specimens; all *N. rossii* (13.9–47.5 cm TL) specimens were juvenile.

These data were added to the 1983–2010 series for analysis. Thus, 15,958 *N. coriiceps*, 203 *G. gibberifrons* and 4033 *N. rossii* caught in 1592 hauls are included in the present study. A total of 2362 fish of other species were not considered (i.e. *Trematomus* spp., *Notothenia nudifrons*, *Chaenocephalus aceratus*, *Parachaenichthys charcoti*).

The total number of fish per haul shows the relative importance of each species in the catches (Fig. 2). The effect of different bandwidths (0.99–2.64 years) used to test for overall consistency between observed numbers and relative abundances was evaluated (Fig. 3), but whatever the value of the smoothing parameter the results were the same. It should also be noted that, since relative and absolute abundances behave similarly, the observed process cannot be considered an artefact of the interaction between *N. rossii* and *N. coriiceps*.

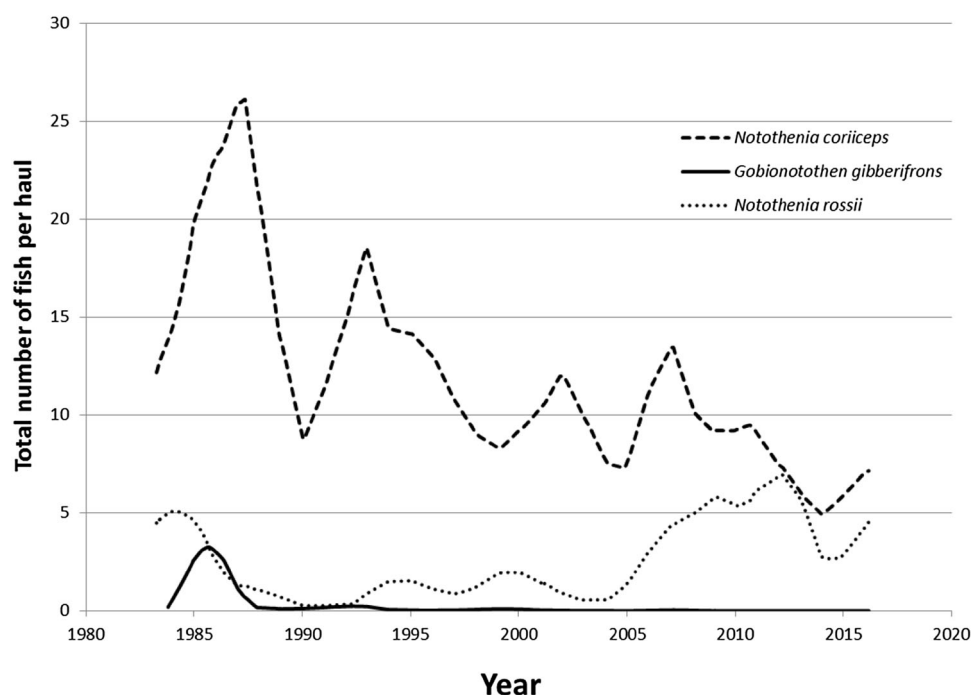
A general increasing trend in the recruitment of *N. rossii* is evident from 2005 to 2012. More recent data (up to 2016) show a fluctuation from which we suggest that recruitment may have stabilized at levels close to those observed in 1983, a few years after the fishery peaked. After 1987 *G. gibberifrons* declined steadily, being caught in very small numbers since 1995 and disappearing in trammel net catches at Potter Cove after 2012 (Fig. 2).

The unexploited *N. coriiceps* remained as the most abundant fish species in terms of numbers and biomass at Potter Cove during the whole monitoring period, despite a long-term declining trend (Fig. 2).

Discussion

Studies of variation in the abundance of Antarctic inshore demersal fish by means of trammel net sampling were initially carried out by Duhamel (1990) in the Morbihan Gulf, Kerguelen Islands. A decrease in catches of juvenile *N. rossii* inshore in those waters was attributed to commercial fishery operations offshore, followed by a recovery of the juvenile stock from 1984 to 1988 after closure of the fishery (Duhamel 1990). Similar declines among inshore juvenile fish have been observed elsewhere, e.g. in coastal waters of northern Canada (McCain et al. 2016). Indicated in that regard was a substantial reduction in juvenile Atlantic cod (*Gadus morhua*) and pollock (*Pollachius virens*) owing to overfishing of adult stocks.

Fig. 2 Estimated number of fish per haul against time at Potter Cove, 1983–2016, smoothed using bandwidth 1.65 for analysis



While there is precedent for using trammel nets to analyse trends of Antarctic fish populations by means of CPUE (catch per unit effort), it is a challenge owing to variable conditions, which cause problems in maintaining constant effort. Moreover, the number of fish caught per net does not increase as a simple function of net soak time. The catch rate depends on many other uncontrolled factors, such as water movements and water transparency (due to wind, tide, and local water circulation), daylight cycle (dependent on season) and prey availability. Regardless, in our study, the total number of fish per haul (Fig. 2) shows the relative importance of each species in the catches. Therefore, instead of working only with CPUE, we also used *N. coriiceps* as a reference species to obtain an index of trends in the abundances of *N. rossii* and *G. gibberifrons*. All three of these fish species exhibit similar ecological habits in the fjords, with *N. coriiceps* being the only one not fished.

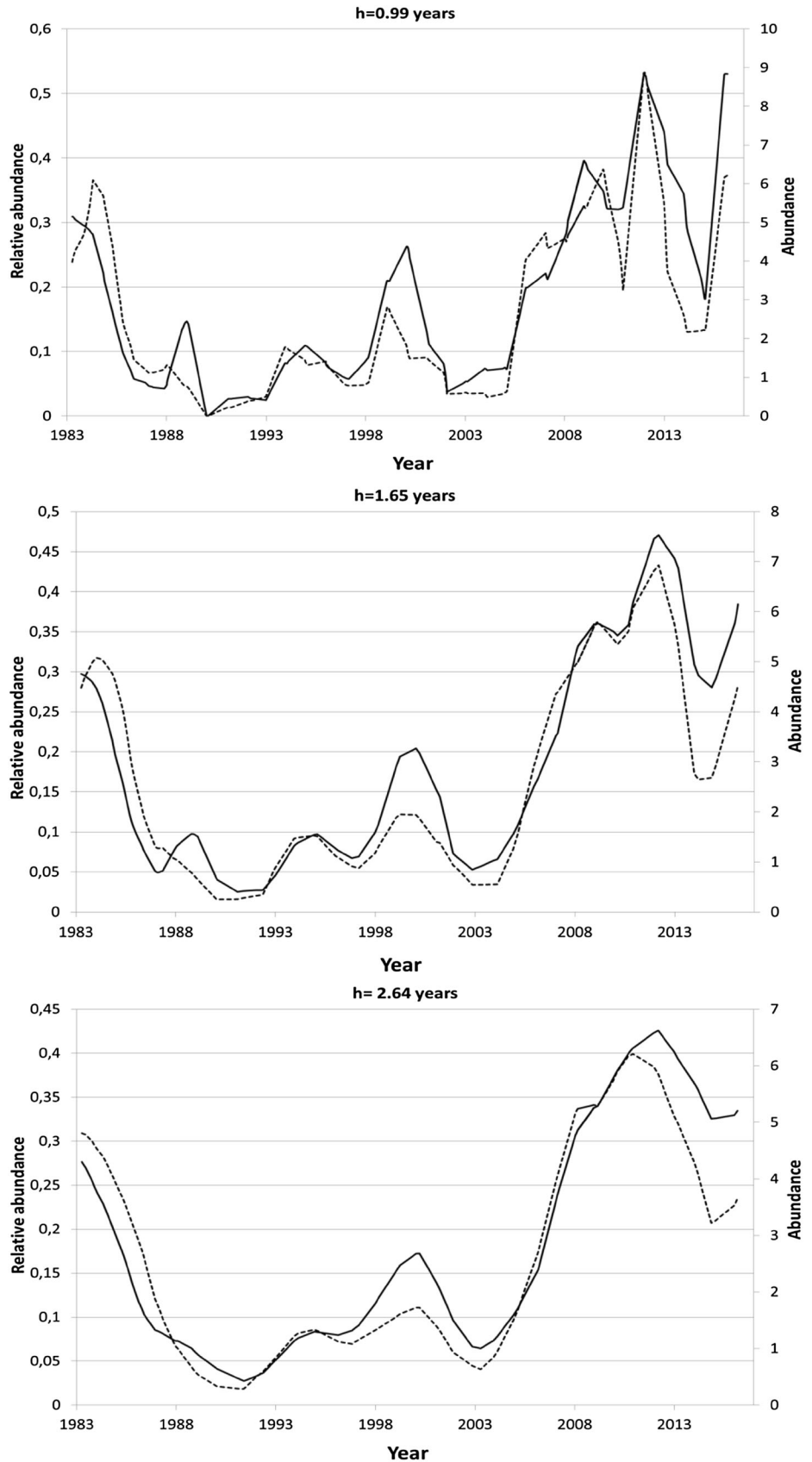
Notothenia coriiceps is the dominant inshore demersal fish of the South Shetland Islands and has a high degree of site fidelity (Barrera-Oro and Casaux 1996; Kock et al. 2012), and that attribute adds to its utility as an index species. The apparent decline observed in the catches of *N. coriiceps* (Fig. 2) might be attributed to variability in fishing effort. A long-term reduction in soak time of the trammel nets was required in order to increasingly use specimens for mark and recapture experiments (Barrera-Oro and Casaux 1996; Moreira et al. 2013). The latter work allows better understanding of factors that could affect recruitment of inshore species.

Besides using *N. coriiceps* as an index species to provide an ecological context to trends in *N. rossii* and *G. gibberifrons*, further corroboration of the trammel net monitoring, and in the context of the CCAMLR Ecosystem Monitoring Program, has been provided by population trends of an important predator of the juveniles of these fishes, the Antarctic Shag, *Phalacrocorax bransfieldensis* King 1828. Casaux and Barrera-Oro (2006) linked the declining numbers of breeding pairs of shags in two colonies on Nelson Island, a locality near Potter Cove, to decreasing availability of these fishes. This trend was evident from the 1990s through the mid-2000s at that locality but not at the Danco Coast colonies, located a long way from the exploited area (Casaux and Barrera-Oro 2016).

Overall, the abundance of *N. rossii* has increased since this study was initiated (Fig. 2) reaching values compatible with acceptable recruitment relative to values observed in 1983. Regardless, the present situation should not be construed as indicating full recovery of the adult population because from our data it is impossible to ascertain the actual position of the population in the stock-recruitment curve. Independent estimates of the abundance of the stock are needed to determine its true status before setting the Total Allowable Catch needed for management of any future fishery. This requisite was later added in a general CCAMLR Conservation Measure (CM 32-02) for a number of species and areas (CCAMLR 2015).

Gobionotothen gibberifrons is abundant in shallow water trammel/gill net catches (5–70 m depth) at the Danco

Fig. 3 Effect of different bandwidths on the evaluation of relative (solid line) and absolute (dashed line) expected catches of *Notothenia rossii* as function of time. Left column relative values; right column absolute means; h interval of years of the means



Coast, western Antarctic Peninsula (Casaux et al. 2003) and at James Ross Island, eastern Antarctic Peninsula (Jurajda et al. 2016). This reflects a different situation for this species in an area remote from the main historical fishing grounds, north of the South Shetland Islands and the tip of the Antarctic Peninsula (Joinville Island). Further separation of these areas results from water in between, which are more than 1200 m deep along with well developed slope fronts.

As reported by several authors (e.g. Everson 1970; Barrera-Oro et al. 2000; Mintenbeck et al. 2012) *N. coriiceps* is the most successful nearshore species in terms of abundance and biomass in the shallow water communities around the islands of the southern Scotia Arc. It is possible, as a hypothesis to explain greater abundance, that *N. coriiceps* may have encountered progressively less competition with exploited and depleted notothenioid species (Nototheniidae and Channichthyidae) and consequently expanded its trophic habitat and niches.

Consistent with our inshore results are those from the most recent bottom trawl research cruises carried out in 2007 and 2012 offshore of the South Shetland Islands (including Elephant Island): (1) a continuous increase in the abundance *N. rossii*; (2) further decline in *G. gibberifrons* recruitment indicated by the low proportion of juvenile fish; and (3) a high abundance of *N. coriiceps* (Kock et al. 2007; Kock and Jones 2012). Results from demersal fish surveys around South Georgia after the heavy commercial fishery in that region during the late 1960s to early 1970s, in comparable periods to our trammel net data series in the South Shetland Islands, agree on the very slow process of recovery observed for *N. rossii* and other demersal species (Belchier 2013).

In addition to harvesting, other possible factors may explain the slow recovery of the exploited Antarctic fish species in Subarea 48.1. As noted in our previous study, and likely still applicable: (1) continued illegal fishing; (2) depletion of the pelagic larval and juvenile fish through by-catch in the fishery for Antarctic krill; (3) depensation, owing to low population size; and/or (4) climate change (Marschoff et al. 2012). Although climate change is often invoked by authors to explain biotic trends of recent years in the Scotia/Antarctic Peninsula region (e.g. Murphy et al. 2007; Montes-Hugo et al. 2009; Schofield et al. 2010; Trathan and Agnew 2010), it is difficult to identify how climate change might be involved in what we have reported for South Shetland fish, given the contrasting trends we have identified among co-habiting species. Also to be considered should be the extraction from the system of >1 M baleen whales during the 1950–60 s, as well as extirpation of fur seals, *Arctocephalus gazella* Peters 1875, both of which may have caused a dramatic increment in the abundance of one of their preys, Antarctic krill (the “krill

surplus hypothesis”) (e.g. Emslie and Patterson 2007), and rippling effects in the food web. These impacts were concurrent with the massive depletion of finfish in the 1970s, constituting important events that should not be ignored to explain, at least partially, subsequent major changes in the food web structure of Antarctic marine ecosystems (Ainley et al. 2007 and references therein). Indeed, the recovery of fur seals is complete, bringing negative repercussions to trophically competing species that did well in their absence in the Scotia region (Trathan et al. 2012; Hofman 2016). Likewise, some species of baleen whales, e.g. humpback *Megaptera novaeangliae* Borowski 1781, are rapidly recovering in Subarea 48.1 (Branch 2011), also having repercussions for trophically competing species (e.g. Trivelpiece et al. 2011; Lynch et al. 2012; Hofman 2016). Within this context of dramatic food web and ecosystem change in the past and currently underway, causes to explain the total lack of recovery of *G. gibberifrons* at Potter Cove (this study) will remain a mystery, begging for further, directed study (see also Kock and Jones 2012).

Regardless of what factors might be involved in facilitating observed trends in the South Shetlands fish, results of the most recent offshore surveys and those of our long-term inshore monitoring program show that *N. rossii* and *G. gibberifrons* populations in the South Shetland Islands region cannot, at present, sustain a commercial fishery. Upon any consideration to reopen the fishery, it should be taken into account that these species have exhibited very low recovery rates, a fact to be considered in complying with Article II.3 (c) of the Convention, i.e. fishing levels can not preclude recovery within 2–3 decades (CAMLR 2012, paragraph 3.93). In retrospect and in light of the trends that we report and summarize, that time frame now seems overly ambitious at least for benthic finfish.

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