



Slow recovery of previously depleted demersal fish at the South Shetland Islands, 1983–2010

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ABSTRACT

Following seals and baleen whales prior to the 1970s, demersal fish stocks were depleted off the South Shetland Islands by intensive industrial fishing during the late 1970s to early 1980s. Little has been reported since about how these stocks have fared, after international agreement closed this fishery in 1990. We report changes in size and abundance of the commercially exploited *Notothenia rossii* and *Gobionotothen gibberifrons* relative to the ecologically similar but unexploited *Notothenia coriiceps* at Potter Cove, South Shetland Islands, over a 28-yr period, 1983–2010. *N. rossii* abundance declined from 1983 to 1991, and an increase in mean size during 1983–1984 is consistent with weak cohorts during preceding years. Modal age changed from 2–3 to 6–7 yr. Length data of *G. gibberifrons*, available from 1986, exhibited a similar pattern, showing a decrease until 1991–1992. After a period of relative stability (1992–1994), a sharp increase in length and a continued decline in relative abundance indicated low recruitment. The length–frequency distribution of unexploited *N. coriiceps* throughout the whole period showed no change in modal size or mean length of the fish. We relate these patterns to the fishery and suggest that a further two decades will lapse before these stocks recover. Using the South Shetland fisheries as an example, current management rules for Southern Ocean fisheries, deemed to be precautionary and disallowing depletion beyond which a stock can recover in 2–3 decades, may be unrealistic in an ocean profoundly altered by numerous stock depletions and rapid climate change.

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1. Introduction

Commercial fishing of finfish in the Antarctic began in the 1960s just as decades of sealing and whaling were ending. In the Scotia Sea region, the fishery started in waters around South Georgia in 1969, the South Orkney Islands in 1977 and the South Shetland Islands in 1978 (Kock, 1992; also Laws, 1977, for sealing and whaling history). Targeted fish included marbled rockcod (*Notothenia rossii*) and humped rockcod (*Gobionotothen gibberifrons*). Little was known of these species prior to commercial fishing (Kock, 1992; Eastman, 1993). Owing to severe depletion, fisheries for these species have been prohibited by CCAMLR (Convention for the Conservation of Antarctic Marine Living Resources) in the waters around South Georgia since 1985 (FAO Statistical Subarea 48.3) and the South Shetland Islands (Subarea 48.1) since 1990 (Kock, 1992; CCAMLR, 2010). CCAMLR management was not responsible for these species' depletion, and the agency strives to manage fisheries such that stocks will not fall below the level at which recovery cannot be

achieved in 2–3 decades (CCAMLR, 2011a). We herein provide a test of that strategy for two exploited fishes.

The diversity (45%) and biomass (95%) of the Antarctic fish fauna is dominated by a single, endemic group, the Notothenioidei, which includes five families of mostly demersal species found generally to depths down to 1200–1500 m (Eastman, 2005). The Antarctic notothenioids are characterized by slow growth and low fecundity, which make them particularly susceptible to overexploitation (Norse et al., 2012). In the seasonal pack ice zone of the Southern Ocean and the islands lying north of it, species diversity and biomass of notothenioids are greatest from 100 to 300 m depth (Tiedtke and Kock, 1989). Thus, fishing down to 450 m, the historical fishery caught mostly these fish. Vessels did not operate inshore because of the many uncharted, subsea rocks present, despite known fish abundance in fjords and bays. For example, existing information indicates that pre-industrial exploitation quantities of *N. rossii* around South Georgia were large; many thousands were caught for local consumption at whaling stations and regularly sold salted in Buenos Aires (Binnie, 1917; Tønnessen and Johnsen., 1982; Matthews, 1978).

In the 1970s the fishery targeted *N. rossii* and the mackerel ice-fish (*Champscephalus gunnari*), but took a considerable by-catch

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of other species as well (Kock, 1992). A maximum of 400,000 and 125,000 t of *N. rossii* and *C. gunnari*, respectively, were caught in the 1970–1971 season around South Georgia (Fischer and Hureau, 1985; Kock, 1992), after which, not surprisingly, these stocks collapsed [Hereafter, an austral summer season, e.g. 1970–1971, is represented by the initial year, e.g. 1970.] For example, until 1992, the stock size of *N. rossii* in that area was estimated to be <5% of its original size in 1969 (Kock, 1992).

Farther south around the South Shetland Islands, heavy fishing was reported mainly from its northern-most island, Elephant Island, during 1977–1980 (CCAMLR, 1986). *N. rossii* (41,000 t) and *C. gunnari* (19,000 t) were the target species, but *G. gibberifrons* was also taken in significant quantities as by-catch (4200 t; Kock, 1992).

For appropriate species, consistent sampling of coastal fish populations can contribute to detecting changes in the offshore portion of the stock (Casaux and Barrera-Oro, 2002). Young specimens of *N. rossii* and *G. gibberifrons*, though exploited offshore, also inhabit inshore littoral waters. Associated with these species, the black rockcod (*Nototothenia coriiceps*) is a species that was not commercially fished but has ecological habits in the fjords similar to the exploited species. In summary, these three demersal notothenioids spend at least part of their life cycles in inshore waters (<120 m deep) and also occur in offshore waters at depths to 200–550 m (Barrera-Oro, 2002; see below).

Earlier research, comparing catches among the three nototheniid species, showed a sharp decline in the abundance of young *N. rossii* and *G. gibberifrons* but not of *N. coriiceps* during 1983–1990 at several inshore South Shetland sites. This paralleled what was happening to the overall stock of the first two species; subsequent sampling to 2007 showed that the inshore populations of the exploited species remained in poor condition (cf. Kock, 1992; Barrera-Oro et al., 2000; Barrera-Oro and Marschoff, 2007). This decrease was attributed to a reduction in recruitment due to the decline of the offshore populations, subsequent to overfishing in the late 1970s and early 1980s.

The alternative hypotheses to explain the decline in the two species is an increase in the abundance of *N. coriiceps* rather than a decline in the recruitment of *N. rossii* and *G. gibberifrons*. To answer this question, we examined size and age of the fish, thus adding to our earlier analyses (Barrera-Oro et al., 2000; Barrera-Oro and Marschoff, 2007). This paper presents a comparison of the mean annual lengths of the three fish species and density distributions of *N. rossii* and *N. coriiceps* through a 28-yr period at Potter Cove, 1983–2010. We predict that, if the two over-fished species are recovering, this should be evident by an increase in the relative number of inshore small fish and a reduction in the mean size indicating stronger year classes (and ultimately better recruitment) among the inshore segments of their respective populations.

1.1. Summary of the fish life cycles

The life cycle of *N. rossii* has been well studied at South Georgia (Olsen, 1954; Burchett, 1983) and at the Kerguelen Islands (Duhamel, 1982). These studies showed that eggs are spawned on the bottom of continental shelf areas at about 120–350 m depth in offshore waters and there, the pelagic larvae, post larvae and young fingerlings inhabit surface layers. After about seven months, the blue phase fingerlings enter the fjords, change morphologically and become demersal, whereupon the fish transform to the brown phase fingerling, that persists for the remaining juvenile period. At about 5–7 yr and 41–45 cm TL, *N. rossii* reaches sexual maturity and migrates to outer shelf waters, where it joins the adult population; they spawn on the bottom at 120–350 m depth. These offshore–inshore phases are similar throughout the species' range (Kock et al., 1985), including the South Shetland Islands region (Barrera-Oro, 2002; Barrera-Oro et al., 2010), and

are similar to many other deep-water demersal species, such as Patagonian toothfish (*Dissostichus eleginoides*; Collins et al., 2010) and the well known subarctic sable fish (*Anoplopoma fimbria*; see Beamish et al., 2006, and references therein).

G. gibberifrons exhibits length stratification by depth. Mainly juveniles and part of the adult population occur in inshore waters 15–30 m deep; adults occur in offshore, deeper waters, at depths down to c. 450 m (Casaux et al., 1990; Kulesz, 1994; Barrera-Oro, 2002). Fish of 4–12 yrs in age were found in Potter Cove, our study area (Barrera-Oro, 1989).

N. coriiceps is the dominant fish in nearshore waters of the Scotia Sea (Everson, 1970; Barrera-Oro and Casaux, 2008), including the Joinville/D'Urville Islands at the tip of the Antarctic Peninsula (Jones and Kock, 2006; Kock et al., 2007). Data on its depth distribution at Potter Cove, at the Danco Coast in the western Antarctic Peninsula and at South Georgia indicate that this species may remain inshore during its whole life cycle (Burchett et al., 1983; Casaux et al., 1990, 2003). However, recent surveys in Elephant Island and southern South Shetland Island waters found pre-spawning aggregations of *N. coriiceps* mainly between 200 and 300 m depth (Jones et al., 2003; Kock et al., 2007), supporting previous indications that the species migrates to deeper waters to spawn (Everson, 1970; Hureau, 1970; Kock, 1989).

2. Methods

2.1. Sample collection

Trammel nets were fished at Potter Cove, King George Island/25 de Mayo Island, South Shetland Islands, close to the Scientific Station "Jubany" (62°14'S and 58°40'W), from 1983 to 2010 (Fig. 1a). A high proportion of the fish collected was used in biological studies (reviewed in Barrera-Oro and Casaux, 2008). Data from ~18,500 specimens belonging to *N. coriiceps*, *N. rossii* and *G. gibberifrons* were obtained. Total length (TL) to the nearest mm and sex of fish were recorded. Nets (length 25, 35 and 50 m; width 1.5 m; inner mesh 2.5 cm; outer mesh 12 cm) were set for 6–96 h at rocky, macroalgal beds at 5–50 m depths at three sites in the outer portion of the cove (Fig. 1b). The inner zone, bordered by a glacier, has a bottom covered by glacial sediments and is devoid of both algae and fish. The nets used are effective in Antarctic shallow waters, where the fish remain inaccessible to trawlers. Their advantages include the capture of a large quantity of fish in a short time, without by-catch of benthic organisms, as well as easy operation from small boats. As a trammel net is a passive sampling device, catches depend mainly, among other things, on fish activity that is assumed to reflect density dependence as reflected by the catches. For a detailed description of the fishing procedure, as well as of the biotic components and abiotic features of Potter Cove, refer to Casaux et al. (1990).

The age references for *N. rossii* were taken from the length-at-age keys given in Barrera-Oro and Casaux (1992), where the age determination was based on scales and whole otoliths readings. Part of the methodology used in that paper is reproduced here for clarification. Scales of 292 specimens and otoliths of 275 were used. Approximately 20 scales of each specimen were placed in a 5% alkaline solution for 48 h. Then, selected scales were cleaned mechanically, dried and mounted between microscope slides. The winter rings (checks) along the axis from the focus to the anterior margin were counted. A binocular microscope with transmitted light was used, at magnifications from 12× to 25×.

Whole otoliths could be observed on both sides, medial and lateral, by direct examination. They were treated either in 100% glycerine for 18–48 h, or in a detergent solution for 5–20 min. These time periods are critical, since after them the otoliths become

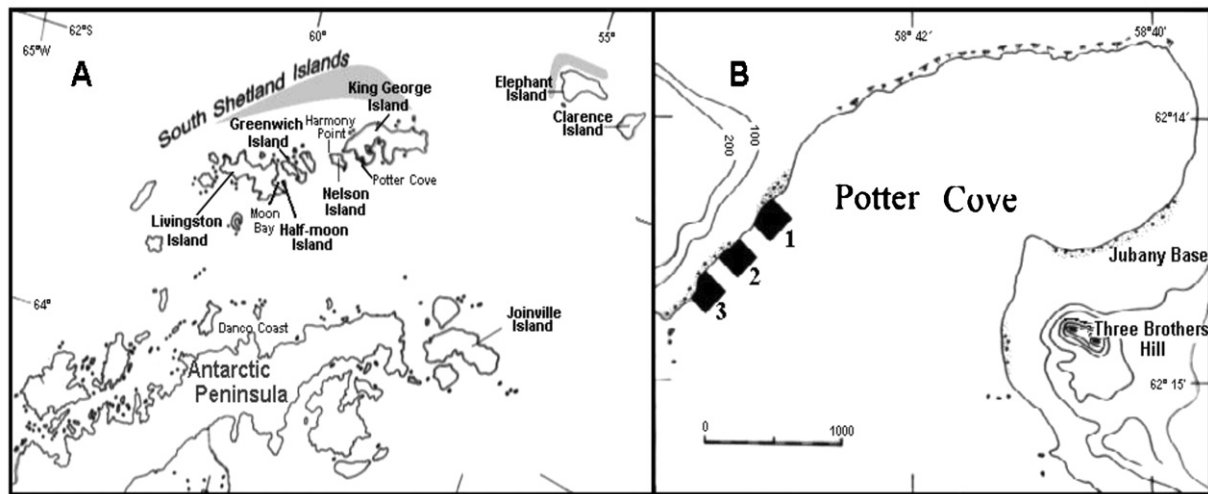


Fig. 1. (a) The South Shetland Islands–Antarctic Peninsula region showing the localities considered in this study. The offshore cruises have been conducted in the shaded area. (b) The sampling sites at Potter Cove.

irreversibly transparent. One opaque and one hyaline ring were considered to represent an annual ring (annulus). Five otoliths were sectioned to confirm the estimates from the whole structure. The sectioning technique and the reading method was similar to that used by Barrera-Oro and Casaux (1996a). Whole otoliths and sections were submerged in 96% alcohol and then examined with the same optical equipment as for the scales, using reflected light against a black background, at magnifications from 6× to 25×.

2.2. Statistical analysis

Abundance time series (see Fig. 2) were constructed as described in Barrera-Oro and Marschoff (2007). The usual proxy for abundance is the catch per unit effort (CPUE), a measure of the quantity of fish caught with a standardized unit of fishing effort (number of nets, soak time of nets, etc.). Since sampling did not utilize a consistent amount of effort between years, the relative abundance (RA) of *N. rossii* and *G. gibberifrons* was standardized for all samples as the proportion of these species in relation to *N. coriiceps* catches:

$$RA(b) = \frac{Nb}{Nc + Nb} \quad (1)$$

where Nb is the number of specimens of the species considered (*N. rossii* or *G. gibberifrons*) and Nc is the number of specimens of *N. coriiceps*, producing the values of CPUE of each species relative to the CPUE of *N. coriiceps* in each haul.

A non-parametric regression (Härdle, 1989) of RA against time was fitted for *N. rossii* and *G. gibberifrons* using the Epanechnikov kernel giving estimates of the RA by time. The smoothing technique allows adjustment of the shape of the curve to the available data near each x (in this case: time). The choice of the bandwidth is a matter of local precision balance, as small values reproduce the data and large values minimize the overall variance. A range of bandwidths were analysed starting from the value minimizing the cross validation sum of squares (in this case, roughly equivalent to one year).

RA was binned in years, with fishing periods extending from May 1 to April 30 in the next calendar year and a one way ANOVA model was fitted for each species. We used the variance stabilizing transformation, $y = \sqrt{RA + 0.5}$ in a nested design, to provide estimates of the annual means.

The annual size distributions were calculated using the maximum likelihood method (Thompson and Tapia, 1990), pooling all specimens caught within the timeframe defined above.

3. Results

Since all *N. rossii* (TL 10.3–47.2 cm) and *G. gibberifrons* (13–36.4 cm) were juveniles, length data from both sexes were combined (Table 1). The same procedure was followed with *N. coriiceps*, because although the samples included immature and mature adults (11.3–58.6 cm), the length by sex was not significantly different ($P > 0.05$).

The non-parametric length-to-time regression lines obtained with bandwidths of 1–3 years show the same pattern in the change of the abundances and mean sizes (Figs. 2 and 3). Several features of the curves differ between species. However, since *G. gibberifrons* was caught in very small numbers, the comparison will be limited to *N. rossii* and *N. coriiceps*. Extremes in the mean size curve of *N. rossii* are associated with extremes in its abundance. Such association is not apparent in *N. coriiceps*. The mean size curve of *N. rossii* has two large maxima; no such large maxima are present in *N. coriiceps*. Finally, the variance component due to years in the ANOVA is much larger in *N. rossii* than in *N. coriiceps*.

Beginning with the commencement of our sampling in 1983, the length of *N. rossii* trended slightly upwards until 1986, followed by a downward trend until 1991, upwards to a second maximum in 2000 and then downwards until 2009 (Fig. 3). Since 1986 (no previous data available), the length of *G. gibberifrons* also showed a downward trend until 1991. After this date the trend was upward (Fig. 3). Finally, the length of *N. coriiceps* exhibited local peaks with maxima in 1993 and 2000 and minima in 1988, 1997 and 2006, and, therefore, oscillated around the expected overall mean size, without any discernible trend (Fig. 3).

N. rossii caught between 1982 and 1985 changed in modal age from 2–3 to 6–7 yr (Fig. 4a). The number of *G. gibberifrons* caught was insufficient for a similar treatment of the data. The increase in *N. rossii* length was closely associated with the predominance of older age classes. Later on, when the 1980 age class should have migrated offshore, a bimodal distribution appeared in the pre-recruit population and the annual mean lengths decreased. It should be noted that size–frequency distributions correlated with age indicate that the fish born in 1980 constituted the strongest year class recorded for the species in Potter Cove during the past

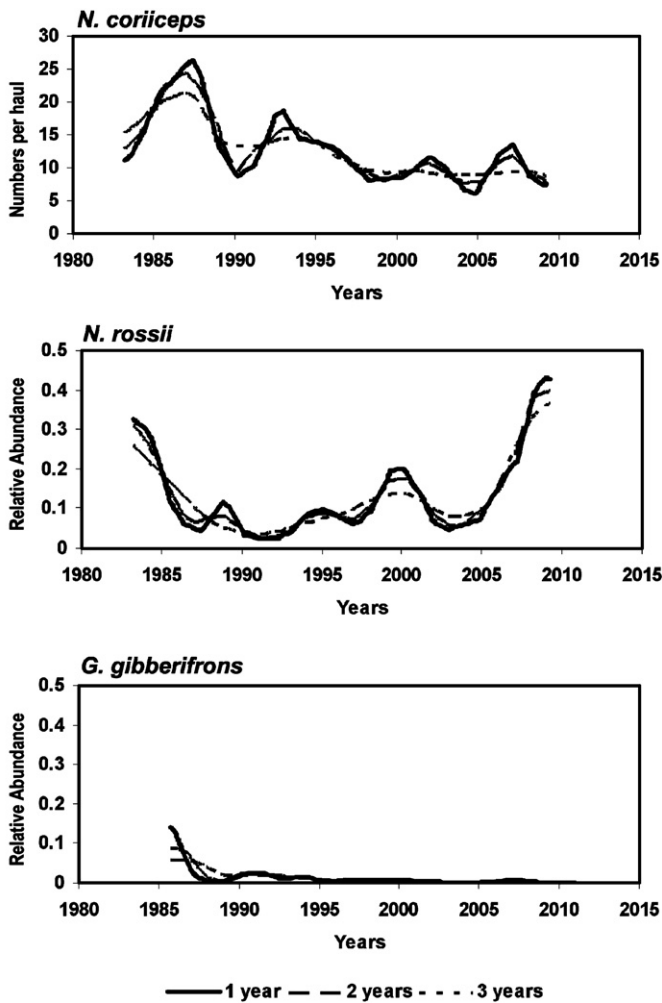


Fig. 2. Comparison of the variation in abundance of *Notothenia coriiceps*, relative abundance of *Notothenia rossii* and relative abundance of *Gobionotothen gibberifrons*, with different bandwidths (1, 2 and 3 years).

few decades (Barrera-Oro and Casaux, 1992), in agreement with the time of heavy commercial fishing in the area.

After 1991, relative abundances (Fig. 2), mean sizes (Fig. 3) and length distributions of *N. rossii* (Fig. 4a) did not depend on a single forcing event (the strong decline in the spawning biomass) but rather on interannual variations in reproductive success, fecundity and age structure of the spawning population.

It is worth noting that after 1991 the relative abundance of *N. rossii* shows oscillations but a discernible increasing trend extending up to 2010 (Fig. 2). On the other hand, the relative abundance of *G. gibberifrons* remains stable at a very low value, if not decreasing.

4. Discussion

The sampling programme conducted at Potter Cove provides results representative of the situation of inshore populations around the South Shetland Islands. Results from other sites like Harmony Cove (Nelson Island), Moon Bay (Livingston Island), Fildes Bay, Admiralty Bay and Discovery Bay over a comparable period are consistent with those at Potter Cove (Barrera-Oro et al., 2000). The results obtained using trammel nets are confirmed by other components of the food web. In Antarctica, shags (*Phalacrocorax* sp.) feed heavily on inshore, demersal fish, as indicated by otoliths in the pellets that they cast daily (Casaux and Barrera-Oro, 1993, 2006; Barrera-Oro and Casaux, 1996b). Not surprisingly, among the prey

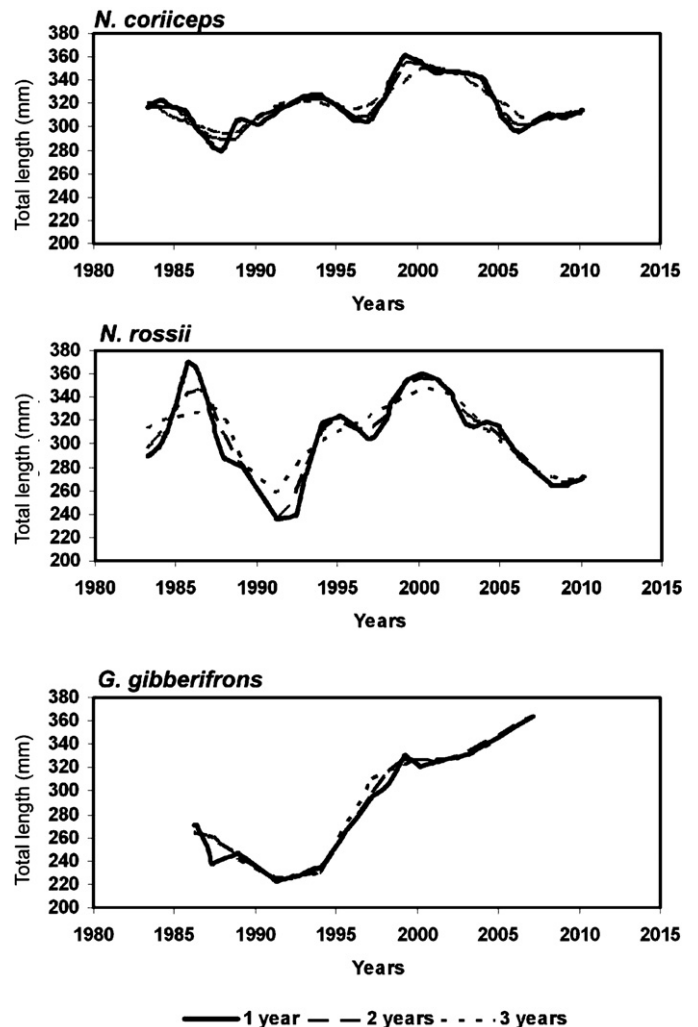


Fig. 3. Comparison of the variation in the annual mean length of *Notothenia coriiceps*, *Notothenia rossii* and *Gobionotothen gibberifrons*, with different bandwidths (1, 2 and 3 years).

species caught with nets inshore at the South Shetlands, only *N. rossii* and *G. gibberifrons* have been absent or scarcely represented in shag's pellets. The absence of otoliths from the two exploited species contrasts with the high occurrence of those from *N. coriiceps*. These results are consistent with the high incidence of *G. gibberifrons* in the diet of shags and in trammel-net catches both at the Danco Coast and the tip of the Antarctic Peninsula (Joinville-D'Urville Islands; Casaux and Barrera-Oro, 2006), reflecting higher availability of this fish in an area remote from the main historical fishing grounds.

In fish such as *N. rossii* and *G. gibberifrons*, which include a juvenile inshore and an adult offshore phase in their life cycles (see Section 1), a sharp decrease in recruitment to inshore waters, leading to a decline in density, should result in an increase in the mean size as the fish grow and are not replaced by smaller specimens. For *N. rossii*, the sharp increase that we observed in mean size between 1983 and 1986 (Fig. 3) and the duration of the species' inshore phase, which in the South Shetland Islands is 6–7 yr (Linkowski and Zukowski, 1980; Barrera-Oro and Casaux, 1992), is consistent with the sharp decline in abundance during 1983–1991 (Fig. 2). We hypothesize this to be a consequence of the depletion of the adult population offshore during earlier years. The fish born in 1980, immediately after the last spawning during the commercial fishing era, left inshore waters in 1986 (at 6–7 yr of age), coincident

Table 1
Results of one-way nested ANOVA performed on total lengths of fish at Potter Cove.

Source	Sum of squares	D.O.F.	Mean square	F	P
(a) <i>N. coriiceps</i>					
Years	6,835,179.4646	28	244,113.50	22.5411	$<1 \times 10^{-12}$
Hauls	12,627,413.5676	1166	10,829.69	4.3611	$<1 \times 10^{-12}$
Error	31,594,302.1942	12,723	2483.24		
(b) <i>N. rossii</i>					
Years	3,747,554	26	144,137	51.3314	$<1 \times 10^{-12}$
Hauls	7,270,817	601	12,098	4.3084	$<1 \times 10^{-12}$
Error	5,624,349	2003	2808		
(c) <i>G. gibberifrons</i>					
Years	98,365.65	12	8197.14	3.42	0.0015
Hauls	113,800.94	39	2917.97	1.22	0.265
Error	100,604.35	42	2395.34		

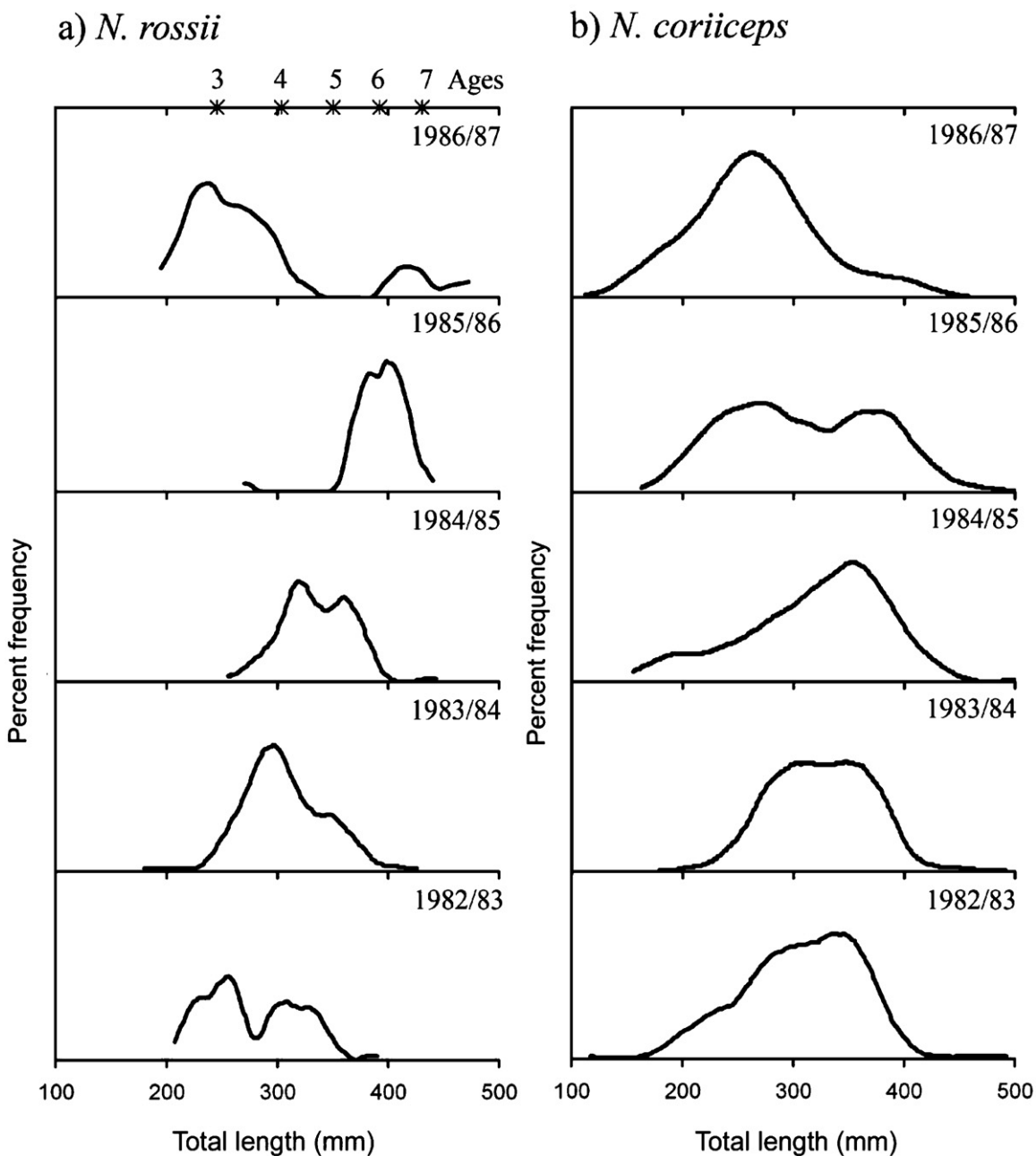


Fig. 4. Density function of *Notothenia rossii* (a) and *N. coriiceps* (b) (Epanenchikov kernel, bandwidth = 0.1) during the first five years sampled (1982/3–1986/7) at Potter Cove.

with the peak in mean length (Fig. 3). In the following years, until 1991, the decreasing abundance is consistent with the entrance of only weak cohorts and the consequent reduction in mean size. Such changes cannot be attributed to an increase in the abundance of *N. coriiceps*, which may have affected only the relative abundance but not the size distributions of the other species.

Our results indicate that from 2000 onwards *N. rossii* showed both a trend in the mean size, indicative of recruitment to the inshore population (Fig. 3), and an increase in the abundance in 2008–2010, suggesting that recruitment might have recovered to values close to those at the start of the monitoring programme (Fig. 2).

The situation of *G. gibberifrons* was more compromised. The few specimens caught inshore in the last decade have been large (>36 cm), indicating lack of significant recruitment (Fig. 3).

The evidence accumulated points to a single event responsible for declines in the populations of *N. rossii* and *G. gibberifrons*. Such an event would be the brief but heavy fishing in the area in the late 1970s to early 1980s, consistent with historical information from bottom trawl surveys. Trawl data show a decline in the commercially exploited stocks in the early 1980s, followed in the end of the 1990s by a slow recovery of *N. rossii* but further decline of *G. gibberifrons* stocks (Jones et al., 2001, 2003; Kock et al., 2002; Barrera-Oro and Marschoff, 2007). A cruise carried out in December 2006–January 2007 found a reproductive concentration of *N. rossii* for the first time since the proposed depletion event (Kock et al., 2007); see Fig. 1. The population of *G. gibberifrons* was found to be still in poor condition, with a low proportion of juveniles (<30 cm) caught offshore, while that of *N. coriiceps* had continued to increase (Kock et al., 2002, 2007). This offshore sampling included the northeast shelf of King George/25 de Mayo Island, very close to our inshore sampling site in the same island, Potter Cove.

Why these decadal trends are apparent in these fish, and especially lack of or slow recovery since the depletion in the 1970s, could be related to four factors: (1) continued illegal fishing; (2) depletion of the pelagic larval and juvenile fish through by-catch in the fishery for Antarctic krill (*Euphausia superba*); (3) depensation, owing to low population size; and/or (4) climate change.

In regard to the first possibility, illegal, unregulated and unreported (IUU) fishing has been significant in the CCAMLR area, particularly until about 2000 (Österblom et al., 2010) and still operates at an uncertain but lower level (CCAMLR, 2011b). Of course, owing to the nature of IUU fishing, we have little knowledge of how common this has been specifically around the South Shetlands. However, IUU fishing targets the commercially more valuable deep water species (toothfish *Dissostichus* spp.) using longlines and gill nets in the Indian and Pacific sectors. It is unlikely that this fishery might have affected the shallower species in the South Shetland Islands.

The second possibility, by-catch of pelagic young stages of notothenioids in the krill fishery is a well recognized event that has been discussed by CCAMLR biologists (CCAMLR, 2006). The krill catch in Area 48 has averaged 182,022 t (SD 124,131 t) between 1980 and 2003, with catches peaking during 1984–1991 when 250,000–400,000 t was taken annually (otherwise ~ 100,000 t; CCAMLR, 2005). However, except for 2009 the fishery within Area 48 has not been concentrated in the 48.1 portion, which includes the waters off the South Shetland Islands (CCAMLR, 2011b). Therefore, we do not think this possibility is a likely explanation of the results we have presented.

The third possibility, depensation or depressing effects owing to low population size (Allee effect), has been detected in a number of heavily fished demersal species, including Atlantic cod (*Gadus morhua*) and Nassau grouper (*Epinephelus striatus*), as reviewed by Longhurst (2010). These fishes, and perhaps demersal species in general, have highly evolved behavioural mechanisms involved in

fertilization of eggs. These mechanisms are compromised at low density. Given the relative health of the unexploited *N. coriiceps*, depensation must be seriously considered as an explanation. To assess this possibility further, much more information is required about the behaviour during spawning of our study species.

In regard to climate change (which becomes more important in species having depressed populations, thus, facilitating depensation; Longhurst, 2010), since the mid-1970s, the Southern Ocean, and especially the Antarctic Peninsula area, has been influenced by a long-term stability of the positive state of the Southern Annular Mode (SAM). This is an atmospheric pressure index representing differences between the Ross Sea and Antarctic Peninsula region. One result has been altered winds and circulation and changes in sea ice extent, including the waters around the South Shetland Islands (Stammerjohn et al., 2008; Thompson and Solomon, 2002; Russell et al., 2006). What specifically changed to possibly affect reproductive success of the study species awaits a time when SAM fluctuates between negative and positive more frequently, as it did in earlier decades. In that way we could correlate fish life history variation to interannual climate (e.g., Belchier and Collins, 2008). Nowadays, SAM just oscillates from being less to more positive.

Finally, probably exacerbating the latter two factors is the fact that this portion of the Southern Ocean has been severely altered, first by almost complete extraction of once unimaginably abundant fish- and krill-eating fur seals (*Arctocephalus* spp.) followed by baleen whales. More recently, fur seals have largely recovered and baleen whales are beginning to do so (Ainley et al., 2010), but climate change increasingly is altering ecological processes, indicated by an invasion of gelatinous zooplankton (salps *Salpa* Thompson; Atkinson et al., 2004) and changes in phytoplankton quality (Montes-Hugo et al., 2009). As a result of reduced prey owing to climate change and recovering competitors, penguin populations are decreasing (Ducklow et al., 2007; Ainley and Blight, 2009; Ainley et al., 2010; Trivelpiece et al., 2010). It is this system, originally implemented over many millennia, that is seeking ecological stability, including recovery of demersal fish. The likelihood that genotypic and phenotypic plasticity has been reduced in these fish owing to severe population reduction (e.g., Allendorf and Hard, 2009) further compromises their recovery potential in such a rapidly changing system.

The exploitation of these demersal fish is now conducted under the CAMLR Convention which, in its article II allows that fishing should not be so intense that a stock cannot recover within two to three decades, a rule established in 1982. The time required for the recovery of these species after an overfishing event, and probably the case for many other Antarctic fish species, largely exceeds these limits, a fact that stresses the relevance of the precautionary approach to the management of fisheries adopted by the CAMLR Commission (Constable et al., 2000; Constable, 2011). Moreover, our findings indicate that what might be deemed precautionary catch limits for one period may not apply in another, and thus very close attention using scientific monitoring, as well as basic life history research, is required to exercise effective management (Longhurst, 2010). Mere precaution may not be enough in the management of the perturbed ecosystem around the South Shetland Islands.

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