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STOCK ASSESSMENT OF SHRIMP *Pandalus borealis* (KRØYER 1838) IN SKJÁLFANDI BAY NORTHERN ICELAND

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ABSTRACT

The shrimp (*Pandalus borealis*) stock in Skjálfandi Bay northern Iceland is assessed in this report. Management and assessment methods of shrimp in Iceland are reviewed and suggestions for improvements are given for Sierra Leone. The analyses are based on time-series data of annual autumn surveys between 1991 and 2007 by the Marine Research Institute in Iceland. Length-frequency analysis was also conducted. Five age classes were identified and their proportions calculated using cohort slicing. Mean lengths corresponding to relative ages were estimated. Growth of *P. borealis* was modelled using non-linear minimisation to find the best fit according to the von Bertalanffy growth curve. Growth parameters estimated were L_∞ =24.28 mm (carapace length), growth k = 0.46 and t_0= -0.42. The dynamic production model from the commercial catches and autumn surveys from1988 to 1999 gave an average biomass of 2047 tons. Maximum Sustainable Yield per recruit was 1.42 g which corrresponds to F_{max} = 1.97. Maximum Sustainable Yield = 566 tons and $F_{0.1}$ = 0.72. The dynamic production model has advantages over the surplus production models generally used in Sierra Leone. A review of this model showed that it could be used to assess penaeid shrimps in Sierra Leone provided that scientific surveys are done which give a more reliable estimate of the abundance index than catch from commercial vessels. Data should be collected monthly or seasonally and because it is very expensive to conduct surveys, seasonal surveys should be done. Periodic collection of data, relating to selectivity, maturity, length and weight are a prerequisite for proper assessment of the shrimps of Sierra Leone. The economy of Iceland is to a large extent dependent on cod and the current management strategy is to enhance the cod stocks. This does not favour shrimp. Perhaps it is time to revise the current management strategy.

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1 INTRODUCTION

The pandalid shrimp of Iceland has been of considerable scientific interest because of its commercial value in the past as well as its ecological interaction with fish species including cod and haddock. In 1998, the value of shrimp (*Pandalus borealis)* caught by vessels fishing in Icelandic waters was about USD 66 million, the second highest fisheries resource in terms of value after cod (Hagstofa Islands 1999).

Shrimp fishing in Iceland was experimented for the first time around 1924 in Ísafjarðardjúp, a fjord in the northwest peninsula of the country (Hallgrímsson 1993, cited by Garcia 2007). Commercial fishery started more than ten years later in 1935, when land based processing plants had been built (Garcia 2007). This probably led to the extension to other inshore fishing grounds, including Arnarfjörður, three years later, and to Húnaflói in northern Iceland (Hallgrímsson and Skúladóttir 1981). The advent of shrimp peeling machines in 1959 as well as bigger and better fishing vessels in the late 1960s resulted in increasing fishing effort. Shrimp fishing was profitable and this resulted in further expansion of these fisheries to other inshore areas around Iceland (Skúladóttir 1981. Catches between 2500 and 7800 tons were recorded during the period 1969-1984. Landings from the inshore fisheries were estimated at around 12,000 tons in 1996 but were on average less than 8,000 tons in most years from 1973 to 2000 (Skúladóttir and Sigurjónsson 2003). During the present 2008/2009 fishing season only 122 tons of shrimp have been landed so far (Fiskistofa 2009) out of a quota of 500 tons allocated to Arnarfjörður.

The low TAC for inshore shrimp is because of a collapse that occurred in recent years, and according to the Icelandic Marine Research Institute (MRI), is due to increased predation by cod (MRI 2008). That said, it should be noted that other shrimp fisheries in the world have been severely affected by fishing pressure including perturbations caused by otter trawls on the bottom substrate in which shrimp mainly burrow (Jennings and Kaiser 1998).

Unlike Iceland, the shrimps of Sierra Leone are of the family Penaeidae. Penaeidae are generally larger than *Pandalus*. The genus *Panaeus* occurs in tropical and subtropical waters between the latitudes 40°N and 40°S. Adult shrimp rarely occur below 180 m (Bailey-Brock and Moss 1992). They are laterally compressed, with the head containing a prominent rostrum. Shrimps belonging to family Penaeidae are dioecious, unlike *Pandalus* which are protandric hermaphrodites. The male *Panaeus* is identified by the two pairs of modified appendages on the first and second abdominal segments (the petasma and appendix masculine) both of which are used in the transfer of sperm to the female. In females there is a sperm receptacle (thelycum) located between the base of the fifth walking legs (Bailey-Brock and Moss 1992). The most commonly exploited shrimp species are *Penaeus kerathurus* (tiger shrimp), *Parapenaeus longirostris* (deep water rose shrimp), *Penaeus notialis* (pink shrimp) and *Parapenaepisis atlantica* (white shrimp). Of the 80 commercially exploited finfish, molluscs and crustaceans in Sierra Leone, shrimps have on average contributed a large percentage to the total income from the fisheries resources in recent years, just as it did for Iceland in the past. The estimated value for shrimp exported from Sierra Leone in 2006 was around USD 8,000,000 which was 56.3% of the total value of fisheries exports in that year (Anon. 2008).

Shrimp stocks in Sierra Leone are mainly exploited by Chinese owned vessels with the primary aim of exporting the catch. Recent estimates by the Statistics Unit of the Ministry of Fisheries and Marine Resources using the Fox version of a surplus yield model, gave a potential yield of 1555 mt for 1996-2005 and F_{msy} of 4950 fishing days (Anon. 2008). Landings during this period were on average about 1590 mt. The estimated potential yield can be misleading due to the fact that the effort was based on fishing days rather than standardised effort from the commercial fishing vessels which have different gross registered tonnage (GRT) and whose catch statistics were used for the estimate. It is worth noting that there is sometimes misreporting of catch statistics.

Around the region of West Africa including the waters of Sierra Leone, illegal unreported and unregulated (IUU) fishing practices are said to be high. However, the extent of IUU fishing and is unknown, posing a problem for stock assessment.

Commercial fisheries for penaeid shrimps have a long history along the coasts of many sub-tropical and tropical countries, often leading to over-exploitation (Garcia and LeReste 1981). The advent of specialised industrial fishery in the 1950s, brought about an increase in the exploitation of shrimp (Leite and Petrere 2006) and by the end of the 1970s, there was an increased demand for shrimp particularly in high value markets like the USA, Europe and Japan (Leite and Petrere 2006) resulting in an annual production to the tune of 700 thousand tons by the early 1980s (Isaac *et al*. 1992, cited by Leite and Petrere 2006).

The expansion of capture fisheries and available (but often inadequate) information on the biology of penaeid shrimps, have led to the conclusion that most stocks are either fully exploited or over-exploited (Leite and Petrere 2006).

Iceland has experienced a situation where inshore pandalid shrimp stocks have collapsed and fishing in these areas has been stopped, including Skjálfandi Bay. The collapses have been associated with cod predation on shrimp rather than the effect of fishing. However, shrimp fishing has been allowed again in the inshore area of Snæfellsnes and in the inshore area of Arnafjörður, a TAC of 150 tons was allocated in the 2007-2008 fishing year and 500 tons for the 2008-2009 fishing year. The MRI continues to monitor the stocks by bi-annual independent surveys in spring and autumn. The Marine Research Institute has also encouraged UNU-FTP fellows, for example Ceesay (2000) and Lam Anh (2007), to use data from the inshore stocks with a view to estimate the biomass and predict stock development. The effect of predation of cod and haddock on shrimp cannot be overemphasised. The economy of Iceland is to a large extent dependent on cod and the current management strategy is to enhance the cod stocks. This does not favour shrimp. Perhaps it is time to revise the current management strategy.

Sierra Leone has never documented a collapse in the shrimp fisheries but it is worth gaining knowledge from Iceland and to see if the monitoring, stock assessment and management measures might be adapted to the situation in Sierra Leone.

The European Union is funding a three year project on Institutional Support for Fisheries Management, ISFM, and part of the project is to estimate the biomass of the fisheries resources and exploitation patterns. For a rational exploitation of any fisheries resources, the knowledge of its population dynamics is very important. This study assesses the stock of *Pandalus borealis* in Skjálfandi Bay (Figure 1) by using a dynamic production model.

1.1 Objective

The main objective of this study is to access the status of the stock of *Pandalus borealis* in Skjálfandi Bay north of Iceland, using a dynamic production model.

1.1.1 Specific goals

- To estimate the trend in biomass of *P. borealis*
- To estimate growth parameters of *P. borealis*
- To estimate equilibrium yield potential and spawner stock biomass
- To compare the shrimp stock assessment and management system of Iceland with that of Sierra Leone and suggest corrective measures where necessary.
- To evaluate the appropriateness of using a dynamic production model for stock assessment in the shrimp fisheries in Sierra Leone, considering the differences in stock identification, and the life history, biology and behaviour of penaeid shrimp to that of inshore stocks of *P. borealis* in Iceland, taking also into account the availability and resolution of data.

2 LITERATURE REVIEW

2.1 The study area

Skjálfandi Bay in the northeast of Iceland is 10 km wide at the inner part and about 51 km between the capes of Gjögutá and Tjörnestá (Figure 1, Gíslason 2004). The maximum depth is about 220 m and the bay is approximately 25 km long. Bottom sediments on the slope are mainly sand or gravel, while the rest of the sea bed is almost entirely covered by silt. River Skjálfandafljót is a glacial river which carries a lot of fine clay on the western side and River Laxá in Aðaldal, which is mainly a spring fed river, is on the eastern side, but carries a lot of volcanic black sand from the barren interior of the country. These two large rivers not only influence the substrate but also result in low salinity in the bay (Gíslason 2004). Warm geothermal water from the nearby lake Mývatn (Ólafsson 1999) regulates the water temperature in Laxá in Aðaldal resulting in increased surface temperature in the bay, particularly at the river mouth.

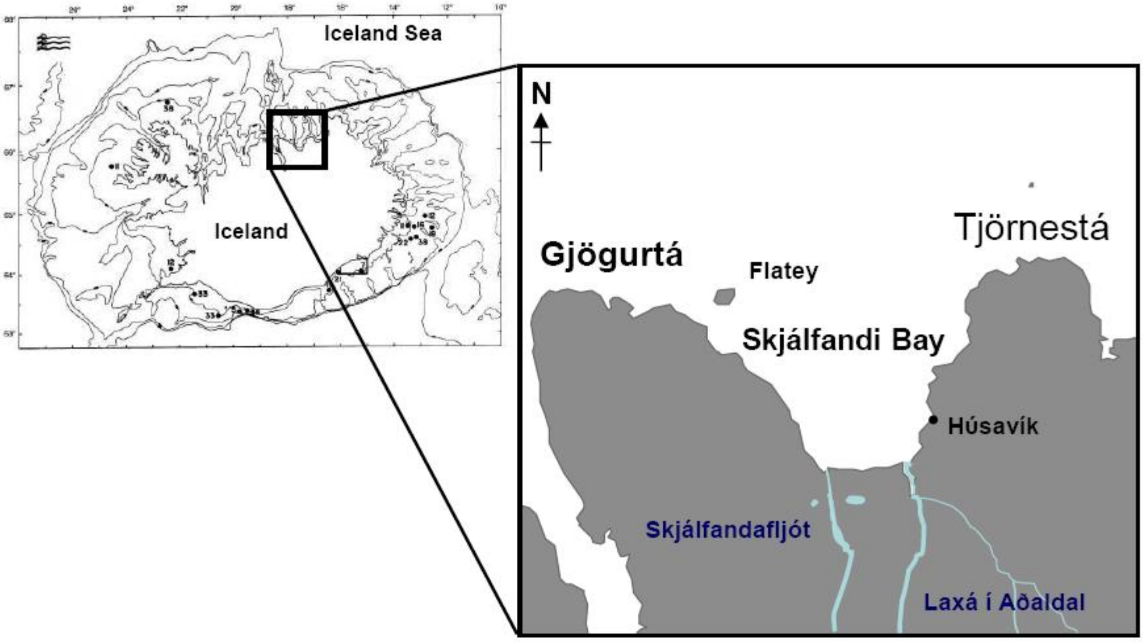

Figure 1: Map showing study area, Skjálfandi Bay northeast of Iceland. (Adapted from Jónsson 1996).

2.2 The ecology of *Pandalus borealis*

2.2.1 Geographical distribution

The shrimp *Pandalus borealis*, known in Icelandic as stóri kampalampi or rækja, is also called the northern shrimp, northern red shrimp, cold-water shrimp, deep-water shrimp or pink shrimp.

It has a wide geographical distribution, spanning from the northern parts of the North Atlantic to the northern Pacific Ocean. *P. borealis* inhabits many waters in these regions as reviewed by Garcia (2007). It is, however, endemic to the sub-arctic region between 35° N and 82° N (Hørsted and Smidt 1956, Bergström 2000). Its distribution has been recorded clockwise from the southwest and on to the east coast of Iceland at depths ranging between 50-700 m but mostly in offshore areas below 300 m (Skúladóttir and Sigurjónsson 2003). *P. borealis* also inhabits inshore bays and fjords where they are caught in shallower waters.

Larvae, juveniles and adults show different distribution patterns. Larger individuals inhabit deeper waters whilst larvae and juveniles are found in shallower waters or near shore, a trend that has also been observed in several fish species. This pattern could be related to temperature, which is relatively more stable as depth increases, and the bigger shrimps prefer such environment. Younger shrimps have a greater tolerance for temperature and salinity fluctuations than adults (Hørsted and Smidt 1956, Aschan 2000a). Seasonal variations in depth distribution have been reported and this could be ascribed to temperature changes. Vertical migration has also been reported in shrimp and this is attributed to feeding, moving up into the water column at night and returning to the bottom substratum in the morning (Shumway *et al*. 1985, Bergström 2000). Shrimp exhibit this vertical migration probably to avoid predators. Depth at which the species is most abundant varies with latitude; being most abundant in much deeper waters at higher latitudes (Shumway *et al*. 1985).

2.2.2 Environmental requirements

Just like for many other marine species, the nature of the substratum, temperature and salinity are factors that have an effect on the distribution of the northern shrimp (Shumway *et al*. 1985). Northern shrimp can tolerate temperatures between -1.6 ^{o}C and +8 ^{o}C (Bergström 2000) but occur mostly at 0 °C. Development of *P. borealis* at all stages in the life cycle increases with temperature, causing shrimp in warmer waters to mature at a younger age and of a smaller size than in colder waters (Nilssen and Hopkins 1991, Skúladóttir and Pétursson 1999).

Northern shrimp prefers salinity between 33-35‰ although it has been reported at a salinity as low as 23.4‰ in some areas (Garcia 2007). Salinity and temperature influence the circulation patterns of water masses in the oceans. The circulation patterns over time influence primary productivity and subsequently the biodiversity of marine ecosystems (Hunt and Drinkwater 2005). In the late 1990s, a warm saline Atlantic water mass north of Iceland (Gudmundsson 1998) resulted in the upwelling of nutrient rich bottom waters and increase in primary productivity. Increased temperature also led to the migration of cod to the northern Icelandic waters away from the inshore areas thus resulting in decreased predation and probably to increased growth and recruitment of shrimp.

The species is reported to inhabit areas where the bottom sediments are mainly soft, muddy or sandy (Hørsted and Smidt 1956, Shumway *et at*. 1985, Bergström 2000), and ocean bottoms with sediments rich in particulate organic carbon (POC), Bergström (2000).

2.2.3 Feeding and growth

P. borealis is a catholic feeder. It filter-feeds as it moves through the water column during night (Wollebæk 1903, Hørsted and Smidt 1956, cited by Garcia 2007) devouring zooplankton. It is known to feed on small benthic invertebrates including polychaetes, whilst foraging or even scavenging on the bottom sediments (Shumway *et al.* 1985).

Offshore, males of *P. borealis* grow throughout the year (Garcia 2007). However, a slower growth rate is observed in winter and this is attributed to food limitation. Males grow throughout the year, although more slowly in winter. Mature females grow during summer and sometimes also every second winter in offshore populations. This is because egg bearing occurs every second year (Skúladóttir *et al.* 1991). In the inshore waters, *P. borealis* may grow from spring to autumn (Garcia 2007), with no growth occurring in winter (Nilssen and Hopkins 1991). Growth rate is therefore faster in warmer waters and slower in colder waters. It was, however, noted by Nilssen and Hopkins (1991) that changes in water temperature, density of the population and recruitment have the potential to trigger differences in growth rates and the length at which *P. borealis* change sex. These effects were observed off the northeast coast of Iceland where shrimp populations found in the area containing north Icelandic winter water with temperatures of 2-3 °C and Arctic bottom water below 0 °C lying under warmer upper layers, spawn for the first time at 6 years and reach L_{50} (carapace length at which 50% of the females are mature) at 23.0-24.9 mm (Garcia 2007). However, in Vestfirðir and Snæfellsnes, west coast of Iceland which received warmer Atlantic water at a temperature of 7 °C, shrimp spawn at age 3 and L_{50} of 17.6-19.8 mm.

Although *P. borealis* is difficult to age due to the fact that shrimp have no hard parts such as otholits and scales, age and growth estimates have been done by many authors (Skúladóttir 1981, Shumway *et al*. 1985, Bergström 2000). It is also known that growth of the species can be modelled using the von Bertalanffy growth equation (Bergström 2000). Length at age of *P. borealis* in Húnaflói, an inshore fjord in Iceland, varies from 12.1 mm carapace length for a 2 year old to 22.1 mm for the 6 year plus group (Skúladóttir, pers. comm.)

2.2.4 Reproductive strategy

In Icelandic waters, *P. borealis* are males when they first reproduce (Berkeley 1930) but later change sex and become female for the rest of their lives (Skúladottir and Pétursson 1999). This is a distinctive feature of pandalid shrimps, but there are reports from the Gulf of Maine and Sweden that some individuals develop into females from birth (Bergström 2000). According to Rasmussen (1953), sex change in the northern shrimp is size dependent but does not depend on age alone. Age at sex change varies from place to place. In Iceland, the inshore shrimp change sex when they are about 3ö5 years old. Sex change varies with location and it is also influenced by temperature (Appollonio *et al*. 1986).

Fertilisation in *Pandalus borealis* is external and the ovigerous period in offshore populations where temperatures near the bottom are around 0-1.2 ^{o}C is 10 months. The warmer inshore populations have a shorter egg carrying period of 5 months. The inshore stocks spawn once a year during summer or early autumn in most places but when temperature is low as in the offshore populations, spawning takes place only once every 2 years (Skúladóttir *et al*. 1991a). Fecundity of the species correlates positively with size and temperature, larger females carrying more eggs range from 600-5000 but mostly around 2000 and regions with relatively low temperatures have been reported to lower fecundity in *P. borealis* (Nilssen and Hopkins 1991, Bergström 1997).

The life cycle of *P. borealis* is reviewed in Bergström 2000 and Garcia (2007). After hatching, the pelagic larvae go through six developmental stages before reaching the post-larval stage. Post-larvae moult six times and finally metamorphose into young shrimps. The species become sexually mature after another seven stages including the juvenile stage. On average the species can live from 3- 8.5 years (Shumway *et al*. 1985).

2.2.5 Mortality

Mortality of *P. borealis* is influenced to a great extent by cod which is one of its many predators. Other predators include haddock, Greenland halibut, redfish and seabirds. Total mortality coefficient, Z, is estimated to be between 0.5-2.0 (Garcia 2007), highest in areas where fishing is intense.

Parasitic infestation is also a contributing factor to mortality in *P. borealis*. The parasite *Phryxus abdominalis*, an isopod, is found on the tail of samples from Greenland (Hørsted and Smidt 1956) and may also attack tissues of the muscle.

2.3 Management of the shrimp fishery in Iceland

Inshore shrimp fisheries are made up of nine separate populations inhabiting six fjords and near-shore areas in Icelandic waters (Skúladóttir and Pétursson 1999, cited by Ceesay 2000). The inshore stocks in the various fjords of Iceland are managed as separate units. Management regulations for inshore and offshore shrimp stocks of Iceland include obtaining a fishing license, minimum square mesh size in cod end of 36 mm, minimum landing size with carapace length, CL>15 mm in the inshore and CL>13 mm for offshore stocks, and a quota system based on Total Allowable Catch (TAC). Less than 30% of the catch can be below the stipulated minimum size. Keeping logbooks and recording of catches is mandatory by law. The TAC was first implemented in the period between 1962 and 1967, but without specifying which vessel should fish in which area. A new TAC system was introduced in 1974 which allocates catch to vessels, inshore areas and restrictions of landings by areas. No one was permitted to land outside the fjord in which they were permitted to fish. TACs for each fjord are set by the MRI, based on independent assessment and divided among fishing vessels by the Fisheries Directorate of Iceland. Bi-annual inshore trawl surveys have been carried out by the MRI since 1973, in late winter (February), sometimes referred to as the spring survey and again in the autumn (September/October). The surveys were standardised in 1988 using fixed trawling stations, gear and towing speed. MRI recommends a harvesting strategy which is based on the biomass indices from the two most recent surveys and long term trend in catches with a view to secure a long-term optimum catch thereby sustainably exploiting the resource.

The management of shrimp stocks has evolved over the years. For example in 1962, the minimum mesh size was 32 mm and in 1995 a 22 mm Nordmøre sorting grid was included in the regulations for the management of shrimps of inshore stocks at Eldey, Skjálfandi Bay, and around the Snæfellsnes peninsula in western Iceland (Skúladóttir and Sigurjónsson 2003). This was, however, not mandatory. Individual Transferable Quota (ITQ) came into existence in 1990, with the limitation that quotas can only be transferred to vessels registered to fish in the same area.

In addition to the above management strategies, “a strict by-catch” regulation is implemented which limits the number of juvenile fish, particularly cod and haddock which can be caught as by-catch. During autumn there is usually an influx of juvenile fish into the inshore to the extent that fishing can be closed until survey results show their disappearance from the fishery (Skúladóttir, pers. comm.).

3 MATERIAL AND METHODS

3.1 Data

The Marine Research Institute inshore research survey database was used in this study. The data covers Icelandic inshore waters from 1973 to the present, which has been collected by researchers at the MRI during biannual spring and autumn surveys of the in inshore shrimp stocks. Catch data from commercial fisheries was also used. The inshore shrimp trawl data include carapace length-frequency by sex, annual catches in weight and numbers, catch and other information on duration of tows, gear used, mesh size, vertical and horizontal opening of the gear, bottom temperature and depth of each survey area.

Carapace length was measured from the posterior margin of the orbit to the posterodorsal margin of the carapace (Bergström 2000, p.186). No direct age determination is carried out for *P. borealis* during the surveys since the species lack otoliths and hard parts. For the purpose of this study, autumn survey data from 1991–2007 in Skjálfandi Bay is used.

3.2 Data analyses

3.2.1 Length-frequency analysis and cohort slicing

Length-frequency distributions were plotted by year and the most distinct modes identified. These modes were assumed to be mean lengths corresponding to relative ages. The data were then pooled for the entire period of study and the length-frequency distribution plotted. Cohorts in the combined length-frequency were sliced using the method of McDonald and Pitcher (1979), which age-disaggregates length distributions. This was done by using functions in the statistical software, R (R 2.8.1). The mean lengths observed from the length-frequency plotted by year were then used in some functions written in R (Stefansson and Taylor 2008) to estimate the true mean-length of the different cohorts together with their standard deviations. The assumption is that each of the combinations of cohorts length distributions are in the form of a Gaussian distribution (Stefansson and Taylor 2008) with mean length, μ and variance, σ^2 in the form:

$$f\ \ x = \frac{1}{\sqrt{2\pi\sigma}} e^{-(x-\mu)^2/(2\sigma^2)} \text{.....................................}(1)$$

and the cumulative distribution was

$$F(x) = \int_{-\infty}^{x} \phi(t)dt = \Phi(\frac{x-\mu}{\sigma}) \quad \text{.....................................}(2)$$

Now assuming that a fixed age group is distributed along the length axis according to the Gaussian density, with mean length, (μ_a) and standard deviation, (σ_a.). The proportion of individuals in a length category l whose width = 1 is given by:

$$\Phi\left(\frac{(l+\frac{1}{2})-\mu_a}{\sigma_a}\right)-\Phi\left(\frac{(l-\frac{1}{2})-\mu_a}{\sigma_a}\right)\text{..}(3)$$

since this is a probability of shrimp having length between $(l+\frac{1}{2})$ and $(l-\frac{1}{2})$.

Suppose the true proportion of shrimp in age group a is π_a then the proportion of shrimp in the length group l, across the entire ages in the length distribution is:

$$\sum_a \pi_a \left\{\Phi\left(\frac{(l+\frac{1}{2})-\mu_a}{\sigma_a}\right)-\left(\frac{(l-\frac{1}{2})-\mu_a}{\sigma_a}\right)\right\}\text{......................... }(4)$$

From the above, we have to estimate the proportions in each age group,π_a the mean length at age, μ_a. and the standard deviation σ_a. From the length-frequency plotted, some mean lengths at age were estimated and a guess was made for a fixed standard deviation corresponding to the mean lengths. Having fixed these values, the proportions for each mean length were estimated using an R command (Stefansson and Taylor 2008). A number of runs were done in the R command first with fixed values, then by changing the values or using the estimated proportions from the model runs, predicted proportions at length were obtained. The predicted length distribution is given by:

$$\hat{y}l=\sum_a \pi_a \left\{\Phi\left(\frac{(l+\frac{1}{2})\mu_a}{\sigma_a}\right)-\Phi\left(\frac{(l-\frac{1}{2})-\mu_a}{\sigma_a}\right)\right\}\text{..................................}(5)$$

and to estimate the unknown parameters, π_a , μ_a and σ_a that gives the best fit, the sums of squares of the observed proportion at length and the theoretical proportions is calculated by:

$$(yl-\hat{y}l)^2 \text{ ....}(6)$$

where yl is the measured, (observed) proportion and $\hat{y}l$ the modelled proportion. This minimises the discrepancy between the observed and the theoretical values.

3.2.2 *Estimation of growth parameters*

Once the mean length at age of each cohort and the standard deviations have been estimated, then the average growth rate can be modelled. The von Bertalanffy growth equation (von Bertalanffy 1938) was applied to the estimated mean length-at-age and the standard deviation data by the non-linear minimisation function in R.

The von Bertalanffy model has been used extensively to describe the average growth of finfish (Dickie 1968) and shrimp (Bergström 1992a) and is often used in fish population dynamics. The von Bertalanffy growth model assumes that growth is faster when the fish and species is young, slows as it matures and levels off at a later age. It is assumed that the growth of *P. borealis* in the Skjálfandi Bay is well described by the von Bertalanffy growth function:

$L_t = L_\infty(1 - e^{-(k(t-t_0))})$ ....(7)

Where L_t is the average length-at-age t, L_∞ is the asymptotic length, *k* is the growth velocity parameter and t_0 is hypothetical age at length equal to 0.

A function to estimate growth parameters was written in R and by non-linear minimisation, estimates of the least sums of squares were obtained. Initial values of L_∞, k and t_0 were set and the least sums of squared error, SSE, was minimised using an iterative process. The estimated growth parameters which gave the minimum SSE were taken as that of *P. borealis* in the study area and for the period of study.

To verify the gross accuracy of the growth parameters, the growth performance index (Phi prime value, $\boldsymbol{ø' = \log_{10}k + 2\log_{10} L_\infty}$ (Pauly 1979, Pauly and Munro 1984) was used to compare the results with those from previous studies.

3.2.3 *Length-weight relationship*

The relationship between length and weight of fish and shellfish including shrimps is expressed as:

$W=aL^b$..(8)

Where W, is the weight in grams, L is the carapace length in mm, and a and b are parameters.

Equation (8) above was log-transformed.

$Ln(W)=Ln(a)+ b*Ln(L)$...(9)

and a simple linear regression applied to get the a (intercept) and b (slope) values.

The length-weight relationship was computed separately for shrimp without sternal spines (mature females) and those with sternal spines (immature females, males and transitional). The length-weight relationship was then used to convert length at age to weight at age.

3.2.4 Selectivity

A selectivity ogive was used to estimate the selectivity for different length classes of *P. borealis* from Skjálfandi Bay. The selection pattern was based on the result of a 36 mm (cod-end) mesh size (Skúladóttir pers. comm.). The results were obtained from the MRI.

3.3 Dynamic production model

The von Bertalanffy growth parameters were used to convert length of shrimps to age in order to run an age-based dynamic model. The basis for this age-structured stock assessment model, as it is sometimes called, is to establish a population dynamics model and then see how the model's predictions of annual catches in this case, relate to the observed data. The model incorporates recruitment, natural mortality and weighting correction factors which are explained below.

3.3.1 *The number model*

The stock size of shrimp for each year was estimated based on the number model:

$$N_{y+1} = N_y + R_y - C_y \quad (10)$$

Where R_y represents the recruitment in year y, N_y is the number of shrimp that survive in the year y, and C_y is the catch in the same year.

The biomass is then:

$$B_y = \sum (N_{ay} * w_a) \quad (11)$$

When modelling a population of numbers of shrimp with true ages 1,......, A may use "age" A+1 as a "plus group". As a result of mortality, the plus group is reduced each year, but also a new age group enters the plus group and this is represented by equation (12)

$$N_{A+1,y+1} = (N_{A,y} + N_{A+1,y})e^{-Z_A} \quad (12)$$

3.3.2 *Recruitment*

The spawning stock per recruit in the fishery will be estimated by using the Beverton-Holt equation:

$$R = \frac{\alpha s}{(1 + \frac{s}{k})} \quad (13)$$

Where R = recruitment

S = spawning stock

α = coefficient used as a multiplier for prospective recruitment

K = size of the spawning stock that produces half maximum recruitment

3.3.3 *Natural mortality*

Natural mortality, *M,* is also needed as one of the inputs in the model. Natural mortality was not estimated because of lack of data and the popular Pauly's empirical formula (Pauly 1979 applies only to fish and not crustaceans. Natural mortality was therefore assumed to be 0.5. This value is usually used in the literature (Fu and Quinn II 2000, Skúladóttir *et al.* 2004).

3.3.4 *Fitting the model*

The model is fitted by first using initial parameters stated above and followed by defining the functions in R which utilises a non-linear minimisation. The model is evaluated by comparing the model output to the data and this is done by computing the sums of squares between each data set and the fitted values they correspond to (Stefansson 2007).

Coefficients of variation, CVs for different parameters are used as weighting factors assigned to the sums of squares which are then minimised in order to estimate the parameters in the model.

Assuming that all terms are log transformed data. Each term is then of the form:

$$\lambda \sum_t (\ln(x_t) - \ln(\hat{x}_t))^2$$

Where x_t's were annual landings, biomass index, recruitment factor, fishing mortality and λ is the weighting factor.

Statistically, the "correct" weighting factor is the inverse of the variance,

$$\lambda = \frac{1}{\sigma^2_{\ln(x_t)}}$$

But in the case of a low dispersion, the standard deviations of the log-transformed quantities and the coefficient of variation of the original numbers are similar. Thus CV(x) is used as $\sigma_{\ln(x_t)}$.

The CV for catch, CV(Y) was assumed to equal 0.1 since catch data are quite precise. The CV(Y) used here thus assumes that 95% of annual catch estimates lie within 20% (two standard deviations) of their true values (we assume here again that the errors are symmetrical).

The coefficient of variation for biomass index, CV(I), was set to 0.1, coefficient of variation of fishing mortality, CV(F), was set to 0.4 and coefficient of variation of recruitment, CV(R), was set to 1000. During the iteration process, these values were allowed to deviate freely until better CVs were obtained to fit the model. There is, however, no general consensus as to how to fit these CVs (Stefansson pers. comm.). The model is fitted by first using the initial parameters stated above, followed by defining the functions in R which utilises a non-linear minimisation. The model is evaluated by comparing the model output to the data and this is done by computing the sums of squares between each data set and the fitted values they correspond to.

Coefficients of variation, CVs for different parameters are used as weighting factors assigned to the sums of squares which will then be minimised in order to estimate the parameters in the model (Stefansson 2007).

3.4 Yield per recruit and spawner stock per recruit

Yield per recruit and spawner stock per recruit were estimated for different fishing mortalities ranging from 0 to 2. The spawner stock biomass was estimated for 3 year old shrimps. This is when they change sex in Skjálfandi Bay (Skúladóttir and Pétursson 1999). The estimation of yield potential from a stock is done in two stages by first estimating the number of recruits and then calculating the yield per recruit, which depends on the size (age) at recruitment and the fishing mortality. These calculations are used to estimate how much of a particular year class could be caught for sustainable utilisation. The catch in numbers of shrimp at age *a* according to Stefánsson (2008b) is given by:

$$C_a = \frac{F_a}{Z_a}(1-e^{-Z_a})e^{-\sum_{a'<a} Z_{a'}} R \text{ ....} (14)$$

where C_a is catch at age, F_a is fishing mortality, Z_a is total mortality and R is recruitment and the catch in weight is given by:

$$Y_a = \frac{F_a}{Z_a}(1-e^{-Z_a})w_a e^{-\sum_{a'<a} Z_{a'}} R \text{ ....} (15)$$

then total yield of the year-class then sums up to:

$$Y = \sum_a Y_a = \frac{F_a}{Z_a}(1-e^{-Z_a})w_a e^{-\sum_{a'<a} Z_{a'}} R \text{ ....} (16)$$

This gives a yield per recruit as:

$$\frac{Y}{R} = \sum \frac{F_a}{Z_a}(1-e^{-Z_a})w_a e^{-\sum_{a'<a} Z_{a'}} \text{ ....} (17)$$

The yield per recruit was calculated for fishing mortalities, *F*, ranging from 0 to 2.0 to find a value of *F* that will provide a high yield to the fishery, while at the same time keep the stock size reasonably high. A yield per recruit curve was then plotted and the fishing mortality F_{max}, which corresponds to maximum yield per recruit, was noted. From the same yield per recruit curve, the fishing mortality at which the slope is one-tenth of the slope at the origin is $F_{0.1}$. This indicates a fishing mortality which is lower than that required to obtain the maximum yield and will result in economic optimisation of the fisheries and prevent recruitment overfishing (Gulland 1983). The model though theoretical, is used to give plausible indications as to whether a heavy or light effort gives a rational exploitation or if it is better to catch young or old shrimp. The assumption in the yield per recruit model is that recruitment is constant, and that the age structure of the population is the same as if a single cohort is followed through time. Hence a measure of yield "per recruit" (Beverton and Holt 1957).

The calculated yield per recruit for different fishing mortalities was then multiplied by the assumed virgin stock to give an estimate of yield.

4 RESULTS

4.1 Length-frequency distributions and mean length at age

Carapace length-frequency distributions plotted for each year did give some modes but only a few (Figures 2 and 3). However, a plot of the combined carapace length-frequency distributions plotted length-frequency distribution of *P. borealis* in autumn surveys in Skjálfandi Bay 1989 – 2007 (Figure 4), revealed five distinct peaks corresponding to discrete age groups when cohort slicing was done. The mean length at age estimates ranged from 11.79 mm (CL) for shrimps that are 1 year old, to 22.04 mm (CL) for those at age 5 (Table 1). Proportions corresponding to relative ages reveal that the 2 year olds were abundant and proportions of 5 year olds in the stock were small.

Table 1: Mean carapace length (mm) of *P. borealis* estimated from the autumn survey data 1989-2007 and estimated average proportion of different year classes, using the cohort slicing method.

Age	Mean carapace length (mm)	Standard deviation	Proportion
1	11.76	1.34	0.346
2	15.44	1.32	0.368
3	19.63	1.38	0.177
4	21.27	2.19	0.108
5	22.04	2.19	0.001

4.2 Length-weight relationships

Length- weight relationships obtained for *P. borealis* in Skjálfandi Bay are given below:
Shrimps with sternal spine: $W=0.000933*L^{2.875}$
Shrimps without sternal spine: $W=0.000887*L^{2.960}$

4.3 Von Bertalanfy growth parameters

Mean lengths were plotted against relative age together with a Von Bertalanffy growth curve (Figure 5). The average growth parameter estimated for *P. borealis* in Skjálfandi are asymptotic carapace length CL_{∞} =24.28 mm, growth velocity parameter, k = 0.46 and t_0= -0.42 with SSE = 0.47. The estimates of the asymptotic carapace length as well as the growth velocity parameters estimated for this area are similar to those estimated from previous studies on *P. borealis* elsewhere (Table 2).

Table 2: Comparison of growth parameter estimates from previous studies with that obtained from this study using Phi prime (Pauly 1979, Pauly and Munro 1984).

Study area	**L∞(mm)**	**K (yr^{-1})**	**ø'**	Reference
Skjálfandi Bay	24.28	0.46	2.43	This study
Húnaflói	24.9	0.29	2.25	Lam Anh (2007)**
Flemish Cap	37.6	0.14	2.30	Skúladóttir *et al*. (2007)
Kachemak Bay	24.6	0.37	2.35	Fu and Quinn II (2000)
Jan Mayen	33.2	0.19	2.32	Ashcan *et al.* (1996)

** Unpublished report

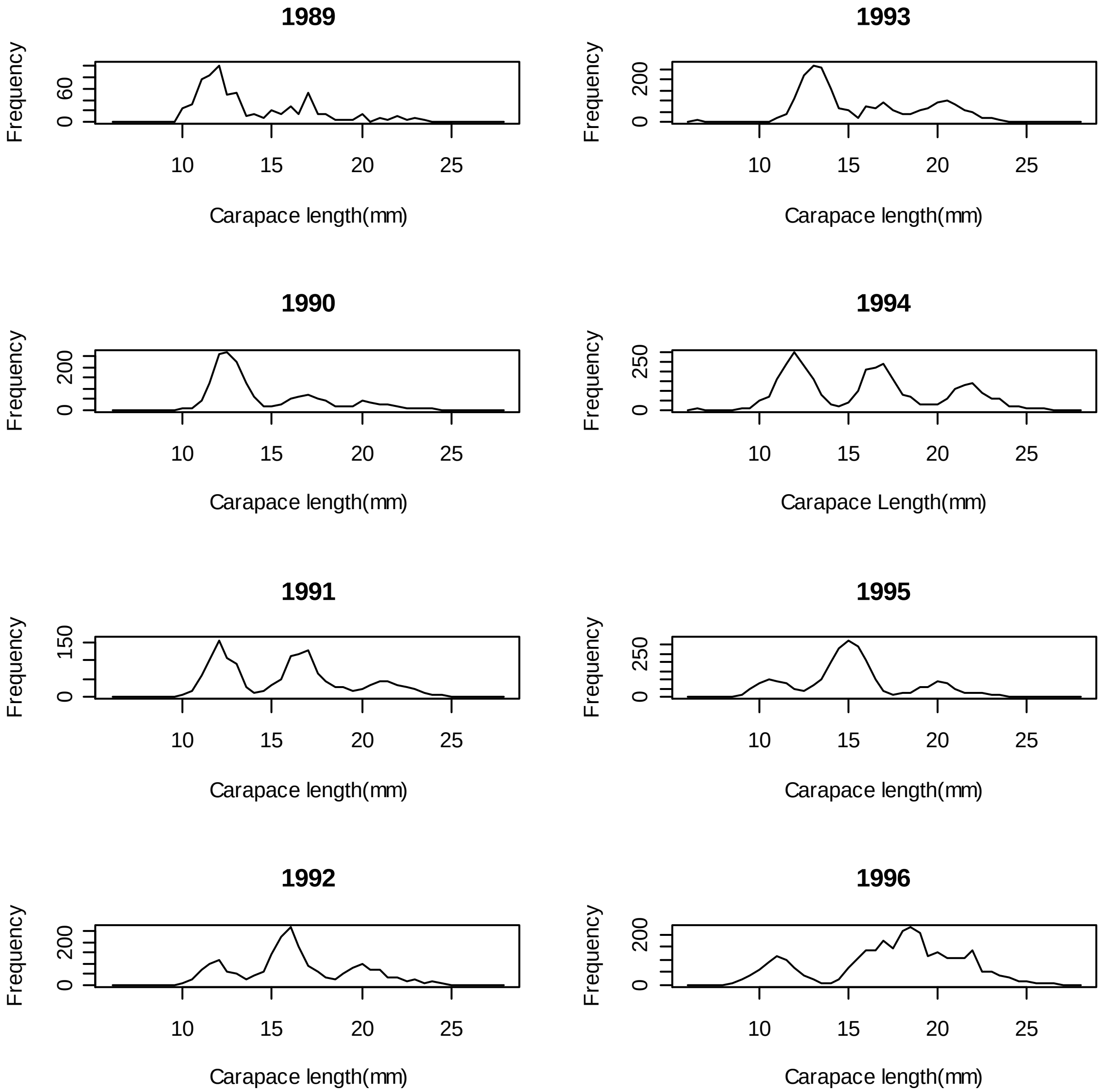

Figure 2: Length-frequency distribution of *P. borealis* plotted from the autumn surveys data for 1989- 1992.

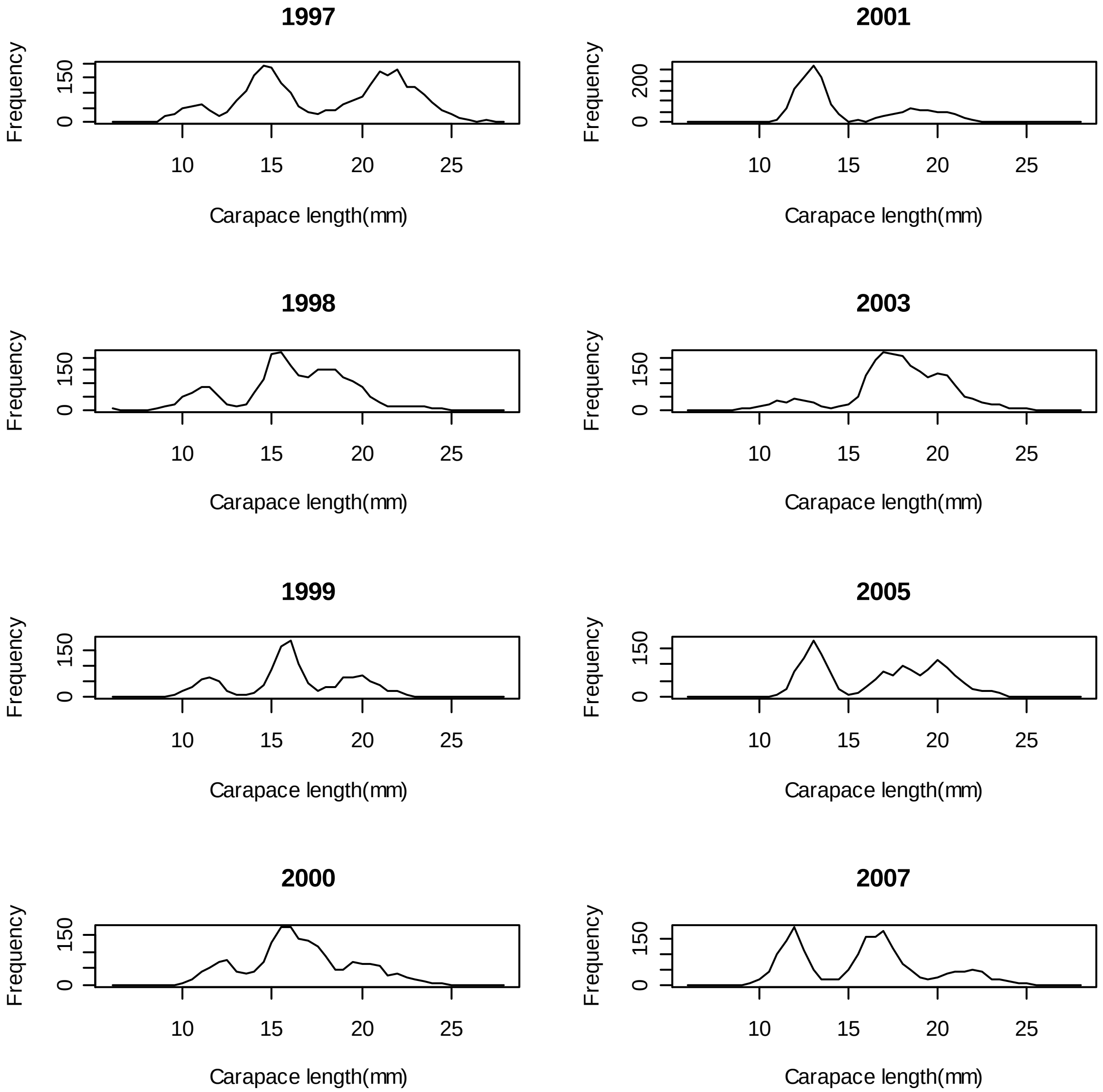

Figure 3: Length-frequency distribution of *P. borealis* in autumn surveys in Skjálfandi Bay 1997- 2007

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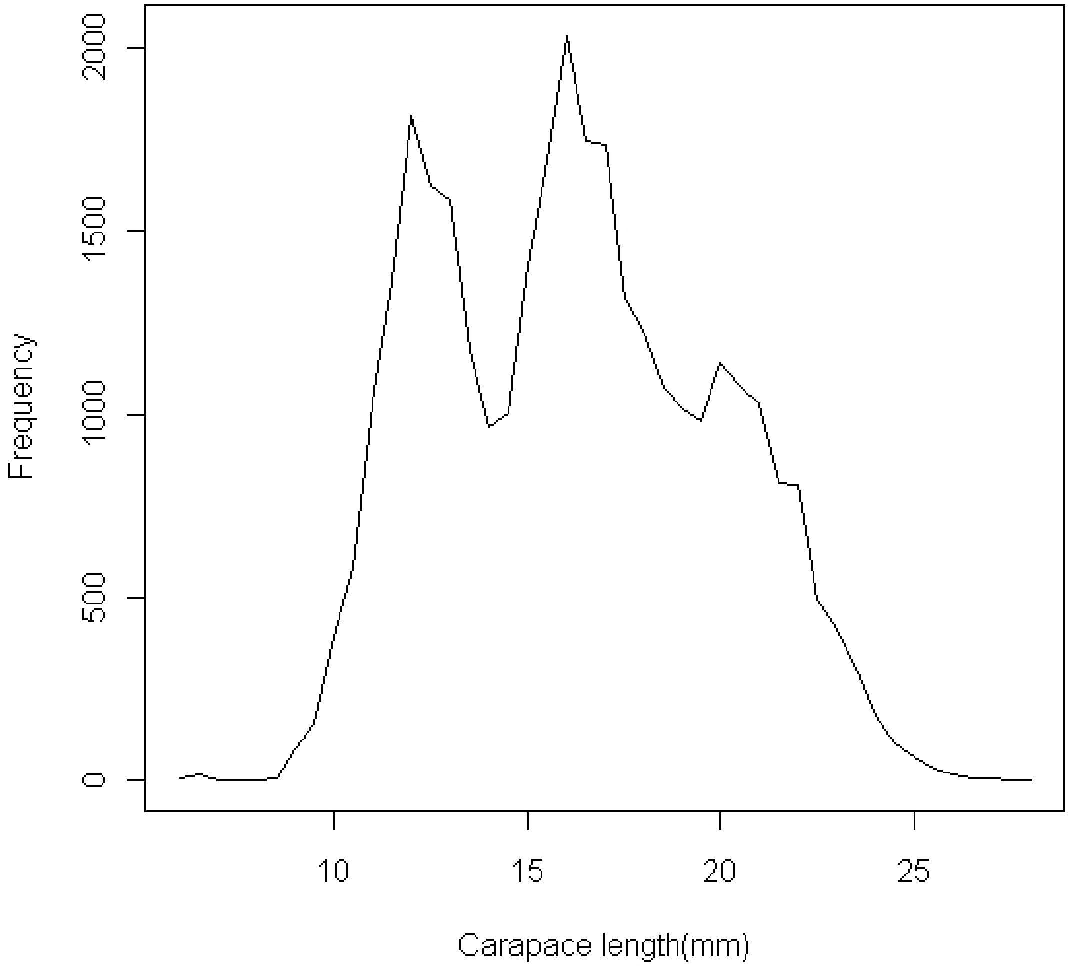

Figure 4: Cumulative length-frequency distribution of *P. borealis* in autumn surveys in Skjálfandi Bay 1989 - 2007

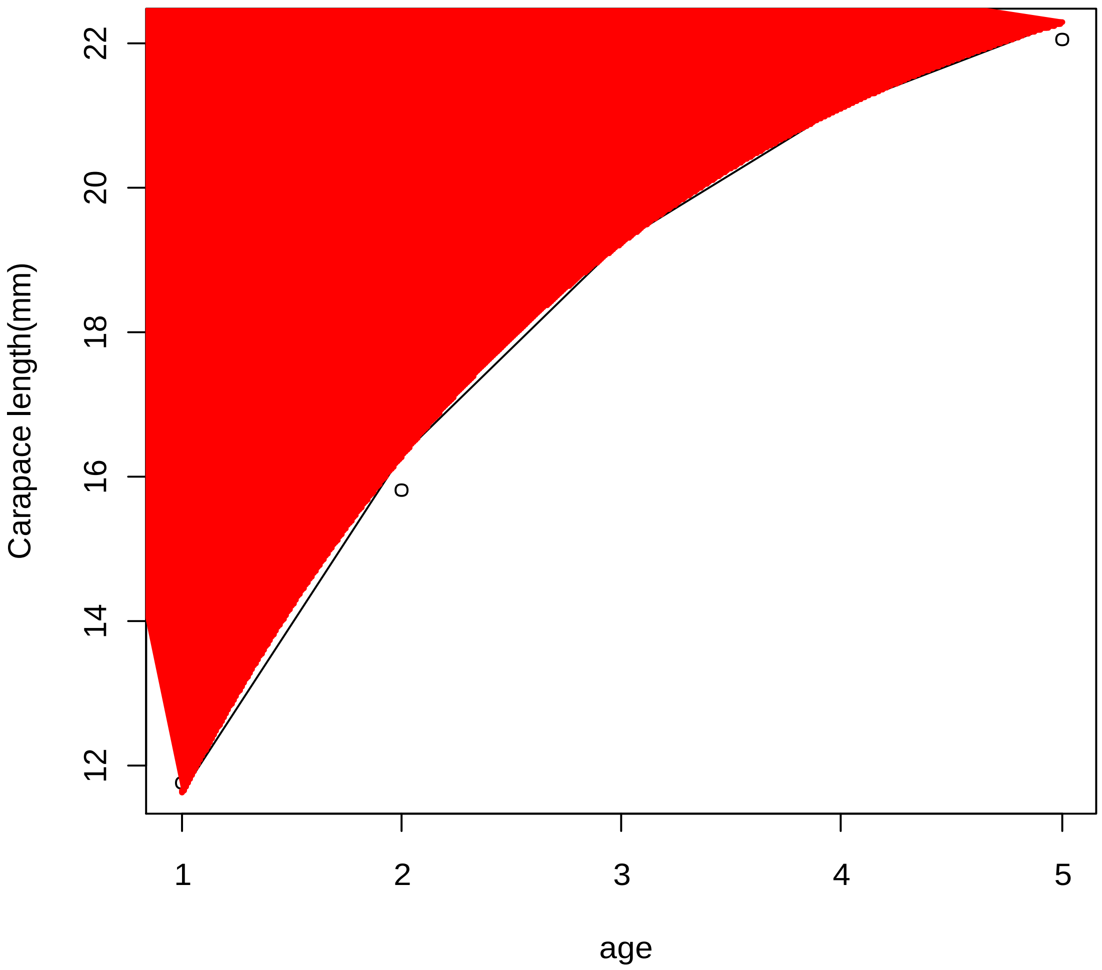

Figure 5: von Bertalanfy growth curve for *P. borea*lis in Skjálfandi Bay. Light curve is fitted to initial growth parameter values and thick curve is the fitted growth parameter estimates.

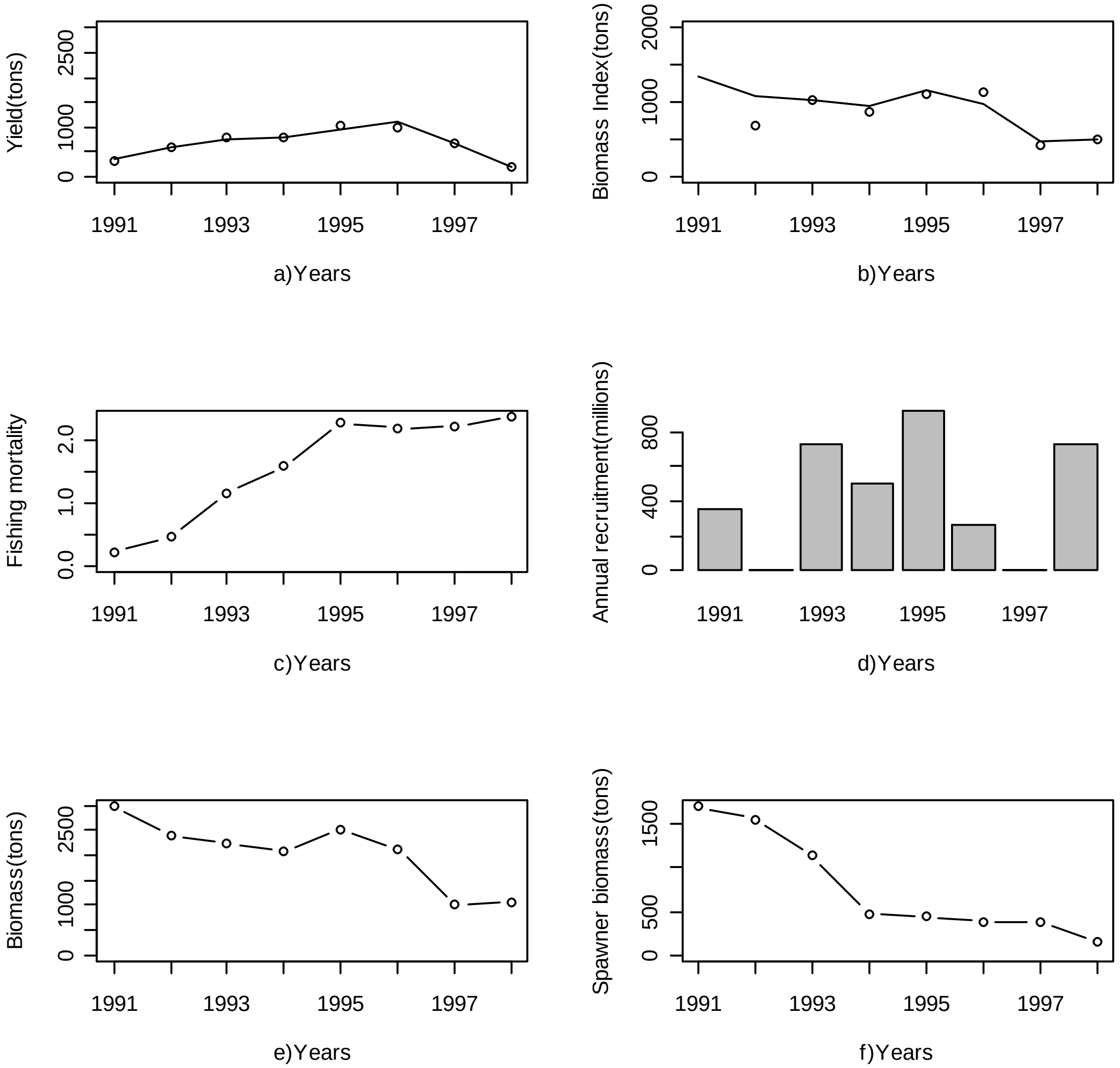

Figure 6: Output from dynamic biomass model. Figures 6a) and 6b) show annual catch and biomass index from surveys respectively (circles) and lines fitting the models. 6 c, d, e, and f are the fishing mortalities, recruitment, biomass and spawner biomass estimated from the model. The sums of squares of errors are SSEI = 0.48, SSEY= 0.03, SSER= 253.22 and SSEF = 1.60

The output gave a better fit of the yield over the years (Figure 6a). The model, however, did not show a good fit with the biomass index. This could be because of biases in the survey. Fishing mortalities rose sharply from 1991 to 1995 and remained high, above 2.0 until closure in 1999 (Figure 6c). Spawner biomass shows a decreasing trend since 1991 (Figure 6f). Stock biomass also decreased from 1991 to 1994, increased in 1995 and finally decreased again (Figure 6e). Average stock biomass was 2047 tons from 1991 to 1998 (Table 3).

Table 3: Biomass estimates of *P. borealis* in Skjálfandi Bay from 1991-1998.

Year	Biomass estimate (tons)
1991	2965
1992	2384
1993	2236
1994	2081
1995	2507
1996	2118
1997	1020
1998	1065

4.4 Yield and spawner stock biomass

The mean recruitment (1 year old shrimp) between 1992 -1997 was estimated to be 399.5 million individuals (dynamic production model) and this was assumed for the virgin stock after recovery.

Yield and spawner stock biomass were estimated for fishing mortalities starting from 0 to 2 (Table 4). The estimated yield increases from 271.7 tons at F =0.2 to 565.9 tons at F = 2.0 whereas the virgin stock spawner biomass was 1766.7 tons (at F = 0, no fishing) and 238.1 tons at F = 2.

Table 4: Yield and spawner stock biomass estimated from the yield per recruit models for different fishing mortalities of *P. borealis* in Skjálfandi Bay.

Fishing mortality	Yield (tons)	Spawner stock biomass (tons)
0	0	1766.7
0.2	271.7	1278.6
0.4	410.1	959.9
0.6	482.7	745.3
0.8	521.7	596.0
1.0	543.1	489.0
1.2	554.9	410.1
1.4	561.2	350.4
1.6	564.3	304.1
1.8	565.7	267.6
2.0	565.9	238.1

From the yield per recruits curves, the estimated Maximum Sustainable Yield per recruit was 1.42 g which corrrespond to a F_{max} = 1.97, Maximum Sustainable Yield = 566 tons and $F_{0.1}$ = 0.72 (Figure 7).

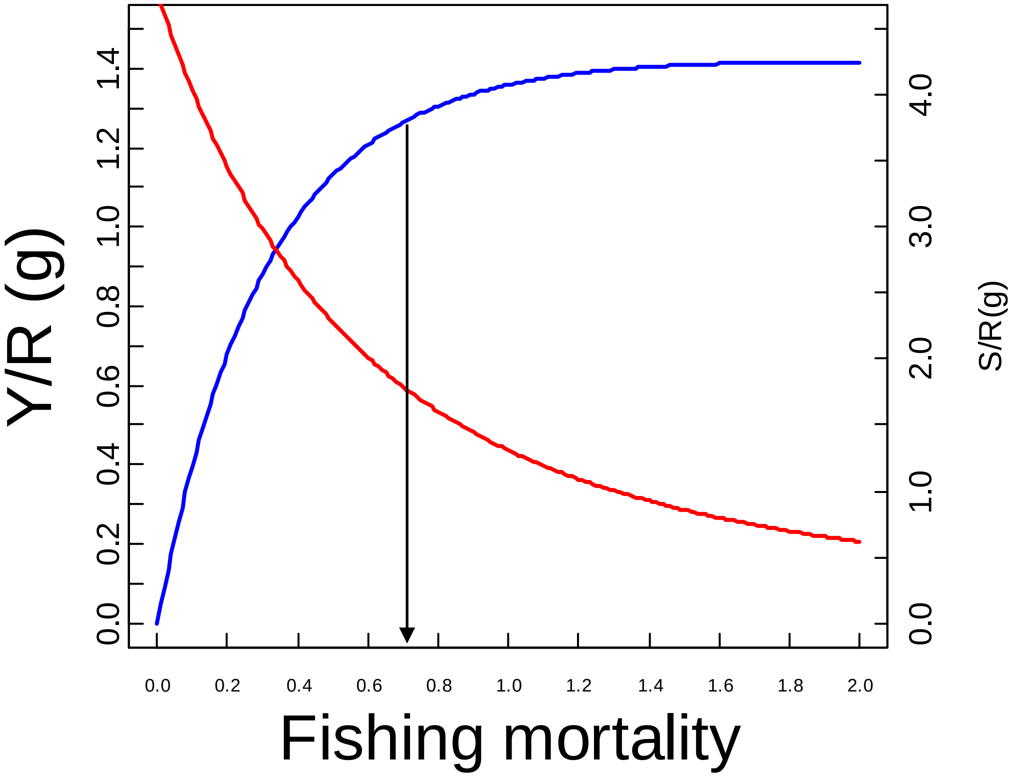

Figure 7: Yield per recruit and spawning stock per recruit curves for fishing mortalities ranging from 0 to 2. Arrow is showing fishing mortality corresponding to $F_{0.1.}$

5 DISCUSSION

P. borealis caught in Skjálfandi Bay had a maximum carapace length of 28 mm. This was is larger than the maximum length of shrimps observed in inner fjords such as Arnarfjörður (26.5 mm, Ceesay 2000) and in Húnaflói (25 mm, Lam Anh 2008. The stock of Skjálfandi Bay is geographically closer to the offshore (deeper water) stocks which are also relatively large sized. Larger shrimps prefer deeper waters where temperature is relatively stable (Garcia 2007). There are claims that the shrimp stock in Skjálfandi sometimes mixes with that of the offshore (Skúladóttir, pers. comm.). In fact one mixed area lies between Skjálfandi and the offshore stocks (Skúladóttir and Pétursson 1999). Even though these areas are well defined, there is the possibility of some mixing of these stocks.

The asymptotic length L_∞ and instantaneous growth parameter, *K*, obtained from this study (L_∞ =24.3, *K*= 0.46) are slightly different from those reported for the inshore stock in Húnaflói, northwest Iceland (Lam Anh 2008). The growth rate estimated in this study is also faster than that observed in Húnaflói (Lam Anh 2008). The slightly higher growth rate (*K*= 0.46) observed in Skjálfandi Bay may correlate with higher natural mortality and shorter life span of the stock (Kinne 1970). It is probable that the predation effects by cod and haddock are greater in Skjálfandi Bay than in Húnaflói but this cannot be conclusive. The inflow of warm geothermal water from the adjacent lake Mývatn, (Ólafsson 1999) which regulates the water temperature in Laxá in Aðaldal resulting in increased surface temperature in the bay, may influence the bottom temperature when mixing takes place. If this were substantial then one would expect a higher temperature in Skjálfandi Bay thus leading to faster growth. Skúladóttir (2007) observed that growth rate of shrimp in Gullmarsfjord, Sweden where temperature was higher was faster than in Ísafjarðardjúp, Iceland and in the Flemish Cap where bottom temperatures were lower.

Estimating growth parameters based on length-frequency data is difficult. Problems arise because during the ovigerous period, moulting does not take place and there is also variability in growth rates of different sexes, age classes and in different seasons.

The growth performance index, ø', assesses the reliability of L_∞ and *K*, obtained in this study. The index (ø') obtained in this study and computed values obtained in previous studies for *P. Borealis* in other areas remain fairly similar (Table 2). The maximum size of relatively fast growing populations will be relatively low (high *K*, low L_∞), but when growth is relatively slow (low *K*) in populations of the same species, L_∞ will increase, resulting in the growth performance index being relatively constant for the species (Pauly 1984. The same appears to be true for shrimp.

Fishing mortalities were high but the stock was able to withstand fishing and predation pressures for some years until the collapse in 1998. It might be that cod was feeding on the juveniles and yet the adults were able to reproduce and increase in biomass but only for some time until recruitment could not compensate for mortality. Another hypothesis might be the match-mismatch hypothesis (Cushing 1975). If when the larvae are hatched and there is plenty of food then they have the tendency to grow and be able to recruit into the fishery later in life provided there is not much predation. If, on the other hand, food is limited at the time of hatching then the survival rate will be reduced. In reality both physical and biological processes interact and can both have an effect on recruitment and other life processes (Heath 1992).

The natural mortality $M = 0.5$ which was used in this model was not calculated but represents a best guess, considering high natural mortality due to predation. The effect of fishing mortality is reduced with increases in natural mortality, as the two types of mortality "compete" for the same individuals. It is advisable to catch the species that suffer high natural mortality at a relatively young age, before they die of natural causes, as this is the only time that exploitation can be optimum (Sparre *et al*. 1989). *P. borealis* is preyed upon to a large extent by cod and haddock resulting in high natural mortality. Unless the shrimp stocks are to be managed as food for cod, it might be advisable to reduce the minimum size at capture while cohort biomass is still relatively high.

When natural mortality is low the increases in biomass from growth of individuals exceed losses due to natural mortality for a long period in the life of the cohort. A high M will result in biomass losses due to natural mortality exceeding gain due to growth at a relatively young age.

The yield in the inshore shrimp of Skjálfandi Bay shows an increase from a low figure (310 tons) in 1991, reaching a peak of 1027 tons in 1995 and 1996 (Figure 6a). After 1996, there was a decline in yield to an all time low of 191 tons in 1998 when the stock collapsed.

In years when biomass is high, recruitment was also high. The shrimp biomass estimate in 1999 was about a third of the biomass in 1991. This shows that there was not much recruitment to compensate for the reduction in biomass since 1992.

Annual recruitment is influenced by density of cod and haddock in the previous year. When the cod (*Gadus morhua*) and/or haddock in the area are less abundant in a given year, shrimp recruitment increases and vice versa. In some years reduced recruitment of shrimps can be observed when there is an increase in cod. For example, when the presence of cod in Skjálfandi Bay was less in 1997, recruitment of shrimp was high in 1998. When a regression was done the relationship was however weak (Figure 8). This is probably due to short time series data that were available.

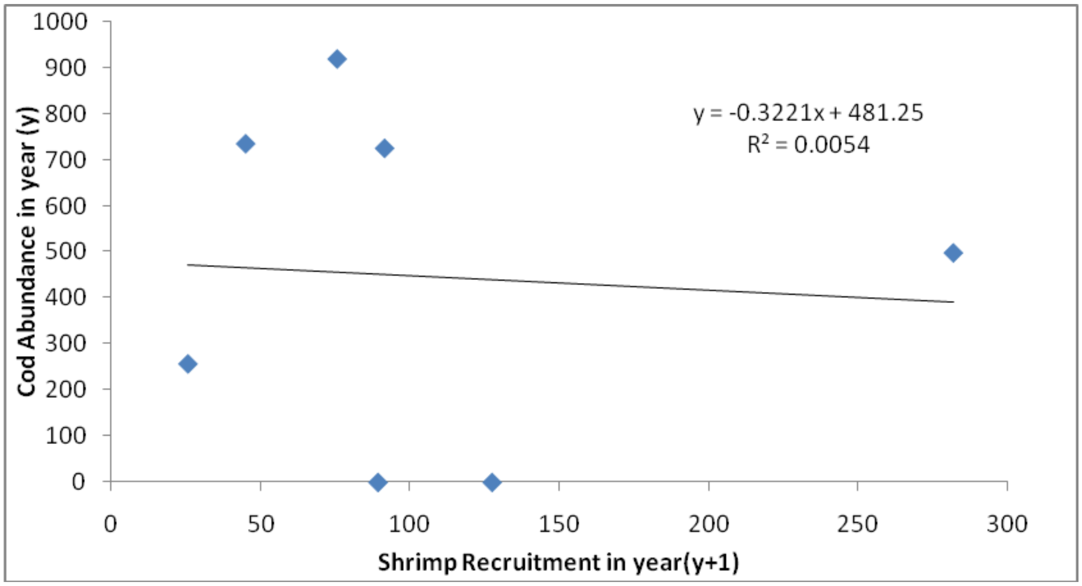

Figure 8: Annual shrimp recruitment estimated from the dynamic production model for each year regressed with the mean number of cod (*Gadus morhua*) per haul in the previous year from the MRI autumn surveys. The general trend is that as cod numbers decrease in a year, the recruitment of shrimp increases the following year.

A similar situation was observed by Lam Anh (2008) in the inshore stocks of Húnaflói. According to Lam Anh (2008) relatively low cod density in 1989 resulted in the increased recruitment of shrimp to the fishery in 1990. Stefánsson *et al*. (1994) also observed the same trend in other Icelandic waters, as the amount of cod increases, the biomass of *Pandalus borealis* is reduced and so is recruitment. In July 1991, Parsson *et al*. (1991) noted that when cod moved into the St Anthony Basin in northeastern Newfoundland there was a drastic reduction in shrimp catches during that time. Lilly *et al*. (2000) also suggested that an increase in shrimp stocks observed in Northeast Newfoundland was to some extent related to decreased cod stocks. Shrimp fisheries off the east coast of Canada are said to have increased after the decline in cod; and the development of the West Greenland shrimp fisheries during the 1970s is believed to have been related to the major decline in the cod stock in that area (Lilly *et al*. 2000). There is evidence from analysis of stomach contents of young cod that shrimp are a major prey (Magnússon and Pálsson 1991).

The predation effect of haddock (*Melanogrammus aeglefinus*) on *Pandalus* cannot be underestimated (Skúladóttir pers. comm.). The higher the numbers of haddock present in the bay in a particular year, the smaller the amount of shrimp recruitment the following year (Figure 9). This trend is similar to the effects shown by cod on shrimp in Skjálfandi Bay (Figure 8).

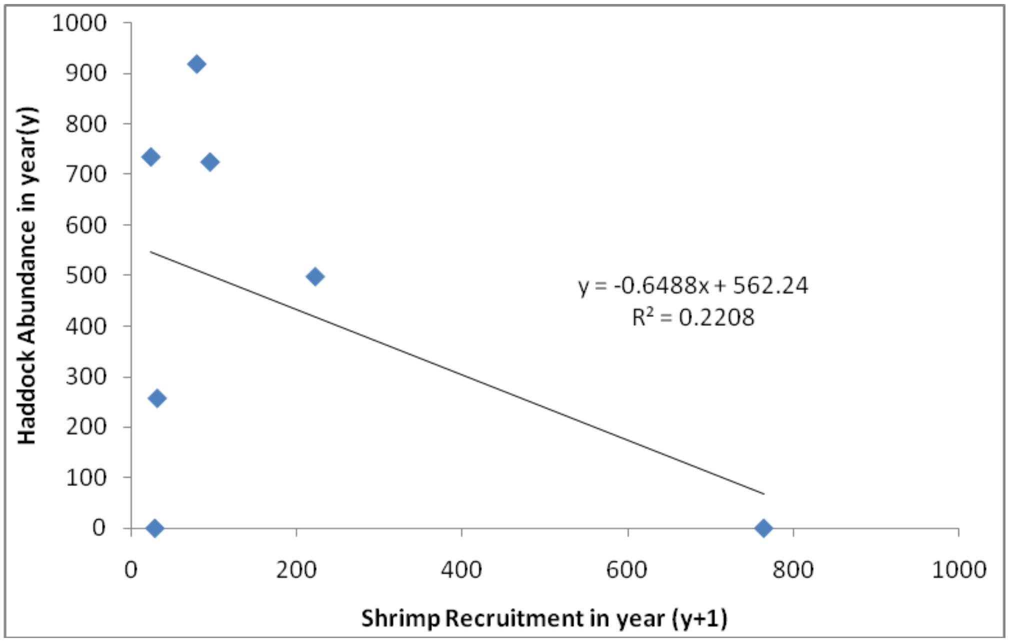

Figure 9: Regression of shrimp recruitment estimated from the dynamic production model in a year with the mean number of haddock per haul in the previous year from the MRI autumn surveys. When numbers of haddock were high in a year, the recruitment of shrimp was low in the year after.

The physical environment can also affect shrimp populations (Andersson 2000, Koelle 2002 and this may be related to variations in stocks size in different areas (Anderson 2000). Wieland (2004), revealed that an increase in the bottom temperature of the waters off West Greenland led to a progressive increase in biomass and catches of *Pandalus* in that area during the period 1991-2002. The present study did not look at the effects of environmental changes such as temperature on the abundance and distribution of *P. borealis* in Skjálfandi Bay. It is worth noting that sea temperatures at Hjalteyri close to Skjálfandi Bay (Jónsson 1999) outside the tolerance level of shrimp (0 ºC – 8 ºC) were not observed during the period under study. The average surface temperature is around 4.9 ºC and a little less at the bottom (Jónsson pers. comm.). Therefore, temperature could not have influenced mortality. However, it is possible that predation pressure and other environmental factors might have affected the biomass and recruitment of shrimp in Skjálfandi Bay.

5.1 Management of the shrimp fishery in Sierra Leone

Four species make up the bulk of the shrimp resources of Sierra Leone. The four main shrimp species are found in different regions in the waters of Sierra Leone. The white shrimp (*Parapenaeopsis atlantica)* is found in the Shebro River estuary in the southern part of the country. The pink shrimp (*Penaeus notialis*), which is also the most abundant species, occurs predominantly off the Freetown peninsula and especially around Banana Island. They are mainly found in coastal waters and have a geographical distribution, from Cape Blanc, Mauritania down to Angola. *P. notialis* occurs mainly between 10 m to 100 m depths and at temperatures between 18-24 °C. The tiger shrimp (*Penaeus* *kerathurus*) is found in the north around Yeliboya whereas the deep water rose shrimp (*Parapenaeus longirostris*) is mainly offshore in deeper waters and it is purported to be exploited on a limited scale even though there is “potential for a whole offshore fishery on its own” (Anon. 2008). The potential of a *P. longirostris* fishery in Sierra Leone is, however, not well known (Anon. 2008). One reason for the limited exploitation is that most shrimp trawlers are small and only capable of trawling in shallow waters around the estuary.

Penaeid species have a wide geographical distribution, occurring in the south Atlantic region of Spain, off the coast of Morocco and in the Mediterranean and adjacent waters as well as in the waters of Angola. During the upwelling seasons, the species migrate to coastal waters.

From a biological point of view, the fact that these species are found in different parts of the waters in Sierra Leone, one would expect management to be based on individual stocks and different management strategies to be implemented for their sustainable utilisation. This is, however, not the case. In fact, in the catch statistic database, these species were all lumped together until 2004 when they were first recorded by species.

The Fisheries Management and Development Act of 1994 contain regulations relating to fishing in Sierra Leone. Effort control in the shrimp fisheries is implemented by limiting the number of fishing vessels that can access the resource at any one time. A mandatory license has to be obtained before one can fish for shrimp and the mesh size (stretched mesh) in the cod-end should not be below 42 mm. The deployment of observers onboard fishing vessels and logging of catch statistics is mandatory in the shrimp fishery. Shrimpers are excluded from fishing in the Inshore Exclusion Zone, IEZ which is the area mostly exploited by the artisanal sector, and part of which serves as breeding and nursery grounds for a number of fish species.

A landing obligation for the local market is to be met by all shrimpers; they should land 5% of shrimp and other shellfish species and 60% of fish caught. This is to make shrimp readily available to the Sierra Leone market. Limited patrols serve as a deterrent to illegal fishing practices and those found to have committed any infraction are fined.

5.2 Assessment and management measures to be incorporated into the shrimp fisheries of Sierra Leone

From a review of the assessment and management measures implemented in Iceland, and those practiced worldwide, there are some that could be useful for Sierra Leone to sustainably exploit the shrimp resources. These are discussed below.

5.2.1 Assessment methods

Shrimp fisheries assessment in Sierra Leone is mainly in the form of a descriptive summary of catch statistics, and abundance indices in the form of CPUE without considering population structure. Although these methods provide information on the status of the shrimp fisheries relative to previous levels and are useful for management purposes, they do not include analytical assessment. Assessment that provides information on stock dynamics and population parameters may be good enough to enable managers to develop management schemes for sustainable exploitation of the resource, whereas assessment based on catch statistics and abundance indices tends to only "lead to reactive management" (Smith and Addison 2003 p.231).

Catch rates (CPUE) from commercial shrimp vessels have been used as indices of relative abundance and for the purposes of assessing stocks (Gulland 1956, Goni *et al*. 1999). CPUE may not be a measure of actual abundance (Hilborn and Walters 1992). Scientific surveys do have some advantage over the use of commercial CPUE data (Anon. 1997) and the uncertainty of biomass and population parameters derived from survey data can to some extent be quantified (Folmer and Pennington 2000), whereas the uncertainty associated with catch-based assessment of commercial data is difficult to calculate as effort is not standardised. Lack of reliable survey data may lead to stock collapse if these uncertainties are ignored (Pennington and Strømme 1998). It is documented also that catch rates in shrimps and fish is influenced by seasons, diurnal movement, efficiency of the captain of the fishing vessel and many more factors other than just stock abundance (Bishop *et al*. 2008); and hence the need for standardisation to correct the effects that are caused by the above factors.

In recent years, new methods have been used for standardising CPUE and this includes the Generalized Linear Model (GLM) (Campbell 2004, Yongshun Xiao 2004, Bishop *et al*. 2008) and other methods such as general additive model, GAM, and generalized linear mixed models (GLMMs) (Helser *et al*. 2004).

When GAM is fitted to catch rates data, factors that have significant effects and the extent to which they influence results can be identified through the analysis of deviance of the GLM and can then be standardised. This standardisation removes effects of such factors as changes in the composition of fleets over time, depths, and area (Hilborn and Walters 1992, Punt *et al*. 2000).

Sierra Leone hardly does scientific surveys for assessment of the stocks of penaeid shrimps and relies on catch statistics of commercial vessels. Given the cost involved in scientific surveys for stock assessment and the fact that Sierra Leone is ranked as one the poorest countries in the world, this state of affairs is likely to continue for quite a while. What could probably be done in the interim is to use statistical standardisation tools now available that will to some extent give a fairly good estimate of stock size other than the usual CPUE estimates from commercial vessels.

The assessment method used in this study could be utilised in the penaeid shrimps in Sierra Leone but with some modifications. This is because of the differences in biology and life history of pandalic shrimps to that of penaeid shrimp. *P. borealis* can reach ages of 5-8 years and so stock assessment can be done on a yearly basis. Penaeid shrimps in Sierra Leone have a fishable life span of one year though it is reported that they can live to between 2-3 years in some areas (Garcia 1988). Apart from a short life span, one of the most distinct features of penaeid shrimps to that of temperate shrimps is their seasonality. Abundance varies greatly with seasons and in Sierra Leone, there are two peak seasons in a year; a major (June-July) and a minor one in March (Showers 1999). Thus assessment of shrimp stocks in Sierra Leone is to be done on a seasonal basis so as to give plausible recommendations based on comparisons with peak seasons of different years for example. According to Garcia (1988) studies on the dynamics of shrimp populations *must refer to time intervals of less than a year and monthly (sometimes weekly) intervals*. For the Sierra Leone shrimps it is advisable to have a monthly basis because considerable growth can take place within a short time frame.

Penaeid shrimps in Sierra Leone migrate to the ocean to grow but return to the estuary to spawn. Seasonality plays a key role in this migration pattern and so time of spawning, recruitment, population age structure and catchability should be taken into consideration rather than taking an average within a year. Growth is seasonal and Pauly (1984) has suggested that a “seasonal oscillating” von Bertalanffy model which takes into consideration the factor of season could be used to give a general description of all cohorts in a stock.

5.2.2 Management

Iceland manages the shrimps stocks as separate units in the fjords and those in offshore as one unit even though they are the same species. The principle of managing a stock on its own is vital in fisheries. Having knowledge of the stock size and amount harvested will enable managers to institute measures to rationally exploit the resource. In the case of Iceland, the fact that shrimp stocks are monitored at stock level has enabled researchers to ascertain which stocks are overexploited and that has led to the closure of some inshore fisheries.

The establishment of different management strategies for each shrimp species in Sierra Leone is to be implemented. New management strategies should include the use of by-catch reduction devices.

5.2.3 By-catch reduction devices

The level of by-catch and discards in world fisheries in general and in shrimp fisheries in and particular is large and has led to high mortalities of non-target species (Blaber *et al*. 1990). Tillman (1992) notes that the reduction in the croaker stocks in the Gulf of Mexico was due to huge discards. Most industrialised nations are therefore using by-catch reduction devices (BRD) as a management tools. Measures to reduce by-catch play an important role in fisheries management today. The Nordmore sorting grates are utilised in Icelandic shrimp fisheries and are considered to be one of many by-catch reduction devices that have resulted in low levels of by-catch in small crustacean fisheries (Broadhurst 2000, Kelleher, 2005). According to Richards and Hendrickson (2006), the Nordmore sorting grate led to a by-catch reduction of 15% in the Northern shrimp fishery in the Gulf of Maine. A review by Broadhurst (2000) indicates a greater efficiency; with by-catch reduction due to the Nordmore grate ranging between 60-90% in many fisheries. From studies observed in other areas the use of BDR can be considered a useful management practice that could be implemented in Sierra Leone.

Another species usually caught as incidental in shrimp fisheries are marine turtles. This issue has attracted the attention of conservation groups worldwide and has led to the development of Turtle Excluder Device (TED). A TED is a metal grid with bars that is attached to a shrimp trawl net and has an opening usually at the top or bottom and creates a hatch to let larger species such as sea turtles, and bigger fishes escape while retaining the shrimps. This is used in many fisheries in the United States of America and has proved to be effective (NOAA 2007).

Sierra Leone has not implemented a regulation on the use of BDR in its shrimp fisheries and the level of by-catch, mostly fish, in the shrimp fisheries is very high. For example in 2006 alone the average by-catch in the shrimp fisheries was about four times the catch of target species (Figure 10).

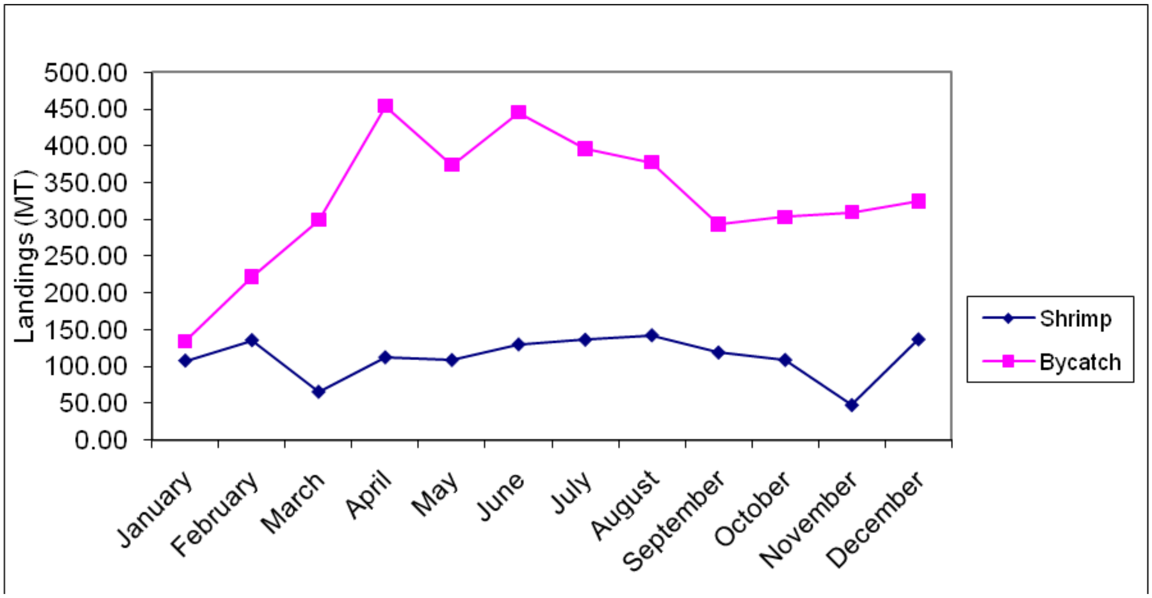

Figure 10: By-catch levels in the shrimp fisheries of Sierra Leone in 2006. On average the by-catch level was about four times the amount of shrimp caught.

Landings of by-catch in the shrimp fishery are usually of marketable sized fish, but the discards are not documented. Even though observers are placed onboard each vessel to monitor fishing activities, they cannot work 24 hours a day and there is reason for one to think that the level of discards in this fishery may be high. Discarded fish, many of which die when thrown overboard, eventually decay and decompose leading to increases in particulate organic matter which in the absence of proper mixing of the water column may result in low levels of oxygen. This will affect marine organisms in the area. One way to resolve the by-catch problems is for the Ministry of Fisheries and Marine Resources of Sierra Leone to legislates the use of BRDs in the shrimp fishery and establish a framework for BDR certification.

Another regulation which could be implemented to reduce by-catch is to mandate the use of square meshes rather than the traditional mesh that is presently used in the fishery. Square shaped meshes have been observed to increase selectivity (Broadhurst *et al*. 2004) thus reducing by-catch. In the Gulf of St. Vincent a reduction in the by-catch rate of small fishes and juvenile king prawns was achieved as a result of the use of square meshes in the cod-end (Broadhurst *et al*. 1999a).

6 CONCLUSION

Biomass estimates in the autumn surveys were on average around 2047 tons. Fishing mortality estimates were very high, increasing rapidly until 1995 when they became relatively stable. The shrimps were able to withstand high fishing mortality for a while before the collapse in 1998. The reason for the collapse could be attributed to other factors rather than predation by cod. Predation by was observed to reduce recruitment of shrimp but not significantly. In comparison, predation by haddock seems to have relatively more effect on recruitment than predation by cod.

Five age groups were observed for shrimps in Skjálfandi Bay (ages 1-5). The proportion of age 5 was small and seems almost non-existent but fitted well to the von Bertalanffy growth curve. The growth rate of *P. borealis* was faster in Skjálfandi Bay than in the inshore stocks of Húnaflói. Maximum Sustainable Yield per recruit was 1.42 g per recruit which corrresponds to a $F_{max} = 1.97$ and $F_{0.1} = 0.72$.

The dynamic production model has the advantage of incorporating age structure into the model. It is very useful for the assessment of pandalic shrimps. It, however, utilises complex data and depends on the assumption that catch rates are reliable estimates of index of biomass. The problem of ageing shrimp may also affect the output of the results. This method does have an advantage over surplus production models used in Sierra Leone. The disadvantages with either the Schaefer or Fox surplus production models is that the equilibrium assumptions of average effort (Fox 1970) for fitting the data often give poor and misleading estimates (Smith and Addison 2003) and are generally not used in most fisheries.

The dynamic production model can be used to assess penaeid shrimps in Sierra Leone provided the scientific surveys are done which gives a more reliable estimate of the abundance index than catch from commercial vessels. Data should be collected monthly or seasonally and because it is very expensive to conduct surveys, seasonal surveys should be done. Also, data relating to selectivity, maturity, and length- weight relationship will have to be periodically gathered if the model is to be used in assessing the shrimps of Sierra Leone.

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