A summary of the assessment and management approach applied to South African abalone *(Haliotis midae)* in Zones A-D

Éva E. Plagányi

MARAM (Marine Resource Assessment and Management Group), Department of Mathematics and Applied Mathematics, University of Cape Town, Eva.Plaganyi-lloyd@uct.ac.za

ABSTRACT

The management of abalone stocks worldwide is complicated by factors such as poaching combined with the difficulties of assessing a sedentary (but not immobile) resource that is often patchily distributed. The South African abalone *Haliotis midae* fishery is faced with an additional problem in the form of a movement of rock lobsters *Jasus lalandii* into much of the range of the abalone. The lobsters have dramatically reduced sea urchin *Parechinus angulosus* populations, thereby indirectly negatively impacting juvenile abalone, which rely on the urchins for shelter. The model developed for abalone is an extension of more standard age-structured assessment models because it explicitly takes spatial effects into account, incorporates the ecosystem change effect described above and formally estimates illegal catches using a novel index, the Confiscations Per Unit Policing Effort (CPUPE). The model is simultaneously fitted to CPUE and Fishery-Independent Abalone Survey (FIAS) abundance data as well as several years of catch-at-age (cohort-sliced from catch-at-length) data for the various components of the fishery as well as for different strata. A basic tenet of fisheries modelling is to not go beyond the information content of the data. The model developed involves the efficient use of data to allow a model of greater complexity (as was essential in this instance) than usual. It has provided the basis for management advice over recent years by projecting abundance trends under alternative future catch levels.

INTRODUCTION

Benthic shell-fisheries world-wide encompass some of the greatest challenges to fisheries managers in that they are relatively sedentary, broadcast spawners and often exhibit a complicated spatial structure (Orensanz *et al.* 2004, Dowling et al. 2004a,b). Their high commercial value renders them particularly vulnerable to overexploitation, with the collapse of several stocks being attributed either to overfishing or environmental factors (Shepherd *et al.* 2001, Tegner *et al*. 2001). The South African abalone *Haliotis midae* fishery is currently declining precipitously due to a combination of these two factors: it has recently been subject to phenomenally high levels of illegal fishing and is at the centre of an intricate ecosystem change in the form of a movement of rock lobsters *Jasus lalandii* into a major part (Zones C and D – Fig. 1) of the range of the abalone (Tarr *et al*. 1996). Predation of urchins by rock lobsters is blamed for the collapse of urchin populations (Mayfield and Branch 2000). As sea urchin abundance declined, so too did the numbers of juvenile abalone. Such abalone (individuals 3-35 mm in length) depend heavily on a commensal association with urchins *Parechinus angulosus*, because the urchins provide protection against predators and may supplement the diet of juvenile abalone (Day and Branch 2002).

The South African abalone fishery dates back to 1949 and is one of the oldest commercial abalone fisheries in the world, with 55-yr records of commercial catch data (Tarr 1992). Four main fishery Zones A-D on the south coast (Fig. 1) have typically yielded some 80 - 90% of the total annual Total Allowable Catch (TAC). The South African abalone fishery (i.e. excluding farmed animals) ranks as one of the top 5-10 biggest abalone fisheries in the world, but is nevertheless substantially smaller than the abalone fisheries of Australia and Japan (Gordon and Cook 2001, Tarbath *et al*. 2002).

This paper summarises the development of a model for *H. midae* that is an extension of more standard age-structured assessment models because it uses a novel approach to formally estimate illegal catches, it incorporates the ecosystem change effect described above and it explicitly takes spatial effects into account. The model is simultaneously fitted to CPUE and Fishery-Independent Abalone Survey (FIAS) abundance data as well as several years of catch-at-age (cohort-sliced from catch-at-length) data for the various components of the fishery as well as for different strata. Prior to this approach, recommendations on TACs for *H. midae* were based solely on a set of decision rules that used data on trends in the commercial CPUE (catch per unit effort), the average and modal sizes of abalone in the commercial catches, and the proportion of the catch represented by the smallest legal sizes.

A basic tenet of fisheries modelling is to not go beyond the information content of the data. The model we developed involves the efficient use of data to allow a model of greater complexity (as was essential in this instance) than usual. Age-structured production models (ASPM) (e.g. Hilborn 1990) have previously been mostly applied to marine fish species other than shell-fish which are generally assessed using simpler methods. The methodology described here is essentially an adaptation of that applied for management purposes for many key South African and Namibian fishery resources, including for the major fisheries for hake (*Merluccius* Spp.) in this region (e.g. Butterworth and Geromont, 2001, Butterworth and Rademeyer, in press). The ASPM approach has also been applied to a number of international stocks such as southern bluefin tuna (e.g. Butterworth and Plagányi 2000).

An ASPM approach was specifically chosen in preference to, for example, Virtual Population Analysis (VPA). VPA necessitates catch-at-age data for all years and essentially reconstructs the history of each cohort, assuming that the observed information is known without error. In contrast the ASPM methodology is more flexible, in that it does not require catch-at-age data for all the years considered, and it can accommodate likely errors in such data, by making assumptions about the selectivity-at-age of the catch (Butterworth *et al.* 2003a). An ASPM assessment involves constructing an age-structured model of the population dynamics and fitting it to all available abundance indices by maximising the likelihood function. Available catch and survey abundance index data can thus be formally incorporated in the analysis.

One of the greatest impediments to the application of stock assessment models to abalone fisheries has been that such models usually rely on commercial catch per unit effort (CPUE) data as an index of abundance. Several authors have stressed that in areas such as South Australia and Tasmania the spatial variation in density is such that, coupled with the added complexities of diver behaviour, CPUE cannot be considered a reliable indicator of stock abundance in these areas (Breen 1986, Prince 1992, Shepherd *et al*. 1992, Keesing & Baker 1998, Worthington & Andrew 1998, Dowling 2004 a,b). Plagányi *et al.* (2001) argue that CPUE data have utility in the South African context (but not necessarily elsewhere in the world) because, *inter alia*, the major South African abalone fishery is located in shallow kelp bed areas, relatively close inshore, along a relatively short stretch of coastline and with relatively easy access to most of the areas. As proposed by Dichmont *et al*. (2000) one option available to resource managers with some confidence in CPUE as an index of abundance for their fishery, is to use a dynamic model tuned to both CPUE data and a survey index.

This paper describes the spatial and age-structured assessment model that is currently applied to the management of the commercially valuable *H. midae* resource in Zones A-D (Fig. 1). The model estimates the reduction in juvenile abalone survival due to the ecosystem change extent and estimates the illegal take using a novel fisheries index – the confiscations per unit of policing effort (CPUPE).

METHODS

The South African abalone fishery is reliant on a single gastropod species, *Haliotis midae*, locally termed *perlemoen*, which is restricted to shallow habitats in beds of kelp, *Ecklonia maxima* (Tarr 1993). Abalone are patchily distributed between Cape Columbine on the west coast, and the Transkei region of the Eastern Cape Province on the east coast (Fig. 1). A second species, *H. spadicea*, is not as abundant, and is taken by recreational fishers only.

The commercial fishery occurs only within the western part of the overall range of abalone, from Cape Columbine to Quoin Point (near Cape Agulhas) (Tarr 2002) (Fig. 1). The commercially fished area has been divided into seven fishing zones, with TACs set separately for each zone since 1986 (see also Discussion). The main fishing areas are Zones A-D (Fig. 1). This section of coastline is naturally divided into zones because a series of sandy beach areas serve as partitions for areas containing sublittoral rocky seabed and hence suitable habitat for abalone (Tarr 1993).

Biology

Abalone are broadcast spawners and the planktonic larvae typically drift with the currents for about a week (McShane 1992). They settle mostly in shallow inshore waters (< 5 m), where they seek shelter under boulders or under the spines of sea urchins *Parechinus angulosus* (Tarr *et al.* 1996). They are slow growing, requiring a period of about 7 years to attain 100% sexual maturity, and 8-9 years to attain the minimum legal size limit of 114 mm shell breadth (Tarr 1995). They become emergent only once they have attained a sufficiently large size (~100mm shell length (SL)) to be afforded some protection from predation (Tarr 1993). With increasing size, animals gradually disperse into deeper water.

Benthic invertebrates typically have fairly limited dispersal potential (Bradbury and Snelgrove 2001). Strong correlations between the abundances of adult and newly recruited abalone at several sites in South Australia suggests that abalone larvae are not widely dispersed (Prince *et al.* 1988, McShane 1992). For black abalone *H. cracherodii* in southern California, average dispersal ranges are thought to be on the order of only 1- 5 km (Tegner 1993). Studies of genetic structure (e.g. Shepherd and Brown 1993) and simulations of larval transport for southern Australian abalone (Black 1993) provide additional support for the notion of dispersal over spatial scales of a few kilometers only. Although it is presently not known to what extent larval mixing occurs throughout the main fishing area in South Africa, larval and post-larval stages are most likely retained in areas close to the parental population, with some larval interchange between adjacent areas. It is considered unlikely that much inter-change occurs between adjacent fishery zones which each have a long-shore extent of some 30 km. Nonetheless, larval dispersal of South African abalone is likely wider (5 – 30 km) than reported for other species (BENEFIT 2002) based on the observation during recruitment surveys of consistent levels of juvenile (2 – 40 mm) abundance in the Betty's Bay Reserve and the neighbouring Zone D commercial grounds (R. Tarr, MCM, South Africa, pers. commn).

Data

A large amount of data are available for each of the main abalone fishery Zones A-D and are presented in full in Plagányi (2004). For assessment and management purposes, Zone C has been split into two subareas, a "poached" subarea (CP) to the west and a "nonpoached" subarea (CNP) to the east, because of substantial differences in the extent of poaching in these subareas. The two areas are approximately equivalent in terms of available habitat for abalone based on kelp bed area estimates (Tarr 1993).

All available data have been reworked in terms of a standard Model year y that is taken to run from October of year y-1 to September of year y. Commercial catch data (in terms of tonnes) are available from 1953 (Fig. 2). Total allowable catches (TACs) for the commercial fishery were set individually for each of seven fishing Zones A to G from the 1986/7 season onwards (Tarr 1992). Moreover, data on the commercial catch of abalone in Zones A-D are available for the period 1977 to 1985. However, prior to 1972 catch data are available only for all Zones (A-G) combined, and hence zonal catch estimates for the period 1953 to 1976 are assumed to be a fixed proportion p_c of this total annual catch, where p_c is taken to be the average of a Zone's proportional contribution to total catch over the period 1977 to 1981. The same approach is used to apportion the Zone C catch estimates for the period 1953 to 1976 between the two subareas CP and CNP.

Recreational catch estimates (in terms of numbers of abalone caught) are estimated from telephonic surveys, conducted since 1992, of selected recreational permit-holders. In contrast to the commercial fishery, where a law in force since 1966 prohibits commercial abalone fishing operations within 185m of the high-water mark (Dichmont *et al.* 2000), the recreational fishery is essentially a shallow-water fishery with divers accessing the resource mostly from the shore. The recreational fishery was closed indefinitely in 2003 (DEAT 2003).

Available indices of abundance for each Zone include a GLM-standardised commercial CPUE series (from 1980 to present) and a Fishery Independent Abalone Survey (FIAS) conducted since 1995. These surveys were designed to provide an index of relative abundance with a CV of some 25% (which is substantially more precise than that achieved in earlier surveys) (Dichmont *et al*. 2000). Only animals larger than 100mm SL are recorded in these surveys so as to reduce uncertainty in the estimates due to the non-emergent/cryptic behaviour of juveniles. The data for Zone C (the commercial CPUE data in particular) exhibit more contrast than the data for the other Zones, suggesting better potential for precise parameter estimation. For Zones such as Zone A, data contrast is insufficient to allow independent estimation of parameters such as natural mortality with great reliability and hence these parameters have been estimated by jointly fitting across all zones.

There is a very large amount of catch-at-age data available per Zone from the various fishery sectors and research surveys. The catch-at-age data are derived from length distribution data that are cohort-sliced using a von Bertalanffy growth curve (from Tarr 1995) that was fitted to tagging data. The catch-at-age data used in the model-fitting process include data from the commercial (approximately 19 years per Zone), recreational (8 years) and poaching sectors (7 years) (the last from confiscations), as well as from the fishery-independent and industry surveys. In several instances age classes have been lumped together to reduce the number of categories containing a proportional abundance less than 2% in any one year.

An important additional data source derives from an industry and MCM co-operative diving survey in 2002 that was carried out *inter alia* to provide information on recruitment strength and population structure in key fishing Zones B and C. Additional data that are currently used in a diagnostic context relate to the extent of population depletion below pristine levels.

THE MODEL

The full details of the spatial age-structured production model (ASPM) are provided in Plagányi (2004) and Plagányi and Butterworth (2004a). A schematic summary of the model is shown in Fig. 3.

Spatial structure

Earlier analyses demonstrated (Plagányi *et al.* 2001, Plagányi 2004) that spatial structure was critical because inshore (within 185m of the highwater mark) and offshore components of the resource are affected differently by the different sectors of the fishery. Moreover, one of the problems in trying to model the abalone resource was the confounding of background natural mortality rate, mortality due to unknown levels of illegal take by the poaching sector and increased juvenile mortality due to the "lobster" effect in Zones C and D. This has been further complicated by the fact that the recruitment failure effect (starting in the early 1990's) and escalated poaching levels (from 1994 onwards) commenced at approximately the same time. Both processes have a large effect on the juvenile abalone age classes with the former mainly affecting ages 0 – 3 yr and the latter affecting age classes as young as 4 years. FIAS samples animals only 5 years and older, but an industry/MCM (Marine and Coastal Management) full population survey was conducted in Zones B and C in 2002, making it possible to discriminate between these effects. By fitting to Zones B (no recruitment failure effect) and C (with recruitment failure effect) simultaneously, it was possible to estimate the extent of the "lobster" effect. In 2003, a combined Zones A, B, C and D model was first attempted. The model simultaneously models the dynamics in each of ten regions (A, B, CNP, CP, D – inshore and offshore regions for each zone/subarea) (Fig. 3). The same natural mortality rate is assumed to apply across all regions. However, the western Zones (CNP, CP and D) are assumed subject to an ecosystem change effect in contrast to Zones A and B which are further to the east and have almost no rock lobsters present.

Incorporating the ecosystem change effect

The ecosystem change effect is not modelled explicitly in the assessment model but has been incorporated by allowing for an increase in the natural mortality rates of 0-yr old abalone in the affected zones (CNP, CP and D) in the model.

In general, age-dependent natural mortality rates M_a are modelled using the following formulation:

(1) $$M_a = \mu + \frac{\lambda}{a+1}$$

where parameter μ is estimated in the model-fitting process and λ is either estimated or set equal to a constant.

In addition, in CNP, CP and D, the model estimates two parameters to describe the rate and extent of the "ecosystem change" effect: a steepness of in increase in mortality parameter ν and a maximum increase in mortality parameter M_{max}. An exponential increase in the M_0 mortality rate is assumed to have occurred as from year y, where different values of the starting year y were tried and the rate of increase in M_0 is determined by parameter ν. M_0 is assumed to increase continuously up to a maximum value M_{max} and then remains constant at this value from years y_{Mmax} forwards.

The Confiscations-Per-Unit-of-Policing Effort (CPUPE)

Data on the numbers of confiscated abalone per Zone are jointly recorded by the South African police service and Marine and Coastal Management, and are available for all years since 1994. These data are used primarily to estimate the trend in poaching over time in each of the abalone fishery management Zones A-D. However, policing effort has not remained constant over time because, for example, in some years the government has provided additional resources in an effort to curb the escalating poaching levels. The poaching confiscation data are thus used to obtain base-case (no adjustment for changes in policing efficiency) estimates of the trend in poaching over time in each of Zones A-D, as well as to derive a somewhat unusual fisheries index – the CPUPE or Confiscations Per Unit of Policing Effort (Fig. 4).

The CPUPE is used as an input to the model and defines the year/s with the assumed highest level of poaching. The maximum poaching level per zone is estimated (in terms of numbers) within the model and the poaching estimates for the remaining years are then computed using the CPUPE trend information. The model estimates of poaching are scrutinised taking into account that the actual number of animals poached is some (unknown) multiple of the observed (i.e. unadjusted for policing efficiency) number of abalone confiscated. A minimum realistic poaching value is determined as the total number (location-known + proportion of location-unknown) of confiscations divided by the confiscation proportion. In instances where the model estimate of poaching is less than the minimum realistic value, the poaching level is fixed to the latter value or some multiple of it instead.

Parameters

The base-case combined ABCD model estimates the following 30 parameters:

1) Pre-exploitation spawning biomass B_0^{sp} for A, B, CNP, CP and D [5 parameters]
2) Inshore-offshore migration parameter ρ (CP) [1 parameter]
3) Inshore-offshore migration parameter ρ (A, B, CNP, D) [1 parameter]
4) Poaching estimate for the year with the assumed highest level of poaching for the zone in question: $CP_{\max}$ estimated for A, B, C (combined) and D. [4 parameters]
5) p_{poach} [1 parameter] – equates roughly to the assumption that 10% of the Zone C poaching take is from CNP and the remainder from CP

6) $M_a : \mu \quad (\lambda = 0.2)$ (see Equation 1). Natural mortality parameters μ and λ assumed common to all zones [1 parameter]
7) "Recruitment failure" parameters common to CNP, CP and D: ν and M_{max} [2 parameters]
8) Three parameters (μ, δ and $\tilde{a}$) for each of five selectivity functions (assumed common to all zones) [15 parameters]

One important aspect of sensitivity relates to the confiscation percentage constraint, and alternative scenarios are always presented in presenting management advice for this resource. In the 2004 base- case model for example, the confiscation percentage success rate when averaged over the past 5 years in Zones A-D is 23% (A: 26%; B: 39%; C: 18%; D:10%). This is similar to the confiscation percentage success rate estimated by policing operations (Marcel Kroese, pers. commn) and to estimates from an attempt to estimate compliance confiscation success rates from data from the NGO Traffic Hong Kong office (Mackenzie and Burgener 2004).

RESULTS

A few selected model results are given below based on assessments conducted in previous years. Note therefore that this information is not directly applicable to the 2005 assessment.

General trends

The observed decline in both catch rates and the fishery independent abundance index in recent years is particularly steep and it has been recognised for some time that continued depletion of the resource is inevitable unless the combined catch by all sectors can be drastically reduced. Considering that total catches in the 1960s were substantially greater than those taken in the 1970s and 1980s (Fig. 2) (this hardly seems likely not to have been the case, even though the details of the zonal-partitioning of catches assumed above could be in error to some extent), it is not too surprising that the CPUE trend shows an increase towards the end of the 1980s. This is the obvious explanation for the observed increase in CPUE estimates during the 1980s despite increases in overall catch levels over this period (Tarr 1993). It is to be expected from a relatively slow-growing long-lived resource afforded a respite as high and unsustainable historic catch levels are reduced substantially to below the then current sustainable yields.

In Zone C for example, if one assumes that the CPUE trend is a reasonable index of stock abundance, this suggests that whereas the Zone C commercial take during at least some of the 1970s and 1980s was below sustainable yield (SY) levels, total catches during the 1990s and early 2000's have again exceeded such yields, resulting in a concomitant decline in CPUE values over this most recent period (Fig. 6). This recent decline in CPUE is fully consistent with the catch data only if the latter are considered to include the considerable poaching component of the overall catches over recent years as estimated by the model. The estimated poaching catches are substantial in all of the scenarios investigated. Moreover, fitting simultaneously to full population survey data for Zones B (with poaching but without ecosystem-change effect) and C (with both poaching and the ecosystem-change effect) has suggested dramatic increases in juvenile mortality rates in Zone C (and Zone D by extrapolation) that has further exacerbated the recent observed declines in both CPUE and FIAS in these regions. In particular, earlier modelling attempts demonstrated an inability of the model to successfully simulate the trends in the Zone C FIAS data without taking into account the "ecosystem-change" effect.

Additional diagnostics

A useful diagnostic is provided by the pattern and scale of the fishing proportion F_y^s (Fig. 6). Note that the fishing proportion underlying these analyses represents the fished proportion of a fully selected age class rather than the more familiar annual fishing mortality rate referred to in fisheries stock assessment literature. The F_y^s values in Fig. 6 have been plotted on approximately the same scale for ease of comparison, and suggest that historically Zones B and CP have been the most heavily fished. Subarea CP is known as a favourite abalone diving spot because of the extended shallow regions. As expected, Zone A appears to have been the least heavily fished of the zones presumably because it is the zone furthest away from the main town in the region, Hermanus (Dichmont *et al.* 2000). This is reinforced by the relatively low recreational fishing proportion evident in Zone A compared to the substantial F_y^{rec} values for subarea CNP, which encompasses locations such as Hermanus and Vermont (Fig. 1). These areas are favoured holiday locations and hence it is to be expected that F_y^{rec} values are higher in these regions. In response to an ever-growing recreational catch during the 1990's, increasingly stringent limitations were imposed on recreational fishers (R. Tarr, MCM, pers. commn) and hence the contribution of the recreational sector to the total catch has gradually decreased and is currently zero by legislation.

Comparisons of time-trajectories of exploitation proportion by sector predicted by the model suggest that efforts to control the illegal poaching sector have largely been fruitless as F_y^{poa} has either increased or remained steady in all zones during the last few years (Fig. 6). In Zones A, B and D, the total fishing proportions as estimated by the model for recent years have exceeded even the initially high *F* values corresponding to the initial "mining out" of the abalone resource. Thus whereas average commercial sector *F* values range between 0.05 and 0.16 (as could be considered reasonable for a fairly productive long-lived resource), recent estimates of F_y^{poa} are 0.2 or higher, with a maximum of 0.64 in Zone B in 2002, which is clearly not sustainable in the medium to long term. Note however that the F values for the different sectors are not precisely comparable – although they all refer to a common age 11, they spread differently over age-classes and apply differentially to inshore and offshore areas.

Parameter estimates

Model results in 2003/4 suggested a pristine spawning biomass, B_0^{sp}, of 2012 [95% C.I.: 1780 ; 2400] and 4724 [4380 ; 5300] tonnes respectively for subareas CNP and CP, and hence a total Zone C spawning biomass of ca. 6740 tonnes. The difference in the pristine spawning biomass estimates $B_0^{sp}[CNP]$ and $B_0^{sp}[CP]$ are in the main due to the partitioning of the historic zone C catch data between the two subareas. The pristine spawning biomass estimates for the other zones are on a similar scale to the Zone C estimates, with 8030 [7030 ; 12 800], 5870 [5450 ; 6300] and 7460 [6800 ; 8950] tonnes estimated for Zones A, B and D respectively.

It is encouraging that a reasonable estimate of the pre-exploitation spawning biomass for Zone A was obtained as previous attempts (fitting a Zone A model in isolation) to estimate this parameter yielded unrealistically high values due to the uninformative nature of the data. The confidence interval for the Zone A pristine biomass estimate is much wider than for the other zones. The Zone A data provide a classical example of a so-called "one-way-trip-trajectory" (Hilborn and Walters 1992) – the CPUE data could derive from either a relatively unproductive population with a high pristine biomass or from a more productive population with a lower pristine biomass. One approach to refine these estimates would be to incorporate prior information on these parameters (K and M) to reduce the space of likely parameter combinations. The approach adopted here is analogous to some extent because the natural mortality parameter for Zone A is constrained to be the same as for the other zones (considered reasonable on biological grounds), thereby restricting the range of likely pristine biomass values. Thus although the Zone A fit is still viewed with some scepticism, it at least represents a step forward.

Mortality estimates and the "ecosystem-change" effect

A wide range of estimates of natural mortality (M) for abalone in the wild have been reported in the literature, ranging from about 0.05 to more than 1.00 yr^{-1} (Shepherd and Breen 1992, McShane and Naylor 1997). There is some (weak) evidence that M decreases with increasing age (Shepherd and Breen 1992). The base-case estimate of (age-dependent) M as ranging from 0.33 for 0-yr old abalone (when not subject to the "ecosystem-change" effect) to 0.14 for age 11 and older individuals is considered reasonable for a relatively long-lived species such as this. The likelihood profile estimate of the 95% CI for the mortality parameter μ (with estimate 0.127) is relatively tight [0.126 ; 0.142], suggesting that natural mortality is well estimated in the model fit.

In earlier model versions, estimates of mortality rates were confounded because, given the limited data available, it was not possible to discriminate between background mortality rates, mortality due to unknown levels of illegal take by the poaching sector and increased juvenile mortality due to the "lobster" effect in Zones C and D. This has been resolved to some extent by fitting to the 2002 industry/MCM full population surveys conducted in Zones B and C.

For "lobster" Zones C and D, the model parameter M_{max} sets the maximum mortality rate of 0-yr old abalone that is assumed to have occurred due to the ecosystem-change effect. The 2003/4 base-case model estimate of this parameter tended to hit whatever maximum constraint was set for the parameter (the base-case maximum constraint was 10 which corresponds to a near zero annual survival rate of 0-yr old abalone in Zones C and D). Based on the likelihood profile method, the lower 95% confidence limit for M_{max} is approximately M_{max} = 1.6, which implies an annual 0-yr old survival rate of 0.2, compared to the pre-1990 0-yr survival rate of 0.72.

Poaching estimates

The Zone C poaching estimate (for 1995) furnished by the 2003/4 base-case model was 556 000 [95% CI: 510 000 ; 695 000] abalone or 319 [267 ; 360] tonnes, which is some three times greater than the average Zone C commercial take during the 1990s and 20% greater than the 2003 commercial TAC for Zones A-D combined. Whereas towards the end of the 1990s poaching estimates were equal to or slightly less than the commercial TAC, the recent explosion of poaching activities has resulted in a total catch for Zones A-D combined which is more than seven times the legal 2003 commercial TAC for these zones. These estimates are phenomenally high but remain quite plausible considering that they correspond to the assumption that, on average, 23% - 36% of all poached abalone are confiscated.

Biomass trajectories

Fig. 6 shows spawning and exploitable biomass trajectories relative to total catches in each zone. Total catches include recorded or model-estimated takes by the commercial, recreational and poaching sectors combined. Note that the “exploitable” biomass trajectories shown are those corresponding to the resource component harvested by the commercial sector.

The model estimates an initial steep decline in the spawning biomass of abalone in all zones, as a result of the high historic exploitation levels in the 1960s. A slight recovery of the stock level is estimated to have occurred during the 1980s, followed by a relatively stable period and then an appreciable downward trend in recent years (Fig. 6). Consistent with the incidence of poaching having commenced earliest (around 1994) in Zones C and D (also subject to the ecosystem-change effect), the recent downward trends in these zones are particularly marked. Given that the CPUE index effectively relates only to animals of ages nine and older, there is a time lag before the negative effects of poaching become evident. The inshore FIAS survey data highlight more clearly the substantial recent decreases in the numbers of small abalone in all zones. Given the long-lived nature of abalone, these decreases are particularly evident when projecting forward in time.

The inshore region is particularly depleted, with a recent estimate for CP of only 12 %. Zones B and D are estimated to have similar current levels of depletion, with the current depletion for Zone A being estimated as somewhat higher. It is to be expected that abundance in Zone A should be slightly less depleted than the other zones as it is the least accessible (in terms of being furthest from human settlements) of the main fishing zones and poaching is thought to have commenced later in this zone than in the other zones.

DISCUSSION

A description is given of the stock assessment approach used for the South African *H. midae* resource to estimate the extent of the illegal take and the reduction in juvenile abalone survival due to an ecosystem shift (in the form of a movement of rock lobster into a major part of the range of the abalone). This resource is subject to ever-increasing problems as a result of these two factors. The current modelling methods are critical for highlighting the extent of the problem and evaluating the consequences of possible changes in harvest regime, but do not represent a management solution (Plagányi 2004). As stressed by Orensanz *et al*. (in press) and Parma *et al.* (2003), sustainability of a fishery is likely to succeed only when the right incentives are provided, such as in the form of secure long-term access rights. The actual solution in this case has been seen to reside in the development of alternative management approaches that better include local communities. In recognition of this, a new policy to define the process of allocating commercial abalone fishing rights was announced by MCM in October 2003. The new management model is based on a Territorial User Rights in Fisheries (TURF) system (Stephenson & Lane 1995, Christy 1996, Caddy 1999).

The severe overexploitation of many Latin American benthic shellfish stocks has been attributed largely to the absence of co-management practices (Castilla and Defeo 2001) and the allocation of TURF's has achieved some success in improving both the quality of the regulatory process and the status of the shell-fisheries in these regions (Castilla *et al.* 1998, Castilla and Defeo 2001, Parma *et al*. 2003). Plagányi and Butterworth (in review) argue that despite being associated with a management regime that has failed to reconcile fisheries with conservation in this instance, in retrospect, there appear to have been clear economic and biological advantages to having continued these classic stock assessment exercises.

Detailed modelling of the abalone – rock lobster – urchin multi-species interactions is complex and not immediately feasible although some progress has been made (Plagányi 2004). However, given pressures to increase rock lobster quotas in the EoH region, an immediate priority relates to gaining an improved understanding of the trade-offs involved in harvesting rock lobster heavily in this region with the aim (in theory at least) of allowing some recovery of the abalone resource. Given the paucity of available data and lack of full ecosystem understanding, it is debatable whether a detailed ecosystem approach to this problem will yield practically meaningful conclusions. The complexity of these interactions is also not easily accommodated within the relatively rigid structure of preset models such as ECOPATH with ECOSIM (Walters *et al*. 1997, see also Plagányi and Butterworth 2004b for a summary of the potential advantages but also the problems of applying the EwE approach). The best approach to shedding further light on the multi-species interactions would likely depend on experimental studies and an adaptive management approach (e.g. Walters 1986, Hilborn and Walters 1992, Sainsbury *et al.* 2000) rather than simply on modelling results. For example, an actively adaptive management strategy applied to the Australian multi-species fishery was successful in resolving key uncertainties about resource dynamics and sustainable resource use (Sainsbury *et al.* 1997). The approach involved identifying four different plausible hypotheses and adopting an experimental process involving the sequential closure of areas to trawl fishing. After a period of a few years, the experiment was successful in discriminating among the competing hypotheses (Sainsbury *et al.* 1997, 2000).

Ideally, an Operational Management Procedure (OMP) (Butterworth *et al*. 1997, de Oliveira *et al.* 1998, Butterworth and Punt 1999) needs to be developed for the abalone resource in the main fishery Zones A-D. As a first step, the population model described in this paper could be used as the operating model for the underlying dynamics. Decision models would then need to be developed to take account of three critical factors:

a) the recent trend in poaching in each secondary zone (or TURF);

b) the recent trend in CPUE and survey indices in each secondary zone, as determined from finer spatial scale data than that input to the operating model; and

c) an assessment of the impact of multi-species interactions.

The last of these could be based on any or several of the following:

i) Data on abalone recruitment success from a dedicated recruitment survey or from a full population survey with coverage in at least one lobster-invaded and one "lobster-free" zone (as was the case for the 2002 MCM/Industry survey).

ii) Information on the EoH proportion of the rock lobster TAC, in the event that it can be demonstrated that sufficient numbers of rock lobsters have been harvested to allow some recovery of the abalone resource. This relates to item 1. above – note also that this would become relevant only in a few years time given the time-scale needed for a noticeable recovery.

iii) Information from models of abalone - rock lobster – urchin interactions. These could either be relatively simple models or more complicated whole ecosystem models.

Indications from these models of a short-term enhancement or reduction of the "rock lobster" effect could be fed into a decision model, provided such multi-species / ecosystem models are carefully parameterised and have demonstrated sufficient robustness of their conclusions to uncertainty in the data as well as to a range of plausible alternative hypotheses. In the case of abalone, the development of a tactical ecosystem model as the basis for computing harvest limits within an OMP itself would seem to be a very long way off.

ACKNOWLEDGEMENTS

Financial support from Marine and Coastal Management (MCM), South Africa, is gratefully acknowledged. Thanks to Rob Tarr and Angus MacKenzie for assisting with access to and analysis of the data.

REFERENCES

BENEFIT 2002. Report of the BENEFIT stock assessment workshop, University of Cape Town, 10-15 December 2002. BENEFIT Programme, Swakopmund. 44 pp.

Breen, P.A. 1986. Management of the British Columbia fishery for northern abalone (Haliotis kamschatkana). Can. Spec. Publ. Fish. Aquat. Sci. **92**: 300-312.

Breen, P.A. 1992. A review of models used for stock assessment in abalone fisheries. *In* Abalone of the World: Biology, Fisheries and Culture. *Edited by* S.A. Shepherd, M.J. Tegner, and S.A. Guzmán del Próo. Fishing News Books, Oxford: pp. 253-275.

Butterworth, D.S. and É.E. Plagányi 2000. Exploratory analyses of Southern bluefin tuna dynamics using production models. CCSBT Document: CCSBT/SWG/0011/16: 28 pp.

Butterworth, D.S. and R.A. Rademeyer. (In press). Sustainable management initiatives for the southern African hake fisheries over recent years. Bull. Mar. Sci.

Butterworth, D.S., Rademeyer, R.A. and É.E. Plagányi 2003a. An age-structured production model based assessment and reference point evaluation for the Gulf of Maine cod stock. United States Northeast Fisheries Management Council document, February, 2003. 41 pp.

Butterworth D.S., Rademeyer R.A. and É.E. Plagányi 2003b. ASPM-based assessment of the Gulf of Maine cod stock: A comparison with ADAPT-VPA and extension to a Bayesian form. United States Northeast Fisheries Management Council document, July 2003. 25 pp

Caddy, J.F. 1999. Fisheries management in the twenty-first century: will new paradigms apply? Rev. Fish Biol. Fisheries **9**: 1-43.

Castilla, J.C., Manríquez, P., Alvarado, J., Rosson, A., Pino, C., Espoz, C., Soto, R., Oliva, D. and O. Defeo 1998. The artisanal caletas as unit of production and basis for community-based management of benthic invertebrates in Chile. Can. Spec. Publ. Fish. Aquat. Sci. **125**: 407-413.

Castilla, J.C. and O. Defeo 2001. Latin American benthic shellfisheries: emphasis on co-management and experimental practices. Rev. Fish Biol. Fisheries **11**: 1-30.

Christy, F.T. 1996. The death rattle of open access and the advent of property rights regimes in fisheries. Mar. Res. Econ. **11**: 287-304.

Clarke, S. 2004. Understanding pressures on fishery resources through trade statistics: a pilot study of four products in the Chinese dried seafood market. Fish and Fisheries **5**: 53-74.

Day, E. and G.M. Branch 2002. Effects of sea urchins (*Parechinus angulosus*) on recruits and juveniles of abalone (*Haliotis midae*). Ecol. Monogr. **72**: 133-149.

Department of Environment and Tourism [DEAT] 2003. Policy for the allocation of commercial fishing rights in the abalone fishery: 2003. 23 pp. (Available from website www.environment.gov.za).

Dichmont, C.M., Butterworth, D.S. and K.L. Cochrane 2000. Towards adaptive approaches to management of the South African abalone (*Haliotis midae*) fishery. S. Afr. J. Mar. Sci. **22**: 33-42.

Gastrow, P. 1999. Main trends in the development of South Africa's organised crime. Afr. Security Rev. 8(6): 4pp.

Gorfine, H.K., Taylor, B.L. and T.I. Walker 2001. Triggers and targets: what are we aiming for with abalone fisheries models in Australia? J. Shellfish Res. **20**: 803-811.

Hauck, M. 1997. Crime, Conservation and Community Development: Ecological Criminology and the Case Study of Abalone Poaching. M.Sc. thesis, University of Cape Town. 285 pp.

Hauck, M. 1999a. Regulating marine resources in South Africa: a case study of abalone poaching. Acta Juridica. 1999: 211-228.

Hauck, M. 1999b. Seawatch: Civil Society's Role in Combating Environmental Crime. S. Afr. J. Environmental Law and Policy **6**: 101-119.

Hauck, M., and N. Sweijd 1999. A case study of abalone poaching in South Africa and its impact on fisheries management. ICES J. Mar. Sci. **56**: 1024-1032.

Hilborn R. 1990. Estimating the parameters of full age-structured models from catch and abundance data. Bull. Int. N. Pac. Fish. Commn **50**: 207-213.

Hilborn, R. and C.J. Walters 1992. Quantitative Fisheries Stock Assessment: Choice, Dynamics and Uncertainty. Chapman & Hall, New York.

Johnston, S.J. and D.S. Butterworth. (In press). The evolution of Operational Management Procedures for the South African West Coast rock lobster. New Zealand Journal of Marine and Freshwater Research

MacKenzie, A.J. and M. Burgener. 2004. A first attempt at trying to estimate compliance confiscation success rates within Southern Africa. Marine & Coastal Management, South Africa, Internal Abalone Working Group report, WG/AB/04/08/08.

Mayfield, S. and G.M. Branch 2000. Interrelations among rock lobsters, sea urchins, and juvenile abalone: implications for community management. Can. J. Fish. Aquat. Sci. **57**: 2175-2185

McShane, P.E. 1992. Early life history of abalone: a review. *In* Abalone of the World: Biology, Fisheries and Culture. *Edited by* S.A. Shepherd, M.J. Tegner, and S.A. Guzmán del Próo. Fishing News Books, Oxford: pp. 120-138.

McShane, P.E. and J.R. Naylor 1997. Direct estimation of natural mortality of the New Zealand abalone (*Haliotis iris*). New Zeal. J. Mar. Fresh. Res. **31**: 135-137.

Newman, G.G. 1973. Assessment and management of some marine stocks in southern Africa. Ph.D. thesis, University of Cape Town. 268 pp.

OLRAC cc 2002. Survey design considerations for a proposed full population structure survey to be conducted jointly between industry and the M&CM. Marine & Coastal Management, South Africa, Internal Abalone Working Group Report, WG/AB/02/03/01.

Orensanz, J.M., Parma, A.M., Jerez, G., Barahona, N., Montecinos, M. and I. Elias. (In press). What are the key elements for the sustainability of "S-Fisheries"? Insights from South America. *In* Proceedings of the conference on the scientific bases for the sustainability of fisheries. *Edited by* N. Erhardt. Miami, November 26-30, 2001.

Parma, A.M., Orensanz, J.M., Elías, I. and G. Jerez. 2003. Diving for shellfish and data: incentives for the participation of fishers in the monitoring and management of artisanal fisheries around southern South America. *In* Towards Sustainability of Data-Limited Multi-Sector Fisheries. *Edited by* S.J. Newman, D.J. Gaughan, G. Jackson, M.C. Mackie, B. Molony, J. St. John, and P. Kailola. Australian Society for Fish Biology Workshop Proceedings, Bunbury, Western Australia, 23-24 September 2001. Fisheries Occasional Publications No. 6, May 2003, Department of Fisheries, Perth, Western Australia, 186 pp.

Plagányi, E.E. 2004. Walking the bridge from single- to multi-species approaches in southern African fisheries management. PhD thesis, University of Cape Town , 460 pp.

Plagányi, É.E., Butterworth, D.S. and A. Brandão 2001. Towards assessing the South African abalone (*Haliotis midae*) stock using an age-structured production model. J. Shellfish Res. **20**: 813-827.

Plagányi, É.E. & D.S. Butterworth 2004a. Preliminary results from the abalone ASPM used to simultaneously represent abalone resource dynamics in Zones A, B, C and D in 2004 WG/AB/04/08/06.

Plagányi, É.E. & D.S. Butterworth 2004b. A critical look at the potential of Ecopath with Ecosim to assist in practical fisheries management. *In* Ecosystem Approaches to Fisheries in the Southern Benguela. Shannon, L.J., Cochrane, K.L. and S.C. Pillar (Eds). *Afr. J. Mar. Sci.* 26: 261-287.

Prince, J.D. 1992. Using a spatial model to explore the dynamics of an exploited stock of the abalone (*Haliotis rubra*). *In* Abalone of the World: Biology, Fisheries and Culture. *Edited by* S.A. Shepherd, M.J. Tegner, and S.A. Guzmán del Próo. Fishing News Books, Oxford.

Prince, J.D. and S.A. Guzmán del Próo 1992. A stock reduction analysis of the Mexican abalone (Haliotid) fishery. Fish Res. **16**: 25-49.

Sainsbury, K.J. 1991. Application of an experimental management approach to management of a tropical multispecies fishery with highly uncertain dynamics. ICES Mar. Sci. Symp. **193**: 301–320.

Sainsbury, K.J., Campbell, R.A., Lindholm, R. and Whitelaw, W. 1997. Experimental management of an Australian multispecies fishery: examining the possibility of trawl-induced habitat modification. *In* Global Trends: Fisheries Management. Edited by E.L. Pikitch, D.D. Huppert, and M.P. Sissenwine. American Fisheries Society Symposium **20**: 107-112. Bethesda, Maryland, USA

Shepherd, S.A., Tegner, M.J. and S.A. Guzmán del Próo (Eds). 1992. Abalone of the World: Biology, Fisheries and Culture. Fishing News Books, Oxford.

Tarbath, D., Mundy, C. and M. Haddon 2003. Fishery assessment report: Tasmanian abalone fishery 2002. Tasmanian Aquaculture & Fisheries Institute, University of Tasmania. 144 pp.

Tarr, R.J.Q. 1992. The abalone fishery of South Africa. *In* Abalone of the World: Biology, Fisheries and Culture. *Edited by* S.A. Shepherd, M.J. Tegner, and S.A. Guzmán del Próo. Fishing News Books, Oxford: pp. 438-447.

Tarr, R.J.Q. 1993. Stock Assessment, and Aspects of the Biology of the South African Abalone (*Haliotis midae*). M.Sc. Thesis, University of Cape Town. VIII+156 pp.

Tarr, R.J.Q. 1995. Growth and movement of the South African abalone (*Haliotis midae*): a reassessment. Mar. Freshwater Res. **46**: 583-590.

Tarr, R.J.Q. and A.J. MacKenzie 2002. The South African abalone fishery: biology, management and poaching. Paper BEN/DEC02/SAA/1a presented to the BENEFIT Stock Assessment Workshop, Cape Town, December 2002.

Tarr, R.J.Q., Williams, P.V.G. and A.J. MacKenzie 1996. Abalone, sea urchins and rock lobster: A possible ecological shift may affect traditional fisheries. S. Afr. J. Mar. Sci. **17**: 311-315.

Tegner, M.J. 1993. Southern California abalones: can stocks be rebuilt using marine harvest refugia? Can. J. Fish. Aquat. Sci. **50**: 2010-2018.

Tegner, M.J., Haaker, P.L., Riser, K.L. and L.I. Vilchis 2001. Climate variability, kelp forests, and the Southern California red abalone fishery. J. Shellfish Res. **20**: 755-763.

Worthington, D.G. and N.L. Andrew 1998. Small-scale variation in demography and its implications for an alternative size limit in the fishery for blacklip abalone (*Haliotis rubra*) in New South Wales, Australia. *In* Proceedings of the North Pacific Symposium on Invertebrate Stock Assessment and Management. *Edited by* G.S. Jamieson, and A. Campbell. Can. Spec. Publ. Fish. Aquat. Sci. **125**: 341-348.

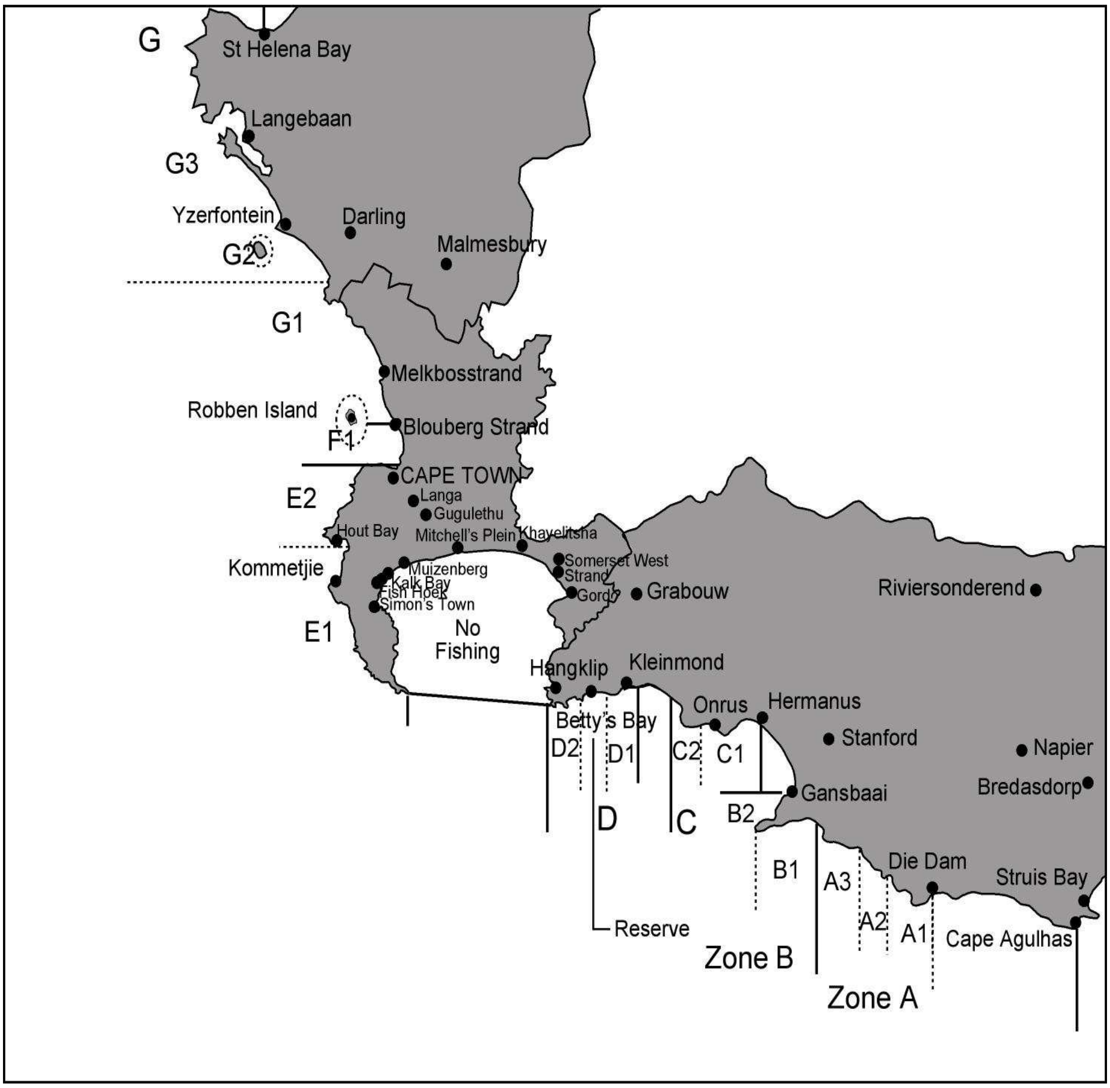

Fig. 1. Map (from DEAT 2003) showing the possible division of primary abalone fishing zones into secondary zones as part of the implementation of the new management policy. Zones A, B and D are currently assessed at the level of primary zone only. The secondary zones C1 and C2 of Zone C correspond to subareas CNP and CP which are differentiated in the current models.

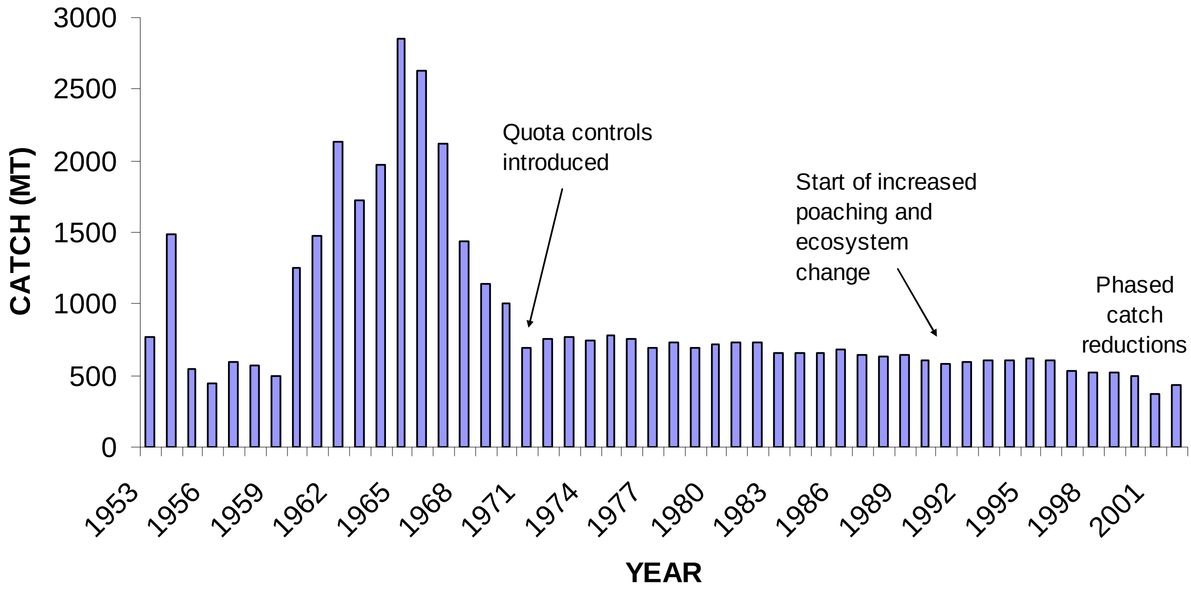

Fig. 2. Total commercial catches (whole wet mass in MT) of abalone *Haliotis midae* from 1953 to 2003.

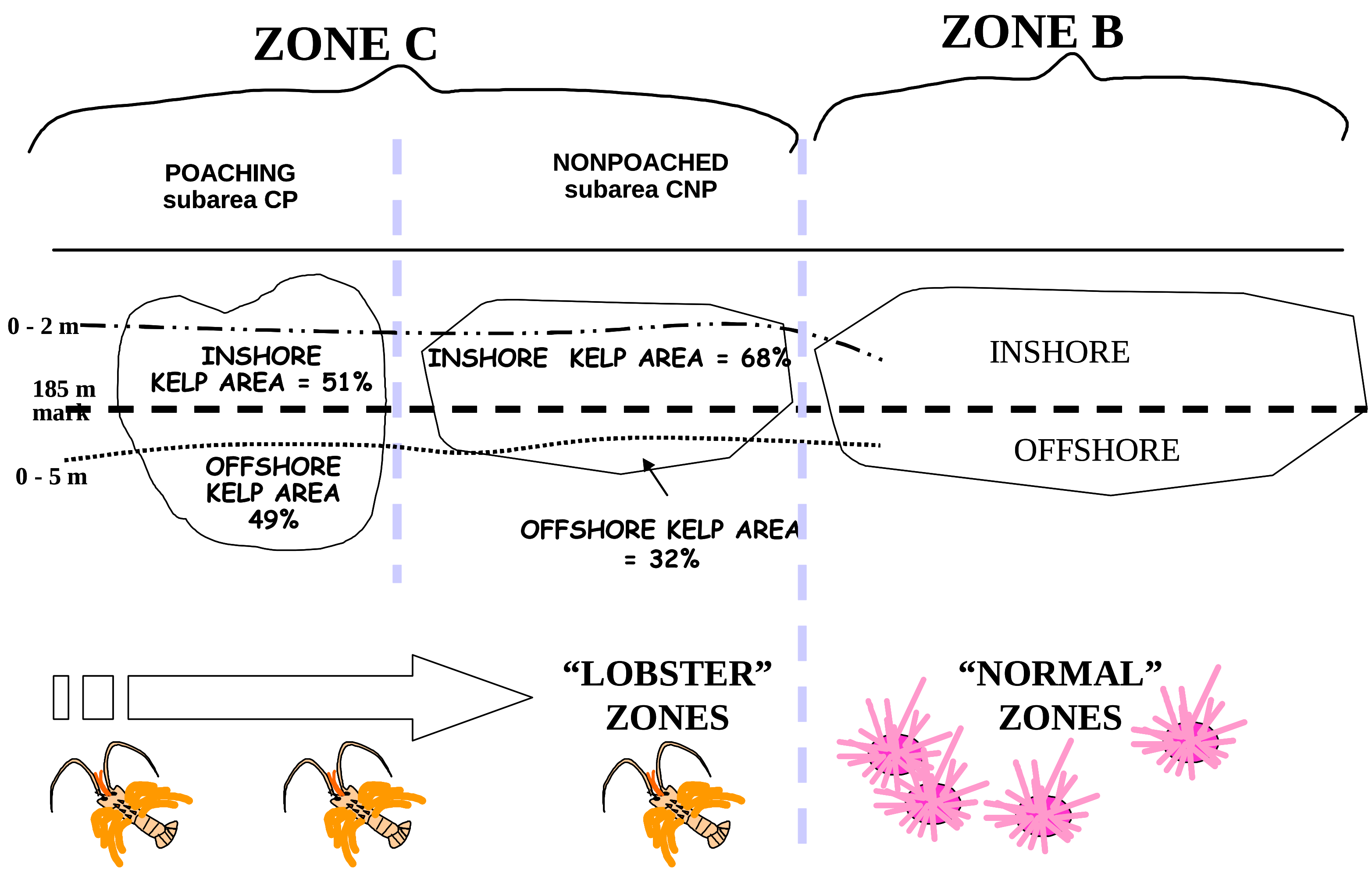

Fig. 3. Schematic summary of the structure of the combined Zones B&C abalone model. This model has subsequently been extended to a combined zones ABCD model by including "lobster"zone D to the west of Zone C and "normal"zone A to the east of Zone B, and simultaneously fitting to all these zones.

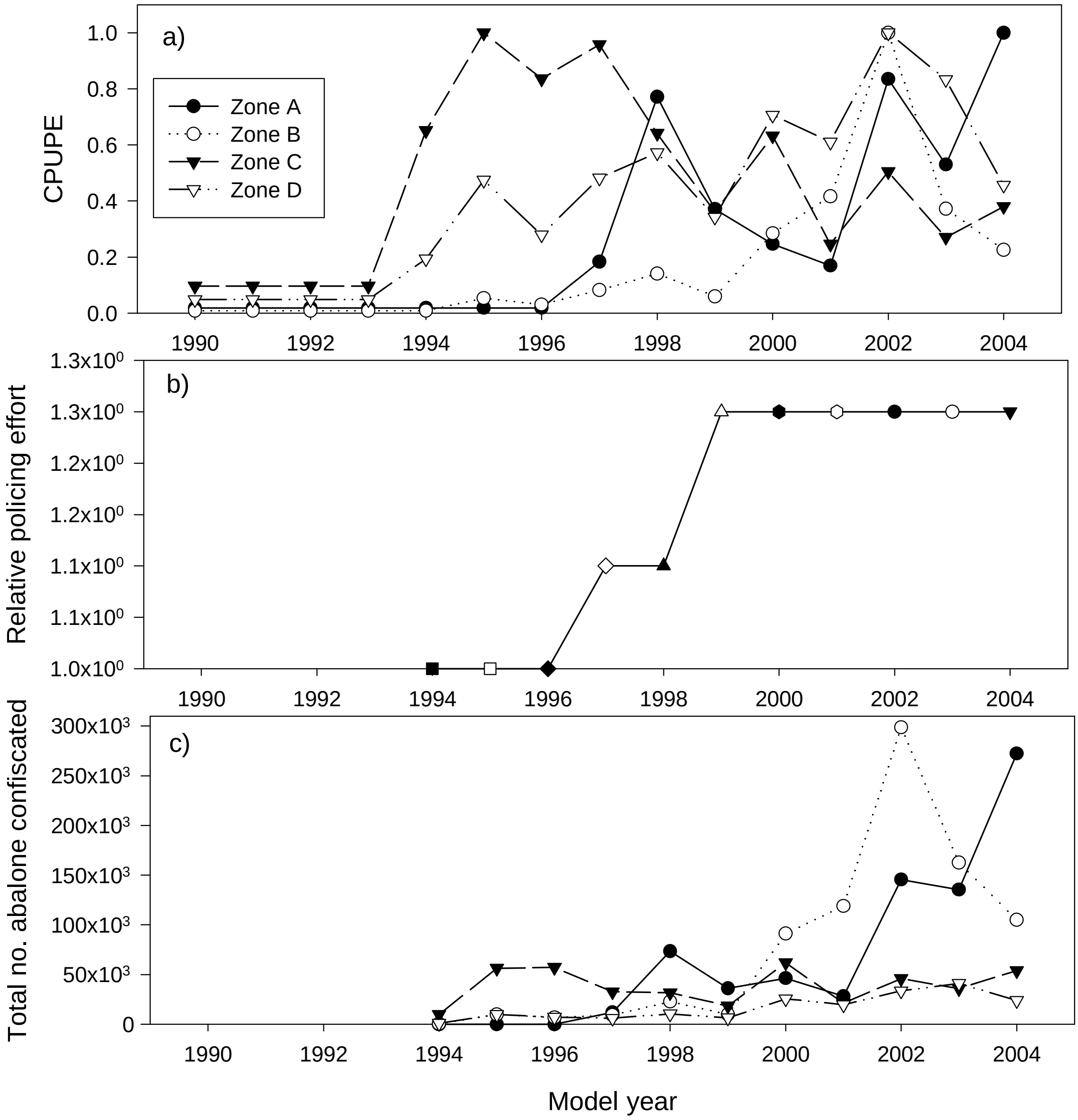

Fig. 4. a) Time series representing the Confiscations-Per-Unit-Policing-Effort (CPUPE) for each of the main abalone fishery Zones A-D. The numbers of abalone confiscated and the "policing efficiency levels" (shown in b) are based on information from law enforcement officers in South Africa from 1994 to present. The proportions represent the poaching intensity in that Zone relative to the maximum poaching level observed for that Zone. A linear increase in poaching is assumed to have occurred from zero in 1990 in Zone C to the 1994 level, and from zero in 1991 to the 1994 level in Zone D. For Zone C, the same pattern of poaching is assumed to apply to subareas CNP and CP. c) The total estimated number of abalone confiscated per Zone. The total number of confiscations is computed as the sum of location-known confiscations and a proportion of contributions from a “Undefined zone” category. These total confiscation values are used in the model to set the minimum number of poached animals that must have been taken from a particular zone in a particular year.

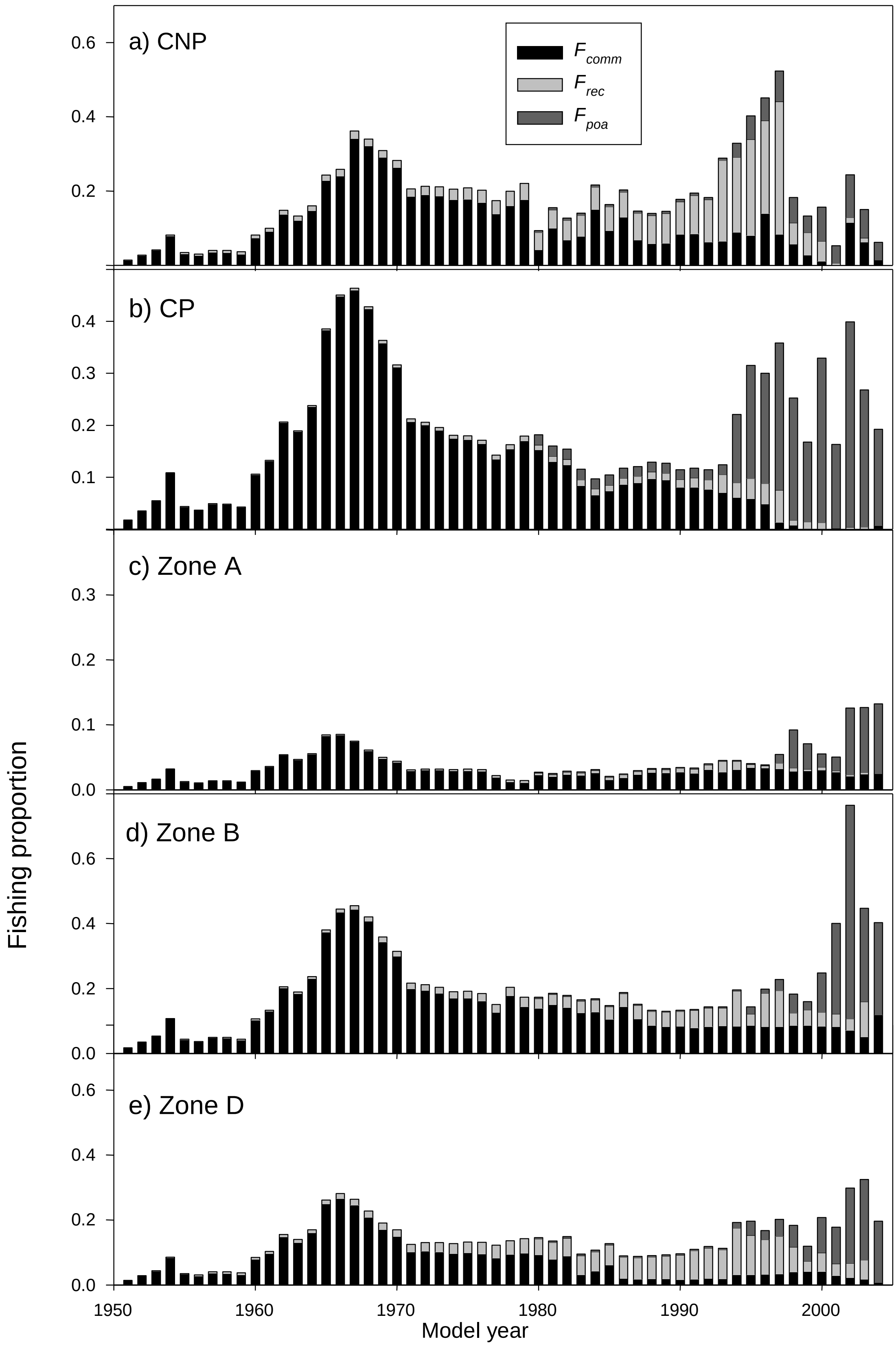

Fig. 5 Changes in the fishing proportion *F* over time for each of a) subarea CNP, b) subarea CP, c) Zone A, d) Zone B and e) Zone D. The figures show the contribution to the total fishing proportion by the three sectors: commercial (comm.), recreational (rec) and poaching (poa) for the base-case combined ABCD model.

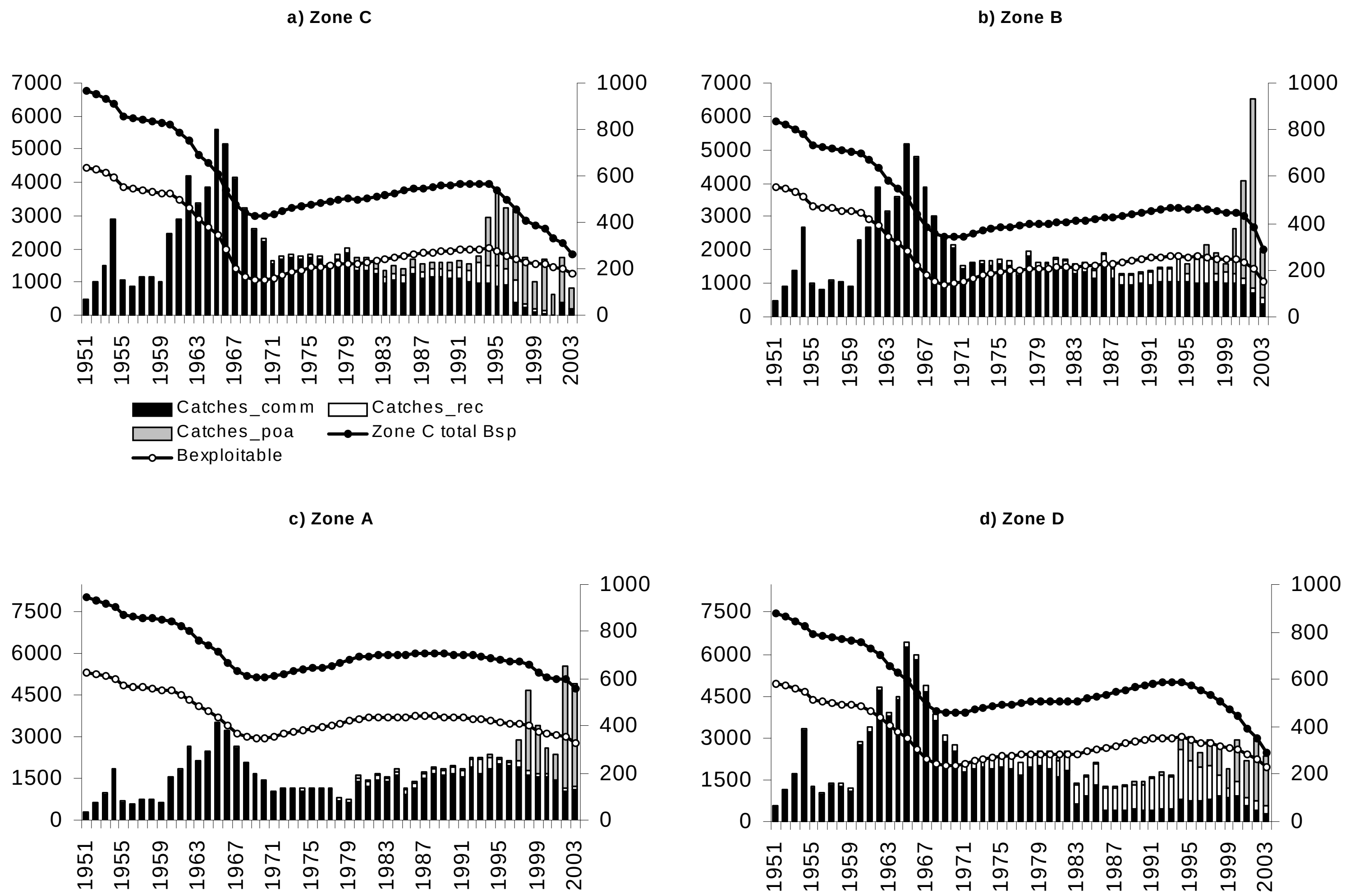

Fig. 6. Spawning biomass and commercial exploitable biomass on the left hand axis (as estimated for the base-case combined ABCD model) are shown relative to total catches (commercial + poaching + recreational) as plotted on the right hand vertical axis, for each of a) Zone C, b) Zone B, c) Zone A and d) Zone D.