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# Assessment of South African *Umbrina robinsoni* based on per-recruit models

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Slender baardman *Umbrina robinsoni* are an important component of recreational shore-angler and spearfisher catches along the eastern seaboard of South Africa. Stocks of *U. robinsoni* at three sites — False Bay, Stil Bay and the KwaZulu-Natal (KZN) coast — were modelled using a per-recruit approach. Total ( $Z$ ) and fishing ( $F$ ) mortality rates were estimated by catch-curve analyses using measures of individual size (length or weight) recorded by researchers, divers (log books) or during spearfishing competitions. Based on estimates of  $F$  during the period 2001–2003, spawner biomass per-recruit ratios were estimated to be either at or below the 25% threshold (False Bay  $SB/R = 21\% SB/R_{F=0}$ , Stil Bay  $SB/R = 25\% SB/R_{F=0}$ , and KZN  $SB/R = 21\% SB/R_{F=0}$ ), suggesting that rates of  $F$  were too high. Reductions in  $F$  necessary to achieve target fishing mortality levels ( $F_{SB40}$ ) at the current minimum size limit ( $I_c$  40 cm total length) were 51% for Stil Bay and the KZN coast and 57% for False Bay. Based on the bag frequencies from 927 diver outings in KZN (1989–2003), a reduction in bag limit from the current five to two fish is predicted to reduce  $F$  in this region by approximately 25%. Increasing the  $I_c$  to 50 cm is predicted to increase  $SB/R$  ratios to 36%  $SB/R_{F=0}$  in False Bay, 43%  $SB/R_{F=0}$  in KZN and 52%  $SB/R_{F=0}$  in Stil Bay, at current levels of  $F$ . Owing to the philopatric nature of *U. robinsoni* and the consequent existence of temporary refugia, catch curves are likely to underestimate fishing mortality. The reductions in  $F$  estimated to attain the target reference points are therefore probably conservative.

**Keywords:** fishing mortality, growth rate, management, Sciaenidae, stock assessment

## Introduction

The slender baardman *Umbrina robinsoni* is a moderately large (maximum mass 12 kg, total length approx. 100 cm) sciaenid that occurs in inshore waters (<45 m depth) along the east coast of Africa, from Cape Point to Oman (Hutchings and Griffiths 2005). The species was previously misidentified as *U. ronchus* throughout its range, and in the Western and Eastern Cape provinces of South Africa frequently confused with the sympatric *U. canariensis* which inhabits deeper shelf waters (Hutchings and Griffiths 2005).

The first reference to the fishery for *U. robinsoni* is found in Biden (1930) where he records that the older Kalk Bay (a fishing harbour in False Bay, Western Cape) fishers were complaining about the almost total disappearance of pufaro *Polyprion americanus* and baardman *Sciaena capensis* (a synonym for *U. robinsoni*) from catches. This anecdotal evidence suggests that *U. robinsoni* were historically a component of the boat-based linefishery in the region, whereas they undoubtedly were also landed by beach-seines (South Africa's oldest commercial fishery that dates back to the 1600s) operating along South Africa's eastern

seaboard. *U. robinsoni* still occur as an occasional bycatch in beach-seines operating along the northern shore of False Bay (Lamberth et al. 1994) and elsewhere along the Cape coast (Marine and Coastal Management, unpublished data).

Although recreational shore-angling has been a popular pastime for at least the past century, *U. robinsoni* only became important in catches relatively recently (1960s) when private ownership of four-wheel drive vehicles and technological improvements in angling gear (such as the introduction of the prawn pump, geared reels, monofilament nylon and fibreglass rods) allowed effective targeting of species that feed in the surf zones of sandy beaches (Bennett 1991, Bennett et al. 1994). Analyses of the records of three angling clubs active in the False Bay region revealed that *U. robinsoni* were the fifth most often caught species over the period 1938–1992 (Bennett et al. 1994). Over the past decade, however, the species has become increasingly scarce in shore-angler catches (Brouwer et al. 1997). Recreational spearfishers also target *U. robinsoni* throughout the species' range in South African waters and

the most recent survey revealed that the species accounts for between 3% and 10% of the total catch (Mann et al. 1997). Despite a relatively long history of exploitation, there have been no previous attempts to assess the stock status of *U. robinsoni*; primarily due to a lack of scientific information on the taxonomy and biology of the species. Current management measures, largely based on perceived vulnerability to exploitation, comprise a minimum size limit of 40 cm total length and a bag limit of five fish per angler per day (RSA 1998).

Marine and Coastal Management (MCM), the South African living marine resources management authority<sup>1</sup>, approved a Linefish Management Protocol (LMP) during 1998–1999 (Griffiths et al. 1999). The LMP was derived from the concept of an Operational Management Procedure (OMP) as applied to large, high-value fisheries internationally, e.g. whale stocks (Kirkwood 1997), South African hake (Punt 1992) and pelagic fisheries (Cochrane et al. 1998). South Africa's LMP, however, recognises the general data deficiency for most species in South Africa's multispecies linefish resource and accounts for risk (associated with management decisions) through the use of biological reference points rather than simulation testing (Griffiths et al. 1999). Although recent improvements in data collection (partly achieved via the implementation of a shore-based observer programme) will permit the use of more sophisticated age-structured production stock assessment models for the more common commercial linefish species in the near future, this is still not possible for scarce species such as *U. robinsoni* (CG Wilke and SE Kerwath, Department of Agriculture, Forestry and Fisheries [DAFF], pers. comm.).

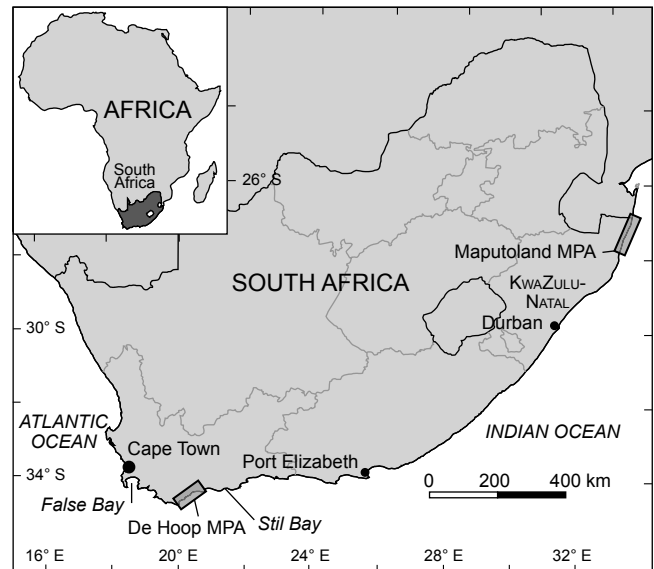
As with most South African linefish species, catch-and-effort data for the *U. robinsoni* fishery is scarce, with the little available data specific to certain time periods and/or user groups. For fisheries without accurate long-term catch-and-effort data or information on the stock-recruitment relationship, spawner biomass per-recruit (SB/R) and yield per-recruit (Y/R) models are considered the most suitable stock assessment methods (Butterworth et al. 1989, Punt 1993, Griffiths 1997, Griffiths et al. 1999).

Recent studies have demonstrated limited coastwise movement (Hutchings 2005) and substantial life-history differences (growth rate, size-at-maturity and longevity) between spatially separate populations of *U. robinsoni* (Hutchings and Griffiths 2010). In this paper, the minimum size limit for *U. robinsoni* and current rates of fishing mortality in three regions — False Bay, Stil Bay and the KwaZulu-Natal (KZN) coast (Figure 1) — are evaluated using per-recruit models.

## Material and methods

### Biological data

Basic data used in the per-recruit analyses were obtained by biological sampling *U. robinsoni* from four sites off South



**Figure 1:** Map of South Africa showing sampling sites and places mentioned in the text

Africa's eastern seaboard during the period 2001–2004. Two of the sites, De Hoop and Kosi Bay, are established marine protected areas (MPAs), whereas *U. robinsoni* at the other two sites, False Bay and Stil Bay, are exposed to intensive recreational angler and/or spearfishing exploitation (Figure 1). A collection of otoliths and total length measurements of False Bay *U. robinsoni* collected during 1991–1993 (SJ Lamberth, DAFF, unpublished data) were also used to determine growth during this period. For further details on biological sampling, determination of size-at-maturity, age determination and growth curve fitting, see Hutchings and Griffiths (2010). Sample size, length–mass relationship, female size-at-maturity and maximum recorded age for each site are shown in Table 1. The estimated von Bertalanffy growth model parameters are given in Table 2. Observed length-at-age data were used to construct age–length keys for *U. robinsoni* sampled from each of the different regions or time periods.

### Mortality

#### Natural mortality estimates

In Hutchings and Griffiths (2010), natural mortality ( $M$ ) for *U. robinsoni* was estimated from the age structure of unexploited populations in large, longstanding marine protected areas. Values of  $0.26 \text{ y}^{-1}$  (CV = 3.6%) and  $0.35 \text{ y}^{-1}$  (CV = 4.8%) were obtained for warm temperate (De Hoop) and subtropical (Kosi Bay) regions respectively. The subtropical estimate was assumed for KZN and the warm-temperate estimate for Stil Bay. False Bay temperatures are substantially cooler than at De Hoop and the *U. robinsoni* from this region appear to live longer and grow faster than those in the latter region (Hutchings and Griffiths 2010). The  $M$  value for False Bay was therefore taken as the average of the De Hoop estimate and the value ( $0.24 \text{ y}^{-1}$ ) calculated using Hoenig's (1983) formula using the maximum

<sup>1</sup> Marine and Coastal Management of the Department of Environmental Affairs and Tourism, recently transferred and renamed Branch Fisheries of the Department of Agriculture, Forestry and Fisheries, is the section of the Government Department that carries responsibility for fisheries management

**Table 1:** Details of biological data for sampled *U. robinsoni* populations used in per-recruit analysis. Sample size ( $n$ ) is the total number of fish that were biologically sampled in each region;  $l_m$  is the estimated female size at 50% sexual maturity;  $t_{max}$  is the maximum recorded age in the sample;  $a$  and  $b$  are the parameters in the total length (TL, mm)–mass (g) relationship

Region	$n$	$l_m$ (cm, TL)	$t_{max}$ (y)	$a$	$b$
False Bay	175	40	18	$10^{-4.94}$	3.02
De Hoop	312	39	16	$10^{-5.16}$	3.10
Stil Bay	74	39	21	$10^{-5.22}$	3.10
KZN	354	48	12	$10^{-4.84}$	2.94

**Table 2:** Von Bertalanffy growth model parameter estimates for *U. robinsoni* used in per-recruit models. Standard errors (SE), coefficients of variation (CV), upper (UCI) and lower (LCI) 95% confidence limits are given. Sample size ( $n$ ) indicates the number of fish used in growth curve fitting

Parameter	False Bay (1991–1993)	False Bay (2001–2003)	De Hoop (2001–2003)	Stil Bay (2004)	Kosi Bay (2001–2002)
$n$	247	139	295	73	152
$L_\infty$ (mm)					
Best fit	770	881	594	741	875
SE	30.8	29.4	22.7	88.9	74.0
CV (%)	4.0	3.3	3.8	11.7	8.3
LCI	719	831	563	663	779
UCI	842	949	653	946	1 064
$k$					
Best fit	0.214	0.164	0.183	0.132	0.151
SE	0.023	0.016	0.032	0.039	0.034
CV (%)	10.6	9.9	17.4	29.1	22.7
LCI	0.169	0.133	0.120	0.064	0.092
UCI	0.259	0.197	0.247	0.21	0.225
$t_0$ (y)					
Best fit	-0.271	-0.864	-2.419	-3.22	-2.498
SE	0.202	0.248	0.762	1.067	0.641
CV (%)	70	28	30	31.6	25
LCI	-0.705	-1.39	-4.29	-5.89	-3.93
UCI	-0.068	-0.435	-1.28	-1.7	-1.46

age recorded from False Bay (18 years). The sensitivity of the per-recruit models to  $M$  was tested by varying the above estimates by plus or minus 15–20% (i.e.  $M = 0.2–0.3 \text{ y}^{-1}$  for False Bay and Stil Bay and  $M = 0.3–0.4 \text{ y}^{-1}$  for KZN).

#### Total and fishing mortality

Total mortality ( $Z$ ) was estimated from the age structure of exploited populations (derived from length frequency data and regional age–length keys) using the Chapman and Robson (1960) method. Confidence intervals were obtained using a non-parametric bootstrapping procedure (see Hutchings and Griffiths 2010). Fishing mortality ( $F$ ) was estimated by subtracting the applicable  $M$ -estimate for each region from the  $Z$ -estimate. Length measurements were obtained from the following sources:

1. Length measurement (TL, cm) of commercial beach-seine catches from False Bay for the period 1991–1993.
2. Length measurements (nearest mm) collected predominantly from commercial beach-seine operators (85%) and supplemented by research seine-netting and recreational anglers catches (data pooled) in False Bay during 2001–2003.
3. Recreational, research and competitive spearfisher competition records from Stil Bay for three different time

periods (1990–1994, 1998, 2000–2004). These were converted from mass (nearest 10 g) to TL (mm) where necessary using the length–mass relationship for this region.

4. Spearfishing competition records and personal dive records from spearfishers active in KZN covering the period 1989–2003. These were converted from mass (nearest 10 g) to TL (mm) using length–mass relationships derived for this region and subdivided into four time periods (1989–1991, 1992–1994, 1995–1997 and 2000–2003).

#### Selectivity and maturity

The catch data collected represented the dominant fishing method active in each area, namely beach-seining in False Bay and spearfishing off Stil Bay and the KZN coast, and knife edge selectivity was considered appropriate for the analysis. The proportion-at-age of fish larger than three different minimum size limits ( $l_c = 30 \text{ cm}$ ,  $40 \text{ cm}$  and  $50 \text{ cm}$  TL) that were modelled for each region was calculated by dividing the number of fish in each age class greater than  $l_c$  by the total number sampled in each age class (Table 3). The proportion-at-age of mature females was estimated by multiplying the number-at-age for each length class (from

**Table 3:** Proportion-at-age of mature female (MatF) *U. robinsoni* and of fish larger than each of the three minimum size limits ( $l_c = 30, 40$  or  $50$  cm TL) that were modelled

Age	Proportion											
	False Bay (1991–1993)			False Bay (2001–2003)			Stil Bay (2004)			KZN (2001–2002)		
	MatF	30 cm	40 cm	50 cm	MatF	30 cm	40 cm	50 cm	MatF	30 cm	40 cm	50 cm
0	0	0	0	0	0	0	0	0	0	0	0	0
1	0	0	0	0	0.01	0	0	0	0.05	0.17	0.64	0.02
2	0.10	0.93	0	0	0.16	0	0	0	0.61	1	1	0.04
3	0.41	1	0.31	0	0.66	0.69	0	0	0.78	1	1	0.09
4	0.86	1	0.97	0.22	0.92	1	0.27	0	0.86	1	1	0.19
5	0.97	1	1	0.64	0.98	1	0.88	0.88	0.87	1	1	0.34
6	1	1	1	0.96	1	1	1	1	0.99	1	1	0.54
7	1	1	1	1	1	1	1	1	1	1	1	0.72
8	1	1	1	1	1	1	1	1	1	1	1	0.85
9	1	1	1	1	1	1	1	1	1	1	1	0.93
10	1	1	1	1	1	1	1	1	1	1	1	0.97
11	1	1	1	1	1	1	1	1	1	1	1	0.99
12	1	1	1	1	1	1	1	1	1	1	1	1
$t_{max}$	1	1	1	1	1	1	1	1	1	1	1	1

age-length keys) by the predicted proportion of mature females in each length class (from logistic ogive fitted to observed maturity data), summing the resultant proportions for each age and dividing by the total number of each age in the age-length key (Table 3).

**Per-recruit analysis**

Yield per-recruit (Y/R) and spawner biomass per-recruit (SB/R) was calculated using the following equations:

$$Y/R = \sum_{t=0}^{t_{max}} \frac{(FS_t)}{(FS_t + M)} \left( e^{-\sum_{i=0}^{t-1} (FS_i + M)} \right) (1 - e^{-FS_t - M}) W_t \tag{1}$$

$$SB/R = \sum_{t=0}^{t_{max}} \left( B_t \left( e^{-\sum_{i=0}^{t-1} (FS_i + M)} \right) W_t \right) \tag{2}$$

where  $F$  is the instantaneous rate of fishing mortality,  $S_t$  is the proportion of fish at age  $t$  larger than  $l_c$ ,  $W_t$  is the predicted individual mass of fish of age  $t$ , and  $B_t$  is the proportion of mature females at age  $t$  (see Table 3).

Per-recruit models for the KZN coast were constructed using biological information collected at Kosi Bay, and for Stil Bay using biological data collected at De Hoop and additional growth information collected at Stil Bay. For False Bay, growth data used in per-recruit models were from samples collected at the two different time periods and biological data (size-at-maturity, length-mass relationship) collected during the latter period. SB/R and Y/R curves were also calculated for three alternative minimum size limits ( $l_c = 30$  cm,  $40$  cm and  $50$  cm).

**Results**

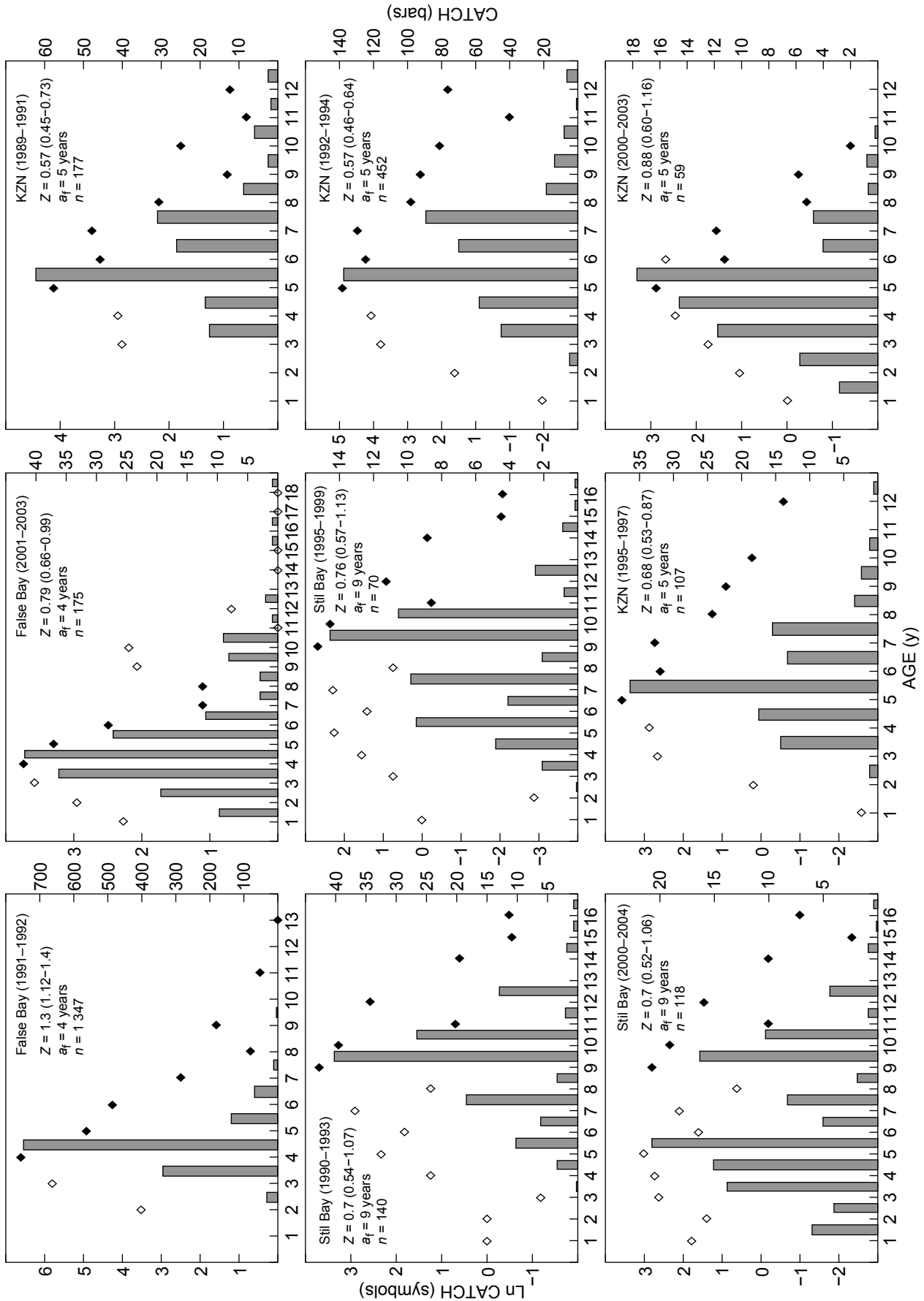
**Total and fishing mortality**

Age distributions for exploited *U. robinsoni* populations at False Bay, Stil Bay and the KZN coast at various intervals over the period 1991–2004 are shown in Figure 2. Estimated  $Z$  and  $F$  rates are given in Table 4. As expected, variation around  $Z$ -estimates increased with decreasing sample size. Estimated  $F$  over the period 1990–2004 appeared to decrease at False Bay, remained relatively constant at Stil Bay and increased along the KZN coast (Table 4).

**Per-recruit analyses**

*False Bay 1991–1993*

Yield and spawner biomass per-recruit curves for *U. robinsoni* sampled from False Bay during the period 1991–1993 at the current minimum size limit ( $40$  cm TL) and three alternative  $M$ -estimates ( $0.2$   $y^{-1}$ ,  $0.25$   $y^{-1}$  and  $0.3$   $y^{-1}$ ) are shown in Figure 3. Estimated  $F$  during the period exceeded both the target ( $F_{SB40}$ ) and threshold reference ( $F_{SB25}$ ) points, indicating that the risk of recruitment overfishing (and stock collapse) at the time was high. A 75% reduction in  $F$  was required at the time to elevate SB/R to the target reference point ( $F_{SB40}$ ).



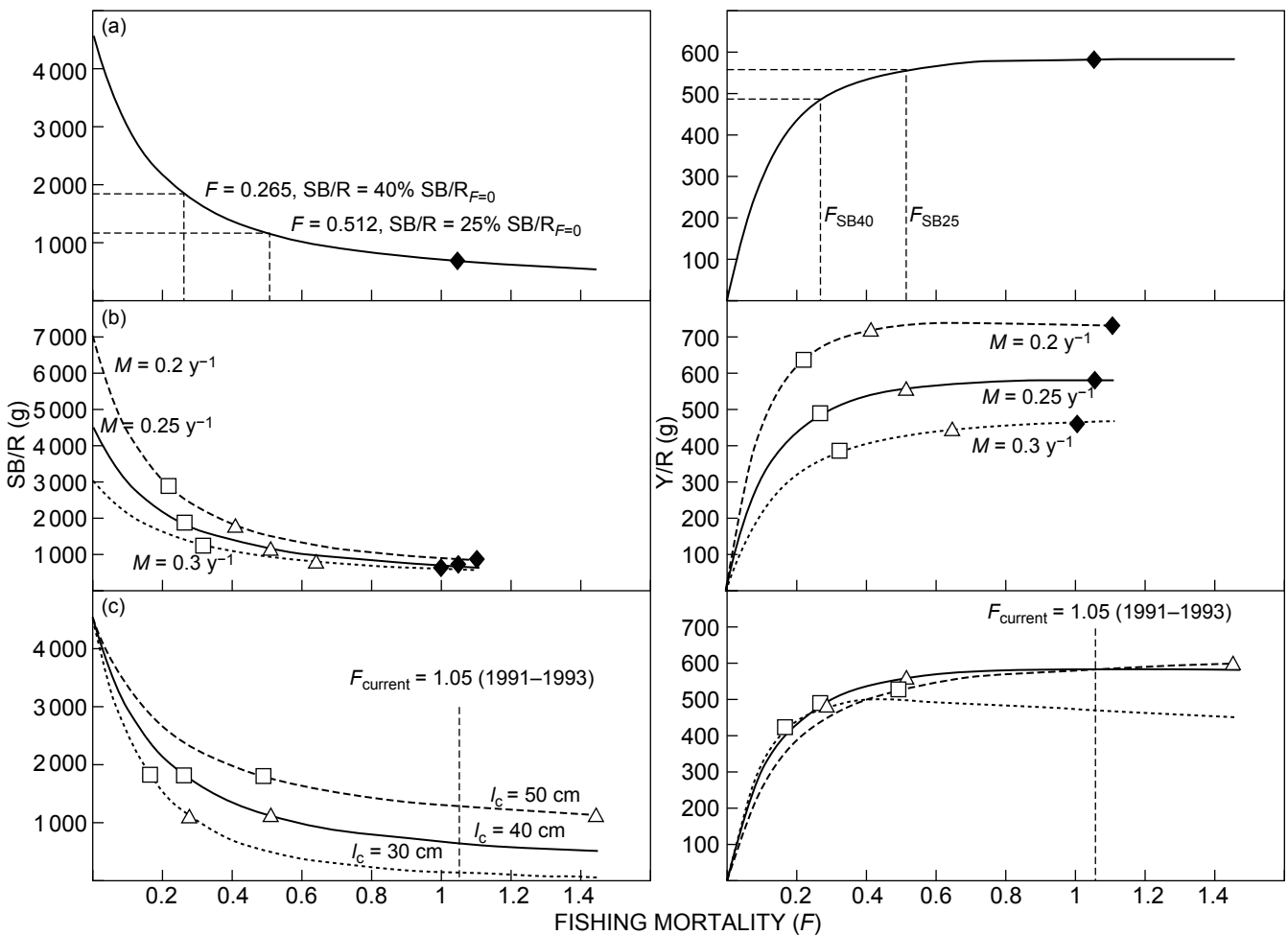
**Figure 2:** Catch curves and age distributions for the *U. robinsoni* stocks assessed showing age at full recruitment (a), the ages used in estimating the total instantaneous mortality rate Z (shaded diamonds) and length frequency sample size (shaded bars)

Decreasing  $F$  to the target reference point would have resulted in a predicted 20% reduction in Y/R ratio, but given that recruitment overfishing was probably occurring at the

time, reduced  $F$  and consequently larger spawner biomass would potentially have resulted in increased annual yield over the longer term. Although maximum SB/R increased greatly

**Table 4:** Total ( $Z$ ) and fishing ( $F$ ) mortality estimates for exploited *U. robinsoni* populations. Number of random lengths ( $n$ ), age at full recruitment ( $a_r$ ), upper (UCI) and lower (LCI) 95% confidence intervals and coefficient of variation (CV) for  $Z$  are given

Period	$n$	$a_r$	$Z$ ( $y^{-1}$ )	LCI	UCI	CV (%)	$F$ ( $y^{-1}$ )
<i>False Bay</i>							
1991–1993	1 389	4	1.3	1.12	1.48	7	1.05
2001–2003	175	4	0.8	0.66	0.99	11	0.55
<i>Stil Bay</i>							
1990–1993	140	9	0.76	0.54	1.07	17	0.51
1998	70	9	0.80	0.57	1.13	18	0.54
2003–2004	118	9	0.73	0.52	1.10	20	0.47
<i>KZN</i>							
1989–1991	177	5	0.57	0.45	0.73	12	0.22
1992–1994	452	5	0.57	0.46	0.64	9	0.22
1995–1997	106	5	0.68	0.53	0.87	13	0.33
2000–2003	63	5	0.88	0.60	1.16	17.4	0.53



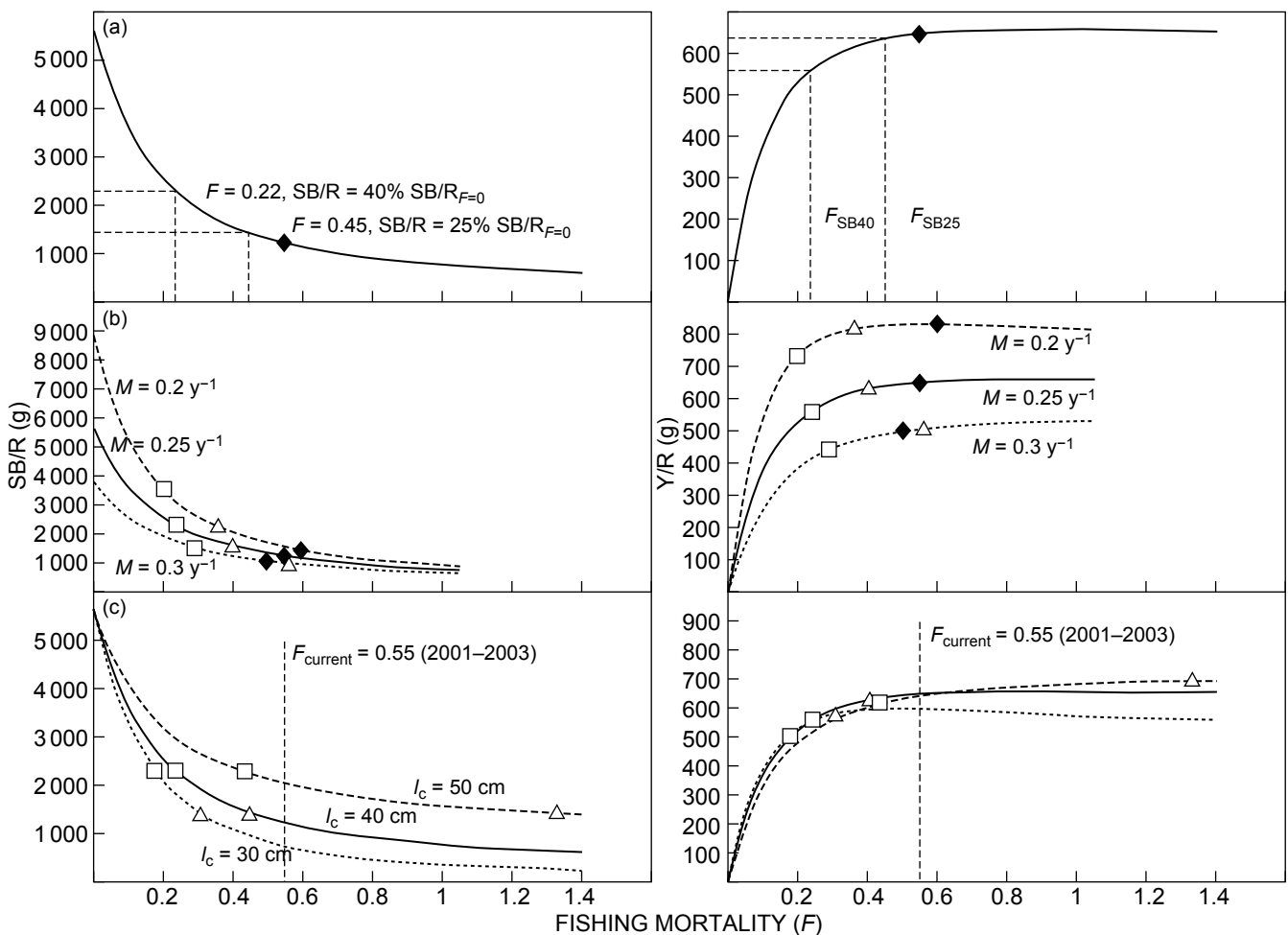
**Figure 3:** (a) Relationship between fishing mortality ( $F$ ) and spawner biomass (SB/R) and yield per-recruit (Y/R) for False Bay *U. robinsoni* sampled during 1991–1993; (b) SB/R and Y/R curves at different  $M$  estimates; (c) SB/R and Y/R curves for alternative minimum size limits ( $l_c$ ). Squares represent the target reference point ( $F_{SB40}$ ), triangles the threshold reference point ( $F_{SB25}$ ) and closed diamonds the estimated current (1991–1993) fishing mortality

with decreases in the  $M$  (Figure 3), biological reference points only showed moderate change ( $F_{SB40} = 0.22\text{--}0.32\text{ y}^{-1}$ ,  $F_{SB25} = 0.4\text{--}0.64\text{ y}^{-1}$  and  $SB/R = 11\text{--}18\%$   $SB/R_{F=0}$ ). Under all three  $M$ -estimates modelled (i.e.  $0.2\text{--}0.3\text{ y}^{-1}$ ), the stock would have been considered collapsed in terms of South Africa's LMP. The effect of altering the minimum size limit ( $l_c = 30\text{ cm}$ ,  $40\text{ cm}$  and  $50\text{ cm}$ ) on  $SB/R$  and  $Y/R$  (using the most realistic  $M$ -estimate of  $0.25\text{ y}^{-1}$ ) is shown in Figure 3c. Decreasing  $l_c$  to  $30\text{ cm}$  reduced  $SB/R$  to  $3.6\%$  of pristine levels, whereas a  $10\text{ cm}$  increase in  $l_c$  elevated  $SB/R$  to above the threshold level ( $SB/R = 29\%$   $SB/R_{F=0}$ ) with little reduction in yield. A further  $53\%$  reduction in  $F$  was, however, required to attain the target ( $SB/R = 40\%$   $SB/R_{F=0}$ ), indicating that additional regulations (e.g. a reduction in bag limit, closed season or area) would have been necessary at the time in order to protect spawner biomass.

**False Bay 2001–2003**

Yield and spawner biomass per-recruit curves for *U. robinsoni* sampled from False Bay during the period 2001–2003

at the current minimum size limit ( $40\text{ cm TL}$ ), three alternative natural mortality estimates ( $M = 0.2\text{ y}^{-1}$ ,  $0.25\text{ y}^{-1}$  and  $0.3\text{ y}^{-1}$ ) and three alternative minimum size limits ( $l_c = 30\text{ cm}$ ,  $40\text{ cm}$  and  $50\text{ cm}$ ) are shown in Figure 4. Although estimates of  $Z$ , and hence  $F$ , were substantially lower ( $48\%$ ) during 2001–2003 than 1991–1993 (Table 4), the estimates of  $SB/R$  were only moderately greater in 2001–2003 ( $21\%$  of  $SB/R_{F=0}$  in 2001–2003,  $14.5\%$  of  $SB/R_{F=0}$  in 1991–1993). Under the current  $l_c$ , and using the most realistic  $M$ -estimate, the stock is still classified as collapsed ( $F > F_{SB25}$ ), with a  $57\%$  reduction in  $F$  required to elevate  $SB/R$  to above the target level (Figure 4a). A reduction in  $F$  of this magnitude would result in a predicted decrease in  $Y/R$  ratio of only  $16\%$  in the short term. Changing the estimate of  $M$  ( $0.2\text{--}0.3\text{ y}^{-1}$ ) resulted in modest changes in target and threshold reference points ( $F_{SB40} = 0.2\text{--}0.29\text{ y}^{-1}$ ;  $F_{SB25} = 0.36\text{--}0.56\text{ y}^{-1}$ ; Figure 4b). Using the highest tested natural mortality estimate ( $M = 0.3\text{ y}^{-1}$ ),  $SB/R$  is estimated at  $27\%$  of pristine; although this is above the threshold reference level, the stock is still classified as overexploited. It is also unlikely that  $M$  for the False



**Figure 4:** (a) Relationship between fishing mortality ( $F$ ) and spawner biomass ( $SB/R$ ) and yield per-recruit ( $Y/R$ ) for False Bay *U. robinsoni* sampled during 2001–2003; (b)  $SB/R$  and  $Y/R$  curves at different  $M$  estimates; (c)  $SB/R$  and  $Y/R$  curves for alternative minimum size limits ( $l_c$ ). Squares represent the target reference point ( $F_{SB40}$ ), triangles the threshold reference point ( $F_{SB25}$ ) and closed diamonds the estimated current (2001–2003) fishing mortality



Bay stock would be this high, considering that the maximum age sampled in this region (18 years) was greater than that recorded at De Hoop (16 years) where  $M$  was estimated at  $0.26 \text{ y}^{-1}$ . A 10 cm increase in the minimum size limit ( $l_c = 50 \text{ cm TL}$ ) is predicted to increase SB/R to 36% of  $\text{SB/R}_{F=0}$  (i.e. above the threshold reference point) at current levels of  $F$  with negligible change in the Y/R ratio. However, a further 20% reduction in  $F$  would still be required to elevate the SB/R ratio to above the target reference point (Figure 4c).

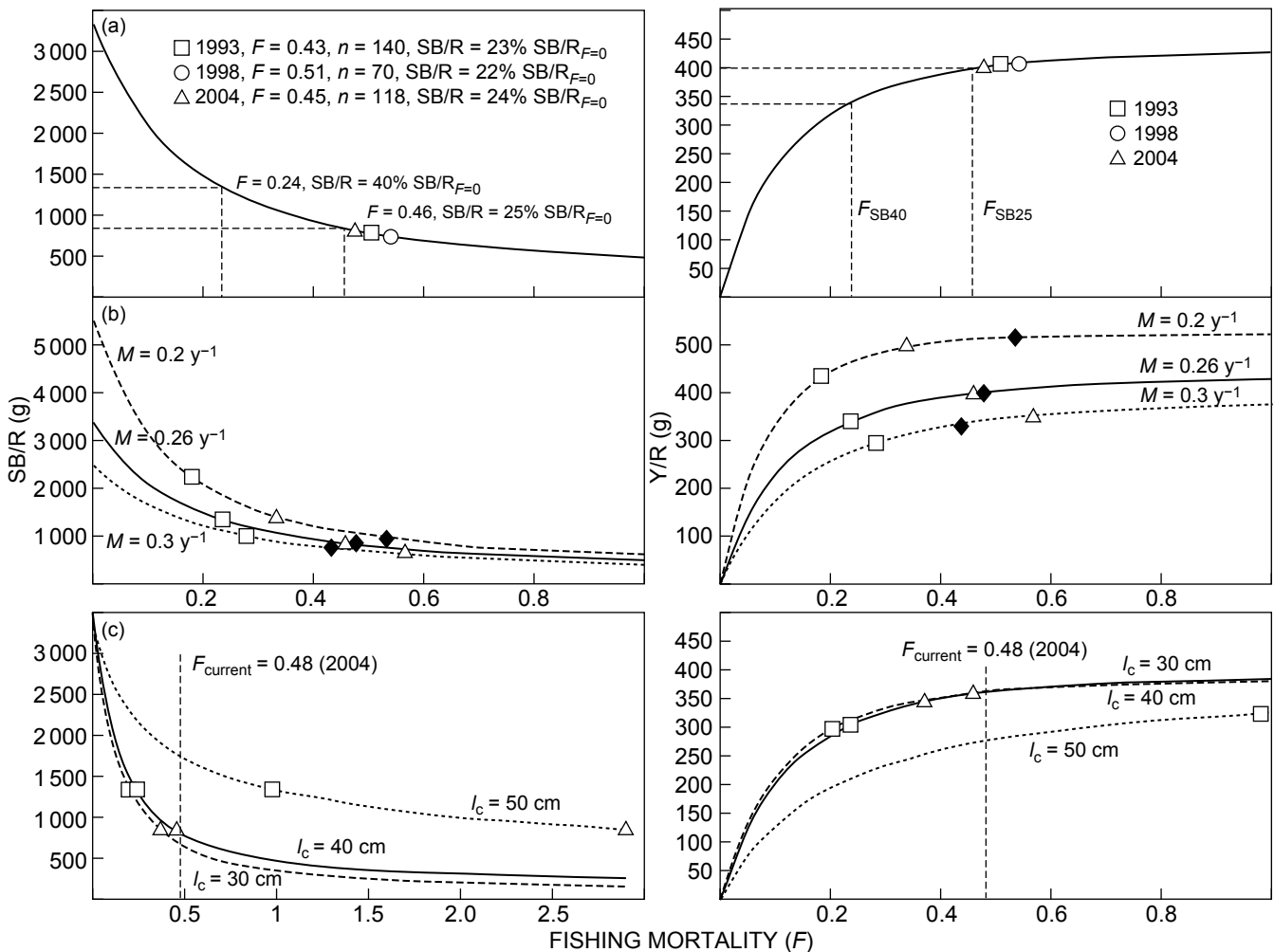
#### Stil Bay 2004

Yield and spawner biomass per-recruit curves for *U. robinsoni* sampled from Stil Bay during 2004 at the current minimum size limit (40 cm TL) and three alternative  $M$ -estimates ( $0.2 \text{ y}^{-1}$ ,  $0.26 \text{ y}^{-1}$  and  $0.3 \text{ y}^{-1}$ ) are shown in Figure 5. Estimates of  $F$  for the three periods produced SB/R ratios of 22–24% of  $\text{SB/R}_{F=0}$ , i.e. just below the threshold reference point. The required reduction in  $F$  to elevate SB/R to the 40% of pristine level ranged from 51% to 57%, with a corresponding

short-term decrease in Y/R of approximately 20%. The effect of altering the minimum size limit on biological reference points and estimated stock status is shown in Figure 5c. A 10 cm decrease in  $l_c$  to 30 cm TL would have little effect on target ( $F_{\text{SB40}}$ ) or threshold ( $F_{\text{SB25}}$ ) reference points, but would decrease current (2001–2004) SB/R to 21%  $\text{SB/R}_{F=0}$ , whereas an increase in  $l_c$  to 50 cm TL would allow SB/R to recover to above the 40% of pristine level (Figure 5c). Yield per-recruit with an  $l_c = 50 \text{ cm TL}$  would be reduced by approximately 25% at current levels of  $F$ , but would be slightly increased if the stock was exploited at  $F_{\text{SB40}}$ .

#### KZN coast

Yield and spawner biomass per-recruit curves for *U. robinsoni* along the KZN coast at the current minimum size limit ( $M = 0.35 \text{ y}^{-1}$ ,  $l_c = 40 \text{ cm TL}$ ), three alternative  $M$ -estimates ( $M = 0.3 \text{ y}^{-1}$ ,  $0.35 \text{ y}^{-1}$  and  $0.4 \text{ y}^{-1}$ ,  $l_c = 40 \text{ cm TL}$ ) and three alternative minimum size limits ( $l_c = 30 \text{ cm}$ ,  $40 \text{ cm}$  and  $50 \text{ cm TL}$ ;  $M = 0.35 \text{ y}^{-1}$ ) are shown in Figure 6.



**Figure 5:** (a) Relationship between fishing mortality ( $F$ ) and spawner biomass (SB/R) and yield per-recruit (Y/R) for Stil Bay *U. robinsoni* sampled during 2004 showing estimated SB/R and Y/R at three different time periods; (b) SB/R and Y/R curves at different  $M$  estimates showing estimated current  $F$  (2004); (c) SB/R and Y/R curves for alternative minimum size limits ( $l_c$ ). In b and c, squares represent the target reference point ( $F_{\text{SB40}}$ ), triangles the threshold reference point ( $F_{\text{SB25}}$ ) and closed diamonds the estimated current fishing mortality

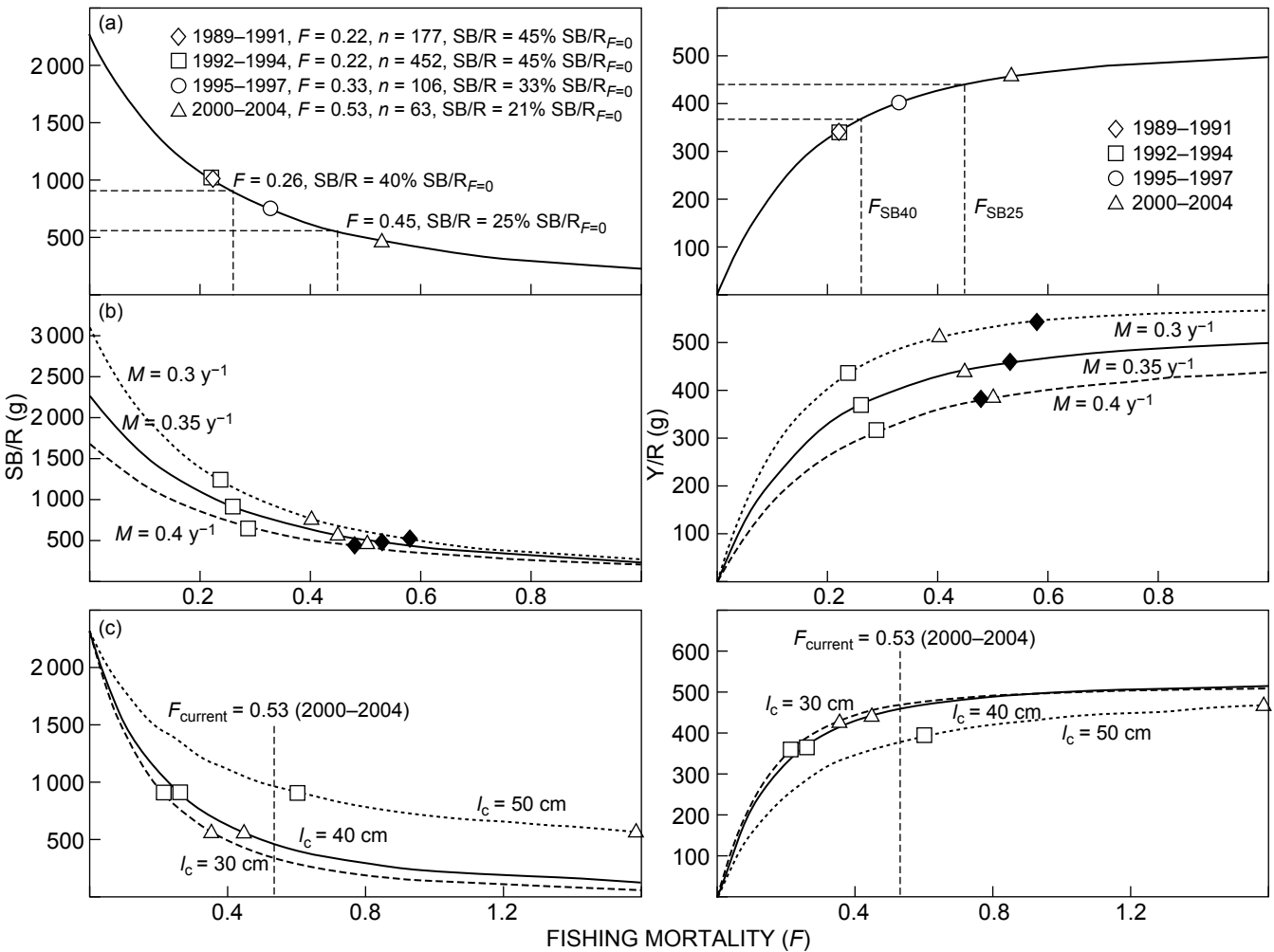
Estimates of  $F$  at four different time periods show SB/R declining from above the target reference point during the period 1989–1994 to below the target reference point over the period 1995–1998 (SB/R = 33% SB/R<sub>F=0</sub>) and below the threshold reference point during the latter four-year period (2000–2004, SB/R = 21% SB/R<sub>F=0</sub>; Figure 6a). Again, biological reference points did not alter much with changes in  $M$ , and under all three of the different  $M$ -estimates the current estimated  $F$  exceeded or was close to the threshold level ( $F_{SB25}$ ; Figure 6b). Under the current  $l_c = 40$  cm TL, a 51% reduction in  $F$  is required to elevate SB/R to the target level (40% of pristine). Analysis of observed bag frequencies of KZN spearfishers (from divers' personal records) indicate that compliance with a decrease in bag limit from the current five to one fish per person per day would reduce  $F$  by an average of 46% (over four time periods), whereas a bag limit of two would reduce  $F$  by an average of only 18% (Table 5). The current bag limit of five fish per person per day is totally ineffective in controlling fishing mortality

(Table 5). An alternative management strategy of increasing  $l_c$  to 50 cm TL would, however, elevate SB/R to above the target reference point with only a small (16%) decrease in Y/R at current (2000–2004) levels of  $F$  (Figure 6c). However, if recruitment overfishing is occurring at present, a larger spawner biomass should produce larger year classes and ultimately a greater annual yield.

**Discussion**

**Natural mortality**

Reliable estimates of natural mortality are essential for per-recruit assessments but are often difficult to obtain in established fisheries. In this study we were fortunate to obtain direct measures of  $M$  through sampling *U. robinsoni* in established MPAs. Both the De Hoop and Maputaland MPAs have been in existence for a sufficiently long time to allow at least one cohort to attain maximum age in the absence of fishing mortality. As *U. robinsoni* are known



**Figure 6:** (a) Relationship between fishing mortality ( $F$ ) and spawner biomass (SB/R) and yield per-recruit (Y/R) for KZN *U. robinsoni* showing estimated SB/R and Y/R at four different time periods; (b) SB/R and Y/R curves at different  $M$  estimates showing estimated current  $F$  (2000–2004); (c) SB/R and Y/R curves for alternative minimum size limits ( $l_c$ ). In b and c, squares represent the target reference point ( $F_{SB40}$ ), triangles the threshold reference point ( $F_{SB25}$ ) and closed diamonds the estimated current fishing mortality

**Table 5:** Observed bag frequencies (Freq.) and the potential percentage reduction in fishing mortality (%F) resulting from the enforcement of various daily bag limits (DBLs) for *U. robinsoni* landed by spearfishers along the KZN coast during four different time periods

DBL	1989–1991		1992–1994		1995–1997		2000–2003		Total period 1989–2003	
	Freq.	%F	Freq.	%F	Freq.	%F	Freq.	%F	Freq.	%F
0	120	100	373	100	58	100	45	100	596	100
1	36	55.8	114	48.6	9	37.0	11	42.6	170	49.7
2	26	31.5	42	26.1	7	7.4	17	5.6	92	25.2
3	6	21.5	20	14.2		3.7	3	0	29	14.6
4	2	14.9	11	7.3	1	0			14	8.5
5	4	9.4	11	3.3					15	4.6
6	2	6.1	2	2.0					4	2.9
7	3	3.9	1	1.3					4	1.8
8		3.3	1	0.8					1	1.4
9		2.8		0.5						1.1
10		2.2		0.3						0.6
11		1.7	1	0					1	0.5
12		1.1								0.3
13		0.6								0.2
14	1	0.0							1	0
Total catch (no.)	181		395		27		54		657	
Mean weight (kg)	2.6		2.5		2.2		1.7		2.4	
Diver-outings	200		576		75		76		927	
Diver-hours	1 034.5		3 018		407		403		4 863	
CPUE (number of fish 100 dive-hours <sup>-1</sup> )	17.5		13.1		6.6		13.4		13.5	
CPUE (kg fish 100 dive-hours <sup>-1</sup> )	44.1		32.7		14.6		22.8		32.1	

to have small (approximately 1.5 km) home ranges and coastwise movement of adults is limited (Hutchings 2005), the populations within the sampled MPAs are unlikely to have been significantly impacted on by fishing through immigration into, or emigration out of, the protected areas.

The *U. robinsoni* population in the subtropical Maputaland MPA was found to have a markedly different natural mortality rate, in addition to other life-history parameters, such as growth rate and, age/size-at-maturity, than that in the warm-temperate De Hoop MPA (Hutchings and Griffiths 2010). It is likely that the exploited populations that were assessed elsewhere along the subtropical KZN coast experience a similar natural mortality rate to the Maputaland MPA population because there are no major oceanographic discontinuities between the areas and *U. robinsoni* occur in very similar subtidal, moderate-profile, patch-reef habitat throughout the region. The *M*-estimate obtained for the De Hoop MPA population is also conceivably very similar to that experienced by nearby warm-temperate southern Cape *U. robinsoni* populations (such as the one assessed at Stil Bay).

The spatial scale over which such extrapolation of *M*-estimates can be made is, however, unknown. *U. robinsoni* sampled from False Bay, approximately 280 km west of De Hoop, were found to have a faster growth rate and obtain a significantly larger asymptotic length than those sampled within the De Hoop MPA. This could be due to a number of factors including physical and biotic environmental differences and/or the effects of exploitation. With such a marked difference in growth rate, which is positively correlated with mortality rate (Roff 1992), it is questionable whether the De Hoop MPA estimate of *M* is transferable to this region. However, good agreement was obtained between

Hoenig's (1983) equation, which is based on maximum age and on catch-curve estimates for both the De Hoop and Maputaland MPA populations (Hutchings and Griffiths 2010). Because the maximum age recorded in False Bay was similar to that recorded at De Hoop (18 and 16 years respectively), the natural mortality rate experienced by the False Bay population is likely to be close to the De Hoop estimate. In any case, sensitivity analyses that involved varying the *M*-estimates did not have very large effect on biological reference point or stock status estimates. The question of transferability of *M*-estimates obtained from sampling in MPAs to other regions in no way reduces the value of having such estimates. It is certainly preferable to using *M*-estimates obtained by extrapolation from studies on other species. As one wouldn't expect substantial changes in natural mortality, particular the 'average' value obtained from catch-curve analysis — regressed over good and bad recruitment and growth years — destructive sampling in MPAs would not need to be repeated. Indeed, our study as a whole could not have been conducted without the valuable life-history information obtained by sampling within MPAs, because procuring sufficient samples for accurate estimates of parameters such as growth and size-at-maturity from exploited areas would not have been possible due to the current scarcity of the species in exploited areas.

#### Fishing mortality

The use of competitive or personal spearfishing records for estimating the total mortality rate is not ideal, because spearfishers inherently select the larger fish encountered, which could potentially lead to positive bias of older year classes in the age composition of the catch relative to the

population and consequently negatively bias estimates of  $Z$ . However, given the philopatric nature of *U. robinsoni* and the fact that spearfishers in areas with reasonably high rates of exploitation (e.g. KZN) will probably be targeting all fish above the legal size limit, biases of this nature are unlikely to be severe, as long as the appropriate age-at-full recruitment is used when estimating  $Z$  from catch data. Spearfishing is by far the dominant method used for catching *U. robinsoni* in KZN (Mann et al. 1997) and reliable data on size composition of *U. robinsoni* in catches made by other methods are extremely scarce, as well as being spatially and temporally discontinuous (Brouwer et al. 1997, Mann et al. 1997, Pradervand 2002).

The Stil Bay *U. robinsoni* population is assessed as over-exploited, at three different time periods spanning the period 1990–2004. The failure to detect a negative trend in  $F$  over this time may be partly explained by the selective nature of competitive spearfishers. The catch data used for the Stil Bay assessment came largely from national spearfishing competition records held in the area. Competitors are allowed to weigh in two fish per species per day, and would attempt to select the larger individuals when encountering a shoal. Serial depletion of philopatric *U. robinsoni* shoals discovered in each competition, with competitors selecting the larger (older) fish from each 'new' shoal encountered probably led to relatively stable  $F$ -estimates over time, even though the catch per unit effort (CPUE) was decreasing.

The possibility that a fish population is not in equilibrium can cause errors when estimating current mortality rates using catch-curve methods. In populations that are growing, e.g. within recently closed areas or due to catch restrictions, increasing recruitment will skew the population age distribution towards young fish, resulting in overestimates of mortality rate based on catch-at-age data, whereas a shrinking population (collapsing recruitment) will have the opposite effect (Attwood 2003). The total catch and CPUE of *U. robinsoni* during three national spearfishing competitions held at Stil Bay during 1993, 1998 and 2004 decreased dramatically from 136 fish at a catch rate of 0.84 fish diver-day<sup>-1</sup> in 1994 to 70 fish at 0.58 fish diver-day<sup>-1</sup> in 1998 and 40 fish at 0.52 fish diver-day<sup>-1</sup> in 2004, (South African spearfishing competition records). This decline occurred despite similar or improving skill levels, suggesting regional overexploitation. This observed decline in CPUE suggests that the Stil Bay population may indeed be shrinking; and the failure to detect an increase in  $F$  over time may be a result of collapsing recruitment. Differences in sea conditions and designated competition areas may also have played a role in causing the observed decline in total catch.

There is no clear trend in CPUE (in terms of number of fish per diver-hour) evident in the dive records of KZN spearfishers, which suggests that the finding of declining stock size over time may not be correct (Table 5). This is particularly concerning as the length frequency sample size used in estimating  $F$  is considerably smaller during the latter period and hence is less accurate (Table 4). An alternative explanation is that KZN spearfishers are still finding *U. robinsoni*, but they are smaller and younger fish and the overall stock biomass has indeed decreased. This does appear to be the case as the average size fish

landed by spearfishers in KZN and CPUE (in terms of mass) does decline over time (Table 5). Another factor that may mask a decline in stock size when using CPUE data, or an increase in mortality rate estimated from catch-at-age/length data, is the resident, philopatric nature of many inshore fish species, such as *U. robinsoni*. An implicit assumption in the catch-curve approach to mortality estimation is that the age structure of the catch is representative of the recruited portion of the population, i.e. fish within the assessment area are homogeneous. For resident species with small home ranges, this will only occur if all sites are fished in proportion to the original biomass. Serial depletion of 'new' sites by fishing will result in the age structure of the catch and CPUE remaining similar to that of an unexploited population, whereas the biomass of the population as a whole is substantially reduced. It is likely that this effect caused underestimates of the true mortality rates experienced by the populations as a whole of all three *U. robinsoni* stocks assessed.

#### Temporal variation

Although SB/R of False Bay *U. robinsoni* was estimated as a greater percentage of pristine during the more recent sampling period (21% of SB/R<sub>F=0</sub> in 2001–2003, 14.5% of SB/R<sub>F=0</sub> in 1991–1993), it is highly unlikely that the False Bay stock had recovered at all over the 10-year period between sampling events because fishing effort has probably increased. Van der Elst (1993) estimated that there were 365 000 participants in South Africa's open access recreational shore-fishery (the main user group catching *U. robinsoni* in False Bay) in 1991 and that this number would increase at approximately 6% per year. McGrath et al. (1997) used a lower rate of increase (2%) due to a finding of a very low income elasticity of demand and a predominance of white participants. False Bay's proximity to the greater Cape Town metropolis, which has shown considerable population growth over the same period (1990–2000, approximately 4% per annum; Statistics South Africa 2001), suggests that the growth in recreational shore-anglers fishing the area increased at a similar rate.

#### Management

Several studies have indicated that the risk of recruitment overfishing and stock collapse is high when the relative SB/R is reduced to <20–30% of the pristine level (SB/R<sub>F=0</sub>) (Clark 1991, Mace and Sissenwine 1993, Thompson 1993, Mace 1994); whereas a fishing mortality rate that reduces the SB/R ratio to 40% SB/R<sub>F=0</sub> ( $F_{SB40}$ ) has been shown to provide high yields with a low risk of stock collapse for species with a wide range of life-history characteristics, regardless of the spawner-recruit relationship (Clark 1993, Punt 1993). The South African LMP adopts the  $F_{SB40}$  level as a target reference point (stocks assessed to have a SB/R ratio = 40–50% SB/R<sub>F=0</sub> are considered to be optimally exploited) and the  $F_{SB25}$  point as a threshold reference point (stocks assessed to have a SB/R ratio <25% SB/R<sub>F=0</sub> are considered heavily overexploited and there is a high risk of recruitment failure and stock collapse). Any stock assessed as having a SB/R ratio of <40% SB/R<sub>F=0</sub> is considered overexploited, and management action is required to reduce

fishing mortality in order to facilitate stock recovery towards the target level. Management action obviously needs to be progressively more urgent in situations where the threshold reference point is approached or exceeded.

Given the overexploited status of the three exploited *U. robinsoni* stocks assessed, it is necessary to implement measures aimed at rebuilding stocks to the target reference level ( $SB/R = 40\% SB/R_{F=0}$ ). Per-recruit model outputs indicate that this would be achieved for stocks occurring off Stil Bay and along the KZN coast through a 10 cm increase in the minimum size limit, from 40 cm to 50 cm TL. However, even with a 10 cm increase in the minimum size limit, further reductions in fishing mortality (20–50%) for heavily fished areas such as False Bay would still be required. There is also a strong possibility that current fishing mortality for all of the stocks was underestimated and hence the current stock status estimates are optimistic. Analysis of observed bag frequencies of KZN spearfishers indicate that the current daily bag limit of five fish person<sup>-1</sup> day<sup>-1</sup> is not limiting fishing mortality in the region (see Table 5). A decrease in the current bag limit to one fish person<sup>-1</sup> day<sup>-1</sup> would reduce  $F$  by an average of 46% and a daily bag limit of two would reduce  $F$  by an average of just 18% (Table 5). Data on the bag frequencies of recreational anglers or spearfishers in other regions are unfortunately not available and the effect of alternative daily bag limits on reducing  $F$  cannot be determined. A bag limit of one fish person<sup>-1</sup> day<sup>-1</sup> would probably be viewed as draconian by most recreational anglers and spearfishers whereas an increase in the minimum size limit by 10 cm to 50 cm TL should be acceptable by these sectors, given that anglers and divers fishing for sport or food consider the size of the fish caught as important. Increasing the minimum size limit could also assist in achieving the goals of ecosystem-based approach to fisheries management by allowing individuals to better fulfil their ecological roles prior to recruitment into the fishery (Froese et al. 2008). The cohort biomass of the *U. robinsoni* stocks assessed here are predicted to reach a maximum biomass at between five and six years of age (45–65 cm TL). Exploiting stocks at sizes greater than that at which the cohort biomass peaks has been shown to greatly reduce the impacts on the stock size structure and biomass (Froese et al. 2008). An increase in the minimum size limit and a reduction in fishing mortality through bag limit restriction appears necessary, although the actual combination of these two management measures should be decided on in consultation with the relevant user groups. Co-management with user groups will hopefully ensure support for the restrictions and facilitate compliance (Bohnsack and Ault 1996, Griffiths 1997).

Although these management measures alone will probably not be sufficient to rebuild stocks to target levels in heavily exploited areas such as False Bay, several other recent management actions will facilitate stock rebuilding, namely: beach-seine operators have since 2004 been prohibited from retaining and selling *U. robinsoni*, and several new marine protected areas have been proclaimed which will be effective in protecting spawning stock due to the resident nature of adult fish. Although a nationally consistent suite of regulations is preferable, it must be borne in mind that the species shows considerable spatial variation in its life

history and stocks should be assessed at relatively small spatial scales. Of particular concern is a complete lack of information on the biology or stock status of the species from the Eastern Cape. Attwood (2003) makes several valid criticisms on the use of per-recruit stock assessment techniques (or more specifically the use of catch-at-age data for estimating mortality rates) for South African linefish and recommends the use of CPUE as a stock status indicator instead. Although in our study attempts were made to address some of the shortcomings of 'typical' per-recruit assessments (i.e. sampling MPAs in different regions for estimates of  $M$ , separate regional assessments based on measured life-history parameters, investigation of temporal variation in growth rate), some of Attwood's (2003) concerns are still valid — specifically, the failure to derive estimates of fishing mortality uniformly across all areas (indeed it is likely that the continued occurrence of *U. robinsoni* in False Bay catches, despite the fact that the stock was already classified as collapsed 15 years ago, is due to the presence of natural refuges) and the fact that assessments based on catch-at length/age data are retrospective and do not allow for proactive management. Furthermore, although our study extended over a considerable period of time, the findings still do not represent a comprehensive assessment of *U. robinsoni* stock status throughout South African waters and realistically, given the financial and manpower constraints, it is not likely to be repeated or expanded, for a species of limited commercial importance, in the near future.

This study has shown that *U. robinsoni* stocks are overexploited at three different sites along South Africa's coast and that changes in management regulations are required. Once such changes are implemented, future management should focus on monitoring of stock status as part of nationwide, multispecies linefish monitoring programmes and implement adjustments to restrictions accordingly. Both CPUE and size/age catch composition data should be collected. Currently, ongoing roving creel surveys and shore-based observer programmes should increase the quantity of data available for future assessments of *U. robinsoni* and other recreationally caught South African marine linefish. In order for CPUE data to provide an accurate measure of relative biomass, information on factors affecting catchability (e.g. targeting, gear type and environmental conditions) should also be collected. The limited effect of temporal variation in growth rate on biological reference point estimates suggests that the per-recruit models presented here can be used to assess the relative future changes in biomass in response to changes in  $F$  and  $I_c$  (Hutchings et al. 2010).

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