# THE EFFECTS OF A MARINE RESERVE ON GALJOEN (*DICHISTIUS CAPENSIS*) AT CAPE POINT, SOUTH AFRICA, AND IMPLICATIONS FOR THE MANAGEMENT OF THE RECREATIONAL FISHERY

**Written by:** Chantel Elston

**Supervisor:** Colin Attwood

## Abstract

Marine protected areas (MPAs) are gaining credibility in the scientific community because of their duality as a conservation and fishery management tool, but sometimes the actual effects of an MPA fall short of the expected outcomes. Case-by-case studies are needed to understand what works and what doesn't and this understanding can then be applied to decision making and adaptive management. The galjoen (*Coracinus capensis*), a surf zone teleost endemic to South Africa, is a popular fish for recreational shore anglers and as such the population has declined to dangerously low levels. Marine reserves were established around Cape Point to try and counteract this decline. This study aimed to determine whether these reserves are having a positive effect on galjoen mortality, density and size. A controlled shore angling program has been running on the Cape Peninsula since 1986 in which anglers use a standardised fishing technique. The anglers record the length of their fishing trip as well as the fork length of the fish they catch. Two sites located in near proximity to each other, one in a reserve and one in an exploited area, were chosen from the data set to compare. This avoided the conflicting problem of major habitat differences. General linear models (GLMs) were used to isolate the effect of area on the catch per unit effort (CPUE) as well as fork length, and a negative log-likelihood function was used to estimate the mortality rate and sex ratio for each area. CPUE increased significantly from of 1.02 ± 0.81 galjoen.angler$^{-1}$.hour$^{-1}$ in the exploited area to 1.48 ± 0.85 galjoen.angler$^{-1}$.hour$^{-1}$ in the protected area and fork length increased significantly from 300.69 ± 34.71mm in the protected area to 329.31 ± 40.19mm in the exploited area. The GLMs revealed that the parameters 'area' and 'year' significantly affected the variation in CPUE and fork length, and that area had the greatest explanatory power in both cases suggesting that the reserve had a positive effect on the galjoen density and size. The mortality estimate for the protected site, which was taken as natural mortality, was 0.55 year$^{-1}$ and the mortality estimate at the exploited site was 1.0 year$^{-1}$. Fishing mortality ($F$) was estimated as 0.45 year$^{-1}$ which was considered to be close to $F_{\mathrm{MSY}}$. Increases in CPUE over time in the exploited area led to the hypothesis that the reserve is re-stocking adjacent exploited areas. A reduction in $F$ over time suggested that effort is reduced in the exploited study area, but it is unknown whether this effort has been displaced to another area adjacent the reserve. Because conventional fishery management tools are difficult or impossible to enforce for the galjoen stock and because the reserves appear to be positively affecting the galjoen within the reserve, it is suggested that the reserves are the optimal and most efficient conservation and fishery management tool for the galjoen stock.

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## Introduction

A Marine Protected Area (MPA) is defined as a “discrete geographic area of the sea established by international, national, territorial, tribal, or local laws designated to enhance the long-term conservation of natural resources therein” (Claudet 2011). MPAs differ in both objectives and the type of extractive uses allowed (Claudet 2011). They have gained much credibility throughout the scientific community due to their duality as both a conservation and fishery management tool (Roberts et al. 2001, Halpern 2003, Micheli et al. 2004, Botsford et al. 2009). Establishing an MPA is a spatial way to not only protect species directly targeted by fishing, but also to protect and restore habitats, entire species assemblages, as well as ecological interactions (Claudet 2011, Micheli et al. 2004). MPAs are also gaining popularity because conventional fishery management tools, such as catch or effort controls, are difficult to enforce and the fisheries that have been managed using these tools have continued to decrease to low levels (Holland and Brazee 1996, Halpern 2003, Botsford et al. 2009). There is the hope that MPAs will counteract the declines in fish populations and protect the ocean’s biodiversity (Attwood and Bennett 1994, Halpern and Warner 2003, Botsford et al. 2009, Gaines et al. 2010).

The effects of an MPA are expected to manifest themselves in long-term increases in four biological measures, namely density, biomass, organism size, as well as diversity (Halpern 2003, Micheli et al. 2004, Claudet 2011). It is also expected that in response to this increase in organism density within the MPA, spill-over of organisms and thus an increased catch per unit effort (CPUE) within adjacent exploited areas will occur (Roberts and Polunin 1991, Roberts et al. 2001, Halpern 2003, Micheli et al. 2004, Claudet 2011). However, these measures will only increase if fishing mortality is reduced within the MPA (Horwood et al 1998, Jennings 2000). Assessments of the actual effects of MPAs relative to these expected outcomes are essential for adaptive management and decision making (Claudet 2011). The main question underlying these assessments is how the state of the ecosystem within the MPA compares to the state of the ecosystem had the MPA never been established (Claudet 2011). This can obviously not be directly observed and must therefore be estimated through indirect means (Stewart-Oaten et al. 1986, Osenburg and Schmitt 1996, Osenburg et al. 2006, Claudet 2011).

The most common method of assessing the effect of an MPA is the Control-Impact design – two sites, one within the MPA and a control site located outside the MPA are compared. These two sites are assumed to be identical in the absence of an effect of the MPA and as such any differences between the two sites are attributed to the effect of the MPA. However, this estimated effect of the MPA is confounded by spatial variation and pre-existing differences in habitat (Attwood 2003, Westera et al. 2003, Claudet 2011). The second method of assessment is the Before-After design where a site within the MPA is sampled before and after its establishment. The differences in the before and after measurements are attributed to the effect of the MPA, but this effect is confounded by temporal variation such as larval supply (Roberts et al. 2001, Claudet 2011). The final method of assessment is the BACI design and involves sampling both an MPA site and a control site before and after

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establishment of the MPA. The changes in the MPA site from before to after its establishment relative to the control site provides the most reliable measure of the effect of an MPA (Lincoln-Smith et al. 2006, Claudet 2011).

Two commonly cited meta-analyses by Halpern (2003) and Micheli et al. (2004) reveal that the actual effects of an MPA sometimes don't result in the expected increase of the four biological measures. In Halpern's (2003) meta-analysis 37% of MPAs produced no increase in density, 10% produced no increase in biomass, 20% showed no increase in mean organism size, and 41% showed no increase in diversity. There has been little focus on why these measures increase in some MPAs but not others (Halpern 2003, Botsford et al. 2009). Like any conservation or management strategy, mistakes will made and without planning, monitoring and evaluation we will never understand what works, what doesn't, and why. Without case by case evaluation and appropriate monitoring programs there is the risk this valuable management tool will never reach its full potential (Hilborn et al. 2004).

The galjoen (*Dichistius capensis*) is a surf-zone teleost endemic to southern Africa and is amongst the most heavily exploited fish species by recreational shore anglers (Attwood and Bennett 1994, Attwood 2003, Attwood and Cowley 2005). The population occurs in two disjunct stocks with some exchange between them; a southern stock is located on the southern coast of South Africa and a western stock is known to occur on the coast of Namibia (Attwood and Bennett 1994, Attwood and Cowley 2005). The galjoen is a medium size fish with asymptotic lengths of males and females being 472 and 677mm respectively, sexual maturity is reached at an age of 5-6 years, and maximum age is 13 years (Bennett and Griffiths 1986). The decline of galjoen in certain areas was first reported by Smith (1935) prior to World War 2. By 1973, a size limit was introduced as the species was considered to be in trouble. Further restrictions were added in 1984 when bag limits and a closed season was established (Attwood 2003). Unfortunately decreases still occurred and a per-recruit analysis by Bennett (1988) revealed that only 16% of the original spawner-biomass-per-recruit remained, classing the galjoen as a collapsed fishery (Attwood 2003). In response to this and to the growing evidence that angling was having detrimental effects on many target species (Westera et al. 2003), portions of South Africa's coastline were set aside as no-take marine reserves, with the objective to protect depleted stocks so that they would recover fully and re-stock adjacent areas (Bennett and Attwood 1991).

The hope that these reserves will become a pristine ecosystem by reducing fishing to zero may be naïve because the dynamics of marine populations are made more complicated by the spatial movement and connectivity of these populations (Botsford et al. 2009). Knowledge about larval dispersal and the movement patterns of juvenile and adult fish, particularly the degree to which there is site fidelity as opposed to pure diffusion, is important when assessing the effectiveness of an MPA (Botsford et al. 2009). Previous work has shown that sedentary organisms or organisms that are mobile to a lesser extent, such as organisms displaying home range behaviour, are more likely to result in a positive outcome of an MPA (Kramer and Chapman 1999, Gell and Roberts 2003, Botsford et al. 2009). Mark-recapture studies have shown that most galjoen display a home ranging behaviour

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(with home ranges found to be not larger than 1.38km) while a small percentage of galjoen display a nomadic behaviour with individuals dispersing up to 1044km away from the area that they were tagged (Attwood and Bennett 1994, Bennett and Cowley 2005). This movement pattern can be explained by two models: the first model puts forward the notion that the population is polymorphic with respects to their dispersal behaviour, with some individuals displaying residency behaviour and others nomadic behaviour (Attwood and Bennett 1994, Attwood and Cowley 2005). The second model is the tourist model which does not differentiate between fish, but explains that each fish spends its time at a small number of widely separated sites and they move between these sites as conditions dictate (Bennett and Cowley 2005).

This home ranging behaviour may one of the causal factors for the apparent success of certain MPAs on this species. A study done by Bennett and Attwood (1991) assessed the effect of the De Hoop MPA in South Africa on the surf-zone fish assemblage by employing a BACI design. Specifically for the galjoen, CPUE rates increased 4 to 5 fold two years after the establishment of the MPA and remained stable at these high levels, which were similar to the unexploited levels. This suggests that the MPA caused an increase in galjoen density. A second study done by Attwood (2003) assessed the effect of MPAs on galjoen by sampling at four different sites along the South African coastline between 1987 and 2000. Three of the sites were in MPAs (the De Hoop Marine Protected Area and the Tsitsikamma National Park) while one site was the control/exploited site at Cape Point. CPUE was higher in the De Hoop MPA but lower in the Tsitsikamma National Park compared to the exploited site, while mortality was highest at the exploited site. Thus these MPAs differ to the exploited site with regard to density and age structure, however, these estimates are not necessarily transferrable between the MPAs and the exploited site because of significant habitat type differences (Attwood 2003).

This study aims to assess what effects the reserves at the Cape Peninsula are having on the galjoen population by comparing two sites that have minor habitat differences. Specifically I will test the hypotheses that (a) mortality rates will be different in the reserve compared to the exploited area, (b) CPUE rates for galjoen will be different in the reserve compared to the exploited area, and (c) mean galjoen size will be different between the reserve and the exploited area.

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## Methods

### *Field methods*

The Cape Peninsula ($34^{o}$20’ S, $18^{o}$24’ E) marks the border between the warm-temperate waters of the South-West coast and the cool-temperate waters of the West coast of South Africa. The western side of the peninsula is influenced by cool upwelling with a temperature range of 9-$16^{o}$C and on the eastern side of the peninsula there are the sun-warmed waters of False Bay with temperatures ranging between 12-$20^{o}$C (Attwood 2003). The shores are mainly composed of quartzitic sandstone rocks and platforms interspersed with small sandy and boulder beaches (Attwood 2003). The inshore waters support extensive kelp forests. Parts of the eastern and western shorelines of the Peninsula are no-take marine reserves, while other parts of the shoreline are exploited by recreational shore-anglers and to some extent spearfishermen.

A controlled shore-angling programme has been running on the Peninsula since 1986. Two anglers have fished in the April through to December months from a number of protected and exploited areas. Anglers used 3-4m fishing rods with multiplier reels loaded with 10-15kg breaking strain nylon fishing line. 100-150g lead sinkers were used to cast bait of white mussels *Donax serra,* red-bait *Pyura stolonifera,* and wonder-worm *Marphya* sp. on or near reefs in broken surf. Mustad 92570 hooks ranging in size from #1 to #2/0 were used.

All fish that were caught were measured to the nearest millimetre fork length with a rigid measuring-board. The number of hours fished per angler and the date for each fishing trip was recorded. Two sites from this data set were chosen in relatively close proximity (roughly 10kms) to each other – one from a reserve and one from an exploited area. Olifantsbos ($34^{o}$ 04’ S, $18^{0}$ 20’ E) was chosen as the protected site and Pegram’s Point ($34^{o}$ 19’ S, $18^{0}$ 26’ E) was chosen as the exploited site, with both sites being located on the western side of the Peninsula. Both sites comprise of small sandy beaches and rocky points and are both subject to regular south-easterly winds in the summer that creates large swell. Only data from the year 2000 onwards was used for standardisation purposes.

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### *Statistical analyses*

*CPUE standardisation using a general linear model:*

CPUE is often used as a proxy for relative fish density but it is affected by numerous other factors including fishing technique, the angler, seasonality, year, and area (Attwood 2003, Maunder and Punt 2004). Many effects were controlled for in the experimental design – fishing technique was standardised and fishing trips were conducted in the same season every year.  However, the other factors need to be excluded in order to compare only the effect of area on CPUE. Thus CPUE was standardised using a general linear model (GLM) in Statistica 11 (Copyright StatSoft, Inc. 1984-2012). Initially, the full model as shown in equation (1) was applied to the CPUE data. Terms that were not found to be statistically significant by themselves or in any interactions involving that term were excluded until the most parsimonious model was found (equation 2).

$$CPUE_{abc} = \alpha_0 + \beta_a + \beta_b + \beta_c + (\beta_a * \beta_b) + (\beta_a * \beta_c) + (\beta_b * \beta_c) + \varepsilon_{abc} \quad \text{with } \varepsilon_{ab} \sim N(0, \sigma^2) \; . \; . \; . \; . \; . \; . \; (1)$$

$$CPUE_{ab} = \alpha_0 + \beta_a + \beta_b + (\beta_a * \beta_b) + \varepsilon_{ab} \quad \text{with } \varepsilon_{ab} \sim N(0, \sigma^2) \; . \; . \; . \; . \; . \; (2)$$

Where, $CPUE_{abc}$ = CPUE (number of galjoen caught per hour per angler)
$\alpha_0$ = intercept (mean CPUE)
$\beta_a$ = effect of area (exploited and protected)
$\beta_b$ = effect of year (2000 – 2012)
$\beta_c$ = effect of angler (SW and CH)
$\varepsilon_{abc}$ = error term

Residuals were checked to be normally distributed using three plots, namely a histogram of the residuals, a normal probability plot, and a predicted vs. residual values plot.

*Length structure standardisation using a general linear model:*

The size of a fish that is caught will not only be affected by the area in which it is found, but by numerous other factors too. These factors ideally need to be excluded in order to compare the fish size structure between the areas. Thus fork length was standardised using a general linear model (GLM) in Statistica 11 (Copyright StatSoft, Inc. 1984-2012). Initially, the full model as shown in equation (3) was applied to the fork length data. Terms that were not found to be statistically significant by themselves or in any interactions involving that term were excluded until the most parsimonious model was found (equation 4). Some interaction terms could not be included in equation (4) because of missing data which created non-convergent problems in the parameter estimation of the GLM process.

$$\text{Log}\,(FL_{abc}) = \alpha_0 + \beta_a + \beta_b + \beta_c + (\beta_a * \beta_b) + (\beta_a * \beta_c) + (\beta_b * \beta_c) + \varepsilon_{abc} \quad \text{with } \varepsilon_{ab} \sim N(0, \sigma^2) \; . \; . \; . \; . \; . \; . \; (3)$$

$$\text{Log}\,(FL_{ab}) = \alpha_0 + \beta_a + \beta_b + \beta_c + (\beta_a * \beta_b) + \varepsilon_{ab} \quad \text{with } \varepsilon_{ab} \sim N(0, \sigma^2) \; . \; . \; . \; . \; . \; (4)$$

Where, $FL_{abc}$ = fork length (mm)
$\alpha_0$ = intercept (mean fork length)
$\beta_a$ = effect of area (exploited and protected)
$\beta_b$ = effect of year (2000 – 2012)
$\beta_c$ = effect of angler (SW and CH)
$\varepsilon_{abc}$ = error term

Residuals were checked to be normally distributed using three plots, namely a histogram of the residuals, a normal probability plot of the residuals, and a predicted vs. residual values plot.

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*Mortality rate*

The catch-at-length data can be transformed into catch-at-age data, of which the declining frequency is used to estimate the mortality rate. However this conversion is difficult in the case of galjoen because males and females grow at different rates (Bennett and Griffiths 1986). In addition, the sex of the fish was not recorded because it cannot be determined without killing the fish (Attwood 2003). However, a large enough random sample of fish should theoretically contain information about the total mortality rate as well as the relative proportion of each sex (Attwood 2003). Thus a likelihood function was used to estimate both these parameters from the catch-at-length data following the protocol set forth by Attwood (2003). In summary, the maximum likelihood estimates of the total instantaneous mortality rate (Z) and the proportion of newborn fish that are male (w), given the vector of catch-at-lengths (L), were obtained by minimizing the negative log-likelihood:

$$LLH\,(\mathrm{L} \mid \mathrm{Z}, \mathrm{w}) = -\sum_{i} ln\; p(\mathrm{L}_i) = -\sum_{i} ln\; (\mathrm{wZe}^{-\mathrm{Z(tm(Li) - tm\hat{}(300))}}\,\mathrm{dt_m/dL} + (1\text{-}\mathrm{w})\mathrm{Ze}^{-\mathrm{Z(tf(Li) - tf\hat{}(300))}}\,\mathrm{dt_f/dL})\;.\;.\;.\;.\;.\;.\;(5)$$

Where $p$(L) is the probability of selecting a fish of length L, $t_m$ is the age of the male, $t_f$ is the age of a female, and dt/dL is the rate of change of age with length. The model also took into account the fact that the sample of fish was not representative of the entire population but it is truncated, eliminating all lengths smaller than 300mm because of the angling technique.

## Results

A total of 2506.6 hours were spent fishing between 2000 and 2012 by the two chosen anglers and 3238 galjoen were caught for both the sites. Fork lengths for 2020 of those fish were recorded - fish that were found to have broken or missing tags, or missing barbs were not measured.

*CPUE*

Values of CPUE were higher at the protected site compared to the exploited site, with a mean $\pm$ standard deviation CPUE of 1.48 $\pm$ 0.85 galjoen.angler$^{-1}$.hour$^{-1}$ for the protected site and of 1.02 $\pm$ 0.81 galjoen.angler$^{-1}$.hour$^{-1}$ for the exploited site, with the difference being statistically significant (t-test for independent samples, $t_{s\ 0.05(1)}$ = -6.26, df = 507, p < 0.001) (Fig. 1).

The factors of 'year' and 'area' and the interaction between the two significantly affected the CPUE in the model described by equation (2). The 'area' parameter had the greatest explanatory power (Table 1) and after standardising other factors 'area' resulted in an increase in CPUE from 0.98 galjoen.angler$^{-1}$.hour$^{-1}$ in the exploited area to 1.53 galjoen.angler$^{-1}$.hour$^{-1}$ in the protected area (Fig. 3). When standardising other factors 'year' resulted in CPUE ranging from a minimum of 0.83 galjoen.angler$^{-1}$.hour$^{-1}$ in 2006 to a maximum of 1.96 galjoen.angler$^{-1}$.hour$^{-1}$ in 2001 (Fig. 3). When modelling the interaction between year and area, the modelled CPUE closely resembles the observed CPUE, which suggests that these two parameters explain most of the

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variation found in CPUE. CPUE was still found to be higher in the protected site compared to the exploited site for each year (Fig. 4). The fact that the ‘angler’ parameter or any interactions with it were found to be insignificant by the model described by equation (1) suggests that the variability in skill by the two anglers did not affect the CPUE. Because of this, the model was simplified by excluding the ‘angler’ parameter.

*Size structure*

The mean ± standard deviation fork length was significantly greater in the protected site (329.31 ± 40.19mm) compared to the exploited site (300.69 ± 34.71mm) (t-test for independent samples, $t_{s\ 0.05(1)} = 10.86$, df = 2017, $p < 0.001$) (Fig. 5).

Only the ‘year’ and ‘area’ parameter significantly affected the fork length in the model described by equation (4). The ‘angler’ parameter was not significant in the model and none of the interactions with ‘angler’ could be modelled with equation (3) due to missing data. This once again suggests that variability in the skill of the two anglers does not significantly affect the size of the fish that they caught, but the ‘angler’ parameter could not be excluded because it was not known whether an interaction with ‘angler’ would be significant or not. The ‘area’ parameter had the greatest explanatory power (Table 2) and after standardising for other factors ‘area’ accounted for an increase in size from 306.29mm in the exploited area to 326.71mm in the protected area (Fig. 6). The ‘year’ parameter caused fork length to fluctuate between a minimum of 301.71mm in 2001 to a maximum of 330.68mm in 2012 (Fig. 7) when other parameters were standardised. The interaction between ‘year’ and ‘area’ was not significant.

*Mortality rate*

The mortality rate estimates calculated from the size distribution can be ascribed to natural mortality in the protected area and natural mortality plus fishing mortality in the exploited area. Ten years has to pass to overcome the effect of earlier fishing in the protected site because the age of first capture for the fish is 4 years and their maximum age is a minimum of 13 years (Attwood 2003). The marine reserve was established in 1979 which is 21 years prior the first sample effort, thus the age structure of the fish in the reserve is devoid of fishing effects.

The smallest negative log-likelihood estimate for the exploited site corresponded to a sex ratio that ranged from 0.3 to 0.7 and a mortality estimate that ranged from 0.9 to 1.1 $\text{year}^{-1}$. The smallest negative log-likelihood estimate for the protected site corresponded to a sex ratio that ranged from 0.1 to 0.2 and a mortality estimate of 0.5 $\text{year}^{-1}$. However, because the two sites are located such a small distance apart, it can be assumed that the sex ratio for the two sites would be the same. Thus, instead of looking at the mortality rate corresponding to the absolute lowest negative log-likelihood value, the mortality rate corresponding to the lowest negative log-likelihood value for a sex-ratio of 0.4 (the median value between the most likely sex ratios for protected and

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exploited site) was chosen (table 3). Therefore the best mortality estimate for the protected site is 0.55 year$^{-1}$ and 1.0 year$^{-1}$ for the exploited site. By difference the fishing mortality in the exploited site is 0.45 year$^{-1}$.

## Discussion

### *Effects of the reserve on galjoen density and size*

CPUE is known to be a reliable and relative measure of reef fish density (Willis et al. 2000, Attwood 2003) and because galjoen are largely resident fish we can expect the CPUE rates to differ between exploited and protected areas (Attwood 2003). The CPUE increased roughly 1.5 times in the protected compared to the exploited area. This is a much lower estimate compared to the 4-5 fold CPUE increase found by Bennett and Attwood (1991) after the establishment of the De Hoop MPA, but is similar to the 1.5-2 fold increase in CPUE in the De Hoop MPA compared to the exploited Cape Peninsula site found by Attwood (2003).

The variation found in CPUE could only be attributed to the effect of area and year and the interaction between the two. A meta-analysis by Babcock et al. (2010) revealed that the increase in density of target species is detected on average 5.13 years after establishment of the MPA and that subsequently their density trajectories over time are variable, with some populations continuing to increase, others remaining stable, and some decreasing. We can expect this variation in density over time because of natural fluctuations due to differences in larval and food supply, fishing effects from outside the reserve, or differences in predation rates (Babcock et al. 2010). Despite this temporal variation, area had the highest explanatory power in the model which provides evidence that the reserve itself is one of the main causes for the increase in CPUE and thus the increase in galjoen density.

The average size of the fish is expected to be higher within the reserve because they are not fished out and thus grow larger (Halpern 2003, Gell and Roberts 2003). The average size of the galjoen increased roughly 1.1 times (or by 9.5%) in the reserve compared to the adjacent exploited area. In theory, this should mean that there is a higher amount of larval supply from the reserve because larger fish are more fecund (Westera et al. 2003). The variability in galjoen size was significantly affected by the year and area. One can expect the average size of the fish within a population to fluctuate annually because of differences in the proportion between juveniles and adults. In addition, one expects the average size of a residential fish to increase within the reserve as time goes on as the fish are allowed to age and grow (Halpern 2003). Despite this, area had the greatest explanatory power for the variability in size which again provides evidence that the MPA is a main causal factor in the increase of average organism size.

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Recreational line-fishing is known to be influenced by the skill of the angler (Attwood 2003, Westera et al. 2003). Despite this, the variable 'angler' was found to have no significant effect on the CPUE or galjoen size. This suggests that the two anglers' skills were similar enough to not cause significant differences.

It must be mentioned that the limitation with this study, like all studies involving a protected and exploited site, is the potential that differences in habitat are confounding these suggested effects of the reserve (Claudet 2011). Despite the fact that the two sites are located in relatively close proximity and the major habitat features are the same (i.e. same currents, water temperature, geographical features etc.), fish are known to respond to the amount of biogenic habitat, such as large kelps (Carr 1994). Thus if there are existing differences in the amount of kelp at the two sites, this could be influencing the variation in galjoen density. Various studies have attempted to quantify the amount of kelp along the South African coastline (Field et al. 1980, Levitt et al. 2002, Rothman et al. 2006, Anderson et al. 2007, Rothman et al. 2010) but there are only estimates of kelp biomass for the protected site and not the exploited site. Regardless, estimates of kelp biomass vary from a minimum of 1.7kg.m$^{-2}$ to a maximum 21.3kg.m$^{-2}$ at various sites along the western coast of the Cape Peninsula (Anderson et al. 2007). At the protected site of Olifantsbos itself, kelp biomass is estimated at 6, 14.4 and 15.1 kg.m$^{-2}$ at different sections of the site (Anderson et al. 2007). Because of these high levels of variability within a site, kelp biomass variability across sites might not be affecting the galjoen density. It is thus still likely that the MPA is having a positive significant effect on the organism density and size.

*Effects of the reserve on adjacent exploited areas*

Because the majority of fish display a home ranging behaviour with a small percentage of fish displaying dispersal behaviour (Attwood and Bennett 1994, Bennett and Cowley 2005), the population will benefit from being spatially protected within the reserve, but it also means that there is the potential of spill-over into adjacent exploited areas. Attwood's (2003) CPUE estimate for the exploited Cape Peninsula site for 1987 to 2000 is much lower than the CPUE for the exploited site found by this study for 2000 to 2012 (0.64 galjoen.angler$^{-1}$.hour$^{-1}$ and 1.02 galjoen.angler$^{-1}$.hour$^{-1}$ respectively). These CPUE values are somewhat comparable because the same fishing technique was used and there are minor habitat differences along the Cape Peninsula. Thus the higher estimate found in this study from 2000-2012 may be a potential indicator that there is spill-over of the stock from the reserves into these exploited areas which is increasing the CPUE and density. This is in line with Kramer and Chapman's (1999) prediction that fish density will rapidly increase in newly established reserves, and only later provide spill-over effects as density-dependent processes come into play. However, this hypothesis would need to be rigorously tested to determine if this increasing trend is not being affected by factors other than time.

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*Effects of fishing*

Galjoen were protected in the reserve for long enough to allow for at least one cohort of fish to reach maximum age without losses to fishing. As such the mortality rate in the reserve can be ascribed to natural mortality. The natural mortality rate estimate of 0.55 year$^{-1}$ falls in the range of Attwood's (2003) three estimates of 0.42 year$^{-1}$, 0.43 year$^{-1}$ and 0.61 year$^{-1}$ for three different protected sites along South Africa's South coast. However, it is important to note that the population structure within the reserve could have been affected by fishing in adjacent areas because of a small amount of exchange between the areas. One can assume that there will be less fish immigrating into the reserve than emigrating out by a factor of at least $e^{-F}$ (where $F$ is the fishing mortality) (Attwood 2003).

Because the mortality of the protected area can be taken as an estimate of natural mortality, the difference between this estimate and the mortality estimate of the exploited area can be ascribed to fishing mortality ($F$). Previous work has shown that female galjoen sustained twice the fishing pressure that male galjoen did in the late 1900's ($F$ of 1.08 and 0.53 year$^{-1}$ respectively) (Bennett 1988). This resulted in the likely scenario that females were being growth overfished while males were not (Bennett 1988, Attwood and Bennett 1990, 1995). Attwood (2003) found $F$ to be between 0.8 and 1.75 year$^{-1}$ in the late 1900's at Cape Point which was considered to be unsustainable. The $F$ estimate of 0.45 year$^{-1}$ from this study is lower than previously found and is close to the $F_{MSY}$ (the fishing rate that allows for maximum sustainable yield from the system) of 0.47 year$^{-1}$ estimated from a recruitment curve by Attwood and Bennett (1990). This may suggest that the stock is not currently over-exploited in the fished area however, estimates for $F_{MSY}$ can be regarded with levels of uncertainty (Botsford et al. 2009).

Reasons for this suggested decrease in $F$ are unclear – $F$ is assumed to have a linear relationship with the catchability of the fish (q) and the effort applied to catch the fish (E) ($F = q \times E$), both of which are largely independent of the amount of fish present (Rijnsdorp et al. 2006). For a reduction in $F$ either the catchability of the galjoen, the effort applied, or both would need to decrease. It is unlikely that the catchability decreased because it is dependent on gear efficiency (which has increased throughout the years) and the distribution of the fish in relation to the fisher (there is no evidence to suggest that galjoen or the anglers have changed their distribution). Thus, the only explanation for this reduction in $F$ is a reduction in angling effort, but angler surveys would need to be conducted to confirm this.

One of the major criticisms of MPA's is that they have the potential to displace rather than reduce fishing effort (Hilborn et al. 2004). If an area becomes closed to fishing there is a reduction in potential yield and if the effort is not reduced outside of the MPA, these stocks would become severely over-fished (Hilborn et al. 2004). The reduction in $F$ from the 1900's to the 2000's suggests that there has been a reduction in effort in the exploited study area, but there is the potential that anglers might be fishing more in other areas adjacent to the reserve.

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This would cause problems associated with effort displacement and would affect whether the increases in galjoen density and size within the reserve will translate to an overall increase in density throughout the Cape Peninsula.

*Conclusion*

Shore angling in South Africa is one of the largest participation sports and is a source of recreation to a wide spectrum of the population (Bennett 1991). However, because shore-angling is open access it is impossible to implement a total allowable catch and effort controls are difficult to monitor and enforce (Bennett 1991). Not only does this study add to the growing body of evidence that MPAs are beneficial to the recovery of targeted species, but it also provides the best evidence thus far that the marine reserves of Cape Point are having a positive effect on galjoen mortality, density and size. Increases in CPUE for the exploited area over time also suggest that the reserve is re-stocking adjacent exploited areas. Reductions in *F* over time suggest that angling effort has been reduced in one area, but it is unknown whether the effort has been displaced to other areas. Despite this, I put forward the notion that in this circumstance, an MPA is the optimal and most effective management and conservation tool for the galjoen stock.

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## Figures and tables

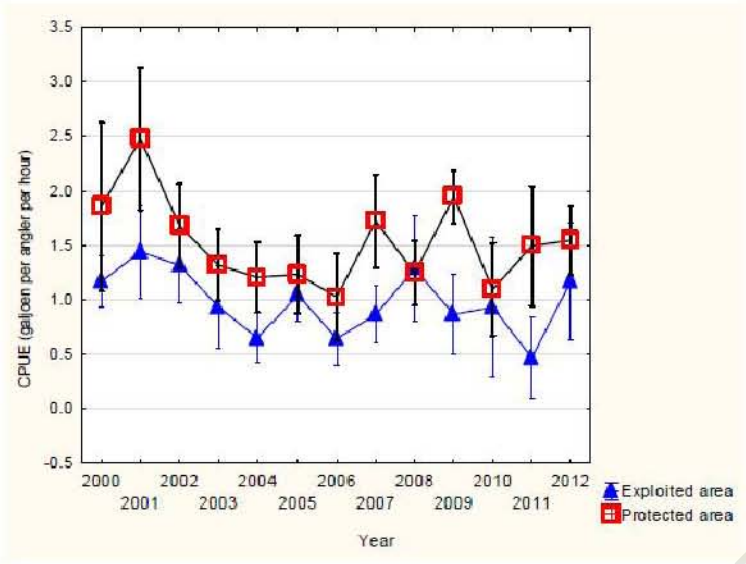

Figure 1: Mean CPUE for galjoen from 2000 – 2012 for the protected and exploited area

Table 1: Results of the GLM applied to the CPUE data for the protected and exploited sites, and the sum of squares and the likelihood ratio statistics calculated by the progressive inclusion of additional parameters in the model in the order listed, $p < 0.05$ indicates statistical significance

| Parameter | Sum of Squares | Degr. Of freedom | F | p |
|---|---|---|---|---|
| Intercept | 682.583 | 1 | 1086.182 | 0.0000 |
| Year*AREA | 14.261 | 12 | 1.891 | 0.0332 |
| Year | 34.177 | 12 | 4.532 | 0.0000 |
| AREA | 31.990 | 1 | 50.905 | 0.0000 |
| Error | 303.529 | 483 | | |

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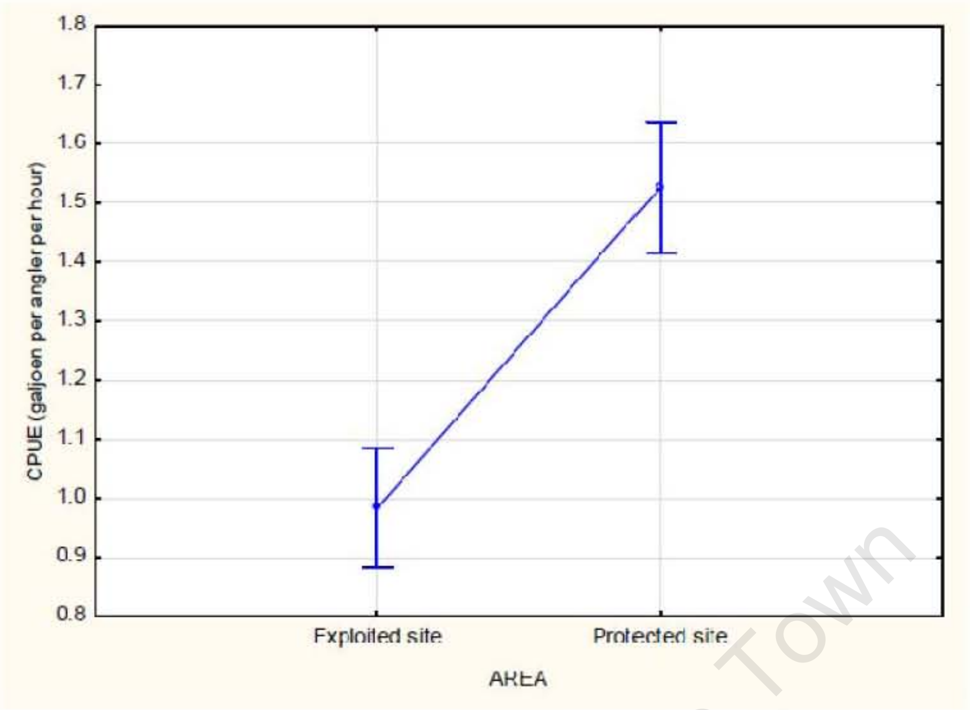

Figure 2: GLM predicted least squares mean CPUE for galjoen for the protected and exploited site, with CPUE modeled only with parameter 'area'. Vertical bars denote 95% confidence interval.

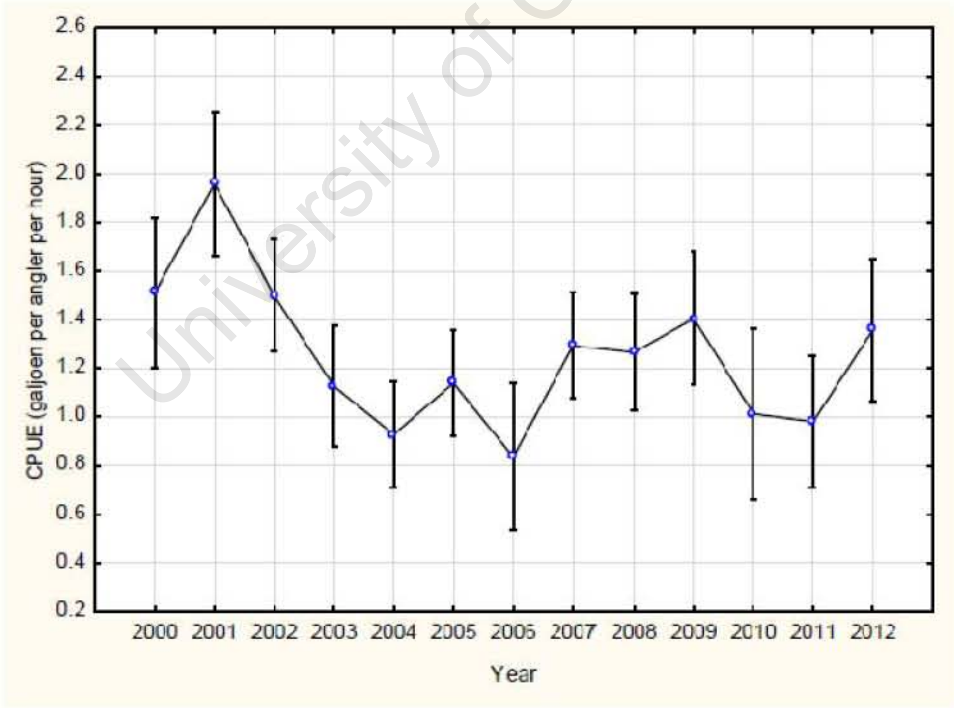

Figure 3: GLM predicted least squares mean CPUE for galjoen for 2000 to 2012, with CPUE modeled only with parameter 'year'. Vertical bars denote 95% confidence interval.

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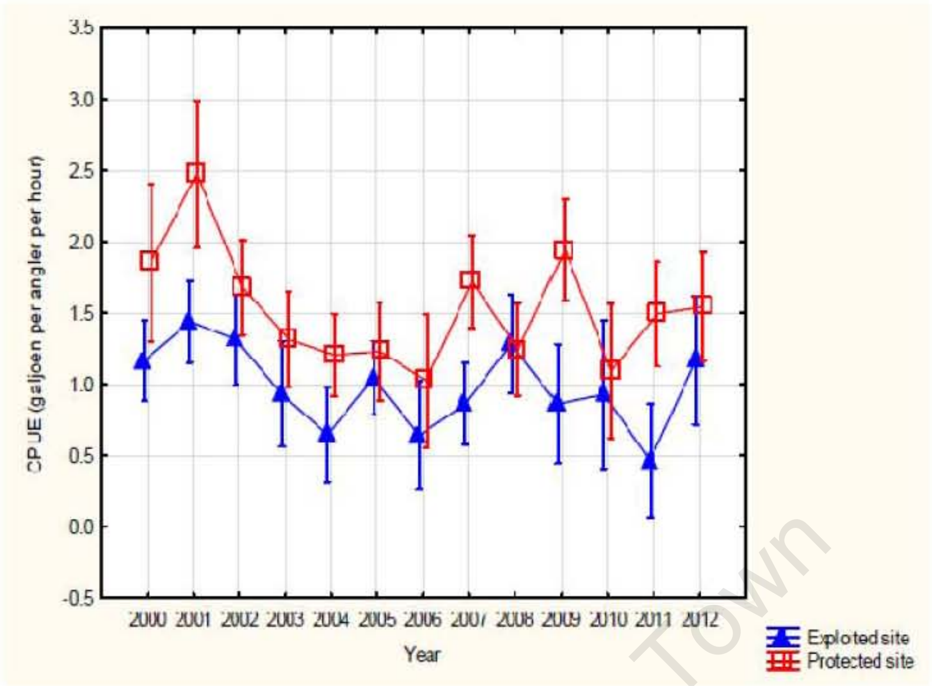

Figure 4: GLM predicted least squares mean CPUE of galjoen for 2000 to 2012 for the protected and exploited site, with CPUE modeled only with interaction between 'area' and 'year'. Vertical bars denote 95% confidence interval.

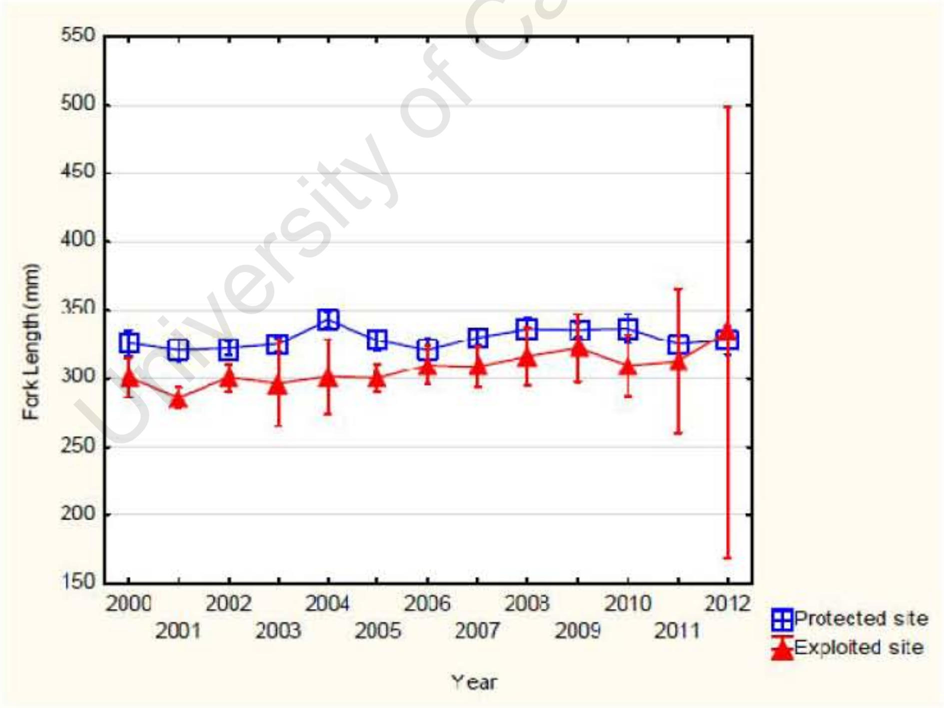

Figure 5: Mean fork length of galjoen from 2000 to 2012 for the protected and unexploited site. Vertical bars denote 95% confidence interval.

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Table 2: Results of the GLM applied to the fork length data for the protected and exploited sites, and the sum of squares and the likelihood ratio statistics calculated by the progressive inclusion of additional parameters in the model in the order listed, $p < 0.05$ indicates statistical significance

| Parameter | Sum of Squares | Degr. Of freedom | F | p |
|---|---|---|---|---|
| Intercept | 1642.639 | 1 | 645008.8 | 0.000000 |
| Year | 0.088 | 13 | 2.7 | 0.001096 |
| Location | 0.051 | 1 | 19.9 | 0.000009 |
| Year*Location | 0.030 | 13 | 0.9 | 0.557841 |
| Angler | 0.003 | 1 | 1.1 | 0.285156 |
| Error | 5.134 | 2016 | | |

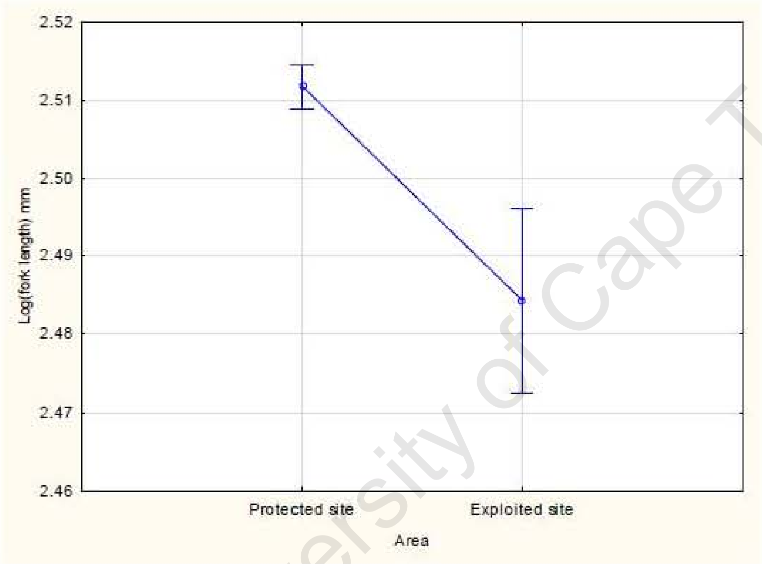

Figure 6: GLM predicted least squares mean fork length of galjoen for protected and exploited site, with fork length modeled only with parameter 'area'. Vertical bars denote 95% confidence interval.

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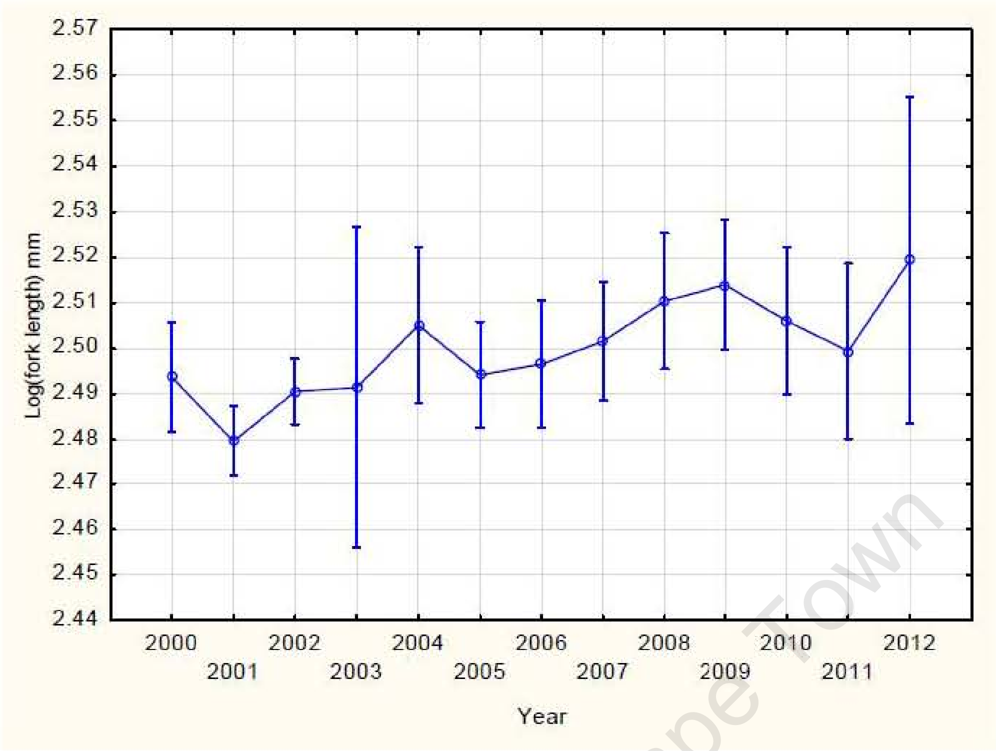

Figure 7: GLM predicted least squares mean fork length of galjoen for 2000 to 2012, with fork length modeled only with parameter 'year'. Vertical bars denote 95% confidence interval.

Table 3: Total mortality rate estimated from galjoen size distribution (total length ≥300mm) for the exploited and protected site

| Site | N | Z | w | -LLH |
|---|---|---|---|---|
| Exploited | 258 | 1 | 0.4 | 930 |
| Protected | 1761 | 0.55 | 0.4 | 8364 |

Where N = Sample size
Z = instantaneous mortality rate (year$^{-1}$)
w = male fraction at recruitment
-LLH = negative log-likelihood

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