REPORT

OF THE INTERNATIONAL REVIEW PANEL REGARDING FISHING CLOSURES ADJACENT TO SOUTH AFRICA'S AFRICAN PENGUIN BREEDING COLONIES AND DECLINES IN THE PENGUIN POPULATION

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OF THE INTERNATIONAL REVIEW PANEL REGARDING FISHING CLOSURES ADJACENT TO SOUTH AFRICA'S AFRICAN PENGUIN BREEDING COLONIES
AND
DECLINES IN THE PENGUIN POPULATION

DEPARTMENT OF FORESTRY, FISHERIES
AND THE ENVIRONMENT (DFFE)

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EXECUTIVE SUMMARY

- The population of African penguins breeding in South Africa has been declining rapidly (approximately 8% per annum since 2005) and is consequently at a high risk of extinction in the wild in the coming decades. It is essential to understand and mitigate the primary factors leading to this decline.

- Considerable effort has been made by the fishing and conservation sectors in collaboration with government to understand the causes of the decline and how they might be mitigated. The Panel commends South Africa on its world-leading efforts to underpin challenging utilisation-conservation policy decisions with sound science.

- Implementation of closures managed within the Island Closure Experiment (ICE) aimed to understand whether reducing fishing around islands with penguin breeding colonies would help to reduce the current rate of decline. This internationally-recognised experiment involved implementing an alternating pattern of closures around four island breeding colonies on the South African west and south coasts. It is now complete and, notwithstanding the difficulties implementing the experiment, has been successful in demonstrating for the west colonies of Dassen and Robben islands (those more intensively studied within the ICE), that excluding fishing around island breeding colonies is likely to reduce the rate of decline in the population to a small extent, mediated through improvements in reproductive success. Excluding purse-seine fishing around island breeding colonies is also likely to have other positive benefits for penguin conservation, such as facilitating higher adult survival, but the ICE was not designed to estimate such effects.

- The Panel recognises that closure of purse-seine fisheries around penguin colonies will provide only a part of the measures required to slow or reverse the population decline of African penguins.

- There is a trade-off amongst maximising benefits to penguins, minimising the costs to the fishing industry, and having a reliable basis to quantify the effects of closures (including no closures) on the penguin recovery rate. The trade-off among closure options is a policy decision related to conservation, economic and social goals and objectives for South Africa. This report outlines some aspects that could form part of a decision-making framework to identify the closure options that will provide the best outcomes for penguins given some level of cost to the fishing industry.

- The effects of alternative fishery closure designs differ amongst the island breeding colonies, in terms of reducing the rate of decline, costs to the fishing industry, and social impacts. Hence, advice related to the effects of possible closure options is presented by island breeding colony, and not simply at the regional or national level; decisions on closures should also be made by colony, taking account of the unique aspects of the fishery and threats at each colony.

- The impacts to the fishing industry can be evaluated using an "opportunity-based model" (OBM) that predicts the proportion of the catch of pelagic fish in closure areas that cannot be "replaced" by fishing outside these areas, together with a Social Accounting Matrix (SAM) model that converts "lost catch" into economic impacts (loss of GDP and jobs) on the fishery, suppliers of goods and services to the fishing industry, and the broader economy. The OBM and SAM model can be used to rank closure options in terms of economic effects but the OBM likely overestimates the potential lost opportunities outside the closed area on a given day. The Panel remains concerned about: (i) the lack of information on how the closures impact fishing costs and fishing behaviour; (ii) the ability of the SAM model to adequately attribute impacts at the scale of fishing communities; and (iii) that there are social impacts that are not estimated using the SAM, but are important to consider in any trade-off analysis.

- Evidence suggests that catches from within closure areas will be more difficult to replace around Dyer Island and St Croix Island than around the other remaining five colonies with important breeding populations. Evidence also suggests that levels of lost catch can be reduced, if closures around penguin preferred habitats are well designed.

- The Panel identified (in this report) recommendations related to future monitoring of penguin colonies and research to understand the effects of closures on the change in penguin numbers and costs to the fishing industry and local communities.

- Further attempts were made to identify consensus closure options among the fishing and conservation sectors during the Panel meeting and ongoing efforts to identify such options are encouraged, particularly as closures may need to be adjusted given the results of future monitoring.

- The Panel strongly encouraged continued communication, and collaboration, with transparency of research data and analyses, as a means to build trust and strengthen these discussions. Working collaboratively will further enhance the effectiveness and social acceptability of management measures and decisions aimed at mitigating the decline of the African penguin.

1. BACKGROUND

1.1 Historical decline of African penguins

The African penguin, *Spheniscus demersus*, breeds only in Namibia and South Africa, where it is restricted to coastal waters, except over the Agulhas Bank where its preferred prey may occur further offshore. Their usual non-breeding habitat is also highly coastal, spanning ~3 200 km of coastal Namibia and South Africa, but with the occasional individual recorded as far north as Gabon, in the west, and Mozambique, in the east (Crawford et al., 2013).

In the 1920s, the African penguin may have had an estimated breeding population as large as between ~500 000 and ~1 000 000 pairs. The population subsequently decreased so that almost a century later less than ~20 000 pairs remained, of which ~25% were in Namibia and ~75% in South Africa (Coetzee et al., 2021a). As a consequence of the marked population declines across both these range states, the species was classified in 2010 as Endangered on the Red List of the International Union for Conservation of Nature (IUCN, 2018). The IUCN has not made regional assessments, but these would almost certainly show the species to be of even greater conservation concern in some parts of its range.

As recently as 2004, ~52 000 pairs of African penguins could be found at 19 breeding localities in South Africa, but by 2019 the population had fallen to ~13 200 pairs, with five colonies becoming extinct (Coetzee et al., 2021a; see Figure 1.1 for a map of the breeding colonies referred to in this report). The latest counts from 2022 show the decline continuing, with an estimated breeding population of ~10 000 pairs (Masotla et al., 2023). Further, the small size of the remaining colonies means that all now face a substantial probability of extinction; indeed, it is anticipated that a further seven colonies will become extinct in the near future (Coetzee et al., 2021a). Coetzee et al. (2021a) also note that stemming the population decline at the larger remaining colonies therefore represents the best means of maintaining the species in the wild, and that if current population trajectories continue, the species could be functionally extinct by 2035.

The latest population surveys in 2022 reported that seven colonies collectively held more than 95% of the remaining population in South Africa (Masotla et al., 2023): Dassen Island (2 513 pairs [25.1%]), Robben Island (991 [9.9%]), Boulders Beach (891 [8.9%]), Stony Point (1 565 [15.6%]), Dyer Island (1 026 [10.25]), St Croix Island (1 262 [12.6%]) and Bird Island (1 437 [14.4%]).

Against this background, it is important to recognise that a decline in the numbers of African penguins is not inevitable. Between 1987 and 2004, the number of adult African penguins at west coast sites in South Africa increased from 7 500 to 33 000 (Sherley et al., 2020; Figure 1.2). It is evident that numbers can increase during periods when conditions are favourable, but that this has rarely been the case in recent decades.

1.2 Summary of basic penguin population and feeding ecology

African penguins generally commence breeding aged around 5 to 6, but unsuccessful breeding attempts at earlier ages are also known. They can continue breeding past age 20, although this is probably uncommon (Crawford et al., 2013). Adult survival, breeding propensity and reproductive output are all highly variable, with reported links to food availability (Crawford et al., 2013). Juvenile survival, as with many seabirds, is lower in the first year after fledging (Crawford et al., 2013).

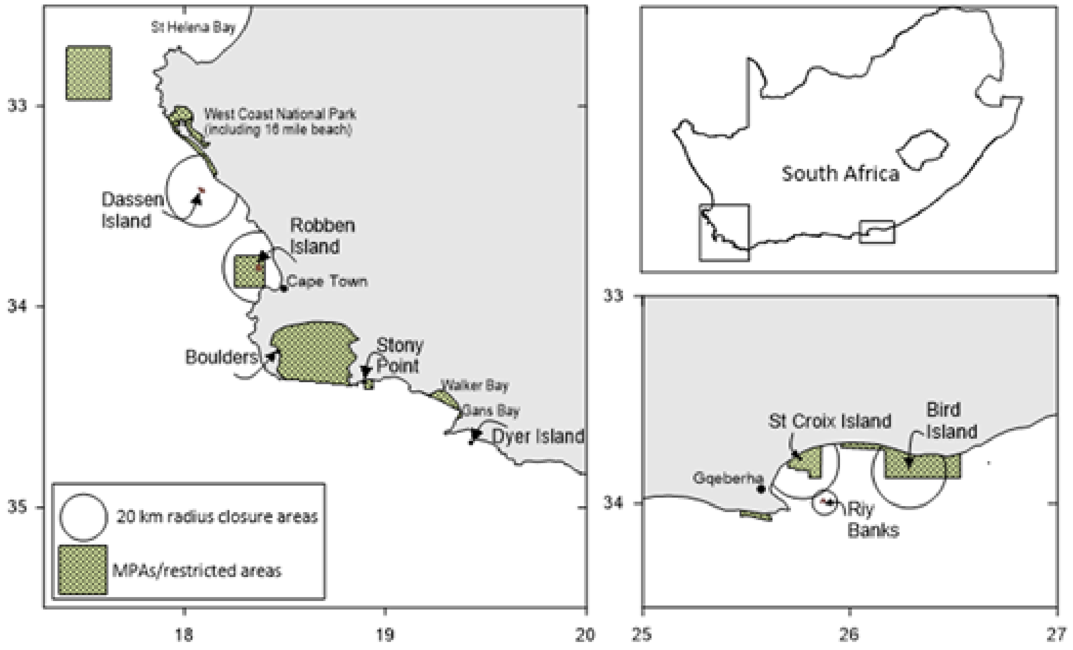

Figure 1.1: Map (courtesy of J Coetzee) of southern Africa showing the location of the breeding colonies for African penguins off South Africa

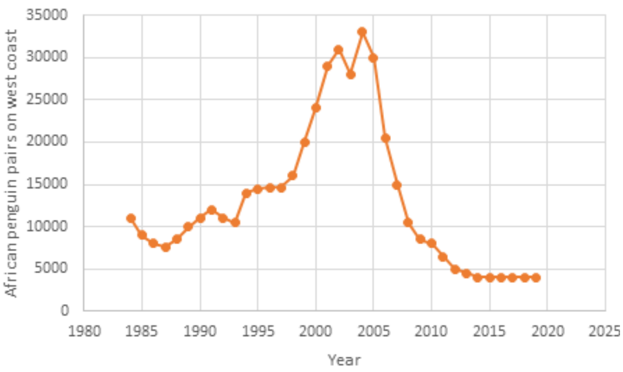

Figure 1.2: Total number of pairs of African penguins at all known west coast sites between 1984 and 2019 (data from Sherley et al., 2020)

Access to energy-dense prey (small pelagic fish) is critical to African penguins at multiple different times of year. Two periods are particularly demanding: moult and breeding. Adult penguins must build up their body reserves prior to moult, as they cannot enter the water to obtain prey within this 21-day fasting period, during which time they replace their entire plumage (Crawford et al., 2013). Moult tends to be synchronized at most individual localities, although the timing varies among localities. At Dassen Island, the peak moult is August–November; at Robben Island and Boulders Beach, most birds moult from November; at Dyer and St. Croix islands, peak moult is October–December; while a large proportion of birds at Bird Island start moult in September (Crawford et al., 2013). At all localities, most immature birds moult in October–March (Crawford et al., 2013).

Adequate prey is also important prior to and during breeding. Females must accumulate the resources necessary for egg production, whilst both parents must accumulate sufficient reserves to ensure they can repeatedly stay ashore whilst incubating, brooding or guarding their offspring. Incubation lasts 38–41 days and is shared equally by both sexes; chicks are brooded by adults until about 10 days after hatching; from 26–30 days, chicks are often left unguarded and may form crèches of up to 25 chicks; chicks fledge when between 55–130 days old (Crawford et al., 2013). During breeding, adults can sacrifice their own body condition to a certain extent, but generally not to the point beyond which their own survival is compromised (c.f. Southwell et al., 2015). Therefore, during breeding, and immediately post breeding, adequate resources are necessary to ensure adult maintenance, chick growth, and eventually to ensure independent chicks can forage successfully whilst still naïve, and adults can recover lost condition. African penguin breeding can occur throughout the year, with a second clutch possible, or with adults relaying if their first clutch is lost (Crawford et al., 2013). At Dassen Island, eggs are mostly laid in December–June, with most chicks during January–August; at Robben Island, eggs are laid in January–August, with chicks abundant in April–September; and at St. Croix Island, egg laying peaks in January (Crawford et al., 2013). Thus, as with moult, peak breeding time differs between sites.

When foraging, African penguins feed alone or in small groups and sometimes in conjunction with other seabirds. They are visual hunters but may use other cues to locate prey. Most dives are shallower than 30 m deep, although some may reach 85 m, lasting up to 2.5 minutes (Crawford et al., 2013). Almost all dives occur during daylight with virtually none at night. Adults provisioning young chicks generally forage within 40 km of their colony, but may travel up to 120 km, swimming at speeds of just under 2 m s^{-1}, or up to 5 m s^{-1} in short bursts (Crawford et al., 2013). Local forage fish abundance based on hydro-acoustic surveys has been shown to explain around 60% of the variation in time spent diving for penguins foraging within two days of the survey (Campbell et al., 2019). Penguin foraging effort (time spent diving, number of wiggles per trip, number of foraging dives and the maximum distance travelled) increased as forage fish abundance declined; in addition, quantile regression revealed that variation in foraging effort increased as prey abundance around the colony declined (Campbell et al., 2019).

Locating prey at sea is complex. Physical ocean features, such as thermoclines, are often used as foraging cues by marine predators, as these concentrate and hence increase the likelihood of locating prey. This is also true for African penguins, which have been shown to forage at and below the thermocline even though its depth and gradient may shift over time; indeed, penguins dive deeper in search of prey when there is no thermocline (van Eeden et al., 2016). Such physical cues are therefore important. However, olfactory cues have also been shown to be important. Dimethyl sulphide (DMS), an organo-sulphur compound released when phytoplankton are grazed, is known to attract seabirds (Nevitt et al., 2004), including African penguins (Wright et al., 2011). DMS-scented oil slicks attracted 2–3 times more penguins than control slicks, whereas penguins showed no response to slicks containing cod liver oil. The number of penguins attracted to DMS increased for at least 30 min, suggesting penguins could travel up to 2 km to reach scent cues. Such results also support the hypothesis that African penguins use DMS as an olfactory cue to locate prey patches at sea from a distance, which is particularly important given their slow commuting speed, relative to that of flying seabirds (Wright et al., 2011).

African penguins are known to hunt either independently or cooperatively, pursuing both solitary as well as schooling pelagic fish (McInnes et al., 2017). The most profitable foraging involves herding of fish, compressing schools upwards during the ascent phase of a dive where most prey

Pelagic fish (photo credit Carl van der Lingen) captures then constitute isolated fish, separated from the main school (McInnes et al., 2017). Catch-per-unit-effort for penguins is significantly improved when targeting schools rather than solitary fish, especially when penguins forage in groups. It appears that African penguins have evolved specialist hunting strategies closely linked to their primary reliance on schooling pelagic fish (McInnes et al., 2017). As penguins drive prey to the surface, it is also likely to enhance the foraging efficiency of flying seabird species (McInnes and Pistorius, 2019). As such, penguins may be integral to important processes that influence the structure and integrity of marine communities. Importantly, if group foraging confers an advantage to African penguins, then dwindling populations may suffer from an Allee effect as colonies become too small to support sufficient densities of birds for foraging groups to form (Ryan et al., 2012).

Predicting how populations respond to their environment requires detailed knowledge of demographic traits, such as survival and reproduction. However, translating foraging efficiency into demographic responses remains challenging for most marine predators, including African penguins. However, for macaroni penguins, Horswill et al. (2017) have shown that when prey availability is low, foraging trips are significantly longer and extend overnight; birds forage farther from the colony, potentially to reach more-distant foraging grounds, and allow for increased search times. These extended foraging trips are also linked to a marked decrease in fledgling weight, most likely associated with reduced rates of provisioning (Horswill et al., 2017). Further, work on the same macaroni penguin population suggests that lowered first-year survival rates are, at least partially, associated with lower fledgling masses (Horswill et al., 2014).

Declines in African penguin numbers might be caused by low survival rates of penguins or by low breeding success, or a combination of these. Survival rates of adult African penguins can be estimated by analysis of re-sightings (either visual or electronic) of individually-marked birds. Survival of adult African penguins has in many recent years been considerably lower than is typical for seabird species, suggesting that factors reducing adult survival are likely to contribute to the observed population decline. Although monitored survival rates do not appear to indicate any correlation with anchovy, *Engraulis encrasicolus*, stock biomass, a strong correlation between adult survival and sardine, *Sardinops sagax*, stock biomass has been reported by Robinson et al. (2015) and by Crawford et al. (2022). Both studies found little relationship between adult survival and sardine stock biomass in years when stock biomass was average, or above average, but found very low adult survival in most years of particularly low sardine stock biomass.

1.3 Hypotheses related to how fisheries can impact penguin populations

1.3.1 Fishery related hypotheses

There is a considerable literature related to the effects of marine capture fisheries on seabird population processes (e.g., Montevecchi, 2002; Cury et al., 2011; Sydeman et al., 2017). However, for some processes relatively few studies have access to data appropriately matched to predator needs in both space and time (see Trathan et al., 2022). Nevertheless, it remains axiomatic that fisheries have the potential to disrupt seabird population processes. The primary impacts on predators can be characterized as either negative (e.g., bycatch, resource competition), or positive (e.g., discard provisioning), whilst converse impacts of seabirds on fisheries also exist (e.g., bait stealing); see Montevecchi (2002) for a more detailed summary. However, in terms of purse seiners targeting small pelagic fish and interactions with African penguins, the most important interactions are likely to be related to bycatch and resource competition, or possibly to disturbance of group foraging by penguins. African penguins have not been recorded as bycatch in South Africa, which may be due to a combination of spatio-temporal separation of foraging (during the day) and fishing (mostly at night) and net avoidance behaviour. In contrast, resource competition is perceived to be a major cause of African penguin decline by some authors (e.g., Sydeman et al., 2021, and cited references therein), although this is contested (Butterworth and Ross-Gillespie, 2022, and cited references therein). Disturbance of group foraging, unrelated to any prey depletion effects, could possibly occur if groups of penguins were disturbed or displaced by fishing vessels, or if their group coordination and communication while hunting was affected because of noise.

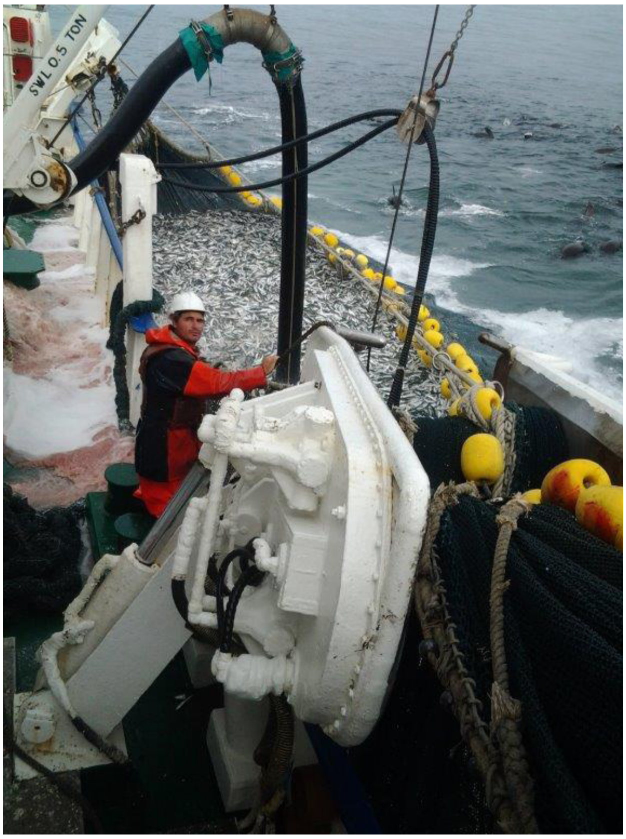

Photo credit SAPFIA – South African Pelagic Fishing Industry Association

Resource competition plausibly could happen through reductions in local prey biomass, or disruption of the prey field so that preferred foraging opportunities are diminished. For example, removal of parts or even whole shoals of schooling fish would diminish local prey biomass and specifically the prey aggregation states thought to be most attractive to penguins. However, key to the realized impact on penguins will be the rates by which local prey are replaced via regional advection or directional movement of prey and diurnal prey migrations. This means that a key aspect of management must be to consider the relative rates of various ecological processes related to prey availability.

Information documenting advection or directional movement of small pelagic fish is sparse. However, along the coast of South Africa, headlands and embayments interact with the oceanographic flow of the coastal countercurrent and shelf-edge jet currents, leading to areas of retention (Hutchings et al., 2002; Kirkman et al., 2016). Such complexities are key to understanding the local movements of fish as they come within the foraging ambit of a given penguin colony, replenishing the prey field depleted by penguins, other predators, or fisheries. Moreover, the African penguin, in common with other penguins, undergoes periods of positive and negative energy balance as they accumulate, or lose, body weight during reproduction (e.g., Southwell et al., 2015). Consequently, depletion of prey, whether due to natural predation or through resource interactions with fisheries, is likely to have variable consequences depending upon the exact timing in relation to breeding, or seasonal prey movement.

Thus, identification of how fisheries impact African penguin populations, particularly foraging, is complex, resulting from interactions between the timing and stage of moult, or breeding, at a given colony (e.g., Crawford et al., 2013; Southwell et al., 2015), the availability of prey locally (e.g., Campbell et al., 2019), advection and transport (e.g., Hutchings et al., 2002; Kirkman et al., 2016), as well as penguin foraging efficiency (e.g., McInnes et al., 2017).

1.3.2 Other hypotheses

1.3.2.1 Forage fish abundance

Butterworth et al. (2015), based on counts of moulting penguins and re-sightings of tagged penguins at Robben Island (Robinson et al., 2015), found that the primary reason for the post-2003 penguin decline was an increase in adult mortality, which they attributed to reduced abundance of sardine off the South African west coast. Analysis of African penguin annual mortality rate at Robben Island in relation to 1+ sardine stock biomass scaled to the maximum November survey estimate of 1 343 000 t in 2003 (Figure 1.4) showed no change in penguin mortality when sardine biomass exceeded about 25 to 30% of the maximum biomass (penguin annual mortality varied among years around a mean of about 15% per annum but with no trend in relation to sardine abundance). However, penguin mortality increased rapidly as sardine biomass fell below 25 to 30% of maximum biomass. Penguin annual mortality was estimated by Robinson et al. (2015) to be about 27% at a sardine biomass index of 20%, and about 55% at a sardine biomass index of 10% (Figure 1.3). Observed (and predicted) mortality exceeding 50% in years with sardine biomass below 10% of maximum represents a very unusual situation for any seabird species, as seabirds are normally long-lived, with adult survival rates typically around 0.8 or more.

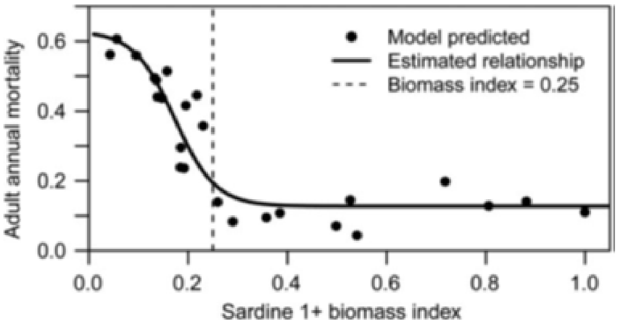

Figure 1.3: The estimated relationship between the 1+ sardine biomass index and penguin adult mortality (from Robinson et al., 2015).

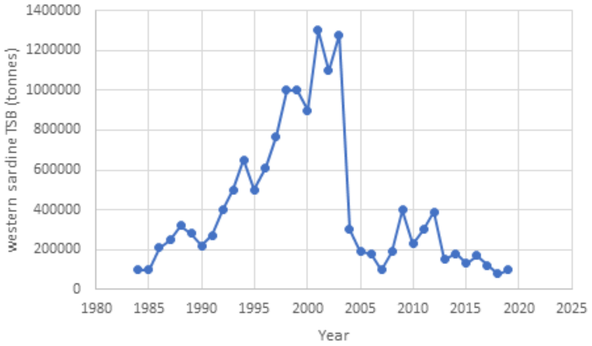

Figure 1.4: Estimated total stock biomass (TSB) of western sardine from 1984 to 2019 (data from de Moor, 2021 and Coetzee et al., 2022).

Crawford et al. (2022) found that penguin survival was around 0.8 when sardine stock biomass was average or above average but declined strongly with sardine standardised stock biomasses below 40% of maximum biomass, results similar to those previously shown by Robinson et al. (2015) but based on more years of data and from two colonies (Dassen and Robben islands). Perhaps surprisingly, there seems to be no clear correlation between African penguin survival and anchovy stock biomass, suggesting that sardine may be the key forage fish determining pen-

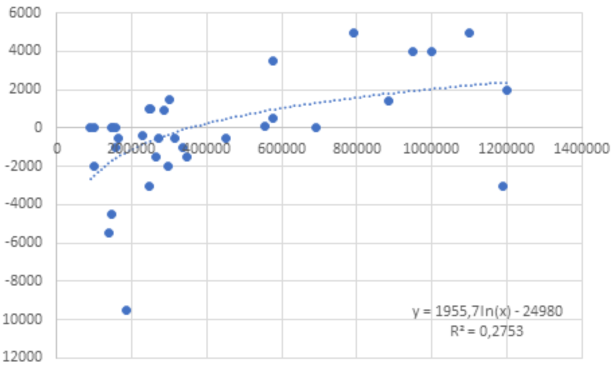

Figure 1.5: Change in numbers of pairs of African penguins between successive years (y-axis) off the South African west coast in relation to sardine total spawning biomass averaged over the year and previous year (x-axis, tonnes of total stock biomass). The dotted line is the best fit logarithmic regression. Penguin data from Sherley et al. (2020), sardine data from de Moor (2021) and Coetzee et al. (2022).

guin survival (possibly due to its higher energetic content; Balmelli and Wickens 1994).

The changes in numbers of African penguins (Figure 1.2) show a close similarity to changes in western sardine total stock biomass (Figure 1.4).

Figures 1.2 and 1.4 suggest that breeding numbers of African penguins may be strongly influenced by western sardine total spawning biomass, although this is correlational evidence so inferring a causal relationship is hazardous. Plotting the change in penguin numbers from one year to the next in relation to western sardine spawning biomass averaged over the year and previous year (Figure 1.5) and fitting a regression line to these data, indicates that breeding numbers of penguins increased in almost all years when sardine spawning biomass averaged more than about 350 000 t but decreased in most years when spawning biomass was below about 350 000 t. As inferred by Butterworth et al. (2015), these data also suggest that western sardine spawning biomass may have been one of the most important drivers of change in west coast African penguin numbers (but noting considerable noise in the data in Figure 1.5).

In relation to sardine stock dynamics, de Moor and Butterworth (2015) concluded *“Importantly, however, average recruitment for the west stock declines for spawning stock biomasses below about 800 000 t”*. Similar strong relationships where recruitment reduces rapidly at low spawning stock biomass exist for other sardines (e.g., Japanese sardine, Bai et al. 2022; Pacific sardine, McClatchie et al., 2010).

In order to ensure long-term sustainability of the western South African sardine stock, it is important to avoid depleting stock biomass below 800 000 t because recruitment from significantly smaller stock biomasses will be likely to be greatly reduced, resulting in prolonged depletion of the stock with limited potential for recovery. In that context, it is noteworthy that, rather than reducing fishing mortality continuously as stock biomass falls to low levels, the harvest control rule (HCR) for this stock allows increasing fishing mortality to be imposed as the stock biomass falls from 524 000 t to 300 000 t (Coetzee et al., 2022). A consequence of this HCR is that the exploitation rate peaked at >70% of estimated stock biomass in 2016 (de Moor, 2021) despite stock biomass being below 200 000 t and therefore already at risk of depressed recruitment. This depletion by the fishery is likely to have reduced the prospects for stock recovery by reducing future recruitment (see, for example, Essington et al., 2015). The implication of that is not only that the available stock biomass for fishing has had limited potential for recovery to allow greater Total Allowable Catches (TACs) because of impaired recruitment, but also that the reduced sardine stock biomass will have impacted African penguin adult survival (Robinson et al., 2015), contributing to the severe decline in breeding numbers of African penguins. Based on the available evidence (de Moor and Butterworth, 2015; Robinson et al., 2015; de Moor, 2021) lower survival and low sardine biomass appears to have been likely to have been one of, and possibly the single, most powerful driver of African penguin population dynamics in recent years, at least at Robben Island.

Further, prey capture, adult survival, the amount delivered to chicks, reproductive success, and other vital rates, all depend upon another set of important ecological interactions, including parental age and experience (e.g., Ainley, 2002). In a declining population, such as for African penguins, juvenile recruitment is vital; indeed, within a given year, penguins fledging with heavier body masses are likely to show higher survival rates than birds fledging lighter (Horswill et al., 2014). Thus, the individual quality of parents and juveniles becomes important, where individual quality is linked to different performance levels consistent throughout life (Lescroël et al., 2009). Seabirds respond to environmental changes by adjusting their breeding and foraging strategies (Cohen et al., 2014), and relationships exist between adult survival and quality, such that population demographic patterns affected by factors at the individuals’ level (e.g., individual quality) may be obscured at the population-scale level (Lescroël et al., 2009). Also, for a given population, life-history trade-offs that connect different aspects of a population’s demography may be important (Horswill et al., 2021).

Life-history theory suggests that long-lived animals (which include seabird species) should buffer their adult survival by abandoning breeding efforts if conditions are likely to have an adverse effect on adult survival, but several studies show empirical evidence of adult survival as well as breeding success of seabirds being reduced by low abundance of their preferred prey (e.g., Oro and Furness, 2002; Frederiksen et al., 2004; Davis et al., 2005). In an analogous manner, fisheries should respond to ecosystem conditions, especially for small pelagic fishes such as anchovy and sardine, which are typified by ‘boom and bust’ population dynamics that arise from inherent variability in their recruitment strength and short life-spans. For example, from the mid-1980s until the early-2020s, sardine biomass on both the west coast and south coast of South Africa was at low historical levels, apart from during a short period from the late-1990s, until the early-2000s (Coetzee et al., 2021a). Subsequently, fishery catches increased, as did the exploitation rate (Coetzee et al., 2021a).

1.3.2.2 Egg collecting and guano harvests

Egg collecting was a pressure but is no longer an issue. Loss of nesting habitat as a result of guano harvesting has reduced the suitability of available nest sites over many decades of guano removal. Guano harvests ended decades ago, but the legacy is that African penguins now breed in sites where they are more exposed to predators, nest flooding or overheating.

1.3.2.3 Predation

Predation by avian predators (especially kelp gulls) and by introduced alien mammal predators (such as feral cats, rats, dogs) occurs at some colonies, mainly affecting survival of eggs and chicks. Predation also occurs at sea, with penguins in some areas vulnerable to predation by Cape fur seals. Predation on adult penguins by Cape fur seals has been particularly frequent at Dyer Island. During 2004 and in 2006–2007 Cape fur seals were estimated to kill about 7% of adult African penguins, mostly when penguins were returning to the colony in the evening to feed chicks

(Makhado et al., 2013). Previous estimates of this mortality were 9% in 1994–1996 and 2 to 2.5% in 1999–2001 (Makhado et al., 2013). The predation is thought to be mainly by a small number of immature male Cape fur seals. It is considered to be a learned behaviour, and Makhado et al. (2009) suggest that the removal of these ‘problem’ seals may be an appropriate management response. That would appear to have the potential to reduce adult mortality by a significant amount at Dyer Island, but possibly would have relatively little benefit at most other colonies.

1.3.2.4. Noise

African penguins are known to be sensitive to underwater noise (Pichegru et al., 2017) and use acoustic communication to increase group feeding efficiency (McInnes et al., 2020). This raises the possibility that African penguin foraging success may be influenced by levels of underwater noise that could compromise group feeding efficiency and consequently result in a form of habitat loss or degradation for foraging penguins. Such impacts could arise from presence of fishing vessels in penguin foraging areas or from the presence of vessel traffic such as tankers and cargo vessels. It has been suggested that increased shipping activity in Algoa Bay may have contributed to the decline in African penguin numbers at St Croix Island, and that increased shipping noise may represent an increasing threat to African penguins in South African waters in general (Pichegru et al., 2022).

1.3.2.5 Nest boxes

African penguins are adapted to nest where they are safe from mammalian predators, historically only on offshore islands. On these islands they nest alongside large numbers of other seabirds. As cold-adapted birds they are vulnerable to overheating on land. They dig burrows in guano in which they nest so that they have a buffered microclimate with high relative humidity, protected from solar heating and safe from avian predators (Frost et al., 1976). Harvesting of guano resources from islands off southern Africa removed most of this preferred nesting habitat decades ago, forcing most penguins to nest on the surface, which exposes them to predators, rain, wind, and especially to solar heating. Solar heating can result in temporary nest desertion by adults forced to go into the sea to cool down, which leaves eggs exposed to predation and overheating, reducing their breeding success (Frost et al., 1976; Randall, 1995; Lei et al., 2014; Welman and Pichegru, 2023). Similar effects also occur in the closely-related Magellanic penguin in South America (Yorio and Boersma, 1994). One solution to this problem is to provide nest boxes that protect penguins from these pressures (see additional details in Appendix C).

1.3.2.6 Other

African penguins are vulnerable to impacts on their survival, ability to achieve breeding condition, and breeding success, of low abundance of their key forage fish (sardine, anchovy), and changes in the geographical distribution of forage fish stocks relative to the locations of penguin colonies and moulting sites. Climate change is widely considered likely to be a main factor influencing abundance and distribution of these key prey. Oil pollution has been a long-term pressure on African penguins and continues to be a pressure. Disturbance at colonies by people, and disturbance at sea by ship traffic are ongoing concerns.

African Penguin in a nest incubating an egg (photo BM Dyer)

1.4 Background to the establishment of the Expert Panel.

An African Penguin Biodiversity Management Plan (BMP-AP; Shaw et al., 2011; Anon, 2010) was developed that aimed to halt the decline of the African penguin population in South Africa within two years of its implementation and after that achieve a population growth that would result in a down-listing of the species in terms of its IUCN Red List status. These objectives were not achieved but the plan did lead to: (i) improved cooperative management; (ii) population reinforcement; (iii) improved breeding-habitat management; and (iv) improved management of the captive population (Table 1 of DFFE, 2021).

Modelling studies suggest that adult mortality is higher when sardine biomass is below a critical threshold (Figure 1.3; Robinson et al., 2015) and low adult survival is a strong driver of the reduction in the population size of African penguins since around 2003. However, projections based on the-then Operational Management Procedure (OMP) for sardine by Robinson et al. 2015; see Figure 1.4) suggested that changing the OMP was unlikely to have a marked impact on penguin growth rate relative to closing the fishery entirely (Figure 1.6). Thus, the focus for potential management actions in recent years has focused on fishing near breeding sites.

Penguins may be especially sensitive to changes in pe-

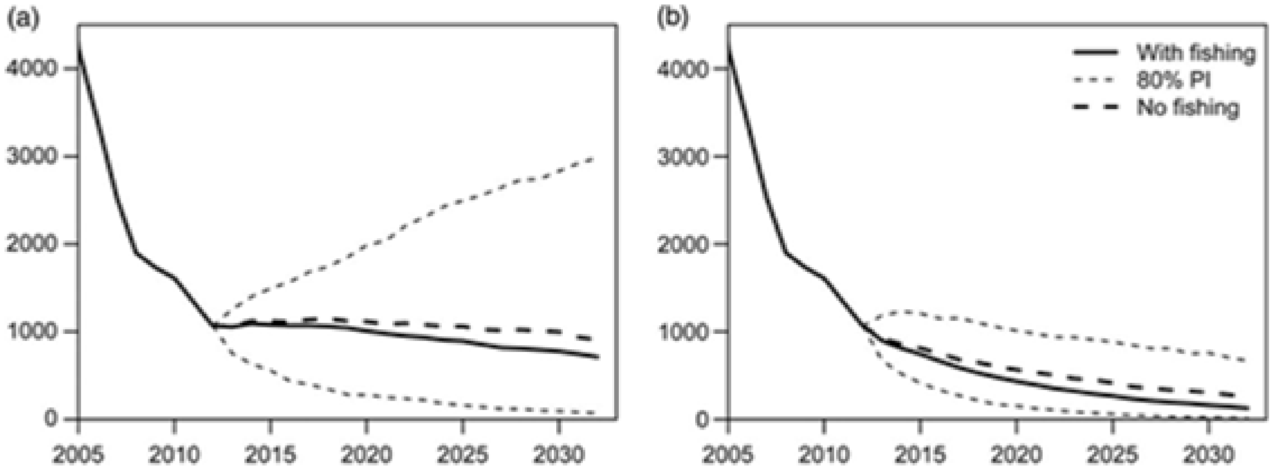

Figure 1.6: Comparison of median projected penguin numbers under Interim OMP-13, and without fishing for future sardine distributions similar to those observed in (a) 1984–1998 and (b) 1998–2012. The 80% probability intervals are indicated for the projections under Interim OMP-13. Projections commence in 2012

lagic fish abundance and distribution as a consequence of their land-based breeding sites and their limited foraging range during breeding (e.g., Sherley et al. 2013; Crawford et al. 2019). For this reason, a study to assess the effects of closure to purse-seine fishing around penguin breeding colonies was initiated in 2008. This study comprised two parts: (i) a feasibility study (2008–2012) during which purse-seine fishing was prohibited around some island breeding colonies and data on penguins and small pelagic fish were collected to determine whether an experiment would have adequate statistical power, within a reasonable time-period, to detect a statistically significant effect of closure, if such existed; and (ii) an Island Closure Experiment (ICE; 2014+), during which data were to be collected to enable a scientific evaluation of whether closures within a distance of 20 km are beneficial to penguin breeding success. In order to maximise contrast for more precise estimation, the study involved a three-year alternation of opening and closing to fishing around islands[1].

Two groups of scientists conducted analyses of the data from the ICE. The analyses were subject to review by the International Fisheries Stock Assessment Workshops (IFSAWs), and over time the differences in terms of methods, data used and results regarding the effects of island closures on penguin reproductive parameters between the two groups declined. However, the two groups of scientists could not reach agreement on some aspects of the analyses and its implications for penguin conservation (see a detailed summary in CAF [2022] and Section 2). This was despite the Minister of Forestry, Fisheries and the Environment tasking the Consultative Advisory Forum (CAF) for Marine Living Resources to develop agreed recommendations on the limiting of small pelagic fishing activities adjacent to penguin colonies. This group considered many documents and held over 50 hours of virtual meetings and several one-on-one meetings in attempts to broker consensus, but this could not be reached and as a last resort they recommended an average of 50% closed and 50% open of the marine Important Bird Areas (mIBA) (CAF, 2022).

The most recent estimates of the effects of closures on penguin reproductive parameters are documented in Sherley et al., (2018, 2021) and Butterworth and Ross-Gillespie (2021a), although these were updated for this report using data up to 2019 and a series of models proposed by the Panel. Models were developed to estimate the implications of changes to each reproductive parameter individually on population growth rate (Butterworth and Ross-Gillespie, 2021b; Sherley et al., 2018, 2021) and attempts were made to infer changes in population growth given the effects of island closures, accounting for the effects on each reproductive parameter (Butterworth and Ross-Gillespie, 2021b; Sydeman et al., 2022).

Options for area closures more aligned with the feeding behaviour of penguins or with the needs of the fishery were developed by a variety of stakeholder groups (e.g., Coetzee et al., 2021a; CAF, 2022). The benefits to penguins were quantified by estimates in the change to the population growth rate and the difference in numbers of penguins expected to be added to the population given the size of the closures (e.g., Butterworth and Ross-Gillespie, 2021b; Sherley et al., 2018, 2021; Bergh, 2022) while costs to the fishery were quantified in terms of catches in areas proposed to be closed, the amount of that catch that would be "lost", and the resulting reduction in jobs in the fishing sector and the general economy (e.g. Coetzee et al., 2021b; Bergh, 2022). Butterworth (2021) outlines a decision table approach to compare the costs and benefits of addressing potential drivers of the dynamics of African penguin. However, there was no agreement amongst the stakeholders on a closure option owing to differences regarding whether the benefits to penguins were meaningful given the predicted change in growth rate (including relative to other potential causes for the decline in abundance), as well as costs to the fishing industry, and all proposals for closures were rejected. However, the stakeholders agreed that an expert panel could help to resolve the technical issues regarding the interpretation of the ICE.

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[1] This time-period was not well-matched to the biology of African penguins, which usually do not breed until aged 4–6 years, so the experiment was designed not to provide information on changes in population size, only on changes in parameters related to reproduction.

1.5 Panel process

A call was made on 28 October 2022 for nominations of qualified individuals to be members of an Expert Panel (henceforth “Panel”), and the Minister selected five scientists with expertise in seabird and penguin ecology, population ecology and ecosystem modeling, and applied statistics (Prof. Robert Furness, Dr. Ana Parma, Dr. Éva Plagányi. Prof. André Punt [Chair], and Prof. Philip Trathan) in December 2022. Recognizing the need for expertise in economics considerations, Prof. James Sanchirico was appointed to the Panel in March 2022. Appendix A lists short biographies for the expert Panel. The Terms of Reference for the Panel are summarized in Appendix B.

The Panel was provided with a list of background documents after a meeting with the Minister of Forestry, Fisheries and the Environment and departmental staff, which was supplemented by documents identified by the stakeholders. The Panel held an online meeting (March 21–23, 2023) at which stakeholders provided input to the Panel in the form of oral presentations and written submissions, after which the Panel met to discuss the implications of the material presented and the necessary next steps. The meeting led to a request for additional information on catches that were reported to have occurred in the closed areas.

A meeting of South African scientists and stakeholders took place on 15 May 2023 during which updated results related to the ICE, the impact of closures on catches and the fishery, as well as how penguin foraging areas could be specified were discussed; one Panel member acted as an observer at the May meeting.

The material from the May and March meetings, along with brief comments by meeting participants, were made available to the Panel, which then met from 5–9 June 2023. The June meeting of the Panel involved a two-day “open” session at which stakeholder groups were provided the opportunity to make presentations to the Panel, followed by a three-day “closed” session during which the Panel reviewed the available evidence, debated conclusions and identified advice and recommendations.

1.6 Current management arrangements

The Department of Forestry, Fisheries and the Environment (DFFE) implemented the following interim closures in September 2022 (Figure 1.7):

1. An L-shaped closure around Dassen Island stretching about 12.5 nm offshore from Yzerfontein and 21.5 nm offshore of Bokpunt, with an extension southward in the offshore area so that the maximum North/South extent is about 20 nm.
2. No additional closure around the Robben Island colony, with only the MPA purse-seine fishery control zone of the Robben Island MPA being closed to fishing.
3. A small closure stretching eastward from Cape Hangklip on the eastern side of False Bay for about 9 miles along the coast and about 3 nm offshore. This includes the small Betty’s Bay MPA and the Stony Point penguin colony.
4. A rectangular area around Dyer Island between Danger Point and Quoin Pt, extending offshore for about 18 nm from Dyer Island and southwards for about 12 nm from the island. This rectangular area is further divided into an inshore area that is closed to all purse seiners and a larger offshore area where only vessels with a total length of less than 26 m may fish.
5. A rectangular area about 20 nm south of St Croix Island in Algoa Bay, with a maximum alongshore extent of about 20 nm, but with fishing allowed around the Riy Banks.
6. A square closure extending about 12 miles south of the Addo MPA in the vicinity of Bird Island with a maximum west/east extent of around 29 nm.

Other restricted areas include the 16-mile beach MPA inshore along the west coast, north of Dassen Island, the entire False Bay, the inshore area in Walker Bay between Stony Point and Dyer Island and the Sardinia Bay MPA, just west of Algoa Bay and the inshore parts of the Addo MPA between the interim closures of St Croix and Bird islands.

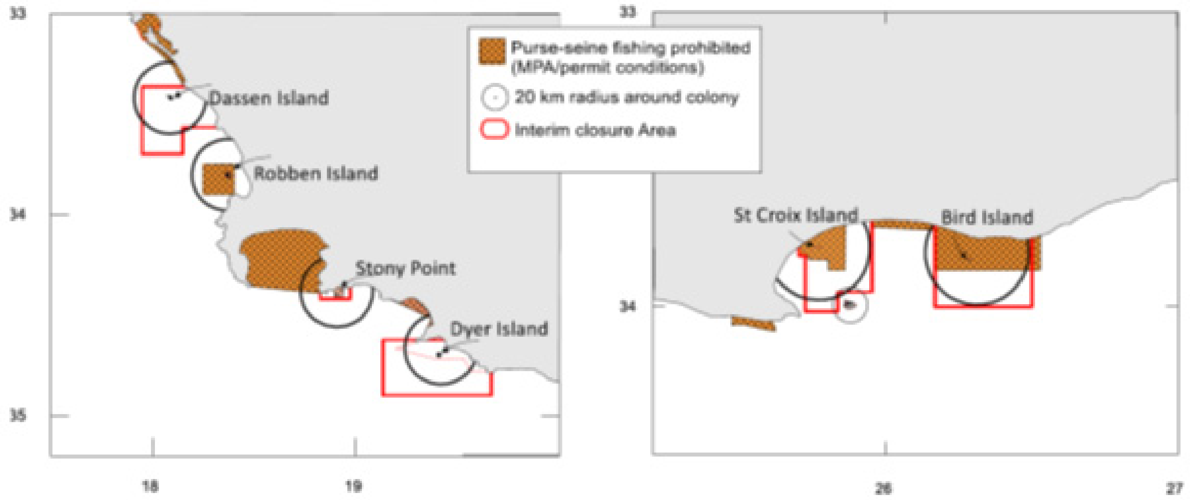


Figure 1.7: Interim closures to fishing (red polygons) as currently implemented. These closures have been implemented since September 2022. Vessels <26 m in length are allowed to fish in the offshore area (outside the red dotted line) of Dyer Island.

2. BENEFITS OF ISLAND CLOSURES TO PENGUINS

2.1 Aims and design of the ICE, and reproductive parameters monitored

The Island Closure Experiment (ICE) was established in 2007 to provide a scientific basis to assess whether closures to pelagic fishing in the neighbourhood of penguin breeding islands might provide a meaningful improvement to penguin reproductive success. The design of the ICE therefore had a basic aim to detect differential reproductive success under open and closed situations during periods when other conditions were unlikely to confound results through having changed themselves.

The ICE comprised two parts: (i) a feasibility study during which purse-seine fishing was prohibited around two pairs of penguin breeding islands: Dassen and Robben islands on the West Coast and St Croix and Bird islands in the Eastern Cape (Figure 1.1); and (ii) an experimental phase (2015–2021) where a series of three-year alternating island closures around the four breeding islands were implemented (Table 2.1). Figure 2.1 summarises the timeline of the ICE and the associated reviews of the analyses conducted.

The three-year alternation of opening and closing to fishing around islands was selected to maximise contrast for more precise estimation of closure effects (CAF, 2022). The duration of three years was selected according to DFFE (2021) to balance conflicting objectives of: (i) rapid alternation to maximise contrast in the data to enable more precise estimation; (ii) a slower alternation to take account of possible autocorrelation in the penguin indices being monitored; and (iii) the desirability to integrate the feasibility study into a possible future experiment to lead to earlier answers.

The feasibility study was originally planned to last two years (2008 and 2009), but that proved to be insufficient time to allow experimental power to be estimated for all the penguin parameters monitored, and analyses of the impacts of purse-seine fishing in the vicinities of breeding islands failed to produce clear-cut results. It was therefore agreed that the feasibility study was to be extended for an additional four years (until the end of 2014).

The penguin parameters that were intended to be measured during the experiment were: chick condition, survival and growth, fledgling success and as measures of foraging behaviour: maximum distance, path length and trip duration (see Campbell et al. [2019] for detailed specifications for how each of these variables are defined and calculated based on monitoring data). Not all response variables could be measured in all colonies; the west colonies (Dassen and Robben islands) were the most intensively monitored while only data on chick condition and foraging-related variables were collected at St Croix and Bird islands (see Table 2.2 for details regarding data availability).

Small-scale acoustic surveys using an inflatable vessel were conducted to provide direct estimates of the biomass of small pelagic fish available to penguins around some of the islands. Those surveys were initially around Robben Island (six surveys were conducted in 2009) but in later years the surveys were extended to around Dassen, St Croix and Bird islands (Coetzee et al., 2016). Fine-scale surveys were also conducted by non-governmental researchers around St Croix and Bird islands from 2014 to 2018 (McInnes et al., 2017). The small-scale surveys were subsequently abandoned at the end of 2018 given their relatively low precision, staff shortages and lack of funding (DFFE, 2021).

2.2 Methods used to estimate effects of closures (catches) on penguin population growth rate

2.2.1. Rationale for models

The impacts of fishing closures on the response variables monitored were quantified using generalised linear mixed-effects models (GLMM). Various model variants were applied since the first analyses of the ICE data were conducted during the initial feasibility period, including an analysis to evaluate the power to detect biologically meaningful impacts caused by the fishery as data accumulated. The power analyses completed in 2016 indicated that meaningful results could be obtained within 20 years of the onset of the experiment (Ross-Gillespie and Butterworth, 2016a).

Table 2.1: Schedule of closures around the four penguin breeding colonies during the ICE. Crosses indicate years in which a 20 km radius area around the island was closed to fishing.

Island	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019	2020	2021
Dassen Island	X	X					X	X	X				X	X[1]
Robben Island				X	X	X				X	X		X	X[2]
St Croix Island		X	X	X				X	X	X				X[3]
Bird Island					X	X	X				X	X	X	

[1] Closed from 15th January to 31st March and from 1st October to 31st December, and open from 1st April to 30th September.

[2] Closed from 15th January to 31st December.

[3] Closed from 1st April to 30th September, and open from 15th January to 31st March as well as from 1st October to 31st December.

Table 2.2: Reproductive parameters monitored at the four breeding colonies that were part of the Island Closure Experiment.

Response variable	Island	Year range
Chick condition	Dassen	2004–2019
	Robben	2004–2019
	Bird	2008–2019
	St Croix	2008–2019
Chick Survival	Dassen	2008–2019
	Robben	2008–2019
	Bird	–
	St Croix	–
Fledging success	Dassen	1995–2015
	Robben	1989–2015
	Bird	–
	St Croix	–
Chick growth	Dassen	1989–2014
	Robben	2004–2014
	Bird	–
	St Croix	–
Max distance	Dassen	2008–2018
	Robben	2008–2018
	Bird	2008–2018
	St Croix	2008–2018
Path length	Dassen	2003–2018
	Robben	2003–2018
	Bird	2007–2018
	St Croix	2008–2018
Trip duration	Dassen	2003–2018
	Robben	2003–2018
	Bird	2007–2018
	St Croix	2008–2018

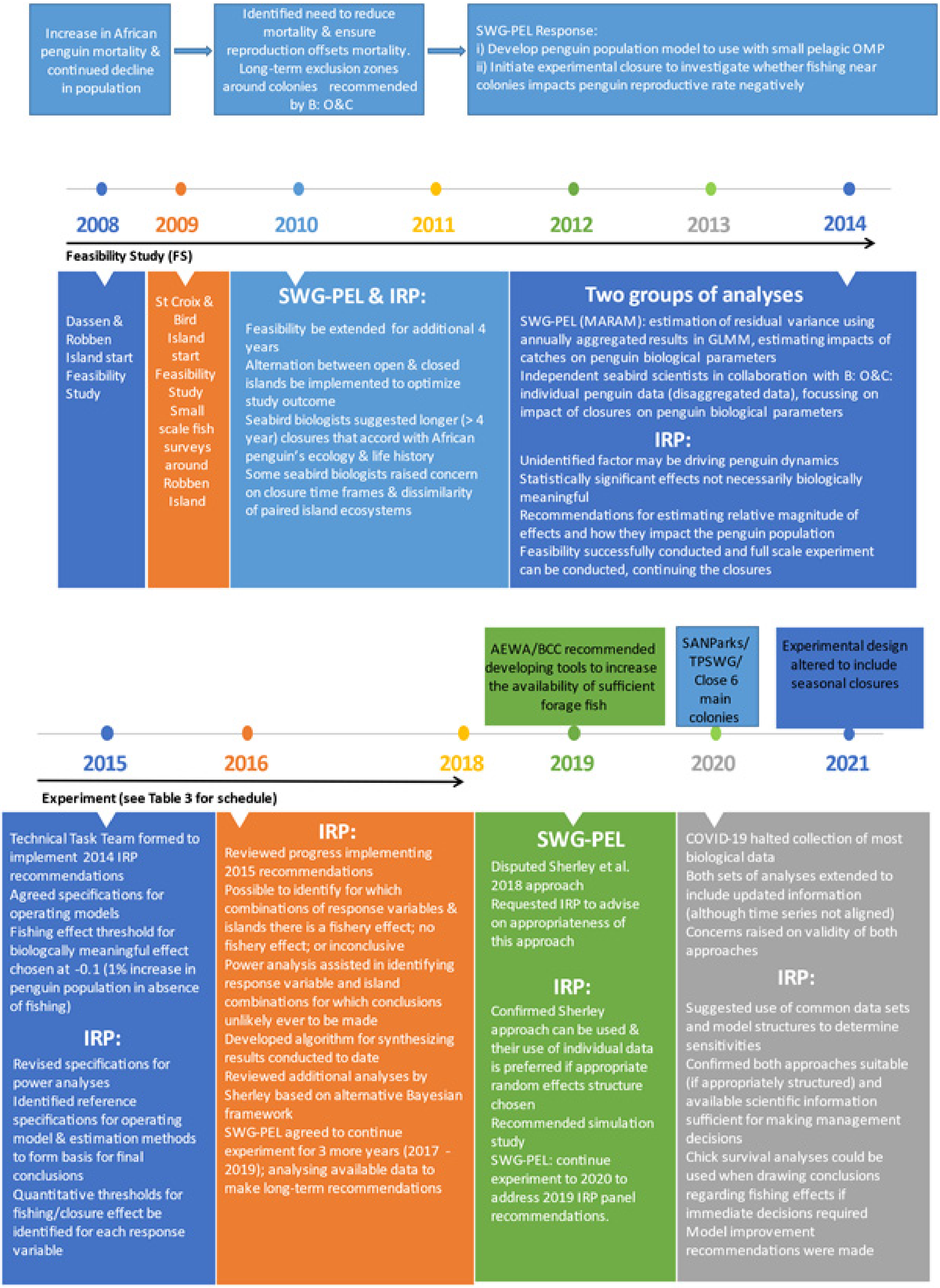
The main features that distinguish the various model variants utilised are summarised in this section. Mathematical specifications and further details are provided in Appendix D and cited documents.

Two main classes of models were considered. These differ in the choice of independent variable used to represent the effect of fishing. In one class, fishing is included as a binary variable having a value of 1 when the island is open to fishing and 0 when it is closed. Predictions from this class of models are referred to as "closure-based estimates" of the impact of fishing. In the alternative class of models, the effect of the actual catches taken within the 20-km areas around the colonies are evaluated as covariates. In this case, the predicted "catch-based estimates" of the impact of fishing within a given closure is calculated using the average catch taken from that closure when the island was open to fishing during the ICE. A concern with the catch-based estimators is that the true impact of fishing may be underestimated if catches tend to be higher when fish biomass is higher due to the confounded effects of fishing and food availability on penguin breeding success. The preference for using the closure-based models as the base for inference regarding the impacts of island closures was supported by the finding of positive correlations between the time-series of catches taken within the 20-km areas (when open) and regional survey estimates of biomasses of anchovy in the west and sardines in the east (Ross-Gillespie and Butterworth, 2023a). In the final set of results presented in Ross-Gillespie and Butterworth (2023a), catch-based models were also examined but they were used only as sensitivity runs requested by the Panel to evaluate the impact of some non-negligible catches apparently taken within the area closed around St Croix Island mainly in 2017 (see section 2.4).

In all cases, separate analyses were conducted for the two pairs of colonies (Dassen and Robben islands on the west coast, and St Croix and Bird islands on the east), assuming that nearby colonies experienced rather similar conditions affecting breeding success, except for the experimental treatment. Separate island-specific effects of the closure were however estimated considering that several factors not controlled by the experimental design may lead to different responses to the closure between the paired islands. The significance of those differences was evaluated by Sherley (2023), and the model with a common effect was selected based on standard model-selection criteria by Sherley (2023). Concerns were expressed that the estimation of a common effect would tend to be biased towards the island with the higher sample size and/or lower variance (Bergh, 2023) and that alternative weights (e.g., size of the colony) could be used to average island-specific estimates. While this is a valid point, the differences between the results were not large and the integrated estimate of a regional impact would not be largely affected.

An important difference between the approaches favoured by different analysts was a preference to analyse the data aggregated as annual means (Ross-Gillespie and Butterworth, 2023a) versus using individual-records-based disaggregated data (Sherley et al., 2018; Sydeman et al., 2021). The relative merits of aggregated and disaggregated data models were the subject of substantial debate (e.g., Butterworth and Ross-Gillespie, 2022; Sydeman et al., 2022). The individual-based approach has the advantage of analysing the data at the level they are collected, but the model needs to appropriately capture the factors and sources of variability (observed or unobserved) impacting the observations, other than closure alone (Haddon et al., 2020). If the model is incorrectly specified and there are unaccounted common random effects that affect all observations from a given stratum (e.g., all observations from a given month, year and colony), individual observations are not independent. This so-called "pseudo-replication" may lead to underestimation of the standard errors of important model outputs. Aggregated models, on the other hand, have the advantage of not requiring assumptions about within-stratum correlation, but are vulnerable to assigning inappropriate weights by stratum (Haddon et al., 2020). Because the two approaches would be statistically equivalent provided that a correct model structure is assumed in the estimation (Butterworth and Ross-Gillespie; 2022, Haddon et al., 2020), the debate centred on the choice of a hierarchical random structure for the disaggregated models that would be able to account for the pseudo-replication.

The choice of random model structure to be used in each of the two approaches was discussed during an international review conducted in 2020 where a recommen-



Increase in African penguin mortality & continued decline in population

Identified need to reduce mortality & ensure reproduction offsets mortality. Long-term exclusion zones around colonies recommended by B: O&C

SWG-PEL Response:
i) Develop penguin population model to use with small pelagic OMP
ii) Initiate experimental closure to investigate whether fishing near colonies impacts penguin reproductive rate negatively

Feasibility Study (FS)

Dassen & Robben Island start Feasibility Study

St Croix & Bird Island start Feasibility Study Small scale fish surveys around Robben Island

SWG-PEL & IRP:

Feasibility be extended for additional 4 years
Alternation between open & closed islands be implemented to optimize study outcome
Seabird biologists suggested longer (> 4 year) closures that accord with African penguin's ecology & life history
Some seabird biologists raised concern on closure time frames & dissimilarity of paired island ecosystems

Two groups of analyses

SWG-PEL (MARAM): estimation of residual variance using annually aggregated results in GLMM, estimating impacts of catches on penguin biological parameters
Independent seabird scientists in collaboration with B: O&C: individual penguin data (disaggregated data), focussing on impact of closures on penguin biological parameters

IRP:

Unidentified factor may be driving penguin dynamics
Statistically significant effects not necessarily biologically meaningful
Recommendations for estimating relative magnitude of effects and how they impact the penguin population
Feasibility successfully conducted and full scale experiment can be conducted, continuing the closures

AEWA/BCC recommended developing tools to increase the availability of sufficient forage fish

SANParks/ TPSWG/ Close 6 main colonies

Experimental design altered to include seasonal closures

Experiment (see Table 3 for schedule)

Technical Task Team formed to implement 2014 IRP recommendations
Agreed specifications for operating models
Fishing effect threshold for biologcally meaningful effect chosen at -0.1 (1% increase in penguin population in absence of fishing)

IRP:

Revised specifications for power analyses
Identified reference specifications for operating model & estimation methods to form basis for final conclusions
Quantitative thresholds for fishing/closure effect be identified for each response variable

IRP:

Reviewed progress implementing 2015 recommendations
Possible to identify for which combinations of response variables & islands there is a fishery effect; no fishery effect; or inconclusive
Power analysis assisted in identifying response variable and island combinations for which conclusions unlikely ever to be made
Developed algorithm for synthesizing results conducted to date
Reviewed additional analyses by Sherley based on alternative Bayesian framework
SWG-PEL agreed to continue experiment for 3 more years (2017 - 2019); analysing available data to make long-term recommendations

SWG-PEL

Disputed Sherley et al. 2018 approach
Requested IRP to advise on appropriateness of this approach

IRP:

Confirmed Sherley approach can be used & their use of individual data is preferred if appropriate random effects structure chosen
Recommended simulation study
SWG-PEL: continue experiment to 2020 to address 2019 IRP panel recommendations.

COVID-19 halted collection of most biological data
Both sets of analyses extended to include updated information (although time series not aligned)
Concerns raised on validity of both approaches

IRP:

Suggested use of common data sets and model structures to determine sensitivities
Confirmed both approaches suitable (if appropriately structured) and available scientific information sufficient for making management decisions
Chick survival analyses could be used when drawing conclusions regarding fishing effects if immediate decisions required
Model improvement recommendations were made

Figure 2.1: Timeline describing major events, decisions etc., during the feasibility study and experiment (Figure 9 of DFFE, 2021). SWG-PEL = small pelagic scientific working group; IRP = International Review Panel; B: O&C = Branch: Oceans and Coasts; AEWA = Agreement on the Conservation of African-Eurasian Migratory Waterbirds; BCC = Benguela Current Commission; SANParks = South African National Parks; TPSWG = Top Predator Scientific Working Group; OMP = operational management procedure; GLMM = generalised linear mixed-effects model.

dation was made to use standard model selection criteria combined with knowledge of the sampling design (Haddon et al., 2020). In both cases, a random Year effect, common to the paired islands, was incorporated to account for year-to-year changes in food availability and other unspecified factors affecting annual breeding success at a regional scale. Monthly differences in chick condition were found to be important and therefore aggregated data were first standardised for the month effect as explained in Ross-Gillespie and Butterworth (2021a), while a random Month effect, nested within Year, was incorporated in the data-disaggregated models (Sydeman et al., 2021). The remaining question, therefore, was which further random effects, if any, would need to be nested within Year (or Year/Month) to account for possible correlation between the individual observations in the disaggregated data models. Sydeman et al. (2021) found that accounting for the identity of the penguin nest (NestID) in the chick survival analysis was significant given that the survival of chicks from the same nest are expected to be correlated. However, their preferred model with random effects Year + Year/NestID did not include Island (nested within Year) and therefore could still be affected by pseudo-replication, as discussed by Butterworth and Ross-Gillespie (2022). The final set of analyses presented by Sherley (2023) used hierarchical model structures suggested by the Panel in the light of previous results presented at its March 2023 meeting. The suggested model structures attempted to address the pseudo-replication by including Island in the random effects in a way that differed depending on the response variable. For the analysis of individual chick condition data, the hierarchical random effects involved Year + Year/Month + Year/Month/Island, i.e., it included the effect of Island nested within the Year × Month interaction. Likewise, the inclusion of Island was suggested for the analysis of chick survival data as Year + Year/Island + Year/Island/NestID, which follows the natural nesting of the data collection program given that different nests are monitored in different years.

The suggested random model structures were preferred based on model selection criteria (Sherley, 2023). In the analysis of chick condition data, the inclusion of the Island random effect nested within Year + Year/Month resulted in wider confidence intervals for the predicted impacts on penguin population growth rate due to a higher standard error of the estimated fixed closure effects (compare models 3 and 3.1 respectively with models 5 and 5.1 in Sherley's Figure 2), as anticipated if observations within year-month-island strata were not independent. Furthermore, the closure effects estimated using these preferred models had very similar precision to those produced using aggregated data (model 8 in Sherley's Figure 2). A difficulty to partition the variance and to estimate the variance attributed to the Year factor was observed so a simpler random structure that excluded the Year factor was selected with no impact on the closure-effect estimates.

For the chick survival data, the inclusion of Island in the nested random structure also decreased the precision of the estimated closure effects (compare models 4 versus 8 and 5 versus 9 in Sherley's Figure 4). In this case, however, the standard errors estimated with the selected data-disaggregated model were larger than those estimated using aggregated data for models containing the equivalent fixed effects. This may be related to the shared frailty (i.e., linked probability of dying) for chicks in the same nest, which was estimated through the NestID random effect in the data-disaggregated models while it was either ignored when generating the annual aggregated survival times series (the A(B) models in Sherley's Figure 4) or it was accounted for prior to evaluating the closure effects in a separate parametric model (the A(S) models).

In conclusion, the Panel **agreed** that the debate about the relative merits of analyses based on aggregated versus disaggregated data was essentially closed based on the final set of results presented at the June 2023 meeting. Although differences in preferences between the analysts remained, the Panel **agreed** that the two approaches would provide similar results (as expected) when appropriately configured (especially to account for pseudo-replication), all other things related to data pre-processing being equal.

2.2.2 Converting impacts on reproductive parameters to changes in penguin population growth rate

Fishing effects on reproductive parameters estimated from the models need to be linked to impacts on penguin population growth rates. A method based on a demographic model described in Ross-Gillespie and Butterworth (2021b) was used by all analysts as a basis to convert changes in chick condition, fledging success and chick survival into absolute effects on annual population growth rate. In the case of chick condition, a relationship between mass at fledging and first-year survival estimated for the macaroni penguin (Horswill et al., 2014) was used to translate changes in chick condition to changes in population growth rate (Sherley et al., 2018). For the other response variables (chick growth, trip duration, maximum distance and path length), whose impact on demography are not straightforward, it was assumed that the estimated relative change in the response variable due to fishing resulted in the same relative change in juvenile survival (Robinson et al., 2014; Butterworth and Ross-Gillespie, 2021a, Table A1). This assumption is not supported by evidence available for other species, which indicates that the relationship between, for example, foraging trip duration or distance travelled with chick survival is nonlinear and involves thresholds. Aside from these nonlinearities, the assumption that the relative impacts on, say, trip duration and chick survival have the same magnitude is highly questionable. The Panel **agreed** to interpret the impacts of fishing in foraging-related parameters only qualitatively, and to not integrate them into the overall impacts on penguin population growth rates.

2.2.3. Integrating fishing impacts predicted from separate analyses into overall fishing impacts on penguin growth rate

The results of the ICE provide estimates of how closing a penguin breeding island will impact the value of a parameter related to penguin reproductive success, and models were developed that related the change in the value of one parameter to a change in population growth rate. Ultimately, it is necessary to 'integrate' the effects for each reproductive parameter to derive an 'overall' estimate of the change in population growth rate due to closing a breeding island. This calculation is complicated because of several factors:

- There are factors that will determine population growth rate other than changes in reproductive rate such as immigration/emigration and changes in survival for post-fledgling animals. Thus, reported changes in population growth rate are those related only to changes in reproductive success, essentially assuming that the survival rate for animals after the first year of life is not impacted by closures to breeding islands and that immigration and emigration balance out.
- Only a subset of the parameters were monitored on all breeding islands and some parameters were not monitored for all years (Table 2.2).
- Some of the parameters (e.g., chick survival and chick condition/growth) are not independent.
- There is a need to infer the effect of closures for breeding islands that were not part of the ICE.
- The estimates of changes in population growth rate derived from the ICE results pertain to a *status quo* of no closure, so changes in population growth rate of half those estimates are pertinent to the recent situation of closures half of the time.

Butterworth and Ross-Gillespie (2021b) provide a “qualitative” scheme for conducting the integration based on the following assumptions/algorithm:

- The three foraging metrics were assumed not to be independent nor were chick condition and chick growth, and measures of uncertainty (standard errors for the estimates of population change by reproductive parameter) were calculated based on dividing the 95% interval for the population growth rate by 4.
- Fledgling success, chick condition, and chick survival are more ‘reliable’ as there is a demographic model relating changes in these variables to changes in population growth rate. Thus, for example, when information about chick condition and chick growth were integrated for Dassen Island, values of 0.06% and 1.74% were averaged qualitatively to get 0.5% and the standard deviation of this value was set to that corresponding to the 0.06% estimate (i.e., 0.42%).
- Of the foraging metrics, maximum distance was considered to be less reliable than path length and trip duration, given there is more uncertainty associated with a maximum than an integrated measure. Thus, inferences regarding changes in foraging distance on population growth rate involved a “qualitative average” of the effects of primarily path length and trip duration, with the standard error set to averages of the standard errors of the change percentages by island.
- No attempt was made to infer changes on chick growth, chick survival and fledgling success for St Croix and Bird islands from the results for Dassen and Robben islands, but estimates of population growth were determined from changes in chick condition/growth and foraging alone.

In their presentation to the Panel, Butterworth and Ross-Gillespie (2023) outlined two alternatives for combining the predicted changes in population growth rate derived from changes in chick condition and chick survival, one in which the effects were averaged and a second in which the effects were added. As explained in section 2.2.2, the relationship between chick condition and juvenile survival used to translate changes in chick condition to changes in population growth rate corresponds to a relationship between mass at fledging and first-year survival (estimated for the macaroni penguin). Therefore, the Panel **agreed** that it is more appropriate to treat those effects as additive when calculating the overall impacts on population growth rates.

2.3 Predicted effects of fishery closures (catches) on penguin population growth rate

2.3.1 Summary of outcomes among analyses

A broad summary of the results in terms of the impacts of fishing around breeding colonies on penguin population growth rates obtained for the west and east colonies included in the ICE is given below. A negative value corresponds to a predicted positive effect of closing the 20-km areas on population growth rate because the reported values correspond to fishing impacts.

Results for three different closure-based estimators are shown for the analyses of chick condition and chick survival in Figures 2.2 and 2.3. The first two estimators involve models fitted to disaggregated data (D) and the third is based on the analysis of aggregated data (A). These estimators correspond to the preferred choices made by the analysts, and use the random-effects hierarchical structure that was recommended by the Panel for the case of models fitted to disaggregated data.

2.3.1.1 Dassen and Robben islands

The two alternative estimates shown in Figure 2.2 obtained using disaggregated data differ with respect to whether the effect of fishing was assumed to be the same on both islands (models W1 and W4) or was allowed to differ between them (models W2 and W5), while separate effects for the two islands were estimated by models W3 and W6, which were fitted to aggregated data. A slight preference for the models that assume the same effect size in both islands was found when the models based on disaggregated data were compared (Sherley, 2023). While some analysts argued that separate effects should be preferred independently of the results of the tests (Butterworth and Ross-Gillespie, 2023a), they acknowledged that the integrated estimates for the western Cape colonies would not be much affected.

The resulting estimates for the three selected alternative models are similar although confidence bounds were narrower when the effects were forced to be the same for both islands, as expected. The exceptions are the results for chick survival for Robben Island, which indicate a larger negative impact of fishing on population growth rate when the analysis is based on disaggregated data than when aggregated data are used. Part of the reason for this difference may be the way the individual data were aggregated to construct the time-series of chick survival.

Larger negative impacts of fishing, close to the -1% value used as a reference, were estimated for Dassen and Robben islands based on chick survival data except for the

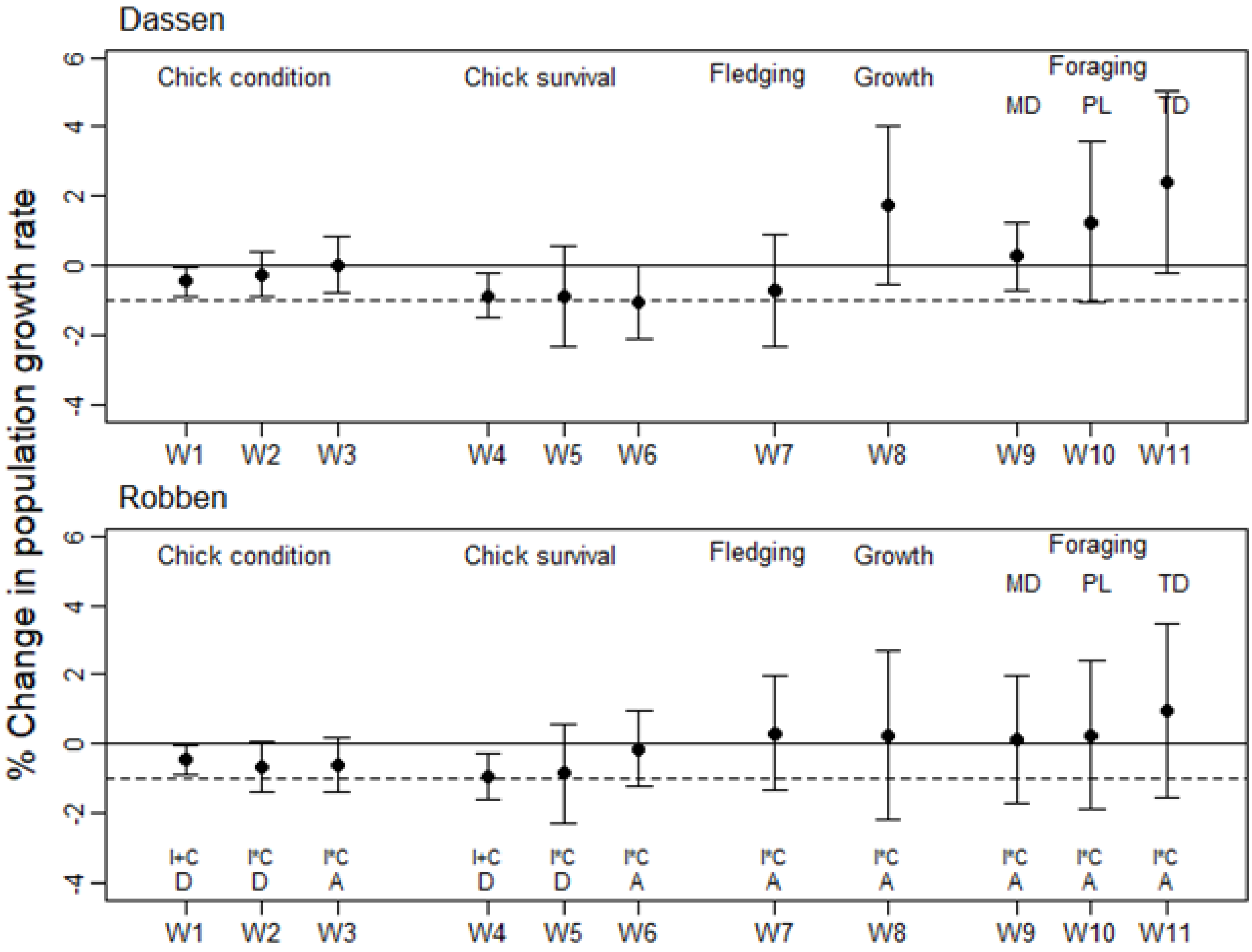


Figure 2.2: Estimates of change in population growth rate for Dassen and Robben islands as a result of fishing (expressed as a percentage per annum) resulting from the analysis of various response variables measured at those colonies: chick condition, chick survival, fledging success, chick growth, and three variables related to foraging behaviour: maximum foraging distance (MD), path length (PL) and trip duration (TD). W1–W11 = model numbers. Model specifications are detailed in Appendix D.

smaller effect estimated for Robben Island using aggregated data. Most estimated effects based on chick condition were negative but somewhat smaller, ranging from 0.04% to -0.67%.

The results based on analyses of chick growth and foraging-related parameters give little indication of a biologically meaningful impact of the closures. A reduction in chick growth rate during years when an island was open to fishing was expected but the opposite was estimated (model W8). Results are not consistent with the generally negative fishing impacts estimated from chick condition and survival, the response variables that are more directly related to population trends.

2.3.1.2 St Croix and Bird islands

The fishing impacts estimated for St Croix and Bird islands based on chick condition data were positive except that for Bird Island based on aggregated data (model E3), which was negative and very small (–0.24%) (Figure 2.3). A negative impact was estimated for some of the foraging variables in some of the island-method combinations, but the estimated impact was positive for other cases. The reliability of foraging metrics as indicators of the impact of fishing on the breeding success of penguins is therefore questionable, particularly given opposite signs of fishing impacts estimated for St Croix Island.

Overall, the Panel did not consider the results for the east colonies to be reliable, given the very little fishing that took place around Bird Island when the area was open except in the early years (Figure 2.4). Also, the first two model results based on disaggregated data included data for the year 2017 when some sizeable catches were taken from within the St Croix Island closure when the area was supposed to be closed. Some sensitivity runs conducted in response to a request by the Panel using the aggregated data (Ross-Gillespie and Butterworth, 2023b) indicate that these catches did not impact the broad results from the ICE for St Croix Island. In particular, the analyses still resulted in positive estimates of fishing impacts for St Croix Island when year 2017 was excluded from the data. This result was not substantially altered in other sensitivity runs reported by Butterworth and Ross-Gillespie (2023a, results not shown here). The only run that resulted in a negative, albeit small, impact (–0.39 in units of % population growth) was when data for 2008–2010 were excluded, Bird Island was treated as closed during all years, and St Croix Island was treated as open in 2017.

The alternative catch-based estimator, which uses actual catches taken within the 20-km areas instead of the open/closed treatment, led to negative but still very small fishing impacts (−0.28 in units of % population growth) at St Croix Island for the chick condition data (Ross-Gillespie and Butterworth, 2023a, results not shown). The results based on foraging-related variables, on the other hand, tended to show smaller negative impacts for St Croix Island than when the open/closed treatment was used.

The existence of other confounding factors not controlled by the ICE add to the difficulties in interpreting the results for the eastern colonies. In particular, the increased number of bunkering operations in Algoa Bay since 2016 may have impacted the penguin population at St Croix Island (Pichegru et al., 2022). A sensitivity run that only included years up to 2015 (Model S5 in Ross-Gillespie and Butterworth, 2023a) failed to identify any impact of the closures on chick condition, and led to lower impacts based on foraging trip parameters.

In summary, the Panel **concluded** that the ICE results for the east colonies were more uncertain and difficult to interpret given that the paired islands did not provide the anticipated contrast, and given the few response variables that could be monitored at those colonies. Notwithstanding these limitations, the Panel **concluded** that the available results only provide indirect evidence of negative impacts of fishing around St Croix Island through increased foraging distances of breeding penguins during years when the colony was open. However, these changes in foraging behaviour were not reflected in estimated poorer chick condition.

2.3.2. Integrated estimates of the overall impact of closures on penguin population growth rate

As discussed in Section 2.2.3, the Panel considered it more appropriate to treat effects estimated from impacts on chick condition and chick survival as additive when calculating the predicted overall impact on population growth rates (Table 2.3). Only the predictions for Dassen and Robben islands are shown given the concerns regarding the use of foraging-related variables (see section 2.2.1) and that fact that, for St Croix and Bird islands, only estimates based on chick condition are available.

Overall, the Panel **concluded** that the results of the ICE for Dassen and Robben islands indicate that fishing closures around the breeding colonies are likely to have a positive impact on population growth rates, but that the impacts may be small, in the range 0.71–1.51% (expressed in units of annual population growth rate). These impacts are small relative to the estimated relative reductions in penguin abundance for these two colonies overthe period 2005–2022, which were estimated by the Panel at −13% for Dassen Island and −10% for Robben Island, using abundance data provided to the Panel.

The ICE in its current form (to estimate the effects of fishing closures on reproductive success) is completed. Future closures of forage-fish fishing around penguin colonies would be likely to benefit penguin conservation, but should be part of a larger package of conservation measures as such closures alone would be unlikely to reverse the current decline in penguin population numbers.

2.4 Caveats associated with the ICE and the associated analyses

The commitment by the South African government to implementing an experimental management scheme (the ICE) to understand whether fishing near breeding colonies negatively affects African penguin populations should be recognised, notwithstanding the caveats in this section because without the ICE, management decisions would have to be based on analogy and expert opinion. The experiment aimed to collect data that could allow the effects of fishing closures on the reproductive parameters of African penguins to be estimated. It implemented several best practices, including paired controls and treatments, monitoring of key reproductive parameters, and an initial period to assess how long it would take for there to be sufficient statistical power to detect a potentially meaningful effect of fishing closures, if one existed. In addition, the data from the experiment were analysed using multiple modelling approaches and the analyses were regularly peer-reviewed within the domestic process as well as by the International Fisheries Stock Assessment Review Workshops (e.g., Haddon et al. 2020), likely increasing the robustness of the

Table 2.3: Overall integrated fishing impacts on penguin population annual growth rates estimated from the data collected during the ICE for the Dassen Island and Robben Island breeding colonies. Three estimates are provided for each island to illustrate the range of results produced by the selection of model runs shown in Figure 2.2. Note that the values provided refer to the predicted effects of fishing around the colonies, so a negative value implies a positive change in population growth rate if the areas were closed relative to if they were kept open to fishing.

Dassen Island	Chick condition	Chick survival	Added fishing impacts on population growth rate	Models	Modelling of closure effect
	−0.43	−0.86	−1.29	W1 & W4	I + C
	−0.24	−0.86	−1.10	W2 & W5	I × C
	0.04	−1.04	−1.00	W3 & W6	I × C
Robben Island	Chick condition	Chick survival	Added fishing impacts on population growth rate	Models	Modelling of closure effect
	−0.43	−0.91	−1.34	W1 & W4	I + C
	−0.67	−0.84	−1.51	W2 & W5	I × C
	−0.59	−0.12	−0.71	W3 & W6	I × C

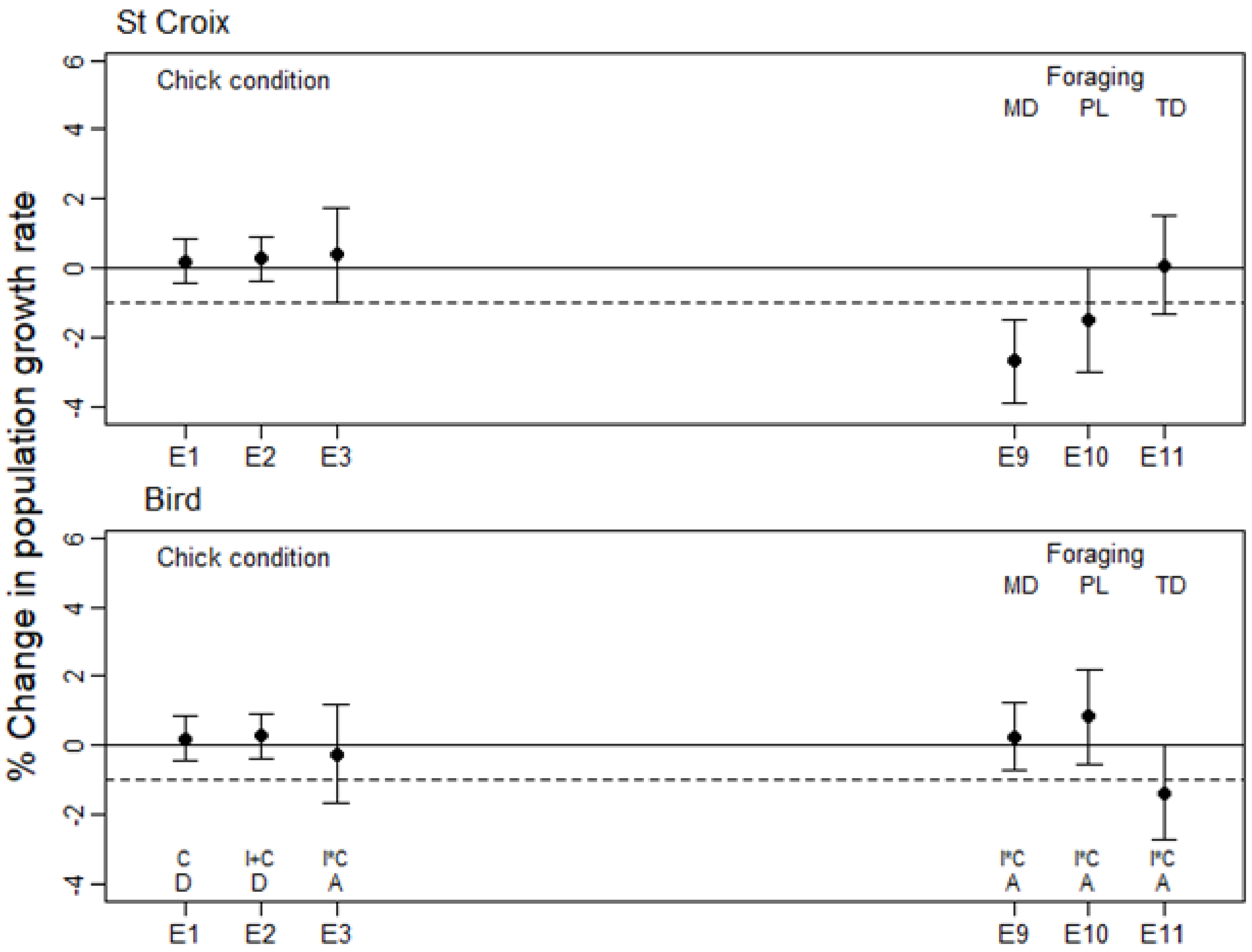


Figure 2.3: Estimates of change in population growth rate for St Croix and Bird islands as a result of fishing (expressed as a percentage per annum) resulting from the analysis of chick condition and three variables related to foraging behaviour measured in those colonies: maximum foraging distance (MD), path length (PL) and trip duration (TD). E1–E11 = model numbers. Model specifications are detailed in Appendix D.

results. The ICE was highlighted by Sydeman et al. (2017) in their review of best practices for assessing forage fish fisheries – seabird resource competition, noting that field experiments are the "holy grail" of seabird-fisheries competition studies because of the potential to detect causal effects. In fact, it is the only case where an experiment has been designed with the aim of detecting fishing effects on reproductive parameters of seabirds. However, Sydeman et al. (2017) note that field experiments can be difficult to design and implement, and the ICE is no exception in this regard.

Notwithstanding that the experiment was designed following best practices, there are several weaknesses of the design and implementation that need to be recognised and their consequences accounted for when interpreting the results in section 2.3 of this report.

- The experiment aimed to estimate the effects of fishing closures on penguin reproductive parameters, meaning that it was necessary to develop models to predict changes in the population growth rate given expected changes in reproductive parameters (see section 2.2.3). While it would have been ideal to relate fishing closures to changes in population sizes directly, it was recognised when the experiment was proposed that the time to detect changes in population size attributable to an island closure would potentially involve a much longer experiment than that needed to detect changes in reproductive parameters. This was due, for example, to the time that penguins take to recruit to the adult population, and that the results in terms of population size might be confounded by the effects of, for example, movement amongst breeding colonies.
- The experiment involved temporal blocks of 3 open and 3 closed periods (Table 2.1). This design was a compromise between longer blocks, which might permit detection of changes in population size and shorter blocks, given the focus on reproductive parameters. The design was implemented nearly as anticipated – the exception was 2021, the data for which are not used in the analyses.
- The closures pertained to 20 km around breeding colonies. However, analyses subsequent to the start of the experiment (e.g., Annexure 1 of CAF, 2022) show that penguin foraging can extend well beyond 20 km (especially for St Croix Island) so while the results of the experiment allow the effect of 20 km closures to be quantified, potentially larger effects may have been observed with closures that more closely reflected foraging areas. The ability to infer changes in reproductive parameters (and

hence population growth rates) for closures that differ from 20 km around islands requires an extra step of interpretation that is necessarily primarily qualitative.

- The experiment relates to four of six major breeding colonies. Closures have been proposed for Dyer Island and Stony Point. Inference of the effect of closures for these colonies requires extrapolation of the effects of the closures for the islands in the experiment, and are consequently more uncertain.
- The experiment manipulated the ability to fish within 20 km of the four islands. It did not specify that catches had to occur when an island was "open". One consequence of this is that catches might be low during open years. This was the case for Bird Island where catches were low irrespective of whether this island was open or closed to fishing due to operational issues. Moreover, analyses provided by Janet Coetzee (DFFE) showed that some catches had occurred inside the closure areas in years when they were supposed to be fully closed to pelagic fishing (in particular, off St Croix Island in 2017; Coetzee, 2023; Figure 2.4). In addition, some recorded catches occurred close to the 20 km closure boundaries. Whether some of these catches actually occurred within 20 km of the islands was not checked given the time available but some of these catches may have occurred inside the closures.
- A primary aim of having two colonies in each region was to enable the effects of factors other than fishery closures on reproductive parameters to be accounted for in the analyses. Given that the ICE is a natural experiment and even though the two islands on each coast are relatively close, there were still differences in distribution of pelagic fish between islands (Coetzee, 2023) that cannot be accounted for in the analyses based on results of the ICE.
- It was not possible to monitor all variables that could affect reproductive success owing to logistical constraints and the possibility that monitoring could have a negative effect on reproductive success of an endangered seabird. Several key parameters, including chick survival and fledging success, were not monitored at the eastern colonies, which reduced the potential to detect the effect of fishing near colonies on reproduction. The choice of parameters to monitor reflected monitoring that was ongoing at the time the experiment was designed. In retrospect (and subject to the constraints of available resources), monitoring of additional variables would have been desirable (see section 5).
- The modelling accounts for the effects of factors other than island, closure, and month of sampling using a year effect. In principle, a key determinant of year-to-variation in reproductive success relates to the biomass of prey species. Acoustic surveys of local biomass were undertaken, but it was found that there is considerable variation over the breeding season and high sampling error (DFFE, 2021) so this variable could not be included in the analyses. Another factor that may have impacted reproduction on St Croix Island is the effect of bunkering near Gqeberha since 2016 (Pichegru et al., 2022).

2.5 Potential but not studied benefits to adult and immature African penguins from the ICE

The ICE measured variables that were considered to be direct measures or proxies for African penguin breeding success or post-fledging survival, but did not measure impacts of island closures on African penguin adult survival or immature survival. Evidence (outlined below) indicates that increases in prey abundance/availability would be likely to result in some gains in adult survival and immature survival.

Seabirds tend to have high adult survival and low fecundity (breeding success). Life history theory predicts that seabird adult survival is likely to be more strongly buffered than breeding success by behavioural responses because seabird population dynamics is driven more strongly by adult survival than by breeding success (Cairns, 1992). The prediction is that long-lived birds will tend to protect their survival by abandoning breeding when times are bad, so low breeding success is likely to be a more conspicuous consequence of low food availability around colonies than is low adult survival. Testing whether there is a relationship between forage-fish stock biomass and adult survival of forage-fish dependent seabirds is made difficult because few studies have collected long-term data on adult survival rates of seabirds in locations where there are matching time-series of forage fish stock biomass data. Nevertheless, several studies have found that adult survival rates are influenced by food availability. While none of the studies listed below are directly comparable to the African penguin situation, they provide an *a priori* basis to raise the expectation that there are fishery-related impacts on adult and immature survival.

- Black-legged kittiwake adult survival is correlated with prey density in the non-breeding area in winter (Reiertsen et al., 2014) as well as in the breeding area in summer (Oro and Furness, 2002; SSERenewables, 2022).
- Black-legged kittiwake adult survival and breeding success at Shetland (north Scotland) were both strongly affected by Shetland sandeel stock biomass (Oro and Furness, 2002).
- Black-legged kittiwake adult survival and breeding success at the Isle of May (east Scotland) were both reduced in years when sandeel fishing occurred on the ICES Sandeel Area 4 stock compared to years when there was no sandeel fishery (Frederiksen et al., 2004).
- Return rates (a proxy for survival) of black-legged kittiwake, Atlantic puffin, common guillemot and razorbill at the Isle of May all show strong asymptotic relationships with ICES Sandeel Area 4 sandeel stock biomass (SSERenewables, 2022).
- Return rate of adult Arctic skuas (parasitic jaegers) at Shetland as well as their breeding success was increased by supplementary feeding of broods, im-

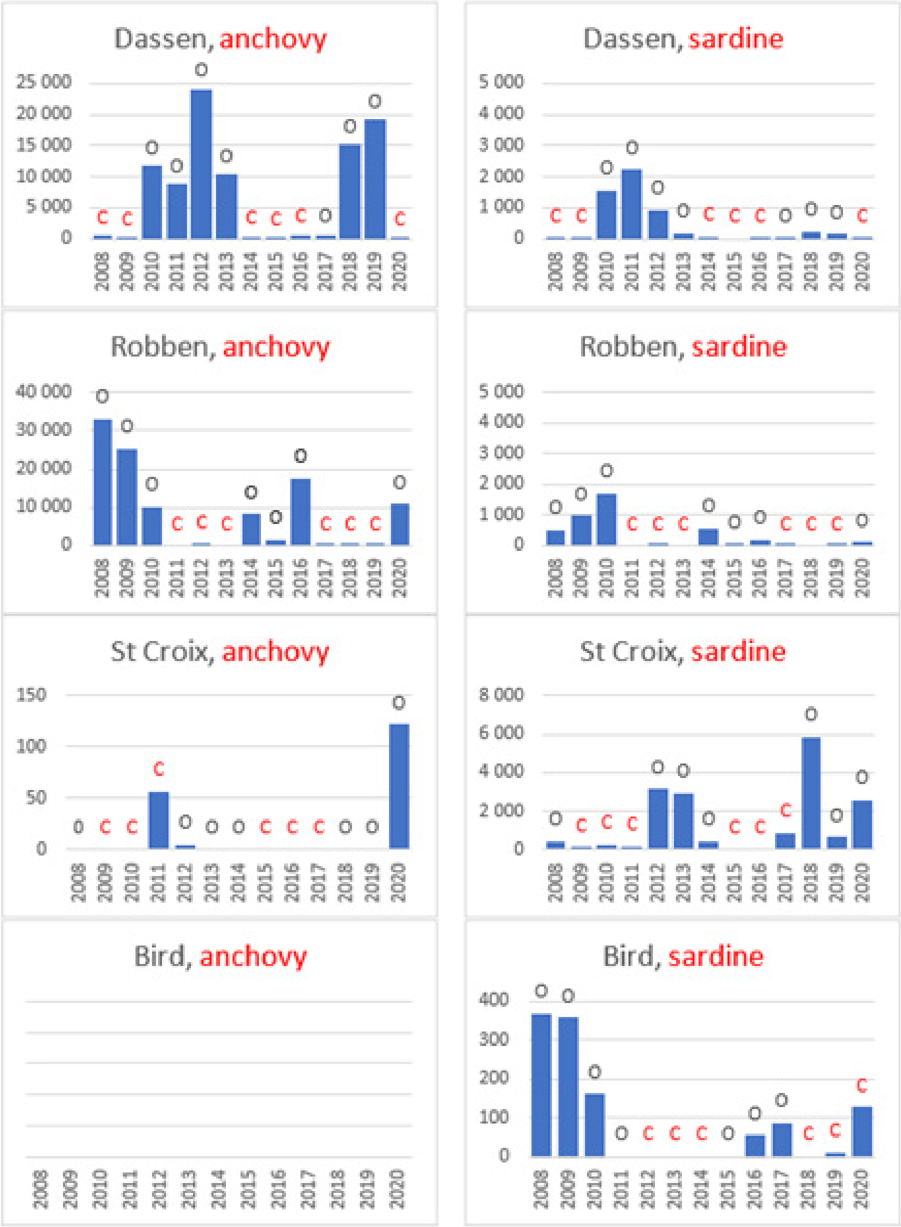


Figure 2.4: Catches of sardine and anchovy taken inside the 20-km closures during the duration of the ICE. Letters above each bar denote years when the areas were open (O) or closed (C). Figure credit to J Coetzee (DFFE, pers. comm.).

plying that low sandeel abundance was likely responsible for low adult survival in that species as a result of the increased costs of breeding when food was scarce (Davis et al., 2005).

- Low food availability reduced adult little auk body condition and reduced adult survival (Harding et al., 2011).
- Increased parental effort by breeding common guillemots (common murres) when foraging conditions deteriorated resulted in reduced adult survival rate and only partly compensated for low prey availability so also resulted in reduced breeding success (Wanless et al., 2023).

Measuring survival of immature seabirds is much more difficult than measuring survival of adults. There is evidence that survival rates of immature seabirds tend to be lower than those of adults (Horswill and Robinson, 2015), presumably because immature animals are less experienced and therefore less competitive. That suggests that low food availability would be likely to impact immature animals more strongly than adults. Therefore, gains from improved prey availability may benefit immature survival more than adult survival. Few studies report examples of change in immature survival rates, but immature survival of crested terns was strongly reduced when forage fish prey biomass was depleted (McLeay et al., 2008).

Evidence from other studies therefore suggests that the ICE is likely to have led to some unquantified improvement to adult and immature African penguins in addition to the quantified gain seen in breeding success for the western breeding colonies. It is impossible to determine the magnitude of any unquantified gain, but it is likely to have

3. BASIS FOR EVALUATING FISHING IMPACTS OF CLOSURES

3.1 Background

The literature investigating the impacts of fishery area closures on commercial fishing fleets and coastal communities highlights the importance of considering the short-run, long-run, and heterogenous effects across communities and fishers (e.g., large- vs small-scale). The short-run impacts on the harvesting sector include the displacement of the vessels from the closed areas that in turn could result in lower (or lost) catches, greater fishing costs, and lower revenues, everything else being equal. The short-run changes to the harvesting sector can also result in changes in throughput into processing facilities, which could lead to fewer shore-side jobs and less product. The long-run impacts include potential changes in shore-based infrastructure (e.g., processing capacity, fueling stations, bait stores, and ice availability), and the number of vessels operating in the fishery.

Both the magnitude and importance of the short- and long-run impacts are unlikely to be uniformly distributed across fishery participants and coastal communities. The placement and size of a closure could, for example, raise the cost of fishing for smaller vessels by increasing their steaming time to the open fishing grounds in a way that results in the exit of these vessels from the fishery over time. Vessel exit can have knock-on effects to the communities in terms of economic activity, shore-side infrastructure, employment, and social wellbeing. Implementing closures, including those to protect ecological processes, in South Africa will impact the fishing industry and local communities to some extent, but accurately quantifying this is challenging.

Economic methods to measure the changes due to a closure differ for the most part on according to whether the focus is on predicting the impacts before the intervention is implemented (*ex-ante* analysis) or measuring the impacts after the intervention is in place (*ex-post* analysis).

Section 3 is organised as follows. Section 3.1 is divided between a summary of the random utility class of model that is generally used to predict the impacts of proposed fishery closures and program evaluation methods that measure the causal impact of a fishery closure on the harvesting sector. Section 3.2 reviews the opportunity-based model (OBM) and section 3.3 reviews the social accounting matrix (SAM) modelling. Section 3.4 assesses the integration of the results from OBM and SAM modelling by highlighting how lost catches on the water are mapped back to coastal communities and regional economies.

3.1.1 *Ex-ante* analysis of the harvesting sector

The literature on the *ex-ante* analysis of the impacts of proposed fishery closures is dominated by random utility models (RUMs), which are statistical models of fleet behaviour (RUMs are a class of discrete choice models (DCMs)). While a RUM can take several forms, often researchers model the decision on whether to go fishing and where to go fishing conditional on taking a trip (see Figure 3.1). Vessels/fishers choose to go on a trip when the economic returns to taking a fishing trip are greater than the outside opportunity cost of not fishing, and fishers choose to fish in site *i* when their expected net returns from fishing in site *i* are larger than the other sites.[1] The expected net returns of a site *i* consist of the vessel's expected catch and price, travel distance to the site from their current location (port or another fishing site), fuel prices, and other variable costs.

RUMs have been applied to a range of fisheries from those for sedentary species (Smith, 2002; 2005; Marcoul

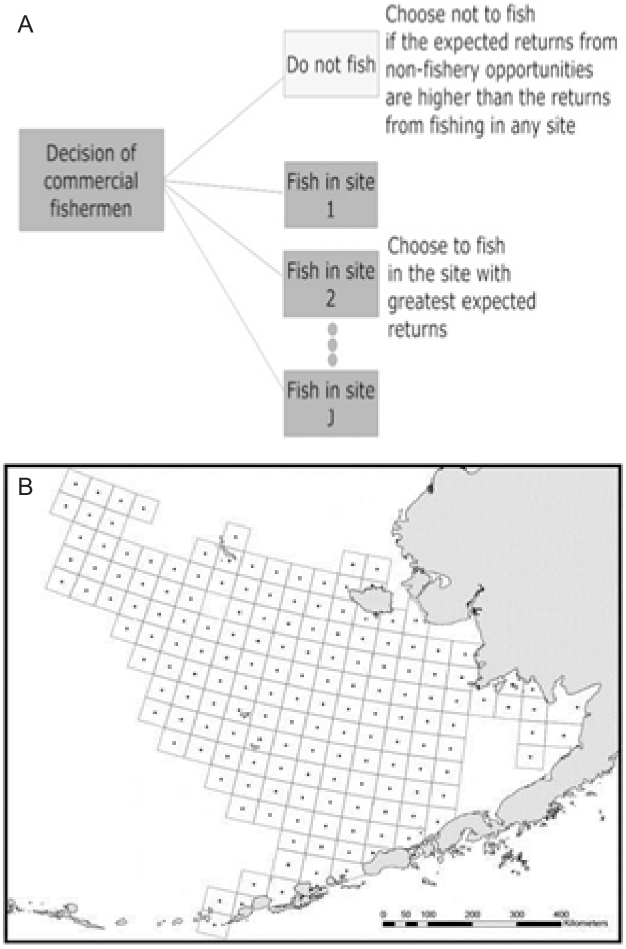


Figure 3.1: Basis of random utility models: Panel A is a stylised decision tree of a commercial fisher (vessel) in any given decision period (Source: Smith et al., 2010). Panel B is an example of the spatial choice of sites available for fishers in the Bering Sea of Alaska (Source: Abbott and Wilen, 2011)

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[1] Extensions of the basic RUMs include variables such as variance of the expected net returns (Dupont, 1993; Mistiaen and Strand, 2000; Hutniczak and Münch, 2018), preference heterogeneity (Smith, 2005), state dependence (your past experience affects future choice) (Holland and Sutinen, 2000; Smith, 2005), evolving information and information sharing (Curtis and McConnell, 2004; Abbott and Wilen, 2011), spatial correlation and learning (Marcoul and Weninger, 2008; Hutniczak and Münch, 2018), bycatch avoidance (Haynie and Layton, 2010; Abbott and Wilen, 2011), and multiple fleets and fisheries (Depalle et al., 2020).

Photo credit SAPFIA – South African Pelagic Fishing Industry Association

and Weninger, 2008) to those for pelagic species (Curtis and Hicks, 2000; Mistiaen and Strand, 2000; Curtis and McConnell, 2004). For nearshore sedentary species, often vessels fish single-day trips choosing a few fishing grounds to visit (Eales and Wilen, 1986; Smith, 2005; Marcoul and Weninger, 2008). For finfish species such as groundfishes or tunas, vessels make multi-day trips (Curtis and Hicks, 2000; Curtis and McConnell, 2004; Hicks and Schnier, 2008; Abbott and Wilen, 2011; Hutniczak and Münch, 2018). When developing RUMs for multi-day trips (e.g., purse-seine tuna fisheries), it is common to treat the choice of the first location separately, and then conditional on that choice, model the subsequent site choices (Sun et al., 2016).

Two interrelated challenges to RUMs are the spatial (definition of a site or fishing ground) and temporal unit (e.g., daily, weekly), and the estimation of a vessel's expected catch at the set of fishing sites when the vessel is on a trip (Smith, 2000; Dépalle et al., 2021). Studies have employed various methods to calculate expected catches that depend on the assumptions about the set of information available to the vessel at a particular time (Abbott and Wilen, 2009 ; Dépalle et al., 2021), including the ephemeral nature of that information (e.g., fish stocks might only stay in a particular location for a short period of time or the distribution of the fish stock in a particular location might be more stable from month to month and across years). For example, it is possible to use only vessel level information (e.g., catches at a particular site within the last week/month and/or the same week/month in the previous year). However, it is also possible to assume that vessels share information by including fleet level information (e.g., catches of similar vessels at a site within the last week/month and/or fleet catches in the same window of time in the prior year). If no vessels have visited a site in the relevant window of time, then expected catches can be assumed to be zero. The formation of expected catches will lack necessary observations if the definition of a fishing site is so small that there are few past observations that fall within it or the window of time is too short (Dépalle et al., 2021). Given that there is no theory on how fishers form expectations of catches at different sites, most analyses carry out robustness checks with different weighted combinations of own and fleet information across different site definitions and time windows (Dépalle et al., 2021).

The estimated RUM can be applied removing from the choice set the sites that are included in the closure area to assess the short-run impact of a proposed closure (e.g., Smith and Wilen, 2003). Conditional on the closure, the RUM predicts the number and timing of trips, the displacement of the fleet due to the closure (the model statistically reallocates the trips to different sites based on the empirical model of fleet behaviour), increases in travel costs, and changes in the catch composition (including different target species).

3.1.2 *Ex-post* analysis on harvesting sector

While RUMs dominate the literature predicting the *ex-ante* impacts of fishery closures, more recently researchers are utilising program evaluation methods that quantify the *ex-post* impacts of closures by estimating the counterfactual (Ferraro et al., 2019). For example, Smith et al. (2006) develop an empirical model to isolate the effects of marine reserves that accounts for multiple gear production technologies, heterogeneity in vessel captain skill, spatial heterogeneity of fish stocks, seasonal patterns in abundance, the effects of coexisting management policies, and the possibility that the harvesting sector anticipates reserve establishment.

Reimer and Haynie (2018) quantified the short-run impact of large-scale closures on the net revenue of the commercial Atka mackerel fishery in the North Pacific using difference in difference (DiD), propensity score matching, and synthetic control methods. DiD measures the counterfactual (what would have happened in the absence of the closure) using the trend over time in a control group (vessels that do not fish in the closure). The assumption is that any differences between the treated group (vessels that fish in the closure area) and the control group are invariant over time and by using their parallel trends before the intervention, these differences will net out leaving the impact of the closure on the treated vessels. Favoretto et al. (2023) employed DiD methods to evaluate the impact of Mexico's Revillagigedo National Park on industrial fisheries.

While DiD assumes that all the control vessels contribute equally to the comparison group, propensity score and synthetic control methods develop a more refined measure of the control unit for each treated unit. Propensity score methods, for example, estimate for each vessel the probability of being in the treated group as a function of pre-treatment observable characteristics, such as vessel size, gear technologies, home ports, boat fixed effects, net revenue, etc. Various criteria (e.g., five nearest neighbours) are then used to match treated and control units based on similar propensity scores, which are estimated predicted probabilities of fishing in the closed area. The assumption is that treatment and control vessels with similar propensity scores are statistically identical except that the treated vessels were impacted by the closure.

Any method of evaluation will need to address the challenges associated with accounting for exogenous time-varying factors, such as stock abundance trends, prices, costs, local and regional labour markets, global market forces (exchange rates), and endogenous time-varying factors such as behavioural responses to the closures that impact the ability to measure the counterfactual. An example of the latter is when impacted vessels are displaced to the fishing grounds occupied by the comparison set of vessels resulting in congestion on the grounds and lower catch rates for the control fleet than otherwise would have occurred had the closure not happened. Ferraro et al. (2019) discuss these challenges along with other biological and market mechanisms that can lead to contamination or biased estimates of the counterfactual.

3.2 Opportunity-Based Model (OBM) estimates of lost catch.

The OBM was used to estimate the impact of closures on catches by the South African pelagic fisheries targeting anchovy and sardine. Because the number of vessels, shoreside infrastructure, and behaviour of the fleet are held fixed over time, the impacts estimated are short-run even though they are calculated over ten years to develop an average loss. The OBM quantifies the impacts of closures under the assumption that catches that occurred in the closed area when it was open are a measure of the catches that would have occurred if the closed area was not closed.

Unlike the early literature on the impacts of marine reserves on catches, which assumed that all catches would be lost when an area is closed, the OBM introduces a set of rules to capture potential behavioural responses of the fleet to the closures. These rules were informed by interviews with fishery operators and include how to replace catches taken within closures with alternative catch opportunities observed across areas and species within a narrow window of time (generally same day and year) considering estimated boat factors (vessel fixed effects from GLMM estimation), boat caps, and potential spillover from other closures. Opportunity catches are also adjusted up or down based on an auxiliary analysis used to evaluate possible biases in predicted aggregate catch in any given year depending on the specific rules used by the OBM.

Using these rules, the OBM develops a measure of the average irreplaceable catch stemming from the proposed closures using catches in the closed areas over ten years and the average catch that could be replaced (opportunity catch) for each species at the island closure level (see Appendix E for further details together with figures and summary tables of the results).

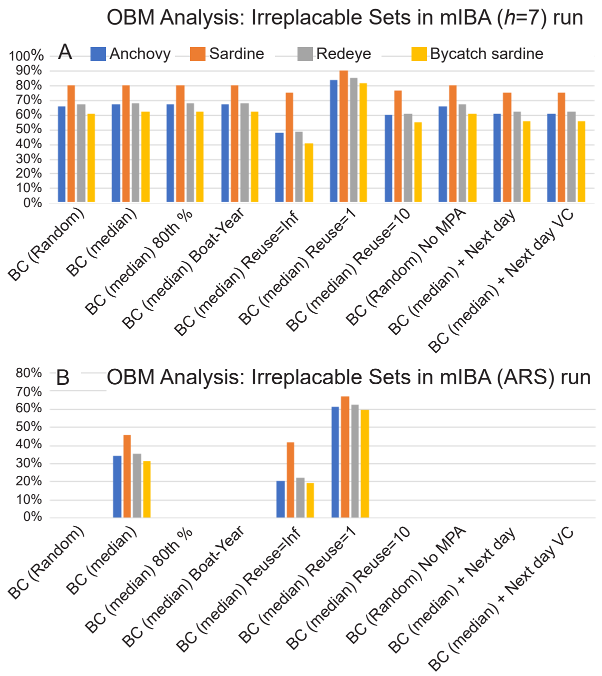

Figure 3.2: Percentage of Irreplaceable Sets in the mIBA (*h* = 7 km) run of the OBM model (Panel A) and in the mIBA ARS run of the OBM model (Panel B) across a set of model sensitivities. In Panel B, a blank corresponds to scenarios that were not run for the mIBA ARS case

The two key modelling assumptions of the OBM are: (a) the observed catches taken in a given day outside a proposed closure provide a complete set of potential alternative fishing opportunities for replacing the catches taken that day within the proposed closure; and (b) there is a maximum number of times each alternative fishing opportunity could be used to replace those catches (referred to as “Reuse”). The former relates to the information set the fishers have at any point in time where the OBM implicitly assumes all vessels fishing on the same day have the same set of information and there were no additional potential opportunities where and when fishing did not take place. The latter is questionable considering that additional fishing opportunities, beyond those used when the areas were opened, could be searched for and identified in response to the implementation of a closure. The search for alternative fishing opportunities would be more effective if the fleet shared the information about fishing locations, as was reported to happen during the June Panel meeting. It also implicitly assumes the lack of seasonality of fishable aggregations from one year to the next and full information decay of fishable aggregations in a location within a day. These assumptions combine to lead to a low of 40% (Reuse = infinity for sardine bycatch) and a high of 90% (Reuse = 1 for direct sardine) of the sets within a closed area (when it is open) being classified as irreplaceable in the marine Important Bird Area (mIBA) (*h* = 7 km) run (Figure 3.2 Panel A). The fraction of irreplaceable sets is lower in the mIBA Area Restricted Search (ARS) run but still ranges from a high over 60% to a low around 20% depending on the scenario (Figure 3.2 Panel B). More detailed calculation of catch losses for different closure proposals and OBM assumptions, summarized in Appendix E (Figure E.3), indicate that the great majority of the estimated catch losses are due to the high fraction of sets classified as irreplaceable under the OBM rules while only a very small fraction of the catch loss was due to lower average catch rates of replacement sets (“opportunity losses”). In common with RUMs, if no vessels have fished at a site in a window of time, the expected catch of a vessel going to that site would be zero. In forming an expectation of catches for use in RUM, analysts consider a wider window of time (fishing within the last month, same month last year, etc.) while allowing for some weighted average of private information (catch rates of the vessel in the sites) and fleet-wide information (perhaps due to sharing of information at sea, observing landings, observing activity at sea) to calculate the expected catches in any site *i* in period *t*. The Panel **agreed** that the current window of same day (or same day plus one) is likely too constraining and recommends further statistical analysis should be undertaken to better understand the seasonal nature of anchovy and sardine sets/catches across the fishing sites, especially along the west coast.

Whether to sample alternative opportunities with or without replacement is an important issue in the OBM analysis. The Panel **agreed** that the OBM would likely underestimate the potential opportunities outside the closed area on a given day (conditional on all the other assumptions being appropriate) if, for example, 100 catches (sets) within a closed area are matched to just a single catch (set). Currently, the results are presented for the case of allowing only one replacement (Reuse = 1 corresponding to sampling without replacement), only five times (sampling with replacement but only five times), and an infinite number of times (sampling with replacement). The Panel **agreed** that the random matching of catches is an improvement over the percentile method but **recommended** that all results should be presented for the Reuse = 1, 5, and infinity cases (see section 6 for additional suggestions on statistical methods to match sets).

The OBM is not able to quantify important potential changes to the net revenue of the fleet due to closures. Net revenue is the total revenue (ex-vessel price*catch) less the variable costs of fishing that include fuel costs (fuel price*fuel used), labour costs, supplies, etc. The fuel costs capture steaming time to and from the grounds, searching efforts, and fuel spent while fishing. Closures can increase fuel costs due to greater travel distances and can also reduce the quality of the catch at the time of landing, leading to lower ex-vessel prices and total revenues (e.g., greater spoilage, lower quality)[2]. The impacts on net revenues are likely not uniform, as smaller vessels might have less ability to travel further due to the riskiness of being out to sea for longer and a more limited fuel capacity. The Panel **agreed** that understanding the impact of closures on the net revenue as well as changes in catches is important for understanding both the short-run impacts and the potential long-run impacts due to changes to the fleet composition, shore-side infrastructure, and coastal community dynamics.

3.3 Social Accounting Matrix (SAM) analysis

Quantifying community economic impacts of fishery policy changes requires understanding about how changes in production on the water translate into changes in the production of goods and services shore-side either directly or indirectly. Economists use several methods to carry out such analysis, such as input-output (IO) models, social accounting matrix (SAM) models, and computable general equilibrium (CGE) models (Seung and Waters, 2006)[3]. Across the methods, the data requirements of the models are extensive, including industrial output, employment, value-added, final demands, and imports. CGE models, which are the most expensive to develop but are the gold standard for quantifying community impacts, allow for changes in relative prices, substitutions across inputs (labour, capital), and compute the welfare implications of the economic shocks (e.g., welfare impacts of job losses rather than just quantifying the number of jobs lost) (Seung and Waters, 2006). SAMs improve on simple IO models by quantifying impacts on the distribution of income, but unlike the CGE framework hold prices fixed and do not allow for substitutions (Seung and Waters, 2006). SAM results, therefore, should be viewed as a very short-run measure of the impact (snapshot) whereas a CGE model can capture more dynamic short-run and medium-run responses of the economy (Seung and Waters, 2006). Because SAMs are designed to analyse demand-driven impacts in the local economy (e.g., change in consumer spending), these models tend to overestimate the impacts of supply-side shocks, such as a reduction of catch (Seung and Waters, 2013; Seung, 2014).

UrbanEcon developed a SAM model that models a shock to the regional economy from a reduction in catches due to the closures as calculated by the OBM (irreplaceable catch). The SAM model traces the shock through the economy by modelling a set of linear relationships that capture the direct, indirect, and induced changes (Figure 3.3). Characterising the value chain of the pelagic fishing industry is a way to decompose the direct and indirect impacts of a change in the total catch of sardine, anchovy, or redeye (Figure 3.4). Vessel owners, captains, and crew experience direct income effects from a reduction in the catch, where the crew are paid on a share system based on the fishmeal price and catches rather than a fixed hourly wage. The lower catch results in less throughput into the shore-side processing facilities, which can be substituted in some situations with import quantities though often for higher prices (depending on exchange rates, and transportation costs). The higher costs of processing fish can result in a reduction in labour demanded by processing

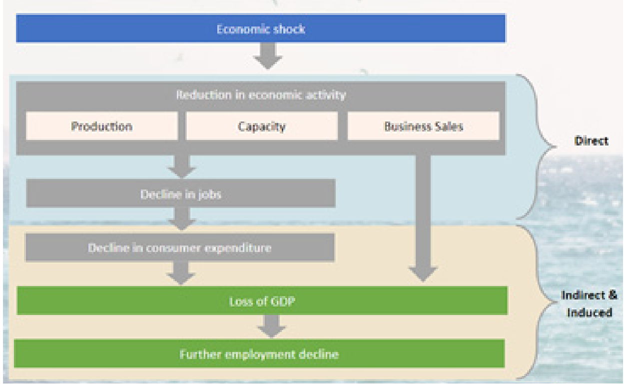

Figure 3.3: Social Accounting Matrix Framework for mapping changes in regional aggregate catches (economic shock) to changes in employment, regional gross domestic product, and regional income. (Source: UrbanEcon June 2023c)

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[2] Bergh (2016) states that fuel costs will increase approximately 23% around Dassen and Robben islands when considering the location of the replaceable sets, which depends on the priority ranking of substitute locations and the assumption regarding the feasible sets from which to search for a replacement.

[3] While the use of IO, SAM, and CGE models dominate the literature in terms of quantifying the impacts of the fishing sector on local communities, a recent paper by (Watson et al., 2021) takes an econometric approach to measuring the impacts using data from Alaska. They find "that a 10% increase in a community's annual resident fishery earnings leads to a corresponding 0.7% increase in resident income. This translates to an increase of 1.54 dollars in total income for each dollar increase in fisheries earnings" where fishery earnings are defined as total revenues of fishing for local permit owners.

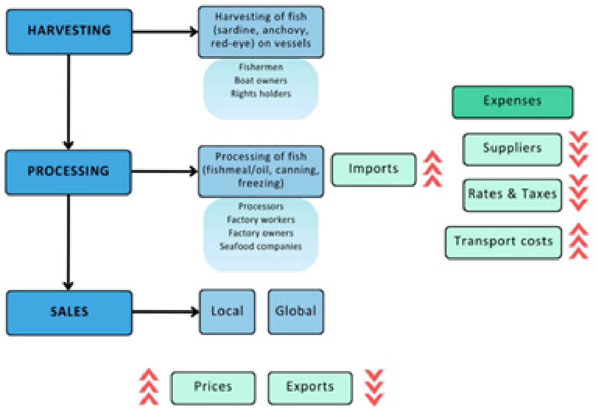

Figure 3.4: Value chain of the pelagic fishing industry, highlighting the pathways for loss in regional catches to the direct impacts in the SAM modelling (Source: UrbanEcon, 2023b).

facilities and lower overall economic performance of the industries. Sales locally or exported might also be impacted if the final output of fishmeal, canned, or bait products is lower due to the lower catches. Lost wages reduce income and purchasing power in the economy, lowering consumer expenditures. Lower expenditures, along with changes in sales, reduce economic output that can have further impacts on employment levels in sectors not directly related to fishing (induced effects in Figure 3.3).

An important impact of the proposed closures is the potential job losses both directly on the fishing industry and the knock-on losses due to lower GDP and income. UrbanEcon (2023a) predicts in the preferred scenario, for example, "full-time employment is expected to decrease substantially, with a reduction of 655 jobs" where the direct impact to harvesters is a loss of 35 with indirect losses of 93, and in the processing sector, the direct losses are 181 out of a total of 527 losses. Using the regional distribution of labour in Table 5.1 of UrbanEcon (2023b) and the direct job losses in Table 5.2 of UrbanEcon (2023b), the direct job losses regionally to the harvesting sector are 11.5 west of Cape Point, 8 between Cape Point and Cape Agulhas, 7 in Mossel Bay, and 5.6 in the east.

How to interpret the significance of job losses on regional economies and welfare depends on the quality of the local labour markets, whether the losses are seasonal workers, and whether the losses are permanent or temporary (Holland et al., 2012). If local labour markets are fluid with low unemployment, then a job loss in one sector could be negated by an increase in another sector, which makes interpretation of the economic costs associated with job losses more difficult. On the other hand, if losses occur in remote locales with incomplete labour markets with high unemployment (as is the case for several of the towns where fishers and processors are based), then these losses contribute directly to the economic costs due to closure rather than being a transfer from one sector to another. In addition, if the job losses are from seasonal workers or temporary layoffs, then the impacts are likely transient and fleeting as opposed to the case where the job losses are due to the closure of the shore-side processing facility (Watson et al., 2021). The latter will have long-run impacts on the local fishing vessels, employment, and incomes, as may be the case for several of the affected local towns. The Panel **agreed** that while the SAM is a useful tool for creating snapshots of the impacts on regional economies it **recommended** that further work needs to be done on the long-run socioeconomic impacts to local communities due to the prospective closures. Moreover, it notes that the predicted effects of closures depend on the reliability of the estimates of lost catch from the OBM, which the Panel **agreed** is likely to provide overestimates given its restrictive assumptions related to the set of opportunities that are available to replace catches in closures (Appendix E). These overestimates are of uncertain magnitude but may be large.

The heterogeneous impacts on fishing operations (e.g., small vs large vessels) are another important factor in understanding the relative significance of the changes to regional economies. In the preferred scenario, UrbanEcon (2023a) shows "that smaller vessels (less than 20 metres) will be the most highly impacted … the largest vessels (above 25 metres) will be the least impacted… meaning that the viability of maintaining operations is variable dependent on boat size, and the larger the boat, the higher level of security it has in its operations." These impacts, however, are not evenly distributed across communities and closures, as some ports will be more dominated by larger vessels (and vertically integrated companies). The Panel **agreed** that while the SAM model provides a measure of the distributional impacts across vessel size it recommends that further work should be done to understand the impacts on local communities more dependent on smaller vessels, such as those operating in the St. Croix area.

Given the complexity of the regional economy, any model (IO, SAM, and CGE) will involve many parameters and relationships, some of which are supported empirically and some of which must be assumed. The UrbanEcon SAM model is not unique in this respect, and the use of interviews with the fishing industry is a best practice to fill in missing data. However, some important questions remain regarding the interpretation of the SAM results. Are the "losses" out of the SAM due to the proposed closures within the standard fluctuations of the local economy due to other kinds of economic shocks, such as fuel prices, exchange rate fluctuations, etc.? Fuel price increases, for example, would be expected to result in less fishing due to higher travel costs, less processing due to higher import costs of products, lower sales, lower consumer expenditures, etc. Are the short-run job losses from a fuel price

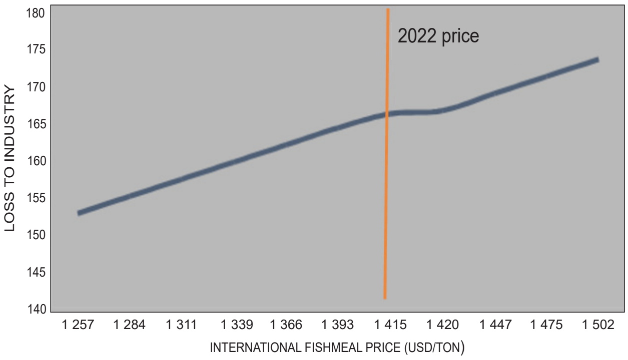

Figure 3.5: Sensitivity on the loss to the industry from the range of fishmeal prices (UrbanEcon, 2023c)

increase greater than the predicted job losses from the preferred scenario? How important for the loss estimates are the assumptions regarding the relative wages of the processing and harvesting sector, especially since most of the job losses occur in the processing sector? How do the results change if the conversion of total employment FTEs is based on a different rate of fishing days per year (currently, 175 fishing days per annum is assumed)? The Panel **agreed** that additional sensitivity analysis of the SAM results should be carried out to have a better understanding of the range of possible regional outcomes from the prospective closures.

In response to queries by the Panel, UrbanEcon carried out additional sensitivity analysis on the range of aggregate outcomes by varying expected catch loss, and fishmeal price. Variations in the global fishmeal price imply that a loss of catch in one year might not have the same economic value as a loss in another year (Figure 3.5). Specifically, UrbanEcon found that "the fishmeal industry performs at its best when international prices are highest – and therefore the largest industry loss will be experienced whereby the island closures negatively affect the level of raw input (anchovies, red-eye, and sardine off-cuts and bycatch) and international prices are highest" (UrbanEcon, 2023c). These results are not surprising, but also highlight the limitations of the SAM modelling assumptions. With the crew paid in proportion to the fishmeal price, as the fishmeal prices increase, the income of the crew increases, but because some crew also lose their job due to the catch reductions, there are then fewer crew members earning more money in a year with higher fishmeal prices. How much the increase in wages to the remaining crew offsets the losses due to fewer workers is an empirical question that cannot be addressed given the linearity and fixed prices (output, input, and wages) assumptions embedded in the SAM framework.

3.4 Downscaling lost catches at sea to regional economies

The critical piece in quantifying the regional impacts of the proposed closures is the mapping of irreplaceable catches that occur at sea to the ports/local communities. Based on responses to a query of the Panel, there appears to be a discrepancy between the regional catch loss totals provided by the OBM based on where the catch is caught, the regional economic impact measurements determined by employment shares in the SAM modelling for 2022, and the breakdown of the lost catch based on shares of regional processing (Table 3.1). The later breakdown is not currently utilised in the SAM analysis and is imputed based on the average lost catch between 2011 and 2019 for anchovy, bycatch sardine, directed sardine, and redeye considering differences in the location of industrial and sardine processing facilities and landings. While the share of catch processed in any facility and port can change from one year to the next, which is the argument UrbanEcon employs when justifying the use of employment shares (Letter from UrbanEcon to Panel dated June 9th, 2023), Table 3.1 highlights the potential for different measures of regional impacts based on the method employed and/or the catch years used in the analysis. The Panel **agreed** that given little empirical justification for one method, each allocation method should be used, and the results compared across the different cases, to better inform discussions on which communities are likely to be most impacted.

Table 3.1: Mapping lost catches to regional economies. Column 1 shows the percentage of lost catch based on the current method for how OLSPS allocates irreplaceable catches in closure areas to regions, Column 2 shows the percentages that UrbanEcon uses based on employment in the fishing sector (harvesting and processing), and Column 3 shows a new set of percentages that OLSPS calculated based on the share of the catch that is processed shore-side by region (Source: Data provided to the Panel by OLSPS on June 9, 2023)

Region	OLSPS lost catch	UrbanEcon employment shares	Regional processing
Western Cape	17%	33.0%	49.4%
Cape Point to Cape Agulhas	60%	27.1%	27.0%
Mossel Bay	0%	23.5%	12.3%
East	23%	16.5%	11.3%

Penguins at Boulders (photo BM Dyer)

4. CRITERIA AND APPROACHES FOR EVALUATING TRADE-OFFS BETWEEN BENEFITS TO PENGUINS AND COSTS TO FISHERY

4.1 Introduction

There are various aspects involved in any decision regarding the locations and duration of island closures intended to conserve African penguins. These include the location and size of the closures, their seasonal duration, and whether and when any closures will be reviewed. The technical review of these aspects is given in sections 2 and 3. There are three primary trade-off axes to consider when selecting closures (see Figure 4.1 for options considered during the Panel discussions):

- The benefit to penguins of the closure.
- The cost (economic and social) to the fishing industry and the communities, especially where fishing and processing operations are based.
- The ability to evaluate the effectiveness of the closures.

The choice of the location and size of closures, and their duration depends on the relative weights placed on the different anticipated outcomes by the decision-makers. Guidance on these weights may be informed by legislation, existing policy frameworks and international agreements. Recommendation of a specific outcome lies outside the scope of the Panel.

The Panel **recommended** that, if designated, closed areas to protect penguins during breeding, should be year-round, unless reasons demonstrate otherwise, primarily because egg laying and chick provisioning occur year-round, and these areas may be important during critical pre- and post-moult periods. The Panel further **recommended** that, if designated, closed areas to protect penguins should be reviewed at a time when results are available to investigate life-history processes such as juvenile recruitment, adult survival and hence population growth rates. This may be at a time between 6 and 10 years after designation. Other reasons to review such closed areas might include major socioeconomic changes in the fishery and processing, or stock abundance, or similar consequences of prey resource change.

4.2 Evaluating effectiveness

The “effectiveness” of a set of closures may be evaluated using a closure program that involves opening and closing areas to fishing in an experimental manner to test hypotheses and quantify changes in the demographic parameters

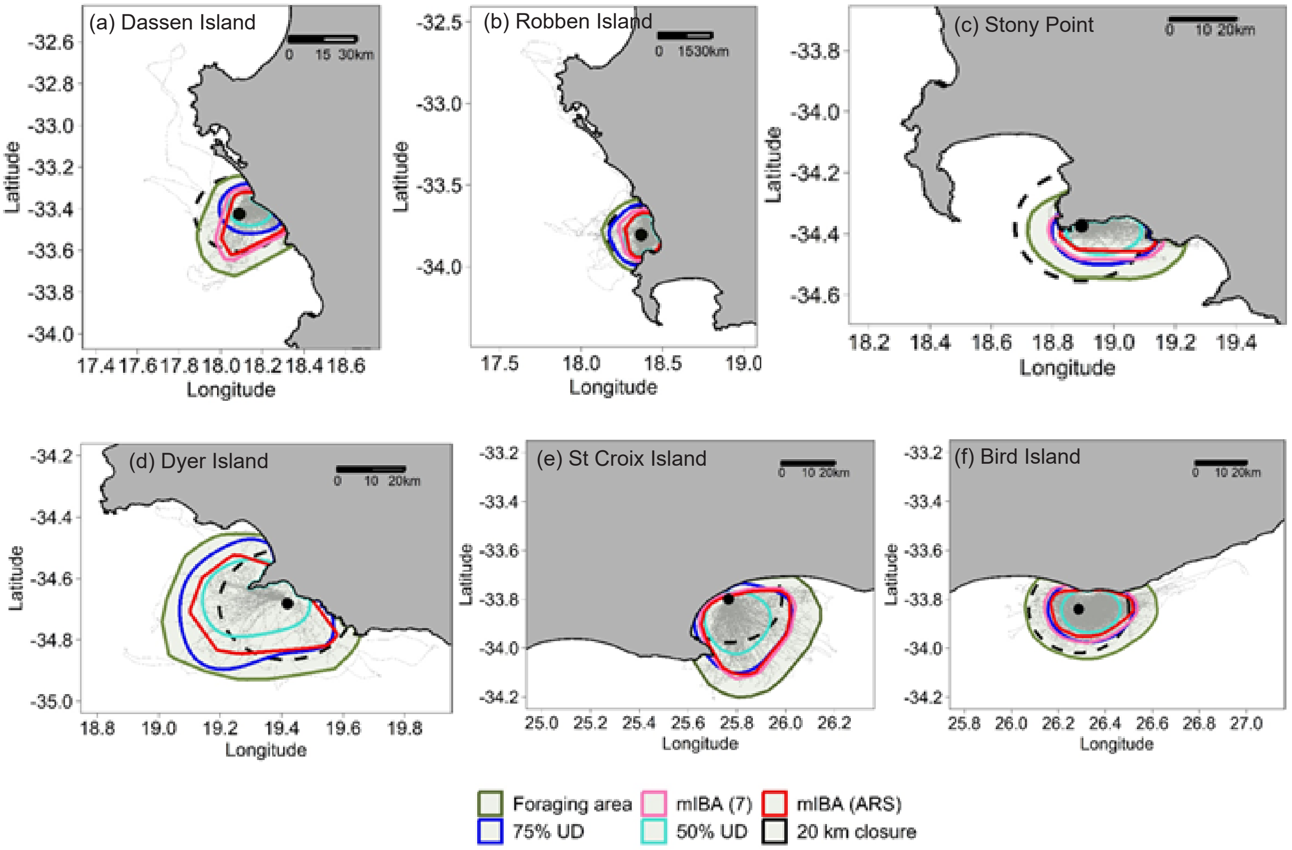

Figure 4.1: Comparison of alternative closure options including the 20-km ICE closures, the inclusive foraging areas defined as the 90% utilisation distribution—UD, (green open polygons), the UD50 and UD75 aggregated kernel density distributions, as well as two mIBA core area versions calculated using a smoothing factor of 7 km (mIBA (*h* = 7 km)) or the ARS scale value calculated for each colony (mIBA(ARS)) using tracking data of African penguins tagged at (a) Dassen Island, (b) Robben Island, (c) Stony Point, (d) Dyer Island, (e) St Croix Island and (f) Bird Island. From McInnes et al. (2023)

of penguins, and hence their population growth rate. However, closure programs are not usually structured in this way, with most such programs involving long-term closures and monitoring of the impacted populations. The Panel strongly **recommended** that monitoring should take place irrespective of whether there is an experimental (alternating open and closed) component to the closure program. Section 5 identifies several ways in which monitoring can be changed to more precisely capture changes in penguin demographics and behaviour and hence the effects of any closures on the penguin population. Section 6 outlines improvements to data collection and analysis to facilitate an evaluation of the effect of any closures on the fishery and associated communities.

The Panel does not consider it essential that there is an ongoing experimental approach (as opposed to monitoring for conservation purposes). However, the Panel provides the following recommendations should there be an experimental component to any future closure program:

- The aim of the experimental structure should be to not only estimate parameters related to reproductive success, but also additional parameters, in particular juvenile recruitment, adult survival and hence population growth rate. This is because there is little value in conducting future experimental manipulations if the aim is simply to estimate the effect of closures on reproductive parameters given this is already adequately informed by the ICE (see section 2).
- There is little benefit in trying to use an experimental framework in regions (e.g., the eastern Cape) where it is (currently) not possible to monitor important parameters such as adult and chick survival. Based on the data already available, and the ability to undertake regular monitoring, the western and southern Cape regions should be the focus of any future experimental closure program.
- Given the necessary focus on adult survival and population growth rate, it is desirable that a power analysis be conducted to identify an appropriate sequence of (possibly alternating open and closed) closures. The existing MPAs around some islands impose some constraints on the experimental use of closures and this should be taken into account in any power analysis.
- Conservation planning software tools, such as Marxan (e.g., Ball et al., 2009; Watts et al., 2017), provide a way to select areas given constraints on either the desired amount of closure by island or the cost to industry.

4.3 Quantify at-sea habitat area

The purpose of closing areas around penguin colonies is to protect penguin foraging habitat. Relatively little was known about the foraging behaviour of African penguins, especially about their preferred foraging habitats at the start of the ICE. The ICE had therefore been set up using a fixed 20 km radius as the open-closed management option (Figure 1.1). With recently available telemetry data, closures may be designed to achieve a more effective protection of the penguins' foraging area.

The at-sea habitat used by seabirds whilst foraging varies throughout the year. Although different seabird species have very different characteristic scales of habitat use, all species show variability in relation to their life-history constraints. Seabirds are most constrained during breeding when they need to return to land to provision their offspring. In general, seabirds, including penguins, forage across spatial scales that differ between incubation, early chick rearing (the brood stage), late chick rearing (the crèche stage) and post breeding (e.g., Warwick-Evans et al., 2018). For African penguins, due to their disturbance sensitivities, most information about foraging is only available during the early chick rearing phase when foraging scales are likely to be most constrained. During this period adults can only travel short distances given their need to return to their chick at short temporal intervals. Thus, resource availability during early chick-rearing is critical, given parents are less flexible. Consequently, all estimates of preferred foraging habitat based on tracking data from early chick-rearing are likely to be conservative.

The marine habitat available to penguins varies spatially and temporally, with some areas being preferred, given the availability of prey. Determining such preferred areas is important, especially if resource competition with fisheries is a concern. Estimating areas of preferred foraging habitat can be achieved through numerical spatial analysis of telemetry (tracking) data. Different analytical approaches are available, but in recent years robust methods that identify marine Important Bird Areas (mIBA) have become widely accepted (Lascelles et al., 2016; Dias et al., 2018), including for the identification of Key Biodiversity Areas (e.g., Handley et al., 2020).

Kernel density analysis calculates the density of locations by fitting a bivariate normal function with a pre-defined radius (smoothing parameter, h) around each location and summing up the values to create a smooth density surface. The kernel utilisation distribution (UD) is the isopleth that contains a certain percentage of the density distribution. To obtain core usage areas for foraging seabirds the 50% UD has often been selected (Lascelles et al., 2016). To align the smoothing parameter (h-value) to the scale at which birds use their marine habitat, behavioural characteristics evident within the telemetry data can be used. For example, periods of Area Restricted Search (ARS) when birds are actually feeding, can be identified through First Passage Time (FPT; Fauchald and Tveraa, 2003). Such methods are now commonly used (e.g., Trathan et al., 2008; Scheffer et al., 2010) in the analysis of penguin telemetry data.

The Panel **recommended** that analyses delineating mIBAs using ARS methods represent the best scientific basis for delineating the preferred foraging habitats during breeding. In the future, additional analyses would further improve understanding, especially with respect to how the spatial scale of any given mIBA might vary by year. The Panel **concluded** that such between-year variation is likely to be important, as the years of the ICE, during which most telemetry data have been collected, have been years of relatively low prey resource abundance.

Further, evidence related to the prolonged African penguin breeding season (e.g., Crawford et al., 2013), also highlights the need to ensure adequate resource availability is maintained within a given mIBA around the year, as the demand is not simply seasonal.

The Panel **recommended** that further validation of mIBAs should occur, in particular using dive data that provide objective identification of foraging locations, rather than commuting (or travelling) locations (see also section 5.9). Such analyses could be included in species distribution models (e.g., Warwick-Evans et al., 2018) that could be used to identify areas of key importance. However, important uncertainties remain, particularly if mIBAs are determined (as they have been) using telemetry data predominantly limited to early chick rearing when breeding adults are most constrained; further, that mIBAs may differ in the future, should prey resource abundance increase.

The life history processes of all species do not completely compartmentalise into distinct time periods or physiological mechanisms. Life-history events are often mediated through carryover effects, with events or activities occurring in one season, habitat, or life-history stage, affecting important processes in subsequent life-history stages (Crossin et al., 2010). Thus, seabirds arriving at a colony to breed must have already initiated certain physiological transitions, including with any associated resource accumulation (Crossin et al., 2010).

For African penguins, such carryover effects almost certainly occur, requiring adults to accumulate resources prior to breeding and prior to moult. This means that adequate prey resources are needed throughout different times of the annual cycle, such that delineating where birds forage and accumulate resources requires spatial information across the complete annual cycle. Outside the breeding season, reductions in resource competition that potentially facilitate reductions in foraging effort may benefit penguins prior to moult and post-moult, especially as these periods are energetically demanding.

Accumulating evidence shows that African penguins undergo predictable movements outside the breeding period (Sherley et al., 2017; Carpenter-Kling et al., 2022), suggesting that preferred habitats are also important at other times of the year. Importantly, it is now apparent that the mIBAs delineated using telemetry data from early chick rearing, are sometimes also important during pre- and post-moult foraging trips (Figure 4.2), even though they may only represent a part of important habitat during these other periods.

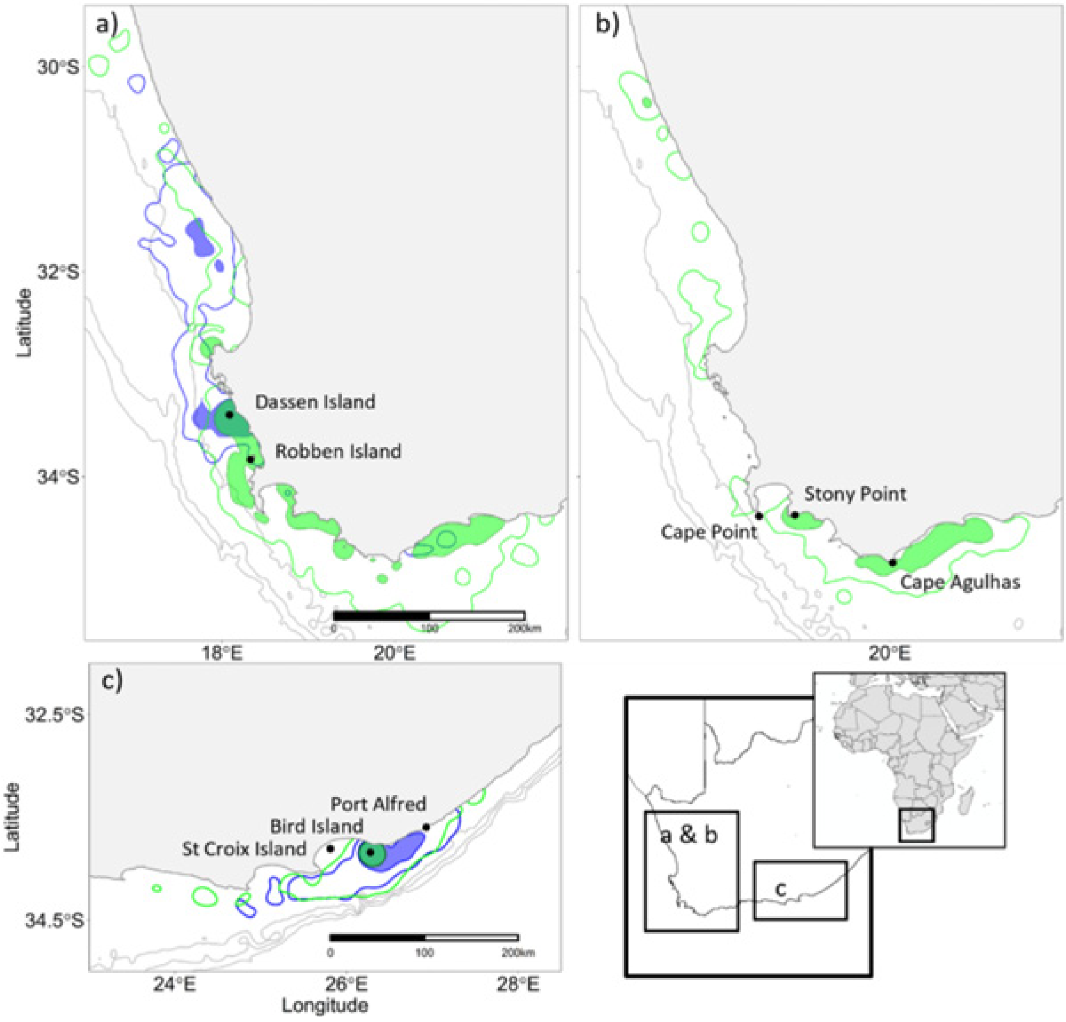

Figure 4.2: The distributional range (90% utilisation distribution—UD, open polygons) and core range (54% UD, shaded areas) of African penguins tagged at (a) Dassen Island, (b) Stony Point, and (c) Bird Island during their pre- (green) and post-(blue) moult foraging trips to the 200, 500 and 1 000 m isobaths (grey lines). Figure from Carpenter-Kling et al. (2022)

4.4 Trade-off space

One way to explore the trade-off between expected benefits to penguins and impacts on fishing is via trade-off plots (see, Hilborn et al. (2021) and Halpern et al. (2013) for examples of trade-off analyses). A trade-off curve (e.g., Figure 4.3) could demonstrate, for example, that the benefits to penguins (as quantified by the proportion of the foraging area that is protected) likely increases rapidly when small areas most used for foraging are closed, with the relative benefits to penguins declining as an increasing proportion of the foraging area is closed to fishing. Because not all closures of the same size are likely to have the same benefit, points A and B in Figure 4.3 demonstrate how a given (hypothetical) 40 km closure (point B) compares with the outcomes of another (hypothetical) closure with the same area but which more closely resembles areas of preferred penguin foraging habitat (point A). Based on the ICE experiment, it is not possible to assign quantitative estimates of the change in population growth rate associated with closed areas that differ from 20 km around colonies, but the qualitative changes in benefits to penguins with increasing closure areas are likely robust (increasing at a decreasing rate). Furthermore, for a given total closure area, closures that more adequately reflect preferred foraging areas will have greater benefits than those that simply close less valuable foraging areas. We also expect that lost fishing catches increase faster when the area closed increases in size, because as demonstrated in the OBM analysis, larger closures lead to more displaced fishing sets and a smaller area available for fishing (and hence fewer fishing opportunities). Based on the OBM results calculated for different alternative closure areas, we developed Figures 4.4 and 4.5, which provide a comparison of closure options across area closed and percent loss in regional catch. Figures 4.4 and 4.5 highlight how not all closures are equal in terms of the predicted lost catch and show that there are potential opportunities to reduce the impact on the fleet while at the same time increasing the amount of area closed (e.g., in Figure 4.4 compare the triangle and square on the blue line for Dyer Island and anchovy).

The Panel provides the following conclusions and recommendations regarding selecting closures given its review of the work identifying foraging areas and lost catch.

- It is desirable to identify a solution that minimizes societal costs and maximizes benefits to penguins; however, an optimal solution (or acceptable "balance") between competing objectives is not simply obtained by closing 50 percent of any given area.
- Conservation actions should be spread throughout the range of the species given each region is subject to different biophysical and anthropocentric threats.
- One approach (if curves such as those in Figure 4.6 can be created) is to find the point at which the change in penguin benefits (by increasing closures) matches the change in costs to society.
- The trade-offs between costs to the fishery and benefits to penguins in terms of the proportion of the foraging area closed will differ among islands and among sectors within the fishery. Consequently, the benefits to penguins and costs to industry should be considered by island (or region) and not simply at the national level (see below). In addition, given the heterogeneity within the industry, expressing costs and job losses by sector (e.g., for small scale operators) would also seem appropriate.

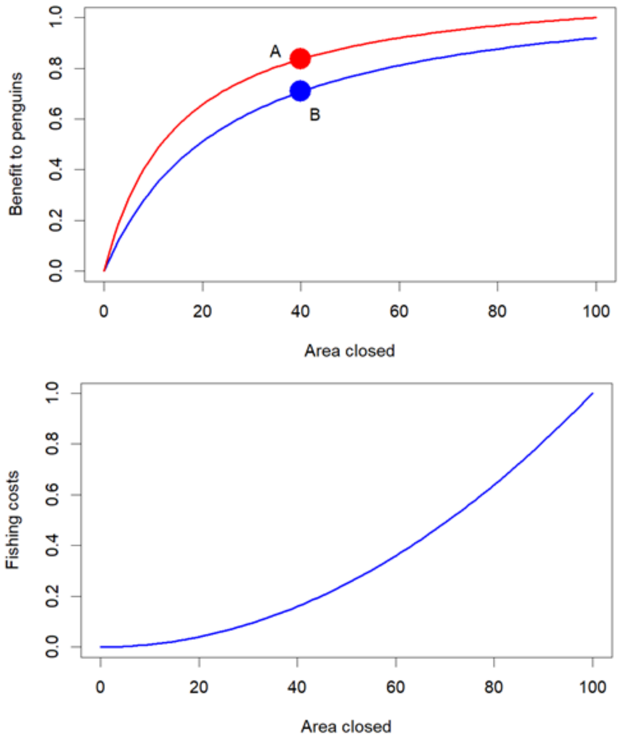

Figure 4.3: Illustrative relationships between benefits to penguins for optimally selected and simple closures given the amount of area closed (upper panel) and between area closed and fishing costs (lower panel). See text for explanations of curves A and B.

- The economic analysis (e.g. Urban-Econ, 2023a,b,c) provides estimates of several types of economic impacts (to the fishery as a direct consequence of the reduction in revenue [direct impacts], that occur due to suppliers of goods and services to the industry [indirect impacts], as well as due to shifts in spending on goods and services due to directly and indirectly impacted parties [induced impacts]), as well as lost jobs. However, the estimates of economic effects to the fishing industry may be more robust than estimates for the rest of the economy and for jobs (see section 3.3).
- Given that the OBM analysis likely provides an overestimate of uncertain magnitude of the loss in catch (see section 3.2) and these losses are then used in the SAM analysis, the results on economic costs (lower GDP, jobs) and lost catches should be considered in a relative sense and hence used for ranking closure options within a region. The relative ranking of the closure may, however, be sensitive to how catches are allocated to local communities (see section 3.4 for additional details). The eco-

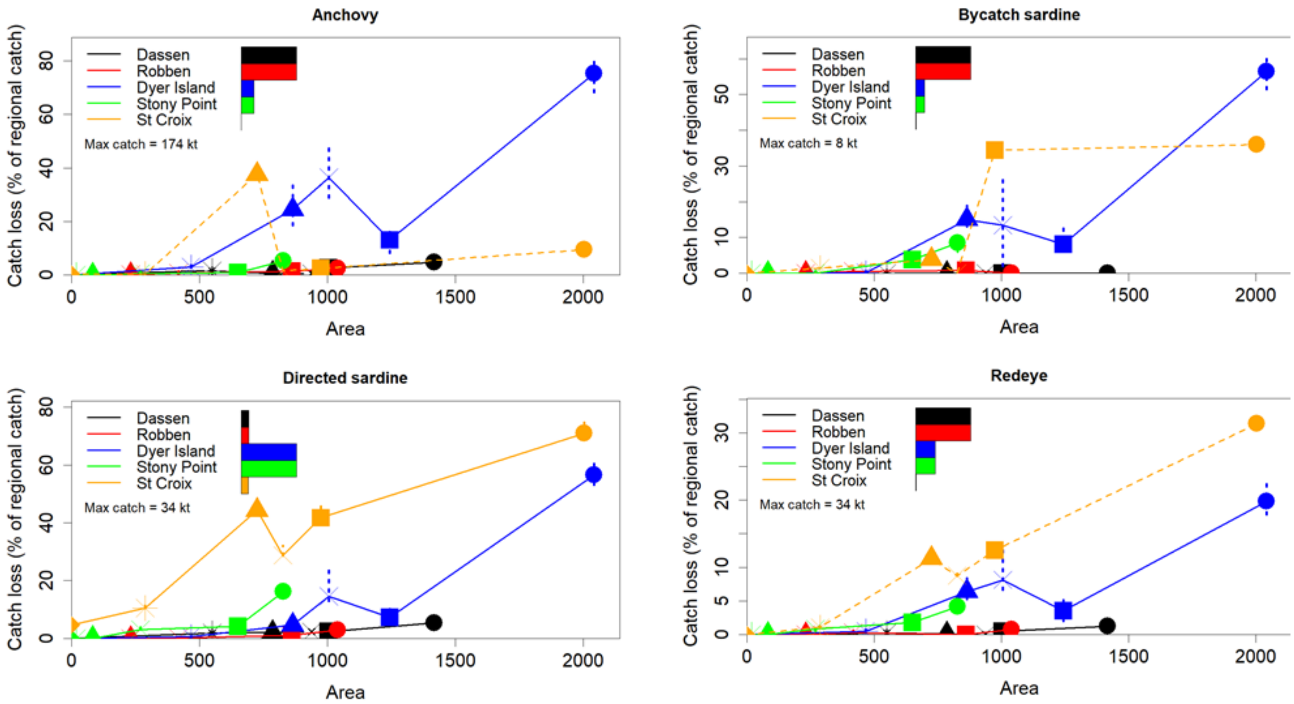

Figure 4.4: Area closed versus loss in catch for five of the six island breeding colonies. Catch losses are expressed relative to the average regional caches during 2011–2020 (west of Cape Point for Dassen and Robben islands; Cape Point to Agulhas for Dyer Island and Stoney Point; east of 24°E for St Croix). The dashed lines indicate results for island breeding colonies with very low catches relative to those for the other island breeding colonies. The different spatial closures considered for each colony are ranked by size on the x-axis: UD90 (closed circle), mIBA (ARS) (closed squares), 20 km (triangle), DFFE (cross), CAF (star), and industry (diamond). The vertical dashed lines cover the range of catch losses computed from the OBM when an alternative set can only be used once or used an infinite number of times. The symbol corresponds to using alternative sets up to five times. The length of the horizontal bars in the legend is proportional to the regional catch

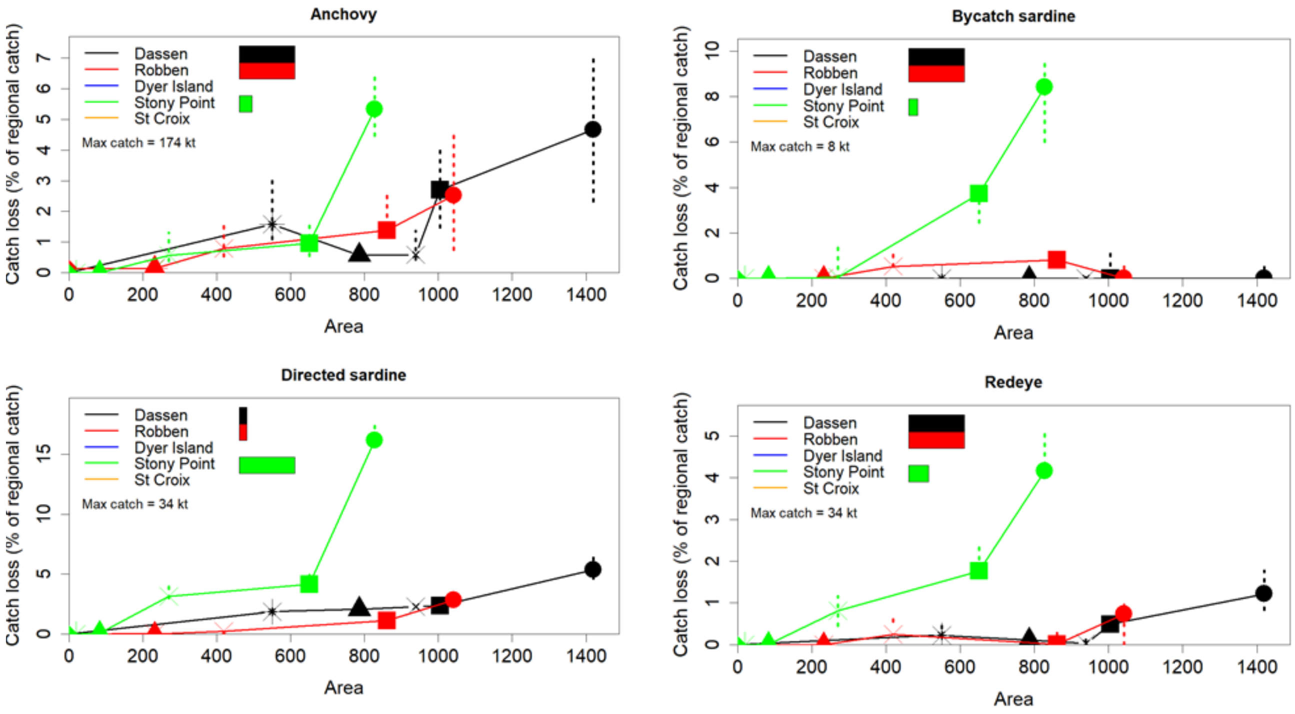

Figure 4.5: As for Figure 4.4, but restricted to Dassen and Robben islands and Stony Point.

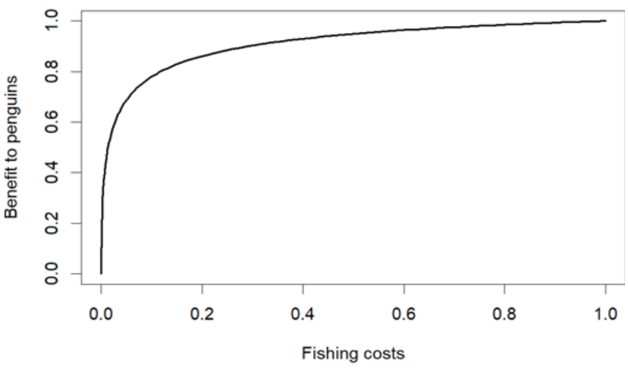

Figure 4.6: Illustrative relationship between the benefit to penguins and fishing costs based on Figure 4.3

nomic analyses are only able to quantify the social effects of closures in terms of job losses. Future work should consider broader social consequences of reduced catches and job losses on community well-being.

- It is necessary to map catch losses back into regional communities to evaluate how vulnerable these communities are because the SAM could be obscuring important local socioeconomic effects.
- The competition among the fishery and penguins would be expected to be greater in years of low prey abundance. An adaptive closure framework that changes closures among years in response to prey abundance could reduce cost to the fishery in years of high prey abundance, as closures in such years would have little or no benefit to penguins.

4.5 Colony-specific considerations

Based on the information provided to the Panel and the results from Figures 4.4 and 4.5, the Panel highlighted the different dimensions of the trade-offs in summary bullets. Across all of the regions, the various penguin foraging areas are important for the small pelagic purse seine fishery.

Dassen Island

- Regionally important for anchovy fishers and redeye fishers.
- Historically important penguin breeding habitat with sufficient habitat for growth; largest remaining breeding population.
- Relatively more susceptible because African penguins are already affected by an overall reduction in regional sardine abundance that, if persistent, may limit their capacity to reverse the declining trend.

Robben Island

- Regionally important for anchovy fishers and redeye fishers.
- Important penguin breeding population.
- Relatively more susceptible because African penguins are already affected by an overall reduction in regional sardine abundance that, if persistent, may limit their capacity to reverse the declining trend.
- Eradication of feral cats should be part of a local conservation management plan.
- Major hub for ecotourism.

Dyer Island

- Regionally important for anchovy fishers, sardine fishers and redeye fishers.
- Important penguin breeding population.
- Relatively important fur seal interactions (predation and/or resource competition) with penguins.
- Figure 4.4 indicates that anchovy catches from within a closure are difficult to replace.

Stony Point

- Regionally important for anchovy fishers, sardine fishers and redeye fishers.
- Important mainland penguin breeding population with logistical access to enhance conservation management.
- Population has increased by 15% pa since 2005.
- Major hub for ecotourism.

St Croix Island

- Fishers rely on sardine due to virtual absence of redeye and anchovy.
- Important penguin breeding population.
- Largest rate of decline since 2016 among the extant penguin colonies.
- Evidence that noise disturbance from bunkering facility is disturbing penguin foraging.
- Figure 4.4 indicates that sardine catches from within a closure are difficult to replace.

Bird Island

- Very little small pelagic fishing.
- Important penguin breeding population but limited scope for major increases.

Boulders Beach

- Fully protected from commercial fishing.
- Important mainland penguin breeding population with logistical access to enhance conservation management.
- Population is healthy and stable (891 breeding pairs in 2022).
- Major hub for ecotourism.

5. FUTURE MONITORING TO EVALUATE EFFECTIVENESS

The Panel makes the following recommendations in relation to potential scientific research questions related to the African penguin population decline, including associated monitoring techniques:

1. Continue to conduct counts of breeding numbers of African penguins at as many colonies as possible in as many years as possible.
2. Monitor adult survival of African penguins using techniques such as passive integrated transponder (PIT) tags and readers at colonies where this is practical to minimise disturbance to colonies. A comparison of time-series of adult survival at different colonies would help resolve which drivers are having the strongest influence on population change. Use of linear ground antennae is feasible when extensive areas of beach need to be monitored for PIT tags; elsewhere antennae can be incorporated into weighbridges where these are in use.
3. Continue monitoring of breeding success where it can be done without disturbance; however, the Panel considers that metrics such as chick weight/body condition/growth rate represent weak proxies of breeding success and may not be cost-effective.
4. Use automatic weighbridges to monitor weights of adult penguins at the start and end of breeding, as this should provide a direct measure of the costs of breeding in terms of the impact on penguin body condition.
5. Use automatic weighbridges to monitor weights of PIT-tagged adult penguins; departure body mass prior to foraging and return body mass subsequent to foraging should provide quantification of foraging efficiency, and potentially meal mass for offspring. Such work will be valuable in itself, but would be especially valuable if complemented by GPS tracking of some individuals.
6. Assess behavioural responses of foraging adult penguins using GPS tracking studies; these will likely remain limited to the period when adults have relatively small chicks. However, deployment of time-depth-recorder tags on these adults (together with GPS units) will provide much improved data on the foraging locations along the path of tracked birds.
7. Conduct foraging studies using telemetry methods, to further determine the impacts of vessel noise (including from bunkering) on foraging behaviour.

5.1 Population counts


Penguins nesting (photo BM Dyer)

African penguins are not easy to count. Breeding birds may be in burrows underground, or in nest boxes, or under bushes, although at most colonies many are visible in the open. Not all pairs breed at the same time, so synoptic counts on any particular date underestimate total breeding numbers. For large colonies, counts have generally been undertaken by teams of people walking through the colony counting occupied nest sites, mostly between February and September, but counts at other times of year are used when they are the only data available (Crawford et al., 2011; Sherley et al., 2020). Because breeding is not fully synchronous, potential sites (apparently not active but showing signs of use) may be included in counts, whilst numbers of unguarded chicks in groups (créches) are divided by two to estimate the (minimum) number of nest sites those birds represent (Sherley et al., 2020). These counts provide relatively low accuracy population estimates but are adequate to demonstrate large changes in population size over time.

Some birds choose not to breed, and so numbers of nests counted at colonies may underestimate the total population, by missing nonbreeding adults, especially when seabirds are under severe pressure (e.g., resource constraints, adverse weather conditions, disturbance). In addition, seabirds tend to become more vulnerable to impacts of human disturbance when already under stress from adverse environmental conditions (Diaz et al., 2021). African penguins are particularly susceptible to human disturbance (Hockey and Hallinan, 1981). Seabirds that would tolerate human activity at a colony when conditions are good may abandon their breeding attempt as a result of a similar level of human disturbance when they are stressed. It is therefore highly desirable to avoid human disturbance at penguin colonies, but especially at those that are in decline and subject to adverse environmental pressures. Use of a drone (unoccupied aerial vehicle; UAV) to overfly a colony and record digital video (or frequent static images that can be mosaiced together) of the breeding sites may allow counts without associated human disturbance, as breeding seabirds show little or no response to an overflying drone providing it is well above the colony[1]. Using drones to count breeding penguins of various species

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[1] Rümmler et al. (2021) found no behavioural reactions of penguin adults or chicks to drones flown more than 70 m above the colony. Recognising that monitoring numbers and breeding success of Sandwich terns *Sterna sandvicensis* by visiting colonies tends to cause excessive disturbance, Spaans et al. (2018) tested the use of a drone, flown 15-–20 m above nesting Sandwich terns at appropriate dates through the breeding season at colonies in the Netherlands, to count breeding numbers and breeding success from photographs. They found that the drone caused “hardly any visible disturbance to the birds” but gave highly accurate data on breeding numbers and breeding success, so was considered much better than using human observations at Sandwich tern colonies. The same conclusion was reached by Valle and Scarton (2021) in Italy. Geldart et al. (2022) showed that drones flying over nesting eider ducks *Somateria mollissima* did not lead to any increase in heart rate of the incubating birds.

has been shown to be highly effective, for some penguin species and in some cases more accurate than human counts, as well as reducing human disturbance (Hayes et al., 2021; Krause et al., 2021; Mattern et al., 2021; Qian et al., 2023).

Because some African penguins nest in locations where they cannot easily be seen or detected from above ground, a complementary approach to census African penguins may be to use drone counts of crèched chicks, or moulting penguin numbers. These are easier to count than breeding birds, as they tend to moult relatively synchronously and in the open, although sometimes these may include small numbers of birds breeding elsewhere. For African penguins, preliminary studies could help determine the efficacy of such techniques.

5.2 Breeding success

Breeding success is an important metric to monitor because it is likely to have a clear influence on population trend and is hence usually a high priority in any seabird monitoring programme. However, this is less straightforward with seabirds that prefer to nest in burrows but may also use open nest sites on the surface. There are likely to be differences in breeding success between nests of different types in different habitats, and this needs to be considered when setting up a monitoring programme. It would be ideal to monitor samples of nests of each type so that annual breeding success can be representative of the colony rather than of just one nest type. Breeding success can be monitored remotely using equipment such as nest cameras or acoustic monitoring, which has the potential to minimise disturbance impacts from people having to visit nests to monitor breeding. Examples of time time-lapse photography are now increasingly common in penguin behavioural studies (e.g., Jones et al., 2018).

5.3 Adult survival

There is evidence that survival of adult African penguins is strongly affected by sardine stock biomass (Robinson et al., 2015; Crawford et al., 2022; Leith et al., 2022), but apparently not to anchovy stock biomass, at least for Robben Island. There is therefore a strong case for increased monitoring of African penguin adult survival, as this is likely to be a major factor determining population trend. Marking of some penguins species with external tags (e.g., flipper bands) has been shown to have adverse effects, so future monitoring of penguin survival should focus on the use of PIT tags and deployment of tag readers at colonies to allow monitoring of adult survival with minimal human disturbance and with tags that do not affect penguin fitness. PIT tag deployments have already been made for African penguins at Robben Island and at Stony Point (Leith et al., 2022). The presence of tagged birds at nests can be determined using a hand-held tag reader carried from nest to nest, but this risks impacts from human disturbance. An alternative is to deploy tag readers at strategic locations within the colony to identify birds as they pass within range of the reader. Both approaches risk missing tagged individuals if readers are not close to particular birds, so provide incomplete assessments of adult survival. In addition, mobile robotic tag readers have also been developed, as well as linear beach antennae, both of which may be feasible to use with African penguins (Trathan and Emmerson, 2014). Experimentation with different approaches will help determine approaches appropriate to African penguins.

5.4 Weigh bridge and PIT tags

It has been possible to set up a narrow “entrance” to the nesting area at some penguin colonies so that when adults approach nests it is possible to monitor each individual’s arrival and departure. This can be achieved with (PIT) tags and Radio Frequency Identification (RFID) tag readers at the entrance to colonies (Kerry et al., 1993; Denhard et al., 2013). Tag deployments can potentially be combined with a weigh bridges used to weigh birds as they arrive and depart (Lescroël et al., 2021) providing data on changes in the weight of known individuals before and after each foraging trip. However, there can be problems associated with such automatic monitoring stations, where, for example, individual penguins use different routes to enter and exit the colony. In such cases, care will be needed to ensure sample sizes are adequate to address key research objectives. Further. there remains the possibility that constrained access to the nesting area could have impacts on the breeding birds, but careful design should be able to avoid such problems.

5.5 Arrival weights of adults

Weights of individual penguins departing from and returning to the colony passing over a weigh bridge can provide data giving evidence on foraging efficiency during individual foraging trips (Lescroël et al., 2021) that could be related to food abundance/availability and other factors (such as noise, vessel traffic, weather conditions, fishing activity). Monitoring of foraging efficiency could be highly informative if such sites can be established.

5.6 Pre-moult weights

Penguins are unusual among birds in having an intense pre-moult fattening period to store resources (energy, protein and perhaps especially sulphur amino acids) to support the process of moult. Unlike most birds that moult slowly while continuing normal daily activities, penguins remain on land through a short period of starvation while a complete moult occurs. During this process they are unable

Moulting penguin chick (photo BM Dyer)

Penguin crèche (photo BM Dyer)

to return to sea because their waterproofing is compromised by the moult process until it has been completed. These birds therefore need a minimum stored amount of resource to successfully complete moult. Weights of penguins at the start of moult may indicate whether environmental conditions have allowed birds to achieve that minimum. Increased adult mortality may in part reflect an inability to achieve the key body reserves needed for moult.

5.7 Chick growth, chick body condition, and chick fledging weights

Chick metrics may provide some indication of how good environmental conditions are for penguin breeding, but they are much less useful than data on breeding success. Chick fledging weights in some seabird species are correlated with post-fledging survival, but that is not the case in all seabirds or in all populations, so fledging weight may not always link to demography. Seabird chicks can show catch-up growth where undernourished chicks end up at a similar fledgling weight because they put on weight at a later developmental stage where other chicks have reached a plateau weight. Chick condition indices may also show rather little correlation with demography, and may be affected by selective mortality of starving chicks at some colonies and during some years. However, these indices may show little relationship with demography if the main determinant of chick survival is predation rather than starvation. Further, even poor quality adults may fledge chicks in years with good environmental conditions, whereas only high quality parents may succeed in poor environmental conditions. The potential therefore exists for inverse relationships where more poor quality chicks fledge in years of abundant resources.

5.8 Recruitment of juveniles

Use of PIT tags in penguin chicks and deployment of tag readers at breeding or moulting sites may provide data on immature survival and seasonal movements of immatures. Relatively little is known about the ecology of immature seabirds as they are much more difficult to study than breeding adults. However, because immatures are less experienced they tend to have lower foraging efficiency than breeding adults and so periods of increased competition (such as during periods of food shortage) are likely to disproportionately affect immature birds. Studies of recruitment of PIT-tagged individual juvenile penguins may therefore help to shed light on population processes driving population growth or decline.

5.9 Studies with TDRs

Time-depth-recorders (TDRs) can provide data on the foraging activity of diving seabirds. For example, deployment of TDRs in combination with PIT tags on penguins that then cross a weigh bridge as they leave the colony and again as they return from a foraging trip can give information on the amount of food obtained in relation to the number of dives made while foraging (Lescroël et al., 2021). This allows foraging efficiency and effort to be related to local environmental variables. The Panel identifies this as a high priority for future research, including for further validation of any mIBA closures designated.

5.10 GPS tracking of breeding adults and video-cam studies

GPS tracking of seabirds is normally limited to short periods during breeding, as GPS tag attachment is usually temporary and devices are removed from the tagged bird after a few days or weeks. Depending on tag design (and therefore cost and battery life) GPS tags can either be designed to store data for download from the tag on recapture of the same bird, or can transmit data to a base station or to the cellphone network or to a satellite. GPS tracking can provide important data on where individuals choose to search for food in relation to local environmental conditions (Sutton et al., 2020). There is also the potential to deploy video-cameras on adult penguins to record foraging behaviour and interactions with forage fish. Such deployments could provide useful understanding of penguin group foraging behaviour. In general, the weight and induced drag of devices (especially if more than one device is deployed on a bird) must be considered, as they could potentially affect the behaviour that is being studied.

5.11 Tracking of nonbreeding season movements of adults

It is possible to use GPS tags to track African penguins before and after the moult period (Carpenter-Kling et al., 2022). Tags remain on the birds for a matter of days or weeks during the breeding season limiting the duration of such studies. Tags would need to be attached more permanently to birds to track movements throughout the nonbreeding period. That is sometimes possible by using a harness, but harnesses are not suitable for most highly marine seabirds, especially those that dive to chase prey. Permanent attachment can be achieved by implanting tags within the bird's body cavity, but such surgical procedures risk injury and increased mortality, so may be better avoided. Long-term overwinter studies on penguins have been undertaken using light-sensing geolocators (e.g., Ballard et al., 2010; Dunn et al., 2011; Thiebot et al., 2011), but care needs to be taken in deployment, not to constrict legs (which engorge with blood) during moult. The Panel recognises that such research would be useful, but also that the concerns about potential tag effects on birds would need to

6. FUTURE RESEARCH OTHER THAN MONITORING

6.1 Refining the estimation of effects of closures on catches, GDP, and jobs

- Further statistical analysis should be undertaken to better understand the seasonal nature of anchovy and sardine sets/catches across the fishing sites, especially along the west coast.
- OBM results for the random case should be presented for the 1, 5, and infinity cases.
- The impact of closures on net revenue as well as changes in catches should be explored because it is important for understanding both the short-run impacts and the potential long-run impacts due to changes to the fleet composition, shore-side infrastructure, and coastal community dynamics.
- Further work needs to be done on the long-run socioeconomic impacts to local communities due to the prospective closures. A key part of this research would be data collection at the scale of local communities to better understand how the fishing sector (onshore and offshore) and penguin tourism contribute to the local economy, jobs, and well-being. Examples of community profiles and analysis that could be used as a guide for such an effort are Colburn et al. (2016), Himes-Cornell et al (2013), and Pollnac et al. (2006).
- Some important questions remain regarding the interpretation of the SAM results:
    - ♦ Are the estimated "losses" due to the proposed closures within the standard fluctuations of the local economy due to other kinds of economic shocks, such as fuel prices, exchange rate fluctuations, fluctuations in total stock biomass etc.?
    - ♦ Are the short-run job losses from a hypothetical fuel price increase (best to consider a range of increases from 5 to 25%) greater than the predicted job losses from the preferred scenario?
    - ♦ How important for the loss estimates are the assumptions regarding the relative wages of the processing and harvesting sector, especially since most of the job losses occur in the processing sector?
    - ♦ How do the results change if the conversion of total full-time equivalent employment is based on a different rate of fishing days per year (currently, 175 fishing days per annum is assumed)? Additional sensitivity analysis of the SAM results should be carried out to have a better understanding of the range of possible regional outcomes from the prospective closures.
- Given little empirical justification for one method, alternative methods for allocating catches to regions should be used, and the results compared across the different cases, to better inform discussions on which communities are likely to be most impacted.
- Given that SAM results should be viewed as a very short-run measure of impacts, a Computable General Equilibrium model (Seung and Waters, 2006) should be developed to capture more dynamic short-run and medium-run responses of the economy.

6.2 Supporting evaluation of trade-offs, including refining estimates of foraging areas

- Further validation of mIBAs should occur, in particular using dive data that provide objective identification of foraging locations, rather than commuting (or travelling) locations.
- Between-year variation in mIBA should be explored.

6.3 Understanding and mitigating reasons for the decline in African penguins due to factors other than fishing near breeding colonies

There is broad agreement that the recent observed decline in African penguin numbers both locally and regionally may be due to a number of factors. The ICE was designed to quantify the impact of sardine and anchovy fishing in the vicinity of penguin breeding islands, and the body of evidence presented to the Panel suggests that this is a contributing factor, but the magnitude of the impacts appears small and could only explain a small part of the recent declines in penguin numbers. Plausible drivers impacting the penguin populations are likely to vary across islands and spatial scales, plus there are variable data available to inform on different impacts, as well as the likely cumulative impacts of different drivers. Future research is needed to address each of the possible drivers. The effects of several drivers could be explored by developing an integrated ecosystem model, such as a MICE (Model of Intermediate Complexity for Ecosystem assessments) (Plagányi et al., 2014; Collie et al., 2016), or so-called MRMs (Minimum Realistic Models – Punt and Butterworth, 1995)[1].

6.3.1 Forage fish abundance

Section 1.3.2.1 summarises information related to the potential for changes in the biomass of prey species to affect population parameters, in particular the effect of sardine biomass on penguin adult survival. Further evaluation of such relationships could involve (a) the development of a new MICE that addresses all of the major penguin colonies off South Africa, and (b) exploration of the consequences of using the current OMP to set catch limits for anchovy, sardine and round herring. The latter exploration may lead to different results than those found by Robinson et al. (2015), given the current (more depleted) status of the sardine population and an OMP that leads to constant catch limits over ranges of low sardine biomass, and spatial constraints. The Panel notes that the current OMP should be tested to evaluate whether it is adequately precautionary in relation to protecting future recruitment prospects of sardine, as it currently allows high exploitation rates when sardine stock

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[1] See Appendix F for details

falls to levels where future recruitment may be impaired. This suggests that further consideration should be given to the role of fishing pressure on sardine stock dynamics.

6.3.2 Guano harvests

Past guano harvesting is recognised as an important possible contributory cause to the penguin decline because of its impact on optimal breeding habitat (see section 1.3.2.2). The impact of reductions in guano as nesting habitat is confounded to some extent with other changes in the system, but could be incorporated in a MICE, expanding on local efforts currently underway.

6.3.3. Resource competition with Cape fur seals

The decline of the penguin population may be related to competition with predators that depend upon small pelagic fish. For example, Cape fur seal populations have increased substantially over the previous century and have expanded into areas used by penguins (see section 1.3.2.3). This is an impact that could usefully be investigated using a MICE both in terms of direct and indirect predation effects, but also to compare the responses of other predators in the system to changes in pelagic fish abundance. Though known to occur, the incidence of predation of penguins by Cape fur seals, is unlikely to have led to the penguin population changes observed. Data on seal diet and changes in regional seal abundance would be particularly informative as inputs to models to quantify the relative contribution of seal predation (and possibly competition) to penguin mortality.

6.3.4 Noise in the marine environment

Disturbance of penguin group foraging, unrelated to any prey depletion effects, could possibly occur if groups of penguins are disturbed or displaced by fishing vessels, or noise associated with bunkering near St Croix Island (Pichegru et al., 2022), especially if their group coordination and communication while hunting is affected by the noise. Continued investigation of the effects of marine noise could involve, for example, using tracking and deployment of TDR tags to understand the changes in foraging behaviour and distribution in response to bunkering noise. Currently, including such investigations in a MICE would not be feasible.

6.3.5 Nest boxes

Although there is evidence that African penguin breeding success can be increased by providing nest boxes (section 1.3.2.5), the ideal design for such nest boxes has not been agreed by all those involved. Nevertheless, widespread gains in penguin productivity might be possible in some areas if a better design were to be found and nest boxes deployed in large numbers at the main colony sites. If deployed at such scales, the cost (including annual maintenance) of individual nest boxes would be an important consideration. Currently, including such investigations in a MICE would not be feasible.

6.3.6 Climate change

Climate change is recognised as a factor impacting seabirds in South Africa (Crawford et al., 2015), including penguins, both directly, such as impacts due to extreme events (Welman and Pichegru, 2022) and indirectly, given potential influence on the recruitment patterns and spatial distribution of anchovy and sardine in the vicinity of penguin colonies (see van der Lingen, 2023 for details). Sea surface temperature (SST) predictions of future increases (or decreases in localised areas) will variably influence different regions and hence penguin colonies. As such, the Panel highlights the need for penguin management strategies (and monitoring) that encompass multiple spatial regions to increase resilience to climate change and fish distribution changes (McInnes et al. 2023).

Given recognition of the impact on African penguins of a continued eastward shift (i.e., from the west to the south coast) in the distribution of anchovy and especially sardine (van der Lingen, 2023), this is an important factor to include in a MICE. Although it may not be possible to precisely model the exact rates of fish movement, available fishery and survey data and/or stock assessment outputs could be used to reasonably represent a restricted number of alternative scenarios to explore the impact on penguin colonies. In particular, attention needs to be paid to the potentially highly influential relationship between adult survival and sardine availability (Robinson et al., 2015; Leith et al., 2022). A MICE should ideally use and fit to all available penguin survival data. By explicitly representing the ages of tagged penguins as well as other confounding sources of mortality, such as due to oiling events and predation, an integrated MICE could assist in separating the alternative sources of mortality. This then provides an objective integrated framework for quantifying and correctly attributing the relative role of different drivers in causing the decline of the penguins. Given an improved understanding – validated to the extent possible - of the relative contributions of each driver to the penguin decline, a MICE is then a useful tool for testing the efficacy of alternative management strategies through forward projecting the effect of future mitigation measures, either on their own or in combination.

The available penguin and fishery data suggest that a pragmatic starting point is to model regional changes in penguin population dynamics due to changes in prey composition and availability. The next step could be to add to the model available environmental and climate data (such as SST, frequency of extreme events), preferably aligned with penguin monitoring data, to explore to what extent spatio-temporal changes in the environment may be contributing to the decline in penguins. Given differences in habitat and climate resilience across colonies, a spatial model structure would be informative in trying to distinguish a reliable signal from the data.

7. SUMMARY AND CONCLUSIONS AND RECOMMENDATIONS

The following sections summarise the key conclusions and recommendations. Table 7.1 provides a prioritised summary of research and other tasks.

7.1 Design, implementation and interpretation of the ICE

- The ICE has been identified as an example of a best practice for assessing forage fish fisheries – seabird resource competition, but the weaknesses of the design and implementation need to be recognised and their consequences accounted for when interpreting the results (section 2.4).
- The debate about the relative merits of analyses based on aggregated versus disaggregated data was essentially closed based on the final set of results presented at the June 2023 meeting. Although differences in preferences between the analysts remain, the two approaches provide similar results when appropriately configured (section 2.2.1).
- The response variables monitored as part of the ICE were considered to be direct measures or proxies for African penguin breeding success or post-fledging survival, but did not measure impacts of island closures on African penguin adult survival or immature survival. The Panel interpreted the estimated impacts of fishing on foraging-related parameters only qualitatively and did not integrate them into the inferences regarding overall impacts on penguin population growth rates (section 2.2.2). Only the predictions for Dassen and Robben islands are discussed in detail given the concerns regarding the use of foraging-related variables (see section 2.2.1) and the fact that only estimates based on chick condition are available for St Croix and Bird islands (section 2.3.2).
- Overall, the results of the ICE for Dassen and Robben islands indicate that fishing closures around the breeding colonies are likely to have a positive impact on population growth rates, but that the impacts may be small, in the range 0.71–-1.51 % (expressed in units of annual population growth rate). These impacts are small relative to the estimated rates of reduction in penguin abundance for these two colonies over recent years (section 2.3.2).
- The change in population growth rate estimated in Section 2.3 did not include impacts of island closures on African penguin adult survival or immature survival, which are likely to exist based on evidence for other situations, but cannot be quantified for African penguins (section 2.4).
- The ICE is completed. Future closures of forage-fish fishing around penguin colonies would be likely to benefit penguin conservation, but will need to be part of a larger package of conservation measures as such closures alone would be unlikely to reverse the current decline in penguin population numbers (section 2.3.2).

Photo credit SAPFIA – South African Pelagic Fishing Industry Association

7.2 Calculating the costs to the fishery associated with closures

- Implementing closures will impact the fishing industry and local communities to some extent, but accurately quantifying this is challenging (section 3.1).
- The OBM and SAM are appropriate methods for estimating costs to the fishery but their results should be considered primarily in a relative sense (section 4.4) and as measures of short-run impacts.
- The OBM quantifies the impacts of closures under the assumption that catches that occurred in the closed area when it was open are a measure of the catches that would have occurred if the closed area was not closed (section 3.2).
- The OBM likely overestimates the loss in catches due to closures, to an unquantified extent, given its assumptions related to the set of opportunities that are available to replace catches in closures, particularly those considered “irreplaceable” because all of the catch on a given day occurred inside a closure (section 3.2; Appendix E).
- Understanding the impact of closures on the net revenue as well as changes in catches is important for understanding both the short-run impacts and the potential long-run impacts due to changes to the fleet composition, shore-side infrastructure, and coastal community dynamics (section 3.2).
- The predicted impacts of closures depend on the reliability of the estimates of lost catch from the OBM, which the Panel agreed is likely to provide overestimates (section 3.3).
- Because SAMs are designed to analyse demand-driven impacts in the local economy (e.g., change in consumer spending), these models tend to overestimate the impacts of supply-side shocks, such as a reduction of catch (section 3.3).

Table 7.1: Prioritised summary of research and other tasks. Short-term tasks pertain to the next 1-2 years, medium-term tasks to the next 2-5 years and long-term tasks the next 6+ years. The relative priorities and timings reflect an integrated outcome of the Panel, which assigned priorities and timings to each task.

Task	Relative priority	Timing
1. Refining the estimation of effects of closures on catches, GDP, and jobs		
a. Explore the seasonal nature of anchovy and sardine sets/catches (West Coast)	Medium	Medium
b. Present OBM results for the 1, 5, infinity cases	High	Short
c. Investigate the impact of closures on net revenue	Medium	Medium
d. Analyse the long-run socioeconomic impacts	Medium	Medium
e. Conduct an in-depth interpretation of the SAM results	High	Short
f. Conduct SAM sensitivity analysis - regional outcomes	High	Short
g. Explore SAM sensitivity to allocation of catches to regions	High	Short
h. Develop a Computable General Equilibrium model	Low	Long
2. Supporting evaluation of trade-offs, including refining estimates of foraging areas		
a. Validate the mIBAs given information on foraging locations	High	Medium
b. Summarise between-year variation in mIBAs	Medium-High	Short
3. Understanding and mitigating reasons for the decline in African penguins due to factors other than fishing near breeding colonies		
a. Develop a MICE/integrated ecosystem model	High	Medium
b. Test that the current OMP is adequately precautionary at low sardine biomass for penguin conservation	High	Medium
c. Collate and collect data on changes in seal diet and regional abundance	Medium-Low	Medium-Long
d. Conduct tracking and deployment of TDR tags to understand the changes in foraging behaviour and distribution in response to bunkering noise	High	Short
e. Optimise nest box design and deployment	Medium	Medium
f. Conduct analyses related to climate change impacts and the variable role of SST on different regions/ penguin colonies	Medium	Medium-Long
g. Further explore the relationship between adult survival and sardine availability (e.g., tagging data preferably matched to estimates of regional sardine abundance)	High	Medium
4. Future monitoring to evaluate effectiveness		
a. Continue counts of breeding numbers at as many colonies as possible	High	Short-Long
b. Monitor adult survival of penguins using low disturbance methods such as PIT tags and readers	High	Short-Long
c. Continue to monitor breeding success	High	Short-Long
d. Use automatic weighbridges to monitor weights of adult penguins	Medium	Medium
e. Use weighbridges to monitor weights of PIT tagged adults + GPS tracking	High	Medium
f. Deploy time-depth-recorder tags (together with GPS units, accelerometers, or video recorders)	Medium	Medium
g. Apply telemetry methods, to examine impacts of vessel noise (including from bunkering)	High	Short-Medium
h. Use drones for monitoring	Low	Medium
j. Use PIT-tagging of juvenile penguins to understand survival	High	Short-Long
k. Conduct video-cam studies of adult group foraging behaviour	Low	Medium
5. Improving communication and collaboration		
a. Improve processes and platforms for sharing data	High	Short-Long
b. Conduct collaborative workshops to share information, jointly discuss compromises and seek solutions	High	Short-Long

Photo credit SAPFIA – South African Pelagic Fishing Industry Association

7.3 Issues pertinent to evaluating trade-offs

- There are three primary trade-off axes to consider when selecting closures: (a) the benefit to penguins of the closure; (b) the cost (economic and social) to the fishing industry and the communities where fishing and processing operations are based; and (c) the ability to evaluate the effectiveness of the closures (section 4.1).
- Closed areas to protect penguins during breeding should be year-round, unless reasons demonstrate otherwise (section 4.1).
- If designated, closed areas to protect penguins should be reviewed at a time when results are available to investigate life-history processes such as juvenile recruitment, and adult survival, and hence population growth rates. This may be at a time between 6 and 10 years after designation. Other reasons to review such closed areas might include major socioeconomic changes in the fishery and processing, or stock abundance, or similar consequences of prey resource change (section 4.1).
- Analyses needed to determine juvenile recruitment, and survival, and adult survival, will require closures of between 6 and 10 years after closure designation, if adequate responses are to be determined (section 4.1).
- Monitoring should take place irrespective of whether there is an experimental (alternating open and closed) component to the closure program (section 4.2).
- If an experimental component is to be part of any closure regime: (a) it should be focused on parameters such as juvenile recruitment and survival, and adult survival in addition to those related to breeding success monitored during the ICE; (b) the western and southern Cape regions should be the focus of any future experimental closure program given data availability and the ability to undertake regular monitoring; and (c) it is desirable that a power analysis be conducted to identify an appropriate sequence of (possibly alternating open and closed) closures (section 4.2).
- Penguin foraging areas should be quantified for trade-off analyses delineating mIBAs using ARS methods (section 4.3).
- Conservation actions should be spread throughout the range of the species given that each region is subject to different biophysical and anthropocentric threats (section 4.4).
- The following considerations are relevant to designing a framework to help decision makers select closed areas (if any):
    - ♦ An optimal solution (or acceptable “balance”) between competing objectives is not simply obtained by closing 50 percent of any given area.
    - ♦ One approach is to find the point at which the change in benefits to penguins (by increasing closures) matches the change in costs.
    - ♦ The trade-offs between costs to the fishery and benefits to penguins in terms of the size of an area closed will differ among islands and among sectors within the fishery. Consequently, the benefits to penguins and costs to industry should be considered by island (or region) and not simply at the national level (see section 4.5 for aspects of each major breeding colony that are relevant for decision making). In addition, given the heterogeneity within the industry, expressing costs and job losses by sector (e.g., for small scale operators) would also seem appropriate.
    - ♦ Care should be taken when interpreting the estimated impacts to the fishing industry given the OBM likely provides an overestimate of uncertain magnitude of the loss in catch (see Section 3.2) so the results of the OBM and hence the SAM model should be considered primarily in a relative sense and hence used for ranking closure options. The relative ranking of a closure may, however, be sensitive to how catches are allocated to local communities.
    - ♦ The economic analyses are only able to quantify the social effects of closures in terms of job losses, and future work should consider broader social consequences of reduced catches, such as measures of community well-being.
- The OBM indicates that the ability to replace catches currently taken in penguin foraging areas, and in turn the impacts of closures on the fishing industry, differs among colonies (most difficult for Dyer Island and St Croix Island) (Figures 4.4 and 4.5).
- The likely effectiveness of closures for mitigating the decline in penguin abundance also differs among colonies given their variable rates of declines (largest declines in St Croix Island) and the presence of other factors unrelated to fishing contributing to those declines (e.g., bunkering close to St Croix Island) (section 4.5).
- It is possible to design closures within the overall foraging area to minimise lost catch for any given choice of percentage of penguin foraging area to be protected (Figures 4.4 and 4.5).

7.4 Monitoring and research to determine causes for the primary reasons for the decline

Section 5 provides details on potential scientific research questions related to the African penguin population decline, including associated monitoring techniques. Key tasks are:

1. Continue to conduct counts of breeding numbers of African penguins at as many colonies as possible in as many years as possible.
2. Monitor adult survival of African penguins. A comparison of time-series of adult survival at different colonies would help resolve which drivers are having the strongest influence on population change. In order to minimise disturbance to colonies; monitoring should use techniques such as PIT tags and readers at colonies where this is practical. Use of linear ground antennae are feasible when extensive areas of beach need to be monitored for PIT tags; elsewhere antennae can be incorporated into weighbridges where these are in use.
3. Continue monitoring of breeding success where it can be done without disturbance; however, the Panel considers that metrics such as chick weight/body condition/growth rate represent weak proxies of breeding success and may not be cost-effective.
4. Use automatic weighbridges to monitor weights of adult penguins at the start and end of breeding, as this should provide a direct measure of the costs of breeding in terms of the impact on penguin body condition.
5. Use automatic weighbridges to monitor weights of PIT-tagged adult penguins; departure body mass prior to foraging and return body mass subsequent to foraging should provide quantification of foraging efficiency, and potentially meal mass for offspring. Such work will be valuable in itself, but would be especially valuable if complemented by GPS tracking of some individuals.
6. Assess behavioural responses of foraging adult penguins using GPS tracking studies; these will likely remain limited to the period when adults have relatively small chicks. However, deployment of TDR tags on these adults (together with GPS units) would provide much improved data on the foraging locations along the path of tracked birds.
7. Conduct foraging studies using telemetry methods, to further determine the impacts of vessel noise (including from bunkering) on foraging behaviour.

7.5 Future research

Sections 1, 4 and 6 summarise hypotheses related to aspects other than fishing near island breeding colonies leading to resource competition, that could explain past and ongoing declines in African penguin populations. Section 6 identifies data sources and analysis methods (including the use of Models of Intermediate Complexity for Ecosystem Assessment – MICE) that could assist in understanding the effect of these aspects and how they can be mitigated.

Section 6.3.1 offers further information related to the potential for changes in the biomass of prey species to affect African penguin population parameters, in particular exploration of the consequences of using the current OMP to set catch limits for anchovy, sardine and round herring. The latter exploration may lead to different results than those found by Robinson et al. (2015), given the current (more depleted) status of the sardine population and an OMP that leads to constant catch limits over ranges of low sardine biomass, and spatial constraints.

7.6. Other

If designated, closed areas to protect penguins should be reviewed at a time when results are available to investigate life-history processes such as juvenile recruitment, and adult survival, and hence population growth rates. This may be at a time between 6 and 10 years after designation. Other reasons to review such closed areas might include major socioeconomic changes in the fishery and processing, or stock abundance, or changes in estimates of core foraging areas, for example, due to mIBAs being based on where foraging occurs and not entire tracks, or similar consequences of prey resource change (section 4.1).

7.7 Communication and collaboration

Continued communication, collaboration, and transparency of research data and analyses, are strongly encouraged to build trust and strengthen progress towards seeking acceptable solutions. Working collaboratively will further enhance the effectiveness and social acceptability of management measures and decisions aimed at mitigating the decline of the African penguin.

Clear, fair and objective communication around this controversial issue is important to ensure the best possible outcomes for penguins whilst respecting that conservation decisions may impact to varying extents on livelihoods and community well-being.

Penguin colony, Bird Island, Algoa Bay (photo BM Dyer)

8. REFERENCES

8.1 Papers and reports

Abadi, F., Barbraud, C., and Gimenez, O. 2017. Integrated population modeling reveals the impact of climate on the survival of juvenile emperor penguins. *Global Change Biology* 23: 1353-–1359.

Abbott, J.K., and Wilen, J.E. 2009. Regulation of Fisheries bycatch with common-pool output quotas. *Journal of Environmental Economics and Management* 57: 195–204.

Abbott, J.K., and Wilen, J.E. 2011. Dissecting the tragedy: A spatial model of behavior in the commons. *Journal of Environmental Economics and Managemen*t 62: 386–401.

Ainley, D.G. 2002. The Adélie Penguin: Bellwether of Climate Change. Columbia University Press, New York.

Anon. 2010.The Biodiversity Management Plan for the African Penguin (*Spheniscus demersus*). DFFE report.

Bai, X., Gao, L., and Choi, S. 2022. Exploring the response of the Japanese sardine (*Sardinops melanostictus*) stock-recruitment relationship to environmental changes under different structural models. Fishes 7: 276.

Ballard G., Toniolo V., Ainley D.G., Parkinson C.L., Arrigo K.R., and Trathan P.N. 2010. Responding to climate change: Adélie Penguins confront astronomical and ocean boundaries. *Ecology* 91: 2056–2069.

Balmelli, M., and Wickens, P.A. 1994. Estimates of daily ration for the South African (Cape) fur seal. *South African Journal of Marine Science* 14: 151–157.

Bell I.R., Possingham H.P., and Watts M. 2009, . Marxan and relatives: software for spatial conservation prioritisation. In: Moilanen A, Wilson KA, Possingham HP (eds) *Spatial conservation prioritisation: quantitative methods and computational tools*. Oxford University Press, Oxford, pp 185–195.

Bergh, M. Lallemand, P., Donaldson, T., and Leach, K. 2016. The economic impact of the west coast penguin island closures on the pelagic fishing industry. DEFF Fisheries document: FISHERIES/2016/JUN/SWG-PEL/18

Bergh, M.O. 2022. Estimates of job losses versus additional penguin pairs from island closures. DEFF Fisheries document: FISHERIES/2022/OCT/SWG-PEL/33.

Bergh, M.O. 2023. Comments on additional documents and presentations submitted for panel deliberations in June 2023. Document FP/PANEL/WP/19 presented to the Panel in June 2023.

Bergh, M., and Horton, M. 2023. Estimates of the impact of closing fishing around six penguin breeding sites on pelagic catches. Document FP/PANEL/WP/01 presented to the Panel in June 2023.

Butterworth, D.S. 2021. A proposed structured framework for providing scientific advice on possible responses to the decline in the numbers of African penguins. DEFF Fisheries document: FISHERIES/2021/MAR/SWG-PEL/12.

Butterworth, D.S., and Ross-Gillespie, A. 2021a. A revised summary of results for the island closure experiment. DEFF Fisheries document: FISHERIES/2021/JUN/SWG-PEL/41.

Butterworth, D.S., and Ross-Gillespie, A. 2021b. A response to some queries concerning the revised summary of results for the island closure experiment provided in FISHERIES/2021/JUN/SWG-PEL/41.DEFF Fisheries document: FISHERIES/2021/SEP/SWG-PEL/59.

Butterworth, D.S., and Ross-Gillespie, A. 2022. Comment on “South Africa’s experimental fisheries closures and recovery of the endangered African penguin” by Sydeman et al. (2021). I*CES Journal of Marine Science* 79: 1965–1971.

Butterworth, D.S., Plagányi, E.E., Robinson, W.M.L., Moosa, N., and de Moor, C.L. 2015. Penguin modelling approach queried. *Ecological Modelling* 316: 78–80.

CAF (Consultative Advisory Forum) 2022. Special Project Report on Penguin and Small Pelagic Fishery Interactions by the Consultative Advisory Forum for Marine Living Resources.

Cairns, D.K. 1992. Population regulation of seabird colonies. *Current Ornithology* 9: 37–61.

Campbell, K.J., Steinfurth, A., Underhill, L.G., Coetzee, J.C., Dyer, B.M., Ludynia, K. , Makhado, A.B., Merkle, D., Rademan, J., Upfold, L., and Sherley, R.B.. 2019. Local forage fish abundance influences foraging effort and offspring condition in an endangered marine predator. *Journal of Applied Ecology* 56: 1–10.

Carpenter-Kling, T., de Blocq, A., Hagen, C., Harding, C., Morris, T., Pichegru, L., Roberts, J., Ryan, P.G., Wanless, R.M., and McInnes, A. 2022. Important marine areas for endangered African penguins before and after the crucial stage of moulting. *Scientific Reports* 12: 9489.

Coetzee, J. 2023. Information on small pelagic purse-seine catches taken within the 20 km radius closure areas around penguin breeding colonies during the Island Closure experiment. DFFE document developed in response to a panel request following the March 2023 meeting. 9pp.

Coetzee, J.C., de Moor, C.L., van der Lingen, C.D., and Butterworth D.S. 2022. A summary of the South African sardine (and anchovy) fishery. MARAM Document MARAM/IWS/2022/Sardine/BG1.

Coetzee, J., Kock, A., Lawrence, C., Makhado, A., Masotla, M., Oosthuizen, H., Shabangu, F., and van der Lingen, C. 2021a. A Synthesis of Current Scientific Information Relating to the Decline in the African Penguin Population, the Small Pelagic Fishery and Island Closures. DFFE (Department of Forestry, Fisheries and the Environment). Unpublished report. Cape Town, South Africa.

Coetzee, J.C., Makhado, A., van der Lingen, C.D., Ebrahim, Z., Kock, A., Lawrence, C., and Shabangu, F.W. 2021b. African penguin colony closures: Finding a balance between minimizing costs to the small pelagic fishing industry while maximizing coverage of foraging area for breeding African penguins. DFFE Document O&C/2021/SEP/Extended Penguin TT/01.

Coetzee J.C., Merkle D., Rademan J., and van der Westhuizen, J.J. 2016. Small scale hydro-acoustic surveys 2013 to 2015. Report No. FISHERIES/2016/DEC/SWG-PEL/73. Cape Town, South Africa: Department of Agriculture, Forestry and Fisheries

Coetzee J.C., van der Lingen, C.D., and Shabangu, F.W. 2021b. A (simple) structured approach for evaluating potential benefits and costs of long-term closures to purse-seine fishing around African penguin breeding colonies. DEFF Fisheries document: FISHERIES/2021/JUL/SWG-PEL/44.

Cohen, L.A., Pichegru, L., Grémillet, D., Coetzee, J., Upfold, L., and Ryan, P.G. 2014. Changes in prey availability impact the foraging behaviour and fitness of Cape gannets over a decade. *Marine Ecology Progress Series* 505: 281–293.

Colburn, L.L., Jepson, M., Weng, C., Seara, T., Weiss, J., and Hare, J.A. 2016. Indicators of climate change and social vulnerability in fishing dependent communities along the eastern and Gulf coasts of the United States. *Marine Policy* 74: 323–333.

Collie, J., Botsford, L., Hastings, A., Kaplan, I., Largier, J., Livingston, P., Plagányi, É.E., Rose, K., Wells, B., and Werner, F. 2016. Ecosystem models for fisheries management: finding the sweet spot. *Fish and Fisheries* 17: 101–125.

Crawford, R.J.M., Altwegg, R., Barham, B.J., Barham, P.J., Durant, J.M., Dyer, B.M., Gekenhuys, D., Makhado, A.B., Pichegru, L., Ryan, P.G. Underhill, L.G., Upfold, L., Visagie, J., Waller, L.J., and Whittington, P.A. 2011. Collapse of South Africa's penguins in the early 21st century. *African Journal of Marine Science* 33: 139–156.

Crawford, R.J.M., Kemper, J., and Underhill, L.G. 2013. African penguin (*Spheniscus demersus*). In: Garcia Borboroglu, P. and Boersma, P.D. (Eds.) Penguins: natural history and conservation. University of Washington Press, Seattle & London.

Crawford, R.J.M., Makhado, A.B., Whittington, P.A., Randall, R.M., Oosthuizen, W.K., and Waller L.J. 2015. A changing distribution of seabirds in South Africathe possible impact of climate and its consequences. *Frontiers in Ecology and Evolution* 3: 10.

Crawford, R.J.M., Sydeman, W.J., Tom, D.B., Thayer, J.A., Sherley, R.B., Shannon, L.J., McInnes, A.M., Makhado, A.B., Hagen, C., Furness, R.W., Carpenter-Kling, R., and Saraux, C. 2022. Food limitation of seabirds in the Benguela ecosystem and management of their prey base. *Namibian Journal of Environment*. 6A: 1–13.

Crawford R.J.M., Sydeman W.J., Thompson S.A., Sherley R.B., and Makhado A.B. 2019. Food habits of an endangered seabird indicate recent poor forage fish availability off western South Africa. *ICES Journal of Marine Science* 76: 1344–1352.

Crossin G.T., Trathan, P.N., Phillips, R.A., Dawson, A., Le Bouard, F., and Williams, T.D. 2010. A carryover effect of migration underlies individual variation in reproductive readiness and extreme egg size dimorphism in macaroni penguins. *American Naturalist* 176: 357–366.

Curtis, R.E., and McConnell, K.E. 2004. Incorporating information and expectations in fishermen's spatial decisions. *Marine Resource Economics* 19: 131–143.

Curtis, R., and Hicks, R.L. 2000. The cost of sea turtle preservation: The case of Hawaii's pelagic longliners. *American Journal of Agricultural Economics* 82: 1191–1197.

Cury, P.M., Boyd, I.L., Bonhommeau, S., Anker-Nilssen, T., Crawford, R.J.M., Furnes, R.W., Mills, J.A., Murphy, E.J., Oserblom, H., Paleczny, M., Piat, J.F., Roux, J-P., Shannon, L., and Sydman, W.J. 2011. Global seabird response to forage fish depletion – one-third for the birds. *Science* 334: 1703–1706.

Davis, S.E., Nager, R.G., and Furness, R.W. 2005. Food availability affects adult survival as well as breeding success of parasitic jaegers. *Ecology* 86: 1047–1056.

DFFE (Department of Forestry, Fisheries and the Environment). 2021. A synthesis of current scientific information relating to the decline in the African penguin population, the small pelagic fishery and island closures. Unpublished report. Cape Town, South Africa.

de Moor, C.L. 2021. Updated assessment of the South African sardine resource using data from 1984-2000. DEFF Fisheries document: FISHERIES/2021/APR/SWG-PEL/23

de Moor, C.L., and Butterworth, D.S. 2015. Assessing the South African sardine resource: two stocks rather than one? *African Journal of Marine Science* 37: 41–51.

Denhard, N., Ludynia, K., Poisbleau, M., Demongin, L., and Quillfeldt, P. 2013. Good days, bad days: Wind as a driver of foraging success in a flightless seabird, the southern rockhopper penguin. *PLoS ONE* 8: e79487.

Dépalle, M., Sanchirico, J.N., Thébaud, O., O'Farrell, S., Haynie, A.C., and Perruso, L. 2021. Scale-dependency in discrete choice models: A fishery application. *Journal of Environmental Economics and Management* 105: 102388.

Dépalle, M., Thébaud, O., and Sanchirico, J.N. 2020. Accounting for fleet heterogeneity in estimating the impacts of large-scale fishery closures. *Marine Resource Economics* 35: 361–78.

Dias, M.P., Carneiro, A.P.B., Warwick-Evans, V., Harris, C., Lorenz, K., Lascelles, B., Clewlow, H.L., Dunn, M.J., Hinke, J.T., Kim, J.H., Kokubun, N., Manco, F., Ratcliffe, N., Santos, M., Takahashi, A., Trivelpiece, W., and Trathan, P.N. 2018.. Identification of marine Important Bird and Biodiversity Areas for penguins around the South Shetland Islands and South Orkney Islands. *Ecology and Evolution* 8: 10520–10529.

Díaz, M., Grim, T., Markó, G., Morelli, F., Ibáñez-Alamo, J.D., Jokimäki, J., Kaisanlahti-Jokimäki, M.L., Tätte, K., Tryjanowski, P., and Møller, A.P. 2021. Effects of climate variation on bird escape distances modulate community responses to global change. *Scientific Reports* 11: 12826.

Dunn M.J., Silk J.R.D., and Trathan P.N. 2011. Post-breeding dispersal of Adélie penguins (Pygoscelis adeliae) nesting at Signy Island, South Orkney Islands. *Polar Biology* 34: 205–214.

Dupont, D.P. 1993. Price uncertainty, expectations formation, and fishers' location choices. *Marine Resource Economics* 8: 219–247.

Eales, J., and Wilen, J.E. 1986. An examination of fishing location choice in the pink shrimp fishery. *Marine Resource Economics* 2: 331–351.

Espinaze, M.P.A., Hui, C., Waller, L., and Matthee, S. 2020. Nest-type associated microclimatic conditions as potential drivers of ectoparasite infestations in African penguin nests. *Parasitology Research* 119: 3603–3616.

Essington, T.E., Moriarty, P.E., Froehlich, H.E., Hodgson, E.E., Koehn, L.E., Oken, K.L., Siple, M.C., and Stawitz, C.C., 2015. Fishing amplifies forage fish population collapses. *Proceedings of the National Academy of Sciences* 112: 6648–6652.

Fauchald, P., and Tveraa T. 2003. Using first-passage time in the analysis of area-restricted search and habitat selection. *Ecology* 84: 282–288.

Favoretto, F., López-Sagástegui, C., Sala, E., and Aburto-Oropeza, O. 2023. The largest fully protected marine area in North America does not harm industrial fishing. *Science Advances* 9: eadg0709.

Ferraro, P.J., Sanchirico, J.N., and Smith, M.D. 2019. Causal inference in coupled human and natural systems. *Proceedings of the National Academy of Sciences* 116: 5311–5318.

Frederiksen, M., Wanless, S., Harris, M.P., Rothery, P., and Wilson, L.J. 2004. The role of industrial fisheries and oceanographic change in the decline of North Sea black-legged kittiwakes. *Journal of Applied Ecology* 41: 1129–1139.

Frost, P.G.H., Siegfried, W.R., and Burger, A.E. 1976. Behavioural adaptations of the Jackass penguin, *Spheniscus demersus* to a hot, arid environment. *Journal of Zoology*, *London* 179: 165–187.

Geldart, E.A., Barnas, A.F., Semeniuk, C.A.D., Gilchrist, H.G., Harris, C.M., and Love, O.P. 2022. A colonial-nesting seabird shows no heart-rate response to drone-based population surveys. *Scientific Reports* 12: 18804.

Goethel, D., Omori, K.L., Punt, A.E., Lynch, P.D., Berger, A.M., deMoor, C.L., Plagányi, É.E., Cope, J.M., Dowling, N.A., McGarvey, R., Preece, A., Thorson, J., Chaloupka, M., Gaichas, S., Gilman, E., Hesp, S.A., Longo, C., Yao, N., and Methot, R.D. 2023. Oceans of plenty? Challenges, advancements, and future directions for the provision of evidence-based fisheries management advice. *Reviews in Fish Biology and Fisheries* 33: 375–410.

Haddon, M., Parma, A., Punt, A.E., and Wilberg, M.J. 2020. Report of international review of some aspects of the Island Closure Experiment. FISHERIES/2020/DEC/SWG-PEL/REVIEW/07. Available from https://science.uct.ac.za/maram/2020-workshop.

Halpern, B.S., Klein, C.J., Brown, C.J., Beger, M., Grantham, H.S., Mangubhai, S., Ruckelshaus, M., Tulloch, V.J., Watts, M., White, C., and Possingham, H.P., 2013. Achieving the triple bottom line in the face of inherent trade-offs among social equity, economic return, and conservation. *Proceedings of the National Academy of Sciences* 110: 6229–6234.

Haltuch, M.A., and Punt, A.E. 2011. The promises and pitfalls of including decadal-scale climate forcing of recruitment in groundfish stock assessment. *Canadian Journal of Fisheries and Aquatic Sciences* 68: 912–926.

Handley, J.M., Pearmain, E. J., Oppel, S., Carneiro, A. P. B., Hazin, C., Phillips, R. A., Ratcliffe, N., Staniland, I.J., Clay, T. A., Hall, J., Scheffer, A., Fedak, M., Boehme, L., Pütz, K., Belchier, M., Boyd, I.L., Trathan, P.N., and Dias, M.P. 2020. Evaluating the effectiveness of a large multi-use MPA in protecting Key Biodiversity Areas for marine predators. *Diversity and Distributions* 26: 715–729.

Harding, A.M.A., Welcker, J., Steen, H., Hamer, K.C., Kitaysky, A.S., Fort, J., Talbot, S.L., Cornick, L.A., Karnovsky, N.J., Gabrielsen, G.W., and Grémillet, D. 2011. Adverse foraging conditions may impact body mass and survival of a high Arctic seabird. *Oecologia* 167: 49–59.

Hayes, M.C., Gray, P.C., Harris, G., Sedgwick, W.C., Crawford, V.D., Chazal, N., Crofts, S., and Johnston, D.W. 2021. Drones and deep learning produce accurate and efficient monitoring of large-scale seabird colonies. *Ornithological Applications* 123: duab022.

Haynie, A.C., and Layton, D.F. 2010. An expected profit model for monetizing fishing location choices. *Journal of Environmental Economics and Management* 59 (2): 165–176.

Hicks, R.L., and Schnier, K.E. 2008. Eco-labeling and dolphin avoidance: A dynamic model of tuna fishing in the eastern tropical Pacific. *Journal of Environmental Economics and Management* 56: 103–116.

Hilborn. R., Akselrud Allen, C., Peterson, H., and Whiterhouse, G.A. 2021., The trade-off between biodiversity and sustainable fish harvest with area-based management, *ICES Journal of Marine Science* 78: 2271–2279.

Himes-Cornell, A., Hoelting, K., Maguire, C., Munger-Little, L., Lee, J., Fisk, J., Felthoven, R., Geller, C., and Little, P, 2013. Community profiles for North Pacific Fisheries - Alaska. U. S. Department of Commerce., NOAA Technical Memorandum. National Marine Fisheries Service-AFSC-259, Volumes 1-12.

Hockey, P.A.R., and Hallinan, J. 1981. Effect of human disturbance on the breeding behaviour of jackass penguins *Spheniscus demersus*. *South African Journal of Wildlife Research* 11: 59–62.

Holland, D.S., and Sutinen J.G. 2000. Location choice in New England trawl fisheries: Old habits die hard. *Land Economics* 76: 133–149.

Holland, D., Sanchirico, J., Johnston, R., and Deepak J.D. 2012. *Economic analysis for ecosystem-based management: applications to marine and coastal environments*. Routledge.

Hollowed, A.B., Holsman, K.K., Haynie, A.C., Hermann, A.J., Punt, A.E., Aydin, K., Ianelli, J.N., Kasperski, S., Cheng, W., Faig, A., and Kearney, K.A. 2020. Integrated modeling to evaluate climate change impacts on coupled social-ecological systems in Alaska. *Frontiers in Marine Science* 6: 775.

Holsman, K.K., Ianelli, J., Aydin, K., Punt, A.E., and Moffitt, E.A. 2016. A comparison of fisheries biological reference points estimated from temperature-specific multi-species and single-species climate-enhanced stock assessment models. *Deep Sea Research Part II: Topical Studies in Oceanography* 134: 360–378.

Horswill1, C., Manica, A., Daunt, F., Newell, M., Wanless, S., Wood, M., and Matthiopoulos, J. 2021. Improving assessments of data-limited populations using life-history theory. *Journal of Applied Ecology* 58: 1225–1236.

Horswill, C., and Robinson, R.A. 2015. Review of seabird demographic rates and density dependence. JNCC Report 552. Joint Nature Conservation Committee, Peterborough.

Horswill, C., Matthiopoulos, J., Green, J.A., Meredith, M.P., Forcada, J., Peat, H., Preston, M., Trathan, P.N., and Ratcliffe, N. 2014. Survival in macaroni penguins and the relative importance of different drivers: individual traits, predation pressure and environmental variability. *Journal of Animal Ecology* 83: 1057–1067.

Horswill, C., Trathan, P.N., and Ratcliffe, N. 2017. Linking extreme interannual changes in prey availability to foraging behaviour and breeding investment in a marine predator, the macaroni penguin. *PLoS ONE* 12: e0184114.

Houston, A.I., Thompson, W.A., and Gaston, A.J. 1996. The use of a time and energy budget model of a parent bird to investigate limits to fledging mass in the thick-billed murre. *Functional Ecology* 10: 432–439.

Hutchings, L., Beckley, L.E., Griffiths, M.H., Roberts, M.J., Sundby, S., and van der Lingen, C. 2002. Spawning on the edge: spawning grounds and nursery areas around the southern African coastline. *Marine and Freshwater Research*, 53: 307–318.

Hutniczak, B., and Münch, A. 2018. Fishermen's location choice under spatio-temporal update of expectations. *Journal of Choice Modelling* 28: 124–136.

IUCN (International Union for Conservation of Nature) 2018. *Spheniscus demersus*. The IUCN Red List of Threatened Species 2018: e.T22697810A132604504. Accessed on 03 April 2023; https://dx.doi.org/10.2305/IUCN.UK.2018-2.RLTS.T22697810A132604504.en.

Jones, F.M., Allen, C., Arteta, C., Arthur, J., Black, B., Emmerson, L.M., Freeman, R., Hines, G., Lintott, C.J., Macháčková, Z., Miller, G., Simpson, R., Southwell, C., Torsey, H.R., Zisserman, A., and Hart, T. 2018. Time-lapse imagery and volunteer classifications from the Zooniverse Penguin Watch project. *Scientific Data* 5: 180124.

Kerry, K., Clarke, J., and Else, G. 1993. The use of an automated weighing and recording system for the study of the biology of Adélie penguins (*Pygoscelis adeliae*). Proceedings of the NIPR Symposium on Polar Biology 6: 62–75.

Kirkman, S.P., Blamey, L., Lamont, T., Field, J.G., Bianchi, G., Huggett, J.A., Hutchings, L., Jackson-Veitch, J., Lett, C., Lipinski, M.R., Madwila, S.W., Pfaff, M.C., Samaii, T., Shannon, L.J., Shin, Y-J., van der Lingen, C.D., and Yemane, D. 2016. Spatial characterisation of the Benguela ecosystem for ecosystem-based management, *African Journal of Marine Science* 38: 7-–22.

Koehn, L.E., Siple, M.C., and Essington, T.E. 2021. A structured seabird population model reveals how alternative forage fish control rules benefit seabirds and fisheries. *Ecological Applications* 31: e02401.

Krause, D.J., Hinke, J.T., Goebel, M.E., and Perryman, W.L. 2021. Drones minimize Antarctic predator responses relative to ground survey methods: An appeal for context in policy advice. *Frontiers in Marine Science* 8: 648772.

Lascelles, B.G., Taylor, P.R., Miller, M.G.R., Dias, M.P., Oppel, S., Torres, L., Hedd, A., Le Corre, M., Phillips, R.A., Shaffer, S.A., Weimerskirch, H., and Small, C. 2016. Applying global criteria to tracking data to define important areas for marine conservation. *Diversity and Distributions* 22: 422–431.

Lei, B.R., Green, J.A., and Pichegru, L. 2014. Extreme microclimate conditions in artificial nests for endangered African penguins. *Bird Conservation International* 24: 201–213.

Leith, F.W., Grigg, J.L., Barham, B.J., Barham, P.J., Ludynia, K., McGeorge, C., Mdluli, A., Parsons, N.J., Waller, L.J., and Sherley, R.B., 2022. Intercolony variation in reproductive skipping in the African penguin. *Ecology and Evolution* 12: e9255.

Lescroël, A., Dugger, K.M., Ballard, G., and Ainley, D.G. 2009. Effects of individual quality, reproductive success and environmental variability on survival of a long-lived seabird. *Journal of Animal Ecology* 78: 798–806.

Lescroël, A., Schmidt, A., Elrod, M., Ainley, D.G., and Ballard, G. 2021. Foraging dive frequency predicts body mass gain in the Adélie penguin. *Scientific Reports* 11: 22883.

Makhado, A.B., Meyer, M.A., Crawford, R.J.M., Underhill, L.G., and Wilke, C. 2009. Efficacy of culling seals seen preying on seabirds as a means of reducing seabird mortality. *African Journal of Ecology* 47: 335–340.

Makhado, A.B., Crawford, R.J.M., Waller, L.J., and Underhill, L.G. 2013. An assessment of the impact of predation by Cape fur seals *Arctocephalus pusillus pusillus* on seabirds at Dyer Island, South Africa. *Ostrich* 84: 191-198.

Marcoul, P., and Weninger, Q. 2008. Search and active learning with correlated information: Empirical evidence from mid-Atlantic clam fishermen. *Journal of Economic Dynamics and Control* 32: 1921–1948.

Masotla, M.M., Visagie, L., and Makhado, A.B. 2023. Estimates of trends in numbers of selected seabird species breeding in South Africa. DFFE Report for the March meeting of the Expert Review Panel To Advise On The Proposed Fishing-Area Closures Adjacent To South Africa's African Penguin Breeding Colonies And The Decline In The Penguin Population.

Mattern, T., Rexer-Huber, K., Parker, G., Amey, J., Green, C-P., Tennyson, A.J.L., Sagar, P.M., and Thompson, D.R. 2021. Erect-crested penguins on the Bounty Islands: Population size and trends determined from ground counts and drone surveys. *Notornis* 68: 37–50.

Maunder, M.N., and Punt, A.E., 2013. A review of integrated analysis in fisheries stock assessment. *Fisheries Research* 142: 61–74.

McClatchie, S., Goericke, R., and Hill, K. 2010. Re-assessment of the stock-recruitment and temperature-recruit relationships for Pacific sardine (*Sardinops sagax*). *Canadian Journal of Fisheries and Aquatic Science* 67: 1782–1790.

McInnes, A.M., and Pistorius, P.A. 2019. Up for grabs: prey herding by penguins facilitates shallow foraging by volant seabirds. *Royal Society Open Science* 6: 190333.

McInnes, A.M., McGeorge, C., Ginsberg, S., Pichegru, L., Pistorius, P.A. 2017. Group foraging increases foraging efficiency in a piscivorous diver, the African penguin. *Royal Society Open Science* 4: 170918

McInnes, A.M., Thiebault, A., Cloete, T., Pichegru, L., Aubin, T., McGeorge, C., and Pistorius, P.A. 2020. Social context and prey composition are associated with calling behaviour in a diving seabird. *Ibis* 162: 1047–1059.

McInnes, A.M., Weideman, E., Waller, L., Sherley, R., Pichegru, L., Ludynia, K., Hagen, C., Barham, P., Smith, C., Kock, A., and Carpenter-Kling, T. 2023. Purse-seine fisheries closure configurations for African Penguin conservation: methods and considerations for optimal closure designs. Document FP/PANEL/WP/09 presented to the Panel in June 2023.

McLeay, L.J., Page, B., Goldsworthy, S.D., Ward, T.M., Paton, D.C., Waterman, M., and Murray, M.D. 2008. Demographic and morphological responses to prey depletion in a crested tern (*Sterna bergii*) population: can fish mortality events highlight performance indicators for fisheries management? *ICES Journal of Marine Science* 66: 237–247.

Medoff, S., Lynham, J., and Raynor, J. 2022. Spillover benefits from the world's largest fully protected MPA. *Science* 378 (6617): 313–316.

Mistiaen, J.A., and Strand, I.E. 2000. Location choice of commercial fishermen with heterogeneous risk preferences. *American Journal of Agricultural Economics* 82: 1184–1190.

Montevecchi, W.A. 2002. Interactions between fisheries and seabirds. In: Schreiber, E.A. and Burger J. (Eds.) Biology of marine birds. CRC Press, Boca Raton, London, New York and Washington D.C.

Nevitt, G.A., Reid, K., and Trathan, P. 2004. Testing olfactory foraging strategies in an Antarctic seabird assemblage. *Journal of Experimental Biology* 207: 3537–3544.

Oro, D., and Furness, R.W. 2002. Influences of food availability and predation on survival of kittiwakes. *Ecology* 83: 2516–2528.

Perriman, L., and Steen, H. 2000. Blue penguin (Eudyptula minor) nest distribution and breeding success on Otago Peninsula, 1992 to 1998. *New Zealand Journal of Zoology* 27: 269–275.

Pichegru, L., Nyengera, R., McInnes, A.M., and Pistorius, P. 2017. Avoidance of seismic survey activities by penguins. *Scientific Reports* 7: 16305.

Pichegru L., Ryan P.G., van Eeden R., Reid T., Gremillet D., and Wanless R. 2012 Industrial fishing, no-take zones and endangered penguins. *Biological Conservation* 156: 117–125.

Pichegru, L., Vibert, L., Thiebault, A., Charrier, I., Stander, N., Ludynia, K., Lewis, M., Carpenter-Kling, T., and McInnes, A. 2022. Maritime traffic trends around the southern tip of Africa – did marine noise pollution contribute to the local penguins' collapse? *Science of the Total Environment* 849: 157878.

Plagányi, É.E., and Butterworth, D.S. 2012. The Scotia Sea krill fishery and its possible impacts on dependent predators: modeling localized depletion of prey. *Ecological Applications* 22: 748–761.

Plagányi, É.E., Blamey, L.K., Rogers, J.G., and Tulloch, V.J. 2022. Playing the detective: Using multispecies approaches to estimate natural mortality rates. *Fisheries Research* 249: 106229.

Plagányi, É.E., Hutchings, L., and Field, J.G., 2000. Anchovy foraging: simulating spatial and temporal match/mismatches with zooplankton. *Canadian Journal of Fisheries and Aquatic Sciences* 57: 2044-2053.

Plagányi, É., Punt, A., Hillary, R., Morello, E., Thebaud, O., Hutton, T., Pillans, R., Thorson, J., Fulton, E.A., Smith, A.D.T., Smith, F., Bayliss, P., Haywood, M., Lyne, V., and Rothlisberg, P. 2014. Multi-species fisheries management and conservation: tactical applications using models of intermediate complexity. *Fish Fisheries* 15: 1–22.

Pollnac, R.B., Abbott-Jamieson, S., Smith, C., Miller, M.L., Clay, P.M., and Oles, B. 2006. A model for fisheries social impact assessment. *Marine Fisheries Research* 68: 1-–18.

Punt, A.E., and Butterworth, D.S., 1995. The effects of future consumption by the Cape fur seal on catches and catch rates of the Cape hakes. 4. Modelling the biological interaction between Cape fur seals *Arctocephalus pusillus pusillus* and the Cape hakes *Merluccius capensis* and *M. paradoxus*. *South African Journal of Marine Science*, 16: 255–285.

Qian, Y., Humphries, G.R.W., Trathan, P.N.N., Lowther, A., and Donovan, C.R.R. 2023. Counting animals in aerial images with a density map estimation model. *Ecology and Evolution* 13: e9903.

Randall, R.M. 1995. Jackass penguins. pp. 244–256 In: Oceans of life off southern Africa. A.I.L. Payne and R.J.M. Crawford (Eds.) Vlaeberg, Cape Town.

Reiertsen, T.K., Erikstad, K.E., Anker-Nilssen, T., Barrett, R.T., Boulinier, T., Frederiksen, M., Gonzalez-Solis, J., Grémillet, D., Johns, D., Moe, B., Ponchon, A., Skern-Mauritzen, M., Sandvik, H., and Yoccoz, N.G. 2014. Prey density in non-breeding areas affects adult survival of black-legged kittiwakes *Rissa tridactyla*. *Marine Ecology Progress Series* 509: 289–302.

Reimer, M.N., and Haynie, A.C. 2018. Mechanisms matter for evaluating the economic impacts of marine reserves. *Journal of Environmental Economics and Management* 88: 427–446.

Robinson, W.M.L., Butterworth, D.S., and Furman, L.B. 2014. Analyses of the results from the island closure feasibility study for the Dassen/Robben and St Croix/Bird island pairs. MARAM Document MARAM/IWS/DEC14/Peng/B4.

Robinson W.M.L., Butterworth D.S., and Plaganyi É.E. 2015. Quantifying the projected impact of the South African sardine fishery on the Robben Island penguin colony. *ICES Journal of Marine Science* 72: 1822–1833.

Rogers, J., and Plagányi, É.E. 2022. Culling corallivores improves short-term coral recovery under bleaching scenarios. *Nature Communications* 13: 1–17.

Ross-Gillespie, A., and Butterworth, D.S. 2016a. Implementation of the algorithm recommended by the panel for the 2016 international stock assessment workshop for assessing whether or not to continue with the penguin island closure experiment. Department of Forestry, Fisheries and the Environment report FISHERIES/2016/DEC/SWGPEL/77rev.

Ross-Gillespie, A. and Butterworth, D.S. 2016b. Penguin power analyses using the approach recommended by the international panel: methods and the complete set of results. MARAM/IWS/DEC16/PENG CLOS/P1a-rev.

Ross-Gillespie, A., and Butterworth, D.S. 2021a. Re-analysis of the island closure experiment results to implement the suggestions of the December 2020 International Panel. Department of Forestry, Fisheries and the Environment Report: FISHERIES/2021/APR/SWG-PEL/35. (available at https://open.uct.ac.za/handle/11427/33665).

Ross-Gillespie, A., and Butterworth, D.S. 2021b. Updated analysis of results from data arising from the Island Closure Experiment. DEFF Fisheries document: FISHERIES/2021/JUN/SWG-PEL/39rev. https://doi.org/10.25375/uct.15073404.v1.

Ross-Gillespie, A., and Butterworth, D.S. 2023a . Results for the section A of sensitivity runs requested by the penguin review panel. Document FP/PANEL/WP/04 presented to the Panel in June 2023 (with Addendum).

Ross-Gillespie, A., and Butterworth, D.S. 2023b . Correlation of the sardine and anchovy catch and biomass series. Document FP/PANEL/WP/03 presented to the Panel in June 2023

Rümmler, M-C., Esefeld, J., Pfeifer, C., and Mustafa, O. 2021. Effects of UAV overflight height, UAV type, and season on the behaviour of emperor penguin adults and chicks. *Remote Sensing Applications – Society and Environment* 23: 100558.

Ryan, P.G., Edwards, L., and Pichegru, L. 2012. African Penguins *Spheniscus demersus*, bait balls and the Allee effect. *Ardea* 100: 89–94.

Shaw K.A., Waller U., Crawford R.J.M., and Oosthuizen W.H. (eds) 2011. Proceedings of the African Penguin BMPs Stakeholder Workshop, 26–28 October 2010, Die Herberg, Arniston, South Africa. Stellenbosch, South Africa: CapeNature.

Scheffer A., Trathan P.N., and Collins M. 2010. Foraging behaviour of King Penguins (*Aptenodytes patagonicus*) in relation to predictable mesoscale oceanographic features in the Polar Front Zone to the north of South Georgia. *Progress in Oceanography* 86: Special Issue SI 232–245.

Seung, C.K. 2010. Evaluating supply-side and demand-side shocks for fisheries: A computable general equilibrium (CGE) model for Alaska. *Economic Systems Research* 22: 87–109.

Seung, C.K., and Waters, E.C. 2006. A review of regional economic models for fisheries management in the US. *Marine Resource Economics* 21: 101–124.

Sherley R.B. 2023. Additional analysis applied to the Western Cape chick condition and survival data to address requests by the 2023 International Panel reviewing the Island Closures Experiment (ICE). Document FP/PANEL/WP/06 presented to the Panel in June 2023.

Sherley R.B., Barham, B.J., Barham, P.J., Campbell, K.J., Crawford, R.J.M., Grigg, J., Horswill, C., McInnes, A., Morris, T.L., Pichgru, L., Steinfurth, A., Weller, F., Winker, H., and Votie, S.C. 2018. Bayesian inference reveals positive but subtle effects of experimental fishery closures on marine predator demographics. *Proceedings of the Royal Society* B. 285: 20172443.

Sherley R.B., Barham, B.J., Barham, P.J., Campbell, K.J., Crawford, R.J.M., Grigg, J., Horswill, C., McInnes, A., Morris, T.L., Pichegru, L., Steinfurth, A., Weller, F., Winker, H., and Votie, S.C. 2021. Correction to 'Bayesian inference reveals positive but subtle effects of experimental fishery closures on marine predator demographics'. *Proceedings of the Royal Society* B. 288: 20212129.

Sherley, R.B., Barham, B.J., Barham, P.J., Leshoro, T.M., and Underhill, L.G. 2012. Artificial nests enhance the breeding productivity of African penguins (*Spheniscus demersus*) on Robben Island, South Africa. Emu 112: 97–106.

Sherley, R.B., Crawford, R.J.M., de Blocq, A.D., Dyer, B.M., Geldenhuys, D., Hagen, C., Kemper, J., Makhado, A.B., Pichegru, L., Tom, D., Upfold, L., Visagie, J., Waller, L.J., and Winker, H. 2020. The conservation status and population decline of the African penguin deconstructed in space and time. *Ecology and Evolution* 10: 8506–8516.

Sherley, R.B., Ludynia, K., Dyer, B.M., Lamont, T., Makhado, A.B., Roux, J.-P., Scales, K.L., Underhill, L.G., and Votier, S.C. 2017. Metapopulation tracking juvenile penguins reveals an ecosystem-wide ecological trap. *Current Biology* 27: 563–568.

Sherley R.B., Underhill L.G., Barham B.J., Barham P.J., Coetzee J.C., Crawford R.J.M., Dyer, B.M., Leshoro, T.M., and Upfold, L. 2013. Influence of local and regional prey availability on breeding performance of African penguins *Spheniscus demersus*. *Marine Ecology Progress Series* 473: 291–301.

Smith, M.D. 2000. Spatial search and fishing location choice: Methodological challenges of empirical modeling. *American Journal of Agricultural Economics* 82: 1198–1206.

Siple, M.C., Koehn, L.E., Johnson, K.F., Punt, A.E., Canales, T.M., Carpi, P., de Moor, C.L., De Oliveira, J.A., Gao, J., Jacobsen, N.S., and Lam, M.E. 2021. Considerations for management strategy evaluation for small pelagic fishes. *Fish and Fisheries*, 22: 1167–1186.

Smith, M.D. 2002. Two econometric approaches for predicting the spatial behavior of renewable resource harvesters. *Land Economics* 78: 522–538.

Smith, M.D. 2005. State dependence and heterogeneity in fishing location choice. *Journal of Environmental Economics and Management* 50: 319–340.

Smith, M.D., and Wilen, J.E. 2003. Economic impacts of marine reserves: The importance of spatial behavior. *Journal of Environmental Economics and Management* 46: 183–206.

Smith, M.D., Lynham, J., Sanchirico, J.N., and Wilson, J.A. 2010. Political economy of marine reserves: Understanding the role of opportunity costs. *Proceedings* *of the National Academy of Sciences of the United States of America* 107: 18300–18305.

Smith, M.D., Zhang, J., and Coleman, F.C. 2006. Effectiveness of marine reserves for large-scale fisheries management. *Canadian Journal of Fisheries and Aquatic Sciences* 63: 153–164.

Southwell, D., Emmerson, L., Forcada, J., and Southwell, C. 2015. A bioenergetics model for estimating prey consumption by an Adélie penguin population in East Antarctica. *Marine Ecology Progress Series* 526: 183–197.

Spaans, B., Leopold, M., and Plomp, M. 2018. Using a drone to determine the number of breeding pairs and breeding success of Sandwich terns *Sterna sandvicensis*. *Limosa* 91: 30–37.

SSERenewables 2022. Berwick Bank Wind Farm Derogation Case. Fisheries compensatory measures evidence report. BERWICK BANK WIND FARM EIA Documents (berwickbank-eia.com)

Sun, J., Hinton, M.G., and Webster, D.G. 2016. Modeling the spatial dynamics of international tuna fleets. *PloS One* 11: e0159626.

Sutherland, D.R., Dann, P., and Jessop, R.E. 2014. Evaluation of artificial nest sites for long-term conservation of a burrow-nesting seabird. *Journal of Wildlife Management* 78: 1415–1424.

Sutton, G., Pichegru, L., Botha, J.A., Kouzani, A.Z., Adams, S., Bost, C.A., and Arnould, J.P.Y. 2020. Multi-predator assemblages, dive type, bathymetry and sex influence foraging success and efficiency in African penguins. *PeerJ* 8: e9380.

Sydeman, W.J., Hunt Jr., G.L., Pikitch, E.K., Parrish, J.K., Piatt, J.F., Boersma, P.D., Kaufman, L., Anderson, D.W., Thompson, S.A., and Sherley, R.B. 2021. South Africa's experimental fisheries closures and recovery of the endangered African penguin. *ICES Journal of Marine Science* 78: 3538–3543.

Sydeman W.J., Hunt Jr. G.L., Pikitch E.K., Parrish J.K., Piatt J.F., Boersma P.D., Kaufman L., Anderson D.W., Thompson S.A., and Sherley R.B. 2022. African penguins and localized fisheries management: Response to Butterworth and Ross-Gillespie. *ICES Journal of Marine Science* 79: 1972–1978.

Sydeman, W.J., Thompson, S.A., Anker-Nilssen, T., Arimitsu, M., Bennison, A., Bertrand, S., Boersch-Supan, P., Boyd, C., Bransome, N.C., Crawford, R.J.M., Daunt, F., Furness, R.W., Gianuca, D., Gladics, A., Koehn, L., Lang, J.W., Logerwell, E., Morris, T.L., Phillips, E.M., Provencher, J., Punt, A.E., Saraux, C., Shannon, L., Sherley, R.B., Simeone, A., Wanless, R.M., Wanless, S., and Zador, S. 2017. Best practices for assessing forage fish fisheries – seabird resource competition. *Fisheries Research* 194: 209–221.

Thiebot J.-B., Cherel Y., Trathan P.N., and Bost C.-A. 2011. Inter-population segregation in the wintering areas of macaroni penguins. *Marine Ecology Progress Series* 421: 279–290.

Trathan, P.N., and Emmerson, L. 2014. News and Views: Animal identification with robot rovers. *Nature* 11: 1217–1218.

Trathan, P.N., Bishop, C., Maclean, G., Brown P., Fleming, A., and Collins, M.A. 2008. Linear tracks and restricted temperature ranges characterise penguin foraging pathways. *Marine Ecology-Progress Series* 370: 285-294.

Trathan, P.N., Fielding, S., Warwick-Evans, V., Freer, J., and Perry, F. 2022. Seabird and seal responses to the physical environment and to spatio-temporal variation in the distribution and abundance of Antarctic krill at South Georgia, with implications for local fisheries management. *ICES Journal of Marine Science* 79: 2373–2388.

Tulloch, V.J.D., Plagányi, E., Brown, C., Matear, R., and Richardson, A.J. 2019. Future recovery of baleen whales is imperiled by climate change. *Global Change Biology* 25: 1263–1281.

Urban-Econ Development Economists. 2023a. The pelagic fishing industry: Socio-economic impact assessment. Document FP/PANEL/WP/13 presented to the Panel in June 2023.

Urban-Econ Development Economists. 2023b. Update of WP13_Pelagic Fishing Industry Impact Report. Document FP/PANEL/WP/30 presented to the Panel in June 2023.

Urban-Econ Development Economists. 2023c. Annexure A. Sensitivity analysis. Document FP/PANEL/WP/21 presented to the Panel in June 2023.

Valle, R.G., and Scarton, F. 2021. Drone-conducted counts as a tool for the rapid assessment of productivity of Sandwich terns (*Thalasseus sandvicensis*). *Journal of Ornithology* 162: 621–628.

van der Lingen, C.L., 2023. Latest understanding of climate change impacts on the spatial distribution of anchovy and sardine off South Africa (in response to a request by the African Penguin International Review Panel). Document 5.1 presented to the Panel in March 2023.

van Eeden, R., Reid, T., Ryan, P.G., and Pichegru, L. 2016. Fine-scale foraging cues for African penguins in a highly variable marine environment. *Marine Ecology Progress Series* 543: 257–271.

Wanless, S., Albon, S.D., Daunt, F., Sarzo, B., Newell, M.A., Gunn, C., Speakman, J.R., and Harris, M.P. 2023. Increased parental effort fails to buffer the cascading effects of warmer seas on common guillemot demographic rates. *Journal of Animal Ecology* 00: 00–00.

Warwick-Evans, V., Ratcliffe, N., Lowther, A.D., Manco, F., Ireland, L., Clewlow, H.L., and Trathan, P.N. 2018. Using habitat models for chinstrap penguins *Pygoscelis antarctica* to advise krill fisheries management during the penguin breeding season. *Diversity and Distributions* 24: 1756–1771.

Watson, B., Reimer, M.N., Guettabi, M., and Haynie, A. 2021. Commercial fisheries & local economies. *Journal of Environmental Economics and Management* 106: 102419.

Watts, M.E., Stewart, R.R., Martin, T.G., Klein, C.J., Carwardine, J., and Possingham, H.P. 2017. Systematic conservation planning with Marxan. In: Gergel, S., Turner, M. (Eds) Learning landscape ecology. Springer, New York, NY.

Welman S., and Pichegru L. 2023. Nest microclimate and heat stress in African penguins *Spheniscus demersus* breeding on Bird Island, South Africa. *Bird Conservation International* 33: e34.

Wilson, R.P., and Wilson, M-P.T. 1989. Substitute burrows for penguins on guano-free islands. *Le Gerfaut* 79: 125–131.

Wright, K.L.B., Pichegru, L., and Ryan, P.G. 2011. Penguins are attracted to dimethyl sulphide at sea. The *Journal of Experimental Biology* 214: 2509–2511

Yorio, P., and Boersma, P.D. 1994. Consequences of nest desertion and inattendance for Magellanic penguin hatching success. *Auk* 111: 215–218.

8.2 Presentations to the Panel

Anon. 2021. Fishing closure options as discussed at the Penguin Extended Task Team _ PETT_10b_Penguin_colony_closure proposals_20211130. Powerpoint file provided to the Panel.

Butterworth, D.S., and Ross-Gillespie, A. 2023. Summary of results and proposals from island closure related analyses. Powerpoint presentation to the June 2023 Panel meeting.

Coetzee, J. 2023. Information provided in response to requests from the Expert Panel to review the science around small pelagic fisheries and penguins – general issues. Powerpoint presentation to the March 2023 Panel meeting.

Adult Penguin and chick (photo credit Dr. Éva Plagányi-Lloyd)

9. ACKNOWLEDGEMENTS

The Panel wishes to express its appreciation to those who prepared documents for the Panel, made presentations during the workshop stages of the Panel process, and contributed to discussions during the March and June workshops and between meetings of the Panel. Janet Coetzee (DFFE), Mike Bergh (Olsps), Matt Horton (Olsps), Alexandra Kempthorne (Urban-Econ Development Economists), Alistair McInnes (Birdlife South Africa), Andrea Ross-Gillespie (UCT), Doug Butterworth (UCT), Lauren Waller (Endangered Wildlife Trust), Azwianewi Makhado (DFFE) and Richard Sherley (University of Exeter) are thanked for responding to Panel requests for additional information and analyses. The Panel would like to thank Ashley Naidoo and Millicent Makoala (DFFE) for their logistical support. Funding was provided by DFFE. We thank SAPFIA for funding a workshop social function.

Penguins on the rocks, Boulders (photo BM Dyer)

APPENDIX A

PANEL BIOGRAPHIES

Robert W. Furness is Principal Ornithologist at MacArthur Green, working mainly on impacts of offshore wind farms on seabird populations and appropriate compensation measures. He was previously Professor of Seabird and Fishing Interactions at the University of Glasgow, Scotland. His scientific publications have been important in influencing policy in marine conservation with over 33,800 citations in Google Scholar and an H index of 100. As International Fish Meal and Oil Manufacturers' Association (IFOMA) Professor he developed plans for a sandeel closed box near kittiwake colonies while allowing continued sandeel harvest in offshore areas of the North Sea with no reduction in the total allowable catch. That closed box was implemented by the European Commission in 2000 and is still functioning. He was appointed by Scottish Government Ministers to the Board of Scottish Natural Heritage (SNH), the statutory adviser to Scottish Government on wildlife conservation and management, where he has played a role in developing government policy in wildlife conservation and management. He chaired SNH's Scientific Advisory Committee, a panel of experts drawn from Universities and Research Institutes to assess the science underpinning conservation policy. He has served as a member of the JNCC Marine Subgroup, developing UK policy on marine protected areas. Professor Furness chaired the International Panel of Experts in Marine Ecology (IAPEME) set up by the Danish Government to review the scientific programme monitoring impacts of the world's first major offshore wind farms (Nysted and Horns Rev). He has chaired several panels of experts set up by the International Council for the Exploration of the Sea (ICES) to advise the European Union on fisheries management, including Benchmark Workshops on sandeel stock assessment and management, assessment methods for short-lived fish stocks (e.g. sprat), and the ICES Working Group on seabird ecology. He was a member of the Marine Stewardship Council certification panels for several important North Pacific fisheries, including Western Bering Sea and Gulf of Alaska walleye pollock, Canadian sablefish, Pacific halibut, and Alaskan sablefish.

Ana M. Parma is a Principal Scientist with the National Scientific and Technological Research Council of Argentina (CONICET), working at the Center for the Study of Marine Systems, the National Patagonic Center in Puerto Madryn. Argentina. She earned her Ph.D. in Fisheries Science in 1989 from the University of Washington, and worked as an assessment scientist at the International Pacific Halibut Commission until 2000, when she returned to Argentina, her home country. Dr. Parma has worked on different aspects of fisheries modelling, assessment and management of a diverse range of fisheries, from artisanal coastal fisheries targeting benthic shellfish to large-scale international fisheries targeting tunas. The main focus of her research has been on the evaluation and design of management strategies, attending to ecological and institutional dimensions, both in data-rich and in data- and capacity-limited contexts. She was awarded a PEW Fellowship in Marine Conservation and a Guggenheim Fellowship for her contributions in this field. Dr. Parma has always worked at the interface between science and management, being involved in several scientific and policy advisory boards and review panels at the national and international levels. She co-chaired a National Research Council (NRC) panel on evaluating the effectiveness of fish stock rebuilding in the United States, was a member of four other NRC committees focused on diverse aspects of stock assessments, marine protected areas and ecosystem-based fisheries management, and was a lead author of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) Assessment Report on the Sustainable Use of Wild Species. She chairs the modelling group of the Commission for the Conservation of Southern Bluefin Tuna, is a member of the Steering Committee of the bluefin tuna program at the International Commission for the Conservation of Atlantic Tunas, and until recently was a member of the Science Council and Global Board of The Nature Conservancy. She has been part of several of the international panels that regularly review South African stock assessments and management strategy evaluations, including those that advised on the design and analyses of the penguin Island Closure Experiment.

Éva Plagányi is a Senior Principal Research Scientist at CSIRO based in Brisbane, Australia since 2009, where she is also Domain leader for Climate Impacts and Adaptation. She has broad experience working on a range of natural ecosystems, mostly marine, from tropical through temperate to Antarctic systems, and focused on species ranging from plankton, sea cucumbers, crustaceans, fish to whales, with a strong focus also on climate change impacts. Her specific skills relate to using mathematics and mathematical models to model the dynamics of natural populations to support sustainable management of natural resources and contribute to conservation outcomes. Her research focuses on stock assessment modelling, ecosystem modelling and management strategy evaluation (MSE) and she has collaborated widely internationally, plus is currently a scientific member of three Australian Resource Assessment Groups: Torres Strait tropical rock lobster, Torres Strait Hand Collectable Fishery and the Northern Prawn Fishery. She has pioneered the approach she coined MICE (Models of Intermediate Complexity for Ecosystem assessments) which is being used to underpin natural resource decision-making in Australia and globally. Dr Plagányi has a joint mathematical-biological background and after completing a Masters degree in the Zoology Department of the University of Cape Town (focussed on anchovy-copepod interactions), she completed a PhD in 2004 from the Department of Mathematics and Applied Mathematics. Before relocating, she was awarded a National Research Foundation (NRF) President's Award and was a senior lecturer in the latter department, whereafter she has remained an Honorary Research Associate of the University of Cape Town. She was a member of the Lenfest Forage Fish Task force (2009-2014) which focussed on global recommendations for forage fish management. She has published over 120 peer-reviewed papers, ca.400 technical reports, 50 popular articles, and is on the editorial board of Ecological Applications and Reviews in Fish Biology and Fisheries.

André E. Punt is a professor in the School of Aquatic and Fishery Sciences at the University of Washington, Seattle, USA and a past Director of the School. He received his B.Sc, M.Sc and Ph.D. in Applied Mathematics at the University of Cape Town, South Africa. Before joining the University of Washington, Dr Punt was a Principal Research Scientist with the CSIRO Division of Marine and Atmospheric Research in Australia. Dr. Punt has been involved in stock assessment and fisheries management for over 35 years and has been recognized for his contributions in this area with awards from the CSIRO, the University of Washington, the Australian Society for Fish Biology, and the American Fisheries Society. The research undertaken by Dr. Punt and the MPAM (Marine Population and Management) group at the University of Washington relates broadly to the development and application of fisheries stock assessment techniques, bioeconomic modelling, and the evaluation of the performance of stock assessment methods and harvest control rules using the Management Strategy Evaluation approach. Currently, projects that Dr. Punt is undertaking with his research group include ecosystem modelling, assessment and management methods for data-poor fisheries, and understanding the impact of climate change and environmental variation on the performance of assessment and management methods. Dr. Punt has conducted stock assessments for a wide range of species, ranging from anchovies and sardines, to groundfish, tunas, and cetaceans. He has published over 420 papers in the peer-reviewed literature, along with over 400 technical reports. He was a member of a National Research Council panel on evaluating the effectiveness of fish stock rebuilding in the United States and has been a member of the panel that reviews aspects of South African stock assessments and management strategy evaluations since its establishment. Dr Punt is currently a member of the Scientific and Statistical Committee of the Pacific Fishery Management Council, the advisory committee for the Center for the Advancement of Population Assessment Methodology, the Standards and Petitions Committee for the International Union for the Conservation of Nature, the Crab Plan Team of the North Pacific Fishery Management Council, and the Scientific Committee of the International Whaling Commission.

James N. Sanchirico is a professor of natural resource economics and policy in the Department of Environmental Science and Policy at the University of California at Davis. His main research interests are the economic analysis of policy design, implementation, and evaluation for marine and terrestrial species conservation, and the development of economic-ecological models for forecasting the effects of resource management policies. Before coming to UC Davis, he was a Senior Fellow at Resources for the Future (RFF) in Washington DC (non-profit think tank on environmental and natural resource policy) and is currently a University Fellow of RFF. He received the Rosenstiel Award for Oceanographic Sciences in 2012 given to researchers who have made outstanding contributions to Ocean Science, the UC Davis Distinguished Scholarly Public Service Award in 2014, and the 2023 Publication of Enduring Quality Award from the Association of Environmental and Resource Economists. He is currently co-Editor at the Journal of the Association of Environmental and Resource Economists and the Journal of Environmental and Resource Economics, member of the U.S. National Academies of Science Ocean Studies Board, chair of the U.S. National Academies Standing Committee on Offshore Wind Energy and Fisheries, and principal investigator on the NSF-funded Sustainable Oceans National Research Training program at UC Davis. Past professional service includes the Lenfest Fishery Ecosystem Task Force, a National Research Council (NRC) committee evaluating the effectiveness of stock rebuilding plans of the 2006 Fishery Conservation and Management Reauthorization, and six years on NOAA’s Science Advisory Board.

Philip Trathan has an extensive publication record on the biology, ecology and management of marine ecosystems in the Southern Ocean. He has wide experience and an established track record of interdisciplinary research, commissioned research and international collaboration. Before retiring from the British Antarctic Survey in 2022, Phil had authored 286 peer-reviewed publications, giving him an academic H-index of 56. Whilst at BAS, Phil was Head of Conservation Biology, a diverse group of marine predator scientists that undertook ecological research on a wide variety of Southern Ocean seabirds and marine mammals. Key components of this research focused upon food-web interactions (e.g. foraging ecology, energetic requirements), critical constraints on life histories (e.g. environmental impacts, breeding performance), circumpolar connections (e.g. post-breeding dispersal) and anthropogenic impacts (e.g. climate change, and resource competition with fisheries). The Conservation Biology group managed BAS long-term monitoring of key marine predators in the Antarctic and Sub-Antarctic. The group provided national capability and advocacy for science into Antarctic policy arenas, including for the Commission for the Conservation of Antarctic Marine Living Resources, the International Whaling Commission, and the Agreement for the Conservation of Albatrosses and Petrels. Phil was personally involved with CCAMLR for over 30 years, providing advice on the management of the fishery for Antarctic krill, on the designation of Scientific Special Areas, on Marine Protected Areas, and on climate change. Phil was the UK’s senior ecological advisor to CCAMLR for more than 16 years. Phil participated in 20 Antarctic field trips, having been involved in predator studies (primarily penguin species, but also marine mammals) for over 20 years. He is particularly interested in how marine predators utilise their available habitat and how this relates to their reproductive output and performance. He has strong links with BirdLife International and WWF (UK) and is a member of the IUCN Species Survival Commission Penguin Specialist Group. Phil was the founding Chair of the Science Advisory Group, Antarctic Wildlife Research Fund, a novel collaboration between the krill fishing industry, NGOs and scientists. Phil is now a Visiting Professor at Ocean and Earth Science, National Oceanography Centre, Southampton, he is a BAS Emeritus Fellow, and an Honorary Fellow of Bangor University. Phil was honoured with an OBE in 2018 for his contribution to Antarctic science and conservation.

APPENDIX B

EXTRACTS FROM THE TERMS OF REFERENCE

1. BACKGROUND

In the mid-2000s, a substantial decrease in the numbers of adult African Penguins was observed off western South Africa. In response to this observed decrease from 2006 and the potential impact of food competition between penguins and fishers in the vicinity of breeding islands, a study to assess the effects of closure to purse-seine fishing around penguin breeding colonies was initiated in 2008. Since the study required income sacrifice from the industry, this study, the Island Closure Experiment (ICE), comprised two parts: (i) a feasibility study (2008– 2014) during which purse-seine fishing was prohibited in an alternating pattern around two pairs of nearby colonies and data on penguins (as well as on small pelagic fish from the routine pelagic fish management process) were collected to determine whether an experiment would have adequate statistical power to detect a significant effect of closure if such existed; and (ii) an experimental phase (2015–2019) where these alternating island closures were continued with the associated continuation of the monitoring during the feasibility study. The results, however, led to a lengthy debate with dichotomous views. The plans for and results of the ICE were regularly reviewed by DFFE's Small Pelagic Scientific Working Group, informed by the advice provided from an annual review, i.e., a DFFE review meeting of world-leading quantitative marine resource scientists on ten occasions since 2006. Most recently, the scientific results have been debated in the peer-reviewed literature (Sydeman et al. 2021, Butterworth and Ross-Gillespie 2022, Sydeman et al. 2022).

A Governance Forum (GF), comprising researchers and managers from the Branches: Oceans and Coasts and Fisheries Management as well as SANParks (South African National Parks), was established in 2021. The aim was to prepare a comprehensive Synthesis Report on the current state of knowledge relating to African Penguins, island closures, fisheries management relevant to African Penguins and the socioeconomics of island closures and penguin-related tourism. The Governance Forum compiled a report titled "A Synthesis of Current Scientific Information Relating to the Decline in the African Penguin Population, the Small Pelagic Fishery and Island Closures" (DFFE 2021) which collated science over the last decade on penguins, small pelagic fisheries and their interactions including the Island Closure Experiments. The Synthesis Report was further scrutinized by two independent reviewers who provided extensive comments; the Governance Forum's Extended Task Team (which added fishing industry and conservation NGO representation to the Governance Forum) and then the Minister's Consultative Advisory Forum for Marine Living Resources (CAFMLR). Comments on that Synthesis Report and recommendations produced by these groups remain contested.

The Department now seeks to establish an international Panel of Experts to—

- a) review the interpretation of the ICE
- b) explore the value of island closures in providing meaningful benefits to penguins
- c) review the processes and outcomes completed through the GF and the CAFMLR process
- d) make recommendations on the implementation of island closures, including spatial delineation, time frames and
- e) advise on further science and monitoring methods.

2. OBJECTIVES

The International Review Panel will—

- a) Review the quantitative scientific analyses of the Island Closure Experiment (ICE) and subsequent publications to evaluate whether the scientific evidence from ICE indicates that limiting small pelagic fishing around colonies provides a meaningful improvement to penguin parameters that have a known scientific link to population demography in the context of the present rate of population decline. Assess the cost-benefit trade-off of 1) costs to fisheries, versus 2) the proportion of penguin foraging range protected during the breeding season, for different fisheries exclusion scenarios. The losses to the fishery should be fleshed out using available economic information, such as was used in the GF and CAF processes. The panel may also comment on the limitations of available information and methods (data collection) to improve the assessment of positive penguin outcomes as well as fishery impact. Costs to fisheries must include an assessment of replacement costs accrued during periods closed to fishing during the ICE.
- b) Within the context of an urgent need to implement timeous conservation actions for the African Penguin and considering the information and rationale of the various scientific reviews and associated documents of the Island Closure Experiment evaluate the evidence supporting the benefits of fishery restrictions around African Penguin colonies to adopt precautionary measures by implementing long-term fishery restrictions.
- c) If closures or fishing limitations are viewed to contribute positively to the support of the African Penguin population, recommend a trade-off mechanism as a basis for setting fishing limitations and mapping. This mechanism must consider a potential positive return to penguins and the impact on fisheries. (As a basis for discussion the Governance Forum Approach and the CAF approach can be considered.) Consideration must also be given to the current state of observations, data and analyses (Penguin, Environmental and Fisheries Economic data). Recommendations on these can be included under future science considerations.
- a. Delineation of fishery no-take areas around six African Penguin colonies (Dassen Island, Robben Island, Dyer Island, Stony Point, St Croix Island and Bird Island) and the duration of the closures, considering life history traits, e.g., age when most

birds start breeding, and associated duration required to signal potential population benefits.

d) Recommendations on the scientific work that is required to evaluate the effectiveness of such no-take areas.

e) Recommendations about what scientific work is appropriate in the short term to determine the dominant causes of the rapid and concerning rate of decline of the penguin population, including recommendations about the use of ecosystem model approaches such as MICE (models of intermediate complexity for ecosystem assessments).

3. PANEL PROCESS AND PROCEDURES

a) The panel should attempt to reach a consensus but if not achieved, names supporting each of the alternative views should be noted. There should be no voting.

b) Virtual and physical meetings are not prescribed at this stage. One option is to have one or two brief virtual meetings to familiarise the panel with the key issues, followed by a week-long physical meeting in Cape Town to wrap it up. Travel expenses will be covered by DFFE. [Panel members may opt to join the weekly session virtually if travelling is not preferred.]

c) Members of the Panel of Experts will be remunerated in accordance with the Republic's Public Finance Management Act, 1999 (Act No.1 of 1999) and the associated Treasury Regulations, and in particular, according to the remunerative structure for non-official members of Commissions and Committees of Inquiry in consultation with the Minister of Finance for this panel's proposed work.

d) Meetings may include closed meetings, meetings with protagonists separately and together.

e) DFFE will appoint the Chair of the Panel and the Chair will report directly to the Minister.

f) DFFE will provide secretarial services.

4. TASKS

The following tasks are required from the panel (administrative and secretarial functions will be supported by DFFE):

a) Panel Members must agree to being available and accepting these Terms of Reference and constitute themselves as a Panel with the Chair.

b) Notification of stakeholders about deadlines for their submissions.

c) Drawing up of a list of attendees at plenary meetings where submissions are heard, indicating who are key participants and who are observers (Sectors will be asked to submit names of observers to be invited).

d) The appointed Panel Members to meet with DFFE Senior Managers to clarify their tasks and outputs.

e) Review documents and information pertaining to proposed island closures for penguin population recovery support. While these will initially be composed of an agreed selection (by local scientists and stakeholders) from the extensive number of documents produced over the last 1.5 years, panel members may request any additional documents such as scientific working group documents. Documents to be categorised into (a) those relevant to the interpretation of the ICE results, (b) documents that propose island closures including stakeholder reports submitted during the ETT and CAFMLR processes and (c) other related documents. This is required to facilitate the panel dividing its focus between

    i. an initial assessment of whether the analysis of ICE supports the view that island closures will benefit penguins, and
    ii. if (i) suggests that island closures will benefit penguins, what closures should be implemented, or what are the trade-offs involved for such closures.

f) Meet with conservation and fisheries sector scientists and where each will be allowed to present their arguments/interpretation of information. (At panel discretion, other scientists, and experts may be invited to make presentations.)

g) Respond to objectives (a) to (e) above.

h) Prepare report on outcomes.

5. OUTCOMES AND RECOMMENDATIONS

a) Recommend whether, based on the results from ICE and other evidence-based information, island closures are likely to benefit penguins.

b) Describe the scientific and evidence-based rationale for recommending implementing/not implementing fishing limitations around penguin colonies

c) Make recommendations about whether a percentage (%) of penguin foraging range and other biological criteria (such as regional representation, population recovery potential, monitoring and evaluation potential) provide a basis for determining benefits from closures for penguins and assess the merits of different proposed methods to delineate important penguin foraging habitat.

d) Make specific recommendations on trade-off mechanisms for island closures in the event that the panel finds that the results of ICE and other evidence demonstrate that island closures are likely to benefit penguins, including specific areas and durations. In addition to recommendations on trade-off mechanisms, the panel must preferably advise on biologically meaningful penguin habitat extents for fishery limitations per island, recommendations must be spatially and temporally explicit, and provided on a map. [DFFE will provide mapping capacity.]

e) Provide advice and recommendations on best estimates and uncertainties of the ratio between penguins gained and losses sustained by the industry as a result of island closures for future suggested closure options.

f) Provide advice on a well-structured analyses framework to monitor the impact of island closures, including what penguin and fish data needs to be collected; how benefits to penguins are to be determined; and how these will be analysed.

g) To recommend scientific analyses, including but not limited to MICE, to determine the reasons for the decline in the penguin population.

APPENDIX C

DEPLOYMENT OF NEST BOXES FOR AFRICAN PENGUINS

At Bird Island in the 2000s, only about 1% of African penguins bred in natural burrows in the remaining patches of guano, so the majority of nests appear to be in suboptimal nesting habitat (Lei et al., 2014). In an effort to mitigate the impacts of guano removal, artificial nest sites (nest boxes) of a variety of designs and materials have been constructed for African penguins at a number of colonies, including Marcus Island (Saldanha Bay), Halifax Island (Namibia), Dyer Island, Boulders Beach, and Robben Island (Western Cape), Stony Point (Betty's Bay), and Bird Island (Algoa Bay) (Sherley et al., 2012; Espinaze et al., 2020). These were first developed in the 1980s by Wilson and Wilson (1989) at Marcus Island and had some success in improving African penguin breeding success. Penguin nest boxes have also been used successfully to increase breeding success of little penguins in New Zealand and Australia (Perriman and Steen, 2000; Sutherland et al., 2014). Sutherland et al. (2014) concluded that 92% of nest boxes installed for more than 6 years for little penguins at Phillip Island, Australia, were occupied, and that nest boxes increased survival of eggs to hatching by 8%, increased survival of chicks to fledging by 9%, and increased fledging weights of chicks (which is likely to increase post-fledging survival) by 11%, leading to a significant local increase in breeding numbers.

At Robben Island, penguin nest boxes were installed (22 triangular plywood boxes in 2001 and a further 37 in 2005 and 10 in 2010, plus 70 fibreglass curved boxes in 2007) and the breeding success of penguins in nest boxes and in other nest sites was monitored each year (Sherley et al. 2012). There was no difference in hatching or fledging success between wooden and fibre-glass nest boxes. Relative to pairs in nests under vegetation, birds nesting in the open had significantly lower egg survival during incubation, but egg survival was no different between birds under vegetation and birds in nest boxes. However, the chicks of birds occupying nest boxes and nests in abandoned buildings had higher survival than chicks in nests under vegetation, with about 10% more chicks fledging per egg laid from nests in nest boxes (Sherley et al., 2012). Chick survival was also higher in nest boxes than in surface nests and nests under shrubs during the chick-guarding stage on Halifax Island (Sherley et al., 2012). Sherley et al. (2012) concluded that "provision of artificial nests can improve breeding productivity for penguins nesting in temperate climes and could help stem the decline of the African penguin".

At Bird Island, some nest box designs provide protection from predators but trap heat and have adverse effects on penguin breeding success (Welman and Pichegru, 2023) and in some cases have now been removed and replaced with new designs intended to perform better. A double-layered ceramic nest chamber installed at Bird Island since 2018 appears to perform better than exposed surface nests, cement nest boxes, or natural nests, by overheating less and by maintaining higher humidity (Welman and Pichegru, 2023). However, penguin breeding success has not yet been compared between ceramic nests and other nests, so the gain in breeding output from such nests is uncertain.

At Stony Point, African penguin adults and chicks were on average heavier in artificial nest boxes than in open nests but for the sample nesting in nest boxes were less heavy in nest boxes with highest soil temperature (Espinaze et al., 2020). There is evidence that ectoparasite abundance can be higher in penguin nest boxes that are warmer and drier than other penguin nests (Espinaze et al., 2020). Fibreglass and cement-fibre nest boxes established at Stony Point in the 2010s had higher soil temperatures and lower relative humidity than did penguin nests under bushes, and held larger numbers of ticks and fleas (Espinaze et al. 2020) and so design of penguin nest boxes needs to consider not only the breeding success achieved by penguins in boxes compared to those in other nest types, but also how penguins might be affected by ectoparasites and stress in boxes that tend to overheat and dry out. Espinaze et al. (2020) suggest that glassfibre, concrete, and other non-porous material nest boxes for African penguins should be re-evaluated and that it may be better to construct nest boxes from much more porous material and with better ventilation designed into the structure.

Triangular nest boxes, Robben Island (photo BM Dyer)

Ceramic nest boxes, Boulders (photo BM Dyer)

APPENDIX D

TECHNICAL SPECIFICATIONS OF MODELS USED TO ANALYSE THE ICE DATA

1. Mixed-effect models used to estimate fishing impacts on penguin reproductive success
Two main classes of mixed-effect models were used, referred to as closure-based and catch-based. Technical specifications are provided below:

1.1 Closure-based models:

The model equation for the closure-based estimator applied to the aggregated data was:

$$f(F_{y,i}) = \propto_0 + \propto_1 I_i + \propto_2 X_{i,y} + \propto_3 I_i X_{i,y} + Y_y + \varepsilon_{i,y} \qquad (1)$$

where $F_{y,i}$ is the average response variable for year y and island i, possibly log-transformed depending on the data source, i = 1,2 is the Island, y = 2008,…,2019 is the Year, $X_{i,y}$ is a binary for the treatment (open = 0, closed = 1) applied at island i during year y, I_i is a binary for the colony (Dassen = 0, Robben = 1 or Bird = 0, St Croix = 1), $\propto_0$, $\propto_1$, $\propto_2$, $\propto_3$ are fixed effects ($\propto_1$ is an island effect, $\propto_2$ is a fishing effect applied when the area around the colony is open, and $\propto_3$ is the treatment × Island interaction), Y_y is a year random effect, and $\varepsilon_{i,y}$ is the residual error.

Details about how the various response variables were pre-processed are provided in Ross-Gillespie and Butterworth (2021a) and Butterworth and Ross-Gillespie (2022).

Models applied to disaggregated data included the same fixed effects, but the random effects varied depending on the response variable.

For chick condition, the random structure requested by the Panel included a Year effect plus Month nested within Year, plus the Island nested within Month and Year.

$$y_{i,y,k,l} = \beta_0 + \beta_1 x_{i,y} + \beta_2 z_l + \beta_3 X_{i,y} z_l + b_y + b_{y,k} + b_{y,k,i} + \varepsilon_{y,k,i,l}$$

where $y_{i,y,k,l}$ is the condition of individual chick l in year y, island i and month k, i = 1,2 is the Island, y = 2008,…, 2019 is the Year, k = 1,…,K is the Month, $X_{i,y}$ is a binary for the closure treatment (open = 0, closed = 1) applied at island i during year y, z_l is a binary for the colony (Dassen = 0, Robben = 1) chick l belongs to, β_0, β_1, β_2, β_3 are fixed effects and b_y, $b_{y,k}$, $b_{y,k,i}$ are random effects, b_y ~Normal$(0, \sigma_1^2)$, $b_{y,k}$ ~Normal$(0, \sigma_2^2)$, $b_{y,k,i}$ ~Normal$(0, \sigma_3^2)$, and $\varepsilon_{y,k,i,l}$ ~Normal$(0, \sigma_\varepsilon^2)$ is the residual error.

In R lmer syntax:

Condition ~ Island/Closure+(1|Year)+(1|Year:Month) +(1|Year:Month:Island)

The significance of the Island × Closure interaction was evaluated by comparing the full model with one where $\beta_3 = 0$ using maximum likelihood (Sherley, 2023).
For chick survival, equation 2 in Shirley (2023) gives the mean hazard function as:

$$\Lambda_{y,i,n,l} = \beta_0 + \beta_1 x_y + \beta_2 z_i + \beta_3 x_y z_i + \omega_y + \omega_{y,i} + \omega_{y,i,n}$$

where n is nest ID, β_0, β_1, β_2, β_3 are fixed effect parameters, and ω_y ~Normal$(0, \sigma_1^2)$, $\omega_{y,i}$ ~Normal$(0, \sigma_2^2)$ and $\omega_{y,i,n}$ ~Normal$(0, \sigma_3^2)$ are random effects for Year, Year × Island and Year × Island × NestID, respectively.

1.2 Catch-based models:

The model equation for the catch-based estimator applied to the aggregated data was:

$$f(F_{y,i}) = \beta_0 + \beta_1 I_i + \beta_2 C_{i,y} + \beta_3 I_i C_{i,y} + Y_y + \varepsilon_{i,y} \qquad (2)$$

where $C_{i,y}$ is the catch (of anchovy and/or sardine) taken within the 20-km area around island i during year y and other variables are as defined for equation (1). Parameters β_0, β_1, β_2, β_3 are fixed effects, the last corresponding to the Catch × Island interaction. A simpler model with a common catch effect for the two paired islands ($\beta_3 = 0$) was suggested for the east colonies given the observed negligible catches around Bird Island except during the early years. For such a model, catches need to be either in absolute values (as in equation (2)), or normalised using a common average catch for the island pair.

Once the parameters are estimated, the effect of fishing around colony i on the response variable (to be translated into the effect of keeping island i open on the island's penguin population growth rate) is predicted using:

$$\Delta y_i = \beta_1 \bar{C}_i + \beta_3 I_i \bar{C}_i \tag{3}$$

where $\bar{C}_i$ is the average catch taken around island i during years when fishing around that island was allowed. Using as predictor the average catch over open years would afford consistency with the closure-based estimator.

The formulation above differs from the catch-based estimators used in the past (e.g., Ross-Gillespie and Butterworth, 2016b) where catches used as covariates were normalised with respect to the average catch taken within each island closure during the years when the island was open.

The effect predicted from equation (3) would be equivalent to the λ_i effect estimated in those previous catch-based analyses that used normalised catches only when a catch × Island interaction is included (i.e., $\beta_3 \neq 0$).

2. Subset of models selected to provide final estimates of fishing impacts on penguin population growth rate

Figures 2.2 and 2.3 show results for a subset of the models presented by Sherley (2023) and Ross-Gillespie and Butterworth (2023b). Tables D.1 and D2 provide a summary of the characteristics of those selected models. Further details about the data preprocessing and the estimation procedures are described in Sherley (2023) and Ross-Gillespie and Butterworth (2023b).

Table D.1: Details of the models applied to the ICE data from Dassen and Robben islands whose results are reported in Figure 2.2.

Model	Response variable	Data aggregation	Fixed effects	Random effects	Reference
W1	Chick condition	Disaggregated	Island+Closure	Year + Year:Month + Year:Month:Island	M6 in Sherley (2023)
W2	Chick condition	Disaggregated	Island×Closure	Year + Year:Month + Year:Month:Island	M5.1 in Sherley (2023)
W3	Chick condition	Aggregated	Island×Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
W4	Chick survival	Disaggregated	Island+Closure	Year + Year:Island + Year:Island:Nest	M9 in Sherley (2023)
W5	Chick survival	Disaggregated	Island×Closure	Year + Year:Island + Year:Island:Nest	M8 in Sherley (2023)
W6	Chick survival	Aggregated	Island×Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
W7	Fledging success	Aggregated	Island×Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
W8	Chick growth	Aggregated	Island×Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
W9	Maximum distance	Aggregated foraging	Island×Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
W10	Path length	Aggregated	Island×Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
W11	Trip duration	Aggregated	Island×Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)

Table D.2: Details of the models applied to the ICE data from St Croix and Bird islands whose results are reported in Figure 2.3.

Model	Response variable	Data aggregation	Fixed effects	Random effects	Reference
E1	Chick condition	Disaggregated	Closure	Year + Year:Month + Year:Month:Island	M7E in Sherley (2023)
E2	Chick condition	Disaggregated	Island + Closure	Year + Year:Month + Year:Month:Island	M6E in Sherley (2023)
E3	Chick condition	Aggregated	Island × Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
E9	Maximum foraging distance	Aggregated	Island × Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
E10	Path length	Aggregated	Island × Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)
E11	Trip duration	Aggregated	Island × Closure	Year	S1 in Ross-Gillespie & Butterworth (2023b)

Penguin nest, Dassen Island (photo BM Dyer)

APPENDIX E

ADDITIONAL DETAILS ON THE OBM AND WHY ITS RESULTS ARE LIKELY OVERESTIMATES

The Panel concluded that the OBM likely overestimates the effects of closures on lost catches given the algorithms used to decide whether a catch in a proposed closure area can be replaced or not. The Panel was less concerned with the method used to replace a catch when it is replaceable (and endorsed the “random” approach).

For each set made in a closure area when the area was open the algorithm involves searching the areas within which it can replace the “lost set”. If there were no sets outside the closure area made on the same day (and in the area considered to be where a replacement set can occur) the set is considered to be irreplaceable. An example of this case is given in Figure E.1. Note that the catches off Dassen Island in Figure E.1 might not be considered irreplaceable if a longer window of time was available (see, e.g., the discussion on the development of expected catches in the RUM subsection in Section 3), and sensitivity is shown in some OBM analyses to a 2-day window rather than only allowing sets on the same day to replace sets in a closure area. A second cause of irreplaceable catches arises when considering how to match the outside sets with the inside sets (with or without replacement). Specifically, even when there are sets outside of the closed area that could be matched with an inside set, it is possible that the inside set is irreplaceable because there is a limit (base case 5) on how often a set outside a closure can replace a set inside a closure area. An example of this case is given in Figure E.2.

The effects in Figures E.1 and E.2 would not be a concern if the proportion of the catch lost due to the set being irreplaceable (i.e., “irreplaceable catch”) was small relative to the catch lost due to catch rates being lower in the alternative sets (i.e., “opportunity loss”), but this is not the case, particularly when the closure area is large (e.g., closures based on mIBA (7 km)). Figure E.3 and Table E.1 illustrate this for a selected set of OBM scenarios and closure proposals. Results correspond to estimated catch losses for anchovy and for directed sardine, summed over the six islands included in the analysis. Several features of the results in Table E.1 are pertinent to note:

- The catch in the closure area (“inside catch”) varies substantially among the closure options (largest for mIBA (7 km) and least for “industry”).
- The catch that is lost due to being unreplaceable ranges from 8.7% to 91.8% of the total lost catch among OBM scenarios and the closure size, and is larger than 50% for some of the closure options (mIBA (7km), mIBA (ARS), and DFFE).
- There is considerable sensitivity of the unreplaceable catch (particularly for the larger closure areas) depending on whether a set can be reused as many times as needed, 10 times. 5 times or only once.
- The irreplaceability percentage is lower when catches on one day can be replaced by catches on the next day (scenario “Plus1day” in Figure E.3), but the effect is smaller than the effect of the reuse value.

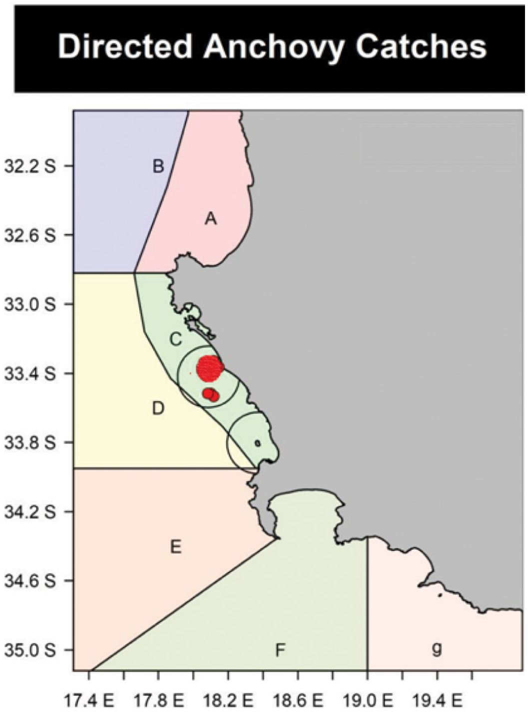

Figure E.1: A (hypothetical) example of catches off Dassen Island on a given day that would be “lost” owing to there being no sets outside the closure on that day.

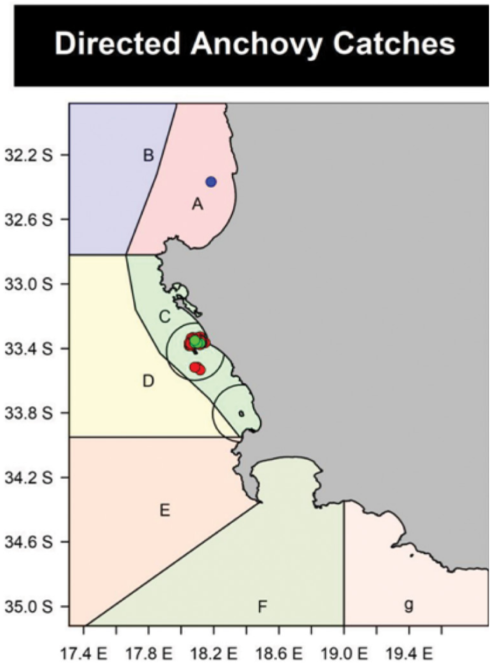

Figure E.2: A (hypothetical) example of catches off Dassen Island on a given day, some of which would be “lost” owing to there being sets outside the closure on that day, but the value of the “reuse” parameter does not allow all of the catches in the closure area to be replaced.

Table E.1: Example results from the OBM. Results are shown for several closure options and several ways to apply the OBM. Blank cells indicate the results concerned are not available.

ANCHOVY	Inside Catch				
Total catch	MIBA (7 km)	MIBA (ARS)	DFFE	CAF	Industry
206 695	65 081.3	44 061.3	20 444.6	12 941.0	2 312.7

	Irreplaceable Catch					Opportunity Loss					Irreplaceability %				
Model	MIBA (7 km)	MIBA) (ARS)	DFFE	CAF	Industry	MIBA (7 km)	MIBA (ARS)	DFFE	CAF	Industry	MIBA (7 km)	MIBA (ARS)	DFFE	CAF	Industry
BC(Random)	40 354.7		4 650.9	555.0	30.0	3 427.7		378.9	576.2	200.2	67.3%		24.6%	8.7%	10.0%
BC(median)	40 694.9	14 330.4	4 703.9			3 820.6	4 427.9	1 723.3			68.4%	42.6%	31.4%		
BC(median) Reuse = Inf	28 697.9	8 477.5	2 744.1	304.8	30.0	5 465.7	5 969.4	1 486.1	1 849.3	209.6	52.5%	32.8%	20.7%	16.6%	10.4%
BC(median) Reuse=1	52 683.6	25 699.3				1 911.6	2 064.4				83.9%	63.0%			
BC(median) Reuse = 10	36 349.9		3 648.2			3 889.8		1 900.0			61.8%		27.1%		
BC(median) + Next day	37 081.3		4 068.5			3 252.0		2 099.0			62.0%		30.2%		

SARDINE	Inside Catch				
Total catch	MIBA (7 km)	MIBA (ARS)	DFFE	CAF	Industry
61 985	33 413.5	17 554.1	7 539.0	2 058.7	436.5

	Irreplaceable Catch					Opportunity Loss					Irreplaceability %				
Model	MIBA (7 km)	MIBA (ARS)	DFFE	CAF	Industry	MIBA (7 km)	MIBA (ARS)	DFFE	CAF	Industry	MIBA (7 km)	MIBA (ARS)	DFFE	CAF	Industry
BC(Random)	26 989.1		3 002.6	463.4	436.5	950.5		493.8	223.5	13.6	83.6%		46.4%	33.4%	41.9%
BC(median)	27 013.1	6 837.6	3 085.4			1 234.9	1 074.1	733.0			84.5%	45.1%	50.6%		
BC(median) Reuse = Inf	25 122.8	5 832.4	2 645.7	463.4	436.5	1 730.0	1 462.2	808.7	319.1	42.1	80.4%	41.6%	45.8%	38.0%	48.5%
BC(median) Reuse=1	30 313.2	11 385.7				375.8	270.8				91.8%	66.4%			
BC(median) Reuse = 10	25 796.6		2 824.6			1 583.7		810.7			81.9%		48.2%		
BC(median) + Next day	25 796.6		2 343.4			1 529.4		942.1			81.9%		43.6%		

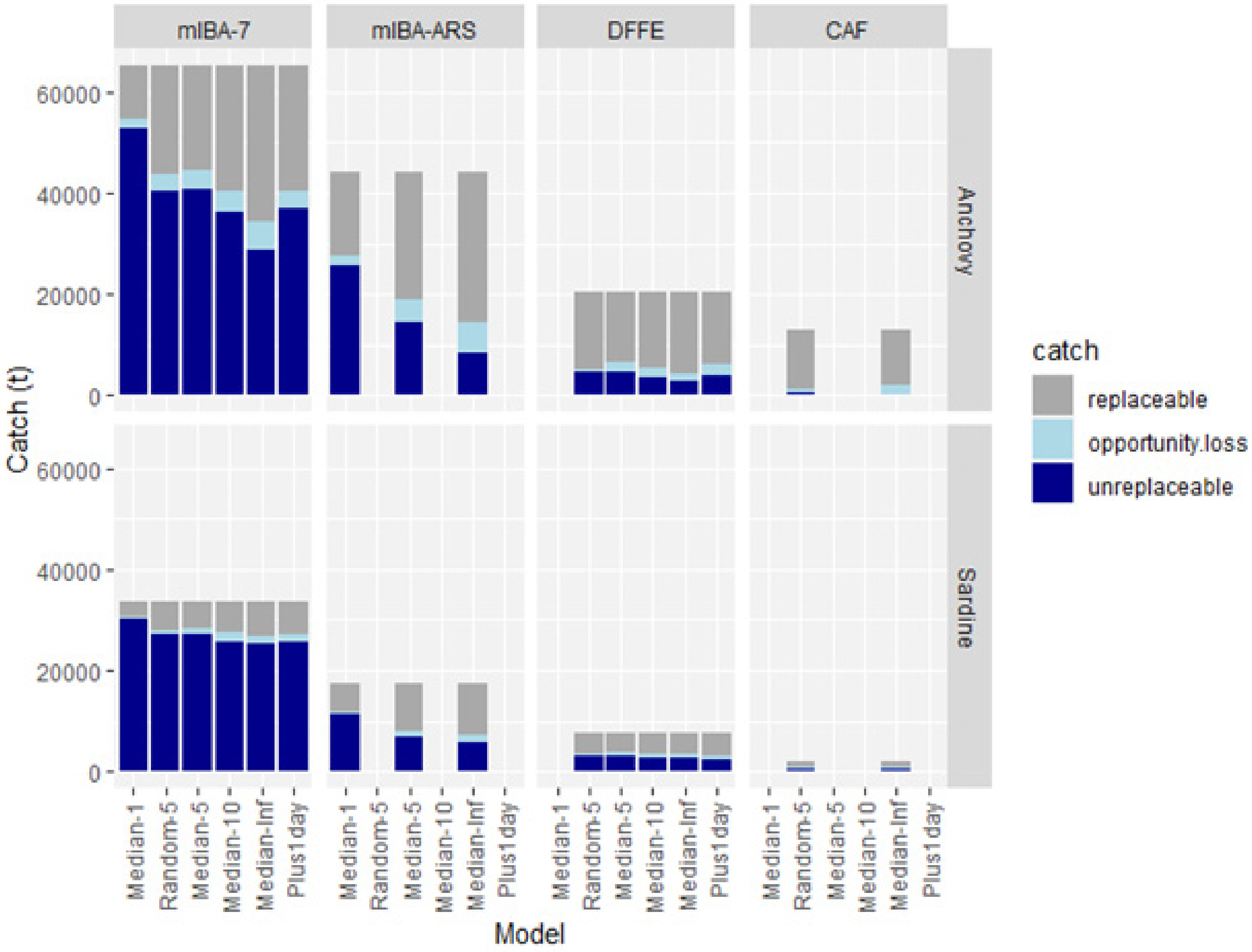

Figure E.3: Catch losses for anchovy and directed sardine estimated by the OBM for four closure proposals (mIBA (h = 7 km), mIBA-ARS, DFFE and CAF) using five model assumptions, four based on the median selection of alternative opportunities and one based on random selection, for Reuse = 1, 5, 10 and Inf (sampling with replacement) specifying the maximum number of times each alternative opportunity can be used as a replacement; the label "Plus1day" refers to the OBM scenario where a 2-day window is used instead of the same day to define the set of alternative fishing opportunities. The height of each stacked bar corresponds to the total annual catch taken inside each closure proposal ("inside catch" in Table E.1), a fraction of which (blue) is estimated to be unreplaceable, a small fraction (light blue) is lost due to lower average catch rates of the replacement sets, and the rest is replaceable (grey). Missing bars indicate the results concerned are not available.

APPENDIX F

OUTLINE OF MICE AND THEIR USE TO ASSESS DRIVERS OF THE DECLINE OF AFRICAN PENGUINS

F.1. Introduction

MICE (Models of Intermediate Complexity for Ecosystem assessments) are recognised as an appropriate tool to address complex science and management issues such as assessing the status of both fisheries and other non-targeted species, including those of high conservation concern, and evaluating the trade-offs among management plans aimed at addressing conflicting objectives (e.g., Tulloch et al., 2019; Goethel et al., 2022). MICE draw on the rigorous quantitative and statistical methodology of stock assessment approaches and extend this to representation of multiple co-existing species and stressors in an ecosystem. MICE have a tactical focus, are context- and question-driven and limit complexity by restricting the focus to those components of the ecosystem needed to address the main effects of the management question under consideration (Plagányi et al., 2014). Stakeholder participation and dialogue is an integral part of this process. MICE estimate parameters by fitting to data, use statistical diagnostic tools to evaluate model performance and account for a broad range of uncertainties. MICE aim to be based on the most appropriate balance between variance and complexity (Collie et al., 2014). These models therefore address many of the impediments to greater use of ecosystem models in strategic and particularly tactical decision-making for marine resource management and conservation.

F.2. A possible structure of an African penguin-centric MICE

The MICE should ideally include a regional sub-structure (i.e., separate western, eastern and southern regions) and be designed based on the data availability and being cognisant that a penguin-centric rather than fishery-centric approach is needed. If focused on a single region, based on data availability, the western region would be an ideal starting point with explicit representation of Dassen and Robben islands. Including paired islands would allow assumptions that some parameters are constant across islands thereby reducing confounding estimation of island-specific effects. Having smaller scale islands embedded in a larger scale model may also be helpful in analysing regional versus local impacts of changes in penguin prey availability, as well as the ability to explicitly model penguin inter-island movements. The key species that will need to be represented in the model include African penguins (age-structure formulation is needed – see Robinson et al. [2015] as an example), sardine, anchovy and Cape fur seals. Other species may be considered based on pre-agreed conceptual models describing plausible hypotheses as to their role as a competitor or predator. In general, it is recommended that MICE and similar ecosystem models be developed in a step-wise manner (Figure F.1) to ensure they remain tractable and only incorporate as much complexity as is needed to explain the available data.

Key processes to be investigated should similarly first be clearly identified via hypotheses and/or conceptual models of the system functioning. Using a structured, step-wise approach enables objective evaluation of the extent to which alternative hypotheses are consistent with, and able to explain, the available data. The model should be fitted to all available data to allow for consistency in assumptions whilst accounting for the uncertainty associated with different data sources and propagating this to the final outputs, as per accepted methods used in integrated analysis (Maunder and Punt, 2013).

In some cases, based on the overall system conceptual model, it may be helpful to develop complementary mechanistic models for more in-depth exploration of system functioning. The outputs of such a model can then be used to inform the functional relationships between different components in a MICE, with the latter being the integrated framework used to evaluate the plausibility of the interaction. For example, a bioenergetic model could be used to investigate how fishing around islands affects penguin foraging behaviour (including cooperative foraging in small groups), performance and travel distance (and hence net energetic budget) when compared with an equivalent no-fishing scenario, taking into account data such as foraging tracks, dive location, etc.

Additional modelling suggestions:

- Ultimately any model will only be as good as the underlying assumptions and the data available to inform them. The ICE has resulted in some very useful data, which needs to be integrated with data on penguin relative abundance as well as tagging and other data sources to inform on survival. Ideally a MICE should be constructed in an iterative fashion so that it is regularly updated with new data and information as these become available.
- A one-way interaction only between penguins and their prey needs to be assumed (i.e., penguin foraging will be assumed to have a negligible effect on their prey)
- As demonstrated in a number of existing MICE (e.g., Plagányi and Butterworth, 2012; Tulloch et al., 2019), it is not always essential to explicitly model the consumption of prey – rather the net effect of relative changes in available prey biomass can be tested as influencing breeding success and/or survival of different penguin stages.
- The relative abundance and energetic content of sardines and anchovy during different times of the year could be evaluated in relation to the peak timing of breeding and moulting of African penguins, as well as when fishing takes place. An annual time time-step may not provide sufficient resolution and it will likely be necessary to use a seasonal or monthly time time-step in the model, together with the role of environmental drivers, discussed below.

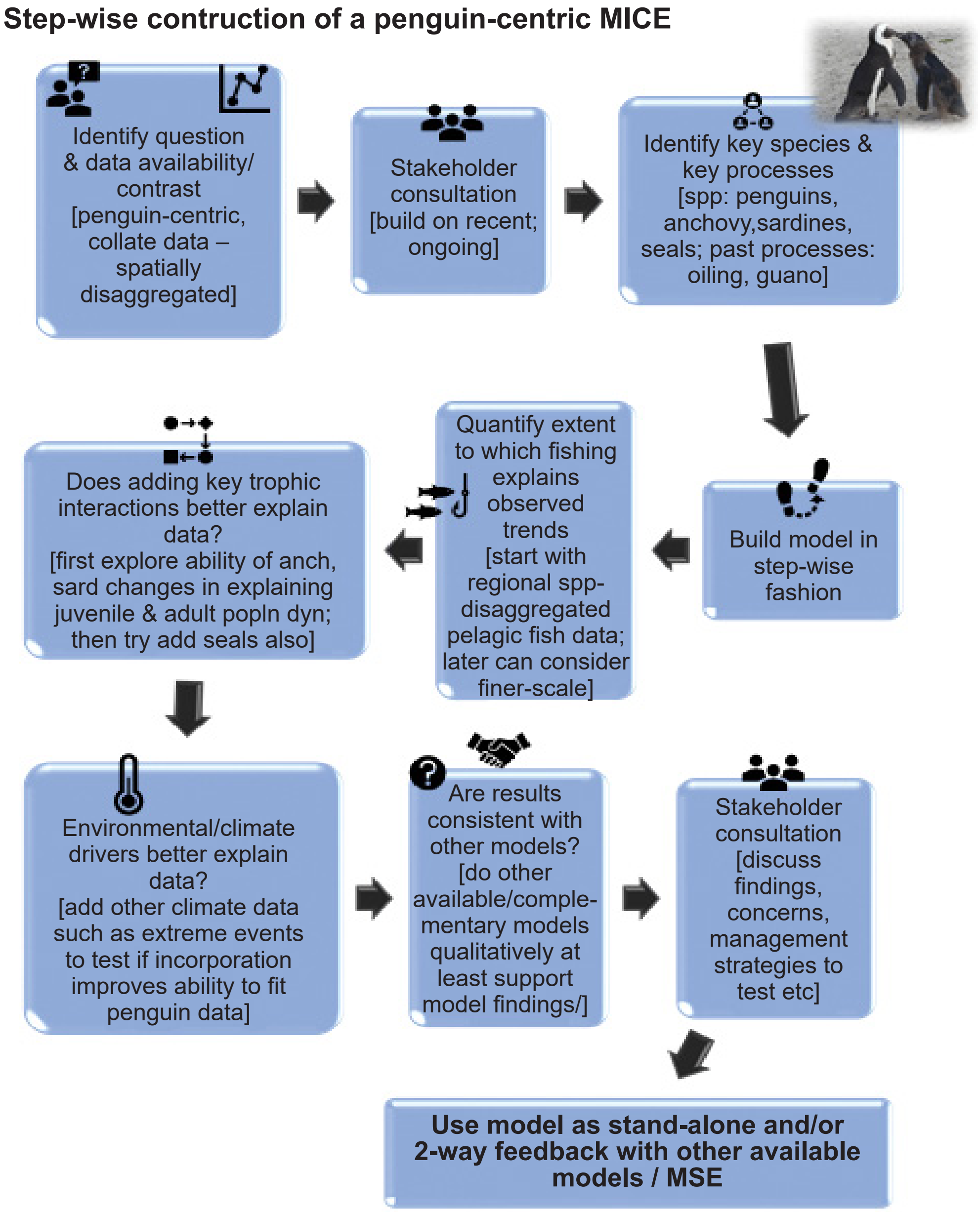

Figure F.1: Schematic summary of step-wise approach to building a MICE, modified from Plaganyi et al. (2022) with illustrative notes shown in square brackets

- Depending on the MICE structure, it would be helpful to distinguish between total regional prey abundance and local abundance (such as that which would theoretically be available within a mIBA(ARS) area), to evaluate match-mismatches between penguin foraging and prey availability, and how fishing might influence this. If there are insufficient data to fully inform explicit spatial modelling, a proxy such as an availability term (parameterised based on what is known) could be used instead (e.g., Tulloch et al., 2019), or a higher variance of prey availability could be used to model situations where foraging is more restricted (see, for example, Koehn et al., 2021).
- Using a fully integrated model and explicitly representing age and stage (e.g., breeding) structure will be important when trying to partition sources of mortality because these operate on different ages, stages and time time-periods, and hence attributing declines to a particular factor needs to involve demonstrating that the data are consistent with the proposed mechanism. Having two or more colonies explicitly represented will further assist with separating confounded sources of mortality and growth.
- A variety of approaches could be used to incorporate measures of foraging behaviour (maximum distance, path length and trip duration) and translate these into population growth in an integrated MICE, although this will likely be a secondary effect that is investigated/sequentially added after first incorporating more direct measures of prey abundance influencing vital rates. There are few studies where

this has been done – for example, Sydeman et al. (2017) note that Robinson et al. (2015) provides one of the few models linking adult survival and prey availability. However, more recently, Koehn et al. (2021) developed a structured seabird model to test the impact of fishing forage fish prey on seabirds and they incorporated both seabird life history and seabird–forage-fish dynamics. Similar to Robinson et al. (2015), they found seabird sensitivity to fishing was mainly dependent on the relationship between adult survival and prey availability, rather than between reproductive success and prey availability. They used a simple equation with two alternative parameter settings to model scenarios of wide vs. limited foraging ranges during the breeding season. A literature search may yield further helpful studies – for example, Houston et al. (1996) developed a model to show the relationship between foraging distance and the maximum size of a chick, which could translate into differences in chick survival; Plagányi et al. (2000) modelled how temporal and spatial match/mismatches between anchovy and their copepod prey could influence anchovy growth rates – conceptually this is similar to how a more detailed penguin foraging model could be used to quantify implications for adult and juvenile energetic budgets and hence growth and survival, with the final relationships (i.e. not the entire sub-model) used as an input to a MICE.

- It may not be necessary to include a detailed representation of Cape fur seal population dynamics to explore the potential role of Cape fur seal predation and competition contributing to the past and current decline in penguin numbers. Rather, it is important to include available data on trends in abundance, especially at the regional scale, relative rates of growth of seal populations (and possibly other predators), diet data and other data to substantiate the intensity and types of competition posited.
- A variety of methods such as described in the literature (see, for example, Haltuch and Punt, 2011; Holsman et al., 2016; Adadi et al., 2017; Hollowed et al., 2020) and used in previous MICE (e.g., Tulloch et al., 2019; Plaganyi et al., 2021; Rogers and Plaganyi, 2022), are available for investigating the role of environmental drivers such as temperature (and extreme events in particular) as well as climate change.
- Once the MICE is adequately validated, it should be a useful tool for testing and quantifying the relative efficacy of alternative penguin conservation measures. Hence the suggested approach is to first develop and fit to data a MICE that includes trophic interactions and key environmental drivers. This will hopefully provide a rigorous framework for quantifying the relative roles of (cumulative) factors causing the decline. The fitted model could then be used to evaluate and compare the likely conservation benefits of a range of mitigation measures such as rehabilitation of adults, predator control, extreme weather risk mitigation and so forth.
- The MICE could also be used as an operating model in a MSE framework (see also Siple et al., 2021), noting that, if coupled with the current small pelagic Operational Management Procedure, consideration needs to be given to aligning in some way the spatial scales that are relevant for the fishery versus the smaller scales that are likely relevant for penguins. Nonetheless, as a first step, the current OMP could usefully be coupled with a penguin population dynamics model to update previous analyses given that sardine biomass is now at much lower levels than was the case during previous testing.

Penguin in full song (photo BM Dyer)

Photo credit Dr. Éva Plagányi-Lloyd