

## **COMPENSATORY MITIGATION: AN INTERIM SOLUTION TO SEABIRD BYCATCH IN FISHERIES?**

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### **ABSTRACT**

The concept of compensatory mitigation is well established as an approach to environmental management. In the past, mitigation programs have been used to conserve wetlands affected by development, and is proposed as a cost-effective approach to offsetting greenhouse gas emissions through reforestation programs. The concept may be equally applicable to environmental management in fisheries. Each year, substantial numbers of non-target species are caught and killed as incidental bycatch during fishing activities. Of increasing concern is the impact this bycatch is having on seabird populations. Substantial areas of fisheries are being closed to protect these species at great cost to the fishing industry. However, other actions may be taken to offset the impact of fishing on these populations, such as reducing predation by invasive species. In this paper, we examine the potential role of compensatory mitigation as a fisheries management tool. An example is provided of the potential cost effectiveness of rodent control relative to fishery area closures for the conservation of a seabird population adversely affected by an Australian tuna fishery. We find that, in the example being examined, invasive rodent eradication is at least 10 times more cost effective than area closures. While this does not solve the actual bycatch problem, it may provide breathing space for both the seabird species and the industry to find longer term means of reducing bycatch.

**Keywords: compensatory mitigation, bycatch reduction, cost effectiveness, seabirds**

### **INTRODUCTION**

Eight percent, or 7.2 million tonnes, of the global fisheries catch consists of non-target species which are subsequently discarded (FAO, 2004). This mortality is having major impacts on species and ecosystems (Baker *et al.*, 2002; Botsford *et al.*, 1997; FAO, 2004; Lewison *et al.*, 2004; Spotila *et al.*, 2000). In some cases, fishing gear modifications and other low-cost measures are effective in reducing bycatch for some species and are being implemented (Gilman *et al.*, 2005). However, in other cases avoiding unacceptable levels of mortality has proven difficult, and costly regulatory interventions are becoming commonplace. For example, New Zealand's squid and Hawaii's swordfish fisheries have both been recently closed due to bycatch of endangered marine vertebrate species (Anonymous, 2006; Lane, 2005; Martin, 2005).

Of primary concern in this paper is the case of seabirds, the incidental catch of which is taken by pelagic fishing fleets such as those that target tuna and squid. However, seabird species that are impacted by fisheries bycatch spend part of their life on land. Events in these terrestrial habitats often lead to significant mortality. For instance, while fisheries bycatch is affecting seabirds globally, particularly albatross, petrels, and shearwaters (Brothers *et al.*, 1999), of greater concern in many instances is the impacts of invasive mammals on breeding colonies. Invasive predators such as feral cats *felis catus* and rats *Rattus* spp. have decimated seabird breeding colonies worldwide, preying on eggs, chicks, and adults of many species (Atkinson, 1985; Nogales *et al.*, 2004). Invasive rodents have been introduced to over 82% of the world's major island groups ranging from the Arctic to the Antarctic, and feral cats occur on most of the world's islands, including Australasia and the islands of the Atlantic, Pacific, and Indian Ocean (Atkinson, 1985; Long, 2003). Three-quarters of seabirds listed by the IUCN are threatened by

invasive species, compared to 47% threatened by fisheries bycatch (Zydalis *et. al.*, 2009). Indeed, invasive mammals are responsible for most vertebrate extinctions over the past six centuries, the overwhelming majority occurring on islands (Groombridge *et. al.*, 1992; MacPhee and Flemming, 1999).

Despite the threat posed by invasive mammal predators to many seabird species, research and management is often directed at anthropogenic sources of mortality such as bycatch. Measures such as closures, restrictions on fishing activities, and gear modifications are aimed at addressing the externality directly, generally resulting in higher costs to the industry and in many instances, lower revenues. While bycatch reduction technology is improving, experimental results are often not translated into actual bycatch reductions in the fishery unless substantial compliance and enforcement measure are introduced (Cox *et. al.*, 2007), all at additional cost to the industry.

For at least some seabird species, greater reductions in mortality (and hence greater increases in benefits) could potentially be achieved by diverting resources from the fishery to other conservation activities. Such alternative measure may include the eradication of invasive species or the creation of new (or restoration of old) breeding habitats. This is akin to the concept of compensatory mitigation, or offsets, used in environmental management in other industries.

The potential for compensatory mitigation as a fisheries management option has received mixed, and mostly adverse, responses. The initial proposal (Donlan and Wilcox, 2008; Wilcox and Donlan, 2007) has received severe criticism, with claims that it may do more harm than good if it diverts attention from the bycatch issue directly (Doak *et. al.*, 2007), that the model used in the analysis was flawed (Finkelstein *et. al.*, 2008a; Priddel, 2007), or that it is limited in its application to only part of the bycatch problem (Zydalis *et. al.*, 2009). These criticisms were largely focused on the assumption that compensatory mitigation may replace the need for bycatch reduction. However, when a species is under threat and bycatch reduction technologies are not sufficient to address the problem, compensatory mitigation may be sufficient to “buy time” for the species while longer term solutions are sought (Iguar *et. al.*, 2009). As the only other remedial measure is a fishery closure,<sup>1</sup> then compensatory mitigation may be a viable option for fisheries management, even if only as a stop-gap measure while bycatch issues are addressed more fully.

In this paper, we examine the potential ecological and economic benefits that may arise through the adoption of a compensatory mitigation approach to the management of bycatch of non-market, but nevertheless valuable, species in fisheries. An example is presented of a potential application of such an approach to seabird conservation. The original fishery examined by Wilcox and Donlan (2007) is revisited, taking into account some of the criticisms raised by Priddel (2007) and others.

## **SEABIRD BYCATCH IN THE EASTERN TUNA AND BILLFISH FISHERY**

The Eastern Tuna and Billfish Fishery (ETBF) operates along the entire east coast of Australia, extending to (and in a few small areas, beyond) the Australian exclusive economic zone. The fishery targets four tuna species (yellowfin, bigeye, skipjack and albacore) as well as several billfish species. In 2007-08, the total value of landings from the fishery was estimated to be around AU\$44m (ABARE, 2009). Until 2005, over three hundred vessels held permits to operate in the fishery, although most of the catch is taken by longliners of which only 113 vessels were active. In 2005 and 2006, a Commonwealth funded buyback program reduced the fishery to only 57 active vessels (Vieira *et. al.*, 2010).

As with many longline fisheries, the incidental bycatch of seabirds is a problem. Flesh-footed shearwaters (*Puffinus carneipes*) suffer the greatest mortality, estimated at 1800-4500 birds per annum (Priddel *et. al.*, 2006a).<sup>2</sup> The east coast population<sup>3</sup> breed exclusively on Lord Howe Island (off the New South Wales north coast) (Priddel *et. al.*, 2006b), with foraging seabirds covering distances of up to 800km from the Island (Thalman *et. al.*, 2009). Studies of foraging behavior found that over half the foraging sites overlapped with tuna vessels, with most of this overlap occurring in areas of highest fishing activity between the Island and the mainland coast (Thalman *et. al.*, 2009) (Figure 1).

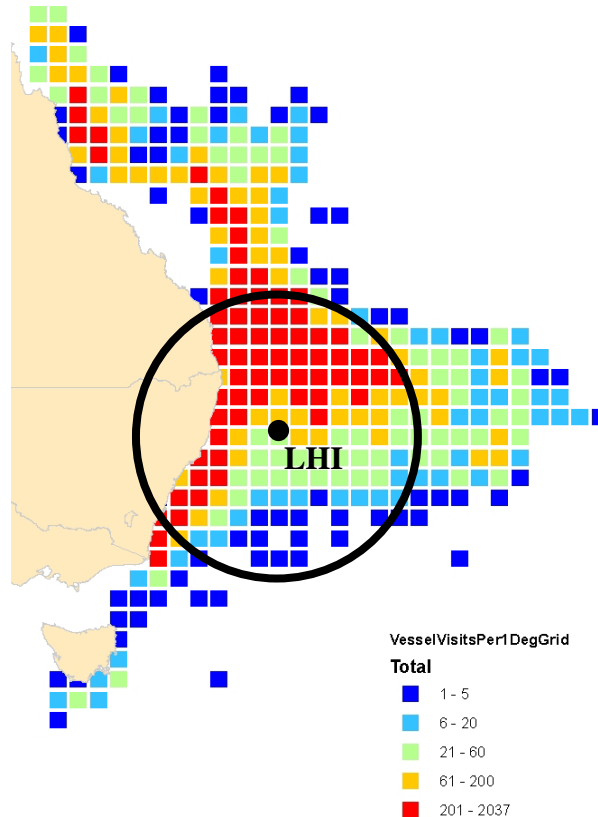


Figure 1 Distribution of total fishing days in the ETBF, 2003-08, and observed foraging range of shearwaters. Lord Howe Island (LHI) lies at the centre of the range.

A management objective has been introduced to reduce seabird bycatch to a target rate of less than 0.05 birds per 1000 hooks (Environment Australia, 1998). In comparison, existing bycatch rates were estimated to average 0.15 birds per 1000 hooks for the fishery as a whole, and average 0.779 shearwaters per 1000 hooks in waters surrounding Lord Howe Island (between 25°S and 35°S) (Baker and Wise, 2005). A number of measures have been trialed to reduce seabird bycatch as part of a bycatch action plan. These include prohibition of setting longlines during daylight hours, the use of heavily weighted lines, and the use of underwater setting chutes. While the measures have reduced bycatch of seabirds in the fishery (AFMA, 2008), mortality rates still far exceed the target rate and bird populations are still expected to decline (Baker and Wise, 2005). Further, they are costly to the industry to implement, may pose health risks to operators<sup>4</sup> and are difficult to enforce.

With the entire eastern Australian population of flesh-footed shearwater breeding on Lord Howe Island and evidence of a population decline (Priddel *et al.*, 2006a), fishery closures may be implemented.<sup>5</sup> Based on fishery observer data on bycatch rates with distance from the nesting colony on Lord Howe, a closure adequate to achieve the Environment Australia (1998) bycatch target would require an area closure of 785 km radius around Lord Howe Island, consistent with the observed foraging range (Thalman *et al.*, 2009). This includes much of the area of high activity in the fishery, and may result in significant losses in total revenue.<sup>6</sup>

Bycatch is not the only threatening process for the seabirds, and demographic impacts on flesh-footed shearwaters occur from on-island threats such as habitat loss, ingestion of plastic, and predation by invasive predators (Priddel *et al.*, 2006a). Rat control is currently undertaken on the Island through poisoning, but evidence of damaged eggs consistent with rat predation persists. While the actual impact of rat predation is debatable (Priddel, 2007), previous modeling work suggests that even modest predation rates may be having a significant impact on the seabird population (Finkelstein *et al.*, 2008b).

## A REVISED MODEL OF THE FISHERY

To illustrate the potential role of compensatory mitigation in fisheries, we conducted a simple bio-economic analysis of a compensatory measure to offset bycatch of flesh-footed shearwaters in the ETBF. The model differs from the original model used by Wilcox and Donlan (2007), and instead is based on an age-structured population model developed by Baker and Wise (2005), and is equivalent in characteristics to the model suggested by Finkelstein et al (2008b). Further, the key parameter estimate changes proposed by Finkelstein et al (2008b) are also adopted, although alternative scenarios of rat predation are also considered. The model equations and key parameters are given in the Annex. The model was run over a 40 year period.

For the purposes of comparison, we estimate the costs to the fishing industry and impacts on the seabird population of eliminating bycatch mortality through a large scale area closure and increasing chick survival rates through eliminating rats on the island.

In the previous analyses, both the closure and rat eradication scenarios implicitly assume that there are no improvements in gear technology over the 40 year period. This assumption is unrealistic, and it is likely that the “bycatch problem” can be solved (or at least substantially reduced) during this period. The analysis allowed for the possibility of technology to reduce bycatch to the required level after 5 years and after 10 years. In terms of the closure scenario, this effectively meant that the costs of the closure were only incurred for the period that the closure was necessary, with no other impact on the seabird population (as the fishing mortality had already been reduced due to the closure). For the rat eradication scenario, fishing mortality was reduced to the same level as the closure after 5 and 10 years respectively. The costs of any new technologies in terms of equipment cost and change in catch rates were not considered as these would be equally applicable to both the closure and rat eradication scenarios.

### Changes in seabird population

The effects of the closure on the seabird population were estimated by assuming catch rates declined to the target of 0.05 birds per thousand hooks. Mortality of seabirds due to fishing was correspondingly reduced from 0.096 (Baker and Wise, 2005) to 0.006 (Finkelstein *et. al.*, 2008a).

The effect of removing rats from the island is uncertain. The original analysis by Wilcox and Donlan (2007) assumed that breeding success (the ratio of eggs to fledges, initially estimated at 0.513) would increase to 1 as a result of removal of rat predation. Rat consumption rates were estimated using allometric relationships for metabolic estimates and reported rat weights from islands (Nagy, 1987; Stapp, 2002), and the average of historic reports of unmanaged rat densities of 45.5 per ha (range 4-94 rats/ha). They further assumed rats met 100% of their daily metabolic requirements from seabird eggs and chicks, when available. Their resultant estimated change in chick survival is consistent with other empirically based studies. For example, reducing the rat population by around 57% resulted in a 61% increase in the breeding success of shearwaters in the Mediterranean (Igual *et. al.*, 2006). Extrapolating from this, complete removal of rats could be expected to result in a 100% increase in breeding success. However, as the rat population is not entirely unmanaged, the assumed current rat density is likely to have been overestimated, as the original study was intended more to illustrate the potential hypothetical benefits of compensatory mitigation than provide a definitive cost-benefit analysis of the eradication program on the Island.

In contrast, Finkelstein et al (2008b) suggested that breeding success would only increase to 0.748 (a 45.8% increase) as a result of rat removal, the minimum observed breeding success in shearwaters on predator free islands. We use this value, but also run the model assuming a slightly higher breeding success rate of 0.831, which was the upper level in the range of observed success rates identified by Finkelstein et al (2008b).

The impact of the different options and survival rate scenarios on the seabird population is illustrated in Figure 2, and summarized in Table 1. For comparison, the projected change in population if no mitigation measures are implemented is also presented. The options are implemented individually. That is, rat mortality is assumed to continue in the closure scenario, while current fishing mortality is

assumed to continue for the rat elimination option. The assumption about technology changes is applied to all scenarios equally (including the do-nothing scenario that will also benefit from the improved bycatch reduction technology).

Table 1. Estimated numbers of seabirds under different scenarios after 40 years

	Baseline (Do nothing)	Closure	Rat eradication		
			$p_f=1$	$p_f=0.748$	$p_f=0.831$
Population after 40 years					
• no gear improvement	6385	27831	22668	12934	15900
• improvements in 5 years	24432	27831	64531	42026	49217
• improvements in 10 years	21601	27831	59717	38130	44986
Increment against baseline					
• no gear improvement		21445	16282	6549	9515
• improvements in 5 years		3399	40099	17594	24785
• improvements in 10 years		6229	38116	16529	23384

Note:  $p_f$  is the probability that an egg will eventually become a fledgling.

The time trajectory of recovery for the two conservation actions differs. Closing the area of the fishery results in an immediate reduction in adult mortality due to bycatch. With more birds of breeding age surviving each year, the population starts to increase immediately. In contrast, reducing chick mortality through eradicating rats has no impact on the breeding population until they reach maturity, so no immediate improvement in the population is observed (Figure 2). However, once these juveniles reach maturity, the population is expected to increase relative to the do-nothing scenario. A key observation from Figure 2 is that rat eradication, even at the most pessimistic outcome in terms of egg-to-fledgling survival, is likely to stabilize the population, albeit at a lower level than the starting point. This suggests that, at worst, the bycatch from the fishery is sustainable if the land-based source of mortality is removed.

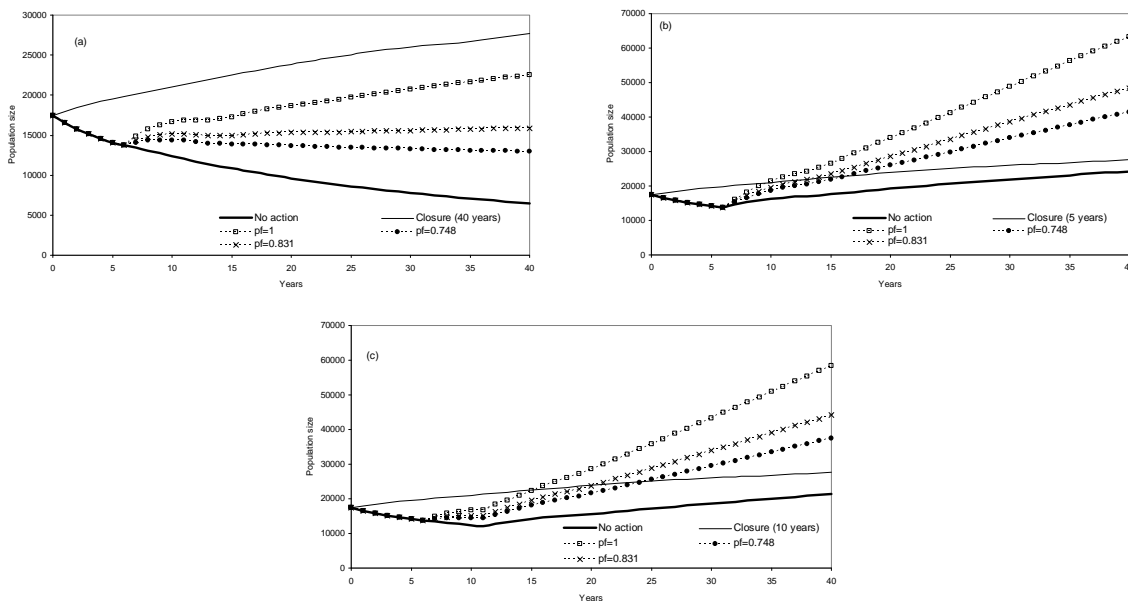


Figure 2. Change in seabird population over time a) no gear improvements; b) gear improvements after 5 years; c) gear improvements after 10 years

An early introduction of bycatch reduction technologies (e.g. after 5 years) greatly reduces the benefits of a closure in the longer term (relative to the benchmark scenario that also experiences the technological

improvement), but enhances the benefits of the rat eradication. In the analysis, rat-related mortality is assumed to continue under the closure scenario, and this continues to depress the population growth.

### Costs of Mortality Reductions

The potential economic impact of a closure of the size required to achieve the target bycatch rates was estimated using a location choice model (Wilcox *et al.*, 2010). The simulation model was based around a multinomial logit model of fisher location choice, as well as information on costs structure of the fleet and fleet size. Given that the fishery is based on a mobile resource that has a different spatial stock structure from year to year, the model was run using two years of trip level data (relating to 2004 and 2007). Based on the model results, the economic impact of the closure (estimated as reduction in fishery profits) is very specific to the underlying stock distribution, ranging from \$0.6m (under 2004 stock conditions) to \$2.2m (under 2007 stock conditions).<sup>7</sup> To place this in context, the total economic profits in the fishery in 2007-08 was estimated to be only \$2m (Vieira *et al.*, 2010).

In contrast, the cost of eradicating ship rats and mice from the Island has been estimated to be only AU\$0.92m (in 2009-10 dollars) (Parkes *et al.*, 2004). These costs, while appearing relatively low, are consistent with rodent eradication costs experienced elsewhere (Martins *et al.*, 2006). This is a one-off cost, and assumes that re-infestation does not occur. The monitoring and surveillance costs of ensuring that re-infestation does not occur are unknown, and are likely to be borne by vessels visiting the island. As a result this is most likely an underestimate of the true cost of rat eradication.

The net present value of the costs of each option under the different scenarios is given in Table 2. The net present value was estimated at both 5% and 10%. The cost of the closure option was varied depending on the length of time required for a technological solution to the bycatch problem to be introduced. As all of the costs with the rat eradication occur at the start of the program, the net present value is equivalent to the upfront costs.

Table 2. Net present value of the costs of different options

	Up-front cost	Annual Cost	Discount rate					
			5%			10%		
			Closure length			Closure length		
			40 year	5 year	10 year	40 year	5 year	10 year
Closure - lower estimate	-	0.64	\$11.0	\$2.8	\$4.9	\$6.3	\$2.4	\$3.9
Closure - higher estimate	-	2.34	\$40.2	\$10.1	\$18.1	\$22.9	\$8.9	\$14.4
Rat eradication	\$0.92	-						

### Cost effectiveness of the measures

A cost benefit analysis requires both the outcomes and the costs of the alternative mitigation measures to be valued in monetary terms. However, as the value of the ecological outcomes is unknown, and it is beyond the scope of this example to derive such values, cost effectiveness analysis is used to determine the most efficient mitigation measure. An implicit assumption, however, is that the value of the stock recovery is considered by society to exceed the costs. Hence, the least cost method to deliver stock recovery is considered the most efficient.

Determining an appropriate ecological outcome for the purposes of comparing the costs is not straightforward. For simplicity, we assume more seabirds are preferred to less, and that the marginal value of a seabird is constant,<sup>8</sup> such that the increase in the number of birds reflects the value of the mitigation activity. Also for simplicity, we take the incremental change in the seabird population relative to the baseline in year 40 as our output measure (Table 1). This ignores the potential time preference relating to seabirds, in that a closure “produces” more additional seabirds earlier than the rat eradication program.

The cost effectiveness ratio was estimated as the average cost per additional seabird relative to the benchmark scenario (Table 3). From Table 3, the rat extermination is the most cost effective method of maintaining or recovering the seabird population. The cost per bird further decreases if bycatch can also

be removed though new technologies, whereas new technologies effectively increase the cost per bird under a closure scenario. While this result appears counter intuitive, the benefits of the closure if gear modifications are introduced relatively early are limited (Figure 2), whereas the gear modifications compliment the reduced predation mortality in the rat eradication option.

Table 3. Cost effectiveness of the alternative options (\$/seabird)

	Closure		Rat eradication		
	Low cost	High cost	$p_f=1$	$p_f=0.748$	$p_f=0.831$
5% discount rate					
• no gear improvement	519	1897	57	142	98
• improvements in 5 years	812	2970	24	54	38
• improvements in 10 years	784	2866	25	57	41
10% discount rate					
• no gear improvement	296	1081	57	142	98
• improvements in 5 years	711	2600	24	54	38
• improvements in 10 years	624	2281	25	57	41

## DISCUSSION

The example above illustrates that, if conservation of species impacted by fishing is an objective, then non-fishing related options may be more cost effective than limiting fishing activity, particularly if technical solutions are likely to be found to reduce bycatch in the near future. In this case, conservation benefits could be achieved at lower cost if the fishing industry funded the rat eradication through a bycatch levy, for example, rather than ceased fishing in the area. Such a levy would also have the additional benefit in that it would create an incentive for fishers to avoid bycatch of seabirds. This is not to say that the externality imposed by fishing should not be eliminated or reduced, and indeed the analysis assumes that it will be reduced at some point. Bycatch is a highly visible form of mortality, and pressures on the industry to reduce bycatch will persist even if compensatory mitigation actions are undertaken. However, compensatory mitigation activities may give the fishing industry “breathing space” in which new technologies can be developed that are themselves cost effective in reducing the bycatch problem. It also provides an opportunity for the fishing industry to engage with conservation groups, and be seen to be concerned about the ecological problems that it is contributing to.

The example assumes a one-off program to eradicate rats from the Island. No doubt, on-going monitoring of both rats and seabirds will be required to ensure that the eradication has been successful and that seabird stocks are recovering. This will be particularly important given that recovery may not be obvious for several years. Given that the alternative – closure of the fishery – imposes far greater costs on the industry, then fishers have an incentive to continue paying a levy for continuing mitigation related activities, effectively as an insurance premium against closures. Such a levy, if directly linked to bycatch, also provides an incentive for fishers to adopt any bycatch reduction technologies that develop over time, and indirectly create an incentive for such technologies to be developed. From the above example, even if the one-off eradication costs were recovered from the industry as an ongoing annual levy, the mitigation option is still between 10 and 20 times more cost effective than closures (depending on the discount rate assumptions).

The results of the analysis are also consistent with those of Finkelstein et al (2008b), who found that combining rat eradication with bycatch reduction resulted in the greatest increase in population size. What mainly differs is the interpretation of the results. Finkelstein et al (2008b) concluded that as bycatch reduction has a bigger impact than rat eradication when viewed separately, bycatch reduction should be the priority option, effectively discounting the potential role of compensatory mitigation. They also conclude that cost should not be a consideration when protecting species (Finkelstein *et. al.*, 2008a). Here, we have demonstrated that compensatory mitigation can achieve conservation objectives at a considerably

lower cost than draconian measures such as closures, and can provide greater conservation benefits if used with bycatch reducing technologies than the latter alone.

## CONCLUSIONS

The key critics of the original Wilcox and Donlan (2007) paper argued that bycatch was the major threatening process for seabirds, and that compensatory mitigation did not address this problem (Finkelstein *et al.*, 2008a; Priddel, 2007; Zydalis *et al.*, 2009). This point is not refuted in this paper, and the model simulations also support the rapid adoption of bycatch reduction technologies. However, like world peace, bycatch elimination cannot be achieved over night. If the technologies currently existed to eliminate seabird bycatch then no doubt they would be in place already. Compensatory mitigation may play an important role as a “stop-gap” measure to provide initial relief for at least some seabird populations that have threats other than fishing affecting their populations also. While these may not be an appropriate long term solution, they may prevent more drastic and costly measures (e.g. fishery closure) being introduced while more suitable technologies are being developed.

When designed under the proper framework, compensatory mitigation requires a mechanism for generating revenue from common pool resources that can be transferred to support high impact conservation actions. Returning to the fisheries context, using individual vessel levies for bycatch 1) provides regulatory certainty for operators, an essential ingredient for effective businesses; 2) creates individual incentives for fishers to avoid bycatch; and 3) could fund mitigation actions that at least partially offset the bycatch that does occur. Unlike other compensatory mitigation programs that have been criticized as leading to reductions in environmental quality, for example through substituting lesser (ecologically) valued wetlands to those removed, fishers have an incentive for the mitigation actions to achieve outcomes greater than, or at least equal to, that of the alternative – closure.

As opposed to command-and-control approaches (e.g., fisheries closures), incorporating market externalities into the costs of fishing allows fishers an opportunity to develop innovative ways of avoiding bycatch, and a derived demand for the development of new technologies to assist in bycatch reduction. The lack of such opportunity is a common complaint in the fisheries sector, and individual incentives have been shown to be the single most important factor determining the sustainability of fisheries (Hilborn *et al.*, 2005). Further, compensatory mitigation would have significant marginal benefits since not only bycatch species but an entire suite of species, and frequently entire ecosystems, would benefit from the removal of invasive mammals and other on-island restoration actions. Compensatory mitigation provides an opportunity to constructively address a global conservation concern, and forge an alliance between conservation and fisheries organizations.

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## ANNEX. POPULATION MODEL DESCRIPTION AND PARAMETERS

The model used was a simple population dynamics model, adapted from Baker and Wise (2005). The number of adult birds in age class  $7 \leq i \leq 40$  in year  $t > 1$ ,  $N_{i,t}$ , is given by

$$N_{i,t} = N_{i-1,t-1}(1 - F - M), \quad 7 \leq i \leq 40, t > 1 \quad (1)$$

where  $F$  is the rate of fishing mortality due to bycatch and  $M$  is the rate of natural mortality. The maximum age of the birds was assumed to be 40 years. The model specified by Baker and Wise (2005) was based on the average number of female fledglings per female, and the population was modelled in



terms of female numbers. As the sex ratio is approximately 1:1 (Baker and Wise, 2005), the model is effectively based on pairs of breeding birds rather than individual birds.

The number of fledglings in time  $t$ ,  $N_{1,t}$  is given by

$$N_{1,t} = fT_{t-1}, \quad t > 1 \quad (2)$$

where  $f$  is the average number of fledglings per pair (the effective fecundity rate), and  $T_t$  is the total number of adult breeding pairs ( $T_t = \sum_{i=7}^{40} N_{i,t}$ ). The effective fecundity rate takes into account natural

mortality of eggs and chicks, including predation mortality, and is defined by

$$f = 0.5 \cdot e \cdot p_f \quad (3)$$

where  $e$  is the average number of eggs per breeding pair and  $p_f$  is the probability that the egg will hatch and the chick survive to become a fledgling. Only half this product is taken as two chicks are required to survive to form one new breeding pair.

The number of juveniles,  $N_{i,t}$ , where  $1 < i < 7$ , was subject to density dependent survival,  $s_t^*$ , such that

$$N_{i,t} = s_t^* N_{i-1,t-1}, \quad 1 < i < 7, t > 1 \quad (4)$$

Following Baker and Wise (2005), the density dependent survival rate was given by

$$s_t^* = \left[ s^{k_t} / (1 - M)^{6(k_t-1)} \right]^{1/6} \quad (5)$$

where  $s$  is the base rate of juvenile survival, and  $k_t$  is a density dependent parameter relating to changes in the population size relative to the first year, given by

$$k_t = [T_t / T_1]^{0.5} \quad (6)$$

The initial number of breeding pairs in each age class in time 1 was estimated based on the total number of birds, and the observed mortality and survival rates within the different ages. The simulations were undertaken by varying the values of  $p_f$  and  $F$ , as described in the main body of the paper. The values of these key parameters are given in Table 3.

Table 3. Model parameters used in the simulations

	Baseline	Closure/bycatch reduction	Rat eradication	Rat eradication and bycatch reduction
$T_1$	17462	17462	17462	17462
$s$	0.766	0.766	0.766	0.766
$M$	0.06	0.06	0.06	0.06
$e$	1.027	1.027	1.027	1.027
$p_f$	0.513	0.513	0.748, 0.831, 1	0.748, 0.831, 1
$F$	0.079	0.006	0.079	0.0006

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## ENDNOTES

<sup>1</sup> A ban on daylight fishing as well as some changes in fishing gear in the area has been trialled. While seabird bycatch has been reduced, it still occurs at a rate higher than the allowable rate, and complete area closure is still under consideration. The daylight ban is not supported by the industry due to the substantial impact it has on target species catch rates.

<sup>2</sup> There is considerable discrepancy between “official” estimates of seabird bycatch from the fishery (AFMA, 2008) and estimates derived from other studies (e.g. Baker and Wise, 2005; Priddel et al., 2006).

<sup>3</sup> Another population exists off South Australia.

<sup>4</sup> Since the introduction of the measures in the fishery, several severe injuries and one death have been recorded as a result of using the new gear.

<sup>5</sup> Temporary closures were implemented in 2008 and 2009 for part of the season.

<sup>6</sup> Such extensive closures have been implemented elsewhere. For example, an even larger area was closed to longline fishing around Hawaii between 1999 and 2004 in order to reduce bycatch of turtles (Huang and Leung, 2007).

<sup>7</sup> All values are in 2009-10 Australian dollars. This also took into account the reduction in fleet size between 2004 and 2007 due to a buyback program that reduced the fleet from around 119 active vessels to 57. With the larger fleet, the cost of the closure was estimated to have been substantially greater.

<sup>8</sup> The marginal value of an additional animal is likely to decrease with increasing population size (Haraden *et. al.*, 2004). However, information on how these values may change is not available. This is a common problem with conservation values for wildlife (Eiswerth and van Kooten, 2009), so is not unique to the case in hand.