

## APPENDIX 1

**This PDF file includes:**

Supplementary text

Figs. S1 to S6

Tables S1, S2

Datasets S2 – S3

## Methods

Trait analysis was conducted using parameter estimates for 81 species, obtained from the primary and reknowned grey literatures (e.g., Birdlife International & NZ Birds Online websites). Values for  $\alpha$  and  $s$  were taken as given in BIDDABA ( $n = 13$  species); for the six species from COMADRE,  $\alpha$  was calculated with methods described in Caswell (2001), whereas  $s$  was calculated as the mean column-sums of the sub-matrix of survival-dependent transitions corresponding to adult reproductive stages, weighted by its stable stage distribution (see Salguero-Gómez *et al.* 2016). The IUCN Red List threatened species ranking and global population size data were obtained from the IUCN Red List (IUCN 2021).

The dataset of colonies on islands where predators were removed were selected because there were estimates of the population size, which are needed for the Demographic Invariant Method (DIM). The regional population estimates were obtained from The Northern New Zealand Seabird Trust (Unpublished; Supplement S2) and from Brooke *et al.* (2018). We excluded colonies that were smaller than 50 individuals because smaller colonies are thought to be more prone to extinction, because they are more affected by demographic stochasticity, which may be related to minimum viable population sizes (Caughley 1994).

The ecological and morphometric information of each species was used to correlate key attributes of the species niche to its vulnerability to at-sea threats. These include foraging strategy (i.e., the primary method of feeding, such as pursuit diving, surface seizing, pattering), primary prey type (i.e., cephalopods, fish) collated from Ashmole (1971), Schreiber & Burger (2002), Del Hoyo *et al.* (2011), and NZ BirdsOnline. Mean (female) adult body mass (B) data were obtained from the CRC Handbook of Avian Mass files 2<sup>nd</sup> Edition (Dunning 2013). No uncertainty for body size estimates were included in the model. At-sea distribution data were from Birdlife International (2016). We used taxonomy and nomenclature from Birdlife International (Birdlife International 2021), which differed for some species in Jetz *et al.* (2012) that was used for the phylogenetic analysis. All parameters used in the risk analysis model for 36 colonies of 16 species can

be found in Supplement S2, and parameters for the trait analysis including 81 procellariiform species in Supplement S3. During data wrangling we fitted each parameter estimate (adult survival, at-sea distribution, age at maturity etc) to multiple distributions and used the most appropriate for each parameter, noted in the main text.

We did not account for density dependence in both positive and negative directions when calculating  $\lambda_{\max}$ . Negative density dependence is where  $\lambda$  declines as the population density increases, in this instance it is assumed that fecundity is higher at low densities and decays at a constant rate (Morris & Doak 2002). Although rare, the opposite can also occur, where rapid declines in fecundity occur at low population densities, and then increase at higher densities, given no limiting factors (e.g., no resource or habitat limitations). The magnitude of negative density dependence across a range of densities can vary due to external or internal population level effects (Morris & Doak 2002). Positive density dependence, sometimes referred to as Allee effects, lead to an increase in population growth rate as the population increases. Such responses are likely when resources are not limited, mating success is improved, group defense reduces predation (Morris & Doak 2002).

Determining density dependence extent or type in a population in either direction is inherently difficult due to data limitations (Morris & Doak 2002). Because the current knowledge of processes and effects on population growth rates is lacking, given the long-term and detailed studies required, there is uncertainty in estimating how density dependence may be affecting our results. Finally, we did not address the potential carrying capacity of the population, and assumed that the length of time to reach this point for most populations is beyond current temporal management plans (e.g., 200 years), also that other threats or changes in levels of mortality from threats (e.g., climate change/prey depletion) will adjust the results of the model.

## Marine Threat Data

Detecting at-sea mortality is challenging because seabirds are scattered widely across their foraging ranges, and carcasses may float just below the surface, sink or be consumed by predators (Laist 1997). Furthermore, in the incidence of birds being entangled in fishing line or ropes may be mistaken for fisheries related bycatch – where animals are incidentally caught in active fishing gear, or in some cases in ghost nets rather than from mortality from plastic ingestion, although this is likely a very small proportion (Laist 1997). On land many species are understudied, and land-based surveys provide no indication of the number of at-sea mortalities. In this paper, we address only the impacts of plastic pollution, climate change/fisheries depletion (associated with prey distribution and abundance changes) and commercial fisheries bycatch to our seabird populations. We acknowledge that our model does not include the full suite of marine threats that seabirds are exposed including such as disease, oil-spills, water-bound contaminants, hunting (for comprehensive reviews of the full suite of threats to seabirds see, Provencher *et al.* 2018 and Rodríguez *et al.* 2019). The methods used to estimate the potential fatalities from each of the marine threats included in our analysis are as follows:

**Fisheries bycatch:** We used the mean annual potential mortality from fisheries bycatch from Richard *et al.* (Richard *et al.* 2017) for 12 of the seabird species included in the colony analysis (Appendix 2). Richard *et al.*'s (2017) estimate assumes that all birds killed in the fisheries were adults (98% of the necropsied birds were adults). The estimates for fisheries related mortality reported here do not account for mortality associated with international (beyond the EEZ), illegal and unregulated, or recreational fisheries, which may present a significant source of mortality for some species. Estimates for *Calonectris diomedea* were from Belda & Sanchez (2001) (Supplement S2). The species *Bulweria bulwerii*, *Pterodroma ultima*, and *Puffinus puffinus* were assumed to be low risk from fisheries because are not highly reported as bycatch in the literature (IUCN 2021). These three species were assumed to have 0.1% of the population killed by fisheries. The estimation of at-sea mortality due to a particular threat typically results in a high degree of imprecision. For example, when estimates of adult mortality in a fishery are reported, they are often calculated from a small number (typically < 10%) of ship-

board observations (Richard *et al.* 2011). In addition, there are a lack of data on cryptic mortalities in commercial fishing operations, that is, birds that are killed may not be bought back on board the ship, may fail to be reported when the observer is off-duty, or not seen by the observer. Specific information on the relation between observed captures and total fatalities needs to be improved in order to improve the reliability of our risk assessment (Richard & Abraham 2013). Therefore, it is possible that our model will either fail to adequately quantify the risk of at-sea threats to a seabird species, or will classify species as being at risk when in they may not be.

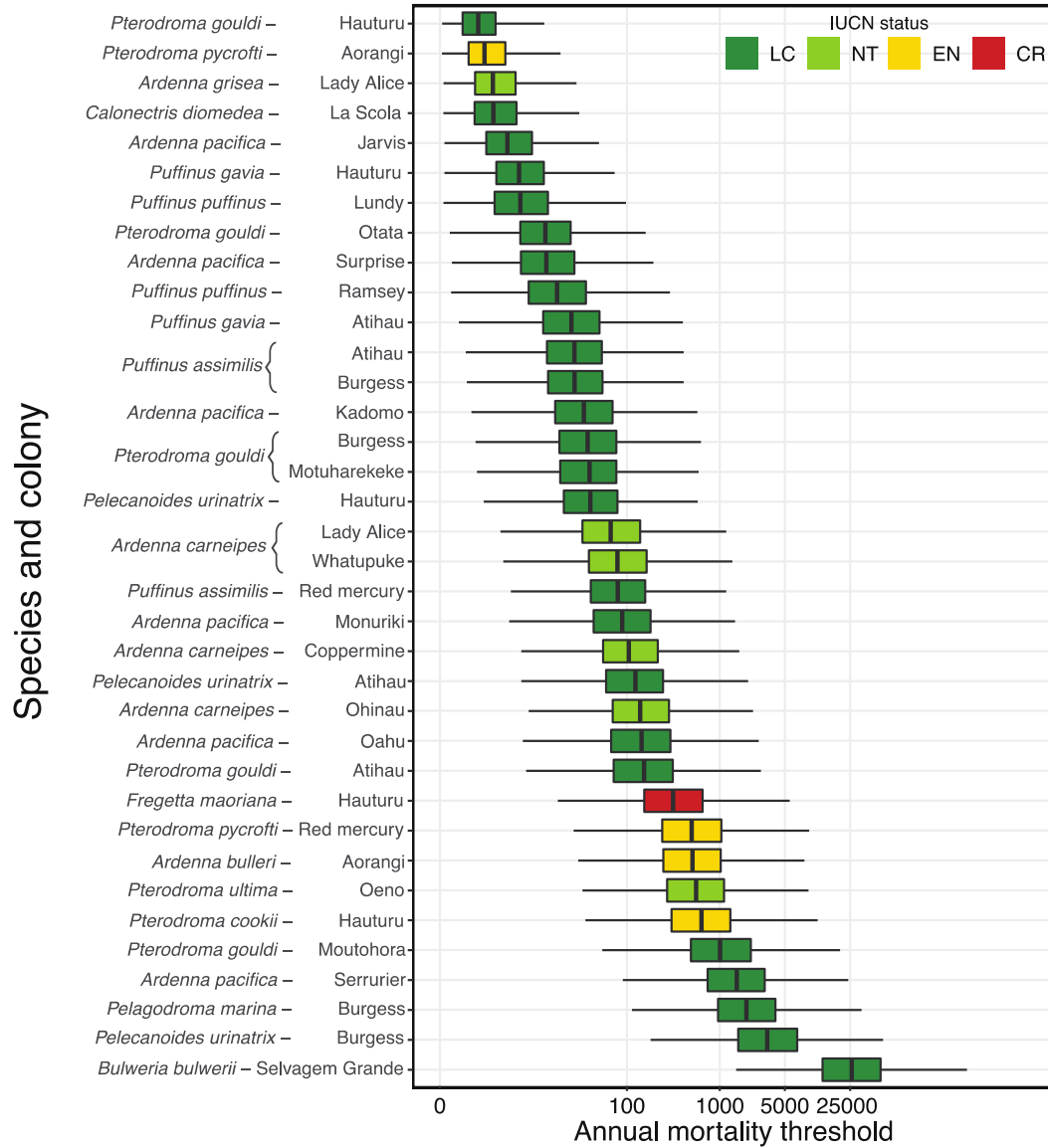
**Plastic ingestion:** The physiological effects of plastic debris ingestion on seabirds may include; internal and external wounds, skin lesions and ulcerating sores, ingestion causing general debilitation, inhibiting feeding capacity, eventually leading to starvation, reductions in reproductive capacity, drowning, and impairment of predator avoidance (Auman *et al.* 1997, Ryan 1987, Vannela 2012). We used the proportion of adults reported in the literature to ingest plastic – the frequency of occurrence - and estimated the proportion of the population on the islands affected. We assumed that the colonies would be affected at the same rate as the frequency of occurrence reported in the literature (Supplement S3), and that of the proportion that ingest plastic, 0.5% of the affected population would die as a direct result of plastic ingestion. This approach may over- or under-estimate the impact of plastic ingestion on adult mortality as there is a lack of understanding about plastic retention in animals and what the long-term impacts may be on adult survival and populations (Rochman *et al.* 2016, Ryan 2016). We tested the sensitivity of each of the species and colonies to plastic ingestion related mortality at 1% and 5% of the proportion of a population that is expected to ingest plastic (Figure 2 & S3).

**Climate Change and Prey depletion:** Despite impressive research efforts that indicate seabirds are the most vulnerable group of avian fauna to prey abundance and distribution changes due to fisheries pressures (Grémillet *et al.* 2018) and climatic changes (Jenouvrier 2013, Oro 2014), there is high uncertainty in our model to predict adult mortality from these pressures. This is due to the difficulty in quantifying adult mortality directly due to the complex interactions affecting prey distributions and abundance (Oro 2014, Sæther & Engen 2010), confounded by the lack of published studies on the effects of climate change/prey depletion for the 16 species included in our marine threats risk

analysis. The influence of climatic changes on seabird populations may exert either positive or negative changes to a population in response to resource availability and distribution, breeding phenology or impacts on habitat (Engen & Sæther 2016, Jenouvrier 2013). In addition, other factors, such as density dependence, inter- and intra-specific competition, and scale dependent variability in climatic stressors will influence how an individual or population will respond (Jenouvrier 2013, Oro 2014). Thus, attributing changes to adult survival directly to a specific climate driver is complex, and generalizing among species can lead to erroneous assumptions (Oro 2014). Until reliable estimates of adult mortality from anthropogenic marine threats to seabirds exist, accurately estimating the population-level effects on seabirds will remain challenging. Because of the uncertainty in estimating the level of mortality caused by climate change, we estimated the impact of climate change as causing 0.5% mortality in a population and tested the sensitivity to risk of 1% and 5% adult mortality for each of the 36 colonies in our risk analysis (Figs. S3 & S5).

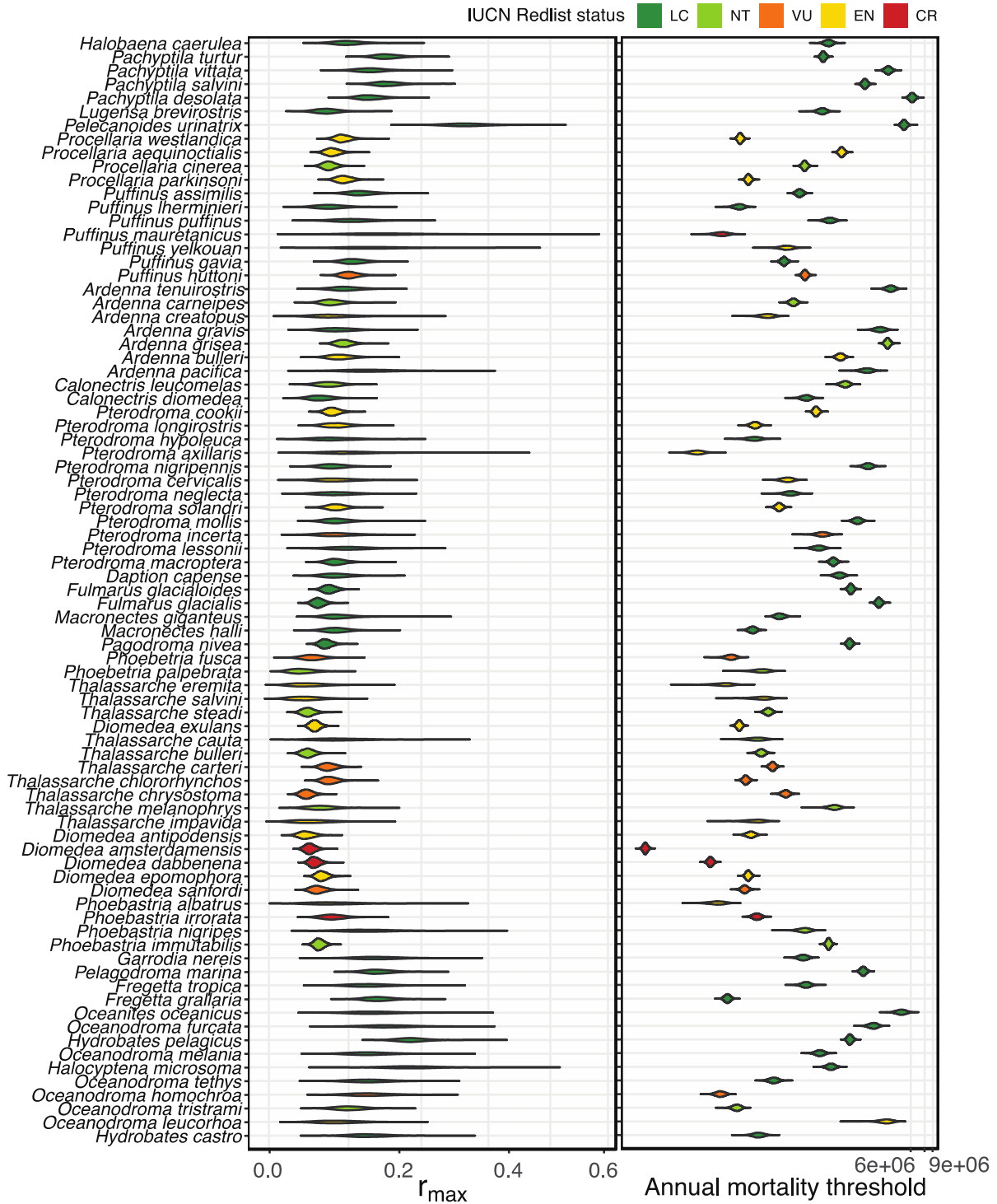
### **Extended results**

Annual mortality threshold for colonies of seabird species included in our analysis.



**Fig. S1:** The model calculated annual mortality threshold (number of individuals) for 16 species from 36 colonies on 23 islands where invasive predators have been eradicated (map inset). The annual mortality threshold for each colony is ranked from lowest (top) to highest. Colors for each species correspond to the IUCN Red List status: LC Least Concern in dark green; NT Near threatened in light green; VU Vulnerable in yellow; CR Critically Endangered in red.

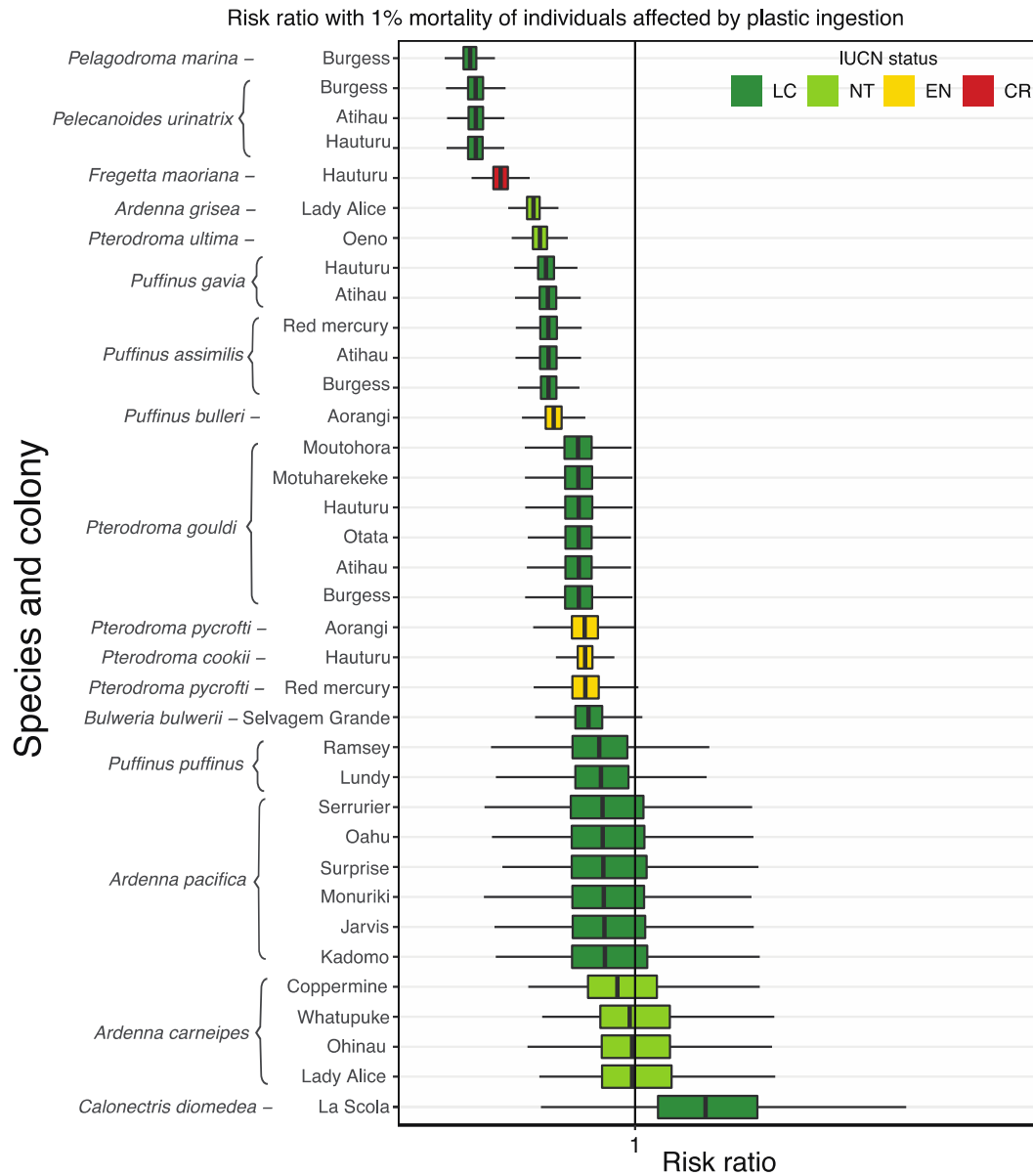
Annual mortality threshold and trait analysis for 81 Procellariiforme seabirds.



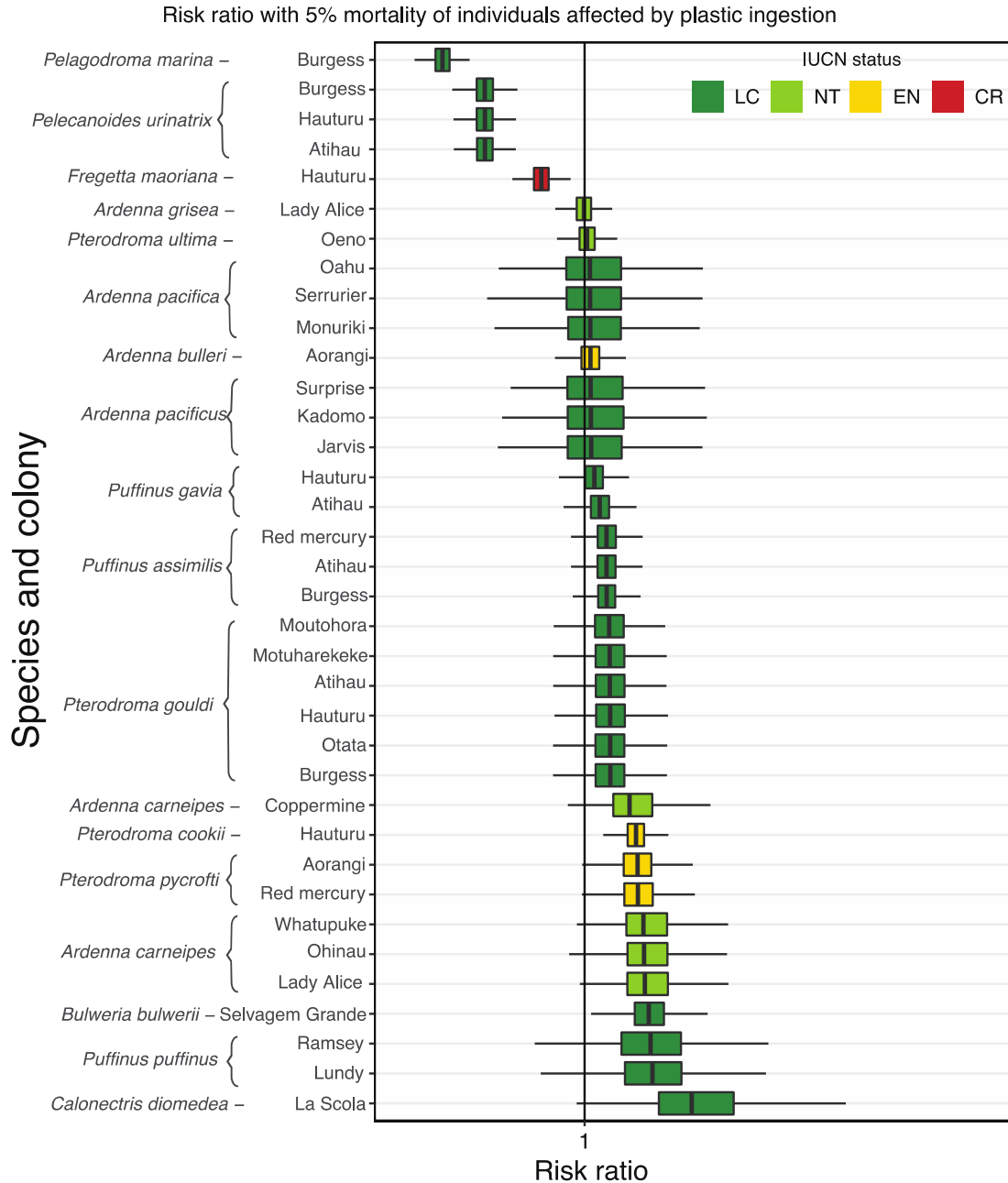
**Fig. S2:** Annual population growth rate ( $r_{max}$ ) and potential annual mortality threshold for the 81 Procellariiformes (phylogenetically ordered as per Jetz *et al.* (2012)) included in the phylogenetic comparative analysis to investigate if shared traits can inform resilience to marine threats.



Plastic ingestion related mortality risk ratio sensitivity analysis.

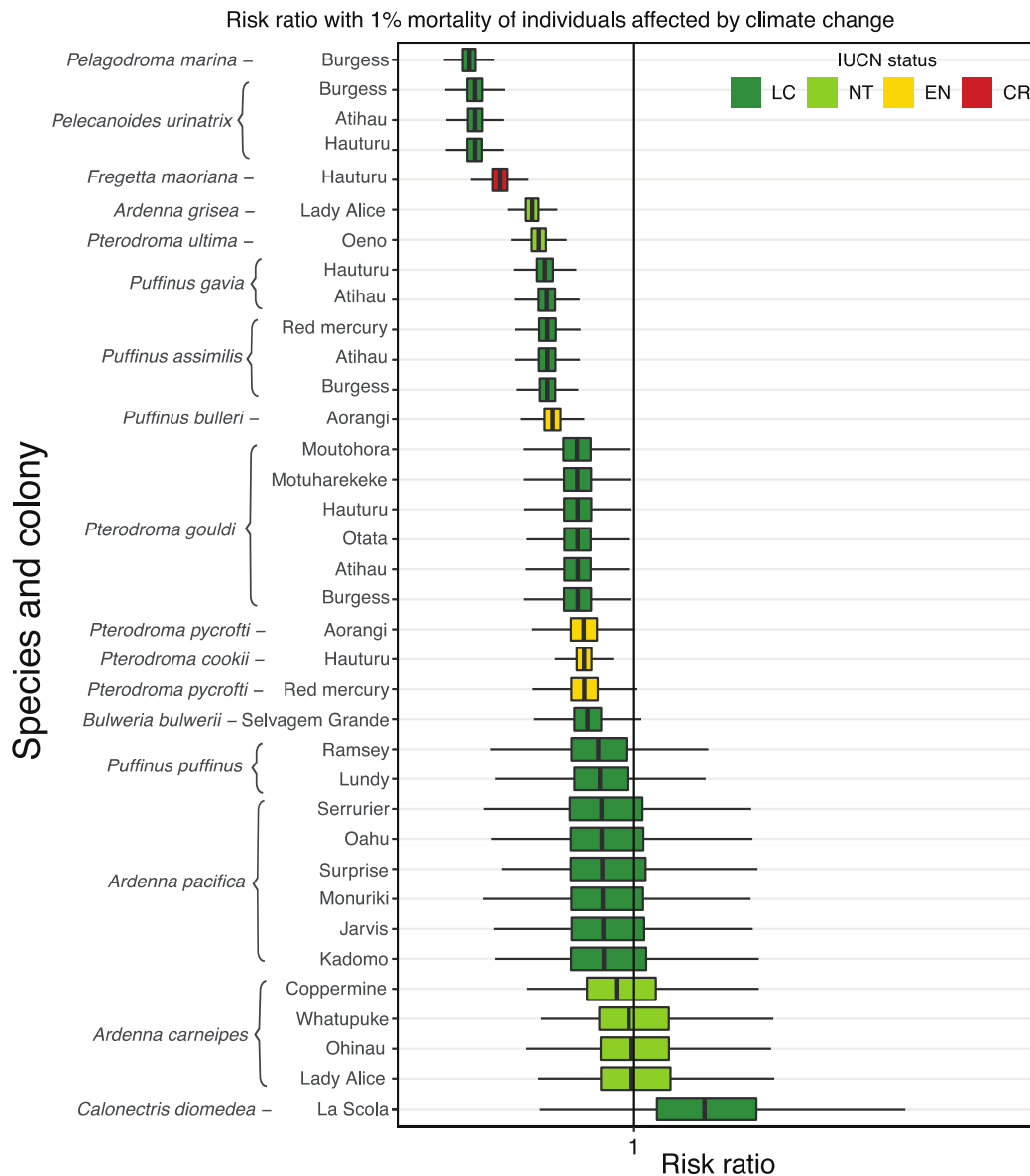


**Fig. S3:** Sensitivity analysis showing the risk for each of the 36 colonies for the 16 species to plastic ingestion mortality at 1% for the proportion of individuals affected (Appendix 1). The risk ratio was calculated as potential mortalities yr<sup>-1</sup> / annual mortality threshold (Richard & Abraham 2013); when this risk ratio  $\geq 1$ , adult mortality from each of the evaluated threats may impede the recovery of a colony even after predator eradication. Colors for each species correspond to the IUCN Red List status: LC Least Concern in dark green; NT Near threatened in light green; VU Vulnerable in yellow; CR Critically Endangered in red.

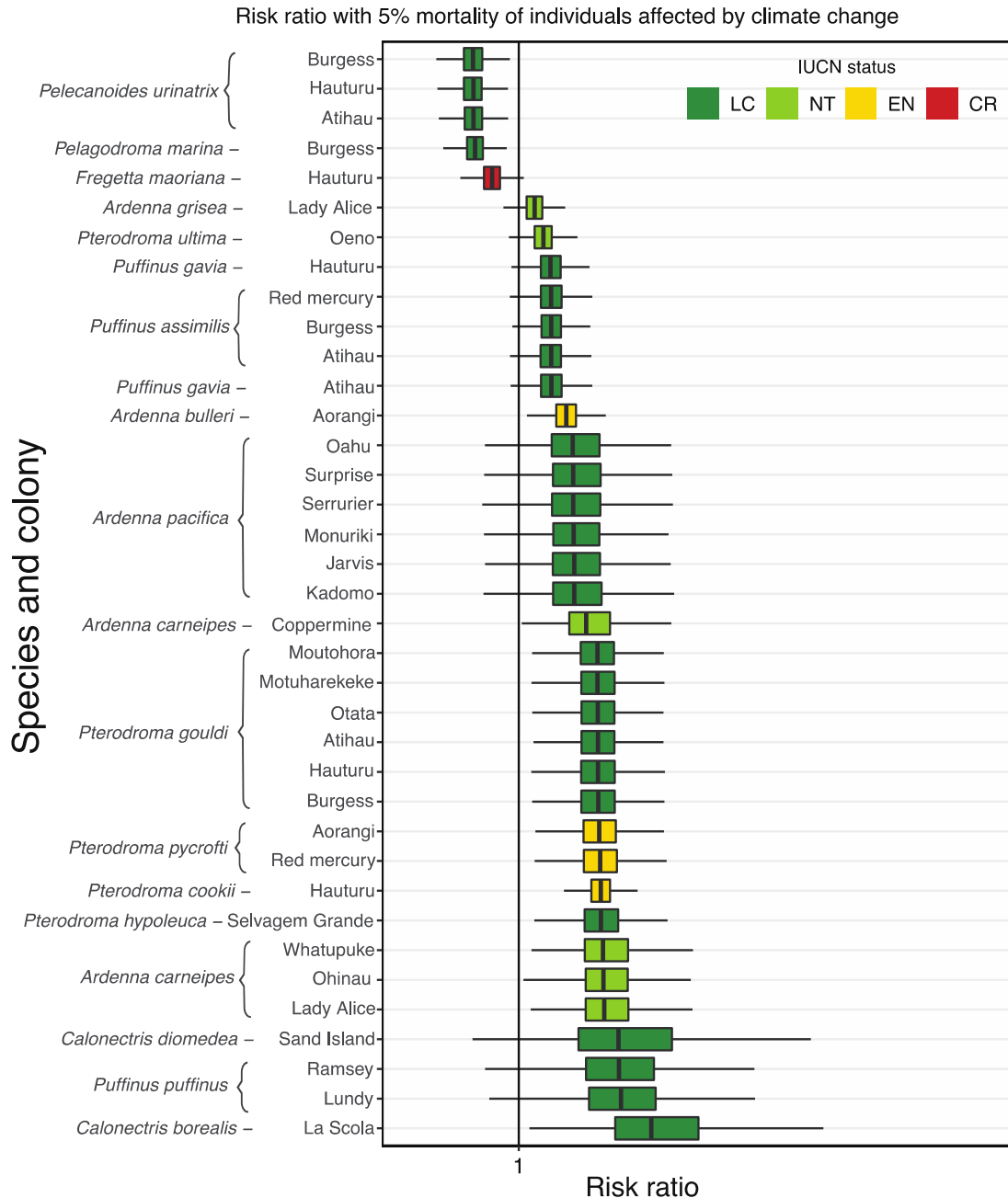


**Fig. S4.** Sensitivity analysis showing the risk for each of the colonies for the 16 species to plastic ingestion mortality at 5% for the proportion of individuals affected. The risk ratio was calculated as potential mortalities yr<sup>-1</sup> / annual mortality threshold (Richard & Abraham 2013); when this risk ratio  $\geq 1$ , adult mortality from each of the evaluated threats may impede the recovery of a colony even after predator eradication. Colors for each species correspond to the IUCN Red List status: LC Least Concern in dark green; NT Near threatened in light green; VU Vulnerable in yellow; CR Critically Endangered in red.

Climate change/Prey depletion risk ratio sensitivity analysis.



**Fig. S5:** Sensitivity analysis showing the risk for each of the 36 colonies for the 16 species to climate change/prey depletion mortality at 1%. The risk ratio was calculated as potential mortalities yr<sup>-1</sup> / annual mortality threshold (Richard & Abraham 2013); when this risk ratio  $\geq 1$ , adult mortality from each of the evaluated threats may impede the recovery of a colony even after predator eradication. Colors for each species correspond to the IUCN Red List status: LC Least Concern in dark green; NT Near threatened in light green; VU Vulnerable in yellow; CR Critically Endangered in red.



**Fig. S6.** Sensitivity analysis showing the risk for each of the 36 colonies for the 16 species to climate change/prey depletion mortality at 5%. The risk ratio was calculated as potential mortalities yr<sup>-1</sup> / annual mortality threshold (Richard & Abraham 2013); when this risk ratio  $\geq 1$ , adult mortality from each of the evaluated threats may impede the recovery of a colony even after predator eradication. Colors for each species correspond to the IUCN Red List status: LC Least Concern in dark green; NT Near threatened in light green; VU Vulnerable in yellow; CR Critically Endangered in red

**TABLE S1.** Species traits ecological traits of at-sea distribution, adult body size, and foraging strategies of pursuit diving and surface filtering predict a species' annual mortality threshold (in bold). The annual mortality threshold is the limit of individuals in a population that can be 'harvested' for the population to remain stable or increase. The phylogenetic generalized least squares models below present the relative influence of the foraging strategies of surface filtering, and pursuit diving, and morphometric variables (adult mean body size) on the annual mortality threshold for the 81 seabird species examined. Pagel's  $\lambda$  is a proxy to phylogenetic signal, with values close to 1 indicating high phylogenetic inertia (i.e. trait under consideration is highly preserved in the species pool). Non- significant results are shown.

Model	t-statistic (df=79)	Pr	R <sub>Adj</sub> <sup>2</sup>	Pagel's $\lambda$
<b>AMT_mean ~ surface_filtering</b>	<b>5.02</b>	<b>2.20E-05</b>	<b>0.24</b>	<b>1.00</b>
<b>AMT_mean ~ pursuit_diving</b>	<b>4.20</b>	<b>0.0004</b>	<b>0.18</b>	<b>0.42</b>
<b>AMT_log ~ range_log</b>	<b>3.66</b>	<b>0.002</b>	<b>0.14</b>	<b>0.742</b>
<b>AMT_log ~ biomass_log</b>	<b>-2.82</b>	<b>0.025</b>	<b>0.091</b>	<b>0.42</b>
AMT_mean ~ scavenging	-1.87	0.169	0.042	0.48
AMT_mean ~ surface_seizing	-1.63	0.261	0.032	0.54
AMT_mean ~ pattering	-1.27	0.398	0.020	0.54
AMT_mean ~ fishfood	-0.99	0.573	0.012	0.55
AMT_mean ~ crustaceans	0.82	0.676	0.008	0.54
AMT_mean ~ pursuit_plunging	0.78	0.688	0.007	0.56
AMT_mean ~ plunging	0.62	0.788	0.005	0.56
AMT_mean ~ dipping	0.47	0.819	0.003	0.55
AMT_mean ~ cephalopods	-0.41	0.819	0.002	0.56
AMT_mean ~ other_inverts	0.36	0.819	0.002	0.55
AMT_mean ~ carrion_birds	0.14	0.921	0.0002	0.56
AMT_mean ~ piracy	-0.13	0.921	0.0002	0.56

**TABLE S2.** Annual mortality threshold model validation Tukey's Test results (p=0.0001). See Main text; Fig. 3A.

IUCN Category	Means	Group
Least Concern (LC)	10.93	a
Near Threatened (NT)	9.32	ab
Vulnerable (VU)	7.93	bc
Endangered (EN)	7.68	bc
Critical (CR)	4.76	c

## Supplementary discussion

### *Model limitations*

While models such as ours are highly sensitive to parameter uncertainty, and may inadequately account for demographic species-specific variation (Dillingham & Fletcher 2011, Richard *et al.* 2017), in the absence of empirical data on population level impacts, they can be used to help inform conservation management actions and prioritize species or site specific monitoring (Niel & Lebreton 2005, Robertson *et al.* 2014). The key assumptions in the risk ratio model include: the target species has constant adult survival, operates at low densities,  $\lambda_{\max}$  is constant across generations, and female fecundity is constant from age of first maturation (Niel & Lebreton 2005, Dillingham 2010, Dillingham *et al.* 2016). Further, our model is unable to capture species specific nuances in behaviour and life-stage, which will likely influence the resilience of a species to a threat. For example, immature birds have a higher probability of dying in fisheries bycatch than breeding adults (Genovart *et al.* 2017). Similarly, young and immature birds are more likely to have higher loads of plastic ingested (van Franeker & Law 2015). Some species are more gregarious when foraging, thus interactions with fisheries operations, or other human activities are likely to cause additional adult mortalities (Genovart *et al.* 2017).

In the same way that oceanic features vary across latitudes and water masses influencing resource distributions for seabirds, the intensity or existence of a threat is not distributed evenly (Ryan 2016). Species that have large spatial distributions are likely to have variable population level responses to marine threats due to differences in spatial exposure, interspecific phenology, and dispersal patterns (e.g., climate change, Genovart *et al.* 2017). Complicating the strength of range as a risk predictor for seabirds is environmental stochasticity, which is closely linked to demographic stochasticity. That is the random variation of population dynamics due to discrete events (i.e., changes to births and deaths from variable environmental factors, such as climate anomalies, prey availability (Tuljapurkar 1990). Environmental stochasticity is widely recognized as being an important consideration in population growth models, particularly with small populations, where one event has the potential for catastrophic results (Weimerskirch

2001). Thus, the effects of environmental stochasticity on vital rates for small populations, coupled with anthropogenic sources of adult mortality or reductions in reproductive output due to poor body condition (i.e., plastic ingestion related) may be more pronounced (Lebreton & Clobert 1991).

**Supplement 2, 3 (separate files)**

**Supplement 2\_colonyriskanalysis.pdf:** Model inputs and estimates for the colony risk assessment for 36 colonies of 16 species and the impact of marine threats to recovery post-predator eradication.

**Supplement 3\_traitanalysis.pdf:** Parameter estimates including 81 procellariiform seabird species for the phylogenetic generalized least squares regression analysis to evaluate the influence of shared traits on the annual mortality threshold of a species.

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