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journal homepage: www.elsevier.com/locate/bioconPelagic citizen science data reveal declines of seabirds off south-eastern Australia[☆]

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ABSTRACT

Many seabird communities are declining around the world, a trend frequently linked to climate change and human impacts on habitat and prey. Time series observations of seabirds away from breeding colonies are generally rare, which limits our understanding of long-term changes for conservation actions. We analysed a dedicated citizen science dataset of pelagic seabird abundance (86 species – 30 used for modelling analysis - from 385 trips) from two locations over 17 years (2000–2016) and a third for seven years, over the continental shelf and slope of south-eastern Australia. To estimate temporal trends and environmental drivers, we used generalised additive modelling and species archetype modelling for groups. Almost half (43%) of the most abundant seabird species declined in our study area over the 17 years. The declines may be associated with human-induced ecosystem change and represent poleward shifts in distribution out of our study area, changes in population abundance, or both. Winter-dominant groups, primarily species rarely frequenting warmer water, were often negatively associated with SST_{anom} , while summer-dominant groups, composed of species more tolerant of temperate and tropical environments, were generally positively associated with SST_{anom} . Widespread local declines in seabird populations are of increasing concern. Understanding the extent to which these observed declines represent real declines in abundance, or range shifts, should be a priority. Changing sea temperatures are probably contributing to both. These results from the coast of south-eastern Australia need to be placed in the context of the highly mobile study organisms and the vast spatial scale of the ocean. Long-term citizen science observations, from an array of locations around the world, promise to provide valuable insights into seabird ecology, playing a key part in seabird conservation.

1. Introduction

Seabirds are critical components of marine ecosystems. They disperse plants, replenish nutrients in succession processes on islands (Lorrain et al., 2017), prey on marine resources (Chambers et al., 2011) and scavenge (Magnusson and Magnusson, 2000; Sekercioglu, 2006;

Veit and Harrison, 2017). They also exhibit a diverse range of life-histories, reflected in their varied foraging guilds and differential responses to environmental drivers. Because seabirds are so widely distributed, collectively dependent on a wide range of food-sources, and highly sensitive to anthropogenic threats, an understanding of their distribution and abundance is a key integrating piece of information

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used to inform our understanding of the status of our ocean ecosystems (Grémillet and Boulinier, 2009).

Long-term trends of seabirds depend in part on their shifts in distribution and abundance in response to changes in environmental factors (Ainley et al., 2005; Cox et al., 2018; Goyert et al., 2016) including sea surface temperature, salinity, wind speed, wind direction, and plankton distributions (Abrams, 1985; Pakhomov and McQuaid, 1996; Pocklington, 1979; Quillfeldt et al., 2015; Schneider, 1990). For example, sea surface temperature differentially influences foraging distributions of Antarctic prions (*Pachyptila desolata*), slender-billed prions (*Pachyptila belcheri*), and blue petrels (*Halobaena caerulea*) in the Southern Ocean (Quillfeldt et al., 2015). Elsewhere, high wind speeds affect foraging and migration patterns of different seabird species (Gibb et al., 2017; Pennycuik, 1969; Weimerskirch et al., 2016). Frontal zones, where warm and cool ocean currents intercept, also create eddies and upwellings, providing productive foraging areas for seabirds (Abrams, 1985; Cox et al., 2018; Kai and Marsac, 2010; Kai et al., 2009; Pakhomov and McQuaid, 1996; Schneider, 1990).

While they have been traditionally regarded as remote (Merino et al., 2012), these pelagic environments are increasingly affected by humans via overfishing (McCauley et al., 2015; Merino et al., 2012; Worm et al., 2006), ocean pollution, invasive predators, climate change, and coastal land degradation (Croxall et al., 2012). The harvest of forage fish, in particular, can affect the food supply of top predators, including seabirds (Cury et al., 2011; Sydeman et al., 2017). Therefore, the conservation of seabirds requires an oceanographic context to interpret anthropogenic impacts such as fishing or a warming ocean.

The vast majority of our understanding of seabird populations comes from monitoring and restoration of seabird colonies (Bakker et al., 2018; Brooke et al., 2018; Priddel et al., 1995, 2000; Prince et al., 1994). However, this is only one stage of the seabird lifecycle and without data from their time at sea (e.g., breeding and foraging), an understanding of anthropogenic impacts on seabirds will remain incomplete (Croxall et al., 2012). Full-annual cycle research is essential for proper conservation efforts (Marra et al., 2015). While focus on successful breeding populations is critical, so too is understanding the factors affecting seabird distribution and movements at sea (Grémillet and Boulinier, 2009), with the potential to inform prioritized marine conservation areas (Lascelles et al., 2016).

Currently, data on seabirds at sea are patchily distributed across the world: most seabird research comes from developed countries in the Northern Hemisphere (Aragones et al., 1997; Mott and Clarke, 2018), leaving knowledge gaps in other parts of the world. For example, relatively little research has investigated the drivers and trends in high seabird diversity regions of the south-west Pacific (Mott and Clarke, 2018) or the global warming hotspot of south-eastern Australia (Cai et al., 2005; Popova et al., 2016; Wu et al., 2012). Current understanding of seabird dynamics in these regions has relied on at-sea observations or tracking data (e.g. Berlincourt and Arnould, 2015; Chambers et al., 2011; Jungblut et al., 2017; Miller et al., 2018; Priddel et al., 2014; but see Brandis et al., 1991), generally at fine temporal and spatial extents. But, broad, generalisable trends informing conservation applications are still lacking.

There are new sources of information that could fill this gap: long-term and large spatial scale data are increasingly available through citizen science initiatives (Bonney et al., 2014; Callaghan et al., 2018; Kobori et al., 2016), providing new opportunities to answer ecological and conservation questions (e.g. Freeman and Miller, 2018; McCormack et al., 2010; Sparks et al., 2017). The quality of these data is comparable to that collected professionally (Aceves-Bueno et al., 2017), allowing informed conservation measures, especially when augmented with rigorously collected ecological data (Callaghan and Gawlik, 2015; Sullivan et al., 2017). Yet studies demonstrating the validity of these citizen science data are still necessary for broad-scale uptake by ecologists and conservationists (Burgess et al., 2017). Concomitantly, publicly-available ocean data are increasing in quality and extent (Riser

et al., 2016), providing an oceanographic context to citizen science observations (Faghmous et al., 2015). However, up until now, the integration of broad-scale citizen science seabird data with publicly available ocean data has been sparse.

We curated and analysed a unique temporal and spatial dataset, regularly collected by skilled, voluntary birdwatchers (385 pelagic trips, 2000–2016), recorded year-round from three locations in pelagic waters of south-eastern Australia – an important foraging region for a high diversity of seabirds (Mott and Clarke, 2018). We evaluated temporal trends in seabird abundance, as well as associations with potential oceanographic drivers (i.e., sea-surface temperature anomaly, chlorophyll-a concentrations, wind speed and eddy kinetic energy). We also applied a species mixing analysis to group seabird species with similar patterns in abundance, highlighting the possibility of functional groups for improving seabird monitoring and conservation (Dunstan et al., 2011). Specifically, we had three objectives: (1) to identify relationships of different environmental and temporal variables on abundances of species and groups of seabirds; (2) to identify overall trends of seabird abundances and; (3) to qualitatively compare these trends to trends in breeding colonies.

2. Material and methods

2.1. Seabird data and study area

Our study area encompassed three sites along the south-east Australian coast (Fig. 1): Wollongong (including Kiama) (34°33'S, 150°54'E), Sydney (33°50'S 151°18'E), and Port Stephens (including Swansea) (32°44'S 152°13'E). During pelagic trips, the boat traversed in a south-easterly direction to and from the edge of the continental shelf (50–65 km from port, Fig. 1), and every observed seabird was identified, to species if possible, by experienced observers. At the shelf, the boat remained for 2–4 h of further observation before returning. Chumming was used to attract birds during the pelagic trips which may bias sampling, but we note that this bias was systematic, affecting all long-term observations and trends in abundance. Further, many birds were also observed in transit. About nine trips occurred during the year, varying in frequency among sites with an average of seven, six and thirteen trips per year respectively from Port Stephens, Sydney and Wollongong. We collated, error-checked, and organised records (courtesy of Southern Ocean Seabird Study Association (SOSSA - <http://www.sossa-international.org/>)) from a total of 385 pelagic trips including respectively 226, 108, 51 trips from Wollongong (2000–2016), Sydney (2000–2016) and Port Stephens (2010–2016).

In this study area, the oligotrophic East Australian Current (EAC) drives most marine ecosystem processes as it flows poleward (Suthers et al., 2011), before diverting east and forming the Tasman Front (Hamilton, 2006) while the remainder of the EAC flows south as a field of eddies (Everett et al., 2012). Wind, current-driven upwellings, and increased eddy occurrences increase productivity of plankton and fishes along this current, attracting seabirds and other marine life (Everett et al., 2014; Kai and Marsac, 2010; Ridgway, 2007; Shulzitski et al., 2016).

2.2. Bird data

In order to have enough information to build reasonable models, we selected for analysis the 30 most common seabird species (Table 1) out of the total 86 species (Table A1). An analysis of total seabird abundance was not done, as this would be driven by few, abundant species (especially shearwaters, see Table 1 for maximum counts). We also excluded six species which were predominantly distributed near the coast and not expected to respond to our drivers: silver gull (*Chroicocephalus novaehollandiae*), kelp gull (*Larus dominicanus*), greater crested tern (*Thalasseus bergii*), common tern (*Sterna hirundo*), white-fronted tern (*Sterna striata*) and little penguin (*Eudyptula minor*). Finally, we

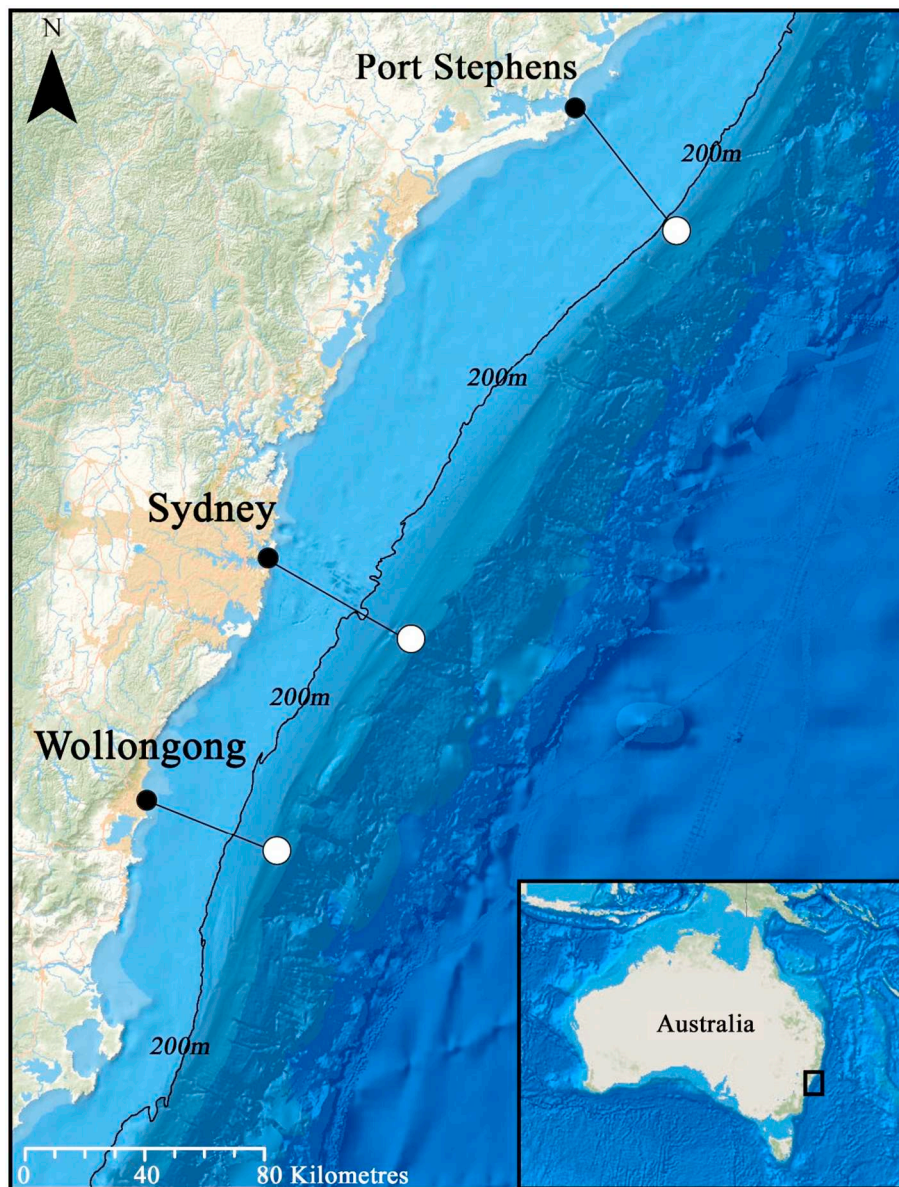


Fig. 1. Approximate pelagic routes from Wollongong (2000–2016), Sydney (2000–2016) and Port Stephens (2010–2016) (black dots) to and from the continental shelf (white dots). Approximate 200 m isobath is marked representing the continental shelf.

also compared our trends in pelagic counts (i.e., abundance) with breeding trends of the 30 modelled seabird species (Table 1), based on literature reviews and elicited expert opinion. These trends were identified from accessible literature and, where possible, taken from local colonies or those from which the majority of birds in the study area were expected to breed. These represented the trend in abundance at monitored breeding colonies, primarily derived from surveys of adult birds.

2.3. Environmental data

We used a combination of temporal, spatial, and oceanographic variables to describe patterns in seabird abundance, analysing trends in seabird abundance and occurrence (presence/absence), in relation to our environmental drivers, at species and multi-species levels. The seven variables evaluated were: Day-of-year (*DOY*; 1–365; continuous), *Time* (decimal years starting from 22/01/2000; continuous), SST-anomaly (SST_{anom} ; degrees C; continuous), chlorophyll-a (*Chl*; $mg\ m^{-3}$; continuous), an index of eddy activity and turbulent flow, Eddy Kinetic

Energy (*EKE*; $cm^2\ s^{-2}$ continuous), mean wind speed (*Wind*; $km\ h^{-1}$; continuous), and *Site* (categorical; Port Stephens, Sydney, Wollongong). Other variables were evaluated, including sea-surface temperature (SST), current speed and mean sea-level anomaly, but were not included due to collinearity with included variables. The variable included from a collinear pair or set was the variable with the highest information content (as measured by a decline in model Akaike information criterion (AIC)). Wind direction was expected to have a significant effect on seabird abundance and presence, however as we could not separate the effect of seasonality from wind direction, it was not analysed. For details on the variables and their sources, see Table B1. Environmental data were obtained and averaged for a $\sim 100\ km^2$ region ($\sim 10\ km \times \sim 10\ km$) with the south-east corner aligned with the end of the NW-SE transect to ensure the box bisected the transect (Fig. 1) and encompassed the variation in trip routes to the shelf-edge.

2.4. Statistical analyses

We used generalised additive models (GAMs) for single species

Table 1

List of the 30 most recorded seabird species and the number of records from pelagic trips off Sydney and Wollongong (2000–2016) and Port Stephens (2010–2016), excepting coastal and not primarily pelagic species (e.g. silver gull, common tern, little penguin, see Appendix A for records and counts for all species, and Appendix B for breeding trend justifications). Group number (#) represents the groups or archetypes species were assigned to using species-archetype modelling (see Fig. 3a–b). South-eastern Australia trends were derived from individual species GAMs (see Appendix D). Trends marked with ^{NS} represented those which had a non-significant relationship between counts and time ('days' in GAM outputs; Appendix D). Breeding colony trends were identified from relevant literature (see Appendix E).

Common name (group #)	Scientific name	No. records	Max. count	SE-Aus. trend	Breeding colony trend
Australasian gannet (2)	<i>Morus serrator</i>	355	650	Decreasing	Uncertain
Fluttering shearwater (1)	<i>Puffinus gavia</i>	312	15,000	Decreasing	Stable
Wedge-tailed shearwater (4)	<i>Ardeanna pacificus</i>	287	3500	Decreasing	Uncertain
Shy albatross (6)	<i>Thalassarche cauta</i>	275	31	Increasing	Uncertain
Black-browed albatross (5)	<i>Thalassarche melanophrys</i>	272	150	Increasing	Increasing
Providence petrel (2)	<i>Pterodroma solandri</i>	261	75	Stable	Increasing
Short-tailed shearwater (11)	<i>Puffinus tenuirostris</i>	258	35,000	Stable ^{NS}	Decreasing
Grey-faced petrel (4)	<i>Pterodroma gouldi</i>	231	250	Decreasing	Uncertain
Hutton's shearwater (1)	<i>Puffinus huttoni</i>	230	1200	Stable	Uncertain
Indian yellow-nosed albatross (10)	<i>Thalassarche carteri</i>	216	180	Decreasing	Decreasing
Flesh-footed shearwater (7)	<i>Ardeanna carneipes</i>	205	600	Decreasing	Uncertain
Wilson's storm-petrel (8)	<i>Oceanites oceanicus</i>	184	1000	Stable	Uncertain
Pomarine jaeger (3)	<i>Stercorarius pomarinus</i>	182	125	Decreasing	Uncertain
Campbell albatross (6)	<i>Thalassarche impavida</i>	175	50	Increasing ^{NS}	Increasing
Wandering albatross (5)	<i>Diomedea exulans</i>	160	30	Decreasing	Decreasing
Antipodean albatross (1)	<i>Diomedea antipodensis</i>	141	55	Decreasing	Decreasing
Brown skua (2)	<i>Stercorarius antarcticus</i>	141	20	Stable ^{NS}	Stable
Arctic jaeger (3)	<i>Stercorarius parasiticus</i>	140	11	Decreasing	Uncertain
Sooty shearwater (4)	<i>Ardeanna grisea</i>	134	50	Stable	Decreasing
Fairy prion (9)	<i>Pachyptila turtur</i>	128	550	Decreasing	Stable
White-faced storm-petrel (8)	<i>Pelagodroma marina</i>	105	112	Increasing	Decreasing
Northern giant petrel (2)	<i>Macronectes halli</i>	100	10	Increasing	Increasing
Buller's albatross (2)	<i>Thalassarche bulleri</i>	97	40	Increasing	Increasing
Cape petrel (1)	<i>Daption capense</i>	97	345	Decreasing ^{NS}	Stable
Long-tailed jaeger (3)	<i>Stercorarius longicaudus</i>	76	23	Decreasing	Stable
Southern giant petrel (1)	<i>Macronectes giganteus</i>	69	12	Decreasing	Uncertain
Sooty tern (12)	<i>Onychoprion fuscatus</i>	39	450	Decreasing	Uncertain
Black petrel (4)	<i>Procellaria parkinsoni</i>	35	6	Stable ^{NS}	Uncertain
Buller's shearwater (3)	<i>Ardeanna bulleri</i>	28	3	Decreasing ^{NS}	Uncertain
Gould's petrel (7)	<i>Pterodroma leucoptera</i>	25	16	Increasing ^{NS}	Stable

analysis, given strong seasonality of many seabirds' abundance and occurrence (Brandis et al., 1991). GAMs were ideal for modelling data with multiple temporal signals, including residual temporal autocorrelation (Wood, 2017). Species were also grouped to reflect similar responses to the environmental variables, modelling them using species mixture modelling, with hierarchical methods which clustered species into groups that respond similarly to the predictor variables, followed by generalised linear modelling (GLM) which fits a single model to represent how an 'archetype' of each species group responded to environmental variables (Dunstan et al., 2011). This approach has been termed 'species archetype modelling' (Hui et al., 2013), and it improves confidence in describing the archetype response, borrowing strength from common species with high information content (Hui et al., 2013), and simplifying the assemblage to assist management and conservation applications (Hui et al., 2013; Woolley et al., 2013). This is also a valuable model-based approach for identifying components of a species assemblage that are not easily clustered due to large inter-specific variation in environmental responses. GAM and mixture GLM approaches were fundamentally similar, relying on the same data preparation. *Site* was fitted as a fixed factor because our study made inferences across the entire study area as random factors with less than five levels can be inaccurate (Bolker et al., 2009; Harrison et al., 2018).

2.5. Single-species analysis

Following standard practices, the four environmental variables were standardised before analyses, which means that their coefficients fitted by the GAMs and GLMs are interpreted as the unit change in the response per unit change in the standard deviation of the environmental variable. Single species abundance and occurrence (presence-absence) were fitted to negative-binomial (log link) and binomial (logit link)

GAMs, respectively, taking the following form:

$$y = s(DOY) + s(Time) + SST_{anom} + Chl + EKE + Wind + Site,$$

where *s* indicated a smoother (cyclic cubic regression spline for *DOY*, and thin plate regression spline for *Time*). The basis dimension of these smoothers was fixed at *k* = 6 for *DOY* and *k* = 3 for *Time* to avoid overfitting the general trends. Temporal autocorrelation was evaluated by including a continuous-time AR1 structure (corCAR1) of *Time*, nested in *Site*, using a GAMM. It had little influence on the fitted model, not altering the significance of the *Time* variable for any species, and so was not included in the final analysis. This improved comparison between GAMs and GLMs (corCAR1 was not possible in the mixture modelling GLM). Residuals were examined for normality and homoscedasticity to evaluate the suitability of smoothed variables and the negative binomial family. GAM (and GAMM) modelling was done using the 'mgcv' package (Wood, 2011) in R (v3.5.0; R Core Team, 2018).

2.6. Species mixture modelling

We used a negative-binomial GLM for species' mixture modelling of seabird abundance:

$$y = \cos\left(\frac{2\pi}{365}DOY\right) + \sin\left(\frac{2\pi}{365}DOY\right) + Time + Time^2 + SST_{anom} + Chl + EKE + Wind + Site.$$

Given this approach used GLMs, the variables with non-linear responses (*DOY* and *Time*; represented by smoothers in the GAMs) were modelled with a cyclical cosine-sine function for *DOY*, and a quadratic term for *Time*. Comparison of fitted GLMs and GAMs showed that both types of model fitted the data similarly well. We acknowledge that by using a quadratic term for *Time* in the GLMs, and a *k* = 3 basis for the *Time* smoother in the GAMs, that the *Time* variable represented a long-term temporal trend that was monotonic or oscillated at a frequency of

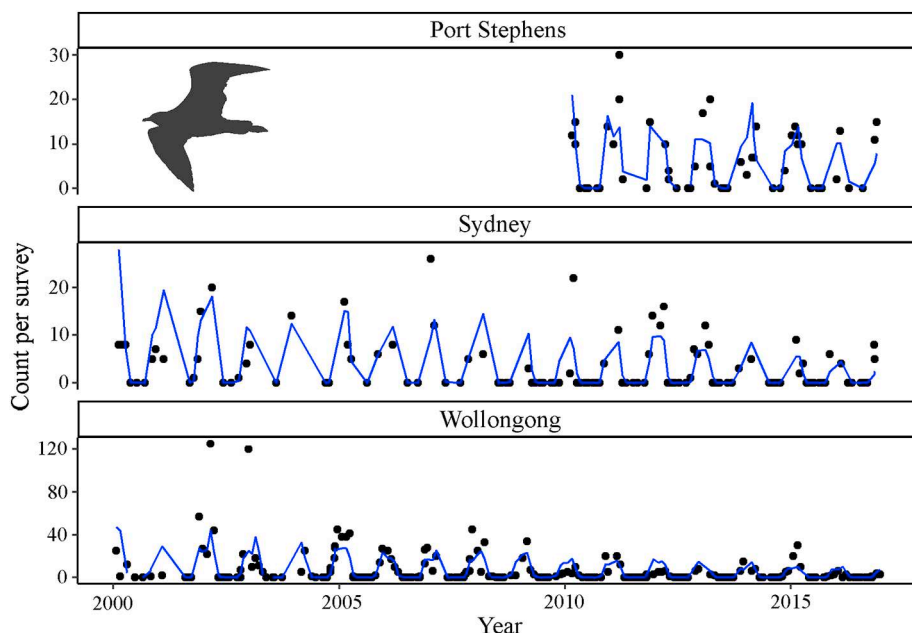


Fig. 2. Example figure of single-species (pomarine jaeger) counts (black dots) at the three study sites (Port Stephens, Sydney and Wollongong), and the fitted counts from the negative-binomial GAM (line). Site was included as a categorical variable in the models. Similar figures for all of the 30 analysed species can be found in Appendix C, Fig. C1, and comparisons of overall long-term trends and breeding colony trends can be found in Table 1.

at least ~30 years. The Time variable was fitted to the entire time series, but due to the different lengths of the time series among sites, the 2000–2009 part of the *Time* variable was fitted to fewer data (and with reduced spatial coverage) than the 2010–2016 part. This meant that we were more confident about the latter part of the time series, however as we generalised our results across all sites, inferences across the entire series remain valid. Species' mixture modelling was done using the 'SpeciesMix' R package (Dunstan et al., 2011). To determine the ideal number of archetypes for species' clustering, the full model was run with varying numbers of archetypes, identifying the best number from the model, using Bayesian Information Criteria (BIC). We also based this decision on the number of archetypes with one species, and the probability of membership in these archetypes, to avoid fitting too many archetypes (Dunstan et al., 2011). Fitted variable coefficients for each archetype from the best model were compared against their standard error, and coefficients with high standard error were not considered important (Woolley et al., 2013). Prediction was used to visualise the non-linear responses for *DOY* and *Time*, whereby a prediction for one year (*DOY*) or for the entire time series (*Time*) was made with environmental variables held at their median values. Predictions were each rescaled from 0 to 1 to represent proportional change from maximum predicted seabird abundance, which removed the influence of *Site* and allowed better comparison of the predicted responses between the GLMs and GAMs.

3. Results

Eighty-six species were recorded on 385 SOSSA pelagic trips from 2000 to 2016 (Table A1). Fifty percent of all eighty-six species were recorded fewer than ten times, considered to be rarities or vagrants in the dataset. The most frequently recorded species was Australasian gannet with 355 records, while the least recorded species in the top thirty was Gould's petrel, with 25 records (Table 1). Shearwaters were the most numerous and drove overall trends, with four species recording abundances greater than 1000 on any single trip (short-tailed shearwater, fluttering shearwater, wedge-tailed shearwater and Hutton's shearwater; Table 1).

3.1. Single species

There was substantial variation among species in the amount of

deviation explained by GAM modelling of abundance (see fitted GAM for pomarine jaeger, Fig. 2; other species' fitted GAM figures, Fig. C1). Some species showed strong seasonal trends (e.g. pomarine jaeger, southern giant petrel, sooty tern) while others showed relatively little variation in abundance throughout the year (e.g. fluttering shearwater and Australasian gannet; Fig. D3). Abundance varied considerably in seasonality and occurrence among the 30 most common species, and among sites for 17 species (Appendix D). There was significant seasonal (*DOY*) and long-term (*Time*) variation in abundance of 29 and 22 species respectively (Appendix D). The strongest seasonal abundance signals (i.e., modelled counts) were for short-tailed shearwater and Wilson's storm-petrel while the strongest long-term abundance signals were exhibited by wandering albatross and cape petrel. There was also significant seasonal and long-term variation in occurrence (i.e., presence/absence) for 30 and 12 species respectively. Wilson's storm-petrel and providence petrel demonstrated the strongest seasonal fluctuation, while wandering albatross and cape petrel exhibited the strongest long-term occurrence trends (both decreasing) over the 17 years (Table D1). This reduction in seabird species showing significant long-term trends in occurrence indicates that seabird presence was more stable than abundance over the 17 years.

Abundance of many seabird species was significantly related to environmental variables (see details in Appendix D). Abundance of 10 species was significantly related to SST_{anom} : five positively and five negatively, with sooty tern showing the strongest positive relationship and long-tailed jaeger the strongest negative relationship. Abundance of another four species was significantly related (positively) to *Chl*, with white-faced storm-petrel abundance most strongly related. Abundance of only two species was significantly related to *EKE*: fluttering shearwater and Wilson's storm-petrel, respectively positively and negatively related. Abundance of another eight and two species was respectively significantly positively (Gould's petrel strongest relationship) and negatively related to *Wind* (cape petrel strongest relationship). GAMs explained different amounts of variation in abundance across the species, but these models had generally explained high deviance (Appendix D supplies full model results).

There were significant temporal trends in abundance of twenty-two (73%) of the species analysed. Abundance for 13 species (43%) decreased; five species (17%) increased; and four species (13%) were stable/variable (Fig. C1; Table D1). Breeding colony data from the literature and other available monitoring data (Table E1) suggested that,

of the 17 species with sufficient data to show trends, seven, five and five were respectively decreasing, increasing and stable/variable. Five albatrosses (excluding shy albatross which showed uncertain breeding trends and Campbell albatross which showed non-significant agreement) and northern giant petrel had consistent significant trends and in the same direction, between our study and those of others (Table 1). For the remaining thirteen species, there were no documented trends because of either high variability or insufficient data (Table 1; Table E1).

3.2. Species groupings

We identified 12 archetypes, or seabird groups, clustered from the 30 species (evaluated using BIC; see Fig. F1), with eight groups of one or two species, and four groups of more than two (Fig. 3a–b; Appendix F). The former were not easily clustered, with their diverse seasonal and inter-annual trends in counts and responses to the environment. Three of these groups were summer-associated (3, 4, 7), four were winter-associated (1, 2, 5, 6) and one showed a broad seasonal abundance

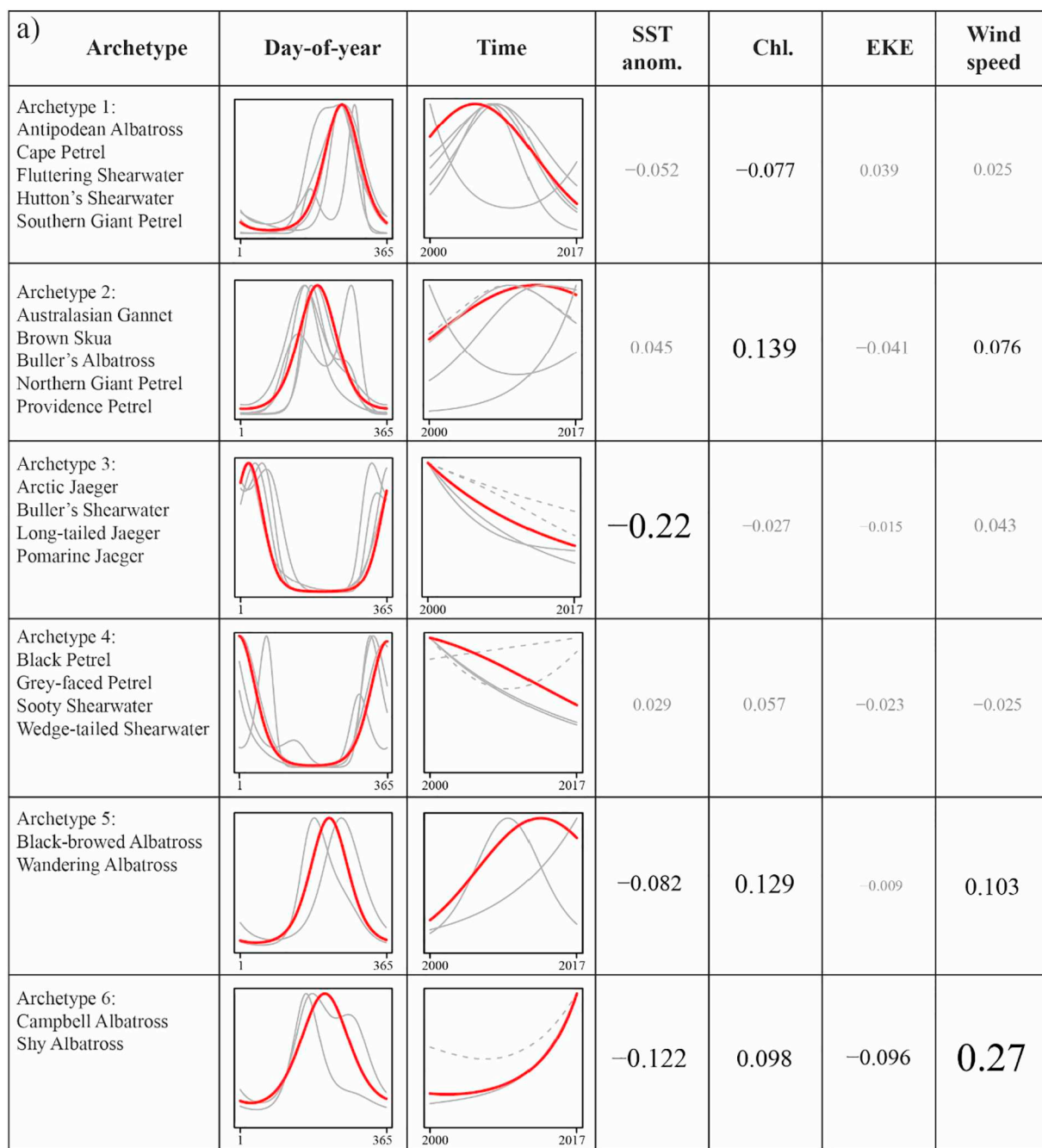


Fig. 3. Archetypes (i.e. groups) of seabird species, with similar responses to environmental variables after species mixture modelling of counts – groups 1–6 (a) and groups 7–12 (b). ‘Day-of-year’ (one year) and ‘Time’ (entire time series), with non-linear responses, represented by mean response for the group (red line, fitted by the mixture GLM), with responses of each species (grey lines, fitted by a GAM (non-significant smoothers are dashed)). All other variables were held at their median values, and predictions of each rescaled from 0 to 1 to represent proportional change from maximum predicted seabird counts. Variables included: sea-surface temperature anomaly (SST anom.), chlorophyll-a concentration (Chl.), eddy kinetic energy (EKE) and wind speed. For linear-response variables, the mean coefficient, fitted by the mixture model, shown for each group (coefficients considerably smaller than mean value shown in grey), with font size proportional to coefficient magnitude. See Appendix F for full mixing model results. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

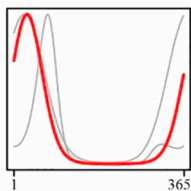
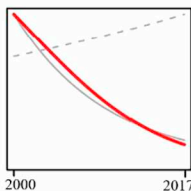
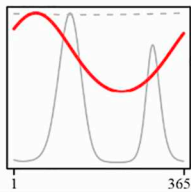
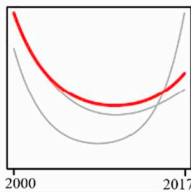
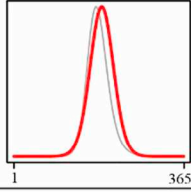
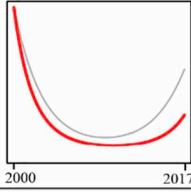
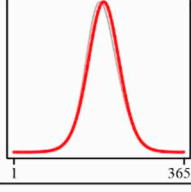
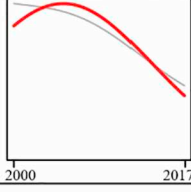
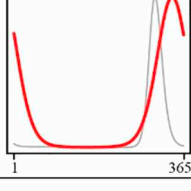
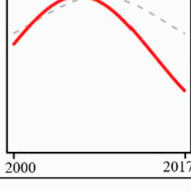
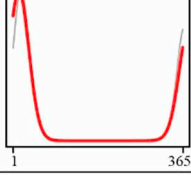
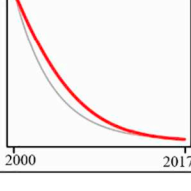
b) Archetype	Day-of-year	Time	SST anom.	Chl.	EKE	Wind speed
Archetype 7: Flesh-footed Shearwater Gould's Petrel			0.40	-0.158	0.022	0.028
Archetype 8: White-faced Storm-petrel Wilson's Storm-petrel			0.31	0.5	-0.30	0.189
Archetype 9: Fairy Prion			-0.25	0.27	0.029	0.133
Archetype 10: Indian Yellow-nosed Albatross			-0.119	0.149	-0.087	0.031
Archetype 11: Short-tailed Shearwater			0.31	0.160	0.204	0.7
Archetype 12: Sooty Tern			0.9	-0.32	0.098	-0.4

Fig. 3. (continued)

trend (8; Fig. 3ab). Abundance of all fitted seabird groups mostly decreased (1, 3, 4, 7, 8) but some increased (2, 5, 6) (Fig. 3a–b), over the 17 years. Comparison of group models and single-species GAM smoothers showed that the structure of the GLM described the non-linearity of *DOY* and *Time* well (Fig. 3a–b).

These groups also responded positively and negatively to environmental variables, with stronger mean responses for groups of two species, than other groups with more species. Day-of-year explained most variation in abundances of archetypes, followed by the four environmental variables, and then *Time* (Table F3). Groups of only one species could be interpreted equally by looking at the single-species GAMs (i.e. Appendix D), but for compatibility are included in Fig. 3a–b. There was no obvious general connection between the effect of SST_{anom} and temporal trends in abundance. In general, all winter-associated groups were generally negatively associated with SST_{anom} or minimally positive

(Fig. 3a–b), while all summer-associated groups were positively associated with SST_{anom} (except for group 3, negative association).

4. Discussion

Seabird populations are in decline around the globe, with 30% of global seabird species now threatened (Croxall et al., 2012; Spatz et al., 2017). Our analysis contributes to this concerning trend, with thirteen (43%) of the thirty seabird species for which we had sufficient data declining in abundance, as did five of the eight (63%) seabird groups with more than one species. Of the five declining groups, four (1, 3, 4 and 7) showed strong declines over the 17-year study-period (noting that a group may decline while one or more constituent species do not e.g. Gould's petrel in group 7).

Within these groups, species declining in south-eastern Australia

included Antipodean albatross, arctic jaeger, cape petrel, flesh-footed shearwater, fluttering shearwater, grey-faced petrel, Hutton's shearwater, pomarine jaeger, southern giant petrel and wedge-tailed shearwater. Declines in three albatross species (Antipodean albatross, Indian yellow-nosed albatross and wandering albatross) and increases in two albatross species (black-browed albatross and Buller's albatross) and northern giant petrel reflected similar trends in breeding colonies (Table 1), suggesting that citizen science data may track changes in abundance of seabirds away from their breeding grounds. Clearly, data from pelagic trips represent new lines of evidence for seabird population trends, limited to species encountered in the surveyed region. We note that trends from the breeding colonies could also be related to local factors or trends of seabirds which were not foraging at sea in the study area.

As well as real declines in abundance, some species may have declined on our surveys because of a range shift in their foraging behaviour, given distributional changes in environmental variables (e.g., Ridgway, 2007), or a combination of declining abundance and range shifts. This was reinforced by our data analyses, strongly indicating that environmental drivers are changing seabird distribution and abundances (Appendix D; Fig. 3a–b; Table F2), as in other parts of the world (Cox et al., 2018; Goyert et al., 2016). In particular, climate change is increasingly altering abiotic drivers as our oceans are increasingly affected by warming waters (Cai et al., 2005; Popova et al., 2016; Wu et al., 2012). This was reflected in a strengthening East Australian Current, increasing the flow of warm, oligotrophic water southwards along the coast of eastern Australia (Everett et al., 2012; Ridgway, 2007; Suthers et al., 2011). This has shifted the distribution of fish populations (Brander, 2010), as some species shift south with climate change and the strengthening East Australian Current (Ridgway, 2007).

As seabird populations and distributions are strongly linked with their prey (Barbraud et al., 2018; Frederiksen et al., 2006), warming and less-productive ocean conditions are likely to have negative consequences for many seabird populations, highlighting the need for conservation of our marine ecosystems. Our summer-associated groups of more than one species (groups 3, 4 and 7, Fig. 3a–b) exhibited negative long-term trends, in response to different environmental drivers. Group 7 (flesh-footed shearwater and Gould's petrel) associated with positive SST_{anom} values (warmer than average water), suggesting warming sea-surface temperature may not be the key driver of group-level declines for this group; Group 3 (representing summer species: Arctic jaeger, pomarine jaeger, long-tailed jaeger and Buller's shearwater) was entirely associated with negative SST_{anom} values (cooler than average water), suggesting that this group is already responding to a warming ocean given the strong evidence of abundance declines over the 17 year period; and Group 4 (black petrel, grey-faced petrel, sooty shearwater, wedge-tailed shearwater) was not associated with SST_{anom} (group 4; Fig. 3a–b), suggesting group-level declines are more closely linked to factors other than warming waters. Winter-associated groups of more than one species were more often associated with negative SST_{anom} values (cooler than average water). Some were also associated positively with Chl , and negatively with EKE . The latter factor reflects the strengthening East Australian Current, increasing eddy activity and transport (Oliver and Holbrook, 2014; Rykova and Oke, 2015). However, winter-associated groups 5 and 6 with only albatross species which breed and forage mostly in cooler waters of the Southern Hemisphere (Baker et al., 2007; Wakefield et al., 2011; Weimerskirch et al., 1994; Sztukowski et al., 2018) showed group-level increases over the study period (Fig. 3a). As only two of the four constituent species showed significant trends (black-browed albatross increased and wandering albatross decreased over the 17 year study period), we recommend further investigation of the impacts of warming sea surface temperature on cold water species, which is dependent on high quality long-term datasets.

Long-term data are essential for proper monitoring and conservation (Lindenmayer et al., 2012), and adaptive monitoring requires an

ability to incorporate updated data, as well as to refine data collection techniques (Lindenmayer and Likens, 2009). Further, long-term citizen science data provide opportunities to investigate the full-annual cycle of seabirds (Marra et al., 2015), by complementing the wealth of tracking and breeding colony data already widely utilised in seabird conservation research (Berlincourt and Arnould, 2015; Chambers et al., 2011; Jungblut et al., 2017; Miller et al., 2018; Priddel et al., 2014; see Table E1). Our citizen science data, when integrated with environmental variables, provided a new data-rich perspective on how oceanographic conditions influenced the population and foraging trends of seabirds. Most of the declining species identified in our study are either poorly monitored or showed stable population trends at their breeding grounds (Table 1; Table E1), potentially reflecting the cost of monitoring populations at breeding grounds (Mott and Clarke, 2018). The declines we detected are cause for concern, given that local effects at one or a few breeding sites are unlikely to be the cause. Our results are restricted to populations of seabirds within our study area (south-eastern Australia); similar data for other regions are required to identify proportions of populations which visit coastal and shelf regions. Pelagic birding trips could be a substantial source of such data, given they occur on every continent except Antarctica. These data would benefit from also recording coordinates, sea surface temperature, or wind speed and direction of seabird sightings. Trips using chumming to attract seabirds should record how many individuals and of which species were attracted, so that the potential effects of attraction bias can be assessed. Citizen science seabird data may also prove to be important in identifying marine protected areas (Ainley et al., 2005; Cox et al., 2018; Goyert et al., 2016), and management of fisheries by contributing to ecosystem-based fisheries management approaches (see Chambers et al., 2011). By more accurately considering the impact of fisheries on seabirds in this region, the impacts of changing oceanic conditions in the Tasman could be reduced in the future, and the 'real-time' data collection of citizen scientists could be implemented in an adaptive management framework (e.g., Lindenmayer and Likens, 2009) for marine ecosystems.

5. Conclusions

Globally, marine ecosystems are increasingly threatened by environmental pressures, usually directly or indirectly human-related (Croxall et al., 2012; McCauley et al., 2015; Merino et al., 2012; Worm et al., 2006). For the vast majority of wide-ranging animals, we still lack an adequate understanding of species' spatial and foraging interactions. It is critical that this knowledge gap is rectified to ensure that biodiversity is appropriately managed and conserved. Our findings indicate a decrease in a large number of seabirds in a climate-sensitive, fast-changing ecosystem, while also highlighting a substantial lack of research occurring on many of these species. Time, and more data, will determine the extent to which regional declines in abundance are indicative of species-wide declines (e.g. Hyrenbach and Veit, 2003), but the long-term trends identified here highlight cause for conservation concern for at least some seabird species. Our use of mixture modelling to cluster species into groups with similar environmental responses further allows for prioritising of seabirds for monitoring programs. We highlight the utility of citizen science data to inform ecology and conservation of seabirds, with an example from south-eastern Australia. We stress that citizen science has a vital role for conservation and management in marine ecosystems, and a valuable next step will be to integrate other similar seabird datasets world-wide.

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.biocon.2019.05.007>.

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