Review of climate change mechanisms affecting seabirds within the INTERREG VA area.

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i.   Executive summary

1.   The seas between the west coast of Scotland, Northern Ireland and the Republic of Ireland (INTERREG VA region), support internationally important populations of 25 species of breeding seabirds. These seabird populations are considered vulnerable to climate change impacts. In order to inform effective conservation measures, it is important to identify and understand the main mechanisms by which these seabirds may be impacted by climate change in the region. As part of the Marine Protected Area Management and Monitoring (MarPAMM) project, this literature review synthesises current knowledge of the mechanisms underpinning potential impacts of climate change on seabird demography within the INTERREG VA region (study region). This review will inform the management plans being developed within the MarPAMM project (under work package T5).

2.   This structured literature review was performed through a standardised literature search using species name and key words (e.g. "ocean warming", "extreme weather"). Potentially relevant studies were identified from their titles and abstracts, selecting studies from the North Atlantic to best represent the study region. The number of studies providing support for each mechanism was used to identify the weight of evidence underpinning current available knowledge.

3.   For each species or taxonomically similar group of species, studies were synthesised into a single account detailing confirmed ecological and demographic impacts of climate change, along with the underlying mechanisms. For each account, impacts were categorized by the climate drivers identified and demographic parameters affected (breeding productivity, breeding population size, survival, and phenology). Potential future climate change scenarios were also summarised, with flow charts used to visualise mechanisms linking climatic variables to demographic impacts.

4.   A total of 156 studies were identified and included in the review, of which 64 considered multiple species. The number of studies available varied greatly between species, with the most (35) for common guillemot and the fewest (1) for roseate tern. In general the most studied species were those which provide good indicators of marine prey stocks, while those which breed in inaccessible locations or that are sensitive to disturbance were studied least.

5.   The majority of studies investigated indirect impacts of climate change on seabird demography (116 studies), with relatively few documenting direct impacts (24 studies). Long-term (>10 years) datasets (110 studies) were generally used to relate demographic parameters to broad-scale climatic variables, most commonly sea surface temperature (SST) and the North Atlantic Oscillation (NAO), through indirect mechanisms (breeding productivity: 58 studies, breeding phenology: 32 studies; breeding abundance: 24 studies; adult survival: 20 studies). In contrast, short-term (<10 years) datasets (38 studies), were generally used to quantify impacts associated

with individual events, such as extreme weather (anomalous temperatures and storms), operating directly on seabird demography (breeding failure; mass mortality).

6. Mechanisms with the strongest evidence to link long-term climate change to seabird populations were indirect, arising from temperature-mediated changes on prey populations. There was also support, though considerably less, for mechanisms associated with single or short-term events, often leading to complete breeding failures in some seasons or mass mortality events (wrecks), particularly during winter.

7. Foraging range was an important determinant of species/species group responses to climate change. For example, the demography of long-ranging species, such as Northern fulmars, was influenced by broad-scale climatic factors (e.g. NAO), while for species with a more restricted range, such as European shags, demography was generally determined by local conditions (e.g. SST). Foraging behaviour was also a key driver of species responses. The demography of surface seizing species, such as kittiwakes, was generally linked to variations in prey abundance, while diving species, such as common guillemots, were more commonly affected by direct impacts of adverse weather on foraging ability and prey availability/accessibility.

8. Differences in climate change responses between colonies of the same species likely represented variation in diet, which varied substantially throughout studies considered (N Atlantic), or differences in exposure to extreme weather events, with low lying coastal nesting species being particularly vulnerable.

9. A key limitation of this review is that much of the literature available is focussed on seabird populations that breed on the east coast of Scotland, within or adjacent to the North Sea. This region differs substantially from the study region in its oceanographic and physical features. Further, several colonies dominate the literature within this study, with 54 studies considered in this review originating from the Isle of May, Firth of Forth. In contrast, only 13 studies were based within or included the entire study region. As such, caution must be taken when generalising to the study region from mechanisms found at other colonies or regions.

10. In summary, this review identifies substantial evidence for climate change impacts on seabird populations with relevance to the study region. The strongest support was for long-term, indirect, bottom-up mechanisms, through changes in prey populations. This highlights the importance of an ecosystem-based approach to the marine environment as impacts cascade through trophic levels. Short-term, direct mechanisms were also important, by way of extreme weather events, often resulting in catastrophic demographic consequences. With the frequency and severity of such events predicted to increase, long-term changes in population abundance or breeding distribution may occur, with important consequences for seabird conservation.

11. Climate change adaptation options for seabird populations in the region may focus around management of prey stocks and breeding site protection/creation to increase resilience to storm events. However, although pointing to numerous useful areas where management may be targeted, more work is required to support this assessment.

1. Introduction

Climate change is impacting natural systems across the globe, leading to detectable changes in the abundance and distribution of many species, with impacts on ecological communities (Scheffers et al., 2016). These impacts are projected to become significantly greater through time (e.g. Warren et al., 2018), so it is important to identify the species that are most vulnerable to climate change. In order to inform potential conservation responses, it is important to identify the main mechanisms by which climate change may affect species (Pearce-Higgins and Green, 2014), which in many instances may occur through indirect impacts on species-interactions rather than direct effects (Ockendon et al., 2014).

Altered conditions within the marine environment, attributed to climate change, have been linked to temperature trends and frequency of extreme weather events (Scaife et al., 2008). Long-distance migrants, which many seabirds are, may potentially be affected by these adverse conditions across their global range (Pearce-Higgins & Green 2014). Seabird abundance, breeding success, survival rates and phenology at colonies have been shown to relate to a suite of climate variables. However, less is known of how demographic processes such as dispersal and emigration, age structure, and sex ratios respond to climate change and what impact this would have on population dynamics (Mitchell et al. 2020).

In the British Isles, seabird populations are regarded as particularly sensitive to climate change, with many projected to suffer long-term population declines during the course of this century (Johnston et al., 2013). Seabirds have already been identified as responding to climate change, with relationships established through a number of long-term studies (Harris et al., 2006; Lauria et al., 2012). For example, declines in population abundance and breeding success, have been observed potentially in response to climate change and adverse weather (Dias et al., 2019; Mitchell et al., 2020).

While studies have shown correlations between seabird demographic and phenological measurements and climatic variables such as Sea Surface Temperature (SST) and the North Atlantic Oscillation (NAO) Index (An index summarizing the air pressure differential between Iceland and the Azores) (Frederiksen et al., 2004a; Burthe et al., 2012), fewer have explicitly identified the direct or indirect mechanistic links between climate variables and possible changes in seabird populations.

1.1 Direct and indirect impacts

Direct effects of climate change result from physical interactions between aspects of the weather and birds. These include high storm winds blowing incubating cliff-nesting adults off eggs, storm swell washing away nests (Newell et al., 2015) and high levels of rainfall flooding nesting burrows (Harris and Wanless, 2011). With rising sea levels there is also

increased incidence of flooding of low-lying colonies (Norris and Buisson, 1994). Seabirds may also be exposed to enhanced physiological stress through overheating (Gaston et al., 2002; Oswald et al., 2008) or hypothermia (Frederiksen et al., 2008). Impacts of climate change may manifest as changes in seabird behaviour. High winds and precipitation may reduce foraging efficiency, through impaired foraging and flight ability (Birkhead, 1976; Taylor, 1983; Finney et al., 1999; Lewis et al., 2015). High onshore winds may also increase incidences of 'groundings' when combined with certain phases of the lunar cycle and the presence of artificial light (Syposz et al., 2018).

Indirect effects can arise through bottom-up impacts on the abundance, quality, or distribution of prey, which in turn are influenced by changes in primary productivity and oceanographic features (Frederiksen et al., 2006). Abundance of prey may be altered by the strength and timing of the spring phytoplankton bloom, which is closely tied to water-column stratification and SST (Scott et al., 2006). Spatial distribution of phytoplankton may shift in response to SSTs, additionally altering planktonic composition (Bedford et al., 2020). Changes in temporal abundance of prey may also be related to variation in the timing of the prey species migration (Røttingen, 1990). As seabirds tend to adjust the timing of chick rearing to match the availability and growth rates of prey (Burthe et al., 2012), there is an increasing risk of temporal mismatch if they fail to respond quickly enough to such changes in their food supply resulting in reduced breeding success. However, on a global scale seabirds have, on average, not shifted breeding phenology (Keogan et al., 2018) and there is a current lack evidence of temporal mismatches occurring (Samplonius et al., 2021.). Impacts of prey abundance during the winter are less well understood but are likely to have an impact on of the body condition of seabirds with possible carry over effects into the subsequent breeding season (Salton et al., 2015). Other indirect effects on seabirds may occur through increased top-down pressures, such as higher predation rates at breeding colonies as a result of climate related changes in prey availability (Church et al., 2019). Mammalian predators may also undergo temperature-mediated range shifts due to changes in habitat suitability, bringing them into contact with seabird colonies (Hof et al., 2012; Buchadas and Hof, 2017a). There is also evidence that top down and bottom-up pressures may arise simultaneously which could further compound negative impacts . Changes in oceanographic and other physical processes can cause a reduction on fish prey availability and some seabirds will switch to predation of other seabird species at or nearby their colonies (Votier et al., 2004; Perkins et al., 2018). There is also evidence for increased risk of disease transmission or higher levels of parasitism with higher air temperatures (Gaston et al., 2002; Cooper et al., 2009), which are predicted as part of modelled climate change scenarios (NOAA, 2020).

Crucially, multiple direct and indirect effects may act on a species simultaneously (Oro, 2014). An additional difficulty exists in differentiating the impacts of climate from other anthropogenic effects such as oil spills (Votier et al., 2005), plastic ingestion (O'Hanlon et al., 2017), and fisheries, through prey depletion (Furness and Tasker, 2000; Frederiksen et al.,

2004b), changes in discard rates (Church et al., 2019), or by-catch (Žydelis et al., 2009). Moreover, each of these impacts are regarded as major threats to these species in their own right.

1.2 Vulnerability to climate change

The extent to which seabirds are impacted by changes in their environment is largely determined by their physiology, foraging ecology, and behaviour (Mitchell et al., 2020). A number of different approaches exist to assess this vulnerability (Foden et al., 2019). Surface feeding species may be more vulnerable to changes in food availability or distribution in the surface waters, due to their inability to exploit greater depths as used by the deep diving species (Furness and Tasker, 2000). Similarly, seabirds with more restricted diets will show less flexibility to adjust to changes in their food supply (Howells et al., 2018). High winds and wave heights may also impair pursuit plunging species ability to forage which may lead to starvation (Birkhead, 1976; Taylor, 1983). Similarly, the physiological traits of species (e.g. relative body weight or water repellence of the plumage), or breeding behaviour (e.g. sheltered versus exposed nests) may dictate their ability to cope with high winds, storm swell, and heavy rainfall which can flood nests, wash away eggs, or blow incubating adults off eggs or nestlings (Newell et al., 2015). The extent to which birds are resident or migratory will influence the scale of the range over which they are affected by climatic impacts. Local conditions will influence non-migratory species all year round (Frederiksen et al., 2004a; Greenwood, 2007), while migrants are affected by the environmental conditions throughout their range varying with their annual cycle (Tavares et al., 2019).

1.3 Detection of impacts

The detection of climate-mediated changes on seabird demography and phenology generally require established long-term monitoring of seabird breeding success, recruitment, phenology, survival, or abundance (Oro, 2014). While for many marine species most of their annual cycle is spent entirely at sea, seabirds must come to land to breed. During this period, the conspicuous nesting habits of cliff and ground nesters make them relatively accessible species from which to collect data on key demographic rates, diet or behaviour. Consequently, seabirds are often used as sentinels for the state of the marine environment (Furness and Camphuysen, 1997). Longevity and the high adult survival rate characteristic of seabirds - which typically take several years to reach maturity (Horswill and Robinson, 2015) - allow populations to be buffered from short-term fluctuations in deleterious environmental conditions, and may also delay a measurable impact on populations (Heath et al., 2009). Therefore, changes in population abundance due to anthropogenic impacts such climate change, unless catastrophic, are not easily identified.

Additionally, phenotypic plasticity within populations may alter demographic responses to environmental change (Gremillet and Charmantier, 2010). Breeding success is generally regarded as being a useful demographic rate to detect changes in the marine environment such as a reduction in key prey species (Furness and Tasker, 2000). Adverse weather may also have direct consequences for breeding productivity through reduced provisioning (Finney et al., 1999) or increased need for chick brooding. Breeding success provides an indicator of the environmental conditions experienced not only during the summer, but also in spring, as adult body condition prior to the breeding season may have an influence on success (Frederiksen et al., 2006). Measurements of adult and immature survival can also be related to long-term trends in climatic change (Oro, 2014) , or short-term over-winter survival (Harris et al., 2007). However, when considering immature survival rates, variations in natal philopatry must be considered on a species (Coulson, 2016), and spatial (Johnston et al., 2019) levels. Survival rates may be directly affected by starving or direct mortality due to storms (Harris and Wanless, 1996; Morley et al., 2016; Tavares et al., 2019). However, adults may trade an increased likelihood of survival with reproductive success, making bottom-up effects of prey or harsh environmental conditions harder to detect using adult survival rate (Lewison et al., 2012).

Phenology, particularly the timing of breeding measured through laying or hatch dates, is driven by the endocrine system (Bradshaw and Holzapfel, 2010; Visser et al., 2010) and breeding may be initiated by environmental conditions or prey abundance during the pre-breeding period (Frederiksen et al., 2004a; Greenwood, 2007). NAO is often used as proxy for winter conditions which migratory seabirds may use as a cue for breeding, with the NAO index being assumed to relate to foraging conditions pre-breeding (Moe et al., 2009). Studying phenology in relation to the timing of peak prey abundance may identify the extent to which trophic mismatching may occur (Burthe et al., 2012). It is yet to be determined how birds may adapt to these shifts. At two colonies in the North Sea, the timing of breeding was shown to have shifted, the direction of which was dependent on species concerned (Wanless et al., 2009). However, on a global scale, Keogan et al. (2018) illustrated through meta-analysis of seabird phenology that laying and hatching has not shifted over time on average. While laying date has not shifted significantly, arrival date has been shown to have advanced in common guillemots *Uria aalge* and Brünnich's guillemots *Uria lomvia* in the North-West Atlantic (Merkel et al., 2019), potentially related to environmental conditions.

Time lags may be present between the changes in the marine environment and observed impacts on seabird populations (Thompson and Ollason, 2001). Black-legged kittiwakes *Rissa tridactyla*, for example, have shown that their breeding success is correlated with the winter SST with a 1 year time lag (Frederiksen et al., 2004b). This lag linking mechanisms is likely mediated through prey, and varies between seabird species dependent on diet composition. Time lags may also be present due to the influence of climate on breeding

success, when changes in population numbers of breeding adults through recruitment are not reflected until several years later (Thompson and Ollason, 2001; Oro, 2014).

1.4 The INTERREG VA region

This review addresses the species and associated with the INTERREG VA region (Figure 1). Within the United Kingdom (UK) the region encompasses the west of Scotland, including the Outer and Inner Hebrides; and the entire coast of Northern Ireland. Within the Republic of Ireland (Ireland hereafter), the north-west and north-east counties are encompassed. The area therefore incorporates both the North Atlantic and the Irish Sea, areas which experience temporally varying oceanographic conditions, and spatially varying hydrographic characteristics (Ellett and Edwards, 1983; McClatchey, 2014; Howarth, 2016). In particular the north-west of Scotland, Ireland, and Northern Ireland are more exposed and experience higher wave heights than coasts within the Irish Sea (Baxter et al., 2011; Masselink et al., 2016).

Much of the literature cited in this report is derived from seabird populations that breed on the east coast of Scotland, within or adjacent to the North Sea. This region differs substantially in its oceanographic and physical features when compared to the region covered by the INTERREG VA region (hereafter "study region"). The west of Scotland, in general, experiences higher winter SSTs than the North Sea, however this contrast reverses in the summer (Baxter et al., 2011). The North Sea has also experienced a stronger trend in increasing SSTs than the North Atlantic regions of the UK and Ireland (Bedford et al., 2020). Throughout the study region there has been a shift in copepod taxonomic composition between 1958 and 2017, with potential bottom-up consequences on seabird prey species (Bedford et al., 2020). While diet of seabird populations within the study region is relatively understudied, the primary prey resources for seabirds include lesser sandeels *Ammodytes marinus*, gadoid spp., clupeoid spp., and Sprats *Sprattus sprattus* (Swann et al., 1991, 2008; Halley et al., 1995; Anderson et al., 2014a). There is some evidence of alterations in prey abundance with the appearance of snake pipefish *Entelurus aequoreus* in the diet of seabird species for a few years (Harris et al., 2007). Pipefish are of poor nutritional quality (Harris et al., 2007) and consumed as an alternative prey (Swann et al., 2008; Anderson et al., 2014a).

1.5 Aims

Several literature reviews have already outlined the current knowledge on the effect of climate-mediated change on seabirds (Heath et al., 2009; Sydeman et al., 2012; Oro, 2014), and several reviews focus on the waters of the UK and Ireland (Daunt and Mitchell, 2013; Daunt et al., 2017; Mitchell et al., 2020). With particular focus on studies carried out in the North Atlantic, this review aims to provide a synthesis of the specific mechanisms by which climate change influences seabird populations and how they impact the different seabirds in

the INTERREG VA region (Figure 1 and Table 1). In doing so, we also highlight the weight of evidence underpinning our current knowledge of these mechanisms. We do not seek to present the extensive information which already exists on the conservation status of seabirds breeding in the UK (Eaton et al., 2015; Booth Jones, 2020a; JNCC, 2020a) and Ireland (Colhoun and Cummins, 2013) and background on whether a species is globally threatened or not (https://www.iucnredlist.org/). For each species covered we will additionally present the published predicted population sizes and distributional range, as well as the vulnerability assessments, produced from climate change scenarios made for the mid-to-late 21st century.
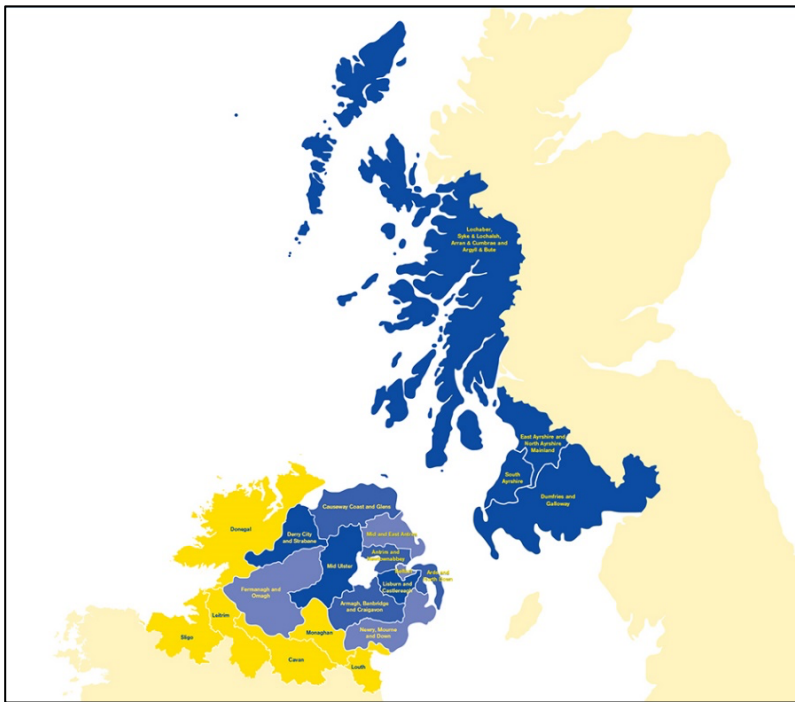
Figure 1. Map of the INTERREG VA region (as shown by the labelled areas). Image take from: The EU Special Programmes Body https://www.seupb.eu/iva-overview

2. Methods

2.1 Literature search

A structured literature review was performed, through a standardized internet search, using Google Scholar, by species (indicated by common name and Latin name/s) and using the key words: "climate change", "sea-surface temperature", "precipitation", "adverse weather", "extreme weather", "NAO", "stratification", "global warming", and "ocean warming"). Multiple common or Latin names which are applicable to single species were additionally included within the search terms. Potentially relevant studies were identified from their

titles and abstracts, explicitly selecting studies from the North Atlantic to best represent the INTERREG VA study area. The literature search resulted in a total of 206 studies (Table 1).

Relevant studies were collated into a database which delineated studies by author, date, study sites, and study length. From each study the following information was derived: study location, duration, the climate variable (e.g. SST, NAO, wind speed), other pressures (e.g. oil spills, fisheries), mechanism (e.g. plankton phenology, abundance, composition), mechanism impact on spp. (e.g. prey abundance), measurement method (e.g. phenology, breeding productivity). Also recorded was whether a covariate displayed a statistically significant effect or, where model weighting was used, a covariate was contained within the most parsimonious model. Additionally, the direction of that effect of a covariate on demography was recorded.

Table 1. Species list and number of identified studies relevant to the INTERREG VA study area. 156 publications were collated in total, 64 of which considered multiple species.

Common name	Latin name	No. Studies
Black-legged kittiwake	*Rissa tridactyla*	25
Black-headed gull	*Chroicocephalus ridibundus*	8
Mediterranean gull	*Ichthyaetus melanocephalus*	3
Common gull	*Larus canus*	6
Great black-backed gull	*Larus marinus*	3
Herring gull	*Larus argentatus*	9
Lesser black-backed gull	*Larus fuscus*	6
Sandwich tern	*Thalasseus sandvicensis*	7
Little tern	*Sternula albifrons*	7
Roseate tern	*Sterna dougallii*	1
Common tern	*Sterna hirundo*	10
Arctic tern	*Sterna paradisaea*	9
Great skua	*Stercorarius parasiticus*	5
Arctic skua	*Stercorarius skua*	3
Common guillemot	*Uria aalge*	35
Razorbill	*Alca torda*	9
Black guillemot	*Cepphus grylle*	2
Atlantic puffin	*Fratercula arctica*	15
European storm petrel	*Hydrobates pelagicus*	5
Leach's petrel	*Oceanodroma leucorhoa*	3
Northern fulmar	*Fulmarus glacialis*	8
Manx shearwater	*Puffinus puffinus*	4
Northern Gannet	*Morus bassanus*	8

European shag	*Phalacrocorax artistotelis*	23
Great cormorant	*Phalacrocorax carbo*	6

Mechanisms were were linked to the underlying climatic characteristic responsible and the ecological and demographic impacts on species. Following previous studies of this type, it is apparent that the wide-range of potential biological responses make it impossible to sufficiently standardise responses to look at the magnitude of a biological response (Ockendon et al., 2014; Franks et al., 2018). Therefore, we used the number of studies providing support for each mechanism and species, to identify weight of evidence underpinning current available knowledge. This literature was summarised into a short account for each species or grouped by poorly studied but taxonomically similar species where life-history traits such as foraging and nesting behaviour were deemed to be comparable. These species and species groupings accounts form the bulk of this review.

2.2 Flow charts

For each species, or species grouping, flow charts were created to visualise the mechanisms linking a climatic variable to a resulting demographic impact, as identified in the literature. Additionally, a flow chart encompassing the mechanisms, across all the reviewed species, was created (Figure 14). Lines depict the direction of 'flow' of a mechanism and are sized by the number of studies which cite a particular parameter. Links between climate variables displayed without an arrow indicate where studies reported a combined effect. To investigate the direction of effect by mechanisms on aspects on demography, the number of studies which found a negative or positive significant effect, or an insignificant effect are indicated above each demographic parameter. The direction of these effects was collated for each demographic impact and summed according to whether they were significantly negative, significantly positive or non-significant. In order to provide a significance score. This score was calculated by adding up the number observed effects (negative values per each negative effect, positive values for a positive effect, 0 values for a non-significant effect) and dividing by the total number of effects.

2.3 Future projections

Several studies have made future projections of seabird population size, distribution, or productivity in the UK and Ireland or Europe in the late 21st century under climate change, based on climate envelope models (e.g. Huntley et al. 2007; Johnston et al. 2013). Whilst these tend to be based on the same data sets (e.g. the periodic censuses in the UK), they differ slightly in their projections due to variation in the emissions scenarios, geographical areas, future time-steps and bird data used, and whether they assumed unrestricted or zero dispersal. In addition, not all studies modelled the same set of species. These future

projections for each species have been summarised in the following species and species grouping accounts. On the basis of these and other projections, other studies have assessed the vulnerability of species to climate change (e.g. Pearce-Higgins et al. 2017; Wheatley et al. 2017). These climate change vulnerability assessments differ in their assessments because they use slightly different projections and criteria for assessing risk. Furthermore, some studies incorporate additional ecological information (such as dispersal ability and habitat availability) into their assessments, while studies using more generic assessments typically solely use the projections from climate envelope models.

3. Species Accounts

3.1 Black-legged kittiwakes *Rissa tridactyla*

3.1.1 Breeding population

Black-legged kittiwakes (hereafter referred to as kittiwakes) maintain a circumpolar distribution within the Northern Hemisphere spanning both the Atlantic and Pacific Oceans. Within the North Atlantic they have a maximum estimated population of 3,000,000 breeding pairs (Mitchell et al., 2004a). Kittiwakes have undergone an overall population decline in the UK and Ireland from 505,465 breeding pairs in 1985-1988, to 379,895 in 1998-2002 (Mitchell et al., 2004a). Breeding abundance has continued to decline within the UK and Ireland (JNCC, 2020b) with population change of -50% between 2000 - 2018 (JNCC, 2020a). This decline is potentially linked to commercial fisheries (Frederiksen et al., 2004b; Daunt et al., 2008; Wanless et al., 2018), climate related bottom-up trophic processes, and adverse weather conditions affecting kittiwake breeding productivity (Wanless et al., 2007), phenology (Burthe et al., 2012), and adult survival (Oro and Furness, 2002).

The close association between the explicitly surface-feeding kittiwake, and the pelagic phase of lesser sandeels has been researched extensively within the UK, and the dynamics between these two species are recognised as a primary influence on kittiwake breeding at colonies in the North Sea(Frederiksen et al., 2004a, 2006, 2007a; Daunt et al., 2008; Wanless et al., 2009; Burthe et al., 2012; Lauria et al., 2013; Carroll et al., 2015; Olin et al., 2020). Kittiwake's dependence on sandeels is reflected in the distribution and size of their colonies around the UK, which have been shown to reflect local availability of sandeel aggregations (Lewis et al., 2001; Frederiksen et al., 2005). Catastrophic declines in the kittiwake population on Shetland (circa -70%) in the 1990s have been related to an absence in sandeel larvae due to oceanic current shift reducing larval transport from Orkney (Frederiksen et al., 2005). Moreover, decline in the number of breeding pairs of kittiwakes on Skomer, Wales, a colony less dependent on sandeels, was attributed to adverse weather conditions stemming from a positive NAO index, potentially mediated via changes in prey/productivity (Lauria et al., 2012). The high dependence of kittiwake populations on sandeels, in combination with their surface feeding behaviour, has made this species particularly vulnerable to changes in sandeel availability. Consequently kittiwakes are considered as sensitive bioindicators of the North Sea ecosystem (Furness and Tasker, 2000; Wanless et al., 2007).

3.1.2 Breeding productivity

Within the UK, the Seabird Monitoring Programme (SMP) showed that breeding productivity declined from 1986 to 2008 but stabilized from 2009 to 2018 (JNCC, 2020b). Within the North Sea, optimal breeding conditions in kittiwakes are related to an abundance of +1 group sandeels on which pre-breeding adults maintain their body condition before breeding, upon returning to the colony (Rindorf et al., 2000; Frederiksen et al., 2005;

Wanless et al., 2007). Following the transfer of 1+ group sandeels from a pelagic to a buried phase, 0-group sandeel abundance and size becomes integral to chick feeding (Frederiksen et al., 2006; Wanless et al., 2007). A strong 0-group sandeel year class is a predictor of increased kittiwake breeding success in the following year (Frederiksen et al., 2007b). Changes to sandeel quality and availability have marked effects on kittiwake breeding success (Rindorf et al., 2000; Frederiksen et al., 2013), and may be altered by bottom-up impacts of SST (Harris and Wanless, 1990; Frederiksen et al., 2005, 2007a), and winter NAO (Frederiksen et al., 2004a; Moe et al., 2009; Wanless et al., 2009). These climatic variables may alter foraging conditions of the planktivorous sandeel by altering the nutritional quality of the spring plankton bloom, and in particular the distribution and abundance of the copepod *Calanus finmarchicus* (Frederiksen et al., 2013; Carroll et al., 2015; Régnier et al., 2017). With rising SSTs, *C. finmarchicus* is being gradually replaced by the temperate water associated *C. helgolandicus,* which is of lower nutritional value (Frederiksen et al., 2007a).

The lack of alternative prey in the North Sea potentially compounds the effects of variable sandeel abundance (Frederiksen et al., 2007b). However, kittiwake breeding success across the UK has not shown a consistent significant trend related to SST (Carroll et al., 2015), indicating the presence of colony specific variation in prey ecology (Harris and Wanless, 1990; Frederiksen et al., 2007a; Olin et al., 2020). Where the effects of climate, such as winter SST, on breeding productivity or adult condition were weakest, birds were able to exploit other locally abundant prey species such as: gadoids, Canna, North Atlantic (Harris and Wanless, 1990; Swann et al., 1991, 2015); Clupeids, Irish Sea (Chivers et al., 2012a); and sprat, Celtic Sea (Lauria et al., 2013).

The consequence from direct impacts of adverse weather have been observed on the Isle of May, Scotland, where kittiwakes nesting on exposed cliffs were vulnerable to breeding failure – a 15% reduction in nests was caused by a storm in a one year (Newell et al., 2015). Indirectly, during years of poor prey availability kittiwake breeding success may be affected by top-down predation from great black-backed gulls substituting their diets on seabirds due to a lack of marine prey (Regehr and Montevecchi, 1997).

3.1.3 Phenology

The timing of the spring bloom and its influence on sandeel availability and quality has potential effects on kittiwake phenology (Burthe et al., 2012), and may be altered by the winter NAO (Frederiksen et al., 2004a; Moe et al., 2009; Wanless et al., 2009). A trend for later breeding has been recorded at colonies within the North Sea (Frederiksen et al., 2004a; Wanless et al., 2009) and the high-arctic (Moe et al., 2009). Later breeding is associated with lower prey availability and therefore lower breeding success (Moe et al., 2009). Negative NAO (colder, drier, less stormy, wind dominated winters) values correlated with later laying dates in the North Sea (Frederiksen et al., 2004a; Wanless et al., 2009), while high SSTs had a negative effect on Svalbard within the Arctic (Moe et al., 2009). These responses to climate are potentially bottom-up driven as temporal shifts in the habitat suitability of *C.*

finmarchicus, are negatively influenced by a high NAO index (Planque and Fromentin, 1996). Stronger and earlier spring stratification of the water column, which is projected to become more frequent with climate change, provides less suitable habitat for *C. finmarchicus* which prefer weaker, more oxygenated stratification during the spring bloom (Carroll et al., 2015). Temporal shifts in copepod abundance have a knock on influence on the growth of 0-group sandeels, for which the peak availability has become later (Rindorf et al., 2000). While kittiwakes have similarly shifted phenology to later in the season (Frederiksen et al., 2004a), this has been insufficient to align peak sandeel quality with the timing of chick rearing (Burthe et al., 2012).

3.1.4 Survival

Kittiwake survival rates are highly variable (Sandvik et al., 2005), and within the UK displayed fluctuating trends across long-term study sites of Skomer and the Isle of May (JNCC, 2020b). Adult survival in kittiwakes is potentially negatively related to higher winter SST (Frederiksen et al., 2004b), potentially due to prey availability. Higher winter SSTs have an increased metabolic cost for sandeels inhibiting winter growth and forcing prolonging periods of burial (Carroll et al., 2015; Wright et al., 2020). Additionally, years of positive NAO (associated with warm, wet winters, and westerly/variable winds) have been linked to poorer overwinter survival and condition in sandeels (Wanless et al., 2009). However, in Norway, kittiwake survival was influenced by NAO and SST to a lesser extent, potentially due to the availability of alternative prey stocks (Sandvik et al., 2005).

During the breeding season, top-down impacts of great skua predation have been shown to negatively affect adult survival, particularly during years of high skua productivity (Oro and Furness, 2002). Potentially indirectly related to climate, toxic algal blooms have also been identified as another source of mortality in kittiwakes (Coulson and Strowger, 1999).

3.1.5 Future projections

Kittiwakes distributional range is projected to decline in extent in European waters over the 21st century by 19% (Huntley et al., 2007) to 20-25% (Russell et al., 2015). The future distribution in Europe is projected to overlap only partly with the current distribution by 65-70% (Russell et al., 2015) to 77% (Huntley et al., 2007). Currently occupied colonies in the south of their range (e.g. France, Spain and Portugal) are expected to become climatically unsuitable (Huntley et al., 2007). In the UK, kittiwake is projected to be at very high risk from climate change ([*]Ausden et al., unpubl.) with a projected 55.5% decline in abundance within UK SPAs (Johnston et al., 2013). In the North Sea and Britain's continental shelf, kittiwake population size is predicted to change relatively little by the middle of the 21st century; by -3.2 to +0.8% (Sadykova et al., 2020). However, over the same time period, the

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[*] *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.

overlap of kittiwake's at-sea distribution with the distribution of its prey species (herring *Clupea harengus,* sandeel) is expected to decrease considerably (Sadykova et al., 2020). Additionally, in the UK and Ireland, kittiwake breeding success is projected to decline by 21-43% (Carroll et al., 2015). A range of more generic risk assessments for kittiwake in Britain project a medium to low risk for this species (Wheatley et al., 2017) and medium benefit of climate change (Pearce-Higgins et al., 2017).

3.1.6 INTERREG VA region summary

Kittiwakes are found throughout the study region (Figure 1.) (Mitchell et al., 2004a; Balmer et al., 2013). The species has displayed declines in abundance and productivity at colonies on the West of Scotland and Irish Sea between 1998-2018 (Frederiksen et al., 2007a; JNCC, 2020b). There is currently less data available on diet trends for the West of Scotland and Irish Sea compared to the North Sea (Frederiksen et al., 2007a). However, available knowledge suggests there is less reliance on sandeels by kittiwakes in this region (Frederiksen et al., 2005), and their diet contains greater prevalence of clupeids (Chivers et al., 2012a) and gadoids (Harris and Wanless, 1990). The effect of environmental change has been studied more extensively in the North Sea, which has experienced stronger rises in SST than the West of Scotland and Irish Sea (Bedford et al., 2020). While productivity is declining in the study region, this was not linked to SSTs (Frederiksen et al., 2007a). Due to the disparity in SST rise and diet, studies from the North Sea may not be suitably representative of colonies within the study region. Additionally, due to the greater exposure of the North-West coasts of Ireland, Northern Ireland, and Scotland to storm swell (Baxter et al., 2011; McClatchey, 2014; Masselink et al., 2016), breeding abundance with the region may be susceptible to increasing frequency of adverse conditions (Lauria et al., 2012).
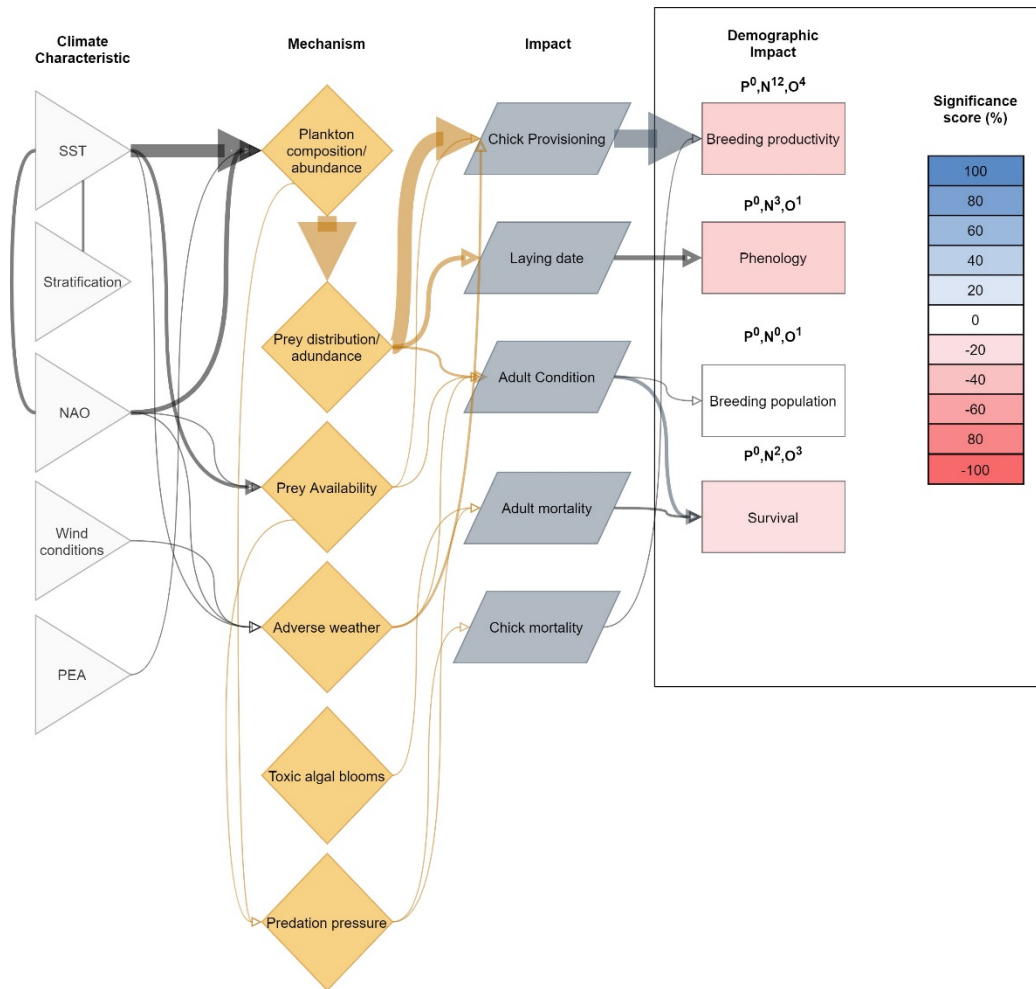
Figure 2. Kittiwake related mechanism flow chart. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea Surface Temperature; PEA = Potential Energy Anomaly; P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = number of studies where non-significant effects found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.2 Gulls

3.2.1 Breeding populations

Six species of gull (excluding kittiwake) breed within the study region: black-headed gull *Chroicocephalus ridibundus,* Mediterranean gull *Ichthyaetus melanocephalus*, common gull *Larus canus*, great black-backed gull *Larus marinus*, herring gull *Larus argentatus*, and lesser black-backed gull *Larus fuscus*. Within the North-East Atlantic, black-headed gulls, common gulls, lesser black-backed gulls, and herring gulls range from France and the Iberian Peninsula, and from Norway and Iceland (BirdLife International, 2020a, 2020b, 2020c, 2020d). While Mediterranean and black-headed gulls additionally breed throughout Europe, with a southern limits in Turkey, the UK and Denmark form the northern periphery of Mediterranean gull's breeding distribution (BirdLife International, 2020e, 2020d). The

breeding distribution of great black-backed gulls reaches furthest north, to Svalbard (BirdLife International, 2020f).

The species have been combined into a single account as they all exhibit similar life history traits of nesting, clutch size, migratory behaviour, foraging behaviour, and generalist diets which incorporate the terrestrial environment, allowing for a degree of comparison. Kittiwakes have been excluded as they exhibit entirely marine based diets, and research in relation to this species is primarily focussed on marine related environmental covariates. Herring gulls traditionally forage in intertidal and terrestrial habitats, whereas lesser and great black-backed gulls may forage offshore (Gotmark, 1984). Black-headed and common gulls forage within intertidal and terrestrial habitats (Kubetzki and Garthe, 2003). Gulls are regarded as having a low vulnerability to the influence of climate change, potentially as gulls have more generalist diets which may buffer them from indirect climate impacts, by allowing them to switch food sources (Burthe et al., 2014). This is visible in diets across several gull species which are now primarily dependent on anthropogenic sources such as discards, agriculture, and landfills (Foster et al., 2017; O'Hanlon and Nager, 2018; JNCC, 2020c). Gulls have also shown increased use of the urban environment for nesting and for foraging opportunities, notably lesser black-backed and herring gulls (Rock and Vaughan, 2013). Dietary differences between the species may be one of drivers underlying their variable responses to environmental change.

The global number of black-headed gulls was last estimated as 2,800,000 breeding pairs (Mitchell et al. (2004a). Within the UK and Ireland, black-headed gull populations showed no major changes between 1986-1988 to 1998-2002 with 79,392 pairs recorded at the last census (Mitchell et al., 2004a). More recent data from the SMP show an increase of 35% between 2000-2018 (JNCC, 2020a).

Global estimates of the Mediterranean gull are around 100,000 breeding pairs (Mitchell et al., 2004a). Between 1986-1988 and 1998-2002 the Mediterranean gull population has increased from 1 to 110 pairs within the UK and Ireland (Mitchell et al., 2004a). Recent SMP data display an increase of 2,059% between 2000-2018, with an additional 54 breeding pairs recorded in the Republic of Ireland 2015-2018 (JNCC, 2020a). Increase in the UK's population of black-headed and Mediterranean gulls may be related to climate due to improved habitat suitability (Burthe et al., 2014).

Globally the number of common gulls is thought to be 316,000 breeding pairs (Mitchell et al., 2004a) The UK population of common gulls increased by 36% from 1986-1988 to 1998-2002, to 21,475 pairs (Mitchell et al., 2004a; JNCC, 2020a). However, following the increase between 1986-2002, they underwent a decline in breeding abundance from 2002-2018 (JNCC, 2020d). Whilst they are susceptible to predation by American mink Neovison vison, the underlying cause of the decline is currently unknown (JNCC, 2020d). The decline in the North Sea population was found to be unrelated to climate, and the species is regarded as having low vulnerability to climatic influence (Burthe et al., 2014).

The world population estimate of lesser black-backed gulls is thought to be 650,000 pairs (Mitchell et al., 2004a). The coastal population of lesser black-backed gulls in the UK and Ireland increased by 40% from 1986-1988 to 1998-2002 to 91,323 pairs (JNCC, 2020e). However, since the Seabird 2000 survey they have undergone a decline in breeding abundance at their traditional sites within the UK, particularly in England and Wales, while populations have fluctuated in Scotland (JNCC, 2020e). In Ireland, lesser black-backed gulls have increased (Creme et al., 1997), this was primarily attributed to anthropogenic changes in land use increasing availability of suitable habitat. While increases in the use of urban sites for breeding have been recorded (Balmer et al., 2013), the size of the urban nesting population is yet to be estimated. Changes in populations are potentially related to changes in availability of anthropogenic food resources from fisheries discards or human waste (Maciusik et al., 2010; Burthe et al., 2014; Tyson et al., 2015; Foster et al., 2017). Population declines within the Norwegian sea are significantly correlated with winter air temperature and NAO, potentially related to subsequent declines in the abundance and quality of herring stocks. Mild winters were associated with high breeding numbers of gulls in the Norwegian study islands, potentially due to increase availability of prey resources such as gadoids (Bustnes et al., 2010). Rising SSTs were suggested to have a bottom-up influence on increasing populations in Belgian and French based colonies of lesser black-back gulls, as SST correlated with decapod larval abundance which in-turn influenced the abundance of swimming crabs *Polybius henslowii as* prey (Luczak et al., 2012; Schwemmer et al., 2013).

Global estimates of herring gulls are around 799,000 breeding pairs (Mitchell et al., 2004a). Within the UK and Ireland the breeding abundance of herring gulls decreased by 13% from 1986-1988 to 1998-2002 (Mitchell et al., 2004a) and these declines have continued over the period of 2000-2018 (JNCC, 2020c). This decline has been related to changes in the management of resources previously scavenged for food, including covering refuse tips (JNCC, 2020c), and a reduction in fisheries discards (Foster et al., 2017). Changes in land use have been proposed to be the underlying drivers of a decline in herring gulls in Ireland (Creme et al., 1997). For herring gulls, energy-rich resources such as landfills and fisheries discards have become restricted (Bicknell et al., 2013; Foster et al., 2017), leading to a switch to foraging at select farmland and intertidal sites (O'Hanlon and Nager, 2018; Pennycott et al., 2020).

Global estimates of great black-backed gulls are around 110,000 breeding pairs (Mitchell et al., 2004a). Within the UK and Ireland the breeding abundance of great black-backed gulls decreased by 4% respectively 1986-1988 to 1998-2002 (Mitchell et al., 2004a) and continued to decline from 2000-2018 (JNCC 2020f). For herring gull, declines in population size have been linked to increasing covering landfill (JNCC, 2020c), and a reduction in discards (Foster et al., 2017). In comparison to herring gulls, great black-backed gulls have not experienced a decline to the same extent, due to changes in the availability of fisheries discards, potentially due to their diet also containing a large proportion of seabirds (Buckley, 1990; Russell and Montevecchi, 1996; Regehr and Montevecchi, 1997; Westerberg et al.,

2019), and terrestrial mammals (JNCC, 2020f). On Cape Clear in the southwest of Ireland, declines in great black-backed gulls were related to potential food shortages or poisoning from landfill refuse, though no clear cause could be identified (Buckley and Kelly, 1994). Gull predation of other seabirds, such as Leach's storm petrel, can fluctuate with the availability of fish such as capelin, with increases predation corresponding with fewer capelin, with implications for other species populations (Stenhouse and Montevecchi, 1999).

It is apparent that gull breeding population size and distribution in the UK and Ireland have primarily been linked to the availability of human based resources rather than climate driven processes (JNCC, 2020c, 2020e). Within the Wadden Sea, which adjoins Germany, NAO correlated with increasing breeding populations of black-headed-, common-, herring-, and lesser black-backed gulls (Reid et al., 1999). This relationship was assumed to be correlative and not causative, as periods of positive NAO values also corresponded, potentially coincidentally, with recovering seabird colonies in the Wadden Sea (Reid et al., 1999).

3.2.2 Winter populations
Gulls are one of the few groups of seabirds within the UK and Ireland where wintering populations are regularly found onshore (e.g. at roost sites), which can be surveyed and therefore provide an assessment on size the non-breeding population of gulls in the winter.

While breeding abundance of gulls within the study region are generally related to changes in human activity rather than climate, a combination of these factors may be acting upon wintering populations. Numbers of wintering black-headed gulls declined in the 2000's following a peak in presence during the 1980s and early 1990s. This is potentially related to changes in landfill and agricultural practices, but also milder winters on the European continent which may have reduced migration to the UK (Banks et al., 2009). On a local scale, distribution also varies with temperature. In Poland, during periods of lower winter temperatures, black-headed gulls were more likely to be found at rubbish dumps (Jurinović and Kralj, 2013) and urban areas, which tend warmer and more likely to be snow free (Maciusik et al., 2010).

Within the UK, wintering herring gulls have decreased. These losses have been attributed to anthropogenic related food availability, increased culling, and disease (specifically *Clostridium botulinum* associated with landfills) (Mitchell et al., 2004b; Coulson, 2015) rather than climate change. However, there has also been site specific variation in the numbers of wintering herring gulls, with marked declines in the north of Scotland and an increase in southwest England potentially related to an increase in urban nesting behaviour (Banks et al., 2009). In the Canadian Northwest Atlantic, despite noted long-term increases in SST, herring gulls have displayed no change in their migratory patterns, which may have arisen due to shifts in habitat suitability (Robertson, 2016). Within Finland, NAO was found to have a significant effect on shifts in wintering grounds of both great black-backed gulls

and herring gulls, with winter ranges shifting significantly northwards while breeding ranges shifted southward (Potvin et al., 2016).

Within the UK, the numbers of wintering lesser black-backed gulls have increased, potentially related to mild winters, however there have also been site specific variation potentially related to the locations of anthropogenic food sources (Banks et al., 2009). In the north and west of Scotland, Banks et al., (2009) showed an increase in wintering lesser black-backed gulls between 1983 and 2004.

3.2.3 Breeding productivity

Within the UK and Ireland from 1989 to 2018, black-headed gull, herring gull, lesser black-backed gull, great black-backed gull productivity has been shown to fluctuate (JNCC, 2020g, 2020c, 2020e, 2020f), while common gull productivity has generally declined (JNCC, 2020d).

In Poland in 2012, breeding black-headed gulls experienced a dramatic drop in April air temperatures. This resulted in a catastrophic loss of eggs in early nesting birds, due to adults leaving their nest to feed which caused their eggs to overcool and crack (Indykiewicz, 2015). Alternatively, in warm wet years in Poland, outbreaks of black flies *Simuliidae* have negatively affected breeding success of common gulls, through harassment by flies causing direct injury, or indirectly, a decrease in nest attendance leading to predation (Bukaciński and Bukacińska, 2000). It is unclear the extent to which these observations from mainland Europe can be regarded as relevant in the context of the UK.

Productivity in gulls may primarily be influenced by the availability of discards and refuse tips for species which typically rely on anthropogenic waste for foraging. For example, the breeding success of herring gulls on Isle of Canna, Scotland, in the North-East Atlantic, has significantly declined following changes in local fisheries landings (Foster et al., 2017). Ground nesting gulls within Scotland may also be susceptible to American mink predation (Craik, 1997, 2000). In relation to direct, climate related impacts, low lying nests are susceptible to flooding and high tides which can inundate nests (Bonter et al., 2014). A rise in the frequency of storms may increase occurrences of flooding related to tidal swell (JNCC, 2020h).

Marine foraging great black-backed and herring gulls have been found to be susceptible to anomalous changes in ocean conditions. Breeding success in both of these species in Newfoundland was found to decline related to reduction in the availability of capelin at the sea surface due to the potential formation of a thermocline from anomalously cold temperatures (Regehr and Montevecchi, 1997; Rodway and Regehr, 1999). However, great black-backed gull breeding success was less effected as they were able to supplement their diet by predating other seabirds, including herring gulls (Regehr and Montevecchi, 1997; Rodway and Regehr, 1999). Physiologically, herring gull egg volume in the Bay of Fundy is potentially influenced by climate, correlating negatively with spring SST and winter NAO. This was linked to fluctuations in prey and adverse weather (Bennett et al., 2017).

3.2.4 Phenology

Few studies have looked at the influence of climate on the phenology of gulls. However, it is known that lesser black-backed gulls display both long- and short-distance migration strategies, which influence early and late breeding timings. This population-level behavioural plasticity may buffer the species from varied climate conditions (Shamoun-Baranes et al., 2017). In black-headed gulls the timing of arrival at wintering sites in Poland was significantly positively related to winter NAO while precipitation had no effect on the timing of migration (Jurinović and Kralj, 2013).

During the breeding season in Estonia, NAO had no effect on breeding phenology in common gulls, though laying date became earlier in warmer years (Brommer et al., 2008)**.** This study suggested that local SST influenced the availability of invertebrate prey of common gulls. Warmer springs were also related to increased variation in laying dates between years (Brommer et al., 2008)**.** Earlier breeding in common gulls related to warmer winters and springs have been linked to higher recruitment (more offspring which live to breed), and fitter offspring, which bred earlier (Rattiste and Tartes, 2005). Later breeding has been linked to lower breeding success in herring and great black-backed gulls in Newfoundland, where extreme cold spring SSTs significantly delay breeding possibly as a consequence of later arrival of accessible capelin due to a potential thermocline (Regehr and Montevecchi, 1997; Rodway and Regehr, 1999).

3.2.5 Survival

No information is available from the SMP on black-headed gull, common gull, Mediterranean gull, and great black-backed gull survival (JNCC, 2020f, 2020d, 2020g, 2020h). Data on the survival rate of herring gulls and lesser black-backed gulls from Skomer, Wales, display gradual declines in both species 1986-2018 (JNCC, 2020c, 2020e). No studies attributing survival to climate change were found.

3.2.6 Future projections

Black-headed gull is projected to decline in total range extent by 43% across Europe over the 21st century (Huntley et al. 2007). The future distribution in Europe is expected to overlap by 50% with their current distribution (Huntley et al. 2007). The southern range of the current European distribution has been projected to become climatically unsuitable (Huntley et al., 2007). By contrast, in the UK, black-headed gull is projected to experience very high benefit from climate change (Ausden et al., unpubl.), with a projected increase in abundance of 91.8% on UK SPAs (Johnston et al., 2013). A range of more generic climate change vulnerability risk assessments for black-headed gull within Britain span all risk levels, but include more support for medium or low risk for the species (Pearce-Higgins et al., 2017; Wheatley et al., 2017).

Future projected changes in Mediterranean gull total range extent in Europe over the 21st century vary considerably in direction: from a 28% decrease (Huntley et al., 2007) to a 95-

100% increase (Russell et al., 2015). The future distribution in Europe is projected to overlap little with the current distribution: by 7% (Huntley et al., 2007) to 16-18% (Russell et al., 2015). Future projected changes in Mediterranean gull range and abundance within the UK vary: ranging from complete absence from the UK (Huntley et al., 2007), to very high benefit (*Ausden et al., unpubl.). A range of more generic risk assessments for Mediterranean gull in Britain span the full range of potential risk levels (Wheatley et al., 2017) but also project high benefit of climate change (Pearce-Higgins et al., 2017).

Common gull is projected to decline in total range extent by 48% across Europe over the 21st century (Huntley et al. 2007). The future distribution in Europe is expected to overlap by 49% with the current distribution (Huntley et al. 2007). Most of the south of the current European distribution is projected to become climatically unsuitable (Huntley et al., 2007). In the UK, projections vary: Johnston et al. (2013) project a 96.1% decline in abundance on SPAs, while Ausden et al. (unpubl.) project only moderate risk from climate change. A range of more generic risk assessments for Britain tend towards medium to low risk (Wheatley et al., 2017) or a balance of potential risks and benefits (Pearce-Higgins et al., 2017).

Great black-backed gull is projected to decline by 33-37% in total range extent in Europe over the 21st century (Huntley et al., 2007; Russell et al., 2015). The future distribution in Europe is projected to overlap only partly with the current distribution: by 41% (Huntley et al., 2007) to 58-60% (Russell et al., 2015). Most of great black-backed gull's distribution around the Baltic and France is projected to become climatically unsuitable over the 21st century (Huntley et al., 2007). Future projected changes in great black-backed gull range and abundance within the UK vary: from a 5.8% decline in abundance on SPAs (Johnston et al., 2013), to projections of low risk with high benefit (*Ausden et al., unpubl.). A range of more generic risk assessments for great black-backed gull in Britain project medium to low risk (Wheatley et al., 2017) or high benefit (Pearce-Higgins et al., 2017).

Herring gull is projected to decrease in total range extent in Europe over the 21st century: by 28-30% (Russell et al., 2015) to 37% (Huntley et al., 2007). The future distribution in Europe is projected to overlap only partly with the current distribution: by 45% (Huntley et al., 2007) to 66-69% (Russell et al., 2015). Most of the south of herring gull's European distribution is projected to become climatically unsuitable (Huntley et al., 2007). By contrast, in the UK, herring gull abundance is projected to increase by 290.5% on SPAs (Johnston et al., 2013), and here the species is projected to be at low risk with high benefit (*Ausden et al., unpubl.). A range of more generic risk assessments for herring gull in Britain project medium to low risk (Wheatley et al., 2017) and high benefit (Pearce-Higgins et al., 2017).

Ausden et al., unpubl. is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.

Lesser black-backed gull is projected to decline in total range extent in Europe over the 21st century: by 22-23% (Russell et al., 2015) to 37% (Huntley et al., 2007). The future distribution in Europe is projected to overlap with the current distribution by 39% (Huntley et al., 2007) to 70-71% (Russell et al., 2015). More than 60% of lesser black-backed gull's current European distribution is projected to become climatically unsuitable (Huntley et al., 2007). Future projected changes in lesser black-backed gull range and abundance within the UK vary: from an 18% decline in abundance on SPAs (Johnston et al., 2013), to projections of low risk with very high benefit ([*]Ausden et al., unpubl.). A range of more generic risk assessments for lesser black-backed gull in Britain project medium to low risk (Wheatley et al., 2017) and high benefit (Pearce-Higgins et al., 2017).

3.2.7 INTERREG VA region summary

Within the study region breeding abundance and productivity has varied both between gull species, and spatially within species from 2000-2018 (JNCC, 2020c, 2020f, 2020g, 2020d, 2020e). While herring gull, lesser black-backed gull, great black-backed gull populations have primarily declined in Scotland, in Ireland some increases have been seen, however this is potentially related to improved survey effort. Though maintaining a comparatively small population, the number of Northern Ireland based Mediterranean gulls have increased in the past decades (JNCC, 2020h). Declines in some species, have coincided with distributional trends away from coastal areas, and increasing prevalence at inland and urban sites (JNCC, 2020c, 2020g, 2020e). Population and distributional changes within the study region have not been linked to climate variation, but anthropogenic changes in the management of fisheries discards (Foster et al., 2017), refuse sites (Coulson, 2015), and terrestrial predators (Craik, 1997, 2000). The generalist diet of gulls may buffer several species from the effects of a changing climate (Burthe et al., 2014). Though some aspects of agricultural (Pennycott et al., 2020) and predatory foraging (Regehr and Montevecchi, 1997; Rodway and Regehr, 1999), may be influenced by climate conditions. Given their adaptability and generalist nature, future projected responses to climate change are uncertain for many gull species. Most studies suggest largely stable or increasing trends are likely, although across studies common gull may be most negatively affected by future climate change impacts.
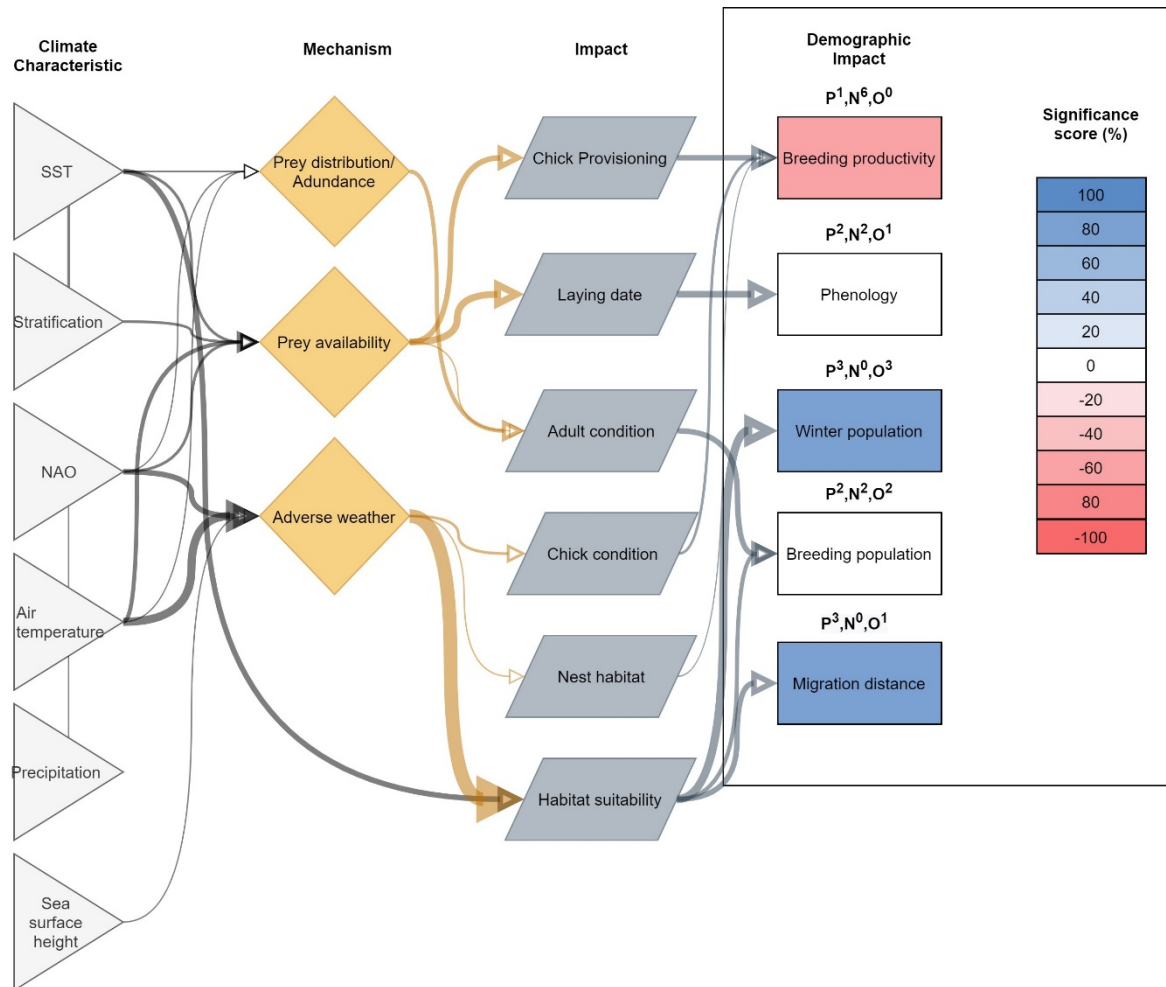
Figure 3. Grouped gull species related mechanism flow chart. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.3 Terns

3.3.1 Breeding populations

All five of the species of tern breeding in the UK and Ireland – Arctic *Sterna paradisaea*, common *S. hirundo*, little *Sternula albifrons*, roseate *S. dougallii* and Sandwich tern *S. sandvicensis* – breed in the study region. In the north-east Atlantic region, the five tern species vary considerably in their geographical ranges. Arctic tern is the most northerly distributed tern, with its southern range edge in Ireland, England and the Netherlands; this is mostly a coastal species but increasingly breeds inland at higher latitudes. Common tern breeds widely across the region from France north, in both coastal and inland areas, but does not breed in Iceland. Little tern has a highly patchy but wide distribution in the region,

from Finland to Portugal. The Scottish populations of roseate tern are the northernmost in the world; within the region, this species only breeds in the UK, Ireland and France. Sandwich tern, like little tern, has a highly patchy but wide distribution in the region.

At the turn of the 21st century, the global population sizes were estimated at 800,000-3,000,000 pairs for Arctic terns, 420,000-620,000 pairs for common terns, 44,000-130,000 pairs for little terns, 120,000-130,000 pairs for roseate terns and 160,000-170,000 pairs for Sandwich terns (Mitchell et al., 2004b).

In north-west Europe, tern populations were nearly extirpated in the late 19th century due to egg collection and hunting of adults (Mitchell et al., 2004b). Legislation and management since the early 20th century has allowed the populations to recover. In the North-east Atlantic, surface-feeding seabirds (such as terns) are generally of poorer current conservation status than those species that feed deeper in the water column (Mitchell et al 2020).

National populations of all UK and Irish tern species declined between the most recent two censuses (Mitchell et al., 2004b). It is difficult to make accurate population size estimates for terns over large areas, because populations fluctuate dramatically from year to year and move between colonies. In the UK and Ireland, terns were inevitably counted over multiple years during each of the national censuses, and so population size estimates and derived trends should be treated with some caution (Mitchell et al., 2004b).

Arctic tern is the most abundant tern species in the UK and Ireland, with most of the population found in Scotland. Arctic tern declined from 78,764 Apparently Occupied Nests (AON) in 1984-1988 to 56,123 AON in 1985-2002 (Mitchell et al., 2004b), with a further decline of 13% between 2000-2018 (JNCC, 2020a). Common tern is the next most abundant tern species in the UK and Ireland. The species is evenly spread around the region, with 8% and 19% breeding inland in the UK and Ireland respectively.

Common tern declined very slightly between 1984-1988 and 1995-2002, from 14,861 AON to 14,497 AON (Mitchell et al., 2004b), however numbers have since increased 15% between 2000-2018, as indicated by SMP data (JNCC, 2020a).

Little tern is thinly distributed around the UK and Ireland, with breeding concentrated around south and east England. From 1984-1988 to 1995-2002, the little tern population decreased from 2,857 AON to 2,153 AON (Mitchell et al., 2004b). SMP data estimates little terns to have decreased a further 25% between 2000-2018 (JNCC, 2020a).

Sandwich tern is the third-most abundant tern species in the UK, with most of the population located in England. In the UK and Ireland, Sandwich tern declined from 16,047

AON in 1984-1988 to 14,252 AON in 1995-2002 (Mitchell et al., 2004b), however have since increased 12% between 2000-2018 (JNCC, 2020a).

In the UK and Ireland, roseate tern is the rarest breeding tern and the second-rarest breeding seabird. Most of the population breeds in just three colonies. The population size increased from 550 AON in 1986 to 790 AON in 1999-2000 (Mitchell et al., 2004b), increasing a further 106% between 2000-2018 (JNCC, 2020a). This increase was a partial recovery after a major decline from 2,384 AON in 1969-1970.

Although there is widespread evidence of sometimes high magnitude effects of climatic variation on tern demographic parameters, in particular breeding success and phenology, it is less clear that these effects are passed on to changes in population size. However, most studies used NAO to test for relationships between climate and breeding numbers. NAO does not necessarily capture all meaningful climatic variation for terns for the entirety of Europe. Relationships between local climatic variation and population size remain relatively unstudied. In the southern North Sea, between 1909 and 1996, Sandwich tern breeding numbers in Germany were weakly positively related to December-March NAO although there was no similar relationship for Arctic and common tern breeding numbers (Reid et al., 1999). More specific effects of weather parameters were not investigated, but it was speculated that the mechanism of the effect could be through food supply. There was a step-change in the number of Arctic, common and roseate terns ringed (mostly pulli) in Ireland within 0-1 years related to March mean air temperature in coastal east Ireland (Donnelly et al., 2009) indicating a change to breeding success or population size. The number of little tern breeding pairs at colonies in south Portugal were unrelated to April-May NAO (Lopes et al., 2015), but were related to local vegetation cover, which in turn is likely to be affected by terrestrial climatic conditions.

Little terns nest just above the high tide mark on beaches, and are expected to be particularly threatened by sea level rise although the magnitude of this impact will depend on future trends in coastal erosion and storm activity (Gill, 2004). Although coastal defence strategies will further mediate the impact of sea level rise, the building of defence infrastructures will alter sediment transport patterns and change where beaches occur. Scarton (2010) speculated that different apparent responses by tern species to flooding at Venice lagoon from 1995-2008 could arise from their different breeding site use; for example that Sandwich tern may be more resilient to flooding than common tern, but did not formally test such relationships. Nesting habitats used by little and common tern breeding area in Slovenia is predicted to decline dramatically due to sea level rise over the 21st century (Ivajnšič et al., 2017). At this location, little tern presence was best predicted by distance to anthropogenic disturbance and vegetation density, while common tern presence was best predicted by elevation and overall habitat type. Due to their different habitat requirements, the rate of habitat loss in response to sea-level rise and storm surges may

therefore vary between tern species. As some colonies can cover extensive areas of low-lying substrate, relatively large amounts of land can be lost rapidly with only minor sea level rise. Given the mobility of tern populations, the magnitude of the future impact on populations is therefore likely to depend upon a combination of wider substrate availability in response to changes in sea-level rise and storm surges, and how that interacts with the human protection of coastal habitat or the pre-emptive creation of new breeding habitat.

3.3.2 Breeding productivity

Tern productivity in the UK and Ireland is highly variable, but generally stable: most regions show no trend in recent decades (JNCC, 2020i, 2020j, 2020k, 2020l, 2020m). The exception is common tern in Ireland: here, productivity declined gradually from 1990 to 2014 (JNCC, 2020j). Productivity varies considerably between species. Arctic tern productivity is the lowest of any seabird in the UK and Ireland (JNCC, 2020i), while roseate tern productivity in the UK is relatively high due to conservation effort (JNCC, 2020l).

There have been a few studies which have linked climatic variables to parameters relating to aspects of productivity. On the Atlantic coast of Portugal, local February SST was negatively related to the proportion of sand smelt (*Atherina* spp.) in little tern breeding season diet; this proportion in turn was positively correlated to little tern clutch size (Ramos et al., 2013; Correia, 2016). On the Atlantic coast of Portugal, little tern egg volume was negatively related to both local April/May SST and April NAO (Ramos et al., 2013; Correia, 2016). In the southern North Sea, common tern clutch size was strongly negatively related to rainfall in the preceding days at some German colonies, but not others (Becker et al., 1985). At the affected colonies, the effect of rainfall on subsequent clutch size was considered to be due to an inability of males to adequately provision females. It was proposed that at the unaffected colonies, males were facultatively able to feed at sites where prey capture was less influenced by rainfall. It is not known whether these effects of weather on tern clutch size and egg volume are passed on to hatching success, fledging success, post-fledging survival or combinations of these effects.

Studies relating tern productivity to climate were only identified for two species: common and little tern. In the southern North Sea between 1982 and 1996, common tern breeding success in Germany was unrelated to December-March NAO (Reid et al., 1999). In Maine (USA), rainfall had a negative relationship with common tern nest success (Scopel and Diamond, 2018). Furthermore, rainfall moderated the relationship between krill diet prevalence and common tern nest success: when rainfall was high, high krill prevalence in the diet was associated with low nest success. This suggests that when the weather is poor, food quality becomes relevant to breeding success.

The foraging success of terns also appears to be sensitive to variation in wind speed and sea states. Common and Sandwich terns, from colonies in Northumberland, were found to

increase prey capture rates and daily fishing success from low to moderate wind speeds (Sandwich tern only) and sea choppiness (Dunn, 1973). For common and Sandwich terns in Aberdeenshire, prey capture rate was negatively related to increasing wind speeds (Taylor, 1983). This was because both species fed closer to the surface at higher wind speeds, meaning that fish deeper in the water column were not available. It therefore seems that much of the effects of weather on tern productivity is due to the effects of weather on provisioning rate.

3.3.3 Phenology

Arrival and breeding phenology of terns has generally advanced in recent years. The rate of advance in phenology has varied between species, and between different populations of the same species. Breeding date of Arctic terns in Denmark advanced by 18 days between 1929-1998 (Møller et al., 2006). From 1971-2006, Arctic, common and Sandwich terns advanced their arrival dates on the Farne Islands, but only Arctic tern also advanced their first egg dates (Wanless et al., 2009). Arrival dates of Sandwich tern in eastern Ireland advanced over the period 1980 to 2003 (Donnelly et al., 2015). In Denmark in recent years, Arctic, little and Sandwich tern both advanced and shortened their breeding season; similar relationships were non-significant in common tern (Møller et al., 2010). Common terns breeding on the North Sea coast of Germany from 1996-2010, were found to delay breeding arrival and laying dates corresponding to warmer SSTs occurring at wintering sites (Dobson et al., 2017).

Several studies, but not all, have found evidence that climate drives variation in phenology in terns. No effect of spring temperature or December-February NAO was found on Sandwich tern arrival date in eastern Ireland (Donnelly et al., 2015). Little tern arrival date in Finland (Baltic sea; at the northern edge of the species' range) is positively related to April/May NAO index and advanced in the early 21st century (Pakanen, 2018). First egg date did not advance in the same time period, and was negatively related to local temperature in the week preceding the earliest recorded laying date (2006-2016). Mean egg laying date was negatively related to local temperature in the week typically contemporary with peak laying, and positively related to April/May NAO index. Therefore, little tern arrival and breeding phenology in Finland are respectively influenced by different weather conditions, with local temperature seemingly only important to breeding phenology.

In Portugal, little tern early laying was also negatively related to April/May NAO index (Ramos et al., 2013), even though a positive NAO index brings different weather conditions in the two areas. This could indicate that little tern has a variable relationship to climate variables across its range. Alternatively, little tern phenology could be more strongly influenced by variation in broad-scale factors affecting the entire north Atlantic than by weather variation in the local area. Overall, little tern provides an important example of the influence of climate at two different times of the year on demographic parameters: timing

of breeding and egg volume are influenced by spring conditions; clutch size is influenced by winter conditions through food.

When considered as a single species group, there was no relationship between Arctic, common and Sandwich tern phenology in the North Sea for birds breeding on the Farne Islands and February-March SST or winter NAO (Wanless et al., 2009). Arctic tern date of chick ringing (a proxy of laying date) in Denmark (North and Baltic Seas) was negatively related to May NAO and Danish April and May temperatures (Møller et al., 2006). The variance in date of chick ringing was positively related to April NAO and Danish temperature.

3.3.4 Survival

No information is available from the SMP on tern survival (JNCC, 2020j, 2020m, 2020k, 2020l, 2020i), additionally no studies attributing survival to climate change were found.

3.3.5 Future projections

Arctic tern is projected to decline considerably in total range extent across Europe over the 21st century: by 48-52% (Russell et al., 2015) to 57% (Huntley et al., 2007). The future distribution is expected to overlap only moderately with current distributions: by 39% (Huntley et al., 2007) to 47-50% (Russell et al., 2015). Much of the range contraction is projected to be within the UK and Ireland (Huntley et al., 2007). In the UK, the Arctic tern has been regarded as being at high risk ([*]Ausden et al., unpubl.) with a projected 92.1% decline in abundance on UK SPAs (Johnston et al., 2013). A range of more generic risk assessments for Britain tend towards low risk (Wheatley et al., 2017) or a balance of potential risks and benefits (Pearce-Higgins et al., 2017).

Common tern is projected to decline in total range extent in Europe over the 21st century: by 21-25% (Russell et al., 2015) to 29% (Huntley et al., 2007). In Europe, the future distribution is projected to overlap only partly with the current distribution: by 55% (Huntley et al., 2007) to 68-71% (Russell et al., 2015). The European distribution is projected to extend northwards, and become more patchy in the south and east (Huntley et al., 2007). In the UK, common tern is expected to increase in range ([*]Ausden et al unpubl.) and abundance (by 66.7% on SPAs - Johnston et al., 2013). A range of more generic risk assessments for Britain span the full range of potential risk levels (Wheatley et al., 2017) or project high benefit (Pearce-Higgins et al., 2017).

Little tern is projected to decline in total range extent in Europe over the 21st century: by 0-3% (Russell et al., 2015) to 29% (Huntley et al., 2007). The future distribution in Europe is projected to overlap with the current distribution by 17% (Huntley et al., 2007) to 57-61% (Russell et al., 2015). The European distribution is projected to become more patchy and to shift northwards (Huntley et al., 2007). In the UK, climate change is projected to present

little tern with high benefit ([*]Ausden et al., unpubl.) with a projected 158.4% increase in abundance on UK SPAs (Johnston et al., 2013). A range of more generic risk assessments for little tern in Britain span the full range of potential risk levels (Wheatley et al., 2017) or project high benefit (Pearce-Higgins et al., 2017).

Future projected changes in roseate tern total range extent in Europe over the 21st century vary in direction: from a 33% decrease (Russell et al., 2015) to a 4% increase (Huntley et al., 2007). The future distribution in Europe is projected to overlap only partly with the current distribution: by 10-11% (Russell et al., 2015) to 41% (Huntley et al., 2007). Colonies in France are projected to become climatically unsuitable (Huntley et al., 2007). In the UK, climate change is projected to present roseate tern with high benefit (Pearce-Higgins et al., 2017); similarly in the UK, climate change is projected to present roseate tern with very high benefit ([*]Ausden et al., unpubl.). A range of more generic risk assessments for roseate tern in Britain span the full range of potential risk levels (Wheatley et al., 2017) or project high benefit (Pearce-Higgins et al., 2017).

Future projected changes in Sandwich tern total range extent in Europe over the 21st century vary in direction: from a 37% decrease (Huntley et al., 2007) to a 0-13% increase (Russell et al., 2015). The future distribution in Europe is projected to shift northwards (Huntley et al., 2007), overlapping only partly with the current distribution: by 30% (Huntley et al., 2007) to 42-47% (Russell et al., 2015). Future projected changes in Sandwich tern range and abundance within the UK vary: from an 4.5% decline in abundance on SPAs (Johnston et al., 2013), to projections of low risk with moderate benefit ([*]Ausden et al., unpubl.). A range of more generic risk assessments for Sandwich tern in Britain span the full range of potential risk levels (Wheatley et al., 2017) or project high benefit (Pearce-Higgins et al., 2017).

3.3.6 INTERREG VA region summary

Within the study region, Arctic and common terns are prevalent throughout, while little and sandwich terns are found in restricted areas (Balmer et al., 2013). Roseate tern remains the rarest, with a singular pair in Northern Ireland (Booth Jones, 2020a). The populations within the region have been varied between 2000-2018, with increasing abundance of common and sandwich terns, and decreases in arctic and little terns (JNCC, 2020a). In the West of Scotland, variation in trends in abundance have been largely been linked to the management of mammalian predators (Craik, 1997; JNCC, 2020i). Outside the region, adverse wind conditions have influenced foraging efficiency (Dunn, 1973; Taylor, 1983), and productivity, through altered sandeel abundance (Ratcliffe, 1997). Relatively little is known about the diet of tern species breeding within the study region however. Breeding

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[*] *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.

phenology has been shown to have shifted within the North Sea (Wanless et al., 2009), shifts which may have concurrently occurred at colonies in the West of Scotland and Irish Sea. However, more knowledge is required of tern diet and ecology within in the region to be able to assess potential climate related impacts. The impacts of terrestrial predators such as mink may currently outweigh climate influences on productivity. Potential impacts of sea-level rise upon habitat availability for nesting colonies are likely to differentially affect species according to their habitat requirements and the impact that human coastal management will have on this availability. This could result in significant redistribution of individuals between colonies. The direction and magnitude of future projections of climate change varies between species and studies, but having the most northerly distribution, Arctic terns seem most vulnerable and may be most likely to decline, whilst more southerly common and little terns appear likely to benefit from warmer climates and may increase in abundance.
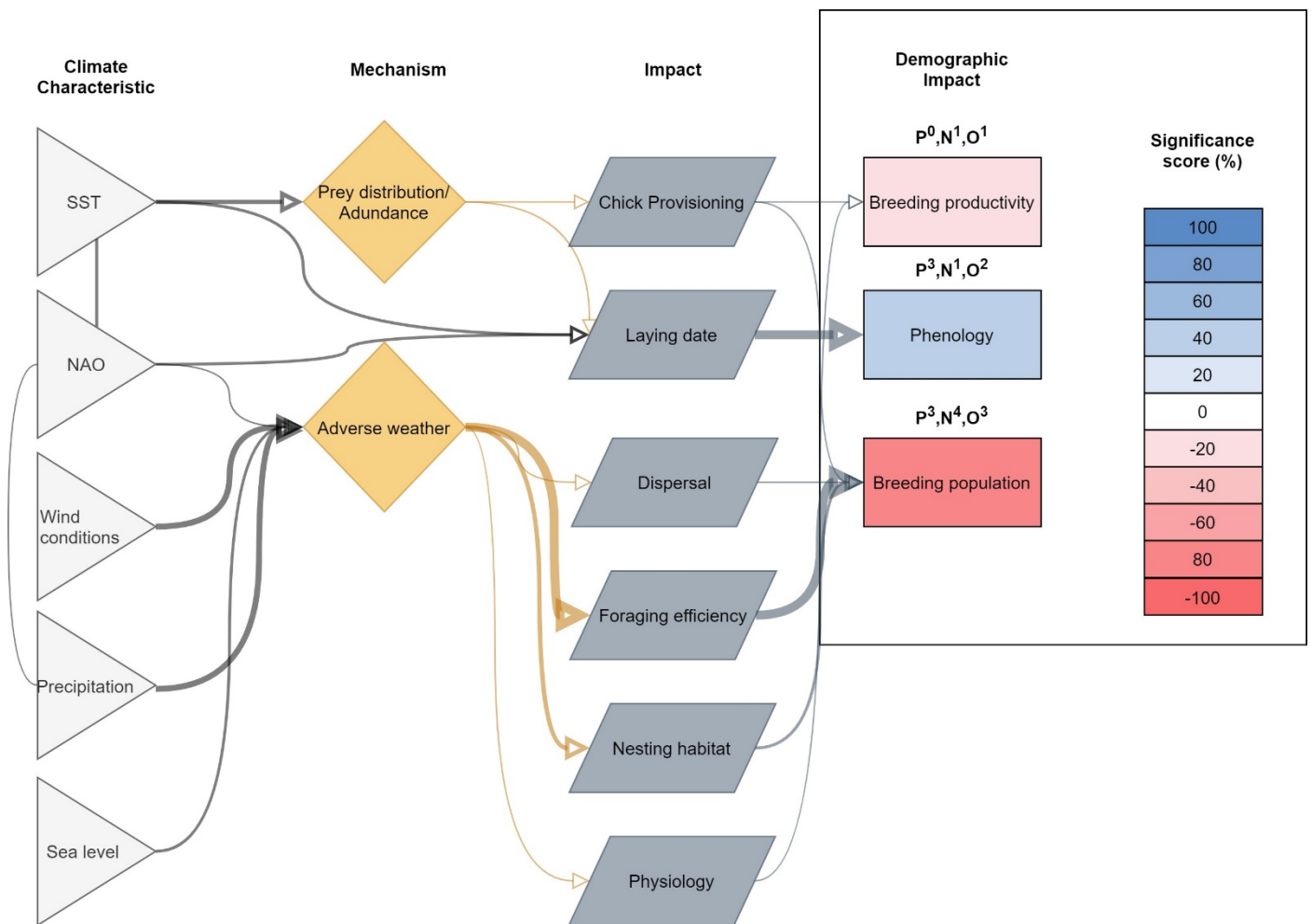
Figure 4. Grouped tern species related mechanisms flow chart. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.4 Skuas

3.4.1 Breeding population

Within the study region, Arctic skuas *Stercorarius parasiticus* breed exclusively within Scotland (Balmer et al., 2013; JNCC, 2020n) and great skuas *Stercorarius skua* breed primarily within Scotland (Balmer et al., 2013; BirdLife International, 2020g), with disparate breeding pairs in Ireland and Northern Ireland (Cummins et al., 2019; Booth Jones, 2020b).

Estimated maximum populations of Arctic and great skua within the Northeast Atlantic are 35,000 and 16,000 breeding pairs respectively (Mitchell et al., 2004a). Arctic skuas have experienced marked declines in their population between 1985-1988 and 1998-2002 of -

37% to 2,100 breeding pairs (Mitchell et al., 2004a). In contrast, great skuas experienced an increase of 26% to 9,600 breeding pairs (Mitchell et al., 2004a). These contrasting trends in breeding abundance have continued from 2000 to 2018 (JNCC, 2020o, 2020n, 2020a). The UK and Ireland are areas of international importance for great skuas, supporting 60% of the global population (Mitchell et al., 2004a; JNCC, 2020a).

The population decline in Arctic skuas is primarily attributed to notable reductions in their breeding success, which in turn is related to the breeding success of the species they prey or kleptoparasite on, via bottom up processes. Additionally, the density of nesting great skuas sharing a colony assemblage with arctic skuas may negatively influence breeding success (Perkins et al., 2018). Increases in great skuas have been linked their ability to foraging on alternate prey resources (Church et al., 2019). While the diet of breeding Arctic skuas has been recorded at colonies in Shetland as being primarily based on sandeels (Davis et al., 2005), great skuas at the same sites also forage on fisheries discards, and other seabirds (Phillips et al., 1997; Votier et al., 2003).

3.4.2 Breeding productivity

Arctic and great skua populations on Shetland have experienced periods of low productivity from 1986-2018 (JNCC, 2020n, 2020o). Their breeding success is largely dictated by the availability of sandeels (Hamer et al., 1991; Davis et al., 2005). Breeding success in Arctic skuas has been linked to the indirect influence of climate and fisheries related fluctuations in sandeel abundance acting upon the species on which skuas act as kleptoparasites, such as: black-legged kittiwake, common guillemot, Atlantic puffin *Fratercula arctica*, Arctic tern (Perkins et al., 2018). The seabird component of great skua's diet also includes Arctic skua chicks which may therefore experience additional top down predation pressure from great skuas during years of low sandeel abundance (Jones et al., 2010). The extent of this predation pressure has caused Arctic skua breeding productivity to correlate negatively with great skua density (Perkins et al., 2018). Additionally, great skuas have been shown to outcompete Arctic skuas for nesting habitat (Dawson et al., 2011). However, the degree to which great skuas may respond to bottom-up shift in prey may be dependent on the species composition of seabirds at their host colony (Phillips et al., 1997). Shifts in target seabird host species may also indicate bottom-up shifts, with sandeel-dependent kittiwakes becoming less frequent in the diet of Shetland-based skuas, being replaced by clupeoid-foraging auks (Church et al., 2019). Although great skuas have been able to substitute their diet, this may increase energy expenditure required to travel and forage on alternate resources, and may result in a reduction in breeding success by chick predation from other skuas due to longer periods of absence from the nest (Hamer et al., 1991). Dawson et al. (2011), noted the primary driver of reduced productivity of great skuas, alongside reduced productivity of prey seabirds such as auks, was chick predation by con-specifics and great black-backed gulls. In addition to such bottom-up and top-down pressures on chick rearing great skuas, they may experience direct heat stress from excessively warm weather (Oswald et al., 2008). Heat stress is potentially exacerbated within the study region as skuas reside at

the southern limit of their range within the Atlantic (BirdLife International, 2020g, 2020h). In years of low sandeel availability, the potential for heat stress on adults has been shown to be higher, due to more time spent foraging, less time spent guarding chicks from con-specific predators and fewer opportunities to bathe for thermoregulation (Oswald et al., 2008).

3.4.3 Phenology

No information is available from the SMP on great skua and Arctic skua phenology (JNCC, 2020o, 2020n). Additionally, no studies attributing phenology to climate change were found.

3.4.4 Survival

No information is available from the SMP on trends in great skua and Arctic skua survival (JNCC, 2020o, 2020n), however studies of return rates are being carried out (Poltz Faggiani and Jones, 2019). Additionally, no studies attributing survival to climate change were found.

3.4.5 Future projections

Although Arctic skua is predicted to decline considerably in total range extent across Europe over the 21st century, the magnitude of this change is uncertain, varying from 28% (Huntley et al. 2007) to 76-80% (Russell et al. 2015). Similarly, there is considerable uncertainty in the predicted overlap between current and future distributions, varying from 70% (Huntley et al. 2007) to 19-24% (Russell et al. 2015). More southerly current populations of Arctic skua are not predicted to occur on the basis of climatic modelling (Huntley et al. 2007), suggesting that Arctic skua is likely to be unable to be adjust to changes at the fringes of their range. Huntley et al. (2007) predict that Arctic skua will no longer breed in the UK and Ireland by the latter 21st century. In the UK, climate change is projected to present Arctic skua with very high risk ([*]Ausden et al., unpubl.).

Great skua is projected to decline in total range extent in Europe over the 21st century: by 15-32% (Russell et al., 2015) to 36% (Huntley et al., 2007). The future distribution in Europe is projected to overlap little with the current distribution: by 15-22% (Huntley et al., 2007; Russell et al., 2015). Great skua is not predicted to breed in the UK and Ireland by the late 21st century (Huntley et al., 2007). No climate vulnerability risk assessments have been carried out for great skua.

3.4.6 INTERREG VA region summary

Within the study region great skuas are found predominately on the West of Scotland (Balmer et al., 2013), although some breeding pairs exist in the North-West of Ireland (Cummins et al., 2019) and a single breeding pair is present on Rathlin Island, Northern Ireland (Booth Jones, 2020a). Within the region, Arctic skuas are found in smaller numbers on the Outer and Inner Hebrides of Scotland (Balmer et al., 2013; Poltz Faggiani and Jones,

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[*] *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.

2019). Though the skua species have been monitored much more widely outside the study region (Perkins et al., 2018), information on their ecology within the region has largely arisen from Handa Island in the North-West of Scotland (Jones et al., 2010; Poltz Faggiani and Jones, 2019). Declines in the numbers of Arctic skuas, and variable population trends of great skuas may be attributed to the foraging success of their host seabirds. Predation by great skuas on con-specifics or Arctic skuas may also influence productivity (Dawson et al., 2011). These bottom-up and top-down pressures may be linked to climate's influence on the diet of seabird prey species. While catastrophic declines have been linked to sandeel availability, variation in the diet of seabird prey species on the West of Scotland may make the influence of climate trends, such as increasing SST, less certain (Harris and Wanless, 1990; Chivers et al., 2012a; Anderson et al., 2014b).
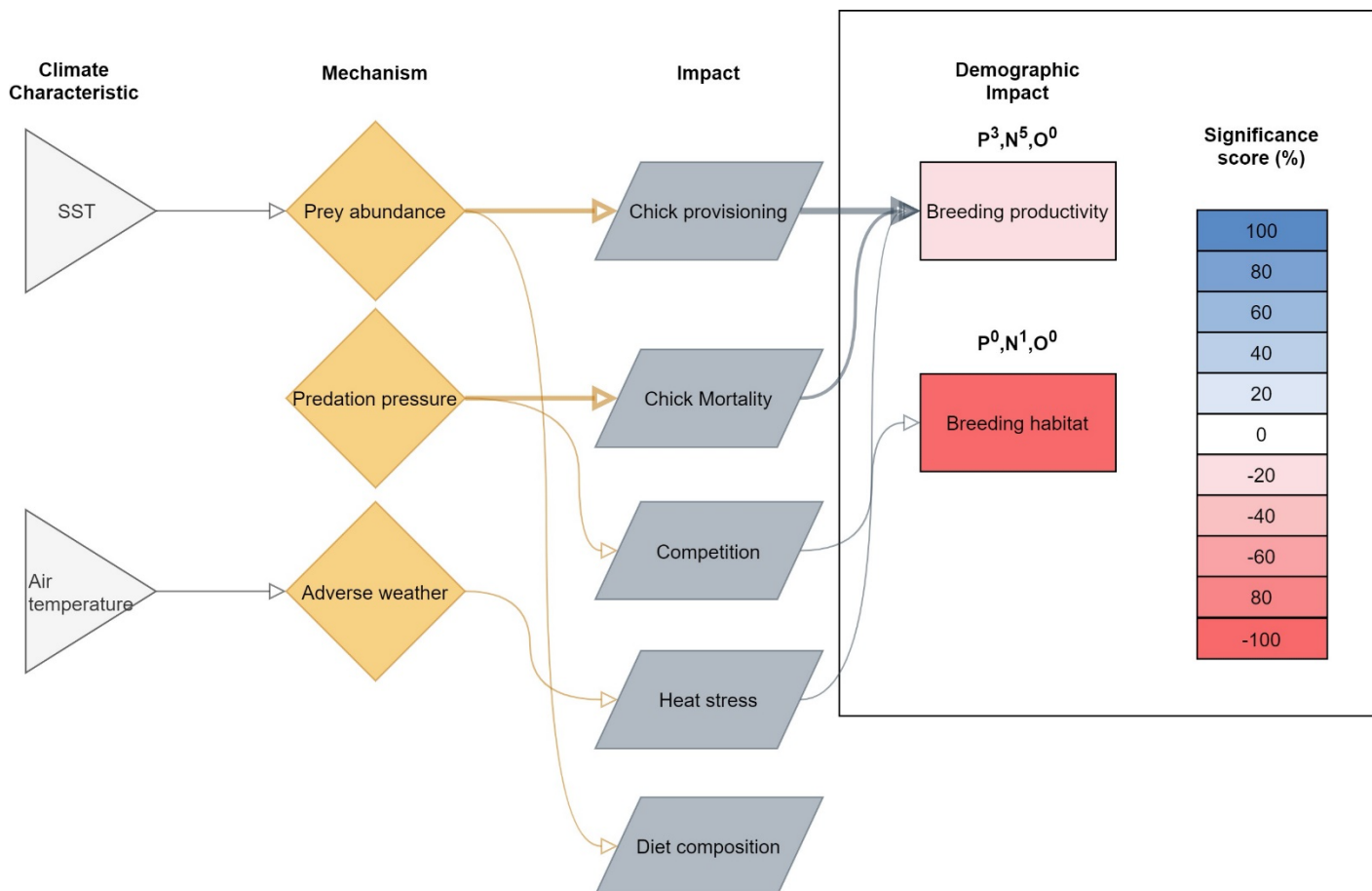
Figure 5. Grouped skua species related mechanism flow chart. Lines sized by number of respective studies. SST = Sea-Surface Temperature. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation-significance score = P-N/(P+N+O).

3.5 Common guillemots *Uria aalge*

3.5.1 Breeding population

Common guillemots *Uria aalge* (hereafter called guillemots) are one of the most populous seabird species within the North Atlantic, with a population of 2,900,000 breeding pairs (Mitchell et al., 2004a). Within the North-East Atlantic they range from the temperate Iberian peninsula to the Arctic (JNCC, 2020p). The UK and Ireland contain around 36% of the global population, which, despite some colony specific variation, has displayed an overall increase in numbers between 1986-1988 and 1998-2002 (+24%) (Mitchell et al., 2004a).

The population trends of guillemots within the UK and Ireland has been recorded as stable between 2000-2018 with very slight increases of around +1% (JNCC, 2020a), although some colonies display continued increases (Meade et al., 2013). Population abundance in guillemots has been previously shown to be negatively affected by fisheries bycatch (Munilla et al., 2007; Regular et al., 2010) and oil pollution (Cadiou et al., 2004; Votier et al., 2005). However, population collapse and recovery has more recently also been related to climate mediated bottom-up impacts on prey (Erikstad et al., 2013). It would appear that the influence of anthropogenic impacts from events such as oils spills or bycatch on population and demographic trends are more important than those related to climate change. When affected by direct climatic impacts such as adverse winter weather (Harris and Wanless, 1996) or summer storms (Newell et al., 2015), the extent of negative demographic impacts have been potentially buffered by their largepopulation size.

3.5.2 Breeding productivity

Breeding productivity within the UK remained relatively stable until 2002, then subsequently undergoing a decline until 2009, and then increased and stabilized thereafter (JNCC, 2020p).

Of the studies investigating the environmental characteristics driving trends in breeding success within the UK, climatic variables such as NAO, SST, or stratification are often found to have a negligible effect (Reid et al., 1999; Frederiksen et al., 2006; Scott et al., 2006; Kadin et al., 2012; Lauria et al., 2012). This may be explained by guillemots pursuit diving behaviour allowing foraging at a range of depths, providing resilience to fluctuations in prey availability (Furness and Tasker, 2000). While their foraging behaviour may buffer guillemots from the worst effects of influencing prey availability, direct impacts from adverse weather have been shown to decrease chick feeding frequency (Birkhead, 1976) and prey quality (Finney et al., 1999). Rough at-sea conditions correlate with lengthened intervals between feeds, potentially due to prey dispersal in energetically demanding conditions.

While changes to prey availability may be mitigated by guillemot foraging behaviour, prey abundance is repeatedly cited as correlating with breeding success, as evidenced by studies

on sprat, within the Baltic Sea (Kadin et al., 2012); sandeel, within the North Sea (Rindorf et al., 2000; Frederiksen et al., 2006; Burthe et al., 2012); cod *Gadus morhua,* in the Norwegian Sea (Barrett and Erikstad, 2013) and herring, in the Celtic and Norwegian Seas (Lauria et al., 2012; Barrett and Erikstad, 2013). There is further support for bottom-up influence of climatic conditions operating. On the Isle of May, Frederiksen et al. (2013) found breeding success correlated positively with environmental suitability of *C. finnmarchicus*, a lipid rich copepod resource for sandeels, with a one-year lag. The habitat suitability for this copepod related strongly to the abundance of 1-group sandeels, an important prey for pre-breeding and chick feeding guillemots. However, when compared to a colony on Røst, Norway, *C. finnmarchicus* suitability did not correlate with breeding success, potentially due to their diet relating primarily to herring, which do not feed on the same prey.

3.5.3 Phenology

Climatic variables have been shown to correlate with changes in guillemot phenology, which have shown variable trends between colonies within the UK (JNCC, 2020p). Negative NAO values corresponded to later laying dates/hatching dates (Frederiksen et al., 2004a; Crespin et al., 2006; Harris et al., 2006; Wanless et al., 2009) and more synchronous (Reed et al., 2006) breeding in the North Sea, while positive values of NAO were associated with significantly later breeding at Skomer within the Celtic Sea (Votier et al., 2009). Long term trends towards earlier breeding have been shown in the North Sea and the seas surrounding Shetland (Frederiksen et al., 2004a; Harris et al., 2006; Wanless et al., 2009; Burthe et al., 2012). The mechanism of NAO's influence on phenology is likely related to contrasting effects on colony specific prey abundance affecting pre-breeding adult condition (Harris et al., 2006; Anderson et al., 2014b), and the alignment of breeding to the timing of the spring plankton bloom (Burthe et al., 2012).

3.5.4 Survival

Return rates (as a proxy for survival rates) within the UK, as recorded on the Isle of May and Skomer, have shown respective declines and fluctuations, though have generally displayed no strong trends (JNCC, 2020p). Overwinter survival rates in guillemots was found to be effected primarily by intrinsic fitness of an individual bird (Harris et al., 2007b), potentially related to age (Crespin et al., 2006), the carry-over effect from previous breeding attempts (Reed et al., 2015), or parental quality (Harris et al., 2007b). The additional effects of climate displayed variable influence between studies, with NAO influencing return rates in Crespin et al. (2006), and skipping propensity found to be higher in years with higher winter SSTs by Reed et al. (2015). However, Harris et al. (2007b) found winter NAO and storminess had no correlation with first year survival. Parental quality was found to be the overriding influence on first year survival (Harris et al., 2007), likely related to high quality parents who lay earlier and fathers who better prepare chicks to survive their first winter (Harris et al., 2007b). High winter SST and NAO values effecting prey availability, compounded by anthropogenic impacts of oil spills, have been shown to have negative effects on guillemot survival (Votier et al., 2005, 2008b).

In a trait unique among seabirds, common guillemots display a distinct phenotypic variation. Bridled birds exhibit a white eye stripe, and unbridled birds have a fully black head. These contrasting morphs have varying winter distributions and traits advantageous to contrasting climatic conditions (Birkhead, 1984; Reiertsen et al., 2012). The occurrence of bridled birds, and breeding success of pairs containing a bridled individual, are generally correlated with lower temperatures (Reiertsen et al., 2014). Therefore, climate change may have an underlying impact on the phenotypic composition of guillemot populations.

3.5.5 Future Projections

Future projected changes in guillemot total range extent in Europe over the 21st century vary: from a 14% decrease (Huntley et al., 2007) to a 3-11% increase (Russell et al., 2015). The future distribution in Europe is projected to overlap only partly with the current distribution by 54-62% (Huntley et al., 2007; Russell et al., 2015). France and much of the southern UK are projected to become climatically unsuitable (Huntley et al., 2007). Future projected changes in guillemot range and abundance within the UK vary: from a 75.5% decline in abundance on SPAs (Johnston et al., 2013), to projections of low risk with moderate benefit ([*]Ausden et al., unpubl.). In the North Sea and the British continental shelf, there is uncertainty in the sign and magnitude of the predicted change in guillemot population size by the middle of the 21st century: from -13.0 to +4.6% (Sadykova et al., 2020). Over the same time period, there is predicted to be considerable geographical change in the overlap between guillemot's at-sea distribution and the distribution of its prey species (herring, sandeel; Sadykova et al. 2020). A range of more generic risk assessments for guillemot in the UK project medium to low risk (Wheatley et al., 2017) or a combination of risks and benefits (Pearce-Higgins et al., 2017).

3.5.6 INTERREG VA region summary

Guillemots are found throughout the study region (Balmer et al., 2013). Colonies displayed varying population trends within the region between 2000-2018 (JNCC, 2020p). Increasing populations recorded at colonies within the Irish Sea contrast with declines on the West of Scotland. Diet composition potentially varies between these areas, as sandeels form an important component of the diet in the West of Scotland, while clupeids are more prevalent within the Irish Sea (Chivers et al., 2012b; Anderson et al., 2014b). The dietary variation between these colonies may underpin their response to increasing SSTs during the breeding season. Varying prevalence of low quality snake pipefish between colonies within the region (Anderson et al., 2014b) may indicate that colonies are experiencing detrimental alterations in prey abundance (Harris et al., 2007a). While diets between breeding areas are distinct, within the study region, wintering areas used by guillemots may be similar. This may lead to the negative impacts of adverse weather on prey availability affecting adult condition similarly across the study region (Votier et al., 2005; Morley et al., 2016).

Figure 6. Common guillemot related mechanism flow chart. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature; NHT = Northern Hemisphere Temperature anomaly. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.6 Razorbill *Alca torda*

3.6.1 Breeding population

Razorbills *Alca torda* are found exclusively within the North Atlantic, within the North-East Atlantic their range extends from Brittany to the Arctic (JNCC, 2020q). They have been estimated to have a global population of 630,000 breeding pairs, with surveys 1998-2002 in estimating the UK and Ireland to contain 216,087 individuals (Mitchell et al., 2004a; JNCC, 2020q). Their population increased in the UK and Ireland between 2000 and 2018 by 33% but there have been differences between regions with some losses being reported (JNCC,

2020a). Their diet is comprised predominantly of sandeels at more southerly latitudes of the North Atlantic (Harris and Wanless, 1986) and herring further north (Barrett, 2015). The association between Razorbills and sandeels has been related to a northward range expansion within Hudson Bay, Canada, which correlated with the arrival of sandeels (Gaston and Woo, 2008). Regarding climatic variables, population growth in the Wadden Sea was found to correlate with positive winter NAO values, though this could not be biologically explained, and was attributed to an coinciding recovery in the population (Reid et al., 1999).

3.6.2 Breeding productivity

Productivity within the UK has been shown to fluctuate throughout 1986-2018, with periodic declines associated indirectly with low prey abundance (JNCC, 2020q). Direct nest failure caused by high winds was found to disproportionately affect razorbill breeding productivity compared to other exposed cliff nesting species (Newell et al., 2015). This was attributed to the lighter weight of razorbills, making them more susceptible to being lifted off their nests by high winds.

Studies of the indirect effect of climate on razorbill breeding success are more numerous, and are primarily attributed to bottom-up effects on prey. Spring SST temperatures were found to correlate with razorbill productivity (Lauria et al., 2012). High SSTs in winter and spring may have a potential negative effect on the growth rates of sandeels (Burthe et al., 2012), or abundance of herring (Lauria et al., 2012) which have both correlated positively with razorbill productivity. However, based on long term monitoring, the biomass of 0-group (Burthe et al., 2012) and 1-group (Frederiksen et al., 2006) sandeels was not found to have an effect on razorbill productivity.

Spring NAO had a negative influence on breeding productivity in the Celtic Sea (Lauria et al., 2012), potentially through influence on prey availability, or the direct impacts of high winds and storm frequency related to positive NAO values.

3.6.3 Phenology

Currently no phenological trends are reported by the SMP (JNCC, 2020q). However, later laying dates have been correlated with lower SST (Harris and Wanless, 1989; Barrett, 2001) and negative winter NAO (Wanless et al., 2009), which in turn could be influenced by timing of prey availability. However, Burthe et al. (2012) found predator and prey trophic levels did not synchronously react to changes in winter and spring SST, concluding that changes in phenology of razorbills may be insufficient to avert trophic mismatching.

3.6.4 Survival

At two colonies which monitor long-term survival rates within the UK, the Isle of May has reported fluctuating return rates between 1986-2018, while on Skomer no trend was seen over a similar period (JNCC, 2020q). Local SSTs to have been found to have a larger influence on survival than NAO (Sandvik et al., 2005; Lavers et al., 2008). This was attributed to SST potentially having a greater effect on prey availability than NAO for inshore foraging

razorbills. Colony specific foraging home-ranges exhibited by razorbills were attributed to variation in survival between colonies in North-East Canada (Lavers et al., 2008). Lavers et al. (2008) cited the spatially varying influence of local oceanography on bottom-up effects on prey availability to vary survival rates between colonies. Local currents and hydrodynamics are therefore an important influence on the prey of inshore foraging razorbills.

3.6.5 Future projections

Razorbill is projected to decline in total range extent in Europe over the 21st century: by 12-17% (Russell et al., 2015) to 28% (Huntley et al., 2007). The future distribution in Europe is projected to overlap only partly with the current distribution: by 41% (Huntley et al., 2007) to 44-48% (Russell et al., 2015). Much of the south of the range, including within the UK and Ireland, is projected to become climatically unsuitable. Future projected changes in razorbill range and abundance within the UK vary: from an 79.5% decline in abundance on SPAs (Johnston et al., 2013), to projections of moderate risk with low benefit ([*]Ausden et al., unpubl.). A range of more generic risk assessments for razorbill in Britain span the full range of potential risk levels (Wheatley et al., 2017) or project a combination of risks and benefits (Pearce-Higgins et al., 2017).

3.6.6 INTERREG VA region summary

Razorbills are found throughout the study region (Balmer et al., 2013), however populations trends within the region are not uniform. Between 2000-2018, populations declined on the West of Scotland, and increased within the Irish Sea (JNCC, 2020q). Of the few studies on razorbill diet carried-out within the region, sandeels have been found to be the predominant prey (Swann et al., 1991; Chivers et al., 2012b). This diet composition is similar to that further south in the Celtic Sea (Shoji et al., 2015), and the North Sea (Harris and Wanless, 1986, 1989). This comparable diet to more intensively studied colonies in the Celtic and North Seas suggest razorbills within the study region may respond in a similar manner to environmental change.

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[*] *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.
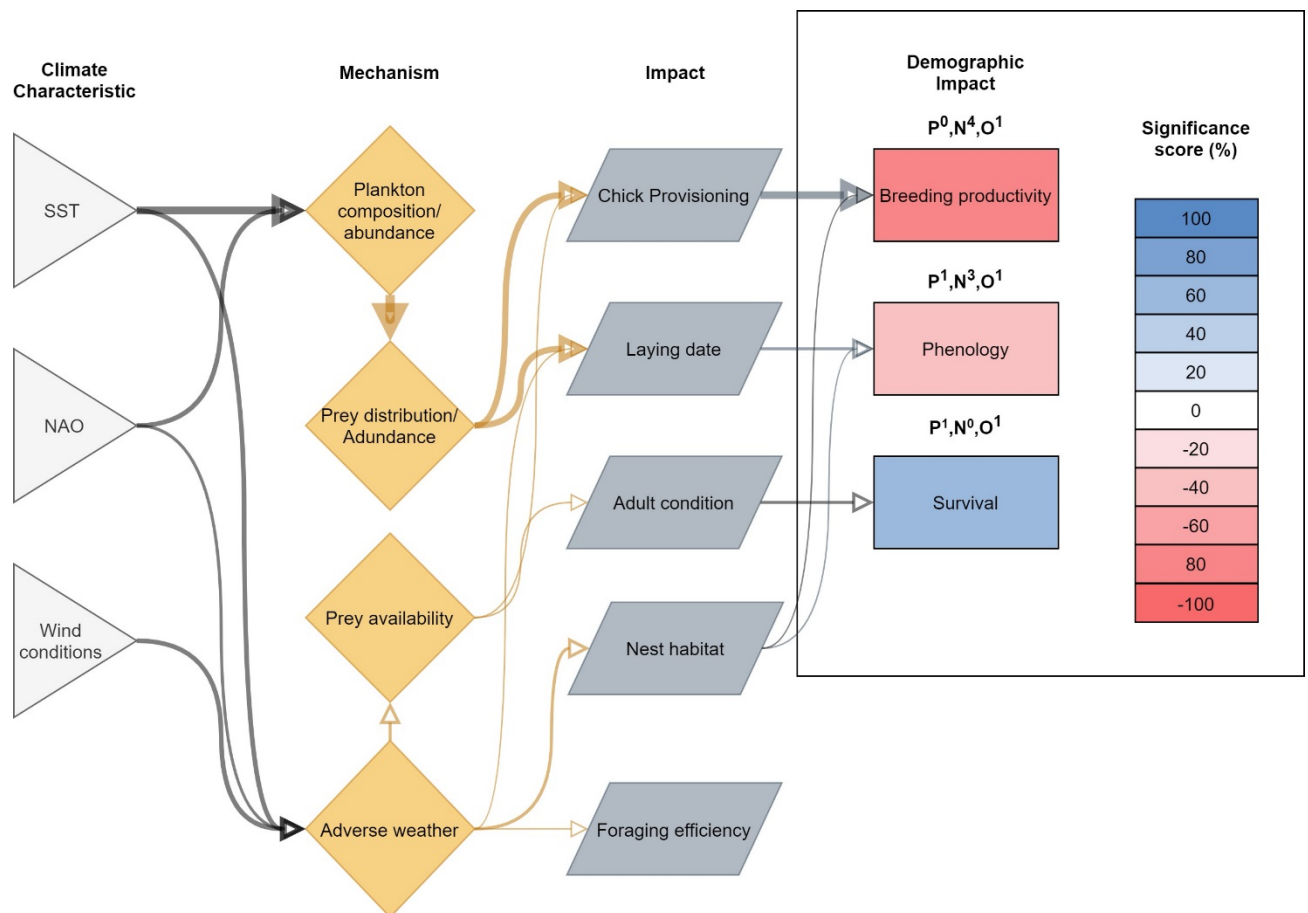
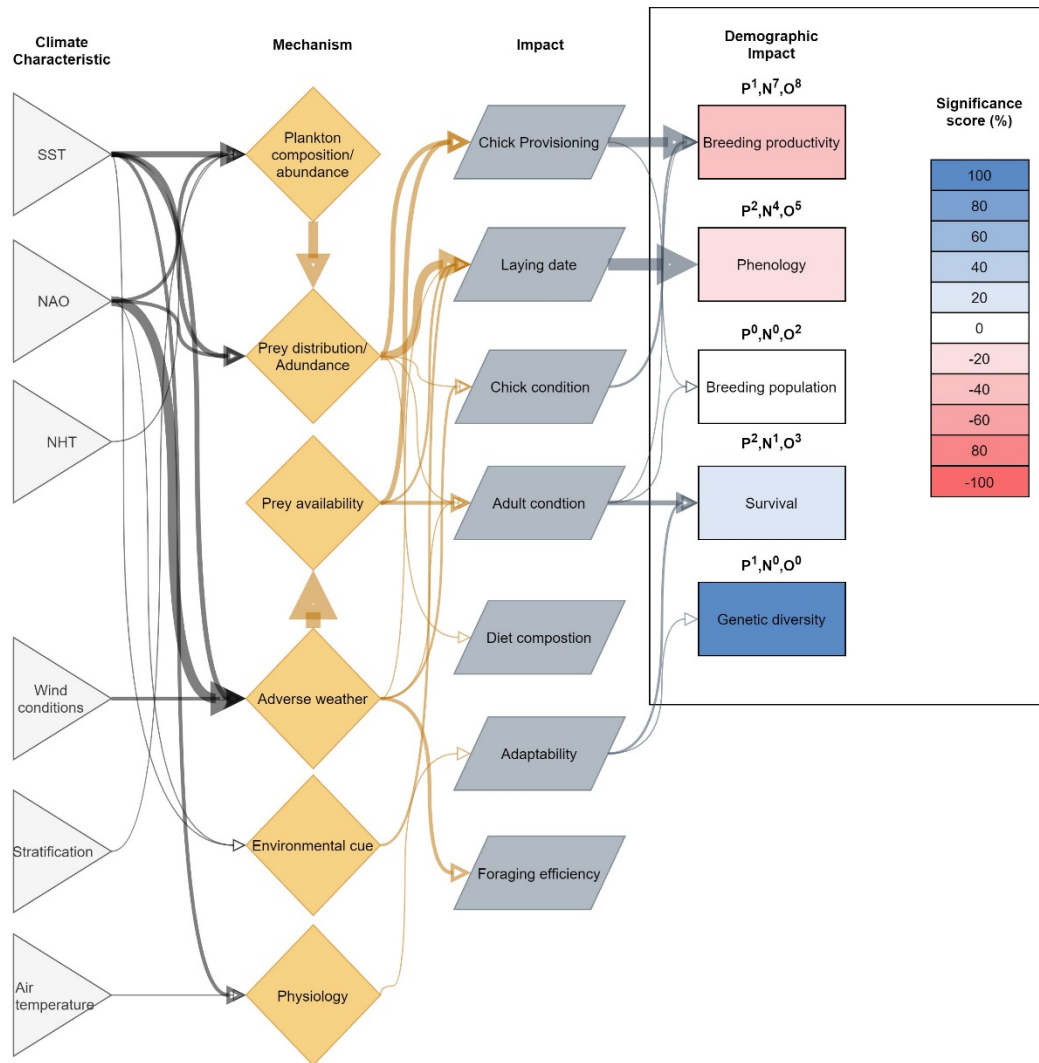**Figure 7. Razorbill related mechanism flow chart. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).**

3.7 Black guillemots *Cepphus grylle*

3.7.1 Breeding population

Black guillemots maintain circumpolar distribution within the Northern Hemisphere. Globally they have been estimated to have a maximum population of 410,000 breeding pairs (Mitchell et al., 2004a). Within the UK and Ireland, their breeding population has remained stable between 1988-2002 with the most recent estimate of 42,683 individuals, the majority of which breed in Scotland (Mitchell et al., 2004a; JNCC, 2020r). The stable population of black guillemots within the UK and Ireland is largely attributed to their generalist inshore foraging behaviour and lack of reliance on prey susceptible to climatic fluctuations, namely butterfish *Pholis gunnellus* (Ewins, 1990). Their population is potentially restricted by nest site availability which may be negatively influenced by human disturbance (Cairns, 1980) and mammalian predation, including from American mink

Neovison vison (Craik, 1997; JNCC, 2020s). Black guillemots within the study region reside within their extreme southern limit (BirdLife International, 2020i) and therefore may be more susceptible to shifting environmental conditions. Within the Baltic sea, the foraging range of black guillemots was projected to expand, while the available breeding habitat was suggested to constrict with rising air temperatures, and increased precipitation (Buchadas and Hof, 2017b).

3.7.2 Breeding productivity

Breeding productivity is stable in the UK and Ireland (JNCC, 2020s), though few studies have related productivity to climatic variables. Coastal nesting black guillemots may be susceptible to direct effects of adverse weather through flooding from heavy precipitation or extreme storm swell (Hario, 2001). Long-term studies in the Alaskan Arctic have shown indirect negative effects of sea ice retreat (Divoky et al., 2015) and range shifts in competing seabirds (Divoky, 2011). Decline in the availability of sea ice associating Arctic cod *Boreogadus saida,* and replacement with lower lipid nearshore sculpins, has been related to a decline in the condition of black guillemot chicks, in particular slower growth, decreased weight at fledging, and lower breeding success (Divoky et al., 2015; Harwood et al., 2015).

3.7.3 Phenology

While few studies examine black guillemot phenology in relation to climate, at Bangor, Northern Ireland; Greenwood (2007) reported earlier laying dates in years of warmer April SSTs. While no long-term trend (1985-2006) was reported, the variation in laying date was potentially attributed to the timing in availability of butterfish. In the Alaskan Arctic, rising temperatures have potentially extended the breeding season with the earlier melting of snow restrictive to nest building. While earlier laying was associated with increased breeding success and adult survival, at the Alaskan colony breeding success has declined due to prey availability (1976-2017) (Sauve et al., 2019).

3.7.4 Survival

No information is available from the SMP on black guillemot survival (JNCC, 2020s), additionally no studies attributing survival to climate change were found.

3.7.5 Future projections

Black guillemot is projected to decline considerably in total range extent across Europe over the 21st century: by 41% (Huntley et al. 2007) to 36-41% (Russell et al. 2015). The future distribution is expected to overlap only moderately with the current distribution: by 50% (Huntley et al., 2007) to 57-62% (Russell et al., 2015). Most of the current distribution in the UK and Ireland is projected to become climatically unsuitable (Huntley et al., 2007). In the UK, black guillemot is regarded as being at high risk under climate change ([*]Ausden et al.,

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[*] *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.

unpubl.) although a range of more generic risk assessments in Britain place black guillemot at medium to low risk (Pearce-Higgins et al., 2017; Wheatley et al., 2017).

3.7.6 INTERREG VA region summary

Within the study region, black guillemots from colonies in Scotland, Ireland, and Northern Ireland, have been found to be increasing (JNCC, 2020s), potentially related to availability of nest sites. While few colonies are monitored on the West of Scotland (Johnston et al., 2019), long-term studies have been carried out in Northern Ireland. Such monitoring has been conducted in Bangor Marina, Northern Ireland which reported laying date to correlate with SST (Greenwood, 2007). Within the UK and Ireland, butterfish forms a ubiquitous prey component (Ewins, 1990; Leonard et al., 2014). Environmental changes that influence butterfish habitat, may have a detrimental impact on black guillemot foraging throughout the UK and Ireland. This may be compounded in the study region, as it forms the southern limit of black guillemots global range (BirdLife International, 2020i), and they may therefore reside at the limit of physiological and habitat suitability.
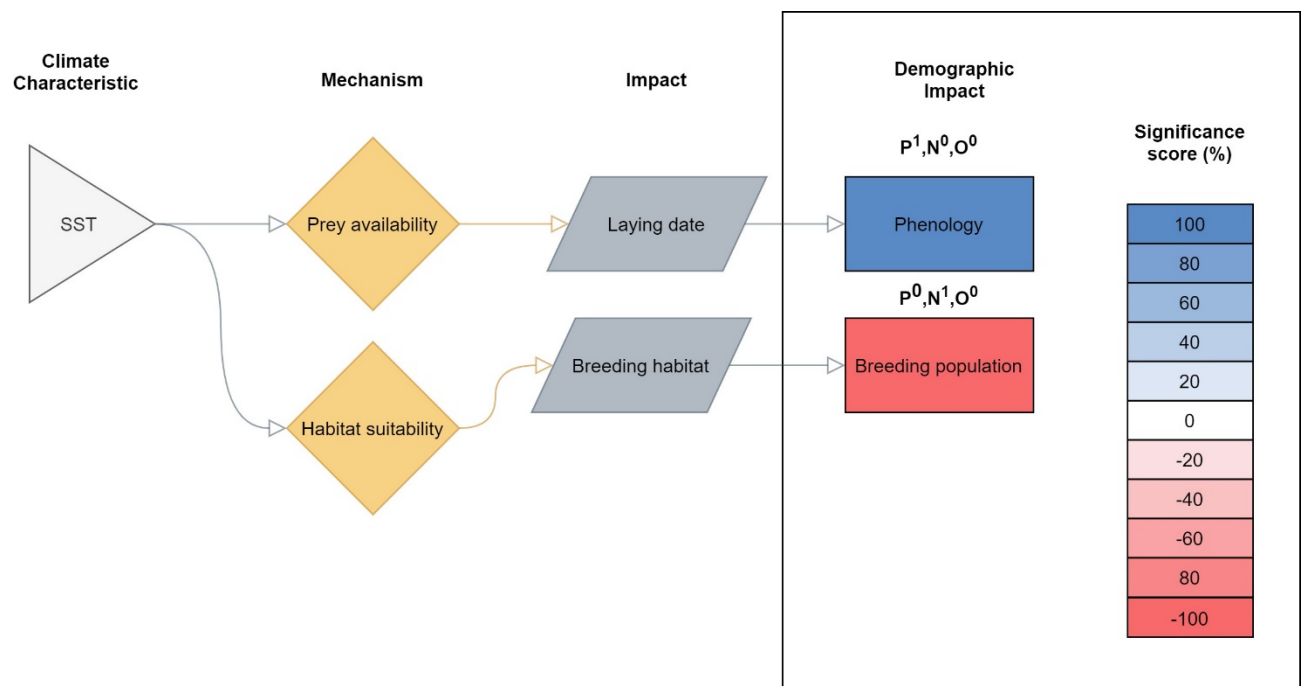
Figure 8. Black guillemot related mechanism flow chart. Lines sized by number of respective studies. SST = Sea-Surface Temperature. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O =number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.8 Atlantic puffin *Fratercula arctica*

3.8.1 Breeding population

Atlantic puffins *Fratercula arctica* (hereafter referred to as puffins) maintain a distribution within the North-East Atlantic ranging from the North-West of France to the archipelago of Svalbard within the Arctic (Harris and Birkhead, 1985; BirdLife International, 2020j). The population within the UK and Ireland has shown an increase of 19% from 1985-1988 to 1998-2002, to population of 581,110 breeding pairs (Mitchell et al., 2004a). However, colonies throughout the UK and Ireland display varying population trends, with some declines recorded in Ireland (JNCC, 2020t) and Shetland (Miles et al., 2015). Due to the difficulties associated with monitoring burrow nesting seabirds (Arneill, 2018) such as puffins means fewer annual surveys are carried out, with little data available for the study region (JNCC, 2020t).

3.8.2 Breeding productivity

Monitoring of breeding productivity within the UK from 1986-2018 has shown this parameter to fluctuate, however with an overall decline (JNCC, 2020t). Variation in puffin breeding success has been suggested to primarily be influenced by changes in prey abundance and composition. This may stem from reliance on single species dominated prey resources including: sandeels, within the North Sea (Martin, 1989; Wanless et al., 2004); herring, within the Celtic and Norwegian Seas (Barrett, 2001; Lauria et al., 2012; Kress et al., 2017); white hake *Urophycis tenuis*, within the Northwest Atlantic (Kress et al., 2017); and Atlantic Capelin *Mallotus villosus*, within the Barents Sea (Barrett, 2001). The reduction in sandeel stocks around Shetland in the late 1980's coincided with a reduction in the load weight of feeds to chicks, leading to breeding failure at Hermaness, Shetland (Martin, 1989). Using data spanning 30 years, it has also been shown the sandeel biomass of the 0-group from the previous year (a proxy for 1+ group) correlates positively with breeding productivity in puffins breeding on the Isle of May (Frederiksen et al., 2006). At the same colony, a decline in the size of 0-group sandeels was observed through long-term monitoring of puffin diet (Wanless et al., 2004). Increased availability of 1-group sandeels during the pre-breeding period may also positively affect breeding success through improved adult body condition, which in turn will influence breeding productivity (Frederiksen et al., 2006).

Shifts in prey abundance may be related to variation in SST, and have been observed to affect productivity. Fledging success of puffins on Røst, in the Norwegian Sea, positively correlated with spring SST (Durant et al., 2003, 2006), though negatively on the Isle of May (Burthe et al., 2012) and St. Kilda (Harris et al., 1998). In Norway, the positive correlation was suggested to relate to low temperatures and higher salinity around the period of the spring bloom. This correlated with reduced plankton abundance during the spring bloom, and influenced growth of first year herring migrating north to the Barents sea, and thereon fledging success of puffins (Durant et al., 2003, 2006). While, on the Isle of May, breeding

success correlated positively with environmental suitability of *C. finnmarchicus* with a one year lag, closely associated with cooler SSTs (Frederiksen et al., 2013), and higher SSTs correlating lower sandeel abundance (Burthe et al., 2012). However in Norway, *C. finnmarchicus* suitability did not correlate with breeding success, potentially due to a herring based diet (Frederiksen et al., 2013). Similarly, winter and spring NAO, SST, and herring stocks were not seen to correlate with population size or breeding success on Skomer, with puffins showing no long-term trend in breeding success (1986-2007) (Lauria et al., 2012). In Maine, northward range shifts in prey, comprising of a decrease in white hake and increase in American butterfish *Poronotus triacanthus* (which is not digestible) in chick diet related to warming SSTs, have been shown to effect chick condition (Kress et al., 2017).

3.8.3 Phenology

Climate variation has also been linked to phenological trends in puffins, though trends in timing of breeding vary spatially, and in their relationship with climate variables. On the Isle of May, laying dates have been shown to occur later (Burthe et al., 2012), though no long term phenological trends were apparent at colonies in Norway within the Norwegian and Barents sea (Barrett, 2001; Durant et al., 2004). In the North Sea and Norwegian Sea, breeding timing correlated with winter NAO but not local winter SST (Durant et al., 2004; Wanless et al., 2009); while on St. Kilda in the North Atlantic high SST correlated with later breeding, and also declines in the mass and success of chicks (Harris et al., 1998). However, lower air temperatures and SST significantly correlated with later laying date on Hornøya (Barrett, 2001). This was related to ice remaining in burrows and the timing of prey spawning altering in warmer years, potentially allowing earlier breeding (Barrett, 2001; Durant et al., 2004). While delayed breeding may be in response to the timing of peak prey abundance (Durant et al., 2003), shifts in phenology may be insufficient, leading to trophic mismatch between peak chick rearing demands and the quality and size of prey (Burthe et al., 2012). This could potentially explain how later breeding is related to declines in the mass and success of chicks (Harris et al., 1998).

3.8.4 Survival

Long term monitoring on the Isle of May and Skomer, have shown respectively no trend in return rates, and a downward trend in survival rates between 1986-2018 (JNCC, 2020t). However, previous fluctuations in survival have been reported on the Isle of May, where a general decrease in the survival rates was observed from 1973 to 1994 (Harris et al., 1997). In contrast to other seabird species, survival appears not to be related to winter conditions, rather carry-over effects from previous breeding seasons (Harris et al., 2013). Correlations between survival and winter/spring SST in relation to breeding success were also highlighted in Harris et al. (2005). This study showed survival rates at sandeel dependent colonies in the North Sea correlated negatively with SST, while the opposite was seen on herring dependent colonies within the Norwegian and Barents Sea. This spatial variation in the response of puffin survival to SST is related to the position within the species' range, with responses tending to be negative at southerly or warmer colonies, such as Skomer and Fair

Isle, but positive in cooler or northern colonies such as Røst in Norway (Grosbois et al., 2009). In Maine, adult puffin survival has been declining since the 1990s, winter conditions, as indicated by NAO, was found to have little effect on this decline (Breton and Diamond, 2014). Instead, May SSTs, declining commercial fisheries landings, and quantity of group-1 herring in diet were shown to be significant predictors of adult annual survival.

3.8.5 Future projections

Puffin is projected to decline in total range extent in Europe over the 21st century: by 28-39% (Huntley et al., 2007; Russell et al., 2015). The future distribution in Europe is projected to overlap only partly with the current distribution: by 32% (Huntley et al., 2007) to 35-43% (Russell et al., 2015). Much of the south of the range, including within the UK and Ireland, is projected to become climatically unsuitable (Huntley et al., 2007). In the UK, puffin is projected to be at very high risk ([*]Ausden et al., unpubl.) with a projected 67.9% decline in abundance on British SPAs (Johnston et al., 2013). A range of more generic risk assessments for puffin in UK span the full range of potential risk levels (Wheatley et al., 2017) or project medium risk (Pearce-Higgins et al., 2017).

3.8.6 INTERREG VA region summary

Puffins breed disparately throughout the study region (Balmer et al., 2013), however, few long-term surveys within the region are available. Existing surveys display puffin populations to be stable on the Shiant Islands between 2000-2015 (Taylor et al., 2018), and to have declined on Rathlin Island, Northern Ireland; and Puffin Island, Ireland; between 1999-2011 (JNCC, 2020t). Few studies of puffin diet are available for the region, however sandeels are potentially a predominant prey species across the UK (Harris and Hislop, 2009; Shoji et al., 2015). Similarities between diet within the study region and that of puffins on the intensively studied Isle of May (Harris, 1997; Harris et al., 2005) may indicate that colonies on the West of Scotland and Irish Sea respond in a similar manner to changes in prey. Wintering areas of puffins from different regions of around the UK and Ireland may overlap, therefore similar foraging conditions may be experienced across colonies, in-turn influencing productivity, phenology, and adult survival (Harris et al., 2010; Jessopp et al., 2013; Morley et al., 2016). Given the sensitivity of puffins at British colonies to warmer SST, this species seems particularly vulnerable to future climate change impacts within the region.

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[*]*Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.
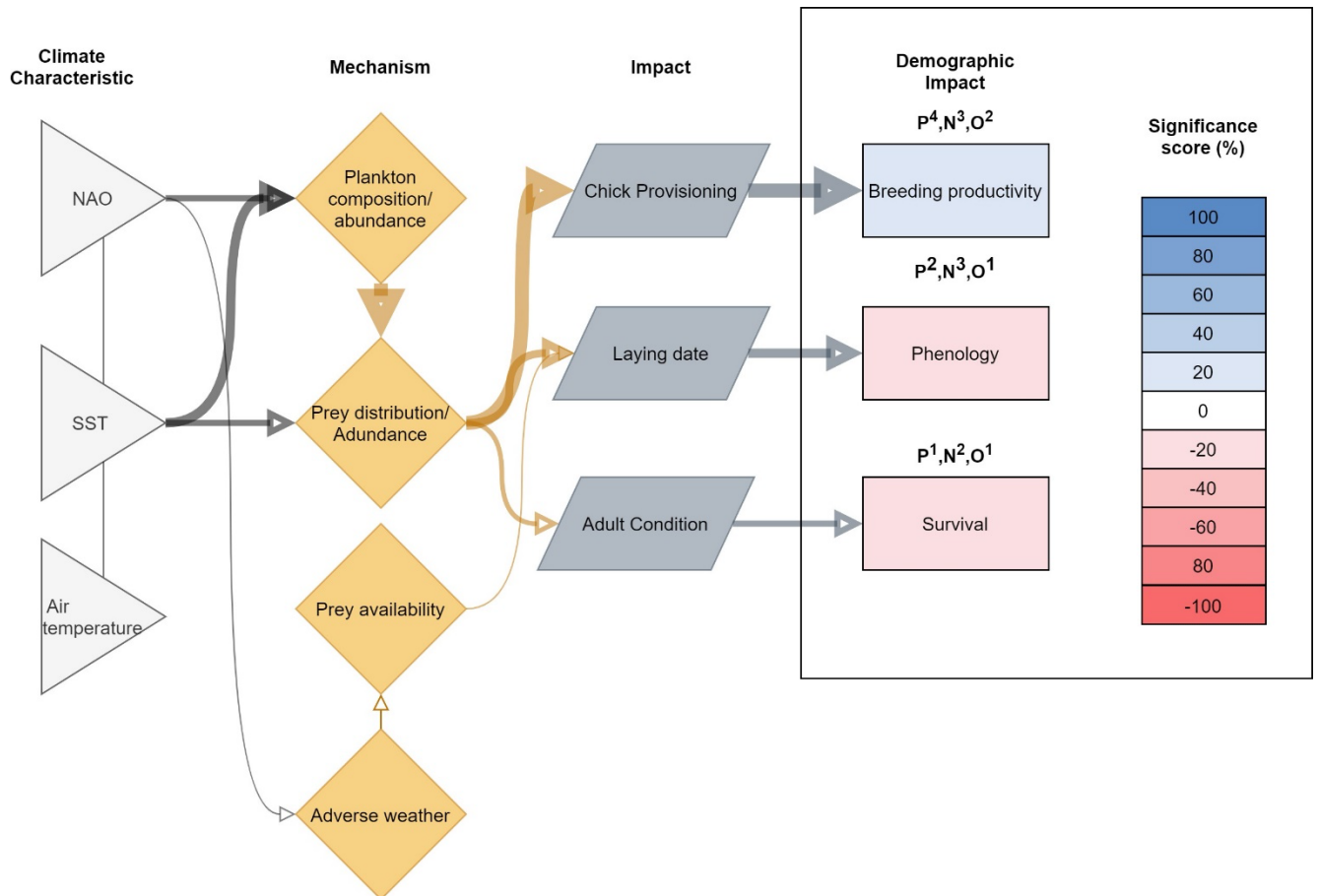
Figure 9. Puffin related mechanism flow chart. Lines sized by number respective of studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O= number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.9 Procellariiforms

3.9.1 Breeding populations

Procellariiforms which breed within the study region include Leach's storm petrels *Oceanodroma leucorhoa*, European storm petrels *Hydrobates pelagicus*, Manx shearwaters *Puffinus puffinus*, and Northern fulmar *Fulmarus glacialis* (hereafter referred to as fulmar). Due their similar pelagic/surface seizing foraging behaviour and broadly similar breeding behaviour, they have been considered as a group within this review.

Within the North Atlantic, maximum estimated populations in breeding pairs are: 5,000,000, Leach's storm petrels; 680,000, European storm petrels; 410,000, Manx shearwaters; and 4,100,000, fulmars. In the UK and Ireland, the maximum number of breeding pairs are: 65,310, Leach's storm petrels; 162,000, European storm petrels; 371,000, Manx

shearwaters; and 539,000, fulmars. The UK and Ireland are an area of international importance for Manx shearwaters, supporting 80% of the global population (JNCC, 2020a). Fulmar populations have been recorded to have declined within the UK and Ireland by 36% between 2000 and 2018. No data is available for the other 3 species which are burrow nesters and therefore harder to survey (JNCC, 2020a).

Crevice and burrow nesting storm petrels and shearwaters are found at a few select, inaccessible colonies around the coast. Indeed, specific colonies may hold large proportions of species populations within the UK and Ireland, with St. Kilda in particular holding <90% of breeding Leach's storm petrels, and the Isle of Rum and the island of Skomer holding <60% of breeding Manx shearwaters (Mitchell et al., 2004a). However, the primarily cliff nesting fulmars can be found around the coasts of the UK and Ireland.

The UK fulmar population experienced an increase in abundance and range during the 20th century which primarily related to increased availability of offal from whaling and fisheries discards (Phillips et al., 1999). In recent decades, the fulmar breeding population has experienced declines within the UK (JNCC, 2020u). Colony specific fluctuations in the population have been related to NAO with a 5 year lag (Thompson and Ollason, 2001). Additional procellariiform range shifts have been associated to climate variables, as found in range expansions in Balearic shearwaters, related to shifts in SST and prey distribution (Votier et al., 2008a; Luczak et al., 2011).

The four procellariiform species are primarily pelagic surface foragers, however shearwaters are capable of extended deeper dives to a maximum 55m (Shoji et al., 2016; Mitchell et al., 2020). The procellariform diet includes: fish, squid, zooplankton, crustaceans, molluscs, and fisheries discards (Thompson et al., 1995; Hedd and Montevecchi, 2006; BirdLife International, 2020k, 2020l, 2020m). While few studies exist on the potential climate driven shifts in diet of the four species, a study in the Northwest Atlantic found changes in SST anomalies, potentially drove a bottom-up decrease in the diversity of krill and amphipod species in Leach's storm petrel diet (Hedd et al., 2009).

3.9.2 Breeding productivity

The declines in the breeding population of fulmars in the UK and Ireland (JNCC, 2020u) have also coincided with declines in breeding productivity over time (Thompson and Ollason, 2001; Lewis et al., 2009). Breeding success was found to negatively correlate with higher NAO Index values, which have increased over time (Thompson and Ollason, 2001; Lewis et al., 2009). The mechanism for this decline is potentially related to changes in prey availability related to storm frequency. However, Lewis et al. (2009) found intrinsic inter-individual heterogeneity additionally influences the extent of the effect of NAO. Individuals which had low previous breeding success were shown to experience pronounced declines in breeding success with increased winter NAO, opposed to pairs with high previous breeding success which were less effected (Lewis et al., 2009). Regarding physiology of fulmars, NAO was found not to have a significant effect on egg size (Michel et al., 2003). Inter-individual

heterogeneity is further exemplified with egg size varying with female breeding experience, and body size which is also likely to be related to climate (Michel et al., 2003).

In Manx shearwaters, growth rates and peak masses of chicks have shown a degree of decline (1965-2008). Peak mass of chicks correlated significantly with the winter SSTs from the previous year and the mass of 2 year-old herring, but did not vary with winter NAO (Riou et al., 2011). In years of low prey availability, it was assumed that adults foraged further and with less frequent returns, indicating adults were searching for prey at large distances from the colony (Riou et al., 2011). Direct impacts through high precipitation during the incubation period of Manx shearwaters have been found to have a significant negative effect on breeding success through flooding (Thompson and Furness, 1991). Birds were found to have an adaptive response to flooding, being more likely to lay eggs in nests less susceptible to flooding, and may move to new burrows following failure (Thompson and Furness, 1991).

In comparison to fulmars and shearwaters, there is currently a lack of studies on the indirect effect of climate change on storm petrels and breeding success, forming a potential knowledge gap. Storm petrels may be indirectly affected by climate change through shifts in both native and invasive predators or species that may destroy nest sites. Both Leach's and European storm petrels are subject to several species of avian and mammalian predators (Phillips et al., 1997; Bicknell et al., 2009; Cadiou, 2013). Great skua *Stercorarius skua* may alter their levels of predation to seabird species including petrel species following fluctuations of fish prey stocks (Phillips et al., 1997). On St. Kilda in the North-East Atlantic, predation rates by great skuas were shown to be unsustainable if breeding adult Leach's storm petrels formed a substantial proportion of their diet (Votier et al., 2006). Additionally, predation of Leach's storm petrel eggs on St. Kilda by mice *Apodemus sylvaticus hirtensis*, was found, although having no clear population level effect (Bicknell et al., 2009). However, combined mortality of adults by avian predators and breeding failure due to mice could have negative impacts on storm petrel populations (Votier et al., 2006; Bicknell et al., 2009). In European storm petrels, inadvertent habitat destruction through erosion by introduced species, or the arrival of expanding species, such as great cormorants, could lead to reduction in breeding pairs through emigration or direct population decline (Cadiou et al., 2010).

3.9.3 Phenology

The breeding timings of Procellariiforms resident within the UK and Ireland vary in start date and period length. In the North Sea, fulmars were shown to have the least variable range in laying dates within and between years compared to other seabird species, and were one of the latest species to breed on the Isle of May (Wanless et al., 2009). No long-term trend in phenology was found in either the fulmars in the North Sea (Wanless et al., 2009), or Manx shearwaters in southwest Wales in the Celtic Sea (Riou et al., 2011). However timing of breeding in Manx shearwaters may influence fledgling success, as chicks which fledged

earlier were heavier than their later counterparts (Riou et al., 2011), potentially related to the timing of peak prey abundance dictated by SST. In contrast, European storm petrels display large variation in laying dates, potentially related to oceanographic drivers to plankton during the pre-breeding period, however no direct link was explored (Cadiou, 2001).

3.9.4 Survival

The breeding procellariiforms of the UK and Ireland undertake winter migrations of varying extent, with Leach's storm petrels and Manx shearwaters making distant migratory journeys to the South Atlantic (Guilford et al., 2009; Pollet et al., 2014). These expansive ranges may expose birds to a range of climate impacts additional to those experienced within the North Atlantic. Off the coast of Brazil, migratory Manx shearwaters were found to have increased mortality with the positive El Nino Southern Oscillation (ENSO) phase, which is associated with increased wave height (Tavares et al., 2019). These storm conditions may potentially reduce foraging ability, or result in exhaustion or injury from storm swell and winds. However, due to the high cost of flapping flight in shearwaters and fulmars, high winds have been found to have a positive relationship with energetic costs. As found in fulmars, high winds reduced energetic costs and increased range during the breeding season (Furness and Bryant, 1996; Spivey et al., 2014). Additionally, in combination with the lunar New Moon periods, high onshore winds were found to impact increased incidences of groundings, due to increased susceptibility to artificial light in adverse conditions (Syposz et al., 2018).

Survival rates in Orkney-based fulmars correlated with climate, and were found to have a negative relationship with winter NAO (Grosbois and Thompson, 2005). While female survival significantly declined in relation to the NAO Index, overall combined survival of both males and females decreased over the study period (1962-1995) suggesting that there had been a degradation of environmental conditions driven by the NAO (Grosbois and Thompson, 2005; Cordes et al., 2015).

In European storm petrels, spring/winter NAO did not have an overall effect on survival at a breeding colony within the Bay of Biscay (Zuberogoitia et al., 2016). However, recapture rates varied between seasons, with spring NAO positively correlating with the probability of recapture, while winter NAO had negative correlations (Zuberogoitia et al., 2016). In the Mediterranean, the probability of European storm petrels returning to breed has been found to relate to climatic conditions, with cold winters increasing the occurrence of skipped breeding (Soldatini et al., 2016). However, Soldatini et al (2014) reported that high pre-breeding SSTs and low levels of chlorophyll may increase skipping propensity, potentially due to reduced prey availability. Precipitation may also influence European storm petrel survival, as reductions in rainfall related to warmer and dryer conditions within the Mediterranean have been correlated with increased juvenile survival. Therefore, the

mechanisms driving survival in European storm petrels are potentially complex, and remain to be investigated thoroughly within the study region.

3.9.5 Future projections

Leach's storm petrel is projected to decline in total range extent in Europe over the 21st century: by 4-16% (Huntley et al., 2007; Russell et al., 2015). The future distribution in Europe is projected to overlap little or not at all with the current distribution: by 0-4% (Russell et al., 2015) to 11% (Huntley et al., 2007). Leach's storm petrel is not projected to remain present in the UK and Ireland in the late 21st century (Huntley et al., 2007). Within the UK, climate change is projected to present Leach's storm petrel with both high risk and high benefit ([*]Ausden et al., unpubl.), reflecting a high likely turnover in suitability at individual colonies.

The European storm petrel is projected to increase in total range extent by 1-24% in Europe over the 21st century (Huntley et al., 2007; Russell et al., 2015). The future distribution in Europe is projected to overlap little with the current distribution: by 10% (Huntley et al., 2007) to 15-17% (Russell et al., 2015). Much of European storm petrel's current range in the Mediterranean is predicted to become climatically unsuitable (Huntley et al., 2007). In the UK, climate change is projected to present storm petrel with high benefit ([*]Ausden et al., unpubl.). A range of more generic risk assessments for storm petrel in Britain span the full range of potential risk levels (Wheatley et al., 2017) or project high benefit (Pearce-Higgins et al., 2017).

Future projected changes in Manx shearwater total range extent in Europe over the 21st century vary in direction: from a 5% decrease (Huntley et al., 2007) to a 68-69% increase (Russell et al., 2015). The future distribution in Europe is projected to overlap little with the current distribution: by 24% (Huntley et al., 2007) to 2[†]8-29% (Russell et al., 2015). In the UK, climate change is projected to present Manx shearwater with high benefit ([*]Ausden et al., unpubl., Pearce-Higgins et al., 2017). A range of more generic risk assessments for Manx shearwater in Britain span the full range of potential risk levels (Wheatley et al., 2017) or project high benefit (Pearce-Higgins et al., 2017).

Future projected changes in fulmar total range extent in Europe over the 21st century vary: from a 19% decrease (Huntley et al., 2007) to a 7-41% increase (Russell et al., 2015). The future distribution in Europe is projected to overlap only partly with the current distribution: by 56-69% (Russell et al., 2015) to 72% (Huntley et al., 2007). In the UK, fulmar has been regarded as being at high risk ([*]Ausden et al., unpubl.) with a projected 87.6% decline in abundance on UK SPAs (Johnston et al., 2013). A range of more generic risk assessments for

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[*] *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.

UK tend towards medium to low risk (Wheatley et al., 2017) or a balance of potential risks and benefits (Pearce-Higgins et al., 2017).

3.9.6 INTERREG VA region summary

The study region holds large proportions of the UK and Ireland's populations of Manx shearwaters, Leach's storm petrels, and European storm petrels (JNCC, 2020a). Within the region, European storm petrel colonies are distributed on the West of Scotland and the North-West of Ireland. The majority of breeding Leach's storm petrels are found within the study region on St. Kilda and the Flannan Isles (Balmer et al., 2013). Manx shearwaters breed on select islands throughout the West of Scotland and Northern Ireland, while fulmars are found throughout the region (Balmer et al., 2013). Due to the logistical challenge of surveying burrow/crevice nesting species (Arneill, 2018), there is minimal information available on the population trends of the storm petrel species and Manx shearwaters within the region. However, there is some indication that Manx shearwaters have increased on the Treshnish Isles (JNCC, 2020v). Cliff nesting fulmars, have been found to be declining at colonies throughout the Irish Sea and West of Scotland (JNCC, 2020u). While few studies have been capable of examining trends in European and Leach's storm petrel breeding abundance or productivity, it is apparent that they may be susceptible to top-down impacts from avian (Votier et al., 2006), and mammalian predation (Bicknell et al., 2009). Burrow and crevice nesting procellariiforms may be exposed to direct impacts of nest flooding linked to high precipitation, which may increase with storm frequency (Thompson and Furness, 1991). Though few studies exist, much of the research on Manx shearwaters and storm petrels, during the breeding season, is directly applicable to breeding colonies within the study region (Thompson and Furness, 1991; Votier et al., 2006; Bicknell et al., 2009). Due to the wide ranging nature of wintering fulmars and Manx shearwaters, research undertaken outside the region, which links winter conditions to breeding abundance and productivity (Thompson and Ollason, 2001; Lewis et al., 2009) is potentially applicable to colonies in Irish Sea/ West of Scotland. Therefore, climatic impacts related to high NAO and SST (Thompson and Ollason, 2001; Riou et al., 2011) on foraging efficiency and prey availability are potentially applicable to the study region. Given all of these factors and the wide-ranging behaviour of these species, both during the breeding season and as part of their annual life-cycle, their future vulnerability to climate change is particularly uncertain.
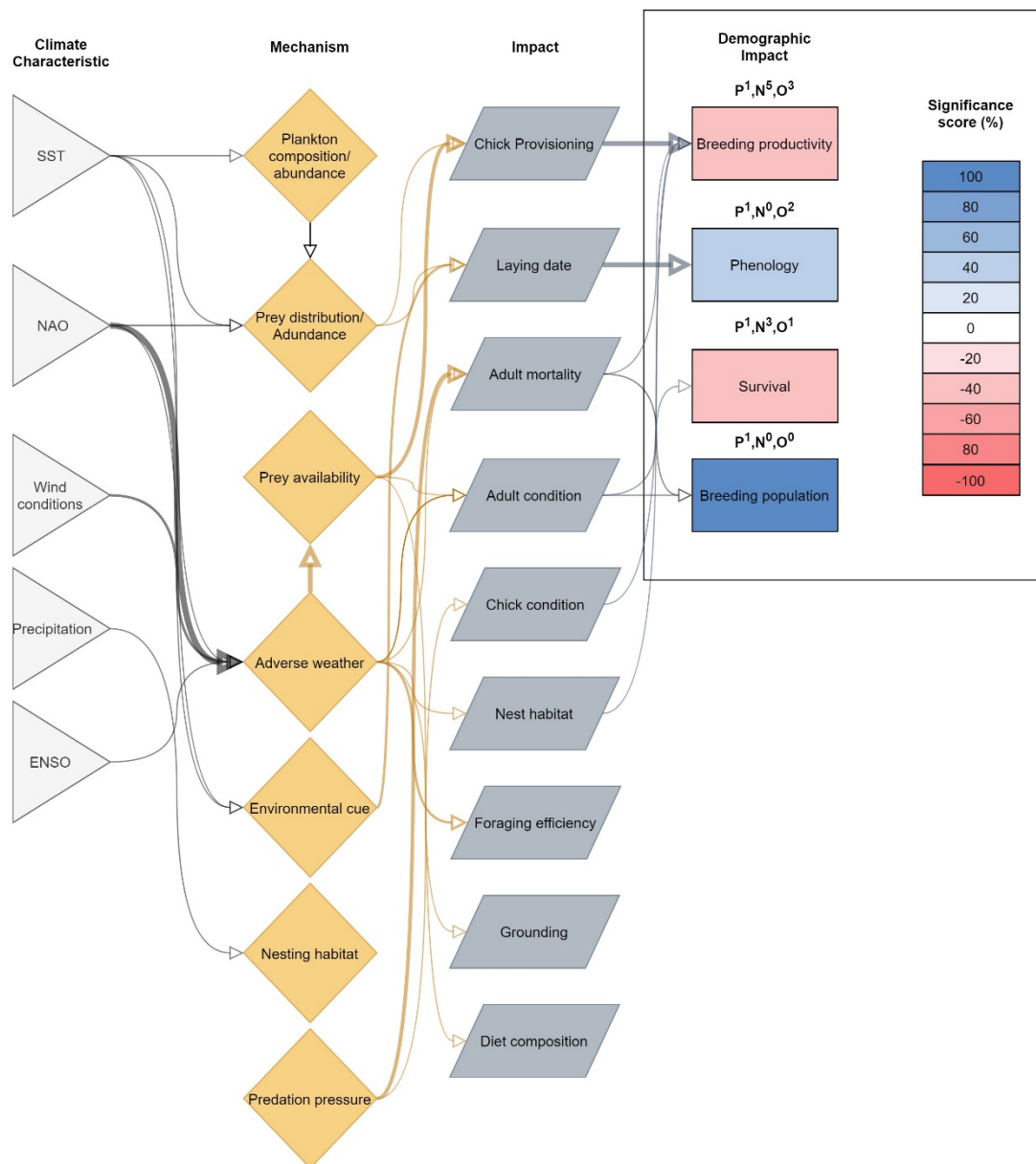
Figure 10. Grouped Procellariiform species related mechanism flow chart. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature; ENSO = El Nino Southern Oscillation. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = indicates the number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.10 Northern gannet *Morus bassanus*

3.10.1 Breeding population

Northern gannets (hereafter referred to as gannets) are found throughout the North Atlantic, breeding within Europe from the north of France, to Iceland and Norway (BirdLife International, 2020n). They were estimated to have a global population of 390,000 breeding

pairs at the turn of the century (Mitchell et al., 2004a). Within the UK and Ireland their population was recently recorded at 293,161 (2013-2015) apparently occupied sites, with breeding abundance steadily increasing from 1986-2018, most recently recorded as increasing from 34% from 2003-2004 to 2013-2015 (JNCC, 2020r). The UK and Ireland are an area of international importance for gannets, supporting 56% of the global population (Mitchell et al., 2004a; JNCC, 2020a)

Gannets have been suggested to largely be buffered from climate change. This is attributed to their opportunistic foraging ability, a wide range of prey types, diving and surface seizing foraging behaviour, and ability to forage long-distances from the colony (Hamer et al., 2001). Colonisation and increasing populations in gannets within the North-East and North-West Atlantic have correlated closely with increases in mackerel *Scomber scombrus* landings by fisheries and long-term trend of warming SST (Montevecchi, 1997; Barrett et al., 2017b). However, they may be able to switch to cold water prey during temperature anomalies (Montevecchi, 1997). Foraging behaviour may be buffered from the effects of adverse weather, and gannets may be able to take advantage of strong winds by modulating flight height (Lane et al., 2019).

3.10.2 Breeding productivity
Breeding productivity of gannets has remained stable between 1986-2018, with consistent mean productivity at 0.69 chicks per nest (JNCC, 2020a). Long term (1986-1996) trends of breeding success of gannets in the UK did not correlate with winter NAO Index (Reid et al., 1999). However extreme temperature anomalies have been shown to negatively affect gannets elsewhere. Anomalously high SSTs during the breeding season within the Gulf of St. Lawrence, Northwest Atlantic, were found to reduce prey availability and accessibility, due to the formation of thermocline and the thermal physiology of fish prey. This reduction in prey negatively affected adult condition, shortened the chick rearing period (Franci et al., 2015), and reduced breeding success (Montevecchi et al., 2013).

Gannets can exploit both warm and cold-water prey species, and, outside temperature extremes, are generally well suited to temperature fluctuations between years. At colonies in Newfoundland, within the Northwest Atlantic, gannets foraged on capelin *Mallotus villosus* in colder years, and alternated to mackerel in warmer years, when capelin may be negatively affected by higher SSTs (Davoren et al., 2012).

3.10.3 Phenology
Gannets have a relatively long incubating and chick rearing period of around 4.5 months, and northward colonisation by gannets may be potentially constrained by the length of the Arctic summer (Barrett et al., 2017a). The latitudinal variation in the timing of spring has also been correlated with later breeding timing in Northerly colonies (Wanless et al., 2008). Over the period 1980-2007, the onset of breeding was found to become later across multiple colonies within the North-East Atlantic (Wanless et al., 2008). Latitude was also shown to be the primary factor influencing timing of breeding, with a stronger effect than

SST. Additionally, winter SST and NAO anomalies were unrelated to hatch date. Therefore, photoperiod was suggested to be more influential on the timing of breeding, though the trend in phenology could not be explained. Later breeding is generally associated with reduced breeding success and adult condition, and a trend in later hatching dates may pose long term negative effects on productivity and survival.

3.10.4 Survival

No information is available from the SMP on gannet survival (JNCC, 2020r), additionally no studies attributing survival to climate change were found.

3.10.5 Future Projections

Gannet is projected to decline by 6-16% in total range extent in Europe over the 21st century (Huntley et al., 2007; Russell et al., 2015). The future distribution in Europe is projected to overlap only partly with the current distribution: by 17% (Huntley et al., 2007) to 29-41% (Russell et al., 2015). A north-west shift in range is projected, with the loss of sites in France and gain of sites in Iceland (Huntley et al., 2007). In the North Sea and British continental shelf, gannet population size is predicted to change relatively little by the middle of the 21st century; by -5.7 to +2.3% (Sadykova et al., 2020). Future projected changes in gannet range and abundance within the UK vary: from a 6.3% decline in abundance on SPAs (Johnston et al., 2013), to projections of moderate risk with very high benefit ([*]Ausden et al., unpubl.) and high benefit (Pearce-Higgins et al., 2017). A range of more generic risk assessments for gannet in Britain span the full range of potential risk levels (Wheatley et al., 2017) or project high benefit (Pearce-Higgins et al., 2017).

3.10.6 INTERREG VA region summary

Several gannet breeding sites on the West of Scotland are encompassed within the study region (Balmer et al., 2013). However, no gannetries are present in Northern Ireland or the counties of Ireland held within the study region. The majority of these gannetries displayed increases in breeding abundance in surveys carried out 2013-2015 (JNCC, 2020r). Few diet studies exist for gannets foraging within the study region. Other studies primarily from the North Sea indicate primary prey likely to be consumed include: mackerel, sandeels, sprat, clupeids, and gadoids (Hamer et al., 2000; Lewis et al., 2003; Votier et al., 2010). The incorporation of mid-trophic level fish in the diet of gannets may buffer prey resources from direct bottom-up impacts of environmental variation. However, diet may vary between colonies (Lewis et al., 2003), and therefore, depending on the diet composition, colonies may vary in their response to climate change. Given current trends and their biology, they appear to be at relatively low risk from climate change in the region.

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[*] *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.
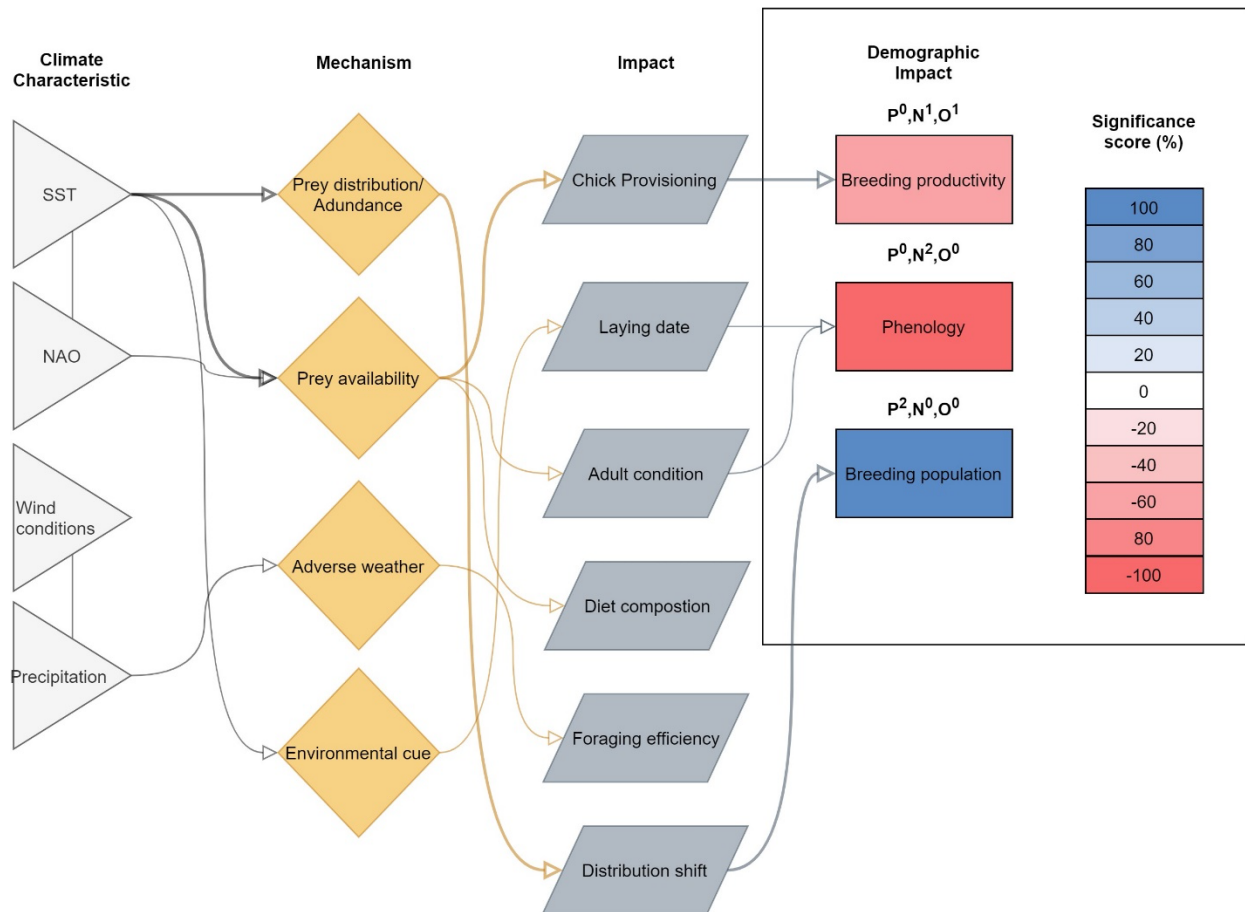
Figure 11. Gannet related mechanism flow chart. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.11 Shags Phalacrocorax artistotelis

3.11.1 Breeding population

Within the North-East Atlantic, European shags (hereafter referred to as shags) *Phalacrocorax artistotelis* range from the Mediterranean to northern Norway and Iceland (BirdLife International, 2020o; JNCC, 2020w). Within this range they have an estimated population of 73,000 breeding pairs (Mitchell et al., 2004a). Of these, 32,306 pairs were surveyed in the UK and Ireland between 1998-2002, displaying an overall decline of -25% from the previous 1985-1988 survey (Mitchell et al., 2004a), declining a further -24% between 2000-2018 (JNCC, 2020a). However, within Ireland shags have displayed an increase of 45% between 1998-2002 and 2015-2018 surveys (JNCC, 2020w). Declines have also been noted at colonies in Spain (Velando and Freire, 2002), which contrasts to the increasing Norwegian population (Bustnes et al., 2013). This pattern of declining southerly populations, contrasting with increases in the north, is consistent with expected climate

driven trends. Prey composition and availability is potentially driving these trends, as demographic parameters of breeding productivity, phenology, and survival have been linked to herring and sandeel abundance and quality in the North Sea (Aebischer and Wanless, 1992). Shags are generalist predators and therefore the importance of specific prey types varies between colonies in relation to availability. In the Norwegian sea, changes in the abundance of more southerly breeding pairs correlated with the abundance of 1 year old saithe *Pollachius virens*, while Barents sea colonies correlated with capelin *Mallotus villosus* abundance (Bustnes et al., 2013).

However, in relation to shags breeding on the Isle of May, sandeels within North Sea have been found to be of decreasing quality (Burthe et al., 2012) and importance (Wanless et al., 2018) between 1986-2015. Benthic species, such as Pholidae, also comprise a large proportion of the diet (Wanless et al., 2018). The proportion of benthic species in comparison to sandeels was shown to vary with SST (Howells et al., 2017). Howells et al. (2017) found the proportion of 1+ age group relative to 0-group sandeels in shag chick diet to be greater following a period of low SST, whereas prey richness was higher following warmer SST in the previous year. Similarly, Fortin et al. (2013) found number of breeding pairs significantly decreased with increases in SST, but also sea surface height, suggesting altered prey availability may influence this species on a colony level.

3.11.2 Breeding productivity

Within the UK there has been no definitive trend in the breeding productivity of shags (JNCC, 2020w), however productivity may be influenced directly and indirectly through weather conditions. Indirectly, breeding productivity has been closely tied to prey abundance, specifically sandeels (Rindorf et al., 2000; Frederiksen et al., 2006, 2007b; Burthe et al., 2012) and herring (Aebischer and Wanless, 1992). Within Scotland a gradual decline in productivity has been observed (JNCC, 2020w). On the Isle of May, high SSTs have been shown to negatively influence breeding success of shags, potentially due to an influence on prey stocks (Burthe et al., 2012). Adverse conditions from increased wave height, precipitation and wind speeds associated with gales, may alter prey availability (Lewis et al., 2015). This was evidenced by Howells et al. (2017), with the reduction of the proportion of sandeels during high winds in North Sea shag diet. Inefficient foraging conditions arising from adverse weather are potentially related impacts on shag flight, or reduced visibility in the water column, as the burrowing of sandeels during periods of low light reduces prey availability (Daunt et al., 2006). Therefore, chick mortality, first year survival and breeding success have each been correlated negatively with high precipitation and wind (Aebischer and Wanless, 1992; Velando et al., 1999; Frederiksen et al., 2004a). Velando et al. (1999), found larger broods lost more chicks, suggesting adverse weather correlates with increased food demands, and a higher feeding rate imposed on adults.

Regarding broader climatic proxies, winter NAO, a potential proxy for storminess, has been found to have no influence on breeding success in shags in the Barents Sea, France, and

Spain (Bustnes et al., 2013; Fortin et al., 2013; Barros et al., 2014). However, Fortin et al. (2013) found number of breeding pairs correlated negatively with increased sea surface height. Additionally, accounts of storms flooding and damaging nests detail the process of direct impacts on breeding pair numbers and nest success (Aebischer, 1993; Velando and Freire, 2003; Newell et al., 2015).

Adverse conditions also have an unexpected significant effect on brood sex ratio and dispersal behaviour. Years with positive values correlated with increased proportions of female chicks, and NAO negative years correlated with increased proportion of males. This may be due to sexual dimorphism, leading to increased demand for parental investment in male chicks during years of adverse conditions (Barros et al., 2014).

3.11.3 Phenology

Shags display relatively large annual and inter-individual variation in laying dates in comparison to other species of seabird (Wanless et al., 2009; Burthe et al., 2012). Trends in the timing of breeding vary between colonies, with advancement in laying on the Isle of May and Northern Spain (Wanless et al., 2009; Álvarez and Pajuelo, 2011), and no trend on the Farne Islands (Wanless et al., 2009). Climatic conditions may influence breeding timing, particularly through indirect effects on prey availability (Aebischer, 1986; Lorentsen et al., 2015), and foraging efficiency, as the amount of time spent foraging during the pre-breeding period is closely associated with phenology (Daunt, 2006). Reduced prey availability during the pre-breeding season was assumed to be correlated with later breeding on the Isle of May, potentially driven by number of days with strong easterly winds in March (Aebischer and Wanless, 1992). On the Isle of May, shag phenology has responded differently to climatic variables in comparison to other seabird species (Frederiksen et al., 2004a). Other species altered their phenology in relation to the winter NAO index, with variability in phenology primarily associating with SSTs. It was speculated that shags adjust laying date in relation to the appearance of sandeels on a local scale (Frederiksen et al., 2004a). However, winter NAO may influence phenology when closely tied with prey availability, as was found in Norway at both Sklinna and Rost. At these colonies, hatching date was primarily influenced by the abundance of 1-year old saithe which was closely related to winter NAO (Lorentsen et al., 2015).

The advancement in laying date in Northern Spain correlated with the Temperature Anomaly Index (TAI), indicative of local conditions, while SST and NAO had no influence. While SST also reflects local conditions, the demersal diet exhibited by shags in this area potentially means they are more influenced by local fronts and upwellings rather than large scale oceanographic processes (Álvarez and Pajuelo, 2011). Wide ranging seabird species, that migrate to pelagic areas, may experience large scale oceanographic conditions and anticipate breeding season conditions. However, the timing of breeding in partially-migratory, coastal resident, shags (Grist et al., 2017), may predominantly be related to the influence of local environmental conditions on prey stocks.

3.11.4 Survival

Long term studies of shag return rates on the Isle of May have seen no definitive trend between 1987-2018 (JNCC, 2020w). Indirectly, in the Norwegian Sea climate has been linked to survival through positive 1-year lagged winter NAO, which correlated negatively with the mortality of non-breeding individuals in their first year. This was potentially caused by reduced recruitment of saithe (Bustnes et al., 2013). Direct impacts from gales are potentially attributed to the partial waterproof plumage of shags and cormorants, leaving individuals vulnerable to hypothermia following periods of extreme precipitation (Grémillet et al., 1998). Through this mechanism, high precipitation and onshore winds in winter have been found to reduce immature and adult survival in shags (Frederiksen et al., 2008; Bustnes et al., 2013).

3.11.5 Future projections

Future projected changes in shag total range extent in Europe over the 21st century vary in direction: from a 16% decrease (Huntley et al., 2007) to a 34-36% increase (Russell et al., 2015). The future distribution in Europe is projected to overlap to some extent with the current distribution: by 44% (Huntley et al., 2007) to 75% (Russell et al., 2015). Much of the current Mediterranean range is projected to become climatically unsuitable (Huntley et al., 2007). Future projected changes in shag range and abundance within the UK vary: from an 1.6% decline in abundance on SPAs (Johnston et al., 2013), to projections of very high risk ([*]Ausden et al., unpubl.). A range of more generic risk assessments for shags in Britain project medium to low risk (Wheatley et al., 2017) or project a combination of risks and benefits (Pearce-Higgins et al., 2017).

3.11.6 INTERREG VA region summary

Shags breed throughout the study region (Balmer et al., 2013). While variation in population trends exists within this region, the population has primarily exhibited declines at colonies on the West of Scotland and Irish Sea (JNCC, 2020w). Although few diet studies exist for the region, on the Isle of Canna diet was dominated by sandeels (Swann et al., 1991, 2008), a predominant prey in the North Sea (Howells et al., 2018). While shags may supplement their diet with a range of prey species, these alternate resources may not support previously recorded breeding abundance. Therefore, demographic trends within the region may respond in a similar manner to environmental drivers recorded on the intensively studied Isle of May. Shags are susceptible to the effects of high precipitation and winds on both physiology, and nest success. The degree to which colonies are affected by adverse conditions may be defined by storm direction and strength. Therefore, the greater degree of exposure too (Baxter et al., 2011; Masselink et al., 2016), and frequency of (McClatchey, 2014), storms across the west of Ireland, Northern Ireland and Scotland may vary the

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[*] *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.

impacts experienced by wintering and breeding shags within the study region in comparison to the North Sea.
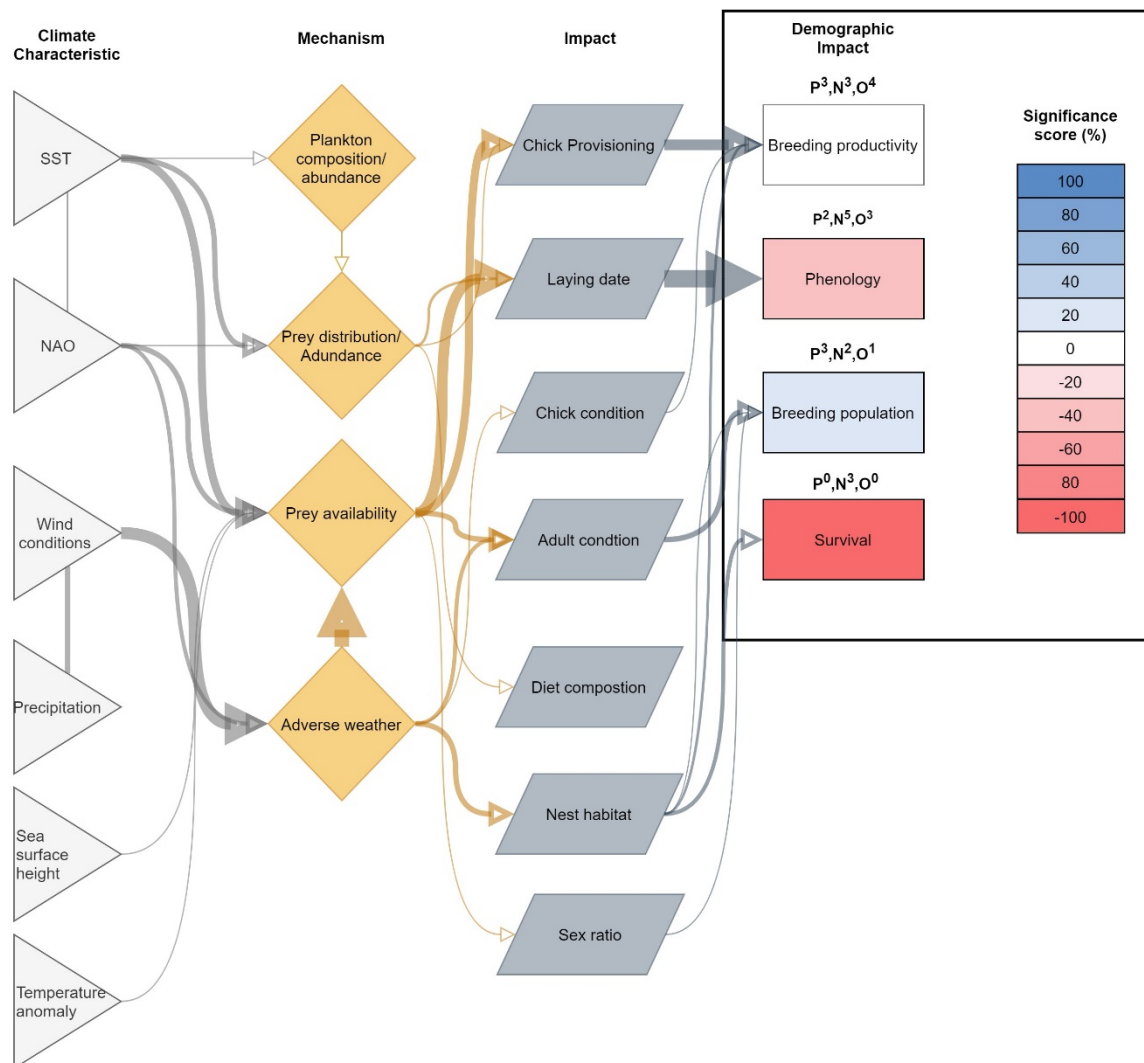
Figure 12. Shag related mechanism flow chart. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature. P = number of studies where significant positive effect found; N= number of studies where significant negative effect found; O= number of studies where non-significant effect was found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

3.12 Great cormorants *Phalacrocorax carbo*

3.12.1 Breeding population

Great cormorants *Phalacrocorax carbo* (hereafter referred to as cormorants) maintain a global distribution, with a range within the North-East Atlantic extending from the Bay of Biscay to the Arctic. Two sub-species of great cormorant breed within the UK and Ireland,

including: the coastal associated *P. c. carbo*; and the inland dwelling *P. c. sinensis.* Coastal populations (likely *P. c. carbo*) within the UK and Ireland have broadly remained stable (11,560 coastal, AON, Mitchell et al., 2004). Though their population has shown variable trends, displaying increases from 1986-1988 to 1998-2002 (+10), and a decrease of -7% from 2000 to 2018 (JNCC, 2020a). The most notable population change between censuses involved the inland colonisation by *P. c. sinensis* from continental Europe (Mitchell et al., 2004a; JNCC, 2020x). In general, studies of the mechanisms driving cormorant demography are fewer than those conducted for shags. This may be due to the feasibility of studying shags, in comparison to the relatively shy behaviour of cormorants when breeding (Gardarsson and Jónsson, 2019); and the identification of shags as a viable indicator of marine fish stocks (Lorentsen et al., 2015).

The majority of studies of cormorants in relation to climate change investigate the drivers of their range shifts. Population trends of *P. c. carbo* in Iceland in 1996–2015 were found to correlate positively with the Subpolar Gyre (SPG) (Gardarsson and Jónsson, 2019). This relationship is assumed to stem from enhanced prey abundance and milder conditions associated with negative SPG values. However, it was noted that the population is reaching carrying capacity in relation to availability of nest sites, and philopatry/dispersal distance. This is also reflected in studies of cormorants in Greenland, which have displayed a positive relationship between population and winter SSTs (White et al., 2008). However, it was highlighted that birds may hit an energetic threshold between range expansion into the Arctic, and winter migration distances to areas outside of the Arctic night (White et al., 2013). This increase in *P. c. sinensis* migrants has seen an increase in the winter population in France (Marion and Bergerot, 2018). Investigating the effect of climatic conditions on range expansion, Marion and Bergerot (2018) found winter air temperature had little effect on the northward shifts. However, sub-optimal roosting areas further north, during cold periods, were potentially used more frequently due to competition for roost sites driven by an increase in wintering population in France (Marion and Bergerot, 2018).

3.12.2 Breeding productivity and phenology

Of the few studies of breeding success and phenology, low SSTs and icing of local lagoon habitat were found to delay *P. c. sinensis* breeding within the Baltic Sea adjoining Poland (Stempniewicz et al., 2000). In reflection of the relationship between foraging efficiency and adverse weather displayed in shags, Stempniewicz et al. (2000) noted high winds to negatively affect breeding success. Additionally, delayed breeding was suggested to cause a degree of synchrony which may have reduced breeding success due to competition between pairs (Stempniewicz et al., 2000). No information is available from the SMP on cormorant phenology within the UK (JNCC, 2020x).

3.12.3 Survival

No information is available from the SMP on cormorant survival (JNCC, 2020x), additionally no studies attributing survival to climate change were found.

3.12.4 Future projections

Future projected changes in cormorant total range extent in Europe over the 21st century vary: from a 14% decrease (Huntley et al., 2007) to a 28% increase (Russell et al., 2015). The future distribution in Europe is projected to overlap little with the current distribution: by 27% (Huntley et al., 2007) to 43-47% (Russell et al., 2015). The European distribution is projected to move north-eastwards (Huntley et al., 2007). Future projected changes in cormorant range and abundance within the UK vary: from a 0.9% decline in abundance on SPAs (Johnston et al., 2013) to a projection of very high benefit (*Ausden et al., unpubl.). A range of more generic risk assessments for cormorant in Britain tend towards medium to low risk (Wheatley et al., 2017) or project high benefit (Pearce-Higgins et al., 2017).

3.11.6 INTERREG VA region summary

Cormorants are distributed throughout the coasts of the study region, with populations being primarily coastal based (Balmer et al., 2013). In Ireland, Northern Ireland, and Scotland populations have displayed varying annual breeding abundances, with some declines (Booth Jones, 2020a; JNCC, 2020x). However, dispersal to other colonies could not be ruled out as a cause of variation in abundance (JNCC, 2020x). Very little information is available on cormorant diet within the region. However, their generalist diet (Grémillet, 1997), and perceived ability to colonise a range of habitats may provide resilience to future environmental change, supported by relatively low levels of vulnerability to climate change in published assessments. More exposed regions on the North-West of Scotland, Northern Ireland, and Ireland may be susceptible to disturbance from storm swell and high winds originating from the North Atlantic (Baxter et al., 2011; Masselink et al., 2016), than Irish Sea based colonies.

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* *Ausden et al., unpubl.* is an unpublished extension of Pearce-Higgins et al. (2017), conducting climate vulnerability assessments for additional species within the UK.
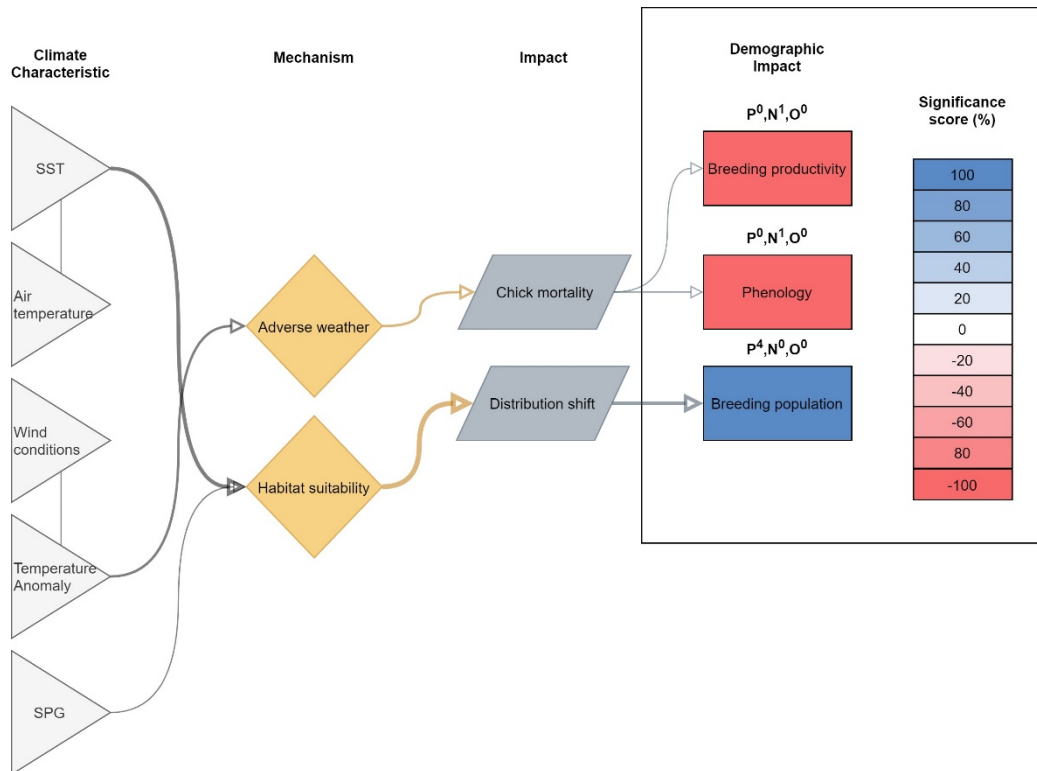
Figure 13. Cormorant related mechanism flow chart. Lines sized by number respective of studies. NAO = North Atlantic Oscillation; SST = Sea-Surface Temperature; SPG = Subpolar Gyre Index. P = number of studies where significant positive effect found; N = number of studies where significant negative effect found; O = number of studies where non-significant effect found. Demographic impacts are coloured according to the "significance score" indicating the cumulative direction of effect as evidenced through the available studies. Significance scores are calculated through the equation- significance score = P-N/(P+N+O).

4. Discussion

We have summarised the mechanisms of potential climate change impacts on 25 species of seabird which breed within the INTERREG VA area. Of the 156 references collated, from studies assessing the response of each species to environmental covariates across Europe and the UK and Ireland, only 13 studies were conducted within the study region. Due to this relative lack of research originating from within the region, the species accounts in this report included relevant studies from outside the region. These accounts are designed to provide an important source of evidence to support the development of management plans, covered by Technical Work Package 5 of the MarPAMM project. Moreover, they summarise the current state of knowledge with respect to the future climate change scenarios for seabird populations and underpin the next phase of the project (MarPAMM Lot 5 Task 2.1) which will involve new bespoke predictions being generated with respect to population size and distribution at sea. In this discussion we firstly highlight the general patterns in the mechanisms affecting different seabird species before secondly considering shortcomings in the available literature which may have implications for their interpretation with respect to the study region and for future work.
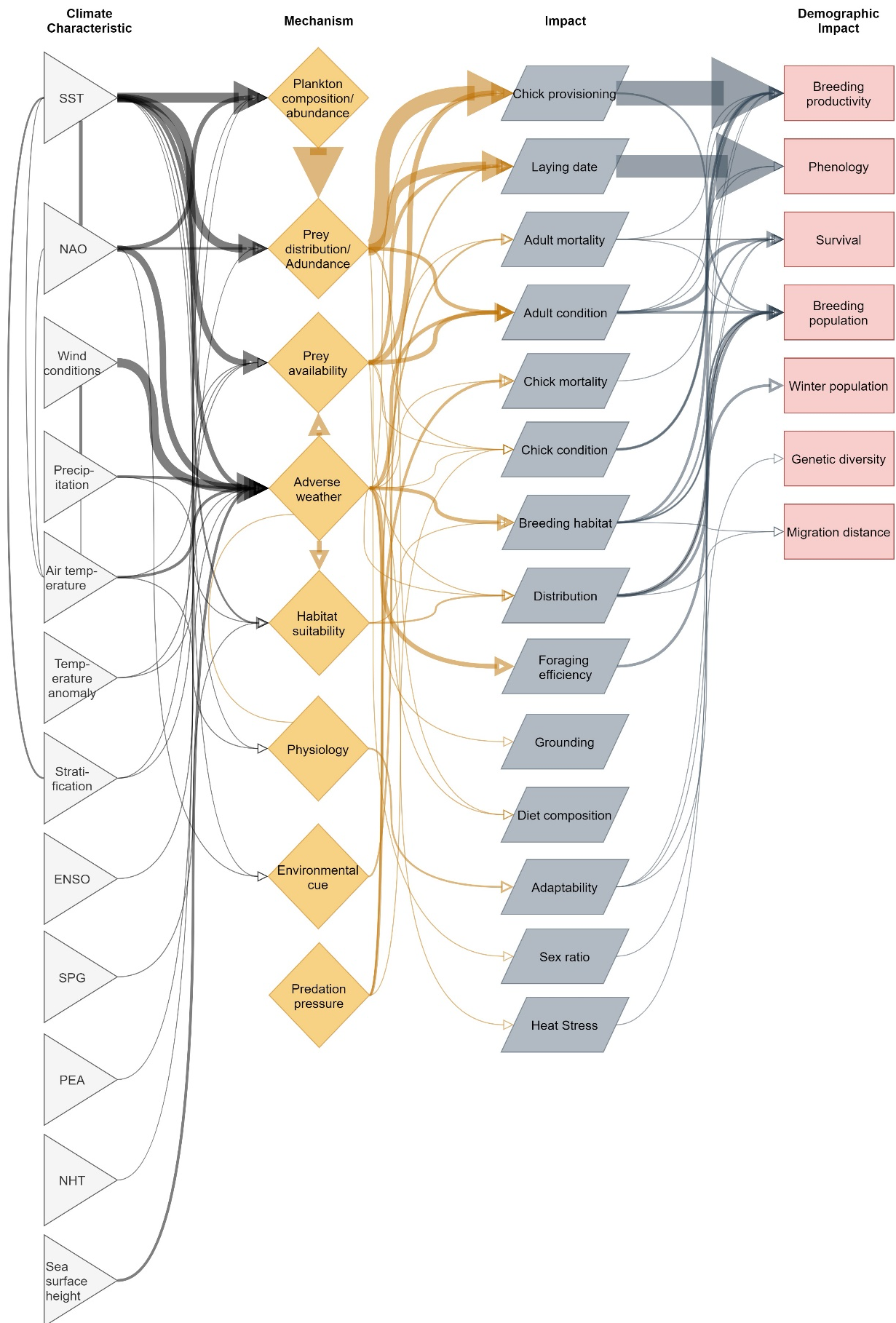
Figure 14. Mechanism flow chart encompassing all species. Lines sized by number of respective studies. NAO = North Atlantic Oscillation; SST = Sea Surface Temperature; PEA = Potential Energy Anomaly; Northern Hemisphere Temperature anomaly; ENSO= El Nino-Southern Oscillation;and SPG= Sub-polar Gyre.

4.1 General patterns

Analysis of literature linking climate variation to seabird demography highlighted several mechanisms acting both directly (24 studies) and indirectly (116 studies). Of the studies, 220 analysis were carried out within 156 publications. The majority of studies (110 papers) investigated indirect mechanisms through long-term (>10 years) monitoring programmes of breeding productivity (58 studies), breeding timing (32 studies), breeding abundance (24 studies), and adult survival (20 studies). Studies within the North-East Atlantic, based on these time-series of data, primarily related demographic parameters to climatic variables of SST and NAO. One, or both, of these variables were contained within 94 studies. These variables were often used to investigate indirect climate processes affecting seabirds, including prey abundance, through effects on the quality and quantity of prey, or through prey accessibility, altered by adverse weather influencing prey behaviour, or seabird foraging efficiency. Given the large scale that it represents, NAO was frequently cited as a proxy for at-sea winter conditions which may affect breeding timing, adult survival, or breeding productivity through pre-breeding adult condition (Wanless et al., 2007, 2009; Frederiksen et al., 2013). SST was primarily indicative of local oceanographic conditions (Álvarez and Pajuelo, 2011; Burthe et al., 2012) and phytoplankton habitat suitability (Frederiksen et al., 2013), affecting bottom-up prey abundance. This was most notable in the North Sea where stochastic shifts in SST, cascade through the food chain to reduce the abundance of high lipid plankton *C. finmarchicus,* and in turn reduce the abundance and size of sandeel prey (Frederiksen et al., 2013). Select studies directly investigated the influence of prey stocks, such as sandeels on seabirds (Frederiksen et al., 2005, 2006), with aspects of SST and NAO discussed to some degree.

4.1.1 Study lengths
Studies based on short-term periods were fewer (38 papers) and often reported events which related to catastrophic declines in breeding success. This included impacts of anomalous temperatures (air or sea-surface) indirectly affecting prey availability (Regehr and Rodway, 1999), or chilling eggs (Indykiewicz, 2015), and gales fouling incubating/nesting attempts (Thompson and Furness, 1991; Newell et al., 2015). High wind and precipitation are also examined in short-term studies in the context of foraging efficiency (Birkhead, 1976; Blake, 1984; Furness and Bryant, 1996; Finney et al., 1999).

4.1.2 Varying responses
Responses to climate variables can deviate between species, and between colonies (Grosbois et al., 2009). While overlap was often found in species responses to climate variables (Burthe et al., 2012; Lauria et al., 2013), the life-history traits of foraging behaviour (diving or surface seizing) or foraging range often determined species-specific responses. One notable deviation between two well studied species was between coastal associating

European shags (Grist et al., 2017), and long-ranging Northern fulmars (Edwards et al., 2013). Whereas fulmar productivity, survival, and phenology were significantly influenced by NAO (Thompson and Ollason, 2001; Grosbois and Thompson, 2005; Lewis et al., 2009; Wanless et al., 2009), shag phenology and diet composition was largely dictated by local SSTs (Frederiksen et al., 2004a; Wanless et al., 2009; Álvarez and Pajuelo, 2011; Howells et al., 2017). Additionally, survival and productivity of surface seizing species, such as kittiwakes, were often linked to variations in prey abundance (Frederiksen et al., 2006; Wanless et al., 2007); while diving common guillemots, which are able to forage throughout the water-column, were less affected by prey abundance and studies more frequently attributed negative effects of climate to reductions in prey availability from adverse weather (Votier et al., 2005, 2009; Munilla et al., 2007). Investigation of homogeneous or heterogeneous responses of seabird species to climate change is only possible through similar sample sizes. Gaps are prevalent for certain species; this bias is potentially due to a species' ease of study. This is highlighted in the contrast in number studies between shags (23 studies) and cormorants (6 studies) potentially due to cormorant's comparative shyness during breeding season. Additionally species which have been identified as good indicators of marine prey stocks such as shags (Lorentsen et al., 2015) and kittiwakes (Daunt et al., 2008) may also have had more dedicated research attributed to them.

4.1.3 Demographic responses

Across species, prey abundance was found to have an overriding influence on seabird demographics, and when included within analysis alongside NAO or SST variables, the latter were found to have no additional influence (Frederiksen et al., 2006; Álvarez and Pajuelo, 2011; Barros et al., 2014), thus highlighting the potential of additional oceanographic variables acting on prey, such as local upwellings (Álvarez and Pajuelo, 2011). The strongest evidence for a long-term mechanism linking climate change to seabird populations is therefore mediated through changes in prey species associated with changes in sea temperature. This is best evidenced by the multiple studies from the Isle of May within the sandeel dominated North Sea (e.g. Frederiksen et al., 2006). The same principles also apply to other seabird systems, whether puffins and herring (Durant et al., 2003), or guillemots and sprat (Österblom et al., 2006), and indeed across seabird systems globally from the Southern Ocean to the Pacific (Pearce-Higgins and Green, 2014). This means that understanding the impacts of climate change on the marine system from plankton to fish can play an important part in identifying the vulnerability of different seabird species to climate change, at least where the main prey species of different colonies are understood. The importance of diet is seen in variable kittiwake population trends between the east and west coast of the UK, with more positive trends in the west where sandeels are a smaller component of the diet and responses to changes in sea-surface temperature have been weaker (Frederiksen et al., 2005; Cook et al., 2014). Thus, in common with studies of climate change impacts on birds (Pearce-Higgins and Green, 2014), and on species more generally

(Ockendon et al., 2014), the main mechanisms appear to operate through altered species interactions, and particularly through reductions in prey populations.

Shorter-term and more catastrophic events such as storms can result in the complete loss of productivity in particular breeding seasons, or even be associated with large mortality 'wreck' events during the winter, affecting adult survival (Morley et al., 2016). Given the k-selected nature of seabirds, occasional breeding failures do not have a major impact on population trajectories providing productivity in the intervening years is high (Ricklefs, 1990). However, if climate change results in an increase in the frequency of severe storms during the breeding season, either affecting low-lying tern colonies or even some cliff-nesting species, then there is the potential for a run of bad breeding seasons to begin impacting colony size, or alternatively for some more mobile species such as terns to relocate colonies in response to such events. Occasional impacts of severe storm events, in both summer and winter, on seabird mortality are more likely to impact populations as this could result in significant and sudden drops in abundance, which if associated with an increase in frequency of such events, or a long-term reduction in productivity, could drive significant long-term decline (Robinson et al., 2014).

4.1.4 The INTERREG VA region

There is a current lack of available diet data for the North Atlantic and Irish Sea within the study region, compared to the North Sea (Frederiksen et al., 2007a; Wanless et al., 2018). Although few diet studies exist for the region, for certain species including shags, razorbills, and puffins breeding on Canna, off the West Coast of Scotland, the diet was dominated by sandeels (Swann et al., 1991, 2008), a predominant prey in the North Sea (Howells et al., 2018; Wanless et al., 2018). Therefore, demographic trends within the region may respond in a similar manner to environmental conditions recorded on the intensively studied Isle of May. For kittiwakes and guillemots, available knowledge suggests there is less reliance on sandeels in this region (Frederiksen et al., 2005; Anderson et al., 2014b), and a greater prevalence of clupeids (Chivers et al., 2012a), and gadoids (Harris and Wanless, 1990). Diet composition also varies within the study region, and sandeels may form an important component of diet in the West of Scotland, while clupeids are more prevalent within the Irish Sea (Chivers et al., 2012b; Anderson et al., 2014b). The presence of snake pipefish in the diet of individuals breeding within the region (Anderson et al., 2014b) may indicate species experiencing low abundance of preferential/higher quality prey (Harris et al., 2007). Dietary variation between colonies, or species, within the region may dictate a population's response to climate change.

Several species of seabird, particularly those which nest in low-lying coastal areas or cliffs, are susceptible to the effects of high precipitation and winds on nest success (Newell et al., 2015). The degree to which species nests are affected by these adverse conditions, may be defined by storm direction and strength. Coastal regions of North-West of Scotland, Northern Ireland, and Ireland may be more exposed to storm swell and high winds

originating from the North Atlantic, than coasts within the North and Irish Sea (Baxter et al., 2011; Masselink et al., 2016). With storms potentially increasing in frequency and strength in the North Atlantic (Lowe et al., 2001; McClatchey, 2014), available breeding habitat may decrease in exposed and frequently disturbed areas of coast, confining nest sites to most sheltered areas.

4.2 Shortcomings and next steps

As this review has highlighted, there is considerable spatial variation in the population trends of species between colonies, and in the mechanisms by which different colonies are affected by climate change. Spatial comparisons have been carried-out between colonies that reside within similar waterbodies (e.g. Celtic Sea, North Sea, Norwegian Sea) (Wanless et al., 2009; Lauria et al., 2013; Lorentsen et al., 2015), and between colonies in differing seas (Harris et al., 2005; Harris et al., 2006; Frederiksen et al., 2013; Reiertsen et al., 2014). This is important because there is strong variation in the extent to which different colonies have been studied. For example, a total of 54 studies considered in this review originated from the Isle of May, located in the Firth of Forth, whilst only 13 of the 156 studies reviewed were based within or included the study region. Anderson et al. (2014a), Lauria et al. (2013) and Frederiksen et al. (2007a), are some of the only studies that compare regions outside and within the study region. Given the spatial variation in the diet, population trends and responses to climate change that exist between colonies (see above), this means that the specific details in this review for individual species may not necessarily be applicable to the study region. Specifically, it seems likely that colonies in this region are less reliant on sandeels than in the North Sea (e.g. Frederiksen et al. 2005), and therefore are potentially less vulnerable to the SST – plankton – sandeel cascade of impacts than those that border the North Sea (see Cook et al., 2011; Lauria et al., 2012).

Similarly, there is considerable variation in the extent to which different seabirds have been studied (Table 1). Whilst auks, most terns, kittiwake, and shag have been reasonably well studied, some of the gulls, skuas, black guillemot, and petrels, for example, were the subject of fewer studies, and therefore the degree of confidence in the mechanisms identified for these species will be lower. Thus, there is an urgent need for more studies of the impacts of climate change on those species highlighted in this report, which have been relatively poorly-studied to date.

Whilst mechanisms associated with direct impacts of severe events and impacts of prey abundance have been relatively well-studied, some of the other mechanisms, such as potential impacts of disease or habitat change on colonies have been studied to a lesser extent, but could be important in some circumstances. In the face of the uncertainty associated with potential climate change impacts, a good approach is to monitor populations or colonies of interest, and to analyse such data to identify potential climate change impacts. More detailed work may then be required to identify the mechanisms

underpinning those relationships, which could then be used to inform potential conservation responses (Pearce-Higgins, 2011).

4.3 Climate projection modelling

In MarPAMM Lot 5 Task 2.1 we aim to predict late 21st-century abundance for all breeding seabirds within the study region under climate change. The methods used are based upon those used in Johnston et al. (2013), but with changes made to take advantage of recently updated climate projections, to include oceanographic data and to refine the modelling approach. Projected abundances from this analysis will then be used in projection of at-sea distribution for a subset of these species (Task 2.2). In combination with this literature review, these information sources will be used to produce a vulnerability assessment for breeding seabirds within the study region.

4.4 Conclusions

Despite these caveats and limitations of the review, it has crucially highlighted the importance of climate change impacts on seabirds being mediated through cascades through the food chain, affecting plankton, fish and then birds. Severe storm events may also be important for some species. Given the relative paucity of studies specifically from within the study region, it is difficult to be confident about the precise nature of climate change on seabirds breeding in the West of Scotland, North-East of Ireland and Northern Ireland, but it would seem likely that impacts on key prey species will be a key mechanism, whilst some colonies may also be particularly vulnerable to storm events. This would suggest that climate change adaptation options may best focus around the management of fish stocks (e.g. Frederiksen et al. 2004) and the protection of colonies from storm events (e.g. Rounds et al., 2004), but more work is required to support this assessment.

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6. Disclaimer

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Project partners

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