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Final report

Terrestrial biological monitoring on Fildes Peninsula (Maxwell Bay, Antarctic)

by:
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On behalf of the German Environment Agency

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Abstract: Terrestrial biological monitoring on Fildes Peninsula (Maxwell Bay, Antarctic)

Current climate change is having an increasing impact on polar ecosystems. In addition, cumulative effects such as fishing, tourism, pollution etc. harbour further risks for the Antarctic and the surrounding Southern Ocean. Long-term monitoring programmes on population and distribution data of fauna and flora, especially in areas with recent, clearly detectable changes, such as the West Antarctic Peninsula, can provide valuable insights into the effects of changing environmental conditions on ecosystems and also help to assess future developments and develop protective measures. The study area of the present study is the Fildes Region and adjacent ice-free areas of Maxwell Bay in the south-west of King George Island (South Shetland Islands, Maritime Antarctic). The Fildes Region in particular, as one of the largest ice-free areas in the Maritime Antarctic, is strongly characterised by diverse human activities and the resulting conflicts between the interests of the various user groups and legally agreed environmental protection requirements. Based on long-term monitoring in the Fildes Region that began in the 1980s, a systematic survey of breeding bird and seal populations was carried out in the summer months of the 2021 to 2023/24 seasons. According to the results, species such as the gentoo penguin and southern giant petrel show strong population growth in a long-term comparison and recently reached maximum breeding pair numbers. In contrast, the populations of the brown skua and south polar skua have remained stable. In contrast, long-term population declines were recorded for the Adélie penguin, kelp gull and Antarctic tern. Due to a very sharp population decline, the chinstrap penguin and cape petrel were recently only present in the Fildes Region with very few breeding pairs. In addition, the seal population, the distribution of larger elephant seal resting sites and all observed seal pupping sites were recorded. The data set on breeding bird populations in selected areas of Maxwell Bay was supplemented with new data. Furthermore, the documentation of glacier retreat areas in selected areas of Maxwell Bay was continued using satellite images, as was the evaluation of regional climatic development. The continuation of marine debris monitoring also provided valuable information on the type and quantity of marine debris washed ashore. In addition, the interaction of seabirds with anthropogenic material and the current state of knowledge on introduced, non-native species in the Fildes Region are presented. Finally, the need for further research in this area is explained.

Kurzbeschreibung: Terrestrisch-biologisches Monitoring auf der Fildes Peninsula (Maxwell Bay, Antarktis)

Die aktuellen Klimaveränderungen wirken sich zunehmend auf polare Ökosysteme aus. Zusätzlich bergen kumulative Effekte wie z.B. Fischerei, Tourismus, Verschmutzung etc. weitere Risiken für die Antarktis und das umgebende Südpolarmeer. Langzeitmonitoring-Programme zu Populations- und Verbreitungsdaten von Fauna und Flora, insbesondere in Gebieten mit jüngsten, klar nachweisbaren Veränderungen, wie z. B. die Westantarktische Halbinsel, können wertvolle Erkenntnisse über die Auswirkungen veränderter Umweltbedingungen auf Ökosysteme liefern und auch helfen, künftige Entwicklungen zu beurteilen und Schutzmaßnahmen zu erarbeiten. Das Untersuchungsgebiet der vorliegenden Arbeit ist die Fildes-Region sowie angrenzende eisfreie Gebiete der Maxwell Bay im Südwesten von King George Island (Südshetland-Inseln, Maritime Antarktis). Insbesondere die Fildes-Region als eines der größten eisfreien Areale der Maritimen Antarktis ist stark geprägt von vielfältigen menschlichen Aktivitäten und daraus resultierender Konflikte zwischen den Interessen der verschiedenen Nutzergruppen und gesetzlich vereinbarten Umweltschutzbestimmungen. Basierend auf einem in den 1980er Jahren begonnenen Langzeitmonitoring in der Fildes-Region erfolgte in den Sommermonaten der Saisons 2021 bis 2023/24 eine systematische Erfassung der Brutvögel- und Robbenbestände. Demnach zeigen Arten wie der Eselpinguin und Südlicher

Riesensturmvogel im Langzeitvergleich ein starkes Populationswachstum und erreichten zuletzt maximale Brutpaarzahlen. Die Bestände der Braunen Skua and Südpolarskua blieben dagegen stabil. Im Gegensatz dazu waren langfristige Bestandsrückgänge für Adéliepinguin, Dominikanermöwe und Antarktisseeschwalbe zu verzeichnen. Aufgrund einer sehr starken Bestandsabnahme waren Zügelpinguin und Kapsturmvogel zuletzt nur noch mit sehr wenigen Brutpaaren in der Fildes-Region vorhanden. Daneben erfolgte die Erfassung der Robbenbestände, der Verbreitung größerer Ruheplätze des Seeelefanten sowie aller beobachteten Robbenwurfplätze. Der Datensatz zum Brutvogelbestand in ausgewählten Bereichen der Maxwell Bay konnte um neue Daten ergänzt werden. Ferner wurde die Dokumentation der Gletscherrückzugsgebiete ausgewählter Bereiche der Maxwell Bay anhand von Satellitenbildern sowie die Evaluierung der regionalen klimatischen Entwicklung fortgesetzt. Die Fortsetzung des Strandgutmonitorings lieferte zudem wertvolle Informationen zur Art und Menge von angespültem Meeresmüll. Zusätzlich werden die Interaktion von Seevögeln mit anthropogenem Material sowie der aktuelle Kenntnisstand zu eingeschleppten, nicht-heimischen Arten in der Fildes-Region dargestellt. Abschließend wird der weitere Forschungsbedarf in diesem Gebiet erläutert.

Table of content

List of figures

List of tables

List of abbreviations

Abbreviation	Explanation
ACAP	Agreement on the Conservation of Albatrosses and Petrels
AON	Apparently occupied nests
ASPA	Antarctic Specially Protected Area
ASMA	Antarctic Specially Managed Area
ATCM	Antarctic Treaty Consultative Meeting
ATS	Antarctic Treaty Secretariat
bp	Breeding pair
CCAMLR	Commission for the Conservation of Antarctic Marine Living Resources
CEP	Committee of Environmental Protection
COMNAP	Council of Managers of National Antarctic Programmes
ENSO	El Niño-Southern Oscillation
ept	estimated pair type
ESA	European Space Agency
GIS	Geographical Information System
GPS	Global Positioning System
IAATO	International Association of Antarctica Tour Operators
IBA	Important Bird Area
IOC	International Ornithological Congress
IPCC	Intergovernmental Panel on Climate Change
IUCN	International Union for Conservation of Nature
juv/bp	successfully raised chicks per breeding pair
KGI	King George Island
LDPE	Low-density polyethylene
LLDPE	Linear low-density polyethylene
MARPOL	International Convention for the Prevention of Marine Pollution from Ships
mIBA	marine Important Bird Areas
m/a	metres per year
NOAA	National Oceanic and Atmospheric Administration
ONI	Oceanic Niño Index
SAM	Southern Annual Mode
SCAR	Scientific Committee on Antarctic Research

Abbreviation	Explanation
SCARCGA	SCAR Composite Gazetteer of Antarctica
SOI	El Niño Southern Oscillation Index
UN	United Nations
USGS	United States Geological Survey
WAP	Western Antarctic Peninsula

Summary

Introduction

The Antarctic is of great global importance as a result of its unique biological, geological and hydrological characteristics. Increasingly, the Antarctic and the surrounding Southern Ocean are under pressure due to the cumulative impacts caused by of climate change, pollution, fishing, tourism and a variety of other human activities with consequences for both local polar ecosystems and the regulation of global climate and sea level rise. Within the Antarctic, the strongest current changes in recent decades can be observed on the Antarctic Peninsula. The effects of this temperature increase on polar ecosystems in the Antarctic have been the subject of numerous studies. Of particular importance here is the variability of sea-ice cover and the resulting changes in the food web. The role of seabirds as valuable indicators of the state of marine and coastal ecosystems is widely recognized. In particular, food availability for seabirds is influenced by the effects of current climate change, such as increased water temperature and changes in sea ice cover, as well as increased fishing activities. In addition, seabirds often reflect anthropogenic influences at local, regional and global levels. Long-term monitoring programmes in the Antarctic are particularly valuable in this regard, as their great importance for assessing the state of ecosystems and for forecasting future developments is widely recognized and reminded. The study area Fildes Region in southwest King George Island, Maritime Antarctica is characterized by a comparatively high terrestrial biodiversity and a high level of diverse human activities. Negative impacts of the high concentration of human activities in the Fildes Region have been repeatedly described, including disturbance of seabirds and seals, habitat destruction and significant contamination of soil, water and air. The present study is based on several research projects on seabird and seal monitoring in the Fildes Region and ice-free area of the Maxwell Bay and other research questions and aimed to continue the long-term monitoring. This study presents the results of the continued seabird and seal monitoring and thus supports the high value of such long-term surveys in the Antarctic. In addition, the state of knowledge on breeding birds in selected ice-free areas of Maxwell Bay was updated, the input of marine debris into the study area and the influence of anthropogenic material on seabirds was analysed. This work contributes to a comprehensive assessment of ecological processes as a result of current climatic changes and the increasing anthropogenic influence in the Antarctic.

Monitoring of breeding birds and seals in the Fildes Region

The Fildes Region is currently home for 14 breeding bird species, whose population and distribution were recorded in the 2021/22 to 2023/24 seasons using standardised methodology. Ardley Island, located to the east of the Fildes Peninsula, is currently home to eleven breeding bird species, among them three representatives of the *Pygoscelis* genus chinstrap penguin, Adélie penguin and gentoo penguin, which breed there sympatrically. Since the start of long-term monitoring in the 1979/80 season, data on this penguin population is now available for up to 44 years. Based on that the number of breeding pairs of chinstrap penguins on Ardley Island continued to decline in the current study period. In the 2023/24 season, only 3 breeding pairs of chinstrap penguins were counted, the lowest number since long-term monitoring began. The local population of this species thus declined by 98.3 % in a long-term comparison. The Adélie penguin also recorded very low breeding pair numbers on Ardley Island. With only 184 breeding pairs in the 2022/23 season, the lowest value since the start of long-term monitoring was recorded for this species. Overall, the decline in the local population of the Adélie penguin was -71.1 %. In contrast, the gentoo penguin population on Ardley Island continued to increase during the study period, reaching the maximum to date of 7,942 breeding pairs in the 2023/24 season. The number of gentoo penguins breeding on Ardley Island thus increased by 122 % in a long-term comparison.

The breeding success of chinstrap penguins on Ardley Island showed strong fluctuations in a long-term comparison, although the significance is limited due to the recently very low number of breeding pairs. In contrast, the values for the breeding success of Adélie and gentoo penguins fluctuated much less, but showed a negative trend over time. Nevertheless, the long-term mean for the breeding success of both species was significantly higher (Adélie penguin: 1.15 ± 0.25 successfully raised chicks per pair (juv/bp), gentoo penguin: 1.27 ± 0.17 juv/bp) than for the chinstrap penguin (1.09 ± 0.44 juv/bp). No fledged chinstrap penguin chicks were registered in the 2023/24 season, with only three pairs breeding. Although a declining reproductive success of a population, as it was observed for the gentoo penguin on Ardley Island, may indicate that its growth is slowly approaching capacity, the still rapidly growing population of this species proves that the maximum capacity of this breeding area has not yet been reached. The extent to which ongoing immigration of birds from other areas plays a major role in the population growth of the gentoo penguin is not known. The results shown here on the population dynamics of the three penguin species on Ardley Island once again demonstrate that the long-term trends already described are continuing. These local trends are consistent with the development of the three *Pygoscelis* penguin species in large areas of Antarctica, especially in the northern WAP region. This development reflects the different responses of the various species to the complex ecological processes resulting from progressive climatic warming. The breeding success of penguins is significantly influenced as well by regional factors such as sea ice cover, food availability as well as local weather extremes or predation. In comparison, the breeding success of *Pygoscelis* penguins on Ardley in recent decades were often higher than in neighbouring colonies on Barton and Potter Peninsula and in more remote colonies, such as Livingston Island, Deception Island, Peterman Island or the South Orkney Islands. The higher breeding success of gentoo penguins on Ardley Island compared to the other *Pygoscelis* species is presumably related to the higher variability of this species in relation to various environmental factors. The results of the CEMP site monitoring from Ardley Island showed that the body weight of Adélie penguins at fledging in the 2023/24 season was significantly higher than in previous years, although the sample sizes were not the same. In contrast, the body mass of the gentoo penguins showed no difference between the seasons but decreased slightly within each season. This indicates that chicks that fledge earlier in the season have a higher body weight and consequently have a higher probability of survival in their first winter. Based on our own, older spatial surveys of the penguin colony and current aerial photographs, it was shown, that the central, higher part of the island is no longer occupied at all. In contrast, numerous new and larger breeding groups of gentoo penguins can now be found in the south of Ardley Island. This area has a dense, partly peat-like moss cover, especially in the higher areas, which is increasingly dying off as a result of the dense faecal cover. The environmental factors responsible for these spatial changes in the penguin colony should be the subject of future research.

Based on long-term monitoring in the Fildes Region, data from up to 35 years is now available for the population of the southern giant petrel. This data clearly shows that the population of this species has been steadily increasing in this area for several years, reaching its maximum breeding pair number of 872 bp in the 2022/23 season. This means that the number of southern giant petrels breeding in the study area has increased by 200 % since the 1983/84 season. The ten individual colonies of southern giant petrel in the Fildes Region harbour colonies of varying sizes, ranging from 5 to 177 breeding pairs, largely reflecting the current string population growth across the study area showing a positive population trend. In the three seasons analysed, a comparatively high overall breeding success was again recorded for the Fildes Region, with 0.46 to 0.59 juv/bp, which was well above the long-term average. The stabilisation of the previously declining long-term trend thus continued. For the period 2003/04 to 2023/24, there was a slight but significant increase in overall breeding success. The highest breeding success of

0.81 juv/bp in one colony contrasts with unusually low breeding success or complete brood failure in other colonies, although the reasons for this are not known, as the latter colonies are in areas with little human disturbance, other factors than these appear to be responsible. Nevertheless, with regard to the reaction of the southern giant petrel to human disturbance, previous results can be largely confirmed. Southern giant petrels in the Fildes Region are exposed to a high level of anthropogenic disturbance due to the wide range of human activities, particularly affecting breeding individuals during austral summer. The extent of human disturbance in the giant petrel colonies in the Fildes Region depends on the distance of the breeding sites to the stations and accessibility on foot or by boat. Thus, colonies in areas with a low level of disturbance differ significantly from those with a medium and high level of disturbance in terms of their breeding success. In line with the strong population increase and the development of the level of human disturbance after the construction of several stations in the 1980s, most of the individual breeding sites within the colonies show a clear positive development. Numerous previously abandoned breeding sites have been recolonised and those that had been occupied for some time but where no young were successfully raised are now showing breeding success. This indicates habituation effects towards certain human disturbances. The development of the southern giant petrel population in the Fildes Region and other areas of Maxwell Bay shown here is consistent with published information on regional positive population trends. For the South Shetland Islands, for example, numerous studies on stable or increasing populations contrast with relatively few reports of decreasing trends. As a result of the strong population increase, the Fildes Region appears to be the second largest breeding site in the South Shetland Islands, although the reasons for the global increase in the population of the southern giant petrel are not yet fully understood. In conclusion, with regard to the population development of the southern giant petrel in the Fildes Region, possible influences of large-scale environmental factors probably outweigh those of local environmental conditions and negative anthropogenic influences.

The cape petrel in the Fildes Region has shown a drastic decline in the breeding population in recent decades. The population decline since the start of long-term recording in the 1984/85 season is currently around 98 %. In the 2021/22 to 2023/24 seasons, very low numbers of 3 to 6 breeding pairs were again recorded. In each case, one to three of the 36 known breeding sites were occupied. Due to a lack of necessary data, the reasons for the observed decline in the cape petrel population in the Fildes Region are difficult to determine. Possible causes include high predation pressure from skuas, but above all environmental changes such as food shortages and poor weather conditions. Human activities, on the other hand, play a subordinate role, as the majority of breeding areas are located in remote and unvisited areas. The few assessments of population development from other areas in Antarctica are contradictory and range from decreasing trend, no change, to population increase.

After the first documented sightings of the light-mantled sooty albatross, whose range of the sooty albatross extends circumpolar to various Subantarctic islands near the Antarctic Convergence between 46° and 55° S, in the Fildes Region in the 1984/85 season, numerous further sightings of this species have been documented almost every year. The first evidence of breeding of this species dates back to the 2008/09 season, including the documentation of chicks, was followed by several suspicion of breeding in subsequent seasons. In the 2021/22 to 2023/24 seasons, synchronised flights of albatross pairs were observed several times, but no other signs that could indicate breeding were observed. So far, almost all observations have been made on the west coast of the Fildes Peninsula, particularly in the area of the known breeding site. In contrast, numerous observations of flying light-mantled sooty albatrosses were recorded for the first time on the east coast of the Fildes Peninsula in the 2022/23 season. On several days, one to two albatrosses were observed there, either flying synchronous display flight or showing courtship behaviour at a scree slope. It is possible that these observations represent the first signs of this species becoming established in the east of the Fildes Peninsula.

Following initial surveys of the two native storm-petrel species Wilson's storm-petrel and black-bellied storm-petrel in the Fildes Region in the 1980s which was followed by a systematic survey based on nocturnal calls during the breeding season in the 2003/04 to 2005/06 seasons, the data set was supplemented during the study period based on individuals heard from the breeding cavities during the day. All additional breeding sites were smaller spots with only a few individuals. A total of 344 breeding sites have now been recorded. At most breeding sites (39 %), both species bred sympatrically, compared to 35 % with only Wilson's storm-petrels and 26 % with only black-bellied storm-petrel. Furthermore, small breeding sites with up to 10 breeding pairs predominate. Overall, the breeding population in the Fildes Region is still estimated at 3,500 to 5,000 Wilson's storm-petrels and 500 to 1,000 black-bellied storm-petrels.

The current survey of skua breeding pair numbers in the Fildes Region revealed a breeding population of between 32 and 55 bp of brown skuas, between 84 and 149 bp of south polar skuas and between 5 and 13 mixed and hybrid pairs for the study period. Thus, after earlier collapses, the local skua population was in the average range. Overall, the populations of the different skua pair types remained stable in the long-term comparison despite considerable fluctuations and showed no significant temporal trend. Furthermore, the two species showed a similar reaction to certain environmental influences. There was a significant negative correlation between the average snow depth in November and December and the number of breeding pairs of brown skuas in the Fildes Region. For south polar skuas, such a correlation was indicated, but fell just short of the significance level. This illustrates the importance of snow-free breeding sites for these ground-nesting species, which, unlike other species, do not breed on cliffs but mostly on low hills, which are often still covered in snow well into spring. According to this, the lowest breeding pair numbers of skuas in the Fildes Region were recorded in seasons with high snow cover at the beginning of the breeding season. The breeding success of the two skua species in the Fildes Region showed considerable fluctuations over the years studied and again differed significantly from each other. Brown skuas successfully raised relatively more chicks than south polar skuas. A temporal trend could not be determined for any pair type. The spatial distribution of skua breeding pairs in the three seasons studied largely corresponds to the data from previous years with regard to the greater concentration of brown skuas near penguin colonies and the coast. In contrast, there are significantly more south polar skuas in the southeast of the Fildes Peninsula. In addition to the number of breeding pairs, the number of recorded skua territories in the Fildes Region that were occupied but where breeding had not initiated has been recorded since the 2006/07 season. It becomes clear that there were significantly fewer fluctuations in the total number of pairs of skuas present in the breeding area in summer than when only breeding pairs were considered. A long-term comparison clearly shows that south polar skuas in particular often occupied territories in large numbers but did not start breeding. This clearly indicates a greater dependence of south polar skuas on environmental factors that influence the decision to breed or not. In all three studied seasons, active feeding of skuas by station personnel was again observed in the majority of stations on the Fildes Peninsula, or inferred by corresponding findings of fresh food remains on nests near the station, although this practice contradicts the mandatory requirements of the Madrid Protocol.

During the breeding bird survey in the Fildes Region in the 2021/22 to 2023/24 seasons, a total of 46, 53 and 51 breeding pairs of kelp gulls were recorded, respectively. These values are well below the long-term average, so that the negative trend in the population has intensified in the long-term comparison since the start of the survey in the 1984/85 season, caused by the higher numbers of breeding pairs in the 1980s and 1990s. In the period between 2004/05 and 2023/24, however, the kelp gull population remained stable, although significant fluctuations were recorded. A significant negative correlation between the number of breeding pairs of kelp gulls in the Fildes Region and local snow cover in the spring months of October and November, which had been indicated, has now been proven. This shows that a snowy spring, in which the breeding sites are still snow-covered at the time relevant for the decision to breed, leads to a low number of breeding pairs in the respective breeding season. In contrast to earlier results, the current data did not show any correlation between the number of breeding pairs of kelp gulls in the Fildes Region and the cyclical atmospheric-oceanic phenomenon 'El Niño-Southern Oscillation'. Although significantly reduced numbers of breeding pairs were recorded in seasons with a moderate to very strong El Niño, whereas in La Niña years relatively many gulls bred in the area, there was no relationship between this climate parameters and the local kelp gull population.

The local population of Antarctic terns can fluctuate greatly due to meteorological conditions. Furthermore, breeding terns tend to relocate or abandon their breeding sites during the breeding season due to their high susceptibility to disturbance. Nevertheless, despite the resulting methodological inadequacies, long-term monitoring can provide valuable information on the general development of the population of this species in an area. In the 2021/22 and 2022/23 seasons, only 73 and 125 breeding pairs of Antarctic terns were recorded respectively, which are the two lowest values since the start of long-term monitoring in the season 1984/85. In the 2023/24 season, the population of this species recovered slightly to 196 breeding pairs. This means that the Antarctic tern population in the three breeding seasons analysed was well below the long-term average of 423 ± 262 BP. Despite considerable population fluctuations, there is evidence of a significant downward trend in the population of Antarctic terns in the Fildes Region in a long-term comparison since the start of the survey in the 1984/85 season. If the number of breeding pairs between the 2000/01 and 2023/24 seasons is considered, this negative trend becomes even more pronounced. In all three studied seasons, more than one breeding pair was mostly recorded at the majority of breeding sites in the Fildes Region. The mean number of pairs per breeding site was 3.87 ± 0.58 during the study period. The proportion of different size classes of breeding sites varied considerably between years. As in previous years, the proportion of single breeders was relatively high at around 49 % to 52 %. However, large colonies with more than 20 and up to 300 breeding pairs were not recorded any more. Furthermore, the human influence in the Fildes Region on the Antarctic tern is evident in the form of the use of anthropogenic material such as metal, pieces of bone and glass or fruit pits for nest building. The cause of the sharp decline of the breeding pair number of the Antarctic tern in the Fildes Region area is not known. The extent to which the population decline can also be observed in other breeding areas, including Subantarctic islands in the Indian and Pacific Ocean, is not known due to the limited comparative population data published. Statements on stable or growing populations are contrasted with information on shrinking populations.

A small number of snowy sheathbills have been regularly observed in the study area. However, direct evidence of breeding is rare due to the inaccessibility of the breeding sites, so various parameters, such as repeated observations of two adult birds at the same site, were used to conclude that breeding was suspected. So far, a few breeding sites have been identified on the west coast of the Fildes Region. In the 2021/22 and 2023/24 seasons, there were several observations of single individuals as well as a breeding record based on the observation of a moulting chick together with two adult birds in the 2022/23 season. In contrast, another long-standing breeding site in the south-west of the Fildes Peninsula appeared unoccupied.

The blue-eyed shag was considered a potential breeding bird in the Fildes Region for many years. Despite regular observations of flocks of up to 30 individuals in the area, no more broods of this species have been recorded since the 1980s. Breeding attempts in the study area, as most recently in the 2015/16 and 2019/20 seasons, were not recorded in the 2021/22 and 2022/23 seasons. In contrast, in the 2023/24 season, successful breeding of this species was recorded for the first time in a long time. A total of four breeding pairs bred on a rock in the south-east of Ardley Island in the midst of numerous breeding gentoo penguins. In February 2024, a total of 4 chicks were counted in 3 nests.

Data on sightings of vagrant and migrant bird species in the Antarctic can indicate a potential expansion of the breeding range of Subantarctic species and may provide valuable clues to unusual environmental conditions that allow vagrants to reach regions beyond their usual distribution range. During the current study period, numerous representatives of non-native birds were recorded, in some cases exclusively as dead finds. The most frequently recorded dead finds were of seabird species that are frequently observed in the Southern Ocean but are not found as breeding birds in the Fildes Region, in particular prions, snow petrels and southern fulmars. Emperor penguins were observed in all three seasons of the study period. Other bird species were observed either in two (cattle egrets, white-rumped sandpiper, Baird's sandpiper) or one of the three studied seasons (macaroni penguin, Arctic tern). King penguins were not observed in the study region, although there were frequent observations or breeding attempts in neighbouring penguin colonies.

Continuing the long-term monitoring, monthly seal surveys were conducted at the coastline of the Fildes Peninsula and Ardley Island in the 2021/22 to 2023/2024 seasons. Thus, data on the seal population in the austral summer in the Fildes Region is available for up to 28 years. The monthly changes in the local seal population correspond to those described in previous studies, i.e. the number of registered southern elephant seals is highest in January, while the number of Weddell seals is highest in December and January, with their numbers decreasing significantly at the end of the austral summer. In contrast, the number of Antarctic fur seals does not reach its maximum until February. During the monthly seal counts in the 2021/22 to 2023/24 seasons, a total of between 1,899 and 2,565 southern elephant seals were recorded. These total numbers were thus within the long-term average since 2000/01 of 2,115 ± 324, with the total for the 2023/24 season being the second highest in this period. As in previous years, the highest number of individuals changing fur was again recorded in January. Remarkable is the minimum value of only 339 southern elephant seals in February 2023, the lowest value for this month since the start of regular monitoring in the 1982/83 season. However, monthly trends are not discernible for the period 2000/01 to 2023/24, which indicates a stabilization of the population after the decline in the 1980s. During the current seal counts, all elephant seal haul-out sites with at least ten individuals were also recorded. The largest concentration of southern elephant seals has been documented in January in the northernmost part of the Fildes Peninsula, with 144, 239 and 189 individuals, respectively, the latter being one of the highest number of individuals since the beginning of the seal monitoring in the Fildes Region (maximum: January 2012 with 255 individuals). Analysis of the spatial data, which now spans 17 years, shows considerable seasonal and annual fluctuations. The spatial distribution of haul-out sites showed no significant changes when compared to previously published results. Spatial changes or declines in haul-out sites due to human disturbance, as have been detected in neighbouring Admiralty Bay, are not evident on the Fildes Peninsula. Although this area has a much higher concentration of human activity overall, the elephant seals' haul-outs are located exclusively on the west coast, which, in contrast to the station surroundings, is hardly affected by direct human disturbance, as is regularly observed around stations. One exception is a larger haul-out site in a bay, which is regularly visited by tourists and station members in the austral summer. However, no spatial or quantitative changes were detected there either.

The total number of Weddell seals recorded on the Fildes Peninsula and Ardley Island in the 2021/22 to 2023/24 seasons was 74, 94 and 147 individuals respectively and thus in some cases well below the long-term average of 151 ± 53 individuals since the 2000/01 season. As in previous years, the number of Weddell seals observed in the study area decreased towards the end of the austral summer. In addition, both January and February, as well as December 2022, recorded the lowest numbers of Weddell seals for the respective month since regular surveys began in the 1982/83 season. In February 2022, no seals were recorded at all for the first time. Accordingly, the 2021/22 season saw the lowest total number of this species ever recorded in austral summer in the study area and the 2022/23 season the second lowest. On average, more Weddell seals were observed in the Fildes Region in the 1980s than in the period 2000/01 to 2023/24. While the total number of seals and the number in December and February remained stable during this period, a downward trend is evident for the month of January.

Both the monthly and total numbers of Antarctic fur seals counted during the austral summer in the 2021/22 to 2023/24 seasons showed notable fluctuations. A total of 815 and 456 individuals were counted in the 2021/22 and 2023/24 seasons, respectively, with the long-term average since 2000/01 being 713 ± 399 individuals. In contrast, only 275 individuals were counted in the 2022/23 season. Nevertheless, the number of fur seals in the 1980s was significantly lower than it is today. Due to the remarkable fluctuations in the last two decades, no significant trends for the local population of this species in the austral summer between the 2000/01 and 2023/24 seasons can be identified. However, a strong decline in the total numbers can be seen since the season 2014/15. In the season 2021/22, a leucistic Antarctic fur seal was documented in the study area. In total, only two leucistic specimens out of a total of 15,381 individuals were recorded during the monthly seal counts in the seasons 2000/01 to 2023/24 in 82 counts in the summer months. For the 1980s, on the other hand, there were no reports of observations of leucistic individuals in 131 counts carried out twice a month throughout the year, with a total of 3,266 animals recorded. Observations of other atypical fur colorations are not known for the Fildes Region.

During the austral summer of the 2021/22 to 2023/24 seasons a small number of new-born seals were again recorded on the Fildes Peninsula and Ardley Island. Although the main reproduction season for elephant seals and Weddell seals is outside the austral summer and most of the pups have already left the area, a single elephant seal pup was observed in January 2023 at the north-west coast of the Fildes Peninsula. Furthermore, four Antarctic fur seal pups were recorded in the 2021/22 and 2023/24 season, respectively, and three pups in the seasons 2022/23.

Breeding bird monitoring in the ice-free areas of Maxwell Bay

Due to major logistical constraints, not all relevant ice-free areas could be visited during the study period. However, data on breeding birds for some areas were kindly provided by Czech, Korean and Ukrainian scientists respectively. Therefore, only limited data can be added to those listed in previous studies. Based on the available data for Barton Peninsula, it can be stated that the population of chinstrap penguins did not show a temporal trend over the survey period, although the number of breeding pairs of this species has decreased over the past ten years. In contrast, the population of gentoo penguins continued to increase. Since the 2018/19 season, the gentoo penguin has been the most abundant penguin species in this colony. As on Ardley Island, the highest number of breeding pairs of this species since the start of the long-term survey was recorded in the 2023/24 season. On the small Green Point peninsula to the northeast of the Fildes Peninsula, which can be accessed by foot, a small number of breeding pairs of kelp gulls were recorded during the study period. In the 2021/22 season, a pair of south polar skuas bred there. Recent publications on the penguin population of the Stranger Point colony on Potter Peninsula indicate a sharp decline in the Adélie penguin population in selected nest groups at the Stranger Point colony between the 1995/96 and 2022/23 seasons, totalling -89.8 %. Similarly, the number of Adélie penguins on Ardley Island fell by 85 % in the same period. The number of fledged chicks fell by 93.1 % at Stranger Point and by 82.8 % on Ardley Island. These comparative results indicate a very uniform regional development in response to changing environmental conditions. Breeding success showed no decreasing trend in either colony. The extent to which the increasing trend of the gentoo penguin population in the colony at Stranger Point has continued in recent years can only be assumed due to the lag of available data, as it corresponds to the supra-regional development in the WAP region. Statements about the possible presence of chinstrap penguins are also currently not possible due to a lack of available data. With regard to the king penguin, annual broods of one pair have now been documented for the Potter Peninsula in the 2011/12 to 2020/21 seasons. Although neither of the two chicks that have hatched so far have been successfully raised, the ongoing breeding attempts, including in neighbouring Admiralty Bay, are interpreted as an incipient expansion of the species' range to the south. Based on the available data for the Stansbury Peninsula, Nelson Island, there were significantly fewer southern giant petrel breeding pairs in the study period than in the previous decade. In the 2022/23 season, only 3 chicks were documented at the beginning of February and in the following 2023/24 season, not a single chick was recorded in January 2024. The cause of the current decline is not known, especially as the immediately neighbouring Fildes region recorded the highest breeding bird population of the last four decades. However, the long-term trend of the breeding population of the southern giant petrel on Stansbury Peninsula is still positive, possibly indicating the interim breeding site shift due to anthropogenic disturbance, as has also been shown for the Fildes Region. Like the number of breeding pairs, the breeding success of the southern giant petrel on Stansbury Peninsula showed a decline. The colonies of the Fildes Region neighbouring the Stansbury Peninsula do not reflect the current collapse in breeding pair numbers and breeding success, with the exception of a very small colony nearby, where a decline in breeding pair numbers and a complete loss of broods was also recorded in the last two years. All other comparable colonies in the neighbourhood, showed no such development, but on the contrary an above-average breeding success in the 2022/23 and 2023/24 seasons. The breeding population of skuas on Stansbury Peninsula could only be recorded in the 2023/24 season. The count in January 2024 revealed a total of 2 bp of brown skua and 11 bp of south polar skua. In addition, 3 territories (1 brown skua, 2 south polar skua) and 3 older nests (1 brown skua, 2 south polar skua) were recorded. In line with the population trend of the cape petrel on the Fildes Peninsula, the sharp decline of this species - despite the observation of some flying individuals - was also confirmed for the Stansbury Peninsula, although this could not be quantified.

Occurrence of the Antarctic pearlwort (*Colobanthus quitensis*) in the Fildes Region

In the Fildes Region only a few locations of the Antarctic pearlwort have been documented to date. However, the populations that were recorded in the 2000s were completely extinct after just a few years. A population of pearlwort known since 2018 was monitored and documented in order to comparing the size and condition of the cushions with previous data. It can be seen that this stand had both dead and additionally overgrown areas. Overall, the stand appeared slightly larger than in 2018. Another small population of Antarctic pearlwort first recorded in the far north of the Fildes Peninsula in the 2020 season was no longer found during the study period.

Climatic changes in the study area

The annual mean temperature in the study area between 1969 and 2023 showed a significant positive trend and increased by 1.21 °C. This corresponds to an annual warming of 0.022 °C and to a further increase compared to previous periods. The annual precipitation in the study area showed a high variability without a significant trend. Furthermore, the negative correlation between annual precipitation and mean annual temperature, which was shown before could be confirmed. Furthermore, the mean temperature of the individual seasons showed a significant positive trend for winter and autumn, but not for summer and spring. The temperature anomaly, i.e. the deviation from the annual average, illustrate that after a phase of colder months between the 2009/10 and 2015/16 seasons, there has been a significant increase in the occurrence of warm months in both winter and summer since then.

Glacier retreat areas of selected areas of Maxwell Bay

The results of the change in glacier areas in the regions bordering Maxwell Bay illustrates the substantial horizontal retreat of the glacier fronts from 1956 to 2023, which increased in recent years and is particularly evident in the area of tidewater glaciers. The current strong glacier retreat in recent years is attributed to the high temperatures in both summer and winter. Thus, the area of the Bellingshausen Dome/Collins Glacier decreased from 12.29 km^2 to 9.14 km^2 by 35 %. In addition, the glacier front showed a retreat by approx. 375 m in 67 years, which corresponds to an annual retreat of 5.6 metres. After stagnating between 2006/07 and 2011/12, the Weaver and Barton Peninsulas showed a renewed significant retreat of the glacier front between 2012 and 2023. Barton Peninsula has gained a very large ice-free area with 1.41 km^2. Overall, the ice-free area of the Barton Peninsula increased from 6.23 km^2 in 1956 to 11.06 km^2 in 2023 by 44 %. Consequently, this peninsula experienced the highest total glacier retreat of all areas with 1123.7 metres and 16.8 m/a. In addition to the land-based glaciers, two ‘tidewater glaciers’ in the prominent bays of Collins Harbour and Marian Cove showed a glacier retreat in Collins Harbour of 1.70 km^2 and 2.14 km^2, respectively. Furthermore, a recent colonisation of glacier retreat areas by seabirds was documented for the Fildes Peninsula with 122 known breeding sites of four species (skuas, kelp gulls, Antarctic terns and Wilson’s storm-petrels) in a total area of 3.15 km^2.

Documentation of permafrost degradation in the Fildes Region

As a result of the global temperature increase, increasing degradation of the permafrost is being recorded in polar regions. For the first time, clearly visible degradation of the permafrost in terms of thaw slumps was recorded at several locations in the Fildes Region in the 2022/23 season. These observations are attributed to the unusually high temperatures in the that season.

Input of marine debris into the study area

In recent years, awareness of the risks of plastics entering the environment has increased significantly, as they can have a negative impact on the environment due to the widespread use of plastics of all kinds, their flexibility and durability as well as the pollutants they contain or adhere to. With the continuation of the current systematic survey of marine debris in the Fildes Region, five years of data on marine debris in a selected bay on the west coast of the Fildes Peninsula are now available. The recorded densities of 0.85 and 0.90 objects/metres in the 2023/23 and 2023/24 seasons were the highest since systematic recording began in 2019. Thus, the comparison with the data from the 2018/19 season onwards indicates an increase. Throughout the study period the proportion of items made of plastic clearly outweighs those made of wood, glass, metal or other materials at around 98 %. Overall, small items dominated the majority of collections, followed by medium-sized and large properties. During the study period, a significant amount of styrofoam and other foam material (42 %) was found at the sampled beach section, although the entry and dumping of styrofoam in the Antarctic is prohibited under the legally binding regulations of the Madrid Protocol and the SCAR Code of Conduct. Furthermore, a clear link to fishing activities (e.g., nets, plastic ropes or strings, floats, packing tapes, etc.) was found in an average of 11 % of the finds. Burnt plastic/pyroplastics was recorded throughout the study period and probably originates from dumping of waste incineration residues on board ships. Plastic bottles, their closures or parts thereof again made up a significant proportion of marine debris. It was noticeable that there were numerous smaller water bottles of the same type and manufacturer. Among the collected marine debris were several industrial plastic pellets or granules. The annual accumulation rates between recorded from the Fildes Region were relatively high and mostly higher than from other areas on the South Shetland Islands, the Antarctic Peninsula and Subantarctic Islands. Due to the local topography, the north-western exposure to the Drake Passage as well as the prevailing westerly winds there is a considerable input of marine debris washed up from the Southern Ocean at the beach section investigated. Local currents and tides also have an impact, even on a small scale. In contrast, the amount of marine debris on the east side of the Fildes Peninsula towards Maxwell Bay is significantly lower. It can be assumed that the input of marine debris takes place via the resident station and via intensive shipping traffic in this area.

Influence of anthropogenic material on seabirds in the Fildes Region

Various negative effects of anthropogenic material on animals in terrestrial, aquatic and marine environments have been repeatedly documented. Analyses of stomach analysed of bird, which were found dead in the Fildes Region, revealed that anthropogenic material was recorded in around 46.2 % of the animals analysed, including Wilson’s storm-petrels, Adélie and gentoo penguins, a south polar skua chick and an adult southern giant petrel. Furthermore, skua pellets with indigestible remains of seabirds were collected and analysed,. On average, clearly identifiable plastic particles were found in approx. 37.4 % of all skua pellets. The frequency of microplastics differed significantly depending on the species. Wilson's storm-petrels contained the highest abundance of microplastics, including industrial granules, with a frequency of approx. 28.3 % of the samples, followed by black-bellied storm-petrel with approx. 30.4 %. The extraordinarily high frequency of microplastic particles (approx. 87.1 %) in skua pellets containing the remains of prions were striking. On average, between 1.6 and 4.6 particles per pellet containing microplastics were found, depending on the bird species. The maximum number of 22 microplastic particles was found in a skua pellet with residues of prion Pellets of this species contained significantly more and larger particles on average, and the mean plastic load per skua pellet also differed depending on the species. The total of 168 microplastic particles found also included a high proportion of industrial granules (approx. 22 %), which were also found in the stomach contents of storm-petrels and in the coastal drift line of the study area. The colours yellow and orange predominated in the 40 industrial granules found in stomach contents and skua pellets. Looking at both colour distributions, it is noticeable that this essentially corresponds to the colour spectrum of the industrial granules collected along the coastline, which also predominantly showed yellow shades. Other examples of the influence of anthropogenic material were the presence of long-lining hooks, glass, plastic film or strings in bird nests as well.

Introduced non-native species

As biological invasions are considered one of the greatest challenges for the conservation of species and the maintenance of the health of ecosystems and their structures, especially in geographically and evolutionary isolated ecosystems, attention was paid to the presence on any non-native introduces species. Several records of non-native grasses and insects have already been documented for the Fildes Peninsula. Repeated checks of the known location of introduced non-native grass species in the area of two Antarctic stations did not reveal any evidence of a re-occurrence of the plants which were manually removed in the 2000s. In January 2022, a conspicuous tuft of grass was noticed in the area of another station under the stairs to a residential building. It was removed, including the surrounding soil in accordance with existing recommendations. Unfortunately, there is currently no species identification available. Furthermore, a single acorn was found in a station area on the main route taken by tourists on the air-cruise programme on their way between the airport and the beach. This indicates that seeds or dispersal units of non-native plants - including larger ones - can still enter the study area as a result of human activities. Regarding introduced non-native insects, specimens of the winter crane fly *Trichocera maculipennis*, which is now considered to be persistent in the Fildes Region, were repeatedly observed inside various station buildings, either flying or in adhesive traps. In addition, up to three individuals of this species were found in the adhesive trap set up in a field hut on Ardley Island. Numerous specimens of the winter crane fly were also detected outside the station buildings.

Open questions, need for further research and development of the monitoring programme

The high value of long-term monitoring programmes of fauna and flora is widely recognised, as they can provide useful insights into the effects of changing environmental conditions on the ecosystem and also help to assess future developments and develop future conservation measures. This is particularly important in areas that have undergone major environmental changes in recent decades as a result of global warming, such as the Western Antarctic Peninsula. The majority of population surveys of seabird and seal populations in the Antarctic are based on only a few counts and, due to frequent natural population fluctuations, have only very limited informative value regarding actual developments. The long-term, largely continuous data series resulting from systematic long-term monitoring in the Fildes Region are therefore of the greatest value. Furthermore, the location of the study area, with its high level of human activity, offers the opportunity to assess the effects of certain anthropogenic influences, such as human disturbance, marine debris or introduced non-native species, on Antarctic ecosystems. Furthermore, recent results show that direct human impacts should be considered within monitoring programmes of certain seabird species in anthropogenic areas to ensure a reliable interpretation of population data and to define indicator species. This can help to address conservation measures that may be necessary to preserve Antarctic ecosystems. For the reasons mentioned above, a continuation of the long-term surveys of the presented topics is urgently recommended. Furthermore, increased use of remote sensing data to avoid logistical constraints and reduce the number of visits in seabird colonies is highly recommended. There is also a need for research on various topics, e.g. investigating the reasons for spatial changes in the penguin colony on Ardley Island, monitoring the occurrence of new taw slumps and the further development of permafrost degradation.

Zusammenfassung

Einleitung

Die Antarktis ist aufgrund ihrer einzigartigen biologischen, geologischen und hydrologischen Eigenschaften von großer globaler Bedeutung. Zunehmend stehen die Antarktis und das sie umgebende Südpolarmeer jedoch aufgrund der kumulativen Auswirkungen, verursacht durch Klimawandel, Umweltverschmutzung, Fischerei, Tourismus und einer Vielzahl anderer menschlicher Aktivitäten, unter Druck, was sowohl für die lokalen polaren Ökosysteme als auch für die Regulierung des globalen Klimas und den Anstieg des Meeresspiegels Folgen hat. In der Antarktis sind die stärksten aktuellen Veränderungen der letzten Jahrzehnte auf der Antarktischen Halbinsel zu beobachten. Die Auswirkungen dieses Temperaturanstiegs auf die polaren Ökosysteme in der Antarktis sind Gegenstand zahlreicher Studien. Von besonderer Bedeutung sind dabei die Variabilität der Meereisbedeckung und die daraus resultierenden Veränderungen im Nahrungsnetz. Die Rolle von Seevögeln als wertvolle Indikatoren für den Zustand von Meeres- und Küstenökosystemen ist allgemein anerkannt. Insbesondere die Verfügbarkeit von Nahrung für Seevögel wird durch die Auswirkungen des derzeitigen Klimawandels beeinflusst, z. B. durch erhöhte Wassertemperaturen und Veränderungen der Meereisbedeckung, sowie durch verstärkte Fischereiaktivitäten. Darüber hinaus spiegeln Seevögel häufig anthropogene Einflüsse auf lokaler, regionaler und globaler Ebene wider. Langfristige Monitoringprogramme in der Antarktis sind in dieser Hinsicht besonders wertvoll, da ihre große Bedeutung für die Beurteilung des Zustands der Ökosysteme und für die Vorhersage künftiger Entwicklungen weithin anerkannt und ihre Durchführung regelmäßig angemahnt wird. Das Untersuchungsgebiet Fildes-Region im Südwesten von King-George-Island in der maritimen Antarktis zeichnet sich durch eine vergleichsweise hohe terrestrische Biodiversität und ein hohes Maß an unterschiedlichen menschlichen Aktivitäten aus. Negative Auswirkungen der hohen Konzentration menschlicher Aktivitäten in der Fildes-Region wurden wiederholt beschrieben, darunter die Störung von Seevögeln und Robben, die Zerstörung von Lebensräumen und die erhebliche Verschmutzung von Boden, Wasser und Luft. Die vorliegende Studie basiert auf mehreren Forschungsprojekten zum Monitoring von Seevögeln und Robben in der Fildes-Region und im eisfreien Gebiet der Maxwell-Bucht sowie auf anderen Forschungsfragen und zielt auf die Fortsetzung des langfristigen Monitorings ab. Die vorliegende Studie stellt die Ergebnisse des fortgesetzten Seevogel- und Robbenmonitorings vor und unterstreicht damit den hohen Wert solcher Langzeituntersuchungen in der Antarktis. Darüber hinaus wurde der Kenntnisstand über Brutvögel in ausgewählten eisfreien Gebieten der Maxwell Bay aktualisiert, der Eintrag von Meeresmüll in das Untersuchungsgebiet und der Einfluss von anthropogenem Material auf Seevögel analysiert. Diese Arbeit trägt zu einer umfassenden Bewertung ökologischer Prozesse als Folge der aktuellen klimatischen Veränderungen und des zunehmenden anthropogenen Einflusses in der Antarktis bei.

Monitoring der Brutvögel und Robben in der Fildes Region

In der Fildes-Region leben derzeit 14 Brutvogelarten, deren Bestand und Verbreitung in den Saisons 2021/22 bis 2023/24 nach einer standardisierten Methodik erfasst wurden. Die östlich der Fildes Peninsula gelegene Ardley-Insel beherbergt derzeit elf Brutvogelarten, darunter drei Vertreter der Gattung *Pygoscelis* mit dem Zügel-, Adélie- und Eselspinguin, die dort sympatrisch brüten. Seit dem Beginn der Langzeitbeobachtung in der Saison 1979/80 liegen nun Daten über diese Pinguinpopulation für bis zu 44 Jahre vor. Demnach ist die Zahl der Brutpaare (bp) der Zügelpinguine auf Ardley Island im aktuellen Untersuchungszeitraum weiter zurückgegangen. In der Saison 2023/24 wurden nur 3 Brutpaare von Zügelpinguinen gezählt, die niedrigste Zahl seit Beginn des Langzeitmonitorings. Damit ist die lokale Population dieser Art im Langzeitvergleich um ca. 98 % zurückgegangen. Auch der Adéliepinguin verzeichnete auf Ardley

Island aktuell einen sehr niedrigen Brutpaarbestand. Mit nur 184 Brutpaaren in der Saison 2022/23 wurde für diese Art der niedrigste Wert seit Beginn des Langzeitmonitorings verzeichnet. Insgesamt betrug der Rückgang der lokalen Population des Adéliepinguins ca. 71 %. Die Eselspinguinpopulation auf Ardley Island nahm dagegen während des Untersuchungszeitraums weiter zu und erreichte in der Saison 2023/24 den bisherigen Höchststand von 7.942 Brutpaaren. Damit stieg die Zahl der auf Ardley Island brütenden Eselspinguine im Langzeitvergleich um 122 %. Der Bruterfolg der Zügelpinguine auf Ardley Island zeigte im Langzeitvergleich starke Schwankungen, wobei die Aussagekraft aufgrund der zuletzt sehr geringen Anzahl von Brutpaaren begrenzt ist. Im Gegensatz dazu schwankten die Werte für den Bruterfolg von Adélie- und Eselspinguinen deutlich weniger, zeigten aber im Zeitverlauf einen negativen Trend. Dennoch war der langfristige Mittelwert für den Bruterfolg beider Arten deutlich höher (Adéliepinguin: 1,15 ± 0,25 erfolgreich aufgezogene Küken pro Paar (juv/bp), Eselspinguin: 1,27 ± 0,17 juv/bp) als für den Zügelpinguin (1,09 ± 0,44 juv/bp). In der Saison 2023/24 wurden keine flüggen Zügelpinguin-Küken registriert, bei nur drei Brutpaaren. Obwohl ein abnehmender Reproduktionserfolg einer Population, wie er für den Eselspinguin auf Ardley Island beobachtet wurde, darauf hindeuten kann, dass sich ihr Wachstum langsam der Kapazität nähert, beweist die immer noch schnell wachsende Population dieser Art, dass die maximale Kapazität dieses Brutgebiets noch nicht erreicht ist. Inwieweit die anhaltende Einwanderung von Vögeln aus anderen Gebieten eine wichtige Rolle für das Populationswachstum des Eselspinguins spielt, ist nicht bekannt. Die hier dargestellten Ergebnisse zur Populationsdynamik der drei Pinguinarten auf Ardley Island zeigen einmal mehr, dass sich die bereits beschriebenen langfristigen Trends fortsetzen. Diese lokalen Trends stehen im Einklang mit der Entwicklung der drei *Pygoscelis*-Pinguinarten in weiten Teilen der Antarktis, insbesondere in der nördlichen WAP-Region. Diese Entwicklung spiegelt die unterschiedlichen Reaktionen der verschiedenen Arten auf die komplexen ökologischen Prozesse wider, die sich aus der fortschreitenden Klimaerwärmung ergeben. Der Bruterfolg der Pinguine wird auch von regionalen Faktoren wie der Meereisbedeckung, der Nahrungsverfügbarkeit sowie lokalen Wetterextremen oder Prädatoren maßgeblich beeinflusst. Im Vergleich dazu war der Bruterfolg von Eselspinguinen auf Ardley in den letzten Jahrzehnten oft höher als in benachbarten Kolonien auf der Barton und Potter Peninsula und in weiter entfernten Kolonien wie Livingston Island, Deception Island, Peterman Island oder den Süd-Orkney-Inseln. Der höhere Bruterfolg der Eselspinguine auf Ardley Island im Vergleich zu den anderen *Pygoscelis*-Arten hängt vermutlich mit der größeren Variabilität dieser Art in Bezug auf verschiedene Umweltfaktoren zusammen. Die Ergebnisse des CEMP-Site-Monitoring auf der Ardley-Insel zeigten, dass das Körpergewicht der Adélie-Pinguine beim Ausfliegen in der Saison 2023/24 deutlich höher war als in den Vorjahren, wenngleich die Stichprobengrößen nicht einheitlich waren. Im Gegensatz dazu zeigte das Körpergewicht der Eselspinguine keinen Unterschied zwischen den Saisons, sondern nahm innerhalb jeder Saison leicht ab. Dies deutet darauf hin, dass Küken, die früher in der Saison flügge werden, ein höheres Körpergewicht haben und folglich eine höhere Überlebenswahrscheinlichkeit in ihrem ersten Winter haben. Anhand eigener, älterer räumlicher Erhebungen der Pinguinkolonie sowie aktueller Luftbilder konnte ferner belegt werden, dass der zentrale, höher gelegene Teil der Insel überhaupt nicht mehr besetzt ist. Dagegen finden sich im Süden von Ardley Island inzwischen zahlreiche neue und größere Nestgruppen von Eselspinguinen. Dieses Gebiet weist vor allem in den höheren Lagen eine dichte, zum Teil torfartige Moosdecke auf, die durch die dichte Bedeckung mit Guano zunehmend abstirbt. Die Umweltfaktoren, die für diese räumlichen Veränderungen in der Pinguinkolonie verantwortlich sind, sollten Gegenstand künftiger Forschungen sein.

Auf der Grundlage des Langzeitmonitorings in der Fildes-Region liegen nun Daten aus bis zu 35 Jahren für die Population des Südlichen Riesensturmvogels vor. Diese Daten zeigen deutlich, dass die Population dieser Art in diesem Gebiet seit mehreren Jahren stetig zunimmt und in der Saison 2022/23 eine bisherige maximale Brutpaarzahl von 872 BP erreicht hat. Das bedeutet, dass die Zahl der im Untersuchungsgebiet brütenden Südlichen Riesensturmvögel seit der Saison 1983/84 um 200 % gestiegen ist. Die zehn einzelnen Kolonien des Südlichen Riesensturmvogels in der Fildes-Region beherbergen Kolonien unterschiedlicher Größe, die von 5 bis 177 Brutpaaren reichen, und spiegeln im Wesentlichen das derzeitige starke Populationswachstum im gesamten Untersuchungsgebiet wider und einen positiven Trend aufweisen. In den drei untersuchten Saisons wurde in der Fildes-Region erneut ein vergleichsweise hoher Bruterfolg verzeichnet, mit 0,46 bis 0,59 juv/bp, was deutlich über dem langfristigen Durchschnitt lag. Damit setzte sich die Stabilisierung des vormals rückläufigen Langzeittrends fort. So ist für den Zeitraum 2003/04 bis 2023/24 ein leichter, aber signifikanter Anstieg des Gesamtbruterfolgs festzustellen. Dem höchsten Bruterfolg von 0,81 juv/bp in einer Kolonie standen ungewöhnlich niedrige Bruterfolge bzw. völlige Brutausfälle in anderen Kolonien gegenüber, wobei die Gründe hierfür nicht bekannt sind, da sich die letztgenannten Kolonien in Gebieten mit geringen menschlichen Störungen befinden und somit andere Faktoren verantwortlich zu sein scheinen. Was die Reaktion des südlichen Riesensturmvogels auf menschliche Störungen angeht, so können die bisherigen Ergebnisse jedoch weitgehend bestätigt werden. Südliche Riesensturmvögel in der Fildes-Region sind aufgrund der vielfältigen menschlichen Aktivitäten einem hohen Maß an anthropogenen Störungen ausgesetzt, von denen vor allem die im Südsommer brütenden Individuen betroffen sind. Das Ausmaß der menschlichen Störungen in den Kolonien der Riesensturmvögel in der Fildes-Region hängt von der Entfernung der Brutplätze zu den Stationen und der Erreichbarkeit zu Fuß oder mit dem Boot ab. So unterscheiden sich die Kolonien in Gebieten mit geringem Störungsgrad deutlich von denen mit mittlerem und hohem Störungsgrad in Bezug auf ihren Bruterfolg. Im Einklang mit der starken Zunahme der Population und der Entwicklung des Störungsgrads nach dem Bau mehrerer Stationen in den 1980er Jahren zeigen die meisten der einzelnen Brutplätze innerhalb der Kolonien eine deutlich positive Entwicklung. So wurden zahlreiche zuvor aufgegebene Brutplätze inzwischen neu besiedelt, und auch solche, die schon länger besetzt waren, aber wo keine Jungen aufgezogen werden konnte, verweisen nun einen Bruterfolg. Dies deutet auf Gewöhnungseffekte bzw. Habituierung gegenüber bestimmten menschlichen Störungen hin. Die hier gezeigte Entwicklung der südlichen Riesensturmvogelpopulation in der Fildes-Region und anderen Gebieten der Maxwell Bay stimmt mit den publizierten Informationen über positive regionale Populationstrends überein. Für die Süd-Shetland-Inseln beispielsweise stehen zahlreichen Studien über stabile oder wachsende Populationen relativ wenige Berichte über rückläufige Trends gegenüber. Infolge des starken Populationsanstiegs scheint die Fildes-Region der zweitgrößte Brutplatz auf den Süd-Shetland-Inseln zu sein, obwohl die Gründe für den weltweiten Anstieg der Population des Südlichen Riesensturmvogels noch nicht vollständig geklärt sind. Zusammenfassend lässt sich sagen, dass im Hinblick auf die Populationsentwicklung des Südlichen Riesensturmvogels in der Fildes-Region die möglichen Einflüsse großräumiger Umweltfaktoren wahrscheinlich stärker ins Gewicht fallen als die lokalen Umweltbedingungen und negativen anthropogenen Einflüsse.

Der Kapsturmvogel in der Fildes-Region hat in den letzten Jahrzehnten einen drastischen Rückgang der Brutpopulation verzeichnet. Dieser beträgt seit Beginn der Langzeitaufzeichnungen in der Saison 1984/85 derzeit etwa 98 %. In den Saisons 2021/22 bis 2023/24 wurden erneut sehr geringe Zahlen von 3 bis 6 Brutpaaren festgestellt. Jeweils ein bis drei der 36 bekannten Brutplätze waren besetzt. Die Gründe für den beobachteten Rückgang der Kapsturmvogelpopulation in der Fildes-Region sind mangels entsprechender Daten schwer zu ermitteln. Mögliche Ursachen sind ein hoher Prädationsdruck durch Skuas, aber vor allem Umweltveränderungen wie Nahrungsmangel und schlechte Wetterbedingungen. Menschliche

Aktivitäten spielen dagegen eine untergeordnete Rolle, da sich die meisten Brutgebiete in abgelegenen und nicht besuchten Gebieten befinden. Die wenigen Einschätzungen der Populationsentwicklung aus anderen Gebieten der Antarktis sind widersprüchlich und reichen von einem rückläufigen Trend über keine Veränderung bis hin zu einer Zunahme der Population.

Nach der ersten dokumentierten Sichtung des Rußalbatros, dessen Verbreitungsgebiet sich zirkumpolar bis zu verschiedenen subantarktischen Inseln in der Nähe der antarktischen Konvergenz zwischen 46° und 55° S erstreckt, in der Fildes-Region in der Saison 1984/85 wurden fast jährlich zahlreiche weitere Sichtungen dieser Art. Der erste Brutnachweis dieser Art stammt aus der Saison 2008/09, einschließlich des Nachweises von Küken, gefolgt von wiederholtem Brutverdacht in folgenden Jahren. In den Saisons 2021/22 bis 2023/24 wurden zwar mehrmals Synchronflüge von Albatros-Paaren beobachtet, aber keine sonstigen Anzeichen, die auf eine Brut hindeuten könnten. Bislang wurden fast alle Beobachtungen an der Westküste der Fildes Peninsula gemacht, insbesondere im Bereich des bereits bekannten Brutplatzes. Im Gegensatz dazu wurden an der Ostküste der Fildes Peninsula in der Saison 2022/23 erstmals zahlreiche Beobachtungen von fliegenden Rußalbatrossen registriert. An mehreren Tagen wurden dort ein bis zwei Albatrosse beobachtet, die entweder synchrone Balzflüge durchführten oder an einem Geröllhang balzten. Es ist möglich, dass diese Beobachtungen die ersten Anzeichen dafür sind, dass sich diese Art im Osten der Fildes Peninsula etabliert.

Nach ersten Erhebungen der beiden einheimischen Sturmschwalbenarten Buntfußsturmschwalbe und Schwarzbauchmeerläufer in der Fildes-Region in den 1980er Jahren, denen eine systematische Erhebung auf der Grundlage nächtlicher Rufe während der Brutsaison in den Jahren 2003/04 bis 2005/06 folgte, wurde der Datensatz während des Untersuchungszeitraums anhand von Individuen ergänzt, die tagsüber aus den Bruthöhlen gehört wurden. Bei allen zusätzlichen Brutplätzen handelte es sich um kleinere Areale mit nur wenigen Individuen. Insgesamt sind inzwischen 344 Brutplätze erfasst worden. An den meisten Brutplätzen (39 %) brüteten beide Arten gemeinsam, gegenüber 35 % mit nur Buntfußsturmschwalbe und 26 % mit nur Schwarzbauchmeerläufer. Außerdem überwiegen kleine Brutplätze mit bis zu 10 Brutpaaren. Insgesamt wird die Brutpopulation in der Fildes-Region nach wie vor auf 3.500 bis 5.000 Buntfußsturmschwalbe und 500 bis 1.000 Schwarzbauchmeerläufer geschätzt.

Die aktuelle Erhebung der Skua-Brutpaarzahlen in der Fildes-Region ergab für den Untersuchungszeitraum eine Brutpopulation von 32 bis 55 bp brauner Skuas, 84 bis 149 bp Südpolarskuas und 5 bis 13 Misch- und Hybridpaare. Damit lag die lokale Skuapopulation nach früheren Einbrüchen im durchschnittlichen Bereich. Insgesamt blieben die Bestände der verschiedenen Skua-Paartypen im Langzeitvergleich trotz erheblicher Schwankungen stabil und zeigten keinen signifikanten zeitlichen Trend. Allerdings war der Bestand der Südpolarskua zu Beginn der Erhebung in den 1980er Jahren noch deutlich niedriger als in den Jahrzehnten zuvor. Ferner zeigte sich eine ähnliche Reaktion der beiden Arten auf bestimmte Umwelteinflüsse. So gab es eine signifikante negative Korrelation zwischen der mittleren Schneehöhe im November und Dezember und der Anzahl der Brutpaare von Braunen Skuas in der Fildes-Region. Bei den Südpolarskuas deutete sich ein solcher Zusammenhang zwar an, verfehlte aber knapp das Signifikanzniveau. Dies verdeutlicht die Bedeutung schneefreier Brutplätze für diese bodenbrütenden Arten, die im Gegensatz zu anderen Arten nicht auf Klippen, sondern meist auf niedrigen Hügeln brüten, die oft noch bis weit ins Frühjahr hinein mit Schnee bedeckt sind. Dementsprechend wurden die niedrigsten Brutpaarzahlen von Skuas in der Fildes-Region in Saisons mit hoher Schneedecke zu Beginn der Brutsaison verzeichnet. Der Bruterfolg der beiden Skua-Arten in der Fildes-Region wies im Laufe der untersuchten Jahre erhebliche Schwankungen auf und unterschied sich wiederum deutlich voneinander. Braune Skuas zogen relativ mehr Küken erfolgreich auf als Südpolarskuas. Ein zeitlicher Trend konnte für keinen Paartyp festgestellt werden. Die räumliche Verteilung der Skua-Brutpaare in den drei untersuchten Saisons entspricht weitgehend den Daten aus den Vorjahren, was eine stärkere Konzentration von Braunen Skuas in der Nähe von Pinguinkolonien und der Küste unterstreicht. Dagegen brüten im Südosten der Fildes Peninsula deutlich mehr Südpolarskuas. Neben der Zahl der Brutpaare wird in der Fildes-Region seit der Saison 2006/07 auch die Zahl der gemeldeten Skua-Territorien erfasst, die zwar besetzt waren, in denen keine Brut begonnen wurde. Es wird deutlich, dass die Gesamtzahl der im Sommer im Brutgebiet anwesenden Skua-Paare deutlich weniger schwankte, als wenn nur die Brutpaare betrachtet wurden. Dies deutet eindeutig auf eine größere Abhängigkeit der Südpolarskuas von weiteren Umweltfaktoren hin, die die Entscheidung für oder gegen eine Brut, beeinflussen. In allen drei untersuchten Saisons wurde in der Mehrzahl der Stationen auf der Fildes Peninsula erneut eine aktive Fütterung der Skuas durch das Stationspersonal beobachtet bzw. durch entsprechende Funde von frischen Nahrungsresten auf Nestern in Stationsnähe belegt, obwohl diese Praxis den rechtlich verbindlichen Vorgaben des Madrider Protokolls widerspricht.

Während der Brutvogelerfassung in der Fildes-Region in den Saisons 2021/22 bis 2023/24 wurden insgesamt 46, 53 bzw. 51 Brutpaare von Dominikanermöwen erfasst. Diese Werte liegen deutlich unter dem langjährigen Durchschnitt, weshalb sich der negative Trend der Population im Langzeitvergleich seit Beginn der Erfassung in der Saison 1984/85 verstärkt, verursacht durch die höheren Brutpaarzahlen in den 1980er und 1990er Jahren. Im Zeitraum zwischen 2004/05 und 2023/24 blieb der Bestand an Dominikanermöwen jedoch stabil, wenngleich erhebliche Schwankungen zu verzeichnen waren. Eine vormals sich andeutende signifikante negative Korrelation zwischen der Anzahl der Brutpaare der Dominikanermöwe in der Fildes-Region und der lokalen Schneedecke in den Frühjahrsmonaten Oktober und November konnte nun nachgewiesen werden. Dies zeigt, dass ein schneereiches Frühjahr, in dem die Brutplätze zu dem für die Brutentscheidung relevanten Zeitpunkt noch schneebedeckt sind, zu einer geringen Anzahl von Brutpaaren in der jeweiligen Brutsaison führt. Im Gegensatz zu früheren Ergebnissen zeigten die aktuellen Daten keine Korrelation zwischen der Anzahl der Brutpaare von Dominikanermöwen in der Fildes-Region und dem zyklischen atmosphärisch-ozeanischen Phänomen „El Niño-Southern Oscillation“.Obwohl in Saisons mit einem mäßigen bis sehr starken El Niño eine deutlich geringere Zahl von Brutpaaren verzeichnet wurde, während in La-Niña-Jahren relativ viele Möwen in dem Gebiet brüteten, ließ sich kein Zusammenhang zwischen diesem Klimaparameter und der lokalen Seemöwenpopulation nachweisen.

Die lokale Population der Antarktisseeschwalben kann aufgrund meteorologischer Bedingungen stark schwanken. Außerdem neigen brütende Seeschwalben aufgrund ihrer hohen Störanfälligkeit dazu, ihre Brutplätze während der Brutsaison zu verlegen oder aufzugeben. Trotz der sich daraus ergebenden methodischen Schwierigkeiten kann ein langfristiges Monitoring jedoch wertvolle Informationen über die allgemeine Entwicklung der Population dieser Art in einem Gebiet liefern. In den Saisons 2021/22 und 2022/23 wurden nur 73 bzw. 125 Brutpaare der Küstenseeschwalbe gezählt, was die beiden niedrigsten Werte seit Beginn des Langzeitmonitorings in der Saison 1984/85 darstellen. In der Saison 2023/24 erholte sich der Bestand dieser Art leicht auf 196 Brutpaare. Damit lag der Bestand der Antarktisseeschwalbe in den drei untersuchten Brutsaisons deutlich unter dem langjährigen Durchschnitt von 423 ± 262 BP. Im Langzeitvergleich seit Beginn der Erfassung in der Saison 1984/85 lässt sich trotz erheblicher Bestandsschwankungen ein deutlicher Abwärtstrend des Seeschwalbenbestandes in der Fildes-Region feststellen. Betrachtet man die Anzahl der Brutpaare zwischen den Saisons 2000/01 und 2023/24, so wird dieser negative Trend noch deutlicher. In allen drei untersuchten Saisons wurde an den meisten Brutplätzen in der Fildes-Region mehr als ein Brutpaar festgestellt. Die durchschnittliche Anzahl der Paare pro Brutplatz betrug während des Untersuchungszeitraums 3,87 ± 0,58. Der Anteil der verschiedenen Größenklassen an den Brutplätzen variierte von Jahr zu Jahr erheblich. Wie in den Vorjahren war der Anteil der Einzelbrüter mit etwa 49 % bis 52 % relativ hoch. Große Kolonien mit mehr als 20 und bis zu 300 Brutpaaren wurden jedoch nicht mehr erfasst. Darüber hinaus zeigt sich der Einfluss des Menschen in der Fildes-Region auf die Antarktisseeschwalbe in Form der Verwendung von anthropogenem Material wie Metall, Knochen- und Glasstücken oder Obstkernen zum Nestbau. Die Ursache für den starken Rückgang der Brutpaarzahl der Antarktisseeschwalben im Gebiet der Fildes-Region ist nicht bekannt. Inwieweit der Bestandsrückgang auch in anderen Brutgebieten, einschließlich der subantarktischen Inseln im Indischen und Pazifischen Ozean, zu beobachten ist, ist aufgrund der wenigen veröffentlichten vergleichenden Bestandsdaten nicht bekannt. Zudem stehen Aussagen über stabile oder wachsende Populationen Informationen über abnehmender Populationen gegenüber.

Eine kleine Anzahl von Weißgesichtsscheidenschnäbeln wurde regelmäßig im Untersuchungsgebiet beobachtet. Direkte Brutnachweise sind jedoch aufgrund der Unerreichbarkeit der Brutplätze selten. Daher wurden verschiedene Parameter, wie z. B. wiederholte Beobachtungen von zwei Altvögeln am selben Ort, herangezogen, um auf eine Brut zu schließen. Bislang wurden an der Westküste der Fildes-Region nur einige wenige Brutplätze identifiziert. In den Saisons 2021/22 und 2023/24 gab es dort mehrere Beobachtungen einzelner Individuen sowie einen Brutnachweis aufgrund der Beobachtung eines mausernden Kükens zusammen mit zwei Altvögeln in der Saison 2022/23. Im Gegensatz dazu schien ein anderer langjähriger Brutplatz im Südwesten der Fildes Peninsula unbesetzt zu sein.

Die Blauaugenscharbe galt viele Jahre lang als potenzieller Brutvogel in der Fildes-Region. Trotz regelmäßiger Beobachtungen von Schwärmen von bis zu 30 Individuen in dem Gebiet wurden seit den 1980er Jahren keine Bruten dieser Art mehr festgestellt. Brutversuche im Untersuchungsgebiet, wie zuletzt in den Saisons 2015/16 und 2019/20, wurden in den Saisons 2021/22 und 2022/23 nicht registriert. In der Saison 2023/24 wurde dagegen zum ersten Mal seit langer Zeit wieder eine erfolgreiche Brut dieser Art registriert. Insgesamt vier Brutpaare brüteten auf einem Felsen im Südosten von Ardley Island inmitten von zahlreichen brütenden Eselspinguinen. Im Februar 2024 wurden insgesamt 4 Küken in 3 Nestern gezählt.

Daten über Sichtungen von Irrgästen und Durchzüglern in der Antarktis können auf eine potenzielle Ausweitung des Brutgebiets subantarktischer Arten hinweisen und wertvolle Hinweise auf ungewöhnliche Umweltbedingungen liefern, die es diesen Vögeln ermöglichen, in Regionen außerhalb ihres üblichen Verbreitungsgebiets zu gelangen. Während des aktuellen Untersuchungszeitraums wurden zahlreiche Vertreter nicht-heimischer Vögel erfasst, in einigen Fällen ausschließlich als Totfunde. Die häufigsten Totfunde betrafen Seevogelarten, die im Südlichen Ozean häufig beobachtet werden, aber in der Fildes-Region nicht als Brutvögel vorkommen, insbesondere Entensturmvögel, Schneesturmvögel und Silbersturmvögel. Kaiserpinguine wurden in allen drei Saisons des Untersuchungszeitraums beobachtet. Andere Vogelarten wurden entweder in zwei (Kuhreiher, Weißbürzelstrandläufer, Bairdstrandläufer) oder in einer der drei untersuchten Saisons (Goldschopfpinguin, Küstenseeschwalbe) beobachtet. Königspinguine wurden im Untersuchungsgebiet nicht beobachtet, obwohl es in benachbarten Pinguinkolonien häufig zu Beobachtungen oder Brutversuchen kam.

Im Rahmen der Fortsetzung des Langzeitmonitorings wurden in den Saisons 2021/22 bis 2023/2024 monatliche Robbenerhebungen an der Küste der Fildes Peninsula und auf Ardley Island durchgeführt. Somit liegen Daten über die Robbenpopulation im Südsommer in der Fildes-Region für bis zu 28 Jahre vor. Die monatlichen Veränderungen in der lokalen Robbenpopulation entsprechen denen, die in früheren Studien beschrieben wurden, d. h. die Zahl der registrierten Südlichen Seeelefanten ist im Januar am höchsten, während die Zahl der

Weddellrobben im Dezember und Januar am höchsten ist, wobei ihre Zahl zum Ende des Südsommers deutlich abnimmt. Im Gegensatz dazu erreicht die Zahl der Antarktischen Seebären ihr Maximum erst im Februar. Bei den monatlichen Robbenzählungen in den Saisons 2021/22 bis 2023/24 wurden insgesamt zwischen 1.899 und 2.565 Südliche Seeelefanten erfasst. Diese Gesamtzahlen lagen damit im langfristigen Durchschnitt seit 2000/01 von 2.115 ± 324, wobei die Gesamtzahl für die Saison 2023/24 die zweithöchste in diesem Zeitraum war. Wie in den Vorjahren wurde die höchste Zahl von Individuen, die das Fell wechselten, erneut im Januar registriert. Bemerkenswert ist der Minimalwert von nur 339 Südlichen See-Elefanten im Februar 2023, der niedrigste Wert für diesen Monat seit Beginn des regelmäßigen Monitorings in der Saison 1982/83. Für den Zeitraum 2000/01 bis 2023/24 sind jedoch keine monatlichen Trends erkennbar, was auf eine Stabilisierung der Population nach dem Rückgang in den 1980er Jahren hindeutet. Bei den aktuellen Zählungen wurden auch alle Seeelefanten-Liegeplätze mit mindestens zehn Individuen erfasst. Die größte Konzentration von Südlichen Seeelefanten wurde im Januar im nördlichsten Teil der Fildes Peninsula mit 144, 239 bzw. 189 Individuen dokumentiert, wobei letztere eine der höchsten Individuenzahlen seit Beginn des Robbenmonitorings in der Fildes-Region darstellen (Maximum: Januar 2012 mit 255 Individuen). Die Analyse der räumlichen Daten, die sich nunmehr über 17 Jahre erstrecken, zeigt erhebliche saisonale und jährliche Schwankungen. Die räumliche Verteilung der Ruheplätze zeigte keine signifikanten Veränderungen im Vergleich zu früher veröffentlichten Ergebnissen. Räumliche Veränderungen oder ein Rückgang der Ruheplätze aufgrund menschlicher Störungen, wie sie in der benachbarten Admiralty Bay festgestellt wurden, sind auf der Fildes Peninsula nicht erkennbar. Obwohl dieses Gebiet insgesamt eine viel höhere Konzentration menschlicher Aktivitäten aufweist, befinden sich die Ruheplätze der Seeelefanten ausschließlich an der Westküste, die im Gegensatz zur Umgebung der Station kaum von direkten menschlichen Störungen betroffen ist, wie sie in der Nähe der Stationen regelmäßig zu beobachten sind. Eine Ausnahme bildet ein größerer Ruheplatz in einer Bucht, die im Südsommer regelmäßig von Touristen und Stationsmitgliedern besucht wird. Allerdings wurden auch hier keine räumlichen oder quantitativen Veränderungen festgestellt.

Die Gesamtzahl der auf der Fildes Peninsula und Ardley Island in den Saisons 2021/22 bis 2023/24 erfassten Weddellrobben lag bei 74, 94 bzw. 147 Individuen und damit teilweise deutlich unter dem langjährigen Durchschnitt von 151 ± 53 Individuen seit der Saison 2000/01. Wie in den Vorjahren nahm die Zahl der im Untersuchungsgebiet beobachteten Weddellrobben gegen Ende des Südsommers ab. Darüber hinaus wurden sowohl im Januar und Februar als auch im Dezember 2022 die niedrigsten Zahlen an Weddellrobben für den jeweiligen Monat seit Beginn der regelmäßigen Erhebungen in der Saison 1982/83 verzeichnet. Im Februar 2022 wurden zum ersten Mal überhaupt keine Robben erfasst. Dementsprechend wurde in der Saison 2021/22 die niedrigste Gesamtzahl dieser Art, die jemals im Südsommer im Untersuchungsgebiet erfasst wurde, und in der Saison 2022/23 die zweitniedrigste gezählt. Im Durchschnitt wurden in den 1980er Jahren mehr Weddellrobben in der Fildes-Region beobachtet als im Zeitraum 2000/01 bis 2023/24. Während die Gesamtzahl der Robben und die Zahl der Robben im Dezember und Februar in diesem Zeitraum stabil blieben, ist für den Monat Januar ein rückläufiger Trend zu beobachten.

Sowohl die monatliche als auch die Gesamtzahl der während des Südsommers in den Saisons 2021/22 bis 2023/24 gezählten Antarktischen Seebären wiesen bemerkenswerte Schwankungen auf. In den Saisons 2021/22 und 2023/24 wurden insgesamt 815 bzw. 456 Individuen gezählt, wobei der langfristige Durchschnitt seit 2000/01 bei 713 ± 399 Individuen lag. Dagegen wurden in der Saison 2022/23 nur 275 Tiere gezählt. Dennoch war die Zahl der Seebären in den 1980er Jahren deutlich geringer als heute. Aufgrund der bemerkenswerten Schwankungen in den letzten beiden Jahrzehnten lassen sich keine signifikanten Trends für die lokale Population dieser Art im Südsommer zwischen den Saisons 2000/01 und 2023/24 erkennen. Allerdings ist seit der Saison 2014/15 ein starker Rückgang der Gesamtzahlen zu beobachten. In der Saison 2021/22 wurde ein leukistischer Seebär im Untersuchungsgebiet dokumentiert. Insgesamt wurden bei den monatlichen Robbenzählungen in den Saisons 2000/01 bis 2023/24 bei 82 Zählungen in den Sommermonaten nur zwei leukistische Exemplare von insgesamt 15.381 Individuen erfasst. In den 1980er Jahren hingegen wurden bei 131 Zählungen, die zweimal im Monat über das ganze Jahr verteilt durchgeführt wurden, bei insgesamt 3.266 erfassten Tieren keine leukistischen Individuen dokumentiert. Beobachtungen von anderen atypischen Fellfärbungen sind für die Fildes-Region nicht bekannt.

Während des Südsommers der Saisons 2021/22 bis 2023/24 wurde auf der Fildes Peninsula und Ardley Island erneut eine kleine Anzahl neugeborener Robben beobachtet. Obwohl die Hauptfortpflanzungszeit für Seeelefanten und Weddellrobben außerhalb des Südsommers liegt und die meisten Jungtiere das Gebiet bereits verlassen haben, wurde im Januar 2023 ein einzelnes Seeelefantenjungtier an der Nordwestküste der Fildes Peninsula beobachtet. Darüber hinaus wurden in der Saison 2021/22 und 2023/24 jeweils vier Jungtiere des Antarktischen Seebären und in der Saison 2022/23 drei Jungtiere registriert.

Brutvogelmonitoring in den eisfreien Gebieten der Maxwell Bay

Aufgrund erheblicher logistischer Einschränkungen konnten nicht alle relevanten eisfreien Gebiete während des Untersuchungszeitraums besucht werden. Daten über Brutvögel für einige Gebiete wurden jedoch freundlicherweise von tschechischen, koreanischen bzw. ukrainischen Wissenschaftlern zur Verfügung gestellt. Daher können die in früheren Studien aufgeführten Daten nur begrenzt ergänzt werden. Auf der Grundlage der verfügbaren Daten für Barton Peninsula kann festgestellt werden, dass die Population der Zügelpinguine während des Erhebungszeitraums keinen zeitlichen Trend aufwies, obwohl die Zahl der Brutpaare dieser Art in den letzten zehn Jahren abgenommen hat. Im Gegensatz dazu hat die Population der Eselspinguine weiter zugenommen. Seit der Saison 2018/19 ist der Eselspinguin die am häufigsten vorkommende Pinguinart in dieser Kolonie. Wie auf Ardley Island wurde auch hier in der Saison 2023/24 die höchste Anzahl von Brutpaaren dieser Art seit Beginn der Langzeituntersuchung verzeichnet. Auf der kleinen Halbinsel Green Point im Nordosten der Fildes Peninsula, die zu Fuß erreichbar ist, wurde während des Untersuchungszeitraums eine geringe Anzahl von Brutpaaren der Dominikanermöwe registriert. In der Saison 2021/22 brütete dort ein Paar Südpolarskuas. Jüngste Veröffentlichungen über die Pinguinpopulation der Stranger-Point-Kolonie auf der Potter Peninsula weisen auf einen starken Rückgang der Adélie-Pinguinpopulation in ausgewählten Nestgruppen der Stranger-Point-Kolonie zwischen den Saisons 1995/96 und 2022/23 um insgesamt 89,8 % hin. Auch die Zahl der Adéliepinguine auf Ardley Island ging im gleichen Zeitraum um 85 % zurück. Die Zahl der flüggen Küken sank am Stranger Point um 93,1 % und auf Ardley Island um 82,8 %. Diese vergleichenden Ergebnisse deuten auf eine sehr einheitliche regionale Entwicklung als Reaktion auf sich verändernde Umweltbedingungen hin. Der Bruterfolg zeigte in beiden Kolonien keine abnehmende Tendenz. Inwieweit sich der steigende Trend der Eselspinguinpopulation in der Kolonie am Stranger Point in den letzten Jahren fortgesetzt hat, kann aufgrund mangelnder Daten nur vermutet werden, da er der überregionalen Entwicklung in der WAP-Region entspricht. Auch Aussagen über das mögliche Vorkommen von Zügelpinguinen sind mangels verfügbarer Daten derzeit nicht möglich. Für den Königspinguin wurden für die Potter Peninsula in den Saisons 2011/12 bis 2020/21 nun jährliche Bruten von einem Paar dokumentiert. Obwohl keines der beiden bisher geschlüpften Küken erfolgreich aufgezogen wurde, werden die laufenden Brutversuche, auch in der benachbarten Admiralty Bay, als beginnende Ausdehnung des Verbreitungsgebiets der Art nach Süden interpretiert. Ausgehend von den verfügbaren Daten für die Stansbury

Peninsula, Nelson Island, gab es im Untersuchungszeitraum deutlich weniger Brutpaare des Südlichen Riesensturmvogels als im vorangegangenen Jahrzehnt. In der Saison 2022/23 wurden Anfang Februar nur 3 Küken dokumentiert, und in der darauffolgenden Saison 2023/24 wurde im Januar 2024 kein einziges Küken erfasst. Die Ursache für den aktuellen Rückgang ist nicht bekannt, zumal in der unmittelbar benachbarten Region Fildes der höchste Brutvogelbestand der letzten vier Jahrzehnte verzeichnet wurde. Der langfristige Trend des Brutbestands des südlichen Riesensturmvogels auf der Stansbury Peninsula ist jedoch nach wie vor positiv, was möglicherweise auf eine zwischenzeitliche Verlagerung des Brutplatzes aufgrund anthropogener Störungen hinweist, wie sie auch für die Fildes-Region nachgewiesen wurde. Wie die Zahl der Brutpaare zeigte auch der Bruterfolg des Südlichen Riesensturmvogels auf Stansbury Peninsula einen Rückgang. Die an die Stansbury Peninsula angrenzenden Kolonien der Fildes-Region spiegeln den aktuellen Einbruch der Brutpaarzahlen und des Bruterfolgs nicht wider, mit Ausnahme einer sehr kleinen Kolonie in der Nähe, wo in den letzten beiden Jahren ebenfalls ein Rückgang der Brutpaarzahlen und ein vollständiger Verlust von Bruten zu verzeichnen war. Alle anderen vergleichbaren Kolonien in der Nachbarschaft zeigten keine solche Entwicklung, sondern im Gegenteil einen überdurchschnittlichen Bruterfolg in den Saisons 2022/23 und 2023/24. Der Brutbestand der Skuas auf der Stansbury Peninsula wurde nur in der Saison 2023/24 erfasst. Die Zählung im Januar 2024 ergab insgesamt 2 bp brauner Skuas und 11 bp Südpolarskuas. Darüber hinaus wurden 3 Territorien (1 braune Skua, 2 Südpolarskua) und 3 ältere Nester (1 braune Skua, 2 Südpolarskua) erfasst. In Übereinstimmung mit der Populationsentwicklung des Kapsturmvogels auf der Fildes Peninsula wurde der starke Rückgang dieser Art - trotz der Beobachtung einiger fliegender Individuen - auch für die Stansbury Peninsula bestätigt, wenngleich er nicht quantifiziert werden konnte.

Vorkommen der Antarktischen Perlwurz (*Colobanthus quitensis*) in der Fildes Region

In der Fildes-Region wurden bisher nur wenige Standorte der Antarktischen Perlwurz dokumentiert. Die Bestände, die in den 2000er Jahren erfasst wurden, waren jedoch nach wenigen Jahren vollständig erloschen. Ein seit 2018 bekannte Bestand dieser Pflanze wurde kontrolliert und dokumentiert, um die Größe und den Zustand der Polster mit früheren Daten zu vergleichen. So ist zu erkennen, dass dieser Bestand sowohl abgestorbene als auch zusätzlich bewachsene Bereiche aufwies. Insgesamt erschien der Bestand etwas größer als 2018. Ein weiterer kleiner Bestand der Antarktischen Perlwurz, der erstmals in der Saison 2020 im äußersten Norden der Fildes Peninsula erfasst wurde, wurde im Untersuchungszeitraum nicht mehr gefunden.

Klimatische Veränderungen im Untersuchungsgebiet

Die Jahresmitteltemperatur im Untersuchungsgebiet weist zwischen 1969 und 2023 einen signifikant positiven Trend auf und steigt um 1,21 °C. Dies entspricht einer jährlichen Erwärmung von 0,022 °C und einem weiteren Anstieg im Vergleich zu früheren Zeiträumen. Der Jahresniederschlag im Untersuchungsgebiet zeigte eine hohe Variabilität ohne einen signifikanten Trend. Außerdem konnte die zuvor gezeigte negative Korrelation zwischen Jahresniederschlag und Jahresmitteltemperatur bestätigt werden. Darüber hinaus zeigte die mittlere Temperatur der einzelnen Jahreszeiten einen signifikanten positiven Trend für Winter und Herbst, nicht jedoch für Sommer und Frühling. Die Temperaturanomalie, also die Abweichung vom Jahresmittel, verdeutlicht, dass nach einer Phase kälterer Monate zwischen den Saisons 2009/10 und 2015/16 seither das Auftreten warmer Monate sowohl im Winter als auch im Sommer deutlich zugenommen hat.

Gletscherrückzugsgebiete ausgewählter Bereiche der Maxwell Bay

Die Ergebnisse der Veränderung der Gletscherflächen in den an die Maxwell Bay angrenzenden Regionen verdeutlichen den beträchtlichen horizontalen Rückzug der Gletscherfronten von 1956 bis 2023, der in den letzten Jahren zugenommen hat und besonders im Bereich der Gezeitengletscher deutlich wird. Der derzeitige starke Gletscherrückgang in den letzten Jahren wird auf die hohen Temperaturen sowohl im Sommer als auch im Winter zurückgeführt. So ging die Fläche des Bellingshausen Dome/Collins Glacier von 12,29 km^2 auf 9,14 km^2 um 35 % zurück. Zudem zeigte die Gletscherfront einen Rückzug von ca. 375 m in 67 Jahren, was einem jährlichen Rückzug von 5,6 Metern entspricht. Nach einer Stagnation zwischen 2006/07 und 2011/12 ist auf den Halbinseln Weaver und Barton zwischen 2012 und 2023 wieder ein deutlicher Rückzug der Gletscherfront zu verzeichnen. Die Barton Peninsula hat mit 1,41 km^2 eine sehr große eisfreie Fläche gewonnen. Insgesamt nahm die eisfreie Fläche der Barton Peninsula von 6,23 km^2 im Jahr 1956 auf 11,06 km^2 im Jahr 2023 um 44 % zu. Folglich erlebte diese Halbinsel mit 1123,7 Metern und 16,8 m/a den höchsten Gesamtgletscherrückgang aller untersuchten Gebiete. Zusätzlich zu den landgestützten Gletschern zeigten zwei „Gezeitengletscher“ in den markanten Buchten von Collins Harbour und Marian Cove einen Gletscherrückgang von 1,70 km^2 bzw. 2,14 km^2 in Collins Harbour. Darüber hinaus wurde für die Fildes Peninsula eine rezente Besiedlung von Gletscherrückzugsgebieten durch Seevögel mit 122 bekannten Brutplätzen von vier Arten (Skuas, Dominikanermöwe, Antarktisseeschwalbe und Buntfußsturmschwalbe) auf einer Gesamtfläche von 3,15 km^2 dokumentiert.

Dokumentation der Permafrostdegradation in der Fildes-Region

Als Folge des globalen Temperaturanstiegs wird in den Polarregionen eine zunehmende Degradation des Permafrosts festgestellt. Zum ersten Mal wurde in der Saison 2022/23 an mehreren Stellen in der Fildes-Region eine deutlich sichtbare Degradation des Permafrosts in Form von Tauwassereinbrüchen festgestellt. Diese Beobachtungen werden auf die ungewöhnlich hohen Temperaturen in dieser Saison zurückgeführt.

Eintrag von Meeresmüll in das Untersuchungsgebiet

In den letzten Jahren ist das Bewusstsein für die Risiken des Eintrags von Kunststoffen in die Umwelt deutlich gestiegen, da diese aufgrund der weit verbreiteten Verwendung von Kunststoffen aller Art, ihrer Flexibilität und Haltbarkeit sowie der Schadstoffe, die sie enthalten oder die an ihnen anhaften, negative Auswirkungen auf die Umwelt haben können. Mit der Fortführung der aktuellen systematischen Erhebung von Meeresmüll in der Fildes-Region liegen nun fünf Jahre Daten über Meeresmüll in einer ausgewählten Bucht an der Westküste der Fildes Peninsula vor. Die erfassten Dichten von 0,85 und 0,90 Objekten/Meter in den Saisons 2023/23 und 2023/24 waren die höchsten seit Beginn der systematischen Erfassung im Jahr 2019. Ein Vergleich mit den Daten ab der Saison 2018/19 zeigt somit eine Zunahme der Dichte. Während des gesamten Untersuchungszeitraums überwiegt der Anteil der Gegenstände aus Kunststoff mit rund 98 % deutlich gegenüber denjenigen aus Holz, Glas, Metall oder anderen Materialien. Insgesamt dominierten kleine Gegenstände die Mehrheit der Sammlungen, gefolgt von mittelgroßen und großen Objekten. Während des Untersuchungszeitraums wurde am beprobten Strandabschnitt eine beträchtliche Menge an Styropor und anderen Schaumstoffen (42 %) gefunden, obwohl die Einfuhr sowie Verklappung von Styropor in der Antarktis nach den rechtsverbindlichen Vorschriften des Madrider Protokolls und des SCAR-Verhaltenskodex untersagt sind. Darüber hinaus wurde bei durchschnittlich 11 % der Funde ein eindeutiger Zusammenhang mit Fischereiaktivitäten festgestellt (z. B. Netze, Kunststoffseile oder -schnüre, Schwimmer, Verpackungsbänder, usw.). Verbranntes Plastik/Pyroplastik wurde während des gesamten Untersuchungszeitraums festgestellt und stammt wahrscheinlich von der Verklappung von Müllverbrennungsrückständen an Bord von Schiffen. Plastikflaschen, ihre

Verschlüsse oder Teile davon machten wiederum einen bedeutenden Anteil des Meeresmülls aus. Auffällig war, dass es zahlreiche kleinere Wasserflaschen desselben Typs und Herstellers gab. Unter dem gesammelten Meeresmüll befanden sich mehrere industrielle Kunststoffpellets. Die jährlichen Akkumulationsraten in der Fildes-Region waren relativ hoch und meist höher als in anderen Gebieten auf den Süd-Shetland-Inseln, der Antarktischen Halbinsel und den subantarktischen Inseln. Aufgrund der örtlichen Topographie, der nordwestlichen Lage zur Drake-Passage sowie der vorherrschenden Westwinde kommt es an dem untersuchten Strandabschnitt zu einem beträchtlichen Eintrag von aus dem Südpolarmeer angeschwemmtem Meeresmüll. Lokale Strömungen und Gezeiten haben ebenfalls einen Einfluss, wenn auch nur in geringem Umfang. Im Gegensatz dazu ist die Menge an Meeresmüll auf der Ostseite der Fildes Peninsula in Richtung Maxwell Bay deutlich geringer. Es ist davon auszugehen, dass der Eintrag von Meeresmüll über die ansässige Station und über den intensiven Schiffsverkehr in diesem Gebiet erfolgt.

Einfluss von anthropogenem Material auf Seevögel in der Fildes-Region

Verschiedene negative Auswirkungen von anthropogenem Material auf Tiere in terrestrischer, aquatischer und mariner Umgebung wurden bereits vielfach dokumentiert. Analysen von Magenanalysen tot aufgefundener Vögel aus der Fildes-Region ergaben, dass in etwa 46,2 % der untersuchten Tiere anthropogenes Material nachgewiesen wurde, darunter Buntfußsturmschwalben, Adélie- und Eselspinguine, ein Küken der Südpolarskua und ein erwachsener Südlicher Riesensturmvogel. Darüber hinaus wurden Skua-Pellets mit unverdaulichen Überresten von Seevögeln gesammelt und analysiert. Im Durchschnitt wurden in ca. 37,4 % aller Skua-Pellets eindeutig identifizierbare Kunststoffpartikel gefunden. Die Häufigkeit von Mikroplastik war je nach Art sehr unterschiedlich. Buntfußsturmschwalben enthielten mit einer Häufigkeit von ca. 28,3 % der Proben am meisten Mikroplastik, einschließlich Industriegranulat, gefolgt von Schwarzbauch-Sturmschwalben mit ca. 30,4 %. Auffallend war die außergewöhnlich hohe Häufigkeit von Mikroplastikpartikeln (ca. 87,1 %) in Skua-Pellets, die Reste von Prionen enthielten. Im Durchschnitt wurden je nach Vogelart zwischen 1,6 und 4,6 Partikel pro mikroplastikhaltigem Pellet gefunden. Die höchste Anzahl von 22 Mikroplastikpartikeln wurde in einem Skua-Pellet mit Resten von Entensturmvögeln gefunden. Pellets dieser Art enthielten im Durchschnitt deutlich mehr und größere Partikel, und auch die mittlere Kunststoffbelastung pro Skua-Pellet war je nach Art unterschiedlich. Unter den insgesamt 168 gefundenen Mikroplastikpartikeln befand sich auch ein hoher Anteil an Industriegranulat (ca. 22 %), das auch im Mageninhalt von Sturmschwalben und an der Küste des Untersuchungsgebiets gefunden wurde. Bei den 40 im Mageninhalt und in den Skua-Pellets gefundenen Partikeln Industriegranulat überwogen die Farben Gelb und Orange. Betrachtet man beide Farbverteilungen, so fällt auf, dass dies im Wesentlichen dem Farbspektrum der entlang der Küstenlinie gesammelten Industriegranulate entspricht, die ebenfalls überwiegend helle bzw. gelbliche Farbtöne aufwiesen. Weitere Beispiele für den Einfluss von anthropogenem Material waren das Vorhandensein von Langleinenhaken, Glas, Plastikfolie oder Schnüren in Vogelnestern.

Eingeschleppte nicht-heimische Arten

Da biologische Invasionen als eine der größten Herausforderungen für die Erhaltung der Arten und die Gesunderhaltung der Ökosysteme und ihrer Strukturen gelten, insbesondere in geografisch und evolutionär isolierten Ökosystemen, wurde das Augenmerk auf das Vorhandensein nicht heimischer einführender Arten gerichtet. Für die Fildes Peninsula wurden bereits mehrere Nachweise von nicht einheimischen Gräsern und Insekten erbracht. Wiederholte Kontrollen des bekannten Standorts von nicht-einheimischen Gräser im Bereich zweier antarktischer Stationen ergaben keine Hinweise auf ein Wiederauftreten der Pflanzen, die in den 2000er Jahren manuell entfernt wurden. Im Januar 2022 wurde im Bereich einer anderen Station unter der Treppe zu einem Wohngebäude ein auffälliges Grasbüschel entdeckt. Es wurde einschließlich des umgebenden Bodens gemäß den bestehenden Empfehlungen entfernt. Leider liegt derzeit noch kein Ergebnis der Artbestimmung vor. Außerdem wurde eine einzelne Eichel in einem Bahnhofsbereich an der Hauptroute gefunden, die von den Touristen des Air-Cruise-Programms auf ihrem Weg zwischen Flughafen und Strand genutzt wird. Dies deutet darauf hin, dass Samen oder Ausbreitungseinheiten nichtheimischer Pflanzen - auch größerer Exemplare - immer noch durch menschliche Aktivitäten in das Untersuchungsgebiet gelangen können. Was die eingeschleppten nichtheimischen Insekten betrifft, so wurden in verschiedenen Stationsgebäuden immer wieder Exemplare der Wintermücke *Trichocera maculipennis* beobachtet, die in der Fildes-Region inzwischen als hartnäckig gilt, entweder fliegend oder in Klebefallen. Darüber hinaus wurden bis zu drei Individuen dieser Art in der Klebefalle gefunden, die in einer Feldhütte auf Ardley Island aufgestellt war. Zahlreiche Exemplare der Wintermücke wurden auch außerhalb der Stationsgebäude nachgewiesen.

Offene Fragen, weiterer Forschungsbedarf und Weiterentwicklung des Monitoringprogramms

Der hohe Wert langfristiger Monitoringprogramme für Fauna und Flora ist weithin anerkannt, da sie nützliche Erkenntnisse über die Auswirkungen veränderter Umweltbedingungen auf das Ökosystem liefern und auch dazu beitragen können, künftige Entwicklungen zu bewerten und künftige Schutzmaßnahmen zu entwickeln. Dies ist besonders wichtig in Gebieten, die in den letzten Jahrzehnten infolge der globalen Erwärmung große Umweltveränderungen erfahren haben, wie z. B. die westliche Antarktische Halbinsel. Die meisten Bestandserhebungen von Seevogel- und Robbenpopulationen in der Antarktis beruhen auf wenigen Zählungen und haben aufgrund der häufigen natürlichen Bestandsschwankungen nur eine sehr begrenzte Aussagekraft über die tatsächliche Entwicklung. Die langfristigen, weitgehend kontinuierlichen Datenreihen aus dem systematischen Langzeitmonitoring in der Fildes-Region sind daher von größtem Wert. Darüber hinaus bietet die Lage des Untersuchungsgebietes mit seiner hohen menschlichen Aktivität die Möglichkeit, die Auswirkungen bestimmter anthropogener Einflüsse, wie z.B. menschliche Störungen, Meeresmüll oder eingeschleppte nichtheimische Arten, auf die antarktischen Ökosysteme zu beurteilen. Darüber hinaus zeigen jüngste Ergebnisse, dass direkte menschliche Einflüsse im Rahmen von Monitoringprogrammen für bestimmte Seevogelarten in anthropogenen Gebieten berücksichtigt werden sollten, um eine zuverlässige Interpretation von Populationsdaten zu gewährleisten und Indikatorarten zu definieren. Dies kann dazu beitragen, Schutzmaßnahmen zu ergreifen, die zum Erhalt der antarktischen Ökosysteme erforderlich sein könnten. Aus den oben genannten Gründen ist eine Fortsetzung der langfristigen Erhebungen zu den vorgestellten Themen dringend zu empfehlen. Darüber hinaus wird eine verstärkte Nutzung von Fernerkundungsdaten empfohlen, um logistische Einschränkungen zu vermeiden und die Zahl der Besuche in Seevogelkolonien zu verringern. Darüber hinaus besteht Forschungsbedarf zu verschiedenen Themen, z.B. zur Untersuchung der Gründe für räumliche Veränderungen in der Pinguinkolonie auf Ardley Island, zur Beobachtung des Auftretens neuer Tauwassereinbrüchen und der weiteren Entwicklung der Permafrostdegradation.

1 Introduction

Antarctica is considered to be one of the last relatively untouched areas on earth. It is defined as the area south of 60 degrees south latitude. Due to its great global importance as a result of its unique biological, geological and hydrological characteristics, the Antarctic Treaty was adopted in 1959. This forms the foundation for the protection of Antarctica on the basis of peaceful use, emphasizing its global value for science. Increasingly, Antarctica and the surrounding Southern Ocean are under pressure due to the cumulative impacts caused by climate change, pollution, fishing, tourism and a variety of other human activities with consequences for both local polar ecosystems and the regulation of global climate and sea level rise (Chown & Brooks, 2019; Siegert et al., 2020; Cavanagh et al., 2021). As a result of increased attention to the growing negative impacts of human activities in Antarctica, the Protocol on Environmental Protection to the Antarctic Treaty (hereafter Madrid Protocol), which came into force in 1998, was adopted by the Antarctic Treaty states to ensure common binding minimum standards for environmental management. Within the Antarctic, the strongest current changes in recent decades can be observed on the Antarctic Peninsula (Kejna, 2003; Meredith & King, 2005; Ducklow et al., 2007; McClintock et al., 2008; Steig et al., 2009; Znój et al., 2017; González & Fortuny, 2018; Siegert et al., 2019; Turner et al., 2020). The effects of this temperature increase on polar ecosystems in the Antarctic have been the subject of numerous studies (e.g. Chapman & Johnson, 1968; Walther et al., 2002; Meredith & King, 2005; Convey & Smith, 2006; Clarke et al., 2007; Bokhorst et al., 2008; Forcada & Trathan, 2009; Korczak-Abshire, 2010; Convey, 2011; Trivelpiece et al., 2011; Flores et al., 2012; Constable et al., 2014; Pinkerton et al., 2021). Of particular importance here is the variability of sea-ice cover and the resulting changes in the food web (e.g. McClintock et al., 2008; Rossi et al., 2019; Testa et al., 2022; Ferreira et al., 2024).

The role of seabirds as valuable indicators of the state of marine and coastal ecosystems is widely recognized (van Franeker, 1992; Reid & Croxall, 2001; Weimerskirch et al., 2003; Piatt et al., 2007; Krüger, 2019a; Thibault et al., 2019; Velarde et al., 2019; Bird et al., 2022). In particular, food availability for seabirds is influenced by the effects of current climate change, such as increased water temperature and changes in sea ice cover, as well as increased fishing activities (Bertrand et al., 2012; Trathan et al., 2015; Hinke et al., 2017a; Watters et al., 2020; Krüger et al., 2021; Santa Cruz et al., 2022; Wethington et al., 2023). In addition, seabirds often reflect anthropogenic influences at local, regional and global levels (Micol & Jouventin, 2001; Pfeiffer & Peter, 2004; Tin et al., 2009; Chwedorzewska & Korczak, 2010; Braun et al., 2012; Cusset et al., 2023). Long-term monitoring programmes in the Antarctic are particularly valuable in this regard, as their great importance for assessing the state of ecosystems and for forecasting future developments is widely recognized and also called for by the research community as well as at international level, e.g. at the Antarctic Treaty Consultative Meeting (ATCM), Scientific Committee on Antarctic Research (SCAR) and Council of Managers of National Antarctic Programs (COMNAP) (Lewis-Smith, 1990; ATS, 2005b; COMNAP, 2005, 2006; Hughes, 2010; Xavier et al., 2016; Brooks et al., 2019). Furthermore, the current United Nations Decade of Ocean Science for Sustainable Development 2021 - 2030 emphasizes the need for international cooperation in research with the aim of reversing the cycle of declining ocean health (source: https://www.oceandecade.org/). In this context, the Southern Ocean Task Force was set up to develop the Southern Ocean Action Plan, coordinated by the Scientific Committee on Antarctic Research (SCAR, source: https://www.sodecade.org/about/).

The present study was conducted in the Fildes Region in southwest King George Island, Maritime Antarctica. This area is characterized by a comparatively high terrestrial biodiversity and a high level of diverse human activities due to the presence of six Antarctic stations of different nations and an airstrip for intra- and intercontinental flights. Negative impacts of the high concentration of human activities in the Fildes Region have been repeatedly described, including disturbance of seabirds and seals, habitat destruction and significant contamination of soil, water and air (Tin & Roura, 2004; Tin et al., 2009; Braun et al., 2012; Braun et al., 2014; Braun et al., 2017b). For this reason, a comprehensive study was initiated - based on unpublished data from the 1980s - during which a GPS- and GIS-based, systematic survey of breeding bird populations, reproduction and resting sites of seals as well as the spatial and temporal use of the area by human activities and their environmental impact was documented (Peter et al., 2008). The resulting environmental risk assessment highlighted the need for effective management of all human activities in the Fildes Region (Peter et al., 2008). It was also shown that, in view of the complexity and intensity of human activities in the Fildes Region, the management measures taken in this area to date were largely inadequate from an environmental perspective. As a consequence, a designation as an Antarctic Specially Managed Area (ASMA) was recommended and a discussion process was initiated at international level to introduce efficient management measures with the aim of reducing negative environmental impacts (Peter et al., 2008). A further research project served to update the data basis during the ongoing discussion process, whereby current developments in the area were documented (Peter et al., 2013). In the meantime, the international discussion process to solve the environmental problems has been considered dormant for some time, as unfortunately no agreement could be reached between the stakeholders involved.

Subsequently, three further research projects were carried out to continue the long-term monitoring of seabirds and seals in the Fildes Region and to investigate other research questions, such as the spread of the native grass species, the introduction of non-native species, the documentation of glacier retreat areas and the spread of plants (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). Seabird monitoring was also extended to several ice-free areas of the adjacent Maxwell Bay.

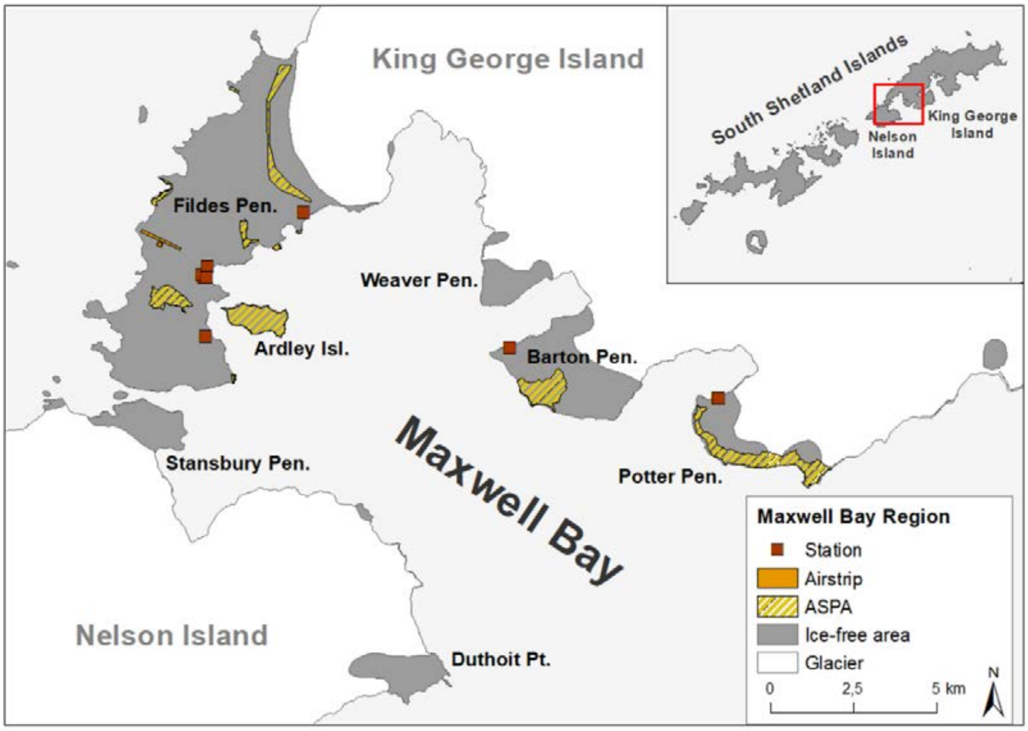
This study presents the results of the continued seabird and seal monitoring and thus supports the high value of such long-term surveys in the Antarctic. In addition, the state of knowledge on breeding birds in selected areas of Maxwell Bay was updated and supplemented, and the input of marine debris into the study area was analysed. This work contributes to a comprehensive assessment of ecological processes as a result of current climatic changes and the increasing anthropogenic influence in the Antarctic.

All publications, presentations and poster contributions produced as part of this project can be found in Appendix G.

2 Study area

The study area, is located in the south-west of King George Island (Figure 1), a largely ice-covered island that is part of the South Shetland Islands in the Western Antarctic Peninsula (WAP), which is a region undergoing rapid environmental changes (Turner et al., 2020). There are several designated Antarctic Specially Protected Areas (ASPA) in the Maxwell Bay area: ASPA No. 125 Fildes Peninsula, ASPA No. 150 Ardley Island, ASPA No. 171 Narębski Point, ASPA No. 132 Potter Peninsula (Figure 1). The latter were designated on the basis of rich seabird populations (ATS, 2013, 2014). A total of eight Antarctic stations from Argentina, Chile, China, Russia, South Korea and Uruguay are located here (Figure 1), as well as numerous field huts.

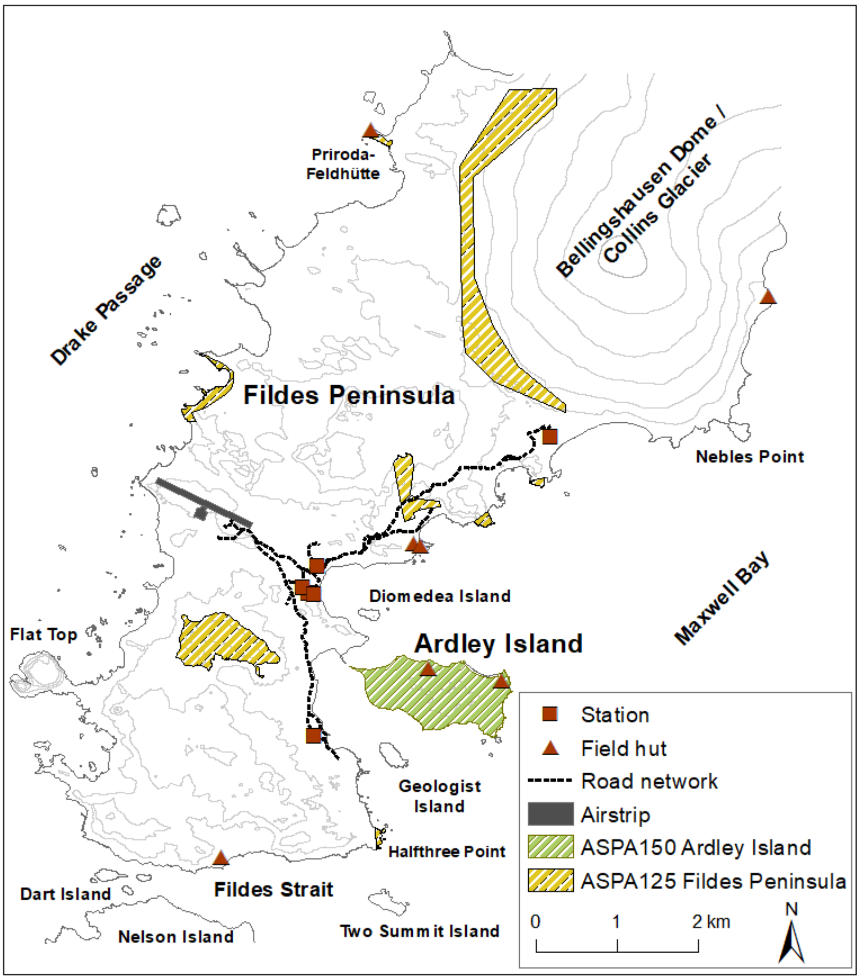
Figure 1: Overview map of the study area with locations of the Antarctic Special Protection Areas (ASPAs)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

In the extreme south-west of King George Island are the ice-free areas of the Fildes Peninsula and Ardley Island (62°08‘ to 62°14’ S, 59°02‘ to 58°51’ W). Both areas and all larger neighbouring islands within a maximum distance of 0.5 km off the coast form the Fildes Region (Figure 2). This area borders Drake Passage to the west and Maxwell Bay to the east. The Bellingshausen Dome and Collins Glacier spurs are adjacent to the north, while the area is separated from Nelson Island by the approximately 400 m wide Fildes Strait to the south. The Fildes Region is one of the largest areas on the Antarctic Peninsula (Henriques et al., 2018), covering an area of around 35 km^2. It is of great importance for the local fauna, as it is home to a total of 14 breeding bird species and is a reproduction area for five of the six seal species living in the Antarctic. ASPA No. 125 Fildes Peninsula, consisting of eight sub-areas, was designated due to rich fossil deposits (Figure 2; Peter et al., 2008; ATS, 2009a). The designation of ASPA No. 150 Ardley Island is based on an extraordinarily high biodiversity of flora and fauna by Antarctic standards (Figure 2; ATS, 2009b). The approximately 1.3 km^2 island is connected to the Fildes Peninsula at low tide by an isthmus that can be accessed on foot. The designated protected area covers the entire island, with the exception of a narrow coastal strip in the north-east, which serves as a visitor zone (Chile & Argentina, 2011). Another notable feature is the presence of distinct beach terraces of high scientific importance (Peter et al., 2008; Nopnakorn et al., 2023). Furthermore, there is a current management approach to protect marine resources in the form of the designation of marine Important Bird Areas (mIBA). Three mIBAS (No. 9, 11 and 12) have been proposed in the vicinity of the Fildes Region (Handley et al., 2021; Soutullo et al., 2024).

Figure 2: **Overview of the Fildes Region study area with representation of the Antarctic Special Protection Areas (ASPAs)**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Due to its close proximity to South America, the high concentration of six stations of different nations and an airstrip for inter and intra-continental flights, the Fildes Peninsula area plays a key logistical role in the Antarctic Peninsula region. A wide range of research, logistics and tourism activities also take place here. This includes intensive air, sea and land transport as well as tourism in the form of guided tours and boat trips, visits by delegations and film teams, educational programmes and leisure excursions by station members (Braun et al., 2012). The resulting spatial and temporal overlaps of these activities often cause conflicts of interest between the user groups as well as contradictions with the legally agreed environmental protection requirements, e.g. Madrid Protocol of the Antarctic Treaty (Braun et al., 2012). In recent years, the intensity of air and sea transport has increased significantly (Braun et al., 2020b), as have scientific, logistical and tourist activities.

3 Preliminary comments

In the 2023/24 season, due to the expected high risk of the occurrence of avian influenza (Dewar et al., 2023), various measures were implemented in accordance with the protocol drawn up by the Instituto Antártico Uruguayo (IAU) to prevent the spread of this disease in the study area. This included the consistent disinfection of footwear, clothing and equipment as well as the wearing of face masks when handling penguins. Since skuas (*Stercorarius* spec.), as apex predators, are at high risk of contracting avian influenza, animals of this species were not caught, measured or ringed due to the high risk of the occurrence and further distribution of H5N1 avian influenza. All skua nest checks were carried out by visual inspection at the greatest possible distance.

Despite increased attention, no signs of Highly Pathogenic Avian Influenza (HPAI) H5N1 in the form of birds with typical symptoms or increased mortality were detected by our team in the study area in the Fildes Region in the 2023/24 field season. In addition, several teams of scientists from different nations analysed numerous samples of various kinds (cloacal swabs, blood and tissue samples, etc.) of seabirds and seals as well as soil samples on Fildes Peninsula, Ardley Island and on the adjacent Barton Peninsula (pers. comm. L. Krüger) between November and February. All samples from the Fildes Region proved to be negative in the PCR test (Bennett Lazo et al., 2024; Lisovski et al., 2024). In contrast, all samples of south polar skuas (*Stercorarius maccormicki*) from James Ross Island tested positive for HPAI H5N1 during a mortality event in March 2024 (Bennett Lazo et al., 2024). This is consistent with other findings, after the first detection of avian influenza in the Subantarctic on the Falkland Islands of South Georgia in a mortality event of seabirds and in October 2023 (Banyard et al., 2024; Bennison et al., 2024), the highly pathogenic virus also reached Antarctica towards the end of the austral summer (Lisovski et al., 2024). The 'Sub-Antarctic and Antarctic Highly Pathogenic Avian Influenza H5N1 Monitoring Project' was initiated to track the occurrence of the disease in the Subantarctic and Antarctic (https://scar.org/library-data/avian-flu). As a result, a constantly updated map is made available that lists all suspected and confirmed cases of the virus in Antarctic wildlife.

4 Methods

The standardised methodology applied in this study corresponds to the methodology used in previous projects based on the seabird and seal monitoring introduced in the 1979/80 season. A first detailed description can be found in Peter et al. (2008). The field season of the current project covered the austral summer (December, January and February) of the 2021/22, 2022/23 and 2023/24 seasons. The timetable of the seabird and seal monitoring and other tasks within this study during the austral summer can be derived from Table 11 in Appendix B. All maps were created using ESRI ArcGIS Desktop 10.8.1 and refer to the projection WGS 1984, UTM 21S. The topographical data basis is largely taken from the SCAR KGIS project (Vogt et al., 2004) and the SCAR Antarctic Digital Database (source: https://www.add.scar.org/). All place names used are based on the names listed in the SCAR Composite Gazetteer of Antarctica (SCARCGA).

The statistical analyses and the creation of graphs were carried out using the SigmaPlot 14.5 programme, based on a significance level of 5 %. Linear regression was used to check for temporal trends. The year indicated in the graphs refers to the second half of the austral summer (the term ‘2022’ corresponds to ‘2021/22 season’). If missing data occur in the graphs, no census data were available for the respective season. As many of the figures and tables are a continuation of the previous projects, references to the latter have been omitted to improve readability. All sources of the monitoring data used are listed in Table 9 in Appendix A.

1. Seabird monitoring in the Fildes Region

   In each of the three field seasons, an area-wide systematic GPS-based mapping of all breeding sites in the Fildes Region was carried out. The study area covers approx. 35 km^2. and was monitored at least weekly. Species with a high breeding site fidelity were mapped using specially created monitoring maps. As the start of breeding and raising of the chicks of the individual species vary greatly, these mapping and monitoring activities sometimes extended over the entire summer season. Depending on the species, the number of breeding pairs (bp) was recorded as individual nests or as a combination of individual nests and colonies. For some bird species, breeding success was calculated in the form of the number of fledged or ready-to-fledge young per breeding pair.

   a. Penguins

   The number of breeding pairs of penguins on Ardley Islandwas recorded in November and December and was partly supported by Uruguayan scientists. If, for logistical reasons, the census could only be carried out relatively late after the start of breeding (December), the parameter ‘apparently occupied nests’ (AON) was recorded in order to include active nests that had already been abandoned. To record the spatial distribution, the long-standing categorisation of the colony into zones and breeding groups was continued. To determine the breeding success of the penguins, the fledged juveniles were counted in January or February, depending on the species. The census of the fledged chicks was partly supported by Korean and Chinese scientists.

   As part of the CCAMLR-CEMP site monitoring, the juveniles of Adélie and gentoo penguins on Ardley Island were weighed in the 2021/22, 2022/23 and 2023/24 seasons according to the specified methodology (CCAMLR, 2014). In each of the three pentads, 20 fledglings were caught daily with a hand net before leaving the colony and weighed to the nearest 0.1 kg using a Pesola scale. The young animals to be weighed were selected at random. However, as Adélie penguins have a much more synchronised breeding phenology, in which the juveniles leave the colony almost simultaneously, it was only possible to weigh them exclusively on one day or at most in a pentad. In the 2023/24 season, the juvenile gentoo penguins were weighed on a single day instead of in pentades, but in large numbers. For this reason, and due to the very late fledging of gentoo penguins, an alternative methodology was used in the 2023/24 season, in which a larger number of juveniles were weighed on one day (22 Adélie penguins, 270 gentoo penguins).

To assess possible changes in the distribution of penguin breeding groups on Ardley Island, an older ground-based spatial survey was compared with current aerial photographs taken by Uruguayan scientists in the 2023/24 season.

The recorded breeding pair numbers of chinstrap penguins on the Drake Coast of the Fildes Peninsula are based partly on counts and partly on estimates from afar, as some breeding sites are inaccessible or difficult to access, which makes a clear interpretation of the data difficult.

b. Southern giant petrel

The number of breeding pairs of the southern giant petrel was counted in December using the parameter AON, as this species begins breeding as early as mid-October (Peter et al., 1991), in order to record active nests that have already been abandoned (Creuwels et al., 2005). In order to determine the breeding success of this species, all surviving chicks were counted at the end of February, as the fledging of this species only takes place late in March or April (Peter et al., 1991) and such a later survey was not possible for logistical reasons.

Unpublished expedition reports from the 1980s and a comprehensive risk assessment were used to classify the level of human activity in the Fildes Region and associated disturbance in the individual giant petrel colonies based on the frequency and intensity of disturbance using detailed information on the spatial distribution of air and sea traffic as well as vehicle and pedestrian movements (Peter et al., 2008). This information was kept up to date through further observations during the subsequent projects (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). The main observations here were aircraft movements that did not comply with the 'Guidelines for Aircraft near Concentrations of Birds in Antarctica' (ATS, 2004) and visits by station members or tourists, whereby the recommended minimum distance to giant petrels (Pfeiffer & Peter, 2004) was often not observed.

c. Cape petrel, kelp gull, Antarcic tern

For these species, the breeding population was recorded in the form of both individual breeders and colonies. This made it possible to derive the average colony size and breeding pair density. Due to the inaccessibility of the breeding sites from a distance, the number of breeding pairs of the Cape Petrel was determined on the basis of the adult birds sitting at the breeding site. Due to the high susceptibility of kelpgulls and Antarctic terns to disturbance, the nests were not searched and instead the minimum breeding pair density was derived from the number of attacking adults (Peter et al., 2008). As Antarctic terns can breed throughout the summer, the entire study area was also regularly monitored during the months of January and February. To determine the main breeding areas of kelp gulls and Antarctic terns, the kernel density was determined.

d. Light-mantled sooty albatros, snowy sheathbill, blue-eyed shag

For these species, direct evidence of breeding is difficult due to the inaccessibility of the breeding sites. Therefore, various parameters, such as repeated observations of two adult birds at the same site or of juveniles at the suspected breeding site, were used to draw conclusions about suspected breeding. So far, a few breeding sites have been identified on the west coast of the Fildes Region.

e. Wilson´s storm-petrel, black-bellied storm-petrel

In the present project, the existing knowledge of the breeding population of the two storm-petrel species, which is based on systematic mapping based on nocturnal calls during the breeding season (Peter et al., 2008), was supplemented with the help of daytime sightings or calls of birds at the breeding site.

f. Skuas

The breeding population of the skuas were recorded throughout the entire austral summer. In February, the number of fledged chicks was recorded to determine breeding success. All nest checks were supplemented by ring readings throughout the season in order to record the present adult birds. In addition to the number of breeding pairs, the number of occupied territories in which no eggs were laid was also recorded. A territory is considered to be occupied by non-breeders if the typical territorial behavior was observed (Furness, 1987). In the 2023/24 season, no skuas were caught, measured or ringed due to the high risk of H5N1 avian influenza. Then, all nest checks were conducted by visual inspection at the greatest possible distance and without physical contact. As the study area in the Fildes Region is located in the hybridisation zone of the brown skua and the south polar skua (Ritz et al., 2006), mixed pairs also occur here. This makes pair assignment difficult if only the species of one partner is known. On average, the proportion of mixed pairs in the total number of breeding skuas in the Fildes Region amounted to approx. 6 %. Due to logistical constraints and early breeding failures, the number of skua pairs of unknown type in pairs where only one partner is known has recently increased. Therefore, the "estimated pair type (ept)" according to Krietsch et al. (2016) was used to minimize inaccuracies. This refers to the type of known partner, as partners of the other spieces are relatively rare. In the following, only the pair type "undetermined" is listed if species identification was generally dispensed with or no species identification was possible in individual cases.

g. Vagrant and migrant birds

In addition to the breeding birds, all sightings and mortalities of vagrant and migrant bird species were documented, in some cases exclusively as dead finds. In addition, any information on additional observations provided by station personnel staff was used.

2. Seal monitoring on Fildes Peninsula and Ardley Island

To record the populations of the various seal species in the austral summer, the entire coastline of the Fildes Peninsula and Ardley Islands was surveyed around the middle of the month. The categorisation of the bays established in the 1980s was used for this purpose (see Figure 49 in Appendix B). This allowed the number of all seals on land to be recorded with bay accuracy. In addition, all groups of elephant seals with at least ten individuals were measured. Attention was also paid to evidence of newborn seal pups in the area, although births of most seal species in the Fildes Region take place mainly in September and October.

3. Breeding bird monitoring in the ice-free areas of Maxwell Bay

For the survey of breeding bird populations in selected ice-free areas of Maxwell Bay, these areas were visited if possible or information provided by other scientists was used.

4. Climatic changes in the study area

In order to analyse monthly meteorological data on air temperature in the study area, the Met READER database for Bellingshausen station, WMO index 89050 (source: https://legacy.bas.ac.uk/met/READER/surface/stationpt.html) was used. Snow depth and precipitation data was taken from National Oceanic and Atmospheric Administration (NOAA, source: https://www.ncei.noaa.gov/access/metadata/landing-page/bin/iso?id=gov.noaa.ncdc:C00516). Data on the El Niño phenomenon, the 'Oceanic Niño Index' (ONI) and the global mean temperature values were also obtained from the NOAA data base (source: https://origin.cpc.ncep.noaa.gov/products/analysis_monitoring/ensostuff/ONI_v5.php & ncei.noaa.gov/data/global-summary-of-the-day/access/).

5. Glacier retreat areas of selected areas of Maxwell Bay

To document the changes in horizontal glacier extent, the glacier margin zones of the 'land-terminating laciers' of all larger ice-free areas within Maxwell Bay were analysed using remote sensing data. The Potter Peninsula was excluded from the study, as a comprehensive study of this area already exists (Rückamp et al., 2011). The tidewater glaciers of Collins Harbour and Marian Cove were also taken into account. For this purpose, free aerial and satellite images from the United States Geological Survey (USGS, source: http://earthexplorer.usgs.gov/), the European Space Agency (ESA, source: https://scihub.copernicus.eu/, https://sentinels.copernicus.eu/web/sentinel/home) and the commercial satellite platform DigitalGlobe WorldView-2, WorldView-3 and GeoEye-1 (source: https://discover.digitalglobe.com/) were analysed. The first available aerial photographs date from 1956 and 1984, although the relatively heavy snow cover at the time of the photographs makes interpretation difficult. Other remote sensing images come from Landsat (USGS), Copernicus Sentinel (ESA), WorldView-2, WorldView-3 and GeoEye-1 (Digital Globe) satellites.

The number of suitable images was severely limited, as the prerequisite for optical analysability is low cloud cover and low snow cover throughout the year. For this reason, a large number of aerial photographs and satellite images had to be checked in advance for their suitability, whereby attention had to be paid to sufficient coverage of the desired areas and, with regard to cloud and snow cover, to sufficiently good recognisability of the glacier edge zones. Although the study area only rarely has cloud-free days, the long-term data series means that there are now enough meaningful satellite images available to assess the horizontal development of glacier extent in the study area over the past decades. However, suitable images for the year 2024 were not available due to high cloud cover, so the most recent images were taken in 2023. Older aerial photographs and satellite images were georeferenced before being analysed. The systematic documentation of the glacier retreat areas was carried out using manual image interpretation, based on delineation using colour tone, surface structure or other delineation features using ESRI ArcGIS Desktop 10.8.1 software. Manual delineation is sometimes made more difficult by snow cover and overlying or overlying moraine material (Braun et al., 2017a). The delineation of the boundary zone of the glacier was supported by changes in contrasts and the selection of spectral channels (true or false colour representation). The points at which the glacier crosses the coastline were always used as the boundary of the ice-free area. The absolute accuracy of the method is limited by the resolution of the satellite images (Landsat: 30 m, Sentinel: 10 m, WorldView: 0.5 m). The change in area between 1956, 1989 and 2023 was calculated based on the horizontal glacier extent. In contrast, the glacier area extent was calculated directly for Bellingshausen Dome and the tide-water glaciers of Collins Harbour and Marian Cove.

However, due to the fact that it is sometimes difficult to recognise the exact glacier boundary due to snow cover and moraine material in front of or on top of it, an error that cannot be quantified further is to be expected.

6. Input of marine debris into the study area

   Since the 2018/19 season, marine debris has been recorded in a selected bay on the west coast of the Fildes Peninsula (bay no. 12, position: 62° 12‘ 09‘‘ S, 58° 59’ 55‘‘ W, see Figure 49 in Appendix B). This 450 m long stretch of beach is topographically separated from the neighbouring bays and the hinterland. As a large proportion of the beach debris often does not remain on the immediate coastline but is dispersed by the strong prevailing winds, the neighbouring area between the coast and the cliff bordering the bay towards the inland was also included in the survey. As a result, the surveyed area totalled approx. 53,000 m^2, where all recognisable anthropogenic material was collected. Larger objects that could not be removed, such as buoys, long ropes or large pieces of wood, were left on site but comprehensively documented in order to avoid duplication in subsequent seasons. All collected material was cleaned and dried at the station, sorted and classified, weighed and photographed according to the standardised methodology specified by CCAMLR. The following materials were categorised: plastic, wood, paper, metal, glass and other. The following size classes were used in the classification: S = ‘small’ (< 2.5 cm), M = ‘medium’ (2.5 cm - 10 cm), L = ‘large’ (> 10 cm - 1 m), VL = ‘very large’ (> 1 m). In addition, all recognisable inscriptions on the objects were noted for a possible determination of the origin, manufacturer or originator of the marine debris. It should be noted that other scientists collected marine debris in the 2021/22 season, which limits the significance of the data presented here for this season. We were kindly provided with the data for a further collection in December 2023, i.e. shortly before our own survey. This allowed the data from the two collections to be combined and included in the analysis.

7. Influence of anthropogenic material on seabirds in the Fildes Region

   To investigate the influence of anthropogenic material on seabirds, all complete dead animals found in the study area were dissected and their stomach contents checked for anthropogenic material. In addition, skua pellets with indigestible remains of seabirds were collected, dried and then carefully analysed for all visible anthropogenic material, as this method might otherwise under-represent smaller particles (Ryan & Fraser, 1988). However, only compact, closed pellets with seabird remains were used for this purpose, as parts of the contents of older disintegrating pellets may already have been dispersed by the wind. The species identification of the seabirds was based on the colour of the feathers and the size of the bones, especially the legs. Furthermore, all recognisable anthropogenic material in the birds' nests and other relevant finds or observations were documented.

8. Introduced non-native species

   The known location of an introduced non-native grass species in the area of the Russian Bellingshausen station was checked at least once in each of the 2021/22 to 2023/24 seasons. Due to temporary pandemic regulations, the location of several tufts of a non-native grass near the Chinese station Great Wall could only be visited in the 2022/23 and 2023/24 seasons. The dataset was supplemented by observations of non-native insects, both inside and outside station buildings.

5 Monitoring of breeding birds and seals in the Fildes Region

5.1 Overview of the bird species recorded

The Fildes Region is currently home for 14 breeding bird species, 13 of which are seabird species. All scientific species names used in this study correspond to the current specifications of the World Bird List project IOC, source: IOC World Bird List – Version 14.1, Gill et al., 2024). A complete list of all breeding bird species of the Fildes Region, their global population status and conservation status as defined by the International Union for Conservation of Nature (IUCN) are given in Table 1. Almost all of these species are currently listed as 'Least Concern (LC) – unlikely to become endangered or extinct in the near future' due to a stable or growing global population. The same applies to species that show a declining trend but whose number of individuals is estimated to be very high, meaning that the species is not currently threatened. Only the light-mantled sooty albatross is currently categorised as 'Near Threatened (NT) – close to being endangered in the near future' (Table 1).

Table 1: Overview of all breeding bird species recorded in the Fildes Region since 1979/80, their global population trend, IUCN conservation status (IUCN, 2024) and current population size and trend in the Fildes Region in the seasons 2021/22 to 2023/24

Species	Global population trend	IUCN status	Population size (bp) Fildes Region 2021/22 – 2023/24	Population trend Fildes Region 1980 – 2023/24
Adélie penguin (*Pygoscelis adeliae*)	increasing	‚Least Concern‘	184 – 220	decreasing
Chinstrap penguin (*Pygoscelis antarcticus*)	decreasing	‚Least Concern‘	3 – 10	decreasing
Gentoo penguin (*Pygoscelis papua*)	stable	‚Least Concern‘	7,065 – 7,942	increasing
Southern giant petrel (*Macronectes giganteus*)	increasing	‚Least Concern‘	780 – 872	increasing
Cape petrel (*Daption capense*)	stable	‚Least Concern‘	3 – 8	decreasing
Light-mantled sooty albatross (*Phoebetria palpebrata*)	decreasing	‚Near Threatened‘	present (non-breeding)	unknown
Wilson's storm-petrel (*Oceanites oceanicus*)	stable	‚Least Concern‘	~ 3500 – 5000	unknown
Black-bellied storm-petrel (*Fregetta tropica*)	decreasing	‚Least Concern‘	~ 500 – 1000	unknown
Brown skua (*Stercorarius antarcticus lonnbergi*)	decreasing	‚Least Concern‘	32 – 55	stable
South polar skua (*Stercorarius maccormicki*)	stable	‚Least Concern‘	84 – 121	stable

Species	Global population trend	IUCN status	Population size (bp) Fildes Region 2021/22 – 2023/24	Population trend Fildes Region 1980 – 2023/24
Kelp gull (*Larus dominicanus*)	increasing	‚Least Concern‘	46 - 53	decreasing
Antarctic tern (*Sterna vittata*)	unknown	‚Least Concern‘	73 - 196	deecreasing
Antarctic shag (*Leucocarbo bransfieldensis*)	unknown	‚Least Concern‘	4	unknown
Snowy sheathbill (*Chionis albus*)	stable	‚Least Concern‘	1	unknown

5.2 Penguins (*Pygoscelis* spec.)

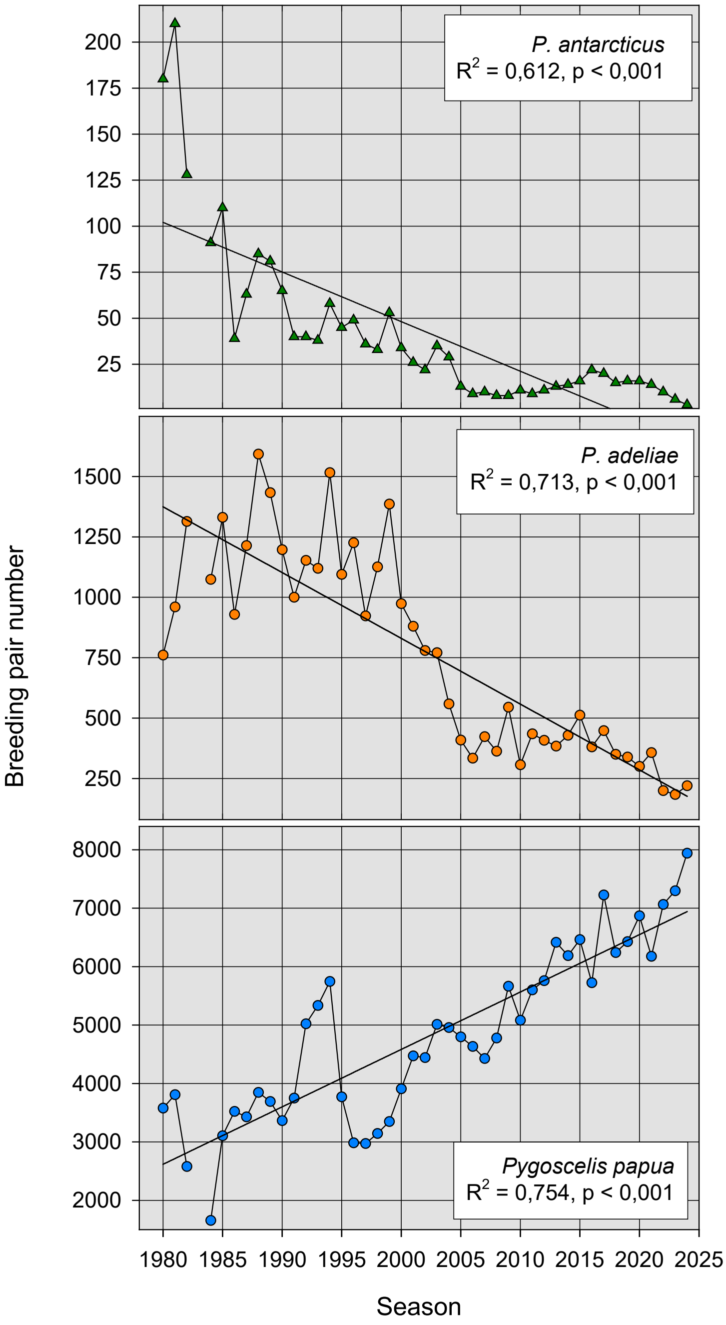
5.2.1 Ardley Island

Ardley Island, located to the east of the Fildes Peninsula, is currently home to eleven breeding bird species. The eastern part of the island hosts a colony of three representatives of the *Pygoscelis* genus, the chinstrap penguin (*Pygoscelis antarcticus*), Adélie penguin (*P. adeliae*) and gentoo penguin (*P. papua*), which breed there sympatrically. As the penguin colony there harbours more than 1 % of the global population of gentoo penguins (IBA selection criterion A4ii), the island is listed as 'Important Bird Area' (IBA) No. 48 (until 2011 IBA No. 53) in addition to its status as ASPA No. 150 (Harris et al., 2015; BirdLife International, 2024c). Despite its protected status, negative impacts from numerous human activities continue to be observed on Ardley Island during the austral summer, which contradict the legally binding requirements of the management plan (ATS, 2009b). These include inadequate coordination of the diverse scientific activities in the protected area, aircraft flying below minimum altitudes and distances, non-compliance with the visitor zone by tourists and suspected unauthorised visits by members of the surrounding stations (Peter et al., 2008; Peter et al., 2013; Braun et al., 2014; Braun et al., 2017b; Braun et al., 2020c; Braun et al., 2022). Since the start of long-term monitoring in the 1979/80 season, data on this penguin population is now available for up to 44 years.

5.2.1.1 Population dynamics

Based on the penguin monitoring carried out, it is clear that the decline in the number of breeding pairs of chinstrap penguins on Ardley Island continued in the current study period (Figure 3). In the 2023/24 season, only 3 breeding pairs of chinstrap penguins were counted, the lowest number since long-term monitoring began. The local population of this species thus declined by 98.3 % in a long-term comparison, with an annual rate of change of -8.7 %. The Adélie penguin also recorded very low breeding pair numbers on Ardley Island. With only 184 breeding pairs in the 2022/23 season, the lowest value since the start of long-term monitoring was recorded for this species (Figure 3). Overall, the decline in the local population of the Adélie penguin was 71.1 % (annual rate of change: -2.7 %). In contrast, the gentoo penguin population on Ardley Island continued to increase during the study period, reaching 7,942 breeding pairs in the 2023/24 season, the highest number since the start of long-term monitoring (Figure 3). The number of gentoo penguins breeding on Ardley Island thus increased by 122 % in a long-term comparison (annual rate of change: 1.8 %).

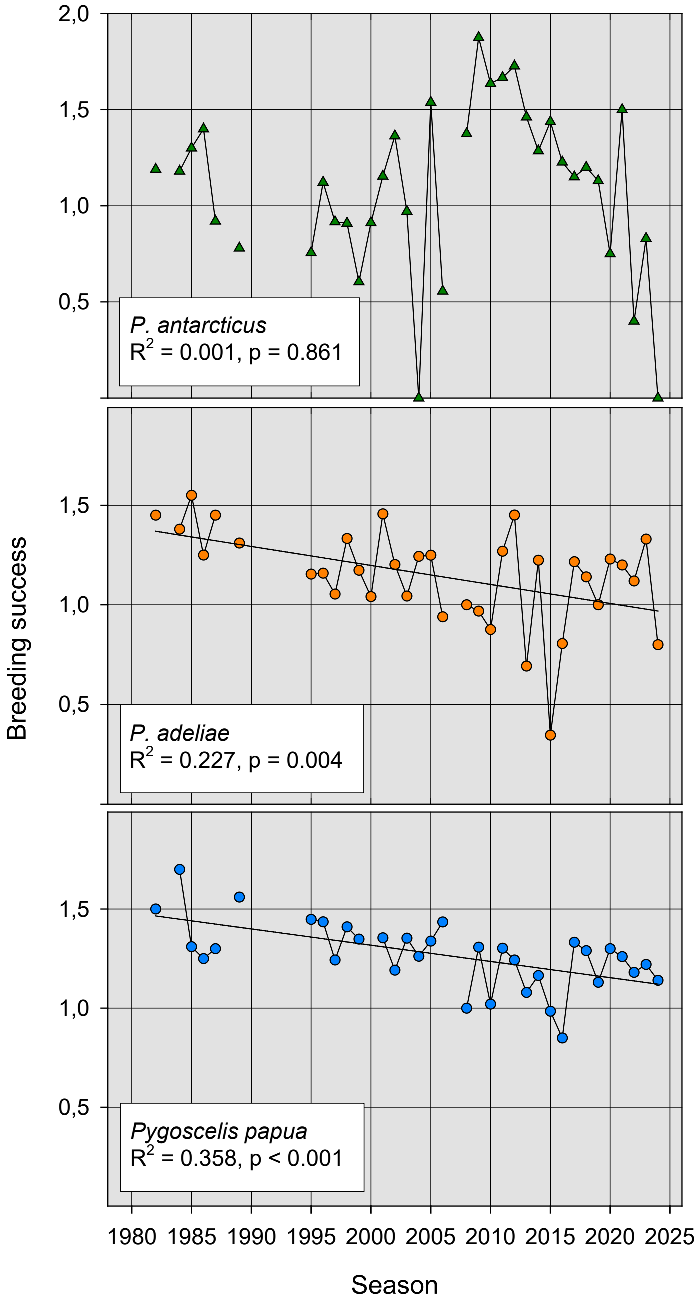
Figure 3: **Breeding pair numbers of chinstrap (*Pygoscelis antarcticus*), Adélie (*P. adeliae*) and gentoo penguins (*P. papua*) on Ardley Island in the seasons 1979/80 to 2023/24 with presentation of significant trends**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

In the studied three seasons, the breeding success of chinstrap penguins on Ardley Island showed strong fluctuations in a long-term comparison (Figure 4), although the significance is limited due to the recently very low number of breeding pairs. In contrast, the values for the breeding success of Adélie and gentoo penguins fluctuated much less, but showed a negative trend over time (Figure 4). Nevertheless, the long-term mean for the breeding success of both species was significantly higher (Adélie penguin: 1.15 ± 0.25, gentoo penguin: 1.27 ± 0.17) than for the chinstrap penguin (1.09 ± 0.44). No fledged chinstrap penguin chicks were registered in the 2023/24 season, with only three pairs breeding. Although a declining reproductive success of a population, as it was observed for the gentoo penguin on Ardley Island, may indicate that its growth is slowly approaching the colony`s capacity, the still rapidly growing population of this species proves that the maximum capacity has not yet been reached. The extent to which ongoing immigration of birds from other areas plays a major role in the population growth of the gentoo penguin is not known (Herman & Lynch, 2022).

Figure 4: Breeding success of chinstrap, Adélie and gentoo penguins on Ardley Island in the seasons 1981/82 to 2023/24 with presentation of significant trends



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The breeding success of penguins is significantly influenced by regional factors such as sea ice cover, food availability as well as local weather extremes or predation (Trathan et al., 1996;

Cobley & Shears, 1999; Forcada et al., 2006; Ducklow et al., 2007; Hinke et al., 2007; Trathan et al., 2008; Lynch et al., 2010; Juáres et al., 2015; Olmastroni et al., 2022; Salmerón et al., 2023). In comparison, the breeding success of *Pygoscelis* penguins on Ardley in recent decades were often higher than in neighbouring colonies on Barton and Potter Peninsula (see chapter 6.2 & 6.5) and in more remote colonies, such as Livingston Island, Deception Island, Peterman Island or the South Orkney Islands (Cobley & Shears, 1999; Holmes et al., 2006; Hinke et al., 2007; Lynch et al., 2010; Barbosa et al., 2012; Republic of Korea, 2014; Dunn et al., 2016a; Dunn et al., 2019; Juáres et al., 2024). The reasons for this are still unclear.

The higher breeding success of gentoo penguins on Ardley Island compared to the other *Pygoscelis* species is presumably related to the higher variability of this species in relation to various environmental factors. Various studies have shown that gentoo penguins are significantly more flexible than Adélie penguins in terms of a higher temporal and spatial responsiveness regarding their breeding and foraging strategy (Boersma, 2008; Miller et al., 2009; Juáres et al., 2013; Juáres et al., 2015). As a result, they are probably much less affected by unfavourable environmental conditions, such as high snow cover in the colony at the beginning of the breeding season (Bost & Jouventin, 1990; Lynch et al., 2009; Hinke et al., 2012; Juáres et al., 2013; Juáres et al., 2020).

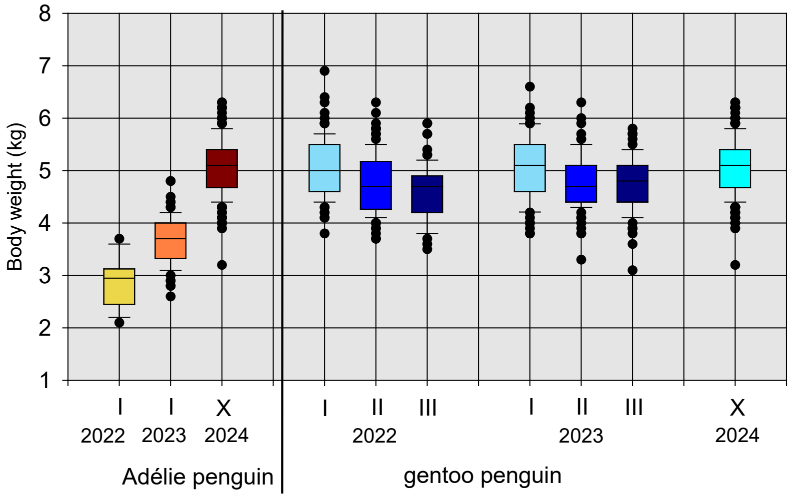
The results shown here on the population dynamics of the three penguin species on Ardley Island once again demonstrate that the long-term trends already described (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022) are continuing. These local trends are consistent with the development of the three Pygoscelis penguin species in large areas of Antarctica, especially in the northern WAP region (e.g. Hinke et al., 2007; Chesalin et al., 2009; Barbosa et al., 2012; Lynch et al., 2012; Naveen et al., 2012; González-Zevallos et al., 2013; Casanovas et al., 2015; Juáres et al., 2015; Dunn et al., 2016a; Fountain et al., 2016; Petry & Benemann, 2017; Herman et al., 2020; Juáres et al., 2020; Pascoe et al., 2020; Strycker et al., 2021; Talis et al., 2023; Wethington et al., 2023). This development reflects the different responses of the various species to the complex ecological processes resulting from progressive climatic warming (Smith et al., 2003; Lynnes et al., 2004; Forcada et al., 2006; Ducklow et al., 2007; Ainley et al., 2010; Ratcliffe & Trathan, 2011; Trivelpiece et al., 2011; Trathan et al., 2012; Clucas et al., 2014; Hinke et al., 2017b). For example, the regional decrease in winter sea ice extent as a result of the temperature increase has a negative impact on the Adélie penguin as a species heavily dependent on ice, while gentoo penguins appear to benefit in the long term from an expansion of available habitat to the south (Forcada et al., 2006; Ballerini et al., 2009; Ainley et al., 2010; Trivelpiece et al., 2011; Clucas et al., 2014; Hinke et al., 2017b; Wethington et al., 2023). In contrast, the chinstrap penguin population recorded a significant decline in its main distribution areas on the Antarctic Peninsula and the South Orkney Islands (Krüger, 2023).

Food availability appears to play an important role in the population development of penguins. For example, penguins of the genus *Pygoscelis* avoid excessive interspecific competition for limited food resources in overlapping distribution areas, in which they largely use the same food, through spatial or temporal niche partitioning (e.g. Kokubun et al., 2010; Miller et al., 2010; Polito et al., 2015; Valls & Petry, 2015; Rosciano et al., 2016; Dimitrijević et al., 2018; Pickett et al., 2018; Gorman et al., 2021; Wawrzynek-Borejko et al., 2022; Machado-Gaye et al., 2024; Soutullo et al., 2024). Nevertheless, minor differences in food preferences appear to have different effects with regard to current environmental changes. For example, changes in the food web as a result of sea ice reduction and a regional increase in krill fishing have negative consequences for Adélie and chinstrap penguins, primarily due to reduced availability of krill (Loeb et al., 1997; Trivelpiece et al., 2011; Clucas et al., 2014; Strycker et al., 2020). It has been suggested that Adélie penguins, in contrast to the currently declining chinstrap penguins, appear to be partly avoiding this impact by shifting their breeding areas southwards, with the overall population increasing in recent decades (Saiff, 1974; LaRue et al., 2013; Sailley et al., 2013; Lynch & Larue, 2014; Lyver et al., 2014; Southwell et al., 2015; Che-Castaldo et al., 2017; Talis et al., 2023). Based on modelling of climate change effects, both future population growth and decline were predicted (Ainley et al., 2010; Che-Castaldo et al., 2017). However, a recent study showed a significant decline in a metapopulation of Adélie penguins on the East Antarctic coastline around 63°E, which was caused by a decrease in breeding success due to an increase in coastal sea ice (Emmerson & Southwell, 2022). This was exacerbated by density-dependent feedback processes due to reduced survival of young birds in smaller cohorts. In contrast, gentoo penguins, whose diet usually contains a higher proportion of fish and which forage in open water (Ratcliffe & Trathan, 2011; Lynch et al., 2012), show a southward expansion of their range (Lynch et al., 2008; Chesalin et al., 2009; Lynch et al., 2012; Ducklow et al., 2013; Clucas et al., 2014; Fountain et al., 2016; Korczak-Abshire et al., 2021). Nevertheless, reduced krill availability may lead to an expansion of gentoo penguin foraging areas and overlap with krill fishing regions, resulting in competition (Ratcliffe et al., 2021). Databases of the penguin abundance and distribution, such as the Antarctic Penguin Biogeography Project, could be used for an updated assessment of the global population with regard to the conservation needs of Antarctic penguin species (Che-Castaldo et al., 2023).

5.2.1.2 CEMP Site Monitoring

The results of the previous CEMP site monitoring from Ardley Island showed that the body weight of Adélie penguins at fledging in the 2023/24 season was significantly higher than in previous years (Figure 5), although the sample size was not the same (2021/22: 14 individuals, 2022/23: 80 individuals, 2023/24: 22 individuals). In contrast, the body mass of the gentoo penguins showed no difference between the seasons but decreased slightly within each season (Figure 5). This indicates that chicks that fledge earlier in the season have a higher body weight and consequently have a higher probability of survival in their first winter. So far, the data series and the sample size are too small to make any clear statements. However, these data are very valuable in the context of the long-term Antarctic-wide CEMP site monitoring.

Figure 5: Body weight per pentade of fledged Adélie and gentoo penguin chicks on Ardley Island in the 2021/22 to 2023/24 seasons

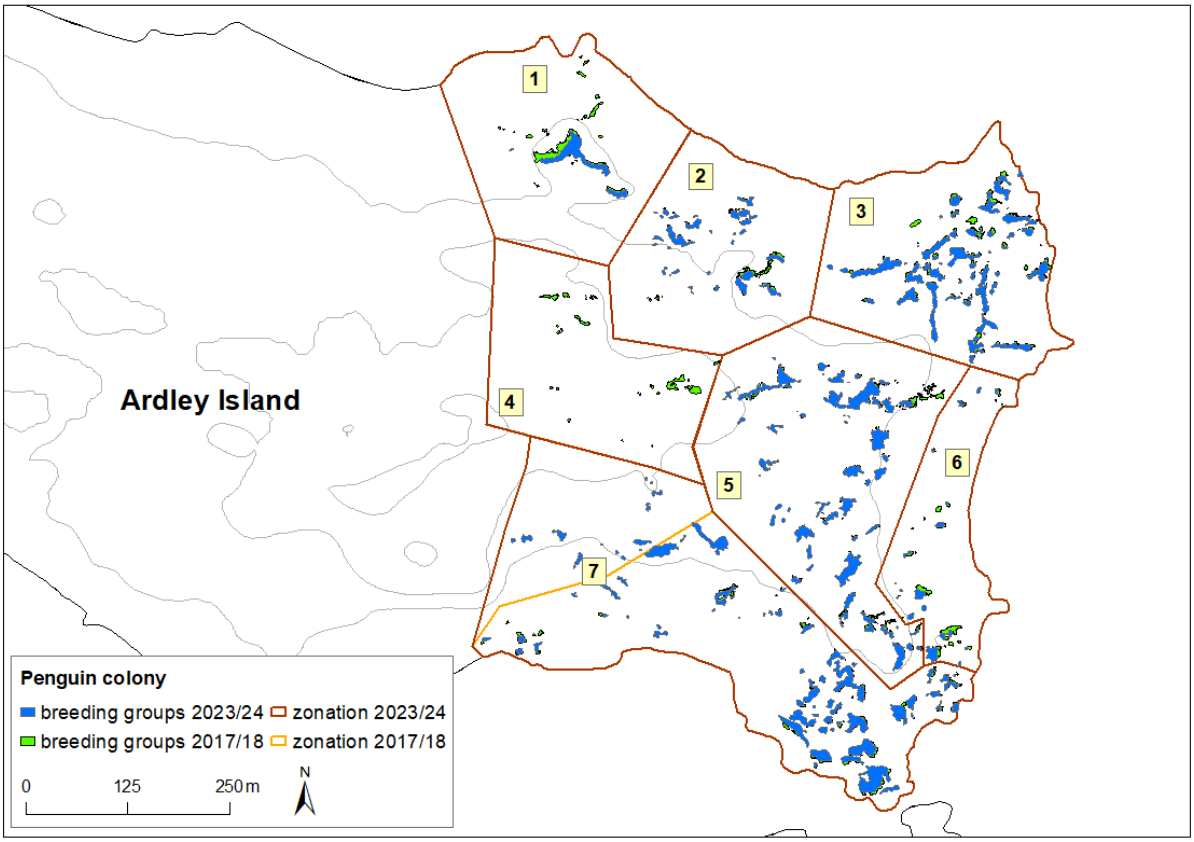


Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

5.2.1.3 Spatial changes in the Ardley Island penguin colony

Based on our own, older spatial surveys of the penguin colony and aerial photographs taken by Uruguayan scientists in the 2023/24 season, noticeable changes in the distribution of the penguin breeding groups on Ardley Island become clear. It is striking that the central, higher part of the island (Zone 4) is no longer occupied at all (Figure 6). In contrast, numerous new and larger breeding groups of gentoo penguins can now be found in the south of Ardley Island (Zone 7, Figure 6). This area has a dense, partly peat-like moss cover, especially in the higher areas, which is increasingly dying off as a result of the dense faecal cover. The environmental factors responsible for these spatial changes in the penguin colony should be subject of future research.

Figure 6: Spatial distribution of penguin breeding groups on Ardley Island in the 2017/18 and 2023/24 seasons



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

5.2.2 Chinstrap penguins on the west coast of the Fildes Peninsula

In addition to the penguin colony on Ardley Island, the chinstrap penguin populations in three other smaller colonies on the west coast of the Fildes Peninsula are being analysed. Despite the problems associated with population surveys, monitoring over the last three seasons indicates a stable population of around 59 to 71 breeding pairs. This population roughly corresponds to that of previous years (Braun et al., 2022). Overall, the population in these colonies is of lesser regional significance, as there are considerably larger colonies with tens of thousands of breeding pairs of this species in the surrounding area on the Drake Coast of King George and Nelson Island (Pfeifer et al., 2019) and these represent an important food source for brown skuas breeding in the Fildes Region (Kopp et al., 2010). There was also evidence of a declining chinstrap penguin population at each of the surveyed breeding sites - with the exception of Rzepecki Island - as well as a decline in the overall population since the early and mid-1980s and the disappearance of several smaller colonies on the west coast of the Fildes Peninsula (Pfeifer et al., 2019). Further monitoring of the smaller colonies on the west coast of the Fildes Peninsula therefore continues to make sense, as the extinction of another small chinstrap penguin colony in the south-east of the Fildes Peninsula has also been recorded here in the past (Peter et al., 2008; Peter et al., 2013). Such events may indicate changes in environmental conditions.

5.3 Southern giant petrel (*Macronectes giganteus*)

The global population of the circumpolar breeding southern giant petrel (*Macronectes giganteus*) has declined significantly in the past by 20 % to 30,000 bp (Patterson et al., 2008), but has recently increased again slightly to around 100,000 adults (BirdLife International, 2024e) or around 48,000 bp (Phillips et al., 2016). More recent population studies show contradictory trends, with declining breeding pair numbers, e.g. on Signy Island, South Orkney Islands (Dunn et al., 2016b), contrasting with stable or growing populations in Patagonia (Quintana et al., 2006), the Falkland Islands (Reid & Huin, 2005; Reid & Huin, 2008), the Subantarctic (Ryan et al., 2009a; Cuthbert et al., 2014) and the northern part of the Danco Coast Peninsula (González-Zevallos et al., 2013). Opposing trends are also described for the Antarctic Peninsula and East Antarctica (e.g. Micol & Jouventin, 2001; Creuwels et al., 2005; Lynch et al., 2008; Korczak-Abshire et al., 2019; Krüger, 2019a; ACAP, 2024a). However, as the populations in the two most important breeding areas, the Falkland Islands and South Georgia, are currently increasing (Stanworth & Crofts, 2017; Poncet et al., 2020), the overall population is currently considered to be growing (Phillips et al., 2016; BirdLife International, 2024e).

Like many other seabird species, the southern giant petrel is subject to direct and indirect impacts from fishing activities (Kock, 2001; Copello & Quintana, 2003; Sullivan et al., 2006; Krüger et al., 2017; Gianuca et al., 2019; BirdLife International, 2024e). This seabird species is therefore listed in the Agreement on the Conservation of Albatrosses and Petrels (ACAP), which came into force in 2004. The main objective of this agreement is to coordinate international activities to minimise known threats to populations of albatrosses and petrels (ACAP, 2024a). Various measures to reduce fishing-related mortality can minimise negative impacts on foraging seabirds, including the southern giant petrel (e.g. Pierre & Norden, 2006; Dasnon et al., 2022; Tamini et al., 2023). On the other hand, giant petrels appear to benefit to some extent from the use of fishery discards (Krüger et al., 2017).

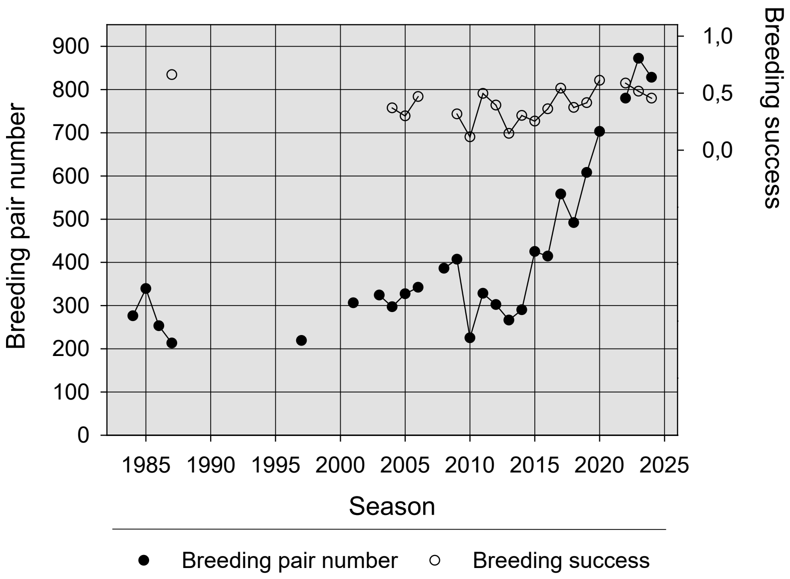
Furthermore, the southern giant petrel is considered to be extremely sensitive to anthropogenic influences in the breeding areas (e.g. Peter et al., 1991; Cooper et al., 2001; Micol & Jouventin, 2001; Pfeiffer & Peter, 2004; Chwedorzewska & Korczak, 2010). When disturbed, the parent birds often react by flying up without quickly returning to the nest, so that eggs or young chicks are exposed to the danger of hypothermia and a high risk of predation by skuas (Warham, 1962; Hunter, 1984; Peter et al., 1991; Chupin, 1997; Schulz et al., 2014). Local population declines or breeding site shifts have already been described several times in the Antarctic, and also in the Fildes Region (Peter et al., 1991; Chupin, 1997; Micol & Jouventin, 2001; Nel et al., 2002; Chwedorzewska & Korczak, 2010; Braun et al., 2012; Peter et al., 2013). The southern giant petrel can therefore be considered a suitable indicator species for the extent of human impact.

In the Fildes Region, southern giant petrels are exposed to a high level of anthropogenic disturbance due to the high level of diverse human activities (Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). This particularly affects breeding animals during the austral summer. The degree of human disturbance in the individual giant petrel colonies in the Fildes Region differs depending on the distance to the resident stations and accessibility on foot or by boat.

Based on long-term monitoring in the Fildes Region, data from up to 35 years is now available for the population of the southern giant petrel. This data clearly shows that the population of this species has been steadily increasing in this area for several years (Braun et al., 2020c; Braun et al., 2022). The survey of the giant petrel population in the 2021/22 to 2023/24 seasons consistently revealed the highest breeding pair numbers of this species since long-term surveys began (Figure 7). A maximum value of 872 bp was recorded in the 2022/23 season. Thus, the number of southern giant petrels breeding in the study area has increased by 200 % since the 1983/84 season, with an average annual growth rate of 4.9 % ($R^2 = 0.499$, $p < 0.001$, $n = 26$).

In the three seasons analysed, a comparatively high overall breeding success was again recorded for the Fildes Region, with 0.46 to 0.59 juv/bp, which was well above the long-term average of 0.39 ± 0.15 juv/bp. The stabilisation of the previously declining long-term trend thus continued (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). For the period 2003/04 to 2023/24, there was a slight but significant increase in overall breeding success ($R^2 = 0.223$, $p = 0.048$, $n = 18$, Figure 7).

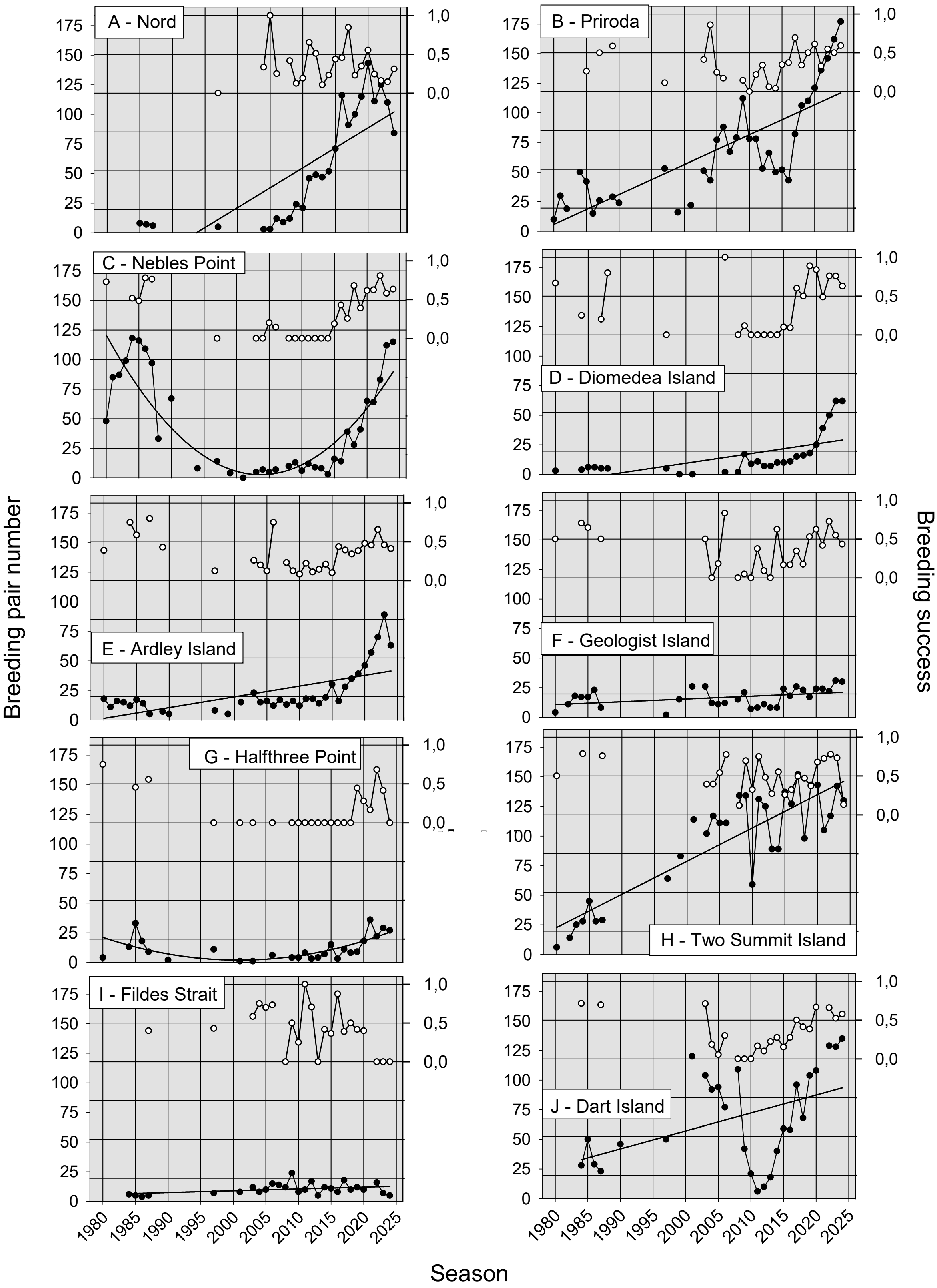
Figure 7: Total breeding pair numbers and breeding success of the southern giant petrel (*Macronectes giganteus*) in the Fildes Region between the 1983/84 and 2023/24 seasons



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The ten southern giant petrel colonies monitored in the Fildes Region harbour colonies of varying sizes, ranging from 5 (I – Fildes Strait) to 177 (B – Priroda) pairs (Figure 8). In addition, the respective breeding pair numbers largely reflected the current strong increase in the number of breeding pairs in the entire study area. Accordingly, the highest numbers of breeding pairs were recorded in five of the ten monitored colonies during the study period: B – Priroda, D – Diomedea Island, E – Ardley, F – Geologist Island and J – Dart Island. The majority of the colonies show a positive linear trend in their population development (Figure 8). Compared to the previous study (Braun et al., 2022), the colony on Geologist Island (colony F) also showed a significant positive trend. In contrast, the population development in colonies C – Nebles Point and G – Halfthree Point can be described using quadratic regression, as the breeding pair numbers there have recently returned to the level of the 1980s after a drastic collapse.

Figure 8: Breeding pair numbers (filled symbol) and breeding success (unfilled symbol) of the southern giant petrel (*Macronectes giganteus*) in the colonies of the Fildes Region

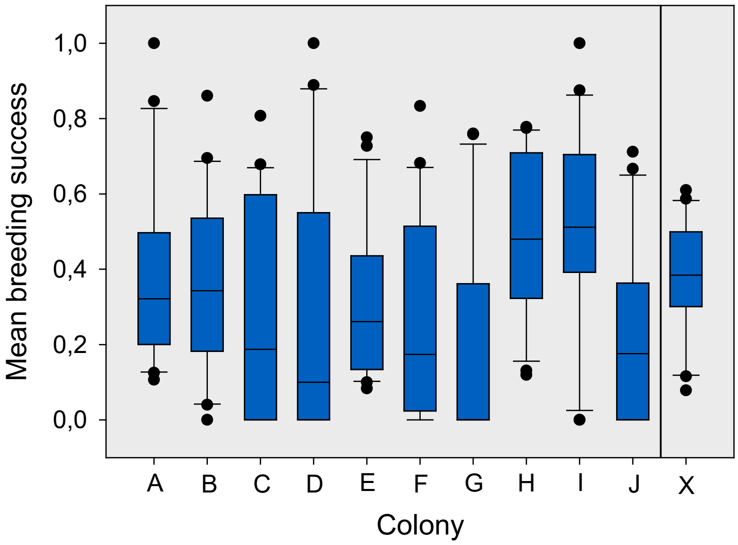


Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Compared to the previous study, the long-term development of breeding success no longer shows a significant trend in any of the individual colonies, but is characterised by clear annual fluctuations (Figure 8; Braun et al., 2022). Thus, previously declining breeding success has also stabilised in the colonies C – Nebles Point, E – Ardley Island and G – Halfthree Point (Figure 8 & 11; Braun et al., 2020c; Braun et al., 2022). The highest breeding success value of 0.81 juv/bp was recorded in the 2021/22 season at colony C – Nebles Point. In contrast, breeding success was unusually low in colonies A – North and I – Fildes Strait during the study period, with no young being successfully raised at the latter breeding site in all three seasons (Figure 8). Also striking was the low breeding success on Two Summit Island (colony H, Figure 8) in the 2023/24 season. As all these colonies are areas with a very low degree of anthropogenic influence, anthropogenic disturbance cannot explain these low breeding success values (Figure 11). Thus, other factors appear to be responsible. For example, significant predation by skuas may also have a negative impact on breeding success locally. On the other hand, the density of skua breeding pairs in the area was not higher than in previous years, so predation is unlikely to be the sole cause. Whether human disturbance was responsible for the complete breeding failure in the 2023/24 season at colony G – Halfthree Point (Figure 8), an area close to the station with a high level of human activity (Figure 11), remains to be seen.

With regard to the reaction of the southern giant petrel to human disturbance, the previous results can be largely confirmed (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). As in previous years, the individual colonies differed considerably in terms of their mean breeding success (Figure 9). The highest mean breeding success over the years was recorded at Two Summit Island (colony H, Figure 8).

Figure 9: Differences in breeding success between the individual southern giant petrel colonies in the Fildes Region; colony designation according to Peter et al., 2008, X = total breeding success in the Fildes Region, the boxplot shows the median, lower and upper quartiles and outliers (dots).



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

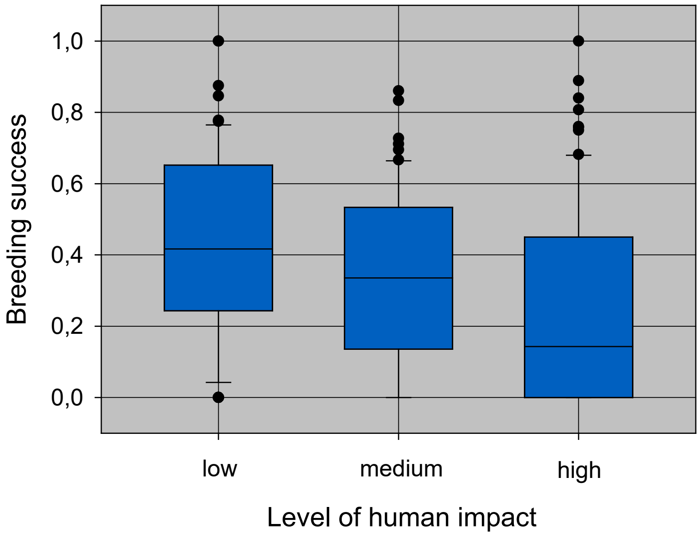
The previously shown correlation between the number of breeding pairs in the individual colonies and the number of breeding pairs in the previous year (Braun et al., 2022) can still be seen in several colonies (Table 2).

Table 2: **Relationship (Spearman correlation) between the number of breeding pairs of the southern giant petrel and the breeding success in the respective previous year in the Fildes Region; categories of human disturbance correspond to Figure 11**

colony	Degree of human disturbance	r_S	p	n
A	low	0.012	0.957	19
B	medium	0.308	0.171	21
C	high	**0.814**	**<0.001**	**24**
D	high	**0.692**	**<0.001**	**21**
E	high	0.357	0.085	24
F	medium	**0.665**	**<0.001**	**21**
G	high	**0.775**	**<0.001**	**20**
H	low	-0.274	0.238	20
I	low	0.301	0.221	18
J	low	**0.593**	**0.001**	**18**
total Fildes Region	-	0.223	0.314	22

In summary, only colonies with a low level of disturbance differ significantly from those with a medium and high level of disturbance (Kruskal-Wallis One-Way ANOVA on ranks: H = 15.733, $p < 0.001$; Figure 10).

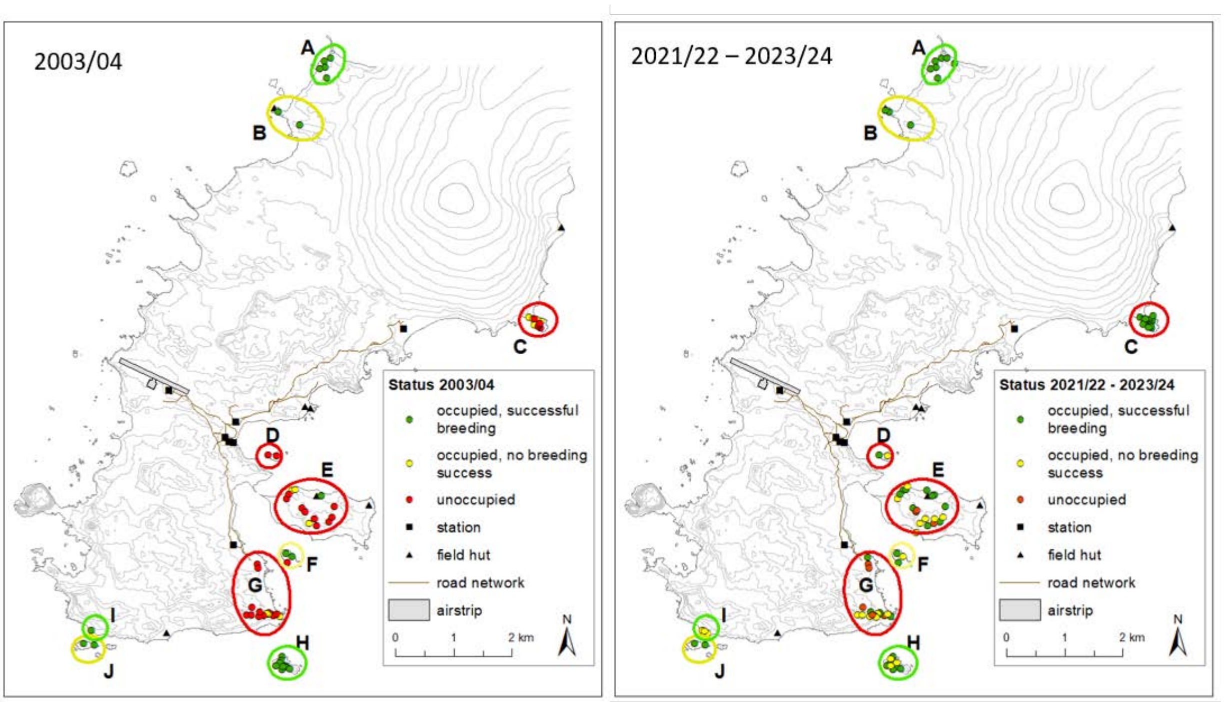
Figure 10: **Comparison of the breeding success of the southern giant petrel (*Macronectes giganteus*) in colonies in the Fildes Region between 1988/89 and 2023/24; the boxplot shows the median, lower and upper quartiles and outliers (dots)**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

A comparison of the distribution, breeding status and degree of human disturbance of all known breeding sites of the southern giant petrel since the 2003/04 season reveals clear changes (Figure 11). For example, numerous previously abandoned breeding sites were recolonised or successfully raised chicks for the first time (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). In addition, new breeding sites that were previously occupied by gulls, for example, but had been abandoned in the meantime, were increasingly occupied in the course of the strong population increase in the recent past. The most significant changes can be found in the colonies A – North, C – Nebles Point, D – Diomedea Island, E – Ardley Island and G – Halfthree Point, where numerous new breeding sites were recorded, as well as those where young were successfully raised for the first time in recent years (Figure 11). The strong growth in the number of breeding pairs and the high breeding success at colony D – Diomedea Island, which is regularly overflown by landing aircrafts (see below), is remarkable. In contrast, the lower breeding success in the 2021/22 to 2023/24 seasons is reflected in colonies I – Fildes Strait and H – Two Summit, where no young were raised at several occupied breeding sites (Figure 11). There were no signs of a breeding site shift triggered by intensive anthropogenic influences, as observed in the past (Peter et al., 1991; Chupin, 1997; Braun et al., 2012; Peter et al., 2013; Braun et al., 2017a), during the study period.

Figure 11: Title of figure Breeding sites of the southern giant petrel (*Macronectes giganteus*) in the Fildes Region, indicating the breeding status in the 2003/04 and 2021/22 to 2023/24 seasons (points indicate the breedings status); the degree of human disturbance is marked with coloured ovals: green - low, yellow - medium, red - high (based on own evaluation)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

For the Fildes Region, it had already been shown that there was a clear correlation of a rapid decline in breeding pair numbers of the giant petrel with the construction of new stations in the 1980s and the resulting high level of human disturbance in the surrounding area caused by frequent visits by station members and low overflights (Peter et al., 1991; Chupin, 1997; Braun et al., 2012). However, as populations in more distant, undisturbed areas increased at the same time, a breeding site shift due to the disturbance was inferred. Similar effects were observed in the 2000s due to intensive disturbance by visitors in the breeding areas, whereby the giant petrel populations recovered quickly after the disturbance was reduced or eliminated (Braun et al., 2012; Braun et al., 2017a). In line with the observations from the Fildes Region, similar declines in southern giant petrel populations have also been documented in other areas of Antarctica as a result of station activities (Jouventin & Weimerskirch, 1991; Micol & Jouventin, 2001; Woehler et al., 2001; Nel et al., 2002; Sander et al., 2005; Chwedorzewska & Korczak, 2010). In addition, tourism activities can also have a negative impact on local breeding populations, as shown, for example, for Harmony Point, Nelson Island, where the number of breeding pairs recovered after visitor routes were restricted to a defined path (Silva et al., 1998; Krüger, 2019a). However, as tourist activities in the Fildes Region are largely limited to the station areas and guided tours, a negative impact on breeding giant petrels appears negligible.

Another positive development is the recent consistently high overall breeding success of the southern giant petrel in the Fildes Region. In the past, this was significantly below the values from other breeding areas (Woehler et al., 2003; Patterson et al., 2008; Dunn et al., 2016b; ACAP, 2024a), and was partly attributed to the high level of human disturbance (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). Low breeding success over many years can have a long-term negative impact on the population despite low mortality in adult and immature birds (Woehler et al., 2003; Patterson et al., 2008; Cuthbert et al., 2014). However, as there has been a strong population increase in the Fildes Region, the previously reduced breeding success does not appear to have had such a negative impact or appears to have been compensated for by immigration by breeding birds from other areas.

The currently observed strong population growth is particularly remarkable after the strong impact of human activities in the past (Peter et al., 1991; Chupin, 1997; Braun et al., 2012). However, as these continue in certain areas, especially in the form of visitors to breeding areas or intensive air traffic, the current results of long-term monitoring point to a clear habituation effect with regard to certain human disturbances (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). Habituation can occur in animals as a gradual decrease in behavioural response as a result of repeated stimulation without subsequent reinforcement (Hinde, 1970). It has been shown several times that seabirds also show reduced behavioural responses to permanent or regular stressors, such as noise or visitors (e.g. Young, 1990a; Nimon, 1997; Cobley & Shears, 1999; Pfeiffer & Peter, 2004; Holmes et al., 2006; Walker et al., 2006; De Villiers, 2008; Viblanc et al., 2012).

In line with this, both short- and long-term habituation effects have been experimentally demonstrated in individual giant petrel colonies in the Fildes Region (Pfeiffer, 2005). In contrast, the probability of a weakening of behavioural responses is lower in less affected areas, where disturbances occur at more irregular intervals and from different directions (Pfeiffer, 2005; Pfeiffer et al., 2007). Habituation is most clearly reflected in the sharp increase in the number of breeding pairs and the recent consistently high breeding success on Diomedea Island (colony D), as this island is located in the approach zone for the local airport and is frequently overflown by low-flying aircraft taking off or landing. After the ‘Guidelines for Aircraft near Concentrations of Birds in Antarctica’ (ATS, 2004) came into force, a reduction in the number of helicopters, which had previously often operated irregularly, was recorded (Peter et al., 2008; Peter et al., 2013). This colony was recolonised during this period and now has a steadily increasing number of breeding pairs and a high breeding success.

Whether the current increase in the number of breeding pairs of the southern giant petrel in the Fildes Region is also reflected in other ice-free areas of Maxwell Bay remains unclear due to a lack of data from there (see chapter 6). For example, the population of this species on Stansbury Peninsula and Cabo Cariz in the north of Nelson Island has increased since the 1980s (Braun et al., 2022), but has declined significantly in recent years (see chapter 6.7). There is insufficient current information available for other areas (see chapter 6.2, 6.5 & 6.6; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022; ACAP, 2024a).

The development of the southern giant petrel population in the Fildes Region and other areas of Maxwell Bay shown here is consistent with published information on regional positive population trends (BirdLife International, 2024e). For the South Shetland Islands, for example, numerous studies on stable or increasing populations (Lynch et al., 2008; Gil-Delgado et al., 2013; Cuthbert et al., 2014; Petry et al., 2016; Krüger et al., 2017; Petry et al., 2018; Krüger, 2019a) contrast with relatively few reports of decreasing trends (Korczak-Abshire et al., 2019; Krüger, 2019a). According to the last available population estimate for the South Shetland Islands, a total of 5,400 breeding pairs breed here (Patterson et al., 2008). More recent data is not available for this region. As a result of the strong population increase, the breeding area in the Fildes Region appears to be the second largest breeding site in the South Shetland Islands after Stinker Point, Livingston Island (Harris et al., 2015; Petry et al., 2018; Krüger, 2019b). However, as no more recent population estimates are available, it is not possible to make a reliable statement about the proportion of giant petrels breeding in the Fildes Region in the regional or global population of the southern giant petrel.

The reasons for the global population increase of the southern giant petrel are not yet fully understood. Environmental factors such as climatic conditions, food availability, diseases or disturbances are of great importance for bird populations (van Franeker et al., 2001; Creuwels et al., 2005; Descamps et al., 2015; Dunn et al., 2016b; Petry et al., 2016; Ainley et al., 2024). The southern giant petrel may be benefiting from the current climatic changes (Krüger et al., 2012; Gianuca et al., 2019). However, the extent to which they are or could be affected by expected or already occurring changes in the food web is still unclear. Furthermore, southern giant petrels, like other seabird species, feed to a considerable extent on fishery waste (Favero et al., 2003; Copello & Quintana, 2009; Bugoni et al., 2010). For example, the sharp increase in fishing activities appears to be responsible for the population increase of the southern giant petrel in South America (Quintana et al., 2006; Copello et al., 2008; Copello & Quintana, 2009), Îles Crozet (Voisin, 1988) and Elephant Island, South Shetland Islands (Krüger et al., 2017). The increased intake of fishery waste as food leads to a higher survival rate of adult animals (Kock, 2001; Krüger et al., 2017). Although bycatch causes additional mortality of birds (González-Solís et al., 2000; Patterson & Hunter, 2000; Kock, 2001; Gianuca et al., 2019), this may be too low in the case of giant petrels to have a significant impact on the overall population (Quintana et al., 2006; Krüger et al., 2017). However, another study emphasises possible negative population effects due to increasing pelagic longline fishery through increased bycatch of female adults (Gianuca et al., 2019). As both sexes differ in terms of their habitat selection (Gianuca et al., 2019; Finger et al., 2023), different responses to changing environmental conditions are to be expected.

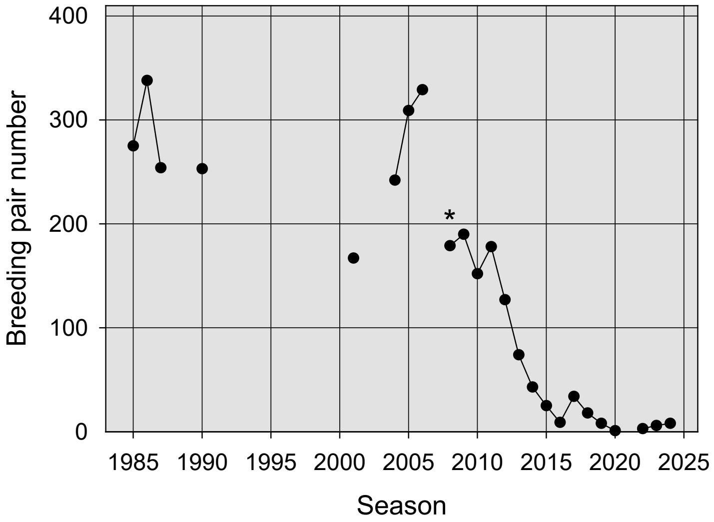
In conclusion, with regard to the population development of the southern giant petrel in the Fildes Region, possible influences of large-scale environmental factors probably outweigh those of local environmental conditions and negative anthropogenic influences.

Several studies are currently underway on southern giant petrels breeding in the Fildes Region (pers. comm. S. Lisovski, L. Krüger), which could provide valuable data on feeding and wintering areas and thus on the causes of current population trends. In the future, the use of a UVA/drone is recommended for the implementation of giant petrel monitoring despite existing logistical limitations. The feasibility of this has already been demonstrated in various studies (Braun et al., 2017a; Fudala & Bialik, 2022; Larsen et al., 2024). In addition, this can reduce the number of visits to the giant petrel colonies and thus further minimise the level of human disturbance.

5.4 Cape petrel (*Daption capense*)

The cape petrel (*Daption capense*) in the Fildes Region has shown a drastic decline in the breeding population in recent decades (Braun et al., 2020c; Braun et al., 2021; Braun et al., 2022). The population decline since the start of long-term recording in the 1984/85 season is currently around 98 %. In the 2021/22 to 2023/24 seasons, very low numbers of 3 to 6 breeding pairs were again recorded (Figure 12). In each case, one to three of the 36 known breeding sites were occupied. The majority of these were located in the extreme south-west of the Fildes Peninsula (Figure 13).

Figure 12: Development of the number of breeding pairs of the cape petrel (*Daption capense*) in the Fildes Region since the 1980s (* incomplete survey)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Due to a lack of data required for this purpose, the reasons for the observed decline in the cape petrel population in the Fildes Region are difficult to determine. Possible causes include high predation pressure from skuas, but above all environmental changes such as food shortages and poor weather conditions (Braun et al., 2021). Human activities, on the other hand, play a subordinate role, as the majority of breeding areas are located in remote and unvisited areas. The relationship between the number of breeding pairs of the cape petrel and the number of the visitors of the International Association of Antarctica Tour Operators (IAATO) claimed by Jiang et al. (2022) is therefore incorrect.

Figure 13: **Location and size of cape petrel (*Daption capense*) breeding colonies in the seasons 2021/22 to 2023/24**

Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

A comparative supraregional assessment of population development is only possible to a limited extent due to a lack of data on breeding populations of this species in the Antarctic. In addition, population estimates are usually based on older population surveys or only on a few, sometimes widely separated survey years (e.g. Woehler & Croxall, 1997; Silva et al., 1998; González-Zevallos et al., 2013; Sierakowski et al., 2017). However, initial indications of a declining population trend were provided by surveys from Admiralty Bay, King George Island (Petry et al., 2016, Stinker Point, Elephant Island (Petry et al., 2018) or Danco Coast (González-Zevallos et al., 2013). However, a more recent study from East Antarctica showed no significant changes in the cape petrel population between the 1970s and the 2017/18 season in the Vestfold Hills, where one of the largest populations of this species is found (Kliska et al., 2022). In contrast, a growing population of the cape petrel was detected in a long-term study of a larger colony at Pointe Géologie, Terre Adélie (Sauser et al., 2023). Here, the population has grown despite declining breeding success in recent decades. It was also shown that the population regulation was mainly controlled by the southern annular mode (SAM) (Sauser et al., 2023). The Southern Annular Mode (SAM) or Antarctic Oscillation describes the atmospheric circulation of air masses between higher and middle latitudes and is responsible for a considerable part of the total Southern Hemisphere climate variability (Marshall, 2003, 2007). There is a positive correlation between SAM and the sea surface temperature in the northern Antarctic Peninsula (Marshall, 2007). Thus, a positive phase of SAM leads to cold sea surface temperature and high chlorophyll-a values around 60 °S (Sauser et al., 2023), which is considered a proxy for food availability in the Western Antarctic Peninsula (Silk et al., 2016). For the colony at Pointe Géologie, Terre Adélie, it was shown that an increased survival probability of immature and adult cape petrels caused by the SAM led to population growth despite a long-term decline in breeding success (Sauser et al., 2023). However, such an effect is not recognisable for the breeding population of the Fildes Region in view of the drastic decline.

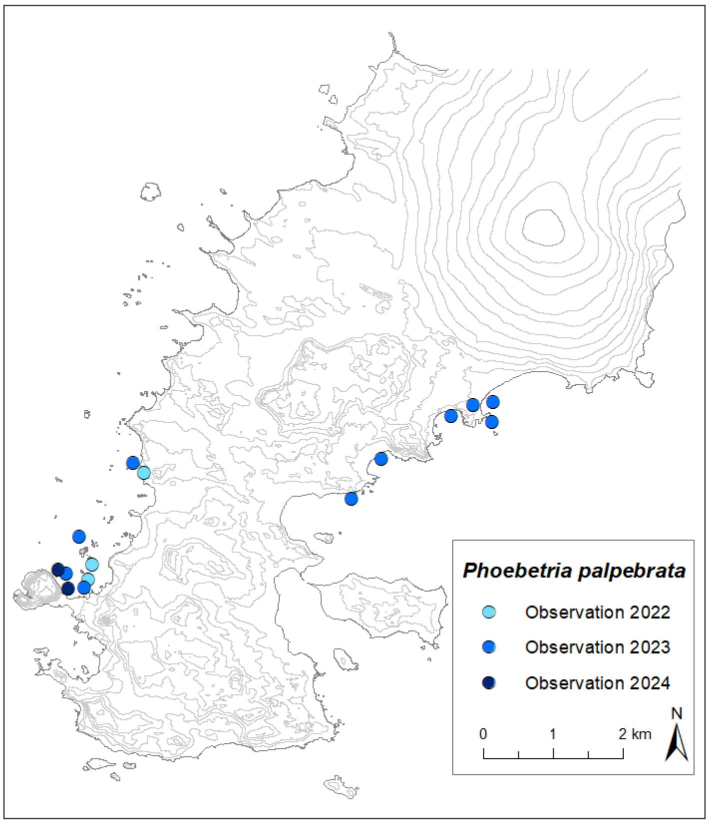
5.5 Light-mantled sooty albatross (*Phoebetria palpebrata*)

The light-mantled sooty albatross (*Phoebetria palpebrata*) is subject to various risks that could threat the population of this long-lived species in the long term. In particular, additional mortality risks due to the negative consequences of longline fishing and predators introduced into the breeding area have led to this species currently being classified as ‘Near Threatened’ (BirdLife International, 2024b) and listed in the Agreement on the Conservation of Albatrosses and Petrels (ACAP; ACAP, 2024b). The global population of this species is currently estimated at 19,000 to 24,000 breeding pairs (BirdLife International, 2024b).

The first documented sightings of the light-mantled sooty albatross, which is considered philopatric (Marchant & Higgins, 1990), in the Fildes Region date back to the 1984/85 season (Peter et al., 1988). Thereafter, numerous further sightings of this species, whose range of the sooty albatross extends circumpolar to various Subantarctic islands near the Antarctic Convergence between 46° and 55° S (Marchant & Higgins, 1990), have been documented almost every year (see Table 11 in Annex C). In the 2008/09 season, the first evidence of breeding of this species, including the documentation of chicks, was found at the breeding site on Flat Top on the west coast of the Fildes Peninsula (Lisovski et al., 2009). This is the only breeding record of this species in the Antarctic to date and therefore well outside its traditional range.

In the 2021/22 to 2023/24 seasons, synchronised flights of albatross pairs were observed several times (Figure 14). However, no other signs that could indicate breeding, such as clear droppings at the known breeding site on the Flat Top or adult birds repeatedly perching or flying in, were observed. This means that the last suspected breeding of the sooty albatross in the Fildes Region dates back to the 2018/19 and 2019/20 seasons (Braun et al., 2022). So far, almost all observations have been made on the west coast of the Fildes Peninsula, particularly in the area around the Flat Top breeding site. Thus, only one observation each of light-mantled sooty albatrosses from the east coast of the Fildes Peninsula was reported from the seasons 1987/88 (Nadler & Mix, 1989) and from the south of the Fildes Peninsula in the season 2011/12 (Lumpe & Weidinger, 2000). In contrast, numerous observations of flying light-mantled sooty albatrosses were recorded for the first time on the east coast of the Fildes Peninsula in the 2022/23 season (Figure 14). On five days in December, January and February 2023, one to two flying albatrosses were observed there, including synchronous display flights (Figure 15). In addition, two adults were observed sitting on a scree slope near the Uruguayan station of Artigas. Both individuals showed clear courtship behaviour and stayed at this location for a longer period of time (Figure 15). It is possible that these observations represent the first signs of this species becoming established in the east of the Fildes Peninsula.

Figure 14: Observations of light-mantled sooty albatrosses (*Phoebetria palpebrata*) in the Fildes Region in the 2021/22 to 2023/24 seasons



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Figure 15: Light-mantled sooty albatrosses with synchronised flight and courtship behaviour, observed on the east coast of the Fildes Peninsula in the 2022/23 season

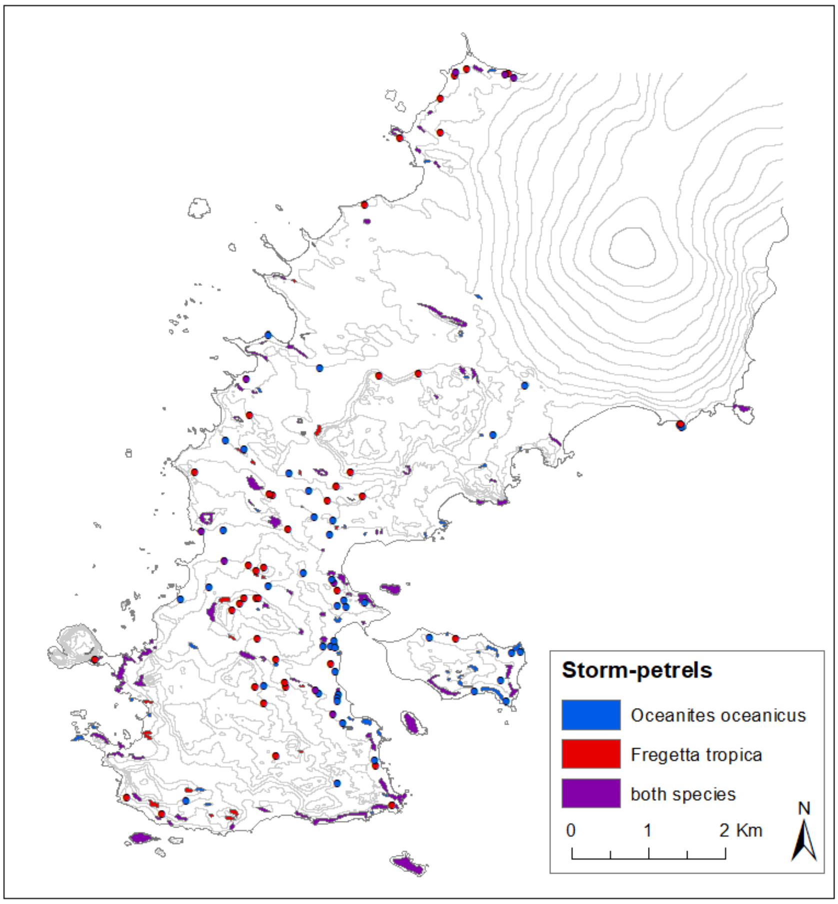
Photos: H. Grämer, Institute of Ecology and Evolution, Friedrich Schiller University Jena

5.6 Wilson’s storm-petrel (*Oceanites oceanicus*) and black-bellied storm-petrel (*Fregetta tropica*)

Following initial surveys of the two native storm-petrel species Wilson's storm-petrel (*Oceanites oceanicus*) and black-bellied storm-petrel (*Fregetta tropica*) in the Fildes Region in the 1980s (Bannasch & Odening, 1981; Roby et al., 1986; Lange & Naumann, 1989; Nadler & Mix, 1989; Peter et al., 2008), a systematic survey based on nocturnal calls during the breeding season was carried out in the 2003/04 to 2005/06 seasons. The survey revealed the widespread distribution in the area and the high number of breeding pairs, although the latter was based on estimates due to the storm-petrels' hidden lifestyle (Peter et al., 2008). In the following years, the data set was supplemented accordingly based on individuals heard from the breeding cavities during the day. These were always smaller breeding sites with only a few individuals.

In the 2021/22 to 2023/24 seasons, the existing information (Braun et al., 2022) on the breeding distribution of the two native storm-petrel species was supplemented by three additional, smaller sites. These were two breeding sites for the Wilson’s storm-petrel and one breeding site for the black-bellied storm-petrel, each with a maximum of 10 breeding pairs. These are located on the west and east coasts of the Fildes Peninsula and added to the existing knowledge of the distribution of all known storm-petrel breeding sites in the Fildes Region (Figure 16).

Figure 16: Distribution of breeding sites of the Wilson’s storm-petrel (*Oceanites oceanicus*) and black-bellied storm-petrel (*Fregetta tropica*) in the Fildes Region; data from surveys between seasons 2003/04 and 2023/24



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

A total of 344 breeding sites have now been recorded. At most breeding sites (39 %), both species bred sympatrically, compared to 35 % with only Wilson's storm-petrels and 26 % with only black-bellied storm-petrel. Furthermore, small breeding sites with up to 10 breeding pairs predominate. Overall, the breeding population in the Fildes Region is still estimated at 3,500 to 5,000 Wilson's storm-petrels and 500 to 1,000 black-bellied storm-petrels (Peter et al., 2008). Based on the high breeding site fidelity or philopatry of storm-petrels (Marchant & Higgins, 1990; Warham, 1990), the present dataset is considered to provide largely complete information on the spatial distribution of these storm-petrel species in the study area.

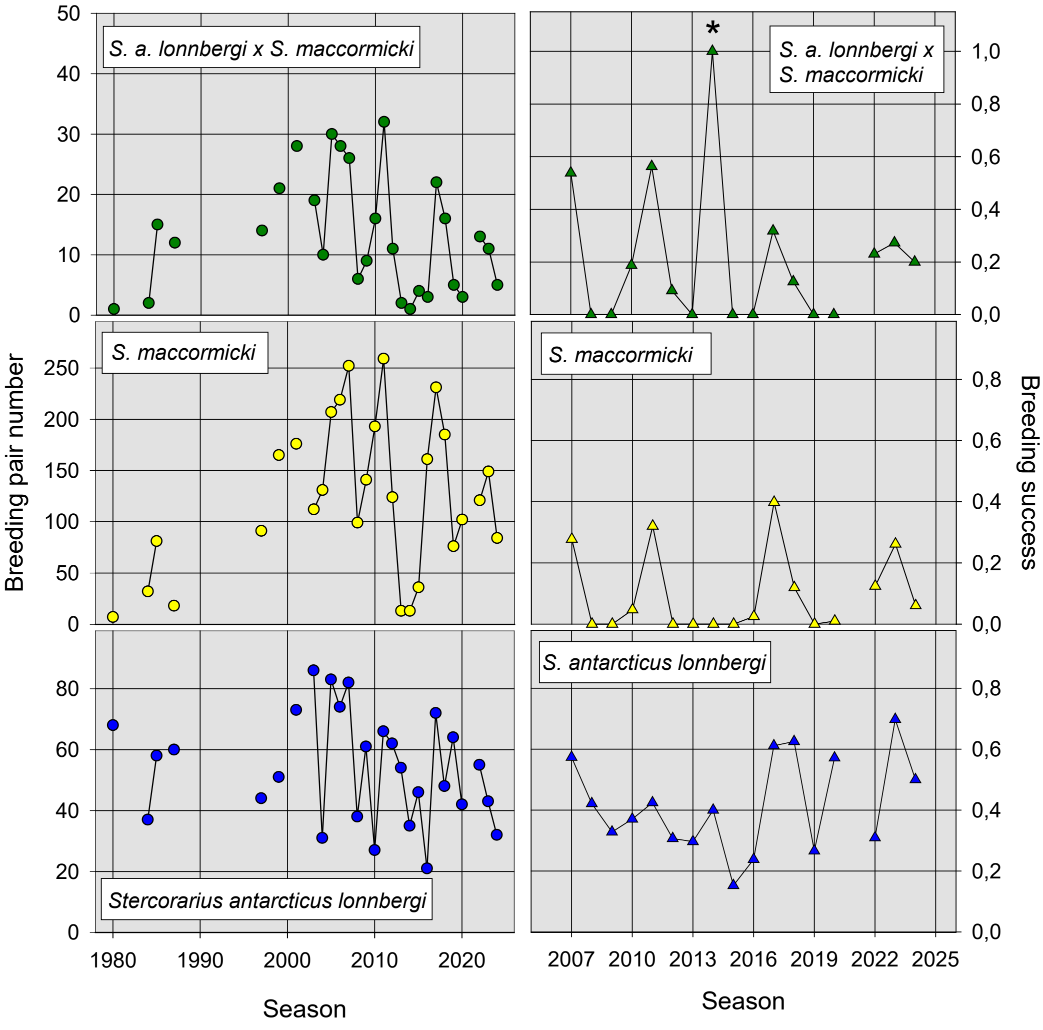
As both storm-petrel species birds return to their breeding burrows at night due to the high predation pressure, hardly any published mapping results are known, making the comprehensive data set on breeding distribution presented here a rarity. With regard to spatial distribution, the method used offers advantages over randomised surveys, in which highly localised species that can easily be missed within the landscape, despite the high effort involved (Bird et al., 2022). However, a major disadvantage is the lack of accuracy in population estimation, which makes it difficult to detect any population changes (Arneill et al., 2019). However, a current comprehensive study on temporal population trends of Wilson's storm-petrels on King George Island is another speciality (Ausems et al., 2023). A 90 % decrease in population size and significant changes in reproductive success and chick growth were observed, with these changes being associated with interannual environmental variation (Ausems et al., 2023).

5.7 Skuas (*Stercorarius* spec.)

A data set of up to 35 years over a period of 45 years is now available on the breeding pair population of the brown skua (*Stercorarius antarcticus lonnbergi*), the south polar skua (*S. maccormicki*) and their hybrid and mixed pairs in the Fildes Region. As the study area in the Fildes region is located in the hybridisation zone of the skua species (Ritz et al., 2006), where both species produce viable and fertile offspring, the separation of species is now increasingly being discussed (Mota et al., 2023). Hence, mixed pairs or pairs in which one partner has intermediate characteristics, i.e. is presumably a hybrid of both species, are also regularly found in the Fildes Region. To minimize inaccuracies due to an increased number of pairs of unknown pair type, , the “estimated pair type” according to Krietsch et al. (2016) was used below for pairs where only one partner is known, which refers to the respective species affiliation of the known partner (see chapter 4).

The survey of skua breeding pair numbers in the Fildes Region revealed a breeding population of between 32 and 55 bp of brown skuas, between 84 and 149 bp of south polar skuas and between 5 and 13 mixed and hybrid pairs for the study period. Thus, after earlier collapses, the local skua population was in the average range, with the long-term mean of the breeding pair numbers for brown skuas at 55 bp ± 18, for south polar skuas at 126 bp ± 74 and for mixed and hybrid pairs at 14 bp ± 11 (Figure 17). Overall, the populations of the different skua pair types remained stable in the long-term comparison despite considerable fluctuations and showed no significant temporal trend (brown skua: $R^2 = 0.055$, $p = 0.221$; south polar skua: $R^2 = 0.083$, $p = 0.130$; mixed & hybrid pairs: $R^2 = 0.009$, $p = 0.632$). Nevertheless, the population of the south polar skua was still significantly lower at the beginning of the survey in the 1980s than in previous decades (Figure 17). Overall, the number of breeding pairs of brown skua and south polar skua in the Fildes Region tended to develop evenly, although the significance level was narrowly missed (Pearson corr.: $r = 0.335$, $p = 0.076$, $n = 29$). Nevertheless, there were differences in the reaction of both species to certain environmental influences. Low snow cover in the austral spring and early summer therefore is considered to be of great importance for ground-nesting birds for the decision to start breeding. There was a significant negative correlation between the mean snow depth in November and December and the number of breeding pairs of brown skuas in the Fildes Region (Spearman corr.: r = -0.501, p = 0.007, n = 28). For south polar skuas, such a correlation was indicated, but fell just short of the significance level (r = -0.360, p = 0.059, n = 28). This illustrates the importance of snow-free breeding sites for these ground-nesting species, which, unlike other species, do not breed on cliffs but mostly on low hills, which are often still covered in snow well into spring. Thus, the lowest breeding pair numbers of skuas in the Fildes Region were recorded in seasons with high snow cover at the beginning of the breeding season (Braun et al., 2022). The positive correlation between the air temperature in spring before the start of breeding and the total number of breeding pairs of skuas, as shown for the neighbouring Admiralty Bay (Petry et al., 2016; Phillips et al., 2019), was also confirmed (Spearman corr.: r = 0.392, p = 0.020, n = 35).

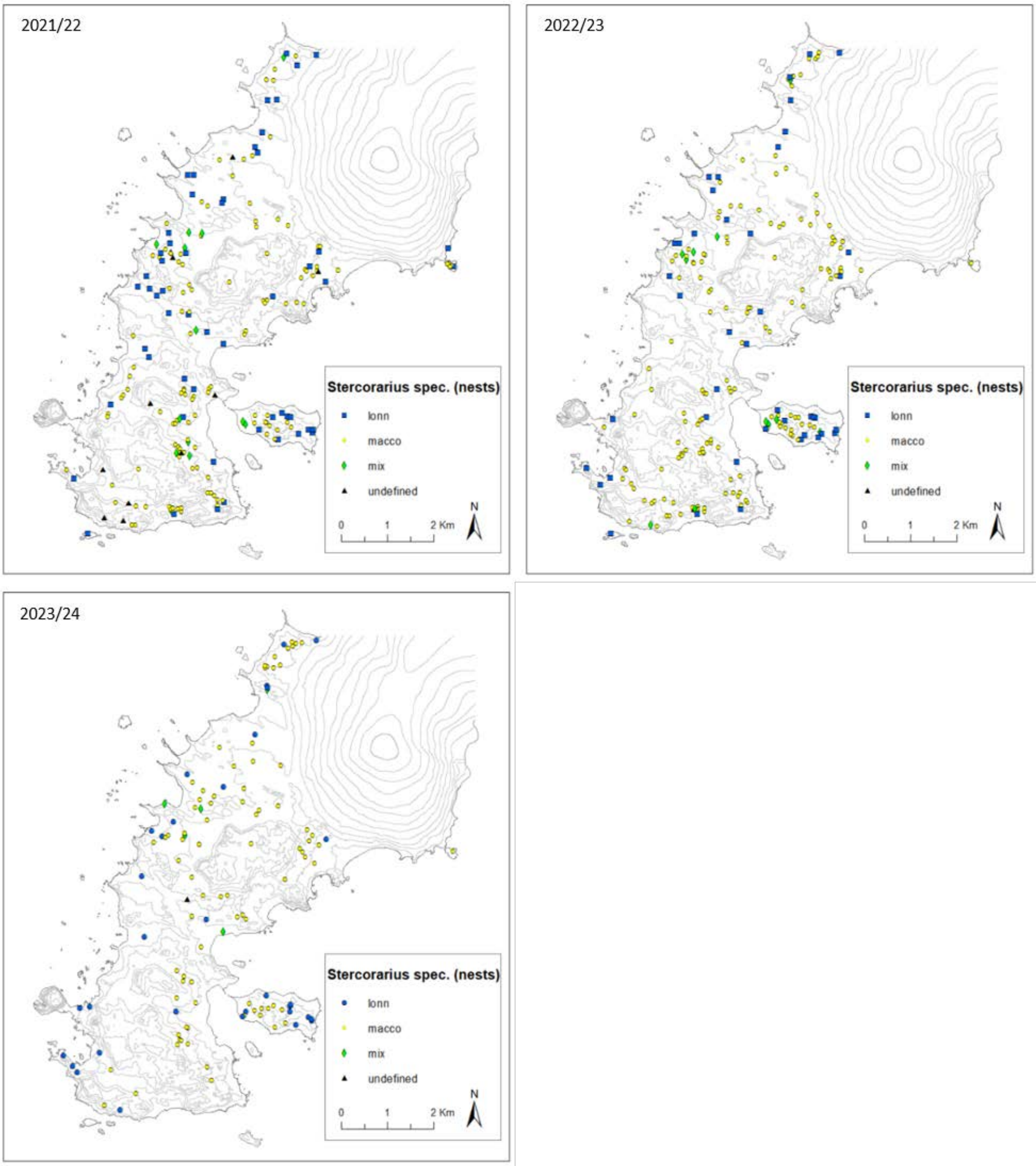
Figure 17: Development of breeding pair numbers and breeding success of brown skuas (*Stercorarius antarcticus lonnbergi*), south polar skuas (*S. maccormicki*) and their mixed and hybrid pairs (*S. antarcticus lonnbergi* x *S. maccormicki*) in the Fildes Region between the 1979/80 and 2023/24 seasons, respectively. (* value is based on one breeding pair only). Note the different scaling of the X and Y axes; assignment of the pairs as 'estimated pair type'.



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The breeding success of the two skua species in the Fildes Region showed considerable fluctuations over the years studied and again differed significantly from each other (Figure 17). Brown skuas successfully raised relatively more chicks than south polar skuas. In line with this, the breeding success of brown skuas, with an average of 0.42 ± 0.16 juv/bp, was significantly higher than that of south polar skuas with 0.10 ± 0.13 juv/bp. A temporal trend could not be determined for any pair type (brown skua: $R^2 = 0.030$, $p = 0.494$; south polar skua: $R^2 = 0.006$, $p = 0.763$; mixed & hybrid pairs: $R^2 = 0.021$, $p = 0.571$). Both values of breeding success showed a positive correlation (Spearman corr.: $r = 0.678$, $p = 0.002$, $n = 18$) and thus developed in a similar way regarding the response to environmental conditions.

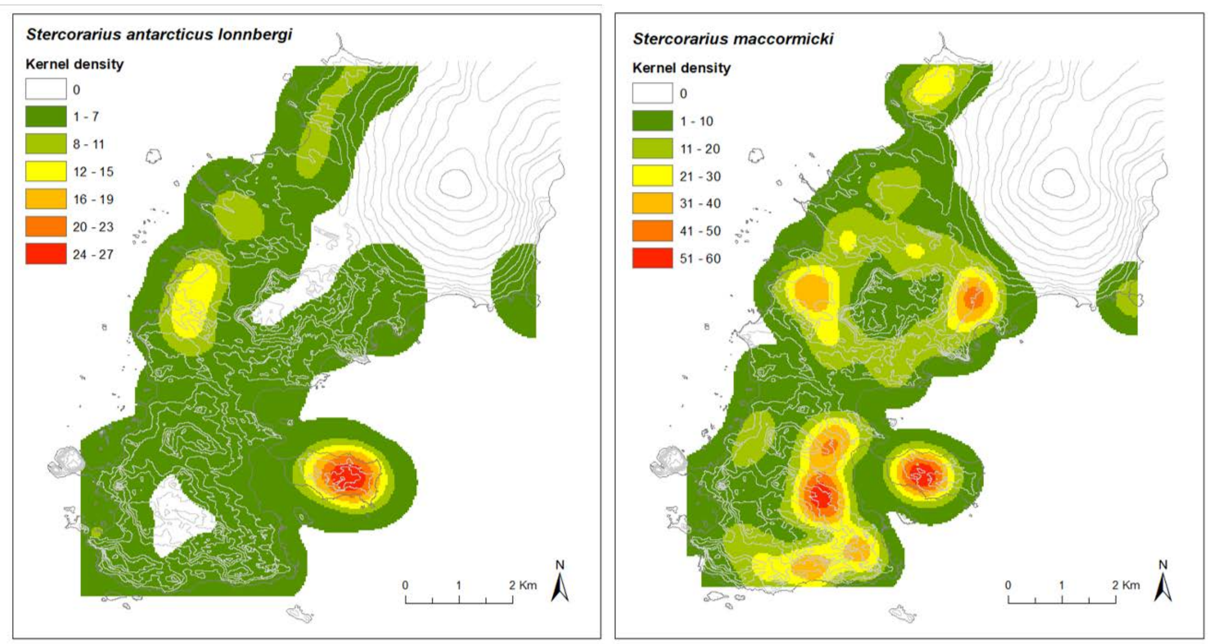
Figure 18: Spatial distribution of skua nests (*Stercorarius antarcticus lonnbergi*, *S. maccormicki*, hybrid and mixed pairs) in the Fildes Region in the 2021/22 to 2023/24 seasons; assignment of the pairs as 'estimated pair type'.



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The spatial distribution of skua breeding pairs in the three seasons studied, shown in Figure 18 & 19, largely corresponds to the data from previous years with regard to the greater concentration of brown skuas near penguin colonies and the coast (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). In contrast, there are significantly more south polar skuas in the southeast of the Fildes Peninsula (Figure 19).

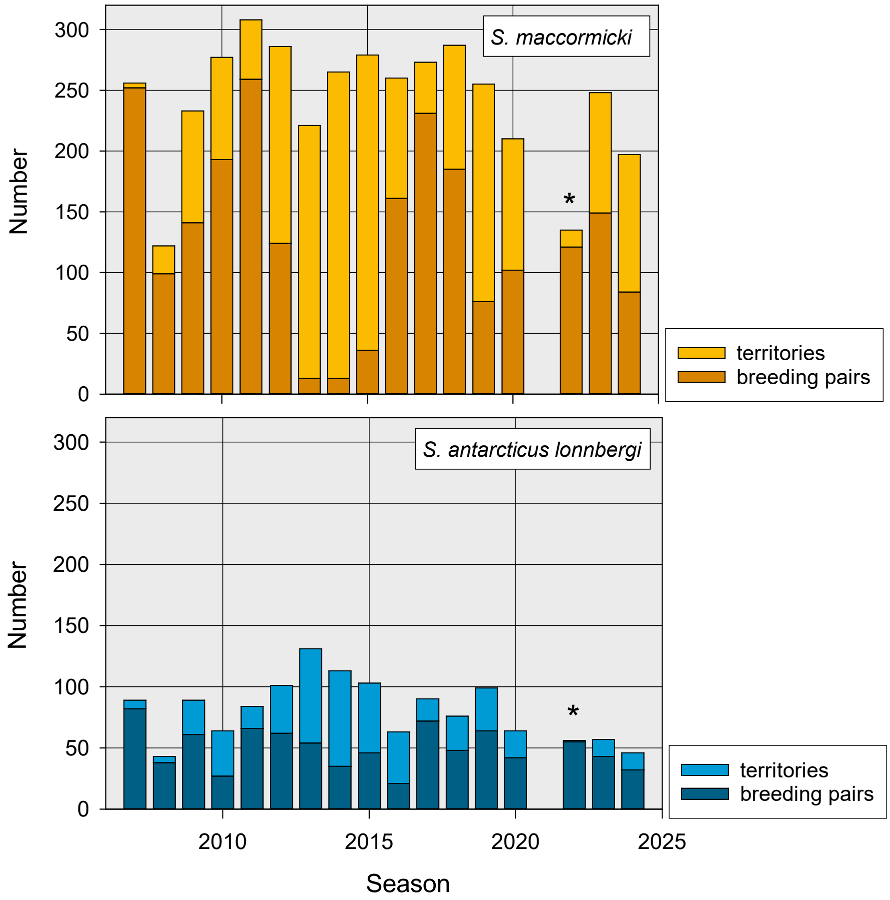
Figure 19: Kernel density of the nest of brown skua (*Stercorarius antarcticus lonnbergi*) and south polar skua (*S. maccormicki*) in the Fildes Region in the 2021/22 to 2023/24 seasons; assignment of the pairs as 'estimated pair type'.



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

In addition to the number of breeding pairs, the number of recorded skua territories in the Fildes Region that were occupied but where breeding had not initiated has been recorded since the 2006/07 season. In the 2021/22 season, however, it was not possible to record the full number of territories for personnel reasons. It becomes clear that there were significantly fewer fluctuations in the total number of pairs of skuas present in the breeding area in summer than when only breeding pairs were considered (Figure 17 & 20). A long-term comparison clearly shows that southern skuas in particular often occupied territories in large numbers but did not start breeding (Figure 20). This clearly indicates a greater dependence of south polar skuas on environmental factors that influence the decision to breed or not.

Figure 20: **Overview of the numbers of breeding pairs and territories of brown skuas (*Stercorarius antarcticus lonnbergi*), south polar skuas in the Fildes Region between 2007/08 and 2023/24 seasons (* incomplete data on territories); assignment of the pairs as 'estimated pair type'.**



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Furthermore, the correlation shown between the breeding success of brown skuas and south polar skuas indicates that the reproduction of both species is similarly influenced by environmental factors. As described in Braun et al. (2022), the reason for the different strong fluctuations in the breeding population and breeding success of the two species is more likely to be a different response to environmental factors and the use of different food sources by both skua species during the breeding season. According to isotope analyses, there is a certain overlap in the trophic niches of brown and south polar skuas and thus potential food competition between the two species (Gal et al., 2021). Nevertheless, a greater dependence of south polar skuas on marine food sources, whose availability is generally more variable (Reinhardt et al., 2000), appears to lead to strong fluctuations in breeding pair numbers and breeding success (Hamer et al., 1991; Phillips et al., 1996; Reinhardt, 1997; Hahn et al., 2007; Hahn et al., 2008; Montalti et al., 2009; Krietsch et al., 2016). In contrast, brown skuas primarily use more stable food sources such as colonies of penguins and other seabirds (Hahn & Peter, 2003; Hahn & Bauer, 2008; Carneiro et al., 2015; Krietsch et al., 2016; Reis et al., 2021). It has been shown several times that brown skuas that breed near a penguin colony and thus have direct access to this resource can have a reproductive advantage, e.g. an earlier hatching time (Kim et al., 2022). In contrast, brown skuas that breed further away from penguin colonies or parents towards the end of the breeding season also appear to make greater use of marine food

(Reinhardt, 1997; Reinhardt et al., 2000; Hahn & Bauer, 2008; Carneiro et al., 2015; Ibañez et al., 2022; Kim et al., 2022; Graña Grilli et al., 2023). Reduced food availability can affect seabirds, including skuas, in poorer body condition, to which they may respond by abandoning breeding (Ainley et al., 1990b; Phillips et al., 2019), or lead to a reduction in breeding success in the year in question (Hamer et al., 1991; Phillips et al., 1996; Reinhardt, 1997; Pezzo et al., 2001; Graña Grilli, 2014; Krietsch et al., 2016; Graña Grilli et al., 2018). Increased predation pressure from conspecifics can also significantly reduce the breeding success of both species (Hamer et al., 1991; Phillips et al., 1998; Reinhardt et al., 2000; Graña Grilli, 2014; Krietsch et al., 2016), as prolonged foraging flights lead to a reduced presence at the nest. Furthermore, it has been shown that younger skuas have lower vital rates than older age classes, i.e. individual quality and environmental variability should be considered together (Pacoureau et al., 2019).

Based on the data presented here, it is clear that the skua population of both species is stable over the long term in the Fildes Region. A further increase in the south polar skua, as previously described for both the Fildes Region (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a) and other areas of the Antarctic (Micol & Jouventin, 2001; Woehler et al., 2001; Costa & Alves, 2008; Carneiro et al., 2010; Krietsch et al., 2016; Phillips et al., 2019), is currently no longer detectable. The severe population declines and repeated complete breeding failures of south polar skuas that have been documented for the Fildes Region have also been described for the neighbouring Potter Peninsula, KGI, (Krietsch et al., 2016; Graña Grilli et al., 2018) or in other areas of Antarctica (Descamps et al., 2023).

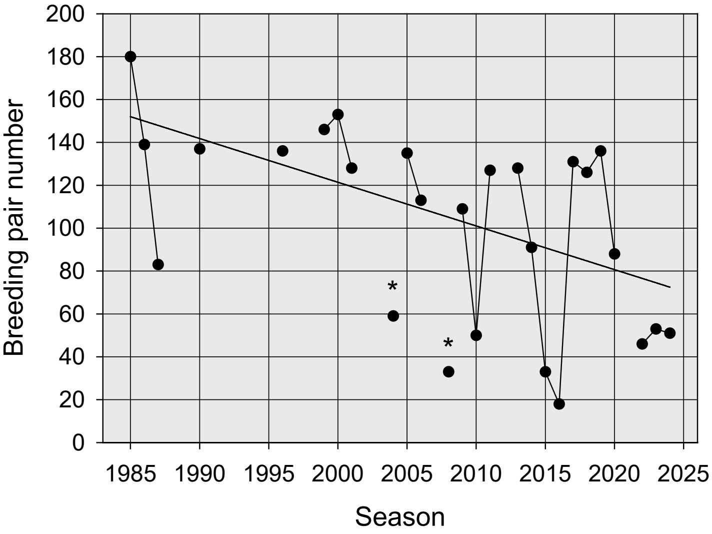
In all three studied seasons, active feeding of skuas by station personnel was again observed in the majority of stations on the Fildes Peninsula, or inferred by corresponding findings of fresh food remains on nests near the station (see also Reinhardt, 1997; Peter et al., 2008; Braun et al., 2012; Peter et al., 2013; Braun et al., 2014; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022), although this practice contradicts the mandatory requirements of the Madrid Protocol (Annex II and III).

5.8 Kelp gull (*Larus dominicanus*)

The last, but now outdated, estimate of the global population of the kelp gull, a widespread breeding bird in the southern hemisphere, was 3.3 to 4.3 million individuals, with a positive trend (BirdLife International, 2024a).

During the breeding bird survey in the Fildes Region in the 2021/22 to 2023/24 seasons, a total of 46, 53 and 51 breeding pairs of kelp gulls (*Larus dominicanus*) were recorded, respectively (Figure 21). These values are well below the long-term average of 107 ± 43 recorded, covering 24 years since the start of the survey in the 1984/85 season. Based on the current breeding pair numbers populations, the negative trend in the population has intensified in the long-term comparison, caused by the higher numbers of breeding pairs in the 1980s and 1990s ($R^2 = 0.330$, $p = 0.003$, $n = 24$, Braun et al., 2022). In the period between 2004/05 and 2023/24, however, the kelp gull population remained stable ($R^2 = 0.162$, $p = 0.122$, $n = 14$, Braun et al., 2022), although significant fluctuations were recorded (Figure 21).

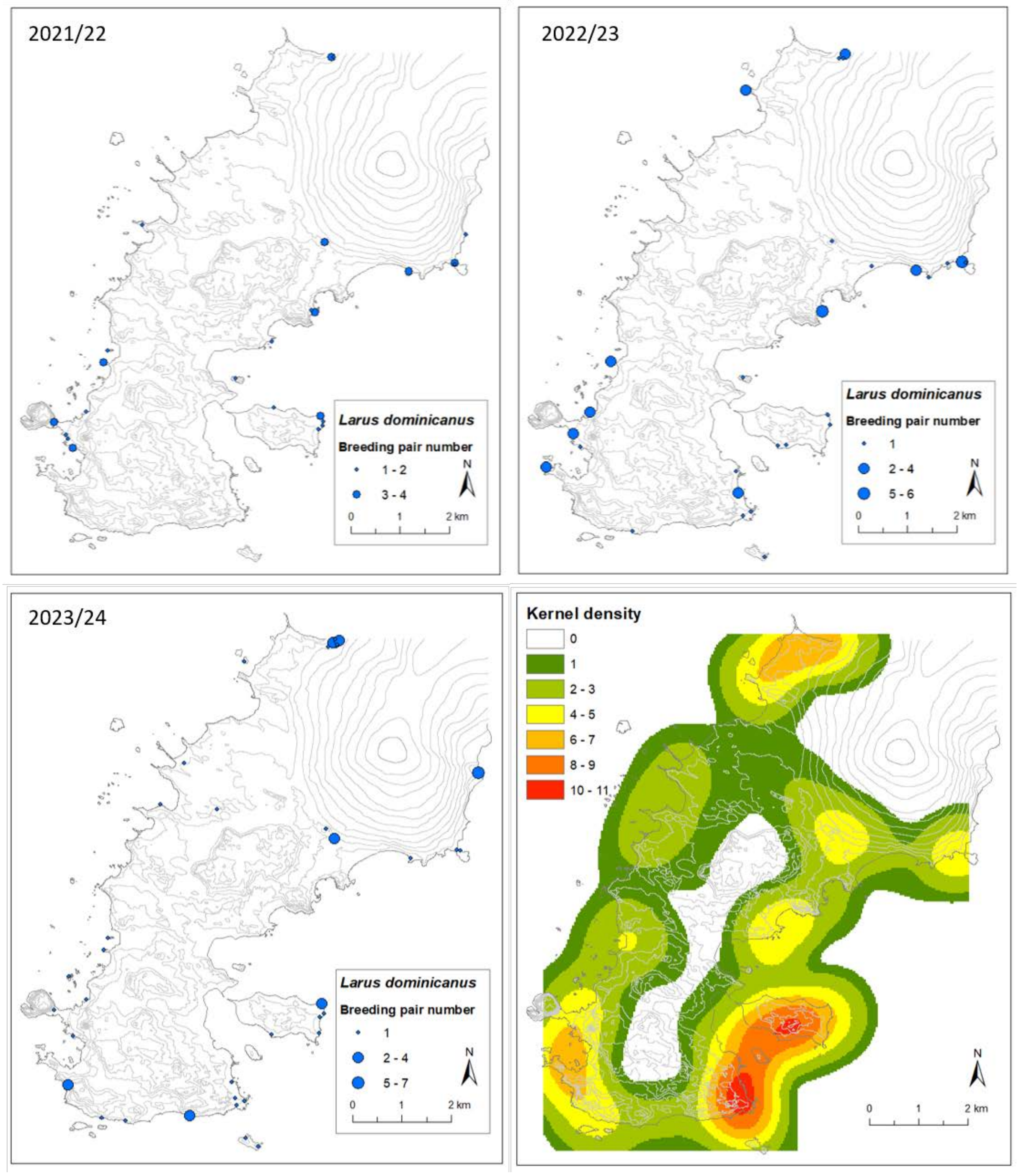
Figure 21: **Development of the number of breeding pairs of the kelp gull (*Larus dominicanus*) in the Fildes Region since the 1980s (* incomplete data)**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Since the 2003/04 season, 163 breeding sites of the kelp gull are now known. Of these, between 28 and 34 breeding sites were occupied during the study period (Figure 22). A new breeding site was recorded in the 2023/24 season, which is one of the few breeding sites located several hundred metres from the coast. Summarising all 162 known breeding sites since the 2000/01 season, the overall distribution of the kelp gull in the Fildes Region becomes clear. The resulting kernel density map shows the predominant concentration of gull breeding sites along the coast, but also numerous breeding sites on the edge of the Bellingshausen dome glacier (Figure 22). The maximum distance of a gull breeding site to the coast was more than 1.6 km.

Figure 22: Location and size of breeding colonies of the kelp gull (*Larus dominicanus*) in the 2021/22 to 2023/24 seasons and kernel density map of all breeding sites recorded since the 2000/01 season

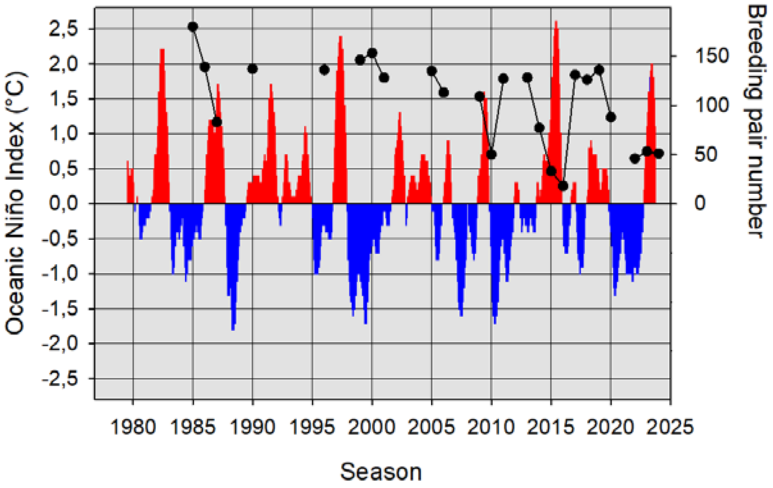


Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

A significant negative correlation between the number of breeding pairs of kelp gulls in the Fildes Region and local snow cover in the spring months of October and November, which had been indicated before (Braun et al., 2022), has now been proven (Spearman corr.: r = -0.437, p = 0.042, n = 22). This shows that a snowy spring, in which the breeding sites are still snow-covered at the time relevant for the decision to breed, leads to a low number of breeding pairs in the respective breeding season.

In contrast to earlier results (Braun et al., 2020c; Braun et al., 2022), the current data did not show any correlation between the number of breeding pairs of kelp gulls in the Fildes Region and the cyclical atmospheric-oceanic phenomenon 'El Niño-Southern Oscillation' (ENSO). The corresponding mean values of the 'El Niño Southern Oscillation Index' (SOI) and the 'Oceanic Niño Index' (ONI) for the respective year of breeding onset were used, which track the atmospheric and oceanic parts of the pattern, respectively. Although significantly reduced numbers of breeding pairs were recorded in seasons with moderate to very strong ONI (seasons 1986/87, 2009/10, 2014/15, 2015/16, 2023/24), whereas in La Niña seasons relatively many gulls bred in the area (seasons 1984/85, 1998/99, 1999/2000, 2010/11, 2012/13, Figure 23, Braun et al., 2020c; Braun et al., 2022), there was no statistical correlation with different parameters of this climatic phenomenon (Spearman corr. SOI: r = 0.108, p = 0.609, n = 24, Spearman corr. ONI: r = 0.104, p = 0.623, n = 24). This finding of a lack of statistical association is consistent with a study from nearby Admiralty Bay (Petry et al., 2016), albeit the latter based on few census data.

Figure 23: Comparison of the Oceanic Niño Index (ONI) and the number of breeding pairs of kelp gull in the Fildes Region; an El Niño event is defined as an ONI anomaly ≥ +0.5 °C over 5 consecutive 3-month periods (red - El Niño, blue - La Niña)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The exact mechanism by which the El Niño phenomenon affected the number of breeding pairs of kelp gulls in the Fildes Region is still unknown. It is likely that certain environmental factors, such as winter mortality of adult birds, snow cover at the start of the breeding season and/or food availability, influence the decision to breed (Braun et al., 2020c; Braun et al., 2022). A relationship between the number of breeding pairs of kelp gulls and the temperature during the early breeding season, as suggested by Petry et al. (2016), could not be demonstrated using the temperature mean for the months of November, October and December (Spearman corr. ONI: r = 0.222, p = 0.294, n = 24; Braun et al., 2022). The availability of prey in the breeding area seems to play a minor role, however, as kelp gulls in the area of the South Shetland Islands feed on *Nacella concinna* during the breeding season (Favero et al., 1997; Favero & Silva, 1998) and these are abundant in the tidal zone (Braun et al., 2022). Furthermore, it has been shown that kelp gulls in the Antarctic colonise new habitats created by glacier retreat (see also chapter 9.3; Lee et al., 2017b).

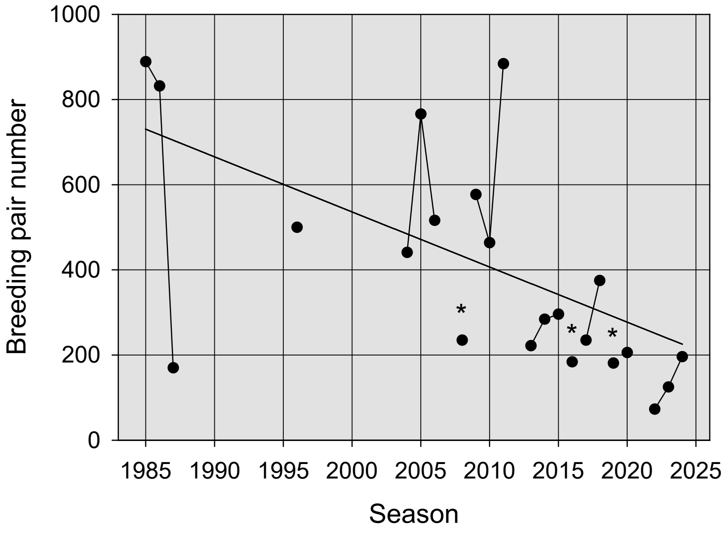
For the Antarctic and Subantarctic, there is unfortunately little information on longer-term population trends of this species (e.g. Jablonski, 1986; Sander et al., 2006; Branco et al., 2009; Whittington et al., 2009; González-Zevallos et al., 2013; Petry et al., 2016; Petry et al., 2018). As a generalist species, the kelp gull, like other gull species, can benefit from additional food provided by fishery discards or urban sources, which can lead to an increase in the population (Coulson & Coulson, 1998; Crawford et al., 2009; Whittington et al., 2016; Ceia et al., 2023). However, this exposes kelp gulls to risks such as ingesting human waste or anthropogenic materials such as plastic, which can have negative consequences for their survival (Lenzi et al., 2016; Witteveen et al., 2017; Seif et al., 2018; Yorio et al., 2020; Seco Pon & Pereyra, 2021; Reusch et al., 2022; Yorio et al., 2022, see also chapter 12.3).

5.9 Antarctic tern (*Sterna vittata*)

The local population of Antarctic terns (*Sterna vittata*) can fluctuate greatly due to meteorological conditions. Furthermore, breeding terns tend to relocate or abandon their breeding sites during the breeding season due to their high susceptibility to disturbance (Weidinger & Pavel, 2013). Nevertheless, despite the resulting methodological inadequacies, long-term monitoring can provide valuable information on the general development of the population of this species in an area.

In the 2021/22 and 2022/23 seasons, only 73 and 125 breeding pairs of Antarctic terns were recorded respectively, which are the two lowest values since the start of long-term monitoring in the season 1984/85 (Figure 24). In the 2023/24 season, the population of this species recovered slightly to 196 breeding pairs. This means that the Antarctic tern population in the three breeding seasons analysed was well below the long-term average of 423 ± 262 BP. Despite considerable population fluctuations, there is evidence of a significant downward trend in the population of Antarctic terns in the Fildes Region in a long-term comparison since the start of the survey in the 1984/85 season ($R^2 = 0.370$, $p = 0.006$, $n = 19$, Figure 24). If the number of breeding pairs between the 2000/01 and 2023/24 seasons is considered, this negative trend becomes even more pronounced ($R^2 = 0.562$, $p = 0.001$, $n = 15$).

Figure 24: **Development of the number of breeding pairs of the Antarctic tern (*Sterna vittata*) in the Fildes Region since 1985 (* incomplete data).**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

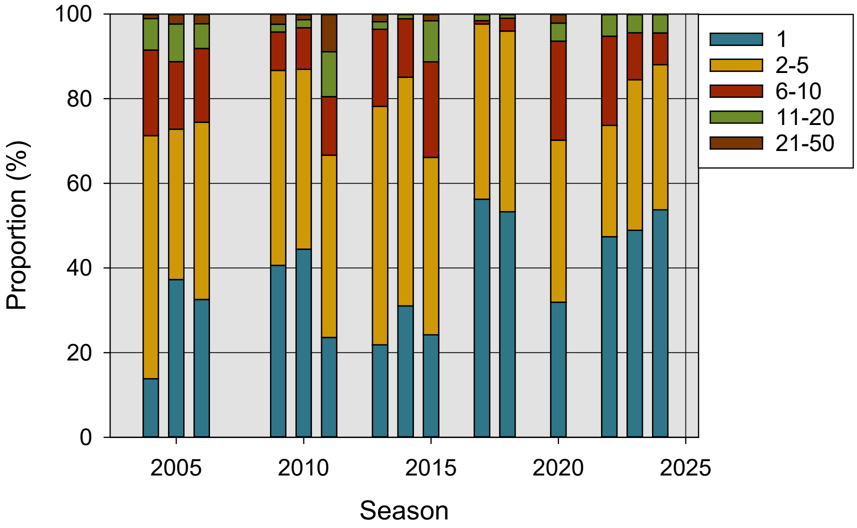
The current breeding pair density of the Fildes Region was significantly below or in the range of various published values, e.g. for neighbouring Admiralty Bay, KGI, or James Ross Island (Table 3). Due to the once again low breeding pair numbers, the long-term average breeding pair density in the Fildes Region fell further compared to previous years and currently stands at 11.9 ± 7.5 bp/km^2 (Braun et al., 2022).

Table 3: **Comparison of Antarctic tern density at different locations; note the different measurement of density (individuals/km^{2}vs. nest/km^2, * probably overestimated due to line transect method)**

Locality	Season	Density	Source
Fildes Region, KGI	1984/85	25.64 bp/km^2	Peter et al., 1988
Fildes Region, KGI	2021/22 –2023/24	2.11 – 5.71 bp/km^2	this study
Admiralty Bay, KGI	1978/79	84.4 ind./km^2	Jablonski, 1986
Admiralty Bay, KGI	?	34 – 96 bp/km^2	Jablonski, 1995 cited in Weidinger & Pavel, 2013
Admiralty Bay, KGI	2004/05	2.21 bp/km^2	Sander et al., 2005
Byers Peninsula, Livingston Island, KGI	2008/09	61.6 – 64.2 ind./km^2	Gil-Delgado et al., 2013 *
James Ross Island	2008/09 & 2010/11	2.2 – 2.9 bp/km^2	Weidinger & Pavel, 2013
Seymour Island	2010/11	12.6 bp/km^2	Weidinger & Pavel, 2013

In all three studied seasons, more than one breeding pair was mostly recorded at the majority of breeding sites in the Fildes Region (Figure 25). The mean number of pairs per breeding site was 3.87 ± 0.58 during the study period. The proportion of different size classes of breeding sites varied considerably between years (Figure 25). As in previous years, the proportion of single breeders was relatively high at around 49 % to 52 % (Braun et al., 2017a; Braun et al., 2022). However, large colonies with more than 20 and up to 300 breeding pairs were not recorded any more. These have been regularly observed in the past both in the Fildes Region (Peter et al., 1988; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2022) and in other Antarctic regions (Gil-Delgado et al., 2013; Weidinger & Pavel, 2013).

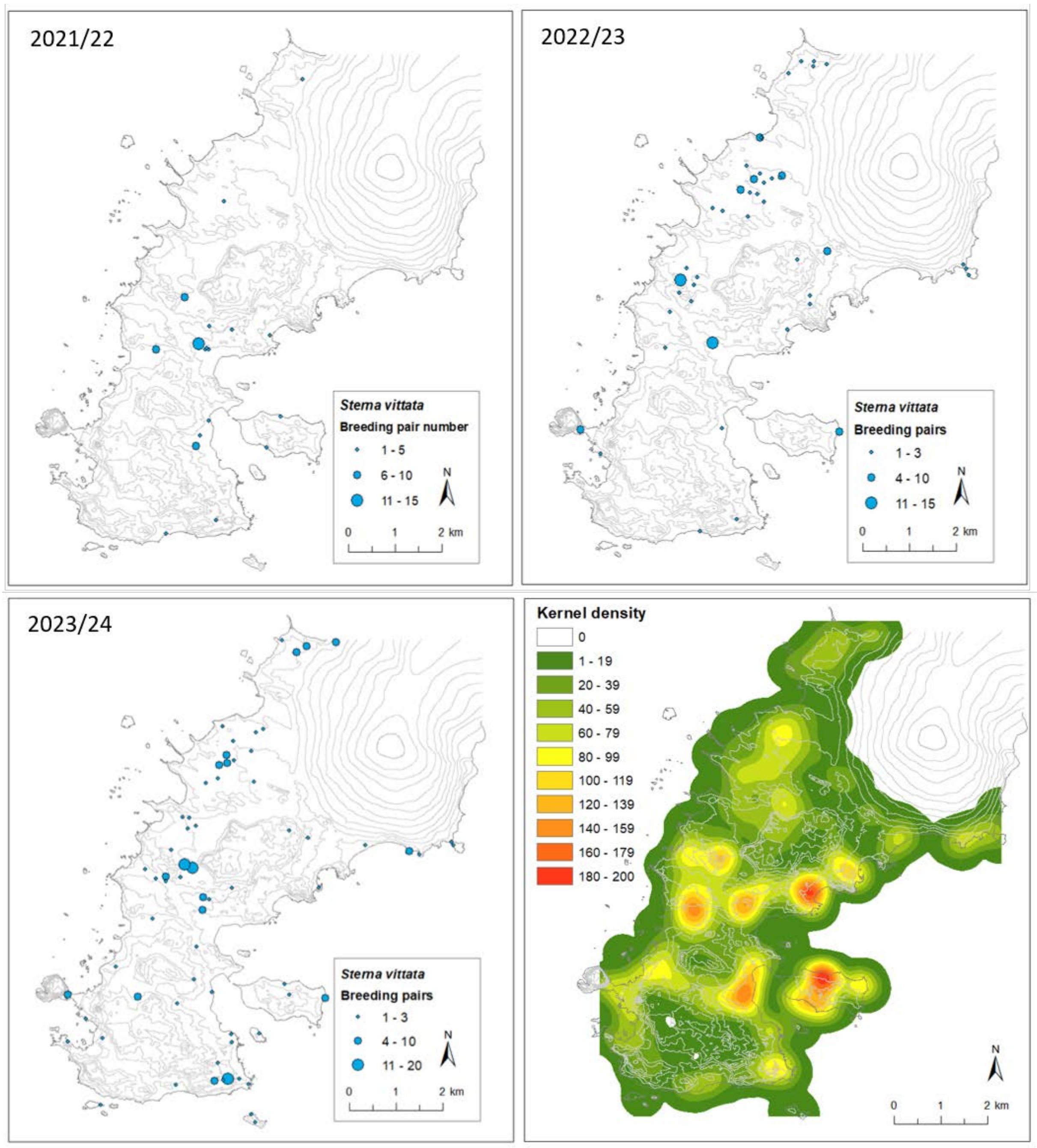
Figure 25: Proportions of different size classes of Antarctic tern (*Sterna vittata*) breeding sites in the Fildes Region between 2003/04 and 2023/24 (only complete surveys included)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The spatial distribution of all mapped breeding sites of the Antarctic tern in the Fildes Region differed between the studied seasons. It becomes apparent that large parts of the study area were not colonised by terns in the 2021/22 and 2022/23 seasons, while a more extensive distribution was recorded in the 2023/24 season (Figure 26). The data of all 1,678 breeding sites recorded since the 2000/01 season show the overall distribution of Antarctic terns in the study area based on the determined kernel density. This clearly shows that most breeding sites are found both inland along river valleys and on the coast or on plateau (Figure 26). Antarctic terns avoid higher areas, but also flatter, permanently wet areas where the gravelly substrate, which is a prerequisite for nesting, is lacking. The mean distance of all known breeding sites to the coastline was 399 ± 226 m, while the maximum distance from a breeding site to the coast was 1,849 m. Breeding Antarctic terns are considered inshore foragers avoiding far inland areas (Casaux et al., 2008). In accordance with this, more than 90% of the breeding sites were located at a distance of < 1 km from the coast. Only a minority of breeding sites were found further inland (Figure 26). In contrast, a higher proportion of inland breeders (29%) was documented on James Ross Island (Weidinger & Pavel, 2013).

Figure 26: **Location and size of Antarctic tern (*Sterna vittata*) breeding colonies in the Fildes Region in the 2021/22 to 2023/24 seasons and kernel density map of all breeding sites recorded since the 2000/01 season**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The cause of the sharp decline of the breeding pair number of the Antarctic tern in the Fildes Region area is not known. The extent to which the population decline can also be observed in other breeding areas, including Subantarctic islands in the Indian and Pacific Ocean, is not known due to the limited comparative population data published. Statements on stable or growing populations are contrasted with information on shrinking populations (Crawford & Cooper, 2003; Crawford et al., 2003; Tree & Klages, 2004; Sander et al., 2005; Whittington et al., 2009; Sierakowski et al., 2017). However, since most of these studies are older, no clear statements can be made about the superregional population development of this species. The global population was last estimated at 44,500 breeding pairs with an unknown population trend (Tree & Klages, 2004; BirdLife International, 2024f). As a significant proportion (approx. 78.7 %) of the global population breed on the South Shetland Islands (Tree & Klages, 2004), monitoring the population development there is of great importance.

There is a high level of human activity in the Fildes Region and many breeding sites are located close to human infrastructure such as stations, the road network and the airstrip. Therefore, a relatively high level of human disturbance is regularly observed there (z. B. Peter et al., 1988; Peter et al., 2013). However, it is not possible to make clear statements about breeding success in these areas, as regular nest checks were not carried out. For Arctic terns (*Sterna paradisea*), however, it was shown that breeding birds from a colony in a settlement on Svalbard returned to the nest significantly faster after a human disturbance than birds from a colony with little human influence (Syrová et al., 2020). This indicates the ability of Antarctic terns to habituate, as has also been demonstrated for other bird species (e.g. Pfeiffer & Peter, 2004, 2006; Vennesland, 2010). This is a trade-off between remaining at the nest and the frequent flying up and the resulting risk of egg cooling (Syrová et al., 2020). Furthermore, it was shown that both colonies showed no difference in hatching success despite the different response to disturbance (Syrová et al., 2020). A comparative study of Arctic and Antarctic terns showed a remarkable consistency in the behavioural responses of both species to human disturbance (Hromádková et al., 2023). For example, the reactions of breeding terns to human disturbance did not exceed those caused by native predators. From this it was concluded that in both polar regions a coexistence of breeding terns and humans can be sustainable (Hromádková et al., 2023).

Furthermore, the human influence in the Fildes Region on the Antarctic tern is evident in the form of the use of anthropogenic material such as metal, pieces of bone and glass or fruit pits for nest building (Braun et al., 2022, see also chapter 12.4).

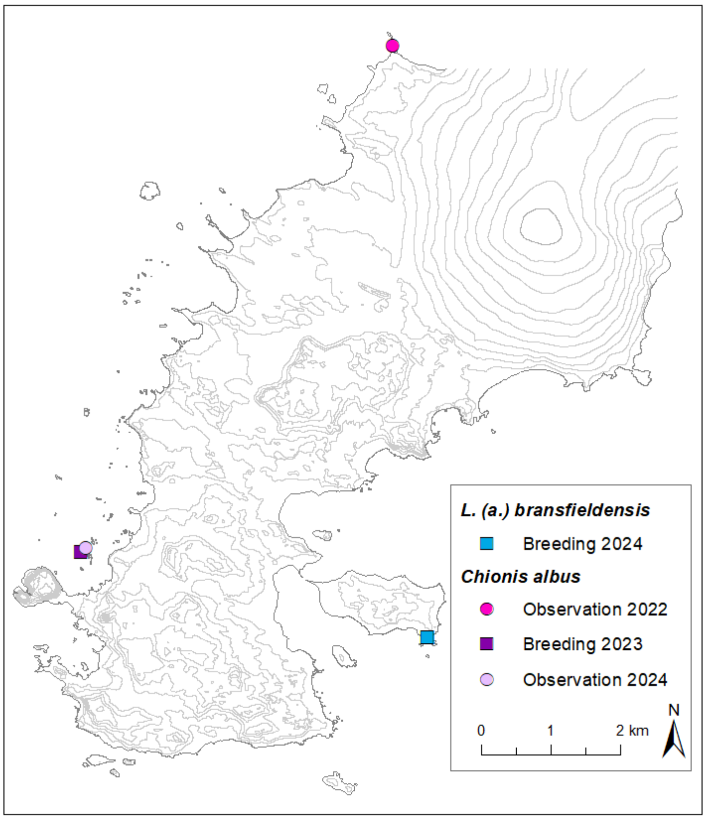
5.10 Snowy sheathbill (*Chionis albus*)

The snowy sheathbill (*Chionis albus*) is the only breeding bird species in the Fildes Region that is not considered a seabird. A small number of individuals of this species were regularly observed as part of the breeding bird monitoring program (Peter et al., 1988; Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). However, direct evidence of breeding is rare due to the inaccessibility of the breeding sites, so various parameters, such as repeated observations of two adult birds at the same site, were used to conclude that breeding was suspected. So far, a few breeding sites have been identified on the west coast of the Fildes Region.

In the 2021/22 and 2023/24 seasons, there were several observations of single individuals as well as a breeding record based on the observation of a moulting chick together with two adult birds in the 2022/23 season (Figure 27). In contrast, another long-standing breeding site in the south-west of the Fildes Peninsula appeared unoccupied.

As this omnivorous species feeds mainly on a kleptoparasitic diet of predatory penguin food (del Hoyo et al., 1996), it is assumed that the large chinstrap penguin colonies in the south-west and north-west of the Fildes Peninsula (Pfeifer et al., 2019) serve as the main food source for the white-faced sheathbills breeding in the study area. Foraging white-faced sheathbills were not observed in the penguin colony on Ardley Island during the study period, despite its considerable size.

Figure 27: Observations of snowy sheathbill and blue-eyed shags in the 2021/22 to 2023/24 seasons



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

5.11 Blue-eyed shag (*Leucocarbo* (*atriceps*) *bransfieldensis*)

The blue-eyed shag cormorant (*Leucocarbo* (*atriceps*) *bransfieldensis*), the southernmost representative of the cormorants, was considered a potential breeding bird in the Fildes Region for many years (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). Despite regular observations in the area, no more broods of this species have been recorded since the 1980s (Braun et al., 2022). The last unsuccessful breeding attempts observed, took place in the north-east of the island (Braun et al., 2020c; Braun et al., 2022).

During the study period, blue-eyed shags were observed several times on the land bridge between the Fildes Peninsula and Ardley Island in a flock of up to 30 animals. Breeding attempts in the study area, as most recently in the 2015/16 and 2019/20 seasons (Braun et al., 2020c; Braun et al., 2022), were not recorded in the 2021/22 and 2022/23 seasons (Figure 27). In contrast, in the 2023/24 season, successful breeding of this species was recorded for the first time in a long time. A total of four breeding pairs bred on a rock in the south-east of Ardley Island in the midst of numerous breeding gentoo penguins. In February 2024, a total of 4 chicks were counted in 3 nests (Figure 28).

Figure 28: Blue-eyed shag (*Leucocarbo* (*atriceps*) *bransfieldensis*) breeding on Ardley Island in the 2023/24 season

Photo: J. Engelhardt, Institute of Ecology and Evolution, Friedrich Schiller University Jena

It is possible that the new broods of the blue-eyed cormorant in the study area are the result of a population increase in the region, as has been proven for several breeding colonies on the rocky north-west coast of Nelson Island and the south-west coast of King George Island, for example (Pfeifer et al., 2021). Overall, the global population trend of this species is described as very uncertain, as some populations are growing while others are shrinking (Lynch et al., 2008; Chesalin et al., 2009; Casanovas et al., 2015; Casaux & Barrera-Oro, 2016; Schrimpf et al., 2018; Phillips et al., 2019; BirdLife International, 2024d; Casaux et al., 2024). A similar situation can be observed for the closely related species (*Leucocarbo* (*atriceps*) *georgianus*), which also breeds in the Antarctic, and which recorded opposing trends in two breeding colonies on Signy Island, South Orkney Islands, as well as a significant decline on Bird Island, South Georgia (Dunn et al., 2021). Reduced food availability or climate change are assumed to be the cause of shag population declines and/or climatic change (Ryan et al., 2003; Crawford et al., 2014; Casanovas et al., 2015; Casaux & Barrera-Oro, 2016; Schrimpf et al., 2018; Dunn et al., 2021). For example, a north-south shift in the range of the blue-eyed cormorant already reflects a gradient of environmental change along the WAP (Casanovas et al., 2015; Schrimpf et al., 2018). However, after the genus *Phalacrocorax* was subdivided taxonomically on the basis of genetic analyses (Kennedy & Spencer, 2014), it is currently hardly possible to estimate the population size (BirdLife International, 2024d). Therefore, following an extensive census of local populations of this species using remotely piloted aircraft systems (Pfeifer et al., 2021), a machine learning methodology was developed to conduct large-scale surveys of unknown shag colonies on Antarctic islands (Cusick et al., 2024).

5.12 Vagrant and migrant bird species

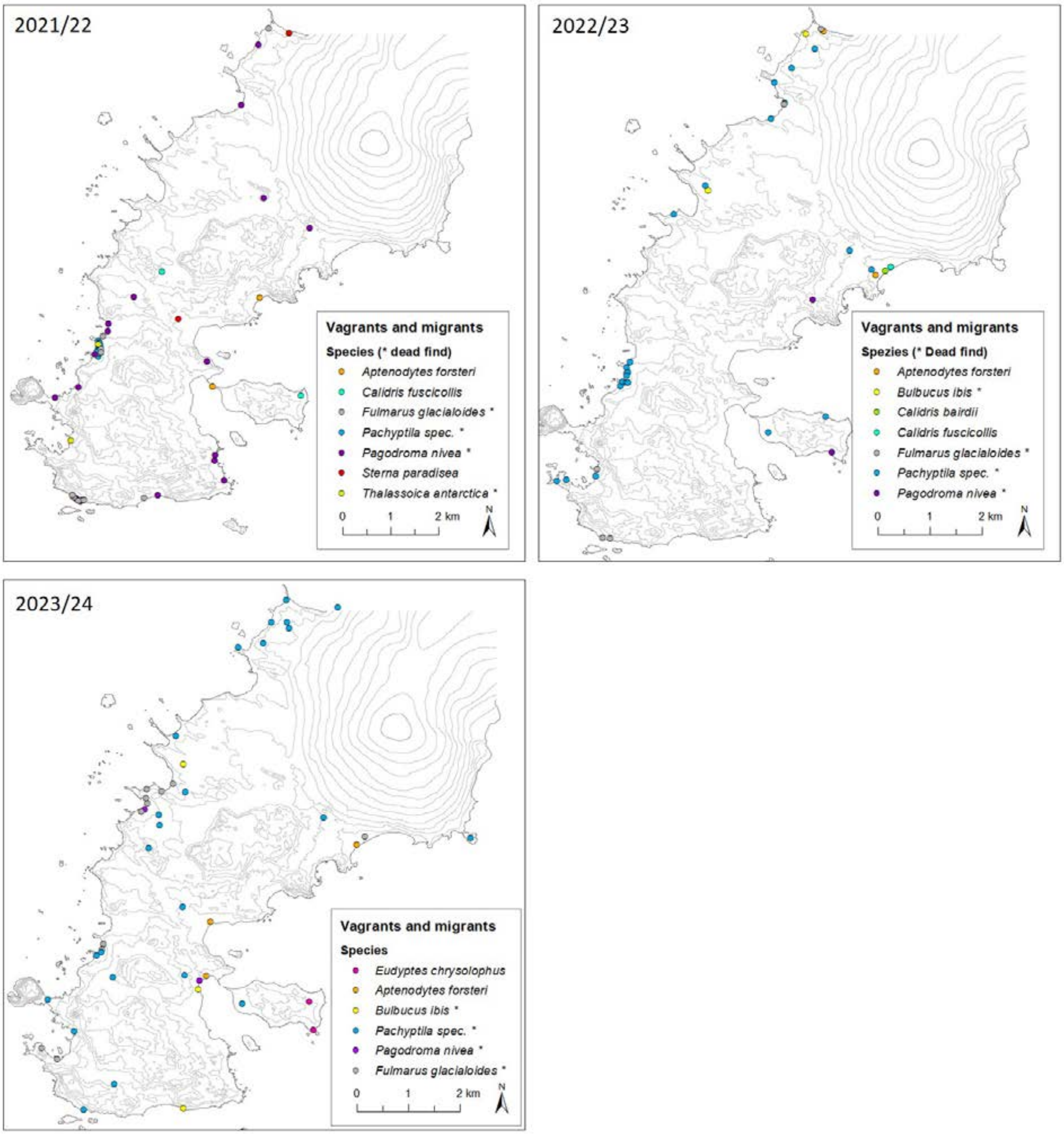
Vagrancy is usually observed in migratory bird species due to their high mobility, but also applies to all other species that stay outside their home or distribution area (Newton, 2008). It can be caused by normal long-distance dispersal, drift due to strong winds, overshooting during migration movements, reversed-direction migration, and the consequences of environmental change (Milius, 2000; Raya Rey et al., 2007; Newton, 2008). Vagrancy is often observed in young birds as they disperse more frequently outside their usual population routes (Newton, 2008; Golubev, 2020b). It can also serve to explore new breeding habitats (Veit et al., 2021; Veit et al., 2022). Ship-based transport may play a role as well (Hahn et al., 1998; Korczak-Abshire et al., 2011; Petersen et al., 2015). Hence, long-term data on rare bird sightings in the Antarctic can indicate a potential expansion of the breeding range of Subantarctic species (Juáres et al., 2014; Juáres et al., 2017; Gryz et al., 2019; Braun et al., 2023; Hallet & Lynch, 2024). Furthermore, this information may provide valuable clues to unusual environmental conditions that allow vagrants to reach regions beyond their usual distribution range.

During the current study period, numerous representatives of non-native birds were recorded, in some cases exclusively as dead finds (Table 4 & Figure 29). The most frequently recorded dead finds were of seabird species that are frequently observed in the Southern Ocean but are not found as breeding birds in the Fildes Region, in particular prions (*Pachyptila* spec.), snow petrels (*Pagodroma nivea*) and southern fulmars (*Fulmarus glacialoides*). Emperor penguins (*Aptenodytes forsteri*) were observed in all three seasons of the study period and macaroni penguins (*Eudyptes chrysolophus*) only in the 2023/24 season (Table 4). The remains of several cattle egrets (*Bubulcus ibis*) were found as vagrant species in the 2022/23 and 2023/24 seasons. In addition, white-rumped sandpiper (*Calidris fuscicollis*), the far more common species of the Scolopacidae family, was recorded in the 2021/22 and 2022/23 seasons. Baird's sandpiper (*Calidriis bairdii*), another member of this family, for which no evidence had previously been available from the study area, was observed for the first time in the 2021/22 and 2022/23 seasons. Furthermore, several Arctic terns flying in flocks were observed as migrant species in the 2021/22 season.

Table 4: Overview of observations of vagrant and migrant bird species in the 2021/22 to 2023/24 seasons (* dead finds)

Species	Number of observations / individuals		
	2021/22	**2022/23**	**2023/24**
Emperor penguin (*Aptenodytes forsteri*)	1/1	2/2	3/3
Macaroni penguin (*Eudyptes chrysolophus*)	0	0	2/2
Southern fulmar (*Fulmarus glacialoides*)	11 / 13 *	7 / 7 *	11 / 11 *
Antarctic petrel (*Thalassoica antarctica*)	1 / 1*	0	0
Snow petrel (*Pagodroma nivea*)	17 / 17 *	2 / 2 *	2 / 2 *
indet. prion (*Pachyptila* spec.)	8 7 8 *	11 / 22 *	26 / 39 *
Arctic tern (*Sterna paradisea*)	2 / ≈45	0	0
Cattle egret (*Bubulcus ibis*)	0	2 / 2	3 /3
Baird’s sandpiper (*Calidris bairdii*)	0	2 / 7	0
White-rumped sandpiper (*Calidris fuscicollis*)	2 / 2	2 / 3	0

Figure 29: **Observations and dead bodies of vagrants and migrants in the Fildes Region in the 2021/22 to 2023/24 seasons**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

A detailed overview of all vagrant and migrant bird species recorded in the Fildes Region between the 1979/80 and 2022/23 seasons was recently published in Braun et al. (2023). An update that includes the 2023/24 season is provided in Table 5.

Table 5: Records of vagrants (va) and visitors (vi) in the Fildes Region between 1979 and 2024 (classification following SCAR Expert Group on Birds, 2005, Sierakowski et al., 2017 & Golubev, 2020b), abundance: regularly (>15 seasons), occasionally (6–15 seasons), rarely (1–5 seasons), * minimum numbers due to occurrence in swarms or difficulties of identification of dead bird remains (modified after Braun et al., 2023)

		Status					
Family	**Species**	**regularly**	**occasionally**	**rarely**	**Seasons with sightings**	**Sightings**	**Individuals**
Spheniscidae	Emperor penguin (*Aptenodytes forsteri*)	va			30	54	57
	King penguin (*Aptenodytes patagonicus*)	va			12	22	22
	Macaroni penguin (*Eudyptes chrysolophus*)	vi			20	32	33
	Rockhopper penguin (*Eudyptes chrysocome*)			va	3	3	3
	Magellanic penguin (*Spheniscus magellanicus*)			va	1	1	1
Diomedeidae	Wandering albatross (*Diomedea exulans*)		vi		1	1	1
	Black-browed albatross (*Thalassarche melanophris*)		vi		5	7	8
Procellariidae	Northern giant petrel (*Macronectes halli*)			vi	2	2	3
	Sooty shearwater (*Ardenna grisea*)			vi	1	1	1
	Southern fulmar (*Fulmarus glacialoides*)	vi			28	149*	371*
	Soft-plumaged petrel (*Pterodroma mollis*)			va	2*	2*	2*

		Status					
	Blue petrel (*Halobaena caerulea*)	vi			1*	8*	9*
	Snow petrel (*Pagodroma nivea*)	vi			22	222	>4000*
	Antarctic prion (*Pachyptila desolata*)	vi			5*	5*	9*
	Broad-billed prion (*Pachyptila vittata*)			va	1	1	1
	Atlantic petrel (*Pterodroma incerta*)			va	1	2	3
	Antarctic petrel (*Thalassoica antarctica*)		vi		11	32	418*
Pelecanoididae	Diving petrel (*Pelecanoides* spec.)			va	1	1	1
Laridae	Pomarine jaeger (*Stercorarius pomarinus*)			va	1	1	1
	South American tern (*Sterna hirundinacea*)			va	1	1	1
Ardeidae	Cattle egret (*Bubulcus ibis*)	va			16	48	60
Scolopacidae	White-rumped sandpiper (*Calidris fuscicollis*)	va			15	23	85
	Pectoral sandpiper (*Calidris melanotos*)			va	1	1	8
	Baird’s sandpiper (*Calidris bairdii*)			va	2	2	7
	Black-necked swan (*Cygnus melancoryphus*)			va	1	1	11
Anatidae	Yellow-billed pintail (*Anas georgica*)			va	1	1	10
	Chiloé wigeon (*Mareca sibilatrix*)			va	2	2	3

The following section provides a few additional details on selected species to supplement the information given in Braun et al. (2020c) the current state of knowledge.

One of the most common species considered to be a regular vagrant in the Fildes Region is the emperor penguin. In recent years, several new colonies have been discovered in the Antarctic with the help of satellite images, increasing the number of currently known colonies to 66 (Fretwell & Trathan, 2021; Fretwell, 2024a). As these are mostly small and located in areas that are considered highly endangered in view of the current climatic changes (Fretwell & Trathan, 2021), they do not significantly affect the overall situation for this species. In particular, a low sea ice extent in connection with an early break-up of the fast ice poses a major risk for the emperor penguin colonies (Barbraud & Weimerskirch, 2001; Fretwell et al., 2023; Fretwell, 2024b). If CO_2 emissions continue to rise at the same rate, current models predict that almost all emperor penguin colonies will be completely extinct by the end of this century (Jenouvrier et al., 2021).

The Subantarctic king penguin (*Aptenodytes patagonicus*) is another penguin species that is affected by the current environmental changes. This was shown, for example, in a study on king penguins from the Crozet Islands in the southern Indian Ocean. According to the study, breeding animals will be forced to travel twice the distance in search of food by the end of the century as a result of the expected warming of the surface water and the resulting southward shift of suitable feeding areas (Péron et al., 2012). Furthermore, there are numerous reports of the king penguin attempting to colonise the area of the South Shetland Islands (e.g. Petry et al., 2013; Gryz et al., 2019; Borowicz et al., 2020; Juáres et al., 2023). This change of distribution pattern and breeding range is often interpreted as a reaction to effects of climate change (Libertelli et al., 2024). The currently most southerly observation of a king penguin comes from the 2019/20 season from Marguerite Bay, 68°S (Libertelli et al., 2024). However, there is no evidence of frequent observations or breeding attempts in the Fildes Region.

The macaroni penguin is another Subantarctic species that reaches higher latitudes in the south. The southernmost observations of this species in the Antarctic come from Haswell Archipelago, East Antarctica (66° S, Golubev, 2016) and Avian Island, Marguerite Bay, Antarctic Peninsula (67° S, Gorman et al., 2010). Older reports even state an observation of a macaroni penguin from Cape Hallett in the western Ross Sea (72° S) in 1964 (Marchant & Higgins, 1990). Breeding macaroni penguins have been documented in the WAP region for some time now (Crossin et al., 2013). Cormorant Island, Palmer Archipelago (64° S) is currently the southernmost known breeding site (Gorman et al., 2010). The total breeding population of this species in the Antarctic is currently estimated at 8,053 breeding pairs (Hallet & Lynch, 2024). With over 8,000 breeding pairs, Elephant Island and neighbouring smaller islands account for the largest proportion, with several of these breeding colonies showing an increase (Hallet & Lynch, 2024). For the remaining South Shetland Islands, 7 breeding pairs are currently reported at 3 of the 12 historically known colonies, with several breeding sites showing a decline compared to previous records (Hallet & Lynch, 2024). In the course of further warming of the sea surface temperature, a poleward shift redistribution of this species is widely expected (Gore et al., 2020; Hallet & Lynch, 2024) and is also indicated by habitat preference models (Green et al., 2022).

In the 2021/22 season, an unusually high number of dead snow petrels were recorded by Brazilian scientists, including 49 dead animals at Harmony Point, Nelson Island, and 7 dead animals on the Fildes Peninsula (Finger et al., 2022). This is consistent with our own observations from the Fildes Region, where a total of 20 dead animals were found (Table 5). There are no other known reports of increased mortality of this species in the WAP region during this period. Although several dead snow petrels were recorded each year in the Fildes Region, their numbers were always significantly lower. The extent to which this increased mortality is related to a lower sea ice extent in winter, as has already been shown for Terre Adélie (Barbraud et al., 2000), is not known, nor is the question of whether a pathogen was responsible (Finger et al., 2022). A connection with Highly Pathogenic Avian Influenza (HPAI) H5N1 does not seem likely, as it was first detected in Antarctica in the 2023/24 season (Bennett Lazo et al., 2024).

In conclusion, the presented data set is the most comprehensive known of its kind from the Antarctic (see also Braun et al., 2023). It may provide valuable clues to unusual environmental conditions allowing vagrants to reach regions beyond their usual distribution range. Furthermore, long-term data on rare bird sightings can indicate a potential expansion of the breeding range of individual species or changing migration routes. Given the ongoing environmental changes, an increase in the number of bird species visiting the area and the frequency of their occurrence is expected. Thus, reports on rare bird sightings may not only contribute to the knowledge of rare bird sightings in the Antarctic Peninsula region, but also stimulate further over-regional studies looking at changes in the range of vagrant and visitor bird species and the frequency of rare bird observations that indicate unusual environmental conditions in Antarctica that may be related to current climatic changes (Braun et al., 2023).

5.13 Seal monitoring on Fildes Peninsula and Ardley Island

Continuing the long-term monitoring, monthly seal surveys were conducted at the coastline of the Fildes Peninsula and Ardley Island in the 2021/22 to 2023/2024. To obtain spatial information on seal distribution, the subdivision of the coastline from the 1980s was used (see Figure 78 in Appendix B). Based on these counts, data on the seal population in the austral summer in the Fildes Region is available for up to 28 years. The current census data are presented in Table 6.

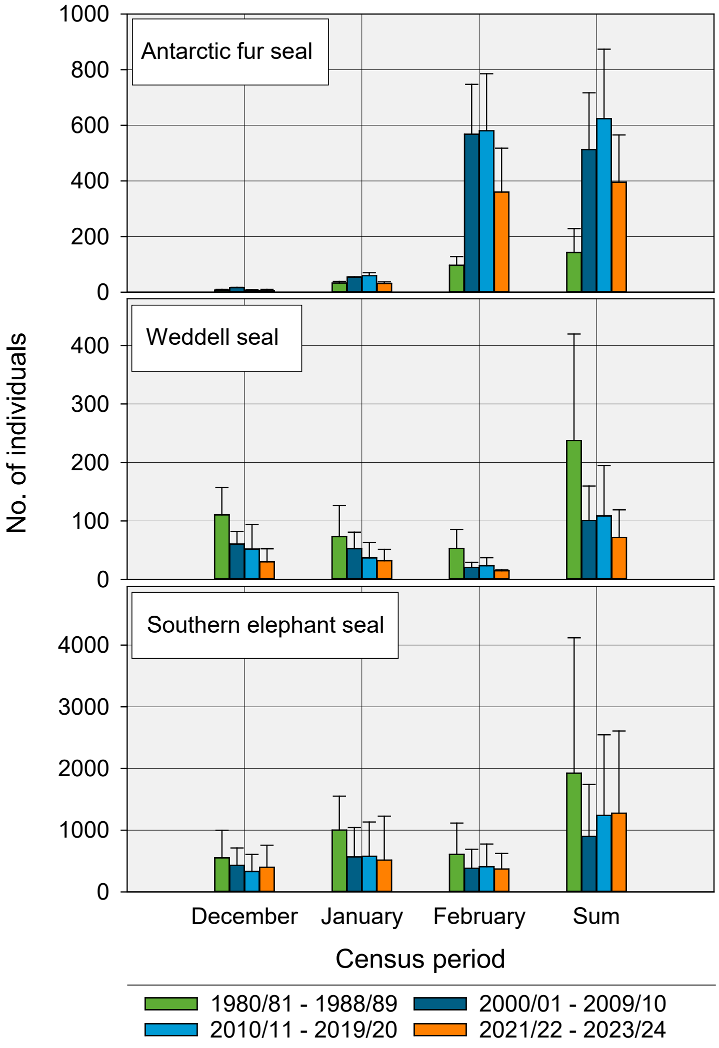
Table 6: Results of the monthly seal census on the Fildes Peninsula and Ardley Island in the 2021/22 to 2023/24 seasons

	Season 2021/22			Season 2022/23			Season 2023/24		
Species	**Dec.**	**Jan.**	**Feb.**	**Dec.**	**Jan.**	**Feb.**	**Dec.**	**Jan.**	**Feb.**
Southern elephant seal (*Mirounga leonina*)	584	1018	587	550	1010	339	818	1028	719
Weddell seal (*Leptonychotes weddelli*)	49	25	0	30	54	10	58	58	31
Antarctic fur seal (*Arctocephalus gazella*)	11	67	737	8	23	244	7	16	433
Crabeater seal (*Lobodon carcinophagus*)	1	0	0	0	1	0	0	0	0
Leopard seal (*Hydrurga leptonyx*)	4	1	0	0	2	1	2	0	0

Since crabeater seals (*Lobodon carcinophagus*) and leopard seals (*Hydrurga leptonyx*) are only found very sporadically on the coast of the Fildes Peninsula and the significance of the observations recorded is very limited, these species are not considered in detail.

The monthly changes in the local seal population correspond to those described in previous studies, i.e. the number of registered southern elephant seals (*Mirounga leonina*) is highest in January, while the number of Weddell seals (*Leptonychotes weddelli*) is highest in December and January, with their numbers decreasing significantly at the end of the austral summer. In contrast, the number of Antarctic fur seals (*Arctocephalus gazella*) does not reach its maximum until February (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022).

Figure 30: **Comparison of decadal average of monthly seal populations on Fildes Peninsula and Ardley Island between 1980/81 and 2022/23; standard deviation is given; note the different scaling of the Y-axes.**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

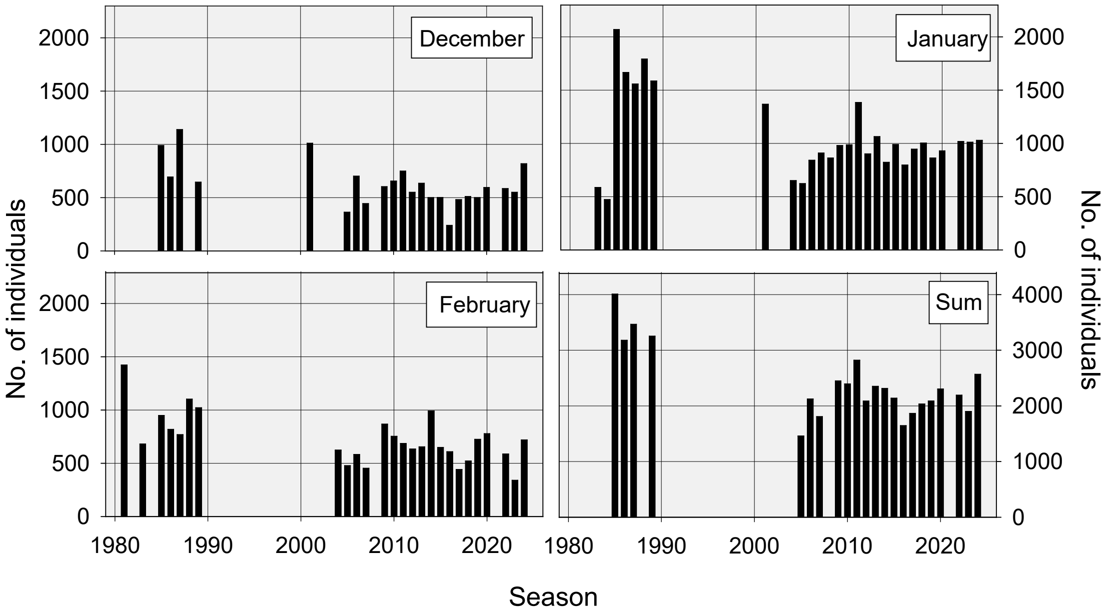
The current data shows, that the trends previously presented in Braun et al. (2017a; 2020c; 2022) can mostly be confirmed. Hence, the number of southern elephant seals and Weddell seals observed during the austral summer were higher in the 1980s, compared to the following decades and also between the 2021/22 and 2023/24 seasons (Figure 30, see chapter 5.13.1 & 5.13.2). The population of the Antarctic fur seal on the other hand demonstrated an opposing trend and experienced a rise during the long-term monitoring (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022), although the average in the seasons 2021/22 to 2023/24 was lower than in the two decades before (Figure 30). Hence, the population growth of the Antarctic fur seal in the study area reflects the global trend that is related to the extensive hunting and subsequent population recovery (see chapter 5.13.3; Yang et al., 2010; Braun et al., 2017a).

Environmental factors associated with current climate warming, such as changes in sea-ice conditions, are likely to negatively impact those seal species that are ice-bound, such as Weddell seals, leopard seals, crabeater seals and Ross seals (*Ommatophoca rossii*) (Siniff et al., 2008; Costa et al., 2010; Van den Hoff et al., 2014; Hückstädt et al., 2020). Conversely, species that avoid the ice, such as southern elephant seals and Antarctic fur seals, could experience certain benefits (Siniff et al., 2008; Costa et al., 2010; Van den Hoff et al., 2014; Hückstädt et al., 2020; Convey & Hughes, 2023). Profound changes in the food web resulting from global warming and industrial fishing may have a significant impact on the population development of seals in the Southern Ocean (Siniff et al., 2008; Costa et al., 2010). A recent study also showed that all fecal samples of crabeater seals, leopard seals and Weddell seals collected from the Antarctic Peninsula contained microplastics, highlighting the presence of anthropogenic contamination in Antarctic seals (Cebuhar et al., 2024).

5.13.1 Southern elephant seal (*Mirounga leonina*)

During the monthly seal counts in the 2021/22 to 2023/24 seasons, a total of between 1,899 and 2,565 southern elephant seals (*Mirounga leonina*) were recorded (Table 6, Figure 31). These total numbers were thus within the long-term average since 2000/01 of 2,115 ± 324, with the total for the 2023/24 season being the second highest in this period. As in previous years, the highest number of individuals changing fur was again recorded in January (Figure 31). Remarkable is the minimum value of only 339 southern elephant seals in February 2023, the lowest value for this month since the start of regular monitoring in the 1982/83 season (Figure 31; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). However, monthly trends are not discernible for the period 2000/01 to 2023/24, which indicates a stabilization of the population after the decline in the 1980s.

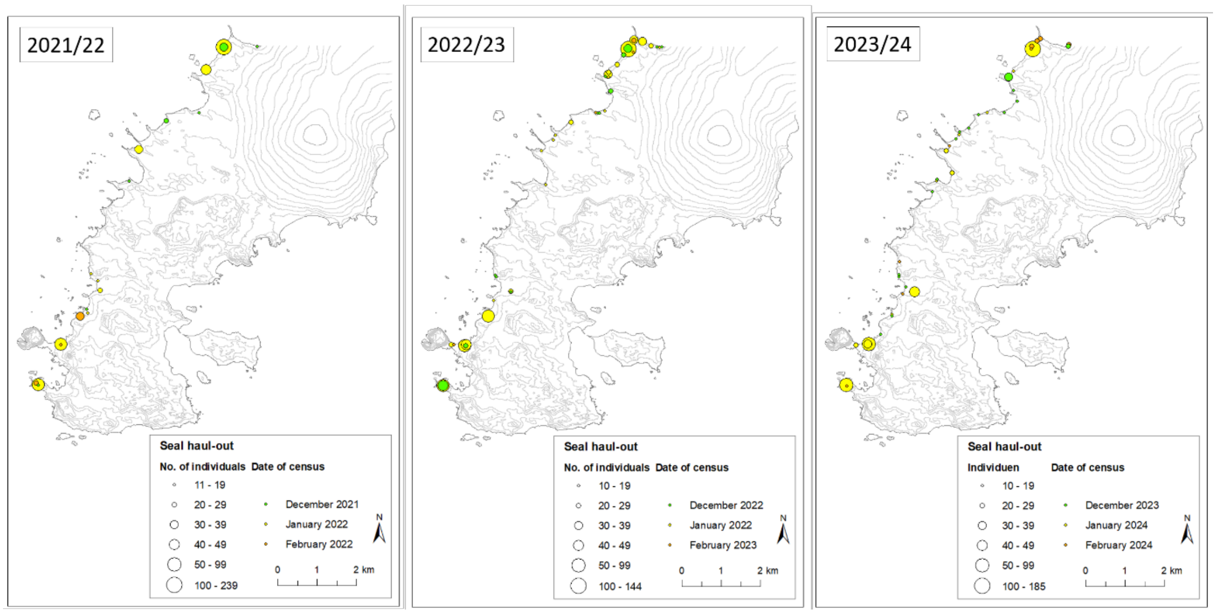
Figure 31: **Number of southern elephant seals (*Mirounga leonina*) recorded during monthly summer seal counts on Fildes Peninsula and Ardley Island between the 1980/81 and 2023/24 season**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

During the current seal counts, all elephant seal haul-out sites with at least ten individuals were also recorded. The largest concentration of southern elephant seals has been documented in January in the northernmost part of the Fildes Peninsula, with 144, 239 and 189 individuals, respectively (Figure 32), the latter being one of the highest number of individuals since the beginning of the seal monitoring in the Fildes Region (maximum: January 2012 with 255 individuals). Analysis of the data, which now spans 17 years, shows considerable seasonal and annual fluctuations. The spatial distribution of haul-out sites showed no significant changes when compared to previously published results (Figure 32; Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). Spatial changes or declines in haul-out sites due to human disturbance, as have been detected in neighbouring Admiralty Bay (Chwedorzewska & Korczak, 2010), are not evident on the Fildes Peninsula. Although this area has a much higher concentration of human activity overall, the elephant seals' haul-outs are located exclusively on the west coast, which, in contrast to the station surroundings, is hardly affected by direct human disturbance, as is regularly observed around stations. One exception is a larger haul-out site in Biologists' Bay south of the airstrip, which is regularly visited by tourists and station members in the austral summer. However, no spatial or quantitative changes were detected there either.

Figure 32: **Southern elephant seal (*Mirounga leonina*) haul-outs with at least ten animals on the Fildes Peninsula and Ardley Island in the 2021/21 to 2023/24 seasons**



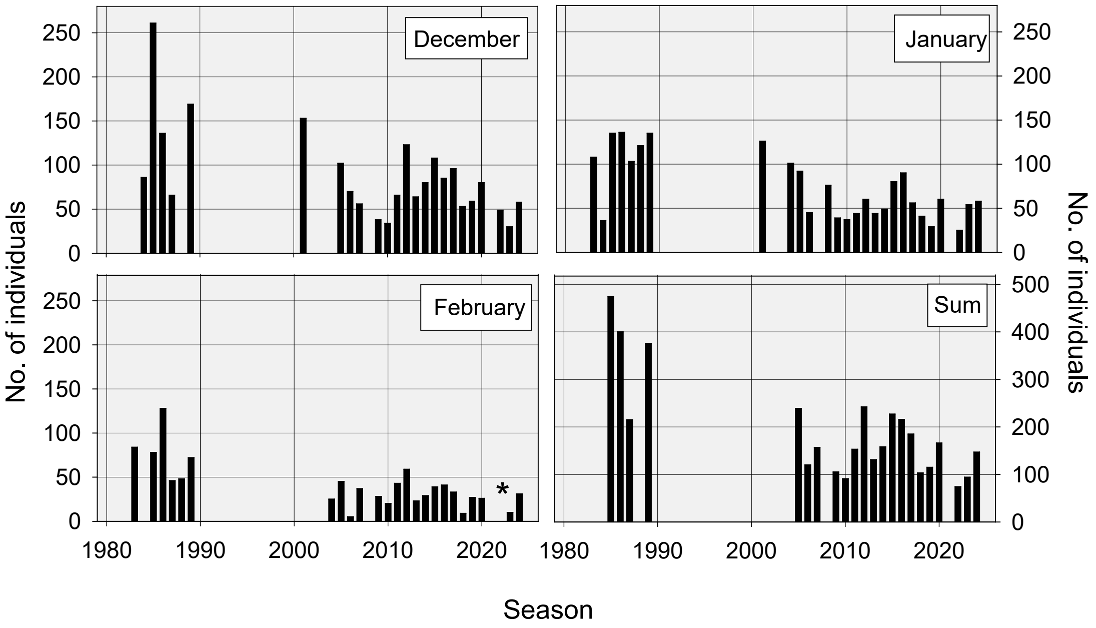
Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The overexploitation of marine resources and current climate change, which may negatively impact the food chain, may potentially threaten the southern elephant seal population (Hanchet et al., 2003; Learmonth et al., 2006; Kovacs et al., 2012; Hofmeyr, 2015). Although the global population of the southern elephant seal is currently classified as Stable or Least Concern by the International Union for Conservation of Nature (IUCN), based on the latest published figures of 325,000 adult seals (Hofmeyr, 2015), there is some preliminary evidence to suggest that such changes are already having a corresponding ecological impact (Mestre et al., 2020). For example, a correlation between the number of reproducing females and the extent of sea ice in the winter feeding grounds has been demonstrated for the reproductive area on the neighbouring Potter Peninsula (Negrete et al., 2022).

5.13.2 Weddell seal (*Leptonychotes weddelli*)

The total number of Weddell seals (*Leptonychotes weddelli*) recorded on the Fildes Peninsula and Ardley Island in the 2021/22 to 2023/24 seasons was 74, 94 and 147 individuals respectively (Table 6, Figure 33) and thus in some cases well below the long-term average of 151 ± 53 individuals since the 2000/01 season. As in previous years, the number of Weddell seals observed in the study area decreased towards the end of the austral summer (Figure 33). In addition, both January and February, as well as December 2022, recorded the lowest numbers of Weddell seals for the respective month since regular surveys began in the 1982/83 season (Figure 33). In February 2022, no seals were recorded at all for the first time. Accordingly, the 2021/22 season saw the lowest total number of this species ever recorded in austral summer in the study area and the 2022/23 season the second lowest. On average, more Weddell seals were observed in the Fildes Region in the 1980s than in the period 2000/01 to 2023/24 (Figure 31 & 33). While the total number of seals and the number in December and February remained stable during this period, a downward trend is evident for the month of January ($R^2 = 0.287$, $p = 0.012$, $n = 21$; Figure 33).

Figure 33: **Number of Weddell seals (*Leptonychotes weddelli*) recorded during monthly seal counts in the austral summer on the Fildes Peninsula and Ardley Island between 1980/81 and 2022/23); * no seals were recorded; if missing, no census census data were available for the respective season.**



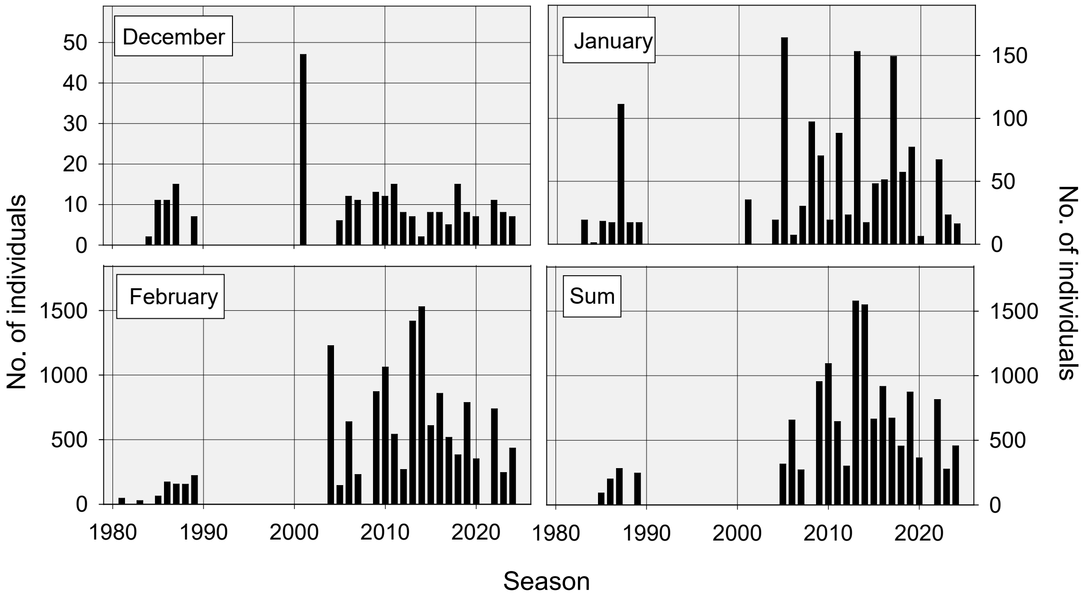
Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Although the Weddell seal is not currently classified as threatened or endangered (IUCN category 'Least Concern' (Hückstädt, 2015), a decline in sea ice due to climate change and the exploitation of marine resources pose significant potential risks to this species (Learmonth et al., 2006; Siniff et al., 2008; Costa et al., 2010; Garrott et al., 2012; Ainley et al., 2015; Salas et al., 2017).

5.13.3 Antarctic fur seal (*Arctocephalus gazella*)

Both the monthly and total numbers of Antarctic fur seals (*Arctocephalus gazella*) counted during the austral summer in the 2021/22 to 2023/24 seasons showed notable fluctuations. A total of 815 and 456 individuals were counted in the 2021/22 and 2023/24 seasons, respectively, with the long-term average since 2000/01 being 713 ± 399 individuals. In contrast, only 275 individuals were counted in the 2022/23 season (Table 6, Figure 34). Nevertheless, the number of fur seals in the 1980s was significantly lower than it is today (Figure 31 & 34). Due to the remarkable fluctuations in the last two decades, no significant trends for the local population of this species in the austral summer between the 2000/01 and 2023/24 seasons can be identified ($R^2 = 0.007$, $p = 0.75$, $n = 18$). However, a strong decline in the total numbers can be seen since the season 2014/15 (Figure 34).

Figure 34: Numbers of Antarctic fur seals (*Arctocephalus gazella*) recorded between 1984/85 and 2023/24 during monthly seal counts in the austral summer (December to February) on Fildes Peninsula and Ardley Island; if missing, no data were available for the respective season.

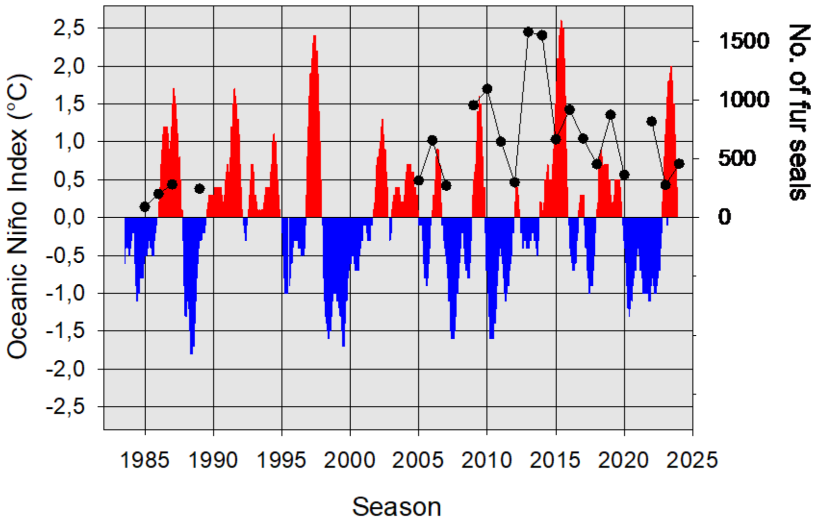


Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

While the low numbers of January and February 2023 and 2024 are consistent with the overall low number of individuals in these seasons, the population for these summer months has remained stable from 2000/01 to 2023/24 (Figure 34). However, a trend in the monthly population numbers can only be observed for December, as a high number was observed in the 2000/01 season ($R^2 = 0.269$, $p = 0.023$, $n = 18$).

The increase in the number of fur seals as the summer progresses is caused by immigration of young males from local breeding populations or from South Georgia and South Orkney Islands (Grebieniow et al., 2020). In contrast to results from the neighbouring Admiralty Bay (Salwicka & Rakusa-Suszczewski, 2002; Grebieniow et al., 2020), a correlation between the number of fur seals recorded in the Fildes Region and the cyclical climate phenomenon El Niño-Southern Oscillation (ENSO) could still not be established, even on the basis of updated data (Braun et al., 2020c; Braun et al., 2022). A moderate or low La Niña phenomenon, i.e. a cooling of the sea surface in a certain area in the east-central Pacific region, was recorded during the study period (see chapter 8.3). However, this is not reflected in a correlation between the annual mean of the Oceanic Niño Index (ONI) and the observed number of fur seals (Spearman corr.: $r = 0.221$, $p = 0.319$, $n = 22$, Figure 35).

Figure 35: Comparison of the Oceanic Niño Index (ONI) and the total number of Antarctic fur seals recorded on the Fildes Peninsula and Ardley Island during austral summer (red - El-Niño, blue - La-Niña).



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

On February 18, 2022, a leucistic Antarctic fur seal was documented in the study area (Figure 36). In total, only two leucistic specimens out of a total of 15,381 individuals were recorded during the monthly seal counts in the seasons 2000/01 to 2023/24 in 82 counts in the summer months. For the 1980s, on the other hand, there were no reports of observations of leucistic individuals in 131 counts carried out twice a month throughout the year, with a total of 3,266 animals recorded. Observations of other atypical fur colorations, as described in Acevedo et al. (2009), are not known for the Fildes Region.

The described observation of leucistic individuals of the Antarctic fur seal is remarkable in that there are only a few published observations of such individuals in the South Shetland Islands (Acevedo et al., 2009; Romero & Tirira, 2017; Grebieniow et al., 2020; Braun et al., 2022). However, the number of leucistic seals in neighbouring Admiralty Bay appears to have increased in recent years (Grebieniow et al., 2020). Increased monitoring with short counting intervals may have contributed to a higher number of observed seals with aberrant coloration. Leucism is the rare, partial absence of pigment in mammals caused by a rare expression of a recessive gene (Hoffman et al., 2018). The fact that leucistic Antarctic fur seals are observed relatively frequently in the Scotia Arc region and especially on South Georgia, the main distribution area of this species (Bonner, 1968; Hofmeyr et al., 2005a; Wege et al., 2015; Hoffman et al., 2018), is attributed to the “founder effect” caused by the very small remaining population due to intensive hunting in the 19th and early 20th centuries (Grebieniow et al., 2020). The few surviving individuals on South Georgia thus formed a genetic 'bottleneck' (Hoffman et al., 2011). The sharply decreasing frequency of the gene segment responsible for leucism with geographical distance from South Georgia indicates a locally limited gene flow, especially to South Shetland Island and Bouvetøya (Hoffman et al., 2018). Thus, in more distant areas, migration from South Georgia does not appear to play a major role in the recovery of populations there, but may be locally significant (Hoffman et al., 2018).

Figure 36: Leucistic specimen of an Antarctic fur seal observed at 18.02.2022 at the north-western coast of the Fildes Peninsula

Photo: J.E. Findeisen, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The Antarctic fur seal, which was declared an important indicator species by CCAMLR in 1980, plays a crucial role in the Antarctic ecosystem. After intensive hunting in the 19th century, which brought it to the brink of extinction, the species' population has recovered strongly since the 1930s, mainly based on a few surviving animals on South Georgia (Hofmeyr, 2016; Foley & Lynch, 2020; Forcada et al., 2023; Krause et al., 2024). As a result of the rapid population growth, a population of 3.5 million animals was reported for South Georgia alone, the main distribution area of this species, between 2007 and 2009 (Forcada et al., 2023). More recent genetic studies have shown that at least four distinct relict populations survive around Antarctica, including on the South Shetland Islands (Wynen et al., 2000; Hofmeyr et al., 2005b; Bonin et al., 2013; Paijmans et al., 2020). Furthermore, the reduction in the extent and duration of sea ice as a result of regional climate change allows seals to move further south (Convey & Hughes, 2023). This has already led to damage to the fragile terrestrial habitats in previously less- or non-visited areas on the South Orkney Islands or the Antarctic Peninsula, e.g. through over-fertilization or trampling during the summer (Convey & Hughes, 2023).

Although the Antarctic fur seal is classified as “Least Threatened” by the IUCN, it is currently assumed that the global population trend is now in decline. A decline in populations has been observed in almost all distribution areas, indicating a drastic downward trend (Forcada et al., 2023; Krause et al., 2024). In addition to other environmental factors, increasing competition for food with recovering whale populations is suspected to be a cause of this (Convey & Hughes, 2023). The sub-population from the South Shetland Islands, which lies at the south-western limit of the species' range, also showed a drastic population decline (Krause et al., 2020; Krause & Hinke, 2021; Krause et al., 2022). The number of Antarctic fur seal pups on the San Telmo Islands and Cape Shirreff on Livingston Island, two main pupping sites in the South Shetland

Islands, has declined by 90 % between 2008 and 2023 (Krause et al., 2024). Predation of pubs by leopard seals appears to play a certain role here (Schwarz et al., 2013). It is possible that the significant decline in the number of fur seals observed in the Fildes Region in austral summer since the 2014/15 season reflects this drastic population decline.

Within the Antarctic fur seal population on South Shetland Island, an astonishingly high genetic diversity has been demonstrated, which is roughly equivalent to that on South Georgia, although the population is many times smaller (Krause et al., 2022). The hunting phase here was probably not long and extreme enough to have caused a strong bottleneck effect (Stoffel et al., 2018; Paijmans et al., 2020; Krause et al., 2022). As there is little genetic exchange between Antarctic fur seals on South Georgia and the South Shetland Islands, the latter form a reservoir of genetic diversity and may thus be important for the survival of this species in a warming Antarctic (Krause et al., 2022). For this reason, targeted protection of this population should be carefully considered in light of current risks (Krause et al., 2024). Over the last century, the threats to the Antarctic fur seal population have changed significantly (Krause et al., 2024); while in the past extensive hunting of this species was the main threat, today climate change, pollution, competition for resources with krill fisheries and recovering whale stocks, as well as tourism and disease, threaten the global population of this species (Convey & Hughes, 2023; Krause et al., 2024). Signs of a change in food use underline the role of the Antarctic fur seal as an indicator of environmental changes in the Southern Ocean ecosystem (Lima et al., 2022b). In particular, juvenile and sub-adult male Antarctic fur seals are subject to greater competition with krill fisheries and food resources in the waters off the western Antarctic Peninsula and the Scotia Sea due to their specific winter distribution, while females overwinter in areas further north (March et al., 2021). It also remains to be seen to what extent a reduced genetic diversity of the population on South Georgia, resulting from the small initial population due to the historical "bottleneck" as a result of commercial exploitation, will have a negative impact on resistance to H5N1 avian influenza.

5.13.4 Seal pupping sites in the Fildes Region

During the austral summer of the 2021/22 to 2023/24 seasons a small number of new-born seals were again recorded on the Fildes Peninsula and Ardley Island. Although the main reproduction season for elephant seals and Weddell seals is outside the austral summer and most of the pups have already left the area, a single elephant seal pup was observed in January 2023 at the north-west coast of the Fildes Peninsula (bay no. 2, see Figure 79 in Appendix E). This was clearly recognisable by its size and fur colour and was located next to the mother. Furthermore, four Antarctic fur seal pups were recorded in the 2021/22 and 2023/24 season, respectively, and three pups in the seasons 2022/23. In all three seasons, the area of the Russian field hut (bay no. 2) was again the birthing site of a small number of Antarctic fur seals (see Figure 80 in Appendix E). This makes this area the most consistent reproduction or pupping site in the study area (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). This site was first recorded in the 1986/87 season (Mönke & Bick, 1988) and in seal pups have been observed in 20 of the 22 years studied since the 2000/01 season. This indicates a remarkable breeding site fidelity of the Antarctic fur seal in the Fildes Region. In contrast, pups of this species have rarely been found at other parts of the coastline (season 2023/24: bay no. 3 & 14), so that there has been no significant expansion of the breeding sites in the study area since the 1980s (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). The observation of a juvenile fur seal in the 2023/24 season at the Flat Top peninsula at the western coast (bay no. 14/15) was only the second evidence of a pupping site of this species at this location after the 2019/20 season (Braun et al., 2022). Further fur seal pupping sites in the southern or eastern area of the Fildes Peninsula or on Ardley Island were not recorded in the seasons 2021/22 to 2023/24.

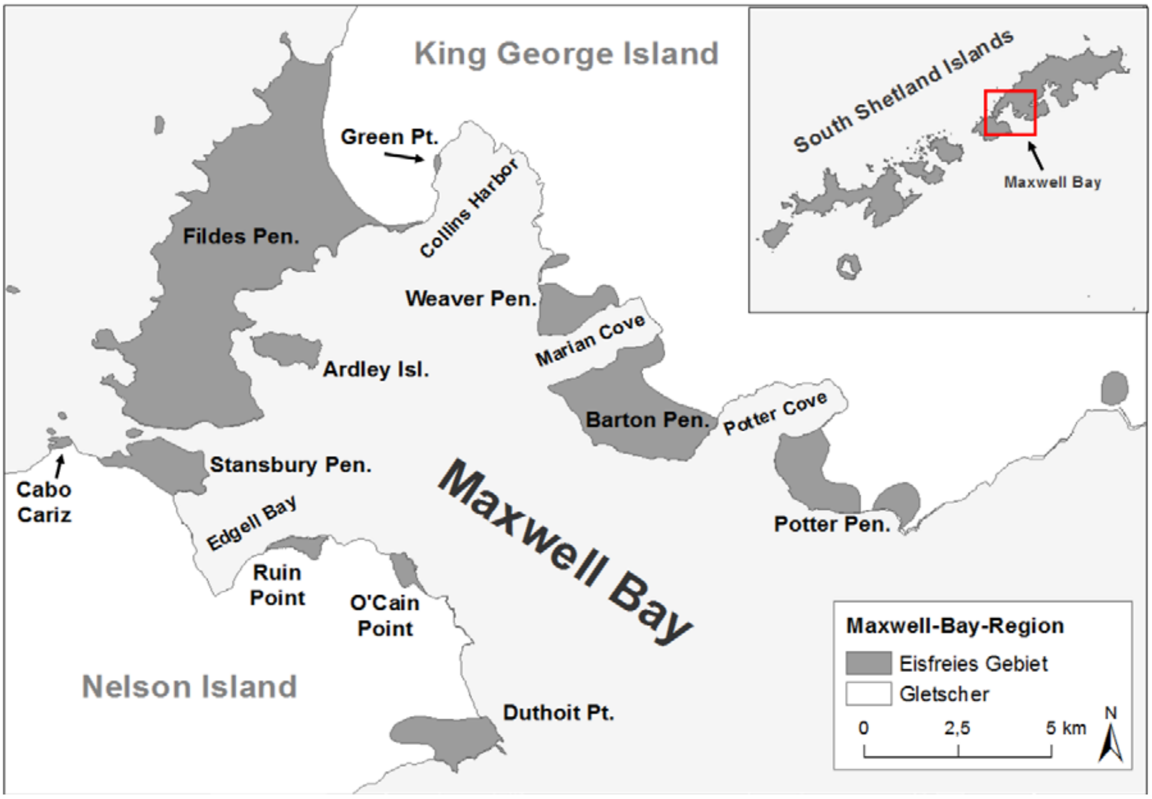
Reproduction of other seal species such as Weddell, leopard and crabeater seal were not recorded in the 2021/22 to 2023/24 seasons, as they mainly give birth in the months of September and October and thus outside the period when project staff were in the study area. Overall, knowledge of seal breeding sites in the Fildes Region can be confirmed based on current observations (Braun et al., 2022; see Figure 79 to 81 in Appendix E). However, the breeding numbers of seals in the Fildes Region are too low to be able to make general statements about reproduction trends of species.

6 Breeding bird monitoring in the ice-free areas of Maxwell Bay

6.1 Introduction

Due to major logistical constraints, the ice-free areas Green Point and Stansbury Peninsula could be visited during the study period (Figure 37). Data on breeding birds on Stansbury and Barton Peninsula were kindly provided by Czech, Korean and Ukrainian scientists respectively (see chapter 6.2 & 6.6). Further ice-free areas of the Maxwell Bay, namely Weaver and Potter Peninsula, Cabo Cariz, Ruin Point and Duthoit Point neither could be accessed nor data were available from other scientists or literature (Figure 37). Therefore, only limited data can be added to those listed in Braun et al. (2022).

Figure 37: Overview of the ice-free areas of Maxwell Bay, King George Island; representation of the South Shetland Islands without the islands of Clarence, Elephant and Gibbs

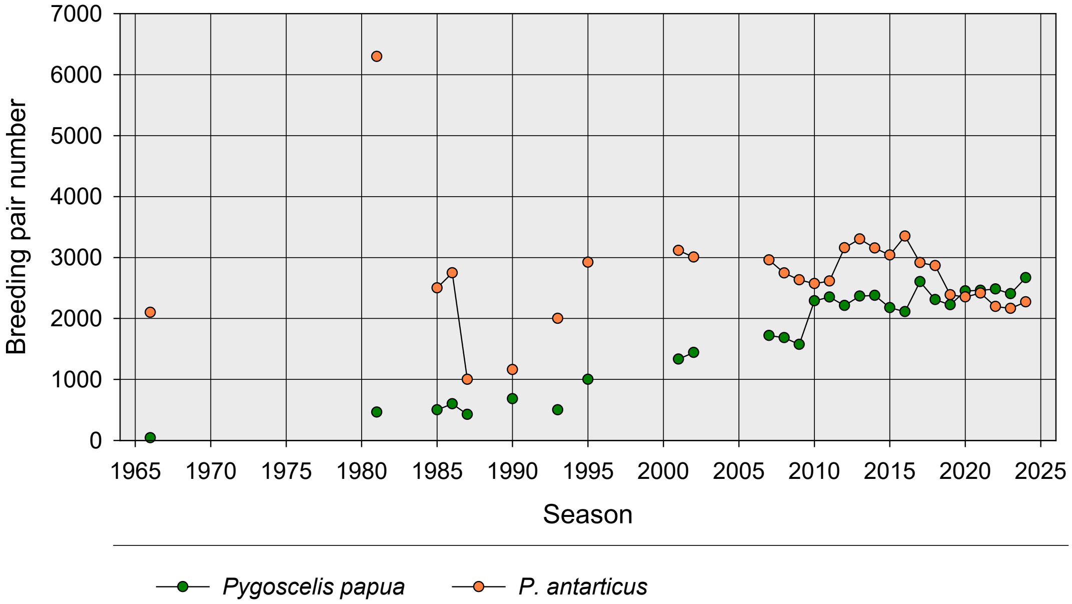


Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

6.2 Barton Peninsula, KGI

Although the Barton Peninsula could not be visited by project staff for logistical reasons, we were provided with data for the penguin colony at Narębski Point (ASPA No. 171) by Korean scientists as part of a long-term cooperation (Figure 1 & Figure 37). In addition, data on breeding birds in the ASPA No. 171 published in the revised Management Plan were used (Republic of Korea, 2024). Thus, it can be stated that the population of chinstrap penguins did not show a temporal trend over the survey period ($R^2 = 0.002$, $p = 0.818$, $n = 28$), although the number of breeding pairs of this species has decreased over the past ten years (Figure 38). In contrast, the population of gentoo penguins continued to increase ($R^2 = 0.926$, $p < 0.001$, $n = 28$). Since the 2018/19 season, the gentoo penguin has been the most abundant penguin species in this colony (Republic of Korea, 2024). As on Ardley Island, the highest number of breeding pairs of this species since the start of the long-term survey was recorded in the 2023/24 season (Republic of Korea, 2024).

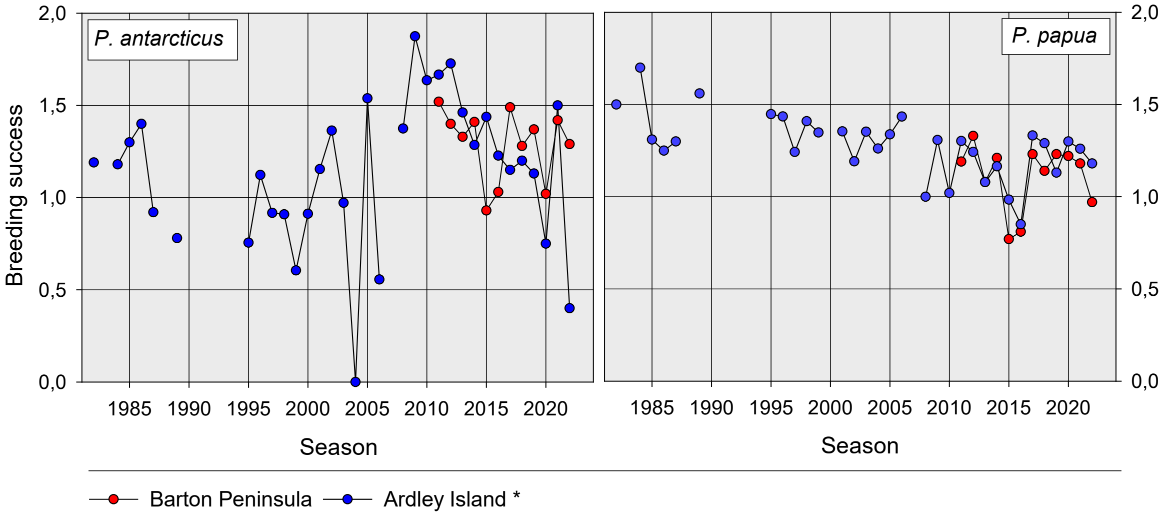
Figure 38: Development of breeding pair numbers of chinstrap penguins (*Pygoscelis antarcticus*) and gentoo penguins (*P. papua*) on Barton Peninsula (for data sources see Table 22 in Appendix F)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The breeding success of gentoo penguins at the Narębski Point penguin colony in the 2021/22 season was 0.97 juv/bp, well below the long-term average of 1.12 ± 0.17. In the following 2022/23 season, it was higher again at 1.19 juv/bp (Figure 39). The success of the chinstrap penguin in the Narębski Point penguin colony was in the average range with 1.29 and 1.46 juv/bp respectively, while only 0.4 chicks were raised on Ardley Island (Figure 39). However, the number of breeding pairs on Ardley Island was very low at only 4 pairs. Data on the number of fledged chicks for the 2023/24 season has not yet been provided, so no comparison with the breeding success data from Ardley Island is possible.

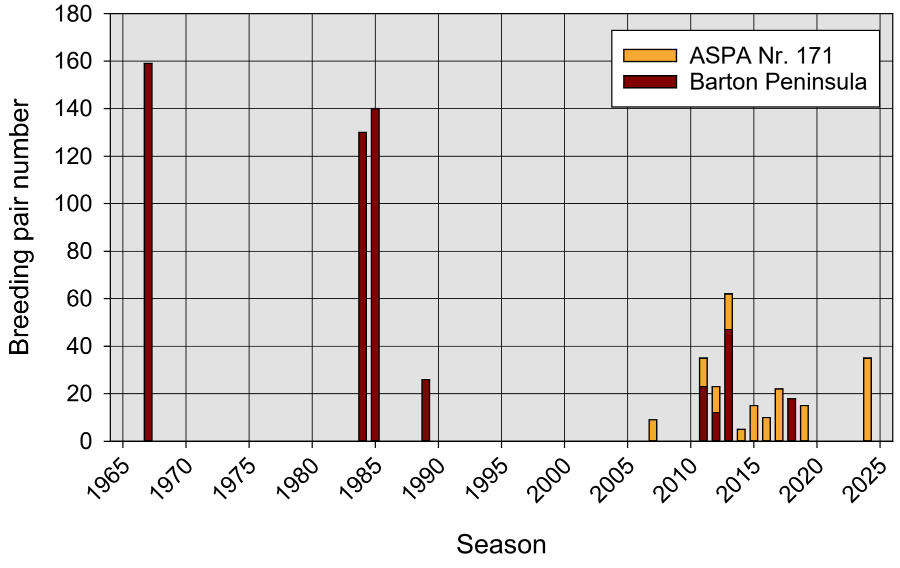
Figure 39: **Comparison of breeding success of gentoo and chinstrap penguins on Barton Peninsula and Ardley Island (data sources for Barton Peninsula given in Table 22 in Appendix F; * only a few breeding pairs)**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Updated information on all other breeding bird species on the Barton Peninsula is only available from the 2023/24 season for the area of ASPA No. 171 Narębski Point. According to this, 35 pairs of southern giant petrel were recorded. It can be seen here that the population of this species was significantly higher in the past, but the number of breeding pairs within ASPA No. 171 reached a maximum in the 2023/24 season (see Table 22 in Appendix F).

Figure 40: **Development of breeding pair numbers of southern giant petrels (*Macronectes giganteus*) on Barton Peninsula (for data sources see Table 22 in Appendix F)**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Furthermore, 7 bp of brown skua, 10 bp of south polar skua, 2 bp of kelp gull, 2 bp of Antarctic tern, > 10 bp of black-bellied storm-petrel, > 132 bp of Wilson's storm-petrel and 6 bp of snowy sheathbill bred within ASPA No. 171 in the 2023/24 season (Republic of Korea, 2024, see Table 22 to 24 in Appendix F). Breeding of the blue-eyed shag has never been documented on the Barton Peninsula (season 1986/87 - Mönke & Bick, 1988, season 2012/13 – J.-H. Kim, pers. comm.).

6.3 Weaver Peninsula, KGI

Due to logistical constraints, the Weaver Peninsula, King George Island, could not be visited during the study period (Figure 37), so more recent data on breeding bird populations are not available. All available information on breeding birds documented on this peninsula is listed in Table 25 & 26 in Appendix F.

6.4 Green Point, KGI

On the small Green Point peninsula to the northeast of the Fildes Peninsula (Figure 37), which can be accessed by foot, a smaller number of breeding pairs of kelp gulls were recorded during the study period, with between 4, 7 and 5 bp respectively (Table 27 in Appendix F). In the 2021/22 season, a pair of south polar skuas bred there. No other breeding birds were recorded.

6.5 Potter Peninsula, KGI

The Potter Peninsula in the east of Maxwell Bay (Figure 37) could not be visited in person during the study period in the study area, which is why the information on the breeding bird population was updated based on available literature (see Table 28 in Appendix F). Recent publications indicate a sharp decline in the Adélie penguin population in selected nest groups at the Stranger Point colony between the 1995/96 and 2022/23 seasons, totalling -89.8 % (Juáres et al., 2024). Similarly, the number of Adélie penguins on Ardley Island fell by 85 % in the same period. The number of fledged chicks fell by 93.1 % at Stranger Point (Juáres et al., 2024) and by 82.8 % on Ardley Island. These comparative results indicate a very uniform regional development in response to changing environmental conditions. Breeding success showed no decreasing trend in either colony (Juáres et al., 2024).

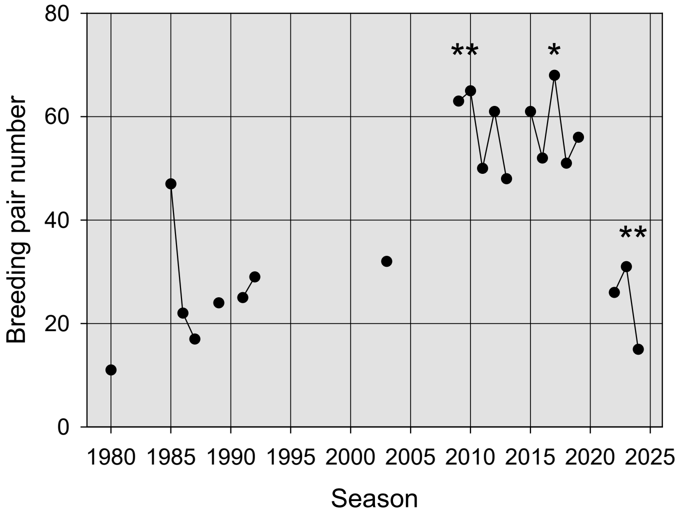
The extent to which the increasing trend of the gentoo penguin population in the colony at Stranger Point has continued in recent years (Juáres et al., 2020; Braun et al., 2022) can only be assumed, as it corresponds to the supra-regional development in the WAP region. Statements about the possible presence of chinstrap penguins are also currently not possible due to a lack of available data.

With regard to the king penguin, annual broods of one pair have now been documented for the Potter Peninsula in the 2011/12 to 2020/21 seasons (Juáres et al., 2023). Although neither of the two chicks that have hatched so far have been successfully raised, the ongoing breeding attempts, including in neighbouring Admiralty Bay, are interpreted as an incipient expansion of the species' range to the south (e.g. Gryz et al., 2019; Borowicz et al., 2020; Juáres et al., 2023).

6.6 Stansbury Peninsula & Cabo Cariz, Nelson Island

The Stansbury Peninsula in the north of Nelson Island could only be visited on 21 December 2021. As in the past, only the area adjacent to Fildes Strait was visited, but not the area further south-east in the vicinity of the Czech refuge. In the 2022/23 season, the breeding population of the southern giant petrel was recorded by Czech scientists in the entire ice-free area. In the 2023/24 season, a Czech-Ukrainian team of scientists carried out a comprehensive survey of breeding birds on the Stansbury Peninsula, which also included areas of this ice-free region that had not previously been covered. According to their own data and the data provided, 26, 31 and 15 bp, respectively, were breeding here in the seasons 2021/22 to 2023/24, significantly fewer pairs than in the previous decade (pers. comm. V. Pavel; Figure 41; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). In the 2022/23 season, only 3 chicks were documented at the beginning of February, which equates to a breeding success of just 0.1 juv/bp. In the following 2023/24 season, not a single chick was even recorded in January 2024. This means that the population of this bird species in this area was only at the level of the 1980s and 1990s. The cause of the current decline is not known, especially as the immediately neighbouring Fildes region recorded the highest breeding bird population of the last four decades. However, the long-term trend in breeding parity of the southern giant petrel on Stansbury Peninsula is still positive ($R^2 = 0.238$, $p = 0.025$, $n = 21$, Figure 41), possibly indicating a breeding site shift due to anthropogenic disturbance, as has also been shown for the Fildes Region (Peter et al., 1991; Chupin, 1997; Braun et al., 2012; Peter et al., 2013; Braun et al., 2017a). Like the number of breeding pairs, the breeding success of the southern giant petrel on Stansbury Peninsula has fallen drastically in recent years and now averages only 0.26 ± 0.20 juv/bp. The overall breeding success for the Fildes Region in the same period was higher at 0.37 ± 0.14 juv/bp (see chapter 5.3). The colonies neighbouring the Stansbury Peninsula do not reflect such a collapse in breeding pair numbers and breeding success, with the exception of the very small colony at Fildes Strait, where a decline in breeding pair numbers and a complete loss of broods was also recorded in the last two years (Figure 8). All other comparable colonies in the neighbourhood, such as Dart and Two summit Island, showed no such development, but on the contrary an above-average breeding success in the 2022/23 and 2023/24 seasons (see Figure 8).

Figure 41: Development of breeding pair numbers of the southern giant petrel (*Macronectes giganteus*) on Stansbury Peninsula, Nelson Island (* late census date in January or February)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The breeding population of skuas on Stansbury Peninsula could only be recorded in the 2023/24 season. The count in January 2024 revealed a total of 2 bp of brown skua and 11 bp of south polar skua (pers. comm. V. Pavel & I. Parnikoza). In addition, 3 territories (1 brown skua, 2 south polar skua) and 3 older nests (1 brown skua, 2 south polar skua) were recorded.

In the 2021/22 season, one breeding pair of kelp gulls was recorded in the area bordering Fildes Strait. In contrast, more than 50 adult birds and 20 older chicks were observed in the western area and more than 25 adult birds and 5 older chicks in the eastern area during an area-wide survey in mid-January in the 2023/24 season (pers. comm. V. Pavel). As these data were collected using a different methodology, the number of breeding pairs is not given.

A complete survey of Antarctic terns only took place in the 2023/24 season, with more than 30 bp recorded in the north-western area of the Stansbury Peninsula. In addition, more than 30 bp and 2 bp were recorded at two other breeding sites in the south-east of the peninsula (pers. comm. V. Pavel).

In line with the population trend of the cape petrel on the Fildes Peninsula (see chapter 5.4), the sharp decline of this species - despite the observation of some flying individuals - was also confirmed for the Stansbury Peninsula, although this could not be quantified (pers. comm. V. Pavel). Furthermore, an adult king penguin was observed in the western area of the Stansbury Peninsula on 13 February 2023 (pers. comm. O. Stanický, D. Jindra).

All available breeding bird data from Stansbury Peninsula are listed in Table 35 to 37 in Appendix F.

6.7 Duthoit Point, Cabo Cariz & Ruin Point

A survey of breeding birds at Stansbury Peninsula Cabo Cariz and Ruin Point on Nelson Island (Figure 30) could not be realised during the study period due to logistical constraints. Therefore, no new information compared to Braun et al. (2022) could be collected. All breeding bird data available to date are listed in Table 38 & 39 in Appendix F. However, one adult Macaroni penguin was reported observed in the chinstrap penguin colony at Martin Point on 10 January 2024 (pers. comm. V. Pavel).

6.8 Conclusions

Despite the logistical difficulties of reaching all the ice-free areas of Maxwell Bay, the aim should be to continue the surveys in the future, as all the data obtained here can provide information on the development of local breeding bird populations. In this way, statements can be made about population trends and possible breeding site shifts or recolonization of areas caused by recent glacier retreat.

7 Occurrence of the Antarctic pearlwort (*Colobanthus quitensis*) in the Fildes Region

In addition to the Antarctic hairgrass (*Deschampsia antarctica*), which has experienced a strong increase in the Fildes Region in recent decades (Braun et al., 2020a; Braun et al., 2022), the Antarctic pearlwort (*Colobanthus quitensis*) is one of the two vascular plants native to the Antarctic. Its distribution extends along the Antarctic Peninsula to 69° S (Convey et al., 2011). Evidence for the species is available for wide areas of the South Shetland Islands (e.g. Pfeiffer & Peter, 2003; Pérez-Torres et al., 2004; ATS, 2005a; Vera, 2011; ATS, 2012; Vera et al., 2013; ATS, 2018a, b, 2022).

For King George Island, occurrences of the Antarctic pearlwort have been described at several locations, where they sometimes form dense populations, e.g. in Admiralty Bay (Brazil & Poland, 2005; Kozeretska et al., 2010; Torres-Díaz et al., 2016; Sanhueza et al., 2017), on Potter Peninsula (Flocco et al., 2009; ATS, 2013) and Barton Peninsula (Kim et al., 2007; Cho et al., 2018).

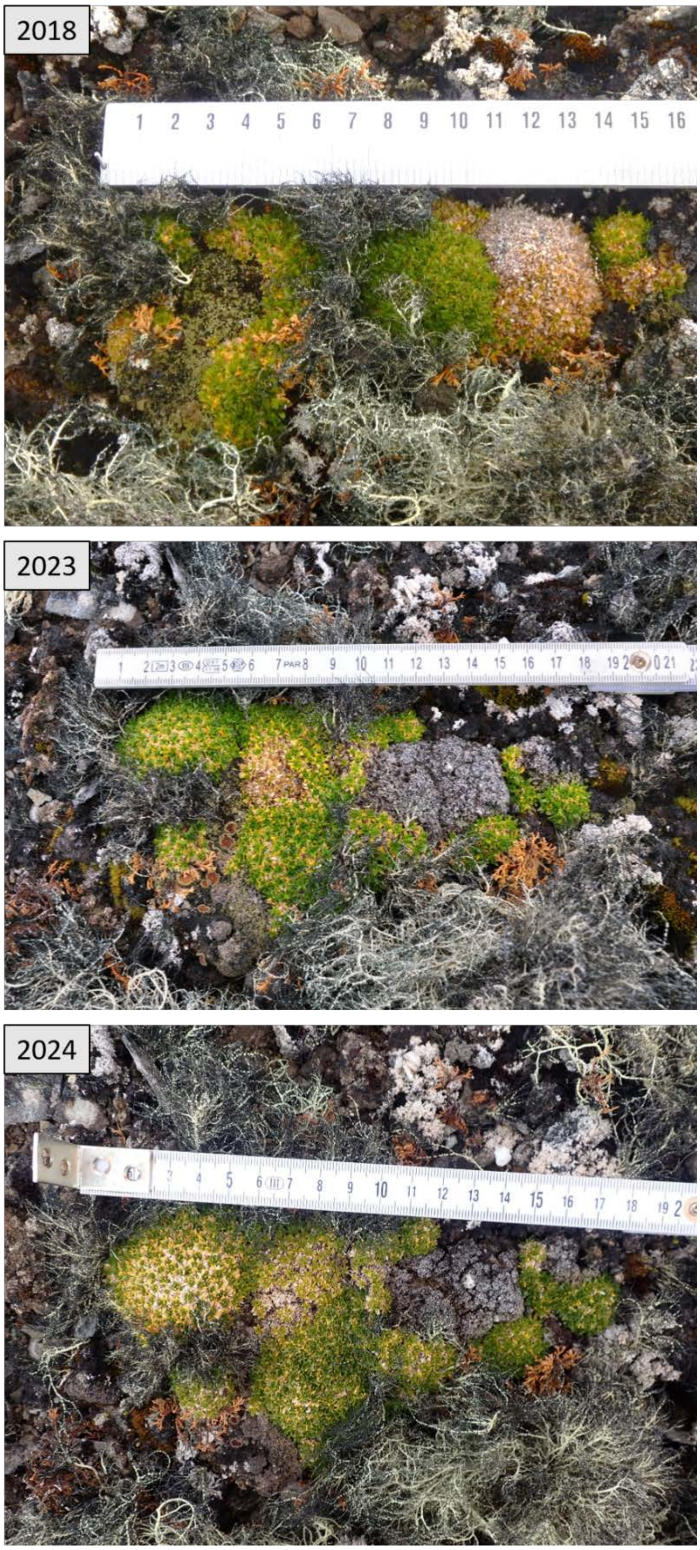
In the Fildes Region, located in south-west King George Island, only a few locations of the Antarctic pearlwort have been documented to date (Peter et al., 2008; Peter et al., 2013; Braun et al., 2020c; Braun et al., 2022). However, the populations that were recorded in 2000 and 2008 were completely extinct after just a few years (Peter et al., 2008; Peter et al., 2013). A population of pearlwort known since 2018 near the Uruguayan station of Artigas on Lake Uruguay was monitored and documented in January 2023 and 2024. The size and condition of the cushions of this plant were compared with the time the population was discovered in 2018. It can be seen that this stand had both dead and additionally overgrown areas (Figure 42). Lighter-coloured, faded areas possibly indicate dying areas. Overall, the stand appeared slightly larger than in 2018 (160 cm^2) with a final area of approx. 17.5 x 11 cm or 193 cm^2 (Figure 42; Braun et al., 2020a). Flowers were not observed in 2023 and 2024, but this may be due to the timing of the control of the stand.

The small population of Antarctic pearlwort first recorded in the far north of the Fildes Peninsula near the edge of the glacier in the 2020 season (Braun et al., 2022) was no longer found during surveys in February 2023 and 2024.

The nearest stand of Antarctic pearlwort on Cabo Cariz, Nelson Island, which was first localised in the 2007/08 season (pers. comm. M. Andreev) and last documented in 2018 (Braun et al., 2020c), could not be visited during the study period for logistical reasons. Therefore, no further information is available on the status or possible spread of the plant at this site.

In view of the numerous populations in the immediate vicinity on King George Island (see above), the environmental conditions for the establishment and spread of this plant species in the Fildes Region still do not appear to be favourable, with the prevailing strong westerly winds being suspected as the main cause (Braun et al., 2022).

Figure 42: **Comparison of the status of the currently only known population of pearlwort in the Fildes Region (*Colobanthus quitensis*) in the Fildes Region in 2018, 2023 and 2024**



Photos: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

8 Climatic changes in the study area

8.1 Introduction

The polar regions are of significant importance to the global climate system, regulating ecosystem functions at both the local and regional levels (Abram et al., 2014). In recent decades, there have been notable changes in the environment of Antarctica, including the retreat of glaciers and sea ice in the Antarctic Peninsula region (e.g. Turner et al., 2005a; Turner et al., 2005b; Schofield et al., 2010; Turner et al., 2014). These changes are mainly attributed to the substantial and rapid temperature increase that have occurred in recent decades (e.g. Kejna, 2003; Meredith & King, 2005; Ducklow et al., 2007; Turner et al., 2007; Steig et al., 2009; Stastna, 2010; Bromwich et al., 2013; Kejna et al., 2013; Bromwich et al., 2014; Znój et al., 2017; González & Fortuny, 2018; Carrasco et al., 2021; Bello et al., 2022; Gorodetskaya et al., 2023). An indication of this is the 3°C increase in temperature that was observed between 1955 and 2004 on the Antarctic Peninsula (Turner et al., 2005a). A warming trend was also observed in the sub-Antarctic islands during the mid to late 20th century (Bergstrom & Chown, 1999; Jacka et al., 2004). The primary driver of ocean and atmospheric warming can be attributed to the anthropogenic intensification of the Southern Annular Mode (Pohl et al., 2021). This phenomenon results in the strengthening and poleward shift of westerly winds in the southern hemisphere (Perren et al., 2020; Nel et al., 2023). Despite a temporary reversal of the warming trend between 1995 and 2016, the Antarctic Peninsula is now exhibiting a renewed and intensified phase of warming (Turner et al., 2016; Oliva et al., 2017; Sancho et al., 2017; González & Fortuny, 2018; Plenzler et al., 2019; Carrasco et al., 2021). Furthermore, there has been an increase in temperature extremes (Turner et al., 2021). The temperature extremes of 2015 and 2020 (e.g. 24 March 2020 at the Argentinian station Esperanza, where the air temperature reached 18.3 °C (World Meteorological Organization, 2021)) were accompanied by a new extreme warming event with record temperatures in February 2022 (9.3 at Bellingshausen station, 13.7 °C at King Sejong station). This event can be attributed to an intense atmospheric river which impacted the northern/northwestern Antarctic Peninsula (Gorodetskaya et al., 2023). In addition, results demonstrate that the 2020-like heatwaves over the Antarctic Peninsula are currently at least 0.4 °C warmer than in the past, representing a 25 % increase (González-Herrero et al., 2022). Recent temperature and precipitation projections for the Antarctic Peninsula indicate an increase in mean annual near-surface temperatures of approximately 0.5 – 1.5 °C across the entire peninsula, accompanied by an increase in mean annual precipitation of approximately 5 – 10 % (Bozkurt et al., 2021).

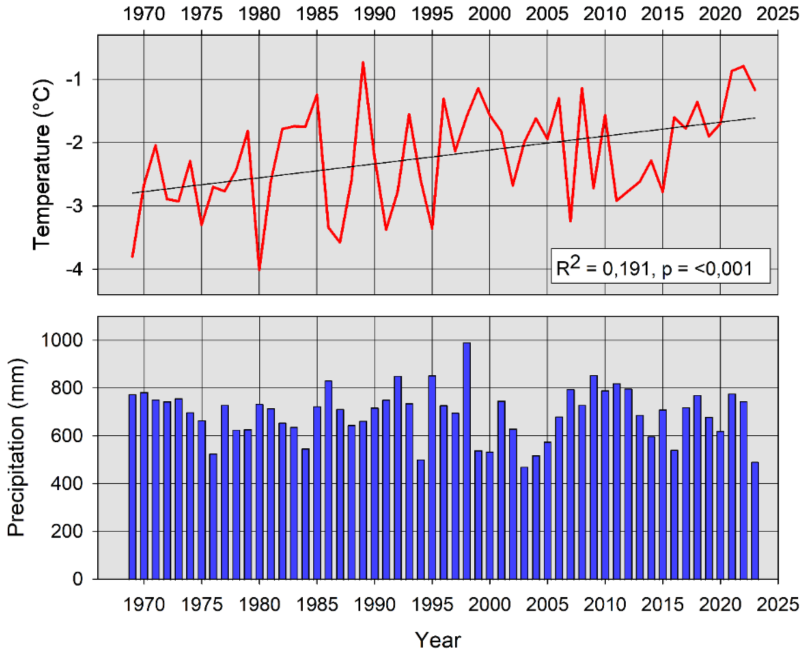
8.2 Evaluation of local meteorological data

The evaluation of the local meteorological data, measured at Bellingshausen station, was determined by analysing the monthly mean values of the air temperature above ground as well as the daily values of snow depth and precipitation. In addition, the global annual mean temperatures were used to assess the temperature development in the study area. Changes in the local climatic conditions between 1968 and 2013, 2018 and 2021 were already described in previous studies (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022).

The annual mean temperature in the study area over the past 55 years showed a significant positive trend and increased by 1.21 °C between 1969 and 2023 (Figure 43). This corresponds to an annual warming of 0.022 °C and to a further increase compared to previous periods (Braun et al., 2020c; Braun et al., 2022). For example, between 1969 and 2020, a change in the annual mean air temperature of 0.889 °C was calculated with an annual growth rate of 0.017 °C. A comparison of the annual mean temperature of the study area with the average global temperature values reveals a similarity in the intensity of the temperature change, as the global annual mean air temperature increased by 1.1 °C between 1969 and 2023. The heat map shown here emphasises the current increase in the mean annual temperature (Figure 44).

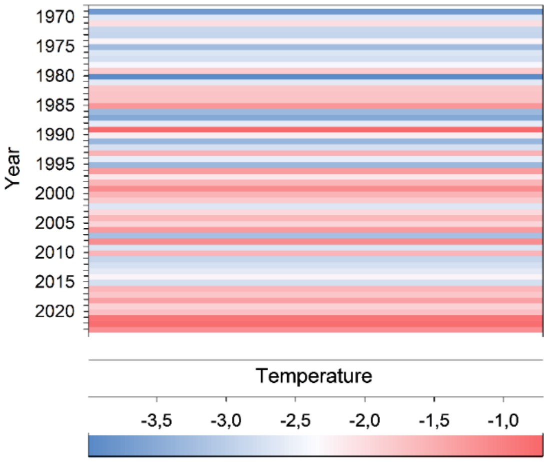
The annual precipitation in the study area showed a high variability without a significant trend ($R^2 = 0.004$, $p = 0.639$, $n = 55$). Furthermore, the negative correlation between annual precipitation and mean annual temperature, which was shown in Braun et al. (2022), could be confirmed (Spearman corr.: $r = -0.320$, $p = 0.0175$, $n = 55$).

Figure 43: Development of the mean annual air temperature and annual precipitation at Bellingshausen station between 1969 and 2024 (modified after Pasik et al., 2021)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

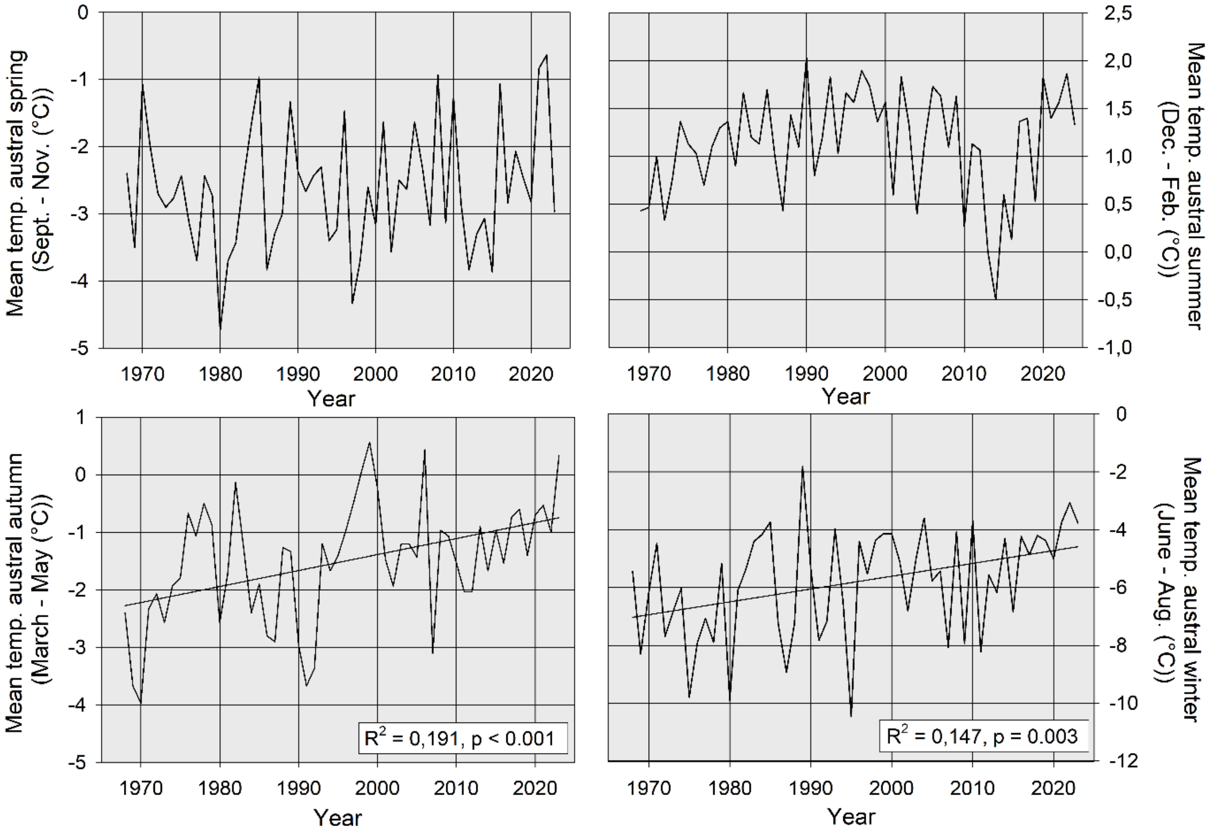
Figure 44: Heat map of the mean annual air temperature at Bellingshausen station between 1969 and 2023



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Furthermore, the mean temperature of the individual seasons showed a significant positive trend for winter (June, July and August: $R^2 = 0.147$, $p = 0.003$, $n = 56$) and autumn (March, April, May: $R^2 = 0.191$, $p < 0.001$, $n = 56$, Figure 45). No significant trends were present for summer (December, January and February: $R^2 = 0.006$, $p = 0.565$, $n = 56$) and spring (September, October, November: $R^2 = 0.024$, $p = 0.251$, $n = 56$, Figure 45). These trends are consistent with the previous results (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022) and has become much clearer, especially in the autumn and winter season.

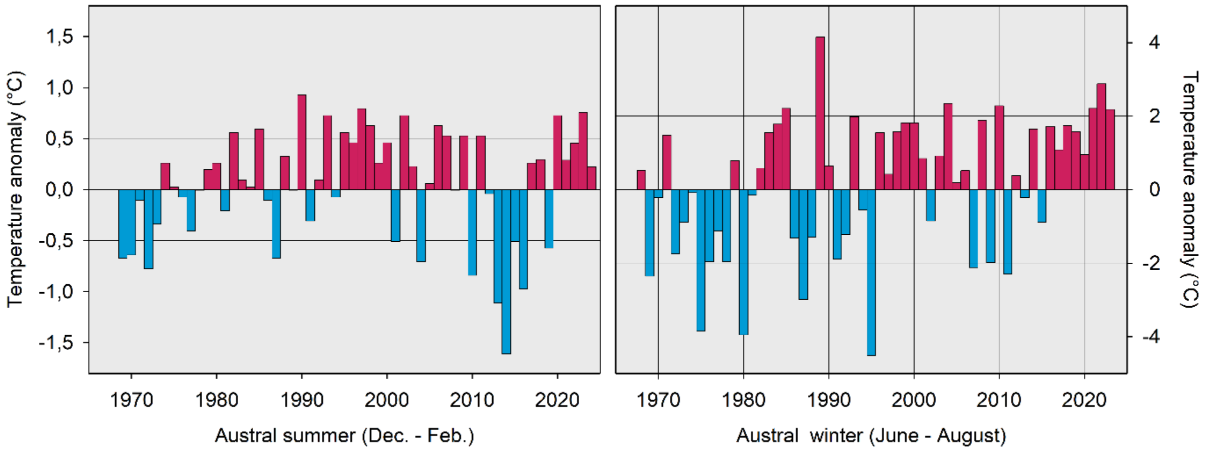
Figure 45: Development of the mean air temperature during the austral sring, summer. autumn and winter at Bellingshausen station between 1969 and 2024



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

When analysing the temperature anomaly (Figure 46), i.e. the deviation from the annual average, it is noticeable that after a phase of colder months between the 2009/10 and 2015/16 seasons, which was described in Braun et al. (2022), there has been a significant increase in individual warm months in both winter and summer since the 2015/16 season.

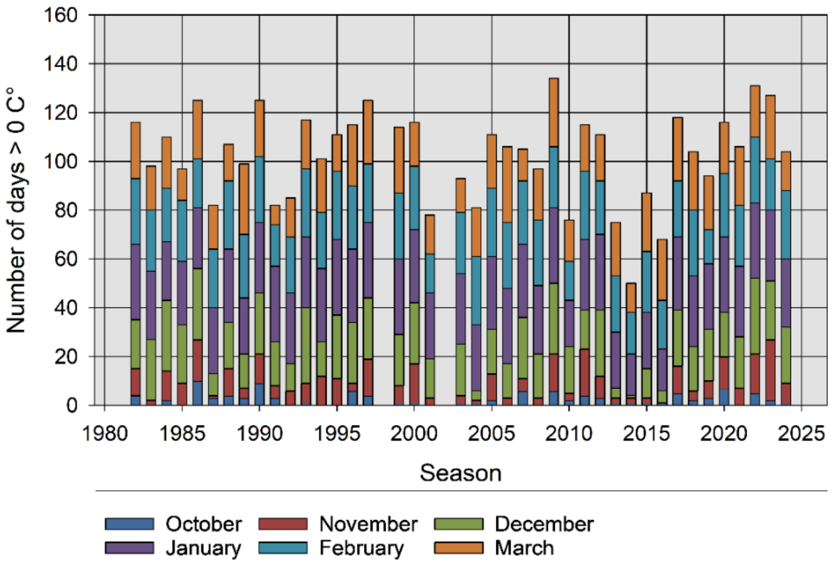
Figure 46: **Air temperature anomaly during the summer and winter months between 1968 and 2024; note the different scaling of the Y-axes**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The number of above-zero days in an area is essential for the development of glaciers, the spread of different plant species and many other processes in the area and can provide an explanation for many processes. They are defined by the days on which the mean temperature exceeds the freezing point. In the long term, however, the number of above-zero days in the study area did not show a linear trend ($R^2 = 0.003$, $p = 0.753$, $n = 41$, Figure 47), although an increase in the number of above-zero days is generally expected as a result of climatic development (Bozkurt et al., 2021). It is also worth noting that after a decline in the number of above-zero days around the 2014/15 season, a new increase has been recorded since then.

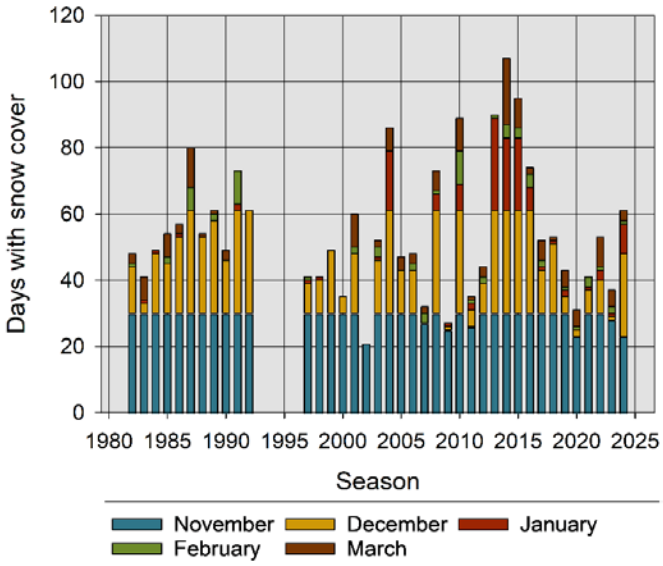
Figure 47: **Development of above-zero days in the months of October to March in the 1981/82 to 2023/24 seasons**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

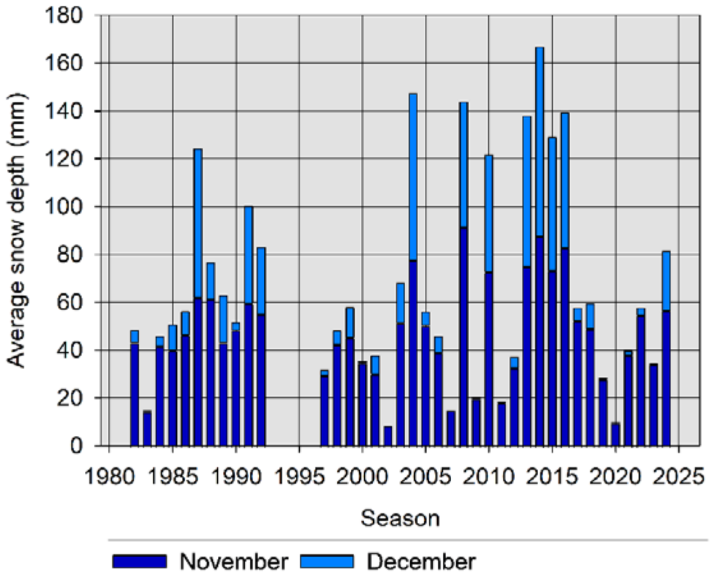
The number of days with snow cover and the average snow depth in November and December are particularly important for the number of breeding pairs and the breeding success of the birds, as the influence of these environmental factors is greatest at the beginning of the breeding season. In this respect, however, the number of days with snow cover has shown considerable fluctuations in recent years and no variable temporal trend ($R^2 < 0.001$, $p = 0.950$, $n = 37$, Figure 48). Nevertheless, as expected, a highly significant negative correlation between the development of above-zero days and the number of days with snow cover could be shown (Spearman corr.: $r = -0.729$, $p < 0.001$, $n = 37$).

Figure 48: Number of days with snow cover in the months of November to March in the 1981/82 to 2023/24 seasons



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Figure 49: Average snow depth in November and December in the 1980/81 to 2023/24 seasons



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The average monthly snow depth in November and December also showed a high variability without a noticeable long-term trend ($R^2 = 0.007$, $p = 0.603$, $n = 39$, Figure 49). It is worth mentioning the recognisable slight increase in average snow depth in the 2023/24 season compared to previous seasons, but did not reach the maximum values (Figure 49).

The meteorological data presented here from the Bellingshausen station underline the frequently described climatic changes in the WAP region (e.g. Kejna, 2003; Meredith & King, 2005; Ducklow et al., 2007; Turner et al., 2007; Steig et al., 2009; Stastna, 2010; Bromwich et al., 2013; Kejna et al., 2013; Bromwich et al., 2014; Znój et al., 2017; González & Fortuny, 2018; Wauchope et al., 2019; Carrasco et al., 2021; Bello et al., 2022; Gorodetskaya et al., 2023). Accordingly, a general increase in air temperature can be observed from the second half of the 20th century onwards. Furthermore, the study area has experienced a resurgence in warming since 2016, a trend that aligns with the prevailing literature on the subject (Carrasco et al., 2021; Bello et al., 2022). The conclusion of the cold period posited by Braun et al. (2022) is now a reasonable assumption, even when compared to other published works (Carrasco et al., 2021). Additionally, it is noteworthy that a temperature extreme in February 2022, which affected the entire Antarctic Peninsula (Gorodetskaya et al., 2023), was also recorded in the study area.

9 Glacier retreat areas of selected areas of Maxwell Bay

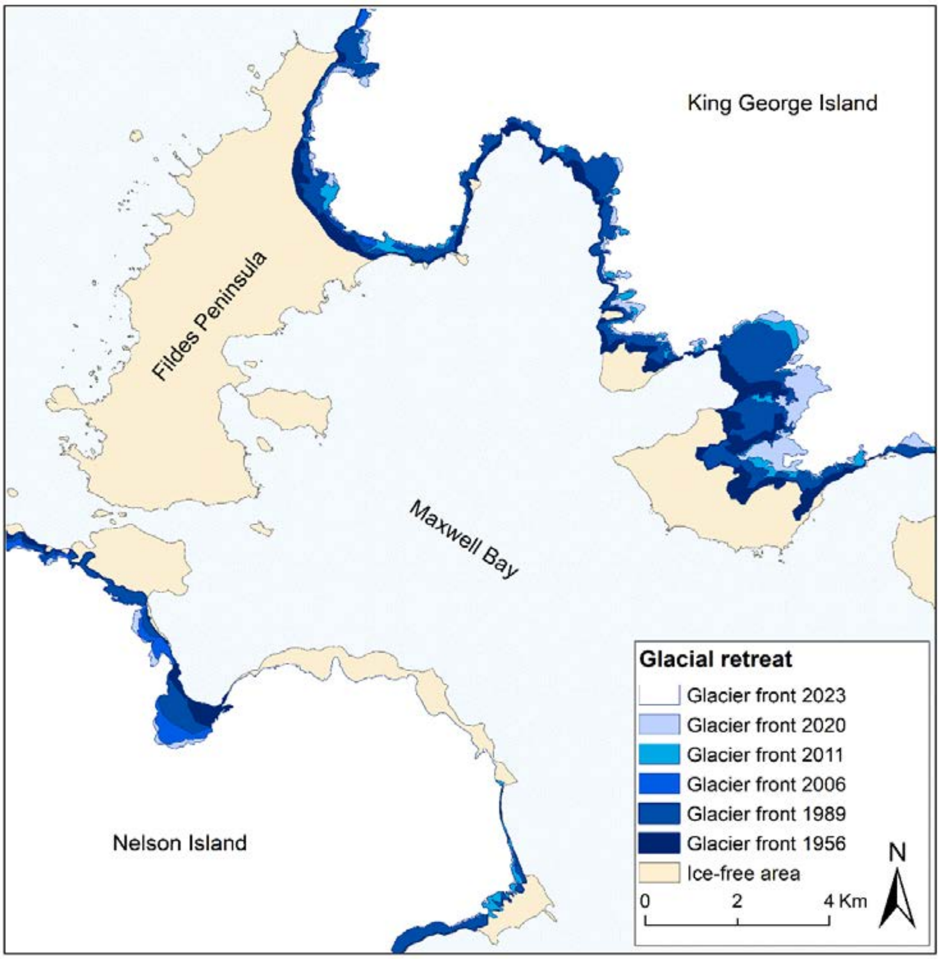
9.1 Introduction

In consideration of the nearly complete glaciation of the Antarctic – and in the case of King George Island, with an estimated 90 % glacial coverage (Pasik et al., 2021) – the development of the ice cover provides insights into the effects of the current climatic change, as the relationships between climate change and the retreat and advancement of glaciers are complicated (Dziembowski & Bialik, 2022). However, there is a reciprocal relationship, as the presence of glaciers and ice domes has a significant impact on atmospheric conditions. Consequently, these interactions result in weather and climate changes at varying scales, from the local to the global (Benn & Evans, 2014). The glaciers of the Antarctic Peninsula, located north of 70°S, have the potential to contribute an additional 69 ± 5 mm to the global mean sea level (Shepherd et al., 2018; Wallis et al., 2023). The formation of glaciers is a process whereby accumulated snow is transformed into ice. A reduction in glacier volume may be observed if the process of ablation as melting, calving, drifting, and sublimation (the direct transition of water from a solid to a gaseous state) exceed the accumulation processes of the snow masses. Ablation processes are caused by solar radiation, air temperature and precipitation. The study area is characterised by two main types of glacier. They can be classified as either 'land-terminating glaciers' or 'tidewater glaciers' (Braun & Goßmann, 2002; Sobota et al., 2015). The type of glacier and the relief of the subsurface are the determining factors in the manner of mass loss. Land-terminating glaciers typically exhibit a gradual retreat, whereas tidewater glaciers calve icebergs, resulting in an accelerated rate of retreat. The morphology of the glacier is a significant factor in determining the rate of retreat (Pasik et al., 2021). The aim here is to analyse and update the frequently described loss of ice mass from the glaciers on King George Island (e.g. Kejna et al., 1998; Park et al., 1998; Macheret & Moskalevsky, 1999; Birkenmajer, 2002; Braun & Goßmann, 2002; Rückamp et al., 2011; Osmanoglu et al., 2013; da Rosa et al., 2015; Sobota et al., 2015; Lee et al., 2017b; Lirio et al., 2017; Pudełko et al., 2018; Szilo & Bialik, 2018; da Rosa et al., 2020; Petsch et al., 2020; Kreczmer et al., 2021; Mojica Moncada et al., 2021; Pasik et al., 2021; Dziembowski & Bialik, 2022) by documenting horizontal changes in the glacier fronts.

9.2 Results

The overview map of the change in glacier areas in the regions bordering Maxwell Bay illustrates the substantial retreat of the glacier fronts, which has continued to increase in recent years (Figure 50, Table 7). This is particularly evident in the area of the Collins Harbour and Marian Cove tidewater glaciers (Figure 50). The current strong glacier retreat in the last 5 years is attributed to the high temperatures in both summer and winter. Furthermore, there were no significantly higher precipitation values in the winter months (see chapter 8.2).

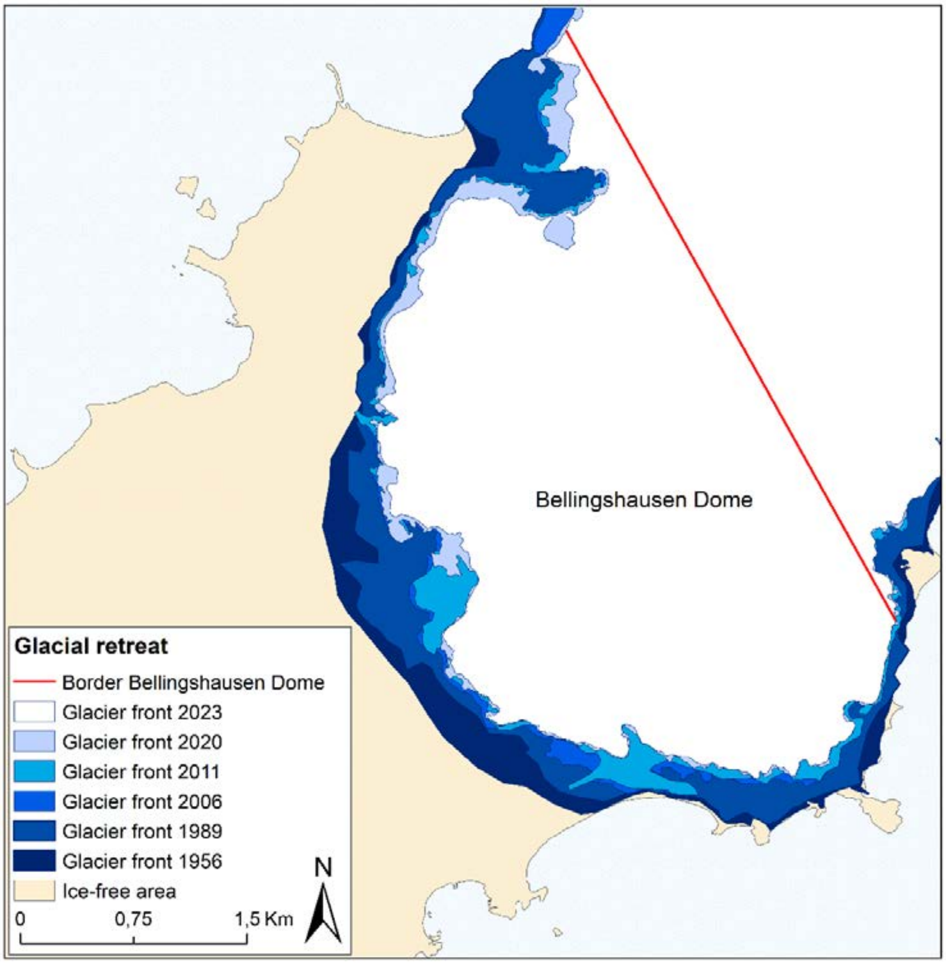
Figure 50: **Glacier retreat areas of the regions bordering Maxwell Bay between 1956 and 2023, documented by overlaying the respective positions of the glacier fronts**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

In detail, the area of the Bellingshausen Dome (also known as Collins Glacier) decreased by 3.15 km² between 1956 and 2023, from 12.29 km² to 9.14 km² (Figure 51, Table 7). This corresponds to a loss of 34.51 %. In addition, the glacier front has shifted by approx. 375 m in 67 years, which corresponds to an annual retreat of 5.6 metres. These findings are not in line with de Oliveira et al. (2021), who stated a loss of glacial area of 0.61 km² between 1988/89 and 2016/18 which corresponds to an annual change rate of 0.022 km²/a which is about half of what is shown here with 0.042 km²/a.

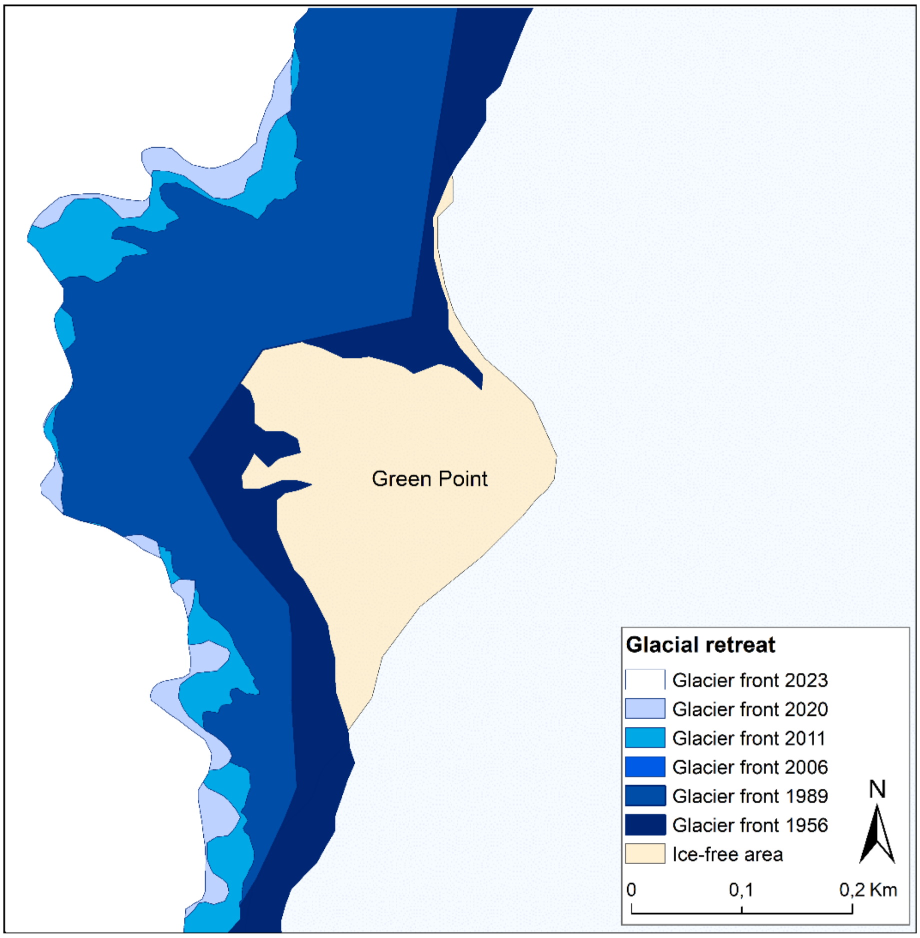
Figure 51: Glacier retreat areas of the Bellingshausen Dome in the north of the Fildes Peninsula between 1956 and 2023, documented by overlaying the respective positions of the glacier fronts; the boundary used to calculate the area of the Bellingshausen Dome is marked in red



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The regional differences in glacier retreat are mainly attributed to the relief in the regions. This can be clearly seen at the small Peninsula Green Point, where hardly any glacier retreat has been observed in recent years. Nevertheless, the ice-free area around Green Point has increased substantially over the entire observation period from 0.06 km² to 0.21 km² between 1956 and 2023 (Figure 52, Table 7). Overall, this corresponds to the largest proportional expansion of the ice-free area of all study areas at approx. 71.59 %. In addition, the glacier front has shifted by 214.8 metres in 67 years, which corresponds to an annual retreat of 3.2 metres.

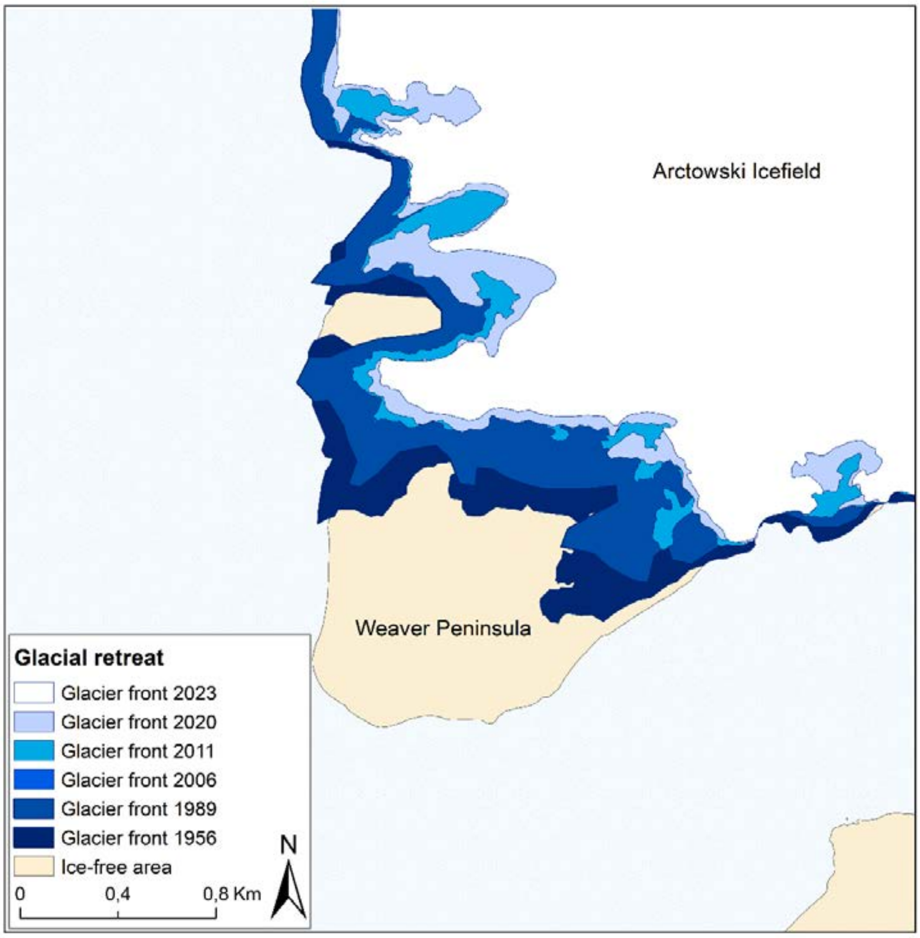
Figure 52: Glacier fronts Glacier retreat areas at Green Point between 1956 and 2023, documented by overlaying the respective positions of the glacier fronts



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Similarly, after stagnating between 2006/07 and 2011/12, the Weaver and Barton Peninsulas showed a renewed significant retreat of the glacier front between 2012 and 2023. Between 1956 and 2023, the ice-free area of the Weaver Peninsula increased from 1.13 km^2 to 2.29 km^2 by a total of approx. 1.16 km^2 and 50.56 % in 67 years (Figure 53, Table. 7). The retreat of the glacier front during this period measured 272.1 metres and averaged 4.1 metres per year (m/a).

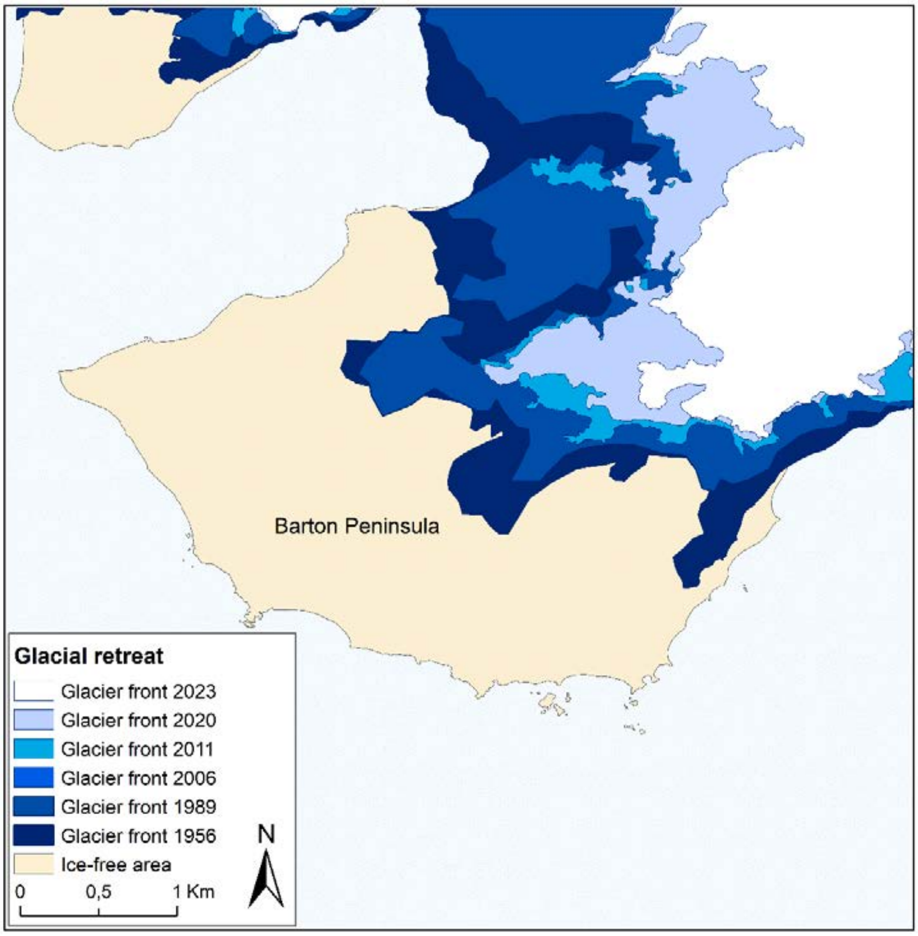
Figure 53: **Glacier retreat areas on Weaver Peninsula between 1956 and 2023, documented by overlaying the respective positions of the glacier fronts**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Barton Peninsula has gained a very large ice-free area with 1.41 km^2 during the studied period (Figure 54, Table 7). Overall, the ice-free area of the Barton Peninsula increased by 3.42 km^2 from 6.23 km^2 in 1956 to 11.06 km^2 in 2023. This corresponds to a growth of approx. 43.67 %. Consequently, this peninsula, experienced the highest total horizontal glacier retreat of all studied areas with 1123.7 metres and 16.8 m/a.

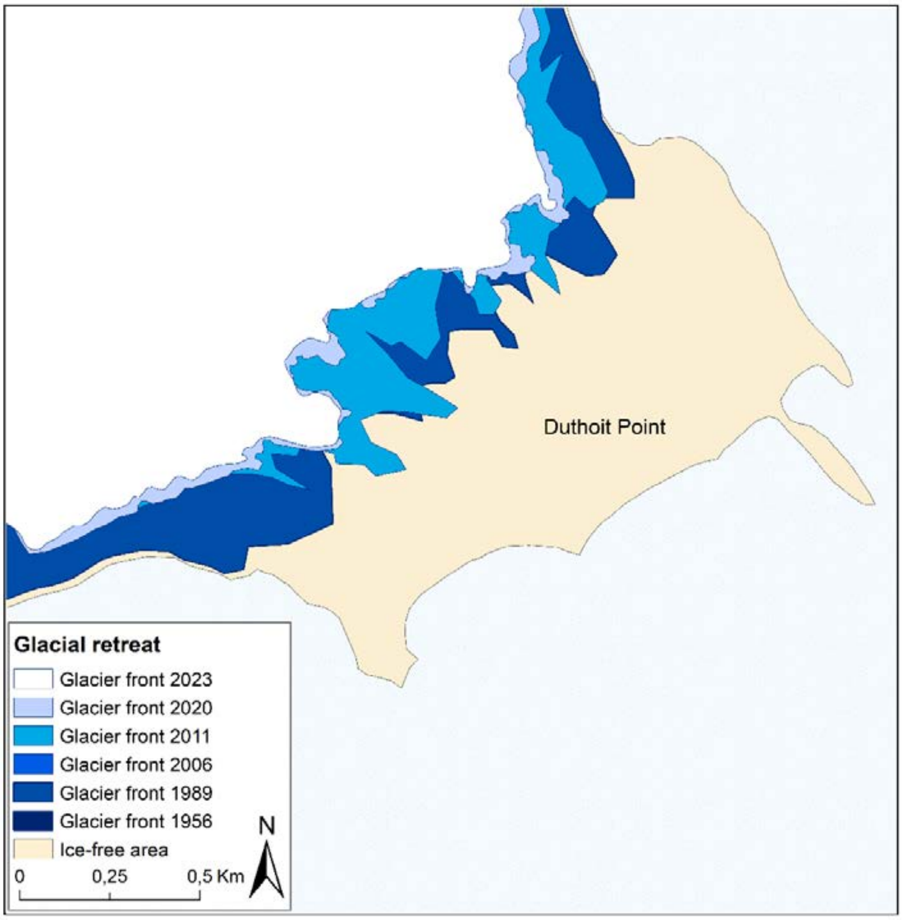
Figure 54: Glacier retreat areas on Barton Peninsula between 1956 and 2023, documented by overlaying the respective positions of the glacier fronts



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Although a significant decrease in glacier area is also recognisable on Duthoit Point, Nelson Island between 2011 and 2021, the decrease between 2021 and 2023 is not very large (Figure 55, Table 7). Due to the uncertain data situation from 1956 for this area, remote sensing data from 1989 was used to calculate the ice-free area. Overall, the ice-free area of 0.99 km^2 increased by 32.37 % to 1.48 km^2 between 1989 and 2021. The retreat of the glacier front with a total of 218.2 m in 34 years at a rate of 4.1 m/a corresponds to the other values in the region.

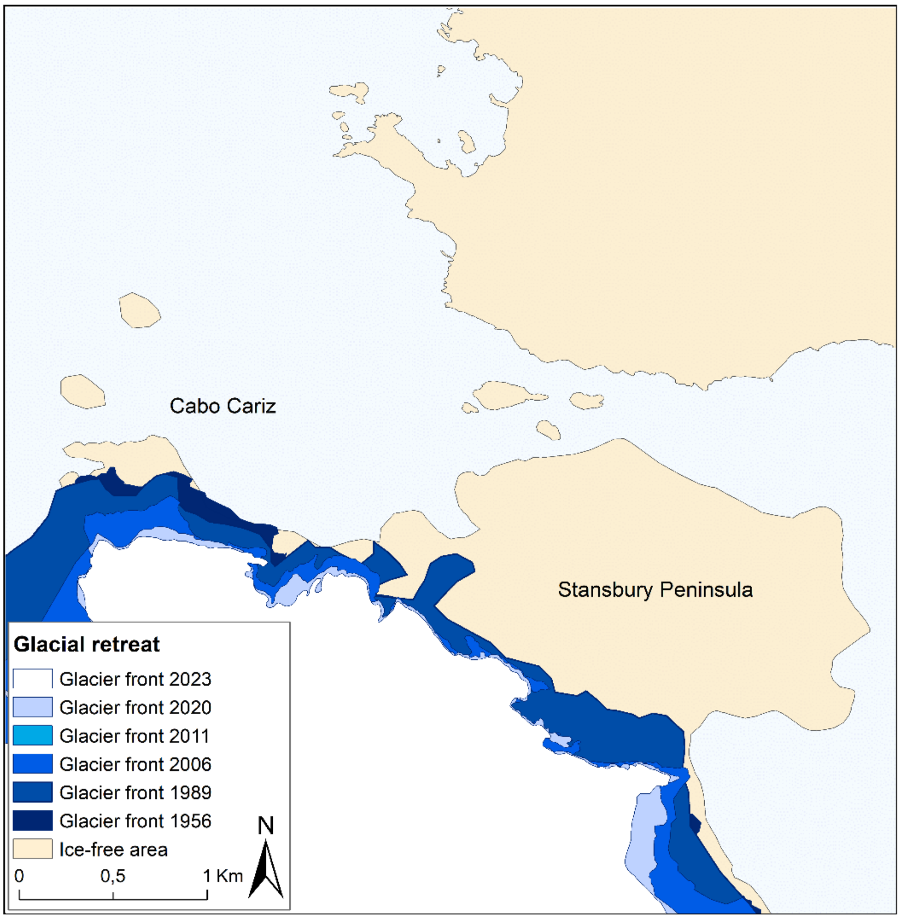
Figure 55: **Glacier retreat areas at Duthoit Point, Nelson Island between 1989 and 2024, documented by overlaying the respective positions of the glacier fronts**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

In contrast to the previously described results, the Stansbury Peninsula, had the lowest values of all analysed glaciers with 5.38 % or 0.17 km^2 increase in ice free area (Figure 56, Table 7). In stark contrat to the study de Oliviera et al. (2021), wo stated a loss of glacial area of 2.11 km^2 between 1988/89 and 2016/18 which is significantly more with an annual change rate of 0.075 km^2/a in comparison to our findings with 0.002 km^2/a.

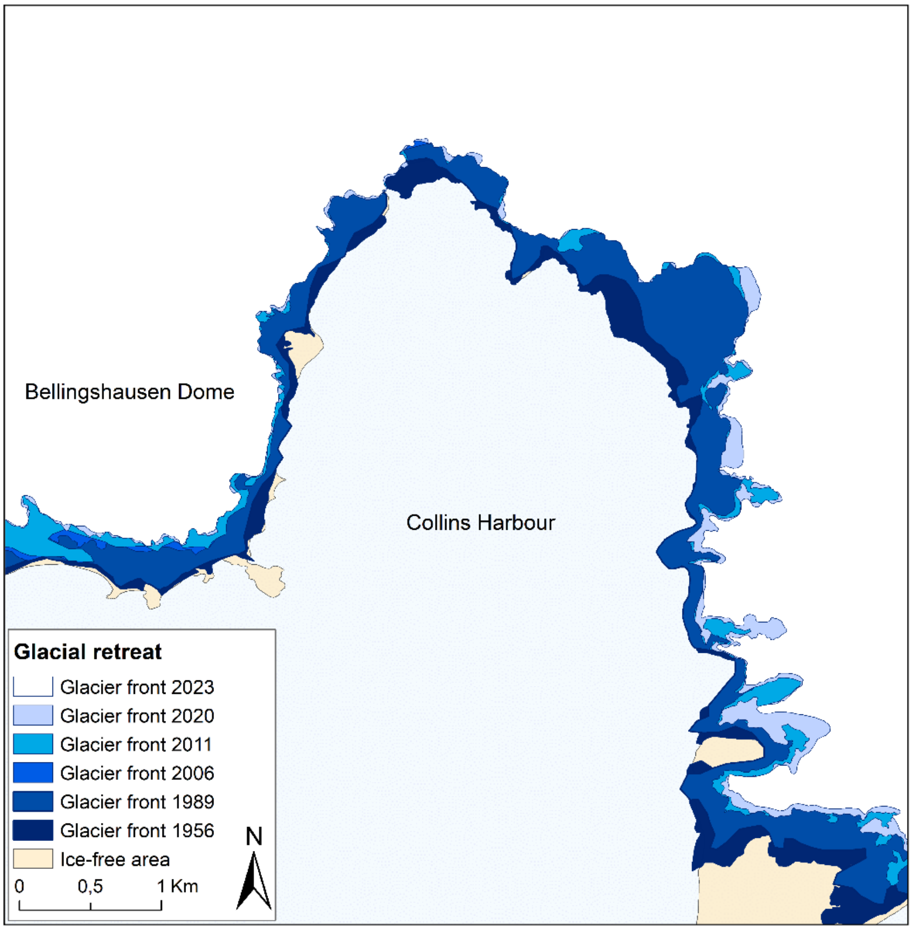
Figure 56: **Glacier retreat areas in the area of Stansbury Peninsula and Cabo Cariz, Nelson Island between 1956 and 2024, documented by overlaying the respective positions of the glacier fronts**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

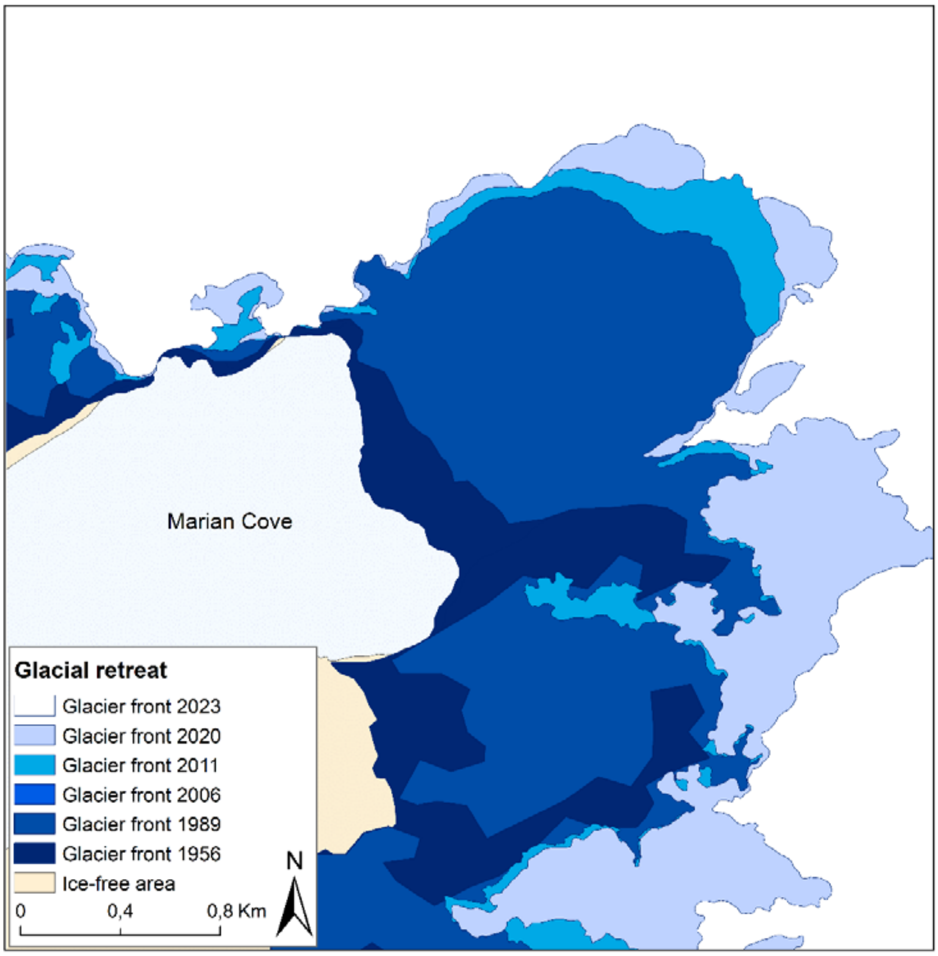
In addition to the land-based glaciers, two ‘tidewater glaciers’ in the prominent bays of Collins Harbour and Marian Cove were investigated and the horizontal glacier retreat there analysed. The glacier retreat from 1956 to 2023 in Collins Harbour of 1.70 km^2 was very high, but distributed over a wider glacier front (Figure 75, Table 7). In Marian Cove, a massive ice area loss of 2.14 km^2 from 1956 to 2023 is evident (Figure 58, Table 7). In particular, the enormous loss of surface area in the past 3 years is worth mentioning.

Figure 57: **Glacier retreat area of Collins Harbour between 1956 and 2023, documented by overlaying the respective positions of the glacier fronts**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Figure 58: **Glacier retreat area of Marian Cove between 1956 and 2023, documented by overlaying the respective positions of the glacier fronts**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Table 7: Overview of the development of the ice-free area (IF) and glacier area (G) in selected areas of Maxwell Bay between 1956, 1989 and 2023

Ice-free area (IF) / glacier area (G)	Area 1956 (km2)	Area 1989 (km2)	Area 2023 (km2)	Difference 1956 – 2023 (km2)	Retreat/ growth	Difference 2020 – 2023 (km2)	Mean annual change rate (km2/a)	Mean glacial retreat (m)	Mean annual glacial retreat rate (m/a)
Bellingshausen Dome (G)	12.29	-	9.1366	-3.1534	-34.51 %	-0.47	-0.042	375.02	5.6
Green Point (IF)	0.06	-	0.2112	0.1512	71.59 %	0.01	0.003	214.8	3.2
Weaver Peninsula (IF)	1.13	-	2.2857	1.1557	50.56 %	0.29	0.013	272.1	4.1
Barton Peninsula (IF)	6.23	-	11.06	4.83	43.67 %	1.41	0.053	1,123.7	16.8
Stansbury Peninsula (IF)	2.98	-	3.1494	0.1694	5.38 %	0.02	0.002	165.9	4.9
Cabo Cariz (IF)	0.11	-	0.2952	0.1852	62.74 %	0.03	0.002	183.5	5.4
Duthoit Point (IF)	-	1.00	1.4787	0.4787	32.37 %	0.06	0.013	218.2	6.4
Collins Harbour (G)	-	-	-	1.7048	-	-	-	-	-
Marian Cove (G)	-	-	-	2.1405	-	-	-	-	-

The results presented here once again emphasise the massive loss of ice mass that has been documented in numerous studies worldwide (e.g. Cook et al., 2005; Davies et al., 2014; Rignot et al., 2014; Baumhoer et al., 2018; Fieber et al., 2018; Hugonnet et al., 2021; Selley et al., 2021), particularly on the Antarctic Peninsula and also on King George Island (e.g. Kejna et al., 1998; Park et al., 1998; Birkenmajer, 2002; Braun & Goßmann, 2002; Rückamp et al., 2011; Osmanoglu et al., 2013; da Rosa et al., 2015; Sobota et al., 2015; Lee et al., 2017b; Lirio et al., 2017; Pudełko et al., 2018; Szilo & Bialik, 2018; da Rosa et al., 2020; Petsch et al., 2020; Pasik et al., 2021). King George Island and its ice fields are influenced by a subpolar and marine climate and are therefore more vulnerable to climate change than continental ice caps (Oerlemans & Fortuin, 1992). In addition to glacier morphology, which significantly influences the rate of glacier frontal retreat, mean annual air temperature, which increased by 1.21 °C between 1969 and 2023 (annual warming of 0.022 °C) during the observation period (see chapter 8.2), is assumed to be the main factor responsible for the observed surface retreat of these glaciers (Dziembowski & Bialik, 2022). As a result, the Bellingshausen Dome has lost about one third of its glacier surface area since 1956 with a rate of 5.6 m/a. If the retreat rates of other glaciers are compared with those of the region, they are generally below or in line with those of the comparison areas (Marr Ice Piedmont, Anvers Island: 7.7 m/year (Cimino et al., 2023), 10 m/year (Bockheim et al., 2013), 8.7 m/year since 1963 (Groff et al., 2023). Only Barton Peninsula, with 16.8 m/year, is well above these reference values. Szilo and Bialik (2018) discussed that glaciers ending in the sea are more sensitive to climate change than glaciers ending on land. This explains why the ice-free areas of the Fildes and Potter Peninsulas are more stable than the tidewater glaciers of Marian Cove and Collins Harbour, as they have a land-based front. The end of the cooling trend between 1995 and 2016 and the current renewed and intensified warming phase are also reflected by the glacier retreat data from the study area. In addition, the positive temperature trends in the austral autumn and winter months (see chapter 8.2) lead to increased precipitation and more frequent thaws (Barrand et al., 2013), which favour the retreat of the glaciers. The consequences of glacial melt have a significant impact on the ecosystem. The addition of freshwater to the ocean as a result of glacial meltwater can affect the composition of the phytoplankton community (Lima et al., 2022a; Jones et al., 2023). The uncovering of new soil can create suitable habitats for native and non-native introduced plants as well as new breeding grounds for seabirds.

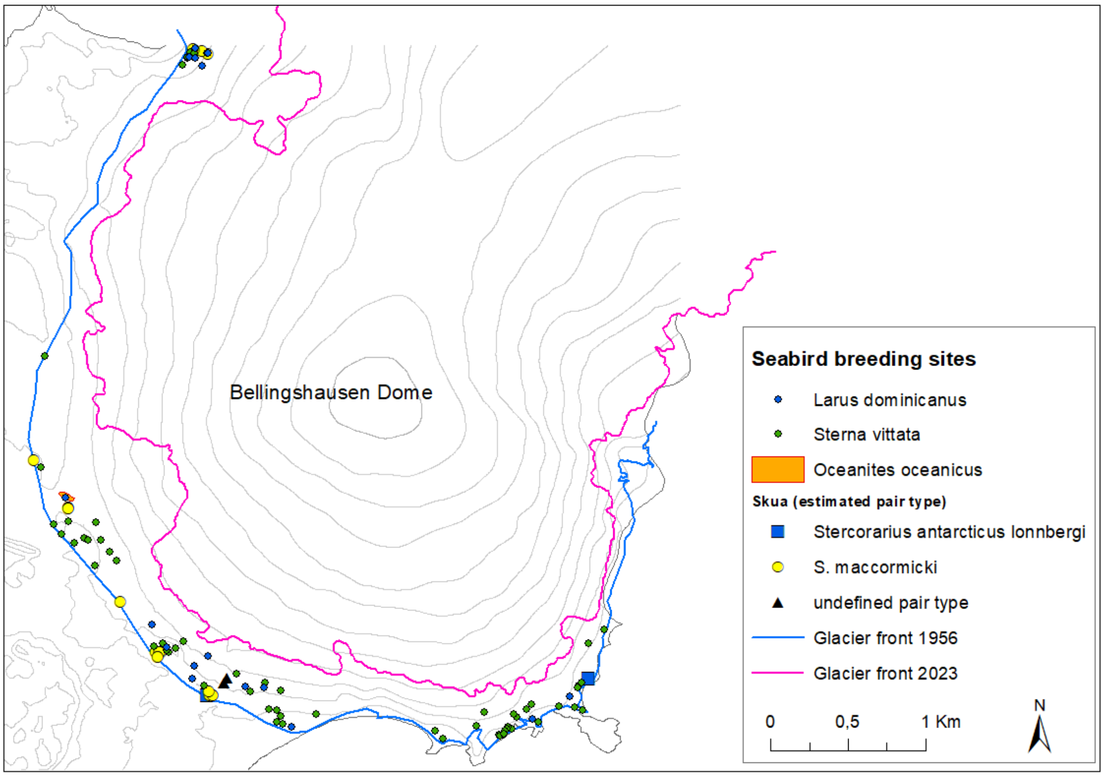
9.3 Recent colonisation of glacier retreat areas by seabirds

Due to the almost complete ice cover of Antarctica, ice-free areas for colonisation by native fauna, flora and microorganisms are very limited. However, as temperatures rise and the resulting recent glacier retreat in large areas of the Antarctic, new ice-free areas are increasingly available for colonisation processes. Corresponding changes can already be observed in the WAP region in particular, which is expected to lead to a merging of habitats and thus a regional homogenisation of biodiversity (Lee et al., 2017a). There are also indications that more ice-dependent species, which appear to be benefiting from new ice-free areas suitable for breeding due to the current climate changes, in particular the Adélie penguin, whose range is shifting further south as a result of the decline in sea ice cover (e.g. LaRue et al., 2013; Clucas et al., 2014).

New ice-free areas in the moraine foreland of a glacier are usually strongly characterised by cryoturbation and waterlogged areas. Accordingly, the colonisation of new ice-free areas by animals, plants and microorganisms requires sufficient protection from meltwater and stabilisation of the soil. Seabirds can serve as vectors for dispersal units or survival forms of plants (e.g. seeds, spores, vegetation fragments) (Parnikoza et al., 2018; Maggio et al., 2022). A detailed description of the ecological significance of glacier retreat areas and the soil and colonisation processes taking place can be found in Braun et al. (2017a).

The systematic breeding bird mapping between 2000/01 and 2023/24 provided comprehensive data on all breeding sites of native seabirds. The documentation of glacier retreat areas in the study area provides detailed information on the new ice-free area in front of the Bellingshausen Dome in the north of the Fildes Peninsula (see chapter 9.2). Based on this data, all recorded breeding sites of seabirds in this area, which covers a total of 3.15 km2, can be visualised (Figure 59). It could be shown here that the number of breeding sites has increased further compared to the results shown previously (Braun et al., 2020a; Braun et al., 2022). While a total of 93 breeding sites of four seabird species were recorded in the 2017/18 season, this figure rose to 106 in the 2019/20 season and 112 in the 2023/24 season. Of these, 16 breeding sites were recently accounted for by kelp gulls, 69 by Antarctic terns, 26 by skuas and one breeding site by Wilson's storm-petrels with at least ten breeding pairs (Figure 59). These included several larger colonies of up to 20 breeding pairs of Antarctic terns and up to 30 breeding pairs of kelp gulls. The updated results shown here underline once again that seabirds colonised new ice-free areas in the study area within a few decades and that the number of breeding sites increased further within a few years (Braun et al., 2020a; Braun et al., 2022). With regard to the spatial distribution of the recorded breeding sites in the new ice-free area, it is clear that they are mainly concentrated in the south-west, south-east and north-west and thus in areas at a greater distance from the glacier front. It can be assumed that the conditions for the colonisation of breeding birds, such as a suitable microtopography and easy availability of nesting material, appear to be fulfilled here (Peter et al., 1990; Quintana & Travaini, 2000; Costa & Alves, 2010; Suárez et al., 2010; Parnikoza et al., 2018).

Figure 59: Breeding sites of native seabirds in the area of the glacier retreat in the north of the Fildes Peninsula in the seasons 2000/01 and 2003/04 to 2023/24



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

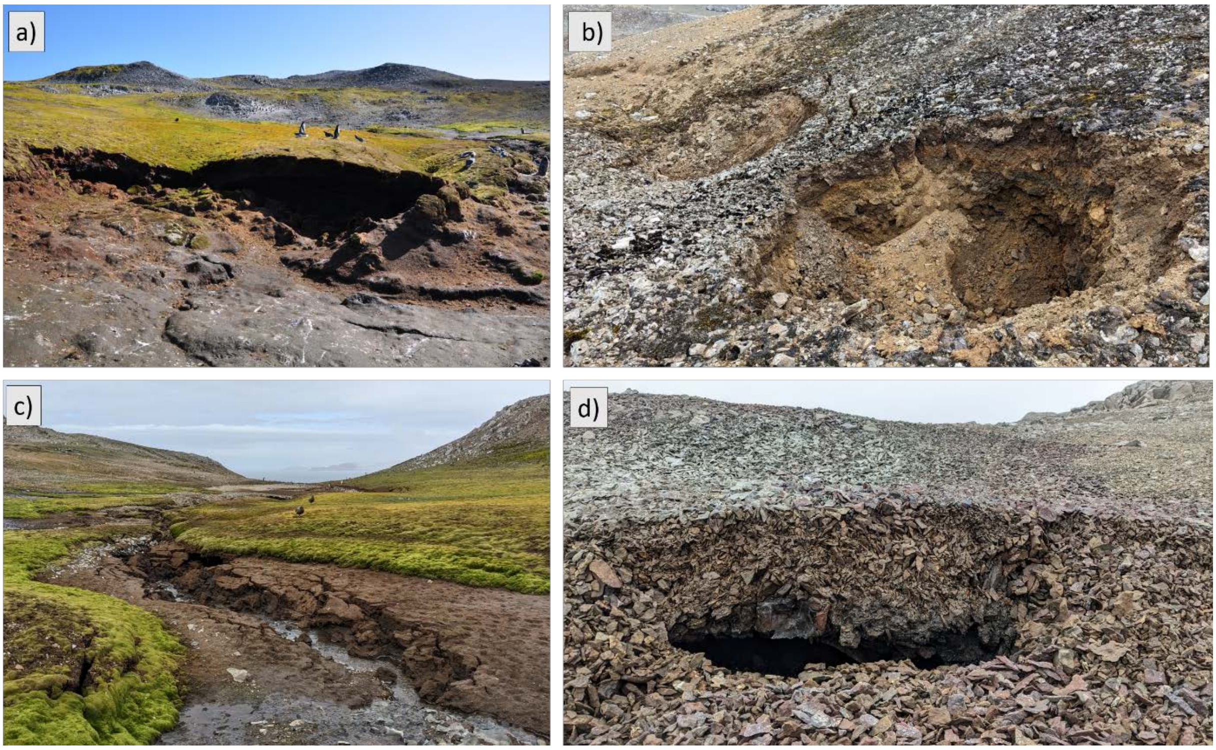
As the results shown here demonstrate the suitability of this area for colonisation by seabirds in the new ice-free area created by the current glacier retreat, a further expansion of possible breeding habitats can be expected as the glacier edge continues to recede. The extent to which this is likely to lead to an increase in the total number of breeding pairs on the Fildes Peninsula remains unclear. A reduction in the competitive pressure for suitable breeding sites is also possible, as all suitable breeding sites are often occupied in breeding seasons with very favourable conditions due to a very high density of breeding birds. This applies in particular to skuas and kelp gulls, which generally return to their traditional breeding sites or can migrate from other areas. Successful colonisation of new, ice-free areas by these species has also been documented in other areas (Costa & Alves, 2012; Lee et al., 2017b).

10 Documentation of permafrost degradation in the Fildes Region

As a result of the global temperature increase, increasing warming and also degradation of the permafrost (year-round frozen ground) is being recorded in the Arctic and mountain areas (e.g. Schaefer et al., 2012; Oliva & Fritz, 2018; Biskaborn et al., 2019). As the melted water escapes when the permafrost thaws, this is an irreversible process (Turner et al., 2007). Among other things, this can lead to instability of the subsurface and thus to considerable damage to infrastructure and accelerate the current global warming due to escaping greenhouse gas emissions (e.g. Walter et al., 2006; Zimov et al., 2006; Melvin et al., 2017; Voigt et al., 2017; Oliva & Fritz, 2018). For the Antarctic, relatively little information is available to date on the effects of global warming on permafrost (Guglielmin & Cannone, 2012; Bockheim et al., 2013; Turner et al., 2014).

For the first time, clearly visible degradation of the permafrost in terms of thaw slumps was recorded at several locations in the Fildes Region in the 2022/23 season (Figure 60). These locations ranged from Ardley Island, the Fildes Strait in the far south of the Fildes Peninsula, to Davies Heights in the central part of the Fildes Peninsula. These observations are attributed to the unusually high temperatures in the 2022/23 season (see chapter 8.2).

Figure 60: Permafrost degradation in the Fildes Region in the season 2022/23: a & c – Ardley Island, b – Fildes Strait, d – Davies Heights



Photos: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

11 Input of marine debris into the study area

11.1 Introduction

In recent years, awareness of the risks of plastics entering the environment has increased significantly, as they can have a negative impact on the environment due to the widespread use of plastics of all kinds, their flexibility and durability as well as the pollutants they contain or adhere to (e.g. Derraik, 2002; UNEP, 2005; Hammer et al., 2012; Avio et al., 2017; MacLeod et al., 2021; Bargagli & Rota, 2022). Accordingly, plastic waste in the oceans and microplastics have been defined as one of the main global environmental problems (UNEP, 2016). Marine litter or debris is any waste generated or processed by humans - regardless of its size - that ends up in the coastal or marine environment. Larger objects (> 25 mm) make up the majority of the plastic floating in the sea (Kaandorp et al., 2023). Microplastics are defined as solid and insoluble plastic particles smaller than 5 mm in size and are categorised as primary or secondary microplastics. Primary microplastics include all particles < 5 mm plastic particles that are used in medical, hygiene or cosmetic products, or industrial granule that is used as a base material for the manufacture of various plastic products. Secondary microplastics, on the other hand, are created through physical, biological and chemical degradation of larger plastic objects or through abrasion in the technosphere (source: https://www.umweltbundesamt.de/service/uba-fragen/was-ist-mikroplastik, accessed 29.07.2024; Andrady, 2022). The majority of microplastics entering the oceans globally originate from land-based activities, while only 2 % are caused by activities at sea (Boucher & Friot, 2017).

In the wake of increased attention to global marine plastic pollution and the drastic threat it poses to the environment, the number of scientific studies on the subject is growing (De-la-Torre et al., 2023; Hamilton et al., 2023). The Alfred Wegener Institute's LITTERBASE platform provides a comprehensive overview of publications on the quantity, distribution and composition of litter/debris in the sea and other bodies of water, as well as the impact of this debris on organisms (Tekman et al., 2024). The growing scientific interest is also reflected in various studies on the distribution of marine debris in the Antarctic environment (e.g. Convey et al., 2002; Barnes et al., 2010; Waller et al., 2017; Anfuso et al., 2020; Waluda et al., 2020; Albarrán et al., 2024). Contrary to the predominantly terrestrial global input of marine debris, fishing, research, supply or cruise ships as well as flotsam from global sources are primarily responsible for the input of marine debris into the Antarctic, as land-based sources of marine debris in the Antarctic play a subordinate role due to the comparatively very low level of human activity on land (research stations, tourism) (do Sul et al., 2011; Waller et al., 2017).

Overall, there are many documented findings of (micro)plastics in the Antarctic, e.g. in the seawater of the Southern Ocean (Cincinelli et al., 2017; Isobe et al., 2017; Lacerda et al., 2019; Suaria et al., 2020; Alurralde et al., 2022), in the deep water and sediment of the Bransfield Strait (De-la-Torre et al., 2023), in the sea ice of East Antarctica (Kelly et al., 2020), on the coasts (Gregory et al., 1984; Convey et al., 2002; Sander et al., 2009; do Sul et al., 2011; Anfuso et al., 2020; Waluda et al., 2020; Albarrán et al., 2024), in an Antarctic freshwater lake (González-Pleiter et al., 2020), in station wastewater (Kim et al., 2023a; Bernard et al., 2024), in sediment samples (Munari et al., 2017; Waller et al., 2017; Reed et al., 2018; Miranda et al., 2023) and in faecal samples from penguins (Bessa et al., 2019; Fragão et al., 2021) and seals (Bravo Rebolledo & van Franeker, 2015; Cebuhar et al., 2024) as well as in the stomach contents of penguins (Braun et al., 2022; Wawrzynek-Borejko et al., 2022; Kim et al., 2023b). In addition, fragments of polystyrene have been detected in Antarctic springtails (Collembola), pelagic amphipods and various benthic invertebrates (Bergami et al., 2020; Jones-Williams et al., 2020; Sfriso et al., 2020). However, many studies differ in terms of their sampling and analytical procedures, which means that sufficient comparability of the results is not always guaranteed (Browne et al., 2015; De-la-Torre et al., 2023).

German scientists have already carried out studies on marine debris on the coast of the Fildes Peninsula in the past, with the first survey dating back to December 1989 (H. Grimm, unpubl. data). A further survey of waste in the region was carried out in the 2003/04 and 2004/05 seasons, but included the mainland as well as the coastal areas (Peter et al., 2008). The current systematic survey with the aim of a long-term study was initiated in a bay on the west coast of the Fildes Peninsula in the 2018/19 season (Braun et al., 2022). Therefore, a bay with a coastline length of 440 m and a total area of 53,000 km^2 was selected (bay no. 12, see Figure 78 in Appendix B), which is bordered by a cliff on the inland side. This means that objects that are spread by the prevailing strong westerly winds are caught by the cliff and can also be recorded (Braun et al., 2022). In addition, the sandy ground of this bay also allows the detection of smaller objects.

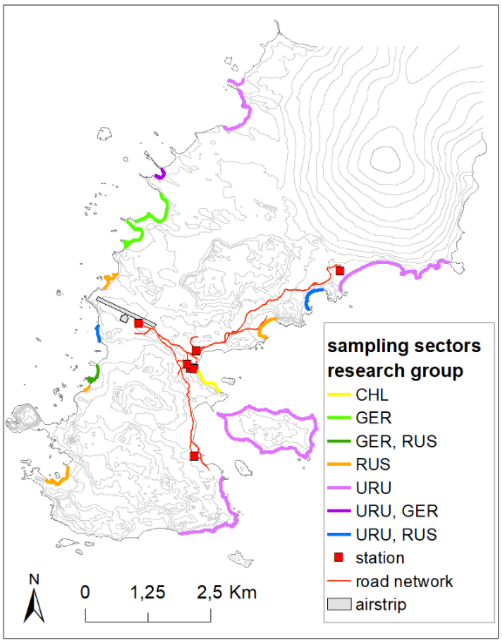
11.2 Coordination with other research groups in the study area

In view of the growing awareness of the effects of marine debris in the Antarctic, scientific studies and clean-up operations are also increasingly being carried out in the Fildes Region. Therefore, knowledge of relevant activities and coordination is of great importance to obtain high-quality scientific results on the occurrence of marine debris in the study area.

With regard to the extent and impact of marine debris, especially that caused by fishing vessels, a long-term monitoring programme was initiated by CCAMLR in 1989. A standardised methodology was developed for this purpose, which was introduced in 1993 (‘Standard Method for Surveys of Beached Marine Debris’, CCAMLR, 2001) and has been revised in 2022. In the past, Ardley Island was also listed among the CCAMLR survey areas (CCAMLR, 2001), although it is still unclear who carried out the surveys.

Several different groups of scientists have been or are currently working on marine debris in the study area, including from Uruguay, Russia and Chile. It is known, that, station personnel and scientists from the Uruguayan Artigas station carried out marine debris monitoring in accordance with the CCAMLR standard at three coastal sections in the north of the Fildes Peninsula once a year from 2000 until at least 2012 (Uruguay, 2006, 2010, 2012). In addition, between the 2015/16 and 2023/24 seasons, Uruguayan scientists collected washed-up marine debris and other litter from other stretches of coast as part of scientific studies (Lozoya et al., 2018; pers. comm. F. Teixeira de Mello & J. P. Lozoya). We also have knowledge of collections of marine debris in the 2021/22 and 2023/24 seasons by Russian scientists (Vesman & Ershova, 2022; pers. comm. P. Chukmasov). Due to a mostly successful local coordination, an overlapping of research activities could be largely avoided. An overview of the spatial distribution of scientific surveys of marine debris and clean-up operations carried out by different research groups or other teams is shown in Figure 61. Continuous coordination of activities is essential, as further marine debris research is planned in the study area, in particular on Ardley Island (Republic of Korea & Chile, 2022a).

Figure 61: Spatial distribution of scientific surveys of marine debris and clean-up operations



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

11.3 Marine debris survey on the coast of the Fildes Region

With the continuation of the systematic survey, five years of data on marine debris in a selected bay on the west coast of the Fildes Peninsula are now available, thus substantiating the initial results published in Braun et al. (2022).

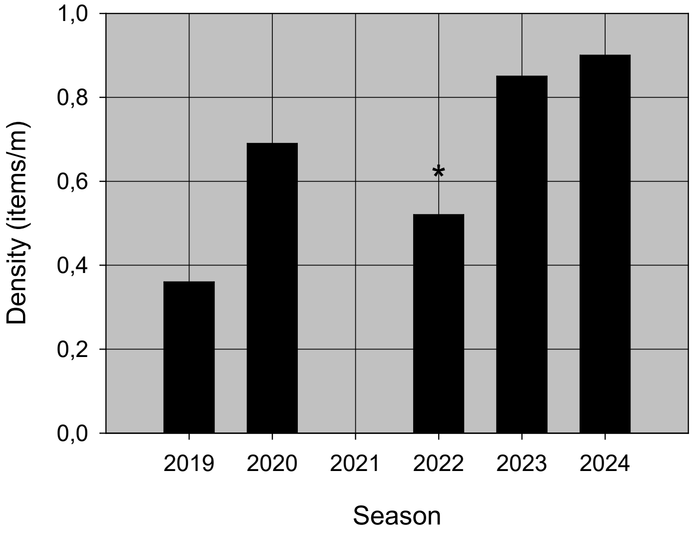
In the 2021/22 season, a total of 222 objects were collected from the approx. 450 m long beach section and the adjacent bay area, although it should be noted that a beach debris collection had been carried out shortly beforehand by other scientists (Vesman & Ershova, 2022), of which no information was available. This value is therefore a minimum figure. In the following seasons 2022/23 and 2023/24, a total of 370 and 396 objects of flotsam were collected, respectively. In the 2023/34 season, the flotsam was also collected one month earlier (pers. comm. P. Chukmasov), whereby the data of the objects were provided and included in the results, but without indicating the weight of the objects. An overview of the objects found can be found in Figure 62.

Figure 62: Items collected during the marine debris monitoring in the 2021/22 to 2023/24 seasons

Photos: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

The recorded densities of 0.85 and 0.90 objects/m in the 2023/23 and 2023/24 seasons were the highest since systematic recording began in 2019 (Figure 63). The mean value for the annual accumulation rate is 0.81 objects/m*year (determined from the 2019/20, 2022/23 and 2023/24 seasons). A comparison with the data from the 2018/19 season onwards indicates an increasing trend, although a statistical analysis is not yet possible due to the data structure.

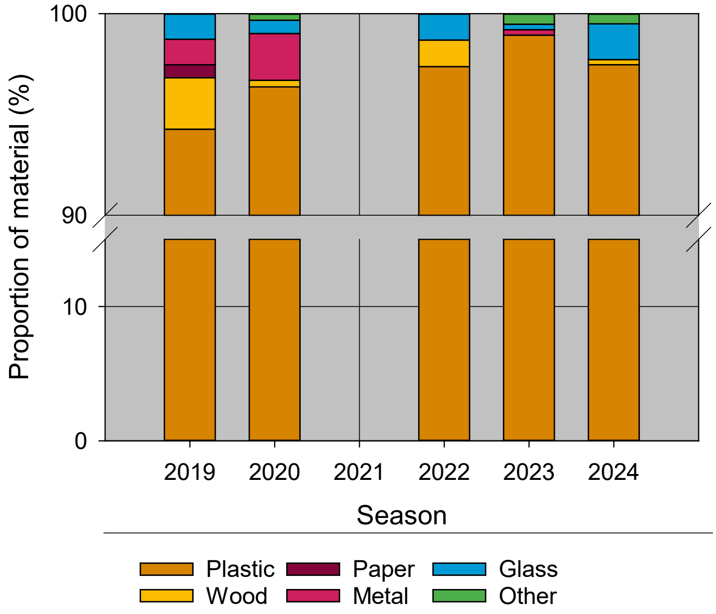
Figure 63: Density of anthropogenic elements per meter, collected during the marine debris survey between 2019 and 2024 (* limited accuracy due to prior external sampling)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

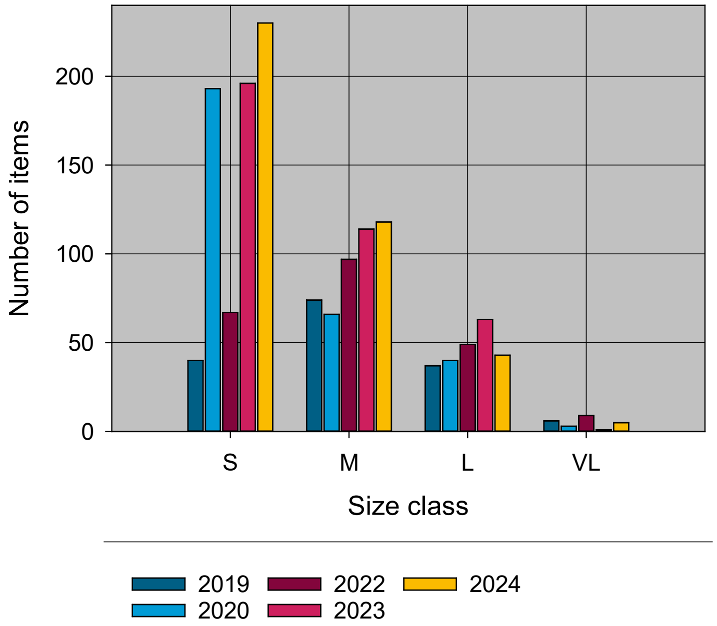
Throughout the study period the proportion of items made of plastic clearly outweighs those made of wood, glass, metal or other materials at around 98 % (Figure 64). Overall, small items dominated the majority of collections, followed by medium-sized and large properties (Figure 65).

Figure 64: Proportion of different materials collected during the marine debris survey between 2019 and 2024 (limited accuracy for 2022 due to prior external sampling)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Figure 65: **Number of items of the different size classes collected during the marine debris survey between 2019 and 2024, classification of size classes: S = small (< 2.5 cm), M = medium (2.5 cm - 10 cm), L = large (> 10 cm - 1 m), VL = very large (> 1 m) (Informative value for 2022 limited by prior external sampling)**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

During the study period, a significant amount of styrofoam and other foam material (42 %) was found at the sampled beach section (Table 8), although the dumping of styrofoam in the Antarctic is prohibited under the legally binding regulations of the Madrid Protocol to the Antarctic Treaty (Annex III, Article 7) and the SCAR Code of Conduct (SCAR, 2018). In addition to washed-up marine debris, styrofoam also enters the environment via the resident stations of the Fildes Peninsula, where it continues to be used as packaging or insulating material (Peter et al., 2008; Peter et al., 2013; Braun et al., 2022). However, a clear distinction of the source of the entry of the styrofoam was not possible. Furthermore, a clear link to fishing activities (e.g., nets, plastic ropes or strings, floats, packing tapes, etc.) was found in an average of 11 % of the finds (Table 8). Burnt plastic/pyroplastics was recorded throughout the study period (Table 8) and probably originates from dumping of waste incineration residues on board ships (Braun et al., 2022).

Plastic bottles, their closures or parts thereof again made up a significant proportion of marine debris finds collected during the entire survey period (Table 8). It was noticeable that there were numerous smaller water bottles of the same type and manufacturer of Chinese origin. It should be noted that a special type of these bottles is used in the local Chinese station, so that it cannot be completely ruled out that these bottles originate from there. Further inscriptions on bottles and other objects indicate South American, Korean and Polish origin.

During the entire period of the current systematic survey of marine debris, particles from industrial plastic pellets or granules were found in all surveys with the exception of the 2018/19 season (Table 8; Braun et al., 2022).

Table 8: **Number of findings in different categories (* Informative value for 2022 limited by prior external sampling)**

Item category	2019	2020	2022*	2023	2024	average
fishing gear (nets, floats, strings, buoys)	26	34	31	43	30	12.1 %
burnt plastic/pyroplastics	5	39	4	6	18	4.8 %
plastic bottles, cups	21	6	29	32	42	9.5 %
Styrofoam, polyurethane foam	49	111	90	183	146	38.7 %
industrial plastic pellets/granules	0	1	1	1	3	0.4 %
total number of items	148	291	222	370	386	n.a.

Marine debris surveys have already been carried out on the coast of the Fildes Peninsula in the past. To the best of our knowledge, the first ever survey in the study area dates back to December 1989, when a total of 25 objects were found in the bay no. 12. Already here, plastic made up the majority of the finds with 84 %, followed by glass (12 %) and metal (4 %) (H. Grimm, unpubl. data). Unfortunately, no further information is available on the methodology used at that time. Nevertheless, compared to the current data, the significantly lower number of marine debris washed up on these coasts several decades ago is evident.

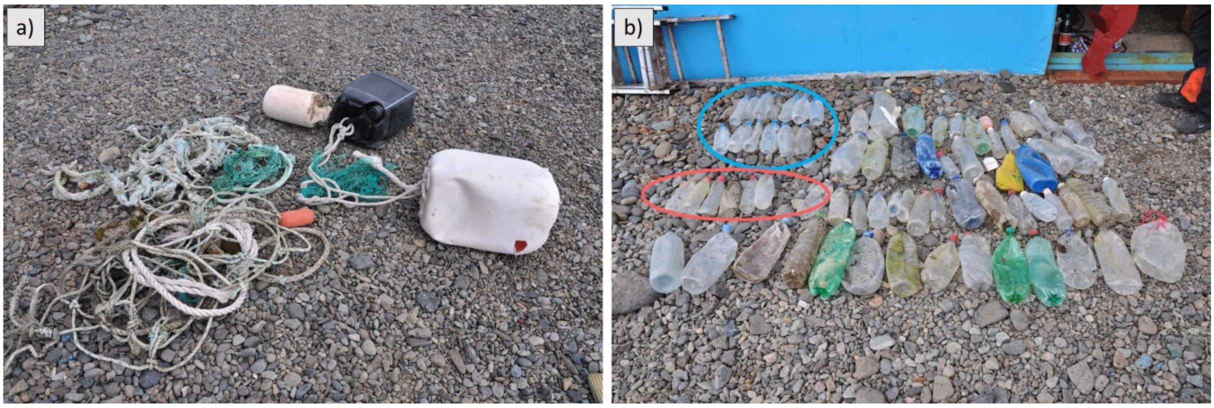
A further survey of human waste was carried out in the 2004/05 season, whereby all objects of at least palm size were recorded (Peter et al., 2008). In contrast to the current study, smaller objects were not included. But here too, the proportion of plastic was highest at 89 % in the bay no. 12, followed by glass and metal (6 %). In contrast to the current marine debris collections, however, only four plastic bottles or remnants of them were found.

Meanwhile, the current marine debris survey on the west coast of the Fildes Peninsula provides valuable information on the type and amount of anthropogenic material drifting in the Southern Ocean and washing up on the coasts of Antarctica in the South Shetland Islands area that can be used for cross-regional comparisons. The annual accumulation rates between 0.69 and 0.90 objects/m*year were relatively high and mostly higher than from other areas on the South Shetland Islands, the Antarctic Peninsula and Subantarctic Islands (e.g. Convey et al., 2002; Otley & Ingham, 2003; Eriksson et al., 2013; Almela & González, 2020; Waluda et al., 2020; Finger et al., 2021). Debris density values between 0 and 1.03 objects/m were determined from the Admiralty area, with the highest value being reached in the immediate vicinity of the station (Anfuso et al., 2020). It is assumed that the accumulation rate decreases with increasing latitude, partly due to successful measures within the Antarctic Treaty region or is related to local conditions (Waluda et al., 2020). The latter seems to play a significant role in the relatively high accumulation rate in the case of the west coast of the Fildes Peninsula. Due to the local topography, the north-western exposure to the Drake Passage as well as the prevailing westerly winds there is a considerable input of marine debris washed up from the Southern Ocean at the beach section investigated (Braun et al., 2022). Local currents and tides also have an impact, even on a small scale (Waluda et al., 2020). In contrast, the amount of marine debris on the east side of the Fildes Peninsula towards Maxwell Bay is significantly lower (Vesman & Ershova, 2022). It can be assumed that the input of marine debris takes place via the resident station and via intensive shipping traffic in this area.

11.4 Other findings along the coastline

In addition to the annual systematic survey of marine debris, during an occasional clean-up operation in February 2023 at the western coast of the Fildes Peninsula, numerous fishing-related objects and plastic bottles were collected from other beach areas (Figure 66 a). Again, the large number of identical bottles was striking, underlining their apparently very large number in the sea (Figure 6 b).

Figure 66: Fishery-related items (a) and plastic bottles (b) collected at the north-western coast of the Fildes Peninsula in February 2023; bottles of the same type are marked with coloured circles



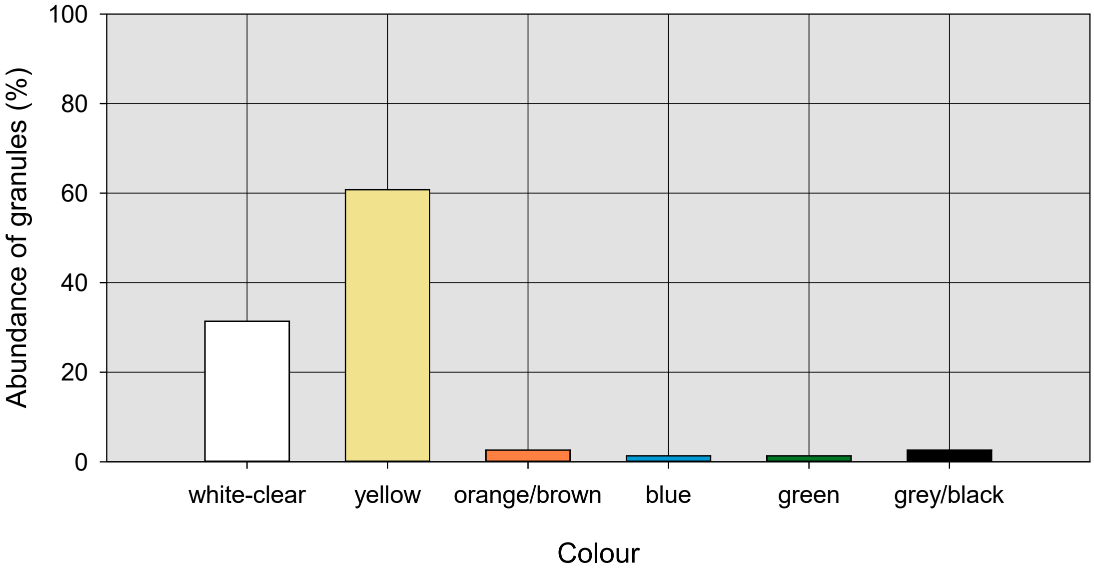
Photos: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Furthermore, attention was also paid to other notable finds along the coast of the Fildes Region. These included plastic granules used as industrial raw material, which had already been detected in previous years (Braun et al., 2022). During repeated coastal surveys in the 2022/23 and 2023/24 seasons, numerous industrial plastic granules were again found in several beach areas, where the topography and sandy subsoil allowed the accumulation and detection of smaller material. Thus, there is now evidence of industrial plastic granules for the bays 2, 3, 8, 10, 12 and 16 (for numbering of the bays see map in Figure 78 Appendix B), all located at the western coast of the Fildes Peninsula. The east coast of the Fildes Peninsula is characterised by gravel and grit beaches, which may mask the possible presence of such granules. The largest quantity of plastic granules was recorded with 77 plastic granules, which were found during a 15-minute search on a 15 m long section of the drift line of bay no. 2 in February 2024 (Figure 67). If all granules found in the 2019/20, 2022/23 and 2023/24 seasons are combined, a total of 153 particles were found. The average size of the particles was 4.2 ± 2.5 mm x 3.5 ± 0.7 mm with an average weight of 17.2 ± 21.0 mg. The predominant colour was yellow, beige or yellowish, followed by white or transparent (Figure 67 & 68). Darker colours were only rarely present. It is not known to what extent there is a bias in the collection accuracy of the particles or whether the particles show signs of weathering and colour change if they remain in the sea for a longer period of time.

Figure 67: **Industrial plastic granules found on a 15 m long section of the drift line of the western coast of the Fildes Peninsula**

Photo: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Figure 68: **Distribution of the colours of the industrial granules found on the west coast of the Fildes Peninsula in the 2019/20, 2022/23 and 2023/24 seasons (modified after Verlis et al., 2013)**

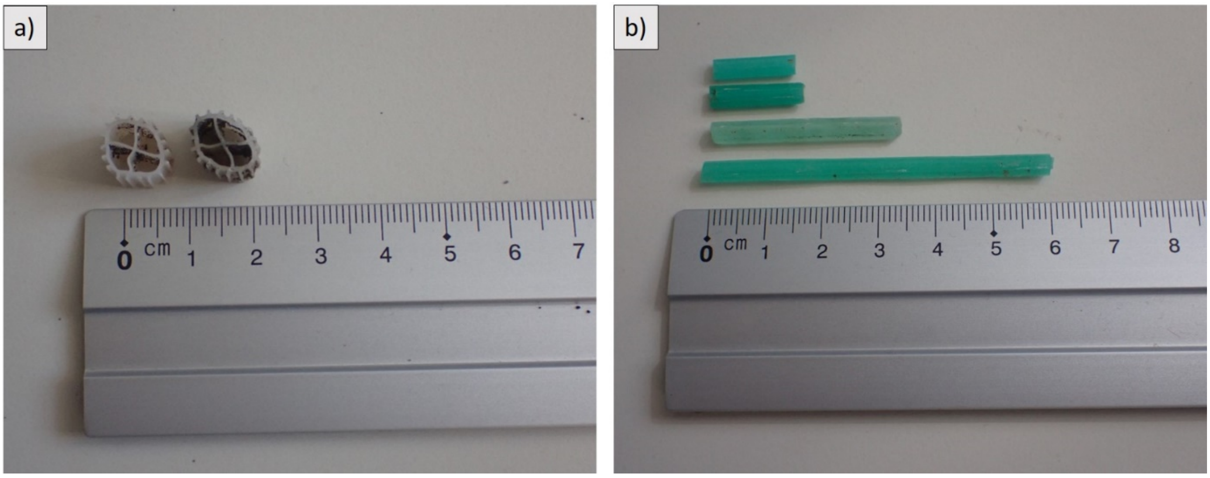


Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

After an initial discovery of three conspicuous cogwheel-like plastic item in the season 2022/23, which were identified as microfilters from the biological cleaning section of sewage water treatment plants, several of these objects were found again in the 2023/24 season in the bay no. 2 and 12 (Figure 69 a). These microfilters were first detected in 2007 from the Atlantic coast of northern France, Spain and Portugal and were subsequently also found on the Mediterranean coast of Morocco, Algeria and Tunisia and on the US and Canadian East Coast. Since the repeated finds of these microfilters in the Antarctic, far away from other sites, represent a peculiarity, a publication (short note) on these finds in a specialist journal is in preparation.

Other conspicuous objects found in the 2018/19, 2021/22 and 2023/24 seasons were green plastic rods with a hexagonal cross-section of 3-4 mm and a maximum length of 10 cm (Figure 69 b). However, the origin and function of these objects is not known.

Figure 69: Microfilters (a) and plastic stick (b), found on the west coast of the Fildes Peninsula



Photos: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

In addition, in the 2022/23 and 2023/24 seasons, large quantities of vegetable waste were found on the north and northwest coast of Ardley Island and thus within ASPA No. 150, presumably from the large number of ships and yachts that are always present on site. Animals entangled in marine debris were not recorded in the 2021/22 to 2023/24 seasons.

11.5 Discussion and conclusions

The results of the marine debris survey presented here show a considerable input of marine debris, mainly plastic, to the coast of the Fildes Peninsula. Even though the Fildes Peninsula is a place of intense and diverse human activity, an external source can be assumed due to the location of the investigated bay and the surrounding area, the activities taking place there and the prevailing winds and currents. Only in exceptional cases can a local source from local logistical, scientific or tourist activities be assumed. This once again proves that the global distribution of marine plastic also extends to the Southern Ocean and penetrates this area of the Antarctic in significant quantities.

The quantity and density of the objects found in the present study is in some cases significantly higher than the values from other areas of Antarctica, the Subantarctic and other Atlantic islands (e.g. Convey et al., 2002; Otley & Ingham, 2003; Barnes & Milner, 2005; Sander et al., 2009; do Sul et al., 2011; Eriksson et al., 2013; Monteiro et al., 2018; Almela & González, 2020; Anfuso et al., 2020; Waluda et al., 2020; Albarrán et al., 2024) and from Svalbard (Meyer et al., 2023). Only in the immediate vicinity of a field hut at Harmony Point, Nelson Island, was the density of objects found higher, while significantly less anthropogenic material was found on the coast (Finger et al., 2021). In line with the high density of marine debris detected here on the west coast of the Fildes Peninsula, Lozoya et al. (2022) emphasise the alarmingly high amount of marine debris in another surveyed area in the north of the Fildes Peninsula, which corresponds more to values from urban areas.

The predominance of plastic of various types and sizes over other materials in marine debris found on the Fildes Peninsula is consistent with other studies from other areas of the Antarctic (e.g. Convey et al., 2002; Anfuso et al., 2020; Waluda et al., 2020; Finger et al., 2021; USA, 2022; Vesman & Ershova, 2022; Albarrán et al., 2024). Another study conducted in a bay in the north of the Fildes confirmed the presence of numerous plastic pellets/granules or primary microplastics for the first time (Lozoya et al., 2022). Here, too, the quantity found corresponded to the values of urban beaches in Uruguay (Lozoya et al., 2016) or Malta (Turner & Holmes, 2011). The occurrence of this material, which can only come from external sources, emphasises the possibility of transport over long distances to areas at higher latitudes. One of the first reports from the South Atlantic and New Zealand dates back to the 1970s and 1980s (Gregory, 1977; Morris, 1980; Bourne & Imber, 1982). Further evidence for the input of this material is provided by several bags found for packaging plastic granulate or pellets of the LDPE or LLPDE type (Braun et al., 2022). A large proportion of the marine debris in the Antarctic, as also shown in the present study, is packaging and objects from the fishing industry, such as nets, ropes or buoys (Peter et al., 2008; Bravo Rebolledo & van Franeker, 2015; Braun et al., 2022). It is not known how these objects end up in the sea. However, the frequent findings of burnt plastic, which presumably comes from waste incineration plants, at the very least point to the deliberate dumping of waste into the sea. This would contradict Annex V of the International Convention for the Prevention of Pollution from Ships (MARPOL 73/78), which came into force in 2013 and prohibits the deliberate dumping of plastic and other waste from ships. This burnt plastics/pyroplastics has also been described from other areas, including the Antarctic (Valdivia et al., 2014). As this material is often very similar to stones, it may often not be recorded in marine debris surveys and often contains high concentrations of pollutants (Turner et al., 2019; Utami et al., 2023). Findings of vegetable waste from ships have also been documented for Livingston Island (USA, 2022).

The occurrence of marine debris in the Antarctic refutes the long-standing hypothesis that the Antarctic Convergence represents an effective circumpolar biogeographical barrier, as north-south dispersal of organisms drifting in the sea, e.g. seaweed, is possible and has been demonstrated several times (Fraser et al., 2017; Fraser et al., 2018; Avila et al., 2020; Dawson et al., 2024).

Although a large proportion of marine debris sinks to the seabed over time (Woodall et al., 2014; Canals et al., 2021; Rowlands et al., 2023), plastic drifting in the sea reaches areas that have so far been relatively untouched by humans. Negative effects of plastic pollution on animals include the entanglement of marine mammals, turtles and seabirds in plastic nets or snares (e.g. Croxall et al., 1990; Arnould & Croxall, 1995; Gregory, 2009; Phillips et al., 2010; Votier et al., 2011; Bravo Rebolledo & van Franeker, 2015; Duncan et al., 2017; Franco-Trecu et al., 2017). Further risks are posed by the ingestion of plastic particles and the resulting negative effects (e.g. Moser & Lee, 1992; Pierce et al., 2004; Gregory, 2009; Brandão et al., 2011; Kühn & van Franeker, 2012; Schuyler et al., 2012; Codina-García et al., 2013; Cole et al., 2013; de Stephanis et al., 2013; Bond et al., 2014; Cousin et al., 2015; Gilbert et al., 2016; Denuncio et al., 2017; van Franeker et al., 2018; Fackelmann et al., 2023; Zimmer-Correa et al., 2024). In addition, contained or adhering chemicals can harm organisms (e.g. Rios et al., 2007; Teuten et al., 2009; Colabuono et al., 2010; Heskett et al., 2012; Tanaka et al., 2013, 2015; Rochman et al., 2019; Tanaka et al., 2019; Yamashita et al., 2021; Charlton-Howard et al., 2023; Menezes et al., 2024). Microplastics ingested by animals pose a further threat to marine life and coastal inhabitants. It has already been detected in a large number of animals, including deep-sea inhabitants (e.g. Cole et al., 2013;

Besseling et al., 2015; Lusher et al., 2017; Bessa et al., 2019; Jamieson et al., 2019; Bourdages et al., 2020; Carlin et al., 2020). Microplastics have also already been detected in ice algae, which play a key role in benthic food webs (Bergmann et al., 2023). A special feature is the fact that susceptible marine organisms are increasingly encouraged to ingest plastic through the evaporation of dimethyl sulphide (DMS), which is vaporised by plastic that has been in the sea for a long time (Savoca et al., 2016), as this odour component is a central component of natural trophic interactions (Savoca & Nevitt, 2014; Savoca et al., 2016). A visual similarity between marine debris or plastic and natural food can also lead to plastic ingestion by marine animals or seabirds (Schuyler et al., 2014; Santos et al., 2016). Another risk is posed by organisms that can colonise objects drifting in the sea and thus reach other areas and potentially colonise there (Barnes & Fraser, 2003; Barnes & Milner, 2005; Bowley et al., 2021; Haram et al., 2023; Mghili et al., 2023; Dawson et al., 2024). A recent study uses ocean modelling to show that much of the marine debris arrives at the tip of the Antarctic Peninsula, where relatively warm conditions increase the likelihood of non-native species settling (Dawson et al., 2024). Furthermore, the importance of the so-called plastisphere in the form of plastic-associated bacterial communities on plastic objects has been pointed out (Cappello et al., 2021). Numerous studies have already demonstrated the occurrence of a high diversity of prokaryotes and eukaryotes, including plastic decomposers and pathogens (e.g. Zettler et al., 2013; Oberbeckmann et al., 2015; Lacerda et al., 2022; De-la-Torre et al., 2023; Silva et al., 2023). In addition, adhering contaminants pose a significant risk to the environment and organisms (Bargagli & Rota, 2022).

Findings of anthropogenic material in bird nests or food remains from the Fildes Region prove that seabirds in the study area interact with marine debris (see chapter 11; Braun et al., 2022) or debris from the local stations (Peter et al., 2008; Peter et al., 2013). It should be noted that compared to the past, significantly less debris is now being discharged from the stations in the Fildes Region into the surrounding area (Peter et al., 2008; Peter et al., 2013). This could be due to increased awareness of the problem of environmental pollution from Antarctic stations (ATS, 2019; Zhang et al., 2020).

For future marine debris surveys, an increasing use of new, image-based methods such as the use of UAVs and deep learning is emerging (Baak et al., 2022; Cocking et al., 2022; Gonçalves & Andriolo, 2022; Pfeiffer et al., 2023). However, standardised protocols for recording plastic pollution in the Antarctic and Southern Ocean are recommended in order to ensure sufficient comparability, which is often required (Browne et al., 2015; Bargagli & Rota, 2022; De-la-Torre et al., 2023; De-la-Torre et al., 2024; Savoca et al., 2024). Furthermore, a selection of bioindicators – analogous to the Arctic (Savoca et al., 2022) – could be used to record the spatial and temporal distribution of certain marine debris.

12 Influence of anthropogenic material on seabirds in the Fildes Region

12.1 Introduction

Various negative effects of anthropogenic material on animals in terrestrial, aquatic and marine environments have been documented (e.g. Gregory, 2009; Teuten et al., 2009; Wilcox et al., 2015; Rochman et al., 2016; Galloway et al., 2017; Almroth & Eggert, 2019; Bucci et al., 2020). Possible consequences range from tissue inflammation, altered gene expression, influence on the gut microbiome, organ damages to reduced growth, reproductive success and survival (see reviews in Derraik, 2002; Kühn et al., 2015; Rochman et al., 2016; Provencher et al., 2019). Furthermore, ingested anthropogenic material can lead to perforation of the gastrointestinal tract or block it completely (e.g. Jacobsen et al., 2010; Brandão et al., 2011; Poli et al., 2015).

Marine mammals and seabirds are also very frequently affected by plastic pollution (Santos et al., 2021; Kibria, 2024). More than 44 % of the world's seabird species are affected by anthropogenic material (Laist, 1997). Ingestion of plastic floating in the sea by seabirds is known from numerous species that feed at or near the water surface (< 2 m), but has also been shown for species that feed at medium (3 - 6 m) and greater depths (20 - 100 m) (Tavares et al., 2017). In addition, bird species with a generalist diet are particularly susceptible to plastic ingestion (e.g. Coulson & Coulson, 1993; Lenzi et al., 2016; Seif et al., 2018; Jardine et al., 2021). Plastic ingestion during foraging by seabirds is strongly influenced by the intensity of plastic dispersal in the oceans, species-specific habitat preferences, foraging characteristics (e.g. food type) and seabird body size (Ryan, 1987a; van Franeker & Law, 2015; Ryan, 2016). Another possible cause is the greater visual similarity of small plastic fragments to crustaceans than to fish or cephalopods (Dell'Ariccia et al., 2017) as well as a strong colour selectivity of the ingested plastic particles, e.g. in petrels (Ryan, 1987a). Chemical signatures of plastic particles floating in the sea could also play a special role, particularly for certain seabird species, especially for representatives of the order of tube-nosed birds or Procellariiformes, which are strongly olfactory orientated when foraging (e.g. Verheyden & Jouventin, 1994; Nevitt et al., 1995; Nevitt, 1999; Cunningham et al., 2003; Nevitt et al., 2008; van Buskirk & Nevitt, 2008; Dell'Ariccia et al., 2014). In particular, odour signatures such as dimethyl sulphide DMS can be used for orientation and alert the birds to areas where prey is supposedly to be found (Nevitt et al., 1995; Dell'Ariccia et al., 2014). As upwelling currents in the sea can bring not only nutrients but also plastic particles that have already sunk back to the surface, aggregations of plastic and zooplankton are often found in the same places, so that foraging seabirds often ingest plastic there (Ryan, 1987a). Procellariform seabirds are particularly affected by this (Ivar do Sul et al., 2011), as they very often have plastic particles in their digestive tract. It has been shown that half of the petrel and storm-petrel species now regularly ingest plastic particles (Kühn et al., 2015), with different species being affected to varying degrees, although some species almost always have ingested plastic (great shearwater: 98 %, blue petrel: 93 %, northern fulmar: > 80 %; Ryan, 1987b; Moser & Lee, 1992). Since Procellariform seabirds only very rarely regurgitate indigestible components of the stomach contents in the form of pellets, plastic particles can accumulate in the stomach.

Plastic pollution has also reached the polar regions and is the subject of a wide range of research (e.g. Avery-Gomm et al., 2018; González-Pleiter et al., 2021; Bergmann et al., 2022; Bernard et al., 2024; De-la-Torre et al., 2024; Lozoya et al., 2024; Savoca et al., 2024). Plastic ingestion by seabirds in the Antarctic has been known for a long time (van Franeker & Bell, 1988; Ainley et al., 1990a). The first evidence from the study area dates back to the 1990/91 season, when plastic was detected in the stomach contents of cape petrels on King George Island, including fishing-related material and industrial granules (Creet et al., 1994). More recent studies have documented the ingestion of plastic by penguins and seals in the Sub-Antarctic and Antarctic (Bravo Rebolledo & van Franeker, 2015; Bessa et al., 2019; Golubev, 2020a; Le Guen et al., 2020; Fragão et al., 2021; Braun et al., 2022; Fógel et al., 2022; Wawrzynek-Borejko et al., 2022; Kim et al., 2023b; Robin et al., 2024). A recent study showed that microplastics were detectable in all of the faecal samples examined from various Antarctic seal species (Cebuhar et al., 2024). In contrast, another study was unable to detect any microplastic in Antarctic fur seal from a hot spot of human activit y(Garcia-Garin et al., 2020).

Due to the high density of stations, field huts and the high level of human activity, the Fildes Region is heavily influenced by anthropogenic activities and is to a large extent negatively impacted by insufficiently optimised waste management at the stations (Peter et al., 2008; Braun et al., 2012; Peter et al., 2013; Braun et al., 2014). In addition, anthropogenic material is also introduced in the form of marine debris, particularly on the west coast (Braun et al., 2022; Vesman & Ershova, 2022; Braun et al., 2024). Negative influences of anthropogenic material on the native fauna can therefore be assumed. Consequently, the interaction of seabirds and seals with anthropogenic material, especially plastic, has already been repeatedly documented in this area, in particular for brown skua and south polar skua (Braun, 2001; Prowaznik, 2019) and the southern giant petrel (Peter et al., 2008; Peter et al., 2013; Braun et al., 2020c).

12.2 Analysis of stomach content of seabird

Since the 2017/18 season, a total of 26 complete animals - either freshly dead or, in the case of storm-petrels, partly as intact mummies - have been collected and their stomach contents analysed for anthropogenic material. Initial results were presented in Braun et al. (2022) and supplemented by further analysed animals. Anthropogenic material was recorded in around 46.2 % of the animals analysed, including Adélie and gentoo penguins, a south polar skua chick and an adult southern giant petrel (Table 9). A total of 36 particles of anthropogenic material were found in all the animals analysed. These included 5 industrial granules, which were only found in Wilson's storm-petrels. Microplastics was most frequently detected in the stomach contents of Wilson's storm-petrels (Table 9). This is consistent with the results already shown for chicks of this species. Since the age of the chicks could be determined on the basis of regular nest inspections and tarsus measurements (Quillfeldt & Peter, 2000), a linear relationship between the age of the chicks and the amount of microplastic in the stomach could be demonstrated (Braun et al., 2022). It was thus possible to prove that Wilson's storm storm-petrels repeatedly ingest microplastic particles when feeding at sea during the breeding season and subsequently pass these on to their offspring, which means an intergenerational transfer (Nordt, 2009; Braun et al., 2022).

Table 9: Number of investigated animals and abundance of microplastic found in the stomach content; *no species identification possible

Species	age class	No. of investigated animals	Abundance of microplastic
Pygoscelis adeliae	juvenile	1	100 %
Pygoscelis papua	adult	2	0 %
Pygoscelis papua	juvenile	6	33.3 %
Macronectes giganteus	adult	1	100 %

Species	age class	No. of investigated animals	Abundance of microplastic
Macronectes giganteus	juvenile	1	0 %
Oceanites oceanicus	adult	4	100 %
Oceanites oceanicus	juvenile	2	100 %
Fregetta tropica	adult	1	0 %
O. oceanicus/F. tropica *	adult	1	100 %
Stercorarius antarctica lonnbergi	juvenile	1	0 %
Stercorarius maccormicki	adult	1	0 %
Stercorarius maccormicki	juvenile	3	33,3 %
Larus dominicanus	juvenile	1	0 %
Sterna vittata	adult	1	0 %

12.3 Microplastics in storm-petrels and petrels using skua pellets

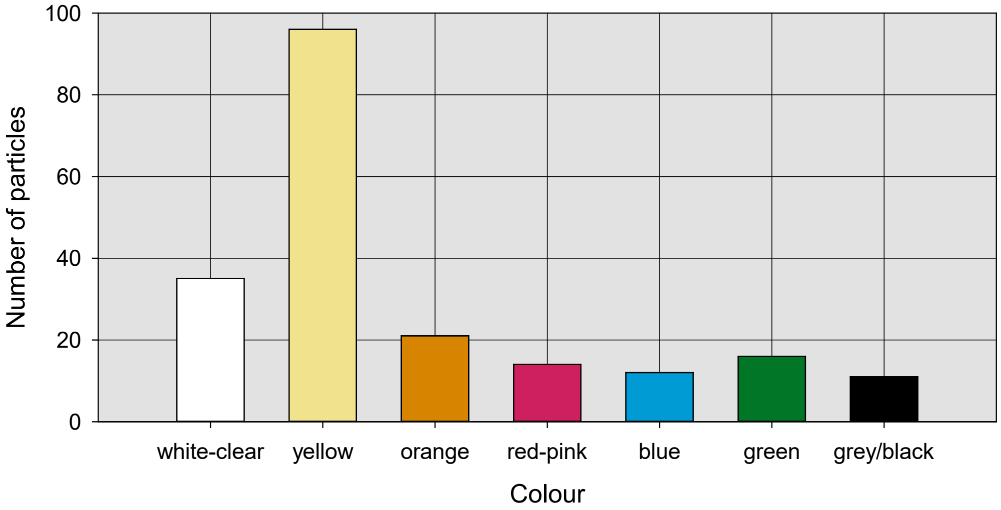
It is known that skuas can specialise as top predators on individual food sources, such as penguins or other seabirds (Pietz, 1987; Young et al., 1988; Ryan & Moloney, 1991; Moncorps et al., 1998; Mougeot et al., 1998; Ryan et al., 2009b; Schultz et al., 2021). Such specialisation on the two native storm-petrel species is also known for brown skuas in the study area and surroundings (Hahn & Quillfeldt, 1998; pers. comm. M. Ritz, M. Kopp, S. Lisovski, J. Esefeld, own observations). As skuas regularly regurgitate the indigestible remains of their food, detailed statements can be made about their food spectrum (Young, 1990b; Reinhardt, 1997; Votier et al., 2001; Santos et al., 2012; Reis et al., 2021; Ibañez et al., 2022; Lozoya et al., 2022). The fact that smaller seabirds are swallowed whole by the skuas due to their small body size and their remains are regurgitated in the form of a compact pellet means that it is possible to draw conclusions about their stomach contents (Ryan & Fraser, 1988; Hammer et al., 2016; Perold et al., 2024).

Therefore, since the 2016/17 season, the focus has increasingly been on the content of skua pellets containing the food remains of smaller seabird species. For the majority of the pellets, it was possible to identify the species of bird eaten. In a first study, it was shown that such skua pellets are suitable for indirect detecting the ingestion of microplastic particles by the seabirds eaten as these plastic particles found differ significantly in shape and size from objects that were directly eaten by skuas (Braun et al., 2022). To substantiate this statement of a trophic transfer, the existing data set was supplemented by a further 149 skua pellets collected in the Fildes Region, which apparently contained the remains of storm-petrels or prions. Thus, a total of 190 pellets containing the remains of Wilson's storm-petrels (*Oceanites oceanicus*), black-bellied storm-petrels (*Fregetta tropica*) and other procellariiform birds (cape petrel - *Daption capense*, prion - *Pachyptila* spec.) were collected and analysed for visible anthropogenic material (Figure 70). In addition to remains of these species, a few pellets also contained remains of seals, penguin limpets or squid. What was striking here was an increasingly large proportion of skua pellets containing the remains of prions *Pachyptila* spec. To what extent this is the result of a possible increase in this species in the prey spectrum of skuas in the Fildes Region or the result of an intensified search cannot be answered.

When analysing the skua pellets, it became clear that this indirect method proves the frequent occurrence of microplastic particles in the stomach contents of petrels and storm-petrels. On average, clearly identifiable plastic particles were found in approx. 37.4 % of all skua pellets. The frequency of microplastics differed significantly depending on the species (Table 10). Microplastics were found in approx. 28.3 % of the samples with residues of Wilsons' storm-petrels, followed by black-bellied storm-petrel with approx. 30.4 %. The extraordinarily high frequency of microplastic particles (approx. 87.1 %) in skua pellets containing the remains of prions *Pachyptila* spec. was striking. On average, between 1.6 and 4.6 particles per pellet containing microplastics were found, depending on the bird species (Table 10). The maximum number of 22 microplastic particles was found in a skua pellet with residues of prion *Pachyptila* spec. pellets of this species contained significantly more and larger particles on average, and the mean plastic load per skua pellet also differed depending on the species (Table 10). The size of the particles was in line with comparable studies (Perold et al., 2024).

The total of 168 microplastic particles found also included a high proportion of industrial granules (approx. 22 %), which were also found in the stomach contents of storm-petrels and in the coastal drift line of the study area (see chapter 11.4 & 12.2). The colour distribution of all microplastic particles found in skua pellets containing seabird remains is shown in Figure 71. The colours yellow (62.5 %) and orange (27.5 %) also predominate in the 40 industrial granules found in stomach contents and skua pellets. Looking at both colour distributions, it is noticeable that this essentially corresponds to the colour spectrum of the industrial granules collected along the coastline, which also predominantly showed yellow shades (see chapter 11.4).

Figure 70: **Distribution of the colours of the microplastic particles extracted from seabird remains found in skua pellets in the Fildes Region between the 2016/17 and 2023/24 seasons**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

A chemical analysis of the microplastic particles found in skua pellets to determine the type of plastic and the detection of possible adhering pollutants will be carried out at a later date.

Figure 71: Examples of skua pellets and microplastic particles found in them

Photos: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Table 10: Details of microplastic items found in skua pellets containing petrel or storm-petrel remains, *no species identification possible

Species in skua pellet	No. of samples	No. & abundance of samples with microplastic	No. of microplastic per sample with microplastic	Mean weight microplastic (mg)	Mean length microplastic (mm)	Mean width microplastic (mm)	Mean stomach load microplastic (mg)	Abundance of granules per sample with microplastic
Oceanites oceanicus	60	17 (28.3 %)	3.3	7.3 ± 5.9	4.0 ± 1.5	2.6 ± 0.6	17.7 ± 36.9	15.8 %
Fregetta tropica	56	17 (30.4 %)	1.6	5.1 ± 6.0	3.3 ± 0.4	2.7 ± 0.6	7.5 ± 6.5	10.0 %
*O. oceanicus/F. tropica? **	30	8 (26.7 %)	1.2	5.2 ± 5.4	3.2 ± 0.6	2.5 ± 0.6	6.6 ± 6.8	9.4 %
Pachyptila spec.	31	27 (87.1 %)	4.6	11.2 ± 6.3	4.4 ± 0.9	3.4 ± 0.5	53.1 ± 56.0	34.0 %
Daption capense	1	1 (100.0 %)	1.0	n.a.	n.a.	n.a.	n.a.	0 %

12.4 Anthropogenic material in seabird nests

The presence of rubbish in bird nests, including for nest building, is known for numerous seabird species (e.g. Votier et al., 2011; Tavares et al., 2016; Grant et al., 2018; O'Hanlon et al., 2019; Tavares et al., 2019; Tavares et al., 2020; Thompson et al., 2020). For the Antarctic, however, there is relatively little evidence to date (e.g. Finger et al., 2021; USA, 2022).

As already documented several times in the past (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022), anthropogenic material was again detected on seabird nests in all three studied seasons. For example, a longline hook, a piece of string, a shard of glass, a paint chip and many pieces of plastic film, similar to a rubbish bag, were found in nests of the southern giant petrel (Figure 72 a, b, c & d). The latter pieces of plastic film were covered with food residues and stomach oil and thus appeared to have been eaten and regurgitated by the birds. Furthermore, two pieces of wire and a nail were found on kelp gull nests. A shard of glass was found in a gentoo penguin's nest, which was presumably used as nesting material, because it has the same size of the stones of the nest (Figure 73).

Figure 72: Anthropogenic material found on nests of the southern giant petrel

Photos: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Due to the nature of the described findings and the fact that all of these nests are located in the south and south-east of the Fildes Peninsula, it is assumed that the high occurrence of rubbish in the vicinity of the stations contributes to the birds' interaction with anthropogenic material. Added to this are fishing activities in the foraging and wintering areas or in the breeding area itself. It has been known for many years that members of Fildes Peninsula stations fish for local fish on a more or less regular basis (Peter et al., 2008; Peter et al., 2013; Braun et al., 2017a). First observations date back to the 1980s (pers. comm. H-U. Peter).

Figure 73: Shard of glass found in a gentoo penguin nest

Photo: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

In the 2022/23 season, a direct interaction with breeding birds of the area was recorded for the first time. Thus, on 15.01.2023, a brown skua with a fishing line sticking out of its beak was observed at a research station. Due to a request by the station manager, the bird was to be examined by the staff of this project. Even from a distance, the animal showed signs of a weakened physical condition. After the catch by a loop, it was tried to remove the line with the hook. It became clear that the skua had swallowed a fishing hook extremely deep. However, after a few minutes of handling, the animal died. It was then dissected and the hook was located in the stomach. Furthermore, the skua's stomach contained a peanut and the typical red paint chips that are found in large quantities around buildings and tanks. This confirms that it was one of the skua pairs, which are present and regularly fed in each of the Fildes Peninsula stations. The fishing hook was clearly different in shape and size from the longline hooks used by the fishing industry (Figure 74), which have already been found several times in the study area (Peter et al., 2008; Peter et al., 2013; Braun et al., 2020c). Using the existing skua database, it was determined that the bird was ringed as an adult by staff of the Polar & Bird Ecology working group of the University of Jena in the season 2006/07 and thus had a minimum age of 17 years.

Another find related to local fishing activities was a hook set on Ardley Island on 08.02.2023. It was found on the beach area of the penguin colony located there and was only conspicuous because a penguin got its feet tangled in the fishing line. In addition to a line, the hook set had two hooks of the same type as the one swallowed by the skua, as well as a treble hook, which posed a considerable risk of injury to animals (Figure 75).

Figure 74: Fishing hook swallowed by a brown skua

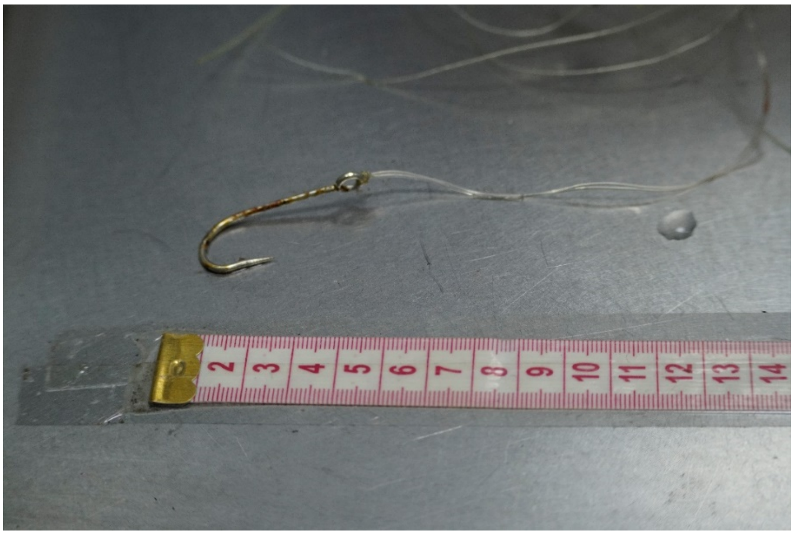


Photo: N. Heid, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Figure 75: Fishing hook set found at Ardley Island penguin colony

Photos: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

12.5 Discussion and conclusions

The results shown here once again demonstrate frequent interactions of the seabirds native to the study area with anthropogenic material. The high ingestion rate of plastic by storm-petrels shown here - determined from both stomach contents and skua pellets - confirms previous findings (e.g. Acampora et al., 2014; Bond et al., 2014; Lavers et al., 2014; Trevail et al., 2015; Kain et al., 2016; Youngren et al., 2018; Baak et al., 2020). However, due to the pelagic diet of petrels and whale petrels, it can be assumed that the plastic particles found in the stomach contents and in skua pellets were ingested from the sea surface and not in the coastal area. Thus, marine debris washed up in the Fildes region is unlikely to be a significant source of ingested microplastics. The size and weight of the plastic particles found in the digestive tract of petrels and in skua pellets with seabird remains in the present study are consistent with the measurements from a comparative study, which shows that the majority of plastic particles ingested by numerous petrel species are between 2 and 10 mm in size (Roman et al., 2019). As it is the most widespread plastic material floating in the ocean (Morét-Ferguson et al., 2010), the risk of ingestion by seabirds, especially by pelagic plankton feeders such as petrels or prions, is particularly high. The intergenerational transfer of microplastics from parent birds of petrels to offspring (Nordt, 2009; Braun et al., 2022) is also known for several representatives of the order Procellariiformes of the southern hemisphere, e.g. for the coloured-footed storm-petrel, southern giant petrel, Scopoli's shearwater (*Calonectris diomedea*) or short-tailed shearwater (*Ardenna tenuirostris*) (e.g. van Franeker & Bell, 1988; Copello & Quintana, 2003; Carey, 2011; Rodríguez et al., 2012; Cousin et al., 2015).

In addition, the results presented here confirm the presence of microplastics in the stomach contents of penguins, which has recently been demonstrated several times for the Antarctic (Bessa et al., 2019; Golubev, 2020a; Fragão et al., 2021; Fógel et al., 2022; Wawrzynek-Borejko et al., 2022). However, there is no known evidence of microplastics in southern giant petrels in the Antarctic. The occurrence of anthropogenic material in skuas is not congruent with the fact that they often search for food near the stations as opportunistic feeders and frequently ingest anthropogenic material (Peter et al., 2008).

Furthermore, in the past three seasons, anthropogenic material was documented more frequently in seabird nests than in previous years (Braun et al., 2017a; Braun et al., 2020c; Braun et al., 2022). All described findings confirm the continuing interaction of seabirds in the Fildes Region with anthropogenic material (Braun et al., 2022). For the seabird species described in the presented study, there are only a few comparable records of anthropogenic material in nests from the Antarctic (e.g. Chile, 1996, 1997; Finger et al., 2021; USA, 2022). In contrast, anthropogenic debris has been detected in nests of kelp gulls in South America and South Africa (e.g. Witteveen et al., 2017; Seco Pon & Pereyra, 2021; Yorio et al., 2022). However, an entanglement of birds or seals or injuries caused by this were not recorded in the Fildes Region. The last entanglement of a seal by a fishing rope dates from the January 2006. Few incidents of this species have also been documented for other areas of the South Shetland Islands, such as Cape Shirreff, Livingston Island (Bravo Rebolledo & van Franeker, 2015). Thus, this particular risk appears to be relatively low in the study area compared to South Georgia (Waluda & Staniland, 2013).

Although various negative effects of anthropogenic material on the animals are to be expected, serious effects can only be proven in exceptional cases, such as in the case of the skua that died from a fishing hook. Nevertheless, sublethal effects can have a significant impact on the survival and fitness of animals (e.g. Lavers et al., 2019; Puskic et al., 2019; Fackelmann et al., 2023; Rivers-Auty et al., 2023; Kibria, 2024).

In conclusion, the results presented here emphasise that seabirds and seals can serve as effective indicators of the extent of ubiquitous pollution from anthropogenic material in an area (Arnould & Croxall, 1995; Lopes et al., 2022). They are also vectors for macro- and microplastics, as well as adhering pollutants from one area to another. It has already been shown that an accumulation of microplastics can be observed in areas with seabird colonies due to guano or pellet accumulation and nesting materials (e.g. Hartwig et al., 2007; Votier et al., 2011; Verlis et al., 2014; Provencher et al., 2018a; Provencher et al., 2018b; Jagiello et al., 2019; Brentano et al., 2020; Finger et al., 2021). In addition, the methods used here (use of dead finds and food pellets from skuas) are inexpensive, non-invasive methods that are particularly advantageous in inaccessible areas (e.g. van Franeker & Law, 2015; Provencher et al., 2017; Perold et al., 2020; Hamilton et al., 2021). Standard sampling and analytical procedures for plastic research (particularly for microplastics) are strongly recommended (De-la-Torre et al., 2024; Savoca et al., 2024).

However, the foraging strategy of the various seabirds (Caldwell et al., 2020) and the length of time the particles remain in the birds' digestive tract must also be taken into account (Ryan, 2015; Dell'Ariccia et al., 2017). While numerous procellariform seabirds in particular are suitable indicators of the frequency of floating marine debris of a certain size (e.g. van Franeker & Law, 2015; Perold et al., 2020; Phillips & Waluda, 2020; Bond et al., 2021), there is very little evidence of plastic ingestion for cape petrels (Creet et al., 1994). Evidence of interactions of terns with anthropogenic material is also rare (Tavares et al., 2017; Carrillo et al., 2023). If only a limited number of complete dead seabird are available, e.g. due to such a high predator density as in the Fildes Region, the indirect detection of the presence of plastic in seabirds via food pellets from skuas described here can provide valuable information about plastic floating in the sea (Ryan & Fraser, 1988; Hammer et al., 2016; Quer & Gilbert, 2018; Perold et al., 2024).

13 Introduced non-native species

13.1 Introduction

Biological invasions are considered one of the greatest challenges for the conservation of species and the maintenance of the health of ecosystems and their structures, especially in geographically and evolutionary isolated ecosystems (United Nations, 2008; Chown et al., 2012; McGeoch & Jetz, 2019). Current climate change favours bioinvasion processes by the enhancement of oceanographic connectivity (Fraser et al., 2018) and accelerating the establishment of non-native species, thereby threatening the survival of native species and ecosystems (Stachowicz et al., 2002; Chown et al., 2015; Lembrechts et al., 2016; Duffy et al., 2017; Liu et al., 2017; Cuba-Diaz et al., 2023). Among the non-native species, invasive alien species, defined as ‘those non-native species that threaten ecosystems, habitats or species’, pose a particular threat to local ecosystems (Mack et al., 2000; United Nations, 2008). Especially due to the comparatively low terrestrial biodiversity of Antarctica, where ecosystem functions are occupied by few species, potentially vacant niches can be relatively easily occupied by alien and invasive species (Convey & Stevens, 2007). A recent study indicates that approx. 13 % of species are considered locally invasive (Leihy et al., 2023). Here, marine invertebrates dominate among the highest risk species alongside flowering plants and terrestrial invertebrates (Hughes et al., 2020). In addition, there is generally less interspecific competition for habitat and resources, so native organisms may not be able to compete with introduced species with greater or different physiological flexibility (Hughes & Convey, 2012). Furthermore, many non-native species cope better with environmental changes such as temperature increase or anthropogenic soil disturbance (Molina-Montenegro et al., 2014; Molina-Montenegro et al., 2019; Pertierra et al., 2021). This can result in consequences for community composition of plants and animals and ecosystem functioning in cold environments (Atala et al., 2019; Bartlett et al., 2023; Martin et al., 2023).

Despite the relatively isolated geographical location of Antarctica, the geographical barrier has been broken several times by non-native species (Hughes & Convey, 2012; Hughes et al., 2015; Potocka & Krzemińska, 2018; Bergstrom, 2022; Leihy et al., 2023). Although transport of non-native species or their propagules to new habitats may also occur via atmospheric transport or through natural vectors such as migratory birds or detached algae (Lewis-Smith & Richardson, 2011; Avila et al., 2020; Macaya et al., 2020; Rodrigues et al., 2023), anthropogenic introduction is much more likely (Frenot et al., 2005; Hughes & Convey, 2010; Osyczka, 2010; Hughes et al., 2011; Chown et al., 2012; Litynska-Zajac et al., 2012). Thus, human-mediated transport of organisms, seeds or other dispersal units to Antarctica may be indirect, e.g. via marine debris (Barnes & Fraser, 2003; Mghili et al., 2023) or ballast water (Cárdenas et al., 2020; Duliére et al., 2022), or via direct introduction by people or cargo (Hughes et al., 2010; Osyczka, 2010; Hughes et al., 2011; Litynska-Zajac et al., 2012; Chwedorzewska et al., 2013; Hughes et al., 2018). Besides the tourism industry, the National Antarctic Programmes play an essential role in this as operators of the stations, aircraft and numerous ships (Whinam et al., 2005; Huiskes et al., 2014; Remedios-De León et al., 2021; Fuentes-Lillo & Cuba, 2022). Hence, the Antarctic Peninsula region, with its rapid temperature increase and high level of human activity, is particularly at risk (Chown et al., 2012; Duffy et al., 2017; Pertierra et al., 2017; Hughes et al., 2020; McCarthy, 2021). Terrestrial biodiversity in Antarctica is increasingly threatened by the expansion of ice-free areas (Duffy & Lee, 2019). Therefore, the changing environmental conditions together with the observed further increase in human activities, ship traffic and the rapidly growing number of visitors at the Antarctic Peninsula favours the introduction, spread and establishment of non-native species (Molina-Montenegro et al., 2014; McCarthy et al., 2022; Cuba-Diaz et al., 2023).

Although the relatively short human presence in Antarctica means, various deliberately or accidentally, introduced plant and animal species have been recorded, among them the annual bluegrass (*Poa annua*) and various invertebrate species, including some insect species (Block et al., 1984; Hughes et al., 2005; Peter et al., 2008; Peter et al., 2013; Russell et al., 2013; Chwedorzewska et al., 2015; Hughes et al., 2015; Bartlett et al., 2020). Several records of non-native grasses and insects have also been documented for the Fildes Peninsula (Peter et al., 2008; Peter et al., 2013; Braun et al., 2014; Braun et al., 2017a; Potocka & Krzemińska, 2018; Braun et al., 2020c; Braun et al., 2022; Republic of Korea & Chile, 2022b). Current results from the monitoring in the Fildes Region in the past three field seasons are presented below.

13.2 Results

13.2.1 Plants

The repeated checks of the known location of an introduced non-native grass species in the area of the Russian Bellingshausen station did not reveal any evidence of a re-occurrence of the plant which was manually removed in the 2008/09 season (Peter et al., 2013). Thus, the species can be considered eradicated at this site.

During the check in the 2022/23 and 2023/24 of the location of several tufts of a non-native grass near the Chinese station Great Wall, no signs of re-occurrence were recorded there either. It should be noted that the area at Great Wall station, where several grass cushions with inflorescences were removed in the 2005/06 season (Peter et al., 2008), was extensively moved and levelled as part of ground work.

In January 2022, a conspicuous tuft of grass was noticed in the area of the Uruguayan Artigas station under the stairs to a residential building (Figure 76). Due to the strong suspicion that it was an introduced species, the grass, including the surrounding soil, was removed in accordance with the Committee for Environmental Protection (CEP) non-native species manual (CEP, 2019) and the Uruguayan regulations. In the following season, the grass was covered with a large wooden board. At present, there are no species identification results for this grass. If the suspicion of a non-native species is confirmed, the site should be carefully monitored in the next few years to prevent reintroduction in time.

Figure 76: Suspected introduced, non-native tuft of grass under the stairs to a station building

Photo: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Furthermore, a single acorn (*Quercus* spec.) was found in the Bellingshausen station area in the 2021/22 season (Figure 77). The site was located on the main route taken by tourists on the air-cruise programme on their way between the airport and the beach. This indicates that seeds or dispersal units of non-native plants - including larger ones - can still enter the study area as a result of human activities

13.2.2 Insects

Specimens of the introduced non-native winter crane fly *Trichocera maculipennis*, which is now considered to be persistent in the Fildes Region, were repeatedly observed inside various buildings at Artigas Station, where numerous traps are still installed. In addition, up to three individuals were found in the adhesive trap set up in the Refugio Ripamonti field hut on Ardley Island. During the three studied seasons, numerous specimens of *T. maculipennis* mosquito species were also detected outside the station buildings. For example, during the marine debris collection (see Chapter 11) in the season 2022/23, a plastic jar was found with a large number (approx. 20 individuals) of *T. maculipennis* attached. Further individuals were observed further north on the west coast in the season 2023/24 (Figure 77). The discovery of a winter crane fly clinging to a rucksack during a boat trip by project staff emphasises the high probability of this species being transported to other areas.

Another insect recorded in a station building was probably a specimen of the moth fly *Psychoda albipennis* (order: Diptera, family: Psychodidae). Other non-native insects were also reported in the station buildings, including clothes moths, their larvae and pupae and food pests. However, an exact species identification by an entomologist is not available.

All known findings of introduced non-native species that have been documented outside station buildings are shown in Figure 77. All individuals presumed to belong to the species *T. maculipennis* are attributed to this now widespread species.

Figure 77: **Locations of all findings of introduced non-native species in the Fildes Region documented by project staff outside station buildings between 2003 and 2024**

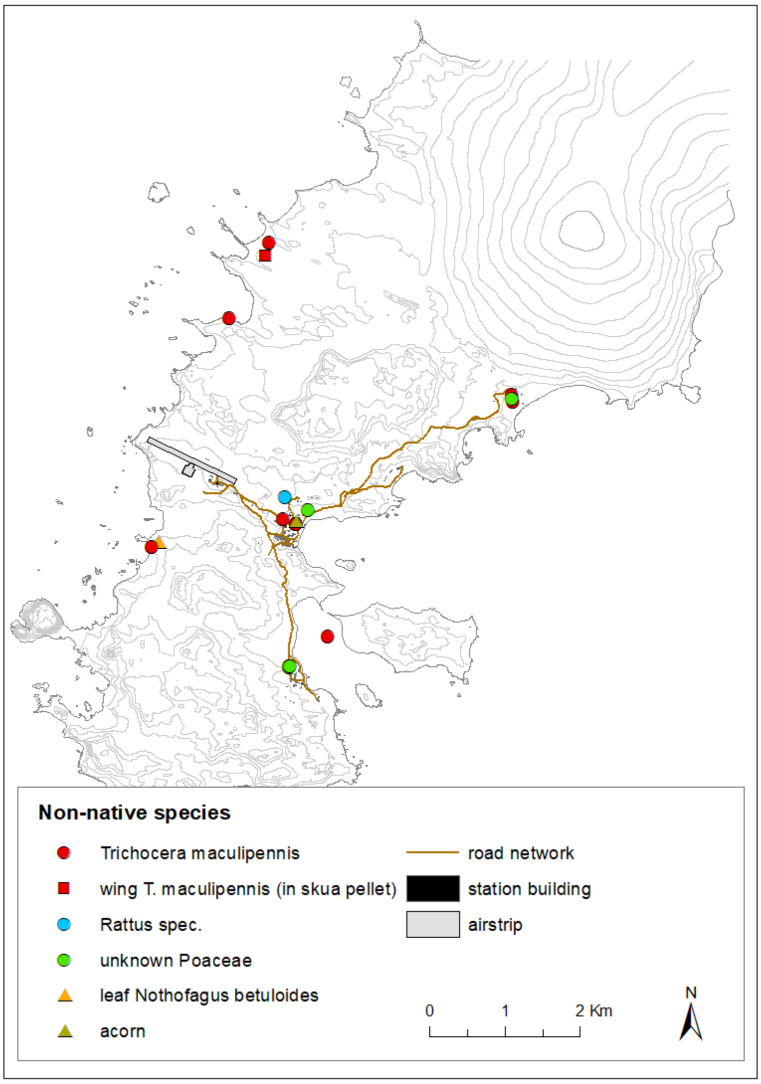


Photo: C. Braun, Institute of Ecology and Evolution, Friedrich Schiller University Jena

13.3 Discussion and conclusions

Since 2004, the occurrence of introduced non-native species on the Fildes Peninsula has received increased attention following the finding of a dead rat, several tufts of Poaceae and the first observation of *T. maculipennis* on a moss carpet (Peter et al., 2008; United Kingdom & Uruguay, 2011).

Due to repeated findings of *T. maculipennis* in wastewater treatment plants at two stations on King George Island, a joint monitoring system was initiated by several Antarctic Treaty Parties in the 2017/18 season (Uruguay et al., 2018). In the meantime, *T. maculipennis* has been documented in at least seven stations on King George Island (Uruguay et al., 2019; Chile et al., 2021; Chile et al., 2023). Specimens of the species *T. maculipennis* were recorded in station buildings, sewage treatment plants and outside the infrastructure (Uruguay et al., 2019; Chile et al., 2021; Chile et al., 2023). The most recent first record comes from a field hut on the Potter Peninsula from February 2022 (Argentina & Uruguay, 2022). Finally, it was stated that since no signs of pupae and larvae have yet been found outside the station buildings in the Fildes region, successful colonisation may not yet have taken place (Chile et al., 2023). In contrast, the numerous finds of adult individuals outside the stations or buildings indicate a high possibility of survival and presumably also reproduction (Remedios-De León et al., 2021). Consistent with this, a large number of exuviae of this species were already found outside the station buildings in the 2018/19 season, albeit in the immediate vicinity (Braun et al., 2022).

Furthermore, the recently proven non-native insect species *P. albipennis*, a cosmopolitan species, has now been detected in large numbers in wastewater treatment plants of two stations on the Fildes and the Barton Peninsula (Republic of Korea & Chile, 2022b; Chile et al., 2023). Furthermore, three beetles of undetermined species were also found, one of which was recorded alive and active outside in the same place as *T. maculipennis* (Chile et al., 2023). A live bedbug of the species *Lygaeus alboornatus* was found in connection with fresh vegetables (Chile et al., 2023). Moth-like insects and mosquitoes have also been recorded from other areas, e.g. Greenwich Island, South Shetland islands, or Doumer Island on the Palmer Archipelago, 400 km further south (Chile et al., 2023).

It can be stated, that the risk of the introduction and spread of non-native species in the Fildes Region remains high. Furthermore, it can be confirmed that there are still deficiencies with regard to measures to prevent the introduction of non-native species to the Fildes Peninsula. The insect eradication attempts of some stations could reduce but not prevent the ongoing colonisation of the station facilities. Therefore, despite some progress regarding the joint monitoring program, an effective, standardized and joint eradication program still needs to be developed and implemented (Chile et al., 2021; Chile et al., 2023). Thus, there is still an urgent need for action to prevent the introduction and expansion of non-native species, especially as King George Island is one of the primary gateways to Antarctica (Chile et al., 2023). The implementation of an overarching non-native invasive species management plan for King George Island or the wider maritime Antarctic region is recommended (Remedios-De León et al., 2021; United Kingdom & Uruguay, 2022). In this respect, too, the Fildes Region represents an important paradigm for other areas in antarctica.

In conclusion, the growing pressure of introduced non-native species on Antarctic ecosystems needs to be countered by efficient measures which should include researching the impact of non-native species, implementing practices to reduce their introduction and spread and developing further response methods (e.g. Bergstrom, 2022; Hughes et al., 2023; Onley et al., 2024).

14 Open questions, need for further research and development of the monitoring programme

The high value of long-term monitoring programmes for the assessment of environmental changes in ecosystems is widely recognised. For example, long-term studies of fauna and flora population and distribution data, particularly in areas with recent, clearly demonstrable changes, can provide valuable insights into the effects of changing environmental conditions on the ecosystem and also help to assess future developments. This is particularly important in areas that have undergone major environmental changes in recent decades as a result of global warming, such as the Western Antarctic Peninsula, and also for future conservation measures. The majority of population surveys of seabird and seal populations in the Antarctic are based on only a few counts and, due to frequent natural population fluctuations, have only very limited informative value regarding actual developments. The long-term, largely continuous data series resulting from systematic long-term monitoring in the Fildes Region are therefore of the greatest value. Furthermore, the location of the study area, with its high level of human activity, offers the opportunity to map the effects of certain anthropogenic influences, such as human disturbance, marine debris or introduced non-native species, on Antarctic ecosystems. Furthermore, recent results show that direct human impacts should be considered within monitoring programmes of certain seabird species in anthropogenic areas to ensure a reliable interpretation of population data and to define indicator species. This can help to address conservation measures that may be necessary to preserve Antarctic ecosystems.

For the reasons mentioned above, a continuation of the long-term surveys of the following topics is urgently recommended. Increased use of remote sensing data to avoid logistical constraints and reduce the number of seabird colonies visited is highly recommended. There is a need for research on the following topics:

- Standardised and GPS/GIS-supported survey of breeding birds and migrant and vagrant birds as well as seals in the Fildes Region
- Recording the breeding success of selected bird species in the Fildes Region
- Breeding bird monitoring in selected areas of Maxwell Bay
- Further investigations into the causes of local population trends of seabird and seals
- Investigating the reasons for spatial changes in the penguin colony on Ardley Island
- Recording non-native species and monitoring their known locations
- Documentation of glacier retreat areas using aerial photographs and satellite images
- Documentation of the colonisation of areas exposed by glacier melt by seabirds
- Monitoring the occurrence of new taw slumps and the further development of permafrost degradation
- Standardised recording of the distribution and extent of marine debris on selected coastal stretches using methods developed by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR)
- Recording the effects of marine debris and other anthropogenic material on seabirds and seals in the study area

- Developing and analysing additional data sources through cooperation with international scientists

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A Appendix: Data sources Fildes-Region

Table 11: Sources of the monitoring data used from the Fildes Region Fildes Region

Season	Source/observer	Recorded data
1967/68	Krylov, 1968	seal census
1973/74	Popov, 1977	seal census
1974/75	Popov, 1977	seal census
1979/80	Bannasch & Odening, 1981	all breeding birds, vagrant & migrant birds
1980/81	Bannasch et al., 1983; Jablonski, 1984	all breeding birds, vagrant & migrant birds, seal census & pupping sites
1981/82	Bannasch et al., 1983	all breeding birds, vagrant & migrant birds, seal census & pupping sites
1982/83	Lorenz, 1984; Peter et al., 1988	all breeding birds, vagrant & migrant birds, seal census & pupping sites
1983/84	Lorenz, 1984; Peter et al., 1988	all breeding birds, vagrant & migrant birds, seal census & pupping sites
1984/85	Rauschert et al., 1987; Peter et al., 1988; Peter et al., 1989	all breeding birds, vagrant & migrant birds, seal census, & pupping sites mapping *Deschampsia antarctica*
1985/86	Rauschert et al., 1987; Zippel, 1987; Mönke & Bick, 1988	all breeding birds, vagrant & migrant birds, seal census & pupping sites
1986/87	Mönke & Bick, 1988	all breeding birds, vagrant & migrant birds, seal census & pupping sites
1987/88	Lange & Naumann, 1989; Nadler & Mix, 1989; Lange & Naumann, 1990	all breeding birds, vagrant & migrant birds, seal census & pupping sites
1988/89	Lange & Naumann, 1989, 1990	all breeding birds, vagrant & migrant birds, seal census & pupping sites
1989/90	Erfurt & Grimm, 1990; pers. Comm. H. Grimm	all breeding birds, seal census & pupping sites
1990/91	pers. comm. J. Valencia & M. J. Roselló	penguins Ardley Island
1991/92	pers. comm. J. Valencia & M. J. Roselló	penguins Ardley Island
1992/93	pers. comm. J. Valencia & M. J. Roselló	penguins Ardley Island
1993/94	pers. comm. J. Valencia & M. J. Roselló	penguins Ardley Island
1994/95	pers. comm. J. Valencia & M. J. Roselló	penguins Ardley Island
1995/96	pers. comm. J. Valencia & M. J. Roselló Soave et al., 2000	penguins Ardley Island skuas, snow sheathbill
1996/97	pers. comm. J. Valencia & M. J. Roselló Chupin, 1997	penguins Ardley Island southern giant petrel
1997/98	pers. comm. J. Valencia & M. J. Roselló	penguins Ardley Island

Season	Source/observer	Recorded data
1998/99	pers. comm. J. Valencia & M. J. Roselló pers. comm. H.-U. Peter & J. Welcker Welcker, 2001	penguins Ardley Island southern giant petrel kelp gull
1999/2000	pers. comm. J. Valencia & M. J. Roselló Welcker, 2001	penguins Ardley Island kelp gull
2000/01	pers. comm. J. Valencia & M. J. Roselló Braun, 2001	penguins Ardley Island all breeding birds, vagrant & migrant birds, seal census & pupping sites, mappping *Deschampsia antarctica*
2001/02	pers. comm. J. Valencia & M. J. Roselló	penguins Ardley Island
2002/03	pers. comm. J Welcker pers. comm. J. Valencia & M. J. Roselló pers. comm. I. Chupin & S. Pfeiffer pers. comm. A. Petrov	skuas penguins Ardley Island southern giant petrel seal pupping sites
2003/04	Peter et al., 2008 pers. comm. M. Ritz	all breeding birds, vagrant & migrant birds, seal census & pupping sites skuas
2004/05	Peter et al., 2008 pers. comm. M. Ritz pers. comm. I. Chupin	all breeding birds, vagrant & migrant birds, seal census & pupping sites skuas kelp gull, seals
2005/06	Peter et al., 2008 pers. comm. M. Ritz	all breeding birds, vagrant & migrant birds, mapping *Deschampsia antarctica*, seal census & pupping sites skuas
2006/07	Fröhlich, 2007 pers. comm. M. Kopp	penguins Ardley Island, southern giant petrel, seal census & pupping sites skuas
2007/08	Braun, 2008 pers. comm. M. Kopp	all breeding birds, vagrant & migrant birds, seal census & pupping sites skuas
2008/09	Peter et al., 2013 pers. comm. S. Lisovksi & M. Kopp	all breeding birds, vagrant & migrant birds, seal census & pupping sites skuas, seal census
2009/10	Peter et al., 2013 pers. comm. M. Kopp & J. Esefeld pers. comm. V. Sjomin	all breeding birds, vagrant & migrant birds, seal census & pupping sites skuas seal pupping sites
2010/11	Peter et al., 2013 pers. comm. A. Soloviev & B. Zatsepin	all breeding birds, vagrant & migrant birds, seal census & pupping sites seal pupping sites
2011/12	Peter et al., 2013 pers. comm. A. Contreras Staeding & R. Eliseev	all breeding birds, vagrant & migrant birds, seal census & pupping sites seal pupping sites

Season	Source/observer	Recorded data
2012/13	Braun et al., 2017a pers. comm. O. Sakharov	all breeding birds, vagrant & migrant birds, seal census & pupping sites seal pupping sites
2013/14	Braun et al., 2017a pers. comm. M. Xing	all breeding birds, vagrant & migrant birds, seal census & pupping sites
2014/15	Braun et al., 2017a	all breeding birds, vagrant & migrant birds, seal census & pupping sites
2015/16	Braun et al., 2020c	all breeding birds, vagrant & migrant birds, seal census & pupping sites
2016/17	Braun et al., 2020c	all breeding birds, vagrant & migrant birds, seal census & pupping sites
2017/18	Braun et al., 2020c	all breeding birds, vagrant & migrant birds, seal census & pupping sites
2018/19	Braun et al., 2022	all breeding birds, vagrant & migrant birds, seal census & pupping sites
2019/20	Braun et al., 2022	all breeding birds, vagrant & migrant birds, seal census & pupping sites
2020/21	pers. comm. A. L. Machado	penguins Ardley Island, southern giant petrel
2021/22	present study pers. comm. A. Soutullo	all breeding birds, vagrant & migrant birds, seal census & pupping sites penguins Ardley Island
2022/23	present study pers. comm. A. Soutullo pers. comm. S. Lisovski pers. comm. V. Pavel pers. comm. E. Krojmal, B. de Feo, B. Mavlyudov pers. comm. P. Chukmasov	all breeding birds, vagrant & migrant birds, seal census & pupping sites penguins Ardley Island southern giant petrel, vagrant & migrant birds southern giant petrel vagrant birds seal census
2023/24	present study pers. comm. A. Soutullo pers. comm. S. Lisovski pers. comm. V. Pavel pers. comm. A. Arsjuchin	all breeding birds, vagrant & migrant birds, seal census & pupping sites penguins Ardley Island southern giant petrel southern giant petrel, vagrant birds vagrant birds

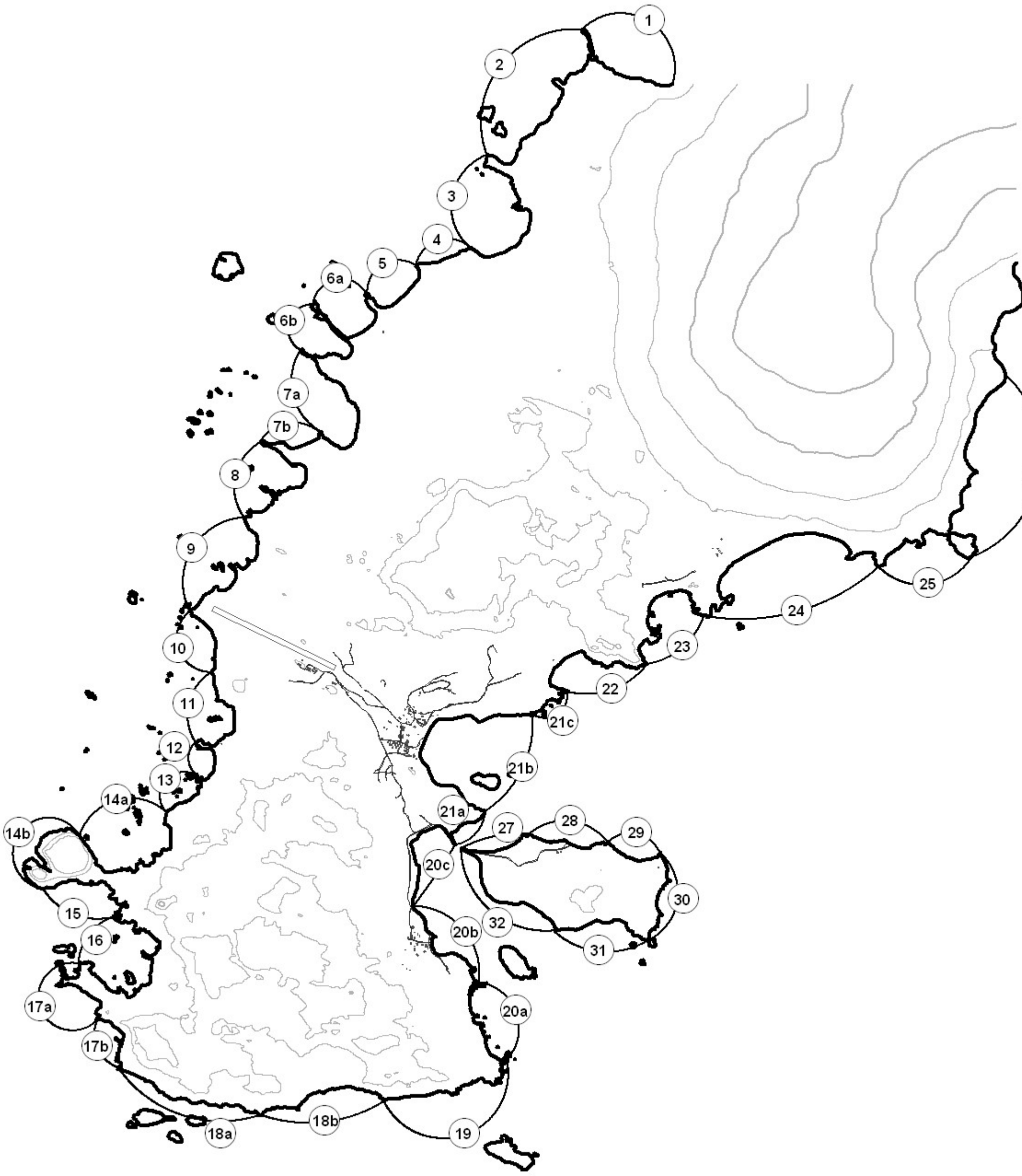
B Appendix: Methodology

Table 12: Timetable of the seabird and seal monitoring and other tasks within this study during the austral summer (Peter et al., 2008)

Species	December	January	February
Adélie, chinstrap & gentoo penguin	breeding pair number	breeding success & chick body weight	breeding success & chick body weight
Southern giant petrel	breeding pair number		breeding success
Cape petrel	breeding pair number		
Light-mantled sooty albatross	breeding pair number	breeding pair number	breeding pair number
Wilson's & black-bellied storm-petrel	breeding pair number	breeding pair number	breeding pair number, chick census
Brown & south polar skua	number of breeding pairs & territories	number of breeding pairs & territories	number of breeding pairs & territories, breeding success
Kelp gull	breeding pair number		
Antarctic tern	breeding pair number	breeding pair number	breeding pair number
Snowy sheathbill	breeding pair number	breeding pair number	breeding pair number
Blue-eyed shag	breeding pair number		chick census
Vagrant & migrant birds	occurrence	occurrence	occurrence
Southern elephant, Weddell, fur, leopard & crabeater seal	number of individuals, haul-out sites, reproduction sites	number of individuals, haul-out sites, reproduction sites	number of individuals, haul-out sites, reproduction sites
Breeding bird monitoring ice-free areas Maxwell Bay	breeding pair number	breeding pair number	
Antarctic pearlwort		occurrence	occurrence
Degradation of permafrost			documentation
Marine debris survey		marine debris collection & classification	
Influence of anthropogenic material on seabirds	occurrence	occurrence	occurrence

Species	December	January	February
Introduced non-native species	occurrence	occurrence, control of known sites with previous occurrence	occurrence, control of known sites with previous occurrence

Figure 78: Numbering of the bays on Fildes Peninsula and Ardley Island as the basis for seal census (Peter et al., 2008)



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

C Appendix: Observations of light-mantled sooty albatrosses in the Fildes Region

Table 13: Overview of all observations of the light-mantled sooty albatross (*Phoebetria palpebrata*) in the Fildes Region; years with confirmed breeding are shown in bold.

Status	Date	Location	No. of birds	No. of nests	Source	Notes
Sighting 1985	19.11.1984	Flat Top rock, Geographers Cove	4	0	Peter et al., 1988	of which 2 in synchronous flight (pair behaviour)
Sighting 1985	17.01.& 01.02.1985	Flat Top rock	2	0	Peter et al., 1988	sailing
Sighting 1985	02.03.1985	Fildes Strait, west of Dart Island	2	0	Peter et al., 1988	sailing
Sighting 1988	27.10.1987	Diomedea Island	1	0	Nadler & Mix, 1989	circling with southern giant petrels
Sighting 1992	04., 14. & 24.01.1992	Fildes Strait, Stansbury Pen.	1	0	Lumpe & Weidinger, 2000	flying
Sighting 2001	05.01.2001	Horatio Stump	2	0	Braun, 2001	in synchronous flight
Sighting 2005	01.01.2005	Flat Top rock	3	0	Peter et al., 2008	circling around Flat Top, two birds in synchronous flight, sitting
Sighting 2008	06.01.2008	Flat Top rock	2	0	pers. comm. H.-U. Peter	in synchronous flight
Breeding 2009	25.12.2008	Flat Top rock	9	5	Lisovski et al., 2009; Peter et al., 2013	first evidence of breeding and young birds
Sighting 2010	21.01.2010	Flat Top rock	1	0	Peter et al., 2013	flying
Sichtung 2011	10.01.2011	Flat Top rock	5	0	Peter et al., 2013	flying
Breeding 2012	15.01.2012	Flat Top rock	1	1	Peter et al., 2013	no offsprings observed
Sighting 2012	10.02.2012	Bellingshausen station	1	0	Peter et al., 2013	overflight
Sighting 2013	14.01.2013	Flat Top rock	1	0	Braun et al., 2017a	flying
Sighting 2013	15.01.2013	Flat Top rock	2	0	Braun et al., 2017a	in synchronous flight

Status	Date	Location	No. of birds	No. of nests	Source	Notes
Breeding 2014	24.01.2014	Flat Top rock	2	2	Braun et al., 2017a	no offsprings observed
Sighting 2015	13.12.2014	Flat Top rock	2	0	Braun et al., 2017a	in synchronous flight
Sighting 2016	13.12.2015	Flat Top rock	2	0	Braun et al., 2020c	flying
Sighting 2016	17.12. 2015 & 05.01.2016	Flat Top rock	3	0	Braun et al., 2020c	flying
Sighting 2016	13.01.2016	Flat Top rock	6	0	Braun et al., 2020c	flying
Sighting 2017	09.01.2017	Flat Top rock	3	0	Braun et al., 2020c	flying
Sighting 2019	25.12.2018	Flat Top rock	1	0	Braun et al., 2022	flying
Sighting 2019	04.01.2019	Flat Top rock	4	0	Braun et al., 2022	flying
Sighting 2019	08.01.2019	coast north of Flat Top rock	3	0	Braun et al., 2022	flying
Sighting 2019	09.01.2019	Flat Top rock	2	0	Braun et al., 2022	in synchronous flight
Sighting 2019	13.01.2019	Flat Top rock	2	0	Braun et al., 2022	flying
Breeding 2019	15.01. & 23.01.2019	Flat Top rock	4-6	2	Braun et al., 2022	sitting at the breeding site, no offsprings observed
Sighting 2020	18.01.2020	Flat Top rock	2	0	Braun et al., 2022	flying
Breeding 2020	21.01.2020	Flat Top rock	3	2	Braun et al., 2022	2 adults sitting at the breeding site, no offsprings observed
Breeding 2020	04.02.2020	Flat Top rock	2	2	Braun et al., 2022	2 adults sitting at the breeding site, no offsprings observed
Sighting 2020	05.02 & 13.02.2020	coast north of Flat Top rock	2	0	Braun et al., 2022	in synchronous flight
Breeding 2020	20.02 & 25.02.2020	Flat Top rock	2	2	Braun et al., 2022	sitting/standing at the breeding site, no offspring observed
Sighting 2022	20.12.2021	Drake coast, near airstrip	3		present study	adults circling

Status	Date	Location	No. of birds	No. of nests	Source	Notes
Sighting 2022	05.01.2022	Flat Top rock	2		present study	flying
Sighting 2022	06.01.2022	Flat Top rock	2		present study	flying
Sighting 2023	22.12.2022	bay east of Artigas station	2		present study	flying
Sighting 2023	04.01.2023	bay east of Artigas station	2		present study	sitting, showing courtship behaviour
Sighting 2023	05.01.2023	Norma Cove	2		present study	flying
Sighting 2023	13.01.2023	east of Bellingshausen station & Neftebasa	1		present study	flying
Sighting 2023	14.01.2023	Gemel Peaks	2		present study	flying
Sighting 2023	14.01.2023	Flat Top rock	3		present study	flying, 2 of them in synchronous flight
Sighting 2023	16.01.2023	Flat Top rock	2		present study	in synchronous flight
Sighting 2023	06.02.2023	Flat Top rock	1		pers. comm. V. Pavel	flying
Sighting 2023	08.02.2023	bay east of Artigas station	1		present study	flying
Sighting 2024	15.12.2023	Flat Top rock	1		present study	fleying
Sighting 2024	28.10.2024	Flat Top rock	1		pers. comm. V. Pavel	landing at breeding site

D Appendix: Observations of vagrant and migrant birds

Table 14: Overview of all observations of the emperor penguin (*Aptenodytes forsteri*) documented in the Fildes Region and Maxwell Bay between 1979/80 and 2023/24 (modified after Braun et al., 2022, changes in bold).

Date	Number	Notes, observer	Source
09.10.1981	1		Bannasch, 1984
07.11.1981	1		Bannasch, 1984
25.03.1984	1	probably immature	Peter et al., 1988
04.05.1984	1	immature	Peter et al., 1988
18.-19.05., 03.06.1984	1	immatur, same individual	Peter et al., 1988
26.07.1984	1	immature	Peter et al., 1988
02.01.1985	1	immature	Peter et al., 1988
17.05.,29.06.,14.7.1985	1	immature, same individual	Rauschert et al., 1987
15.01.1986	1	immature	Rauschert et al., 1987
12./13.4.1986	2		Mönke & Bick, 1988
19.04.1986	1		Mönke & Bick, 1988
01.-03.06.1986	1	was captured, released on 3.6.	Mönke & Bick, 1988
24.01.1987	1		Mönke & Bick, 1988
21.02.1987	1	immature	Nadler & Mix, 1989
13.07.1987	1	immature	Nadler & Mix, 1989
14.07.1987	1	immature	Nadler & Mix, 1989
12.09.1987	1	immature	Nadler & Mix, 1989
09.07.1988	1	immature	Lange & Naumann, 1989
20.10.1988	2	adult	Lange & Naumann, 1989
30.12.1988	1	immature	Lange & Naumann, 1989
13.06.1989	1	immature	pers. comm. H. Grimm
29.09.1989	1	adult	pers. comm. H. Grimm
29.12.1989	1	remains of a freshly killed specimen	pers. comm. H. Grimm
06.01.2004	1	immature	Peter et al., 2008
22.02.2004	1	immature	Peter et al., 2008
11.-26.01.2006	1	immature	Peter et al., 2008
14.12.2008	1	dead specimen	Peter et al., 2013
16.10.2010	1	immature, Russian station personnel	Peter et al., 2013

Date	Number	Notes, observer	Source
28.05.2010	1	adult, Russian station personnel	Peter et al., 2013
20.01.2011	1		Peter et al., 2013
06.01.2012	1	immature	Braun et al., 2017a
10.06.2012	1	immature, O. Sakharov	Braun et al., 2017a
08.01.2013	1	immature, M. Villegas Vira	Braun et al., 2017a
26.11.2013	1	M. Villegas Vira	Braun et al., 2017a
18.12.2013	1		Braun et al., 2017a
28.09.2016	1	immature, I. Besotetscheski	Braun et al., 2020c
12.01.2017	1	immature	Braun et al., 2020c
14.01.2018	1	immature	Braun et al., 2020c
04.05.2018	1	immature, A. Loginov	Braun et al., 2022
16.-17.01.2019	2	immature	Braun et al., 2022
28.01.2019	1	immature	Braun et al., 2022
21.05.2019	1	immature, A. Filippov	Braun et al., 2022
16.12.2019	1	adult, A. L. Machado	Braun et al., 2022
02.-03.02., 05.02.2022	**1**	**adult**	**present study**
17.02.2022	**1**	**immature**	**present study**
18.01.2023	**1**	**immature, B. Mavlyudov**	**present study**
05.02.2023	**1**	**immature, appears weakened**	**present study**
28.01.2024	**1**	**immature**	**present study**
29.92.2024	**1**	**immature, A. Arsjuchin**	**present study**
09.04.2024	**1**	**immature, station personell Artigas**	**present study**

Table 15: **Overview of all observations of king penguin (*Aptenodytes patagonicus*) documented in the Fildes Region and Maxwell Bay between 1979/80 and 2023/24 (after Braun et al., 2022)**

Date	Number	Notes, observer	Source
03.-17.02.1988	1	adult, moulting	Lange & Naumann, 1990
13.05.1989	1	adult, lying on the bank, no normal reaction, movement severely restricted	pers. comm. H. Grimm
15.07.1989	1	adult, injured	pers. comm. H. Grimm
20.10.1989	1	adult	pers. comm. H. Grimm

Date	Number	Notes, observer	Source
15.11.1989	1	adult	pers. comm. H. Grimm
01.12.1989	1	adult, moulting	pers. comm. H. Grimm
17.12.1989	1	adult	pers. comm. H. Grimm
28.12.2000	1	A. Quellmalz	Braun, 2001
31.12.2004	1	A. Conteras Staeding	Peter et al., 2008
08.01.2005	1	dead specimen	Peter et al., 2008
16.02.2006	1	immature	Peter et al., 2008
27.12.2008	1	M. Kopp, S. Lisovski	Peter et al., 2013
Juni 2009	1	Russian station personnel	Peter et al., 2013
Juli 2009	1	Russian station personnel	Peter et al., 2013
19.12.2009	1	dead specimen	Peter et al., 2013
19.12.2009	1	dead specimen	Peter et al., 2013
21.01.2010	1	dead specimen	Peter et al., 2013
15.02.2010	1	dead specimen	Peter et al., 2013
21.02.2010	1	dead specimen	Peter et al., 2013
21.12.2011	1	dead specimen	Braun et al., 2017a
06.02.2012	1	dead specimen	Braun et al., 2017a
15.12.2016	1	adult	Braun et al., 2020c
01.02.2017	1	adult	Braun et al., 2020c
05.05.2018	1	immature, A. Loginov	Braun et al., 2020c
13.02.2023	**1**	**adult, O. Stanický, D. Jindra**	**present study**

Table 16: Overview of all observations of rockhopper penguin (*Eudyptes chrysocome*), Magellanic penguin (*Spheniscus magellanicus*) and macaroni penguin (*Eudyptes chrysolophus*) documented in the Fildes Region between 1979/80 and 2023/24 (modified after Braun et al., 2022, changes in bold)

Species	Date	Number	Notes, observer	Source
Spheniscus magellanicus	23.12.2016	1	immature, Ardley Island	Braun et al., 2020c
Eudyptes chrysocome	30.01.-03.02.2010	1	moulting, Ardley Island	Peter et al., 2013
	19.01.2011	1	immature, between chinstrap penguins, Drake coast	Peter et al., 2013

Species	Date	Number	Notes, observer	Source
Eudyptes chrysolophus	28.12.1979	1		Odening & Bannasch, 1980
	21.03.1981	1	adult, moulting, between chinstrap penguins, Drake coast	Bannasch, 1984
	15.01.1982	1	immatur	Bannasch, 1984
	30.12.1983	1	adult	Torres et al., 1986; Peter et al., 1988
	24.01.-26.02.1984	1	moulting, Exotic Point, in chinstrap penguin colony	Peter et al., 1988
	16.04.1984	1	adult	Peter et al., 1988
	02.-03.12.1984	1	Ardley Island, near gentoo penguin colony	Torres et al., 1986
	21.02.1985	1	Exotic Point, in chinstrap penguin colony, moulting	Peter et al., 1988
	13.&14.12.1985	1	adult, Ardley Island, between gentoo penguin colony, shows nest-building behaviour	Peter et al., 1988
	13.12.1984	1	Ardley Island	Rauschert et al., 1987
	26.-29.01.1986	1	Ardley Island, near chinstrap penguin colony	Torres et al., 1986
	15.11.1987		adult, Ardley Island, in mixed gentoo and chinstrap penguin nesting group	Nadler & Mix, 1989
	02.01.1989	2	immature, Ardley Island, in mixed gentoo and chinstrap penguin nesting group	Lange & Naumann, 1989
	15.01.1989	1	adult, between chinstrap penguins, Drake coast	Lange & Naumann, 1989
	15.01.1989	1	immatur, between chinstrap penguins, Drake coast	Lange & Naumann, 1989
	04.02.1989	1	immature, between chinstrap penguins, Drake coast	Lange & Naumann, 1989
	15.-17.03.1989	1	immature, moulting, Drake coast, between chinstrap and gentoo penguins	pers. comm. H. Grimm
	29.03.-04.04.1989	1	adult, moulting, Drake coast	pers. comm. H. Grimm
	04.01.2001	1	Ardley Island	Braun, 2001

Species	Date	Number	Notes, observer	Source
	16.-25.02.2005	1	moulting, Ardley Island, in penguin colony	Peter et al., 2008
	13.12.2005	1		Peter et al., 2008
	13.01.2010	1	A. Contreras Staeding	Peter et al., 2013
	23.02.2012	1	dead specimen	Braun et al., 2017a
	02.02.2015	1	Ardley Island	Braun et al., 2017a
	04.01.2016	1	A. Contreras Staeding	Braun et al., 2020c
	10.02.2016	1		Braun et al., 2020c
	21.02.2019	1	Ardley Island	Braun et al., 2022
	15.01.2021	1	northern Drake coast, A. L. Machado	Braun et al., 2022
	03.02.2021	1	Ardley Island, A. L. Machado	Braun et al., 2022
	10.01.2024	**1**	**Martin Point, Nelson Island**	**pers. comm. V. Pavel**
	07.02.2024	**1**	**Ardley Island**	**present study**
	08.02.2024	**1**	**Ardley Island**	**present study**

Table 17: Overview of all observations of the black-browed albatross (*Thalassarche melanophris*) documented between 1979/80 and 2023/24 in the Fildes Region and Maxwell Bay

Date	Number	Notes, observer	Source
19.01.1985	1	Maxwell Bay	Peter et al., 1988
14.03.1987	1	Maxwell Bay, near Barton Peninsula	Mönke & Bick, 1988
10.02.1989	1	Maxwell Bay, near Ardley Island, may have followed a ship	pers. comm. H. Grimm; Lange & Naumann, 1989
26.02.1989	2	Maxwell Bay, may have come by ship	pers. comm. H. Grimm; Lange & Naumann, 1989
30.03.1989	1	dead specimen, Drake coast	pers. comm. H. Grimm
26.03.1991	1	Maxwell Bay/Fildes Strait	Lumpe & Weidinger, 2000
09.03.1992	1	Maxwell Bay/Fildes Strait	Lumpe & Weidinger, 2000
27.03.1992	1	Maxwell Bay/Fildes Strait	Lumpe & Weidinger, 2000
05.04.1992	2	Maxwell Bay/Fildes Strait	Lumpe & Weidinger, 2000
05.12.2008	1	Maxwell Bay, near Barton Peninsula	Peter et al., 2013
26.12.2014	1	Fildes Strait	Braun et al., 2020c

Table 18: **Overview of all observations of Arctic tern (*Sterna paradisea*) documented in the Fildes Region between 1979/80 and 2023/24 (modified after Braun et al., 2022, changes in bold)**

Date	Number	Notes, observer	Source
11.10.1984	1	between adult *S. vittata*	Peter et al., 1988
20.10.1984	1	fishing together with *S. vittata*	Peter et al., 1988
21.10.1984	several	between adult *S. vittata*	Peter et al., 1988
12.11.1984	1	clear calls	Peter et al., 1988
12.01.1985	2	fishing, clear calls	Peter et al., 1988
20.11.1986	1 or 2		Mönke & Bick, 1988
24.11.1986	1		Mönke & Bick, 1988
29.12.1986	1		Mönke & Bick, 1988
22.12.2003	2		Peter et al., 2008
25.12.2004	1		Peter et al., 2008
28.12.2004	1		Peter et al., 2008
14.12.2005	28	M. Ritz	Peter et al., 2008
23.12.2005	35		Peter et al., 2008
05.11.2008	ca. 150	S. Lisovski	Peter et al., 2013
21.12.2008	1	leg. V. Pavel	Peter et al., 2013
06.01.2009	mind. 10		Peter et al., 2013
23.12.2009	7	in the swarm	Peter et al., 2013
08.01.2010	1		Peter et al., 2013
16.01.2011	3	fishing	Peter et al., 2013
24.11.2011	1	flying	Peter et al., 2013
01.12.2011	1	flying	Peter et al., 2013
17.12.2011	18	sitting	Peter et al., 2013
22.12.2011	1	flying	Peter et al., 2013
23.12.2011	1	flying	Peter et al., 2013
27.12.2011	5	sitting	Peter et al., 2013
02.01.2012	22	sitting	Peter et al., 2013
12.01.2012	6	flying, in association with native terns	Peter et al., 2013
27.12.2012	3	sitting	Peter et al., 2013
15.01.2013	14	foraging	Peter et al., 2013

Date	Number	Notes, observer	Source
03.02.2013	13	sitting	Peter et al., 2013
11.02.2013	21	sitting	Peter et al., 2013
29.01.2014	3	road between Chinese and Chilean stations, Z. Zhang	Peter et al., 2013
10.02.2014	6		Peter et al., 2013
16.02.2014	2	northern Maxwell Bay, near Bellingshausen Dome glacier	Peter et al., 2013
13.12.2014	1	Flat Top Peninsula	Peter et al., 2013
03.02.2015	3	in breeding colony of *Sterna vittata*	Peter et al., 2013
22.11.2016	2		Braun et al., 2020c
06.12.2016	1		Braun et al., 2020c
10.01.2018	1	flying	Braun et al., 2020c
18.12.2018	150	in breeding colony of *Sterna vittata*	Braun et al., 2022
03.01.2019	20		Braun et al., 2022
21.12.2019	2	flying	Braun et al., 2022
25.01.2020	1	flying	Braun et al., 2022
20.12.2021	**5**	**flying**	**present study**
29.12.2021	**ca. 40**	**in breeding colony of *Sterna vittata***	**present study**

Table 19: Overview of all cattle egret (*Bulbucis ibis*) observations documented in the Fildes Region between 1979/80 and 2023/24 (modified after Braun et al., 2022, changes in bold)

Date	Number	Notes, observer	Source
29.03.1985	1	freshly dead specimen	Zippel, 1987
Januar 1986	1	dead specimen	Zippel, 1987
17.04.1986	3	sitting, soaring	Mönke & Bick, 1990
19.04.1986	2	2 x dead specimen, 1 x severely weakened	Mönke & Bick, 1990
20.04.1986	1	dead specimen	Mönke & Bick, 1990
03.05.1986	10	dead specimen	Mönke & Bick, 1990
08.05.1986	4	taken on board by Soviet fishing research vessel near South Orkney, flown in from a flock of 100-150 ind.	Mönke & Bick, 1990
08.12.1986	1	dead specimen, in kelp gull nest	Mönke & Bick, 1990
07.01.1987	1	dead specimen, in kelp gull nest	Mönke & Bick, 1990

Date	Number	Notes, observer	Source
11.01.1987	1	dead specimen	Mönke & Bick, 1990
09.02.1987	1	dead specimen	Mönke & Bick, 1990
26.12.1990	1	Ardley Island	Lumpe & Weidinger, 2000
April 1991	1	dead specimen, Fildes Strait	Lumpe & Weidinger, 2000
01.05.1991	1	living, Fildes Strait	Lumpe & Weidinger, 2000
08.02.1992	1	Fildes Peninsula	Lumpe & Weidinger, 2000
Dezember 2000	1	dead specimen	Braun, 2001
23.01.2004	1	dead specimen	Peter et al., 2008
07.02.2004	1	dead specimen	Peter et al., 2008
14.02.2004	1	dead specimen	Peter et al., 2008
16.02.2004	1	dead specimen	Peter et al., 2008
10.12.2004	1	dead specimen at skua nest, Nebles Point	Peter et al., 2008
17.12.2004	1	dead specimen	Peter et al., 2008
18.12.2004	1	dead specimen	Peter et al., 2008
10.01.2005	1	dead specimen, wing remnant, M. Flores	Peter et al., 2008
26.01.2005	1	dead specimen, M. Flores	Peter et al., 2008
17.02.2005	1	dead specimen, Dart Island	Peter et al., 2008
19.02.2005	2	dead specimen at skua nest	Peter et al., 2008
02.03.2005	2	dead specimen	Peter et al., 2008
20.02.2006	1	dead specimen	Peter et al., 2008
14.12.2008	1	dead specimen, S. Lisovski, M. Kopp	Peter et al., 2008
15.01.2009	1	dead specimen	Peter et al., 2008
19.02.2010	1	dead specimen	Peter et al., 2008
28.11.2011	1	dead specimen	Peter et al., 2008
05.01.2012	1	dead specimen	Peter et al., 2008
22.03.2013	1	living, M. Xing	Braun et al., 2017a
05.02.2017	1	dead specimen	Braun et al., 2017a
08.02.2017	1	dead specimen	Braun et al., 2020c
11.02.2017	1	dead specimen	Braun et al., 2020c
12.01.2019	2	dead specimen	Braun et al., 2022
30.01.2020	1	dead specimen	Braun et al., 2022

Date	Number	Notes, observer	Source
13.02.2020	2	dead specimen	Braun et al., 2022
15.01.2023	**1**	**dead specimen**	**present study**
13.02.2023	**1**	**dead specimen**	**present study**
13.01.2024	**1**	**dead specimen**	**present study**
16.02.2024	**2**	**dead specimen**	**present study**

Table 20: Overview of all observations of the white-rumped sandpiper (*Calidris fuscicollis*) documented in the Fildes Region between 1979/80 and 2023/2024 (modified after Braun et al., 2022, changes in bold)

Date	Number	Notes, observer	Source
31.10.1981	3	1 of them caught and photographed	Bannasch, 1984; Bannasch et al., 1984
14.10.1985	6		Zippel, 1987
26.10.1985	3		Zippel, 1987
19.01.1986	8		Mönke & Bick, 1988
21.01.1987	4		Mönke & Bick, 1988; Nadler & Mix, 1989
28. & 29.01.1987	3	observed by Soviet zoologist Bulabizew, presumably the same individuals from 21.1.	Mönke & Bick, 1988
27.11.1987	10		Lange & Naumann, 1990
27.02.1988	8		Lange & Naumann, 1990
23.12.1988	1		Lange & Naumann, 1990
25.12.1988	4		Lange & Naumann, 1990
29.12.1988	4		Lange & Naumann, 1990
15.01.1989	1		Lange & Naumann, 1990
29.01.1989	1		pers. comm. H. Grimm
13.02.1989	3		Lange & Naumann, 1990
18.02.1989	3		Lange & Naumann, 1990
13.02.1990	3	escape distance ca. 25 m	pers. comm. H. Grimm
26.12.1990	1		Lumpe & Weidinger, 2000
08.02.1992	1		Lumpe & Weidinger, 2000
30.12.2000	12	Ardley Island, seen several times in quick succession	Braun, 2001
05.01.2001	10	Ardley Island	Braun, 2001

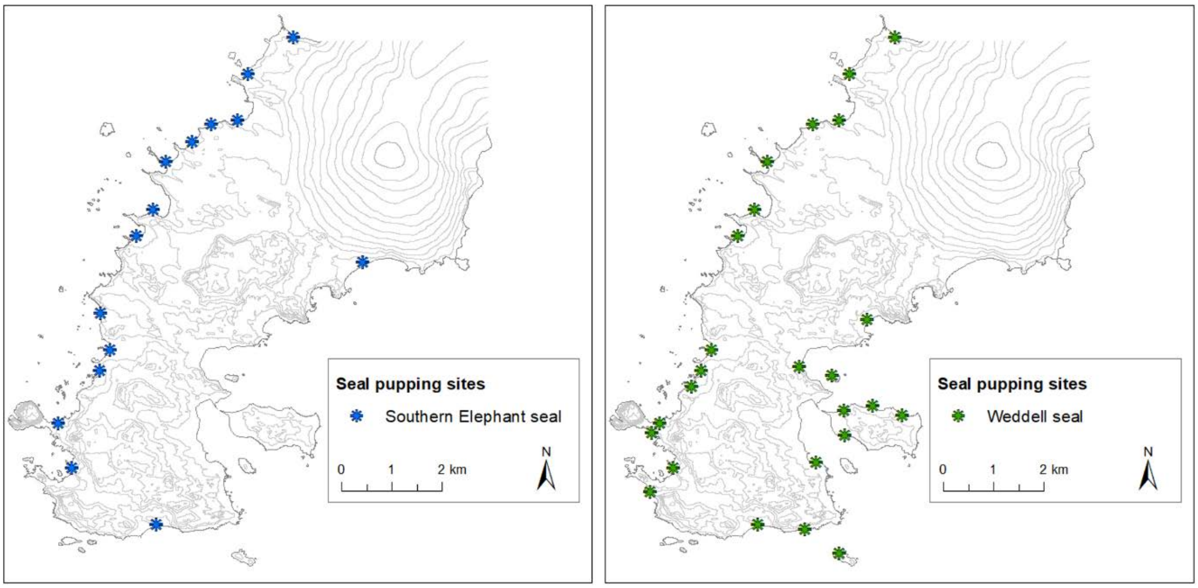
Date	Number	Notes, observer	Source
13.01.2005	8	M. Flores	Peter et al., 2008
22.01.2005	8	M. Flores	Peter et al., 2008
24.01.2005	4	M. Flores	Peter et al., 2008
29.01.2005	4	M. Flores	Peter et al., 2008
09.02.2005	7	M. Flores	Peter et al., 2008
06.02.2009	3		Peter et al., 2013
26.01.2010	1	running, then flying away	Peter et al., 2013
15.01.2011	1	seen and heard flying away	Peter et al., 2013
21.01.2011	1		Peter et al., 2013
29.11.2016	1		Braun et al., 2020c
16.12.2017	1		Braun et al., 2020c
15.12.2021	**1**	**P. Vaz-Canosa**	**present study**
08. & 10.02.2023	**2**	**E. Krojmal, B. de Feo, F. Teixero de Mello, J. P. Lozoya, leg. S. Lisovski**	**present study**

Table 21: Overview of all observations of the Baird's sandpiper (*Calidris bairdii*) documented in the Fildes Region between 1979/80 and 2023/2024

Date	Number	Notes, observer	Source
27.01.2022	1		present study
24.11.2022	6	S. Lisovski	present study

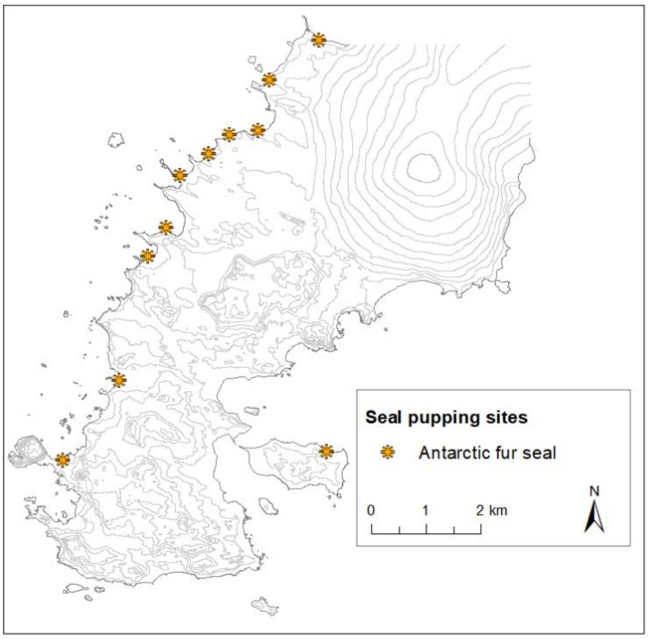
E Appendix: Seal pupping sites at the coast of the Fildes Peninsula und Ardley Islands

Figure 79: Location of southern elephant seal (*Mirounga leonina*) and Weddell seal (*Leptonychotes weddelii*) pupping sites on the coast of the Fildes Peninsula and Ardley Island, compiled from data from the 1984/85, 1986/87 - 1989/90, 2000/01, 2002/03 - 2006/07, 2008/09 - 2019/20 and 2021/22 – 2023/24 seasons)



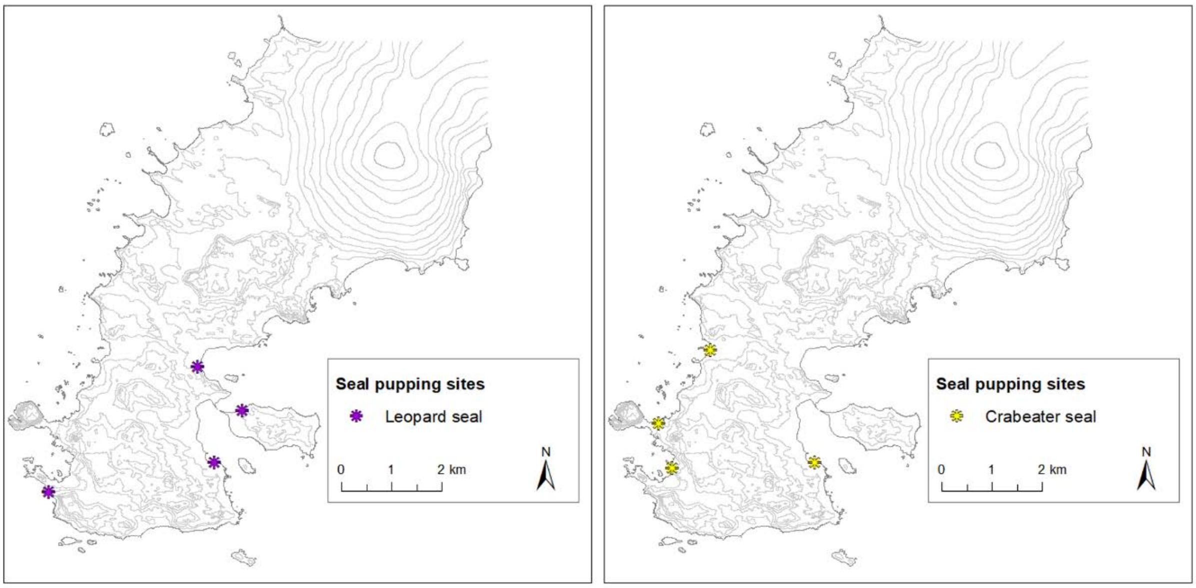
Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Figure 80: Location of the pupping sites of the Antarctic fur seal (*Arctocephalus gazella*) on the coast of the Fildes Peninsula and Ardley Island, compiled from data from the 1984/85, 1986/87 - 1989/90, 2000/01, 2002/03 - 2006/07, 2008/09 - 2019/20 and 2021/22 – 2023/24 seasons



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

Figure 81: **Location of the pupping sites of the leaopard seal and (*Hydrurga leptonyx*) und crabeater seal (*Lobodon carcinophaga*) on the coast of the Fildes Peninsula and Ardley Island, compiled from data from the 1984/85, 1986/87 - 1989/90, 2000/01, 2002/03 - 2006/07, 2008/09 - 2019/20 and 2021/22 – 2023/24 seasons)**



Source: Own illustration, Institute of Ecology and Evolution, Friedrich Schiller University Jena

F Appendix: Seabird monitoring in the ice-free areas of the Maxwell Bay

Table 22: Overview of breeding pair numbers of gentoo penguin (*Pygoscelis papua*), chinstrap penguin (*P. antarcticus*), southern giant petrel (*Macronectes giganteus*), Wilson's storm-petrel (*Oceanites oceanicus*) and black-bellied storm-petrel (*Fregetta tropica*) on Barton Peninsula, King George Island; if not the entire Barton peninsula was counted, a minimum value (min.) is given, data for ASPA No. 171 are in brackets (except penguins, as they exclusively breed withih the ASPA), Table modified after Braun et al., 2022, changes in bold

	P. papua		***P. antarcticus***		***M. giganteus***		***O. oceanicus***		***F. tropica***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1965/66	43	White in: Croxall & Kirkwood, 1979	2,100	White in: Croxall & Kirkwood, 1979						
1966/67					159	Araya & Arrieta, 1971				
1980/81	464	Jablonski, 1984	6,298	Jablonski, 1984						
1983/84					120-140	Lorenz, 1984				
1984/85	~500	Peter et al., 1988	2,500	Peter et al., 1988	140	Peter et al., 1988	~25	Peter et al., 1988	0	Peter et al., 1988
1985/86	600	Rauschert et al., 1987	2,500-3,000	Rauschert et al., 1987						
1986/87	**400-450/566**	**Mönke & Bick, 1988/Republic of Korea, 2024**	1,000/1,750	Mönke & Bick, 1988/ Shuford & Spear, 1988a	min. 20	Mönke & Bick, 1988				
1988/89					26	Lange & Naumann, 1989				
1989/90	682	Republic of Korea, 2014	1,161	Republic of Korea, 2014						

	P. papua		***P. antarcticus***		***M. giganteus***		***O. oceanicus***		***F. tropica***	
1992/93	500	Republic of Korea, 2014	2,000	Republic of Korea, 2014						
1994/95	1,001	Republic of Korea, 2014	2,924	Republic of Korea, 2014						
2000/01	1,333	Republic of Korea, 2014	3,117	Republic of Korea, 2014						
2001/02	1,441	Republic of Korea, 2014	3,008	Republic of Korea, 2014						
2006/07	1,719	Republic of Korea, 2008, 2024	2,961	Republic of Korea, 2008, 2024	**(9)**	**Republic of Korea, 2008, 2024**	**(19)**	**Republic of Korea, 2008, 2024**	**(0)**	**Republic of Korea, 2024**
2007/08	1,684	Republic of Korea, 2014	2,747	Republic of Korea, 2014						
2008/09	1,573	Republic of Korea, 2014	2,632	Republic of Korea, 2014						
2009/10	2,289	Republic of Korea, 2014	2,572	Republic of Korea, 2014						
2010/11	2,351	Republic of Korea, 2014	2,612	Republic of Korea, 2014	**23 (12)**	**Kim et al., 2011; J.-W. Jung, pers. comm.**	**(>50)**	**Republic of Korea, 2014**	**(>5)**	**J.-W. Jung, pers. comm.**
2011/12	2,212	Republic of Korea, 2014	3,161	Republic of Korea, 2014	**12 (11)**	**Republic of Korea, 2014; J.-W. Jung, pers. comm.**	**(0)**	**Republic of Korea, 2014**	**(0)**	**J.-W. Jung, pers. comm.**
2012/13	2,366	Republic of Korea, 2014	3,304	Republic of Korea, 2014	**62 (15)**	**J.-H. Kim & J.-W. Jung, pers. comm.**	**(>5)**	**Republic of Korea, 2014**	**(>1)**	**J.-W. Jung, pers. comm.**
2013/14	2,378	Republic of Korea, 2014	3,157	Republic of Korea, 2014	**36 (5)**	**Republic of Korea, 2014, 2024**	**(>10)**	**Republic of Korea, 2014**	**(0)**	**J.-W. Jung, pers. comm.**

	P. papua		***P. antarcticus***		***M. giganteus***		***O. oceanicus***		***F. tropica***	
2014/15	2,176	J.-W. Jung, pers. comm.	3,041	J.-W. Jung, pers. comm.	**23 (15)**	**J.-W. Jung, pers. comm.**	**(>1)**	**Republic of Korea, 2014**	**(>1)**	**J.-W. Jung, pers. comm.**
2015/16	2,112	W. Y. Lee, pers. comm.	3,352	W. Y. Lee, pers. comm.	**(10)**	**J.-W. Jung, pers. comm.**	**(>3)**	**J.-W. Jung, pers. comm.**	**(>1)**	**J.-W. Jung, pers. comm.**
2016/17	2,604	J.-W. Jung, pers. comm.	2,918	J.-W. Jung, pers. comm.	**(22)**	**J.-W. Jung, pers. comm.**	**(>2)**	**J.-W. Jung, pers. comm.**	**(>1)**	**J.-W. Jung, pers. comm.**
2017/18	2,309	J.-H. Kim, pers. comm.	2,867	J.-H. Kim, pers. comm.	18	J.-H. Kim, pers. comm.				
2018/19	2,224	Republic of Korea, 2019, 2024	2,388	Republic of Korea, 2019	**(15)**	**Republic of Korea, 2019**	**(>7)**	**Republic of Korea, 2019, 2024**	**(>10)**	**Republic of Korea, 2024**
2019/20	2,451	W. Y. Lee, pers. comm.	2,353	W. Y. Lee, pers. comm.						
2020/21	2,460	W. Y. Lee, pers. comm.	2,416	W. Y. Lee, pers. comm.						
2021/22	**2,482**	**H. J. Noh, pers. comm.**	**2,197**	**H. J. Noh, pers. comm.**						
2022/23	**2,405**	**J.-H. Kim, pers. comm.**	**2,165**	**J.-H. Kim, pers. comm.**						
2023/24	**2,669**	**Republic of Korea, 2024**	**2,271**	**Republic of Korea, 2024**	**(35)**	**Republic of Korea, 2024**	**(>132)**	**Republic of Korea, 2024**	**(>10)**	**Republic of Korea, 2024**

Table 23: Overview of breeding pair numbers of cape petrel (*Daption capense*), snowy sheathbill (*Chionis albus*), kelp gull (*Larus dominicanus*) and Antarctic tern (*Sterna vittata*) on Barton Peninsula, King George Island; if not the entire Barton peninsula was counted, a minimum value (min.) is given, data for ASPA No. 171 are in brackets (except penguins, as they exclusively breed withih the ASPA), Table modified after Braun et al., 2022, changes in bold

	D. capense		*C. albus*		*L. dominicanus*		*S. vittata*	
Season	BP	Source	BP	Source	BP	Source	BP	Source
1981/82			**min. 1**	**Bannasch et al., 1983**				
1982/83			**min. 1**	**Lorenz, 1984**				
1983/84			min. 1	Lorenz, 1984				
1984/85			1	Peter et al., 1988	5	Peter et al., 1988	20	Peter et al., 1988
1985/86					23	Rauschert et al., 1987		
1986/87			0	Mönke & Bick, 1988				
2006/07			**2**	**Republic of Korea, 2008, 2024**	**(6)**	**Republic of Korea, 2008, 2024**	**(41)**	**Republic of Korea, 2008, 2024**
2010/11			1	Republic of Korea, 2014	**(5)**	**Republic of Korea, 2014**	**(38)**	**Republic of Korea, 2014**
2011/12			1	Republic of Korea, 2014	**(5)**	**Republic of Korea, 2014**	**(21)**	**Republic of Korea, 2014**
2012/13	0	J.-H. Kim, pers. comm.	**(1)**	**J.-H. Kim, pers. comm.**	**min. 33 (3)**	**Lee et al., 2017b & J.-H. Kim, pers. comm.**	**(15)**	**Republic of Korea, 2014**
2013/14			**min. 2 (1)**	**Republic of Korea, 2014, 2024**	**min. 32 (0)**	**Lee et al., 2017b; Republic of Korea, 2024**	**>1 (0)**	**Republic of Korea, 2014, 2024; J.-W. Jung, pers. comm.**
2014/15	2	J.-W. Jung, pers. comm.	**(3)**	**J.-W. Jung, pers. comm.**	**36 (2)**	**Lee et al., 2017b; J.-W. Jung, pers. comm.**	**13 (6)**	**J.-W. Jung, pers. comm.**
2015/16			**(3)**	**J.-W. Jung, pers. comm.**	**min. 34 (0)**	**Lee et al., 2017b; J.-W. Jung, pers. comm.**	**(2)**	**J.-W. Jung, pers. comm.**

	D. capense		***C. albus***		***L. dominicanus***		***S. vittata***	
2016/17	0	J.-W. Jung, pers. comm.	**1 (1)**	**J.-W. Jung, pers. comm.**	**25 (0)**	**J.-W. Jung, pers. comm.**	**(>1)**	**J.-W. Jung, pers. comm.**
2017/18			**(3)**	**J.-H. Kim, pers. comm.**	**29**	**J.-H. Kim, pers. comm.**		
2018/19			**(5)**	**Republic of Korea, 2019, 2024**	**(0)**	**Republic of Korea, 2019, 2024**	**(4)**	**Republic of Korea, 2019, 2024**
2023/24			**(6)**	**Republic of Korea, 2024**	**(2)**	**Republic of Korea, 2024**	**(2)**	**Republic of Korea, 2024**

Table 24: **Overview of breeding pair numbers of brown skua (*Stercorarius antarcticus lonnbergi*), south polar skua (*S. maccormicki*), mixed and undefined skua pairs (*Stercorarius* spec.) on Barton Peninsula, King George Island; if not the entire Barton peninsula was counted, a minimum value (min.) is given, data for ASPA No. 171 are in brackets (except penguins, as they exclusively breed withih the ASPA), Table modified after Braun et al., 2022, changes in bold**

	S. a lonnbergi		***S. maccormicki***		***Stercorarius* mixed pairs**		***Stercorarius* spec.**	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1983/84							30	Lorenz, 1984
1984/85	10	Peter et al., 1988	1	Peter et al., 1988	0	Peter et al., 1988	0	Peter et al., 1988
1986/87							6-8	Mönke & Bick, 1988
2004/05	11	Kim et al., 2005	86	Kim et al., 2005	12	Kim et al., 2005	0	Kim et al., 2005
2005/06	11	Kim et al., 2009	100	Kim et al., 2009	11	Kim et al., 2009	0	Kim et al., 2009
2006/07	**12 (4)**	**Kim et al., 2009; Republic of Korea, 2019**	**82 (27)**	**Kim et al., 2009; Republic of Korea, 2024**	**9 (0)**	**Kim et al., 2009; J.-W. Jung, pers. comm.**	0	Kim et al., 2009
2007/08	4	Kim et al., 2009	25	Kim et al., 2009	1	Kim et al., 2009	0	Kim et al., 2009
2008/09			32	Kim & Ahn, 2011				

	S. a lonnbergi		***S. maccormicki***		***Stercorarius* mixed pairs**		***Stercorarius* spec.**	
2009/10			98	Kim & Ahn, 2011				
2010/11	**min. 10 (5)**	**Republic of Korea, 2011; J.-W. Jung, pers. comm.**	**min. 27 (18)**	**Republic of Korea, 2011; J.-W. Jung, pers. comm.**	**(1)**	**J.-W. Jung, pers. comm.**		
2011/12	**(5)**	**Republic of Korea, 2014; J.-W. Jung, pers. comm.**	**(22)**	**Republic of Korea, 2024; J.-W. Jung, pers. comm.**	**(1)**	**J.-W. Jung, pers. comm.**		
2012/13	**(4)**	**J.-H. Kim & J.-W. Jung, pers. comm.**	**1 (0)**	**J.-H. Kim & J.-W. Jung, pers. comm.**	0	J.-H. Kim, pers. comm.	0	J.-H. Kim, pers. comm.
2013/14	**(7)**	**Republic of Korea, 2019, 2024**	**2 (0)**	**J.-W. Jung & J.-W. Jung, pers. comm.**	0	J.-W. Jung, pers. comm.	0	J.-W. Jung, pers. comm.
2014/15	**10 (8)**	**J.-W. Jung, pers. comm.**	**11 (1)**	**J.-W. Jung, pers. comm.**	**1 (0)**	**J.-W. Jung, pers. comm.**	0	J.-W. Jung, pers. comm.
2015/16	**(6)**	**J.-W. Jung, pers. comm.**	**min. 100 (47)**	**W. Y. Lee & J.-W. Jung, pers. comm.**	**(3)**	**J.-W. Jung, pers. comm.**	0	J.-W. Jung, pers. comm.
2016/17	**18 (10)**	**J.-W. Jung, pers. comm.**	**127 (43)**	**J.-W. Jung, pers. comm.**	**18 (7)**	**J.-W. Jung, pers. comm.**	0	J.-W. Jung, pers. comm.
2017/18	9	J.-H. Kim, pers. comm.	153	J.-H. Kim, pers. comm.	3	J.-H. Kim, pers. comm.	0	J.-H. Kim, pers. comm.
2018/19	**(5)**	**Republic of Korea, 2019, 2024**	**(7)**	**Republic of Korea, 2019, 2024**				
2023/24	**(7)**	**Republic of Korea, 2024**	**(10)**	**Republic of Korea, 2024**				

Table 25: **Overview of breeding pair numbers of southern giant petrel (*Macronectes giganteus*), kelp gull (*Larus dominicanus*) and Antarctic tern (*Sterna vittata*) on Weaver Peninsula, King George Island**

	M. giganteus		***L. dominicanus***		***S. vittata***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1966/67	45	Araya & Arrieta, 1971				

	M. giganteus		***L. dominicanus***		***S. vittata***	
1984/85					20	Peter et al., 1988
2008/09	0	Braun et al., 2020c				
2012/13	0	J.-H. Kim, pers. comm.	5	J.-H. Kim, pers. comm.		
2013/14	0	J.-H. Kim, pers. comm.	8	J.-H. Kim, pers. comm.		

Table 26: **Overview of breeding pair numbers of brown skua (*Stercorarius antarcticus lonnbergi*), south polar skua (*S. maccormicki*) and their mixed and hybrid pairs on Weaver Peninsula, King George Island**

	S. a lonnbergi		***S. maccormicki***		***Stercorarius* mixed and hybrid pairs**	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
2004/05	2	Kim et al., 2005	15	Kim et al., 2005	1	Kim et al., 2005
2012/13	0	J.-H. Kim, pers. comm.	0	J.-H. Kim, pers. comm.	0	J.-H. Kim, pers. comm.
2013/14	0	J.-H. Kim, pers. comm.	0	J.-H. Kim, pers. comm.	0	J.-H. Kim, pers. comm.

Table 27: Overview of breeding pair numbers of kelp gull (*Larus dominicanus*), south polar skua (*Stercorarius maccormicki*), mixed, hybrid and undefined skua pairs (*Stercorarius* spec.) on Green Point, King George Island (modified after Braun et al., 2022, changes in bold)

	L. dominicanus		***S. maccormicki***		***Stercorarius* mixed & hybrid pairs**		***Stercorarius* spec.**		***S. vittata***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1984/85									15	Peter et al., 1988
1985/86	2	Rauschert et al., 1987					2	Rauschert et al., 1987		
2010/11	30	Braun et al., 2017a	0	Braun et al., 2017a	1	Braun et al., 2017a	1	Braun et al., 2017a		
2011/12			1	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a		
2012/13	26	Braun et al., 2017a	1	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a		
2013/14	14	Braun et al., 2017a	0	Braun et al., 2017a	1	Braun et al., 2017a	0	Braun et al., 2017a		
2014/15	16	Braun et al., 2017a	0	Braun et al., 2017a	1	Braun et al., 2017a	0	Braun et al., 2017a		
2015/16	5	Braun et al., 2020c	0	Braun et al., 2020c	0	Braun et al., 2020c	1	Braun et al., 2020c		
2016/17	28	Braun et al., 2020c	1	Braun et al., 2020c	1	Braun et al., 2020c	0	Braun et al., 2020c		
2017/18	35	Braun et al., 2020c	0	Braun et al., 2020c	0	Braun et al., 2020c	0	Braun et al., 2020c		
2018/19			1	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022
2019/20	11	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022
2021/22	**4**	**present study**	**1**	**present study**	**0**	**present study**	**0**	**present study**	**0**	**present study**
2022/23	**7**	**present study**	**0**	**present study**	**0**	**present study**	**0**	**present study**	**0**	**present study**
2023/24	**5**	**present study**	**0**	**present study**	**0**	**present study**	**0**	**present study**	**0**	**present study**

Table 28: **Overview of breeding pair numbers of gentoo penguin (*Pygoscelis papua*), Adélie penguin (*P. adeliae*), chinstrap penguin (*P. antarcticus*) and king penguin (*Aptenodytes patagonicus*) on Potter Peninsula, King George Island (modified after Braun et al., 2022, changes in bold)**

	P. papua		***P. adeliae***		***P. antarcticus***		***A. patagonicus***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1965/66	2,920	White in: Croxall & Kirkwood, 1979	6,440	White in: Croxall & Kirkwood, 1979	2,500	White in: Croxall & Kirkwood, 1979		
1966/67	ca. 1,500	Araya & Arrieta, 1971	ca. 7,500	Araya & Arrieta, 1971	ca. 1,100	Araya & Arrieta, 1971		
1971/72	ca. 1,000	Müller-Schwarze & Müller-Schwarze, 1975	ca. 18,000	Müller-Schwarze & Müller-Schwarze, 1975	ca. 200	Müller-Schwarze & Müller-Schwarze, 1975		
1980/81	2,584	Jablonski, 1984	18,412	Jablonski, 1984	495	Jablonski, 1984		
1984/85	ca. 1,900	Peter et al., 1988	ca. 17,000	Peter et al., 1988	350	Peter et al., 1988		
1985/86	2,500	Rauschert et al., 1987	16,000-17,000	Rauschert et al., 1987	500	Rauschert et al., 1987		
1986/87	750-1,000	Shuford & Spear, 1988b			75-100	Shuford & Spear, 1988a, b		
1987/88	900	Nadler & Mix, 1989	14,100/ 15,491	Mönke & Bick, 1988/Aguirre, 1995				
1988/89	2,325	Aguirre, 1995	14,554	Aguirre, 1995	265	Aguirre, 1995		
1995/96	2,236	Carlini et al., 2009	9,087	Carlini et al., 2009				
2000/01	3,083	Juáres et al., 2020	ca. 7,300	Schuster, 2010	present	Schuster, 2010		
2001/02	**3,223**	**Juáres et al., 2020**	ca. 5,500	Schuster, 2010	present	Schuster, 2010		
2002/03	**3,778**	**Juáres et al., 2020**			45	Silvestro et al., 2015		

	P. papua		***P. adeliae***		***P. antarcticus***		***A. patagonicus***	
2003/04					10	Silvestro et al., 2015		
2004/05	**3,782**	**Juáres et al., 2020**						
2005/06	**3,585**	**Juáres et al., 2020**						
2006/07	**3,764/ 3,815**	**Carlini et al., 2009/Juáres et al., 2020**	3,412	Carlini et al., 2009				
2007/08			2,003 (estimation)	Juáres, 2013				
2008/09	4,321	Juáres et al., 2020	3,910	Juáres, 2013				
2009/10	3,192	Juáres et al., 2020	2,938	Juáres, 2013	0	Schuster, 2010		
2010/11	4,613	Juáres et al., 2020	3,426	Juáres, 2013				
2011/12	**3,934**	**Juáres et al., 2020**	3,254	Juáres, 2013			**1**	**Juáres et al., 2023**
2012/13	**4,105**	**Juáres et al., 2020**					**1**	**Juáres et al., 2023**
2013/14	**4,991**	**Juáres et al., 2020**	3,703	Juáres, 2013	1	Silvestro et al., 2015	**1**	**Juáres et al., 2023**
2014/15	**4,864**	**Juáres et al., 2020**					**1**	**Juáres et al., 2023**
2015/16	**4,413**	**Juáres et al., 2020**					**1**	**Juáres et al., 2023**
2016/17	**5,148**	**Juáres et al., 2020**					**1**	**Juáres et al., 2023**
2017/18	**4,419**	**Juáres et al., 2020**					**1**	**Juáres et al., 2023**
2018/19	5,383	Juáres et al., 2020					**1**	**Juáres et al., 2023**
2019/20							**1**	**Juáres et al., 2023**
2020/21							**1**	**Juáres et al., 2023**
2021/22							**0**	**Juáres et al., 2023**

Table 29: **Overview of breeding pair numbers of southern giant petrel (*Macronectes giganteus*), cape petrel (*Daption capense*), Wilson's storm-petrel (*Oceanites oceanicus*) and black-bellied storm-petrel (*Fregetta tropica*) on Potter Peninsula, King George Island; If not the entire peninsula was counted, a minimum value (min.) is given (* recording only within the ASPA).**

	M. giganteus		***D. capense***		***O. oceanicus***		***F. tropica***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1958/59	30	Patterson et al., 2008						
1966/67	78 / 95	Araya & Arrieta, 1971 / Patterson et al., 2008			ca. 530	Araya & Arrieta, 1971	0	Araya & Arrieta, 1971
1969/70	39	Patterson et al., 2008						
1983/84	ca. 120	Lorenz, 1984						
1984/85	ca. 59	Peter et al., 1988	0	Peter et al., 1988	ca. 100	Peter et al., 1988	0	Peter et al., 1988
1985/86			3	Rauschert et al., 1987	82	Rauschert et al., 1987		
1988/89	75	Aguirre, 1995	0	Aguirre, 1995	100-200	Aguirre, 1995	present	Aguirre, 1995
1995/96	78	Patterson et al., 2008			1,520-2,280	Hahn et al., 1998	160-213	Hahn et al., 1998
1997/98	46	Hahn et al., 1998						
2006/07	87	ACAP, 2024a						
2010/11	min. 44	Republic of Korea, 2011						

Table 30: **Overview of breeding pair numbers of snowy sheathbill (*Chionis albus*), kelp gull (*Larus dominicanus*), Antarctic tern (*Sterna vittata*) and blue-eyed shag (*Leucocarbo bransfieldensis)* on Potter Peninsula, King George Island; If not the entire peninsula was counted, a minimum value (min.) is given; for blue-eyed shag, the colony on the offshore rock Low Rock was counted after 1990.**

	C. albus		***L. dominicanus***		***S. vittata***		***L. bransfieldensis***	
Season	**BP**	**Sources**	**BP**	**Sources**	**BP**	**Sources**	**BP**	**Sources**
1966/67	ca. 30	Araya & Arrieta, 1971	min. 10	Araya & Arrieta, 1971				
1984/85	12	Peter et al., 1988	25	Peter et al., 1988	55	Peter et al., 1988	2	Peter et al., 1988
1985/86	14	Rauschert et al., 1987	15	Rauschert et al., 1987				
1987/88	6	Favero, 1993			358	Favero, 1994		
1988/89	15	Aguirre, 1995	44	Aguirre, 1995	100-200	Aguirre, 1995	present (Low Rock)	Aguirre, 1995
1991/92							8 (Low Rock)	Casaux & Barrera-Oro, 2006
1992/93							7 (Low Rock)	Casaux & Barrera-Oro, 2006
1993/94							8 (Low Rock)	Casaux & Barrera-Oro, 2006
1997/98					ca. 250	Hahn et al., 1998	present	Hahn et al., 1998
1998/99							5 (Low Rock)	Casaux & Barrera-Oro, 2006
1999/2000							3 (Low Rock)	Casaux & Barrera-Oro, 2006
2000/01							1 (Low Rock)	Casaux & Barrera-Oro, 2006

Table 31: **Overview of breeding pair numbers of brown skua (*Stercorarius antarcticus lonnbergi*), south polar skua (*S. maccormicki*), hybrid, mixed and undefined skua pairs (*Stercorarius* spec.) on Potter Peninsula, King George Island; if the entire peninsula was not counted, a minimum value (min.) is given (* recording only within the ASPA).**

	S. a lonnbergi		***S. maccormicki***		***Stercorarius* mixed & hybrid pairs**		***Stercorarius* spec.**	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1966/67	8 (incl. 1 trio)	Araya & Arrieta, 1971	1	Araya & Arrieta, 1971				
1984/85	6	Peter et al., 1988	1	Peter et al., 1988	0	Peter et al., 1988	0	Peter et al., 1988
1985/86							6	Rauschert et al., 1987
1988/89	20	Aguirre, 1995	2	Aguirre, 1995	0	Aguirre, 1995	0	Aguirre, 1995
1993/94	35	Hahn et al., 1998	40	Hahn et al., 1998	10	Hahn et al., 1998	0	Hahn et al., 1998
1994/95	29	Hahn et al., 1998	41	Hahn et al., 1998	7	Hahn et al., 1998	0	Hahn et al., 1998
1995/96	31	Hahn & Peter, 2003	13	Hahn & Peter, 2003	3	Hahn & Peter, 2003	0	pers. comm. M. Ritz
1997/98	26	Hahn et al., 1998	44	Hahn et al., 1998	13	Hahn et al., 1998	0	Hahn et al., 1998
1998/99	26	Hahn & Peter, 2003	45	Hahn & Peter, 2003	7	Hahn & Peter, 2003	0	M. Ritz, pers. comm.
1999/2000	30	Hahn & Peter, 2003	44	Hahn & Peter, 2003	17	Hahn & Peter, 2003	0	M. Ritz, pers. comm.
2000/01	32	Hahn & Peter, 2003	54	Hahn & Peter, 2003	16	Hahn & Peter, 2003	0	M. Ritz, pers. comm.
2001/02	35	Hahn et al., 2007	63	Hahn et al., 2007	14	Hahn et al., 2007	0	M. Ritz, pers. comm.
2002/03	29	Hahn et al., 2007	33	Hahn et al., 2007	8	Hahn et al., 2007	0	M. Ritz, pers. comm.
2003/04	14	Hahn et al., 2007	49	Hahn et al., 2007	4	Hahn et al., 2007	0	M. Ritz, pers. comm.
2004/05	34	Krietsch et al., 2016	68	Krietsch et al., 2016	17	Krietsch et al., 2016	4	M. Kopp, pers. comm.
2005/06	28	Krietsch et al., 2016	76	Krietsch et al., 2016	17	Krietsch et al., 2016	3	M. Kopp, pers. comm.

	S. a lonnbergi		***S. maccormicki***		***Stercorarius* mixed & hybrid pairs**		***Stercorarius* spec.**	
2006/07	35	Krietsch et al., 2016	81	Krietsch et al., 2016	18	Krietsch et al., 2016	1	S. Lisovski, pers. comm.
2010/11	min. 6*	Republic of Korea, 2011	min. 2*	Republic of Korea, 2011	0*	Republic of Korea, 2011	0*	Republic of Korea, 2011
2012/13	11	Graña Grilli, 2014	0	Graña Grilli, 2014	0	Graña Grilli, 2014	0	Graña Grilli, 2014
2013/14	14	Graña Grilli, 2014	0	Graña Grilli, 2014	0	Graña Grilli, 2014	0	Graña Grilli, 2014

Table 32: Overview of breeding pair numbers of gentoo penguin (*Pygoscelis papua*), Adélie penguin (*P. adeliae*), chinstrap penguin (*P. antarcticus*), southern giant petrel (*Macronectes giganteus*) and cape petrel (*Daption capense*) at Duthoit Point, Nelson Island.

	P. papua		***P. adeliae***		***P. antarcticus***		***M. giganteus***		***D. capense***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1985/86	ca. 5,000	Rauschert et al., 1987	0	Rauschert et al., 1987	*(3,000-4,000)*	*(Rauschert et al., 1987 = presumably incorrect information)*	194	Rauschert et al., 1987	5	Rauschert et al., 1987
1986/87	1,700/800-850	Mönke & Bick, 1988 / Shuford & Spear, 1988b	0	Mönke & Bick, 1988	0	Mönke & Bick, 1988	54	Mönke & Bick, 1988	0	Mönke & Bick, 1988
1993/94	1,828	Coria et al., 1995	1	Coria et al., 1995	0	Coria et al., 1995	118	Coria et al., 1995	14	Coria et al., 1995
2012/13	ca. 1,800	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a	ca. 55	Braun et al., 2017a		
2017/18	2,382	Braun et al., 2020c	0	Braun et al., 2020c	0	Braun et al., 2020c	22	Braun et al., 2020c	0	Braun et al., 2020c

	P. papua		***P. adeliae***		***P. antarcticus***		***M. giganteus***		***D. capense***	
2018/19	ca. 2,860	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022	13	Braun et al., 2022	0	Braun et al., 2022

Table 33: **Overview of breeding pair numbers of Wilson's storm-petrel (*Oceanites oceanicus*), black-bellied storm-petrel (*Fregetta tropica*), snowy sheathbill (*Chionis albus*), kelp gull (*Larus dominicanus*) and Antarctic tern (*Sterna vittata*) at Duthoit Point, Nelson Island; if not the entire peninsula was counted, a minimum value (min.) is given.**

	O. oceanicus		***F. tropica***		***C. albus***		***L. dominicanus***		***S. vittata***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1985/86	ca. 150	Rauschert et al., 1987			2 / 3	Rauschert et al., 1987 / Mönke & Bick, 1988	0	Rauschert et al., 1987	22	Rauschert et al., 1987
1986/87					1	Mönke & Bick, 1988				
1993/94	min. 2	Coria et al., 1995	min. 4	Coria et al., 1995	2	Coria et al., 1995	18	Coria et al., 1995	29	Coria et al., 1995
2012/13					1	Braun et al., 2017a	5	Braun et al., 2017a		
2017/18	min. 2	Braun et al., 2020c			0	Braun et al., 2020c	29	Braun et al., 2020c	20	Braun et al., 2020c
2018/19					1	Braun et al., 2022	8	Braun et al., 2022	0	Braun et al., 2022

Table 34: **Overview of breeding pair numbers of brown skua (*Stercorarius antarcticus lonnbergi*), south polar skua (*S. maccormicki*), mixed, hybrid und undefined skua pairs (*Stercorarius* spec.) and blue-eyed shags (*Leucocarbo bransfieldensis*) at Duthoit Point, Nelson Island; if not the entire peninsula was counted, a minimum value (min.) is given.**

	S. a. lonnbergi		***S. maccormicki***		***Stercorarius* mixed & hybrid pairs**		***Stercorarius* spec.**		***L. bransfieldensis***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1985/86							4	Rauschert et al., 1987		
1986/87									96/100	Mönke & Bick, 1988 / Shuford & Spear, 1988b
1990/91									163	Casaux & Barrera-Oro, 1993
1992/93									140	Casaux & Barrera-Oro, 2006
1993/94	5	Coria et al., 1995	1	Coria et al., 1995	0	Coria et al., 1995	0	Coria et al., 1995	133	Coria et al., 1995
1994/95									120	Casaux, 1998
1995/96									104	Casaux, 1998
1996/97									79	Casaux, 1998
1997/98									73	Casaux & Barrera-Oro, 2006
1998/99									77	Casaux & Barrera-Oro, 2006
1999/2000									64	Casaux & Barrera-Oro, 2006
2000/01									66	Casaux & Barrera-Oro, 2006
2001/02									48	Casaux & Barrera-Oro, 2006
2002/03									42	Casaux & Barrera-Oro, 2006
2003/04									36	Casaux & Barrera-Oro, 2006

	S. a. lonnbergi		***S. maccormicki***		***Stercorarius* mixed & hybrid pairs**		***Stercorarius* spec.**		***L. bransfieldensis***	
2004/05									32	Casaux & Barrera-Oro, 2006
2006/07	2	S. Lisovski, pers. comm.	3	S. Lisovski, pers. comm.	2	S. Lisovski, pers. comm.	2	S. Lisovski, pers. comm.		
2008/09									41	Casaux & Barrera-Oro, 2016
2012/13	2	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a	ca. 55	Braun et al., 2017a
2017/18	3	Braun et al., 2020c	7	Braun et al., 2020c	3	Braun et al., 2020c	14	Braun et al., 2020c	69	Braun et al., 2020c
2018/19	2	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022	81	Braun et al., 2022

Table 35: Overview of breeding pair numbers of southern giant petrel (*Macronectes giganteus*), cape petrel (*Daption capense*), Wilson's storm-petrel (*Oceanites oceanicus*) and black-bellied storm-petrel (*Fregetta tropica*) on Stansbury Peninsula, Nelson Island; if not the entire peninsula was counted, a minimum value (min.) is given; later count date marked by * (modified after Braun et al., 2022, changes in bold).

	M. giganteus		***D. capense***		***O. oceanicus***		***F. tropica***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1979/80	11	Bannasch & Odening, 1981	ca. 75	Bannasch & Odening, 1981				
1984/85	47	Peter et al., 1988	10	Peter et al., 1988	2	Peter et al., 1988	1	Peter et al., 1988
1985/86	22	Rauschert et al., 1987	25	Rauschert et al., 1987				
1986/87	15-20	Mönke & Bick, 1988						
1987/88	min. 9	Lange & Naumann, 1989						

	M. giganteus		***D. capense***		***O. oceanicus***		***F. tropica***	
1988/89	24	Lange & Naumann, 1989						
1990/91	25	Lumpe & Weidinger, 2000						
1991/92	29	Lumpe & Weidinger, 2000	ca. 800	Lumpe & Weidinger, 2000	100-500	Lumpe & Weidinger, 2000	6-10	Lumpe & Weidinger, 2000
2002/03	32	S. Pfeiffer, pers. comm.						
2008/09	63	Braun et al., 2017a						
2009/10	65	Braun et al., 2017a						
2010/11	50	Braun et al., 2017a						
2011/12	61	Braun et al., 2017a						
2012/13	48	Braun et al., 2017a	min. 8	Braun et al., 2017a				
2013/14	min. 71*	Braun et al., 2017a						
2014/15	61	Braun et al., 2017a	min. 2	Braun et al., 2017a				
2015/16	min. 52 *	Braun et al., 2020c						
2016/17	68	Braun et al., 2020c						
2017/18	51	Braun et al., 2020c						
2018/19	56	Braun et al., 2022						
2021/22	**26**	**present study**						
2022/23	**31**	**pers. comm. V. Pavel**						
2023/24	**15**	**pers. comm. V. Pavel**						

Table 36: **Overview of breeding pair numbers of snowy sheathbill (*Chionis albus*), kelp gull (*Larus dominicanus*) and Antarctic tern (*Sterna vittata*) on Stansbury Peninsula, Nelson Island; if not the entire peninsula was counted, a minimum value (min.) is given; (modified after Braun et al., 2022, changes in bold); * deviating methodology**

	C. albus		***L. dominicanus***		***S. vittata***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1984/85	1	Peter et al., 1988	0	Peter et al., 1988	0	Peter et al., 1988
1985/86	1	Rauschert et al., 1987	5	Rauschert et al., 1987	28	Rauschert et al., 1987
1991/92	0	Lumpe & Weidinger, 2000	8	Lumpe & Weidinger, 2000	100	Lumpe & Weidinger, 2000
2009/10	0	Braun et al., 2017a				
2010/11	0	Braun et al., 2017a			min. 6	Braun et al., 2017a
2012/13	0	Braun et al., 2017a	min. 1	Braun et al., 2017a		
2013/14	0	Braun et al., 2017a				
2014/15	0	Braun et al., 2017a				
2017/18	0	Braun et al., 2020c			min. 2	Braun et al., 2020c
2018/19	0	Braun et al., 2022				
2021/22			**1**	**present study**		

	C. albus		***L. dominicanus***		***S. vittata***	
2022/23						
2023/24			**~ 25 ***	**pers. comm. V. Pavel**	**30 ***	**pers. comm. V. Pavel**

Table 37: Overview of breeding pair numbers of brown skua (*Stercorarius antarcticus lonnbergi*), south polar skua (*S. maccormicki*), mixed, hybrid and undefined skua pairs (*Stercorarius* spec.) on Stansbury Peninsula, Nelson Island; if not the entire peninsula was counted, a minimum value (min.) is given; (modified after Braun et al., 2022, changes in bold). * deviating methodology and extended area

	S. a lonnbergi		***S. maccormicki***		***Stercorarius* mixed & hybrid pairs**		***Stercorarius* spec.**	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1985/85	0	Peter et al., 1988	0	Peter et al., 1988	0	Peter et al., 1988	0	Peter et al., 1988
1985/86							5	Rauschert et al., 1987
1989/90							6	H. Grimm, pers. comm.
1991/92	9	Lumpe & Weidinger, 2000	20	Lumpe & Weidinger, 2000	1	Lumpe & Weidinger, 2000	0	Lumpe & Weidinger, 2000
2009/10	0	Braun et al., 2017a	1	Braun et al., 2017a	0	Braun et al., 2017a	4	Braun et al., 2017a
2010/11	0	Braun et al., 2017a	12	Braun et al., 2017a	0	Braun et al., 2017a	15	Braun et al., 2017a
2012/13	0	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a
2013/14	0	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a	0	Braun et al., 2017a
2014/15	min. 1	Braun et al., 2017a						
2015/16							min. 1	Braun et al., 2020c
2016/17	min. 1	Braun et al., 2020c					min. 1	Braun et al., 2020c
2018/19	1	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022

	S. a lonnbergi		***S. maccormicki***		***Stercorarius* mixed & hybrid pairs**		***Stercorarius* spec.**	
2021/22								
2022/23								
2023/24			**2 ***	**pers. comm. I. Parnikoza**	**11 ***	**pers. comm. I. Parnikoza**		

Table 38: **Overview of breeding pair numbers of gentoo penguin (*Pygoscelis papua*), chinstrap penguin (*P. antarcticus*), southern giant petrel (*Macronectes giganteus*), cape petrel (*Daption capense*) and Wilson's storm-petrel (*Oceanites oceanicus*) on Cabo Cariz, Nelson Island**

	P. papua		***P. antarcticus***		***M. giganteus***		***D. capense***		***O. oceanicus***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1984/85	0	Peter et al., 1988	1,000	Peter et al., 1988	17	Peter et al., 1988	10	Peter et al., 1988	ca. 25	Peter et al., 1988
1985/86	*100*	*Rauschert et al., 1987 (presumably incorrect information)*	8,000-10,000	Rauschert et al., 1987	34	Rauschert et al., 1987	55	Rauschert et al., 1987	65	Rauschert et al., 1987
1986/87	0	Mönke & Bick, 1988	300/250-500	Mönke & Bick, 1988/Shuford & Spear, 1988b	11	Mönke & Bick, 1988				
1991/92	0	Lumpe & Weidinger, 2000	800	Lumpe & Weidinger, 2000	8	Lumpe & Weidinger, 2000	ca. 300	Lumpe & Weidinger, 2000	10-50	Lumpe & Weidinger, 2000
2015/16	0		549	Braun et al., 2020c						
2016/17	0		791 ± 48	Pfeifer et al., 2019						
2017/18	0	Braun et al., 2020c			30	Braun et al., 2020c	4	Braun et al., 2020c		

	P. papua		***P. antarcticus***		***M. giganteus***		***D. capense***		***O. oceanicus***	
2018/19	0	Braun et al., 2022	ca. 330	Braun et al., 2022	29	Braun et al., 2022	0	Braun et al., 2022		
2019/20	0	Braun et al., 2022	ca. 350	Braun et al., 2022	39	Braun et al., 2022	0	Braun et al., 2022		

Table 39: **Overview of breeding pair numbers of snowy sheathbill (*Chionis albus*), kelp gull (*Larus dominicanus*), brown skua (*Stercorarius antarcticus lonnbergi*), undefined skua pairs (*Stercorarius* spec.) and Antarctic tern (*Sterna vittata*) on Cabo Cariz, Nelson Island**

	C. albus		***L. dominicanus***		***S. a. lonnbergi***		***Stercorarius* spec.**		***S. vittata***	
Season	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**	**BP**	**Source**
1984/85	1	Peter et al., 1988	30	Peter et al., 1988	1	Peter et al., 1988	0	Peter et al., 1988	0	Peter et al., 1988
1985/86	4	Rauschert et al., 1987	12	Rauschert et al., 1987			2	Rauschert et al., 1987	13	Rauschert et al., 1987
1986/87	0	Mönke & Bick, 1988								
1991/92	1	Lumpe & Weidinger, 2000	1	Lumpe & Weidinger, 2000	3	Lumpe & Weidinger, 2000	0	Lumpe & Weidinger, 2000	0	Lumpe & Weidinger, 2000
2017/18	0	Braun et al., 2020c	9	Braun et al., 2020c	1	Braun et al., 2020c	1	Braun et al., 2020c	min. 1	Braun et al., 2020c
2018/19	0	Braun et al., 2022	min. 1	Braun et al., 2022	0	Braun et al., 2022	1	Braun et al., 2022	0	Braun et al., 2022
2019/20	0	Braun et al., 2022	0	Braun et al., 2022	1	Braun et al., 2022	0	Braun et al., 2022	0	Braun et al., 2022

G Appendix: Publications and contributions by project staff in the reporting period (2021 – 2024)

G.1 Publications

Martínez Rendón, C., **Braun, C.**, Kappelsberger, M. Boy, J. Casanova-Katny, A. Glaser, K. Dumack, K. (submitted): Enhancing Microbial Predator-Prey Detection with Network and Trait-Based Analyses. Microbiome

Bennett Lazo, B., Berazay, B., Muñoz, G., Ariyama, N., Enciso, N., **Braun, C.**, Krüger, L., Barták, M., González-Aravena, M., Neira, V. (2024): Confirmation of Highly Pathogenic Avian Influenza (HPAI) H5N1 Associated with an Unexpected Mortality Event in South Polar Skuas (*Stercorarius maccormicki*) during 2023-2024. Surveillance Activities in Antarctica. Frontiers in Veterinary Science, 11: doi: 10.3389/fvets.2024.1423404

Lisovski, S., Günther, A., Dewar, M., Ainley, D., Aldunate, F., Arce, R., Ballard, G., Bauer, S., Belliure, J., Banyard, A.C., Boulinier, T., Bennison, A., **Braun, C.**, Cary, C., Catry, P., Clessin, A., Connan, M., Correia, E., Cox., E., Cristina, J., Elrod, M., Emerit, J., Ferreiro, I., Fowler, Z., Gamble, A., Grandeiro, J.P., Hurtado, J., Jongsomjit, D., Lesage, C., Lejeune, M., Kuepfer, A., Lescroël, A., Li, A., McDonald, I.R., Menéndez-Blázquez, J., Morandini, V., Moratorio, G., Militão, T., Moreno, P., Perbolianachis, P., Pennycook, J., Raslán, M., Reid, S.M., Richards-Babbage, R., Schmidt, A.E., Sander, M.M., Smyth, L., Soutullo, A., Streith, L., Tornos, J., Varsani, A., Herzschuh, U., Beer, M., Wille, M. (2024): Unexpected delayed incursion of highly pathogenic avian influenza H5N1 (clade 2.3.4.4b) in the Antarctic region. Influenza and other respiratory viruses, 18: e70010

Braun, C., Grämer, H., Peter, H.-U. (2023): Records of vagrant and visitor bird species in the Fildes Region, King George Island, Maritime Antarctic, between 1980 and 2023. Ukrainian Antarctic Journal, 21: 210-219

Ausems, A.N.M.A., Kuepper, N.D., Archuby, D., **Braun, C.**, Gębczyński, A., Gladbach, A., Hahn, S., Jadwiszczak, P., Kraemer, P. Libertelli, M.M., Lorenz, S., Richter, B., Ruß, A., Schmoll, T., Thébault, J., Thorn, S., Turner, J., Wojczulanis-Jakubas, K., Jakubas, D., Quillfeldt, P. (2023): Where have all the petrels gone? 40 Years of Environmental Change and Population Dynamics of Wilson`s Storm-Petrels. Polar Biology, 46: 655-672

Lenzi, J., Bresesti, F., Lozoya, J.P., De Feo, B., Krojmal, E., Lacerot, G., **Braun, C.**, Teixeira de Mello, F. (2022): Diet and debris ingestion of skuas on Fildes Peninsula, King George Island, Antarctica Marine Pollution Bulletin, 185: 114211

Kuepper, N.D., Böhm, L., **Braun, C.**, Bustamante, P., Düring, R.-A., Libertelli, M.M., Quillfeldt, P. (2022): Persistent organic pollutants and mercury in a colony of Antarctic seabirds: higher concentrations in 1998, 2001, and 2003 compared to 2014 to 2016. Polar Biology, 45: 1229-1245

G.2 Conference presentations (lecture)

Braun, C. (2024): Ecological long-term research of the Jena University, Germany, Scientific Meeting at Great Wall Station, King George Island, Antarctic

De Feo, B., Braun, C., Krojmal, E., Lozoya, J.P., Lacerot, G., Russ, A., Teixeira de Mello, F. (2023): Transferencia de microplásticos intra e interespecífico por alimentación y depredación en aves marinas de la Península Fildes, Isla Rey Jorge, Antártida. VII Uruguayan Zoology Congress

Ausems, A.N.M.A., Archuby, D., Braun, C., Gębczyński, A. Hahn, S., Jadwiszczak P., Jakubas, D., Kraemer, P., Kuepper, N.D., Libertelli, M.M., Lorenz, S., Richter, B., Ruß, A., Schmoll, T., Thébault, J., Thorn, S., Turner, J., Wojczulanis-Jakubas, K., Quillfeldt, P. (2022): Where have all the petrels gone? 40 years of environmental change and population dynamics of Wilson's storm-petrels. 15th International Seabird Group Conference, Cork/Ireland

G.3 Conference presentations (poster)

Braun, C., Grämer, H., Hertel, F. (2024): Results of annual marine debris surveys at Fildes Peninsula, King George Island, Maritime Antarctic between 2019 and 2024. Presented at: 11th SCAR Open Science Conference Pucón/Chile.

Grämer, H., Braun, C., Hertel, F. (2024): Up-date of the long-term seabird monitoring in the Fildes Region, King George Island, Maritime Antarctic. Presented at: 11th SCAR Open Science Conference Pucón/Chile

Braun, C., Grämer, H., Peter, H.-U. (2022): Interaction of seabirds with anthropogenic material in the Maritime Antarctic, Presented at: 10th SCAR Open Science Conference 2022