Teledetekcja obrazowa jako narzędzie do monitorowania wielkości populacji wybranych gatunków zwierząt morskich w obrębie Antarktycznych Obszarów Szczególnie Chronionych i Zarządzanych oraz Ostoi Ptaków

Remote sensing imagery as a tool for monitoring the abundance of selected marine animal species within the Antarctic Specially Protected, Managed and Important Bird and Biodiversity Areas

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PhD Dissertation

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Teledetekcja obrazowa jako narzędzie do monitorowania wielkości populacji wybranych gatunków zwierząt morskich w obrębie Antarktycznych Obszarów Szczególnie Chronionych i Zarządzanych oraz Ostoi Ptaków

Streszczenie

Antarktyka, wbrew powszechnemu przekonaniu o jej całkowitej izolacji od wszelkich stresorów, zmaga się z wieloma zagrożeniami dla jej unikalnych w skali planety wartości przyrodniczych. Właściwa współpraca naukowego ciała doradczego z antarktycznymi strukturami politycznymi, decydującymi o przyszłości białego kontynentu, wymaga stanowczego podejścia naukowego polegającego na artykułowaniu konkretnych celów, mających dostarczać antarktycznym decydentom rzetelne dane i wnioski, niezbędne do podejmowania trafnych decyzji politycznych. W projektowaniu badań, stanowiących podstawę niniejszej dysertacji, przyświecającą ideą były założenia ukierunkowanego monitoringu środowiskowego, który jest definiowany przez jego integrację z praktyką ochrony, projektowaniem i wdrażaniem opartym na hipotezach *a priori* i powiązanych modelach reakcji systemu na zarządzanie.

Głównym celem pracy jest zaproponowanie procedur monitorowania wybranych gatunków zwierząt antarktycznych z wykorzystaniem narzędzi teledetekcyjnych oraz algorytmów automatycznej detekcji obiektów. Cztery publikacje wchodzące w skład niniejszej dysertacji przedstawiają wyniki realizacji zadanych celów szczegółowych: (1) zaproponowanie procedury monitoringu populacji petrelca olbrzymiego (*Macronectes giganteus*) z zastosowaniem bezzałogowych systemów powietrznych (BSP); (2) ocena przydatności zdjęć satelitarnych World-View-3 o rozdzielczości 31 cm do przeprowadzania monitoringu populacji rozrodczej mirung południowych (*Mirounga leonina*); (3) rozważenie potencjału zastosowania BSP do przeprowadzania inwentaryzacji ostoi ptaków; (4) zastosowanie modelu uczenia maszynowego YOLO do automatycznej detekcji i liczenia gniazd kormoranów antarktycznych (*Leucocarbo bransfieldensis*) na poddanych georeferencjonowaniu mozaikach lotniczych. Użyte w niniejszej rozprawie metody teledetekcyjne mogą być upowszechnione do prowadzenia inwentaryzacji antarktycznych obszarów cennych przyrodniczo i wymagających szczególnego zarządzania.

Słowa kluczowe: teledetekcja; zdjęcia satelitarne i lotnicze; monitoring zwierząt; Antarktyka; ochrona przyrody; uczenie maszynowe; automatyczna detekcja obiektów

Remote sensing imagery as a tool for monitoring the abundance of selected marine animal species within the Antarctic Specially Protected, Managed and Important Bird and Biodiversity Areas

Abstract

The Antarctic, contrary to the common misconception that it is completely isolated from all stressors, is in fact facing a number of threats to its globally unique natural values. Effective collaboration between the scientific advisory body and the Antarctic political structures responsible for determining the future of the white continent necessitates a rigorous scientific approach to formulating specific objectives. This approach should be designed to provide Antarctic decision-makers with the robust data and conclusions essential for making informed policy decisions. In designing the research that forms the basis of this dissertation, the guiding idea was the premise of targeted environmental monitoring. This is defined by its integration with conservation practice, design and implementation based on *a priori* hypotheses and related models of system response to management.

The principal objective of this study is to propose methodologies for the monitoring of selected species of Antarctic animals utilising remote sensing technologies and automated object detection algorithms. The four publications included in this thesis present the results of the following specific objectives: (1) to propose a procedure for monitoring populations of southern giant petrels (*Macronectes giganteus*) using remotely piloted aircraft systems (RPASs); (2) to assess the suitability of World-View-3 satellite imagery at 31 cm resolution for monitoring the breeding population of southern elephant seal (*Mirounga leonina*); (3) to evaluate the feasibility of utilising RPASs for the purpose of conducting bird sanctuary (Important Bird and Biodiversity Areas (IBAs)) inventories; (4) to assess the efficacy of the YOLO machine learning model in the automatic detection and enumeration of Antarctic shag (*Leucocarbo bransfieldensis*) nests on georeferenced aerial mosaics. The remote sensing methodologies employed in this dissertation can be disseminated for the purpose of conducting inventories of Antarctic areas of natural value and special management needs.

Keywords:

remote sensing; satellite and aerial imagery; wildlife monitoring; the Antarctic; wildlife conservation; machine learning; automatic object detection

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Rozszerzone streszczenie pracy

1. Wstęp

1.1 Ochrona przyrody Antarktyki

Polityczno-prawny status Antarktyki reguluje zestaw umów o zasięgu międzynarodowym, budujących funkcjonalną całość Systemu Układu Antarktycznego. Kluczowym z dokumentów i stanowiącym podstawę sprawczości Systemu jest Układ Antarktyczny, podpisany w 1959 roku w Waszyngtonie (The Antarctic Treaty, 1959), wzbogacany na przestrzeni lat o dodatkowe akty prawne kluczowe dla ochrony dziedzictwa naturalnego Antarktyki: Protokół o ochronie środowiska (Protocol on Environmental Protection to the Antarctic Treaty, 1991), Konwencję o ochronie fok antarktycznych (Convention for the Conservation of Antarctic Seals, 1972), Konwencję o zachowaniu żywych zasobów morskich Antarktyki (Convention on the Conservation of Antarctic Marine Living Resources, 1980) czy, często pomijaną na liście powiązań z Układem (choć bezsprzecznie z nim powiązaną), Konwencję o ochronie albatrosów i petreli (Agreement on the Conservation of Albatrosses and Petrels, 2001). W 1991 roku w Madrycie Antarktyka została uznana *„rezerwatem naturalnym przeznaczonym dla pokoju i nauki"* (Art. 2 Protokołu o ochronie środowiska). Samo to stwierdzenie, zawarte w Protokole madryckim, nie pociągnęło za sobą wartkiej kaskady konsekwencji, należy jednak śmiało podkreślić, że nadanie takiego statusu całemu kontynentowi wraz z otaczającymi go wodami oceanicznymi było wydarzeniem bezprecedensowym w historii ludzkości (co więcej, prawdopodobnie nierealnym do powtórzenia w epoce Antropocenu) i stanowiło fundament dla bardziej szczegółowych postanowień Protokołu, które od momentu jego wejścia w życie w roku 1998, służą ochronie wyjątkowego ekosystemu Antarktycznego. Jednym ze sprawczych narzędzi Protokołu madryckiego stał się Załącznik V, na podstawie którego możliwe jest tworzenie Antarktycznych Obszarów Szczególnie Zarządzanych (ang. Antarctic Specially Managed Areas (ASMAs)) oraz Antarktycznych Obszarów Szczególnie Chronionych (ang. Antarctic Specially Protected Areas (ASPAs)). ASMAs mogą być tworzone na obszarach (zarówno lądowych jak i morskich), gdzie prowadzona jest (lub może być w przyszłości) działalność, w celu ułatwienia jej planowania i koordynacji, uniknięcia możliwych konfliktów, udoskonalenia współpracy między Stronami bądź zmniejszenia wpływu na środowisko. Obecność w obrębie ASMA, w przeciwieństwie do obecności na ASPA, nie wymaga uzasadnienia ani zezwolenia, co w kontekście ochrony przyrody nie nadaje im równoważnej mocy sprawczej, jednak w praktyce Plany Zarządzania ASMA zawierają spis wartości wymagających szczególnej ochrony i pomagają skoordynować działania na danym obszarze tak, by odbywały się za cenę najmniejszego kosztu środowiskowego. Plan Zarządzania ASMA może zawierać rekomendacje dotyczące ograniczenia działalności na danym, w ocenie Zarządzających cennym przyrodniczo lub historycznie obszarze. Pomimo iż, zgodnie z Artykułem 3 Załącznika V Protokołu *„Jakikolwiek obszar, w tym obszar morski, może zostać wyznaczony jako Szczególnie Chroniony Obszar Antarktyki w celu ochrony szczególnych wartości środowiskowych, naukowych, historycznych, estetycznych lub pierwotnych, wszelkiego połączenia wyżej wymienionych wartości bądź prowadzonych lub planowanych badań naukowych"*, obecnie ścisłą ochroną objętych jest zaledwie 75 wycinków Antarktyki co sprawia, że udział Antarktycznych Obszarów Szczególnie Chronionych w całości powierzchni Antarktyki jest nieznaczny (~ 0.076%, stan na rok 2024, na podstawie powierzchni ASPAs uwzględnionych w Planach Zarządzania (ATS, 2024)) i w dużej mierze, z przyczyn logistycznych lub kierowanych interesem poszczególnych państw, powiązany lokalizacyjnie z bliskością stacji badawczych. Może to być uzasadnione faktem, iż Strona sugerująca objęcie ochroną danego obszaru, jest zobowiązana do przedłożenia na forum Państw konsultatywnych propozycji Planu Zarządzania obszaru, jednocześnie prezentującego jego szczególne wartości oraz, w przypadku jednogłośnej akceptacji Planu przez pozostałe Strony, zobligowana jest do jego realizacji a także regularnego (co najmniej co 5 lat) przeglądu założeń w nim zawartych i ich aktualizacji. Wymaga to ciągłego monitorowania elementów ożywionych i nieożywionych przyrody danego obszaru, co jest przedsięwzięciem odpowiedzialnym i kosztownym. Powołanie ASPA wymaga również od jego depozytariusza wyznaczenia odpowiednich władz państwowych, które będą upoważnione do wydawania zezwoleń na prowadzenie jakichkolwiek działalności w granicach ASPA oraz zobligowane do przeprowadzania oceny ich oddziaływania na środowisko. Antarktyczne Obszary Szczególnie Chronione mają więc wprawdzie ogromną moc sprawczą, jednak obarczają swych depozytariuszy szeregiem powinności związanych z procesem zarządzania obszarem, przez co w praktyce sieć ASPA nie jest w stanie chronić w sposób systemowy i równomierny wszystkich wartości, które z założenia powinna. Tyczy się to w szczególności obszarów morskich, które w teorii mogą być objęte statusem szczególnej ochrony, jednak ze względu na ich potencjał komercyjny, zasadność ich powstawania jest podważana przez Strony Układu szczególnie aktywne w zakresie połowów.

Ocean Południowy i morza otaczające Antarktydę są zarządzane przez Komisję ds. Zachowania Żywych Zasobów Morskich Antarktyki (ang. Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR)). Kluczowym celem CCAMLR jest minimalizowanie ryzyka związanego z zagrożeniem dla różnorodności biologicznej wynikającego ze zmiany klimatu (IPCC, 2023) i rosnącego zapotrzebowania na żywe zasoby morskie, poprzez ich ochronę i racjonalne wykorzystanie na Oceanie Południowym (Convention on the Conservation of Antarctic Marine Living Resources, 1980). Wyznaczanie morskich obszarów chronionych (ang. Marine Protected Area (MPA)) jest jednym z mechanizmów mających wspierać ten cel. Do tej pory przyjęto dwa MPA na wodach otaczających Antarktydę kontynentalną i pobliskie wyspy (CCAMLR, 2009, 2016) co spowodowało, że szereg fragmentów morskich zostało objętych wzmocnionymi środkami ochrony lub zarządzania. Trzy kolejne MPA i potencjalny czwarty wciąż pozostają w fazie planowania (CCAMLR, 2021). Od lat nie osiągnięto jednak konsensusu (co jest wymogiem CCAMLR dotyczącym wdrożenia środków zarządzania) w sprawie wyznaczenia proponowanych MPA. W celu dostarczenia informacji na temat wpływu połowów na gatunki zależne, w 1989 roku Komisja ustanowiła Program Monitorowania Ekosystemu (CEMP) (CCAMLR, 2013). Główne cele CEMP to: wykrywanie i rejestrowanie znaczących zmian w krytycznych elementach ekosystemu morskiego na obszarze objętym Konwencją, aby służyć jako podstawa ochrony żywych zasobów morskich Antarktyki oraz rozróżnienie między zmianami wynikającymi z odłowu gatunków komercyjnych a zmianami wynikającymi ze zmienności środowiska. Głównym narzędziem CEMP jest monitorowanie kluczowych parametrów historii życia wybranych gatunków zależnych w celu wykrycia zmian w liczebności odławianych gatunków. „Gatunki zależne" to drapieżniki morskie, dla których gatunki będące celem połowów komercyjnych stanowią główny składnik diety. „Gatunki zależne od kryla" objęte monitoringiem CEMP to gatunki możliwe do obserwacji na lądzie, takie jak ssaki płetwonogie i ptaki.

Jeszcze przed wejściem w życie Protokołu o ochronie środowiska, Birdlife International we współpracy z Naukowym Komitetem do spraw Badań Antarktyki (ang. Scientific Committee on Antarctic Research (SCAR)) wspólnie podjęli pierwsze kroki w celu stworzenia listy obszarów antarktycznych kluczowych dla populacji ptaków oraz zachowania bioróżnorodności (ang. Important Bird and Biodiversity Areas (IBAs)), dalej nazywanych w tekście ostojami ptaków. Pierwszy raport, wraz z propozycją wartości progowych dla poszczególnych gatunków ptaków obejmował 101 ostoi (Harris i in., 2011). Cztery lata później powstał raport aktualizujący, który zawierał już 204 ostoje o zasięgu lądowym (Harris i in., 2015), co było wynikiem wzbogacenia bazy o znaczną liczbę nowych lokalizacji i jednocześnie deklasacji tych, które nie posiadają rzetelnego zaplecza danych. Wszystkie antarktyczne ostoje powołane w roku 2015 posiadają aktywny status i od momentu ich publikacji, baza wzbogacona została o 64 obszary o zasięgu morskim (Handley i in., 2021; Birdlife International, 2024a) i zaledwie dwa nowe obszary o zasięgu lądowym: Zatoka Ryder (Birdlife International, 2024b na podstawie Philips i in., 2019) i Przylądek Melville (Birdlife International, 2024c na podstawie Fudala i Bialik, 2023). Niestety, objęcie danego obszaru statusem ostoi ptaków nie wiąże się z konkretnymi narzędziami wzmacniającymi jego ochronę. Mimo tego, prowadzenie inwentaryzacji lokalizacji kluczowych agregacji ptaków morskich jest niezwykle istotne z punktu widzenia ochrony antarktycznych ekosystemów. Ptaki morskie oferują szczególną możliwość identyfikacji obszarów morskich wysokiej jakości i dodatkowo, jako szczytowi drapieżcy, są uważane za gatunki wskaźnikowe, których populacje odzwierciedlają stan otaczającego środowiska morskiego (Le Bohec i in., 2012). Kompleksowy zestaw danych dotyczący rozmieszczenia kluczowych kolonii antarktycznych ptaków i związanych z nimi liczebności, wzbogacony o pakiety danych ze śledzenia ptaków oraz odpowiednie metody ekstrapolacyjne, może pozwolić na oszacowanie rozmieszczenia żerowisk ptaków na morzu (Dias i in., 2018), a także pozwala na ocenę, w jaki sposób morskie ostoje pokrywają się z obecnie przyjętą i proponowaną siecią kluczowych obszarów zarządzania (głównie MPA) oraz w jaki sposób połowy kryla prawdopodobnie pokrywały się z morskimi ostojami w ciągu ostatnich dekad. Zidentyfikowane przez Handley i in. (2021) morskie ostoje na wodach antarktycznych pozwalają przypuszczać, że gdyby proponowane MPA zostały przyjęte, stała ochrona wysokiej jakości obszarów dla gatunków pingwinów wzrosłaby o 49-100%, w zależności od gatunku. Ponadto dane prezentowane przez powyższy zespół naukowy pokazują, że pomimo zawężającego się zasięgu połowów kryla w Antarktyce w ciągu ostatnich pięciu dekad, konsekwentnie nieproporcjonalnie duża ilość kryla jest odławiana w morskich ostojach ptaków w porównaniu do całkowitego areału, na którym działa rybołówstwo.

1.2 Zagrożenia dla środowiska antarktycznego

Antarktyka zmaga się z człowiekiem już od ponad 200 lat. Przez ten okres region Antarktyki przechodził przez kolejne etapy eksploatacji jego żywych zasobów morskich w postaci polowań na ssaki płetwonogie, wielorybnictwa i rybołówstwa, z których ostatnia aktywność trwa do dziś. Populacje niektórych ssaków płetwonogich oraz wielorybów zostały doprowadzone na skraj zagrożenia wyginięciem przez działalność człowieka prowadzoną w XIX wieku. W ciągu kilku miesięcy od odkrycia Szetlandów Południowych w roku 1819 (Jones, 1975), wyspy te stały się miejscem intensywnej działalności łowców fok. Wynikiem ich nieposkromionej aktywności w latach 20., a następnie latach 70. XIX wieku, było niemal kompletne wytępienie kotików antarktycznych (*Arctocephalus gazella*) z całego archipelagu. Kotiki nie były obserwowane na Szetlandach Południowych aż do 1902 roku, kiedy to jeden osobnik tego gatunku został zaobserwowany (a następnie zabity) na Wyspie Nelson podczas szwedzkiej ekspedycji polarnej (Andersson, 1905). Dopiero w 1959 r. pojawiają się pierwsze doniesienia na temat jednej samicy ze szczenięciem obserwowanej na przylądku Shirreff na wyspie Livingston (O'Gorman, 1961). Okrutny los spotkał również innych przedstawicieli antarktycznych ssaków płetwonogich. Od początku XIX wieku na mirungi południowe (*Mirounga leonina*) polowano dla oleju pozyskiwanego z ich tłuszczu (Bonner, 1982), a w ciągu stulecia ich liczebność znacznie się zmniejszyła, w szczególności w ich subantarktycznym zasięgu rozrodczym. Dokładna liczba odłowionych mirung południowych nie jest znana, ale niektóre szacunki mówią o ponad milionie osobników obu płci (Le Boeuf i Lows, 1994). Inne foki antarktyczne, głównie krabojad foczy (*Lobodon carcinophagus*) i weddelka arktyczna (*Leptonychotes weddellii*), były zabijane na pokarm dla ludzi i psów przez większość XX wieku, aż do przyjęcia Protokołu o ochronie środowiska, który zabraniał między innymi obecności psów w rejonie Antarktyki. Obawa, że w Antarktyce wraz ze stopniową odbudową populacji ssaków płetwonogich rozpocznie się nowy etap ich komercyjnego odłowu przemysłowego, doprowadziła do przyjęcia Konwencji o ochronie fok antarktycznych z 1972 roku. Intensywna działalność człowieka w rejonie Antarktyki nie kończy się na eksploatacji populacji ssaków płetwonogich. Spadek zasobów wielorybów, na które intensywnie polowano na Północnym Atlantyku i Morzu Północnym przyniósł kolejną falę ekologicznej destrukcji indukowanej przez człowieka, przyciągając wielorybników na Ocean Południowy na początku lat 90. XIX wieku (Tønnensen i Johnsen, 1982). Pierwsza ekspedycja wielorybnicza do Antarktyki wypłynęła ze Szkocji w 1892 roku, a wkrótce po niej, w tym samym sezonie, na wody Oceanu Południowego wyruszyła ekspedycja norweska (Tønnensen i Johnsen, 1982). Era antarktycznego wielorybnictwa komercyjnego, w którym oprócz Wielkiej Brytanii i Norwegii uczestniczyło wiele innych krajów, trwała do lat 70. XX wieku. Nieszczęśliwie złożyło się, że wielorybnictwo rozpoczęło się w Antarktyce właśnie wtedy, gdy stało się tak śmiertelnie skuteczne, jak nigdy wcześniej. Efektem końcowym było drastyczne załamanie populacji większości wielorybów Oceanu Południowego (Ballance i in., 2006). W latach 20. XX wieku na arenie międzynarodowej narastały uzasadnione obawy o zrównoważony rozwój komercyjnego połowu wielorybów. W 1930 r. utworzono Biuro Międzynarodowych Statystyk Wielorybniczych w celu monitorowania branży, a w następnym roku powstała Genewska konwencja o regulacji połowów wielorybów (Geneva Convention for the Regulation of Whaling, 1931), która stała się podwaliną dla Międzynarodowej konwencji o regulacji połowów wielorybów (International Convention for the Regulation of Whaling, 1946) oraz Międzynarodowej Komisji Wielorybniczej (ang. International Whaling Commission (IWC)). Międzynarodowa konwencja o regulacji połowów wielorybów nie powinna być mylona z tym, co dziś nazwalibyśmy „umową na rzecz środowiska". Konwencja ta, jak podkreśla Schiffman (2003), była porozumieniem między państwami wielorybniczymi, w interesie wielorybnictwa. IWC wprowadziła moratorium na wszelkie komercyjne połowy wielorybów począwszy od sezonu 1985-1986, tym samym wykonując radykalny zwrot od wcześniej obranego kursu. Jak tłumaczy Sigvaldsson (1996), aby zrozumieć tę nagłą zmianę należy wziąć pod uwagę wieloczynnikowe analizy tej transformacji, uwzględniające postęp gospodarczy i technologiczny, kwestie konstytucyjne, skład negocjujących stron, strukturę władzy w systemie międzynarodowym, organizacje międzynarodowe a także sieci powiązań między nimi. Pomimo wprowadzonego memorandum, Japonia kontynuowała połowy wielorybów w Antarktyce w latach 1986-2018 na podstawie zezwolenia na „naukowe połowy wielorybów" (praktyka budząca wiele kontrowersji, która zakończyła się, gdy w 2019 r. Japonia opuściła IWC). Tym samym na wodach Antarktyki nie prowadzi się już legalnych połowów wielorybów w celach komercyjnych. Całkowity połów wielorybów w Antarktyce w latach 1904-1978 został obliczony na 1 393 254 osobników (Tønnensen i Johnsen, 1982), co było poziomem całkowicie niezrównoważonym ekologicznie, a według niektórych interpretacji, również etycznie (Garner, 2013). Po ssakach płetwonogich i wielorybach, również zasoby ryb zostały doprowadzone do alarmujących poziomów w wyniku intensywnych połowów przemysłowych, które rozpoczęły się pod koniec lat 60. XX wieku. Rybołówstwo rozwijało się głównie na wodach przybrzeżnych wokół Georgii Południowej, Orkadów Południowych i Szetlandów Południowych i doprowadziło do poważnego uszczuplenia zasobów wielu gatunków. Na rok 1992 liczebność nototenii marmurkowej (*Notothenia rossii*) wokół Georgii Południowej szacowano na mniej niż 5% pierwotnie oszacowanej wielkości populacji z 1969 roku (Kock, 1992). Komisja ds. Zachowania Żywych Zasobów Morskich Antarktyki przyjęła środki ochronne w celu wsparcia odbudowy zagrożonych gatunków poprzez zakazanie ukierunkowanych połowów i ustanowienie rygorystycznych limitów przyłowów. Przetrzebienie populacji gatunków nototenii odbiło się negatywnie na populacjach kormorana antarktycznego (*Leucocarbo bransfieldensis*), który jako jeden z dwóch antarktycznych gatunków ptaków (wraz z kormoranem południowym *Leucocarbo georgianus*) żeruje głównie na rybach dennych (Casaux i Barrera-Oro, 2006).

Obecnie, 97% wszystkich połowów Oceanu Południowego dotyczy kryla antarktycznego (*Euphausia superba*), chociaż stanowią pozornie niewielką (około 1%) część szacowanej całkowitej biomasy tego gatunku. Kryl antarktyczny stanowi kluczową biomasę pelagicznych sieci troficznych, szacowaną na 300-500 Mt (Atkinson i in., 2009; Siegel i Watkins, 2016) i odgrywa w nich zasadniczą rolę głównego źródła pokarmu dla różnorodnych drapieżników, w tym wielu endemitów Oceanu Południowego. Połowy kryla w ostatnim raportowanym sezonie (2023) stanowiły prawie czterokrotność połowów z roku 2000 (CCAMLR, 2024). Komercyjne połowy kryla antarktycznego rozpoczęto w roku 1961. W ciągu następnej dekady niewielkie połowy kryla były prowadzone w ramach fazy badawczej rozwoju rybołówstwa. Wielonarodowe połowy kryla były aktywne od początku do połowy lat 70. XX wieku (CCAMLR, 2024). Poziomy tonażu połowów kryla antarktycznego fluktuowały w ciągu ostatnich 50 lat, osiągając 528 000 ton w 1982 r., a następnie spadając do 66 000 ton w 1993 roku. Połowy utrzymywały ogólną tendencję wzrostową w ciągu ostatniej dekady, osiągając kolejny pik w wysokości około 451 000 ton w 2020 r. a następnie delikatny spadek do 416 000 ton w 2022 r. oraz 424 000 ton w 2023 roku (CCAMLR, 2024). W latach 80. połowy miały miejsce w sektorach Atlantyku, Pacyfiku i Oceanu Indyjskiego, ale od lat 90. odbywają się prawie wyłącznie w sektorze południowo-zachodniego Atlantyku. W podobszarze 48.1, który obejmuje wody otaczające Szetlandy Południowe, wysokość połowu w 2023 roku wyniosła 153 587 ton kryla, co stanowi wartość wyższą dwukrotnie od ilości połowu w tym sektorze w roku 2000 (CCAMLR, 2024). Wielkość połowu nie przekraczająca 1% szacowanej całkowitej populacji wydaje się być ilością znikomą, jednak w ocenie wpływu odłowu na stabilność ekosystemu kluczowa jest ocena rozmieszczenia przestrzennego połowów oraz struktury wiekowej poławianych ławic (Panasiuk i in., 2024). Regionalne szacunki biomasy w ramach lokalnych programów monitorowania w głównych obszarach połowowych były wystarczająco regularne, aby zapewnić szeregi czasowe danych ujawniające bardzo duże wahania lokalnej liczebności, co tyczy się cieśniny Bransfield'a i wód rozciągających się na północ od Szetlandów Południowych, gdzie różnice międzyroczne mogą wynosić nawet 2-3 rzędy wielkości (Reiss i in., 2008). Wiedza o takich wahaniach liczebności w kluczowych lokalizacjach działalności rybołówstwa sprawia, że pytanie leżące u podstaw większości dyskusji na temat zarządzania krylem w CCAMLR staje się jeszcze bardziej naglące: w jakim stopniu połowy kryla zagrażają lokalnym drapieżnikom? Aby zwizualizować ekologiczną wartość odławianego tonażu warto uzmysłowić sobie, że szacunki całkowitego zapotrzebowania na kryl populacji wybranych gatunków ptaków mogą być zbliżone do wartości limitu połowowego (620 000 ton). Przykładem może być petrel antarktyczny (*Thalassoica antarctica*), którego zapotrzebowanie na kryl jest szacowane na co najmniej 680 000 ton każdego roku (Descamps i in., 2016).

Zaprezentowane przykłady ingerencji w środowisko Antarktyki poprzez eksploatację jej żywych zasobów nie są jedyną formą obecności człowieka w tym regionie. Na przestrzeni ostatnich trzech dziesięcioleci intensywnie rozwinęła się turystyka antarktyczna. Liczba odwiedzających ten region w celach rekreacyjnych wzrasta sukcesywnie z roku na rok (wyłączając z tej tendencji sezon 2020/21 związany z pandemią COVID-19). Zgodnie z dokumentem Międzynarodowego Zrzeszenia Antarktycznych Operatorów Turystycznych (ang. International Association of Antarctica Tour Operators (IAATO)), które prowadzi statystyki otrzymywane na podstawie raportów od operatorów zrzeszonych w organizacji (nie stanowiących całości działalności turystycznej mającej miejsce w Antarktyce), ponad 120 000 turystów odwiedziło Antarktykę w sezonie 2023/24, z czego prawie 80 000 schodziło na ląd (IAATO, 2024). Dla porównania, w sezonie 1993/94, liczba turystów schodzących na ląd nie przekroczyła 8000 (IAATO, 2024). Oprócz wzrostu liczby turystów, turystyka zróżnicowała się: od lokalnych wycieczek małymi łodziami motorowymi, po rejsy, operacje lotnicze, luksusowe kempingi i różnorodne zajęcia, w tym nurkowanie, wspinaczkę, pływanie na desce, skoki spadochronowe i maratony (Makanse, 2024). Infrastruktura, w tym lądowiska, stacje badawcze i nabrzeża, jest nadal rozbudowywana w celu wspierania działalności naukowej i turystycznej, przy czym znaczna część tej działalności odbywa się na stanowiących niewielki procent całości terenach wolnych od lodu (Pertierra i in., 2017). Poczynając od śladu węglowego i poziomu zanieczyszczeń (również zanieczyszczenia hałasem) jaki niesie za sobą turystyka, ryzyko związane z wciąż rosnącym zainteresowaniem Antarktyką dotyczy niszczenia siedlisk, niepokojenia zwierząt, przenoszenia gatunków obcych i ich propagul, a także patogenów chorobotwórczych. Za pośrednictwem IAATO, sektor turystyczny dąży do ustanawiania standardów środowiskowych, które bywają włączane do zarządzania Systemu Układu Antarktycznego (np. wytyczne dotyczące konkretnych miejsc turystycznych przyjęte na corocznych Spotkaniach Konsultatywnych Układu Antarktycznego (ang. Antarctic Treaty Consultative Meeting (ATCM))), pomagając organizacji uzyskać legitymizację (jako podmiotów samoregulujących) wśród Stron Układu, organizacji pozarządowych i turystów. Według skrupulatnej analizy materiałów promujących turystykę antarktyczną przeprowadzonej przez Varnajot i in. (2024), wyłonić można cztery rodzaje narracji, które stanowią rdzeń „antarktyfikacji” (zjawiska społecznego polegającego na tworzeniu stereotypowych obrazów Antarktyki w celu podniesienia jej atrakcyjności w świadomości potencjalnego turysty): Antarktyka jako miejsce (1) eksploracji; (2) dzikie i puste; (3) pełne superlatyw; oraz (4) wymagające od odwiedzających zarządzania środowiskiem i ambasadorstwa. Wykorzystanie tych narracji zamyka Antarktykę w konkretnych wyobrażeniach mających na celu przyciągnięcie turystów. Jak podkreślają autorzy pracy, identyfikacja tych dominujących narracji ma kluczowe znaczenie, ponieważ mogą one znacząco wpływać na programy ochrony przyrody i antarktyczne dziedzictwo. Narrację opartą na argumentach naukowych, ukazującą zagrożenia płynące ze strony wciąż rosnącej liczby turystów powinno zainicjować środowisko naukowe, nie należy bowiem oczekiwać, że to IAATO, jako stowarzyszenie branżowe, podejmie środki, które ograniczyłyby jego członkom prowadzenie działalności turystycznej.

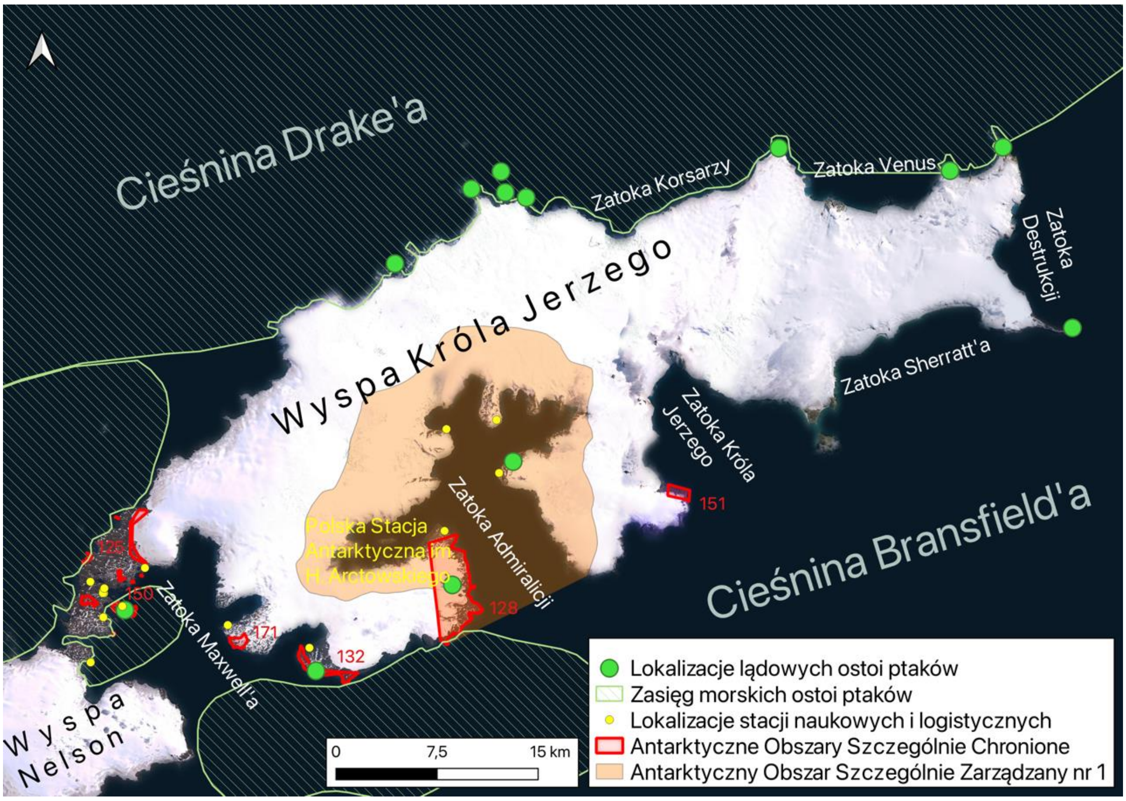
Przytoczone przykłady materialnej obecności człowieka w Antarktyce i jego perturbacyjnego wpływu na jej ekosystem nie są niestety jedynym wkładem naszego gatunku w listę zagrożeń, z jakimi region ten musi się zmagać. Oczekuje się, że globalne ocieplenie spowodowane działalnością człowieka zostanie zaakcentowane ze szczególną siłą w regionach polarnych w porównaniu ze średnią globalną, co jest efektem zwanym wzmocnieniem polarnym (Casado i in., 2023). Świadomość globalnego społeczeństwa w zakresie skali ekstremalnych zjawisk pogodowych wydaje się rosnąć, co jest związane ze szkodami, które zjawiska te wyrządzają w siedliskach człowieka. Zdarzenia te coraz częściej bywają utożsamiane ze spalaniem paliw kopalnych i postępującą degradacją przyrody. Wiele uwagi poświęca się temu tematowi, szczególnie w kontekście rozważania „zysków i strat" dla gospodarek poszczególnych krajów. Warto jednak odnotować, że to nad Antarktyką Wschodnią w marcu 2022 roku wystąpiła najbardziej ekstremalna „fala ciepła" jaką kiedykolwiek zaobserwowano na powierzchni Ziemi (Siegert i in., 2023), która, jak wiele zjawisk skrajnych pociągnęła za sobą całą kaskadę konsekwencji. Zjawisko z marca 2022, nazwane rzeką atmosferyczną, doprowadziło do ocieplenia powierzchni lodu lądowego i szelfów lodowych, rozpadu lodu morskiego i zapadnięcia się szelfu lodowego Congera (Wille i in., 2024). Nie tylko ekstremalne temperatury są zagrożeniem dla środowiska Antarktyki. Do listy zjawisk z kategorii wysoce niepokojących i silnie oddziałujących na stabilność ekosystemową zaliczają się również ekstremalne cyklony (Francis i in., 2022) czy gwałtowne utraty lodu morskiego (Turner i in., 2017). Ekstremalne wydarzenia, a w szczególności ich trudno przewidywalne sploty, mogą prowadzić do obniżenia współczynnika sukcesu rozrodczego antarktycznych zwierząt, czego doskonałym przykładem mogą być pingwiny białookie (*Pygoscelis adeliae*) z kolonii na

Wyspie Petrel (Île aux Pétrels), które w sezonach 2013/14 oraz 2016/17 odniosły całkowitą porażkę rozrodczą, przypisywaną splotowi czynników takich jak: silne i nietypowe dla tego okresu opady deszczu powodujące wychłodzenie piskląt, występowanie rozległego szybkiego lodu morskiego i brak polinii (Ropert- Coudert i in., 2014; 2018). W oceanie ekstremalne zjawiska temperaturowe są zwykle określane jako morskie fale ciepła (ang. Marine Heat Waves) i są definiowane jako wydłużone okresy anomalnie wysokich temperatur, które mogą wywierać znaczący wpływ na bioróżnorodność i ekosystemy morskie. W latach 2002-2018 udokumentowano 19 takich zdarzeń na Oceanie Południowym, a w odpowiedzi odnotowano znaczny wzrost stężenia chlorofilu-a (Montie i in., 2020). Przewiduje się, że w nadchodzących dziesięcioleciach ich częstotliwość i intensywność wzrośnie, a ich skutki będą potęgowane w niektórych obszarach przez ekstremalne zakwaszenie (Burger i in., 2022), chociaż, jak sugerują Siegert i in. (2023), zmienność regionalna i skąpość danych stanowią wyzwanie dla prognoz dla Oceanu Południowego. Będąc przy temacie zjawisk ekstremalnych, trudno nie poruszyć tematu „dziury ozonowej" ponad Antarktydą, której powstanie w atmosferze zostało spowodowane w dużej mierze przez konkretną klasę chemikaliów (Newman i in., 2006). Skutki jej istnienia są nadal odczuwalne i odnoszą się zarówno do biologii (szkodliwy wpływ promieniowania UV na eukarionty i prokarionty (Abbasi i Abbasi, 2017)) jak i klimatologii (ochłodzenie stratosfery, zmiany cyrkulacji troposferycznej półkuli południowej i zmiana jej klimatu powierzchniowego (Thompson i Salomon, 2002, Thompson i in., 2011)).

Biorąc pod uwagę, że intensywność działalności człowieka w rejonie Antarktyki prawdopodobnie nie ustanie w tendencji wzrostowej, a presja na środowisko Antarktyki wynikająca ze zmiany klimatu i działań poza regionem również ulegnie nasileniu, pojawią się dalsze i bardziej naglące wyzwania związane z ochroną przyrody, które będą wymagały opracowania i wdrożenia skutecznej polityki środowiskowej (Hughes i in., 2018; Chown i in., 2022), co wymaga współpracy naukowców z decydentami politycznymi oraz podkreśla wartość długoterminowego monitoringu środowiskowego.

1.3 Wyspa Króla Jerzego

Wyspa Króla Jerzego należy do archipelagu Szetlandów Południowych, rozciągającego się w długiej na 540 km wstędze równoległej do Półwyspu Antarktycznego. Z powierzchnią 1150 km^2 jest największą z 11 głównych wysp wchodzących w jego skład. Na wyspie znajduje się 14 stacji naukowo-logistycznych, z czego 10 ma status baz całorocznych (COMNAP, 2024). Polska Stacja Antarktyczna im. Henryka Arctowskiego należy do placówek operujących przez okres całego roku i jest zlokalizowana w Zatoce Admiralicji. Pomimo, iż na Wyspie Króla Jerzego swoje placówki logistyczne posiada tak wiele antarktycznych programów narodowych, północna i północno-wschodnia część wyspy nie podlega regularnemu monitoringowi środowiskowemu, co jest wynikiem utrudniających działalność terenową warunków atmosferycznych i topografii wyspy (silne wiatry, dryfujące fragmenty lodu, lodowce rozdzielające wolne od lodu obszary, szkiery przybrzeżne) (Rakusa-Suszczewski, 2002), restrykcyjnych procedur bezpieczeństwa związanych z ograniczonymi możliwościami ratunkowymi (Heinrich i Norris, 2024) oraz ograniczeniami finansowymi związanymi z wysokimi kosztami operacji logistycznych i utrzymania narodowych programów antarktycznych (Fowler, 1988; Sánchez i Njaastad, 2014). Na Wyspie Króla Jerzego ustanowionych jest 6 Antarktycznych Obszarów Szczególnie Chronionych (ASPA), 1 Antarktyczny Obszar Szczególnie Zarządzany (ASMA) oraz 13 Ostoi ptaków (Ryc.1). ASPA 151 (ATCM, 2024a) i ASPA 128 (ATCM, 2024b) są zarządzane przez Polskę, podczas gdy ASMA 1 (ATCM, 2023) jest współzarządzany przez państwa aktywne na obszarze, w tym nasz kraj. Spośród 13 lądowych ostoi ptaków na Wyspie Króla Jerzego, 8 zostało ustanowionych na podstawie danych archiwalnych pochodzących sprzed 37 i więcej lat, a więc ich status można uznać za niepewny (Birdlife Data Zone, 2024d). Na Wyspie Króla Jerzego regularnie gniazduje 13 gatunków ptaków, w tym 3 gatunki pingwina z rodzaju *Pygoscelis*. Na brzegach wyspy można spotkać odpoczywające lub liniejące ssaki płetwonogie takie jak kotik antarktyczny, weddelka arktyczna, krabojad foczy, amfitryta lamparcia (*Hydrurga leptonyx*) czy mirunga południowa. Weddelka arktyczna i mirunga południowa prowadzą na Wyspie Króla Jerzego rozród.



Rycina 1. Obszar badań (warstwy wektorowe mapy na podstawie bazy danych Quantarctica (Matsuoka i in., 2021)).

1.4 Wykorzystanie BSP do oceny wielkości populacji zwierząt antarktycznych

Niestety nie nauka, lecz wojna była zapłonem dla rozwoju technologicznego w zakresie zdalnie sterowanych systemów latających. Maszyny do szpiegowania i zabijania po przejściu kolejnych faz swojej ewolucji i odgałęzieniu od funkcji stricte wojskowej, trafiły w ręce użytkownika komercyjnego. Tym sposobem, bezzałogowe systemy powietrzne (BSP, zwane również dronami) stały się w ostatnich dwóch dekadach powszechnymi narzędziami do monitorowania i zarządzania ekologicznego (Ivosevic i in., 2015; Robinson i in., 2022) umożliwiającymi uzyskanie danych o niespotykanej do tego momentu szczegółowości. Relatywnie tanie, łatwe w obsłudze, z wciąż rozwijającym się zapleczem aplikacji przyjaznych użytkownikowi, drony stały się alternatywą dla metod przez dziesięciolecia wykorzystywanych do wyszukiwania i zliczania agregacji zwierząt w rejonie Antarktyki: samolotów, helikopterów czy kontroli naziemnych (np. CCAMLR, 2013; Lyver i in., 2014). Pierwsze loty ponad Antarktyką były prawdziwym przełomem w zakresie eksploracji i otworzyły również antarktyczny rozdział teledetekcji. 20 grudnia 1928 roku Hubert Wilkins zapisał w swoim dzienniku historyczne zdanie: „*Po raz pierwszy* *w historii nowy ląd został odkryty z powietrza"* (Clancy i in., 2014). Hubert Wilkins wraz z Carlem Eielsonem przelecieli z Deception Island ponad Półwysep Antarktyczny robiąc notatki i wykonując dokumentację za pomocą ręcznego aparatu Kodak 3A i dwóch kamer filmowych. Dopiero 7 lat później Herbert Hollick-Kenyon wraz z Lincolnem Ellsworthem po raz pierwszy przecięli Antarktyczny kontynent lotem pomiędzy Dundee Island a Lodowcem Szelfowym Rossa (Joerg, 1937). Po zakończeniu Drugiej Wojny Światowej można mówić o pojawieniu się kartografii z wykorzystaniem fotografii lotniczej, najpierw wraz z programem Ronne Antarctic Research Expedition (RARE), a następnie Falkland Islands Dependencies Aerial Survey Expedition (FIDASE) (Pope i in., 2014). W okresie, gdy każde wzniesienie samolotu skutkowało nowymi odkryciami i zapełnianiem pustego miejsca na antarktycznej mapie, nikt nie zastanawiał się nad skutkiem tych działań w kontekście ekologicznym. W raporcie Wilkinsa (Wilkins, 1929) znajdujemy nawet wzmiankę o kolizjach samolotu z ptakami, rzecz jasna w ramach ubolewania wyrażonego nad stanem technicznym maszyny. Dopiero wiele lat później zaczęło dochodzić do powolnej transformacji myślenia z narracji eksploatacyjno-eksploracyjnej na nastawioną na zachowanie dziedzictwa przyrodniczego Antarktyki jako dobra całej ludzkości.

Potrzeba badań nad skutkami zakłóceń powodowanych przez flotę powietrzną dla antarktycznej i subantarktycznej fauny była uwydatniana przez naukowców (np. Kooyman i Mullins, 1990; Fraser i Patterson, 1997), rezultatem czego było przyjęcie przez Strony Układu podczas ATCM 2004 rekomendacji dotyczących minimalnych wysokości lotów i odległości od kolonii ptaków (ATCM, 2004), które jak podkreśla Harris (2005) były formą wątpliwego „kompromisu" pomiędzy potrzebami operacyjnymi a dobrostanem zwierząt i odbiegały znacząco od wartości sugerowanych przez naukowe ciało doradcze SCAR. Badania naukowe, które analizowały skutki aktywności floty powietrznej w większości przypadków miały, jak konkluduje Harris (2005), charakter oportunistyczny, niemniej jednak sugerowały obecność reakcji behawioralnych i fizjologicznych u pingwinów i fok wynikających z niskich lotów samolotów i helikopterów (np. Culik i in., 1990; Cooper i in., 1994; Giese i Riddle, 1999). Podczas gdy większość badań wykazała niewielki i przejściowy skutek, niektóre powiązały spadki w lokalnych populacjach ze zwiększoną aktywnością w strefie powietrznej (Thomson 1977; Wilson i in., 1991). Zainteresowanie tym, jak pingwiny mogą reagować na przeloty samolotów było łączone z incydentem masowej śmiertelności około 7000 osobników pingwina królewskiego (*Aptenodytes patagonicus*) na wyspie Macquaire, uważanego za przejaw panicznej reakcji na samolot Lockheed C-130 Hercules przelatujący w odległości około 1 mili morskiej od kolonii na stosunkowo niskiej wysokości (Cooper i in., 1994). Według hipotezy stawianej przez Cooper i in. (1994), śmierć ptaków nastąpiła na skutek ucieczki na skraj kolonii ograniczonej przez barierę skalną i wynikającego z nadmiernego stłoczenia spanikowanych zwierząt uduszenia. Sama ucieczka nie została zaobserwowana, a martwe ptaki, zalegające w wysokich stosach, zostały odkryte około 10-12 dni po zdarzeniu.

Zgodnie z przeprowadzoną przez Pina i Vieira, (2022) analizą publikacji dotyczących zastosowania dronów w Antarktyce, pierwszy artykuł naukowy w tym temacie pojawił się w 2006 roku (Funaki i in., 2006), od którego to momentu obserwuje się regularny wzrost przykładów zastosowania dronów do badań antarktycznych. Można śmiało rzec, i potwierdzą to z pewnością naukowcy zaangażowani w badania populacyjne antarktycznych zwierząt, że drony zrewolucjonizowały ten sektor badań.

Eksperymenty przeprowadzane w celu oceny reakcji na obecność drona poszczególnych gatunków antarktycznych ptaków i ssaków wykazywały, że reakcje behawioralne zwierząt na drona mogą być mniej intensywne niż te wynikające z obecności obserwatora naziemnego (Krause i in., 2021) chociaż, jak podkreślają inni autorzy (np. Weimerskirch i in., 2018b), należy w takich rozważaniach uwzględnić gatunek zwierzęcia i jego swoistą odpowiedź na konkretną metodykę badań. Obawy dotyczące zastosowania dronów w Antarktyce w kontekście ich potencjalnego negatywnego wpływu na dziką przyrodę, skłoniły Strony Układu do zainicjowania procesu przygotowywania wytycznych środowiskowych dotyczących wykorzystania BSP w Antarktyce, które zostały przyjęte przez Strony Układu Antarktycznego w drodze rezolucji (ATCM, 2018).

Awifauna Wyspy Króla Jerzego, jak wspomniano w podrozdziale 1.3, składa się z 13 gniazdujących gatunków ptaków, z czego 9 wykazuje potencjał w zakresie monitoringu ich populacji rozrodczej z użyciem drona, swoje gniazda zakłada bowiem na ziemi. Zwyczaje pozostałych 4 gatunków eliminują możliwość rozpoznania ich gniazd na zdjęciach lotniczych, ze względu na fakt, że są ukryte na półkach skalnych, w zagłębieniach skalnych, rumoszu oraz w norach podziemnych. Poza kryterium posiadania gniazda naziemnego, przedstawiciele danego gatunku muszą charakteryzować się wystarczająco dużymi wymiarami ciała lub odpowiednią kontrastowością względem podłoża, by być rozpoznawalnymi na zdjęciach lotniczych przy tak dostosowanym pułapie lotu drona, by nie prowadził do nadmiernego stresu zwierzęcia, reakcji obronnej czy w skrajnych przypadkach porzucenia lęgu. Z uwagi na tundrową specyfikę poszycia Wyspy Króla Jerzego (Olech, 2002), z największą rośliną naczyniową dorastającą do kilkunastu centymetrów, trudno wyobrazić sobie warunki bardziej sprzyjające zastosowaniu drona do mapowania gniazd i przeprowadzania cenzusu populacji zwierząt na lądzie. Potencjał rozpoznania pingwinów z rodzaju *Pygoscelis* na zdjęciach z drona i tym samym ich zastosowania do przeprowadzania inwentaryzacji kolonii tych ptaków, był potwierdzany w publikacjach naukowych (np. Perryman i in., 2011; Ratcliffe i in., 2015; Pfeifer i in., 2019; Shah i in., 2020). Korczak-Abshire i in. (2019) wykorzystali obrazy zarejestrowane przez drona stałopłatowego do rozpoznania czterech gatunków ptaków (pingwinów białookich i maskowych (*Pygoscelis antarcticus*), petrelców olbrzymich (*Macronectes giganteus*) i kormoranów antarktycznych) oraz dwóch gatunków fok (mirung południowych i weddelek arktycznych). Badania przeprowadzone przez Mustafa i in. (2019) miały na celu przetestowanie możliwości rozpoznania na zdjęciu z drona poszczególnych gatunków antarktycznych ptaków i ssaków płetwonogich, z rozróżnieniem na różne wartości wielkości piksela terenowego (ang. Ground Sampling Distance (GSD)). Zastosowanie drona do badań nad antarktycznymi ptakami i ssakami nie ogranicza się do samego ich rozpoznania na obrazie. Zdjęcia uzyskane z nalotu drona mogą pozwolić również na wykonanie pomiarów ciała zwierząt, oszacowanie masy ich ciała, wieku czy określenie kondycji poszczególnych osobników, jak pokazał przypadek badania amfitryt lamparcich i mirung południowych przy użyciu fotogrametrii (Goebel i in., 2015; Krause i in., 2017; Fudala i Bialik, 2020). Drony mogą być również zastosowane do kontroli obszaru w przypadku zaistnienia epidemii zwierząt, czego przykładem może być praca Tyndall i in. (2024). Przedstawiona przez ten zespół nowatorska aplikacja polegała na połączeniu zdjęć z drona z sieciami neuronowymi w celu oceny śmiertelności w kolonii głuptaków (*Morus bassanus*) dotkniętej przez epidemię wysoce zjadliwej ptasiej grypy (HPAI H5N1). Zespołowi udało się zastosować sieć neuronową do skutecznego wykrywania i liczenia żywych i martwych ptaków. Obecność wysoce zjadliwego wariantu ptasiej grypy została potwierdzona również w Antarktyce (Banyard i in., 2024), a rekomendacje wydane przez grupę ekspercką SCAR AWHN (ang. Antarctic Wildlife Health Network), uwzględniają użycie dronów do monitorowania potencjalnych wybuchów epidemii (Dewar i in., 2023).

1.5 Zdjęcia satelitarne w badaniu populacji antarktycznych zwierząt

W tym samym roku, w którym podpisany został Układ Antarktyczny, w kosmos wystrzelony został amerykański satelita Explorer 6, który wykonał pierwsze zdjęcie satelitarne naszego globu (NASA, 2024). Zgodnie z szacunkami Europejskiej Agencji Kosmicznej, w momencie pisania tego tekstu na orbicie ziemskiej znajdowało się około 10 200 aktywnych satelitów (ESA, 2024a). Częściowo ze względu na malejące koszty ich wprowadzania na orbitę, firmy i rządy umieszczają w przestrzeni okołoziemskiej tysiące nowych obiektów rocznie (GAO, 2022). Oczekuje się, że do 2030 roku liczba aktywnych satelitów wzrośnie do ponad 58 000 (GAO, 2022). Abstrahując od niepokojącego faktu braku kontroli nad kosmicznymi śmieciami, które wypełniają systematycznie przestrzeń dobra wspólnego ludzkości (ESA, 2024b), a także ukrytego kosztu środowiskowego jaki za sobą pociągają (Gaston i in., 2023), satelity są niewątpliwym przełomem technologicznym w kontekście narzędzi do monitorowania ekosystemu (Yang i in., 2013), w szczególności w mniej dostępnych rejonach Ziemi, do których należy Antarktyka. W szybko rozwijającym się krajobrazie technologii satelitarnej, zaradne firmy konsekwentnie osiągają kolejne kamienie milowe, rozszerzając swoje konstelacje, aby nie tylko zapewnić wyższe rewizyty, ale także coraz lepsze możliwości obrazowania i rozdzielczości (np. Boeing, 2024; Intelsat, 2024; Spacex, 2024). Przy obecnym tempie rozwoju technologii satelitarnej, oraz wciąż malejących kosztach przedsięwzięć kosmicznych, stosunkowo łatwo wyobrazić sobie, że na przestrzeni kolejnego dziesięciolecia monitoring satelitarny będzie kluczowym i powszechnie dostępnym narzędziem pracy naukowca w Antarktyce, co w połączeniu z użyciem algorytmów sztucznej inteligencji, w tym uczenia maszynowego, może rzucić zupełnie inne światło na niektóre zjawiska przyrodnicze. Aktualne ograniczenia związane z użyciem do badań zdjęć satelitarnych o bardzo wysokiej rozdzielczości (ang. Very High Resolution (VHR)) w rejonie Antarktyki, dotyczą głównie ich dostępności i ceny. Orbita okołobiegunowa (orbita polarna) jest definiowana jako orbita przebiegająca nad biegunami lub w ich pobliżu. Satelity na orbicie okołobiegunowej krążą zazwyczaj na wysokości od 200 do 1000 km nad Ziemią (ESA, 2024c). W trakcie pełnego przelotu satelity, Ziemia wykonuje obrót, w wyniku czego obrazowane są różne fragmenty globu w wąskich pasmach odpowiadających polu rejestracji satelity. Po upływie kilku (np. Sentinel-2: 5 dni (ESA, 2024d)) lub kilkunastu dni (np. Landsat 8: 16 dni (USGS, 2024)), satelita obrazuje ponownie ten sam obszar.

Schwaller i in. (1984) po raz pierwszy przedstawili pomysł zastosowania zdjęć satelitarnych o rozdzielczości 15 m do indeksowania zasięgu rozległych kolonii pingwinów białookich, na podstawie charakterystyki spektralnej guana. Autorzy pracy podkreślali, że przeprowadzenie inwentaryzacji wielkości populacji pingwinów antarktycznych jest konieczne, by dowiedzieć się w jaki sposób antarktyczny łańcuch pokarmowy zareagował na świeżo zakończony etap intensywnego wielorybnictwa, skutkującego prawie całkowitym wyeliminowaniem z układu kluczowych drapieżników kryla. Zdjęcia satelitarne stały się więc już w latach 80. ubiegłego wieku narzędziem, które przynajmniej w teorii, mogło w wymiarze wielkoskalowym pozwolić na oszacowanie wielkości populacji pingwinów. Guinet i in. (1995) wykorzystali zdjęcia satelitarne o rozdzielczości 10 m do wykazania zmian powierzchni zajmowanej przez kolonię pingwina królewskiego na subantarktycznej Wyspie Świni (Île aux Cochons) między sezonami 1962 i 1988. Zgodnie z przeprowadzonymi przez zespół analizami przestrzennymi, powierzchnia kolonii wzrosła o około 56% na przestrzeni tych lat, co pozwoliło na oszacowanie zmiany wielkości populacji z 319 000 par w 1962 r. (na podstawie badań Bauer, 1967) na 494 000-678 000 par, w zależności od przyjętych średnich gęstości gniazd w kolonii tego gatunku. Późniejsze analizy prowadzone przez Weimerskirch i in. (2018a) na podstawie zdjęć satelitarnych i lotniczych, wykazały dramatyczny spadek populacji kolonii z szacowanych 494 000 w roku 1988 na 76 000 par w roku 2015. Enigmatyczny przypadek kolonii z tego zawieszonego między Madagaskarem i Antarktydą skrawka lądu, skłonił francuskich naukowców do pozyskania środków i zorganizowania w roku 2019 pierwszej od 37 lat (i trzeciej w historii) naukowej wyprawy na Wyspę Świni (Kintisch, 2020). Niektóre aspekty trudności związanych z planowaniem i prowadzeniem ekspedycji antarktycznych opisane zostały w artykule Kintisch (2020), przesunięcie lokalizacji zainteresowań badawczych na południe, czyni wszystkie aspekty jeszcze bardziej skomplikowanymi i kosztownymi.

Wyjątkowym przypadkiem zwierzęcia, które rozmnaża się w najsurowszym pod względem warunków klimatycznych miejscu na świecie: w części kontynentalnej

Antarktyki, jest pingwin cesarski (*Aptenodytes forsteri*). Nie istnieje w praktyce żadna inna możliwość zbadania całej populacji tego gatunku, poza zastosowaniem zdjęć satelitarnych. Dotarcie do wszystkich kolonii w sezonie lęgowym, który przypada na antarktyczną zimę, ograniczają nie tylko możliwości techniczne, (wszak gatunek ten prowadzi rozród na lodzie morskim, w środku antarktycznej zimy), ale również, a może przede wszystkim, aspekt finansowy potencjalnego przedsięwzięcia. Trudno byłoby bowiem skonkludować, w dobie ery kosmicznej i obecności człowieka w kosmosie, że wykonanie cenzusu ptaka gniazdującego na Ziemi jest technicznie niemożliwe. Barber-Meyer i in. (2007) byli pierwszym zespołem, który wykorzystał wysokorozdzielcze zdjęcia wykonane przez satelitę Quick-Bird-2, w celu określenia liczebności wybranych kolonii pingwinów cesarskich na Morzu Rossa. Lód morski, na którym tworzą swoje zgrupowania rozrodcze pingwiny cesarskie, tworzy idealne tło do rozpoznawania śladów ich obecności na zdjęciach satelitarnych. Sprawiło to, że zastosowanie technologii kosmicznej do wyszukiwania i oceny wielkości populacji tego gatunku było praktykowane z sukcesem przez różne zespoły badawcze (np. Fretwell i Trathan, 2009; Fretwell i in., 2012; Ancel i in., 2014; Fretwell i Trathan, 2021; Fretwell, 2024). W przeciwieństwie do lodu tworzącego lodowce czy szelfy, lód morski jest jednorodny i wolny od kolorystycznych perturbacji. Spektralna reflektancja lodu morskiego daje obraz czysto białej lub delikatnie niebieskiej powierzchni. Jedynym wyjątkiem są miejsca, gdzie odchody zgrupowań rozrodczych pingwinów tworzą widoczne na obrazie satelitarnym plamy o czerwonawo-brązowym zabarwieniu. Niestety, jak wnioskują Ancel i in. (2014), rozpoznanie depozytów guana na lodzie morskim na zdjęciach satelitarnych może być utrudnione przez ich zasypywanie przez opady śniegu czy ewoluującą topologię lodu morskiego. Antarktyczne gatunki pingwinów są bez wątpienia wyjątkowo predysponowane do ich wyszukiwania na zdjęciach satelitarnych, dzięki ich pokaźnym rozmiarom oraz przez fakt, że w okresie rozrodczym tworzą duże zgrupowania na otwartych przestrzeniach lądu lub na lodzie. Nie są jednak jedynymi ptakami antarktycznymi, których populacje badano z wykorzystaniem satelitów. Przykładem innego, znacznie mniejszego gatunku, tworzącego rozległe kolonie rozrodcze w głębi antarktycznego kontynentu, jest petrel antarktyczny. Zespół Schwaller i in. (2018) dokonał pierwszej w skali kontynentu próby wykrycia lokalizacji rozrodczych petreli antarktycznych na bazie danych satelitarnych

Landsat-8. Dzięki zaproponowanemu algorytmowi zidentyfikowano dużą liczbę potencjalnych kolonii lęgowych petreli antarktycznych. Praca ta stanowi zatem pierwszy krok w kierunku odkrycia nieznanych wcześniej kolonii petreli i bardziej kompleksowego, rutynowego monitorowania tego gatunku. Istnieją przykłady wykorzystania satelitów do monitorowania populacji ptaków z metodyką bazującą na wykryciu poszczególnych osobników. Fretwell i in. (2017) wykorzystali imponujące rozmiary ciała i jasne upierzenie dwóch gatunków albatrosów z rodzaju *Diomedea*, kontrastujące z roślinnym podłożem typowym dla miejsc ich gniazdowania, do ich detekcji na fotografiach satelitarnych w koloniach rozrodczych na Georgii Południowej (albatros wędrowny (*Diomedea exulans*)) oraz wyspach archipelagu Chatcham (albatros królewski (*Diomedea epomophora*)). Do tego zadania wykorzystane zostały zdjęcia World-View-3 o rozdzielczości 31 cm. Jak wykazują zaprezentowane przykłady, zdjęcia satelitarne zostały z sukcesem wykorzystane do badań na antarktycznych gatunkach ptaków. Jednak stosunkowo uboga w różnorodność gatunkową makrofauna lądowa Antarktyki jest tworzona również przez ssaki płetwonogie. Wymiary ich ciała i fakt, że rozród skłania je do nawiązania koneksji z lądem, czyni je wdzięcznym obiektem badań z zastosowaniem telemetrii satelitarnej. Pozbawione swojego żywiołu ssaki płetwonogie, są w środowisku lądowym motorycznie pasywne, ich wydatek energetyczny ogranicza się do niezbędnych strat poświęconych na rozród i linienie. La Rue i in. (2011) w swoich badaniach wykorzystali zdjęcia satelitarne Quick-Bird-2, oraz World-Wiew-1 do przeprowadzenia próby oszacowania wielkości populacji weddelek arktycznych na lodzie Zatoki Erebus. Szeroki zasób danych na temat populacji z Zatoki Erebus, pochodzących z monitoringu prowadzonego przy wsparciu logistycznym stacji McMurdo, znajdującej się w bliskości kolonii, pozwolił na porównanie liczby zwierząt rozpoznanych na zdjęciach satelitarnych z wynikami liczbowymi kontroli naziemnych. Autorzy potwierdzili użyteczność zdjęć satelitarnych o rozdzielczości 60 cm do liczenia weddelek na lodzie, jednocześnie zwracając uwagę na istniejące niedogodności dotyczące użycia zdjęć satelitarnych do badań populacyjnych. Jedną z komplikacji jest fakt, że przedział dobowy, w którym wykonywane jest zdjęcie z poruszającej się po orbicie polarnej satelity, jest z góry zdefiniowany i w tym przypadku wykraczał poza przedział czasowy dnia, w którym większość fok, zgodnie ze zbadaną rutyną dnia tego gatunku, znajduje się na lodzie. Dodatkowo, ze względu na optyczną naturę satelitów, ich użyteczność zależy od zachmurzenia i oświetlenia. W okresie antarktycznej zimy jest ona limitowana długością dnia, co sprawia, że odpowiednie warunki świetlne nie zawsze pokrywają się z czasem ich przelotu ponad pożądaną lokalizacją. Przypadek rozpoznania na obrazie satelitarnym weddelek, z charakterystycznym ciemnym ubarwieniem, leżących na białym tle pokrywy lodowej wydaje się być mniej wymagający niż próba rozpoznania fok leżących na tle mniej kontrastującego, odsłoniętego od śniegu i lodu podłoża. McMahon i in. (2014) podjęli się próby policzenia samic mirung południowych w haremie, na wulkanicznej plaży subantarktycznej Wyspy Macquarie, na podstawie zdjęcia z satelity Geo-Eye-1 o rozdzielczości 50 cm (jednocześnie omawiając użyteczność zastosowania zdjęć o rozdzielczości 60 cm do tego celu). Pomimo, iż mirungi południowe są największym przedstawicielem ssaków płetwonogich w Antarktyce, rozpoznanie samic na zdjęciach satelitarnych generowało pewną dozę niepewności u osób analizujących, co obrazuje średnia liczba rozpoznanych mirung na badanym obszarze, która wynosiła 1790 +/- 306 (n = 3 liczenia; 95% przedział ufności). Autorzy konkludują jednak, iż jest to satysfakcjonujący wynik, czyniący metodę użyteczną, porównując wynik z liczeniem naziemnym wykonanym tego samego dnia, zgodnie z którym na obszarze znajdowało się 1991 dorosłych samic. Temat wykorzystania satelitów niemalże automatycznie przywodzi na myśl największe ze ssaków żyjące na naszej planecie. Fretwell i in. (2014) podjęli się próby rozpoznania waleni południowych (*Eubalaena australis*) na zdjęciach z satelity World-View-2 o rozdzielczości 50 cm z pasmem penetrującym w dalekiej niebieskiej części widma, które pozwala na głębszą widoczność w słupie wody. Autorzy podkreślili, że przypisanie konkretnego gatunku było możliwe dzięki dedukcji i wykluczeniu możliwości obecności innych gatunków podobnej wielkości na zdjęciu z wybranej, poddanej wcześniejszym analizom lokalizacji. Przeskok jakości obrazu, który nastąpił wraz z pojawieniem się satelity World-Wiew-3 w roku 2016, oferującej rozdzielczość przestrzenną 31 cm, umożliwiał już wychwytywanie cech charakterystycznych danego gatunku i tym samym rozpoznawanie gatunku wieloryba na zdjęciach satelitarnych (Cubaynes i in., 2019).

Zdjęcia satelitarne o wysokiej rozdzielczości są bardzo kosztowne, co stanowi główne ograniczenie w ich użyciu do wykrywania zwierząt. Przyszłość w tym temacie jest jednak bardzo obiecująca: ze względu na rosnącą konkurencję pomiędzy firmami branży kosmicznej oferującymi obrazy satelitarne, ceny zdjęć mają potencjał stania się bardziej przystępnymi dla naukowców. Oprócz uruchomionej i rozwijającej się konstelacji World-View firmy Maxar (MAXAR, 2024), również Airbus i Planet rozwijają swoje konstelacje Pléiades Neo (Airbus, 2024) i Pelican (Planet, 2024), wszystkie z rozdzielczością przestrzenną oscylującą w okolicach 30 cm. Konstelacja Clarity, firmy Albedo Space (Albedo Space, 2024), która ma zostać wyniesiona na orbitę w 2025 roku, ma na celu osiągnięcie przełomowej rozdzielczości przestrzennej 10 cm, potencjalnie dorównującej lub nawet przewyższającej niektóre zdjęcia lotnicze. Osiągnięcia te wskazują, że wykrywanie zwierząt w oparciu o zdjęcia satelitarne o bardzo wysokiej rozdzielczości ma coraz większy potencjał.

1.6 Uczenie maszynowe jako narzędzie badań ekosystemów morskich

Uczenie maszynowe przeszło kompleksową ewolucję od wyjściowych teorii Alana Turinga z lat 50. ubiegłego wieku (Bowen, 2016). W momencie, gdy redaguję ten tekst dwóch naukowców: John Hopfield i Geoffrey Hinton, którzy przyczynili się do zainicjowania obecnego, skokowego rozwoju uczenia maszynowego, zostało wyróżnionych najbardziej prestiżową nagrodą, jaką może otrzymać naukowiec *„za fundamentalne odkrycia i wynalazki, które umożliwiają uczenie maszynowe z wykorzystaniem sztucznych sieci neuronowych*" (The Nobel Prize, 2024), pokazując wagę tej ścieżki nauki. Metody uczenia maszynowego przewyższają inne algorytmy, jeśli chodzi o moc predykcyjną i wnioskowanie (Elith i in., 2006), zwłaszcza gdy wymagane jest sprawne podejmowanie decyzji, jak ma to miejsce w przypadku zarządzania zasobami naturalnymi. Nowe algorytmy i metody analizy złożonych danych ekologicznych powstają szybko, z prędkością przekraczającą często możliwości ich bieżącego wdrażania przez ekologów (Humphries i Huettmann, 2018). Przyjęcie uczenia maszynowego w społeczności ekologicznej jest powolne, mimo iż sam termin „uczenie maszynowe" (ang. machine learning (ML)) stał się obecny zarówno w literaturze naukowej, jak i naszej codzienności (Horvitz, 2007). Opór na granicy fuzji ekologii i ML może być spowodowany, oprócz potencjalnego dyskomfortu związanego z wkraczaniem na nieznane ścieżki, również brakiem komunikacji między naukowcami a społecznością ML (Thessen, 2016). W przeciwieństwie do tradycyjnych programów komputerowych, które kodują rozwiązanie zaprojektowane przez programistę, system ML może nauczyć się rozwiązywać problem bez dostarczania wyraźnego przepisu, rozwiązanie jest przez system dedukowane dzięki dostarczaniu mu przykładowych danych. Zdolność do stworzenia rozwiązania problemu może być imponująca, ale zależy przede wszystkim od odpowiedniej prezentacji problemu, czyli „celu" i dostarczenia odpowiednich danych, na podstawie których może zajść proces „uczenia się" (Zhou, 2021). Rubbens i in. (2023) przygotowali zestawienie ~1000 (wyselekcjonowanych na podstawie analiz ilości ich cytowań oraz klarowności opisu metodyki dotyczącej ML) artykułów naukowych prezentujących w swej treści praktyczne zastosowanie ML w ekologii morskiej. Szeroki wachlarz tematyki obejmuje między innymi analizy danych: bentosowych, planktonowych, nektonowych, wyszukiwanie i klasyfikację śmieci i zanieczyszczeń morskich, identyfikację mikroorganizmów morskich, interpretację pomiarów akustycznych, profilowanie społeczności biologicznych za pomocą genomiki środowiskowej, kwantyfikację i mapowanie presji połowowej na podstawie danych geolokalizacyjnych, uzyskiwanie zmiennych biogeochemicznych z obrazów satelitarnych i profili pływaków pomiarowych czy przewidywanie liczebności i rozmieszczenia gatunków zwierząt morskich.

W ostatnich latach wykrywanie i rozpoznawanie obiektów na filmie lub obrazie było ważnym zadaniem wizji komputerowej (ang. Computer Vision (CV)) (Diwan i in., 2023). Wykrywanie obiektów za pomocą modeli uczenia maszynowego odnosi się do zestawu algorytmów, które mogą automatycznie identyfikować i lokalizować obiekty na obrazach lub filmach. Modele te wykorzystują ekstrakcję cech, wybór cech i techniki klasyfikacji do rozpoznawania obiektów w danych wizualnych (Sirisha i in., 2023). Aby wytrenować te modele, dostarczane są oznakowane obrazy, w których każdy obiekt zainteresowania jest przypisany do odpowiedniej klasy. Model wykorzystuje oznakowane obrazy do uczenia się cech specyficznych dla każdej klasy obiektów. Dostępnych jest kilka modeli uczenia maszynowego do wykrywania obiektów, w tym modele wektorów nośnych (Pontil i Verri, 1998), drzew decyzyjnych (Ali i in., 2012) i lasów losowych (Liu i Xiong, 2012). Modele te różnią się podejściem do ekstrakcji cech i klasyfikacji i mogą działać inaczej w zależności od zadania i dostępnych danych (Khan i Al-Habsi, 2020). Niektóre z tych modeli wymagają ręcznej inżynierii cech, podczas gdy inne mogą automatycznie uczyć się cech z danych wejściowych. Modele głębokiego uczenia odnoszą się do klasy sieci neuronowych, które mogą automatycznie identyfikować i lokalizować obiekty na zdjęciach lub filmach. Modele te wykorzystują wiele warstw jednostek przetwarzania do wyodrębniania złożonych funkcji z danych wejściowych, co czyni je skutecznymi w zadaniach wykrywania obiektów (Zhao i in., 2019). Niektóre przykłady modeli obejmują CNN, R-CNN, SSD, ResNet i modele „You Only Look Once" (YOLO) (Xu i in., 2024), które mogą dokładnie rozpoznawać obiekty i wykrywać wiele obiektów na jednym obrazie lub filmie. Jak wynika z pracy przeglądowej przeprowadzonej przez Xu i in. (2024), liczba publikacji na temat wykrywania zwierząt w oparciu o głębokie uczenie i teledetekcję w latach 2017-2023, wyniosła 98, z wyłączeniem prac przeglądowych. Podstawowymi platformami teledetekcyjnymi wykorzystywanymi do wykrywania zwierząt w niniejszych pracach były bezzałogowe i załogowe systemy powietrzne oraz satelity. YOLO stał się jednym z wiodących modeli wykrywania zwierząt, ze względu na fakt, że działa z imponującą dokładnością i szybkością (Redmon, 2016). Dzieląc cały obraz na małą siatkę i dokonując przewidywań bezpośrednio w każdej komórce siatki, YOLO osiągnęło przekonujące możliwości przetwarzania w czasie rzeczywistym, co czyni go szczególnie korzystnym dla aplikacji wymagających szybkiej i dokładnej identyfikacji obiektów w zmiennych środowiskach (Zhao i in., 2019).

Choć w popularnej literaturze „Sztuczna Inteligencja" i „Uczenie Maszynowe" bywają traktowane jako synonimy, ML jest poddziedziną sztucznej inteligencji (ang. Artificial Inteligence (AI)), która obejmuje modele logiczne, algorytmy i metody obliczeniowe zdolne do „inteligentnego zachowania" (Jordan i Mitchell, 2015). W ramach ML, rozwinął się w ciągu ostatniej dekady osobny podrozdział „głębokiego uczenia" (ang. Deep Learning (DL)). Systemy DL wykorzystują rozległe sieci neuronowe do wyodrębniania istotnych cech z surowych danych i uczą się na ich podstawie, zamiast wymagać wyraźnej inżynierii tych cech (Monesinos López i in., 2022). Dane te są często złożone (zaliczają się do nich np. obrazy i dźwięki) i rozbudowane (od tysięcy do milionów rekordów), mogą również mieć charakter Wielkich Danych (ang. Big Data). Wielkie Dane, jak wskazuje sama nazwa, są definiowane przez swój rozmiar, obejmujący rozległy, złożony i niezależny zbiór zestawów danych, z potencjałem wielokierunkowej interakcji pomiędzy nimi. Ponadto, ważnym aspektem Wielkich Danych jest fakt, że nie można ich obsługiwać za pomocą standardowych technik zarządzania danymi ze względu na niespójność i nieprzewidywalność możliwych kombinacji (Sagiroglu i Sinanc, 2013). Doskonałym przykładem mogą być Wielkie Dane satelitarnych zbiorów teledetekcyjnych. Jak podkreśla Liu (2015) mają one kilka konkretnych cech, poza oczywistą ogromną objętością, pozwalających zaliczyć je do tej kategorii tj. mają wiele źródeł i skal, są wielowymiarowe, dynamiczne, izomeryczne i nieliniowe.

2. Motywacja i cel pracy

Właściwe zarządzanie wartościami przyrodniczymi Antarktyki wymaga ciągłego ich monitorowania. Jedynie długofalowe obserwacje, przekładające się na rzetelne dane naukowe i wnioski, mogą być argumentem naukowego ciała doradczego Systemu Układu Antarktycznego i mają szansę modulować ruchy decydentów politycznych, którzy projektują dalsze losy Antarktyki. Monitoring środowiskowy bywa traktowany jako kategoria podrzędna lub jedno z wielu narzędzi w rękach szeroko definiowanej „nauki". Nie wszyscy zgadzają się z tym wykluczeniem (np. Lovett i in., 2007). Carpenter (1998) sugeruje, że nauka o ekosystemach jest jak stół wsparty na czterech nogach: teorii, eksperymentach, porównaniach i badaniach długoterminowych. Można więc pokusić się o hipotezę, że „noga jest integralną częścią stołu".

Głównym celem niniejszej rozprawy doktorskiej jest zaproponowanie procedur monitorowania wybranych gatunków zwierząt antarktycznych z wykorzystaniem bezzałogowych systemów powietrznych (BSP), potocznie zwanych dronami, oraz wysokorozdzielczych zdjęć satelitarnych. Użyte w niniejszej rozprawie metody teledetekcyjne mogą być upowszechnione do prowadzenia inwentaryzacji antarktycznych obszarów cennych przyrodniczo i wymagających szczególnego zarządzania.

Platt (1964) w swojej publikacji otwarcie skrytykował nieukierunkowane gromadzenie szczegółowych danych, które są być może ogólnie istotne dla badania, ale nie dążą w stronę wyodrębniania hipotez. W projektowaniu badań, stanowiących podstawę niniejszej dysertacji, przyświecającą ideą były założenia monitoringu ukierunkowanego, który jest definiowany przez jego integrację z praktyką ochrony, projektowaniem i wdrażaniem opartym na hipotezach *a priori* i powiązanych modelach reakcji systemu na zarządzanie. Na bazie tych założeń wyodrębnione zostały **szczegółowe cele pracy**:

1. Zaproponowanie procedury monitoringu populacji petrelca olbrzymiego (*Macronectes giganteus*) z zastosowaniem bezzałogowych systemów powietrznych (BSP).
2. Ocena przydatności zdjęć satelitarnych World-View-3 o rozdzielczości 31 cm do przeprowadzania monitoringu populacji mirung południowych (*Mirounga leonina*).
3. Rozważenie potencjału zastosowania BSP do przeprowadzania inwentaryzacji ostoi ptaków.
4. Zastosowanie modelu uczenia maszynowego YOLO do automatycznej detekcji, identyfikacji i liczenia gniazd kormoranów antarktycznych (*Leucocarbo bransfieldensis*) na poddanych georeferencjonowaniu mozaikach lotniczych.

3. Zarys pracy

Niniejsza dysertacja stanowi spójną całość dotyczącą oceny wielkości lokalnych populacji wybranych gatunków ptaków i ssaków płetwonogich w obrębie Antarktycznych Obszarów Szczególnie Chronionych (ang. Antarctic Specially Protected Areas (ASPAs)), Antarktycznych Obszarów Szczególnie Zarządzanych (ang. Antarctic Specially Managed Areas (ASMAs)) oraz ostoi ptaków (ang. Important Bird and Biodiversity Areas (IBAs)). Prace terenowe pozwalające na uzyskanie danych do niniejszej rozprawy zostały przeprowadzone na Wyspie Króla Jerzego, wchodzącej w skład archipelagu Szetlandów Południowych w Antarktyce Zachodniej. Ocena wielkości badanych populacji zwierząt opierała się na wykorzystaniu teledetekcji obrazowej: zdjęć z bezzałogowych systemów powietrznych (BSP), oraz wysokorozdzielczych zdjęć satelitarnych, a także modelu uczenia maszynowego YOLO. Wyniki przeprowadzonych badań zostały przedstawione w czterech artykułach naukowych:

1. **Fudala K.**, Bialik R.J. (2022). The use of drone-based aerial photogrammetry in population monitoring of Southern Giant Petrels in ASMA 1, King George Island, maritime Antarctica. *Global Ecology and Conservation* 33, e01990, DOI: 10.1016/j.gecco.2021.e01990.

2. **Fudala K.**, Bialik R.J. (2022). Seals from outer space-Population census of southern elephant seals using VHR satellite imagery. *Remote Sensing Applications: Society and Environment* 28, 100836, DOI: 10.1016/j.rsase.2022.100836.

3. **Fudala K.**, Bialik R.J. (2023). Identifying important bird and biodiversity areas in Antarctica using RPAS surveys—A case study of Cape Melville, King George Island, Antarctica. *Drones* 7(8), 538, DOI: 10.3390/drones7080538.
4. Cusick A., **Fudala K.**, Storożenko P.P., Świeżewski J., Kaleta J., Oosthuizen W. Ch., Pfeifer C., Bialik R.J. (2024). Using machine learning to count Antarctic shag (*Leucocarbo bransfieldensis*) nests on images captured by Remotely Piloted Aircraft Systems. *Ecological Informatics* 82, 102707, DOI: 10.1016/j.ecoinf.2024.102707.

W ramach realizacji zadanego celu pierwszego (1) zaprojektowano prace terenowe przeprowadzane w trakcie dwóch kolejnych sezonów letnich, w obrębie Antarktycznego Obszaru Szczególnie Zarządzanego nr 1 Wyspy Króla Jerzego. W trakcie sezonu pierwszego, dokonano inwentaryzacji lokalizacji rozrodczych petrelców olbrzymich na brzegach Zatoki Admiralicji i wykonano naloty BSP pod kątem modelowania 3D dwóch głównych obszarów zajmowanych przez kolonie ptaków, w fazie poprzedzającej inkubację. Na ich bazie przygotowano modele hipsometryczne terenów. W kolejnym sezonie wykonano serię zaprojektowanych nalotów BSP, w różnych fazach cyklu rozrodczego, w celu ustalenia optymalnego etapu fenologicznego do wykonywania cenzusu petrelców. Dodatkowo, przeprowadzono eksperyment w celu zbadania wpływu obecności drona na reakcje behawioralne petrelców i określenia optymalnej wysokości lotu drona pozwalającej zidentyfikować gniazda oraz pisklęta tego gatunku, przy możliwie najmniejszej ingerencji w zachowanie zwierząt. Wyniki badań przedstawiono w publikacji nr 1.

By zrealizować cel drugi (2) pracy, porównano wyniki liczenia mirung południowych z wykorzystaniem zdjęcia satelitarnego World-View-3 o rozdzielczości 31 cm na piksel oraz ortomozaiki o rozdzielczości 1.39 cm na piksel uzyskanej z misji BSP. Zweryfikowano możliwość wykorzystania wysokorozdzielczych zdjęć satelitarnych: (a) do identyfikacji liczby haremów mirung południowych; (b) określenia struktury wiekowej i płciowej analizowanej agregacji - osobników dorosłych (samice, samce) i młodych; (c) wskazania całkowitej liczby osobników mirungi na obszarze wybranym do porównania. Zwrócono również uwagę na problem niejednoznaczności interpretacji obrazu podczas identyfikowania obiektów do trenowania modeli uczenia maszynowego na podstawie obrazów ze zdjęć satelitarnych. Wyniki badań przedstawiono w publikacji nr 2.

Wyniki zaprezentowane w artykule nr 3 były odpowiedzią na zadany cel trzeci (3) i rezultatem przeprowadzenia inwentaryzacji obszaru Przylądka Melville oraz przybrzeżnych wysp i skał Zatoki Destrukcji położonych na wschodnich obszarach Wyspy Króla Jerzego przy użyciu BSP.

W ramach realizacji celu czwartego (4) nawiązano kooperację z zespołem specjalistów z zakresu uczenia maszynowego oraz dwoma zespołami naukowymi, wykonującymi inwentaryzację populacji kormoranów antarktycznych z użyciem BSP. Wykonano manualne analizy ortomozaik lotniczych uzyskanych z użyciem różnego rodzaju BSP pochodzących od 3 niezależnych zespołów naukowych, z 11 kolonii kormorana antarktycznego rozlokowanych w różnych punktach wysp Króla Jerzego oraz Nelsona. Uskuteczniono trenowanie modelu YOLO na przygotowanym zestawie danych. Omówiono warunki optymalnej wydajności modelu, a także jego ograniczenia. Dokonano analiz wewnątrz- i między- sezonowych zmian liczebności kormoranów w latach 2019-2022 w koloniach Turret Point oraz Shag Rock. Wyniki przedstawiono w publikacji nr 4.

4. Wyniki i wnioski

4.1. Wykorzystanie fotogrametrii lotniczej opartej na BSP w monitorowaniu populacji petrelców olbrzymich na terenie ASMA 1 [na podstawie: Fudala i Bialik, 2022a – Załącznik 1]

W artykule nr 1 (Fudala i Bialik, 2022a) określono parametry lotu optymalne do identyfikacji osobników dorosłych i piskląt petrelców olbrzymich na ortomozaikach otrzymanych z misji drona. Na podstawie przeprowadzonego eksperymentu określono minimalną wysokość lotu, na której nie odnotowano istotnej statystycznie reakcji behawioralnej petrelców. Podano sugerowany piksel terenowy (GSD) dla nalotów dronem DJI Inspire 2 z kamerą Zenmuse X5S i obiektywem 15 mm, umożliwiający bezproblemową identyfikację obu grup wiekowych petrelców (GSD 2.15 cm przy wysokości nalotu ponad ptakami 112-121 m). W ramach realizacji celu pierwszego dysertacji, zaprezentowano kompletną metodologię inwentaryzacji kolonii petrelców olbrzymich z użyciem BSP. Na podstawie inwentaryzacji przeprowadzonych z użyciem drona określono lokalizacje wszystkich obszarów lęgowych petrelców w obrębie ASMA nr 1 i potwierdzono obecność 3 kolonii: Llano Point/Rescuers Hills, Vaureal i Petrel Hill. Inwentaryzacja pozwoliła oszacować rozmieszczenie i liczbę aktywnych gniazd i piskląt na całym obszarze ASMA nr 1 w sezonie 2020/21 (508 aktywnych gniazd i 380 piskląt).

Petrelce olbrzymie charakteryzują się wyjątkowo wysoką wrażliwością na obecność człowieka w bliskości ich gniazd. Liczne podania literaturowe sygnalizowały wysoki koszt środowiskowy badań przeprowadzanych na tym gatunku (Warham, 1962; Conroy, 1972; Jabłoński, 1986), mimo to cenzus populacji tych ptaków na ASPA nr 128 odbywał się do sezonu 2019/20 z zastosowaniem mapowań naziemnych, wymagających wkraczania obserwatorów na teren kolonii rozrodczej. Od sezonu 2020/21 monitoring tego gatunku jest prowadzony w obrębie szerszego kontekstu obszarowego ASMA nr 1 i odbywa się wyłącznie z użyciem bezzałogowych systemów powietrznych, z wykorzystaniem metodyki opisanej w publikacji.

4.2. Zastosowanie zdjęć satelitarnych o bardzo wysokiej rozdzielczości do oceny wielkości populacji mirung południowych na terenie ASPA 128 [na podstawie: Fudala i Bialik 2022b – Załącznik 2]

W artykule nr 2 (Fudala i Bialik, 2022b) oceniono skuteczność rozpoznawania mirung południowych na zdjęciu satelitarnym World-View-3 o rozdzielczości 31 cm. Obraz satelitarny obrazujący kolonię rozrodczą na Przylądku Patelnia pozwolił na rozpoznanie 19 samców, 403 samic i 246 młodych (mediana z liczenia 11 obserwatorów). Wyniki zestawiono z liczbami zwierząt rozpoznanymi na ortomozaice lotniczej wykonanej dronem 15 godzin wcześniej, dla której liczby te wynosiły kolejno 21, 428, 386. Porównaniu poddano analogiczny obszar kolonii, odpowiadający zasięgowi nalotu drona. Obecność dodatkowego haremu poza zasięgiem nalotu drona, który został zidentyfikowany na zdjęciu satelitarnym, zmieniła całkowitą liczbę mirung (obserwowanych w sezonie 2019/20 w piku liczebności samic) na obszarze Przylądka Patelnia o około 13 samic, 8 szczeniąt i 2 samców (mediana z liczenia 11 obserwatorów). W odpowiedzi na zadany cel drugi dysertacji, wskazano błąd względny dla liczeń z obrazów satelitarnych w obrębie poszczególnych grup płciowych i wiekowych mirungi południowej, z podziałem na wyszczególnione strefy w obrębie kolonii, omówiono także potencjalne przyczyny fałszywie pozytywnych i negatywnych predykcji obserwatorów. Błąd względny między medianą wyników uzyskanych z liczenia samic mirungi na podstawie obrazu satelitarnego a wynikami opartymi na liczeniu z drona wahał się od 1,5% do 7,0%, w zależności od strefy kolonii. Skuteczność identyfikacji szczeniąt była mocno zależna od poziomu kontrastowości podłoża względem ciemno ubarwionych młodych, z błędem względnym na poziomie 24,6-40,2%. W badaniach wykazano, że wiele problemów, na które zwrócili uwagę McMahon i in. (2014), którzy podjęli się próby oceny wielkości agregacji mirung południowych na obrazach satelitarnych o rozdzielczości 50 cm, można rozwiązać za pomocą obrazów o rozdzielczości 31 cm. Użyteczność wykorzystania zdjęć satelitarnych do identyfikacji lokalizacji kolonii mirungi południowej nie ulega wątpliwości, uzyskanie szacunkowej liczby osobników dorosłych obu płci w agregacjach jest również możliwe z użyciem tej metody, ze świadomością obarczenia wyniku pewną dozą niepewności, jednak rozpoznanie szczeniąt jest obciążone zbyt dużym błędem, by można było zarekomendować tę metodę do dokonania szacunków ich liczebności w obrębie agregacji rozrodczej.

4.3. **Identyfikacja ostoi ptaków w Antarktyce z wykorzystaniem BSP na przykładzie Przylądka Melville, Wyspa Króla Jerzego** [na podstawie: Fudala i Bialik, 2023 – Załącznik 3]

W artykule trzecim (Fudala i Bialik, 2023) przedstawione zostały wyniki inwentaryzacji ornitologicznej obszaru Przylądka Melville oraz przybrzeżnej strefy Zatoki Destrukcji z użyciem BSP, wykonanej w dwóch terminach sezonu 2022/23. Na bazie zdjęć wykonanych podczas misji drona stworzone zostały wysokorozdzielcze ortomozaiki, na których oznaczono gniazda ptaków (wskazania 3 niezależnych obserwatorów). W dniu 14.12.2022 na obszarze Przylądka Melville znajdowało się 4960 (+/-12) gniazd pingwina maskowego i 456 gniazd kormorana antarktycznego. Analizy potwierdziły również obecność co najmniej jednego gniazda mewy południowej (*Larus dominicanus*), ponad 50 żerujących na terenie kolonii osobników pochwodzioba żółtodziobego (*Chionis albus*) i co najmniej 1 pary wydrzyków brunatnych (*Stercorarius antarcticus*). Strome zbocza i osuwiska skalne obecne na przylądku sugerują również, że lokalizacja może być potencjalnym miejscem gniazdowania oceanników żółtopłetwych (*Oceanites oceanicus*) i czarnobrzuchych (*Fregetta tropica*). Ponieważ nie są to gatunki gniazdujące na powierzchni, ich wykrycie na obrazie lotniczym nie było możliwe. Kontrola w obrębie przyległego przylądkowi fragmentu Zatoki Destrukcji wykazała również obecność zgrupowań rozrodczych ptaków na przybrzeżnych wyspach: 3435 (+/- 16) gniazd pingwina maskowego na Wyspie Trowbridge, 7 gniazd kormorana antarktycznego na skale Trowbridge annex, 627 (+/-6) gniazd pingwina maskowego oraz 10 gniazd kormorana na Wyspie Middle, i 32 gniazda kormorana na niewielkiej bezimiennej wyspie zlokalizowanej między Wyspą Trowbridge a Przylądkiem Melville.

Szacunki wielkości globalnej populacji kormorana antarktycznego sprzed publikacji prezentowanego artykułu nr 3, wynosiły 11 684 pary (na podstawie Schrimpf i in. (2018) po poprawkach uwzględniających wyniki Phillips i in. (2019)). Nie uwzględniały one jednak kolonii Cape Melville w kalkulacjach, co sprawia, że szacowana liczba par tego gatunku wzrosła do 12 191. Opierając postulat na obecności 458 gniazd kormorana antarktycznego, udokumentowanych na ortomozaice z dnia 4.12.2022, w odniesieniu do kryterium B3a, zaproponowano przywrócenie Przylądkowi Melville statusu ostoi ptaków, wraz z podaniem sugerowanego przebiegu jej nowych granic. Wniosek zgłoszony do Birdlife International został pozytywnie rozpatrzony (Birdlife International, 2024c). Chociaż, jak wyjaśniano we wstępie niniejszej pracy, ostoje ptaków nie przekładają się na szczególne środki ochrony danego obszaru, aktywny status ostoi ptaków jest silnym argumentem przy staraniach o wcielenie danego fragmentu Antarktyki do sieci Antarktycznych Obszarów Szczególnie Chronionych (ASPA).

Wykonana w ramach niniejszej pracy inwentaryzacja Przylądka Melville oraz przyległego fragmentu Zatoki Destrukcji była pierwszą aktualizacją danych na temat objętego kontrolą obszaru od 1987 roku (Shuford i Spear, 1988). Potencjał wykorzystania bezzałogowych systemów powietrznych do wykonywania inwentaryzacji ostoi ptaków został potwierdzony w ujęciu praktycznym, tym samym zrealizowany został cel trzeci dysertacji.

4.4. Wykorzystanie uczenia maszynowego do automatycznej detekcji i liczenia gniazd kormoranów antarktycznych na ortomozaikach lotniczych [na podstawie: Cusick & Fudala i in., 2024 – Załącznik 4]

W artykule czwartym (Cusick & Fudala i in., 2024), który powstał w odpowiedzi na postawiony w dysertacji cel czwarty, zaproponowano metodę automatycznej detekcji i liczenia gniazd kormoranów antarktycznych przy wykorzystaniu modelu opartego na architekturze „You Only Look Once" (YOLO). Wyniki uzyskanego modelu sugerują, że wykorzystanie BSP do inwentaryzacji kolonii kormoranów antarktycznych, w połączeniu z algorytmami uczenia maszynowego, może zapewnić wiarygodne i szybkie szacunki liczby gniazd kormoranów (wynik automatycznej detekcji na poziomie F1 > 0,95). Kod, dane i wytrenowany model, pozwalające na pełną odtwarzalność wyników, zostały publicznie udostępnione (https://github.com/Appsilon/Antarctic-nests). Jednocześnie, w suplemencie artykułu czwartego przedstawiono wyniki prowadzonego z użyciem BSP, na terenie ASMA nr 1, monitoringu kormoranów antarktycznych.

Extended abstract

1. Introduction

1.1 Antarctic nature conservation

The legal status of the Antarctic is governed by a set of international agreements that build the functional complexity of the Antarctic Treaty System. The key document and the basis for the System's operability is the Antarctic Treaty (1959), signed in Washington, enriched over the years with additional legal acts crucial for the protection of the Antarctic natural heritage: Protocol on Environmental Protection to the Antarctic Treaty (1991), Convention for the Conservation of Antarctic Seals (1972), Convention on the Conservation of Antarctic Marine Living Resources (1980) or, often overlooked in the list of linkages to the Treaty (although undeniably linked to it), the Agreement on the Conservation of Albatrosses and Petrels (2001). In 1991 in Madrid, the Antarctic was declared *'a natural reserve devoted to peace and science'* (Article 2 of the Protocol on the Environmental Protection). This statement alone, contained in the Madrid Protocol, did not entail a rapid cascade of consequences, but it should be boldly emphasised that the granting of such status to the entire continent, together with its surrounding ocean waters, was an unprecedented event in human history (moreover, probably unrealistic to repeat in the Anthropocene epoch) and laid the foundation for the more detailed provisions of the Protocol, which, since its entry into force in 1998, have served to protect the unique Antarctic ecosystem. One of the causative instruments of the Madrid Protocol became Annex V, under which Antarctic Specially Managed Areas (ASMAs) and Antarctic Specially Protected Areas (ASPAs) can be created. ASMAs can be established in areas (both onshore and offshore) where there is (or may be in the future) activity, in order to facilitate its planning and coordination, avoid possible conflicts, improve cooperation between Parties or reduce environmental impacts. Presence within an ASMA, unlike presence on an ASPA, does not require justification or a permit, which in the context of nature conservation does not give them equivalent enforcement power, but in practice ASMA Management Plans contain an inventory of values requiring special protection and help coordinate activities in the area so that they take place at the least environmental cost. An ASMA

Management Plan may include recommendations to restrict activities in what the Managers consider to be a naturally or historically valuable area. Although, in accordance with Article 3 of Annex V of the Protocol, '*Any area, including any marine area, may be designated as an Antarctic Specially Protected Area to protect outstanding environmental, scientific, historical, aesthetic or wilderness values, any combination of those values, or ongoing or planned scientific research*', only 75 sections of the Antarctic are currently strictly protected, making the share of Antarctic Specially Protected Areas in the total Antarctic area negligible (~ 0.076%, as of 2024, based on the area of ASPAs included in the in the Management Plans (ATS, 2024)) and, largely for logistical reasons or driven by national interests, locationally linked to proximity to research stations. This can be justified by the fact that the Party suggesting that an area be protected is required to submit to the Consultative Parties a proposal for a Management Plan for the area, at the same time presenting its specific values and, in case of unanimous acceptance of the Plan by the other Parties, it is obliged to implement it and to review and update it regularly (at least every 5 years). This requires constant monitoring of the animate and inanimate elements of the area's nature, which is a responsible and costly undertaking. The establishment of an ASPA also requires its depositary to designate the relevant governmental authorities, which will have the power to authorise all activities within the boundaries of the ASPA and will be required to assess their environmental impact. Thus, while Antarctic Specially Protected Areas have a great deal of power, they also burden their depositories with a number of responsibilities related to the area management process, making it impossible in practice for the ASPA network to systematically and equitably protect all the values it is designed to. This is particularly the case for marine areas that are theoretically eligible for special protection status, but the legitimacy of their creation is challenged by Parties particularly interested in their commercial potential.

The Southern Ocean and the seas surrounding Antarctica are managed by the Commission for the Conservation of Antarctic Marine Living Resources (CCAMLR). A key objective of CCAMLR is to minimise the risks to biodiversity associated with climate change (IPCC, 2023) and the increasing demand for marine living resources through their conservation and rational use in the Southern Ocean (Convention on the Conservation of Antarctic Marine Living Resources, 1980). The designation of Marine Protected Areas

(MPAs) is one mechanism to support this goal. To date, two MPAs have been adopted in the waters surrounding mainland Antarctica and nearby islands (CCAMLR, 2009; 2016) resulting in a number of marine fragments being subject to enhanced conservation or management measures. Three further MPAs and a potential fourth are still in the planning stages (CCAMLR, 2021). However, consensus (a requirement of CCAMLR to implement management measures) on the designation of the proposed MPAs has not been reached for years. In order to provide information on the impacts of fishing on dependent species, CCAMLR established the Ecosystem Monitoring Program (CEMP) in 1989 (CCAMLR, 2013). The main objectives of the CEMP are to detect and record significant changes in critical components of the marine ecosystem in the Convention area to serve as a basis for the conservation of Antarctic marine living resources, and to distinguish between changes resulting from the harvesting of commercial species and changes resulting from environmental variability. The main tool of the CEMP is to monitor key life history parameters of selected dependent species to detect changes in the abundance of harvested species. "Dependent species" are marine predators for which species targeted by commercial fisheries are the main component of the diet. "Krill-dependent species" included in the CEMP monitoring are species that can be observed on land, such as pinnipeds and birds.

Even before the Environmental Protocol entered into force, Birdlife International, in collaboration with the Scientific Committee on Antarctic Research (SCAR), jointly took the first steps to create a list of Antarctic Important Bird and Biodiversity Areas (IBAs). The first report, with proposed threshold values for particular bird species, covered 101 IBAs (Harris et al., 2011). Four years later, an update report was published, which already included 204 terrestrial IBAs (Harris et al., 2015), as a result of the enrichment of the database with a significant number of new locations and, at the same time, the declassification of those without a reliable data background. All IBAs established in 2015 have active status and, since their publication, the database has been enriched with 64 marine range sites (Handley et al., 2021; Birdlife International, 2024a;) and only two new terrestrial sites: Ryder Bay (Birdlife International, 2024b based on Philips et al., 2019) and Cape Melville (Birdlife International, 2024c based on Fudala and Bialik, 2023). Unfortunately, the designation of an area as an IBA is not associated with specific tools to strengthen the protection of the area.

Nonetheless, keeping an inventory of the locations of key seabird aggregations is extremely important for the conservation of Antarctic ecosystems. Seabirds offer a particular opportunity to identify high-quality marine areas and, in addition, as top predators, are considered indicator species whose populations reflect the state of the surrounding marine environment (Le Bohec et al., 2012). A comprehensive dataset on the distribution of key Antarctic bird colonies and associated abundances, augmented with bird tracking data packages and appropriate extrapolation methods, can allow for an estimate of the distribution of bird feeding grounds at sea (Dias et al., 2018), and allows for an assessment of how marine IBAs overlap with the currently adopted and proposed network of management areas (mainly MPAs) and how krill fisheries have likely overlapped with marine hot-spots in recent decades. Marine IBAs in Antarctic waters identified by Handley et al. (2021) suggest that if the proposed MPAs were adopted, the permanent protection of high-quality areas for penguin species would increase by 49-100%, depending on the species. In addition, data presented by the mentioned scientific team show that despite the narrowing range of krill fisheries in the Antarctic over the past five decades, a consistently disproportionate amounts of krill are harvested in marine IBAs compared to the total acreage over which fisheries operate.

1.2 Threats to the Antarctic environment

The Antarctic has been struggling with man for more than 200 years. During this period, the Antarctic region has gone through successive phases of exploitation of its living marine resources in the form of sealing, whaling and fishing, the latter activity of which continues to this day. Populations of some pinnipeds and whales were brought to the brink of extinction by human activities carried out in the 19th century. Within months of the discovery of the South Shetlands in 1819 (Jones, 1975), the islands became the site of intense seal hunting activity. The result of this unbridled activity in the 1820s and then the 1870s was the almost complete extirpation of Antarctic fur seals (*Arctocephalus gazella*) from the entire archipelago. Fur seals were not recorded in South Shetland until 1902, when an individual was observed (and subsequently killed) on Nelson Island during a Swedish polar expedition (Andersson, 1905). It was not until 1959 that there are the first reports of a single female with a pup observed at Cape Shirreff on Livingston Island (O'Gorman, 1961). Other representatives of Antarctic pinnipeds have also suffered a cruel fate. Since the early 19th century, southern elephant seals (*Mirounga leonina*) have been hunted for the oil extracted from their fat (Bonner, 1982), and over the century their numbers have declined significantly, particularly in their subantarctic breeding range. The exact number of southern elephant seal caught is not known, but some estimates put the number at over one million individuals of both sexes (Le Boeuf and Lows, 1994). Other Antarctic seals, mainly the crabeater seals (*Lobodon carcinophagus*) and the Weddell seals (*Leptonychotes weddellii*), were killed for human and dog food for most of the 20th century, until the adoption of the Environmental Protocol, which banned, among other things, the presence of dogs in the Antarctic region. Concern that a new phase of the commercial sealing industry would begin in the Antarctic with the gradual recovery of the pinniped population led to the adoption of the Convention on the Conservation of Antarctic Seals (1972). The intensive human activity in the Antarctic region does not end with the exploitation of the pinniped populations. The decline of whale stocks that had been intensively hunted in the North Atlantic and North Sea brought another wave of human-induced ecological destruction, attracting whalers to the Southern Ocean in the early 1890s (Tønnensen and Johnsen, 1982). The first Antarctic whaling expedition sailed from Scotland into the Southern Ocean in 1892, followed shortly after by a Norwegian expedition in the same season. (Tønnensen and Johnsen, 1982). The era of Antarctic commercial whaling, in which many other countries besides the UK and Norway participated, lasted until the 1970s. It was unfortunate that whaling began in the Antarctic just when it had become as deadly effective as ever. The end result was the drastic collapse of most Southern Ocean whale populations (Ballance et al., 2006). By the 1920s, there was growing international concern about the sustainability of commercial whaling. In 1930, the Bureau of International Whaling Statistics was established to monitor the industry, and the following year the Geneva Convention for the Regulation of Whaling (1931) was created, which became the foundation for the International Convention for the Regulation of Whaling (1946) and the International Whaling Commission (IWC). The Convention for the Regulation of Whaling should not be confused with what we would today call an 'environmental agreement'. This convention, as Schiffman (2003) points out, was an agreement between whaling states in the interests of whaling. The IWC introduced a moratorium on all commercial whaling from the 1985-1986 season onwards, thus making a radical turn from the course previously taken. As Sigvaldsson (1996) explains, in order to understand this abrupt change, it is necessary to consider multi-factorial analyses of this transformation, taking into account global economic and technological advances, constitutional issues and the composition of the negotiating parties, the power structure of the international system, international organisations and the linking networks. Despite the introduced memorandum, Japan continued whaling in the Antarctic between 1986 and 2018 under a 'scientific whaling' permit (a controversial practice that ended when, in 2019 Japan left the IWC). Thus, there is no longer legal commercial whaling in Antarctic waters. The total Antarctic whaling catch between 1904 and 1978 was calculated to be 1 393 254 whales (Tønnensen and Johnsen, 1982), a level that was completely unsustainable ecologically and, according to some interpretations, also ethically (Garner, 2013). After pinnipeds and whales, fish stocks were also brought to alarming levels as a result of intensive industrial fishing, which started in the late 1960s. The fishery developed mainly in the coastal waters around South Georgia, South Orkney and South Shetland and led to severe stock depletion of many finfish species. As of 1992, the abundance of marbled rockcod (*Notothenia rossii*) around South Georgia was estimated to be less than 5% of the original stock size in 1969 (Kock, 1992). The Commission for the Conservation of Antarctic Marine Living Resources has adopted conservation measures to support the recovery of these species by prohibiting directed fishing and setting strict bycatch limits. The depletion of cod icefish species has had a negative impact on Antarctic shag (*Leucocarbo bransfieldensis*) populations, which as one of two Antarctic bird species (along with the South Georgia shag *Leucocarbo georgianus*) feeds mainly on benthic fish (Casaux and Barrera-Oro, 2006).

Currently, 97% of all Southern Ocean catches involve Antarctic krill (*Euphausia superba*), although they represent a seemingly small (about 1%) proportion of the estimated total krill biomass. Antarctic krill constitute a key biomass, estimated at 300-500 Mt (Atkinson et al., 2009; Siegel and Watkins, 2016), and play a pivotal role in pelagic food webs as a major food source for a variety of predators, including many Southern Ocean endemits. Krill catches in the last reported season (2023) were almost four times higher than in 2000 (CCAMLR, 2024). Commercial fishing for Antarctic krill began in 1961, and during the following decade small-scale krill fisheries were conducted as part of the research phase of fishery development. Multinational krill fisheries were active from the early to mid-1970s (CCAMLR, 2024). The tonnage of the Antarctic krill fishery has fluctuated over the last 50 years, reaching 528 000 tonnes in 1982 and then falling to 66 000 tonnes in 1993. Catches have maintained an overall upward trend over the past decade, reaching another peak of around 451 000 tonnes in 2020 and then a gentle decline to 416 000 tonnes in 2022 and 424 000 tonnes in 2023 (CCAMLR, 2024). In the 1980s, fishing took place in the Atlantic, Pacific and Indian Ocean sectors, but since the 1990s it has taken place almost exclusively in the Southwest Atlantic sector. In sub-area 48.1, which includes the waters surrounding the South Shetland Islands, the catch in 2023 was 153 587 tonnes of krill, double the catch in this sector in 2000 (CCAMLR, 2024). A catch rate of less than 1% of the estimated total population may seem negligible, but in order to assess the impact of harvesting on ecosystem stability, it is crucial to evaluate the spatial distribution of the catch and the age structure of the stocks caught (Panasiuk et al., 2024). Regional biomass estimates from local monitoring programmes in the main fishing areas have been sufficiently regular to provide time series of data revealing very large fluctuations in local abundance, and this is true in Bransfield Strait and the waters extending north of the South Shetlands, where inter-annual differences can be as much as 2-3 orders of magnitude (Reiss et al., 2008). Knowledge of such fluctuations in abundance in key locations of fishing activity makes the question underlying most discussions of krill management in CCAMLR even more pressing: to what extent does the krill fishery threaten local predators? To illustrate the ecological value of the harvested tonnage, it is worth noting that estimates of the total krill requirements of populations of selected bird species may be close to the catch limit (620 000 tonnes). An example is the Antarctic petrel (*Thalassoica antarctica*), whose krill requirement is estimated to be at least 680 000 tonnes each year (Descamps et al., 2016).

The examples presented of interference with the Antarctic environment through exploitation of its living resources are not the only form of human presence in the region. Antarctic tourism has developed intensively over the last 3 decades. The number of recreational visitors to the region has been increasing consecutively from year to year (excluding the 2020/21 season related to the COVID-19 pandemic). According to a document from the International Association of Antarctica Tour Operators (IAATO), which keeps statistics received from reports from operators affiliated with the organisation (which does not represent all tourism activity taking place in Antarctica), more than 120 000 tourists visited Antarctica in the 2023/24 season, of which almost 80 000 went ashore (IAATO, 2024). This compares with less than 8000 tourists coming ashore in the 1993/94 season (IAATO, 2024). In addition to the increase in the number of tourists, tourism has diversified: from local excursions in small motor boats to ship cruises, airline operations, luxury camping and a variety of activities, including scuba diving, climbing, surfing, skydiving and marathons (Makanse, 2024). Infrastructure, including airstrips, research stations and wharves, continues to be developed to support scientific and tourism activities, with much of this activity taking place on ice-free land, which represents a small percentage of the total (Pertierra et al., 2017). Starting with the carbon footprint and the level of pollution (including noise pollution) that tourism brings with it, the risks associated with the ever-increasing interest in Antarctica relate to habitat destruction, animal disturbance, the transfer of alien species and their propagules, as well as disease pathogens. Through IAATO, the tourism sector seeks to establish environmental standards that are sometimes incorporated into the governance of the Antarctic Treaty System (e.g. tourism-specific guidelines adopted at the annual Antarctic Treaty Consultative Meeting (ATCM)), thus helping the organisation to gain legitimacy (as a self-regulatory body) among Treaty Parties, NGOs and tourists. According to a meticulous analysis of materials promoting Antarctic tourism by Varnajot et al. (2024), four types of narratives can be identified that form the core of 'Antarctification' (a social phenomenon involving the creation of stereotypical images of Antarctica in order to enhance its attractiveness in the minds of potential tourists): Antarctica as a place (1) of exploration; (2) wild and empty; (3) full of superlatives; and (4) requiring environmental stewardship and ambassadorship from visitors. The use of these narratives encapsulates Antarctica in specific imaginaries designed to attract tourists. As the authors of the paper emphasise, identifying these dominant narratives is crucial as they can significantly influence conservation programmes and Antarctic heritage. A narrative based on scientific arguments, demonstrating the threats posed by the ever-increasing number of tourists, should be initiated by the scientific community, as it should not be expected that IAATO, as a trade association, would take measures that would restrict its members from engaging in tourism activities.

The above examples of the material human presence in Antarctica and its disruptive effects on its ecosystem are unfortunately not the only contribution that humans have made to the list of threats facing the region. Anthropogenic global warming is expected to be particularly pronounced in the polar regions compared to the global average, an effect known as polar amplification (Casado et al., 2023). Global public awareness of the magnitude of extreme weather events appears to be increasing, linked to the damage these events cause to human habitats. These events are increasingly equated with the burning of fossil fuels and the progressive degradation of nature. A great deal of attention has been devoted to this issue, particularly in the context of evaluating the economic gains and losses incurred by individual countries. However, it is worth noting that the most extreme 'heat wave' ever observed on the Earth's surface occurred over East Antarctica in March 2022 (Siegert et al., 2023), which, like many extreme phenomena, had a cascade of consequences. The March 2023 phenomenon, known as the atmospheric river, led to the warming of land ice surfaces and ice shelves, the break-up of sea ice and the collapse of the Conger Ice Shelf (Wille et al., 2024). Extreme temperatures are not the only threat to the Antarctic environment. Extreme cyclones (Francis et al., 2022) or rapid loss of sea ice (Turner et al., 2017) are also included in the list of phenomena of high concern and strong impact on ecosystem stability. Extreme events, and in particular their difficult-to-predict combinations, can lead to lower reproductive success rates for Antarctic animals, a prime example being the Adélie penguins (*Pygoscelis adeliae*) from the colony on Petrel Island (Île aux Pétrels), which suffered a complete reproductive failure in the 2013/14 and 2016/17 seasons, due to the combination of factors including: heavy (and unseasonal) rainfall, which cooled the chicks; the presence of extensive fast sea ice; and the absence of polynyas (Ropert- Coudert et al., 2014; 2018). In the ocean, extreme temperature events are commonly referred to as marine heat waves and are defined as prolonged periods of anomalously high temperatures that can have significant impacts on marine biodiversity and ecosystems. Between 2002 and 2018, 19 such events have been documented in the Southern Ocean, with significant increases in chlorophyll-a concentrations in response (Montie et al., 2020). They are predicted to increase in frequency and intensity over the coming decades, with their effects being amplified in some areas by extreme acidification (Burger et al., 2022), although, as noted by Siegert et al. (2023), regional variability and data paucity pose challenges for

Southern Ocean forecasting. While on the subject of extreme phenomena, it is difficult not to mention the 'ozone hole' over Antarctica, the formation of which in the atmosphere was largely caused by a particular class of chemicals (Newman et al., 2006). The effects of its existence are still being felt and relate to both biology (harmful effects of UV radiation on eukaryotes and prokaryotes (Abbasi and Abbasi, 2017)) and climatology (cooling of the polar stratosphere, changes in tropospheric circulation in the southern hemisphere and its surface climate (Thompson and Solomon, 2002, Thompson et al., 2011)).

Given that the intensity of human activities in Antarctica is unlikely to cease its upward trend, and that pressures on the Antarctic environment from climate change and activities outside the region will also increase, there will be further and more pressing conservation challenges that will require the development and implementation of effective environmental policies (Hughes et al., 2018; Chown et al., 2022). This, in turn, requires collaboration between scientists and policy-makers and highlights the value of long-term environmental monitoring.

1.3 King George Island

King George Island constitutes part of the South Shetland archipelago, extending in a 540 km long line parallel to the Antarctic Peninsula. With an area of 1150 km^2, it is the largest of the 11 main islands that comprise the archipelago. There are 14 science and logistics stations on the island, 10 of which have year-round base status (COMNAP, 2024). The Arctowski Polish Antarctic Station is one of the outposts operating year-round and is located in Admiralty Bay. Notwithstanding the fact that numerous Antarctic national programmes have their logistical bases on King George Island, the northern and north-eastern parts of the island are not subject to regular environmental monitoring. This is due to the adverse weather conditions and the island's topography, which include strong winds, drifting ice fragments, coastal skerries and glaciers that separate ice-free areas (Rakusa-Suszczewski, 2002). The restrictive safety procedures associated with limited rescue capabilities, and the financial constraints associated with the high costs of logistical operations (Heinrich and Norris, 2024) and maintenance of national Antarctic programmes also contribute to the situation (Fowler, 1988; Sánchez and Njaastad, 2014). There are 6 Antarctic Specially Protected Areas established on King George Island, 1 Antarctic Specially Managed Area and 13 Important Bird and Biodiversity Areas (Fig.1).

ASPA 151 (ATCM, 2024a) and ASPA 128 (ATCM, 2024b) are managed by Poland, while ASMA 1 (ATCM, 2023) is co-managed by countries active in the area, including Poland. Of the 13 terrestrial IBAs on King George Island, 8 have been established on the basis of archival data dating back 37 years or more, so their status can be considered uncertain (Birdlife Data Zone, 2024d). There are 13 bird species regularly nesting on King George Island, including 3 species of penguin of the genus *Pygoscelis*. Pinnipeds such as the Antarctic fur seals, Weddell seals, crabeater seals, leopard seals (*Hydrurga leptonyx*) and southern elephant seals can be found resting or moulting on the shores of the island. Of the species listed, only southern elephant seals and Weddell seals breed on King George Island.

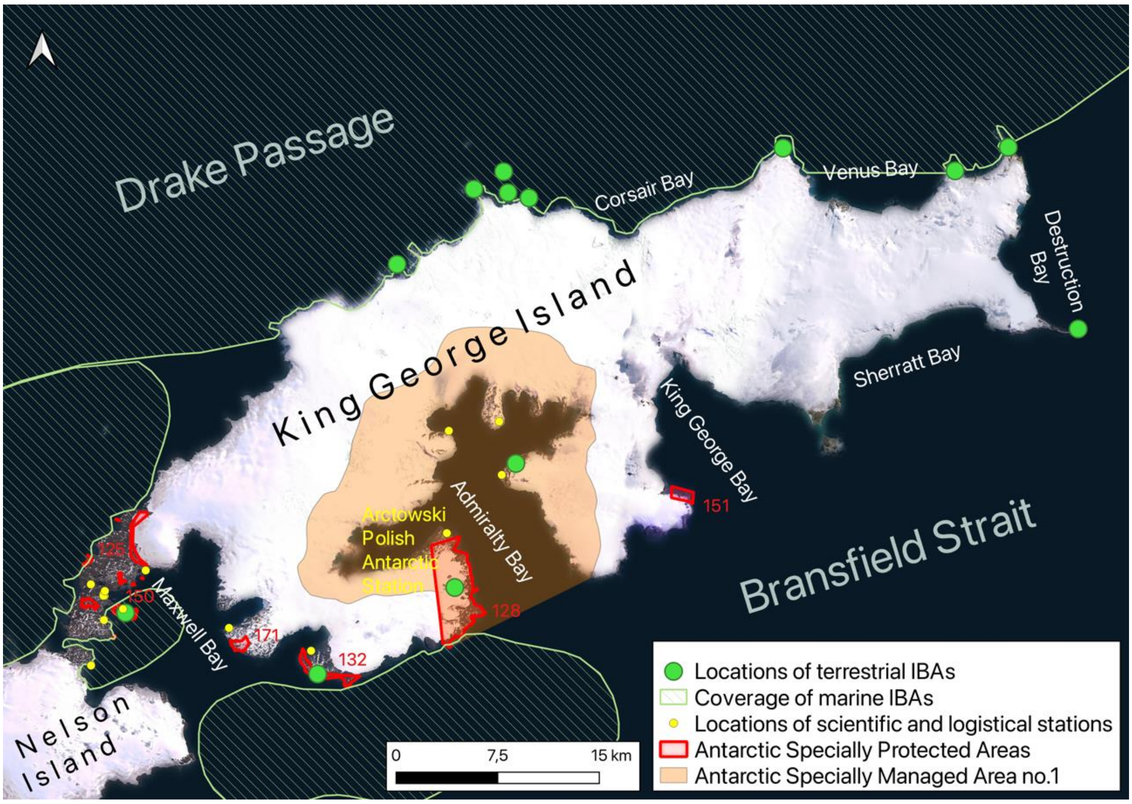


Figure 1. Study Area (map shapefiles based on Quantarctica data package (Matsuoka et al., 2021)).

1.4 Use of RPAS for assessment of Antarctic animal population size

Unfortunately, it was war, not science, that sparked the technological development of remotely piloted aircraft systems (RPAS). These spying and killing machines have gone through several stages of development and have moved from a strictly military function to the hands of commercial users. Thus RPAS (also referred to as drones) have become common tools for ecological monitoring and management over the past two decades (Ivosevic et al., 2015; Robinson et al., 2022), enabling data of unprecedented detail. Relatively inexpensive, easy to use, with an ever-evolving backlog of user-friendly applications, drones have become an alternative to the methods for decades used to find and count aggregations of animals in the Antarctic: aircraft, helicopters or ground controls (e.g. CCAMLR, 2013; Lyver et al., 2014). The first flights over Antarctica were a real breakthrough in terms of exploration and also opened the Antarctic chapter of remote sensing. On 20 December 1928, Hubert Wilkins recorded the historic sentence in his diary: '*For the first time in history new land was beeing discovered from the air*' (Clancy et al., 2014). Hubert Wilkins, along with Carl Eielson, flew from Deception Island over the Antarctic Peninsula taking notes and making documentation with a hand-held Kodak 3A camera and two film cameras. It was not until seven years later that Herbert Hollick-Kenyon and Lincoln Ellsworth first crossed the Antarctic continent with a flight between Dundee Island and the Ross Ice Shelf (Joerg, 1937). After the end of the Second World War, one can speak of the emergence of mapping using aerial photography, first with the Ronne Antarctic Research Expedition (RARE) programme and then the Falkland Islands Dependencies Aerial Survey Expedition (FIDASE) (Pope et al., 2014). At a time when each aircraft ascent resulted in new discoveries and filling a blank on the Antarctic map, no one thought about the ecological impact of these activities. In the Wilkins report (Wilkins, 1929), we even find mention of collisions between aircraft and birds, obviously as part of the lament expressed for the technical state of the machine. It was only many years later that a slow transformation of thinking from an exploitation-exploration narrative to one focused on preserving the natural heritage of the Antarctic as an asset for all mankind began to take place.

The need for research into the effects of air fleet disturbance on Antarctic and subantarctic fauna has been highlighted by scientists (e.g. Kooyman and Mullins, 1990; Fraser and Patterson, 1997), resulting in the adoption by the Parties at ATCM 2004 of recommendations for minimum flight heights and distances from bird colonies (ATCM, 2004), which, as Harris (2005) points out, were a form of questionable 'compromise' between operational needs and animal welfare and deviated significantly from the values suggested by SCAR's advisory body. The scientific studies that have analysed the effects of air fleet activity have mostly been, as Harris (2005) concludes, opportunistic, but have nevertheless suggested the presence of behavioural and physiological responses in birds and seals resulting from low flying aircraft and helicopters (e.g. Culik et al., 1990; Cooper et al., 1994; Giese and Riddle, 1999). While most studies have shown a small and transient effect, some have linked declines in local populations to increased activity in the aerial zone (Thomson, 1977; Wilson et al., 1991). Interest in how penguins may react to aircraft overflights has been linked to an incident of mass mortality of about 7000 individuals of king penguin (*Aptenodytes patagonicus*) on Macquaire Island, thought to be a panic reaction to a Lockheed C-130 Hercules aircraft flying within about 1 nautical mile of the colony at a relatively low altitude (Cooper et al., 1994). The hypothesis put forward by Cooper et al. (1994) is that the birds died as a result of escape to the edge of the colony, which is bounded by a rocky barrier, and subsequent suffocation due to overcrowding by panicked animals. The escape itself was not observed and the dead birds, lying in tall piles, were discovered approximately 10-12 days after the event.

According to an analysis of publications on the use of drones in Antarctica by Pina and Vieira (2022), the first scientific article on the subject appeared in 2006 (Funaki et al., 2006), and since then there has been a regular increase in examples of the use of drones for Antarctic research. It is safe to say, and this will certainly be confirmed by scientists involved in Antarctic population studies, that drones have revolutionised this sector of research.

Experiments conducted to assess the response of individual Antarctic bird and mammal species to the presence of a drone have shown that the behavioural responses of animals to a drone can be less intense than those resulting from the presence of a ground observer (Krause et al., 2021) although, as emphasised by other authors (Weimerskirch et al., 2018b), it is important to take into account the animal species and its specific response to a particular survey methodology in such considerations. Concerns about the use of drones in Antarctica, in terms of their potential negative impacts on wildlife, prompted the Antarctic Treaty Parties to initiate a process to develop environmental guidelines for the use of RPAS in Antarctica, which were adopted through a resolution (ATCM, 2018). The avifauna of King George Island, as previously mentioned in section 1.3, consists of 13 regularly nesting bird species, of which 9 show potential for monitoring their breeding populations with a drone, as they are surface nesters. The habits of the remaining 4 species eliminate the possibility of identifying their nests on aerial photographs, due to the fact that they are hidden on rock shelves, in rock cavities, debris and in underground burrows. In addition to the criterion of having a surface nest, individuals of the species in question must have sufficiently large body dimensions or sufficient contrast to the substrate to be recognisable on aerial photographs at a drone flight ceiling set so as not to cause excessive stress to the animal, a defensive reaction or, in extreme cases, nest abandonment. Given the tundra-like nature of King George Island's vegetation (Olech, 2002), with the largest vascular plant growing to a height of several centimetres, it is difficult to imagine conditions more conducive to the use of a drone for mapping nests and conducting terrestrial animal censuses. The potential of drone imagery to identify *Pygoscelis* penguins, and thus to be used for colony inventories of these birds, has been confirmed in scientific publications (e.g. Perryman et al., 2011; Ratcliffe et al., 2015; Pfeifer et al., 2019; Shah et al., 2020). Korczak-Abshire et al. (2019) used images captured by a fixed-wing drone to identify four bird species (Adélie penguins (*Pygoscelis adeliae*) and chinstrap penguins (*Pygoscelis antarcticus*), southern giant petrels (*Macronectes giganteus*) and Antarctic shags) and two seal species (southern elephant seals and Weddell seals). The study by Mustafa et al. (2019) aimed to test the ability to identify individual species of Antarctic birds and pinnipeds in a drone image, distinguishing between different Ground Sampling Distance (GSD) values. The use of the drone for research on Antarctic birds and mammals is not limited to simply identifying them in images. Images from a drone mission can also be used to measure animals' bodies, estimate their mass and age, or determine the condition of individual animals, as in the case of the photogrammetry study of leopard seals and southern elephant seals (Goebel et al., 2015; Krause et al., 2017; Fudala and Bialik, 2020). Drones can also be used to control an area in the event of an animal outbreak, as exemplified by the work of Tyndall et al. (2024). The novel application presented by this team involved combining drone imagery with neural networks to assess mortality in a colony of northern gannets (*Morus bassanus*) affected by a highly pathogenic avian influenza (HPAI H5N1) outbreak. The team succeeded in using a neural network to effectively detect and count live and dead birds. The presence of a highly pathogenic variant of avian influenza has also been confirmed in the Antarctic (Banyard et al., 2024), and recommendations made by the SCAR

Antarctic Wildlife Health Network expert group, include the use of drones to monitor potential outbreaks (Dewar et al., 2023).

1.5 Studying Antarctic animal populations with satellite imagery

In the same year that the Antarctic Treaty was signed, the US satellite Explorer 6 was launched into space and took the first satellite image of our globe (NASA, 2024). At the time of writing, the European Space Agency estimates that there are around 10 200 active satellites in orbit. (ESA, 2024a). Partly due to the declining cost of launching them, companies and governments are putting thousands of new objects into orbit annually (GAO, 2022). The number of active satellites is expected to increase to more than 58 000 by 2030 (GAO, 2022). Aside from the worrying lack of control over the space debris that is systematically filling up humanity's common space (ESA, 2024b), and the hidden environmental costs it entails (Gaston et al., 2023), satellites are undoubtedly a technological breakthrough in terms of ecosystem monitoring tools (Yang et al., 2013), especially in less accessible regions of the Earth, which includes Antarctica. In the rapidly evolving landscape of satellite technology, ingenious companies continue to achieve milestones, expanding their constellations to provide not only higher revisits, but increasingly better imaging and resolution capabilities (e.g. Boeing, 2024; Intelsat, 2024, Spacex, 2024). At the current rate of development of satellite technology and the ever-decreasing cost of space ventures, it is relatively easy to imagine that within the next decade satellite monitoring will be an important and widely available tool for scientists working in Antarctica, which, combined with the use of artificial intelligence algorithms, including machine learning, could shed a very different light on some natural phenomena. The current limitations of using Very High Resolution (VHR) satellite imagery for research in Antarctica are mainly related to its availability and price. A circumpolar orbit (polar orbit) is defined as an orbit that passes over or near the poles. Satellites in circumpolar orbit typically orbit between 200 and 1000 km above the Earth (ESA, 2024c). During the satellite's full flyby, the Earth performs a rotation, resulting in imaging different parts of the globe in narrow bands corresponding to the satellite's field of registration. After a few (e.g. Sentinel-2: 5 days (ESA, 2024d)) or several days (e.g. Landsat 8: 16 days (USGS, 2024)), the satellite re-images the same area.

Schwaller et al. (1984) first presented the idea of using 15 m resolution satellite imagery to index the extent of large Adélie penguin colonies, based on the spectral characteristics of guano. The authors of the paper emphasise that an inventory of the size of Antarctic penguin populations is needed to understand how the Antarctic food chain has responded to the recently ended phase of intensive whaling, which has resulted in the almost complete removal of key krill predators from the system. Satellite imagery thus became, as early as the 1980s, a tool that, at least in theory, could allow large-scale estimates of penguin population size. Guinet et al. (1995) used satellite imagery with a resolution of 10 m to show changes in the area occupied by a king penguin colony on the subantarctic Pig Island (Île aux Cochons) between the 1962 and 1988 seasons. According to the team's spatial analyses, the area of the colony increased by about 56% over these years, allowing them to estimate a change in population size from 319 000 pairs in 1962 (based on a study by Bauer, 1967) to 494 000-678 000 pairs, depending on the assumed average nest densities of the species' colonies. Subsequent analysis by Weimerskirch et al. (2018a), based on satellite and aerial imagery, showed a dramatic decline in the colony population from an estimated 494 000 pairs in 1988 to 76 000 pairs in 2015. The enigmatic case of the colony on this patch of land between Madagascar and Antarctica has prompted French scientists to raise funds and organise the first scientific expedition to Pig Island in 37 years (and the third ever) in 2019 (Kintisch, 2020). While some aspects of the difficulties involved in planning and conducting Antarctic expeditions are described in Kintisch's (2020) article, the shift in the location of research interests to the south, makes all aspects even more complicated and costly.

The emperor penguin (*Aptenodytes forsteri*) is a unique case of an animal that breeds in the harshest climatic location in the world: the continental part of Antarctica. There is no practical way to survey the entire population of this species, other than using satellite imagery. Reaching all the colonies during the breeding season, which falls in the Antarctic winter, is limited not only by the technical possibilities (the species breeds on sea ice, in the middle of the Antarctic winter), but also, and perhaps above all, by the financial aspect of the potential undertaking. In the context of the space age and the human presence in space, it is challenging to argue that a census of a bird nesting on Earth is technically impossible. Barber-Meyer et al. (2007) were the first team to use high-resolution images taken by the

Quick-Bird-2 satellite to determine the abundance of selected emperor penguin colonies in the Ross Sea. The sea ice on which emperor penguins form their breeding aggregations provides an ideal background for detecting signs of their presence in satellite imagery. This has led to the application of space technology to search for and assess the population size of this species being practised successfully by various research teams (e.g. Fretwell and Trathan, 2009; Fretwell et al., 2012; Ancel et al., 2014; Fretwell and Trathan, 2021; Fretwell, 2024). Unlike ice forming glaciers or shelves, sea ice is homogeneous and free of colour perturbations. The spectral reflectance of sea ice gives an image of a pure white or slightly blue surface. The only exception to this is where the droppings of penguin reproductive aggregations create patches of reddish-brown colour visible on satellite imagery. Unfortunately, as concluded by Ancel et al. (2014), the identification of guano deposits on sea ice in satellite images may be hampered by their burial by snowfall or evolving sea ice topology. Antarctic penguins are undoubtedly uniquely predisposed to being found on satellite imagery, due to their size and the fact that they form large groups on open land or ice during the breeding season. However, they are not the only Antarctic bird species whose populations have been studied using satellites. An example of another, much smaller species that forms extensive breeding colonies deep within the Antarctic continent is the Antarctic petrel. The team of Schwaller et al. (2018) made the first continental-scale attempt to detect the breeding locations of Antarctic petrels using Landsat-8 satellite data. With the proposed algorithm, a large number of potential breeding colonies of Antarctic petrels were identified. This work therefore represents a first step towards the discovery of previously unknown petrel colonies and more comprehensive, routine monitoring of this species. There are examples of the use of satellites to monitor bird populations with methodologies based on the detection of individuals. Fretwell et al. (2017) used the impressive body size and bright plumage of two species of albatrosses in the genus *Diomedea*, contrasting with the vegetation typical of their nesting sites, to detect them on satellite imagery at breeding colonies in South Georgia (wandering albatross (*Diomedea exulans*)) and the islands of the Chatcham archipelago (southern royal albatross (*Diomedea epomophora*)). World-View-3 images with a resolution of 31 cm were used for this task. As the examples presented here demonstrate, satellite imagery has been successfully used to study Antarctic bird species. However, the Antarctic terrestrial macrofauna, which is relatively poor in species diversity, is also formed by pinnipeds. Their body size and the fact that reproduction prompts them to establish a connection with the land make them a grateful object of study using satellite telemetry. Deprived of their element, pinnipeds are motorically passive in the terrestrial environment and their energy expenditure is limited to the necessary losses devoted to reproduction and moulting. La Rue et al (2011) used Quick-Bird-2 and World-Wiew-1 satellite imagery in their study to attempt to estimate the size of the Weddell seal population in Erebus Bay. An extensive body of population data from Erebus Bay, derived from monitoring carried out with logistical support from McMurdo Station, located in close proximity to the colony, allowed comparison of the number of animals identified in satellite imagery with the numerical results of ground-based inspections. The authors confirmed the usefulness of the 60 cm resolution satellite imagery for counting Weddell seals on the ice, while noting the existing disadvantages of using satellite imagery for population surveys. One complication is that the diurnal interval during which an image is taken from a satellite moving in polar orbit is predefined and, in this case, it was beyond the time of day when most seals, according to the studied daytime routine of this species, are on the ice. Additionally, due to the optical nature of satellites, their usefulness depends on cloud cover and lighting. During the Antarctic winter, it is limited by the length of the day, which means that suitable light conditions do not always coincide with the time of their flight over the desired location. The case for identifying Weddell seals, with their distinctive dark colouration, lying against a white background of ice cover on satellite imagery appears to be less challenging than trying to identify seals lying against a less contrasting background of ice-free ground. McMahon et al. (2014) attempted to count female southern elephant seals in a harem, on the volcanic beach of subantarctic Macquarie Island, using 50 cm resolution imagery from the Geo-Eye-1 satellite (and also discussed the utility of 60 cm resolution imagery for the same purpose). Although southern elephant seals are the largest representative of pinnipeds in the Antarctic, the identification of females on satellite imagery generated some uncertainty for the analysers, as illustrated by the mean number of identified females in the study area, which was 1790 +/- 306 (n = 3 counts; 95% confidence interval). However, the authors conclude that this is a satisfactory result, making the method useful, comparing the result with the ground count performed on the same day, according to which 1991 adult females were present in the area. The subject of the use of satellites almost automatically brings to mind the largest of the mammals living on the planet. Fretwell et al. (2014) attempted to identify southern right whales (*Eubalaena australis*) on images from the World-View-2 satellite at 50 cm resolution with a penetrating band in the far blue part of the spectrum, which allows for deeper visibility in the water column. The authors emphasised that the attribution of a specific species was made possible by deduction and by ruling out the possibility of other species of similar size being present in the image from the selected location that had been previously analysed. The leap in image quality that occurred with the advent of the World-Wiew-3 satellite in 2016, offering a spatial resolution of 31 cm, already made it possible to capture the characteristics of a particular species and thus recognise the whale species in satellite images (Cubaynes et al., 2019).

Very high resolution satellite imagery is currently too expensive to become a mainstream tool for studying Antarctic populations. However, the future looks bright for this topic: with increasing competition between space companies offering satellite imagery, the price of images has the potential to become more affordable for scientists. In addition to the launched and expanding World-View constellation from Maxar (MAXAR, 2024), Airbus and Planet are also developing their Pléiades Neo (Airbus, 2024) and Pelican (Planet, 2024) constellations, all with a spatial resolution oscillating around 30 cm. The Clarity constellation from Albedo Space (Albedo Space, 2024), due to launch in 2025, aims to achieve a ground-breaking spatial resolution of 10 cm, potentially matching or even surpassing some aerial photographs. These achievements indicate that animal detection based on VHR satellite imagery has increasing potential.

1.6 Machine learning as a tool to study marine ecosystems

Machine learning (ML) has undergone a complex evolution from the initial theories of Alan Turing in the 1950s (Bowen, 2016). At the time of writing, two researchers: John Hopfield and Geoffrey Hinton, who spearheaded the current breakthrough development of ML, have been awarded the most prestigious prize a scientist can receive *'for foundational discoveries and inventions that enable machine learning with artificial neural networks'* (The Nobel Prize, 2024), demonstrating the importance of this scientific pathway. ML methods outperform other algorithms in terms of predictive power and inference (Elith et al., 2006), especially when agile decision making is required, as is the case in natural resource management. New algorithms and methods for analysing complex ecological data are being developed rapidly, at speeds that often exceed the capacity of ecologists to implement them on an ongoing basis (Humphries and Huettmann, 2018). The adoption of ML in the ecological community has been slow, despite the fact that the term 'machine learning' itself has become present in both the scientific literature and our everyday lives (Horvitz, 2007). Resistance to the fusion of ecology and ML may be due to a lack of communication between life scientists and the ML community, in addition to the potential discomfort of venturing into unfamiliar scientific territory (Thessen, 2016). Unlike traditional computer programs that code a solution designed by a programmer, an ML system can learn to solve a problem without providing an explicit recipe, the solution is deduced by the system by providing it with example data. The ability to produce a solution to a problem can be impressive, but depends primarily on the appropriate presentation of the problem, or 'goal', and the provision of appropriate data from which the 'learning' process can occur (Zhou, 2021). Rubbens et al. (2023) prepared a compilation of ~1000 (selected on the basis of analyses of the number of their citations and the clarity of the description of the ML methodology) scientific articles presenting in their content the practical application of ML in marine ecology. The wide range of topics covered include, among others, analyses of benthic, plankton and nekton data, search and classification of marine debris and pollution, identification of marine microorganisms, interpretation of acoustic measurements, profiling of biological communities using environmental genomics, quantification and mapping of fishing pressure from geolocalisation data, derivation of biogeochemical variables from satellite images and survey float profiles, or prediction of abundance and distribution of marine animal species.

In recent years, the detection and recognition of objects in a video or image has been an important task of Computer Vision (CV) (Diwan et al., 2023). Object detection using machine learning models refers to a set of algorithms that can automatically identify and locate objects in images or videos. These models use feature extraction, feature selection and classification techniques to recognise objects in visual data (Sirisha et al., 2023). To train these models, labelled images are provided in which each object of interest is assigned to a class. The model uses the tagged images to learn features specific to each class of objects.

Several machine learning models for object detection are available, including support vector models (Pontil and Verri, 1998), decision trees (Ali et al., 2012) and random forests (Liu and Xiong, 2012). These models differ in their approach to feature extraction and classification and may perform differently depending on the task and available data (Khan and Al-Habsi, 2020). Some of these models require manual feature engineering, while others can automatically learn features from the input data. Deep learning models refer to a class of neural networks that can automatically identify and locate objects in images or videos. These models use multiple layers of processing units to extract complex features from input data, making them effective in object detection tasks (Zhao et al., 2019). Some examples of models include CNN, R-CNN, SSD, ResNet and 'You Only Look Once' (YOLO) models (Xu et al., 2024), which can accurately recognise objects and detect multiple objects in a single image or video. According to a review paper by Xu et al. (2024), the number of publications on animal detection based on deep learning and remote sensing between 2017 and 2023, was 98, excluding review papers. The primary remote sensing platforms used for animal detection in this work were unmanned and manned aerial vehicles and satellites. YOLO has become one of the leading models for animal detection due to the fact that it operates with impressive accuracy and speed (Redmon, 2016). By dividing the entire image into a small grid and making predictions directly in each grid cell, YOLO has achieved convincing real-time processing capabilities, making it particularly advantageous for applications requiring fast and accurate object identification in changing environments (Zhao et al., 2019).

Although 'Artificial Intelligence' and 'Machine Learning' are sometimes treated as synonyms in the popular literature, ML is a sub-discipline of Artificial Intelligence (AI) that includes logic models, algorithms and computational methods capable of 'intelligent behaviour' (Jordan and Mitchell, 2015). Within ML, a separate subsection of 'deep learning' (DL) has developed over the past decade. DL systems use extensive neural networks to extract relevant features from raw data and learn from them, rather than requiring explicit engineering of these features (Monesinos López et al., 2022). This data is often complex (it includes, for example, images and sounds) and extensive (thousands to millions of records), and can also be Big Data. Big Data, as the name suggests, is defined by its size, encompassing a vast, complex and independent set of data sets, with the potential for multidirectional interaction between them. Furthermore, an important aspect of Big Data is that it cannot be handled by standard data management techniques due to the inconsistency and unpredictability of possible combinations (Sagiroglu and Sinanc, 2013). An excellent example would be Big Data satellite remote sensing collections. As Liu (2015) points out, they have several specific characteristics, apart from the obvious huge volume, that allow them to fall into this category i.e. they have multiple sources and scales, are multidimensional, dynamic, isomeric and non-linear.

2. Motivation and objective of the work

Proper management of Antarctic natural values requires continuous monitoring. Only long-term observations, translated into reliable scientific data and conclusions, can inform the arguments of the Antarctic Treaty System's scientific advisory body and have the potential to modulate the movements of policymakers who project the future fate of Antarctica. Environmental monitoring is sometimes treated as a subordinate category or one tool among many in the hands of a broadly defined 'science'. Not everyone agrees with this exclusion (e.g. Lovett et al., 2007). Carpenter (1998) suggests that ecosystem studies science is like a table supported by four legs: theory, experiments, comparisons and long-term research. One might therefore be tempted to hypothesise that 'the leg is an integral part of the table'.

The main objective of this thesis is to propose procedures for monitoring selected Antarctic animal species using remotely piloted aerial systems (RPAS), commonly known as drones, and high-resolution satellite imagery. The remote sensing methods used in this thesis can be disseminated to conduct inventories of Antarctic areas of natural value and special management needs.

Platt (1964), in his publication, openly criticised the untargeted collection of detailed data, which may be generally relevant to the study, but does not aim towards extracting hypotheses. In designing the research that forms the basis of this thesis, the guiding idea was the assumptions of directed monitoring, which is defined by its integration with conservation practice, design and implementation based on *a priori* hypotheses and related models of system response to management. Based on these assumptions, the **specific objectives of the work** were detailed:

1. Propose a procedure for monitoring population of the southern giant petrel (*Macronectes giganteus*) using RPAS.
2. Assessing the suitability of World-View-3 satellite imagery at 31 cm resolution for conducting population monitoring of southern elephant seal (*Mirounga leonina*).
3. Consider the potential of using RPAS to carry out inventories of Important Bird and Biodiversity Areas.
4. Application of the YOLO machine learning model for automatic detection, identification and counting of Antarctic shag (*Leucocarbo bransfieldensis*) nests on georeferenced aerial mosaics.

3. Outline of the work

This thesis provides a coherent whole for assessing the size of local populations of selected bird and pinniped species within Antarctic Specially Protected Areas (ASPAs), Antarctic Specially Managed Areas (ASMAs) and Important Bird and Biodiversity Areas (IBAs). Fieldwork to obtain data for this thesis was carried out on King George Island, part of the South Shetland archipelago in West Antarctica. The assessment of the size of the animal populations studied was based on the use of remote sensing imagery: images from remotely piloted aircraft systems (RPAS) and high-resolution satellite imagery, as well as the YOLO machine learning model. The results of the conducted research were presented in four scientific articles:

1. **Fudala K.**, Bialik R.J. (2022). The use of drone-based aerial photogrammetry in population monitoring of Southern Giant Petrels in ASMA 1, King George Island, maritime Antarctica. *Global Ecology and Conservation* 33, e01990, DOI: 10.1016/j.gecco.2021.e01990.

2. **Fudala K.**, Bialik R.J. (2022). Seals from outer space-Population census of southern elephant seals using VHR satellite imagery. *Remote Sensing Applications: Society and Environment* 28, 100836, DOI: 10.1016/j.rsase.2022.100836.

3. **Fudala K.**, Bialik R.J. (2023). Identifying important bird and biodiversity areas in Antarctica using RPAS surveys-A case study of Cape Melville, King George Island, Antarctica. *Drones* 7(8), 538, DOI: 10.3390/drones7080538.

4. Cusick A., **Fudala K.**, Storożenko P.P., Swieżewski J., Kaleta J., Oosthuizen W.Ch., Pfeifer C., Bialik R.J. (2024). Using machine learning to count Antarctic shag (*Leucocarbo*

bransfieldensis) nests on images captured by Remotely Piloted Aircraft Systems. *Ecological Informatics* 82, 102707, DOI: 10.1016/j.ecoinf.2024.102707.

As part of objective one (1), fieldwork was planned to be carried out over two consecutive summer seasons within King George Island Antarctic Specifically Managed Area No 1. During the first season, southern giant petrel breeding sites on the shores of Admiralty Bay were inventoried and RPAS flights were used to 3D model the two main areas occupied by the bird colonies during the pre-incubation phase. These were used to produce hypsometric models of the sites. In the following season, a series of designed drone missions were carried out, at different phases of the breeding cycle, to determine the optimum phenological stage for performing petrel censuses. In addition, an experiment was carried out to study the effect of the presence of the drone on the behavioural responses of petrels and to determine the optimal flight altitude of the drone to identify the nests and chicks of this species, with the least possible interference with the animals' behaviour. The results of the study are presented in publication 1.

To meet objective two (2) of the work, the results of the southern elephant seal counts were compared using the 31 cm per pixel resolution World-View-3 satellite image and the 1.39 cm per pixel resolution orthomosaic obtained from the drone mission. The feasibility of using high-resolution satellite imagery was tested: (a) to determine the number of elephant seal harems; (b) to determine the age and sex structure of the aggregation analysed: adults (females, males) and pups; (c) to indicate the total number of elephant seals in the area selected for comparison. Attention was also drawn to the problem of ambiguity in image interpretation when identifying objects for training machine learning models from satellite imagery. The results of the study are presented in publication 2.

The findings presented in publication 3 were in response to the stated objective three (3) and the results of an inventory of the Cape Melville area and the offshore islands and rocks of Destruction Bay in the eastern areas of King George Island using a BSP.

As part of the fourth objective (4), cooperation was established with a team of specialists of ML and two scientific teams performing population inventories of Antarctic shags using RPAS. Manual analysis of aerial orthomosaics collected with different types of drones from 3 independent scientific teams and 11 Antarctic shag colonies at different locations on King George and Nelson Islands. The YOLO model was trained on the prepared dataset. Conditions for optimal model performance were discussed, as well as its limitations. Analyses of intra- and inter-seasonal changes in shag abundance between 2019 and 2022 at Turret Point and Shag Rock colonies were conducted. The results are presented in publication 4.

4. Results and conclusions

4.1 Use of RPAS-based photogrammetry to monitor southern giant petrel populations within ASMA 1 [based on Fudala and Bialik, 2022a - Appendix 1].

In paper 1 (Fudala and Bialik, 2022a), the flight parameters optimal for identifying adults and chicks of southern giant petrels on orthomosaics obtained from a drone mission were determined. On the basis of the experiment, the minimum flight altitude at which no statistically significant behavioural response was recorded in petrels was determined. A suggested GSD for missions using a DJI Inspire 2 drone with a Zenmuse X5S camera and 15 mm lens was provided, allowing for easy identification of both age groups of petrels (GSD 2.15 cm at a flight height above the birds of 112-121 m). In pursuit of the first objective of the thesis, a complete methodology for inventorying giant petrel colonies using RPAS was presented. Based on the drone surveys, the locations of all giant petrel breeding sites within ASMA 1 were determined and the presence of 3 colonies was confirmed: Llano Point/Rescuers Hills, Vaureal and Petrel Hill. The inventory provided an estimate of the distribution and number of active nests and chicks across ASMA No. 1 for the 2020/21 season (508 active nests and 380 chicks).

Southern giant petrels are exceptionally sensitive to human presence in the vicinity of their nests. Numerous reports in the literature have highlighted the high environmental costs of surveying this species (Warham, 1962; Conroy, 1972; Jablonski, 1986), yet population counts of these birds on ASPA No. 128 were conducted using ground-based mapping until the 2019/20 season, requiring observers to enter the breeding colony. From the 2020/21 season onwards, monitoring of this species is carried out within the wider ASMA No. 1 area context and is carried out solely with the use of RPAS, using the methodology described in the publication 1.

4.2 Using very high resolution satellite imagery to assess the size of the southern elephant seal population within ASPA 128 [based on: Fudala and Bialik 2022b-Appendix 2]

Article 2 (Fudala and Bialik, 2022b) assessed the recognition performance of southern elephant seals on a 31 cm resolution World-View-3 satellite image. The satellite image of the breeding colony at Cape Patelnia allowed the recognition of 19 males, 403 females and 246 pups (median of counts by 11 observers). The results were compared with the number of animals detected on an aerial orthomosaic taken by a drone 15 hours earlier, which were 21, 428 and 386 respectively. An analogous area of the colony, corresponding to the range of the drone mission, was compared. The presence of an additional harem outside the range of the drone's flight path, identified by satellite imagery, changed the total number of elephant seals (observed at female haul-out peak during the 2019/20 season) in the Cape Patelnia area by approximately 13 females, 8 pups and 2 males (median of counts by 11 observers). In response to the stated second objective of the thesis, the relative error of satellite imagery counts within each sex and age group of southern elephant seals in subzones within the colony is reported, and potential causes of false positive and negative observer predictions are discussed. The relative error between the median results obtained from counting elephant seal females based on satellite imagery and those based on drone counts ranged from 1.5% to 7.0%, depending on the colony subzone. The identification efficiency of pups was strongly dependent on the level of substrate contrast with their dark colouration, with a relative error of 24.6-40.2%. The study found that many of the issues highlighted by McMahon et al. (2014), who attempted to assess the size of southern elephant seal aggregations using 50 cm resolution satellite imagery, can be addressed using 31 cm resolution imagery. The utility of satellite imagery in identifying southern elephant seal colonies is well established. Furthermore, it is possible to estimate the number of adult seals of both sexes in aggregations using this method, although it should be noted that there is some uncertainty in the result. However, the identification of pups is subject to too much error to recommend this method for estimating their abundance within a breeding aggregation.

4.3 Identification of Important Bird and Biodiversity Areas in Antarctica using RPAS: the example of Cape Melville, King George Island [based on Fudala and Bialik, 2023- Appendix 3].

The third article (Fudala and Bialik, 2023) presents the results of an ornithological inventory of the Cape Melville and Destruction Bay coastal area using RPAS on two dates during the 2022/23 season. High-resolution orthomosaics were created from the drone images and bird nests were marked on them (by 3 independent observers). On 14.12.2022, 4960 (+/-12) nests of chinstrap penguin and 456 nests of Antarctic shag were present in the Cape Melville area. Analyses also confirmed the presence of at least one nest of kelp gull (*Larus dominicanus*), more than 50 foraging individuals of snowy sheathbill (*Chionis albus*) and at least one pair of brown skua (*Stercorarius antarcticus*) in the colony area. The steep slopes and rock slides present on the Cape also suggest that the location may be a potential nesting site for Wilson's (*Oceanites oceanicus*) and black-bellied storm petrels (*Fregetta tropica*). As these are not surface nesting species, their detection on aerial imagery was not possible. Inspection within the Cape-adjacent section of Destruction Bay also revealed the presence of breeding bird aggregations on offshore islands: 3435 (+/- 16) chinstrap penguin nests on Trowbridge Island, 7 Antarctic shag nests on Trowbridge annex rock, 627 (+/-6) chinstrap penguin nests and 10 shag nests on Middle Island, and 32 shag nests on a small unnamed island located between Trowbridge Island and Cape Melville.

Estimates of the size of the global population of the Antarctic shag prior to the publication of the presented article 3, were 11 684 pairs (based on Schrimpf et al. (2018) after corrections to account for the results of Phillips et al. (2019)). However, they did not include the Cape Melville colony in the calculations, bringing the estimated number of pairs of this species to 12 191. Based on the postulate of the presence of 458 Antarctic shag nests documented on the orthomosaic taken on 4.12.2022, in relation to criterion B3a, it was proposed that Cape Melville be restored as an IBA, with a proposed route for its new boundaries. The application submitted to Birdlife International has been approved (Birdlife International, 2024c). As previously stated in the Introduction, IBAs do not directly translate into specific conservation measures for an area. However, active IBA status can be a compelling argument for including a particular section of Antarctica in the Antarctic Specially Protected Areas network.

The inventory of Cape Melville and an adjacent section of Destruction Bay undertaken as part of this thesis was the first update of data on a surveyed area since 1987 (Shuford and Spear, 1988). The potential of using RPAS to undertake IBA inventories was confirmed in practice, thus fulfilling the third objective of the thesis.

4.4 Using machine learning to automatically detect and count Antarctic shag nests on aerial orthomosaics [based on Cusick & Fudala et al., 2024- Appendix 4].

The fourth article (Cusick & Fudala et al., 2024), written in response to the fourth objective of the thesis, proposes a method for the automatic detection and counting of Antarctic shag nests using a model based on the 'You Only Look Once' (YOLO) architecture. The results of the trained model suggest that the use of RPAS for Antarctic shag colony inventories, in combination with machine learning algorithms, can provide reliable and rapid estimates of shag nest numbers (automatic detection score of F1 > 0.95). The code, data and trained model, allowing full reproducibility of the results, have been made publicly available (https://github.com/Appsilon/Antarctic-nests). At the same time, the results of the drone monitoring of the Antarctic shag in ASMA 1 are presented in the supplement to article 4.

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Oświadczenia o wkładzie autorskim /

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22 October 2024

Statement of co-authorship

I confirm that I am co-author of the following article:

Fudala K., Bialik R.J. (2022). The use of drone-based aerial photogrammetry in population monitoring of Southern Giant Petrels in ASMA 1, King George Island, maritime Antarctica. *Global Ecology and Conservation* 33, e01990, DOI: 10.1016/j.gecco.2021.e01990.

My contribution to the article was as follows: I was involved in the conceptualisation of the ideas, the design of the methodology, the collection and analysis of the data, and the writing of the manuscript.

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My contribution to the article was multifaceted and encompassed a range of roles and responsibilities. These included involvement in the methodology, software, validation, formal analysis, investigation, resources, data curation, visualization, project administration, conceptualization of the research idea, and writing the original text of the article.

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Cusick A., Fudala K., Storożenko P.P., Świeżewski J., Kaleta J., Oosthuizen W. Ch., Pfeifer C., Bialik R.J. (2024). Using machine learning to count Antarctic shag (Leucocarbo bransfieldensis) nests on images captured by Remotely Piloted Aircraft Systems. *Ecological Informatics* 82, 102707, DOI: 10.1016/j.ecoinf.2024.102707.

My contribution to the article was involvement in the validation, software, methodology, investigation, formal analysis, data curation, as well as editing of the final text of the article. In particular, I was involved in the development of the machine learning model and the analysis of the results.

Andrew Cusick

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22 October 2024

Statement of co-authorship

I confirm that I am co-author of the following article:

Cusick A., **Fudala K.**, Storożenko P.P., Świeżewski J., Kaleta J., Oosthuizen W. Ch., Pfeifer C., Bialik R.J. (2024). Using machine learning to count Antarctic shag (Leucocarbo bransfieldensis) nests on images captured by Remotely Piloted Aircraft Systems. *Ecological Informatics* 82, 102707, DOI: 10.1016/j.ecoinf.2024.102707.

My contribution to the article was multifaceted, encompassing involvement in the visualization, validation, resources, investigation, formal analysis, conceptualization of the research idea, and original text writing. In particular, I was responsible for the collection of data and the preparation of annotations.

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My contribution to the article was the provision of aerial images of the shag colony on Nelson Island as well as editing of the final text of the article.

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My contribution to the article was the provision of aerial images of the shag colonies on Fregata Island; Kwarecki Island; Rzepecki Island and on Unnamed Island C as well as editing of the final text of the article.

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My contribution to the article was multifaceted, encompassing involvement in the supervision, project administration, investigation, funding acquisition, formal analysis, conceptualization of the research idea, and writing the original text of the article. In particular, I was responsible for the collection of data and the preparation of annotations.

Robert Bialik

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Załącznik 1 / Appendix 1

Fudala K., Bialik R.J. (2022). The use of drone-based aerial photogrammetry in population monitoring of Southern Giant Petrels in ASMA 1, King George Island, maritime Antarctica. *Global Ecology and Conservation* 33, e01990, DOI: 10.1016/j.gecco.2021.e01990.

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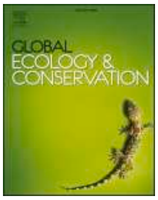

The use of drone-based aerial photogrammetry in population monitoring of Southern Giant Petrels in ASMA 1, King George Island, maritime Antarctica

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ABSTRACT

Southern Giant Petrels (SGPs) are surface nesting birds with a circumpolar Southern Hemisphere breeding distribution. The species tends to have no natural enemies on land, but is sensitive to human disturbance. The search for new methods is crucial and may minimize or exclude stress and risk of nest disturbance, related to ground-based research activities. The aim of this study was to conduct a population census of the SGP of the Antarctic Specially Managed Area no. 1 (ASMA no. 1), Admiralty Bay, King George Island, using an unoccupied aerial system (UAS) based on aerial photogrammetry and to determine the optimal parameters of the aerial mission for the identification of SGP adults and chicks on orthophotos while simultaneously not causing behavioural changes. To this end, in a preliminary survey in the 2019/20 season, the locations of all breeding areas for SGPs in ASMA no. 1 were determined, and the presence of 3 colonies, Llano Point/Rescuers Hills (LP/RH), Vaureal (V) and Petrel Hill (PH), was confirmed. Terrain models for two of the colonies (LP/RH and V) were established, and the flight parameters for the next season were determined. In 2020/21, a total of 23 (DJI Inspire 2 with a Zenmuse X5S camera) drone missions were performed at various stages of the breeding period over the LP/RH and V colonies. This assessment yielded estimation of the number of active nests and chicks over the entire ASMA no. 1 area, and included 508 active nests and 380 chicks for the 2020/21 season. To determine the minimum flight altitude at which no SGP behavioural response was observed, an experiment was performed that showed the vertical distance between the potential nest of SGP and the drone should be greater than 21 m given that lowering the altitude yielded statistically significant differences in bird behaviour. Image analyses showed the possibility of identifying adults and chicks at a ground sampling distance of 2.15 cm, which corresponded to an altitude of 130 m based on the equipment used and the terrain characteristics. The proposed method requires several missions during the incubation phase to determine a reliable number of active nests without using correction factors. To obtain a nesting success factor, it is recommended to perform at least one raid in the post-brooding phase of chick rearing (when the chick is not covered by an adult and is visible in orthophotos). The proposed method is not able to replace traditional methods in the context of many ongoing surveys, but we believe that it may provide a less bird-invasive, human-intensive and time-consuming option to replace ground-based census surveys.

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1. Introduction

The Southern Giant Petrel *Macronectes giganteus* (SGP) is considered a species of 'least concern' according to the International Union for Conservation of Nature (IUCN) guidelines (BirdLife International, 2021). The SGP is recognized as the dominant scavenging seabird species in maritime Antarctic (Conroy, 1972). Warham (1962) characterized the species as a 'surface nester, too large for natural enemies to attack it when ashore'; however, despite the lack of natural enemies, SGP is a bird that tends to be susceptible to human disturbance (Agreement on the Conservation of Albatrosses and Petrels, 2010). Moreover, eggs and chicks may fall prey to skuas, sheathbills and other giant petrels (Conroy, 1972). The phenology and biology of this species was thoroughly studied decades ago (Warham, 1962; Conroy, 1972; Jabłoński, 1986; Voisin, 1988), however, the authors of those studies emphasised the extremely high environmental cost of such projects.

According to Conroy's (1972) report on the SGP colony from Signy Island, human disturbance caused very high chick mortality (90% in 1967, 62% in 1968, 67% in 1969) and interference was so great that data collected from the intensive study areas were not used in survival analyses. Intensive scientific activity, associated with banding and morphometric measurements, was also responsible for egg losses. Conroy (1972) reported that adult birds released after ringing did not return to the nest immediately, resulting in a high percentage of egg losses caused by skuas. In 1966, 30 of 41 eggs abandoned were hunted by predators. Treatments applied the following year to cover the abandoned eggs with fragments of nesting material reduced the percentage of egg loss. Specifically, of 183 eggs, 81 were hunted by predators (Conroy, 1972). Jabłoński (1986) also admitted that in the 1978 season, successful breeding was observed at only 50 of 102 active nests at the Rescuers Hills/Llano Point colony on King George Island, and they attributed the high failure rate to scientific activities that caused birds to escape from incubated nests. Warham (1962) whose investigations occurred on Macquarie Island in seasons from 1959 to 61, reported that many incubating SGPs deserted eggs after human approach and emphasized the high susceptibility of SGPs to being frightened by humans. Therefore, Warham recommends hiding while observing to obtain behavioural data. Bird species of the order *Procellariiformes*, including SGPs, are capable of producing stomach oils with high energy content in proventriculus (Foster et al., 2020) used by birds as a food source for chicks and an energy reservoir enabling them to exploit marine nutrients sources and survive harsh weather conditions (Warham, 1962). When threatened, stomach contents can be ejected by birds, including both adults and chicks, as a defensive measure, resulting in a significant loss to the birds as they lose a source of energy (Warham, 1977). Some works have examined the effects of human disturbances on heart rate and the associated metabolism in incubating albatrosses and petrels. Weimerskirch et al. (2002) reported that the heart rate of wandering albatrosses doubles when human presence is first detected. de Villiers et al. (2006) used heart rate to measure the response of northern giant petrels to human approach and subsequent nest manipulation. The birds' heart rates increased upon detection of a human at a distance of 40 m and continued to increase with the gradual approach of man.

Significant advances in technology in recent years have brought opportunities to minimize or completely reduce the invasiveness of research methods. For example, automatic ground cameras mounted at SGP colonies have been successfully used to monitor breeding phenology (Otovic et al., 2018). In addition, unoccupied aircraft systems (UASs, also called drones) are increasingly being used to monitor bird populations (Lyons et al., 2018; Edney and Wood, 2021; Gallego and Sarasola, 2021), proving that this method is more effective and has a less negative impact on some bird species than do direct observations (Borrelle and Fletcher, 2017; Valle and Scarton, 2020; Krause et al., 2021); however, there are still no legally regulated standards for drone use (Vas et al., 2015; Duffy et al., 2018; Barnas et al., 2020) or flight parameters, so the discussion on this topic at the scientific level is still open (Rümmler et al., 2016; Hodgson and Koh, 2016; Barr et al., 2020). The use of UASs is also considered an option for SGP monitoring (Mustafa et al., 2018; Weimerskirch et al., 2018; Dunn et al., 2021). However, as observed by Weimerskirch et al. (2018), northern giant petrels showed the strongest behavioural response to the presence of the drone. Although the SGPs react less nervously than northern giant petrels, imperial cormorant or brown skua, SGPs were still identified as one of the species most susceptible to stress among the eleven sub-Antarctic seabirds tested in the study. It should be noted, however, that Weimerskirch at al. (2018) approached birds at vertical distances of 3, 10, and 25 m which are short distances. Despite the close distance, sooty albatross showed minimal behavioural responses even when the drone was at the distance of 3 m. Similar experiments with close (4 m) ranges did not seem to disturb birds, such as wild flamingos and greenshanks, as reported by Vas et al. (2015). These researchers did not observe visibly modified behaviour due to the presence of drones and suggested that when used with care, drones can be employed in ornithology for a wide range of observations.

It is extremely challenging to investigate a species as sensitive to human presence as SGPs are, and therefore, there are still gaps in knowledge about the at-sea distribution and survival rates of chicks and adults (Agreement on the Conservation of Albatrosses and Petrels, 2010). Due to the factors complicating the performance of censuses (e.g., high sensitivity of the species or logistical difficulties resulting from conducting surveys under Antarctic or sub-Antarctic conditions), the census procedure has not been standardized (Patterson et al., 2008; Agreement on the Conservation of Albatrosses and Petrels, 2010). The global population size of SGPs has been estimated (based on data from 2005 to 2007) by Poncet et al. (2020) as 50,819 pairs, and BirdLife International (2021) reported the population size of SGPs to be 95,600–108,000 mature individuals. According to Patterson et al. (2008), the South Shetland population is 5409 breeding pairs. For Admiralty Bay, the most recent data were reported by Petry et al. (2016). However, these researchers did not include the areas of the Petrel Hill, Rescuers Hills and Llano Point colonies in the censuses they conducted, and only report data for Vaureal colony, which included 60 breeding pairs in the 2011/2012 season based on their observations. The most recent complete data for Admiralty Bay on the SGP breeding population size is from 1996 (Sierakowski et al., 2017). The number of nests reported for the area of Admiralty Bay varied from 243 to 456 between 1979 and 1996 (Sierakowski et al., 2017), but the current status of the population is unknown.

The main objective of this research was to propose a procedure for monitoring the SGP population using UASs. To establish the optimal timing for performing censuses using this method, a series of photogrammetric missions were performed over the colony area at different stages of bird breeding. The proposed method was used to improve our knowledge about the size and distribution of the breeding population of SGPs in Admiralty Bay. An additional aim of the study was to perform an experiment to investigate the effect of drone presence on SGP responses and to determine the flight altitudes for the drone. In this context, the drones did not disturb animals but still allowed for the easy identification of adults and SGP chicks from orthophoto imagery. In addition, the height from which we could easily identify bird behaviour was determined.

2. Materials and methods

2.1. Study sites

The study area included the entire ice-free shoreline of Admiralty Bay (62°10′S 58°25′W), the largest bay within King George Island, part of the South Shetland archipelago in maritime Antarctica (Fig. 1). Due to its unique environmental, historical, scientific and aesthetic value, the entirety of Admiralty Bay has been established as Antarctic Specially Managed Area no. 1 (ASMA no. 1). The revised total area of both terrestrial and marine areas of ASMA no. 1 is 360 km^2 (Management Plan for Antarctic Specially Managed Area No.1, 2014). Additionally, the region includes (wholly terrestrial) Antarctic Specially Protected Area “Western Shore of Admiralty Bay” (ASPA no. 128) (Management Plan for Antarctic Specially Protected Area No. 128, 2019) and two Important Bird Areas (IBAs no. 045 and 046) (Harris et al., 2015; Fig. 1).

2.2. Field work

The field work was divided into two seasons: 2019/2020 and 2020/2021.

First, during the 2019/2020 season, the locations of SGP breeding colonies on ASMA no. 1 were verified. All potential (both known from the literature and those showing potential in terms of site conditions) localities of SGP colonies were inspected during the pre-laying period by an observer with ground searching methods and from the sea surface with the use of boat and optical equipment. During the 2019/2020 breeding season, SGP breeding colonies were found at three major locations in ASMA no. 1: Rescuers Hills and Llano Point (treated as a single colony, Fig. 1), Cape Vaureal, and Petrel Hill (Fig. 1). Previous literature sources (Jabłoński, 1986; Sierakowski, 1991; Sierakowski et al., 2017) also listed three locations that overlapped with those found in the current study (Fig. 2). The nomenclature that is based on colloquial place names can be confusing to the uninitiated reader; therefore, the colony located at Petrel Hill refers to the part of the previous larger colony located at Point Thomas oasis, which extends from Ecology Glacier to Thomas Point located south of the entrance to Ezcurra Inlet in Admiralty Bay (Fig. 1). The whole Point Thomas colony, which in Trivelpiece et al. (1980) was called Point Thomas West colony in 1977/1978, had 40 nesting pairs (Trivelpiece et al., 1980), including 19 on Petrel Hill (Jabłoński, 1986); however, in 1988, there were only 18 nesting pairs in the whole area, including 17 on Petrel Hill, of which only

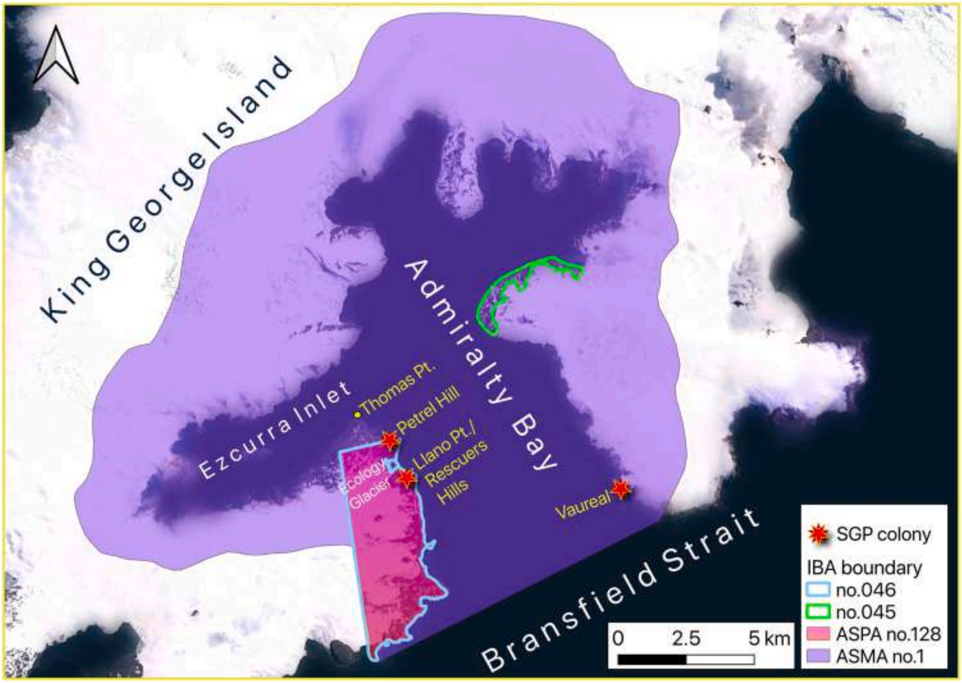

Fig. 1. Study site.

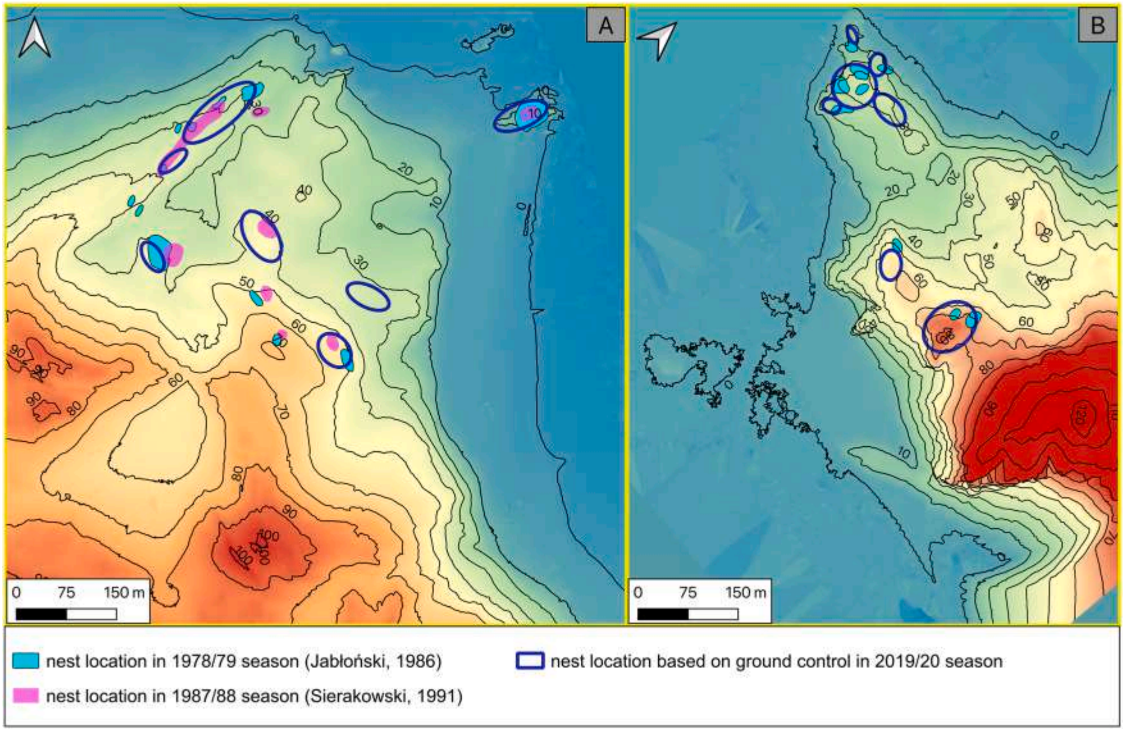

Fig. 2. Hypsometric maps (based on 3D drone missions) of Rescuers Hills/Llano Point colony (A) and Cape Vaureal colony (B) with historical data on the locations of SGP breeding groups.

6 pairs bred successfully (Sierakowski, 1991). In subsequent years, breeding attempts within the Point Thomas colony were made only on Petrel Hill, with no breeding success observed at this colony in the 2018/2019 and 2019/2020 seasons.

At the other two identified locations (Vaureal and Llano Point/Rescuers Hills), in the 2019/2020 season, 3D UAS missions were conducted from a height of 200 m (which was identified by Harris et al., 2019 as the minimal flight distance with no proven disturbance by UAS to SGP) to investigate site conditions and determine the optimal photogrammetry mission heights for the next season. The analysed area was characterized by a variety of elevations represented by glacial moraines and rock formations (Fig. 2). To determine the safe height of the 3D flight while also considering the terrain, the 1:100 000 topographic map of King George Island (Braun et al., 2004) was used. Breeding groups within a single colony were located at varying heights (altitude of the nests above sea level), including from 23 m to 52 m for Rescuers Hills (Fig. 2A), from 9 m to 18 m for Llano Point (Fig. 2A) and from 10 m to 85 m for Cape Vaureal, with two nesting groups clearly identifiable for the situated colony: the first (located N-W, Fig. 2B) with nests at heights from 10 m to 35 m and the second (located S-E, Fig. 2B) with nests at heights from 52 m to 85 m. This difference made it necessary to determine the optimum height at which to carry out the raids while considering the diversity of the terrain, animal safety and optimum ground sampling distance (GSD) resolution.

In the 2020/2021 season, 23 photogrammetric missions were performed over two colonies (19 for Llano Point/Rescuers Hills and 4 for Cape Vaureal), and the flights in all colonies covered the entire breeding cycle of SGPs: pre-laying, incubating, brooding, guarding and post-guarding. According to Otovic et al. (2018), the pre-laying period is the period of pair formation, the incubating period is the period from egg laying to hatching; the brooding period is the period from hatching to the moment when the chick attains homeothermy; the guarding period is the period when at least one parent is visible next to the chick at all times; the post-guarding period is the period when the parent returns to the chick only to feed it. Sierakowski et al. (2017) reported that for the SGP colonies located in Admiralty Bay, the first egg laid was noted 31 October and 10 November, and the first chicks hatched from 31 December to 13 January based on the data from the late 1980s and early 1990s

Moreover, the UAS mission heights were adjusted to the nesting heights, which were verified by 3D missions performed in the previous season, maintaining at least a vertical distance of 50 m between the potential nest of any breeding birds in these areas and the drone (except for raids performed as part of an experiment). The operator responsible for supervising the take-off and landing of the UAS was located in a place not visible to the animals at a horizontal distance of at least 100 m from the breeding group, as suggested by Vas et al. (2015) and Weimerskirch et al. (2018). The operator started and ended the UAS flights at the same point, which was designated separately for the Llano Point/Rescuers Hills and Vaureal colonies. In our planned study, the UAS launch and landing sites were always out of sight of the SGPs.

All UAS missions were conducted using a DJI Inspire 2 drone quadcopter (black/grey body and 4 kg weight) with a Zenmuse X5S 20.8 MP camera (DJI MFT 15 mm/1.7 ASPH lens with a 30-mm-equivalent focal length; DJI, Shenzhen, Guangdong, China). The mission paths (Fig. 3) were programmed in Pix4D Capture (Pix4D S.A., Prilly, Switzerland). The drone automatically started from, returned to and landed at its home position (Fig. 3 - DRONE LAUNCH). After take-off, the drone flew at a vertical speed of 4 m/s to the flight altitude (Table 1) and then flew at a fixed altitude after take-off to the mission start point (Fig. 3 - MISSION START). The overlap of the images (Fig. 3, Table 1) was adjusted to match the length of the mission with the length of the weather window, so the number of photos taken varied (Table 1). The operator maintained eye contact with the drone at all times and took control of the drone if there was a need to land manually by changing flight modes, which also affected the number of photos taken. Of note, although the same mission was repeated many times, the area of the mission itself changed (Table 1), due to the different mission overlaps as well as other factors, such as seawater tidal height. Thus, the number of common points on the land in the images varied. This process allowed the images to be calibrated, thus, fewer images were rejected (Table 1). Although different types of UASs have been used for environmental monitoring in Antarctica, quadcopters are known to be the “least risky” to birds compared to fixed-wing devices (Egan et al., 2020). Moreover, we decided to use the DJI Inspire 2 equipped with Zenmuse X5S Gimbal Camera with 15-mm lens. The significant advantage of this model is that it allows the creation of orthophotos with higher accuracy compared with the other UASs that have already been used in Antarctica. Orthophoto accuracy and detail depend mainly on the field pixel size and resolution, which is called the ground sampling distance (GSD, Table 1). A GSD on the order of 1 cm/px is characteristic of the most accurate orthophotos, allowing for recognition of SGPs at the level of 100% (Mustafa et al., 2019). For comparison, the GSDs for images taken from a 100 m height using different DJI quadcopters with standard cameras were as follows: 4.38 cm (Phantom 3), 2.73 cm (Phantom 4 Pro), 2.34 cm (Mavic 2 Pro), and 2.21 cm (Inspire 2). The dates and technical details of the missions are shown in Table 1.

In the 2020/2021 season, we strictly followed the drone protocol proposed by Barnas et al. (2020). The UAS operators were trained before the start of the study and obtained a UAS operator qualification certificate (UAVO) issued by the Civil Aviation Authority of Poland. The operators were qualified to fly drones weighing up to 25 kg up to 2 km, which is beyond the visual line of sight (BVLOS). All experiments were approved by the Polish Antarctic Programme and performed under permit nos. 2/2019 and 2/2020 given for the period from 25 August 2019–26 February 2022.

2.3. *Experiment of behavioural response to UAS presence*

An experiment was planned for the 2020/2021 breeding season, when the chicks had already reached homeothermia (Otovic et al., 2018) and were in the guarding stage, when at least one parent was still present, usually next to the nest (Otovic et al., 2018), to demonstrate the impact of drone presence at different altitudes above adults and chicks. The experiment was performed on 20 February 2021. The five missions were performed in order of decreasing altitude above the breeding formation starting from 130 m to 30 m over a selected breeding group located on the Llano Point rock formation (Table 1). The duration of all raids was 75 min (the first photo was taken at 12:38, and the last photo was taken at 13:53), taking into account the time required for landing for battery replacement. Each mission started and ended at the same point. The determination of the height of the area occupied for nesting and the height of the planned launch site was based on the previously mentioned 3D terrain model from the previous season. The ground observer, located on a hill adjacent to the study site (with a distance of approximately 250 m from the Llano Point sub-colony), used optical equipment to observe the exchange of individuals, i.e., birds flying away and arrivals of new individuals from the area of the rock, as well as the reaction of birds in the field of view of the telescope.

2.4. *Data processing procedure*

Based on a series of images from the photogrammetric mission, orthophotos were created in Pix4Dmapper (Pix4D S.A., Prilly, Switzerland). The resulting orthophotos were georeferenced (QGIS 3.16.5 ’Hannover’) based on terrain feature points to facilitate comparison of the nests visible on the different maps (Turner et al., 2013). The QGIS Georeferencer plugin was used to perform the transformation, and the Helmert transform was applied to perform simple scaling and rotation transformations. A grid (as a vector layer) delineating transects of 15 m × 15 m was created to facilitate the work of inspecting the site for the presence of nests and the control of nests at subsequent dates.

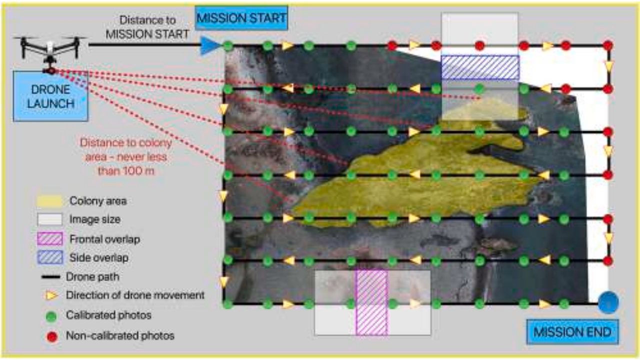

Fig. 3. Conceptual drawing of the drone mission over Llano Point colony.

Table 1
Unoccupied aircraft system (UAS) flight specifications.

Mission Date	Area Coverage	Number of Images Taken (Calibrated)	Flight Altitude	Image Overlap	Ground Sampling Distance (GSD)—Pixel Resolution
Llano Point/Rescuers Hills					
22 Nov 2019 – 3D	1.042 km^2	628 (625)	200 m	75–75%	3.75 cm
14 Oct 2020	0.553 km2	585 (539)	130 m	80–75%	2.13 cm
16 Nov 2020	0.476 km^2	485 (472)	130 m	80–75%	2.24 cm
04 Dec 2020	0.544 km^2	538 (535)	130 m	70–75%	2.28 cm
23 Dec 2020	0.565 km^2	430 (424)	130 m	80–70%	2.30 cm
17 Jan 2021	0.560 km^2	616 (609)	130 m	80–75%	2.23 cm
08 Feb 2021	0.555 km^2	493 (493)	130 m	80–70%	2.39 cm
[a]20 Feb 2021	0.563 km^2	491 (487)	130 m	80–70%	2.15 cm
15 Mar 2021	0.524 km^2	494 (491)	130 m	80–70%	2.30 cm
07 Apr 2021	0.586 km^2	524 (524)	130 m	80–70%	2.41 cm
Llano Point					
14 Oct 2020	0.032 km^2	80 (76)	70 m	78–70%	1.43 cm
16 Nov 2020	0.022 km^2	65 (59)	70 m	78–70%	1.47 cm
04 Dec 2020	0.030 km^2	64 (64)	70 m	78–70%	1.49 cm
23 Dec 2020	0.023 km^2	57 (57)	70 m	78–70%	1.51 cm
17 Jan 2021	0.024 km^2	66 (66)	70 m	78–70%	1.43 cm
08 Feb 2021	0.031 km^2	84 (84)	70 m	78–70%	1.59 cm
[a]20 Feb 2021	0.021 km^2	71 (69)	70 m	78–70%	1.43 cm
Llano Point (experiment)					
20 Feb 2021	0.011 km^2	311 (234)	30 m	80–70%	0.63 cm
20 Feb 2021	0.009 km^2	104 (79)	50 m	80–70%	1.04 cm
20 Feb 2021	0.014 km^2	40 (31)	100 m	80–70%	2.09 cm
Vaureal					
26 Oct 2019 – 3D	1.745 km^2	1553 (1108)	200 m	80–80%	5.56 cm
[b]26 Dec 2020	0.168 km^2	240 (239)	100 m	80–70%	1.98 cm
28 Jan 2021	0.624 km^2	471 (276)	200 m	80–70%	3.96 cm
04 Feb 2021	0.842 km^2	476 (315)	200 m	80–70%	4.08 cm
[b]21 Feb 2021	0.174 km^2	258 (254)	100 m	80–70%	1.98 cm

[a] Data used for the experiment;
[b] Mission done only for the N-W colony.

With an image overlap of 75–80% (Table 1), it is possible to obtain an orthophoto, with no perspective distortion, and the analysed objects (in our case, SGPs) are orthogonally projected onto the base. SGP adults and chicks (from the moment they were no longer covered by an adult) were marked (identified) on each map. This step was performed by both authors twice for each map to minimize the risk of missed bird identifications. The imposed search procedure included searching the image in the order of transects to exclude the possibility of confusion and omission of a given part of the area from the search. In the next step, sites occupied by adult birds at particular time intervals (in more than one chronologically related ortophoto) were selected, thus eliminating one-time findings of an adult at a given location and ruling out the likelihood of a nest there.

2.5. *Data processing of bird response analyses at different photogrammetric raid heights*

The 100 m altitude map was excluded from the analyses due to its insufficient quality resulting from the brief snowfall that occurred during the 100 m altitude raid. Due to the presence of visible precipitation in the images, we were unable to determine the birds' responses.

The following points describe our methodology:

1. At the beginning, the remaining maps (130, 70, 50, 30 m) were georeferenced (analogous to the maps of the whole area);
2. Analyses were conducted by two people independently;
3. In the order of the missions: from the highest to the lowest altitude and again from the lowest to the highest altitude to double check the data;
4. On each map, the location of adults and nestlings was marked, and the type of reaction of the animal between two consecutive images was distinguished according to the following criteria:
   a) Neutral responses were listed as follows:
      - in both images the animal remained in the "relaxation position" with the head resting on the trunk, position indicating sleep or rest (Fig. 4A (chick) and 4B (adult));
      - in both images, the animal remained in the sitting position with neck straight (Fig. 4C (chick) and 4D (adult)) but did not change its location (slight rotation of body position was acceptable);
      - in the first image, the animal was sitting, and on the second image, it showed a position typical for the "relaxation position".
   b) Potentially indicative responses were listed as follows:

Fig. 4. Examples of positions of adult birds and chicks recognized in the pictures: relaxation position -head resting on the trunk (A-chick; B-adult); sitting position- neck straight (C-chick; D-adult).

- in the first image, the animal stayed in the "relaxation position", and in the second image, it was sitting;
- the animal had moved:
  - (i) the animal had joined the colony in relation to the previous image;
  - (ii) the animal flew away in relation to the previous image;
- the animal was looking up showing interest in the drone.

2.6. Statistical analysis of data and control data

The observation data were divided into two groups: (1) a change in the location of an individual according to a change in drone flight altitude and (2) five different behavioural responses of individuals to a change in drone flight altitude. In both cases, we assigned ranks to a given bird behaviour. In case (1), a score of 0 meant that the individual did not change position, while a score of 1 meant that the individual changed position. In contrast, in case (2): a score of 0 meant that no change in the behaviour of the individual was noticed or the individual started to rest between two missions at two different altitudes; a score of 1 meant that the individual interrupted rest; a score of 2 meant that the individual flew (jointed) to the colony; and a score of 3 meant that the bird left the colony. This is a modified rank classification proposed by Rümmler et al. (2016) and was subsequently used by Weimerskirch et al. (2018) and Krause et al. (2021). In addition, prior to the experiment, we made 10 continuous observations of the colony, with a length equal to 15 min corresponding to the average duration of a single mission. Observations were made using optical equipment (an ornithological telescope) from a distance of 250 m so that the presence of the observer did not affect the behaviour of the animals. Observations ranged from 14 to 18 birds, giving a total of 160 observations.

Because our data in both case one (change in location) and case two (5 different behaviours) did not have a normal distribution, were ordinal and independent of the control observations and were not equal between groups, we used the Mann-Whitney-Wilcoxon nonparametric test. It is important that this test can be used when the variable is measured on a dichotomous scale (i.e., 0–1) because this is the case for a nominal variable that is also an ordinal variable. The null hypothesis was that there were no differences between the mean ranks. All calculations were performed in MATLAB Version: R2020a, with Statistics and Machine Learning Toolbox Version 11.7.

3. Results

3.1. Number of chicks and adult individuals and estimated number of occupied nests

Comparative analyses of nest site data for Llano Point/Rescuers Hills from different dates indicated the presence of 206 (Table 2) adult-occupied locations on at least two temporally contiguous maps. The listed temporal range and number of sites considered as potential nests which are highlighted in Table 2 provides a summary of the number of adults, the number of chicks, and an estimate of the number of active nests (made from spatiotemporal analyses of apparently occupied nests visible on orthophotos), divided into colonies and sub-colonies. A graphical representation of nest distribution within colonies and sub-colonies is shown in Fig. 5. At Petrel Hill, the establishment of 2 nests, of which breeding success was observed for 1, was indicated.

3.2. Effect of drone presence

Fig. 6 shows a comparison of the selected Llano Point colony area from different raid heights. The different height maps allowed for the identification of individuals, and the number of individuals in independent counts by two observers was identical (results are presented in Figs. 7 and 8), with differences in interpretation between observers only in determining the behaviour of individuals (see Figs. 7 and 8).

Throughout the experiment, we observed the following number of adults at each height: 64 (130 m), 66 (70 m), 69 (50 m), 68 (30 m), of which 58 were present in all images and 21 of which did not change their position or show any other variable behaviour. The distances between one nest and the others were described by the following statistics: mean: 22.42 m ± 14.20 m; median: 19.34 m; minimum: 1.14 m; maximum: 66.53 m. The exchange of individuals (emergence and departure from the colony site) was recorded by a

Table 2
Number of adults, stage, number of active nests and chicks by date in each sub-colony.

Date	Number of adults present at investigated area	Stage	Number of active nests/Number of chicks			
Llano Point/Rescuers Hills						
Rescuers Hills						
14 Oct 2020	10	Pre-laying				
16 Nov 2020	156	Incubating	Active nests		127	
04 Dec 2020	161	Incubating			143	
23 Dec 2020	141	Incubating/brooding			133	
17 Jan 2021	148	Brooding			111	
08 Feb 2021	119	Brooding/guarding	Chicks		66	
20 Feb 2021	161	Guarding			66	
15 Mar 2021	28	Post-guarding			65	
07 Apr 2021	13	Post-guarding			63	
Llano Point						
14 Oct 2020	17	Pre-laying				
16 Nov 2020	76	Incubating	Active nests		61	
04 Dec 2020	75	Incubating			63	
23 Dec 2020	57	Incubating/brooding			53	
17 Jan 2021	65	Brooding			44	
08 Feb 2021	48	Brooding/guarding	Chicks		39	
20 Feb 2021	66	Guarding			39	
15 Mar 2021	9	Post-guarding			39	
07 Apr 2021	4	Post-guarding			39	
Vaureal						
North-West						
26 Dec 2020	364	Incubating/brooding	Active nests		> 235	
28 Jan 2021	267	Brooding/guarding	Active nests	Chicks	235	142
04 Feb 2021	168	Brooding/guarding	Active nests	Chicks	219	205
21 Feb 2021	282	Guarding	Chicks		209	
South-East						
28 Jan 2021	93	Brooding	Active nests	Chicks	> 65	44
04 Feb 2021	66	Brooding/guarding	Chicks		65	
Petrel Hill- based on ground observations						
16 Nov 2020	10	Incubating	Active nests		2	
04 Dec 2020	5	Incubating	Active nests		2	
23 Dec 2020	3	Incubating/brooding	Active nests		2	
17 Jan 2021	6	Brooding	Chicks		1	
08 Feb 2021	3	Guarding	Chicks		1	
20 Feb 2021	3	Guarding	Chicks		1	

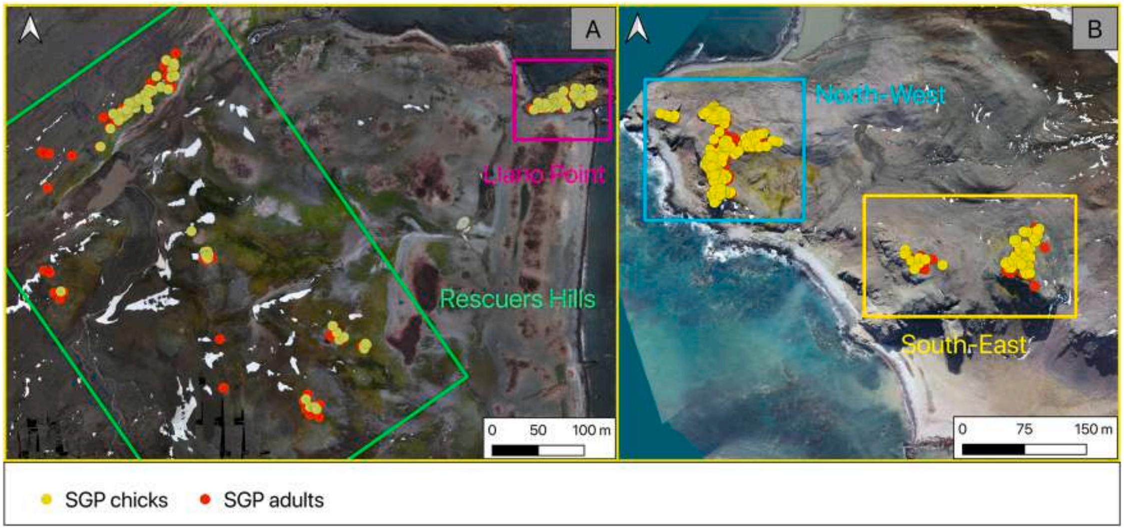

Fig. 5. Distribution of the SGP nests for the Rescuers Hills/Llano Point colony on 08 Feb 2021 (A) and for the Vaureal colony on 04 Feb 2021 (B).

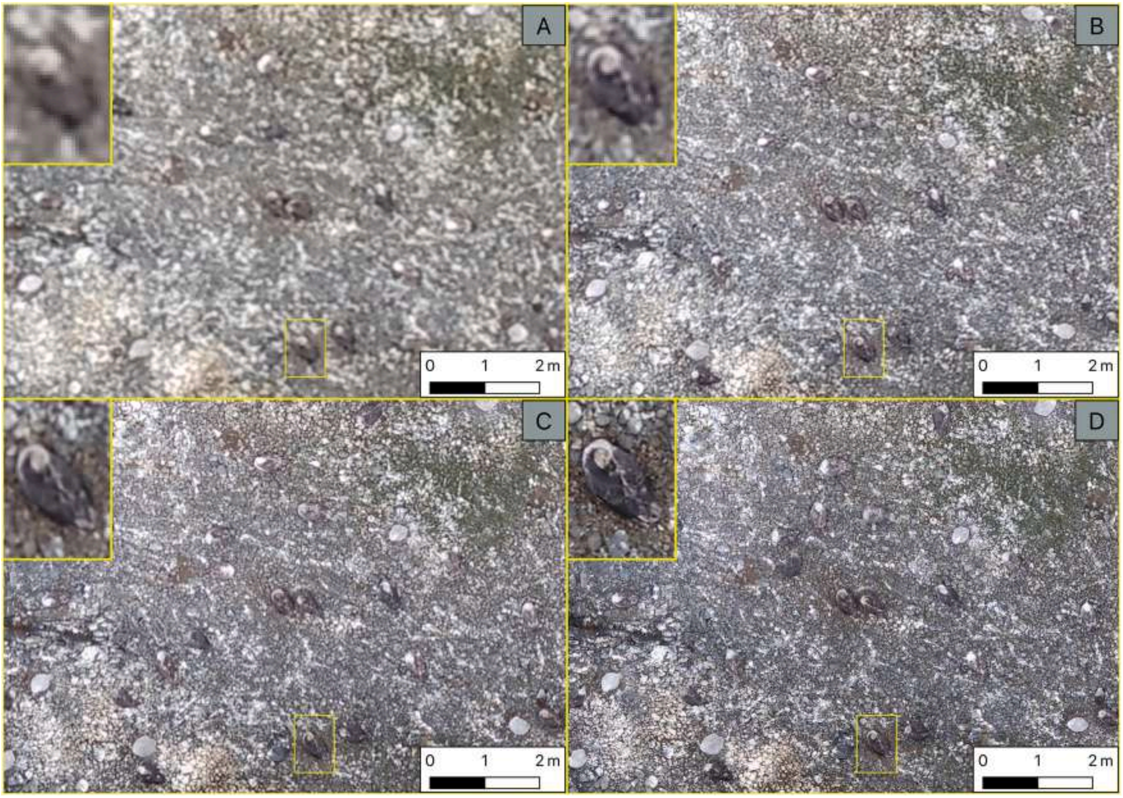

Fig. 6. Comparison of the part of the Llano Point colony area from different raid heights and corresponding GSD: (A) 130 m (2.15); (B) 70 m (1.43 cm); (C) 50 m (1.04 cm); (D) 30 m (0.63 cm).

ground-based observer and confirmed the variable number of individuals recognized at different altitudes. The number of birds that flew away from the colony immediately before the drone launch was 2 individuals, and this value did not statistically differ from that observed during the experiment (two-tailed p-value $>> 0.01$). It should be emphasized that in the case of the 130-m mission, it was not possible to determine the behaviour of the individual for 18 animals; rather, only their position could be determined. Of the responses listed in the Methods section, "the animal was looking up showing interest in the drone" was not noted either from the photo or by the observer on the ground. Among the remainder, a total of 60 responses were recorded when a lower altitude was used, of which 30

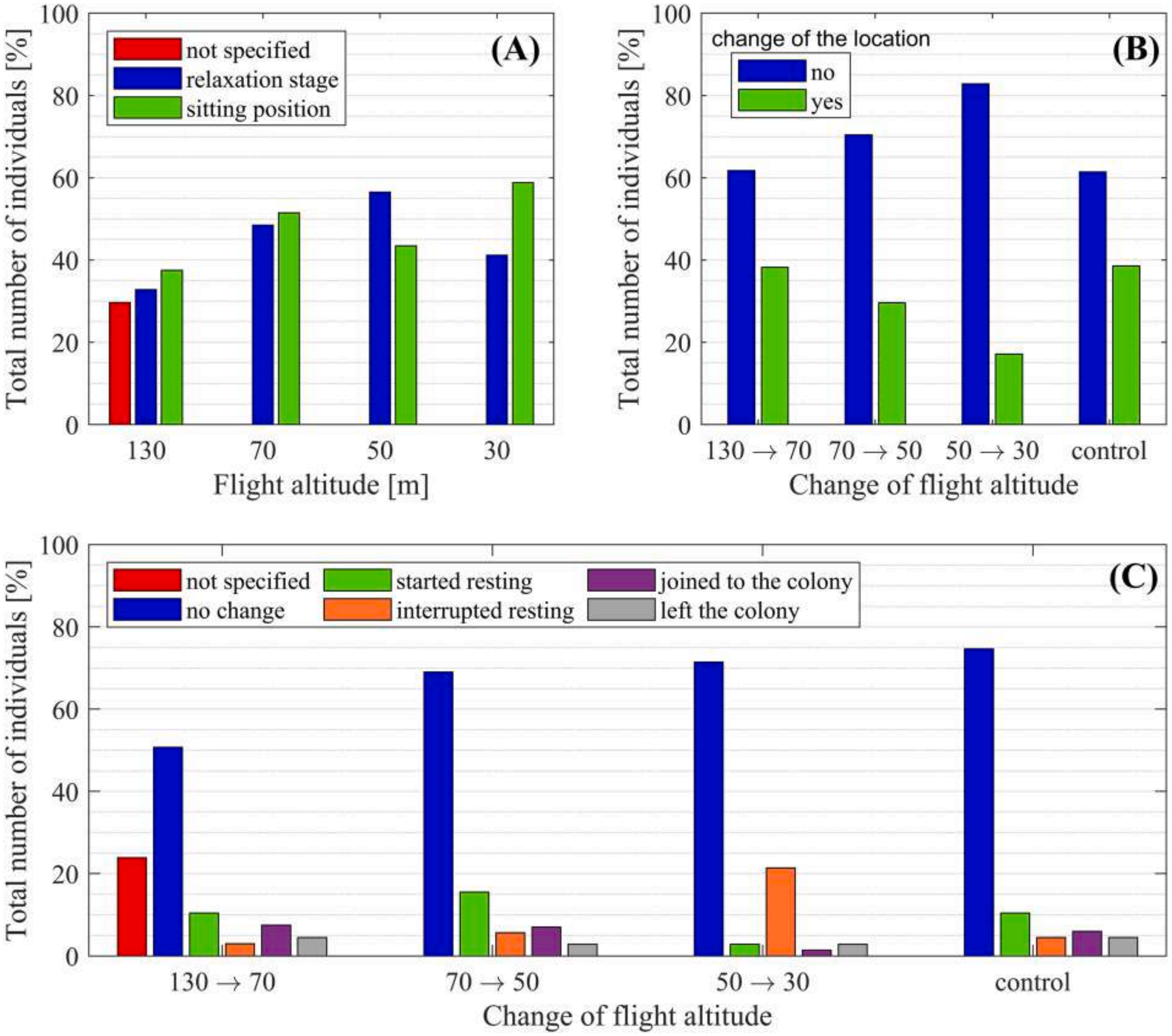

Fig. 7. Total number of adult individuals in maps for each altitude (A); change in position of adult individuals between successive altitudes (B); change in behaviour of adult individuals between successive altitudes (C).

qualified as neutral responses and 30 qualified as potentially negative. For the 39 chicks present in all images (Fig. 8), 16 showed no behaviour change throughout the entire experiment. For the rest, 25 behaviours were recorded, of which 16 were potentially negative and 9 were potentially neutral. Unfortunately, however, it must be stressed that it was not possible to assess the behaviour of 8 individuals during the 130 m mission; additionally, in the 70 m mission, it was not possible to determine the behaviour of 2 individuals. Details of the analysis are presented in Fig. 7 for adults and in Fig. 8 for chicks.

The behavioural responses of adult birds and chicks to all height changes did not differ statistically from the control results (two-tailed p-value $>> 0.01$). Thus, the null hypothesis of similarity of origin between groups could not be rejected. Clearly, in the case of a change in raid height from 50 m to 30 m, an increase in interrupted rest ('1') was noticeable, but for all results, 'no change' or 'start resting' ('0′) was the most frequent result. Furthermore, in the case of a change in position with a change in raid height, for adult individuals, a significant difference (two-tailed p-value $<< 0.001$) occurred for a change in raid height from 50 to 30 m, and we could reject the null hypothesis of equal mean ranks.

To easily identify SGP adults and chicks in the orthophotos, our suggested GSD using a DJI Inspire 2 equipped with a Zenmuse X5S Gimbal Camera and a 15-mm lens was 2.15 cm, which corresponds to a raid height of 112 – 121 m above the birds. Moreover, we could easily identify bird behaviour for a GSD of 1.43 cm, which corresponds to a height ranging from 52 to 61 m above the bird. However, as previously mentioned, for other widely available commercial drones with standard cameras, such identification would require a significant lowering of the vertical distance between the potential nest of SGP and the drone, which, as our experiment showed should be greater than 21 m in order to not disturb animals.

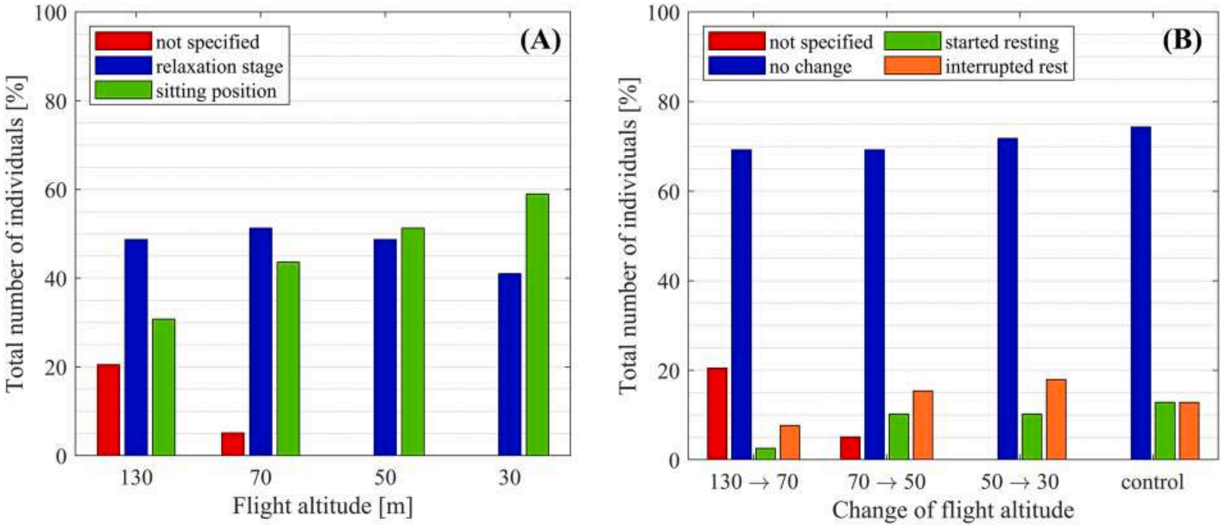

Fig. 8. Total number of chick individuals in maps for each altitude (A); change in behaviour of chick individuals between successive altitudes (B).

4. Discussion and conclusions

Ground counts of adult individuals, chicks, and/or active nests were conducted by different scientific teams at various times during the breeding season, which makes it difficult to identify trends in the breeding population size (Patterson et al., 2008). As Poncet et al. (2020) highlighted, maintaining consistency in research methodology is a major challenge. For giant petrels, factors interfering with this consistency include differences in survey design, area coverage, and timing, lack of corrections for past nesting failures; and poor documentation of historical survey methods. In this context, the method we propose could ensure complete repeatability of the research data in subsequent seasons. The area of the designed mission is stored in a digital record and can be reconstructed and the dates of the missions are noted. Photogrammetric missions facilitate counts of present adult birds and chicks in the composed mosaic of the area and also provide indisputable documentation for future comparative analyses. Obtaining photographic records ensures that the data can be archived and reanalysed years later, perhaps using more modern methods that will be available due to advances in technology. In addition, taking orthophotos can provide documentation of the variability of the environmental background, which, again in a long-term context, can be analysed to determine its effect on the size of the population under study.

An obstacle in the use of orthophoto analyses in the context of identifying active nests is that approximately 15–40% of the adult population capable of breeding may not attend their breeding formation in any one year (Voisin, 1988). Adult birds identified in the mosaic may not necessarily be involved in breeding. However, using visual clues that can be discerned from images (visible nest construction or guano around the nest), the number of apparently occupied nests can be estimated, ignoring one-time sightings of a bird at a given non-nest location. As known from past observations (Creuwels et al., 2005), both non- and failed breeders occupy a certain number of “apparently occupied nests”, so a census based on their counts may give an overestimate of the actual number of breeding efforts. To minimize the risk of overestimation, with the use of several photogrammetric missions during one season, an attempt can be made to estimate the number of “active nests”, that is, nests containing eggs or chicks, in the colony on the basis of spatial analyses. This procedure does not exclude errors but brings us closer to determining the number of adult birds engaged in breeding. The number of nests occupied in consecutive chronological orthophotos provides an estimate of the number of active nests per season. Given that this method is based on several controls, rather than just one, we can identify nests without successful breeding (a measure of success would be a chick present in the orthophoto image in the period of detection); these nests were detected only in a given interval of the period associated with the egg-laying stage and would not have been recorded without comparative analyses of multiple orthophotos. Noting the number of chicks present in successive orthophotos at the guarding stage (when the chicks were no longer covered by their parents as they emerged from the brooding stage) can aid in the determination of nesting success relative to the number of active nests. In our case, the highest number of simultaneously recorded adults was 236 on 4 December 2020 in the Llano Point/Rescuers Hills colony area and 364 on 26 December 2020 in Vaureal North-West sub-colony. Spatiotemporal analyses based on nine orthophotos from different breeding-season dates allowed us to report the maximum number of active nests, which was 206 for the Llano Point/Rescuers Hills colony on 4 December 2020 and at least 300 (only one orthophoto from the incubation, so the number may be underestimated) on 28 January 2021 for Vaureal. Poncet et.al (2020) indicated the validity of using correction factors to account for differences between the number of pairs that attempted to breed and the number counted or estimated on a given survey date. In the case of the Vaureal colony, due to the insufficient number of orthophotos, we can only determine the minimum number of nests without taking into account early incubation and broods that were lost before 26 December for the North-West subcolony and 26 January for the South-East subcolony. In the study by Poncet et al. (2020), correction factor values were based on observations of 141–185 SGPs nests from 4 seasons of intensively monitored study areas on Bird Island, where all nests were marked and monitored daily during the egg-laying period and weekly thereafter to record failures or hatchings. Because we did not have reference data from previous years, we were unable to apply correction factors for areas where we failed to make an adequate number of raids. We did not want to compare the Llano Point/Rescuers Hills colony, for which we had more data from the study season, to the Vaureal colony because we felt that the two colonies had too many differences (site specifics, location relative to the penguin colony, and the substantially different number of skuas nests observed during the initial check (in the season prior to conducting raids)). Therefore, for the Vaureal colony (Table 2) we provide the minimum (>) number of active nests.

An important point to consider that is specific to the genre, is that SGPs have no obvious enemies that perform aerial attacks (Conroy, 1972), which may be a key factor in how they respond to the presence of UASs. According to Brisson-Curadeau et al. (2017), the presence of aerial predators can result in significant behavioural disturbance and hinder the use of UASs, as in cases where the drone may be perceived as an aerial predator. Vas et al. (2015) observed that the drone's angle of descent above the bird species studied was crucial to their behavioural response. Drone approach angles close to a right angle can be associated by birds as a predator attack. However, the question arises as to whether a drone can be compared to a predator or if its presence has a completely different impact on animals. Mulero-Pázmány et al. (2017) noted that UASs are a potential new source of anthropogenic disturbance and can affect wildlife responses in a negative manner. Small UAS with electric motors have short term impacts, and these impacts are comparable to that caused by natural predators. On the other hand, the commercially available drones studied by Egan et al. (2020) elicited lower disturbance compared to that of the predator model. Ditmer et al. (2019) demonstrated the ability of American black bears to habituate and remain habituated to novel anthropogenic stimuli, such as drones, in 3–4 weeks. These researchers indicated that although cardiovascular effects were reduced (Ditmer et al., 2019), and infrequent behavioural changes in animals were observed (Ditmer et al., 2015), frequent disturbances caused by UASs may have other chronic physiological effects (Ditmer et al., 2019). In our case, we did not observe significant behavioural changes, such as changes in bird location, when the flight altitude was lowered. However, it should be emphasized that physiological stress was not measured, and more critical investigations are needed. It is important to consider additional physiological stressors on animals, especially on seabirds, due to UAS flights when developing regulations or new protocols for UAS use based on new experiments. However, with birds as sensitive to human presence as SGPs are, taking this information into consideration seems more difficult than as required for American black bears (Ditmer et al., 2015, 2019) or for chicks of King Penguins (Weimerskirch et al., 2018).

Another factor necessary to consider when estimating the risk of disturbing animals when using a drone appears to be the level of noise generated by the drone in comparison to naturally occurring ambient sound levels (Goebel et al., 2015; Palomino-González et al., 2021), which was a penguin colony with several thousand individuals in the case of the Rescuers Hills/Llano Point colony (Gentoo and Adelie) and a colony of approximately two thousand individuals in the case of the Cape Vaureal (Chinstrap penguins). Additionally, both colonies were affected by waves naturally crashing on rock formations and wind. Although the level of noise was not measured in the current study, according to Goebel et al. (2015), the hexacopter APH-22 emitting a noise of approximately 54 dB at 30 m altitude was completely masked by the noise from a Chinstrap penguin colony of approximately 600 chicks, whose noise was 84.5 dB in close vicinity to the measurements. The DJI Inspire 2 drone used in the study emits between 69 and 50 dB with the vertical distance of 10–100 m, respectively (Palomino-González et al., 2021; Thirtyacre et al., 2021). Additionally, as the horizontal distance increases, the intensity of the sound decreases regardless of the flight height. For horizontal distances above 100 m, the noise generated by the drone does not exceed 50 dB (Palomino-González et al., 2021). For our missions, this was especially important because the raid was performed continuously over a wide area, which meant that the intensity of the sound varied and decreased as the drone moved horizontally away from a particular nest. For example, for the mission at Rescuers Hills/Llano Point, the drone spent approximately 85% of the mission time at a horizontal distance from the nest exceeding 100 m (calculated as a mean time for the individual nest).

Vas et al. (2015) also highlighted potential differences in bird response to raids in relation to reproductive stage. The experiment presented in our paper was performed during the guarding stage, and the response of birds during the incubation stage may differ significantly from the results presented. As an example, in the experiment of Weimerskirch et al. (2018), conducted during a different phase of the reproductive cycle at the Crozet Islands (12 November - 7 December—a term indicating the incubation phase), SGPs were identified as one of the species most susceptible to stress, among the 11 included in the experiment, caused by the presence of the drone. The different breeding phases may explain the differences in stress reactivity shown between the individuals in the Weimerskirch et al. (2018) experiment and those in our experiment. The authors of the publication noted that some of the SGP individuals involved in the experiment exhibited a state of vigilance even before the drone took off. In their study examining the nesting status of the endangered Chaco Eagle and the degree of disturbance of drone flights to individuals, Gallego and Sarasola (2021) observed that the adults exhibited different behaviours in different breeding periods. For example, birds flew away during the nest building stage, even before drone took off, while they remained on the nest despite the presence of the drone during the incubation period. On the other hand, when they had chicks, all types of responses were observed, including alarm calls and vocalization and a single event of escape behaviour, birds only once flew away as a response to the drone. According to Montgomerie and Weatherhead (1988), these differences may be related to the fact that adults made decisions to avoid unnecessary risks (flew away before drone took-off) or not react (behaviourally) based on the presence of drone during the incubation period due to the possibility of losing a chick, which may have a huge reproductive cost. In our case, the experiment was conducted during the guarding stage and we only observed two birds that flew away before the drone took off, which may have been due to natural behaviour as only one adult was at the nest during this period. This change was not statistically significantly different from the observations obtained before the experiment, as well as during the experiment itself. Another factor that may explain these differences in our experiment and that performed by Weimerskirch et al. (2018) is the location of the colonies we analysed in the vicinity of penguin colonies whose noise drowned out the drone, which was discussed above.

Since the animals we observed during the photogrammetry missions did not show any obvious behavioural signs of disturbance, which is unavoidable during ground-based censuses by an observer, we dare to conclude that the use of a drone, although not fully studied in terms of potential harm, seems to be less obviously harmful than close human presence in the colony area. One of the most important advantages is the possibility to obtain observations over a much larger area, which is not achievable even by a group of observers or in locations that are difficult to reach by people. Moreover, another advantage is that these observations are made in a rapid manner. The acquired photogrammetric documentation, which was 9224 images in our study, will be available in later years when technological developments may allow the extraction of information from the images that we cannot even anticipate today.

CRediT authorship contribution statement

KF and RJB conceived the ideas, designed the methodology, and collected and analysed the data; both authors led the writing of the manuscript, contributed critically to the drafts and gave final approval for publication.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Załącznik 2 / Appendix 2

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Seals from outer space - Population census of southern elephant seals using VHR satellite imagery

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ABSTRACT

This work aims to assess the utility of World-View 3 satellite imagery of 31 cm resolution for conducting population censuses of southern elephant seals (SES). For the first time, we present a comparison of the results of image analyses of a 31 cm resolution satellite image and 1.39 cm resolution drone-based orthophoto of a SES breeding colony located at Patelnia Point (King George Island, breeding season 2019). The time distance separating both visualisations of the same area was 15 h 4 min, which allowed us to determine the female recognition from the VHR image with an error of 1.5–7.0%, depending on the density of females in the breeding formation. An important component of this error is false negative matches, which can have a significant impact when selecting the objects needed to train deep learning models. In addition, the pup recognition error was much higher and was 24.6–40.2%, which depended on the density of animals in harems, a large variation in the size of the pups and the diverse substrate on which they were deposited. Counting on VHR images was performed manually by eleven observers familiar with the morphological characteristics of SES. The resolution of the VHR photo allows the identification of females and males involved in breeding due to the clear dimorphism of their sizes. We recommend using VHR satellite imagery to determine the number of breeding formations and spatial relationships between them, as well as for counting females and males during the breeding haul-out. The resolution of satellite imagery available at this time is still insufficient to be used to determine breeding success. With increasing access to satellite resources and an ever-expanding database of VHR images, this is a promising tool to fill the gaps in knowledge about the global population of SESs.

1. Introduction

In the Southern Ocean region, which stretches from the coast of Antarctica to 60˚S, climate change is particularly noticeable due to phenomena such as the changing extent of sea ice and the retreat of glaciers, which open new areas of land for selected groups of organism breeding (Constable et al., 2014; Lee et al., 2017). A small number of pinniped species are endemic to the Southern Ocean region. Among them is the Southern Elephant Seal (SES) *Mirounga leonina*, whose five largest populations, account for approximately 97% of the world SES population (Hindell et al., 2016), reside on the sub-Antarctic islands: South Georgia, Kerguelen, Heard, Macquarie, and the Valdes Peninsula (Slade et al., 1998; McMahon et al., 2005; Hindell et al., 2016). The area south of 60˚S in West Antarctica is where the SES population status for proven breeding sites is unknown (McMahon et al., 2005; Hindell et al., 2016). Information on the presence of SES breeding colonies in the Southern Ocean is speculative and based on incomplete data. This ignorance is largely due to logistical reasons, difficulties in transportation, difficulty functioning in Antarctic conditions, and the ex-

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tremely high costs of such projects. Notably, the SES breeding period, including mating and pupping, takes place in September–November, with 28 October known as the haul-out peak for the South Shetland Archipelago (Carlini et al., 2002); thus, logistic facilities at this time can only be provided by year-round bases. Recent years have shown that population studies in this region should be based on alternative available methods that allow scientists to bypass the logistical inconveniences of research in Antarctica (Zmarz et al., 2018; Wege et al., 2020).

Among the known remote sensing methods used in ecology for more than 10 years, commercial high-resolution satellite images (panchromatic with a pixel <1 m and multispectral with a pixel <2.4 m) have been used to assess the conditions of terrestrial and aquatic animal populations (i.e., LaRue and Knight, 2014; Hindell et al., 2018; Wang et al., 2019). The first synoptic studies of species performed from space concerned specimens inhabiting Antarctica, i.e., estimates of the emperor penguin population size (Fretwell et al., 2012). The problem with using high-resolution satellite images was their high cost and low availability. Currently, the cost of these images is still high; however, the archival image database is becoming so rich that it allows us to search the traditional foraging places of animals, e.g., searching the 18,219 km^2 area of the Weddell Sea to count the crabeater seals (*Lobodon carcinophaga*) (Wege et al., 2020). Notably, in the case of Antarctica, the use of high-resolution satellite images meets logistical challenges and the assessment of locations that were completely inaccessible to humans 10 years ago due to the region's status as one of the most remote and isolated places in the world. Southern elephant seals are a rewarding research subject using remote methods – they meet the basic criteria of using high-resolution photos for their identification (LaRue et al., 2016), i.e., they appear in open spaces, are large enough in relation to the image resolution, and form breeding congregations (harems) that significantly distinguish them from the background. SESs are the largest seals found in the Southern Ocean, but in contrast to crabeater seals (*Lobodon carcinophaga*), Weddell seals (*Leptonychotes weddellii*) and leopard seals *(Hydrurga leptonyx*) are not associated with pack-ice, sharply contrasting with the colour of the seal, which does not allow individuals to be easily identified on the VHR image with insufficient resolution. Although the work of McMahon et al. (2014) indicates the possibility of recognising breeding colonies of SES in satellite images, thus far, this knowledge has not been used to assess the state of the SES population. This is due to the few limitations of such images, already discussed by McMahon et al. (2014), i.e., too low pixel size (0.6 m), making it difficult to distinguish males from females or to distinguish seals from landscape elements of analogous form and size. Certainly, a pixel size of 0.6 m makes it impossible to identify pups. However, since the launch of the WorldView-3 satellite in 2014 (https://earth.esa.int), it has taken 0.31 m panchromatic Very-High-Resolution (VHR) images that are sharp enough not only to allow for the counting of animals (Fretwell et al., 2017), but also to capture the distinctive shapes of different species (Cubaynes et al., 2019), opening new opportunities to address the problems identified by McMahon et al. (2014). Verifying these possibilities is the main objective of this paper.

The particular goals of this paper are (1) to compare the results of counting SES with the use of WorldView-3 image of the resolution of 31 cm per pixel and with the use of the orthophotomap obtained from an Unoccupied Aerial System (UAS) mission with the resolution of 1.39 cm per pixel; and (2) to verify the feasibility of using VHR images: (a) for the identification of the total number of harems; (b) to distinguish SES by age and sex structure-adult (females, males) and pup individuals; and (c) to count the total number of SES individuals in the area selected for comparison. Finally, we wanted to highlight the problem of ambiguity when identifying objects for training machine learning models.

2. Materials and methods

This work was carried out at the Patelnia Point area (Fig. 1C), located on the Western Shore of Admiralty Bay (Fig. 1B) on King George Island (Fig. 1C) in Antarctica, where the SES breed regularly (Fudala and Bialik, 2020).

This comparative study was based on the WorldView-3 satellite image (DigitalGlobe Catalogue ID: 104001005246D800) from October 25, 2019 (Acquisition at 10:59:43 local time at KGI) with a resolution of 0.31 m and the off-Nadir angle equal to 26.27° and the orthophotomap of the same area from October 24, 2019 (19:55:20 local time at KGI for the last photo taken) with a resolution of 1.39 cm that was obtained using a DJI Inspire 2 drone quadcopter with a Zenmuse X5S 20.8 MP camera (Fudala and Bialik, 2020). Fudala and Bialik (2020) have already indicated that for the harems considered in this study, the number of females decreased by only 5 between October 24 and 27, while the number of pups increased by 19 in this time interval. Thus, the short time distance (15 h and 4 min) between the images analysed did not make a significant difference in the number of individuals who were present in each breeding group.

Southern elephant seals were detected and counted on the satellite image manually by eleven experienced Q-GiS users. Observers were asked to mark in separate vector point layers the animals recognisable in the image divided into females, males, and pups. Counting was performed by observers familiar with the morphological differences allowing the animals to be allocated to each group. The exact location of the animals in each harem was not the subject of comparative analysis, as SES may have moved slightly within the time interval separating the two images. The number of animals identified in the satellite image was compared to a reference value determined by the number of animals counted in the drone image (Fudala and Bialik, 2020). An orthophotomap with a model of the area covered by the comparison was analysed for the presence of objects that could be false-positives (non-SES objects classified as SES) and false negatives (SES recognised as non-SES objects) for seal recognition (Fig. 2).

3. Results

Fig. 1 (C,D) shows the localisations (I–V) of all harems within the area of Patelnia Point for October 25, 2019. In the present study, an analogous division of the breeding area into zones as proposed by Fudala and Bialik (2020) was adopted, which was motivated by facilitating the comparison of animal counting results. The satellite image allowed the identification of 5 harems in Zone IV, which coincides with the analyses made on the drone-based orthophotomap of the analogous area. In contrast to the previous

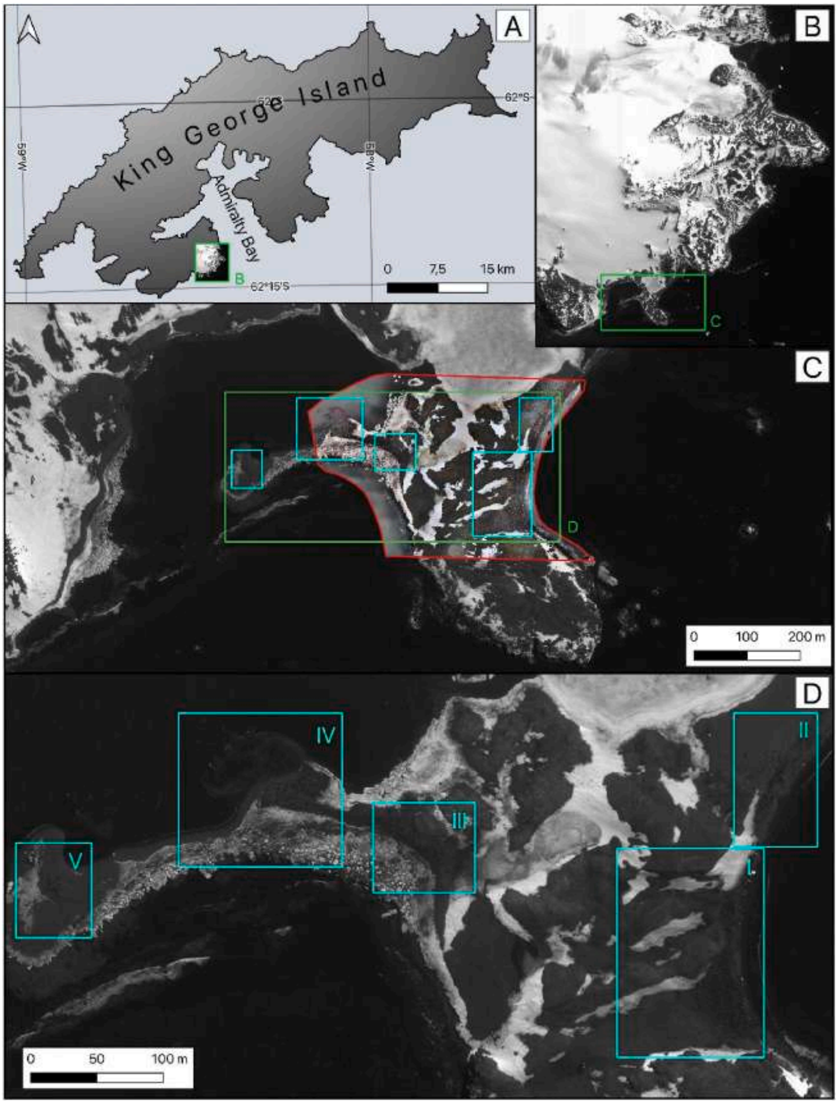

Fig. 1. King George Island (A); West Shore of Admiralty Bay (B) and the Patelnia Point area enlarged (C, D); based on the WorldView-3 images from October 25, 2019 (B, C, D) with a marked boundary of the map coverage (red line) obtained using the drone from October 24 (C). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

work, an additional harem in Zone V was identified outside the orthophoto coverage area. The Zone V area was inaccessible to ground-based observations due to its connection to land in the narrow time band of the lowest tide.

Fig. 3 shows the comparison of exemplary harem at Zone IV between views from the WorldView-3 satellite image from October 25, 2019, and from the drone-based orthophotomap for October 24, 2019. Distinguishing the SESs visible on the satellite image into three age-sex groups (females, males, pups) was feasible, but identifying the exact number of pups was difficult due to the small body size of this subgroup in relation to the resolution of the satellite image. The satellite image allowed 19 males, 403 females, 246 young to be counted (median from observers counting, Fig. 4) for the same area as for the drone-based orthophoto from the day before, for which these numbers were analogously 21, 428, 386. The presence of an additional harem outside the range of the drone image, visible however on the satellite image at Zone V, changes the total number of females by approximately 13, pups 8 and males 2 (median from observers counting, Fig. 4), increasing the total number of females (counts based on the drone mission) at Patelnia Point to at least 440.

The relative error between the median of the results obtained from the counting based on the VHR image and the results based on the counting from the drone (Fig. 4) was 7.0%, 6.5%, 5.7%, and 1.5% for each zone (I; II; III; IV) for females (Fig. 4A), while for pups (Fig. 4B), the error was 40.2%, 34.2%, 29.0%, and 24.6%, respectively. Some of these errors are the result of the following issues that we would like to highlight in this paper: (a) lack of clearly defined and precise criteria for an object to be considered a valid match, resulting in a mixture of false-positive and false negative matches (Fig. 2); (b) too small pixel size of the satellite image in relation to the size of the object (pups); and (c) visual recognition of object on homogeneous and heterogeneous substrates (Fig. 5). Notably, in the

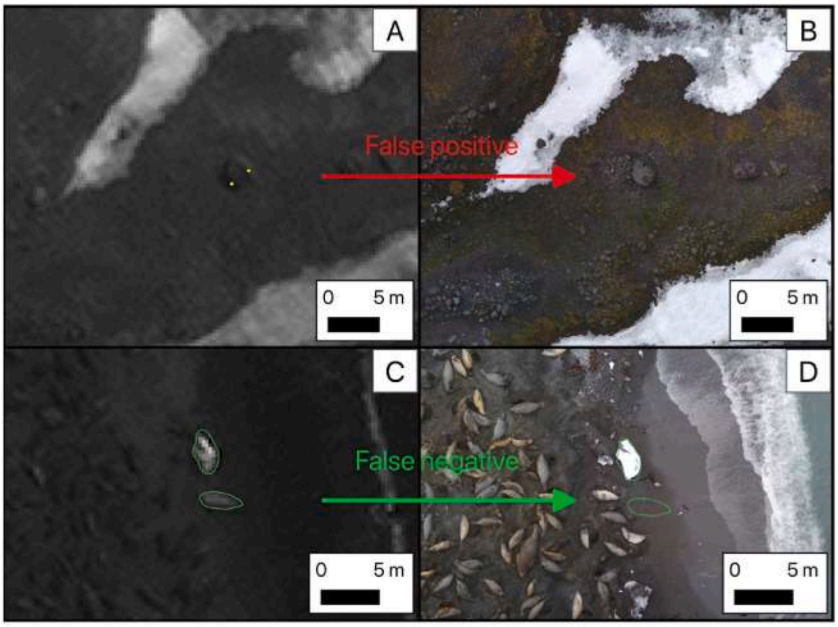

Fig. 2. Examples of false-positive (A, B) and false negative (C, D) matches.

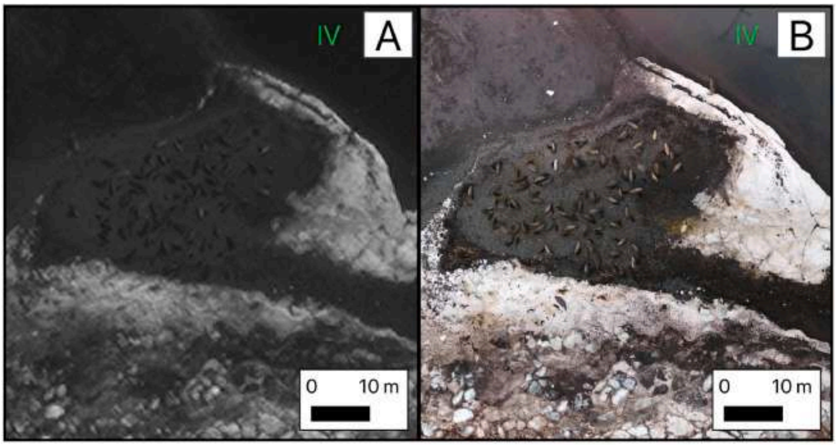

Fig. 3. The harem at Zone IV, view from WorldView-3 satellite image from October 25, 2019 (A); and view from drone-based orthophotomap from October 24, 2019 (B).

case of counting adult males, all observers involved in the count reported the same number of individuals. The difference of two individuals in the counts between the drone and satellite images may be due to the natural behaviour of adult males, which are not alpha-males and which can naturally leave the beach, where harems are located at any time.

Fig. 6 shows the results of the comparison between false-positive and false negative matches made by observers during counting, for which the median of the counts was 3 and 13 for false-positive and false negative, respectively.

4. Discussion

According to the proposed scenarios, climate change and human activity in the next 50 years will further transform the Antarctic environment (Lee et al., 2017; Rintoul et al., 2018; Rogers et al., 2020). These changes will result in, among others, new ice-free land areas (Lee et al., 2017), a reduced extent of sea ice (Rintoul et al., 2018), or changes in the depth of the water mixing zone (Constable et al., 2014). Previous studies do not clearly indicate whether changes in the extent of sea ice have a positive or negative impact on the SES population in the Southern Ocean region (Constable et al., 2014). Understanding the factors responsible for changes in the status of long-lived and slow-reproducing species is an extremely important and difficult task. To detect trends, long time data series are necessary, especially for such wide-ranging species as the southern elephant seal. The detection of changes in population growth is important and may indicate large-scale changes in the function or structure of the whole ecosystem (McMahon et al., 2005). The ice extent growth in East Antarctica (Van den Hoff et al., 2014) and its disappearance in the western part of the Antarctic Peninsula

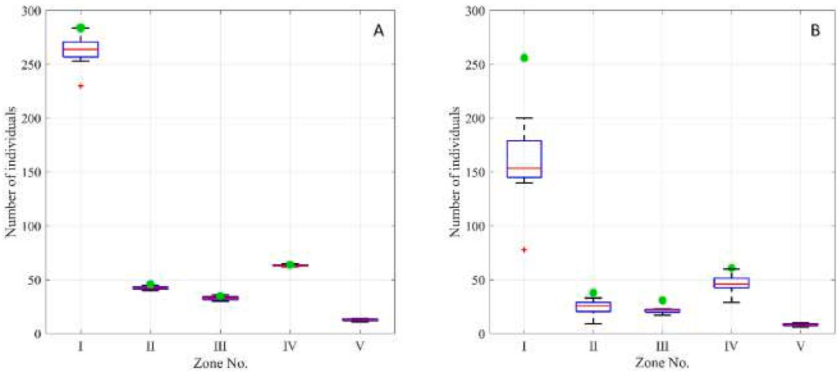

Fig. 4. Results of observers counting based on the VHR image (boxplots) with counting based on the orthophotomap obtained using a drone (green circle) of SES females (A) and pups (B). The line in the middle of the box indicates the median, the box edges represent the first and third quantiles, and crosses represent outliers of the VHR counting. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

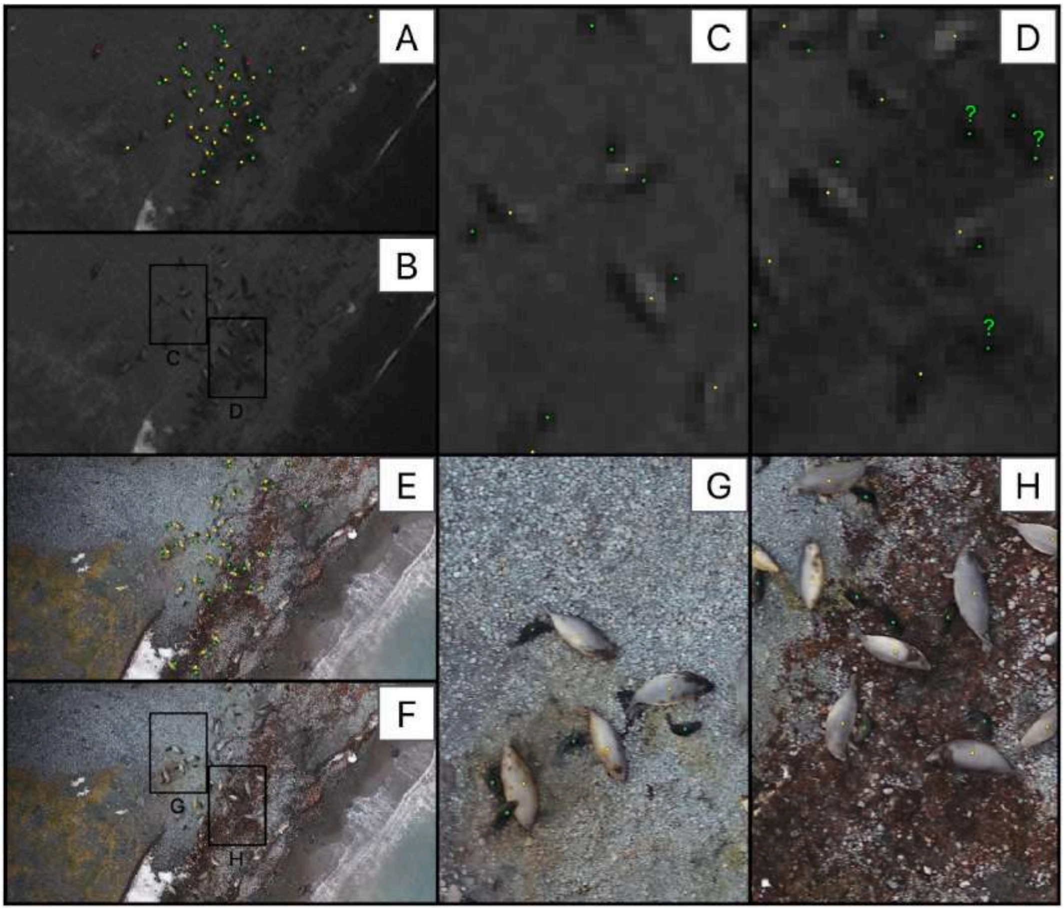

Fig. 5. Comparison of visual recognition of pups on homogeneous (C, G) and heterogeneous (D, H) substrates. Analogous fragments of breeding formation on satellite image 25 Oct (A, B) and orthophoto 24 Oct (E, F). Colour dots: yellow females, green pups, red males. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

(Costa et al., 2010) have had a very negative impact on access to feeding sites for SESs (Bestley et al., 2020). On the other hand, the lack of sea ice resulting from the warm summer of 1983 in the vicinity of Palmer Station on Anvers Island, Antarctic Peninsula, resulted in the first observed births of SES in this area (Helmark and Helmark 1986); according to Siniff et al. (2008) and Corrigan et al. (2016), SESs can benefit from these changes, as exposed areas can create new breeding and molting grounds, and their populations can expand into these new sites. The maximum extent of ice coverage occurs in the Southern Hemisphere at the end of September, the period when SESs begin breeding.

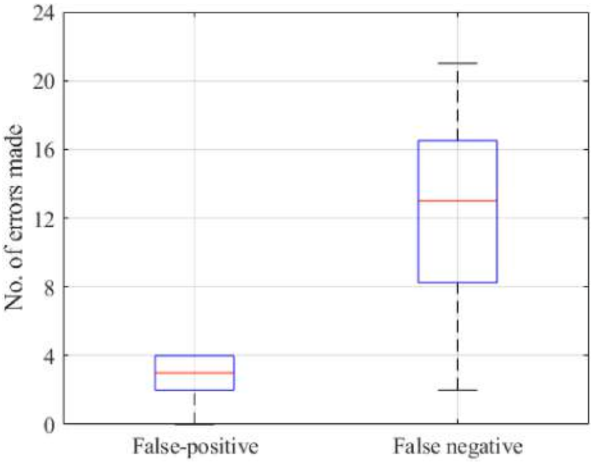

Fig. 6. Comparison between false-positive and false negative matches made by observers during counting of females.

In this paper, we wanted to prove that, satellite remote sensing, offers great possibilities over spatial and temporal scales and addresses the need for a modern SES census technique. We have shown that many of the problems McMahon et al. (2014) pointed out can now be solved with the use of 31 cm resolution images, although some problems still have to wait for higher resolution images.

As already mentioned in Section 3, the relative error between the median of the results obtained from the female counting based on the VHR image and the results based on the counting from the drone varied from 1.5% to 7.0%. Various factors influence the value of this error. Overall, it should be noted that observers undercounted females, making 13 false negative matches (median value). The relative error seems to be an error that cannot be excluded in any way and indicates that with the currently available VHR image resolution, it is not possible to obtain a solution to identify SES females with the use of very popular deep learning models for which precision and recall would be 1. Currently, the selection of objects that are admitted to the training stage of any machine learning model is usually made by two independent observers (i.e., Gonçalves et al., 2020; Duporge et al., 2021), and it is customary for them to require consensus on a given object. Any ambiguity in interpretation that is not identified by both observers is removed (Duporge et al., 2021), or the observers seek consensus by analysing the questionable results (Gonçalves et al., 2020). This naturally leads to further questions: to what extent does the experience of observers influence their identification of animals in satellite images? and can we eliminate objects so freely, with the assumption that in this way we reduce the accuracy of the animal count result based on model used? In the first case, the answer can already be found in our study. All participants were experienced individuals, knowing the terrain, having visited the study site directly; at the same level, they operated the Q-GIS software used to analyse the photo, as well as having seen SES from ground level. This shows that this is a very subjective element, dependent on the observer, which will be difficult to eliminate in the future. However, regarding the second question, it should be remembered that the fundamental issue of using different models in the analysis of the abundance of any animal species is to obtain an accurate count. However, in this work, we show that VHRs can be used to determine the location of animals, but their number can still be determined with some error. Here, again, the question arises as to what kind of error might be acceptable.

It is important that due to the significant difference in size of the females and males present at the breeding colony, it is possible to distinguish them on the satellite image. In addition, it should be noted that sex recognition of adult individuals is only possible from photos during the breeding season, when the animals are gathered in the colony area and only strong adult males, clearly larger than females, participate in the act of reproduction. During molting, it is not possible to identify young males lying on the shore from females, as their body size may be similar for certain age ranges (Le Boeuf and Laws 1994). Unfortunately, it is still not possible to use a satellite photo for counting pups; an error of 40% or even 25% does not allow for effective counting of this age group.

Based on the orthophoto of 24 Oct, we measured the length (using Q-GIS software) of all pups (n = 386) (from tip of nose to tail end) and their maximum width in perpendicular projection. The average length of pups during this period was 1.2 m (min 0.73 m, max 1.56 m), while the average maximum width of individuals was 0.4 m (min 0.24 m, max 0.66 m). Although the average length of the young for the reproductive stage we considered corresponds to more than 3 pixels of the analysed image, the characteristic dark colouration of the SES pups allows them to be visually recognised only against a bright, homogeneous background. Diverse substrates, such as fragments of shoreline covered in patchy algae, are not conducive to pup detection (Fig. 5). Furthermore, around weaning, pups shed their dark lanugo, which, together with their substantial body size, compounded by their large fat reserves to survive postweaning fasting, means that they can be increasingly difficult to recognise from females.

Wide coverage of satellite imagery is certainly a significant advantage of VHR images over a drone mission, which always has limitations related to the lengths of the possible flightpath transects. This made it possible to detect an additional harem present in the Patelnia Point area, which is impossible to detect on the orthophoto due to its limited coverage.

Fortunately, the number of females and males is sufficient to assess the southern elephant seal population during the breeding haul-out, and in this respect, this paper shows the method allowing determination of the SES population size in the Southern Ocean, which in turn will impact the current view of the total population distribution of this species.

Ethical statement

The authors declare that all ethical practices have been followed in relation to the development, writing, and publication of the article.

Authors' contributions statement

KF and RJB conceived the ideas, designed the methodology, and collected and analysed the data; both authors led the writing of the manuscript, contributed critically to the drafts, and gave final approval for publication.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Robert Bialik reports financial support was provided by Ministry of Science and Higher Education of Poland.

Data availability

Data will be made available on request.

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Załącznik 3 / Appendix 3

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Communication

Identifying Important Bird and Biodiversity Areas in Antarctica Using RPAS Surveys—A Case Study of Cape Melville, King George Island, Antarctica

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Abstract: A remotely piloted aircraft system (RPAS) survey of an area containing the eastern extremity of King George Island, including Cape Melville and an extensive part of Destruction Bay, as well as small offshore islands, was undertaken in December 2022. Using RPAS, an inventory of the Destruction Bay area was performed. Chinstrap penguin and Antarctic shag nests were found on Cape Melville and on Trowbridge Island, Middle Island, and an unnamed area located between the Ørnen Rocks formation and Trowbridge Island. During the survey, 507 Antarctic shag nests and over 9000 chinstrap penguin nests were mapped in the investigated area; 458 Antarctic shag nests and 4960 ± 19 chinstrap penguin nests aggregated together on an 8.61 ha land section of Cape Melville were identified. The quantity of Antarctic shag nests found allows for the classification of the area of Cape Melville as an IBA. Among the 175 currently known colonies of Antarctic shags in Antarctica, this is the fifth largest. In this paper, we present the results of the survey, including orthophotos with mapped nest locations. We propose the following recommendations to policy makers and the scientific community: (1) the area of Cape Melville should be classified as an Antarctic Important Bird and Biodiversity Area; (2) based on the RPAS flight, a new boundary of the Cape Melville IBA is proposed; (3) the threshold value (based on >1% of species) to establish an IBA for Antarctic shags should be changed to 122 to reflect the increased estimate of the global population of Antarctic shags; and (4) an inventory of all areas, including previous IBAs that can be qualified as "major colonies of breeding native birds", should be recommended at the Antarctic Treaty Consultative Meeting (ATCM). In logistically inaccessible bird breeding sites, such as the one presented here, RPASs should be used to carry out regular monitoring of Antarctic Important Bird and Biodiversity Areas.

Keywords: Antarctic Important Bird and Biodiversity Area; King George Island; Cape Melville; RPAS in wildlife conservation; Antarctic shag; *Leucocarbo bransfieldensis*; chinstrap penguin; *Pygoscelis antarcticus*; IBA monitoring

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1. Introduction

Scientific claims of the existence of a colony of chinstrap penguins on Cape Melville, King George Island, date back to 1937 (Roberts (unpublished data), cited by Croxall and Kirkwood [1]). Richard Sherratt was captain of the ship, Lady Trowbridge, which, according to available sources, wrecked on Christmas Day, 1820, in the area of the bay and named the area surrounding the Melville Peninsula to the north "Destruction Bay" because of that event. In his notes published in the Imperial Magazine in 1821, he mentioned that innumerable penguins were present in the waters of South Shetland [2]: "The first intimation you have of being near South Shetland is meeting with a great quantity of whales, of the black kind, and what are called the fin-black, you may thence conclude you are about 150 miles from land. Standing on to the southwards, you will meet with innumerable penguins, so many that you would almost conclude that the sea was animated". Aside from isolated details of the description of the topography of the Melville Peninsula, no data on the fauna of Cape Melville were provided. The aerial image taken in December 1965 by the Falkland Islands and Dependencies Aerial Survey Expedition (FIDASE), made available on the USGS Earth Explorer platform as a component of the Antarctic Single Frame Records collection in a joint initiative of the British Antarctic Survey (BAS) and the United States Antarctic Resource Center (USARC) (i.e., photo number X26FID0039127), clearly shows bird breeding groups in the Cape Melville area, delineated by bright patches of guano on the rocky ground. The first estimated figures of the abundance of birds found there date back to 1966 (White (unpublished data), cited by Croxall and Kirkwood [1]). According to White, on 27 January 1966 at Cape Melville, 3250 chinstrap penguin nests were accumulated in an area of 2512 m^2. The same observer also reported the existence of 1100 nests of chinstrap penguins on Ørnen Rocks. According to Jablonski [3], in 1980, the Cape Melville area was occupied by 9970 pairs of chinstrap penguins. In addition, there was a total of 6308 chinstrap penguin nests on Trowbridge Island, which is approximately 4 km away from Cape Melville, and the author’s notes indicated adjacent unnamed islands were included in the Trowbridge Island formation. Jablonski [3], verifying the claims of Croxall and Kirkwood [1], noted that chinstrap penguins do not nest on the steep rocks of Ørnen Rocks, as Kirkwood reported, so it is more likely that he observed Antarctic shags. In 1987, the Antarctic Marine Living Resources Program provided data on the number, size, and location of penguins and blue-eyed shag colonies and the breeding status of other seabirds in South Shetland Islands [4]. According to the data, 8000 to 9000 (rough estimate) adult chinstrap penguins and 200 to 300 (potential breeder) Antarctic shags were found in Cape Melville on 30 January 1987, while approximately 2000 (rough estimate) adult chinstrap penguins were present on Trowbridge Island on the same day. The authors mentioned that they did not perform a census of the offshore islands on the north coast of KGI from Cape Melville to False Round Point.

According to Harris et al. [5], the Cape Melville area was classified as IBA Ant 063 based on criterion A4iii: “The site is known or thought to hold, on a regular basis, at least 20,000 waterbirds, or at least 10,000 pairs of seabirds, of one or more species”. This was based on a publication by Woehler [6], according to which the Cape Melville area held 16,278 breeding pairs of chinstrap penguins. However, Woehler [6] referred to a publication by Jabłoński [3] that reported the number of penguins in the Cape Melville area and deduced that the Cape Melville area (9970 pairs) also included the Trowbridge Island colony (6308 pairs), which did not fall within the boundaries of the area designated as ex IBA 063. In 2015, in a publication that determined the status of Antarctic IBAs [7] by taking all of the abovementioned considerations into account, IBA 063 Cape Melville was declassified due to the inaccuracies and the lack of updated data for the area.

The eastern and northern parts of King George Island are areas not routinely monitored due to remoteness and difficult accessibility, even though there are ten research stations belonging to nine different countries on the island. The purpose of our field survey was to take an inventory of the Cape Melville area and most of the offshore and inshore areas of Destruction Bay using remotely piloted aircraft systems (RPASs). To our knowledge, the survey we performed to update bird abundance represents the first in the area of Destruction Bay since that of Shuford and Spear [4] which was conducted in 1987.

Before we proceed to the next section, it should be noted that in 2020 BirdLife International [8] revised and updated the nomenclature for the criteria used to identify IBAs. For example, criteria A4i and A4iii no longer exist, but these are related to Global IBA Criterion A4: “The site is known or thought to hold congregations of $\geq$1% of the global population of one or more species on a regular or predictable basis” and Regional IBA Criterion B3b: “The site is known or thought to hold, on a regular basis, at least 20,000 waterbirds or at least 6700 pairs of seabirds of one or more species”, respectively.

2. Materials and Methods

According to BirdLife International [9], there are 205 terrestrial IBAs and 74 marine IBAs in the Antarctic, of which 12 and 4 are located on or adjacent to King George Island (Figure 1), respectively. Nine of the currently existing terrestrial IBAs were classified based on demographic data that were collected 35 or more years ago, while four were declassified in 2015 due to a lack of data (Figure 1) [7].

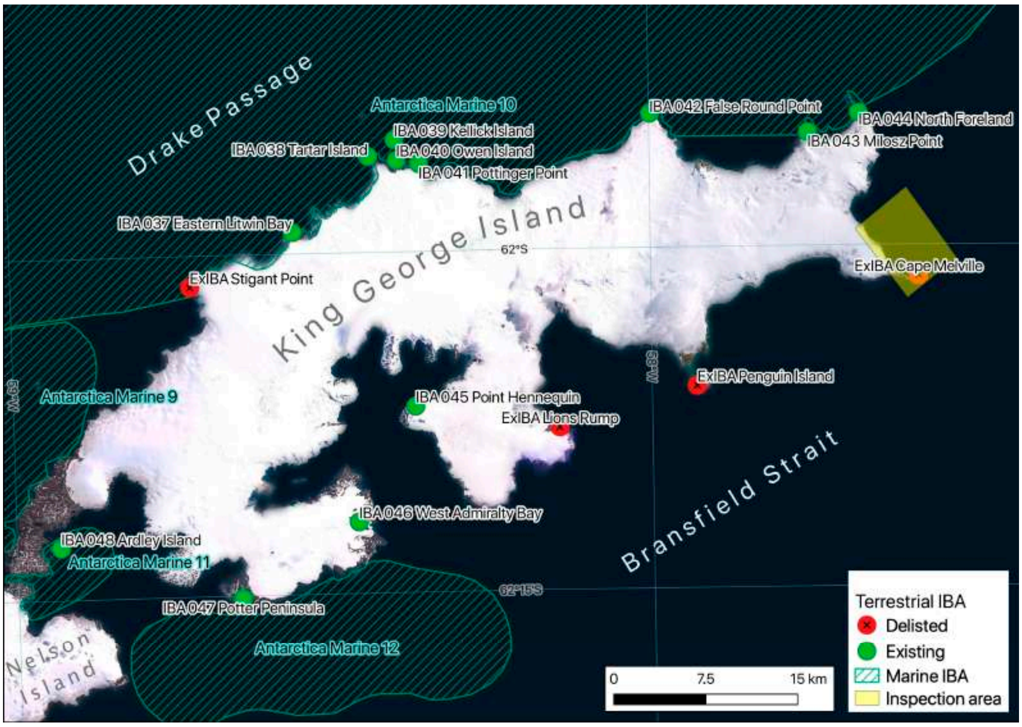

Figure 1. Localisation of terrestrial and marine IBAs within the vicinity of King George Island.

An area of approximately 38 km^2 containing Cape Melville of the King George Islands and offshore islands from Destruction Bay was surveyed (Figure 2). Inspections of the area were launched from the Arctowski Station, located approximately 50 km from Cape Melville, and were carried out using two offshore pontoon boats (Zodiac model MilPro SeaRib SRA-750 and Zodiac MilPro MK6). The boats were equipped with radar, and depth measurements were taken to allow safe manoeuvring in the area where numerous skerries were present. A record of the GPS route from the boat used for shore landings is plotted on the map presented in Figure 2. Except when the observers disembarked, both boats followed the same track. Checks to confirm species recognition were made both from the boat level, using optical equipment in places where landing was not possible, as well as from the land level by ground-based observers in places where landing the boat was feasible. Designed control photogrammetric missions using RPAS were carried out using two drone models:

- DJI Phantom 4 Pro V 2.0 with 20 Mpix camera (DJI, Shenzhen, Guangdong, China), hereafter referred to as Phantom;
- DJI Inspire 2 drone quadcopter with a Zenmuse X5S 20.8 MP camera (DJI MFT 15 mm/1.7 ASPH lens with a 30-mm-equivalent focal length; DJI, Shenzhen, Guangdong, China), hereafter referred to as Inspire.

Both were operated by a pilot who was qualified to fly drones weighing up to 25 kg, for a distance of up to 2 km. All activities were approved by the Polish Antarctic Programme and performed under permit no. 13/2022 for the period from January 2022 to March 2024.

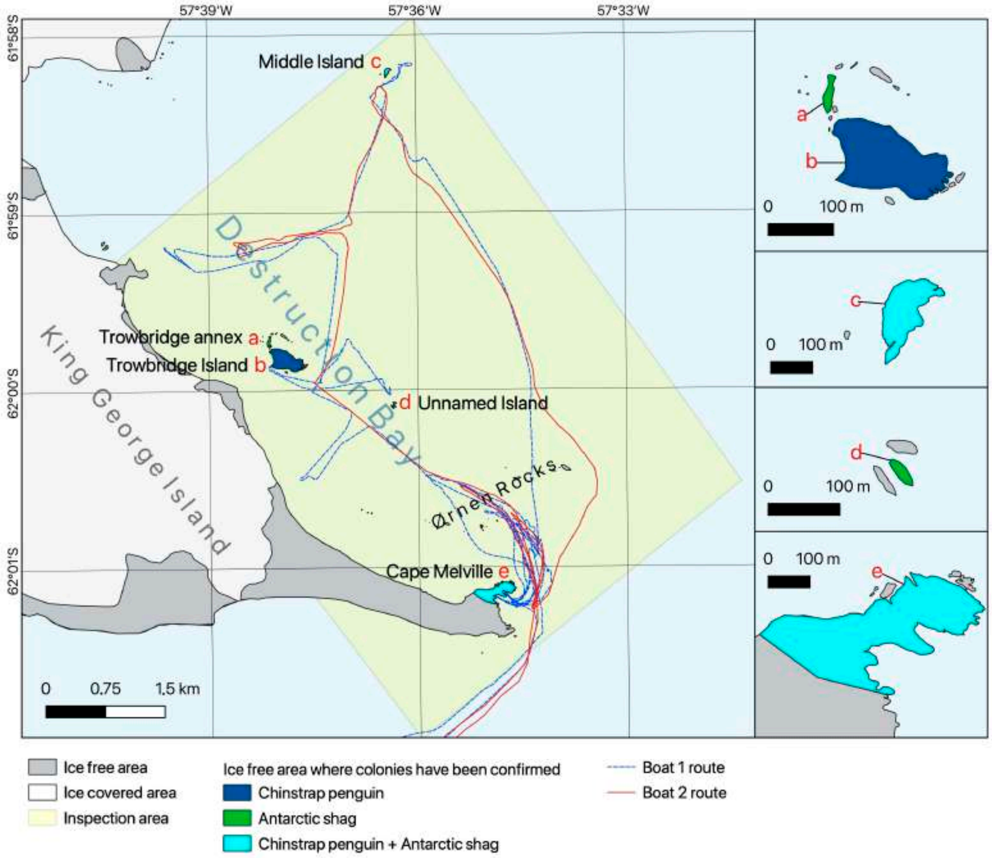

Figure 2. Inspection area at Destruction Bay with breeding sites of investigated fauna species at: Trowbridge annex (**a**); Trowbridge Island (**b**); Middle Island (**c**); Unnamed Island (**d**); and Cape Melville (**e**).

The recommendations presented in Environmental Guidelines for operation of Remotely Piloted Aircraft Systems (RPAS) in Antarctica (v 1.1) [10] and the latest publications on best practices in operating RPASs in Antarctica by Harris et al. [11] were considered when carrying out the RPAS flights.

Visitation of the Cape Melville area was performed three times: the first visit, which allowed confirmation of the presence of colonies on Cape Melville by ground observers, was made on 16 January 2022 (breeding season 2021/2022). In the next breeding season (2022/2023), inspections of Cape Melville were performed using the RPASs on 4 December 2022 and 12 December 2022, which covered the same nest areas. Inspection of the islands from the Destruction Bay area was performed with the Phantom on 12 December 2022. The flight path of each mission was designed using DJI Ground Station Pro (DJI, Shenzhen, Guangdong, China). Flight altitudes and other details of all missions are listed in Table 1. All potential breeding sites in the inspection area marked in Figure 2 were investigated using RPAS, but surveys with pre-planned waypoints were designed and executed only over islands where nests were found after the initial flight. For the Ørnen rocks, a group of islets and sea rocks that was listed in the literature data as a nesting site for chinstrap penguins [1] and a potential nesting site for Antarctic shag [3], the pre-planned waypoint missions were also conducted even though preliminary flights did not confirm the presence of nests. The initial flights were performed in real time using Phantom. The altitude of the survey was set to be at least 40 m above the potential nests and other wildlife.

Table 1. Remotely piloted aircraft systems (RPASs) flight specifications.

Mission Date	Localisation	Area Coverage [ha]	Flight Duration	Flight Altitude [m]	Ground Sampling Distance (GSD)—Pixel Resolution [cm]
4 Dec 2022	Cape Melville	5.36	7 min 04 s	80	1.84
4 Dec 2022	Ørnen Rocks	8.06	4 min 10 s	90	2.51
12 Dec 2022	Cape Melville	8.61	17 min 49 s	80	1.77
12 Dec 2022	Unnamed Island	0.44	4 min 41 s	45	1.20
12 Dec 2022	Trowbridge Island	17.00	5 min 37 s	200	5.01
12 Dec 2022	Trowbridge Island	10.52	9 min 34 s	45	1.22
12 Dec 2022	Middle Island	0.83	6 min 44 s	45	1.22

3. Results

The resolution of the obtained orthophotos ranged from 1.20 cm/px to 5.01 cm/px. First, a mission from 200 m covering all of Trowbridge Island was carried out, followed by another mission from a lower altitude covering the main penguin breeding locations on Trowbridge Island together with the island's surroundings, including small islets and sea rocks (Table 1). In Trowbridge Island surroundings, we found small, unnamed islets, which we call Trowbridge Annex, where nests of shags (AS) were present. Orthophotos of all islands where shags or penguins were identified are shown in Figure 3. They were created in Pix4Dmapper (Pix4D S.A., Prilly, Switzerland) and analysed using Q-GIS software. AS and CHPE nests were counted independently by three people in the form of point vector layers, and the results are presented in Table 2. Only active nests were counted, i.e., those that had a distinctive shape [12]; those that were judged by the observer to have been formed during the 2022/2023 breeding season; and those where at least one bird was clearly visible within the nest. It was assumed that each active nest corresponded to a pair of birds.

Table 2. Results of Antarctic shags (AS) and chinstrap penguins (CHPE) nests at different locations within Destruction Bay.

Date	Localisation	CHPE	AS
		Mean ± SD	
4 December 2022	Cape Melville	4824 ± 12	458 ± 0
	Ørnen Rocks	0	0
12 December 2022	Cape Melville	4960 ± 19	456 ± 0
	Trowbridge Island	3435 ± 16	0
	Trowbridge annex	0	7 ± 0
	Middle Island	627 ± 6	10 ± 0
	Unnamed Island	0	32 ± 0

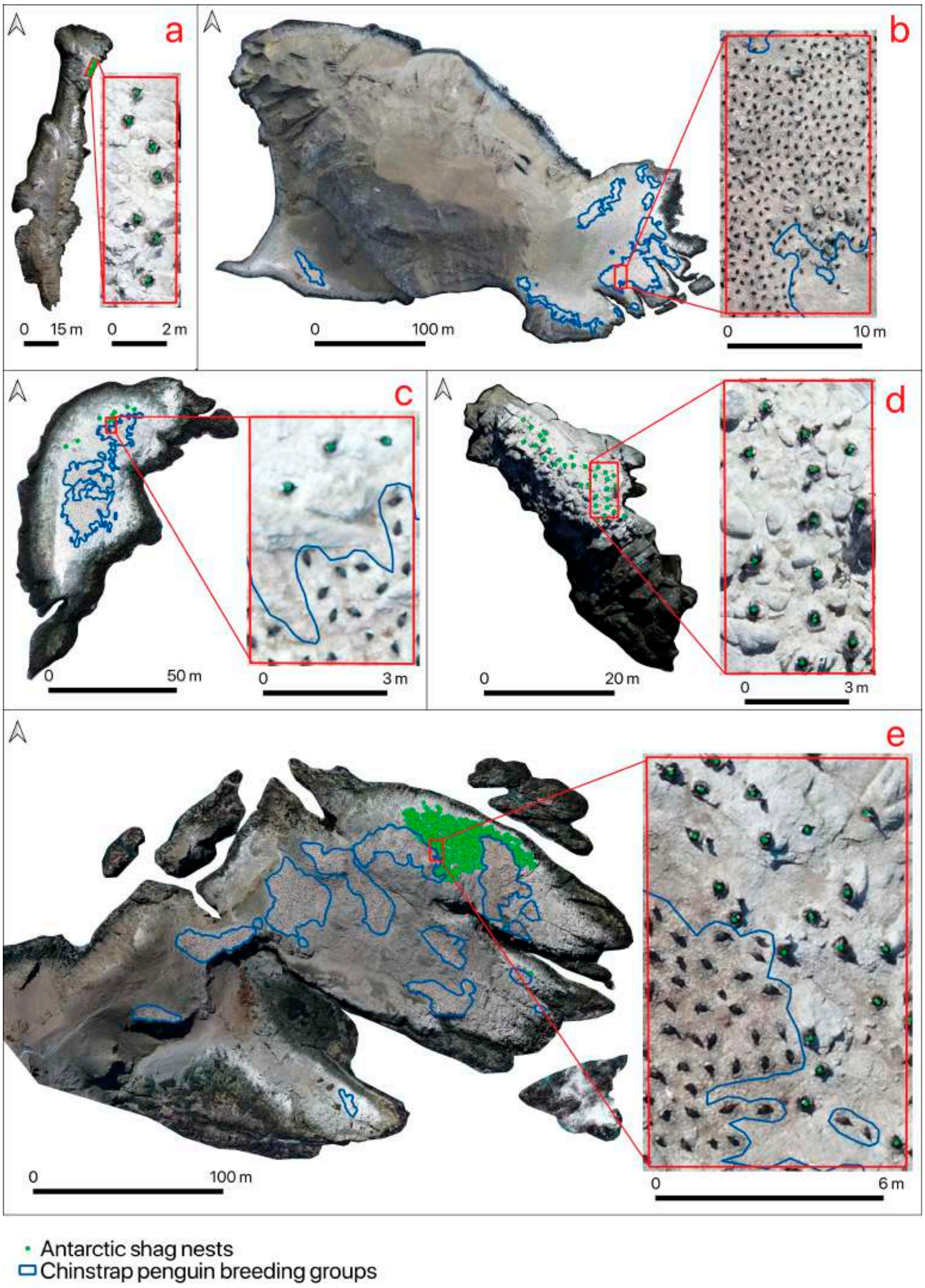

Figure 3. Orthophotos of Trowbridge Annex (**a**); Trowbridge Island (**b**); Middle Island (**c**); Unnamed Island (**d**); Cape Melville (**e**).

Based on the survey documenting the presence of 458 Antarctic shag nests on Cape Melville, we propose restoring IBA status to Cape Melville, but suggest new boundaries for the area, as presented in Figure 4. The proposed area does not include the entire ice-free area of the Melville Peninsula, and excludes the Cape Melville geological formation, as well as the plateau located at an elevation of more than 120 m (Figures 4 and 5). The proposed region covers an area of 23.2 hectares. The former IBA boundary included an area of 344 hectares and two outlying islands from the Ørnen Rocks formation, where our survey ruled out the potential for penguin or Antarctic shag nests. Shuford and Spear [4] noted that several species were possibly breeding in Cape Melville: cape petrel (*Daption capense*), Wilson's storm-petrel (*Oceanites oceanicus*), snowy sheathbill (*Chionis albus*), brown and south polar skuas (*Stercorarius antarcticus* and *Stercorarius maccormicki*) and kelp gull (*Larus dominicanus*). Our surveys confirmed at least one kelp gull nest (Figure 6). Moreover, we found more than 50 snowy sheathbill individuals foraging in this area (possible breeders) and at least one pair of brown skua foraging there. The steep slopes and landslides with rock rubble in the area also suggest that the location may be a potential nesting site for Wilson's and black bellied storm petrels; however, because these are not surface nesting species, their detection on orthophotos is not possible. Importantly, the Melville Peninsula geological formations are extremely valuable sites from a scientific perspective. The Destruction Bay formation provides unique land exposure and a record of late Oligocene warming in West Antarctica [13], which predisposes the entire area of the Melville Peninsula to be placed under special protection in the future.

Figure 4. Boundaries of Ex IBA AQ063 Cape Melville and the newly proposed area of IBA Cape Melville.

Figure 5. Western aerial view of Cape Melville.

Figure 6. Kelp gull nest visible on orthophotos on 4 December 2022 (**left**) and 12 December 2022 (**right**).

4. Discussion

Data obtained through surveys conducted in December 2022 confirmed that the Cape Melville site qualifies as an IBA based on criterion A4, which states [8]: "The site is known or thought to hold congregations of ≥1% of the global population of one or more species on a regular or predictable basis". This was indicated by Harris et al. [5] as the basis for IBA classification with the contribution of the Antarctic shag with the population threshold designated as 133 pairs of this species (threshold determined by Harris et al. [7] based on population size estimates proposed by Wetlands International, Waterbirds Population Estimates, Fourth edition). Schrimpf et al. [14] estimated the global population of this species at 11,366 pairs at 173 active colonies (plus an additional 1984 pairs of uncertain taxonomic status in the South Orkney Islands, which, however, according to a later publication by Rawlence et al. [15], were included in *Leucocarbo georgianus*). Philips et al. [16] reported one additional/new colony of Antarctic shags, which was found in Ryder Bay, increasing their global population to 11,684 breeding pairs. Since then, drones have been used to assess the abundance of these birds at two known locations at Harmony Point on Nelson Island [17] and along the north-western coast of Nelson Island and the south-western coast of King George Island [12]; the numbers were similar, considering seasonal fluctuations, and not significantly different from those previously reported at these locations by Schrimpf et al. [14]. Our study confirmed that Antarctic shags can be easily identified on aerial images with resolutions of 1.84 cm/px. Using both the estimates adopted by Harris et al. [7] and the most recent available data on the species, the number of Antarctic shag nests we detected increased the global population from 11,684 to 12,191 breeding pairs, allowing the area of Cape Melville to be considered an Important Bird and Biodiversity Area, as the number of nests represents approximately 3.8% of the known global breeding population of the species. Finally, according to Schrimpf et al. [14], the colony of Antarctic shags we found at Cape Melville is the fifth largest and is the 175th known colony in Antarctica.

In 2015, a full list of 204 Antarctic IBAs was acknowledged by the Antarctic Treaty Consultative Meeting (ATCM) 2015 with the adoption of Resolution 5, which recommended that representatives consider the recognised IBAs in the planning and conduct of activities in Antarctica, and emphasised that ongoing research is required to improve knowledge of the status and trends of Antarctic bird populations [18]. Although IBA inventory and monitoring should be evaluated regularly [19,20], the last evaluation of the Antarctic terrestrial IBA network took place in 2015 [18]. Since then, only one area (AQ 205 Ryders Bay) has been added to the IBA list based on the suggestion presented by Philips et al. [16], and, recently, more emphasis has been placed on marine areas [20]. As a result, 74 Antarctic marine IBAs were listed in 2020 [9].

For 2023, the age of the source data on which the terrestrial areas were assessed was as follows: 7.8% from 10 years ago or less, 55.1% from 11 to 20 years ago, 5.4% from 21 to 30 years ago, 20.5% from 31 to 40 years ago, 7.8% from 41 to 50 years ago, and 3.4% from 51 to 60 years ago. The evaluation for some IBAs (particularly those classified under criterion A4iii, species group—seabirds) was based on more than one publication, and the most recent publications among those included in the evaluation were accounted for in this percentage distribution. The trigger species for 24 of 205 IBAs was *Leucocarbo bransfieldensis*, while Pygoscelis penguins were the trigger species for 49. Based on our experience with the surveys conducted in the Cape Melville area, we recommend the use of RPASs to conduct an inventory of terrestrial IBAs for the presence of penguin and shag nests, as well as other colonial breeders potentially nesting in open spaces that can be identified on aerial images.

There are currently twelve IBAs identified across King George Island, nine of which were classified based on demographic data collected 35 or more years ago. This includes all IBAs (AQ037-044, with trigger species CHPE) located on the northern ridge of King George Island. This is a logistically complex area due to the large number of coastal skerries which make pontoon boat landings difficult or impossible, difficult wind conditions (part of the island is uncovered from the Bransfield Strait), and the long distance (by sea) from the nearest survey stations. RPAS surveys, especially long-range fixed-wing surveys, seem to be an excellent solution for logistically complex locations [21]. RPAS launches can take place from a ship's landing site or a stretch of coastline accessible to land. As shown in Table 1, the mission time is disproportionate to that required for a ground survey with a ground-based observer, and the process is certainly less invasive to animals than aerial surveys using a helicopter, which, for many Antarctic Specially Protected Areas, whose boundaries coincide with the IBA, such as ASPA 128 (IBA 46) or ASPA 132 (IBA 47), are prohibited.

5. Conclusions

Based on our study we propose the following recommendations to policy makers and the scientific community:

(1) The area of Cape Melville should be classified as an Antarctic Important Bird and Biodiversity Area;
(2) Based on the RPAS flight, a new boundary of the Cape Melville IBA is proposed;
(3) The threshold value (based on >1% of species) for establishing an IBA for Antarctic shags should be changed to 122 because, based on our study, the global population of this species increased to 12,191 breeding pairs;
(4) An inventory of all areas, including previous IBAs that can be qualified as "major colonies of breeding native birds", should be recommended at the Antarctic Treaty Consultative Meeting (ATCM).

In logistically inaccessible bird breeding sites, such as the one presented here, RPASs should be used to carrying out regular monitoring of Antarctic Important Bird and Biodiversity Areas.

To assist in further conservation and management planning, the penguin count data obtained within this study will be added to MAPPPD [22].

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Załącznik 4 / Appendix 4

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Using machine learning to count Antarctic shag (*Leucocarbo bransfieldensis*) nests on images captured by remotely piloted aircraft systems

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ABSTRACT

Using 51 orthomosaics of 11 breeding locations of the Antarctic shag (*Leucocarbo bransfieldensis*), we propose a method for automating counting of shag nests. This is achieved by training an object detection model based on the "You Only Look Once" (YOLO) architecture and identifying nests on sections of the orthomosaic, which are later combined with predictions for the entire orthomosaic. Our results show that the current use of Remotely Piloted Aircraft Systems (RPAS) to collect images of areas with shag colonies, combined with machine learning algorithms, can provide reliable and fast estimates of shag nest counts (F1 score > 0.95). By using data from only two shag colonies for training, we show that models can be obtained that generalise well to images of both spatially and temporally distinct colonies. The proposed practical application opens the possibility of using aerial imagery to perform large-scale surveys of Antarctic islands in search of undiscovered shag colonies. We discuss the conditions for optimal performance of the model as well as its limitations. The code, data and trained model allowing for full reproducibility of the results are available at https://github.com/Appsilon/Antarctic-nests.

1. Introduction

In recent years, Remotely Piloted Aircraft Systems (RPAS) have become common tools for wildlife monitoring (Fischer et al., 2023; Schad and Fischer, 2023) and have proved especially helpful for surveying habitats that are otherwise difficult to access, such as many areas of Antarctica (Pina and Vieira, 2022). RPAS are used to monitor many different wildlife species in the Antarctic environment, including whales, surface-nesting birds and pinnipeds (Fudala and Bialik, 2022a, 2022b; Tovar-Sánchez et al., 2021; Zmarz et al., 2018). Long-term monitoring of wildlife populations with RPAS often generate extensive datasets, frequently high-resolution images that are used to build orthomosaics on which animals (or other features) can be counted (Xu et al., 2024). Manual counting of individuals on an orthomosaic requires a large investment of researchers' time. To minimize counting errors and to obtain a measure of repeatability, counting is often carried out by more than one person, thus multiplying the effort needed (Fudala and Bialik, 2022b; Qian et al., 2023). One solution is to use citizen science to analyse the vast amounts of data provided by satellite and aerial imagery (Francis et al., 2022; LaRue et al., 2020; Wood et al., 2021). On the other hand, machine learning algorithms provide a way to automate the detection and counting of objects in images, increasing the efficiency of wildlife monitoring with RPAS. Computer vision models have been used, for example, to detect seabird guano in satellite imagery (Le et al., 2022; Witharana and Lynch, 2016) and to detect penguins, albatrosses and many other seabird species in images obtained from RPAS (Liu et al., 2020). While the use of RPAS in ecological studies is expanding rapidly, the use of machine learning for visual data collection is growing at a much slower rate, at 50% per year between 2004 and 2018 (Dujon and Schofield, 2019). However, it should be noted that support for aerial image analysis using wildlife detection models has recently become more common, allowing for accurate detection (Qian et al., 2023) and

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estimation (Bakana et al., 2024) of animal numbers to aid wildlife management and conservation (Lyu et al., 2024).

Antarctic shag is a surface-nesting species that breed in the Antarctic Peninsula and surrounding island groups. Antarctic shags often breed in locations that are completely inaccessible to ground-based surveys (e.g., sea cliffs, islets or elevated capes), making aerial photography the only reliable means available to survey some breeding sites. Antarctic shag was a trigger species for the establishment of 24 of 205 Antarctic Important Bird and Biodiversity Areas (IBAs) (Harris et al., 2015). Monitoring their population trends are therefore important (Schrimpf et al., 2018), but to our knowledge, there is currently no method available to automatically detect and count Antarctic shags in RPAS images.

In this study, we use an object detection machine learning model to identify and count individual Antarctic shag nests in georeferenced aerial images collected by RPAS. We chose to count individual nests (a proxy of breeding pairs) rather than individual birds because Antarctic shags and chinstrap penguins sometimes breed in mixed colonies. Antarctic shags have a white patch of plumage on their backs, seen on aerial imagery as a "white spot on a black background", which can be used to distinguish shags from penguins. However, the white spot on the back is only visible when birds sit in certain postures, and the similar body dimensions of shags and chinstrap penguins, therefore necessitates the use of more than one distinguishing feature where these species occupy a common space. Antarctic shag nests are three-dimensional objects that are built of organic material, mainly guano-cemented marine algae and feathers (Bernstein and Maxson, 1982). These nests are highly recognizable on aerial photographs even though their appearance may take different forms through the breeding season. The diameter of shag nests is approximately 49 cm (SD = 5 cm, $n = 43$) (Pfeifer et al., 2021), which is >50 times larger than the image pixel that can be obtained using RPAS when operating at recommended RPAS flying altitudes in Antarctica (Harris et al., 2019; Weimerskirch et al., 2018). Hence, nests are ideal objects for counting on aerial images. We collected 51 RPAS orthomosaics from 11 breeding locations of Antarctic shags and tested whether an object detection machine learning model trained on images from two colonies could accurately count nests separated in time (at the training colonies), or in space (at other colonies). We discuss the performance of the model to count nests and identify colony locations, and the benefits and limitations of applying this method.

2. Data and methods

2.1. Orthomosaics

The data used in the study were collected by three independent

Table 1
Details of the photogrammetry missions. The sequence numbers correspond to the locations in Fig. 1.

No.	location	RPAS used	lon	lat	maps	Data source
1	Trowbridge Island	DJI Phantom 4 Pro v2.0	−61.9939	−57.6357	1	Fudala and Bialik (2023)
2	Unnamed Island A	DJI Phantom 4 Pro v2.0	−62.0016	−57.6061	1	Fudala and Bialik (2023)
3	Cape Melville	DJI Phantom 4 Pro v2.0	−62.0216	−57.5836	3	Fudala and Bialik (2023)[1]
4	Turret Point	DJI Inspire 2	−62.0885	−57.9506	5	this study
5	Shag Rock	DJI Inspire 2	−62.1844	−58.2979	33	this study
6	Unnamed Island B	DJI Phantom 4 Pro v2.0	−62.2451	−58.4937	1	this study
7	Nelson Island	DJI Phantom 4 Adv.	−62.2896	−59.2315	1	Oosthuizen et al. (2020)
8	Fregata Island	Bormatec Ninox	−62.2290	−59.1022	2	Pfeifer et al. (2021)
9	Unnamed Island C	Bormatec Ninox	−62.1937	−59.0552	1	Pfeifer et al. (2021)
10	Kwarecki Island	Bormatec Ninox	−62.1183	−58.8915	1	Pfeifer et al. (2021)
11	Rzepecki Island	Bormatec Ninox	−62.0928	−58.8430	2	Pfeifer et al. (2021)

[1] One of the maps has not been published before.

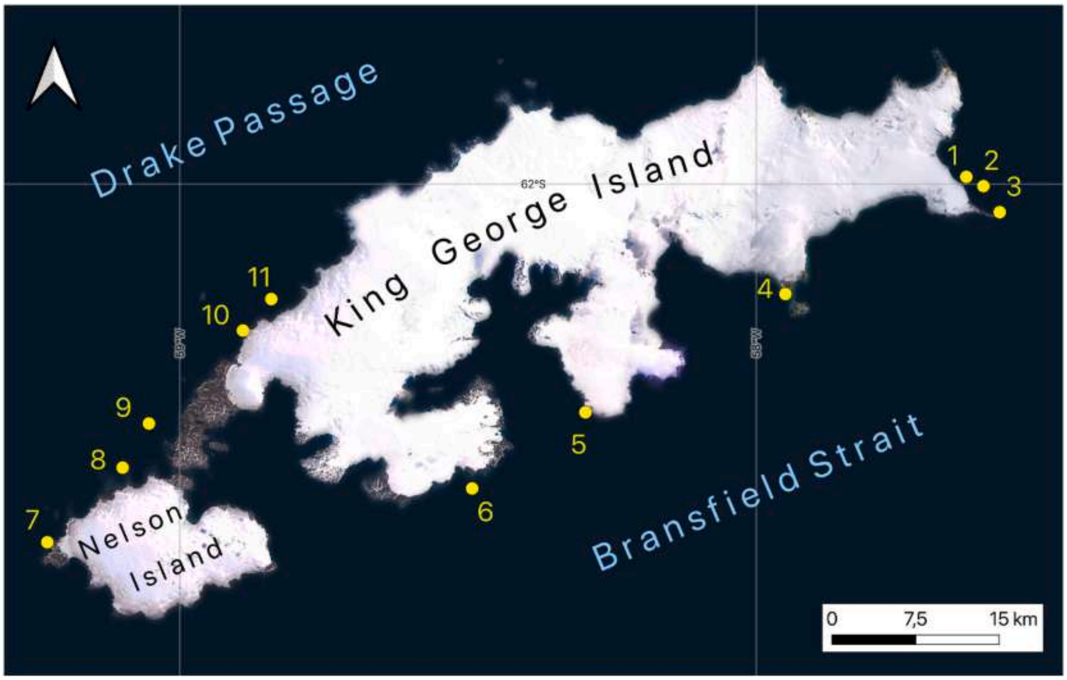

Fig. 1. Locations of Antarctic shag colonies considered in the study. The study colonies (1 to 11) are listed in Table 1.

research groups using various types of RPAS (Table 1). In total, 51 surveys of 11 sites around King George Island and Nelson Island on the South Shetland Islands were flown (Fig. 1). For all the locations, there were multiple images and we used Pix4Dmapper (Pix4D S.A., Prilly, Switzerland) to generate a single orthomosaic of the area of interest. Notably, the orthomosaics varied in spatial resolution and visual characteristics (level of focus, localised spatial distortions, chromatic aberrations, etc.). Details on the size and characteristics of the data used in the study are presented in Table S1.

2.2. Labels

All the shag nests in each orthomosaic were visually identified by an experienced observer who assigned labels to the centre of each nest via QGIS software (QGIS 3.16.5 'Hannover'). When shags and penguins were breeding in close proximity (Fig. 2), we used additional supporting imagery taken from the ground or expert knowledge of the particular location to assist labelling.

Antarctic shag nests have a regular, circular shape and are approximately 50 cm in diameter (Pfeifer et al., 2021). Therefore, the locations of the centres of the nests were translated into square bounding boxes of 50 cm × 50 cm. The bounding boxes were defined geographically, but since the images were georeferenced, they could be translated to specific pixels of relevant images (Fig. 3).

2.3. Dataset creation

The orthomosaics available for the study varied significantly in spatial resolution (from 6 mm to 34 mm per pixel; see Table S1). The images were rescaled to a common resolution of 20 mm per pixel, for two reasons: first, to achieve a model able to perform well also at lower resolutions; second, to leverage the constant size of the nests in terms of occupied pixels. The choice of the resolution used was dictated by the data available for the study (only two orthomosaics had lower resolutions) and the intention to leave at least 25 × 25 pixels per nest. The images used in the study captured significant geographical distances (400 m in the case of Cape Melville), while the objects of interest, i.e., the nests, were only 50 cm in size. Hence, an approach involving tiling (also often referred to as patching) was used. All the orthomosaics were cut into square tiles of 12.8 m × 12.8 m along longitudinal and latitudinal directions, using a custom Python script written for that purpose and available in the shared code repository. In this way, each tile was 640 × 640 pixels. For the orthomosaics that had lower resolution than the one used in the analysis (e.g., Nelson Island and Kwarecki Island), the resulting tiles contained less information than could have been fitted in the tiles, with their pixels stretched to preserve the geographical distances needed. Tiling was performed with 6.4 m overlaps in the longitudinal and latitudinal directions so that the majority of the nests were captured on four individual tiles. The key reason for introducing the overlap and making it so significant is that this way each nest would be fully present in at least one of the tiles.

This approach has two side effects to be considered. First, most of the nests appear multiple times in the data, which requires attention when combining model predictions per image, as they need to be automatically combined into a single prediction per actual nest. To overcome this, non-max suppression is applied after collecting the predictions for a given image (and not for individual tiles). This amounts to a process of filtering candidate (overlapping) predictions by selecting a prediction with the highest confidence, eliminating all other candidates overlapping with it above a threshold and moving on to the next candidate with highest confidence, until no candidates are left (Bodla et al., 2017). Second, using the overlap increases the number of tiles that need to be analysed by a model at inference time. This is a reasonable trade-off for most practical use cases since real-time results are not needed and therefore speed of analysis is considerably less important than model performance (having more undetected nests).

2.4. Training data

Two locations were chosen to constitute the training dataset: Shag Rock and Cape Melville. For Shag Rock, two images were left out of the training dataset and were instead included in the test set. Those images were captured several months after the last image from the training dataset from Shag Rock. Similarly, for Cape Melville, two images captured in December 2022 were used for training, while an image taken in November 2023 was used for testing. In this way, the robustness of the model against temporal separation can be tested. To balance the proportion of tiles including instances of nests, we first selected only tiles with nests; from the remaining tiles, we randomly selected a corresponding number of tiles without nests. With this selection, the training dataset spans 2.5 years and includes images of shag nests in various stages of its annual reproductive cycle, including empty nests. With an extensive dataset covering a cross-section of the entire year, it was possible to train the model on a variety of substrate and shape variants on which nests occurred during the period (Fig. 4).

2.5. Validation data

For validation of the model performance, during the training process (after each training epoch), we used three orthomosaics of Turret Point (with ~2/3 covered by a shag colony), which were spatially distinct from the orthomosaics included in the training dataset (Shag Rock and Cape Melville). Two (temporally distinct) out of five images of Turret Point were left out of the validation set and added to the test set.

2.6. Test data

The performance of the models was tested on the following groups of data:

1. Orthomosaics of Shag Rock, Cape Melville, and Turret Point, temporally distinct from those in the training or validation data (i.e., coming from a consecutive nesting season). Referred to later as "test-over_time".
2. Orthomosaics of locations geographically distinct from both the training and validation datasets in Unnamed Island A, Unnamed Island B and Trowbridge Island. Referred to later as "test-over_location".
3. Orthomosaics of locations geographically distinct from both the training and validation datasets (Kwarecki, Rzepecki, Fregata, Nelson and Unnamed Island C) and collected with older types of RPAS and cameras (with imagery and labels provided by external experts). Referred to later as "test-over_source".

For the test data, the entire set of available images was used for tiling, which in some cases meant significant portions devoid of any nests (e.g., showing water).

3. Machine learning model

3.1. Model definition

To detect shag nests automatically, several of the "You Only Look Once" (YOLO) state-of-the-art object detection framework models (YOLOv5, YOLOv6L and YOLOv8) were tested. Based on the achieved scores of F1 metric and by inspecting visually the quality of the predictions on individual orthomosaics, the model that performed the best was YOLOv6L. We employed the standard architecture of the large YOLOv6 model with the recommended hyperparameters for fine-tuning [https://github.com/meituan/YOLOv6] and standard data augmentation parameters that are largely consistent across YOLO versions. The training time was approximately 10 h on a single Nvidia T4 GPU. We experimented with using different levels of overlap in both the training data and at inference time. We used many different sets of hyperparameters in YOLOv6 but found only marginal improvements; hence, we decided to use the default settings. Finally, we tested working with different spatial resolutions of the imagery, most notably including training runs in which the same images were used at different spatial resolutions. We did not find this approach to be beneficial.

3.2. *Inference method*

Object detection models typically return bounding box proposals, which are equipped with scores indicating the confidence level of a given prediction. The following method was used to combine predictions obtained for individual tiles into a collection of final predictions per orthomosaic:

1. Since we cut the data into overlapping tiles and some nests may have landed at the border of the tile, we removed all predictions closer than 50 cm to the border. Nevertheless, all nests at the tile borders are visible on other tiles.
2. To have only one prediction per nest, we removed lower confidence predictions that significantly overlapped with one another (those that had intersection-over-union above the threshold of 0.2, which for equal areas of two predictions means that one-third is common).
3. We filtered the predicted bounding boxes according to their size (requiring that neither side be smaller than 0.3 m or larger than 0.7 m, the area of the nest be larger than 0.125 m^2 and the ratio of the sides be larger than 0.8).
4. Finally, to assess the counts of the predicted nests, a confidence score threshold of 0.5 was used (notably, this threshold was not used in the calculation of mean average precision (mAP) scores; see Supplement).

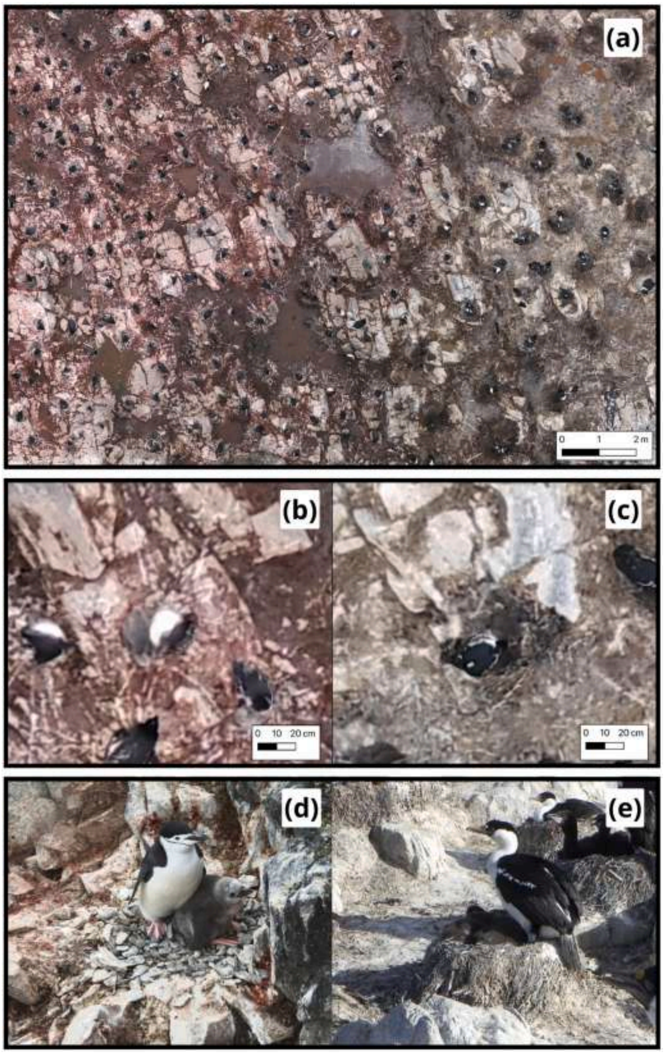

Fig. 2. Section of an orthomosaic depicting a mixed Antarctic shag and chinstrap penguin colony, showing the coexisting nests of both species (a); chinstrap penguin nest, aerial view (b); Antarctic shag nest, aerial view (c); chinstrap penguin nest, ground view (d); Antarctic shag nest, ground view (e).

3.3. *Model evaluation*

After the model predictions for individual tiles were combined into a set of predictions for the entire orthomosaic, the quality of the predictions could be assessed. For the purpose of testing the models, we used the tiling of the entire available test orthomosaics. The quantitative assessment was based on an approach tailored to the use case. First, a threshold of confidence was applied to determine the location the model predicts for the nests (e.g., allowing counting of the nests predicted by the model - a key ecological indicator). Each prediction was considered as a true positive (TP) if there was a nest with which it overlapped significantly (had an intersection over union above the threshold of 0.3). Each nest could be predicted only once. The predictions that were not matched to a nest were treated as false positives (FP), while the nests that were not assigned to any prediction were treated as missed or false negatives (FN). To provide an overview of the results, the F1 score was used. Based on the above definitions, F1 is calculated by the following formula (Van Rijsbergen, 1979):

$$F1 = \frac{TP}{TP + \frac{1}{2}(FP + FN)}$$

This metric is the harmonic mean between precision and recall and is a commonly used (e.g., Peng et al., 2020; Roy et al., 2023), conservative measure of performance that balances both missing predictions (FN) and not predicting too many instances (FP). When F1 takes its highest value of 1.0, it indicates excellent precision and recall; when precision or recall are zero, the value of F1 is 0.

3.4. *Computational requirements*

Inference on a single tile took approximately 60 ms using an Nvidia T4 GPU. Inference on tiles from all the orthomosaics in this study took 15 min on the GPU, and they covered an area of 0.6 km^2. This means that inference for tiles covering an area of 1 km × 1 km, would take approximately 25 min. While the post-processing time of the predictions (merging predictions per tile into a single prediction per image) is negligible, additional time may be required to prepare the tiles. Rescaling and cutting a large orthomosaic into tiles can take significantly longer depending on the starting resolution and hence the size of the image representing the orthomosaic.

4. Results

4.1. *Model performance*

The model achieved F1 scores above 0.991 on all the training and 0.934 on all validation data (Table 2). For the testing data, it generalised exceptionally well over time at all locations ("test-over_time") with scores above 0.934. Moreover, a nearly perfect F1 score was also obtained at locations it had never seen before ("test-over_location") reaching F1 scores of 0.985, 1, and 1 (see an example in Fig. 5). In Table S2, we report the F1 scores as well as the number of actual nests, TP, FP and FN for each individual orthomosaic. Notably, the three orthomosaics in test-over_location contained 48 nests. The model correctly

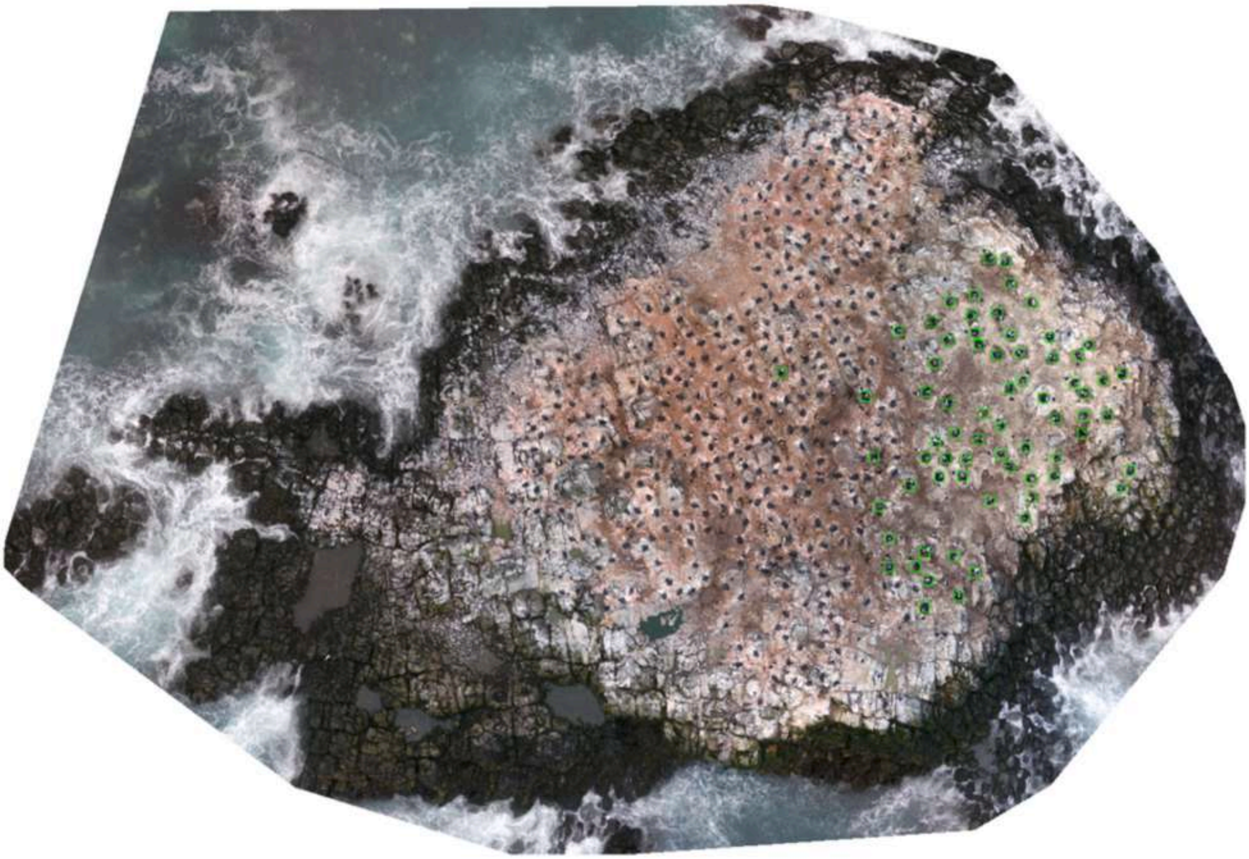

Fig. 3. Orthomosaic of the Shag Rock colony with shag nests labelled using 50 cm × 50 cm bounding boxes (in green). Note the presence of some shag nests within the penguin colony. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

detected all of them and only produced one incorrect prediction — predicting that an apparent shag is sitting on a nest.

Even though the orthomosaics often covered large areas, with colonies of shags occupying a small portion of the field of view, false positive predictions of nests almost never happened, while nests distant from the colony were correctly identified. But, it is informative to inspect the few cases where the model made mistakes. The two lowest F1 scores were obtained for Fregata and Rzepecki Islands. In both cases, the orthomosaics had optical artefacts such as out-of-focus blur and colour aberration present in the entire image, and the area with the nests was distorted, blurred and stretched possibly as a result of being located on a hillside (Fig. 6). Fortunately, in both cases, we had a second orthomosaic each captured within days from the first one. In both cases, the results of the model on the orthomosaic of better visual quality yielded scores as high as those achieved on the remaining studied data. Notably, the orthomosaic of Rzepecki Island captured on 2016-12-30 received a nearly perfect score, with 58 nests out of 62 identified correctly and no false predictions.

Cape Melville is also worth mentioning. While two orthomosaics of this large colony (seven times more nests than the second largest, Shag Rock and Nelson Island) were present in the training data, they were both collected during good weather conditions, with no snow in the colony and the nests clearly visible. The orthomosaic used in the test dataset (collected almost a year later), included large areas covered by snow, which obscured some of the nests. Regardless, the number of predicted nests was within a marginal error of 0.8% of the actual number of 478.

4.2. Issues identified in the data

There were several shortcomings in the quality of the imagery used in the study. First, it is common practice to collect images as individual pictures with a fixed set of camera settings and RPAS altitudes. This, in some cases, resulted in images varying in the appearance of the colours and in parts of the images (e.g., containing parts of the islands that were higher than their neighbourhood) being out of focus. Second, the individual images are combined into a single map with noticeable marks on the patching, including in particular the borders of pixels occupied by shags. Since the birds are often occupying their nests, this had a bearing on the visual quality of the objects studied in this study. Third, the data collected in 2016, which were used as part of the test set, contained in some cases significant optical artefacts, such as motion blur, chromatic aberration and spatial distortions (Fig. 7).

5. Discussion

The possibility of using RPAS for large-area mapping of locations where Antarctic shags nest seems more feasible now than ever before, particularly using fixed-wing drones which can cover longer distances and map much larger areas than quadcopters (Pfeifer et al., 2021; Zmarz et al., 2018). RPAS surveys coupled with computer vision models can therefore be a valuable tool for ecosystem monitoring. Our results show that modern machine learning models can provide reliable counts of Antarctic shag nests. Coupling RPAS surveys with machine learning processing of images can thus be used to monitor species that are difficult to count in other ways (Fischer et al., 2023). Machine learning provides a solution for simplifying the processing of aerial imagery and allowing researchers to more easily, efficiently, and accurately extract ecological data from large amounts of imagery data (Bakana et al., 2024). To make these methods more accessible, we published the data used in this study, the code used in the analysis, and the trained machine learning model.

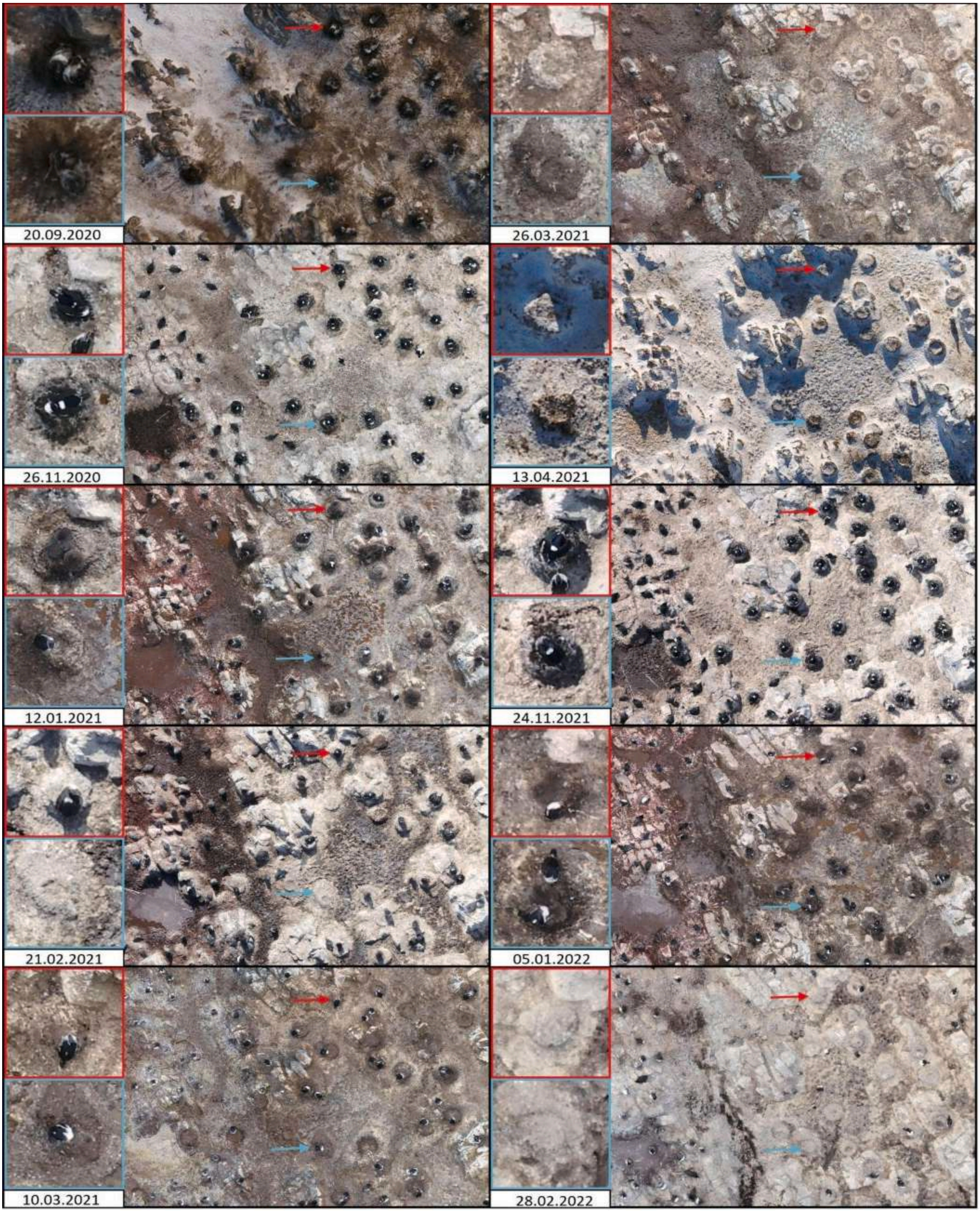

Fig. 4. Antarctic shag nests and variation in the colony substrate at different times during the phenological cycle. Red and blue arrows indicate the same nest on 10 different dates. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 2
Mean F1 scores obtained per dataset split, with standard deviations. The very high scores obtained by the model on the training and validation data were also reproduced on the test data. Details regarding the scores for individual orthomosaics can be found in Table S2.

Dataset split	Training	Validation	Test		
			test-over_time	test-over_location	test-over_source
F1 score	0.998 ± 0.003	0.963 ± 0.034	0.977 ± 0.019	0.995 ± 0.009	0.837 ± 0.145

We used F1 scores to evaluate model performance, and in all cases found that the models were able to predict counts with high accuracy. The performance of object detection models is often measured by a standard metric of mean average precision (often referred to as mAP or mAP[0.5:0.95]) (see Eikelboom et al., 2019; Moreni et al., 2023). We present such an evaluation of the model in the supplement (see Table S3), but due to its focus on the exact overlap between the ground truth and predicted bounding boxes, we do not believe this metric is informative for the current study. For example, one of the lowest mAP values (0.36) was achieved on Unnamed Island B, for which all the nests were identified and no false positives were produced by the model (see Fig. 5). This is because while the model learned to exactly highlight the area of the nests, the ground truth used for the assessment of the model was derived from fixed-size boxes centred on an approximate physical centre of the nests.

In addition to known colonies, our study also shows that RPAS are useful tools to detect new colonies of seabirds. The global population size of the Antarctic shag is estimated to be approximately 12,191 breeding pairs nesting in c. 175 breeding colonies (Fudala and Bialik, 2023) that are often located on rocky slopes (Harrison et al., 2021; Oosthuizen et al., 2020). The need for an evaluation of the status of the population was highlighted by Schrimpf et al. (2018); since then, two new colonies have been inventoried at Ryder Bay (Phillips et al., 2019) and at Cape Melville (Fudala and Bialik, 2023). Orthomosaics of eleven locations where Antarctic shags breed were used in this work. Three of these islands/islets are unnamed (designated Unnamed Islands A-C). The breeding group on Unnamed Island B was identified for the first time in this work. There are hundreds of such islets along the northern parts of Nelson and King George Islands (Pfeifer et al., 2021). Some of them were inventoried decades ago, and the presence of bird colonies needs to be verified, as shown by the example of the location of Smilets Point, where the Antarctic shag nested in 1986/1987 (Shuford and Spear 1988) and no nests were found at this location in 2016/2017 (Pfeifer et al., 2021). Coastal islets inaccessible to terrestrial observers may account for a significant and still severely underestimated share of the total breeding population locations. In the case of the entire range of the species, there could be thousands of such islands/islets.

6. Conclusions

In summary, our work has confirmed that RPAS are a useful tool for detecting new seabird colonies in Antarctica and, with machine learning image processing, can be used to monitor species that are otherwise difficult to count. Most of the orthomosaics used in the study contained

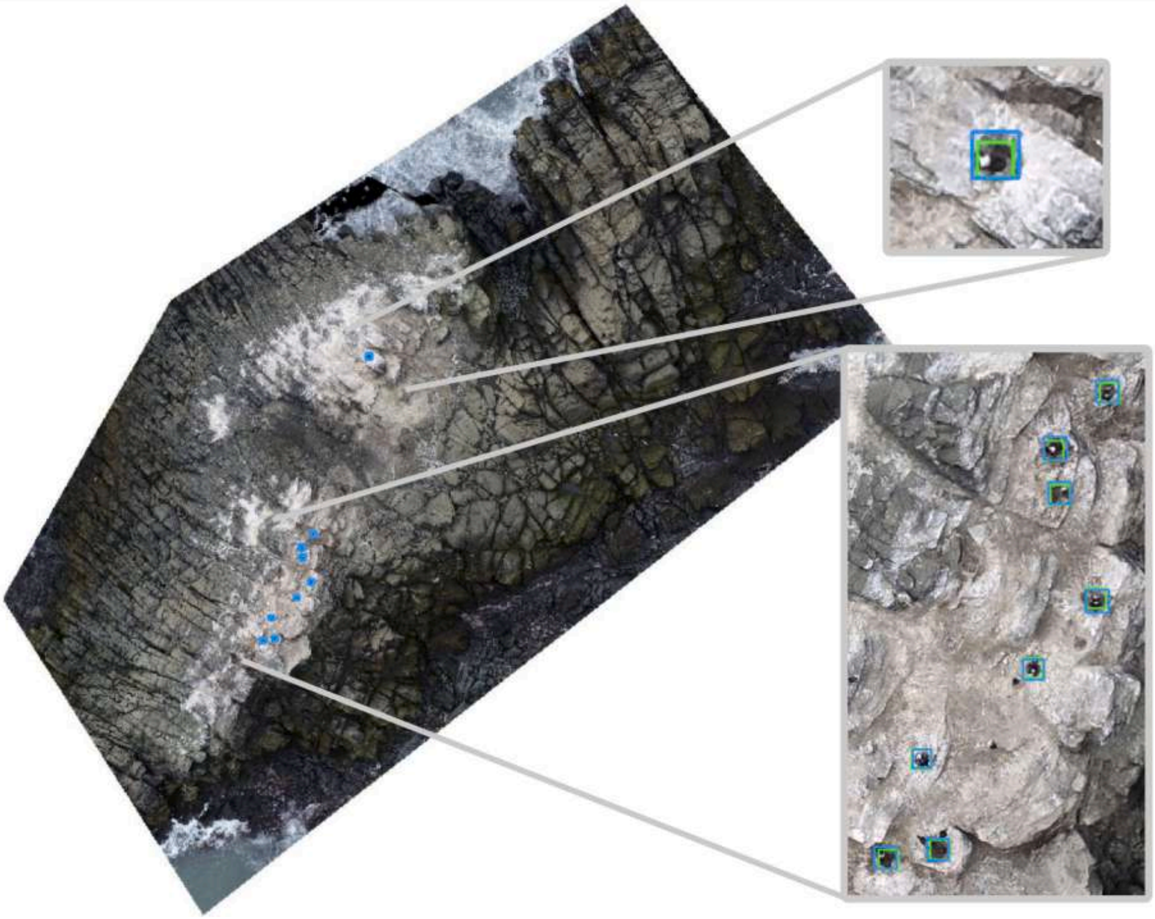

Fig. 5. Ground truth nests (the nine green squares) and the predictions made by the model (blue boxes) for a test-over_location orthomosaic of Unnamed Island B. Note that all the nests are correctly located (even the one that is spatially disjointed from the rest), with no false positive predictions. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

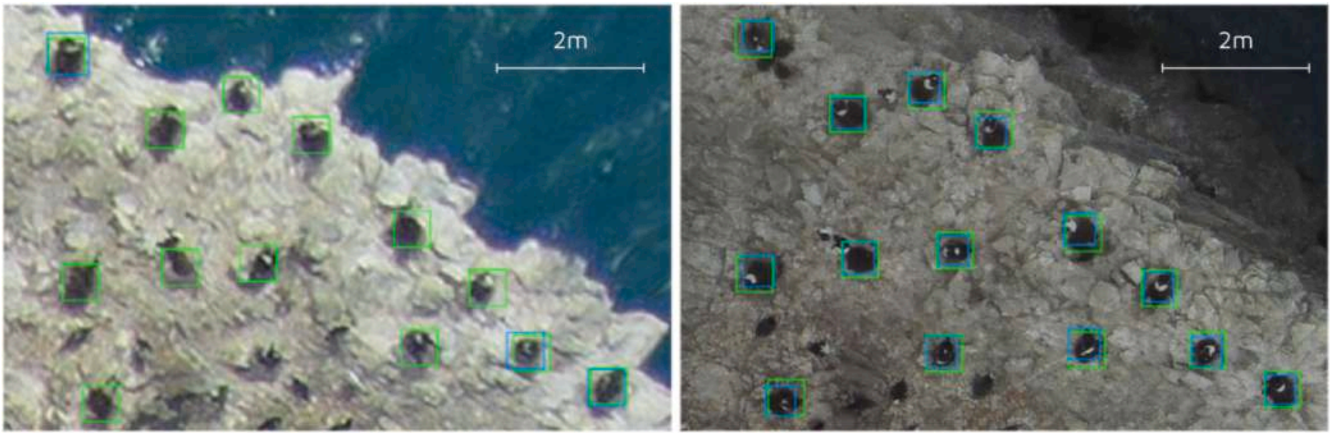

Fig. 6. Close-up image of an area of Rzepecki Island from the test-over_source (ground truth nests are marked in green, and predictions are marked in blue). The image in the left panel is blurred, and only three nests from this part of the island were identified by the model. The image in the right panel shows the same area, which was captured two days later. The improved sharpness of the image allows the model to pick up all the nests correctly (even though the resolution is low and the image is significantly darker than the well-lit training images). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

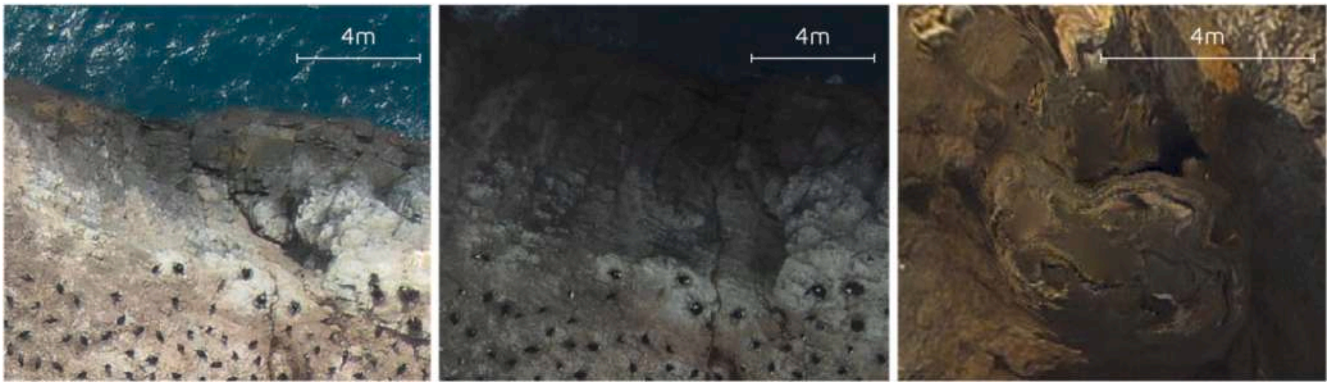

Fig. 7. Examples of shortcomings in the quality of the data used in the study. Left and centre show the same area of Rzepecki Island in orthomosaics taken at 2-day intervals; right shows a close-up of distortion on Kwarecki Island.

large (sometimes very large - reaching tens of thousands of individuals) penguin colonies. Therefore, these orthomosaics can be used for analogous studies of penguin populations. The challenge is to obtain accurate annotations for these much larger numbers of objects to be identified. We plan to address this in future research.

Authors' contributions statement

KF and RJB conceptualised the research idea, collected parts of the data and prepared annotations. AC, JŚ, JK and PPS were involved in the early stages of the machine learning development, while AC and JŚ completed the training and analysed the results. CP and WCO provided parts of the data. KF, RJB and JŚ led the writing of the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

CRediT authorship contribution statement

Andrew Cusick: Writing – review & editing, Validation, Software, Methodology, Investigation, Formal analysis, Data curation. **Katarzyna Fudala:** Writing – original draft, Visualization, Resources, Investigation, Formal analysis, Conceptualization. **Piotr Pasza Storożenko:** Writing – review & editing, Software, Methodology, Investigation, Formal analysis, Data curation. **Jędrzej Świeżewski:** Writing – original draft, Visualization, Validation, Software, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation. **Joanna Kaleta:** Writing – review & editing, Visualization, Data curation. **W. Chris Oosthuizen:** Writing – review & editing, Resources. **Christian Pfeifer:** Writing – review & editing, Resources. **Robert Józef Bialik:** Writing – original draft, Supervision, Project administration, Investigation, Funding acquisition, Formal analysis, Conceptualization.

Declaration of competing interest

None.

Data availability

The code, data and model used to obtain the presented results can be found here: https://github.com/Appsilon/Antarctic-nests

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecoinf.2024.102707.

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SUPPLEMENT

Table S1. Detailed view into the data used in the study

Location and date of capture	resolution [mm/px]	width [m]	length [m]	location (lon)	location (lat)
Shag Rock 2019-11-26	6	62	43	-62.18448	-58.29773
Shag Rock 2019-12-20	6	68	47	-62.18446	-58.29781
Shag Rock 2020-01-30	6	69	50	-62.18444	-58.29784
Shag Rock 2020-02-22	6	60	43	-62.18447	-58.29769
Shag Rock 2020-09-20	9	134	119	-62.18413	-58.29853
Shag Rock 2020-10-08	6	95	64	-62.18433	-58.29824
Shag Rock 2020-11-15	6	67	46	-62.18445	-58.29781
Shag Rock 2020-12-19	7	66	49	-62.18445	-58.29777
Shag Rock 2020-12-26	7	73	56	-62.18441	-58.29789
Shag Rock 2021-01-12	6	114	88	-62.1842	-58.29803
Shag Rock 2021-01-28	6	72	58	-62.18439	-58.29789
Shag Rock 2021-02-04	6	67	45	-62.18447	-58.29781
Shag Rock 2021-03-26	6	74	59	-62.1844	-58.29792
Shag Rock 2021-04-13	6	70	45	-62.18445	-58.29785
Shag Rock 2020-01-10	7	118	90	-62.18415	-58.29802
Shag Rock 2020-03-17	6	63	44	-62.18446	-58.29773
Shag Rock 2020-11-26	6	70	52	-62.18442	-58.29784
Shag Rock 2021-02-21	7	71	49	-62.18444	-58.29786
Shag Rock 2021-03-10	6	75	54	-62.18442	-58.29794
Shag Rock 2021-09-30	6	65	45	-62.18445	-58.29777
Shag Rock 2021-11-03	7	65	44	-62.18447	-58.29776
Shag Rock 2021-11-24	7	84	63	-62.18438	-58.29799
Shag Rock 2021-12-06	6	50	39	-62.18449	-58.2975
Shag Rock 2021-12-30	6	54	38	-62.18449	-58.29758
Shag Rock 2022-01-05	7	78	76	-62.18425	-58.29799
Shag Rock 2022-01-12	6	68	47	-62.18445	-58.29782
Shag Rock 2022-01-23	6	73	52	-62.18444	-58.29791
Shag Rock 2022-02-09	7	67	46	-62.18445	-58.29781
Shag Rock 2022-02-18	6	64	45	-62.18445	-58.29779

Shag Rock 2022-02-28	6	68	45	-62.18446	-58.29782
Shag Rock 2022-02-28	9	68	46	-62.18445	-58.29782
Cape Melville 2022-12-04	20	336	207	-62.02162	-57.5832
Cape Melville 2022-12-12	18	401	257	-62.02157	-57.58436
Turret Point 2020-12-31	14	55	59	-62.08852	-57.95059
Turret Point 2021-11-23	11	55	59	-62.08852	-57.95059
Turret Point 2022-01-06	11	55	59	-62.08852	-57.95059
Shag Rock 2022-09-24	6	67	47	-62.18444	-58.29779
Shag Rock 2022-11-15	6	67	47	-62.18445	-58.29779
Turret Point 2022-10-29	10	57	62	-62.08849	-57.9506
Turret Point 2022-12-12	11	76	76	-62.08842	-57.95083
Cape Melville 2023-11-11	20	397	261	-62.02162	-57.5832
Unnamed Island B 2022-12-26	12	101	101	-62.2451	-58.49369
Unnamed Island A 2022-12-12	12	42	31	-62.00157	-57.60608
Trowbridge Island 2022-12-12	12	29	47	-61.99394	-57.63565
Fregata Island 2016-12-11	20	279	178	-62.22902	-59.10225
Rzepecki Island 2016-12-28	20	139	104	-62.09285	-58.84302
Nelson Island 2018-12-27	25	143	102	-62.2896	-59.2315
Rzepecki Island 2016-12-30	20	139	104	-62.09285	-58.84302
Kwarecki Island 2016-12-28	34	182	136	-62.11832	-58.8915
Unnamed Island C 2016-12-25	15	157	113	-62.19369	-59.05515

Table S2. Detailed performance of the model

Location and date of capture	Split	F1 score	mAP[0.5:0.95]	n_nests	n_preds	TP	FP	FN
Cape Melville 2022-12-04	train	0.997	0.786	459	460	458	2	1
Cape Melville 2022-12-12	train	0.998	0.766	459	461	459	2	0
Shag Rock 2019-11-26	train	1	0.8	69	69	69	0	0
Shag Rock 2019-12-20	train	1	0.746	62	62	62	0	0
Shag Rock 2020-01-30	train	0.992	0.703	59	60	59	1	0
Shag Rock 2020-02-22	train	1	0.694	60	60	60	0	0
Shag Rock 2020-09-20	train	0.991	0.762	54	55	54	1	0
Shag Rock 2020-10-08	train	1	0.745	57	57	57	0	0
Shag Rock 2020-11-15	train	1	0.788	74	74	74	0	0
Shag Rock 2020-12-19	train	1	0.779	71	71	71	0	0
Shag Rock 2020-12-26	train	1	0.764	71	71	71	0	0
Shag Rock 2021-01-12	train	0.993	0.745	70	71	70	1	0
Shag Rock 2021-01-28	train	1	0.735	66	66	66	0	0
Shag Rock 2021-02-04	train	0.992	0.746	62	63	62	1	0
Shag Rock 2021-03-26	train	1	0.791	65	65	65	0	0
Shag Rock 2021-04-13	train	1	0.768	65	65	65	0	0
Shag Rock 2020-01-10	train	1	0.763	58	58	58	0	0
Shag Rock 2020-03-17	train	1	0.7	59	59	59	0	0
Shag Rock 2020-11-26	train	1	0.807	75	75	75	0	0
Shag Rock 2021-02-21	train	1	0.739	68	68	68	0	0
Shag Rock 2021-03-10	train	1	0.764	72	72	72	0	0
Shag Rock 2021-09-30	train	1	0.772	79	79	79	0	0
Shag Rock 2021-11-03	train	0.994	0.768	81	82	81	1	0
Shag Rock 2021-11-24	train	1	0.771	80	80	80	0	0
Shag Rock 2021-12-06	train	1	0.776	80	80	80	0	0
Shag Rock 2021-12-30	train	1	0.762	77	77	77	0	0
Shag Rock 2022-01-05	train	1	0.758	75	75	75	0	0
Shag Rock 2022-01-12	train	1	0.729	76	76	76	0	0
Shag Rock 2022-01-23	train	1	0.731	77	77	77	0	0
Shag Rock 2022-02-09	train	1	0.722	69	69	69	0	0
Shag Rock 2022-02-18	train	1	0.722	74	74	74	0	0

Shag Rock 2022-02-28	train	0.993	0.714	69	68	68	0	1
Shag Rock 2022-02-28	train	1	0.77	69	69	69	0	0
Turret Point 2020-12-31	valid	0.955	0.635	70	64	64	0	6
Turret Point 2021-11-23	valid	1	0.723	67	67	67	0	0
Turret Point 2022-01-06	valid	0.934	0.472	63	59	57	2	6
Shag Rock 2022-09-24	test-over_time	0.993	0.553	73	74	73	1	0
Shag Rock 2022-11-15	test-over_time	0.981	0.513	79	82	79	3	0
Turret Point 2022-10-29	test-over_time	0.991	0.642	58	59	58	1	0
Turret Point 2022-12-12	test-over_time	0.974	0.505	59	56	56	0	3
Cape Melville 2023-11-11	test-over_time	0.946	0.312	478	482	454	28	24
Unnamed Island B 2022-12-26	test-over_location	1	0.36	9	9	9	0	0
Unnamed Island A 2022-12-12	test-over_location	0.985	0.549	32	33	32	1	0
Trowbridge Island 2022-12-12	test-over_location	1	0.467	7	7	7	0	0
Nelson Island 2018-12-27	test-over_source	0.949	0.485	72	65	65	0	7
Fregata Island 2016-12-20	test-over_source	0.65	0.189	27	13	13	0	14
Kwarecki Island 2016-12-28	test-over_source	0.909	0.211	6	5	5	0	1
Rzepecki Island 2016-12-30	test-over_source	0.967	0.332	62	58	58	0	4
Unnamed Island C 2016-12-25	test-over_source	0.842	0.317	11	8	8	0	3
Fregata Island 2016-12-11	test-over_source	0.926	0.328	29	25	25	0	4
Rzepecki Island 2016-12-28	test-over_source	0.615	0.184	63	28	28	0	35

Table S3. A summary of the mean average precision mAP[0.5:0.95] scores obtained on different parts of the data.

Dataset split	Training	Validation	Test		
			test-over_time	test-over_location	test-over_source
mAP[0.5:0.95] score	0.754±0.029	0.610±0.127	0.505±0.121	0.459±0.095	0.292±0.108

The mean average precision is a standard metric (often referred to as the mAP or mAP[0.5:0.95]) used to assess the performance of object detection models (Eikelboom et al. 2019, Moreni et al. 2023). mAP[0.5:0.95] means that the average precision AP is calculated for ten intersection over union (IoU) thresholds, from 0.5 to 0.95 with a step of 0.05, and then averaged over the number of classes (Moreni et al. 2023). The IoU is a number from 0 to 1 that specifies the amount of overlap between the predicted and ground truth bounding boxes. In the present study, the mAP score was not an optimal metric. This is mainly due to the spatial precision of the available annotations; while they correctly indicate the presence of a nest in most cases, they often do not exactly match the shape of the nest (this is especially true for the data captured in 2016 and is reflected in the significantly lower mAP scores). For completeness, we present a summary table with mAP scores (Table S3) and detailed score for each orthomosaic is presented separately (Table S2).