

A preliminary evaluation of the evidence supporting fishery-driven localised depletion effects on the performance and demographic trends of pygoscelid penguins in Subarea 48.1

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Abstract

Two independent lines of evidence have been presented to the working groups and SC-CAMLR that claim to demonstrate that fishery-driven localised depletion of krill around pygoscelid penguin colonies has had a deleterious effect on their performance traits and demographic trends, that are equivalent to the impacts of climate variation. One study utilises 30 years of penguin foraging and reproductive performance measurements collected at two colonies in the South Shetland Islands while the other uses demographic rate changes derived from a comprehensive dataset of penguin population count data across Subarea 48.1 matched against acoustic measurements of krill biomass and krill catches at the gSSMU scale [10]. The second uses estimated population trajectories across a wide range of penguin breeding colonies alongside krill catches within 30km [6]. Both studies then explore the synergistic relationships to measurements of broad-scale climactic variation (El Nino Southern Oscillation; ENSO, and the Southern Annular Mode; SAM). Herein we provide a preliminary assessment of the efficacy of both approaches in drawing conclusions, that are now being used at the Commission level, as representing sound scientific advice. We demonstrate that several underlying assumptions in Watters et al. 2020 are contrary to the published scientific literature, and when the model syntax is re-written to reflect this, predicted penguin performance against long term expected means are substantially different to those presented to CCAMLR. *STUFF ON KRUGER* needed here. While our preliminary assessment focuses on potential issues, future work will centre on considering competitive interactions both at appropriate time and space scales between the fishery as well as between a range of krill dependent predators beyond just pygoscelid penguins.

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Introduction

Concerns over the potential impact of localised depletion of krill through concentrated fishing effort on krill-dependent predators has been a topic of debate within SC-CAMLR and its Working Groups for many years (REF). Recently, two studies have been presented that suggest that local harvesting rates can impact predator performance to the same degree as poor environmental conditions [10], and that synergistic impacts on predators are evident when poor climactic conditions are coupled with locally high harvest rates [6].

While both studies attempt to tackle the same overall problem, they do so using very different methodologies. Watters et al. [10] exploit a considerable dataset; a substantial multi-species time series of *a large number of* penguin performance indices (including those collected under CEMP) collected over three decades at two sites (Cape Shireff on Livingstone Island and Copacabana on King George Island, South Shetland Islands; Figure 1) and more than a decade of summer acoustic *krill* surveys that cover the at-sea distributions of Chinstrap, gentoo and Adélie penguins. Drawing in monthly krill catch statistics from the C1 Catch and Effort dataset and climactic data (Oceanic Niño Index; ONI), the authors use a hierarchical analysis of variance approach to estimate the variance in performance indices as a function of Local Krill Biomass (LKB), Local Harvesting Rates (LHR; the ratio of krill catch to LKB) and ONI. In contrast, Krüger et al. [6] utilise a broader range of penguin colonies across the same three species throughout the Antarctic Peninsula area, in combination with their respective abundance survey estimates (*number of occupied nests*) from an open-source database (www.penguinmap.org). The authors calculate population trends for appropriate sites, and *using use* the CCAMLR C1 Catch and Effort data to extract annual catch values within a 30km radius of each colony. Finally, Krüger et al. [6] use the *trajectory of the trend (positive; increase versus negative; decrease) sign of the difference in number of nests between annual surveys* as a response in a binomial generalised linear mixed *effects* model using the *summed accumulated* annual catch, *a lagged trend of the and mean wintertime Southern Annular Mode (SAM) and the SOMETHING ON ONI ????* to determine the relative contributions of each predictor and their interactive effects on population abundance trends. Both studies draw similar conclusions; that local harvesting levels of krill impact predators, and the degree of impact can either be similar to that of poor environmental conditions or have a synergistic impact when high local harvesting coincides with poor conditions.

These conclusions have been propagated into Commission documentation supporting the reformulation of the D1MPA proposal (CCAMLR-39/BG/02) as well as into Commission discussions (CCAMLR-39, Para 5.48 & Para 5.51). However, while the two studies have moved from Working Papers of EMM into the realms of the peer-reviewed literature, *we have there are* some areas of concern *about regarding* the structuring of these studies that we *feel warrant raising think deserve some extra attention. Our preliminary review raises Some of these* concerns *are unique to each study but also*

while others are common across both, and we structure our paper accordingly. Firstly, we review Watters et al. [10] and Krüger et al. [6] through the lens of some of the ecological assumptions made versus the available evidence pertaining to them. Within the constraints of the data and analytical methods that are available from the studies, we also quantify how rationalising these assumptions to the evidence available impacts on the conclusions drawn. We then highlight some overarching concerns applicable to both papers.

Watters et al. [10] / WG-EMM 2019/11

A key goal for the paper is to highlight the mismatch between the areal scales of fisheries management and ecological *the* interactions between fishing extractions and dependent predators. To do this, the authors create two strata aligned with groups of SSMU (gSSMU); gSSMU #1 including those SSMU inside the Bransfield Strait (APBSE and APBSW and gSSMU #2 incorporating SSMU north of the South Shetlands, including Elephant Island (APDPE, APDPW and APEI) represented in Figure 1. These gSSMU cover 15, 500nm² and 20, 600nm², respectively, and are used to characterise both krill biomass and harvesting rates that are “local” to the penguin colonies for which performance data are used. The reasoning behind scaling to gSSMU are linked to the foraging behaviour of the penguins for which performance data area available i.e. breeding, adult pygoscelids. The authors cite Hinke et al. [4] as the evidence supporting usage of the two gSSMU as appropriate strata.

Pygoscelid penguins exhibit staggered breeding, with Adélie commencing first, followed by chinstraps then gentoos [1]. Adélie penguins are the first to fledge their chicks and thus cease to be centrally foraging, typically departing mid-February for their moulting grounds on the sea ice. Chinstrap penguins depart for a pre-moult foraging trip towards the end of February and return to land in order to moult, before departing again for their overwinter trip (Hinke et al. [3]; Hinke et al. [5]; Figure 2). Conversely, Gentoo penguins appear to remain in close proximity to their breeding colonies overwinter .

We use the Argos-CLS PTT telemetry data provided by the supporting studies to characterise the actual at-sea habitat used, in the context of the relative stage of breeding for each species. For each species, we refrain from undertaking extensive state-space modelling of location errors and merely exclude locations with a “Z” error class, then calculated the 99% Minimum Convex Polygon (home range) and their associated areas in nm². For chinstrap penguins at Cape Shireff, this equated to a home range area of ~4,782nm², or only 23% of the gSSMU to which their performance metrics are indexed against [10]. For the same species at Copacobana the 99% MCP home range is 2,905nm², or ~19% of gSSMU 1 in the Bransfield Strait. Similarly for Adélie penguins, the breeding foraging range occupied 1,139nm² or only ~7% of the area of gSSMU #1. After breeding, available overwinter PTT telemetry and light geolocating data on chinstrap and Adélie penguins suggests a wide dispersal westwards into the Pacific sector of the Southern Ocean, and eastwards into the Weddell Sea and Atlantic sectors, with a relatively small proportion of chinstraps from

the study sites remaining within 500km of their breeding colonies [5]. Yet despite the evidence supporting widespread post-breeding migration of both Adélie and Chinstrap penguins, the model used by Watters et al. [10] constrains both species from Copacabana to gSSMU 1 and Chinstraps from Cape Shireff to gSSMU 2 over winter (Watters et al. [10]; Supplementary Material 1 & 2, code lines 258 to 259). This has the effect of constraining the variability in performance indices from these species to LHR, LKB and ONI over winter in areas where the species has a demonstrated tendency to migrate away from. This is particularly important given that the fishery can now be characterised with a late autumn/early winter start which places a seasonal element on LHR towards increased values in the winter (Figure 4).

Our preliminary review thus far raises two areas of concern. Firstly, that the scales at which “local” predictors are summarised are in some cases five times larger than the habitat exploited by the penguins monitored. Secondly, that the known overwinter migratory behaviour of Adélie and Chinstrap penguins are poorly reflected in the model formulation. To demonstrate the impact that these ecological assumptions have on the model output, we rerun the model of Watters et al. [10] with modified code. To avoid an overly burdensome paper, we shortly summarise those code changes here, and if requested during the meeting we are happy to include the rmarkdown version of this paper with the modified code in place, or submit the modifications to the meeting in some other format.

We also note an additional coding error that may influence how the original (i.e. unmodified) results are interpreted. In summarising the model outputs into boxplots, the code relating to developing Figure 2 (Supplementary Material 1, lines 661-665) seemingly classifies the “Worst Case” ($-0.5^{\circ}\text{C} < \text{ONI} < 0.5^{\circ}\text{C}$; $\text{LKB} > 1 \text{ Mt}$; and $\text{LHR} \geq 0.1$) using column 36 from the output dataframe, which actually reflects “ $\text{ONI} > 0.5^{\circ}\text{C}$; $\text{LKB} > 1 \text{ Mt}$; and $\text{LHR} \geq 0.1$ ” - that is, in the “Worst Case”, ONI is in its “warm” phase as opposed to “neutral”. We thus relabel the output boxplot axes labelling to reflect the conditions representing “worst case” accordingly.

modifications

1. We scale the gSSMU LKB to the SSMU that the summer tracking data indicate penguins occupied. For example, we scale LKB for Cape Shireff chinstrap penguins solely to ADPDW by multiplying the gSSMU LKB by the areal ratio of ADPDW/gSSMU #2. We then select the corresponding SSMU catch values provided in Watters et al. [10] to estimate SSMU-scale LHR.
2. We remove Adélie and Chinstrap penguins from the model formulation over winter; that is, we attribute each species as “NA” during winter, to account for dispersal after breeding.
3. The authors place LKB/LHR values in March into the “summer” period. However fishing effort over the period that performance indices are available is not uniform over the thirty year period, with catch over the

preceding decade showing a nonlinear increase from the middle of March and three years where catch rates increased rapidly from the beginning of the month (Figure 4). Given the highly variable rates of catch throughout the study period, we run scenarios that classify March as either summer or winter to reflect the linkage between March and the breeding state of penguins i.e. Adélie and Chinstrap penguins have either migrated out of the area or have ceased to be centrally foraging species by March.

We run a total of six permutations of the modifications, grouped into three pairs, as follows:

1. Performance indices from all species, using estimates of LKB and LHR at the SSMU level, considering March either in a) summer or b) winter. (Areal scaling down of gSSMU LKB and using SSMU scale C1 monthly aggregated catch data).
2. Performance indices from Adélie and Chinstrap penguins, matched at the SSMU level with March either in a) summer or b) winter
3. Performance indices from Gentoo penguins matched at the SSMU level with March either in a) summer or b) winter.

We present the outputs both in the same boxplot format as Figure 2 in the original manuscript, and as individual cases grouped and colour-coded as ONI “warm / ≥ 0.5 ” (red), ONI “neutral / $-0.5 <> +0.5$ ” (white) and ONI “cold / < -0.5 ” (blue). *These symbols look a bit weird. What did you intend to write?*

results

Krüger et al. [6] / WG-EMM 2019/10

The key objective of the paper by Krüger et al. [6] is to examine the potential synergistic effects of climate change and increased fishing activity in recent decades on the breeding performance of chinstrap and gentoo penguins. The authors make the implicit assumption that there has been a general decrease in krill density in response to climate change, although this is still a topic of debate and is not supported by recent large-scale surveys in the Scotia Sea (REF 2019 Krill survey report).

Perhaps describe the model already here, to set the scene for the data input issues?

To address this topic, the authors use data on the number of occupied nests (breeding pairs) counted at a large number of sites throughout the Antarctic Peninsula between 1980 and 2017. These data are available from the Mapping Application for Penguin Populations and Projected Dynamics (MAPPPD) data archive (REF to website and Humphries et al. 2017). Krüger et al. [6]

include count data on occupied nests based on surveys carried out in November or December, reflecting the early breeding season. They further subset the data to include only colonies for which at least 2 survey estimates are available throughout the 38-year period under consideration (i.e. 1980-2017). These data are provided in the supplementary materials for the paper. Based on the raw counts, they calculate an index of temporal variation in population growth rate using:

$$\lambda_{std} = ((n_b/n_a)/years_{b-a} - 1$$

where n is the number of breeding pairs counted in Nov-Dec of a given year, b and the number of breeding pairs counted in the nearest previous year a in which a Nov-Dec survey was conducted, divided by the interval between surveys (i.e. $b - a$). While this index may be a robust index of population change, the authors then convert λ_{std} into a binary index, $bin\lambda_{std}$ that takes the value 1 for negative growth and 0 for positive growth. The rationale is that this value can be interpreted as the probability of population decline in response to catch and environmental change, but it also completely ignores how the absolute or relative change in population size may respond to fisheries and environmental change.

Another problem with this index is the interval between consecutive surveys. Based on the Krüger et al. [6] supplementary dataset, intervals between surveys exceeding one year are relatively common (intervals >1 year: 27% for Chinstraps and 48% for Gentoos; intervals >2 years: 17% for Chinstraps and 15% for Gentoos, Table 1). As we describe below, these larger intervals may represent a temporal mismatch problem, given the fact that response variables only represent conditions within one year prior to the breeding season.

As in the case of Watters et al. [10], Krüger et al. [6] also use CCAMLR C1 fisheries data to estimate fishing pressure, but in this case only hauls within a 30km radius are considered for each specific colony. This selection is based previous observations that foraging of pygoscelid penguins is more probable within 30 km of the colonies during the breeding season *refer to Warwick-Evans et al. 2018*. While they initially summarise these data within distinct time periods, reflecting different important stages of the penguin annual cycle, these are only used for illustrative purposes [Fig 3 in 6]. However, when modelling the effect of fisheries on population response, the authors use accumulated annual catches within these 30km areas, thereby making the assumption that penguin breeding performance is affected by resource availability in the immediate vicinity of their specific breeding colonies also during the non-breeding period in winter. While this might be valid for gentoo penguins which appear to remain close to the breeding colonies also outside the breeding season, it is questionable how appropriate this is for chinstraps that disperse much more widely during winter (see above). This spatio-temporal mismatch problem is further exacerbated in cases where intervals between consecutive breeding population surveys exceed one year.

The authors use monthly data on the Southern Annular Mode (SAM) index *refer to Kwok and Comiso 2002; Doddridge and Marshall 2017* to

represent environmental variability. Based on an observed 0-3 month lagged correlation between SAM and relevant local climate variables (fractional sea ice cover, open water sensible heat flux and sea level air pressure), the authors exclude SAM values for months coinciding with the breeding season.

To test statistically the effects of local fishing pressure and environmental conditions on penguin breeding performance, Krüger et al. [6] fit a binomial Generalized Linear Mixed Effects model of the form:

$$\text{bin}\lambda_{std} = \text{catch}_y * \text{SAM} + (1|\text{colony ID})$$

where catch_y is the accumulated krill catch within 30km of each colony during the year immediately leading up to the second survey in an interval (i.e. equivalent to year_b in the population growth rate equation above), SAM is the SAM index during the winter prior to the same survey (i.e. temporally overlapping with most of the catch data accumulation period). the term in brackets indicate a random intercept effect on colony. Unlike in the case of the Watters et al. [10] paper, scripts are not provided for the analyses done by Krüger et al. [6], and therefore we have not been able to recreate their analyses exactly and test various aspects of data input, underlying assumptions and alternative model formulations.

NEEDS WORKING ON - A LOT :) BULLET POINTS FROM THE GOOGLE DOC BELOW:

1. Contention that population trends (rate of growth/decline) are related to synergistic impacts of climate change and localized fishing pressure
2. Autumn/winter SAM is an appropriate proxy for krill abundance (i.e no consideration of interactive effects between SAM and ONI)
3. The validity of linear interpolation across multiple years (>1) has not been analysed.
4. 30km buffer around colonies to extract fishing effort is appropriate over winter
5. Using population rates calculated between survey efforts sometimes several years apart are appropriate
6. Assumption that calculated population rates directly reflect mortalities
This one I don't understand. I don't think their assumption actually is that the rate reflects mortality, only that it reflects changes in the number of birds returning to breed (irrespective of if they're still alive or not).
7. No consideration of lagged recruitment (fledging to reproductive age) nor ability to detect lagged recruitment with irregular surveying effort / lack of banding

AND BELOW ARE POINTS FROM THE HTML WORKING DOCUMENT:

Problem 1: There's clearly some discrepancies in amounts of data here that we may need to approach the authors about.

Problem 2: How representative is this rate value for a specific year, in the cases where it has been calculated as a linear change over a period of several years between surveys?

Problem 3: Is it appropriate to use annually accumulated catch as an explanatory variable to explain number of breeding pairs observed in Nov-Dec?

Problem 4: In the case of gentoos, half of the sites never have positive catch rates within 30 km, while for chinstraps the situation is not so bad. How does this explain the model fits and conclusions of Kruger et al.?

Problem 5: If no other climate variables are included in the model, does their argument really make sense? Why exclude SAM during the breeding period?

Problem 6. According to the documentation ,this package only appears to have the capacity to fit a simple LMME, which does not allow for a binomial response

Discussion

Our preliminary review of the evidence supporting localised effects of fishing coupled with broad-scale climactic phenomena having an impact on the vital statistics of pygoscelid penguins (performance and demographic trends) are based on assumptions that potentially do not reflect current knowledge of penguin breeding phenology and movement.

DESCRIPTION OF x6 SCENARIOS RUN WITH STIG

Of greatest concern, however, is that the interpretation of model outputs from both approaches (either from the original studies or the modified parameters we describe) are under boundary conditions that we feel are not appropriate. Both approaches consider only the fishery and broad-scale climate phenomena as the only two causes of krill abundance variability at geographic scales relevant to penguins. Neither study considers, for example, the impact of rebounding baleen whale populations or migratory male Antarctic fur seals beyond brief mentioning. Both taxa have increased in abundance throughout the life of the krill fishery, and there are sufficient telemetry and distance sampling studies in the scientific literature to demonstrate the degree and significance of spatiotemporal overlap with breeding penguin populations (see Santora and Veit [8], Lowther et al. [7], and Oosthuizen et al., Johannessen et al. and Lowther et al. submitted to this meeting, and Figure 3 as examples). Importantly, the distribution of these and numerous other unconsidered competitors is not uniform in either space or time and their impact on local availability of krill is likely to be considerable.

Similarly, the utilisation of broad-scale climatological phenomena to characterise impacts at scales that predators are dependent upon is problematic.

The Amundsen Sea Low (ASL) is the dominant climate feature for the western Antarctic Peninsula. The El Niño Southern Oscillation (ENSO) modulates the ASL, with El Niño (La Niña) shallowing (deepening) its pressure, causing more northwesterly (southeasterly) winds and upwelling (restricted influx) of Circumpolar Deep Water onto the shelf. The Southern Annular Mode also influences the pressure of the ASL, with the current trend of negative SAM constructively (destructively) interfering with ASL when in phase with El Niño (La Niña) events (e.g. Clem et al. [2]). The result is a set of above-surface climate conditions that drive changes in water mass intrusion that are dependent on interactions between two climate processes. However the bathymetry of the Antarctic Peninsula is complex (particularly at scales that are important to centrally-foraging predators such as penguins) and the structuring of krill aggregations in time and space in the WAP have been linked to mesoscale circulation processes [9], which are unlikely to be uniformly affected by macroscale processes.

Our work into the future will progress along three lines, and we welcome any and all offers of collaboration into this work. Firstly, we will progress this debate into the scientific literature in order to ensure a balanced discussion occurs in that forum. Secondly, we will be examining in further detail some of the additional predictors used and their efficacy, the modelling frameworks into which they are brought, and how their incorporation influences the interpretation of the responses. Finally, we shall also be exploring alternative modelling approaches that reflect more of the physical and biological complexity of the system in question. In all cases, our goal is to ensure that the best available objective scientific evidence is presented to our environmental managers and, where appropriate, flag that disagreement exists. Our paper should be viewed in this light to generate constructive dialogue that addresses our common concern of the potential for localised fishing to impact dependent aspects of the ecosystem.

Citations

Tables

Table 1: Frequency distribution (percent) of intervals between consecutive breeding surveys for chinstrap and gentoo penguins used by Krüger et al. [6].

	Chinstrap	Gentoo
1 year	63.03	68.29
>1 year	26.75	48.28
>2 years	16.51	14.67
>3 years	8.95	9.3
>4 years	7	4.26
>5 years	3.64	3.28
>6 years	3.15	2.44
>7 years	2.31	0.98
>8 years	1.82	0.98
>9 years	1.82	0.49
>14 years	1.33	0.49
>15 years	1.33	0
>19 years	0.84	0

Figures

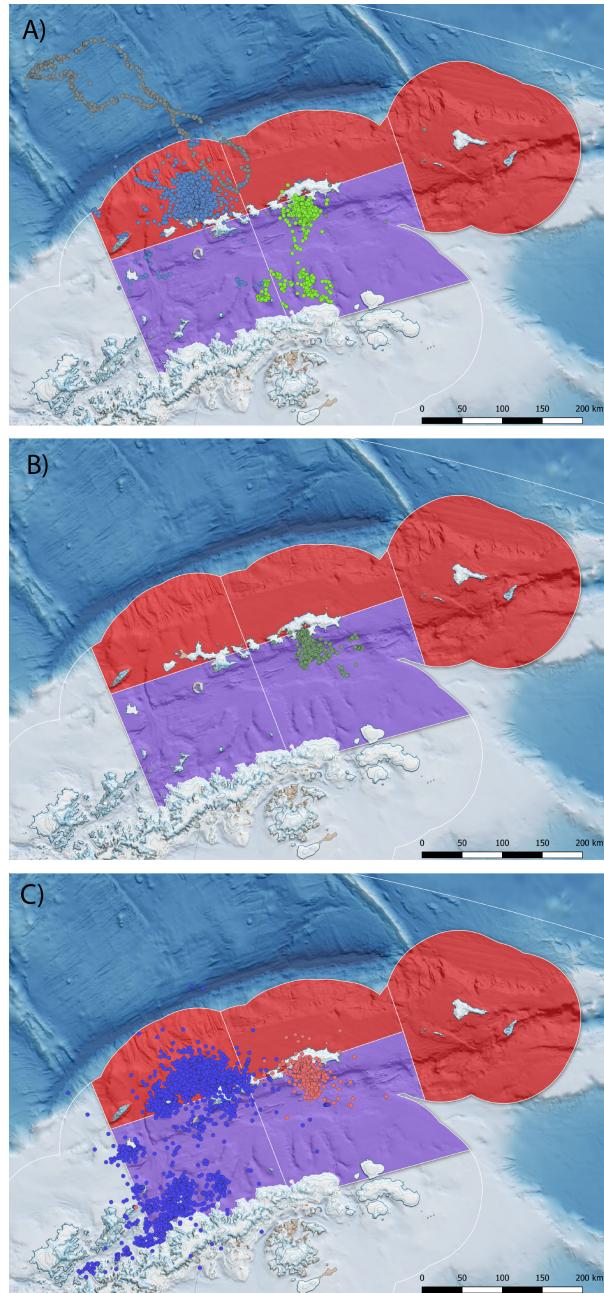


Figure 1: Penguin foraging behaviour derived from available ARGOS-CLS PTT data. A) Chinstrap penguins truncated at 10th March in line with known phenology (elongated grey track represents a single animal) B) Adélie penguins truncated to the end of January and C) Gentoo penguins until August, in line with known overwinter foraging behaviour. The SSMU are combined and coloured according to gSSMU (red; gSSMU 2, purple; gSSMU 1) with Chinstrap and Adélie penguin 99% MCP²home ranges occupying between 7-19% of the gSSMU to which they were assigned.

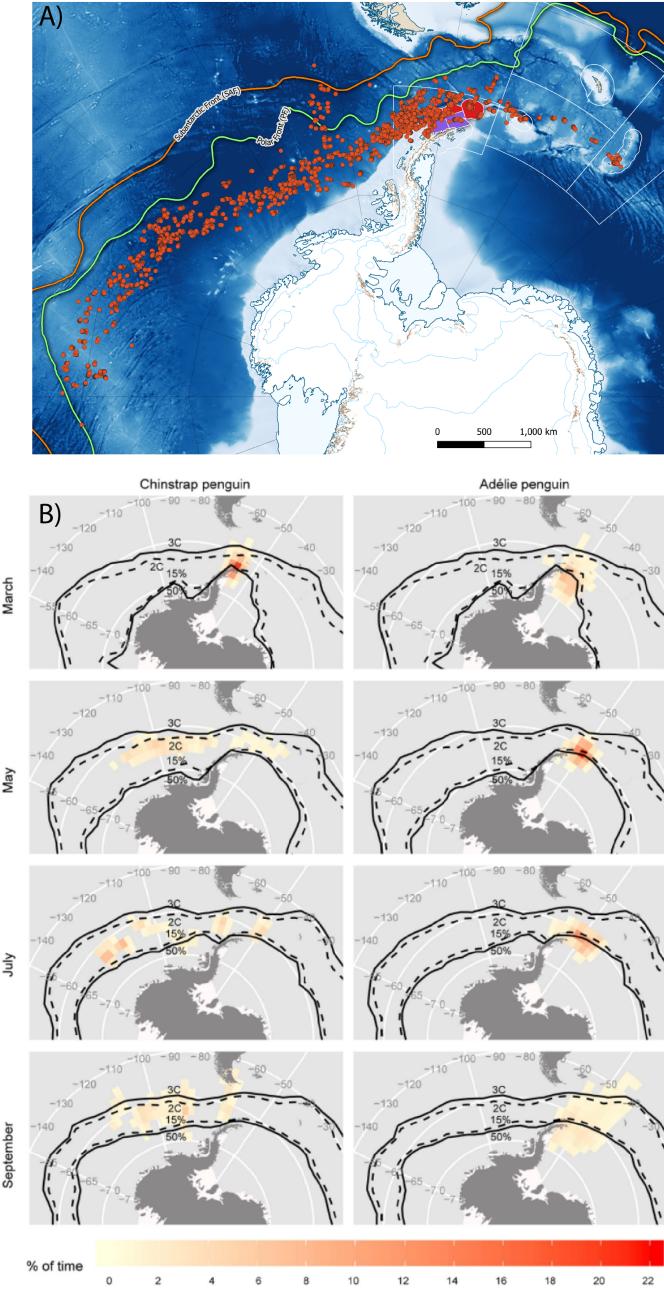


Figure 2: A) Distribution of overwinter movement for Chinstrap penguins, relative to the gSSMU's to which they were attributed, created from telemetry data available in @Hinke2019. B) Adélie and Chinstrap penguin movement recorded by light geolocators, highlighting the large longitudinal range both species disperse through at the end of breeding (taken from @Hinke2015). In the original model formulation by Watters et al. 2020, the performance indices for both species are matched to gSSMU-scale estimates of LKB and LHR, but macroscale levels of ONI variability.

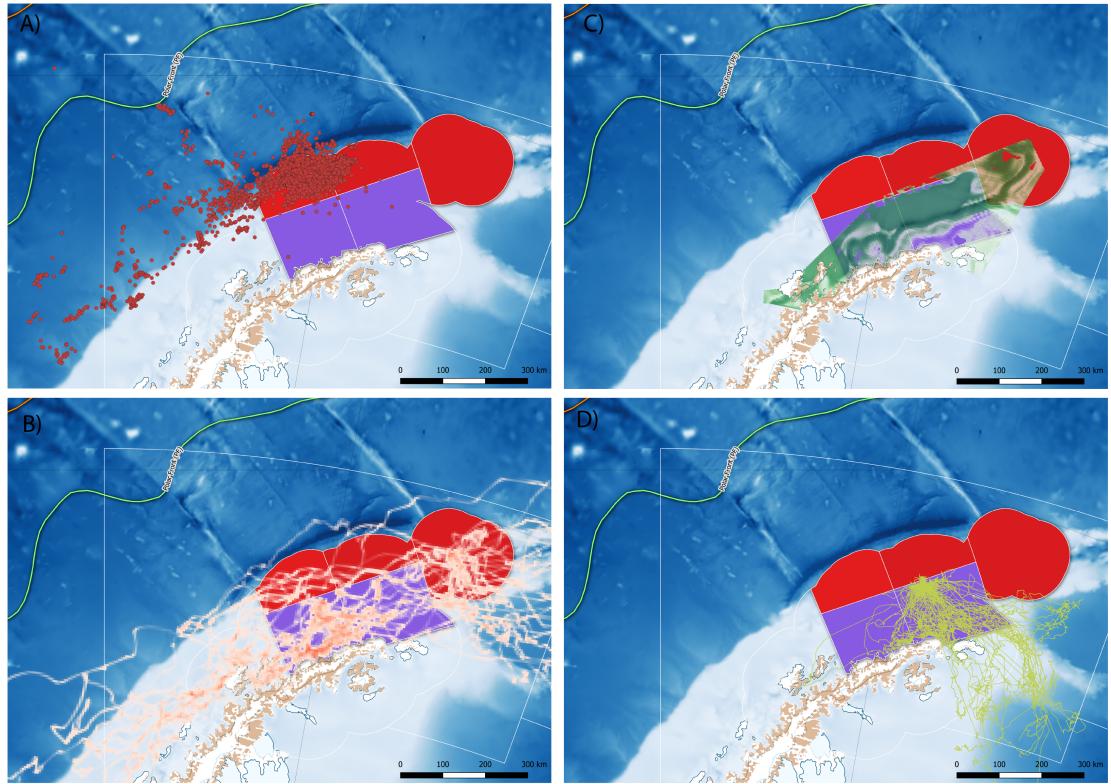


Figure 3: The summer distribution of foraging effort by A) adult female Antarctic fur seals (adapted from telemetry data available in Hinke et al. 2017), B) migratory adult male Antarctic fur seals (adapted from Lowther et al. 2020) C) humpback whales throughout December (adapted from Johannessen et al., this meeting) and D) nonbreeding adult Adélie penguins during the breeding season (adapted from data in Oosthuizen et al., this meeting). Potential effects of competitive overlap between pygoscelid penguins and other krill dependent predators, particularly those who have increased their abundance dramatically over the preceding 40 years, should be considered when attempting to interpret variance in penguin vital rates.

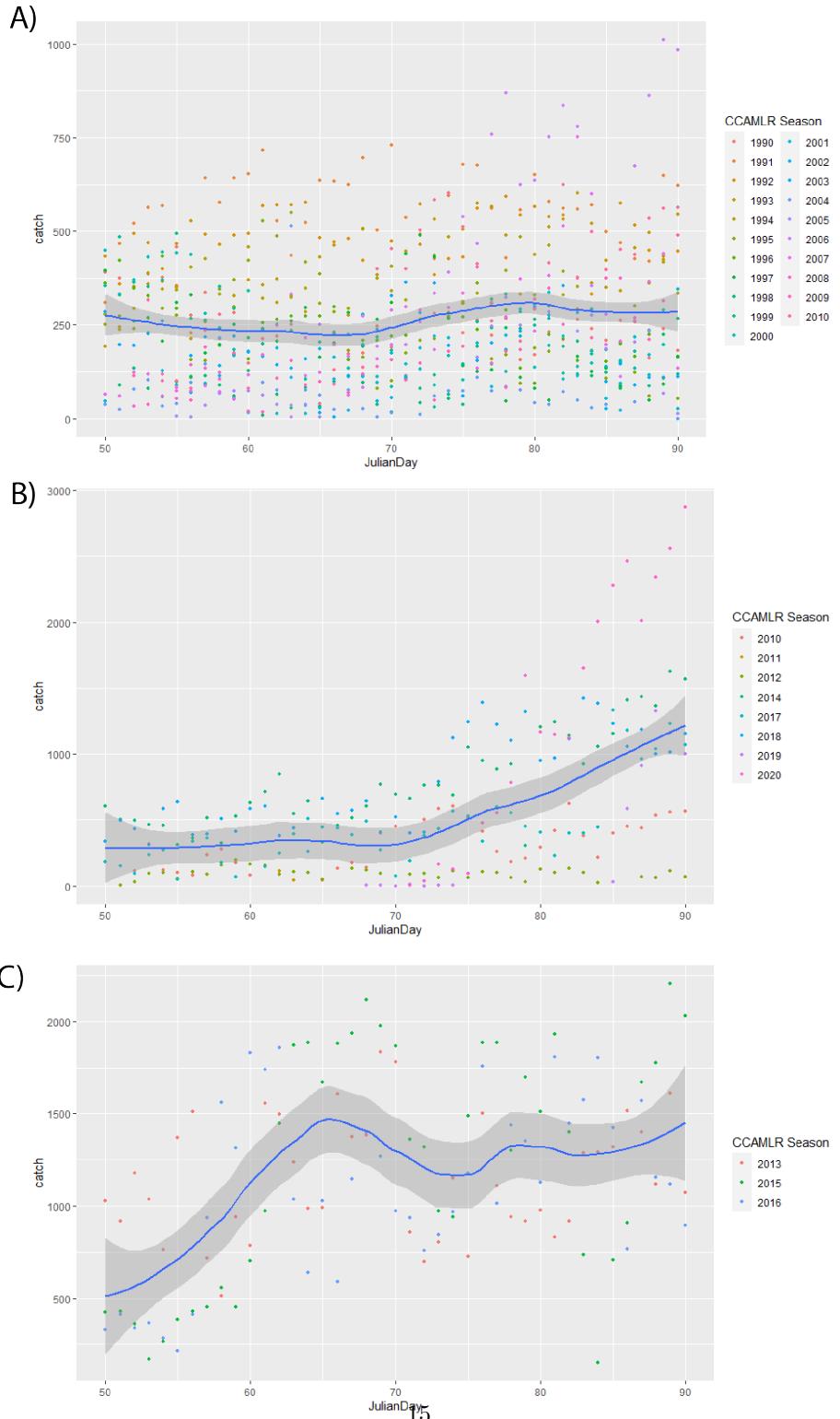


Figure 4: Daily accumulated catch (and fitted LOESS smooth curves) in Subarea 48.1 between 20th February and March 31st between A) 1990-2010 and B) 2010 to 2020. Catch in the Subarea was relatively consistent during the latter stages of penguin breeding at approximately 250tonnes/d. However, since 2010 there has been a tendency for the fishery to increase its effort in the Subarea, starting around the middle of March. C) During this latter period, the fishery increased effort earlier on three occasions (2013, 2015 and 2016), starting before the beginning of March.

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