Marine Characterisation Research Project (MCRP)

Work Package 6: Guillemot and Razorbill Colony Counts and Estimation of Breeding Productivity

Menter Môn-Marine Characterisation Research Project (MCRP)

MCRP-RSPB-DOC-0002 WP6 Final Report 2023

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RSPB

MCRP Document No.: MCRP-RSPB-DOC-0002	Status: Final	Version No.: 1.0	Date: 23/11/2023

Glossary

RSPB Royal Society for Protection of Birds

JNCC Joint Nature Conservancy Committee

SMP Seabird Monitoring Programme

EMMP Environmental Mitigation and Monitoring Plan

AON Apparently Occupied nest Site

GLM Generalised Linear Models HPAI

Highly Pathonogenic Avian Infuenza

Capability Statement

- To inform EMMP Monitoring Indicator-7 (local population effects) standard survey methods were used to estimate whole colony size at RSPB South Stack Reserve for breeding Guillemots and Razorbills, along with the numbers of individuals in each colony sector. We compared these to annual data collected since 1994 for South Stack and other colonies across North Wales.

- Estimates were 13,598 and 11,821 breeding Guillemot individuals and 1,825 and 1,880 breeding Razorbills in 2022 and 2023 respectively. Effects of counter identity were minimal with most variation in colony size estimates likely attributable to variation in colony attendance between the five counts prescribed by the standard monitoring method. The South Stack population has increased markedly since 1994 in line with most but not all other colonies with data in North Wales. This increase was not uniform across the South Stack colony, with the most marked increases for each species taking place in different sectors.

- Auks visited the colony ahead of the breeding season from January based on one full winter of counts, but colony attendance during breeding was highest in the months April-June, after which auks left the colony entirely as is typical for these species. Monitoring in the months following breeding across two years showed no colony attendance from August to December. An outbreak of highly pathogenic avian influenza (HPAI) occurred in June 2023 but coincided with both terrestrial and marine heat waves. These events appear to have had some impact on the seasonal pattern of colony attendance.

- Based on study ledges across three colony sectors, Guillemots productivity was estimated as 0.88 and 0.44 chicks fledged per apparently occupied nest site in 2022 and 2023 compared to 0.71 for the 1989-2019 Wales average. Our figure of 0.75 for Razorbills in 2022 was higher than the figure of 0.53 for Skomer Island for the 1993-2019 period but fell to 0.38 in 2023. The markedly lower productivity in 2023 seems primarily attributable to the environmental events in June, midway through the breeding season. Productivity varied between monitored colony sectors. Furthermore, values varied differently between years for Guillemots, for which one sector appeared minimally impacted by environmental events, an effect that was not significant for Razorbills. Given the contagious nature of HPAI, this could be explained by high levels of onshore physical segregation among species and among birds using different count sectors, providing support for HPAI as the dominant cause in low productivity in 2023.

- Ultimately, any negative population impact of the development on Guillemots and Razorbills will manifest as a decline population growth rate, which includes continuing to grow but at a reduced rate. Repeat monitoring of population size in particular represents a relatively cheap low-tech method of monitoring for such impact that is consistently repeatable. Future abundance monitoring plans could be informed by analyses of levels of statistical power to detect change in population growth rate, and we present an example approach to evaluate this using South Stack data that evaluates the frequency of monitoring (annual, 3 yearly or 5 yearly) and the impacts of different magnitudes of impact (falls in population growth rate of 1%, 3% and 6%). Importantly, future productivity monitoring should continue to occur in multiple count sectors to account for within colony variation, although we lack South Stack specific historical data to inform our ability to detect changes in this demographic rate. Population level impacts of events such as the HPAI outbreak at South Stack and other colonies remain unclear. They are important in the context of this baseline monitoring because, as an additional cause of growth rate reduction, they may influence the ease within which impacts specific to the tidal stream development can be detected.

- Two timelapse camera technologies were trialled. Off-the-shelf trail cameras set to timelapse mode worked reliably but had inadequate magnification for the cliff nest site topography at South Stack even with x60 lenses fitted by the manufacturer. A bespoke camera system based on a professional quality digital single lens reflex camera had greater but still inadequate magnification for informing productivity (constrained by weatherproof housing size). Given the inaccessibility of the majority of nesting areas, we conclude that productivity monitoring using handheld digital single lens reflex and large telephoto lens is the best technological aid, with the additional benefit of allowing multiple sites to be monitored using one camera ensuring estimates are representative of the whole site.

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1 Introduction

Royal Society for the Protection of Birds (RSPB) undertook estimates of colony size (number of individuals) and breeding productivity (number of chicks fledged per Apparently Occupied Nest Site(AON of two species of auk (Guillemots *Uria aalge* and Razorbills *Alca torda*) at South Stack, Anglesey, in 2022 and 2023. This was to establish baseline data for these key metrics of population health. Comparison with future data will inform impacts due to tidal stream power generation infrastructure (Monitoring Indicator 7 – local population effects, EMMP 2020). Although there is nothing specifically relating to local population effects among the Outline Monitoring Questions in the Environmental Mitigation and Monitoring Plan (EMMP), such effects could be a consequence of several of the disturbance issues raised by the 10 monitoring questions, highlighting the importance of effective population and demographic monitoring in assessing impacts of development. The colony counts continue a long-term annual monitoring programme that the RSPB undertakes at South Stack, alongside similar monitoring at other sites in north Wales (often by NRW) which contribute to the Joint Nature Conservancy Committee (JNCC) Seabird Monitoring Program (JNCC 2021). The installation of tidal stream power generation infrastructure effectively makes at South Stack a 'treatment' site in the experimental sense. Future evidence of change in the temporal abundance patterns at South Stack relative to those at 'control sites' elsewhere could be a component of assessing impacts on auks.

Tidal stream generation devices may have multiple potential impacts on marine birds when operating, which can occur separately or in combination, and potentially interact in complex ways with other environmental variables, such as tidal flow regimes and the behaviour of their fish prey. For example, collision risk may be a significant direct impact. The consequences of this include reduced survival of adult birds during breeding or, if they remain in the area, the non-breeding period, in addition to impacts on naive juvenile birds after fledging or when they visit the colony to prospect for nest sites. Potential effects, over and above direct impacts are of two kinds. First, disturbance, which acts by excluding birds from foraging habitat near the devices, representing net habitat loss. Second, impacts on prey fish, which may either avoid or be attracted to the devices, a response which may be tide dependent. The latter could interact with foraging birds to increase their collision risk (where they are not excluded by disturbance). Thus, impacts could be both positive or negative and vary among trophic levels and tide state. Such impacts remain poorly understood and their evaluation is essential if tidal stream power generation is to reduce carbon emissions without negatively impacting biodiversity.

Comprehensive monitoring is essential if any impacts on bird populations are to be detected. Changes in population metrics form part of this, and potentially enable the consequences of impacts to be understood at site, regional and national scales as long as study design is sufficiently powerful for anticipated effects to be detectable. Importantly, breeding abundance and productivity have well-developed field methods with minimum reliance on new technology. There were four objectives from the specification set out in the tender document:

1. Whole colony population estimate (Primary Core Colony Count). Baseline abundance of the breeding Guillemot and Razorbill populations of South Stack RSPB Reserve was estimated prior to tidal generation infrastructure installation.
2. Colony attendance through the breeding season (Option 1 Colony Counts). Seasonal variation in colony attendance was described to determine periods of highest bird activity.
3. Colony attendance outside breeding (Option 2 Colony Counts). Usage outside breeding will be described, during which time birds are not constrained by caring for eggs or chicks.
4. Breeding productivity. This was estimated for these two species at South Stack for the first time.

2 Methods

2.1 Objective 1. Whole colony population estimates

Monitoring of breeding seabirds has been carried out by staff at RSPB South Stack Nature Reserve for many decades (and multiple counter identities) based on counts from fixed viewpoints of named colony sectors to ensure high and standardised coverage of a high proportion of the entire breeding seabird colony. Although a small number of birds not visible from these points will have been missed, these are considered to represent a very small proportion of the total. In 2022 and 2023 sectors were counted by both South Stack RSPB reserve staff and RSPB Conservation Science staff which provided the opportunity to evaluate consistency of counts by different individuals – high consistency is a key requirement of evaluating change over time.

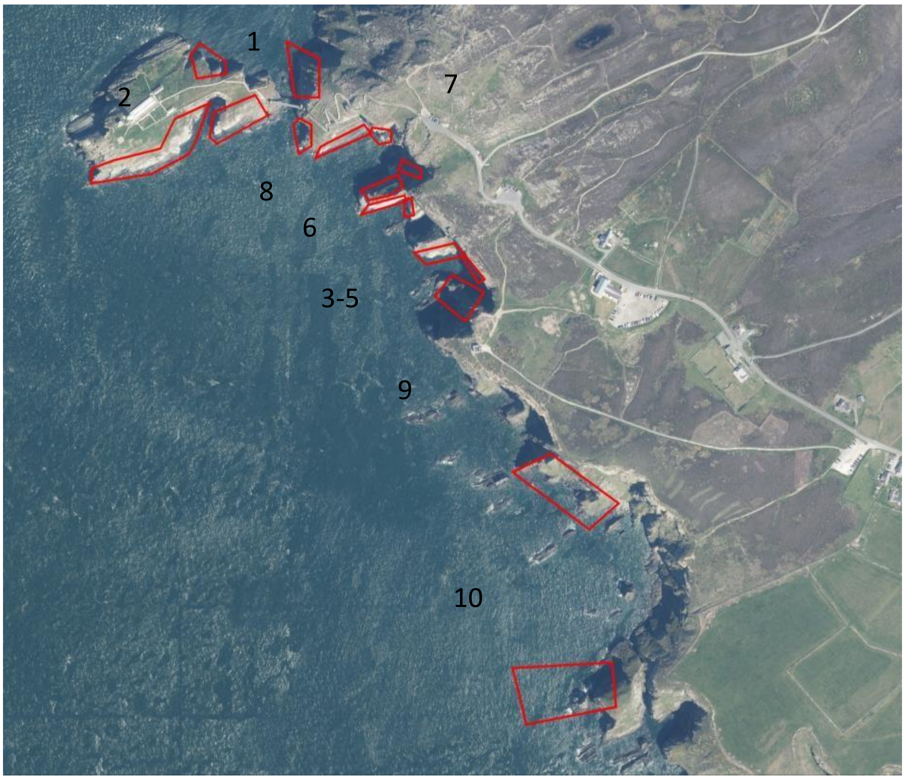
The recommended period for auk abundance monitoring is the first three weeks of June, coinciding with the incubation stage of the breeding season when pairs are closely tied to nest sites and numbers are relatively consistent (the widely used Whole Colony Census Method, Walsh et al. 1995), as used at RSPB South Stack. This requires five counts spaced four days apart and taking place in calm conditions with good visibility and lighting conditions (early morning at South Stack) (Table 2-1). On each survey date, counts were carried out from the same viewpoints (Figure 2-1, Figure 2-2) corresponding to the ten sectors and 18 sub-sectors into which the colony is divided for counting (Figure 6-1). Counts of individuals were averaged for each viewpoint across the five visits to generate estimates of population size for each sector, summing across sectors to provide an estimate of whole colony size. Combining these with historic data for the same sectors provides information about patterns of temporal change in abundance for individual count sectors and the wider colony. Furthermore, whole colony population estimates can be compared with data for other **Guillemot** and **Razorbill** colonies in North Wales using existing data collated by the Seabird Monitoring Program (SMP, JNCC 2021).

TABLE 2-1: DATA COLLECTION ACROSS SOUTH STACK BREEDING SEABIRD COLONY SECTORS IN 2022 AND 2023

Task	Specified visits	Completed visits	Sectors	Comments
Total breeding colony size	5	5	9	
Breeding season counts (Apr-Jul)	9	9	9	
Non-breeding counts (Aug-Mar)	8	8	9	
Productivity photographic surveys	Weekly mid-May to fledging + 2 just prefledging. Min 200 GU+100 RZ prs	9	3	450+ GU AOS[1] on 51 ledges) & 93+ RZ AOS monitored annually
Timelapse cameras	Two camera technologies trialled during breeding	4 images per day	4	Ineffective, additional photographic surveys carried out

FIGURE 2-1: SEABIRD MONITORING COUNT SECTORS AT SOUTH STACK RSPB RESERVE

[1] AOS = apparently occupied sites.



Of these, parts of Mousetrap Buttress N, Mousetrap Wall, Mousetrap Wallside and Redwall were monitoring for productivity. Ellin's Tower was the location of the Timelapse Systems camera. 1 Bridge N&S, 2 Island, 3 Mousetrap Buttress A, 4 Mousetrap Buttress B, 5 Mousetrap Buttress C, 6 Mousetrap Buttress N, 7 Mousetrap Wall, 8 Mousetrap Wallside, 9 Redwall Buttress. Penlas A & B (sector 10) are out of view to the right. See Annex 6.1 for subsectors, their viewpoints and map grid references.

FIGURE 2-2: SEABIRD MONITORING COUNT SECTORS AT SOUTH STACK RSPB RESERVE VIEWED FROM THE SEA. Picture credit ©Lilian Lieber.

1 Bridge N&S, 2 Island, 3 Mousetrap Buttress A, 4 Mousetrap Buttress B, 5 Mousetrap Buttress C, 6 Mousetrap Buttress N, 7 Mousetrap Wall, 8 Mousetrap Wallside, 9 Redwall Buttress. Penlas A & B (sector 10) are out of view to the right. See Annex 6.1 for subsectors, their viewpoints and map grid references.

FIGURE 2-3. SEABIRD COLONIES IN NORTH WALES WITH HISTORIC ABUNDANCE DATA AVAILABLE THROUGH THE JNCC SEABIRD MONITORING PROGRAM.

2.2 Objective 2. Colony attendance through the breeding season

In addition to standard June colony counts, counts of individuals in the same sectors were also carried out bimonthly from 01 April until 15 July each year from the same viewpoints and during suitable weather (Table 2-1). This period covers all breeding activity and provide information on attendance during successive breeding stages.

2.3 Objective 3. Colony attendance outside breeding season

Monthly counts of individuals from the same viewpoints of colony sectors also took place to inform attendance outside the breeding season (August-March) (Table 2-1).

2.4 Objective 4. Breeding productivity

Productivity estimates were based on the interpretation of repeated photographic surveys of selected ledges and nests across multiple colony sectors. Photographs were taken using a digital single lens reflex (DSLR) camera with 25 mega pixel resolution and 500 mm zoom lens. Our method for estimating breeding productivity differed between species.

Forty-eight productivity monitoring ledges were established for **Guillemots**, which typically nest in aggregations on ledges, making it impractical to follow the progress of individual AOS. Study ledges were stratified by three colony sectors (Table 2-2). Although the number of pairs these contained exceeded our target (Table 2-1), the inaccessible nature of the breeding seabirds at South Stack restricted our productivity monitoring to these sectors where the closest views were possible.

Razorbills typically do not nest in dense aggregations, selecting smaller ledges and recesses for egg placement and resulting in segregation of the two species within colony sectors (although areas with the two species were often adjacent). For **Razorbills** a sample of individual AOS were monitored (Table 2-2). Again, the number monitored exceeded our target (Table 2-1) but were restricted to three count sectors with adequate view. Within monitored sectors study ledges and sites were selected to have varied nesting densities, height above sea level, aspect (as far as feasible), and edge vs. core locations within the sector to reflect potential for variation in breeding performance across the wider colony.

In both cases productivity monitoring ledges and sites were marked on photographs and assigned unique identification codes to ensure consistency of monitoring within and across successive breeding seasons (Figure 2-4).

TABLE 2-2: PRODUCTIVITY MONITORING SAMPLE SIZE AND COLONY-LEVEL MEAN PRODUCTIVITY (CHICKS FLEDGED PER AOS[1]) AT SOUTH STACK RSPB RESERVE IN 2022 AND 2023. ALSO SHOWN ARE THE MOST RECENT PRODUCTIVITY ESTIMATES FROM JNCC'S SEABIRD MONITORING PROGRAM (SMP) FOR SITES IN WALES.

		Mousetrap Wall		Mousetrap Wallside		Redwall Buttress		Whole colony		Wales average	Skomer Island
Species	Metric	2022	2023	2022	2023	2022	2023	2022	2023		
Guillemot	n ledges	22	22	16	16	13	13	51	51		
	n AOS	150	151	84	103	239	196	473	450		
	Chicks/AOS	0.88	0.44	0.77	0.74	0.91	0.29	0.88	0.44	0.71	
Razorbill	n AOS	11	6	24	23	65	64	100	93		
	Chicks/AOS	0.82	1.00	0.92	0.43	0.68	0.30	0.75	0.38		0.53

[1] AOS = apparently occupied site.

[2] SMP estimate 1989-2019 https://jncc.gov.uk/our-work/Guillemot-uria-aalge/

[3] SMP estimate 1993-2919 https://jncc.gov.uk/our-work/Razorbill-alca-torda/

Photographic surveys of study ledges and sites were carried out and images interpreted to generate data on the presence of characteristic behaviours (standing upright, incubating eggs, brooding young chicks for **Guillemots**, and presence of an adult pair for **Razorbills**) and visible chicks. Visits were weekly from early May (approximate time of egg laying), and then every 2-4 days from early June until the third week of July, by when chicks had fledged (Table 2-1). Two additional visits in the days just preceding the main fledging period minimised the number of large chicks overlooked and in 2023, visit frequency was increased to 3-4 times per day in the week prior to predicted fledging in lieu of timelapse cameras which proved inadequate (Table 2-1). Whenever possible visits for productivity monitoring were coincident with whole colony counts to minimise risk of additional disturbance.

Because **Guillemots** nest in exposed locations, fledging date in any one year shows high temporal synchronisation to reduce predation risk for individual eggs and chicks. Being less exposed to predation by nesting on smaller ledges and in recesses, **Razorbills** are less synchronised in their timing of breeding as eggs and chicks are less at risk, and this is supported by local evidence from nearby Puffin Island. For both species we would expect peak fledging dates to vary between years and potentially between count sectors within the colony. Given the paucity of chicks detectable in photographs for both species due to close brooding or cavity nesting (<10% of all study nests), chick size could not be consistently used to estimate age and predict fledging date. Consequently, we selected year specific indicative peak fledging dates in the following way. In 2022, 27 June was used given most eggs hatched 1418 June based on observations of the onset of adults provisioning chicks in colony sectors with productivity monitoring (fledging typically takes place 14 days after hatching). The date used in 2023 was 7 July based on peak hatching between 24-28 June in the Island sector, where GPS tracking enabled close observations to take place (WP7).

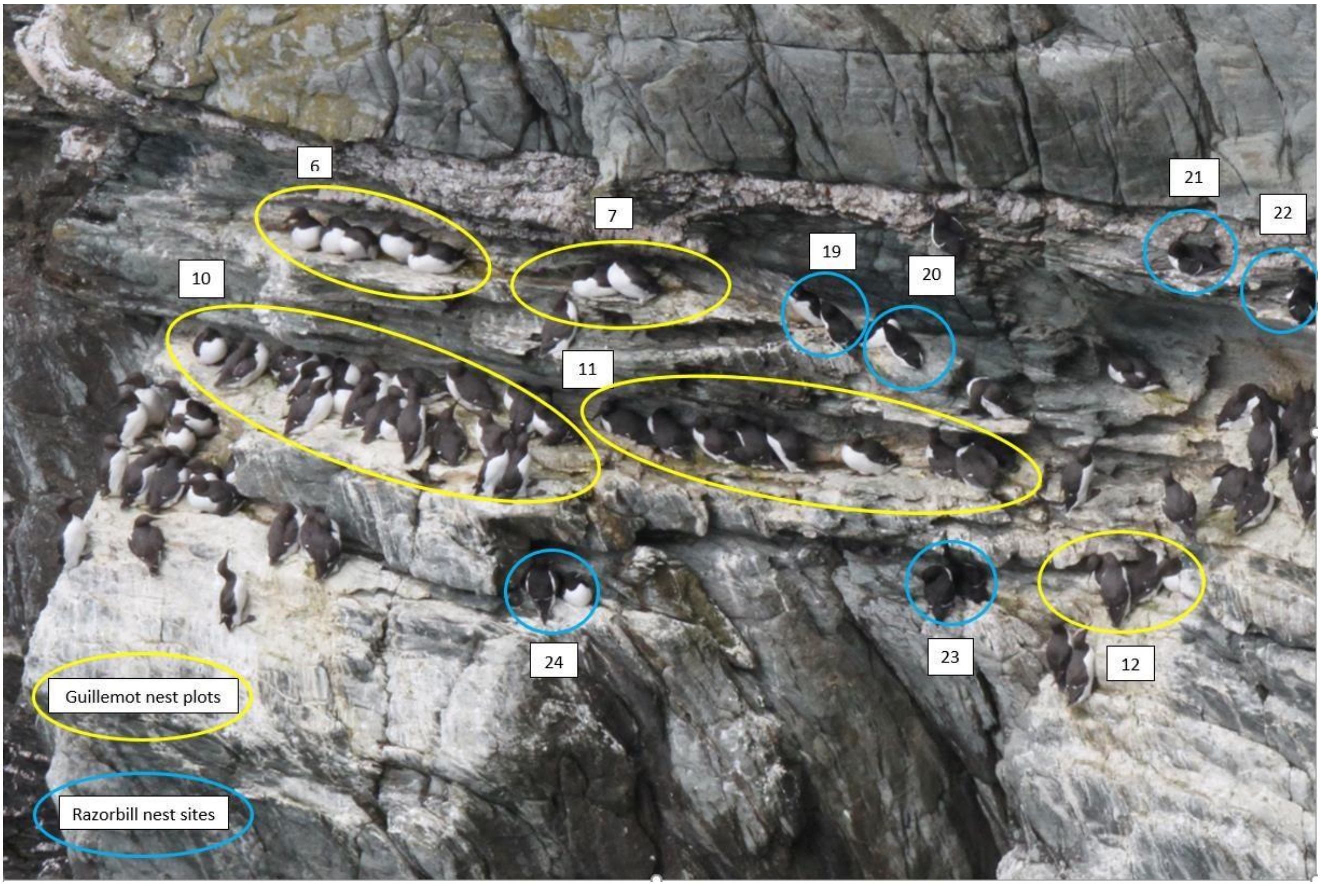
The number of **Guillemot** pairs per study ledge was identified based on maximum consistent counts across at least 2 visits between onset of laying and expected peak fledging 27 June (any outlying maximum was rejected as nonbreeding adults may visit nesting ledges). The number of active AOS on peak fledge dates was taken as the number of pairs with chicks of fledging age. Productivity (chicks fledged per pair) was calculated as the number of AOS with chicks divided by the maximum number of AOS on each study ledge.

As fledging in **Razorbills** is less synchronous than in **Guillemots**, each **Razorbill** AOS was classified as occupied or unoccupied (a pair was considered active if both adults were present or one adult was brooding on two of three consecutive surveys), and as successful or not. Productivity (chicks fledged per pair) across sites in a sector was the number of successful sites divided by the number of occupied sites. For **Razorbills**, the indicative fledging period was 20-24 June in 2022 based on the timing of nests in the Island Sector where close observations were possible as part of WP7 tagging, but unlike the study ledges used for **Guillemots** which supported multiple pairs, success was derived from the activity at individual AOS.

Success was presence of a large chick, continued brooding to an appropriate date (relative to date first incubating: fledging typically takes place 14 days after hatching). Failure was assumed if the site was consistently unoccupied before likely fledging, or there was evidence of confirmed failure (e.g., predated egg seen). Some adults were noted sitting on study ledges and in study sites where we had previously observed total failure evidence by a gap in occupation. As neither species relay after egg or chick loss, we did not include these in our data. Breeding outcome could not be determined for a small number of sites per year (<5%) and these were excluded from analysis. All interpretation of photographs was carried out by the same person.

Time-lapse digital cameras were also deployed in two sectors of the colony, collecting four images per day. One of these overlapped with productivity monitoring ledges and sites. Harris et al (2020) illustrate the potential for underestimate of pair counts, and overestimate of fledged chicks (and therefore overestimation of productivity), when carrying out counts at 2–3-day intervals. One of our planned two time-lapse cameras (Reconyx Hyperfire-2 with focus set to 60m) was deployed approximately 60 m from the breeding birds in the Mousetrap Buttress count sector (Figure 2-1). A third camera (Timelapse Systems unit based on a Canon 24-megapixel DSLR with 55mm lens) was placed at around 120 m to cover a more distant sector within the colony (Redwall Buttress with camera sited on Elin's Tower, (Figure 2-1).

FIGURE 2-4: AN EXAMPLE OF PRODUCTIVITY MONITORING LOCATIONS ON MOUSETRAP WALL BASED ON THE 6 JUNE 2022 MONITORING IMAGE. IDENTIFICATION NUMBERS REFER TO STUDY LEDGES CONTAINING MULTIPLE AOS ON FLAT TYPICALLY CROWDED LEDGES FOR GUILLEMOTS (YELLOW), AND INDIVIDUAL AOS OF RAZORBILLS (BLUE) WHICH FAVOUR ROCK RECESSES AND ARE MORE DISPERSED.



2.5 Statistical analyses

We analysed temporal variation in abundance between and within colonies using Generalised Linear Models (GLMs). As our colony count data showed evidence of over overdispersion, we specified a negative binomial error distribution and log link function. We were interested in the linear effects of YEAR (1994-2023), SITE (a factor specifying colony identity) and their interaction to detect differences between colony-specific rate of change across years. We considered including the 4-level nuisance factor OBSERVER to explore any effects of observer identity on colony counts. However, observer identity was confounded with YEAR as observers periodically changed due to RSPB staff turnover and this was not included in the model. We applied the same approach to our sector-level counts at the South Stack colony, fitting the same model with SITE replaced by SECTOR (a 10-level factor).

Analysis of productivity data also used GLMs. We collected data from successive visits to nesting ledges containing multiple breeding pairs of **Guillemots**, and sites containing single breeding pairs of **Razorbills** and visits increased in frequency (peaking at four per day) as the anticipated peak fledging date approached. We considered analysis of daily nest survival rates based on the number or presence of active AOS on each ledge or site on successive visits. However, although overall breeding status was known with acceptable confidence there was uncertainty about the timing of any breeding failures. Instead, we compared the proportion of study nests that were successful between colony count sectors and between years.

For **Guillemots**, our response variable was the count of successful sites as the numerator and the maximum number of sites present on each ledge as the denominator with a binomial error distribution and logit link function specified.

For **razorbills**, the outcome of each site was our binomial response variable (success or failure) with a logit link function. In both cases we were interested in variation between year (a 2-level factor) and sector (a 3-level factor). We also included their interaction to test whether differences in productivity between sectors was different between the two years. For **Guillemots**, we also included a continuous variable for local population density (the number of AOS on each study ledge) to test for effects on productivity but lacked comparable data for **Guillemots**. Analysis was carried out using SAS software.

3 Results

3.1 Objective 1. Whole colony population estimate

Our estimate of mean breeding **Guillemot** abundance was 13,598 and 11,821 individuals and breeding **Guillemot** was 1,825 and 1,880 individuals in 2022 and 2023 respectively based on reserve staff estimates for continuity with historic data (Table 3-1). Abundance varied between the colony count sectors. The most populous sector for **Guillemot**s was Island, with 43%, while for **Guillemots**, Mousetrap Buttress N contained 47% of the population across the two years (Table 2-1)

Count consistency in 2023 and 2023 was assessed by comparing sector specific counts by each observer where these took place on the same day. On average, Conservation Science staff counted 166 fewer **Guillemots** than Reserve staff. Although statistically significant (paired t-test t_{34}=4.2 P<0.001) this represents a difference of just 8 %.
However, at 4 %, the difference for **Guillemots** was smaller and nonsignificant (t_{38}=0.5 P=0.640). Not all counts took place on the same dates and count differences were more marked when comparing between paired standard visit number, albeit often taking place on different dates, leading to difference in whole colony estimates of 10 and 19% for **Guillemots** and **Razorbills** respectively (Table 3-1).

The populations at RSPB South Stack had increased in abundance by 260 % and 106 % overall for **Guillemots** and **Razorbills** respectively between 1994 and 2023. Our analysis of trends in abundance indicated that rate of increase varied between count sectors for both species (Table 3-2, Figure 3-1). Furthermore, in all cases, individual count sectors showed significant change (Table 3-2). **Guillemots** had increased most in the Island sector (27 in 1998 to 5,320 in 2023, 19,604 %) but had abandoned Penlas North A&B (peak of 15 in 2001 to zero in 2023). **Razorbills** had increased most on Mousetrap Buttress N (from 169 to 592, 252 %) and declined on Penlas North A&B (from 41 to 7; 83 %, (Figure 3-1).

Site-level populations of both species had increased by at least 20% at all other North Wales colonies where comparable data were available (counts beginning by 1997 or earlier and continuing, though not necessarily annually, until at least 2019). We modelled trends in abundance at these colonies following the approach above **Razorbill** abundance had peaked at two of these and has since declined although remaining well above their initial values (Carreg y Llam on Llyn in 2011 and Puffin Island in 2012, Figure 3-2).

3.2 Objective 2. Colony attendance through the breeding season

To aid comparability of seasonal patterns between count sector populations that varied in size by an order of magnitude, we expressed **Guillemot** and **Razorbill** numbers as a percentage of the sector-specific maximum count for each year. Visual inspection of results shows marked synchronicity in colony attendance between sectors through the breeding season with numbers in sectors typically rising to their maxima in June and remaining high until early July when chicks fledge, a pattern that also appears well synchronised between the two species (Figure 33). The onset of breeding was 10 days later in 2023 than in 2022. On allowing for this, there was some evidence than number of birds counted in late summer fell more quickly in the 14 days prior to peak fledging in 2023 compared with 2022 (Figure 3-3 e-f), and effect that varied between sectors illustrated by less overlap in sector trajectories after 1 June in 2023 than 2023 for both species. Furthermore, notable but temporary and localised declines were observed in parts of sectors coincident with the HPAI outbreak and hot weather.

3.3 Objective 3. Colony attendance outside breeding

After chicks have fledged colonies remained empty to the end of December. However, from January birds began visiting the colony again. Furthermore, this reoccupation was not continuous based on our two-year dataset, with the colony vacated for a period in early April in both years (recorded partially in 2022 and in full in 2023, Figure 3-3).

Based on our limited data, **Guillemots** may make their initial colony occupation later than **Razorbills** (mid-March vs.mid-April, Figure 3-3).

TABLE 3-1: COUNTS OF INDIVIDUALS AVERAGED ACROSS FIVE VISITS IN THE FIRST THREE WEEKS OF JUNE IN COLONY SECTORS WITH GOOD COVERAGE ACROSS TWO YEARS BY TWO OBSERVERS AND AT SOUTH STACK FOR A) GUILLEMOTS AND B) RAZORBILLS. OBSERVER 1 WAS RSPB RESERVE STAFF WHILE OBSERVER 2 WAS RSPB SCIENCE TEAM.

A)

Sector	2022		2023	
	Observer 1	Observer 2	Observer 1	Observer 2
Island	5,560	5,320	5,326	5,069
Mousetrap Buttress A	1,366	1,084	1,157	1,100
Mousetrap Buttress B	1,214	1,322	1,048	1,050
Mousetrap Buttress C	1,696	816	1,486	1,434
Mousetrap Buttress N	1,794	1,440	1,312	1,230
Mousetrap Wall	645	603	465	416
Mousetrap Wallside	419	373	381	313
Penlas North A&B	0	0	0	0
Redwall Buttress	904	700	646	527
Colony total	13,598	11,659	11,821	11,139

B)

Sector	2022		2023	
	Observer 1	Observer 2	Observer 1	Observer 2
Island	175	82	357	323
Mousetrap Buttress A	191	158	164	188
Mousetrap Buttress B	16	10	34	28
Mousetrap Buttress C	14	16	17	23
Mousetrap Buttress N	941	595	801	819
Mousetrap Wall	136	121	137	143
Mousetrap Wallside	117	77	220	187
Penlas North A&B	10	7	19	14
Redwall Buttress	225	165	165	154
Colony total	1,825	1,231	1,914	1,880

TABLE 3-2: ABUNDANCE TRENDS FOR COUNT SECTORS AT SOUTH STACK FOR GUILLEMOTS AND RAZORBILLS. A) MINIMAL ADEQUATE MODELS FOR SECTOR TRENDS. B) SECTOR SPECIFIC TRENDS. EARLIEST AND LATEST DATA WERE 1994 AND 2023 UNLESS INDICATED.

A)

Term	Guillemot		Razorbill	
	F	*P*	F	*P*
Year	$817_{1,1536}$	<0.0001	$298_{1,1536}$	**<0.0001**
Sector	$134_{9,1536}$	<0.0001	$61_{9,1536}$	**<0.0001**
Year*Sector	$135_{9,1536}$	<0.0001	$62_{9,1636}$	**<0.0001**

B)

Species	Sector	t	P	Slope	Earliest obs abundance	Latest obs abundance	Obs % change
Guillemot	Bridge N&S	6.8_{1536}	**<.0001**	0.0284	127	170 (2019)	34
	Island	34.4_{1536}	**<.0001**	0.1683	27 (1998)	5320	19,604
	Mousetrap Buttress A	10.3_{1536}	**<.0001**	0.0319	584	1084	86
	Mousetrap Buttress B	7.9_{1536}	**<.0001**	0.0249	740	1322	79
	Mousetrap Buttress C	5.9_{1536}	**<.0001**	0.0191	1035	816	-21
	Mousetrap Buttress N	28.5_{1536}	**<.0001**	0.0859	149	1440	868
	Mousetrap Wall	17.8_{1536}	**<.0001**	0.0577	173	603	249
	Mousetrap Wallside	19.6_{1536}	**<.0001**	0.0601	57	373	555
	Penlas North A&B	-9.0_{1536}	**<.0001**	-0.0890	1	0	-100
	Redwall Buttress	7.9_{1536}	**<.0001**	0.0251	372	700	88
Razorbill	Bridge N&S	4.68_{1535}	**<.0001**	0.0236	80	85 (2019)	6
	Island	7.68_{1535}	**<.0001**	0.0401	42 (1998)	82	98
	Mousetrap Buttress A	11.2_{1535}	**<.0001**	0.0445	64	158	147
	Mousetrap Buttress B	2.2_{1535}	**0.025**	0.0098	23	10	-55
	Mousetrap Buttress C	2.6_{1535}	**0.009**	0.0121	11	16	46
	Mousetrap Buttress N	16.4_{1535}	**<.0001**	0.0618	169	595	252
	Mousetrap Wall	14.6_{1535}	**<.0001**	0.0572	57	121	112
	Mousetrap Wallside	8.3_{1535}	**<.0001**	0.0332	24	77	221
	Penlas North A&B	-12.8_{1535}	**<.0001**	-0.0581	41	7	-83
	Redwall Buttress	3.6_{1535}	**0.001**	0.0142	130	165	27

TABLE 3-3: ABUNDANCE TRENDS FOR NORTH WALES COLONY COUNTS FOR GUILLEMOTS AND RAZORBILLS. A) MINIMAL ADEQUATE MODELS FOR COLONY TRENDS. B) COLONY SPECIFIC TRENDS. EARLIEST AND LATEST DATA WERE 1994 AND 2022 UNLESS INDICATED.

A)

Term	Guillemot		Razorbill	
	F	*P*	F	*P*
Year	$195.5_{1,100}$	**<.0001**	$99.2_{1,91}$	**<.0001**
Colony	$11.7_{4,100}$	**<.0001**	$16.1_{4,91}$	**<.0001**
Year*Colony	$11.7_{4,100}$	**<.0001**	$16.5_{4,91}$	**<.0001**

B)

Species	Sector	t	P	Slope	Earliest obs abundance	Latest obs abundance	Obs % change
Guillemot	Bardsey Island	9.8_{100}	**<.0001**	0.0644	269(1995)	4200 (2021)	425
	Carreg y Llam	2.3_{100}	**<.0001**	0.0228	6185 (1997)	13560	125
	Puffin Island	4.4_{100}	**<.0001**	0.02069	1559 (1997	1413 (2019)	169
	Great Orme	5.4_{100}	**<.0001**	0.0326	1935	11821	20
	South Stack	11.8_{100}	**<.0001**	0.0554	3236	2670	320
Razorbill	Bardsey Island	10.3_{91}	**<.0001**	0.0910	283 (1995)	681 (2021)	577
	Carreg y Llam	1.0_{91}	0.330	0.0266	187 (1997)	367	184
	Puffin Island	1.2_{91}	0.230	0.0023	169 (1997)	520	303

	Great Orme	5.0_{91}	**<.0001**	0.0551	212	370	20
	South Stack	7.3_{92}	**<.0001**	0.0455	599	1880	205

FIGURE 3-1: STATISTICALLY SIGNIFICANT FITTED TEMPORAL TRENDS IN A) GUILLEMOTS AND B) RAZORBILLS FOR INDIVIDUAL COLONY COUNT SECTORS AT SOUTH STACK 1994-2023.

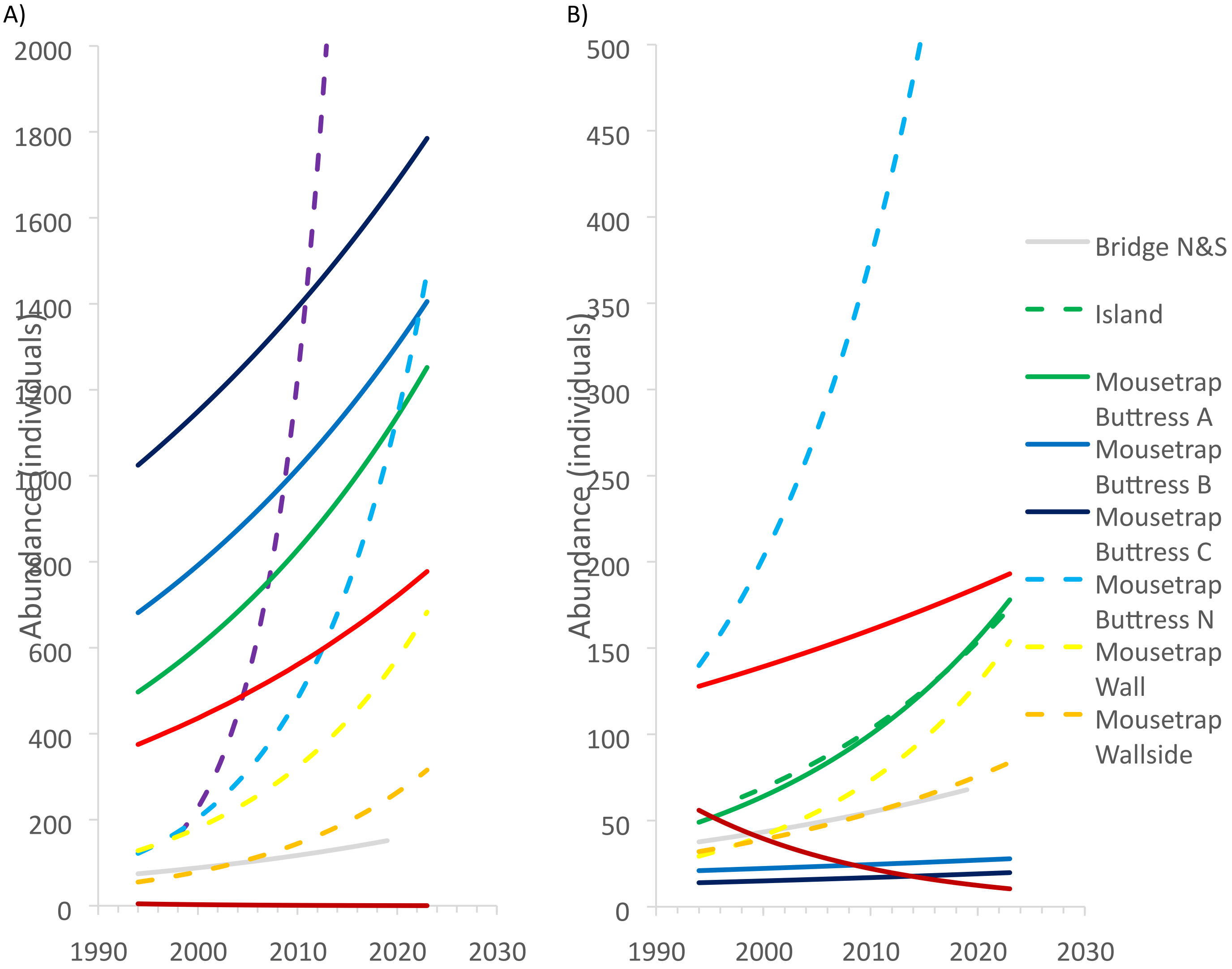
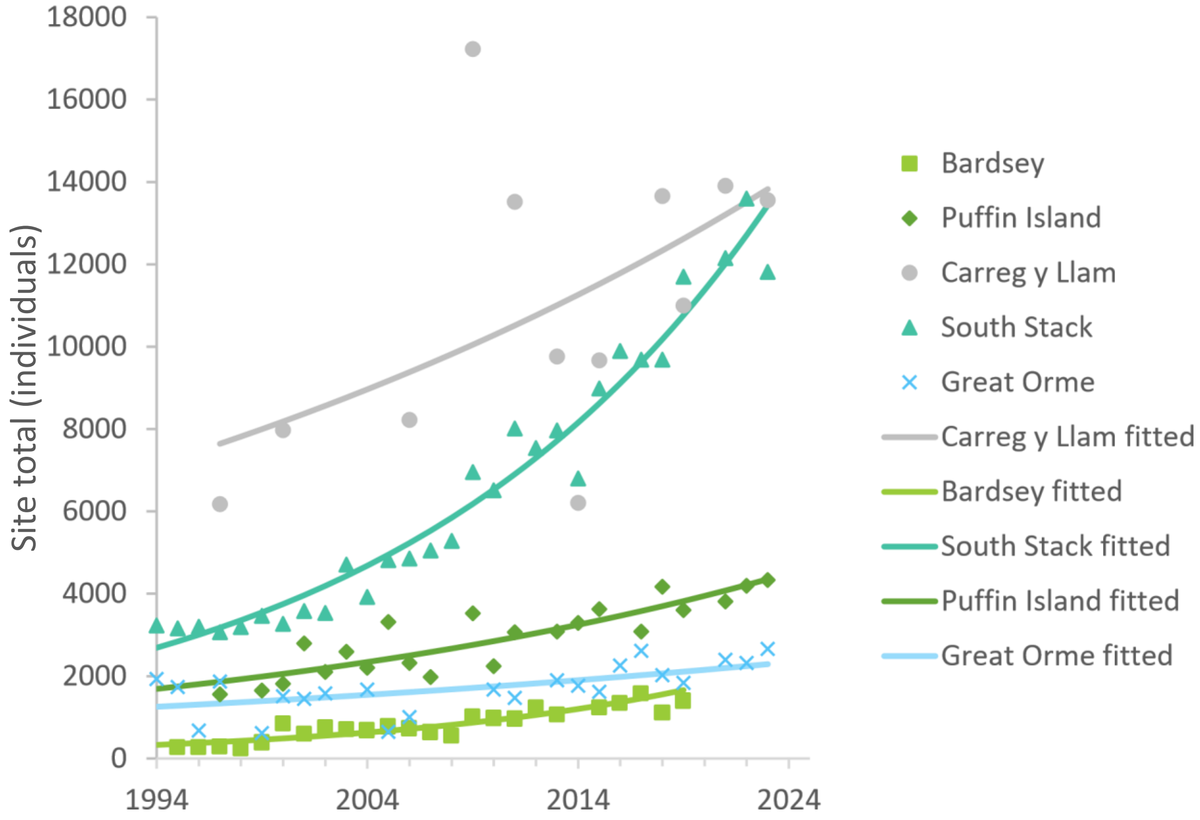


FIGURE 3-2: COUNTS OF INDIVIDUALS AT ADEQUATELY MONITORED A) GUILLEMOT AND B) RAZORBILL COLONIES IN NORTH WALES 1994-2023 (DATA FROM JNCC SEABIRD MONITORING PROGRAM, JNCC 2021). SYMBOLS SHOW AVAILABLE YEAR VALUES, SOLID LINES SHOW SIGNIFICANT TEMPORAL TRENDS, WHILE DASHED LINE JOIN SUCCESSIVE ANNUAL VALUES FOR SITES WHERE SIGNIFICANT TRENDS ARE ABSENT.

A)



B)

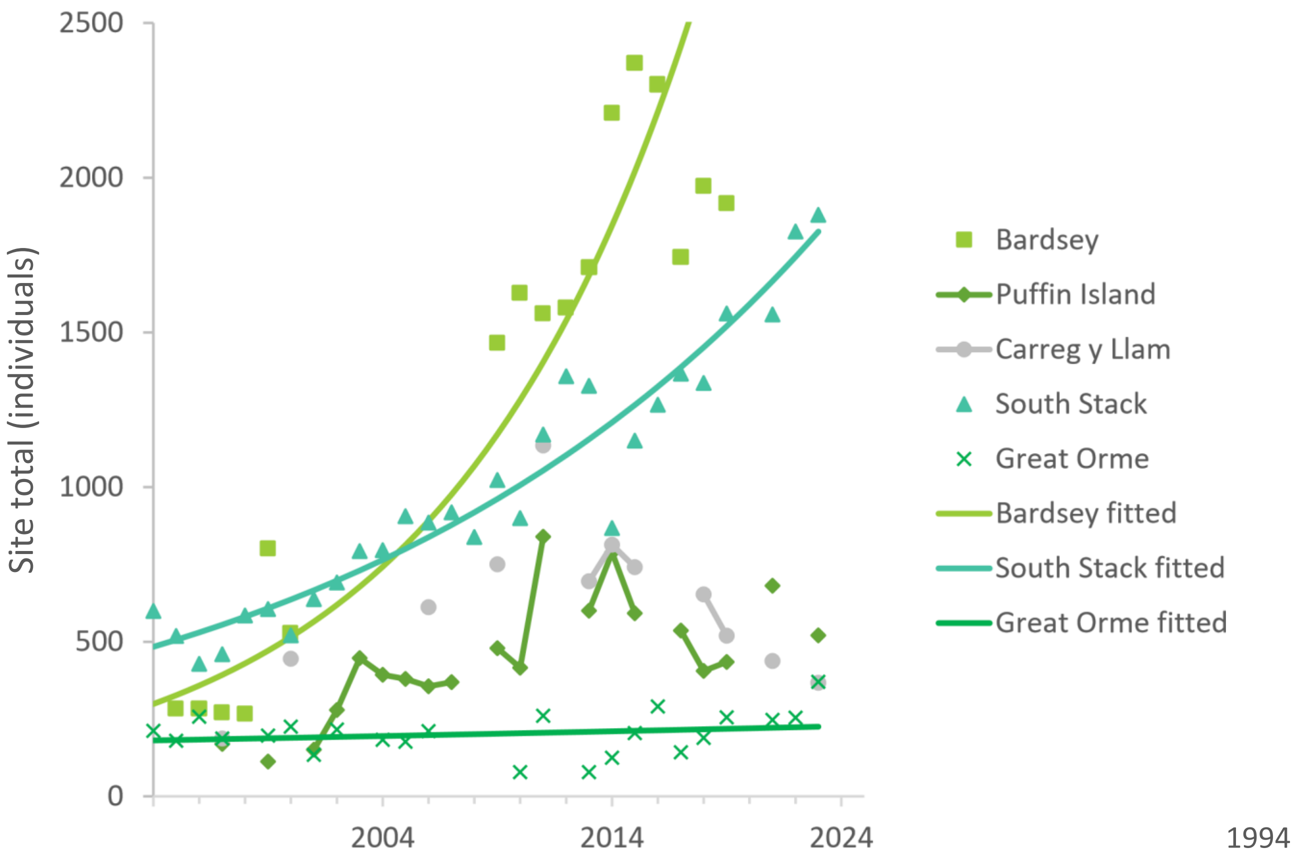
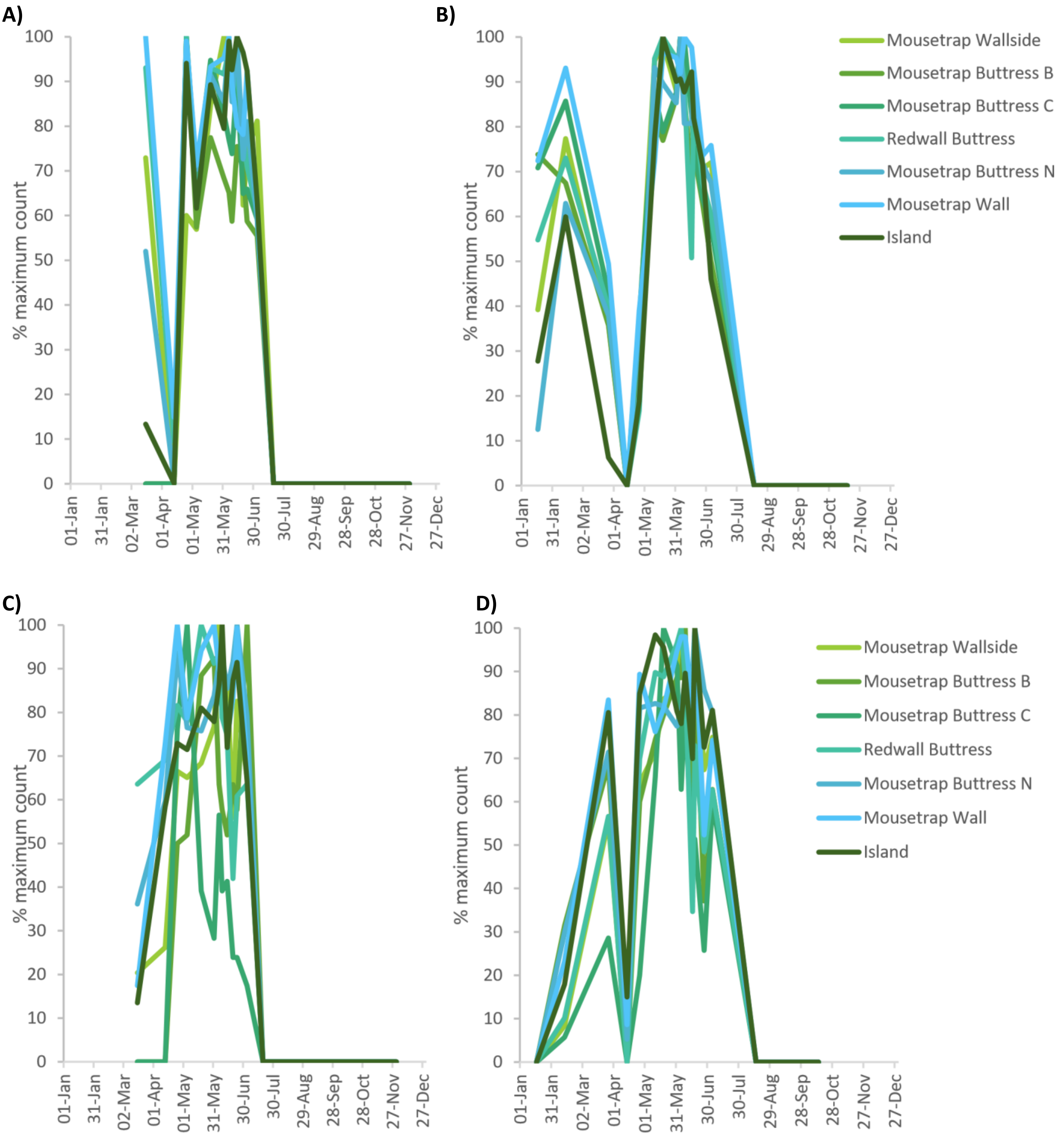
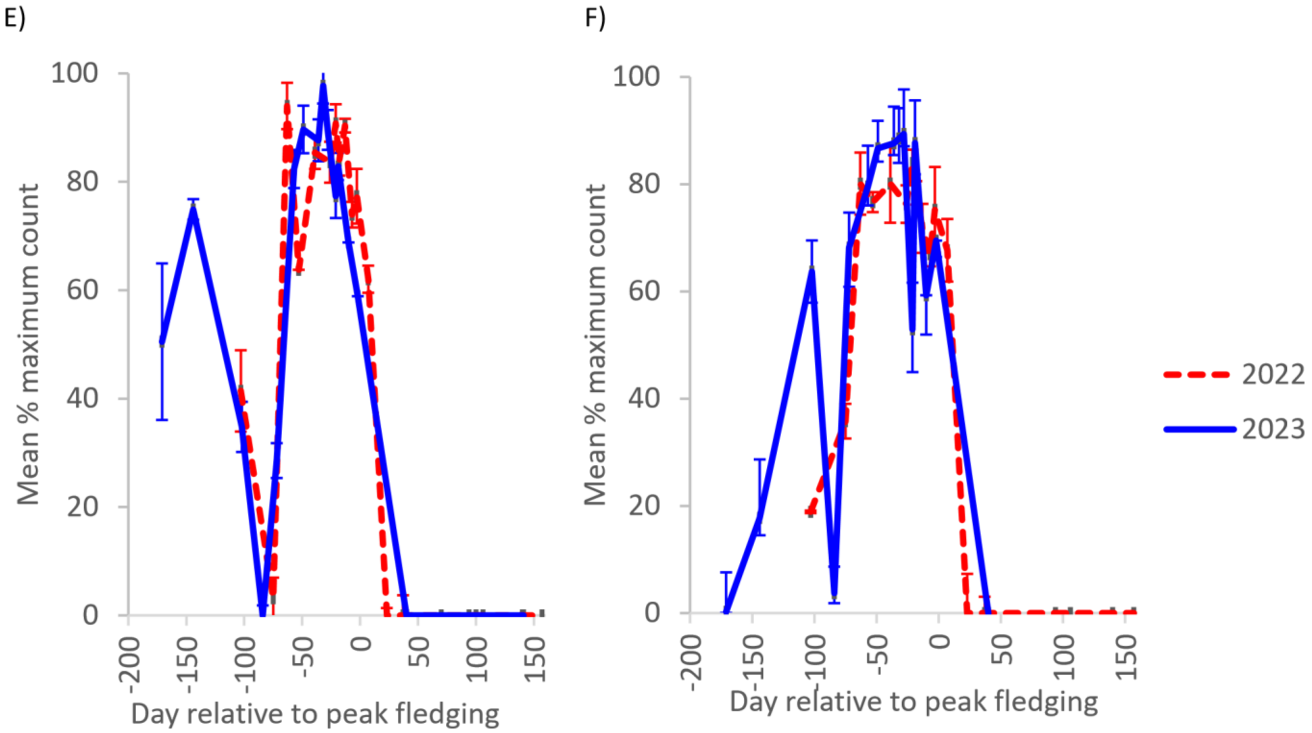


FIGURE 3-3: COUNT SECTOR SPECIFIC SEASONAL VARIATION IN COLONY ATTENDANCE AT SOUTH STACK ACROSS TWO YEARS BY GUILLEMOTS IN A) 2022 B) 2023, AND RAZORBILLS IN C) 2022 AND D) 2023. COUNTS OF

INDIVIDUALS ARE EXPRESSED AS A PERCENTAGE OF THE MAXIMUM COUNT FOR EACH SECTOR IN EACH YEAR. THE % OF MAXIMUM COUNT AVERAGED ACROSS SECTORS (+SE) RELATIVE TO EXPECTED PEAK FLEDGING (DAY ZERO) IS SHOWN FOR E) GUILLEMOTS AND F) RAZORBILLS.





3.4 Objective 4. Breeding productivity

Observed productivity estimates varied between colony sectors and between years (Table 2-2). Furthermore, our analysis of **Guillemot** productivity suggested it varied differently between sectors across the two years (Table 3-4, Figure 3-4), but there was no evidence that it was influenced by local population density (P=0.436). **Razorbill** productivity varied significantly between sectors and years, but these did not interact (Year*Section P=0.511, Table 3-4, Figure 3-4).

TABLE 3-4: MINIMAL ADEQUATE MODELS OF VARIATION IN A) GUILLEMOT AND B) RAZORBILL PRODUCTIVITY AT SOUTH STACK IN 2022 AND 2023. SIGNIFICANT EFFECTS ARE SHOWN IN FIGURE 3-4

A)

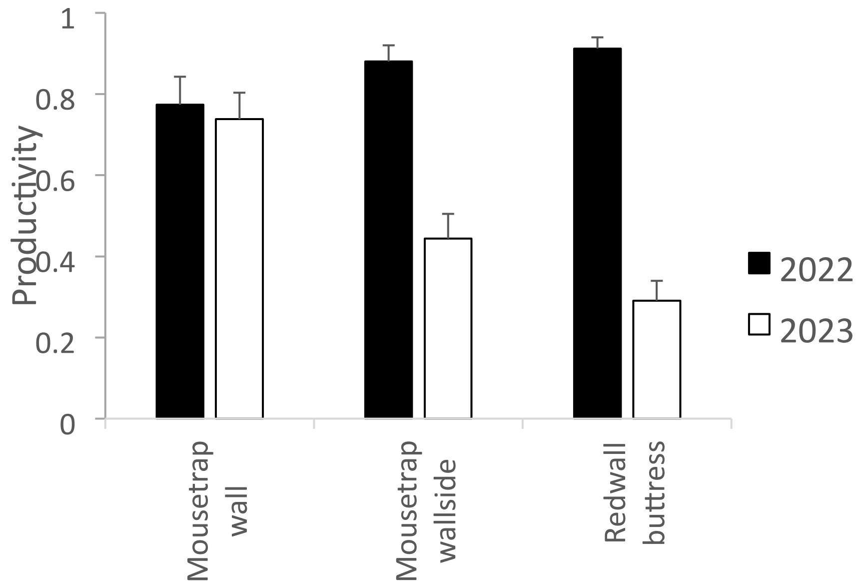
Effect	F	DF	P
Year	49.2	1,92	**<.0001**
Section	0.8	2,92	0.475
Year*Section	10.4	2,92	**<.0001**

B)

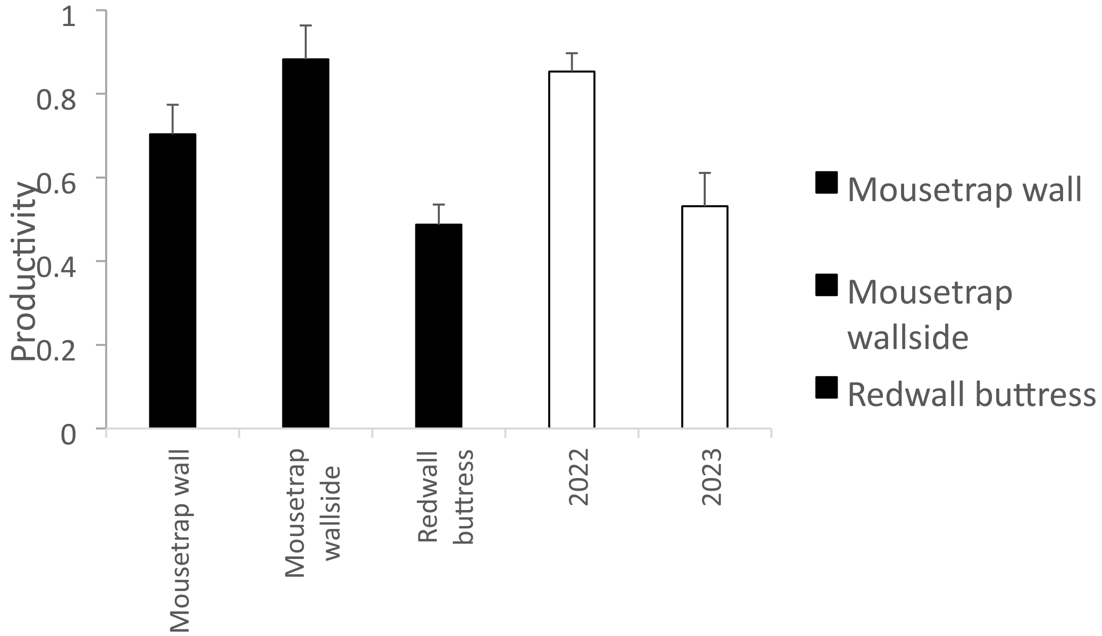
Effect	F	DF	P
Year	24.6	1,188	**<.0001**
Section	5.4	2,188	**0.005**

FIGURE 3-4: SIGNIFICANT PATTERNS OF VARIATION IN BREEDING PRODUCTIVITY AT SOUTH STACK AMONG THREE MONITORED COLONY SECTORS AND TWO YEARS. A) SIGNIFICANT SECTOR AND YEAR INTERACTION FOR GUILLEMOTS. B) MAIN EFFECTS FOR RAZORBILL, WHICH DID NOT INTERACT. BARS SHOW MODEL PREDICTIONS +1SE.

A)



B)



4 Discussion

4.1 Whole colony estimates and patterns of change

Although observer count consistency was extremely high based on same day visits (8 % and 4 % for Guillemots and Razorbills respectively), whole colony population estimates differed by 10 and 19% for the same species. For practical reasons Reserve and Conservation science counts often took place on different dates within the recommending period (as would normally happen between years).

Variation in colony size estimates is not unexpected and given the high consistency among counters this is likely due to day-to-day variation in attendance by the birds – an unavoidable source or error. Temporal variation in colony attendance will occur at hourly and daily scales with the breeding season as not all breeding birds or non-breeding 'loafers' (young birds which have yet to pair and acquire a nesting site) will be present at any one time. Such independent replication of full colony counts rarely takes place in seabird monitoring and its assessment here provides confidence in the repeatability of individual counts Whilst natural variation in colony attendance means there will always be a degree of error in colony size estimates, we can expect effects of counter identity to be minimal.

Historic counts show marked increase in abundance of both species at RSPB South Stack (Figure 3-1). Although most count sectors showed evidence of increasing abundance, temporal patterns were not uniform and varied between species, with a minority of sectors showing very marked increase that differed between species (Figure 3-1). The most marked increase in Guillemots has taken place in the Island count sector, where cliff structure, absence of mammalian predators and human disturbance has likely allowed breeding birds to expand onto flatter ground at the top of the cliff. These are the only breeding auks that are accessible for catching at South Stack Reserve and is where WP7 (Guillemot and Razorbill tagging) focussed its data collection on tracking birds at sea. Mousetrap Buttress N supports by far the largest increase in numbers of Razorbills among the colony sectors. As this species selects small ledges and recesses for nesting, which may be more available in this sector than on the Island. This highlights the role that nest site availability likely plays in structured patterns of abundance change within breeding seabird colonies (e.g., Birkhead 1977). For example, several sectors support few Razorbills ($\leq$15 %), such as Penlas A&B which has always supported few birds and their numbers have declined in recent decades. This count sector is made up of lower cliffs which may represent relatively less attractive or less successful nest sites for these auks than the much taller cliffs of Mousetrap Buttress N. Importantly, the population increases in other sectors are considerably larger than the decline seen in Penlas A&B, indicating that population growth must be driven by recruitment of more new birds to the colony than are being lost through mortality, rather than by redistribution among sectors. Indeed, auks show high fidelity to nesting areas once they have begun making breeding attempts (e.g., Harris et al. 1996).

The RSPB South Stack Guillemot and Razorbill populations have shown consistent population growth since 1994 and are now represent the most populous colonies in north Wales (Figure 3-2). Although data are less complete for the four other colonies with monitoring, which include sites to the west on the Llŷn (Bardey and Carreg y Llam) and to the east (Puffin Island and Great Orme, Figure 2-2), some patterns are clear. Guillemot populations have increased by over 100% since 1994, with the exception of the relatively stable Great Orme colony (20% increase). This suggests local environmental conditions are highly favourable for this species, and that nesting opportunities at these colonies are currently not limiting. This increase is of a similar magnitude to that shown in the wider UK (Figure 4-1, Wales index not available).

Razorbills have also increased nationally, by around 200% both in the wider UK and in Wales (Figure 4-2). However, Razorbills have increased on Bardsey and at RSPB South Stack but show stability (Great Orme) or recent decline (Carreg y Llam and Puffin Island) at other north Wales colonies. Given the high site fidelity of breeding seabirds, reasons for these changes are currently unclear, but could include reduced productivity through locally changing food supplies during breeding and/or increased predation of nest contents. It is notable that following successful removal of Brown Rats (known nest predators in seabird colonies) from Puffin Island in 1998, they recolonised in 2021. Monitoring of productivity at additional sites would be valuable in explaining these local differences. For example, an increase in productivity and subsequently population growth rate would be predicted following planned rat eradication were local population size primarily influenced by levels of productivity.

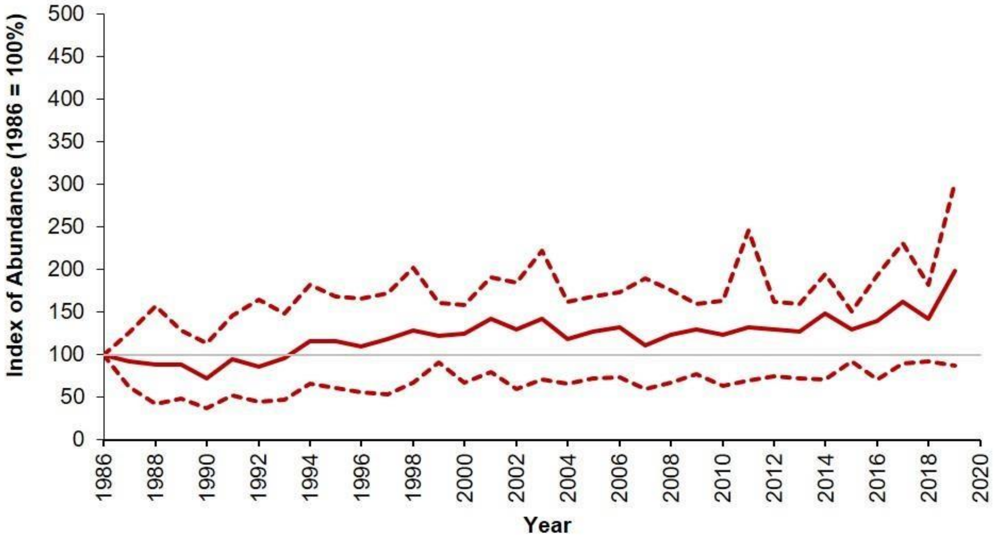
There seems scope for further increase in Guillemot numbers in the Island count sector at South Stack and perhaps for Razorbills at Penlas A&B or on cliffs towards North Stack within the typical operating life span of marine energy generation infrastructure (around 25 years, Cook et al. 2019). However, change in breeding seabird densities and colony extent may have demographic consequences. For example, increasing density and expansion into lower quality nest sites may lead to changes in colony-scale breeding productivity and population growth rate, and the possibility of such effects should be considered within EMMP Monitoring Indicator 7 (local population effects).

Limitations in the spatial and temporal coverage of seabird monitoring can mean that data from ongoing long-term monitoring campaigns lack sufficient statistical power to detect changes in abundance and demography that may be associated with the effects of offshore developments such as wind farms (Cooke et al 2019). Reduced statistical power raises the risk of type II errors: that is, incorrectly reporting no impact when in fact an impact is present.

Statistical power is increased by using larger datasets. For example, analyses based on annual monitoring data has higher power to detect change (at the P<0.05 probability level) than when based periodic data (e.g., monitored every five years) (Cooke et al 2019). The influence of sample size on the power to detect significant change in abundance due to tidal stream installations could be explored for Guillemot and Razorbill abundance (for which we have historic trend data) following the approach of Cooke et al 2019 (Figure 6-2). Achieving sufficient statistical power through appropriate population monitoring design is necessary for Monitoring Indicator 7 to be informative.

FIGURE 4-1: GUILLEMOT ABUNDANCE INDEX AND PRODUCTIVITY LEVELS IN THE UK. DATA FROM JNCC SEABIRD MONITORING PROGRAM (JNCC 2021) (SOLID = INDEX ESTIMATES, DASHED = 95% CLS).

A)



B)

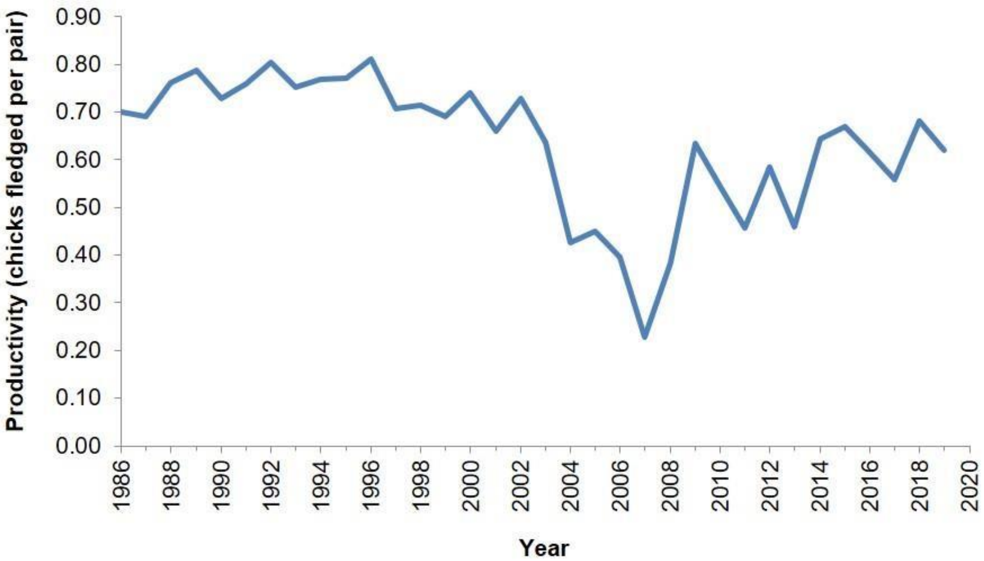
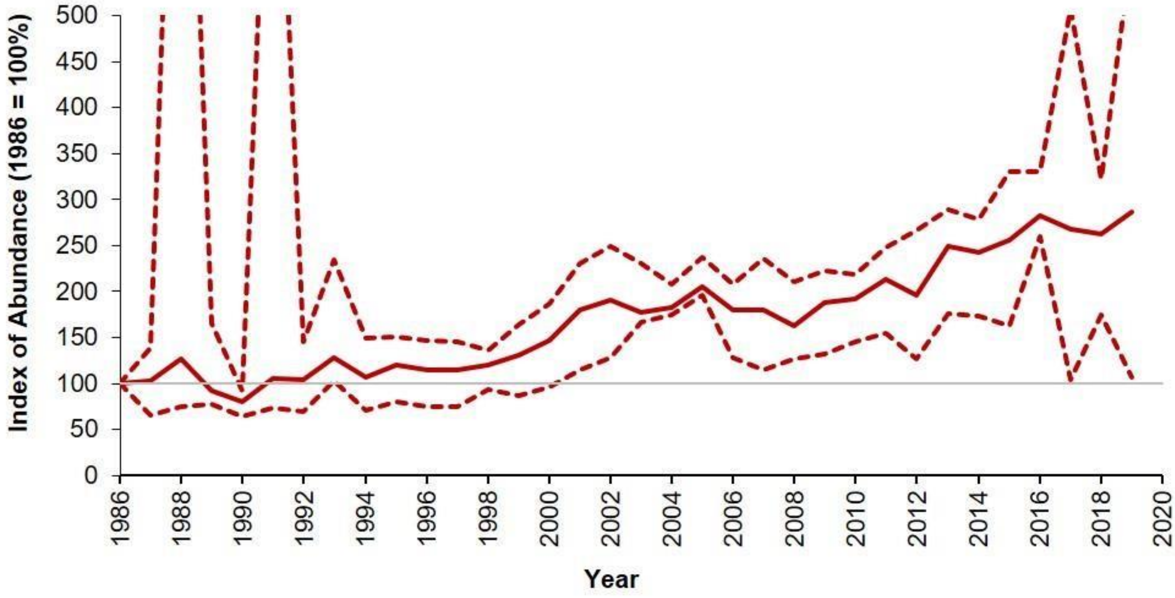
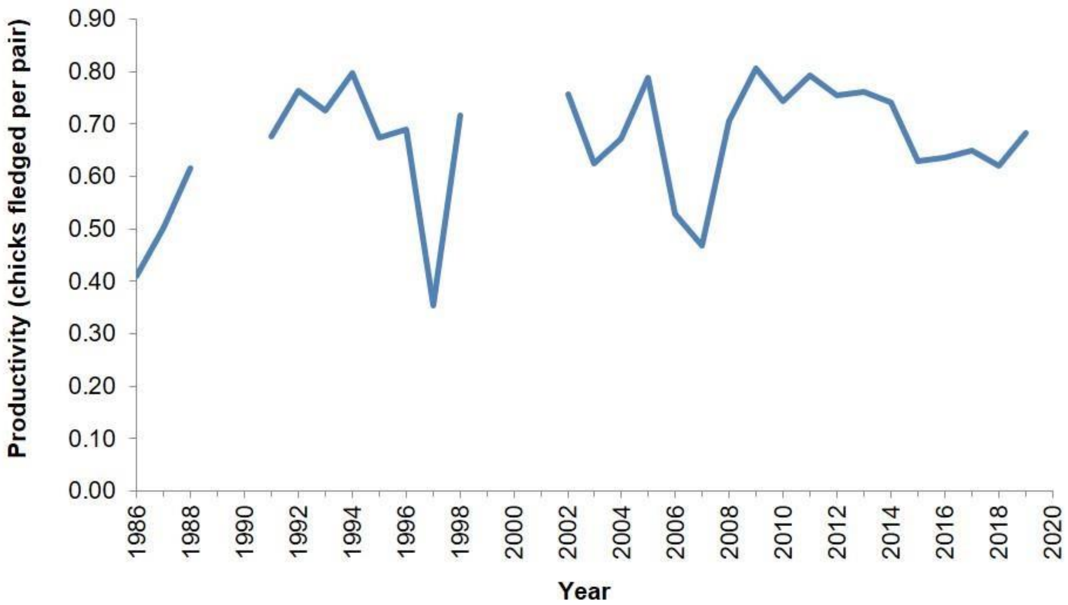


FIGURE 4-2: RAZORBILL ABUNDANCE INDEX IN WALES (A) AND PRODUCTIVITY LEVELS ON SKOMER ISLAND (B) WITHIN ANNUAL ESTIMATES LINKED BY LINES (SOLID = INDEX ESTIMATES, DASHED = 95% CLS). DATA FROM JNCC SEABIRD MONITORING PROGRAM (JNCC 2021). GAPS HIGHLIGHT PERIODS WITH NO DATA OR WHERE VALUES ARE BEYOND THE Y-AXIS RANGE SHOWN.

A)



B)



4.2 Colony attendance

Our data on Guillemot and Razorbill colony attendance show strong similarities to those of other studies (e.g., Harris & Wanless 1988, 1989a). Numbers built up to peak attendance in April until early July when chicks fledge, after which colonies were empty for a long period. Our counts in late March 2022 suggest birds were returning to the cliffs before breeding commenced and our ongoing counts in January-March 2023 provide data on this. Autumn and winter colony attendance varies markedly between sites and is influenced at the level of individuals by a range of factors including breeder age, previous breeding success, and distance to wintering areas (Mudge et al. 1987, Harris & Wanless 1989b, Buckingham et al. 2022).

There was some evidence that the environmental events in 2023 (HPAI, marine and terrestrial heat wave events) appears to have had an impact on seasonal the pattern of colony attendance but anecdotal observations suggest quite localised effects. Furthermore, an increase in adult mortality at South Stack seems plausible given numbers of beached adult auks reported that were ringed as chicks or adults on nearby Puffin Island which also suffered an HPAI outbreak. Breeding seabird abundance can be relatively insensitive to chick mortality, but more sensitive to adult mortality. Further years of abundance monitoring are needed to quantify impact of these environmental events on breeding bird abundance at South Stack and other colonies in north Wales.

4.3 Productivity

Previous studies have estimated productivity using frequent periods of close observation where birds are close enough to look for eggs and chicks when incubating or brooding adults move (Birkhead 2023). This was not practical at South Stack because of the high and inaccessible cliffs. However, our photographic approach enabled productivity to be estimated for a large number of AOS across multiple colony sectors (450+ Guillemots, 93+ Razorbills across three sectors with study ledges and sites stratified by local population density, aspect and hight above water) so is potentially more representative of colony-wide productivity than more intensive but localised observations.

The method of our estimates of Guillemot productivity followed Walsh et al. (1995) in using a generic time to peak fledging informed by a local hatching date. We used information from the Island sector where GPS tracking involved close approach to birds so breeding stage was apparent. However, differences in laying dates between sectors could result in related differences in peak fledging dates on which our productivity estimates are based. Whilst we believe this would have minimal impact on our productivity estimates, effort in documenting magnitude of variation in sector specific peak egg laying dates could be a valuable refinement to future productivity monitoring.

Time lapse camera are used in productivity monitoring because they can provide images of monitored ledges and sites in the days before fledging more frequently than may be possible using human surveyors. If of sufficient resolution, these effectively reduce the time-period between monitoring visits. By doing so, it reduces the chances of failed breeding being misassigned as successful. However, the two camera technologies we trialled proved inadequate for the distant viewed of birds required at South Stack. Additional photographic surveys too place in 2023 to replace these automatic systems.

In 2022 our estimates showed modest variation among sectors in 2022 (0.77-0.91, Table 2-2). The 2022 colony average for Guillemots was very similar to the long-term average figure for Wales (0.71, Table 2-2, year-specific values were not available), and higher than the UK value which has been influenced by poor productivity in Scotland in recent years (JNCC 2021, Figure 4-1). Lower productivity in colonies in north and east Scotland, where sandeels are the main prey, coincided with food shortages. However, Guillemots in Welsh colonies are less reliant on sandeels, feeding instead mostly on Sprats and Gadoids, although more recently there has been an increase in relatively low-quality Gadoid prey, suggesting a change in prey availability, at least in the Skomer Island area (Riordan & Birkhead 2018).

Our productivity data for Razorbills was based on samples of focal nest sites in three count sectors. The resulting estimate of 0.75 chicks fledged per pair in 2022 (range 0.68-0.92) is higher than the value for Skomer Island in 2019 (0.67), although productivity there has varied between years, with our estimate towards the upper limit of the observed range there (Figure 4-2). Productivity was also lower in 2023, also likely linked to HPAI, and also with marked variation in impact among sectors across years (Table 2-2), although this interactive effect was not significant for Razorbills (overall 38% lower in 2023, Figure 3-4). Given the observed productivity values and sample sizes, smaller numbers of study nest sites might in part account for the absence of significant interaction (n=11 and 6 for Mousetrap Buttress A where observed productivity increased from 0.82 to 1 from 2022 to 2023 (Table 2-2).

South Stack suffered an outbreak of highly pathogenic avian influenza (HPAI) midway through the 2023 breeding season. Whilst few dead birds were seen on cliffs, there were observations of large numbers of dead chicks below the cliffs prior to the expected peak fledging data (Figure 4-3) and tests of locally beached dead chicks confirmed the pathogen. In the weeks after the breeding season, atypically higher numbers of adult Guillemots ringed on Puffin Island were reported from beaches in northwest England, indicating increased mortality among adults likely from north Wales colonies, and there was some evidence of reduced counts within the colony. However, impact of food shortage remains a possible contributory factor given the marine heatwave in 2023 or heat stress from period of hot weather (record high June temperature recorded at Valley Met Office surface station). Our analysis suggests Guillemot breeding success was reduced by a significant 50% and 68% in two monitored sectors in 2023, although a third sector was not significantly affected (5% reduction, Figure 3-4). This may support HPAI as a key driver of poor productivity in 2023. A higher level of immunity among birds in that sector seems an unlikely explanation as this was the first occurrence of the disease at South Stack. An alternative explanation is that low levels of interchange of adults among colony sectors or even specific ledges resulted in lower exposure to infection. Chance effects though low sample size seem an unlikely cause of this result (the Mousetrap Wall sector affected contained 22 study ledges supporting 152 pairs). However other extreme environmental events that occurred in 2023 could also have impacted productivity either separately or in combination with HPAI, but we are unable to separate these effects further with the available data.

Timelapse cameras are used for monitoring seabirds, especially in logistically difficult to access areas such as islands. However, they have limitations, especially when used to monitor productivity which requires eggs, chicks and characteristic behaviours to be distinguishable. Primarily, the sites must still be accessible as cameras need to be placed relatively close to representative nesting sites for adequate detail to be recorded. This proved impossible at South Stack where the majority of seabirds nest on vertiginous cliffs. Specifically, although the Reconyx cameras had been focussed to 60m, there are no options for increasing magnification for this product. In contrast, being based on a DSLR, magnification of the Timelapse Systems camera was limited by weatherproof housing size. Whilst a much larger housing could contain the 500mm lens successfully used for the handheld photographic monitoring, the considerably reduced field of view of this focal length would have drastically reduced the number of study ledges and sites monitored. However, the latter system would likely prove successful for monitoring daily variation in colony attendance in which only counts of adults are needed.

FIGURE 4-3: DEAD GUILLEMOT CHICKS IN THE WATER BELOW BREEDING SITES AT SOUTH STACK ON 3 JULY 2023.

4.4 Conclusions

Our recent and historical estimates of colony size for Guillemots indicate a long-term 6% annual population growth rate at South Stack, which forms part of a regional population that is also generally increasing. Guillemots on Skomer Island have showed a comparable period of exponential population growth (5 % per annum) which has been explained without the need for immigration from other colonies (Meade et al. 2013), and the general increase in colony sizes in the North Wales region may similarly be driven by high productivity and survival. The ultimate cause of population growth may lay with increasing prey populations. Whilst there is a lack of evidence on what might have brought this about, it may include changes in fishing practices or climate related oceanographic conditions. Within colony variation in population growth was apparent at South Stack, which varied between auk species, and this is typical of large breeding seabird colonies with varied nest site quality. Seasonal patterns of colony attendance match our understanding of auk ecology, with the absence of autumn colony attendance at South Stack suggesting competition for nest sites is not currently limiting abundance (Bennet et al. 2022).

Although obtaining data on productivity present methodological challenges, our 2022 estimates were consistent with national and local estimates for Wales, but varied between count sectors suggesting population structure at the within-colony scale needs to be considered in future productivity monitoring. If our 2022 productivity estimate is typical of recent years, then its high level compared with recent Wales supports the interpretation that the high rates of annual population growth at South Stack have driven population growth. Environmental events such as those that coincided in 2023 present challenges in detecting trends in both abundance and productivity by creating additional variation in data among years and need to be considered in the design of long-term monitoring programs capable of detecting magnitudes of development impact that are ecologically meaningful.

Ultimately, any negative population impact of the development on Guillemots and Razorbills will manifest as a decline in population growth rate, which includes the possibility of continued growth but at a reduced rate, and this could be the result of a fall in survival, productivity, or both. Monitoring population size in particular represents a relatively cheap low-tech (and low risk) method of monitoring for such impact that is, importantly, consistently repeatable. A key part of planning a long-term monitoring campaign able to detect impacts of the tidal stream development is the use of power analysis. This should include different scenarios of monitoring frequency and plausible magnitudes of change in population growth rate due to the development, and we present an example approach for how this could be done.

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6 Annex

FIGURE 6-1: SOUTH STACK SEABIRD COLONY SECTOR AND SUBSECTOR MIDPOINT LOCATIONS AND THEIR MONITORING VIEWPOINTS.

Sector name	Subsector	Gridref area counted	Gridref viewed from
Bridge N&S	N of bridge cliff	SH20438225	SH20368228
Bridge N&S	South of bridge cliff	SH20428229	SH20368228
Island	Island N of path	SH20358226	SH20478227
Island	Island beyond cove	SH20288224	SH20478227
Island	Island nearside of cove	SH20338232	SH20478227
Mousetrap Butress A	Mousetrap buttress A	SH20528216	SH20598200
Mousetrap Butress B	Mousetrap buttress B	SH20558216	SH20598200
Mousetrap Butress C	Mousetrap buttress B	SH20508216	SH20368228
Mousetrap Butress N	Mousetrap Boulders	SH20528218	SH20478226
Mousetrap Butress N	Mousetrap buttress N	SH20568218	SH20478226
Mousetrap Wall	Hole in the Wall	SH20518224	SH20468224
Mousetrap Wallside	Mousetrap Wallside	SH20478223	SH20578218
Mousetrap Wallside	Mousetrap Wallside	SH20518224	SH20578218
Penlas A&B	Penlas A	SH20758158	SH20848181
Penlas A&B	Penlas B & stack	SH20758158	SH20848181
Redwall Butress	Redwall Buttress	SH20588210	SH20598200
Redwall Butress	Redwall main cliff	SH20628210	SH20598200
Redwall Butress	Redwall below Elin's Tower	SH20168204	SH20598200

6.2. Statistical power to detect change in population growth rate

Knowing the likelihood of detecting significant effects of interest can be evaluated by simulating future data for different of scenarios of change. In this case we are interested in the impact of monitoring frequency on ability to detect decline in population growth rate of a range of magnitudes. Decline in growth rate would be predicted of the development had negative impacts such as increase mortality (e.g., through turbine strikes or foraging habitat loss) and/or reduced productivity (e.g., through decline in prey fish or foraging habitat loss).

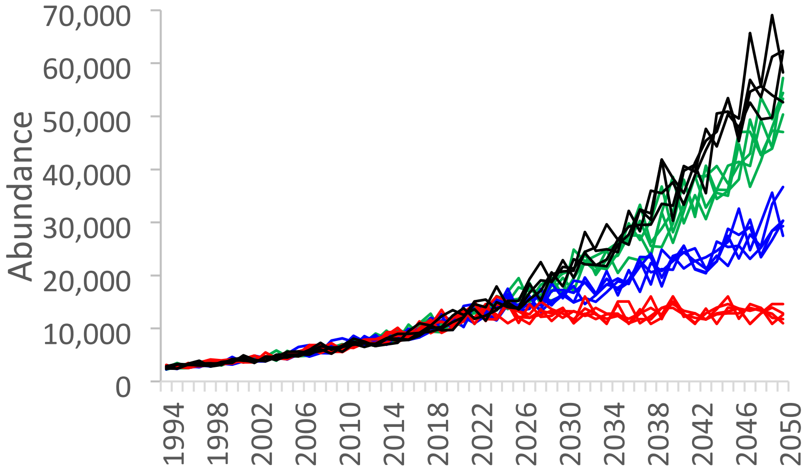
In this example we used existing Guillemot whole colony count data for South Stack to explore our ability to detect significant change in population growth rate relative to time since tidal generator development began for two different dimensions. First, the magnitude of change in annual population growth (5%, 3% and 0%, representing declines of 1%, 3% and 6% of the observed 6% per annum growth rate since 1994), and second, the frequency of colony count estimates (annual, every three years and every five years). Our approach was based on that of Cook et al. (2019).

We did this by comparing the fit of two different statistical models of temporal trends to simulated data for the period 1994 – 2050, covering both the historic pattern and a typical 25-year life of a renewable power generation development (BTO ref). Simulated data followed the observed 6% annual increase from 1994-2023, but after then the rate slowed to 5%, 3% and zero representing increasing magnitudes of impact on Guillemot abundance (Fig.2.4.). Model-1 only contained a YEAR term which generated a function with a common slope and intercept across all years, while Model-2 contained the terms: YEAR, DEVELOPMENT (2-level factor: NO for 1994-2023 and YES for 2024-2050) and the YEAR*DEVELOPMENT interaction, which fits trends of different slope and intercept to the two time periods, with Model-2 being increasingly supported at higher magnitudes of impact (Fig.2.4.B). If the AIC-value of Model-2 was at least two units less than that of Model-1 then it had statistical support and a change in trend over the full the time period was considered to have been detected (Cook et al. 2019).

Simulated data were generated as follows. The trend model fitted to observed data for South Stack 1994-2023 provided residuals and a fitted function (which could be used predict annual values for 1994-2050). We simulated annual data points for 1994-2050 by summing annual fitted values over this period values and randomly selected residual values. This was repeated, resulting in multiple simulated colony count datasets and we compared the fit of Models -1 and -2 for each. We also generated further multiple simulated datasets in which growth rate of fitted values was reduced by 5%, 3% and 0% post 2023 (Fig.2.4.). Finally, by selecting subsets of years post 2023 we could explore the impact on model fit of monitoring in fewer years. An appropriate number of simulated datasets would be in the order of 100-1000 (Cook et al. 2019).

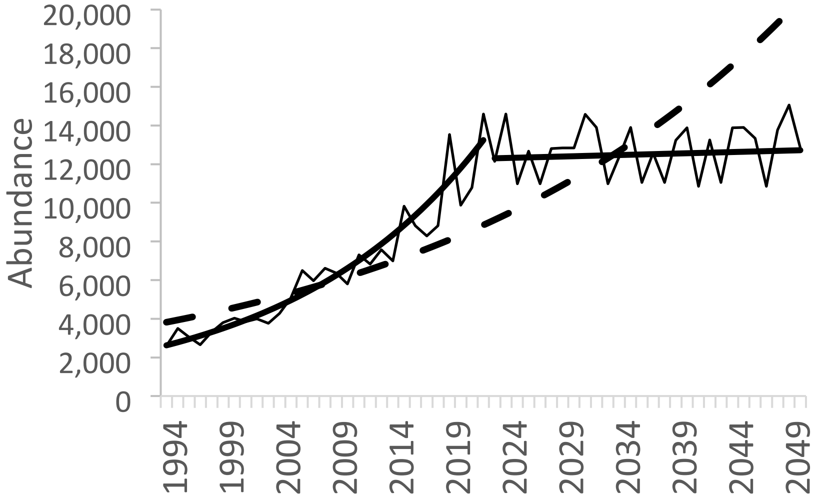
Additional complexities not specifically included are the potentially extended period over which any negative impacts may accumulate as generator designs are progressively installed in contrast to the rapid construction of offshore windfarms. This may result in any impacts in the early years of deployment being small in magnitude and harder to detect. Furthermore, the impact of HPAI on auk abundance in the coming years is unclear, as is the extent to which other colonies in north Wales have been affected and the disease may reappear next year. Separating the effects on population growth rate of HPAI from those of development impact needs some consideration.

FIGURE 6-2: SIMULATED DATA WERE USED TO ASSESS OUR ABILITY TO DETECT SIGNIFICANT IMPACT ON ABUNDANCE. A) FIVE EXAMPLE SIMULATED DATASETS FOR EACH OF I) OBSERVED 6% GROWTH (BLACK), II) 5% GROWTH POST 2022 (GREEN), III) 3% GROWTH (BLUE) AND IV) 0% GROWTH (RED). B) AN EXAMPLE SET OF SIMULATED DATA FOR ZERO POPULATION GROWTH POST 2023, WITH FITTED FUNCTIONS FOR MODEL-1 (DASHED, HERE A POOR FIT) AND MODEL-2 (SOLID, A GOOD FIT). A)



Abundance
70,000
60,000
50,000
40,000
30,000
20,000
10,000
0
1994
1998
2002
2006
2010
2014
2018
2022
2026
2030
2034
2038
2042
2046
2050

B)



Abundance
20,000
18,000
16,000
14,000
12,000
10,000
8,000
6,000
4,000
2,000
0
1994
1999
2004
2009
2014
2019
2024
2029
2034
2039
2044
2049