

Review of Curlew-focused conservation interventions in the UK and Western Europe, and assessment of intervention scenarios using population modelling

Work Package 1 of the Curlew Solutions Trial project, the result of work by the Curlew Recovery Partnership and wider project partners

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The Curlew Solutions Trial Project Summary

This document reports on work conducted for Work Package 1 of the 3-year Curlew Solutions Trial project (CST).

The Curlew is the UK's highest avian conservation priority and a key indicator of progress towards legally binding biodiversity targets under the Environment Act. Although classified as Near Threatened globally, the UK supports approximately a quarter of the world's breeding Curlew population and around a fifth of the wintering population, giving it a disproportionate responsibility for the species' global conservation status. Since 1970, the UK breeding population has declined by around 65%, driven primarily by low productivity caused by high nest and chick predation and long-term degradation of grassland habitats. Recovery at the scale required depends on effective, evidence-based agri-environment schemes (AES) that can be delivered successfully in farmed, grassland-dominated landscapes.

The Curlew Recovery Partnership (CRP) is a collaborative initiative bringing together conservation organisations, statutory bodies, land managers, farmers and communities to halt and reverse Curlew declines. The CST project was developed in response to requests from policymakers for stronger evidence to inform the design and delivery of an AES capable of supporting Curlew recovery at a national scale.

The CST is delivered through three integrated work packages: (1) synthesis of existing UK and European evidence on Curlew breeding ecology and conservation interventions; (2) stakeholder consultation and co-design of practical AES approaches with farmers and advisors; and (3) a multi-year field trial of the effects of fencing on nest survival, the identity of nest predators, variation in fledging success, and the response of Curlews to avian predator activity.

Project Partners

Curlew Recovery Partnership

WWT

Natural England

Curlew Country

Wild Oxfordshire / Upper Thames Wader Group

British Trust for Ornithology (BTO)

Executive summary

Eurasian Curlew *Numenius arquata* populations are in decline, largely due to low breeding productivity. This has stimulated the implementation of a large number of Curlew recovery projects in the UK (at least 50) and elsewhere in Europe, deploying conservation interventions that target increases in Curlew productivity. With several years of outcome data now collected by many projects, it is useful to review outcomes across projects and assess the degree to which these might be able to deliver conservation goals such as population stability at different scales.

In this study, we collate evidence from ongoing Curlew recovery projects on the efficacy of the conservation interventions they are deploying with the goal of improving Curlew productivity. We employ a Leslie matrix-based population modelling approach to explore whether these interventions would be sufficient to deliver population stability if applied in several hypothetical scenarios. We do this by modelling nest-level productivity rates required for population stability, and then examining, through two case studies: (1) the changes in population growth rate that could arise from investing different levels of effort into Curlew conservation interventions in different contexts, and (2) the intervention rates (as the proportion of all nest attempts in the population) that might be required to achieve population stability in different contexts.

Throughout, we model outcomes in terms of numbers of hatched chicks and subsequent survival of young from hatching to one year old, expressing nest-level outcomes in terms of numbers of hatched chicks per breeding pair. This metric makes it simpler to relate to current monitoring approaches, as monitoring nests up to the point of hatching is more straightforward, and can be carried out more reliably between projects to produce more comparable estimates, than monitoring chicks onwards to fledging. However, rates of chick survival from hatching to fledging can vary widely between different contexts, are not necessarily correlated to hatching rates or subsequent juvenile survival, and are themselves the target of some conservation interventions. Therefore, chick survival and fledging success remain critical components of demographic assessment, and the use of 'hatched chicks per breeding pair' as a metric does not remove the need for chick monitoring. Sustainability thresholds modelled using this metric depend on the assumption of average survival rates from hatching to fledging, which (by definition) will be lower than average in some contexts, and this must be taken into account.

The outcomes of ongoing projects suggest that interventions can drive significant improvements in nest success rates, but improving chick survival rates poses a greater challenge. Population modelling suggests that an average rate of 1.06 chicks hatched per breeding pair may be sufficient to achieve population stability. As above, this assumes average chick survival rates from hatching to fledging; so projects in contexts where chick survival is below average will need to achieve a higher rate of chicks hatched per breeding pair than this, and ideally deploy interventions aimed at improving chick survival, to compensate. Models suggest that this rate could be attainable through the deployment of interventions at current intensity in landscapes of relatively high existing Curlew productivity. In landscapes where baseline Curlew productivity is lower, our results suggest that more intensive approaches than those currently being deployed may be necessary in order to achieve population stability.

Key remaining barriers to realising population stability and growth in Curlew populations through conservation interventions include understanding how best to compensate landowners and land managers for the costs of implementing interventions in order to maximise their uptake across a large proportion of the available areas of potentially suitable habitat. Natural England's proposed Wader Recovery Areas appear to offer one viable means by which to ensure that intervention effort is targeted towards areas of relatively high importance to Curlew, where their impact on population growth rates may be largest.

1. Introduction

1.1. Study background

Eurasian Curlew *Numenius arquata* (henceforth, Curlew) is considered to be Near Threatened globally (BirdLife International 2023) and is Red-listed in the UK (Stanbury *et al.* 2021), in light of substantial declines in breeding populations (Heywood *et al.* 2023). Survival of adult Curlews is high, and has increased in recent decades (Taylor & Dodd 2013, Cook *et al.* 2021), and so the main driver of the observed declines is thought to be low productivity (Cook *et al.* 2021, Viana *et al.* 2023), most often attributed to loss of nests and chicks to predation and/or destruction by agricultural operations (Brown *et al.* 2015, Ewing *et al.* 2023a).

In light of this, a number of Curlew recovery projects have commenced in recent years (particularly in the UK and Western Europe), aiming to halt population declines through conservation interventions often targeted at increasing productivity (Wilson *et al.* 2020, Laidlaw *et al.* 2021, Douglas *et al.* 2023). Some such projects have now been applying interventions for several years and collecting outcome data on Curlew productivity under intervention regimes, and some have begun to report their outcomes (e.g. Douglas *et al.* 2023). However, to date, there has been no holistic review of the outcomes of these projects and efficacy of their interventions.

It also remains unclear whether improvements in productivity as an outcome of Curlew recovery projects will be sufficient to deliver key conservation goals; in particular, stabilisation or reversal of recent population declines. One reason for this is the substantial difficulty of relating nest-level observations of productivity to population-level outcomes. Some Curlew projects use productivity thresholds estimated by population modelling as targets against which to measure success. Three such thresholds estimated recently are 0.63 (Taylor *et al.* 2020) and 0.68 (Viana *et al.* 2023) fledged chicks per pair, and 0.43 (Cook *et al.* 2021) fledged chicks per nest respectively, although the use of such estimates dates back substantially further (e.g. Grant *et al.* 1999). However, these values represent annual population means and therefore may not consider factors that are not necessarily observable at site-level or through nest observations. For example, some pairs may not make any attempt to breed in any given year; skipping breeding is poorly understood in Curlew (as indeed in most species; see Sanz-Aguilar *et al.* 2011) but evidence that it may take place in Curlews comes from recent studies that GPS-tagged birds at the pre-breeding stage (Taylor *et al.* 2020). Clearly, productivity of birds that skip breeding will be zero in that year (even if it results in improved survival or reproductive condition in following years), and productivity of birds that do make nesting attempts will need to be accordingly higher to compensate for the effect of this on the population's mean productivity. On the other hand, birds whose first nesting attempt fails may have subsequent nesting attempts if that initial failure is sufficiently early in the season (again demonstrated in the tag data of Taylor *et al.* 2020) and may succeed on their second (or sometimes even third; unpublished data collected by authors of this report) attempt. It is unlikely that all re-nesting attempts will be monitored (Verhoeven *et al.* 2020), potentially leading to underestimation of population-level productivity, although this may be compensated by an opposite tendency for nest attempts that fail early in the nesting process (i.e. during egg-laying or incubation) to be less likely to be discovered, and subsequently monitored, than those that hatch chicks.

An additional challenge with relating nest-observation data to these population-level thresholds is the need to measure productivity in terms of the number of fledged chicks (as typically expressed). Curlews are precocial (i.e. chicks depart the nest and forage independently soon after hatching), can range over large areas prior to fledging, and are cryptic in both appearance and behaviour, often favouring tall vegetation. These factors combine to make established wader monitoring methods inaccurate, particularly in high density areas, such that 'hatching success' is the metric typically used to describe breeding outcomes in Curlews (Jarrett *et al.* 2023). New approaches may facilitate more recording of 'fledging success' (i.e. whether each pair produces at least one fledged chick) in the future (Tománková *et al.* 2024). However, methods for assessing the number of fledged chicks per pair are labour-intensive, require specialist skills and licences, and/or expensive equipment (such as tags). Intensive monitoring may also raise concerns about investigator disturbance effects on chick survival (e.g. Sharpe *et al.* 2009), although it is not certain that any such effects would be negative (Grant *et al.* 1999, Ibáñez-Álamo *et al.* 2012). Therefore, productivity thresholds produced by population modelling that are expressed in terms of numbers of fledged chicks per pair can be difficult to compare to field-based metrics, which may vary depending on how they are assessed, and thus are not reliably comparable either among field studies or to modelled metrics. As a consequence, even though productivity (in birds more broadly) is typically measured as the number of chicks fledged per pair, it made

more sense when modelling Curlew populations in the context of this study to consider the number of chicks hatched per pair, treating survival between hatching and fledging as a component of the survival of young birds to one year old. Doing so would also correspond better to estimates of Curlew juvenile survival, which are chiefly based on the survival of individuals ringed as chicks, mostly within days of hatching (Robinson *et al.* 2014, Cook *et al.* 2021). Nonetheless, chick survival and fledging success are critical components of breeding success, so monitoring chick survival in the field, and if possible generating estimates of “fledging success” (Jarrett *et al.* 2023), is still very important, especially where interventions targeted at improving chick survival are being deployed.

The contribution to conservation at regional and national scales of Curlew recovery projects, which typically operate at relatively focused spatial scales and apply some interventions on a nest-by-nest basis, also remains unclear. It is unknown whether projects focused on a specific local population can have benefits over wider areas, and what proportion of a declining population needs to receive interventions in order to achieve stability at broad scales. The current Curlew distribution encompasses areas under reasonably Curlew-friendly land management where productivity is still relatively high (e.g. nature reserves, islands with low predator densities, and farmland on or adjacent to shooting estates; Calladine *et al.* 2014, Jellesmark *et al.* 2021, Baines *et al.* 2023), and areas where management has changed to be less hospitable to Curlews (e.g. switching from hay to silage production; Vickery *et al.* 2001). Curlews (which are long-lived and site-faithful) may remain in the latter areas for some years, but their productivity is likely to be substantially lower and populations may be at risk of local extinction. A key strategic question is therefore whether, at a national scale, it is better to invest effort into boosting productivity where it is currently very low, or where it remains relatively high – trading off the absolute extent to which productivity could theoretically be improved, against the difficulty of achieving those improvements in an inhospitable landscape (Ewing 2023).

In this study, we collate evidence from ongoing Curlew recovery projects on the efficacy of the conservation interventions they are deploying with the goal of improving Curlew productivity. We employ a population modelling approach to assess whether these interventions are likely to be sufficient to deliver population stability if applied in different contexts and at different scales. We do this by modelling nest-level productivity rates required for population stability, accounting for estimated skip and re-nesting rates and expressed in terms of numbers of hatched chicks rather than numbers of fledglings; and then exploring, through two case studies: i) the changes in population growth rate that could arise from investing different levels of effort into Curlew conservation interventions in different contexts, and ii) the intervention rates (as the proportion of all nest attempts in the population) that might be required to achieve population stability in different contexts.

2. Methods

2.1. Questionnaire

To collate data on interventions deployed and outcomes arising from Curlew recovery projects across the UK and elsewhere, we prepared and circulated a questionnaire with an accompanying request for before/after monitoring data. The questionnaire was sent directly to contacts known to be running Curlew recovery projects and additionally distributed more widely on the African–Eurasian Migratory Waterbird Agreement (AEWA) Eurasian Curlew International Working Group mailing list. An initial response deadline of one month was set, but responses were ultimately accepted until nearly two months after initial circulation to allow projects enough time to gather and process data.

The questionnaire posed a wide variety of questions, most of which were multiple-choice with an option to provide additional written detail. The full list of questions is in Appendix 1. However, all questions were optional and most projects responded to a subset of the whole (this was not unexpected, as the questions were written to request an exhaustive level of detail). Therefore, we only report on the results of questions for which several projects provided answers, permitting some degree of multi-project interpretation. These, collectively, cover the following topics:

1. location, landscape and habitat context of projects;
2. interventions deployed;
3. project and monitoring methods/approaches;

4. effects of interventions and overall project outcomes;
5. for some intervention types only (habitat management, nest fencing, and predator management), additional details e.g. technical approach, cost, and funding.

Sample sizes obtained through the questionnaire were insufficient to permit formal statistical analysis; therefore, we only summarise responses descriptively, supported by basic statistics and data visualisation.

2.2. Population modelling for Curlew conservation

2.2.1. Analysis of nest record data

We calculated estimates of productivity and nest success rates using data from the BTO Nest Record Scheme (NRS; Crick *et al.* 2003) for use in population modelling. We obtained all nest records between the years 1980 and 2018 for Curlew nest attempts in the UK submitted through NRS, amounting to 510 complete records at the time of our data request (May 2023). These were broadly (and relatively evenly) distributed across 41 counties in England, Scotland, Wales, Northern Ireland and the Isle of Man. Larger-than-average contributions (in terms of numbers of nest records) came from Orkney in the early 1990s ($n = 40$ nest records between 1990 and 1995), and northern England in the late 1990s ($n = 70$ nest records between 1996 and 1999 across Cumbria, Northumberland and County Durham combined), and nationally in the more recent years ($n = 66$ nest records between 2015 and 2018 across all counties). Each record included a binary variable indicating whether that nest was thought to have succeeded (i.e. hatched at least one chick), and separate variables for 'minimum young hatched' and 'highest all-contents count'.

We estimated average nest success rates simply by taking the mean of the binary success variable. This approach is likely less accurate than alternative approaches incorporating estimation of daily nest survival rates (Rotella *et al.* 2004), but these require significant quantities of data, including multiple nest visits per record; it was felt that the available nest record data for Curlew would not support such an analysis. Estimating productivity was also challenging, because Curlews are precocial and fledging success hard to monitor, so in our analysis of NRS data we used the term 'chick production' to refer to the number of chicks hatched per successful nest, distinct from the term 'productivity' which typically refers to the number of fledged chicks.

Some observers appeared to have been relatively cautious with regards to the 'minimum young hatched' variable, which appeared to be both zero- and one-inflated as a result. In other words, we noted that it was possible for the same nest to be recorded as successful but with the number of young hatched given as either zero or one, when the actual number may have been higher but with the observer unable to confirm the true number. To account for zero-inflation, we removed 69 nest records which had been recorded as successful, but with a minimum of zero young hatched, from analyses of chick production rates (having previously used these records for estimation of nest success rates). To account for one-inflation, we estimated average rates by fitting a null one-inflated Poisson generalised linear model (GLM) to the remaining data, rather than simply taking an average (using the R package 'VGAM'; Yee 2010). The rates we produced using this method were notably similar to some previous estimates in the literature, albeit those associated with declining populations in unfavourable conditions (e.g. Grant *et al.* 1999). Because variation in chick production rates could be a function of variation in clutch size, we also estimated mean clutch sizes using the 'highest all-contents count' variable.

Guided by our intention to include estimates of rates of re-nesting in our population models, we also investigated whether chick production and nest success rates changed over the course of the breeding season for nests with later initiation dates, which were more likely to be re-nesting attempts after initial failure(s). Nest records contained two estimates of first egg date (FED), which estimate the earliest and latest possible FED respectively (Crick *et al.* 2003). Standard procedure to estimate FED would be to take the midpoint between these dates, but for the purposes of this analysis we used the latest possible FED (FEDL), since a very large proportion of nest records (430 out of 510) had implausibly early estimates of the earliest possible date (FEDE), probably because Curlew nests are often not located until after the first egg or eggs have been laid. FEDL alone is unlikely to provide an accurate estimate of the true FED, but in this case it should be sufficient to enable a distinction between early- and late-initiated nests to be made, with greater accuracy than either FEDE or the midpoint.

We fitted GLMs to describe the relationships between FEDL and each variable describing nest success/chick production: nest success rates (with a binomial error distribution), and minimum young hatched, including both all records (with a zero-inflated Poisson error distribution) and only successful nests (with a one-inflated Poisson error distribution), and highest all-contents count (with a Poisson error distribution). Models were validated by inspection of residuals and significance tested using Likelihood Ratio Tests. We also took weekly mean values across the duration of the breeding season (Table 2; note that we used means, rather than one-inflated null models, to calculate weekly chick production rates for this table, due to the small sample sizes in most weeks).

Based on our estimates for nest success rates and re-nesting rates after failure (see below), we then calculated the expected proportion of nesting attempts that should represent first, second and third attempts respectively. We divided nest records into these proportions chronologically by FEDL, based on the assumption that first attempts are relatively synchronous in Curlew and re-nesting attempts must, by definition, be initiated later in the breeding season. In practice, this meant that we assigned nest records with an FEDL between 31 March and 12 May as first attempts ($n = 336$), those with an FEDL between 13 and 30 May as second attempts ($n = 141$), and those with an FEDL between 31 May and 18 June as third attempts ($n = 33$), based on the findings of a GPS-tagging study in North Wales (Taylor *et al.* 2020; see below). Note, however, that nest record data are likely to contain a greater proportion of first attempts than the true proportion due to a decline in nest-recording effort later in the breeding season, but simultaneously, a proportion of first attempts are likely to have gone undetected by nest recorders (Verhoeven *et al.* 2020). We then re-estimated average nest success and chick production rates for first, second and third nest attempts respectively, using the same approaches as for full-season rates.

2.2.2. Population modelling

Population modelling was conducted using a Leslie Matrix Model (LMM) approach, which draws on a range of demographic rates to estimate year-on-year changes in the number of individuals within a population in different age and breeding classes.

Previous studies using the same or related approaches to model Curlew populations have typically been based on estimates of average productivity per nesting attempt and age-specific survival only (e.g. Taylor *et al.* 2020; Cook *et al.* 2021). For this study, we built LMMs that explicitly incorporated estimates of nest success rates and re-nesting attempts after initial failure, as well as rates of skipping breeding, because we felt that these rates might have some influence on outcomes of Curlew conservation interventions, especially if those interventions reduced rates of nest failure (and thereby increased the relative proportion of nest attempts that were first attempts).

2.2.2.1. Parameters used in population modelling

Parameters for modelling were variously (a) drawn from the literature, (b) estimated by our analysis of NRS data (above), and (c) calculated from the dataset of Taylor *et al.* (2020) as follows.

Survival rates for different age classes were obtained from Cook *et al.*'s (2021) survival analysis for Curlews ringed in the UK under the British and Irish Ringing Scheme. We used their estimates for post-1996 survival based on breeding-season ringing data for both adults (0.922, CI 0.886–0.948) and young birds (i.e. those in their first year of life; 0.390, CI 0.304–0.484). Cook *et al.* (2021) concluded that immature survival (i.e. between one and two years old) was not different to adult survival, so we applied adult survival rates to immature birds in our models.

For young birds, the survival estimate is based on a sample of 14,227 individuals ringed as chicks (Cook *et al.* 2021). Curlews can be ringed as chicks almost immediately after hatching, and indeed most individuals are likely to be ringed within the first few days of life (e.g. Grant 2002), when they are easier to find (still being relatively close to the nest) and to capture. Nevertheless, if we assume the mean ringing age for Curlew chicks to be a day or two after hatching, this leaves an unestimated component of survival for the period immediately after hatching, during which mortality may be particularly high (due to e.g. high vulnerability to predation or poor weather conditions) but Curlew conservation interventions may be particularly effective (because chicks may still be in the zone impacted by interventions focused on nest locations). We set up our LMMs to use a rate of 'young survival' which represents survival from hatching to one-year old, and therefore implicitly is a factor of both the unknown rate of survival immediately post-hatching and the estimated rate

of survival from day of ringing to one-year old (Cook *et al.* 2021), each of which in turn has some overlap with the rate of survival between hatching and fledging. Since young survival rate included this unknown component, we used it to allow the calibration of models to known population trends (see below).

We calculated breeding skip rates and re-nesting rates after failure based on the dataset of Taylor *et al.* (2020), who fitted GPS tags to a sample of 23 adult Curlews caught across three study sites in North Wales over the course of several years, of which 20 individuals tagged in 2018–2019 were considered to have been caught during the pre-breeding phase. From this sample of 20, five birds did not go on to make any attempt to breed (a skip rate of 0.25). Twelve birds failed on their first breeding attempt, of which seven went on to make a second attempt (a re-nesting rate of 0.58). Only three birds definitely failed on their second attempt, of which only one made a subsequent third attempt (a re-nesting rate after two failures of 0.33). Since the sample size from which we calculated these rates was so small, and behaviours could plausibly have been impacted by catching and handling (which in some birds took place during the typical time window for first nest attempts, albeit in all cases fieldworkers were confident that birds targeted for capture had not yet commenced a breeding attempt; pers. comm.), we explored the effects on our models of varying these rates. We varied skip rate between 0.00–0.35 (i.e. if zero to seven birds had skipped breeding, rather than five), because the possibility that a) skipping in this sample was entirely due to catching effects, or b) fieldworkers had in fact not detected early, failed first attempts prior to tagging (Verhoeven *et al.* 2020), could not be entirely ruled out. We varied the re-nesting rate after one failure between 0.42–1.00 (i.e. if five to 12 birds out of 12 had re-nested, rather than seven), because studies of the related Black-tailed Godwit *Limosa limosa* found that 100% of birds re-nested after a failed first attempt (Verhoeven *et al.* 2020), and it was similarly felt that the impact of catching and tagging could plausibly have depressed this rate. We varied the re-nesting rate after two failures between 0.10–0.50 with similar logic; in Black-tailed Godwits this rate was 43% (Verhoeven *et al.* 2020).

Nest success rates, and the number of hatched chicks per successful nest, for first, second, and third nest attempts were those calculated during our analysis of NRS data (above).

2.2.2.2. Population modelling and calibration

We parameterised LMMs using the rate estimates described above, which were intended to reflect average rates within the UK Curlew population (as far as possible, given the limitations of existing knowledge and available data). We then used the estimated population trends for Curlews in the UK from the BTO/JNCC/RSPB Breeding Bird Survey (BBS) to calibrate our models, by varying the rate of young survival (from hatching to one year old) until models estimated population growth rates consistent with BBS trends. The discrepancy between the resultant rate of young survival and the existing estimate of survival from ringing to one year old (0.390, from Cook *et al.* 2021) allowed us to generate an estimate of the rate of post-hatching survival (albeit one that is contingent upon the accuracy of all other estimated rates, including population trend).

We repeated this calibration process using both the UK trend (7% decline over the last 10 years; Heywood *et al.* 2023), for a model representative of the wider picture of Curlew conservation across the substantial populations of England and Scotland, and also the Wales trend. Modelling the latter allowed us to use the example of Wales during Case Study 2 (see below) in order to explore the levels of intervention that might be necessary to recover smaller populations where local declines may be relatively more severe; the Welsh population represents only 3% of breeding Curlew in Britain (Calladine *et al.* 2022) but has undergone a decline of some 40% in the last 10 years (Heywood *et al.* 2023).

We then recalibrated our models again to a population growth rate of one (i.e. stability), by varying in turn both (a) young survival rate and (b) chick production rate (i.e. the product of the number of chicks hatched per successful nest, and the rate of nest success), to understand combinations of rates of chick production and young survival that would be expected to result in population stability, and the degree of uplift required in one or the other compared to current estimated rates (all else being equal). As above, we explored how this was influenced by variation in one source of potential error in our models, by varying skip rates between 0.00–0.35.

2.3. Case Study 1: proposed Wader Recovery Areas

2.3.1. Proportions of breeding wader populations in proposed Wader Recovery Areas

Natural England has drawn up a provisional suite of Wader Recovery Areas (WRAs) that could be made

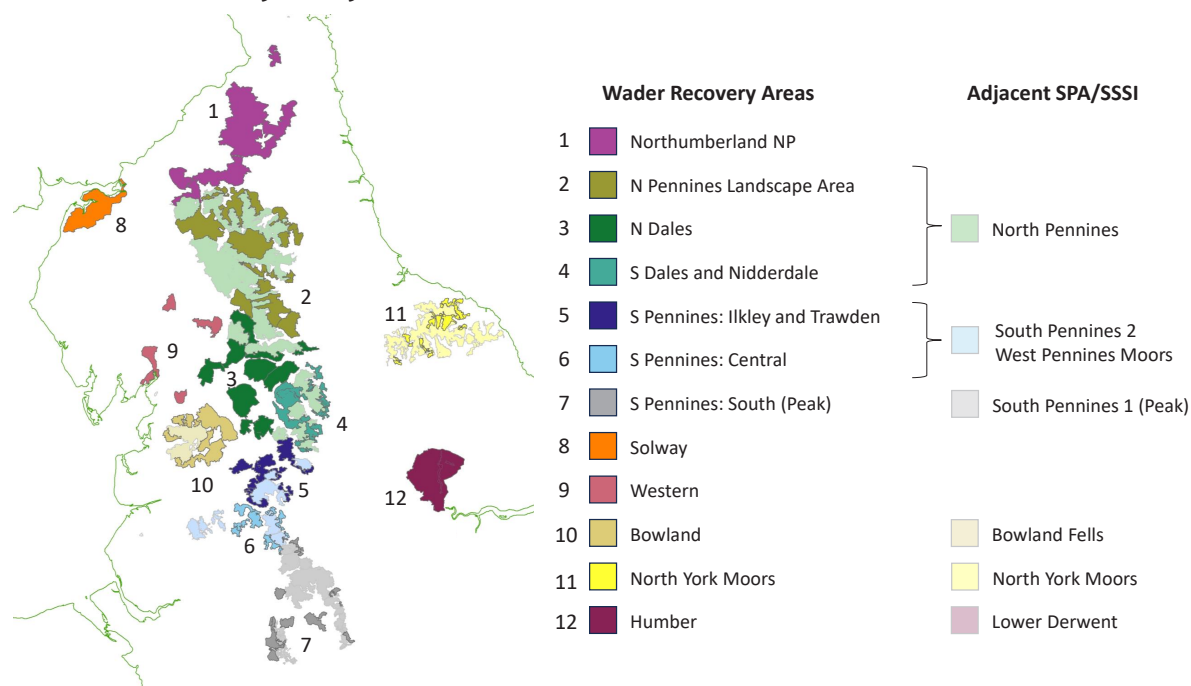
subject to a (yet to be decided) range of measures aiming to increase the breeding productivity of wader populations in these areas, by regulating and incentivising favourable types of management and land use. Within the north of England, these comprise 76 discrete areas ranging in size from less than 1 km² to more than 450 km² at the time of analysis (with a further set of WRAs being drawn up for the south of England). Most of these areas are in and around National Parks or adjacent to existing Special Protection Areas (SPAs) in the Pennines, Northumberland and the North York Moors, and likely comprise a mix of hay meadow, silage, pasture and moorland. However, some WRAs are in more lowland landscapes, such as the Lower Derwent and the South Solway coast. Figure 1 shows all WRAs and adjacent SPAs and relevant Sites of Special Scientific Interest (SSSIs), grouped into 12 geographically-based clusters (Table S1).

We used modelled relative abundance at 1-km² resolution from Calladine *et al.* (2022), which is based on bird atlas data (Balmer *et al.* 2013), to calculate relative breeding abundance for six species of wader (Curlew, plus European Golden Plover *Pluvialis apricaria*, Northern Lapwing *Vanellus vanellus*, Eurasian Oystercatcher *Haematopus ostralegus*, Common Redshank *Tringa totanus*, and Common Snipe *Gallinago gallinago*) in each of the 12 WRA clusters, as well as in five adjacent SPAs and one relevant SSSI. These abundance scores are based on the numbers of birds counted within tetrads during one hour, so are simple indices of abundance rather than inherently meaningful measures. Also, because these counts were carried out between 2008 and 2011, they reflect patterns of abundance around 2010. See Calladine *et al.* (2022) for an explanation of how the 1-km² resolution abundance scores were derived. Relative abundance scores for each area were calculated by summing the abundance scores of the 1-km squares they overlapped, with partially overlapping squares contributing according to the proportion of overlap (e.g. if a square had a score of 1.3 and 0.64 of the square was within an area, its contribution to the abundance score for the area would be $1.3 \times 0.64 = 0.83$). The abundance scores for each area were used to calculate two metrics:

1. relative densities (calculated by dividing relative abundance by land area) which can be used to compare average densities of each species across the different areas; and
2. proportional abundances (calculated by dividing relative abundances for the area by relative abundances for the whole of England) which indicate the approximate proportion of the breeding population of each species in England found within the relevant area.

Relative abundance scores were also used to generate heat maps showing patterns of relative abundance for each species in and around the WRAs. Two maps were produced for each species, both showing heat maps across the whole of northern England. In one of the maps, patterns of abundance within WRAs are highlighted (with heatmap colours outside of the WRAs muted), illustrating spatial variation within and between WRAs in concentrations of these wader species. The enlarged highlighted areas in the second map comprise adjacent protected areas (SPAs and an SSSI) as well as the WRAs, showing what would be included within the overall suite of designated/protected areas if all proposed WRAs were confirmed. In both maps, muting of the colour gradients outside the WRAs was achieved using a transparent white mask (such that patterns are still visible, but less prominent than those within the areas of interest), but patterns across the entire country without this visual effect can be found in Calladine *et al.* (2022).

Figure 1. Proposed distribution of northern Wader Recovery Areas (WRAs), as provided to BTO by Natural England, in relation to existing SPAs (and, where relevant, SSSIs). The 81 individual WRAs have been regionally grouped by BTO for ease of interpretation; the size of each WRA and the regional grouping to which it has been assigned is given in Table S1.



2.3.2. Modelled scenarios for interventions in Wader Recovery Areas

To model the potential outcomes that might arise from Curlew-focused interventions within the proposed Wader Recovery Areas, we designed several scenarios based on hypothetical rates of nest success/failure and young survival/mortality (Table 4). Specifically, we designed three 'baseline' scenarios, representing areas of low, medium and high existing Curlew productivity, based on outcomes of our questionnaire and the literature. We then designed four 'intervened' scenarios, each explicitly linked to a given baseline scenario, again informed by the outcomes of our questionnaire. We do not specify which particular intervention types might lead to these outcomes in terms of nest success and young survival rates. Rather, these analyses should be viewed as thought experiments, which demonstrate the scale of change in population growth rate that could be achieved if a package of interventions, comparable to those currently being applied by Curlew recovery projects, were to lead to the described changes in demographic rates.

The low productivity baseline was informed by questionnaire responses, and specifically rates of nest success and chick survival in some of the worst baseline cases among responding projects. We assigned a nest success rate of 0.1 and a young survival rate of 0.05 to this scenario.

The medium productivity baseline was intended to reflect the national averages used to parameterise our population models, including the estimate of young survival produced in our analysis by calibration to the UK trend. Therefore, we assigned a nest success rate of 0.25 and a young survival rate of 0.3 to this scenario.

The high productivity baseline was directly informed by Baines *et al.*'s (2023) study of Curlew productivity in farmland bordering areas used for grouse shooting, and thereby subject to intensive regimes of predator control. This study reported a nest success rate of 0.81, and 'fledging success' as the proportion of pairs that fledged at least one chick (0.66), from which we derived an estimate for young survival rate of 0.5 (based on the assumption that successful broods are likely to fledge a large proportion of hatched chicks, which we estimated at 0.75)..

Our first scenario applied interventions at high intensity to the low productivity baseline. Rates post-intervention were informed by the outcomes reported in responses to the questionnaire for Curlew recovery projects with relatively low-productivity baseline data; specifically, we identified values that were within

the range of outcome rates reported by the subset of projects that had very low baseline rates (see Section 3.1.4.). We applied an increase in nest success rate to 0.6 (an increase of 500% relative to the baseline of 0.1) and a doubling of young survival rate relative to the baseline, to 0.1, reflecting the questionnaire's finding that increasing reproductive output may be more tractable through nest-focused interventions than increasing subsequent chick survival.

Our second and third scenarios respectively applied interventions at low and high intensity to the medium productivity baseline. Again, intervened rates were informed in part by outcomes reported in response to the questionnaire; we selected values that were within the range of outcome rates reported by projects that had intermediate baseline rates (see Section 3.1.4.). Under the low intensity/medium productivity scenario, we applied an increase in nest success rate to 0.5 (an increase of 100% relative to the baseline of 0.25) but young survival rate was unchanged (remaining at 0.3), again reflecting the apparent difficulty of improving the prospects for Curlew chicks after hatching. Under the high intensity/medium productivity scenario, we applied an increase in nest success rate to 0.75 (an increase of 200% relative to the baseline of 0.25) and an increase in young survival rate to 0.4, the midpoint between the medium and high productivity baseline rates (an increase of 33% relative to the baseline of 0.3).

Our fourth and final scenario did not model application of interventions so much as their removal, reflecting the possibility that favourable management (from a Curlew's perspective) may reduce or cease in existing high-productivity landscapes, for example through a reduction in predator management effort or the planting of trees on or near Curlew breeding grounds (Crowle *et al.* 2022, Baines, 2025). These changes were largely hypothetical, because although Curlews have been shown to decline in local abundance following cessation of favourable management (Baines *et al.* 2008, Baines, 2025), there is (to our knowledge) no evidence for the direct effects on nest success or young survival in waders. We applied the logic that landscapes that had been under favourable management may retain some benefits by comparison to medium-productivity landscapes, at least for a period of time (because, for example, it may take some years for predator populations to recover to average densities after the cessation of predator control); and so that this scenario should be intermediate between the high- and medium-productivity baselines. Therefore, under this scenario, nest success rate and young survival rate both decreased to the midpoints between the medium and high productivity baseline rates, which were 0.53 for nest success (a decrease of 35% relative to the baseline of 0.81) and 0.4 for young survival (a decrease of 20 % relative to the baseline of 0.5).

We then constructed LMMs using each pair of rates for each of the baselines and intervened scenarios, and estimated resultant population growth rates. We also explored the uncertainty in outcomes that arose from allowing one variable to vary; in this case, by changing adult survival rates to the 95% confidence interval of 0.886–0.948 reported by Cook *et al.* (2021).

2.4. Case Study 2: intervention rates to stabilise population declines

To model the potential levels of effort required (in terms of proportion of all nests having interventions applied) in order to stabilise Curlew population declines, we carried forward the three intervened scenarios from Case Study 1 that applied positive interventions to low and medium productivity landscapes, and estimated the proportion of nests that would need to receive the defined interventions in order for the resulting population trend to be stable, at both UK-scale (a large population, in decline) and Wales-scale (a smaller population, in much steeper decline).

We set up LMMs to explicitly model different rates of nest success and young survival depending on whether nests received interventions or not, with the ability to vary the intervention rate between zero and one. Non-intervened nests had the same rates as estimated by calibrating models to the UK and Welsh trends respectively, such that setting the intervention rate to zero produced a population growth rate that reflected the current BBS trend at each scale. Intervened nests had the rates of nest success and young survival associated with each intervention scenario (above, and Table 4) applied to them. We then varied the intervention rate until models estimated a population growth rate of one (i.e. stability).

3. Results

3.1. Survey of Curlew recovery projects

3.1.1. Summary of responding projects

Responses to the structured questionnaire were received from 28 projects. Of these, 12 projects provided detailed outcome data (with or without accompanying baseline data). One additional project provided detailed baseline and outcome data, but without having responded to the structured questionnaire, and one further project which had commenced in the breeding season of 2024, after the questionnaire closed, provided an interview at the conclusion of that breeding season. Responses came from countries across western Europe, with the majority from the UK (Figure 2). Within the UK, responses were received from projects in both low- and high-elevation landscapes and from all four constituent nations.

Projects encompassed a range of habitat types, including arable, heather moorland, and lowland bog, but the majority were situated in grasslands. The majority of these indicated that their study areas were subject to low intensity management, such as hay production or extensive grazing, rather than more intensive management. Most projects (20 out of 28) indicated that local declines in the number of breeding pairs were a key motivation for commencing their Curlew recovery project, with a similar number citing a desire to address known causes of nest or chick loss (evenly split between losses to agricultural operations and losses to predation – 10 out of 28 projects each). Very few projects cited a lack of data or knowledge as a reason for commencing their project (2 out of 28).

Several projects reported that the identity of predators of Curlew eggs and chicks in their project area was unknown. Among projects that did identify predators, Red Fox *Vulpes vulpes* was named by the most projects as predating eggs (9 projects, out of 14 responses), with smaller numbers identifying Carrion/Hooded Crow *Corvus corone* (3 projects), and one project each reporting egg predation by: European Badger *Meles meles*, Beech Marten *Martes foina*, Least Weasel *Mustela nivalis*, Sheep *Ovis aries*, snakes, Northern Raven *Corvus corax*, Western Jackdaw *Coloeus monedula*, European Herring Gull *Larus argentatus*, Western Barn Owl *Tyto alba*, and Western Marsh Harrier *Circus aeruginosus*. Several projects highlighted greater uncertainty around predators of chicks than of eggs. Nonetheless, most projects again highlighted Red Fox as an important chick predator (8 projects, out of 12 responses). A larger number of projects mentioned birds as predators of chicks than of eggs, including Carrion/Hooded Crow (6 projects), Northern Raven, Western Jackdaw, European Herring Gull, Common Buzzard *Buteo buteo*, Western Marsh Harrier, Red Kite *Milvus milvus*, and Common Kestrel *Falco tinnunculus* (one project each).

3.1.2. Interventions deployed by projects

The vast majority of projects deployed multiple interventions aimed at Curlew conservation (or in some cases, wading bird conservation more generally). Interventions consisted of a range of forms of land management, predator-exclusion fencing, and lethal control of mammalian and/or avian predators (Figure 3).

Fencing and mammalian predator control were the most-frequently deployed interventions, with each applied in more than three-quarters of projects, and more often deployed specifically for Curlew conservation than for other or more general purposes, although there was considerable overlap, since many projects reported deploying the same interventions with both Curlew and other purposes in mind (Figure 3a). Avian predator control was similarly deployed specifically for Curlew conservation in most cases (and also other purposes in about half of projects that deployed it), but was utilised by only around half of projects. Whilst Curlew-friendly management of farmland, unfarmed land, and water were all deployed in between a quarter and three-quarters of projects, they were just as frequently carried out for other reasons as for Curlew conservation. Several projects used the write-in 'other' option to indicate that they carried out activities aimed at reducing predator perches or cover (e.g. scrub management); in most cases, this was for both Curlew conservation and other purposes simultaneously (Figure 3a).

Projects estimated variable degrees of coverage in terms of the percentage of pairs within their study area that benefited from interventions. Not all projects responded to this question; of those that did, approximately one-third estimated near-complete coverage, a further one-third estimated over 70% coverage, and most of the remainder estimated between 30% and 60% coverage (Figure 3b).

3.1.3. Project and monitoring methods/approaches

In general, responses to questions around monitoring suggested a considerable upscaling of effort associated with the commencement of Curlew recovery project interventions. Most project monitoring was conducted by a combination of volunteers and paid project staff (Figure 4a), but with a relatively greater contribution from paid staff towards monitoring after interventions began. Similarly, a greater uptake of relatively high-effort monitoring approaches is evident for post-intervention monitoring compared to baseline data. For nest monitoring, this included significant post-intervention deployment of area searching (to locate breeding adult Curlews), nest searching and monitoring, and revisits to conduct chick monitoring (Figure 4b). Nest-focused monitoring methods similarly included an increase in usage of nest cameras and temperature loggers, and analysis of egg remains, including with DNA-based approaches (Figure 4c).

3.1.4. Effects of interventions

Among projects that made estimates of the causes of nest failure and chick mortality, estimates were highly variable both before and after the deployment of interventions (Figure 5). Most projects estimated considerable reductions in the proportion of nests lost to agricultural operations and predation by known predators (mammalian and avian) after deployment of interventions; however, many reported high rates of predation by unknown predators after intervention, which was not the case before intervention (Figure 5a). Projects also reported considerable reductions in the proportion of chicks lost to agricultural operations after intervention (Figure 5b), but there was no consistent pattern for predation, with as many projects reporting increases as decreases in chick predation after intervention (Figure 5b).

These patterns were reflected in overall rates of nest success and chick survival that were reported by projects, consistently across both estimated outcomes and among projects that provided monitoring data (Figure 6). Most projects reported having a positive impact on rates of nest success, often with very large effect sizes (Figure 6a). However, rates of chick survival remained very low after intervention in virtually all projects (Figure 6b). There was no evidence that implementing a greater number of different types of intervention led to more positive outcomes.

3.1.5. Assessments of individual intervention types

3.1.5.1. Habitat management

In most projects, grassland management consisted of reducing conflict between agricultural operations, including cutting of hay/silage and grazing, and nesting Curlews. This was principally achieved either through delaying the timing of grass cutting, or by land-sparing (leaving areas uncut and ungrazed). Spared areas ranged from nest-level protection zones of just a few square metres to larger areas at field- or site-scale. Projects in some countries reported being able to receive subsidies to support land sparing, which were typically valued at between €500–€1,500 (approx. £400–£1,300) per hectare per year.

Relatively few projects used water management for Curlew conservation, and where used it was mainly reported to consist of winter flooding of grassland habitats. Although this was not typically a Curlew-focused intervention, some projects reported that it may improve feeding habitats for Curlew. Scrub management was typically deployed to reduce the availability of predator perches for corvids and raptors near breeding areas for Curlew and other waders. No project reported receiving financial support for either water or scrub management.

3.1.5.3. Fencing

The vast majority of projects deployed fencing in one form or another. In most cases this consisted of stranded, electrified fencing deployed temporarily around individual nests. A small number of projects deployed physical barrier fencing in addition to, or instead of, electric fencing. A few projects instead deployed fencing (of either type) at field- or site-scale, but this was typically reported to be targeted at benefiting Lapwings (which often nest colonially) and was always deployed in addition to individual nest-level fencing targeted at Curlews (it was unclear from responses whether this referred to secondary fencing of individual nests within fenced fields/sites, or fencing of additional Curlew nests situated outside the field- or site-scale fencelines).

Most projects deploying fencing around individual nests fenced areas of either 20 x 20 m or 25 x 25 m.

Some projects deployed smaller areas (down to 10 x 10 m), and one larger (30 x 30 m). Fencing was always implemented as soon as possible after nest discovery, and variously left in place until a few days or weeks after hatching. One project explicitly reported two instances of nest abandonment immediately following the installation of fencing (S. Kenyon, pers. comm.); project staff believed that these abandonments were likely to be attributable to a combination of larger-than-usual teams installing fencing (eight people, rather than the usual three to four), fencing being installed immediately prior to the onset of poor weather conditions (thereby increasing stress to the parent birds), and in one instance possibly the usage of unusually high-density electric fencing (eight strands, rather than the usual strands, creating a more visually imposing barrier).

Reported costs of fencing were highly variable between projects, even when deploying similar-sized fenced areas – generally, projects reported costs in the region of £500–£1,000 per nest for equipment. However, several projects indicated that the majority of the cost of this intervention is associated with initial purchase of equipment, with subsequent maintenance and reuse in following breeding seasons being comparatively inexpensive. Staff costs could be considerable (one project reported €3,000, approx. £2,500, per year, to be the cost of staff time to fence nests), but this depended on other factors including volunteer availability and the number of nests to be fenced. Some projects reported having been able to receive subsidies through agri-environment schemes which covered at least some of the cost of fencing equipment.

3.1.5.4. Predator control

Mammalian predator control was undertaken with variable timing by different projects, but always initiated well in advance of the Curlew breeding season. Primarily, this consisted of shooting Red Foxes *Vulpes vulpes* (hereafter Fox), although some projects also reported trapping mustelids. Several projects reported relying on local farmers or shoots for Fox control, rather than carrying it out themselves.

Avian predator control was conducted by fewer projects, generally targeting Carrion Crows *Corvus corone* and occasionally Eurasian Magpies *Pica pica*. Accordingly, this control happened later in the year than mammalian predator control, to coincide with the corvid breeding season. Like mammalian predator control, several projects reported that avian predator control was carried out by local farmers or shoots, rather than explicitly as part of their project.

Figure 2. Geographical distribution of Curlew recovery projects that provided responses to the questionnaire. Panel a) shows the distribution of projects (n = 28) at country-level, and panel b) further divides UK-based projects into the four constituent nations, with England subdivided again into northern/upland and southern/lowland projects. Shades represent the landscape type in which the largest proportion of each project was described as being situated. Two projects that provided detailed data or interviews but did not respond to the questionnaire are not shown in these graphs, but were based in the Netherlands and the UK (Wales) respectively.

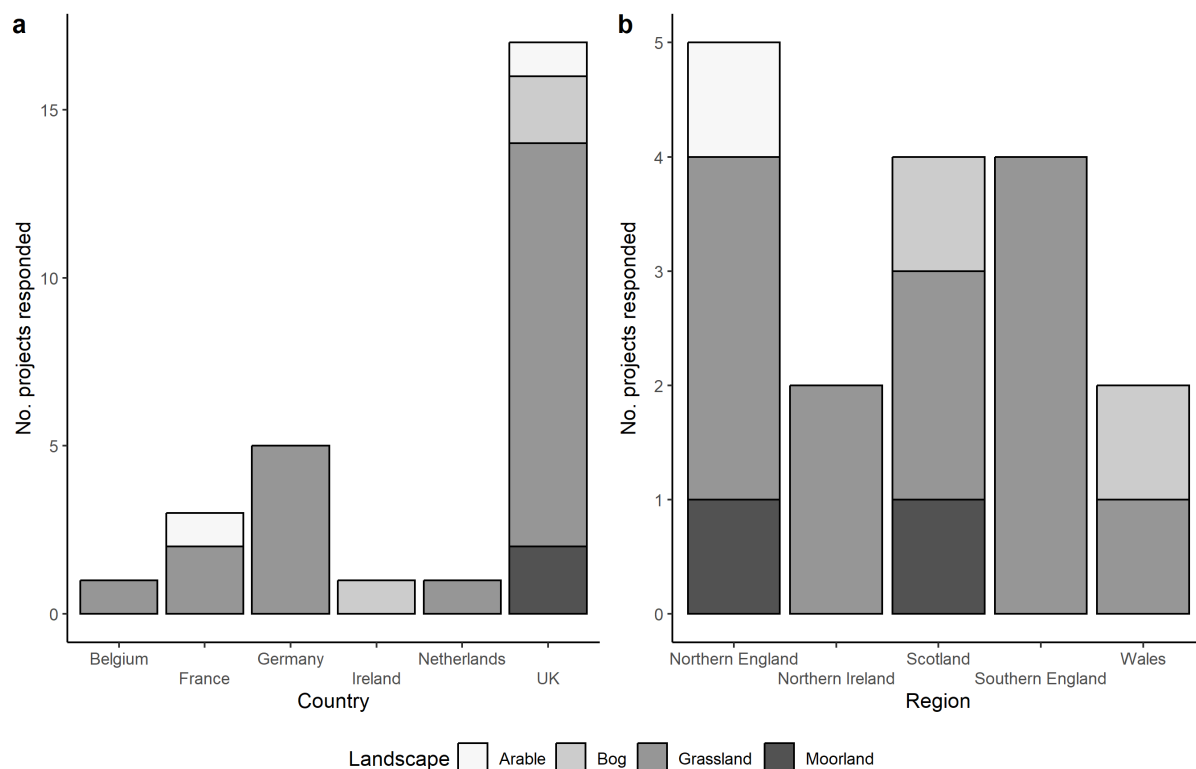


Figure 3. Interventions deployed by Curlew recovery projects. Panel a) summarises the frequency with which different broad groupings of interventions were applied by projects: projects were asked to indicate whether interventions were deployed with Curlew conservation specifically in mind (pale grey bars) and/or for other/more general reasons (darker grey bars). White shadow bars show the total number of projects deploying each intervention, regardless of motivation. Panel b) shows the estimated coverage of interventions, in terms of the percentage of breeding Curlew pairs in the study area expected to benefit. Only a subset of projects (17, out of 28 questionnaire responses) answered this question.

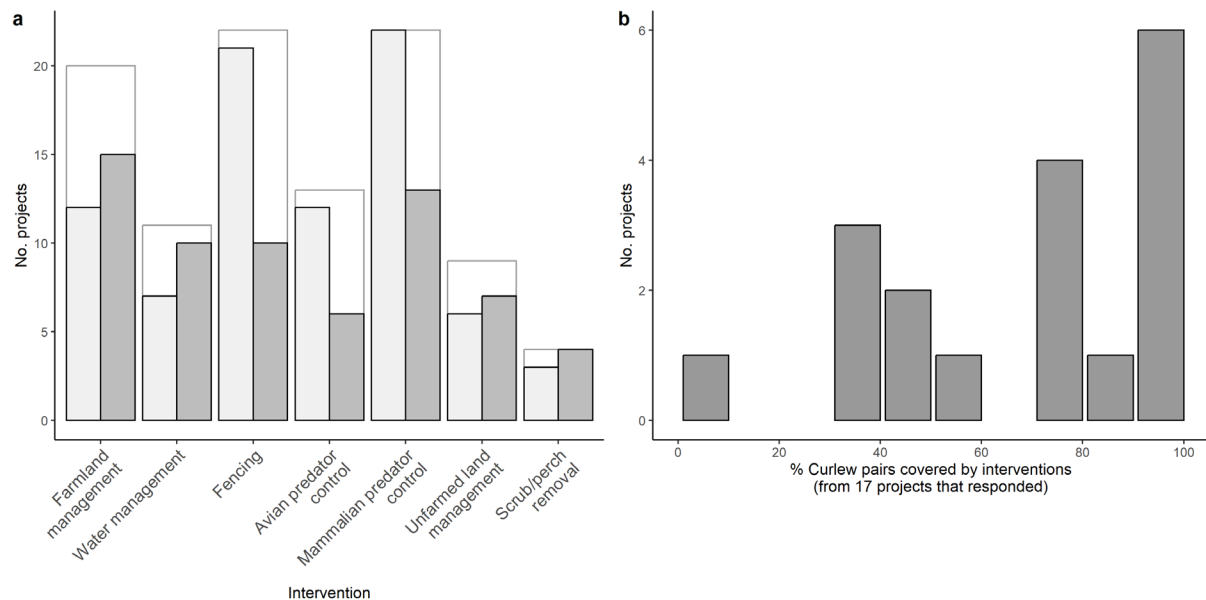


Figure 4. Approaches taken to surveying by projects. Answers referred to both before (pale grey bars) and after (darker grey bars) projects commenced interventions.

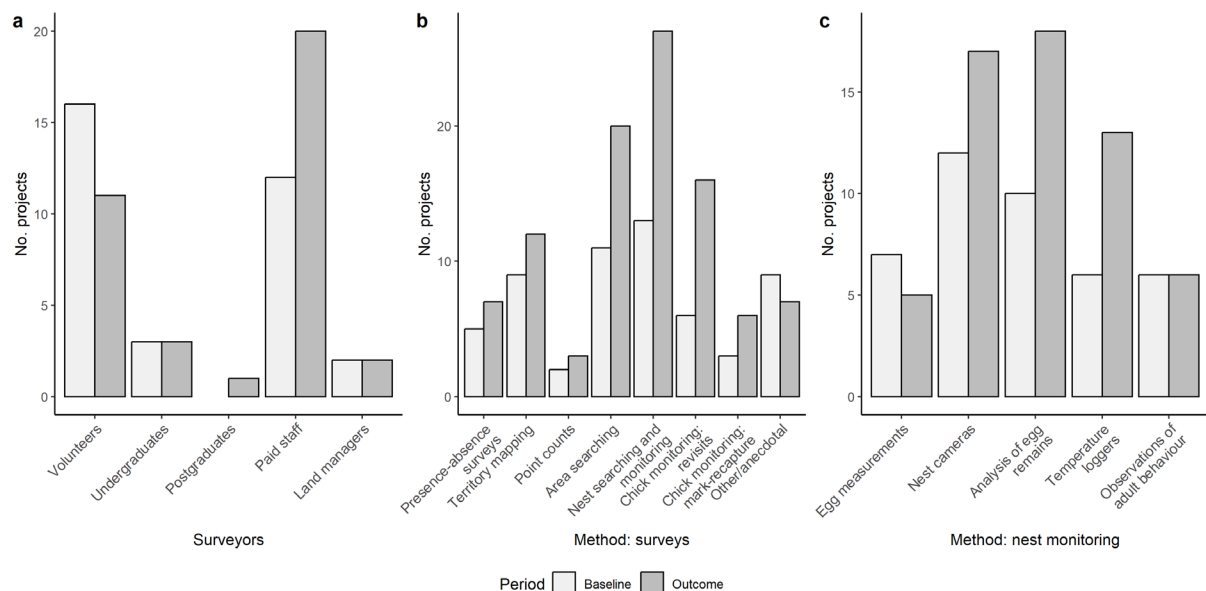


Figure 5. Rates of nest failure and chick mortality attributable to various causes, estimated by projects. Panel a) shows the percentage of nests failed and panel b) shows the percentage of chicks died due to each cause. Estimated rates are shown both before ('Baseline') and after ('Outcome') the deployment of interventions. Points are plotted with partial transparency to indicate where multiple projects estimated the same rates. Estimates made before and after by the same project are joined by lines (not all projects made estimates for both baseline and outcome for every cause).

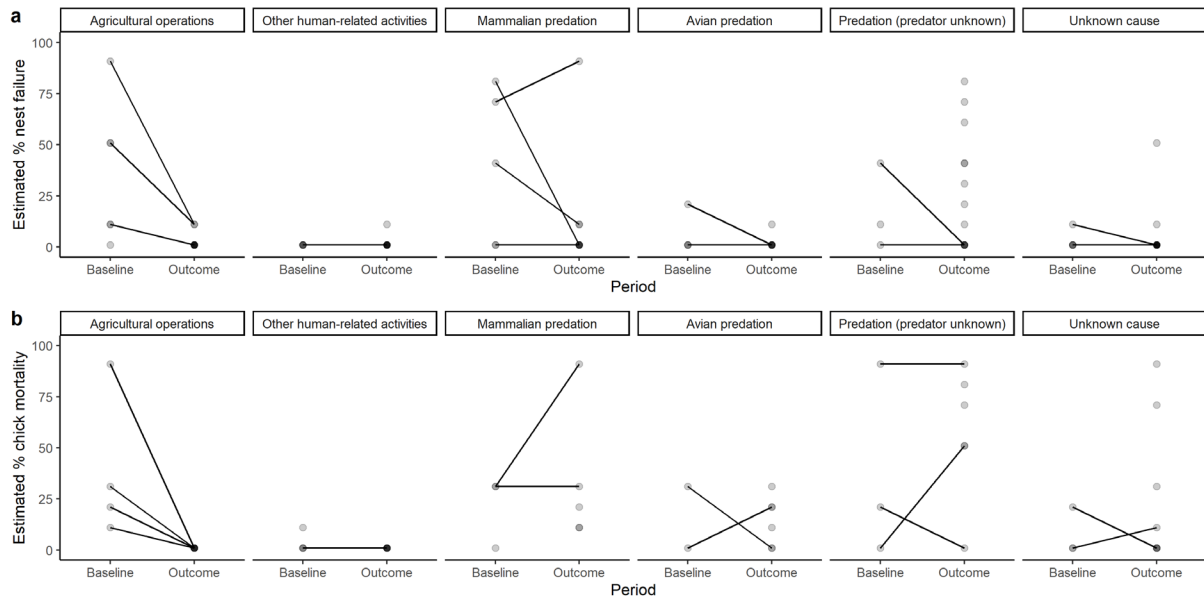
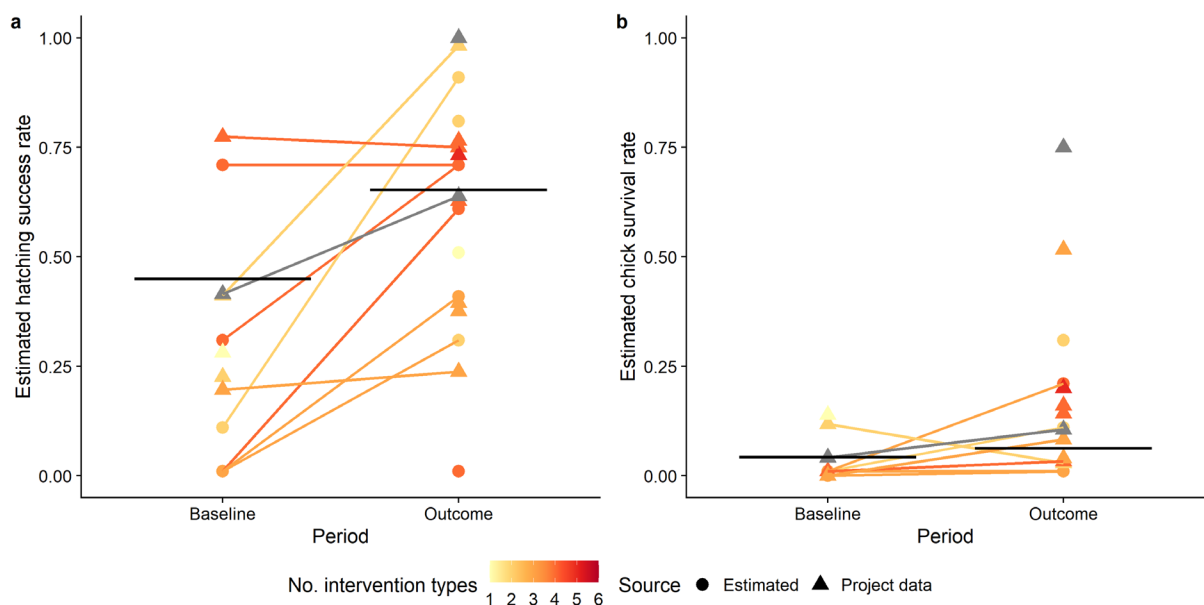


Figure 6. Overall rates of nest success and chick survival. Panel a) shows nest success rates and panel b) shows chick survival rates to fledging. Rates are shown both before ('Baseline') and after ('Outcome') the deployment of interventions. Rates before and after intervention from the same project are joined by lines. Rates were variously estimated by project staff (circles) or calculated from project monitoring data (triangles). The number of intervention types deployed by each project (c.f. Figure 3a) is shown on a colour gradient, with dark red for projects deploying few types of interventions, bright yellow for projects deploying many types of interventions, and grey points for projects that did not respond to relevant questions about intervention deployment.



3.2. Population modelling

3.2.3. Analysis of nest record data

On average (across all nest attempts recorded in NRS, regardless of timing during the breeding season), we calculated a nest success rate (the probability of a nest hatching at least one chick) of 0.275, albeit using a simple method that is more likely to have overestimated than underestimated this rate (due to the fact that many early-failing nests could go undetected in some studies). We estimated that successful nest attempts hatch an average of 2.52 chicks, meaning that the average number of chicks hatched per nest attempt is 0.69 in this dataset; however, other recent studies have estimated substantially higher rates of chicks hatched per nest than this (e.g. Ewing 2023). We estimate this equates to 1.00 chicks hatched per breeding pair (i.e. per pair that does not skip breeding) after accounting for re-nesting.

Nest success rates declined significantly across the course of the Curlew breeding season (Table 1, Figure 7), a finding supported by a clear decline in weekly mean rates (Table 2). The minimum number of chicks hatched per successful nest appeared to decline slightly over the season in terms of weekly means (Table 2); this decline was not statistically significant, but with a small sample size (Table 1). Some wader species show reductions in clutch size in later breeding attempts (Weiser *et al.* 2018), but we found no significant change in highest all-contents count over the season within this dataset (Figure 7d). However, the combined effect of both nest success and chick production rates (minimum number of chicks hatched per nest, including failures) was a significant decline across the season. Therefore, this suggests that average productivity is lower for later-starting breeding attempts, driven principally by variable nest success rates. Given that second and third attempts after failure will commence later in the breeding season, this in turn implies that these re-nesting attempts are likely to be less productive on average than first attempts, an effect shown in other wading birds (Swift *et al.* 2020, Saalfeld *et al.* 2021). It is possible that chicks fledging from these later attempts may also be less likely to ultimately recruit back into the population, which has also been shown in other species of waders (Alves *et al.* 2019).

Combining these average rates with rates of failure and re-nesting (see above; from Taylor *et al.* 2020) to estimate the proportion of nest attempts that were first, second and third attempts; and assuming that first, second and third attempts should be broadly chronologically sorted within the NRS dataset, we estimated separate average rates of nest success and chick production for each nest attempt to use in population modelling (shown within Figure 8).

Figure 7. Change across the breeding season and week-to-week in demographic rates, showing that nests initiated later in the breeding season are a) more likely to fail, and b) likely to produce fewer chicks, c) possibly even after accounting for the increased failure rate, but d) this cannot be explained by changes in clutch size over the breeding season. Points show individual nest records from the Nest Record Scheme, and are plotted with transparency to aid visualisation of overlaid points. Weekly means $\pm$ 1 s.d. (Table 2) are shown by grey bars and whiskers. Fitted model estimates (Table 1) are plotted as solid black lines, with 95 % confidence intervals shown by dashed lines.

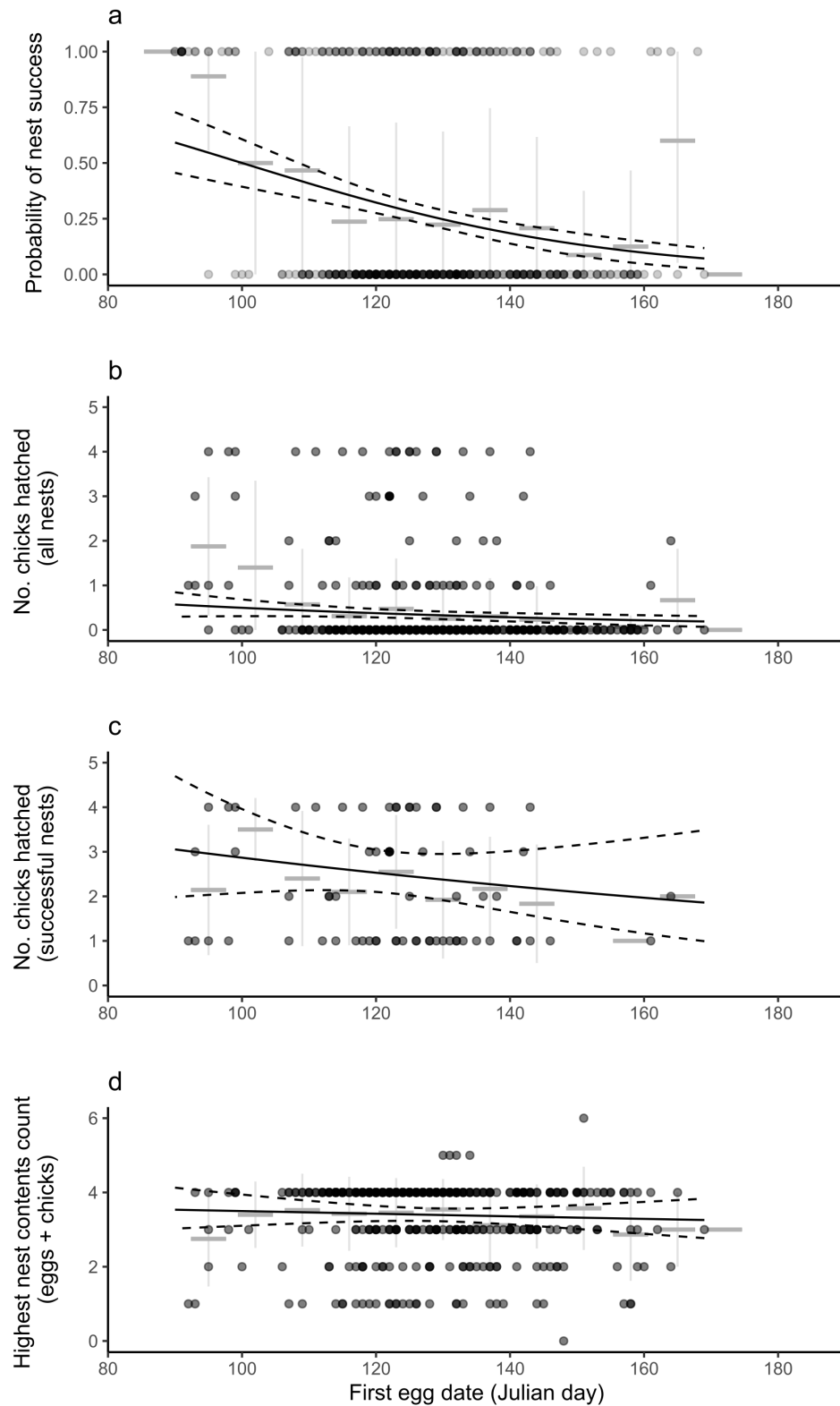


Table 1. Models of in-season variation in demographic rates. Models fitted to Curlew nest records from within the UK, submitted to the BTO Nest Record Scheme.

Variable	n	Error distribution	Effect size (s.e.)	Test statistic	P
Nest success rate	510	Binomial	-0.037 (0.008)	26.43	< 0.0001
Number of chicks hatched (all nests)	441*	Zero-inflated Poisson	-0.014 (0.006)	4.79	0.0287
Number of chicks hatched (successful nests only)	71*	One-inflated Poisson	-0.006 (0.006)	0.98	0.6118
Highest all-contents count (all nests)	441*	Poisson	-0.001 (0.002)	0.28	0.5949

*n.b. Because the number of chicks hatched was estimated using the NRS variable 'Minimum young hatched', it was possible for the same nest to be recorded as successful but with a minimum of zero young hatched. These 69 nest records were included in estimation of success rates, but removed for other analyses.

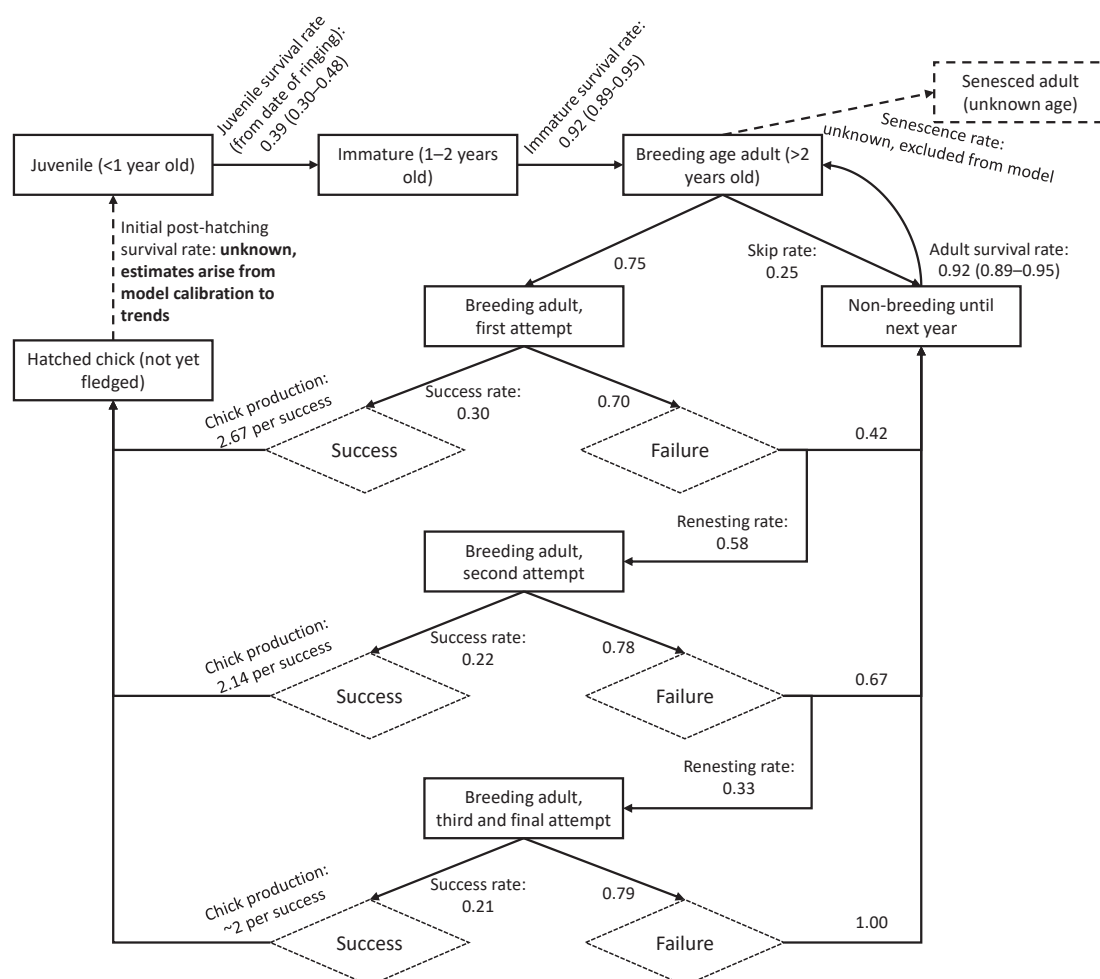
Table 2. Observed weekly means across the breeding season for demographic rates.

Week of year	First egg date of nest attempt (days of year)	Nest success rate (n; +/- s.e.)	Mean number of chicks hatched, all nests (n; +/- s.e.)*	Mean number of chicks hatched, successful nests only (n; +/- s.e.)**	Highest all-contents count, all nests (n; +/- s.e.)
13	Mar 26–Apr 1 (85–91)	1.00 (10; 0.00)	n/a (n = 0)*	n/a (n = 0)*	n/a (n = 0)*
14	Apr 2–8 (92–98)	0.89 (9; 0.11)	1.88 (8; 0.55)	2.14 (7; 0.55)	2.75 (8; 1.28)
15	Apr 9–15 (99–105)	0.50 (6; 0.22)	1.40 (5; 0.87)	3.50 (2; 0.50)	3.40 (5; 0.89)
16	Apr 16–22 (106–112)	0.47 (30; 0.09)	0.57 (21; 0.27)	2.40 (5; 0.68)	3.52 (21; 0.98)
17	Apr 23–29 (113–119)	0.24 (76; 0.05)	0.31 (68; 0.11)	2.10 (10; 0.38)	3.43 (68; 1.00)
18	Apr 30–May 6 (120–126)	0.25 (117; 0.04)	0.47 (108; 0.11)	2.55 (20; 0.29)	3.45 (108; 0.93)
19	May 7–13 (127–133)	0.22 (112; 0.04)	0.25 (100; 0.08)	1.92 (13; 0.37)	3.54 (100; 0.82)
20	May 14–20 (134–140)	0.29 (52; 0.06)	0.30 (43; 0.13)	2.17 (6; 0.48)	3.12 (43; 0.98)
21	May 21–27 (141–147)	0.21 (53; 0.06)	0.23 (48; 0.11)	1.83 (6; 0.54)	3.35 (48; 0.86)
22	May 28–Jun 3 (148–154)	0.09 (23; 0.06)	0.00 (21; n/a)*	n/a (n = 0)*	3.57 (21; 1.12)
23	Jun 4–10 (155–161)	0.13 (16; 0.09)	0.07 (15; 0.07)	1.00 (1; n/a)	2.87 (15; 1.25)
24	Jun 11–17 (162–168)	0.60 (5; 0.24)	0.67 (3; 0.67)	2.00 (1; n/a)	3.00 (3; 1.00)
25	Jun 18–24 (169–175)	0.00 (1; n/a)	0.00 (1; n/a)	n/a (n = 0)	3.00 (1; n/a)

*n.b. Because the number of chicks hatched was estimated using the NRS variable 'Minimum young hatched', it was possible for the same nest to be recorded as successful but with a minimum of zero young hatched. These nest records were included in estimation of success rates, but removed for estimation of chick production.

**n.b. Values presented in this table are means, not estimated using null one-inflated Poisson models as elsewhere; therefore, they may be influenced by the tendency for some recorders to record 'Minimum young hatched' as zero or one.

Figure 8. Underlying structure of population models. All rates applied during modelling are shown, including those extracted from the literature (survival rates of fledged juveniles, immatures and adults), and those estimated in this study by, variously, calculating from NRS data (nest success rates and chick production rates), approximating from Taylor *et al.*'s (2020) tag data (skip rate and re-nesting rates), and estimating by population modelling during this study itself (initial post-hatching survival rate of chicks). N.b. all adults that have had a successful breeding attempt are assumed to subsequently join the non-breeding pool for the remainder of the season (i.e. re-nesting rate after success = 0), but these lines are omitted from the flow chart for visual clarity.



3.2.4. Outcomes of population modelling

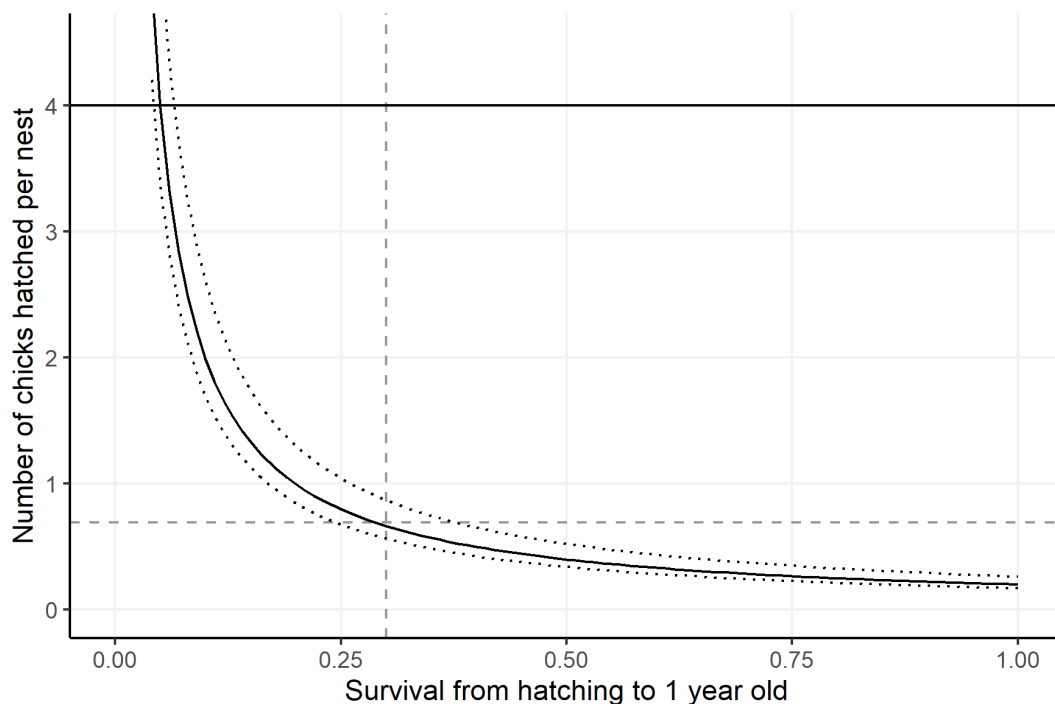
By assuming that all other demographic rates used as parameters in modelling (shown in Figure 8) are broadly representative of national averages, and then calibrating population models to national trends (7% decline over 10 years at UK-scale; Heywood *et al.* 2023), we estimate that survival of young birds from hatching to one-year old in the UK is currently approximately 30% (with estimates ranging from 21–41 % as skip and re-nesting rates were varied). Chicks can be ringed at any age between hatching and fledging, but the majority are ringed at the nest within the first 1–2 days after hatching, when they are easiest to locate and capture. Given survival of ringed chicks to one-year old is estimated to be around 39% (Cook *et al.* 2021), this suggests that around 23% of chicks die within their first approximately two days after hatching (0–46%). This estimate is not incompatible with empirical study suggesting that up to 50% of chicks die within the first week after hatching (Ewing 2023).

Recalibrating our population models to the Welsh population trend (40% decline over 10 years; Heywood *et al.* 2023) suggests that only 11% of young Curlews in Wales survive to one-year old, assuming all other rates are the same as the UK national average. However, it is plausible that nest success rates could currently be lower in Wales than the UK average, and young survival rates may therefore also be slightly higher than this estimate.

We next calibrated population models to stability, to explore the trade-off between increasing nest success and increasing young survival in attaining this target (Figure 9). If survival of young from hatching to one-year old is 30%, then an average of 0.75 chicks would need to be hatched per nest attempt (across all attempts, successful or failed) to attain population stability. We estimate this equates to 1.06 chicks hatched per breeding pair (i.e. per pair that does not skip breeding), accounting for renesting rates. On the other hand, if the average number of chicks hatched per nest attempt remained at 0.69, (or 1.00 chicks hatched per breeding pair), young survival rate would need to increase to 0.34 (or 34%) to attain population stability in the UK.

These statistics rely on the assumption that all other demographic rates (e.g. adult survival) remain constant at the levels used to parameterise our models (Figure 8). Allowing any one of these rates to vary can influence the exact rates associated with population stability; for example, Figure 9 illustrates how varying the skip rate between 0–35%, around its central estimate of 25%, causes the estimate for required number of chicks hatched per nest attempt (with current young survival) to vary between 0.56 and 0.87, which is approximately 0.79–1.22 chicks hatched per breeding pair; or the estimate for required young survival rate (with current numbers of chicks hatched per nest attempt) to vary between 0.26 and 0.38 (26–38%). In this context, it is important to note that substantial variation in demographic rates between different projects, in different regions and with different local contexts (of habitat, land management and predator management) is entirely expected. Therefore, our models based on national average demographic rates cannot be safely assumed to be representative at local scales.

Figure 9. Modelling of young survival and chick production rates required to achieve population stability. Grey dashed lines show the current estimates for survival of young from hatching to one-year old and chicks hatched per nest attempt, based on analysis of Nest Record Scheme data. The solid black line shows combinations of young survival and chick production that our population models estimate would provide a stable population, using central estimates for all other parameters. Dashed black lines illustrate the possible effects of one source of error (in this case, varying skip rate between 0–35%) on our predicted outcomes.



3.3. Case Study 1: Wader Recovery Areas

3.3.1. Proportions of breeding wader populations in proposed Wader Recovery Areas

Assessment of overlap between WRAs and relevant protected areas with predicted relative population densities for breeding waders (Figure 10, Table 3) shows that SPAs + WRAs would contain large proportions of the English populations of Golden Plover (together covering four-fifths of the English population), Curlew and Snipe (accounting for 40% to 50% of the English populations of these species). This suggests that current habitat and/or land management within the existing SPAs and proposed WRAs are already favourable for these species. For Lapwing, Oystercatcher and Redshank, some of the WRAs compare very favourably with the SPAs, and overall densities of these species across the whole WRA suite are similar to or, in the case of Oystercatcher, higher than average densities in the adjacent SPA network. However, overall proportions of the English populations of these species held by the combined WRA + adjacent SPA network are 1% to 20% – much lower than for the other species, which are more strongly associated with upland breeding.

Figure 10a. Heatmaps of predicted wader population density. Heatmaps derived from Calladine *et al.* (2022) are presented for six species; this figure depicts maps for Curlew, Golden Plover and Lapwing. Heat gradients in these maps vary from low (yellow) to high (red) wader densities. A semi-transparent white mask is applied outside the focal areas, such that areas included in proposed WRAs (left) and the combined areas comprising proposed WRAs, existing SPAs, and relevant existing SSSIs (right) stand out more clearly.

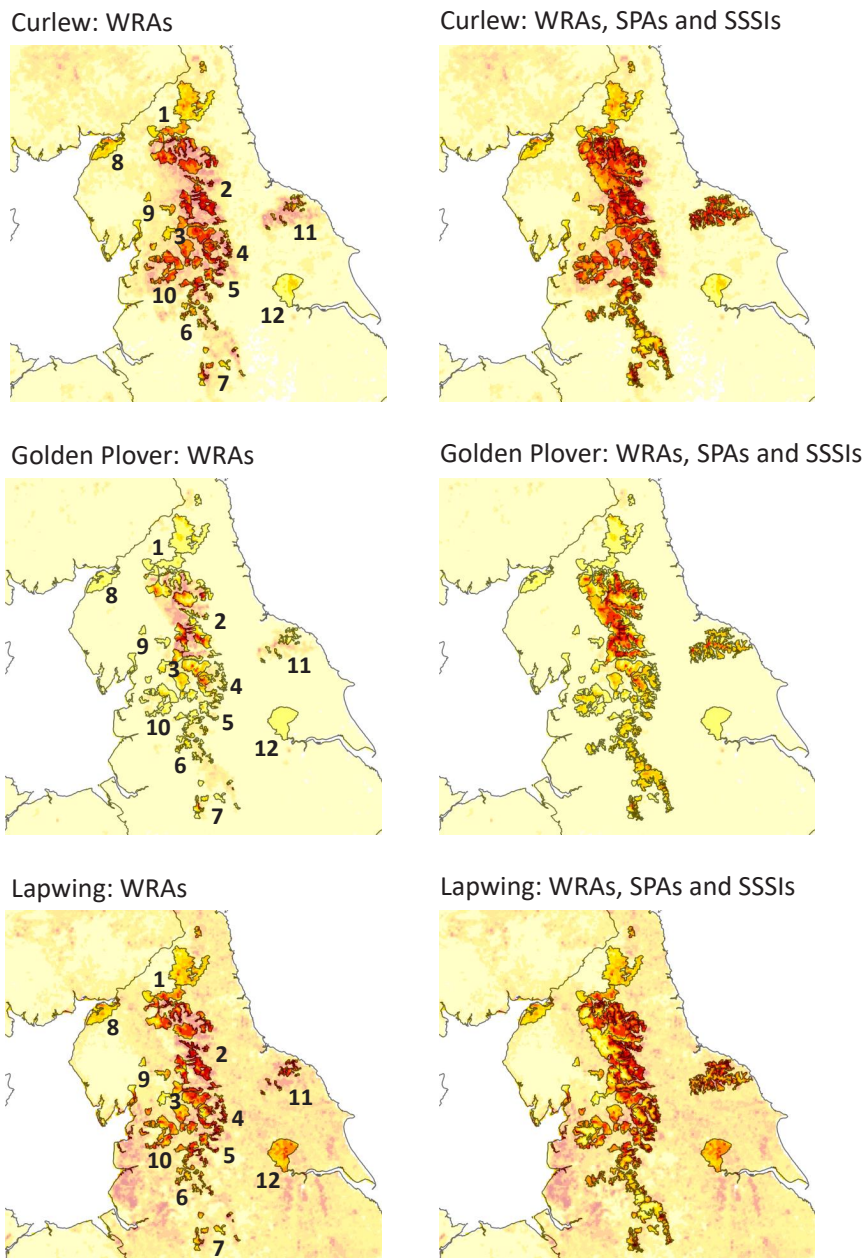


Figure 10b. Heatmaps of predicted wader population density. Heatmaps derived from Calladine *et al.* (2022) are presented for six species; this figure depicts maps for Oystercatcher, Redshank and Snipe. Heat gradients in these maps vary from low (yellow) to high (red) wader densities. A semi-transparent white mask is applied outside the focal areas, such that areas included in proposed WRAs (left) and the combined areas comprising proposed WRAs, existing SPAs, and relevant existing SSSIs (right) stand out more clearly.

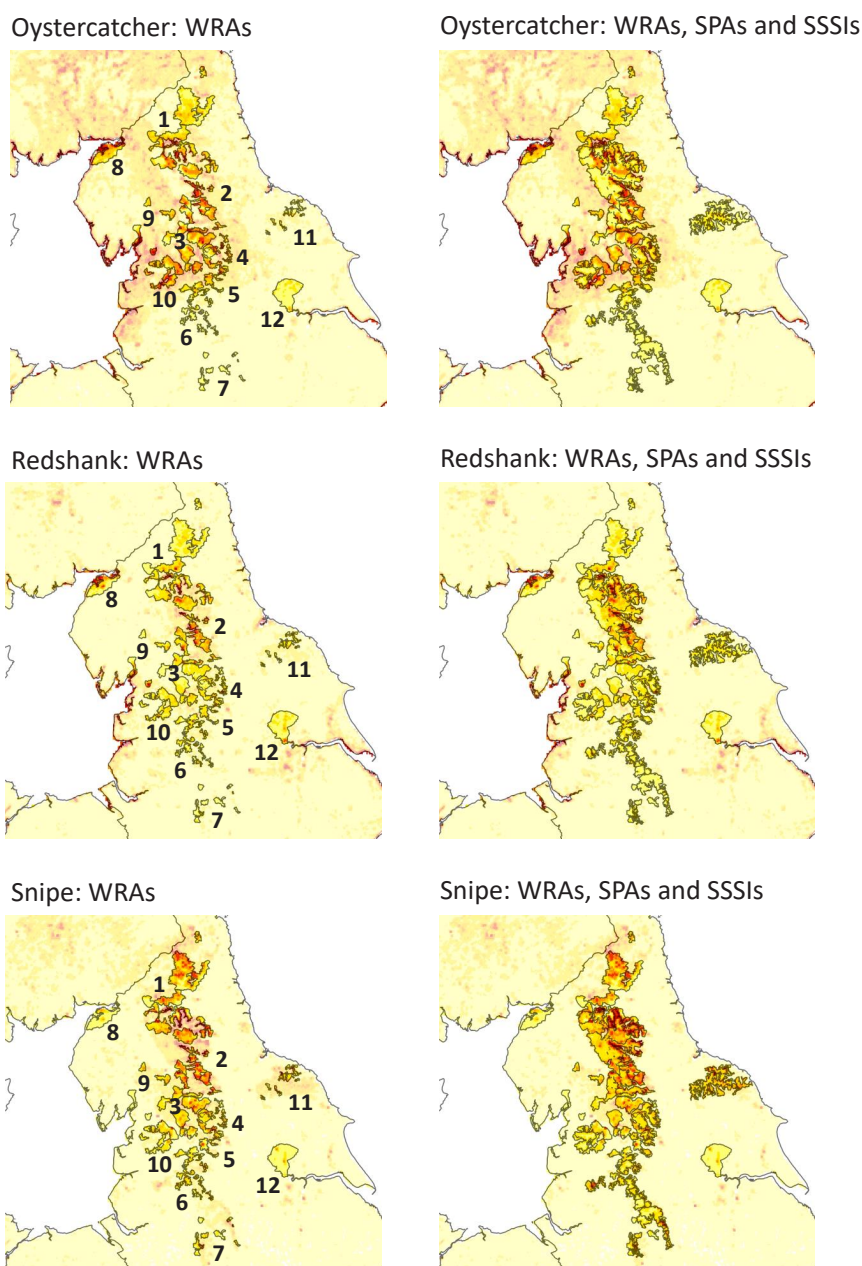


Table 3. Predicted relative density of breeding waders within proposed WRAs and adjacent protected areas. 'Size' gives the land area of each group of WRAs, SPAs and SSSIs, as well as total land areas for combined groupings. The relative density (predicted Bird Atlas count per 1-km square) in each area is shown, with the predicted proportion of the English population that this corresponds to given in brackets.

Areas	Size (km ²)	Lapwing	Curlew	Oystercatcher	Redshank	Snipe	Golden Plover
Wader Recovery Areas							
Northumberland	845	1.54 (1.2%)	1.74 (3.2%)	0.57 (1.8%)	0.12 (1.2%)	0.36 (5.9%)	0.1 (1.2%)
North Pennines Landscape	824	3.73 (2.9%)	4.23 (7.5%)	1.09 (3.4%)	0.35 (3.4%)	0.52 (8.1%)	1.07 (12.5%)
Solway	220	1.32 (0.3%)	1.44 (0.7%)	1.14 (1%)	0.32 (0.8%)	0.07 (0.3%)	0.02 (0%)
Western	153	1.62 (0.2%)	1.53 (0.5%)	1.29 (0.7%)	0.25 (0.4%)	0.14 (0.4%)	0.02 (0.1%)
Bowland	302	2.36 (0.7%)	3.39 (2.2%)	1.31 (1.5%)	0.14 (0.5%)	0.18 (1%)	0.12 (0.5%)
South Pennines North	228	2.98 (0.6%)	3.41 (1.7%)	0.63 (0.5%)	0.13 (0.4%)	0.16 (0.7%)	0.19 (0.6%)
South Pennines Central	121	1.73 (0.2%)	1.83 (0.5%)	0.21 (0.1%)	0.09 (0.1%)	0.18 (0.4%)	0.27 (0.5%)
South Pennines Peak	158	1.8 (0.3%)	2.17 (0.7%)	0.08 (0%)	0.04 (0.1%)	0.16 (0.5%)	0.37 (0.8%)
Humber	372	1.99 (0.7%)	0.51 (0.4%)	0.27 (0.4%)	0.1 (0.5%)	0.05 (0.4%)	0 (0%)
South Dales Nidderdale	262	3.24 (0.8%)	4.01 (2.3%)	0.95 (0.9%)	0.16 (0.5%)	0.27 (1.3%)	0.72 (2.6%)
North Dales	646	2.55 (1.6%)	3.47 (4.9%)	1.03 (2.5%)	0.16 (1.2%)	0.31 (3.9%)	0.83 (7.6%)
North York Moors	108	3.45 (0.4%)	2.57 (0.6%)	0.31 (0.1%)	0.12 (0.1%)	0.28 (0.6%)	0.27 (0.4%)
All Wader Recovery Areas	4,239	2.46 (9.9%)	2.74 (25.2%)	0.81 (13.1%)	0.18 (9.3%)	0.29 (23.6%)	0.45 (26.9%)
Adjacent SPAs & SSSIs							
Bowland Fells SPA	160	1.08 (0.2%)	3.35 (1.2%)	0.54 (0.3%)	0.04 (0.1%)	0.14 (0.4%)	0.48 (1.1%)
South Pennines SPA	662	1.59 (1%)	2.56 (3.7%)	0.2 (0.5%)	0.07 (0.6%)	0.21 (2.7%)	0.81 (7.6%)
North Pennines SPA	1,473	2.64 (3.7%)	4.08 (13%)	0.65 (3.6%)	0.26 (4.5%)	0.46 (13%)	1.82 (38%)
North York Moors SPA	441	2.7 (1.1%)	3.71 (3.5%)	0.31 (0.5%)	0.1 (0.5%)	0.35 (3%)	1.02 (6.3%)
Lower Derwent SPA	11	2.97 (0%)	1.19 (0%)	0.31 (0%)	0.24 (0%)	0.19 (0%)	0 (0%)
West Pennines SSSI	76	2.1 (0.2%)	2.58 (0.4%)	0.33 (0.1%)	0.13 (0.1%)	0.36 (0.5%)	0.28 (0.3%)
All ADJACENT SPAS & SSSIs	2,823	2.3 (6.2%)	3.57 (21.8%)	0.47 (5.1%)	0.17 (5.8%)	0.36 (19.6%)	1.33 (53.3%)
ALL SPAS	4,604	1.8 (7.8%)	2.24 (22.1%)	0.47 (8.2%)	0.27 (14.4%)	0.26 (23%)	0.83 (53.6%)
WRAS & Adjacent SPAs/SSSIs	7,062	2.39 (16.1%)	3.07 (47%)	0.68 (18.2%)	0.18 (15.1%)	0.32 (43.3%)	0.8 (80.2%)
ENGLAND	130,316	0.81 (100%)	0.35 (100%)	0.2 (100%)	0.06 (100%)	0.04 (100%)	0.05 (100%)

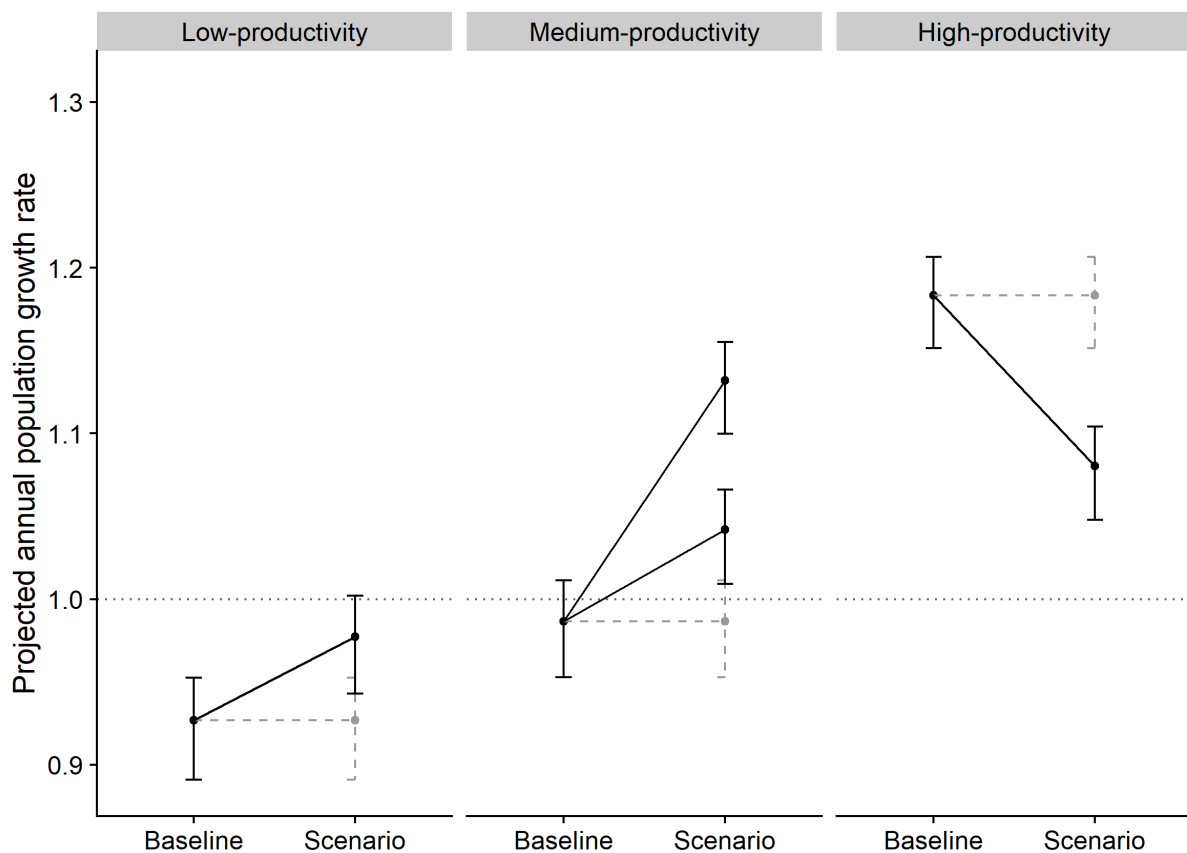
3.3.2. Modelled scenarios for interventions in Wader Recovery Areas

All four modelled scenarios of intervention within landscapes with different levels of baseline Curlew productivity resulted in substantial changes in population growth rate (Figure 11). The scenario which modelled a high intensity of interventions in a low-productivity landscape resulted in a relatively small increase in projected population growth rate that did not prevent continued declines, from 7.3% decline per year to 2.3% decline per year (Table 4). The scenarios which modelled interventions in a medium-productivity landscape resulted in much larger uplifts, from 1.3% decline per year to 4.2% increase per year (with low intensity of interventions) or 13.2% increase per year (with high intensity of interventions). A similarly large effect was found in the scenario which modelled the erosion of beneficial land management in a high-productivity landscape (e.g. through inappropriate tree-planting), which resulted in a reduction of population growth rates from 18.3% increase per year to 8.0% increase per year (Table 4).

As above, these population growth rates rely on the assumption that all other demographic rates remain constant at the levels used to parameterise our models (Figure 8). Allowing these rates to vary can influence the exact population growth rates predicted by the models. Figure 11. illustrates variation if juvenile and

adult survival rates are allowed to vary within their estimated 95% confidence intervals; for two out of seven baselines/scenarios, the range of population growth rates encompasses one (i.e. we cannot be entirely certain whether these scenarios will result in population growth or decline). A number of other sources of potential error exist but were not explicitly modelled here, illustrating the difficulty of predicting whether any specific scenario is more likely to result in population growth or decline. Nonetheless, these sources of error are unlikely to significantly alter the relative change in growth rates between baselines and scenarios.

Figure 11. Modelled population growth rates in three hypothetical landscapes, under baseline conditions and under four scenarios of future land management. Scenarios are joined to their corresponding baseline with lines; grey points and dashed lines represent a 'business-as-usual' case for comparison. The possible effects of one source of error are illustrated (in this case, using 95% confidence intervals for estimated adult survival) on our predicted outcomes using error bars, separated by landscape for visual clarity. The rates of nest success and young survival associated with each baseline and scenario are given in Table 4.



3.4. Case Study 2: intervention rates to stabilise population declines

We estimate that a very significant proportion of nests in Wales would need to receive intervention in order to stabilise the severe ongoing population decline there. Interventions consistent with the medium productivity/low intensity scenario from Case Study 1 would need to be applied at over half of all nest attempts in Wales, whilst the equivalent high intensity scenario could stabilise the population if applied at just under one in four nest attempts (Table 4). If nest-level success and young survival rates could be increased in line with our high productivity baseline for just over one in six nest attempts in Wales, this might also stabilise the population.

At the UK-scale, we estimate rather lower thresholds for stability, reflecting the less negative current population trends at this scale (and accordingly higher estimated baseline young survival rate). Interventions consistent with the medium productivity/low intensity scenario would need to be applied at just over one in five nest attempts, the equivalent high intensity scenario would need to be applied at around one in 20 nest attempts, and rates in line with our high productivity baseline at just over one in 25 nest attempts may be sufficient for population stability.

Table 4. Outcomes of modelled scenarios used in Case Studies 1 and 2, with intervention rates required for population stabilisation.

Landscape productivity	Intervention intensity	Modelled rates		Modelled population growth rate	Intervention rate to achieve population stability (% of all nests)	
		Nest success	Young survival		UK-scale	Wales-scale
Low	Baseline	0.10	0.05	0.927	–	–
	High	0.60	0.10	0.977	> 100%	> 100%
Medium	Baseline	0.25	0.30	0.987	–	–
	Low	0.50	0.30	1.042	17.2%	52.0%
	High	0.75	0.40	1.132	5.2%	22.4%
High	Baseline	0.81	0.50	1.183	3.5%	16.0%
	Decrease in favourable management	0.53	0.40	1.080	–	–

4. Discussion

4.1. Questionnaire main findings

Overall, the combined responses of the 29 Curlew recovery projects who responded to the questionnaire and/or provided data on project outcomes demonstrate that significant uplifts in Curlew nest success rates are achievable, largely through interventions targeting nest failures attributable to agricultural operations and predation by mammals. However, reducing chick mortality (thought chiefly to be driven by predation) appears to present a much greater challenge; most projects reported low chick survival rates even after intervention.

Most projects deployed multiple interventions, including both nest fencing and lethal mammalian predator control in many cases (with each deployed in the vast majority of projects). This limits our ability to assess the efficacy of individual interventions. However, most projects appear to have achieved positive outcomes through deployment of nest fencing (often in conjunction with other interventions), which is typically intended to exclude mammalian predators but may also incidentally serve to protect nests from livestock and agricultural operations. There was nothing to suggest that deploying a greater number of intervention types led to better outcomes (indeed, some of the projects that achieved the largest uplifts in nest success did so with only one or two intervention types), reflecting the findings of previous work (Franks *et al.* 2018). The effectiveness of a suite of interventions is likely to be more strongly influenced by the total level of investment (in terms of time and effort), and by how well the interventions deployed address the main drivers of low productivity in a particular location. For successful breeding, Curlews require a safe nest site (free from predation, and destruction by human activities), enough food for chicks, and a sufficiently low density of predators (both mammalian and avian) combined with enough safe places to hide in order to evade predation. The specific set of interventions required to meet these needs may differ between locations.

Another obstacle to interpretation is that the landscape and habitat descriptors provided by projects suggested the majority were operating in settings that are reasonably hospitable to Curlews (c.f. the medium-productivity landscape in Case Study 1); for example, most projects in grasslands indicated they were in areas of hay production as opposed to silage production. This is perhaps understandable, but nonetheless limits our ability to assess the degree to which interventions can be effective in less favourable landscapes where Curlew populations are in immediate danger of local extinction.

A consistent theme was the significant cost of applying interventions. This was associated with direct costs, such as fencing equipment, but also indirect factors such as the increased uptake of more time-intensive survey techniques (and use of paid staff rather than volunteers to carry these out) in order to conduct post-intervention monitoring in some projects. Only a few projects reported having had these costs covered (even partially) under existing agri-environment schemes. This is likely to represent a significant barrier to the

wider uptake of Curlew-focused interventions across networks of sites at landscape scales, although the need for intensive post-intervention monitoring could be reduced by improved understanding of the efficacy of interventions in different contexts. In other words, Curlew recovery may be possible, but only with sufficient funding. This is likely to be a key reason why wader conservation interventions more generally, which are supported by a good evidence base, are not deployed at sufficient scale to reverse population declines (Franks *et al.* 2018).

4.2. Population modelling caveats and conclusions

There are a number of limitations to the population modelling exercise presented here which must be borne in mind when interpreting the results. Critically, the demographic rates on which we based our models were, in several cases, calculated from small datasets: this was the case for productivity-related rates calculated from around 500 nest records from the NRS, and especially for skip and re-nesting rates calculated from a sample of just 20 tagged birds (Taylor *et al.* 2020). It is possible that the latter, in particular, is not nationally representative, and there is no way to confirm whether these rates vary in time without further equivalent data. These issues are particularly acute when estimating rates associated with second and third nesting attempts, which form a smaller proportion of the total than first attempts.

With that in mind, our population models suggest that a population average rate of 1.06 chicks hatched per breeding pair per year (based on an estimate of 0.75 chicks hatched per nest attempt) may be sufficient to provide population stability, assuming chick survival and other rates remain at current averages. This is not directly comparable to previous estimates (Taylor *et al.* 2020, Cook *et al.* 2021, Viana *et al.* 2023), which are expressed in terms of 'chicks fledged' per nest attempt or per pair. Realistically, 'chicks hatched per breeding pair' is likely to align better with what can be recorded by the majority of Curlew recovery projects, since methods to measure the number of Curlew chicks fledged per pair in the field can be time- and resource-intensive, and vary between study populations such that estimates are not directly comparable between projects (Jarrett *et al.* 2023). This metric should be used in the field in preference to 'chicks hatched per nest attempt', because attempts that fail early are unlikely to go detected, meaning that the sample of nests that are monitored is not a random sample of all nest attempts. Nonetheless, 'fledging success' can be recorded at the brood level (Fletcher *et al.* 2010, Jarrett *et al.* 2023, Tománková *et al.* 2024), and it is vital to consider that chick survival between hatching and fledging is a critical component of breeding productivity: increasing hatching success alone in a landscape that is inhospitable to chick survival is unlikely to result in a stable breeding population. Our models incorporate rates of skipped breeding and re-nesting, enabling rates to be expressed in terms of realised nest attempts and breeding pairs. Again, this may relate more directly to what can be observed in the field by Curlew recovery projects, since it may be easier (a) to consider the productivity of birds that are known to have attempted a nest, without the need to assess the proportion of breeding-age birds in the local population that have skipped in that year, and (b) to consider the outcomes of each monitored nest attempt without the need for information about other attempts made by the same pair in that year.

A further caveat is that our estimate of 1.06 chicks hatched per breeding pair is made with all other rates held at their average estimated levels. Deviation from these levels – either because they have not been estimated accurately or because of genuine ecological variation between sites, regions, or over time – could lead the threshold for stability to increase or decrease significantly. For example, we demonstrate that varying skip rates between 0.00 and 0.35 (around a central estimate of 0.26), holding all other rates constant, changes the threshold for stability to 0.79 and 1.22 hatched chicks per breeding pair (or 0.56–0.87 hatched chicks per nest attempt) respectively. The existence of skipping in Curlew has not been proven beyond doubt, and skip rates used in our modelling are based on a single study with a small sample size (Taylor *et al.* 2020). However, assuming that skipping never occurs (i.e. that all breeding-age individuals successfully pair up and initiate at least one breeding attempt in every year) results in the lowest modelled threshold for stability, and meeting this threshold exactly in a population where skipping does in fact take place would not be sufficient to achieve stability. Therefore, incorporating a level of skipping in our models represents a relatively conservative approach to determining the productivity thresholds required for stability. Demographic rates could also change due to the influence of other, external factors, such as outbreaks of high pathogenicity avian influenza (HPAI) or other diseases, changes to availability/quality of overwintering habitats, change in hunting pressure, and stochastic events such as extreme weather, all of which can influence adult and

juvenile survival rates (Cook *et al.* 2021). This highlights the need for Curlew recovery projects to engage in long-term monitoring of breeding success and local population sizes, during and ideally beyond the lifespan of the project.

In practice, the number of hatched chicks per breeding pair or nest attempt is a function of (i) nest success rates and (ii) the average number of chicks hatched per successful nest. Although Curlews typically lay clutches of four eggs (at least on first attempts), successful nests do not always produce four chicks. Even after accounting for the possibility that some surveyors record the 'minimum number of hatched chicks' as one in cases where more than one chick has hatched, but the surveyor cannot be certain of this (i.e. that the dataset is 'one-inflated', containing a greater proportion of values of one than its true underlying distribution), we estimate the average number of hatched chicks per successful nest to be 2.52 across all attempts. Aside from further impacts of fieldworker caution around recording of a 'minimum' estimate beyond those accounted for by the analytical approach, this shortfall may also be partly due to a combination of partial clutch predation and infertility (e.g. Berg 1992, Grant *et al.* 1999, Marshall *et al.* 2023). Other recent field studies have found substantially higher average rates in the region of 3.5 chicks hatched per successful nest (Ewing 2023 and unpublished BTO data), but it is plausible that the national average over a 40-year period of population decline and local extinctions could be lower than the recent rate in areas where Curlew populations have persisted. NRS data also suggest that the number of hatched chicks per nest attempt falls significantly for nests initiated later in the breeding season (likely to be re-nesting attempts by birds that have already failed once or more), primarily due to increased failure rate but possibly also with a small decrease in number of chicks hatched per successful nest. Whilst this might be interpreted to suggest that focussing on late-initiated nests may yield the greatest benefits (since these nests have a higher baseline risk of failure, which might be mitigated by intervention), this strategy would carry a substantial, perhaps unacceptable, level of risk, since early-hatching chicks may be more likely to ultimately recruit into the breeding population (Alves *et al.* 2019).

Overall, our models suggest that the required uplift in chick production rates and/or young survival rates required to stabilise the population at UK-scale are relatively small. The required increase in chick production could be achievable, given the substantial improvement in nest success rates found by some projects that responded to the questionnaire. Note, however, that the required increase from 1.00 to 1.06 hatched chicks per nest attempt refers to the national average: merely matching this rate in areas where interventions are deployed, whilst rates remain lower elsewhere, will not be sufficient to stabilise population trends. Applying interventions of current intensities across sufficiently large proportions of the population to noticeably increase the national average productivity may present a considerable challenge, or even be impossible, in some contexts (Franks *et al.* 2018), as demonstrated by Case Study 2.

Our models also suggest that a significant proportion of Curlew chicks may die within the first approximately two days after hatching: some 22% (out of a total of 70%) that do not survive their entire first year. This raises the possibility, as yet untested, that interventions targeted around the nest site at medium scales (e.g. field-scale predator fencing) could have disproportionately large benefits, if they provide protection to young Curlew chicks that have hatched and left the nest cup, but stay in the vicinity of the nest over several days. Whilst older chicks (i.e. those closer to fledging) may roam further from the nest and beyond the influence of medium-scale measures, these birds are also more likely to be able to evade predation due to their larger size and greater mobility. Across the entire pre-fledging period, chicks may range up to 1 km, and generally less than 500 m, from the nest site (Ewing 2023). Measures implemented at these scales could carry additional benefits if, for example, they reduce time spent off the nest by adults due to disturbance or the need to forage (Smith *et al.* 2012). If other, non-predation drivers of mortality, such as weather, were of relatively greater importance during the first days post-hatching compared to when chicks are older (as has been recorded in other waders, e.g. Golden Plover; Pearce-Higgins & Yalden 2002), then this may limit the impact on chick survival of the scale at which interventions are targeted around the nest, but multiple studies have found predation to be the predominant driver of chick mortality in Curlews (Berg 1992, Grant *et al.* 1999, Fletcher *et al.* 2010, Zielonka *et al.* 2019, Baines *et al.* 2023, Ewing *et al.* 2023a).

4.3. Wader Recovery Areas

The proportions of the English breeding populations of Golden Plover, Curlew and Snipe found within the network of SPAs and proposed WRAs considered here are much higher than the proportions of Oystercatcher, Redshank and Lapwing populations breeding in these areas. This is because large numbers of Oystercatchers and Redshanks breed in more coastal situations, and Lapwings are still relatively widespread (or were during the 2010 bird atlas surveys) across much of lowland England. However, it is very likely that the recent and ongoing population trends of all these waders will be more positive in most of the WRAs, particularly those in the main Pennine block and in the North York Moors (Border *et al.* 2021), than elsewhere. This means that the relative importance of remaining upland population strongholds for breeding waders is likely to have increased since 2010.

This context compares favourably to the outcomes of our scenario modelling in Case Study 1. Our scenarios were hypothetical, so conclusions should be drawn with caution; parameterising scenarios with different rates of nest success and young survival than those we chose might lead to different predicted outcomes. Nonetheless, the expected magnitude of effects on productivity parameters from the application of interventions was grounded in evidence from the literature and from our survey of Curlew recovery projects, and therefore can provide a decision-support framework. First, the outcomes of Curlew recovery projects may vary depending on the wider context in which they take place. A much higher intensity of interventions may be necessary to achieve population stability in landscapes where the baseline is very challenging. For example, our high intensity/low productivity scenario modelled a doubling of young survival, yet this only equated to an increase from 5% to 10%, and still resulted in a population growth rate below one, i.e. a slower decline, rather than a recovery (Table 4). The low intensity/medium productivity scenario modelled no increase in young survival at all, and an increase in nest success that was also modest compared to the high intensity/low productivity scenario, yet with a comparatively favourable baseline, this scenario did result in a population growth rate above one (Figure 11).

Second, reduction in favourable land management within high-productivity landscapes may result in substantially reduced population growth rates (Table 4). This emphasises that maintaining the quality of high-productivity landscapes for Curlews and other breeding waders is also of importance, as their loss could undermine the benefits of increasing the favourability of medium-productivity landscapes. Land management approaches contributing to existing high productivity scenarios include favourable grassland management, typically in an extensively managed landscape; and high levels of predator management (Fletcher *et al.* 2010, Kentie *et al.* 2015, Franks *et al.* 2018, Laidlaw *et al.* 2021). Site protection, for example through international, national or regional protected area networks, may provide mechanisms to promote such approaches in some locations. Example systems where management contributes to high productivity levels include extensive farmed grassland within or adjacent to areas of high predator control, such as shooting estates in the UK (Baines *et al.* 2023), on grouse moors themselves (Fletcher *et al.* 2010), traditional herb-rich meadows in the Netherlands (Kentie *et al.* 2015), and wet fen meadows in northern Germany (Loonstra *et al.* 2024). In some of these systems, management actions exist as a result of other economic activities which are indirectly beneficial to breeding wader productivity (e.g. gamebird hunting, hay production), whereas in other cases interventions have been undertaken expressly with the intention of biodiversity benefit (e.g. Lake Dümmer in Lower Saxony, Germany) and so might be considered analogous to the outcome over time of the high intensity intervention/medium productivity scenario.

Despite their current favourable state (from a Curlew's perspective), threats to future Curlew breeding success in current high-productivity landscapes are mounting:

1. Tree canopy and woodland cover: Protected Landscapes which comprise much of these high-productivity areas are (at the time of writing) working towards targets to increase tree canopy and woodland cover by 3% of land cover in 2050, compared to a 2022 baseline (DEFRA 2024; n.b. it remains to be seen whether this target will change following the 2024 General Election). For example, the Yorkshire Dales National Park (YDNP) had 4% tree canopy and woodland cover in 2022, so has committed to reaching 7% cover by 2050 (Yorkshire Dales National Park Biodiversity Forum 2023). The YDNP, like many Northern Upland Chain (NUC) Protected Landscapes, is a population centre for all six wader species in this study (Table 3). Increasing tree canopy and woodland cover may benefit some species and aid with meeting national and global environmental and climate policy, but in

general terms, will negatively impact wader populations if implemented in existing areas of high wader productivity, particularly with negative effects of forestry extending 1 km or more from the woodland edge (Douglas *et al.* 2014, Franks *et al.* 2017). There is limited support among upland farms for increasing woodland cover, which is largely at odds with traditional farm management (Iversen *et al.* 2022). Limiting tree canopy and woodland cover targets in WRAs would therefore be likely to be welcomed by these communities, although it would be beneficial if communities whose tree-planting applications might be refused on the grounds of wader conservation are offered comparable streams of income incentivising the preservation of favourable conditions for breeding waders.

2. Changes to farm payments: depending on effectiveness of scheme design, the slow transition to Environmental Land Management (ELM) in England might risk unbalancing the situation that has, so far, maintained high-productivity landscapes for waders. Farm income in these landscapes has been disproportionately dependent on income from alternative sources than agricultural output: direct payments via the Basic Payments Scheme (BPS), agri-environmental schemes (AES), and diversification. Without comparable income available via ELM, farms may turn to intensification, tree-planting, or selling (which may in turn precipitate intensification, tree-planting, or other land use changes). WRAs could secure this balance with well-designed wader options within ELM options, especially if these match or better the income lost from BPS and previous AES options.
3. Wider changes to extensively farmed landscapes: although agricultural management practices in extensively farmed landscapes have changed more slowly than elsewhere (e.g. in the uplands compared to the lowlands, in a UK context; Ward *et al.* 1990) from the mid-20th century to the present day, there have still been shifts in agricultural production in landscapes where waders remain in high abundance, some of which are still ongoing; most notably from hay production to more intensive grass monocultures, and associated increases in dairy production (which require more intensive practices). These landscapes cannot be treated as static, and conservation strategies and policies that mitigate against the detrimental elements of change would be valuable. In synergy with this, there can remain a strong sense of community and tradition within these landscapes, and Curlew conservation strategy will be more effective if this is respected.
4. The future of predator control: reducing local densities of predators of waders (especially Fox and Carrion Crow) through legal lethal control is an oft-applied conservation intervention to improve breeding wader productivity, but nonetheless remains contentious, with a range of moral, ethical and environmental arguments proffered against lethal predator control. Reductions in the efficacy or scale of legal lethal predator control in wader breeding areas will likely result in local reductions in productivity of Curlews and other waders (Fletcher *et al.* 2010, Baines *et al.* 2023). Such reductions would potentially be in line with the Case Study 1 scenario of removal of beneficial management from a high-productivity landscape. This is particularly relevant to the uncertain future of grouse moor management in the UK (Crowle *et al.* 2022). There is widespread evidence of illegal raptor persecution on grouse moors (Murgatroyd *et al.* 2019, Newton, 2021, Ewing *et al.* 2023b), which has driven legislative changes in Scotland (Wildlife Management and Muirburn (Scotland) Act 2024). There is a high spatial overlap between grouse moors and breeding Curlew in the UK, where they are estimated to support 36% of the national population (Calladine *et al.* 2022).

Despite the high Curlew productivity of areas benefiting from predator control (e.g. farmland within or adjacent to shooting estates in the UK), much of the benefit of predator control to wader productivity is diminished within 5 km from the control site (Baines *et al.* 2023). Relative abundance of waders that appears to be ubiquitously high in regions with large amounts of predator control undertaken, like the Northern Upland Chain (Calladine *et al.* 2022), likely belies patchwork variation in productivity (Baines *et al.* 2023), as high-productivity populations within the influence of predator control act as 'source' populations for 'sinks' elsewhere in the same landscape. Curlew natal dispersal and recruitment is poorly understood, but most birds are likely to return to the wider landscape in which they hatched (Gerritsen 2021), so abundance patterns may simply indicate that 'source' populations still exist more widely in these landscapes than in others. Our analyses suggest improving productivity in these landscapes (which equate most closely to our medium-productivity scenario), capitalising on (a) the proximity of professional predator control and (b) retained experience, knowledge, and practice of farming alongside ground-nesting bird nests and broods even where

farming is now intensive, may lift current population decline in these areas into growth. This might further augment the 'source' population effect of these landscapes, increasing the overall national Curlew population and providing the opportunity to recolonise areas within and around the Northern Upland Chain (and potentially within the proposed WRAs).

Our scenarios do not explicitly model the outcomes of any particular intervention type, because there are multiple means by which the uplifts in nest success and young survival that we have modelled might be achieved in practice, and some understanding of the relative merits of these different approaches already exists (Franks *et al.* 2018). However, the most appropriate and effective suite of interventions (and the required level of effort to achieve the modelled thresholds using those interventions) is likely to vary widely on a case-by-case basis between different local situations. This need for flexibility would need to be reflected in any suite of Curlew-focused agri-environment scheme options, ideally with provision of advisory support to ensure interventions are correctly targeted in response to local conditions, if they are to support landscape-scale Curlew recovery effectively.

The proposed Wader Recovery Areas shown and discussed here may be subject to changes that could include areas being removed from the suite, other areas being subject to shifts in boundary that could result in their getting bigger or smaller, and new areas not considered here being added to the list. Areas that could be added to the suite of proposed WRAs include 56 km² of land spanning either side of Duddon Sands to the south of the Lake District, several discrete areas of land (ranging from 2 km² to 71 km²) extending the WRAs currently proposed in and around Forest of Bowland AONB, and a single block of around 75 km² in the Peak District south of Buxton. None of these areas would add much to the numbers of breeding Golden Plovers, Curlews or Snipe within the WRA suite, but the area around Duddon Sands holds relatively high densities of Oystercatchers and may hold some breeding Redshanks. Some of the additional areas in and around the Forest of Bowland, particularly around River Lune in the north and to either side of Grizedale and Bleasdale to the west, are likely to hold moderate densities of Lapwings, Oystercatchers and Redshanks.

4.4. Extent of required interventions

The intervention rates we estimate in Case Study 2 that would be required for population stability (Table 4) suggest that population stabilisation at current levels in Wales would likely demand the application of more intensive interventions than those modelled, e.g. headstarting or landscape-scale predator management. Applying interventions at a rate of over 50% of nest attempts, which would currently represent around 500 nests based on recent population estimates (Taylor *et al.* 2020) – or even 22%, approximately 210 nests – would require an enormous effort simply to locate a sufficient number of nests, let alone to then apply interventions at all of them. A target of 16%, or around 150 nests, is possibly more feasible, but the required intervention intensity for this scenario relates to high productivity-focused land management approaches (e.g. very high background levels of predator control) which are largely absent from Wales at present. Whilst population stabilisation at current levels may not be feasible, prevention of regional extinction (i.e. stabilisation at lower abundance, and probably with a smaller distribution) may be a realistic outcome of current intervention intensities, since the absolute number of nests at which interventions need to be applied in order to meet these percentages of the total will decrease as the population continues to decline.

On the other hand, population stabilisation at UK-scale looks more achievable, especially given the substantial proportion of English populations that are within the SPAs and proposed WRAs. Large-scale Curlew population increases in the Peak District Moorlands from 2004/2005 to 2018 provide evidence that this is possible (Border *et al.* 2021). For example, the proposed WRAs and adjacent protected areas collectively hold some 47% of the English population (Table 3), which corresponds to around 22% of the British population (Calladine *et al.* 2022); clearly, this is larger than the 17.2% coverage (Table 4) estimated for population stability even if applying hypothetical low intensity interventions in medium productivity landscapes, and even if no equivalent measures are taken in Scotland or Wales, under the assumption that the majority of nests within the WRAs and protected areas benefit from interventions.

However, ensuring that the majority of nests within any given area benefit from interventions may be incompatible with nest fencing, the primary intervention type deployed across current Curlew recovery projects, in some contexts. Interventions that are applied at the scale of individuals rather than populations (c.f. 'close order management'; Merton 1992, Clout & Merton 1998), such as nest fencing, can impose

significant constraints of cost and capacity on projects. In this example, ensuring that the majority of nests within an area benefit from nest fencing would require sufficient monitoring effort to locate nests in the first place, followed by sufficient time and materials to deploy fencing to them. In practice, this may only be achievable at relatively focused, local scales (e.g. single sites or small, isolated populations) where the deployment of interventions is limited by the number of nests available at which they can be deployed. With larger populations spread over wider areas (and particularly in upland landscapes), where the deployment of interventions is instead constrained by capacity and some nests do not benefit, other intervention types (predator control, habitat management) may be more effective in terms of increasing the total number of chicks hatched and young fledged, even if their impacts at the level of individual nests are less than could be achieved by nest fencing.

Just as the outcome of Curlew conservation projects may vary depending on context, the perceived success of Curlew conservation projects may also depend on their goals, which are many and varied among projects operating at different spatial scales and in different contexts. For example, it might be possible for a small, spatially-focused project to 'succeed' in stabilising the local population within the project's operational area, but without reducing ongoing declines in the surrounding landscape. Conversely, a project operating at broader scales might 'succeed' in stabilising a regional population but still suffer local extinctions at sites, and overall range contraction, within its operational area.

Other critical assumptions made by this analysis may also be violated in reality. First, the modelling assumes that the Curlew populations at each scale are closed, and therefore that birds emigrating from areas at carrying capacity must recruit somewhere within the population. This is a mostly reasonable assumption for the UK-scale analysis, but not for the Wales-scale analysis, since a proportion of Welsh Curlews are likely to recruit across the border into the populations of western England and vice versa, given that these populations are, in some areas, contiguous and probably not distinct from the perspective of the birds. A second, related assumption is that, should high-productivity areas or those receiving interventions reach their carrying capacity, excess birds are able to emigrate and recruit successfully into the population, rather than forming a pool of non-breeding birds awaiting vacant territories in their natal areas. The degree to which young Curlews tend to recruit into their natal areas or disperse elsewhere, especially when natal areas are close to carrying capacity, is not well understood (but see Gerritsen 2021), so it is unclear whether this assumption is likely to be violated or not. Increased multi-year GPS-tracking of young birds, including in association with the release of headstarted birds, is likely to shed greater light on these questions over the next few years.

4.5. Conclusions

This study demonstrates that interventions currently applied by Curlew recovery projects can have significant positive impacts on rates of nest success, and that these impacts (if applied sufficiently widely) may be sufficient to stabilise population decline in this species at the UK-scale, potentially even in the absence of any increase in the rate of young survival, which appears harder to achieve. However, significant knowledge gaps remain around the effectiveness of specific interventions alone and in combination, and how this varies depending on the context in which they are applied. Ongoing monitoring and evidence collection as Curlew recovery projects continue to apply interventions is critically important, in order to enable more informed decision-making in future.

To maximise the positive effect of interventions, several key remaining barriers must first be addressed: (i) quantifying the cost and effectiveness of interventions in different contexts; (ii) prioritising interventions based on cost-effectiveness (again in different contexts); (iii) identifying the most efficient funding mechanisms to pay landowners, land managers and other actors conducting interventions in order to achieve the desired/required level of uptake; (iv) implementing effective monitoring and evaluation of interventions being deployed; and (v) applying adaptive management where suitable.

Natural England's proposed Wader Recovery Areas appear to offer one viable means by which to ensure that intervention effort is targeted towards areas of relatively high importance to Curlew, where their impact on population growth rates may be largest.

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Supplementary Information

Table S1. All proposed Wader Recovery Areas (as provided to BTO by Natural England). The area of each WRA is given, and the regional groupings assigned by BTO for ease of interpretation are indicated.

WRA number	Area (km ²)	WRA Area
1	97.79	North Dales WRAs
2	90.05	North Dales WRAs
3	16.76	North Dales WRAs
4	4.73	South Dales Nidderdale WRAs
5	53.49	South Pennines North WRAs
6	159.98	North Pennines Landscape WRAs
7	455.39	Northumberland WRAs
8	100.61	Northumberland WRAs
10	8.92	South Pennines Central WRAs
11	29.87	South Pennines Central WRAs
12	3.61	South Pennines North WRAs
13	9.09	South Pennines North WRAs
14	7.91	South Dales Nidderdale WRAs
15	10.8	South Dales Nidderdale WRAs
16	7.07	South Dales Nidderdale WRAs
17	28.85	South Dales Nidderdale WRAs
18	4.42	South Dales Nidderdale WRAs
19	6.49	North Pennines Landscape WRAs
20	17.67	Bowland WRAs
21	17.95	Bowland WRAs
22	16.28	Bowland WRAs
23	127.8	Bowland WRAs
24	116.91	Bowland WRAs
26	0.81	North York Moors WRAs
27	7.96	North York Moors WRAs
28	20.06	North York Moors WRAs
29	2.02	North York Moors WRAs
30	4.78	North York Moors WRAs
31	10.62	North York Moors WRAs
32	19.83	South Pennines Peak WRAs
33	7.89	South Pennines Peak WRAs
34	7.71	South Pennines Peak WRAs
35	7.2	South Pennines North WRAs
36	58.78	South Pennines North WRAs
37	72.45	North Dales WRAs
38	6.57	South Pennines North WRAs
39	122.98	North Dales WRAs
40	112.87	South Dales Nidderdale WRAs
41	12.62	North Pennines Landscape WRAs
42	23.5	North Pennines Landscape WRAs
43	91.79	North Pennines Landscape WRAs

WRA number	Area (km ²)	WRA Area
44	63.86	North Pennines Landscape WRAs
45	57.2	North Pennines Landscape WRAs
46	62.29	North Dales WRAs
47	67.77	North Dales WRAs
50	25.01	Western WRAs
51	32.07	North Pennines Landscape WRAs
52	7.65	North Pennines Landscape WRAs
53	48.79	Western WRAs
54	6.43	South Pennines Peak WRAs
55	5.63	South Pennines North WRAs
56	159.22	North Pennines Landscape WRAs
58	9.37	Northumberland WRAs
59	155.63	Northumberland WRAs
60	9.67	North Pennines Landscape WRAs
61	38.3	Northumberland WRAs
62	76.2	North Pennines Landscape WRAs
63	36.04	North Pennines Landscape WRAs
64	36.42	North Dales WRAs
65	58.72	Western WRAs
66	17.76	South Dales Nidderdale WRAs
67	67.07	North Dales WRAs
68	219.84	Solway WRA
69	1.34	South Pennines Peak WRAs
70	8.92	North York Moors WRAs
71	5.32	North York Moors WRAs
72	8.22	North York Moors WRAs
73	9.4	North York Moors WRAs
74	14.64	South Pennines Central WRAs
75	7.95	South Pennines Central WRAs
76	2.43	South Pennines Central WRAs
77	22.42	South Pennines Central WRAs
78	34.65	South Pennines Peak WRAs
79	15.35	South Pennines Peak WRAs
80	20.98	Western WRAs
81	372.27	Humber WRA



Front cover: Richard Bunce / walking photographer; back cover: Samantha Franks / BTO

Review of Curlew-focused conservation interventions in the UK and Western Europe, and assessment of intervention scenarios using population modelling

Eurasian Curlew *Numenius arquata* populations are in decline, largely due to low breeding productivity. This has stimulated the implementation of a large number of Curlew recovery projects in the UK and elsewhere in Europe, deploying conservation interventions that target increases in Curlew productivity. With several years of outcome data now collected by many projects, it is useful to review outcomes across projects and assess the degree to which these might be able to deliver conservation goals. We collate evidence from ongoing Curlew recovery projects on the efficacy of the conservation interventions and then assess whether these interventions are likely to be sufficient to deliver population stability if applied in different contexts and at different scales. The outcomes of ongoing projects suggest that interventions can drive significant improvements in nest success rates, but improving chick survival rates poses a greater challenge. Key remaining barriers to realising population stability and growth in Curlew populations through conservation interventions include understanding how best to compensate landowners and land managers for the costs of implementing interventions. Natural England's proposed Wader Recovery Areas appear to offer one viable means by which to ensure that intervention effort is targeted towards areas of relatively high importance.

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