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Published version:

DOI:10.1007/s10336-013-0938-3

Chamberlain, D.E., G.E. Austin, R.E. Green, M.F. Hulme & N.I.K. Burton (2013). Improved population trends of Great Cormorants *Phalacrocorax carbo* in England and Wales for effective management of a protected species at the centre of a human-wildlife conflict. Bird Study 60: 335-344.
DOI: 10.1080/00063657.2013.798258

Licensed control does not reduce local Cormorant *Phalacrocorax carbo* population size in winter

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Abstract Cormorants *Phalacrocorax carbo* have increased on European freshwaters, creating conflicts with fishing interests. As a result, control measures have been implemented in several countries, although their effect on the English population has yet to be determined. Wetland Bird Survey data was used to derive population growth rates (PGR) of non-coastal Cormorant populations in England. PGR was analysed in relation to control intensity at different scales (5km to 30km radius) from 2001 to 2009 in order to determine (i) the extent to which control intensity (proportion of the local population shot per winter) was associated with site-level population change, and (ii) whether potential effects of control intensity were evident on Special Protection Areas (SPAs). There were no clear differences in PGR when comparing sites which had experienced control versus sites where control had never been carried out. The few significant relationships between control intensity and Cormorant PGR detected were mostly positive, i.e. population growth was associated with higher control intensity. Control intensity was not related to Cormorant numbers in SPAs. Positive associations with control may arise because control is reactive, or because non-lethal effects cause greater dispersal of Cormorants. These results provide no evidence that Cormorant removal at local scales is having an effect on longer term (i.e. year-to-year) population size at a site level. They also suggest that control measures have not affected national population trends, although a better understanding of site use and movements of individual Cormorants needs to be developed at smaller scales (including those due to disturbance caused by control measures) to more fully understand processes at larger scales. Further research is also needed into the extent to which lethal and non-lethal effects of control on Cormorants are having the desired impact on predation rates of fish, and so help resolve the conflict between Cormorants and fisheries.

Keywords Culling · disturbance · fisheries · human-wildlife conflict · population growth rate · SPA · Wetland Bird Survey.

Introduction

Human-wildlife conflicts are at the root of many current conservation problems and occur when requirements of wildlife overlap with those of human interests. The source of the conflict is often the consumption of resources of value to humans by wildlife, for example predation of livestock (Musiani et al. 2003; Patterson et al. 2004) or game species (Redpath and Thirgood 1997; Valkama et al. 2005), or damage to crops (Naughton-Treves 1997; Weladji and Tchamba 2003). The underlying causes may be increase and expansion in either human or wildlife populations, the latter often arising after conservation interventions (e.g. Vijayan and Pati 2002) or cessation of human activities which formerly limited populations, especially hunting (e.g. Musiani et al. 2003). Measures to resolve such conflicts may include both lethal and non-lethal control of wildlife, but any such measures should take into account potential consequences for the animal populations in question, ideally through thorough *a priori* research. In addition, monitoring programmes to assess effects of management measures both on the animal population and on the resource that is the source of the conflict are necessary to ensure the goals of such measures are being met in a cost-effective way without unintended effects on the target animals.

Cormorants *Phalacrocorax* spp are the source of human-wildlife conflicts in a number of regions where their populations are increasing (e.g. Europe – Lindell et al. 1995; Japan – Kameda et al. 2003; North America – Hebert et al. 2005; Ridgway et al. 2011), both due to damage to trees from guano and potential impacts on fish populations. Within Europe, the Great Cormorant *Phalacrocorax carbo* (hereafter Cormorant) population has shown steep increases over the past few decades. This is particularly true of the subspecies *Phalacrocorax carbo sinensis* which is most numerous in the northern parts of continental Europe and has expanded its range and population rapidly (Lindell et al. 1995, van Eerden and Gregersen 1995, Bregnballe et al. 2011, Keller et al. 2012), partly as a result of reduced persecution in breeding colonies and bans on hunting in the major staging and wintering areas. In addition, the coastal breeding subspecies *P.c. carbo* has also shown a tendency to increasingly winter on inland freshwaters in the UK (Rehfisch et al. 1999; Newson et al. 2004). Consequently the Cormorant population expansion has created conflicts with inland fisheries (Feltham et al. 1999) in the UK but also continental Europe. As a result, control measures to limit the expansion of the Cormorant population and to minimise impacts on inland fish stocks have been implemented in several European countries, although in most, no attempt has been made to assess the impact of such control measures on Cormorant populations (Smith et al. 2008). Attempts at controlling populations of double-crested Cormorant *P. auritus* in North America have, however, had mixed results (e.g. Ridgway et al. 2011), although population reduction has been achieved through combined measures of shooting adults and intensive reductions in breeding success (Bédard et al. 1995).

In the UK, in order to prevent serious damage to fisheries, licences have been made available for limited control of Cormorant populations by shooting since autumn 1996. Initially, the numbers involved were small (up to 517 nationally per year), and shooting was considered largely a technique to aid scaring, rather than as a means of population control (Central Science Laboratory 2005), and at a local level, shooting was shown to have affected Cormorant numbers (Parrott et al. 2003). However, in 2004, there was an increase in the number of birds that could be controlled per year, with an upper limit of 3000 individuals in the first two years, and up to 2000 birds annually thereafter. Modelling of the likely consequences of such levels of control predicted a slightly lower, and more-or-less stable national population (CSL 2005; Smith et al. 2008), although the modelling approach was later criticised, casting doubt on the predictions (Green 2008).

The UK holds internationally important waterbird populations (*sensu* Rose and Scott 1997), particularly in winter, and many Special Protection Areas (SPAs) have been designated under the EC Birds Directive (2009/147/EC) on the basis of the numbers of waterbirds that they support, including Cormorants. There is therefore a risk that control measures carried out to protect fishing interests could negatively impact on SPAs. Indeed, of 20 UK SPAs for which Cormorant is a designated feature, Thaxter et al. (2010) reported a sharp decline in Cormorant numbers on three, and for three more a possible increase in the rate of decline, coincident with increased control under the current control licensing scheme. However, the extent to which such changes on SPAs are statistically linked to control intensity at a site level has yet to be determined. In the view of Natural England (the relevant competent authority) "Cormorant control under licence which might affect a SPA would usually be subject to a site-based appropriate assessment by Natural England if likely significant effects on that SPA could not be ruled out".

In this paper, we consider statistical associations between the number of Cormorants controlled and the year-to-year change in the numbers of non-coastal winter Cormorants at a site level in England, using data from the Wetland Bird Survey (WeBS; Pollitt et al. 2003), which is the main source of data used for deriving the annual population estimates of the national winter Cormorant population (henceforth termed the 'Cormorant index'; Chamberlain et al. 2012). The introduction of control measures has created a natural experiment, with some sites not experiencing any control throughout the period considered, whilst others have been subject to control for some or all of the time period, which enables a thorough assessment of possible impacts on site-level populations. Specifically, we test whether Cormorant control in or around sites has affected the magnitude of apparent population changes at these sites, and whether associations with control intensity are related to the scale at which they are considered. In addition, we also consider whether the number of Cormorants on SPAs is associated with control intensity and over what spatial scale such an effect may be apparent. Finally, we compare the results against national-level population trends and discuss the extent to which inferences can be drawn on effects of control from the local to the national scale.

Methods

Bird data

Bird data were derived from WeBS Core Count data, and were available for Cormorant from 1988 to 2009. WeBS provides the principal source of data for deriving population estimates of the UK's non-breeding waterbirds, for assessing the international importance of UK wetland sites and for monitoring long-term trends and waterbird distributions (Pollitt et al*.* 2003). WeBS Core Counts are made using the so-called 'look-see' methodology (Bibby et al*.* 2000), whereby the observer, familiar with the species involved, surveys the whole of a predefined area, which may vary considerably from site-to-site (for the sites used in this analysis, mean area ± SE = 114.5 ± 20.0 ha, range 0.93 to 5815 ha, n = 466 sites with data available). Counts are made at all wetland habitats, including lakes, lochs/loughs, ponds, reservoirs, gravel pits, rivers, freshwater marshes, canals, sections of open coast and estuaries. Numbers of all waterbird species, as defined by Wetlands International (Rose and Scott 1997), are recorded. Counts are made once per month, ideally on predetermined priority dates. This enables counts across the whole country to be synchronised, thus reducing the likelihood of birds being double counted or missed. For this analysis, Cormorant count was taken as the maximum of December to February counts. It is thus assumed that maximum count is representative of the local site-level winter population ('population' here is used in a broad sense to indicate the number of birds in a defined area). This measure is the most relevant to Cormorant monitoring as it is used in deriving the population index (e.g. Chamberlain et al. 2012). Furthermore, peak counts are used as the basis for SPA site designation (Stroud et al. 2001). The vast majority of conflicts are with inland freshwater fisheries, so only non-coastal sites were considered. The analyses are based around winter counts, and control measures in the non-breeding period (September-April), and throughout the paper 'year' is used to refer to the earlier year of a given non-breeding period, as per WeBS protocol (e.g. 2005 indicates autumn and winter 2005/06).

Licensed control data

The first Cormorant control licences were issued in autumn 1996, although only annual totals were available for analysis prior to autumn 2001. A database of the number of Cormorants killed under licence in England was available from 2001 onwards. This included data for each individual licence application and so was site-specific and spatially referenced. Licences usually ran overwinter from September to mid-April of the following year, although there were exceptions (fisheries with salmon or trout were allowed an extension until 1 May, and there was also scope for licences to be granted outside the normal period under exceptional circumstances). The number of Cormorants killed was known for any given licence period. However, the precise timing of control activity was unknown (i.e. the dates on which any kills took place) which necessarily restricts the analysis to temporally broad scales (i.e. winter-to-winter). This has important implications for the estimation of concurrent control intensity (see below).

Although control was usually allowed only outside of the Cormorant breeding season, licences were sometimes granted for longer periods (i.e. over a year), especially between 2004 and 2005. As it was not possible to assign numbers controlled to a given year in these cases, mean values of total Cormorants killed were used when considering overall trends at the national scale (i.e. England), and any such licences (from any year) were not included in any subsequent site-level analyses (see below). Furthermore, the data were for England only, and no information was available on control measures in neighbouring Wales or Scotland. In order to minimise any potential effects of unknown control measures, only sites that were at least 50km distant from the borders of Wales or Scotland were included.

Environmental data

A number of variables have been shown to influence Cormorant winter population growth rates, including the cover of water bodies, the cover of urban land, whether the site is classed as upland or lowland, and the broad geographical location. Following previous work (Jackson et al. 2006, Chamberlain et al. 2012), categories of urban habitat cover and water cover (high, medium or low), habitat class (upland or lowland) and region of England (southwest, southeast, London, East Anglia, midlands, northwest and northeast) were assigned according to the principal 1-km squares of each WeBS site. In addition, winter severity has been shown to be an important determinant of adult survival (Frederiksen and Bregnballe 2000) and is therefore likely to influence Cormorant population growth. Monthly temperature data were available from 2001 to 2006 at a 5x5-km scale from UKCIP. Mean temperature was calculated per winter (Dec-Feb) and assigned to WeBS sites within each 5x5km square.

Statistical analysis

Year-to-year change in Cormorant count (hereafter termed population growth rate, PGR) per site was modelled in relation to Cormorant control within fixed radii of each WeBS site. Licenses were usually granted for relatively small water bodies, and very few of these were WeBS sites (see below). Cormorant control was therefore determined within set radii of each WeBS site, and Cormorant PGR on the WeBS sites was analysed in relation to control intensity in the surrounding landscape, considering scales of 5km, 10km, 20km and 30km radius around each counted site. The goal was to determine if the presence of control activity and its intensity within the surrounding landscape had any effect on the numbers of Cormorants on a given site in the following year. Cormorant control was expressed as an index between 0 and 1, derived from the proportion of the local population that was culled each winter. The local population was the estimated annual winter population in each set radius within which a given WeBS site was situated. This estimate is that developed by Chamberlain et al. (2012) for derivation of the standard Cormorant population index and is based on the total WeBS count for a given 1-km square plus a model-derived estimate based on Dispersed Waterbird Survey data (Jackson et al. 2006). The control index was therefore the number killed under licence for a given radius divided by the estimated population for the same area. In the few cases (n = 57 out of 5753 observations) where the estimated population was lower than the numbers controlled, the index was set at 1.

The data were analysed following the methods of Freeman and Newson (2008), which uses a recursive relationship to allow the expected count at a site to be dependent upon the expected count at the previous year. We expect some temporal autocorrelation in the data, as Cormorants tend to be site faithful in successive winters (e.g. 85-90% site fidelity – Frederiksen et al. 2002). This approach makes better use of the data than conventional modelling approaches, as a count can still be modelled if the previous count at the same site is missing or zero (cf Thomson et al. 1998), resulting in this study in a sample size which is *c*. 25% larger, and consequently greater precision and power in the analysis. In addition to allowing easy estimation and inference about annual growth rates, the Freeman and Newson (2008) approach allows us to model the effects of covariates on population growth, which may themselves vary in space and time. Here we adopted a similar model structure to Newson et al. (2012), but modelling the rate of change in winter Cormorant count from year t-1 to year t in relation to control intensity and environmental variables, with site identity fitted as a fixed effect (Eqn 1).

$$\ln\left(E\left[N_{i,t}\right]\right) = \sum_{j=1}^{t} R_t + S_i + \boldsymbol{\beta_1} \cdot control_t + \boldsymbol{\beta_2} \cdot control_{t-1} + \boldsymbol{\beta_3} \cdot env \qquad \text{(Eqn 1)}$$

Where $N_{i,t}$ is the winter Cormorant count at site i and time t, R_t defines the recursion parameters denoting years, S_i are site effects, β_1, β_2 and β_3 are vectors of fitted parameters for matrices of the control values in year t, the control values in the previous year, and environmental variables, respectively. The control values and environmental variables are matrices of several variables at different radii, and are cumulative variables, so each represents all values at a site up to year t.

Initially, Poisson models suggested overdispersion in the data, and subsequently, models were fitted specifying a negative binomial error structure using the glm.nb command in R 2.12 (R Development Core Team 2010). Effects of control were considered in two separate analyses. First, sites were classified as control (control had taken place in at least one year), versus non-control sites (control never undertaken), within a given radius. This involved fitting rates of change separately for control and non-control sites within the model. This analysis was not dependent on using sites where the period of control could be identified to a fixed period within a given winter (see below), hence it maximised the sample size (n = 5753 observations from 917 sites).

Second, a more detailed analysis was undertaken considering effects of control intensity, i.e. the proportion of the wider population controlled per site per year. Cormorant control could have effects on PGR from year t-1 to year t through delayed effects, i.e. the population growth is affected by the proportion killed in the previous winter, or through concurrent effects, i.e. by the proportion killed in the same winter as the counts. In considering the former, the number killed and the estimated local population within a set radius of each site was simply summed over the duration of the licence and the control index calculated as described above. For the latter, however, there was a problem in that the count period (December-February) was almost always within the licence period, but it was unknown precisely when control was carried out (i.e. the licence period was known, and the number killed, but the control could have been carried out anytime within that period). The effects of concurrent control were therefore analysed by adjusting the numbers controlled by the number of months prior to the WeBS count. First, the month of the maximum count for each WeBS site was determined. Then, the difference between the date of maximum count and the start of the licence duration was determined, and this was then divided by the duration of the licence to give a correction factor between 0 and 1 (in the few cases where the licence began after the count date, the correction factor was set to zero). This was then multiplied by the total killed, making the assumption that control effort was constant across the duration of the licence. Control intensity was then calculated as previously. Cormorant control intensity in year t-1 is termed delayed control, and the adjusted control index in year t as concurrent control.

All WeBS sites and licence locations were spatially referenced in GIS, and the control intensity within different radii of each site in each year were determined, at 5km, 10km, 20km and 30km (sample sizes were very small ($n < 20$) at larger radii). Data for some licences were not used, either due to evident errors or because licences ran for long periods, hence it was not possible to assign numbers controlled to a given year (see above). These were not included in the analysis, hence any radii that contained such data were excluded. For each WeBS site, the probability that unsuitable control data contributed to the calculation of the numbers controlled increased as the radius around the site increased, hence sample sizes become progressively smaller as radii increase. The sample size for the 5km radius (i.e. the maximum sample size) was 4354 observations from 695 WeBS sites, 167 of which had been subject to licensed control.

In common with previous uses of the Freeman and Newson (2008) model (e.g. Chamberlain et al. 2009; Newson et al. 2012), a statistical hypothesis testing approach was adopted in order to assess whether control had a significant effect on Cormorant PGR in the WeBS sites considered. For control measures, both linear and quadratic effects were fitted to the models, but quadratic effects were only retained if significant. There was a relatively strong correlation between concurrent control and delayed control in most years (mean r = 0.55, n = 8 years and 525-592 sites per year). Furthermore, Variance Inflation Factors were high (>5.0) when both variables were considered simultaneously in a given model. Therefore, control measures were modelled separately, with a focus on delayed control, as this measure represented a known total for a given site, and was not reliant on assumptions about the seasonal distributions of control measures.

All models included land class, urban cover class, water cover class and region as categorical variables. The mean winter temperature of each 5x5km square that contained WeBS sites was available for winter 2001/02 to 2005/06, so effects of temperature were considered in a separate analysis (n = 613 sites 2746 observations). Temperature in year t-1 (i.e. the preceding winter, concurrent with delayed control) was considered in the analysis, although temperature in year t (i.e.

concurrent with the bird survey data) and in year t-1 were very highly correlated ($r > 0.85$ in all years).

In order to determine whether control measures may impact on Cormorant numbers on SPA sites, and hence have implications for SPA designation, the site-level analyses were re-run on the subset of 16 non-coastal WeBS sites in the analysis which were SPAs. The majority of licences for Cormorant control were granted for relatively small commercial fishing enterprises on small water bodies which are not included in WeBS and so do not directly contribute to the Cormorant index (although correction factors are included for the population outside WeBS sites – see Chamberlain *et al*. 2012). There were only 14 WeBS sites where control measures were carried out. The site-level analyses were repeated, but only these 14 sites were considered for the control sites in order to assess whether patterns on these sites were consistent with results from the whole sample.

Spatial autocorrelation was assessed by examining the spatial distribution of the residuals by considering variograms. In neither case was there any strong suggestion of spatial autocorrelation in the data (e.g. Fig S1). Similarly, temporal correlation was examined using the ACF command in R, and was found to be low.

Results

The annual totals of Cormorants controlled under licence in England is shown in Fig. 1, along with the Cormorant index for inland sites in winter (from Chamberlain *et al*. 2012). The Cormorant index showed high variability from year-to-year, but there was a general increasing trend in the late 1980s and the 1990s (Fig. 1). The index stabilised and even showed some declines in more recent years, a pattern also reflected in the trends in mean numbers per site for data considered in the site-level analysis (Fig. 2).

Legal control was initiated in 1996/97, initially at fairly low levels, but there was a sharp rise in 2004/05 which followed a change in the licensing policy (note that due to difficulties in assigning numbers controlled to a given year, mean values are assigned to 2004 and 2005 –nevertheless, the increase in numbers controlled is evident; Fig. 1). There was no evidence that trends in Cormorant populations at the national level from year-to-year were linked in any way to trends in control intensity in that there was no correlation between the Cormorant index and either concurrent control (considering only years where control took place, $r_{12} = -0.03$, $P = 0.91$) or delayed control ($r_{11} = 0.02$, $P = 0.94$). For the site-level analysis, the numbers controlled were expressed as an annual rate per site. The trend suggested that there had been some increase in control rate since 2001, but there was a very large rate of control in 2004, the year the new licensing policy was introduced (Fig.

2). There was no significant correlation between the mean number of Cormorants per site per year for the analysis and concurrent control (r_7 = 0.25, P = 0.52), or delayed control (r_6 = -0.14, P = 0.74).

Control versus non-control sites

The model fitting separate trends to sites with and without control over the period considered showed relatively little difference in trends between the two (Fig. 3). The majority of confidence intervals overlapped 1.0, suggesting no significant increases or decreases in the rate of population change over this period, with a few exceptions – there was a significant (P = 0.0004) positive change from 2002 to 2003, and an almost significant (P = 0.053) positive change from 2005 to 2006, both in non-control sites, and a significant (P = 0.009) positive change from 2001 to 2002 in control sites. Similar patterns were evident at larger scales (Fig. S2).

Control intensity

For the site-level control intensity analysis, of the 167 sites where control took place (out of a total of 695 sites), an average of 5.30 ± 4.71 sd Cormorants were controlled per year (n = 420 observations), equating to an average control intensity of 0.34 ± 0.34 sd. There were no significant relationships between delayed control nor concurrent control at the 5km radius and Cormorant PGR (Table 1; full model details are given in Table S1). For delayed control at larger scales, significant non-linear relationships were found at the 10km radius, whilst there was a positive linear relationship at 30km, and no significant relationship at 20km. For concurrent control, there were significant non-linear relationships at the 20km scale (Table 1). The annual rates of population change for the significant relationships between control intensity and PGR at different scales derived from Table 1 are shown in Fig. 4. In each case, a higher proportion of control of the local population was generally associated with population growth, although at the 20km scale, negative growth rate was predicted when less than c. 20% of the local population was controlled.

Repeat analyses were carried out only considering cases where control was actually carried out on a given WeBS site at the 5km scale. Positive relationships between PGR and both delayed control (parameter estimate = 0.363 ± 0.124, z = 2.919, P = 0.004) and concurrent control (parameter estimate = 0.522 ± 0.240, z = 2.177, P = 0.029) were evident, although only 14 control sites were available for analysis (out a total of 542 sites and 3584 observations).

Effects of temperature

When also including temperature in the models with a reduced data set (*n* = 613 sites, 2746 observations), there was a negative relationship with delayed control that approached significance (*P* = 0.053), and a significant negative relationship concurrent control at the 5km scale (Table 2). At larger scales, there was a significant non-linear relationship with concurrent control at 20km radius, which in common with non-linear associations from the whole data set (Fig. 4), predicted positive a trajectory in PGR above a control intensity of *c*. 0.20. There were no other relationships with control intensity at any scale (Table 2). Temperature was not significant in either of these models, and dropping temperature did not affect the significance of the control intensity measures, indicating that the reduced sample, rather than effects of temperature *per se*, were affecting the results relative to those from the full data set.

Effects of control on SPAs

The analysis was repeated for the subset of SPA sites (a maximum of *n* = 16 sites and 137 observations), up to a radius of 20km (there were not enough sites in the sample to consider larger radii). Due to the small sample size, land class was not considered (all sites were lowland) and only three regions were included (southeast, northeast and East Anglia). There were no significant relationships between delayed control, nor concurrent control and PGR at any scale (Table 3).

Discussion

Based on the results here, there is no evidence that Cormorant removal at local scales (5km to 30km radius) has had an effect on longer term (i.e. year-to-year) population size at a site level – put simply, killing Cormorants in one winter did not appear to impact upon numbers at a site level in the next winter. Furthermore, there were no significant relationships between control intensity and Cormorant PGR on SPAs, and therefore control measures did not have an adverse effect on the objectives under the designation of these sites, although the small sample sizes should be noted. The lack of evidence for negative effects of control, despite a national-level decrease in population growth (Fig. 1), may imply that other factors are influencing the wider population trend, including density-dependent effects (i.e. the population has reached carrying capacity), which have been detected in other populations (Frederiksen et al. 2001), changes in factors affecting reproductive success and/or survival, or changes in immigration (although annual immigration rate is thought to be low anyway – Wernham et al. 1999).

Apparent positive effects of control

A number of models considering different measures of control at different scales showed significant positive relationships between control and PGR, or where the trend was non-linear, showed positive relationships over the greater part of the distribution of control intensity measures, e.g. predicted positive relationships from a control intensity of c. 0.20 onwards (Fig. 3), which is well below the mean of 0.33. There was a single model where there was a significant negative relationship between control and Cormorant population growth, that of concurrent control at a 5km radius when considering the subset with temperature data (Table 2). However, given that this result was based on a restricted number of years, that most analyses indicated either positive relationships or no relationship with control, and that the magnitude of this negative relationship with population growth was effectively balanced by positive relationships at larger scales, these results must be considered at best weak evidence of negative impacts of control on winter Cormorant populations. It should also be noted that that this and several other results were only weakly significant – if applications for multiple testing were applied, then the evidence for relationships between control intensity and Cormorant PGR would be even weaker (although we concur with criticisms of formal adjustments for multiple testing (e.g. Moran 2003) and do not apply them here).

The general pattern of results suggested more Cormorants controlled at a site level was associated with higher rates of population growth. There are four mechanisms by which positive relationships with control intensity may arise. First, the removal of residents may simply result in replacement of more birds via a density-dependent response, which seems plausible in an expanding population. However, if numbers at a site level are limited by density-dependent processes, then the expectation would be that birds replacing those controlled would re-colonise up to the level of the previous population, but results here suggest they may exceed it. Second, there may be significant disturbance caused by control measures which may alter birds' behaviour, for example by making them more dispersive, which may lead to apparent population increases. It is possible that short-term disturbance effects of control carried out in the autumn and early winter could have immediate effects on bird behaviour in the January and February of the following year, as suggested by results using concurrent control. However, similar results were also evident for delayed control, and such disturbance effects seem implausible given the long time span between controls and counts. Third, as control was typically not carried out on WeBS sites, a positive association may arise if control measures force birds to move into WeBS sites, which act as refuges. Although sample sizes were small, this seems unlikely given that on the few WeBS sites that were also subject to control measures, there were also significant positive relationships detected. Fourth, licensed control may be sought in anticipation of increased Cormorant predation prior to enhanced fish stocking or other management changes that increase local fish populations, and which therefore subsequently attract more Cormorants. Fifth, the positive results may arise as control measures may be reactive, i.e. licences are granted at short notice (which is commonplace – Natural England 2012) in response to local increases in Cormorant numbers. This would suggest that control measures are undertaken on the sites with the greatest growth rates, but also that such measures do not have significant impacts on the increasing local population.

Caveats on the analysis

The analytical approach adopted was based on year-to-year change in numbers at the site level in relation to control intensity, thus there is an underlying assumption that populations are linked from one winter to the next. This was supported by previous research which has shown high site fidelity from winter-to-winter (Reymond and Zuchuat 1995; Lekuona and Campos 2000; Frederiksen et al. 2002), and to some extent by the lack of strong spatial autocorrelation (Fig. S1). Nevertheless, wintering Cormorants do sometimes make long distance movements (Schifferli et al. 2011), and are able to respond to locally abundant food supplies (Richner 1995). In order to determine if the statistical approach was in effect too conservative in detecting effects of control on local populations, a further simpler analysis was undertaken where no year-to-year dependence was assumed – a Poisson model considering the effects of numbers controlled on numbers at a given site. The results were qualitatively similar in that all parameter estimates were positive, although there was only a single significant effect (Table S2). Therefore we conclude that the assumption of year-to-year dependence did not affect our main conclusion that there was no negative effect of control on local Cormorant population size.

The analyses used maximum count per winter as the response variable. This was chosen in part because maximum count is the ‘currency’ for Cormorant monitoring in England, being used to derive the Cormorant index (Chamberlain et al. 2012) and also being the basis for SPA designation (which uses the mean of five-year peak counts per site, Stroud et al. 2001). The analysis is therefore underpinned by the assumption that the maximum count is representative of the population using a given site. Using the mean is a possible alternative that would incorporate more the variability in counts, but in fact the mean and maximum counts across sites were very highly correlated (e.g. r =

0.967 across all 5017 sites/years) suggesting a degree of consistency in counts across visits within sites. The use of maximum count was also appropriate for the temporal resolution of the control data, which could only be summarised at the level of the whole winter at best. The approach therefore may detect relatively strong effects of control which affect the year-to-year change in maximum count, but more subtle effects of control would not be detected by this method. For example, there may be short-term effects of mortality followed by rapid recovery by new colonists within a given winter, or numbers may be temporarily reduced at a given site through disturbance effects. Interestingly, Parrott et al. (2003) found an effect of shooting on local Cormorant populations in a relatively small-scale study (13 sites), but there was no difference between lethal and non-lethal shooting, suggesting that disturbance effects may occur. However, from a policy perspective, the effect of the control measures undertaken in England is explicitly linked to year-to-year change in terms of Cormorant monitoring (i.e. through the Cormorant index; Chamberlain et al. 2012) and SPA designation (Stroud et al. 2001). Nevertheless, it would be interesting to develop analytical techniques that can assess potentially more subtle within-winter effects, although the temporal resolution of the control data should ideally be higher for such an approach.

Future research needs

The positive relationships with control intensity detected may suggest more subtle effects involving the part of the population outside of the monitored WeBS sites (usually small water bodies). The Cormorant index is largely based on WeBS sites, although an estimate of the numbers outside of these sites is also included, derived from the Dispersed Waterbird Survey (DWS; Jackson et al. 2006): between 64% and 70% of numbers contributing to the index per year (2001-2009) are from WeBS sites. Furthermore, it should be noted that DWS was from a single year, 2003, and that extrapolations of PGR for DWS estimates are also derived from WeBS trends. We therefore conclude that the English winter Cormorant population as measured by the Cormorant index is not negatively influenced by control measures, but we need to add the caveat that not enough is known about the population outside of WeBS sites (i.e. those not contributing to the national index) which are poorly monitored, but which may nevertheless be crucial in understanding potential responses to control measures. A further survey of Cormorants in the wider countryside, following DWS methods, is needed to understand the effects of control intensity not included in the Cormorant index, and how these interact with those that are (e.g. through disturbance, and non-control sites acting as refugia), is needed before firmer conclusions can be drawn on effects of control on the national population trend.

Lethal control measures undertaken to resolve human-wildlife conflicts often have mixed results (e.g. Donelly et al. 2006; Ridgway et al. 2011), and may only be successful when intensive measures cause very high mortality rates (e.g. Bédard et al. 1995). In order to maximise the chances of success, such approaches need to be underpinned by sound science. Modelling potential effects of such interventions is a potentially useful tool, although assumptions underlying such approaches need careful consideration. Behavioural responses may be particularly difficult to anticipate. For example, badger *Meles meles* culls to reduce their population and hence reduce transmission of badger-borne tuberculosis to cattle have sometimes had the opposite effect, due to unexpected disruption to territorial behaviour which caused badgers to disperse more widely than they would otherwise have done (Carter et al. 2007). In the light of this, we suggest that a better understanding is developed of site use and movements of individual Cormorants (including those due to disturbance caused by control measures) at smaller scales through more intensive research using mark-resighting or remote tracking of individuals. Furthermore, although control measures do not have any apparent effect on local Cormorant populations, we cannot conclude that there is no effect on Cormorant behaviour (including foraging efficiency) at these sites.

Given that ultimately the goal of the control measures is to reduce conflicts with fishing interests, we suggest that a greater priority is needed for research into assessing whether control has the desired impact on predation rates of fish (e.g. either directly through mortality or indirectly through disturbance), and the extent to which the cost of control measures compares against other measures to reduce Cormorant predation, e.g. scaring techniques including non-lethal effects of shooting (Parrott et al. 2003) and providing better fish refuges (Russell *et al*. 2008), and so help resolve the conflict between Cormorants and fisheries.

Acknowledgements This work was funded through the Food and Environment Research Agency (Fera) by the UK Department for Environment, Food and Rural Affairs (Defra). Our thanks go to Tim Andrews and Ashley Smith from Defra and Graham Smith from Fera for their help and comments, and to members of the Fish-eating birds policy review group and to Rhys Green and Sarah Dove (RSPB) for their comments. We are also indebted to Thomas Bregnballe, who provided highly constructive suggestions on a previous draft, and to Verena Keller for providing additional references. Spatial data on numbers of birds licensed to be and actually shot, and advice on policy were provided by Natural England. Site-specific data on Cormorant numbers were obtained from the Wetland Bird Survey (WeBS), a joint scheme of the British Trust for Ornithology (BTO), Royal Society

for the Protection of Birds (RSPB) and Joint Nature Conservation Committee (JNCC), in association with Wildfowl & Wetlands Trust (WWT), that aims to monitor non-breeding waterbirds in the UK.

Supplementary Material

Cormorant control manuscript supplementary material Oct2012.doc – this file contains additional details and results to support the main analysis and is intended for review.

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Table 1. Relationships between the proportion of local population of Cormorants controlled at different radii around the count sites and Cormorant PGR. Models also included urban habitat category, water cover, landscape class, and region (further details in Table S1) of the central 1-km square of each WeBS site. Models assumed negative binomial errors and included fixed site effects. (a) Relationships with control intensity in the previous winter (delayed control - $CONTROL_{t-1}$). (b) Relationships with control intensity in the winter concurrent with the Cormorant counts (concurrent control - $CONTROL_t$). N_{sites} is the number of sites in the model, N_{obs} is the number of observations (i.e. site/years).

Scale	Variable	N_{sites}	N_{obs}	Parameter estimate	SE	*z*	*P*
(a)							
5km	$CONTROL_{t-1}$	695	4354	-0.030	0.068	-0.445	0.657
10km	$CONTROL_{t-1}$	506	3225	0.384	0.184	2.090	0.037
	$CONTROL_{t-1}^2$			-0.190	0.077	-2.469	0.014
20km	$CONTROL_{t-1}$	211	1406	-0.103	0.394	-0.261	0.794
30km	$CONTROL_{t-1}^2$	57	417	5.071	2.112	2.401	0.016
(b)							
5km	$CONTROL_t$	695	4354	-0.071	0.091	-0.777	0.437
10km	$CONTROL_t$	506	3225	-0.094	0.118	-0.796	0.426
20km	$CONTROL_t$	211	1406	-3.335	1.419	-2.351	0.019
	$CONTROL_t^2$			7.462	3.373	2.212	0.027
30km	$CONTROL_t$	57	417	1.114	1.465	0.761	0.447

Table 2. Relationships between the proportion of local population of Cormorants controlled at different radii around the count sites and Cormorant PGR, when including temperature of the previous winter. (a) Relationships with control intensity in the previous winter (delayed control - $CONTROL_{t-1}$). (b) Relationships with control intensity in the winter concurrent with the Cormorant counts (concurrent control - $CONTROL_t$). N_{sites} is the number of sites in the model, N_{obs} is the number of observations (i.e. site/years). Other details as per Table 1.

Scale	Variable	N_{sites}	N_{obs}	Parameter estimate	SE	*z*	*P*
(a)							
5km	$CONTROL_{t-1}$	613	2746	-0.388	0.200	-1.938	0.053
10km	$CONTROL_{t-1}$	448	2044	0.312	0.229	1.357	0.175
20km	$CONTROL_{t-1}$	184	874	-0.480	1.113	-0.426	0.670
30km	$CONTROL_{t-1}^2$	55	265	-0.623	1.860	-0.335	0.738
(b)							
5km	$CONTROL_t$	613	2746	-0.428	0.213	-2.012	0.044
10km	$CONTROL_t$	448	2044	0.017	0.236	0.073	0.941
20km	$CONTROL_t$	184	874	-5.689	2.325	-2.447	0.014
	$CONTROL_t^2$			13.430	6.282	2.138	0.032
30km	$CONTROL_t$	55	265	-0.696	1.799	-0.387	0.699

Table 3. Relationships between numbers of Cormorants controlled at different radii around count sites designated as SPAs and Cormorant PGR(a) Relationships with control intensity in the previous winter (delayed control - $CONTROL_{t-1}$). (b) Relationships with control intensity in the winter concurrent with the Cormorant counts (concurrent control - $CONTROL_t$). Due to the small sample size, some categories used in other models were redundant. Models included region (southeast, northeast and East Anglia), urban cover class (high or medium), and water cover class (high, medium or low).

Scale	Variable	N_{sites}	N_{obs}	Parameter estimate	SE	*z*	*P*
(a)							
5km	$CONTROL_{t-1}$	16	137	0.599	2.432	0.246	0.805
10km	$CONTROL_{t-1}$	13	112	2.660	1.916	1.390	0.164
20km	$CONTROL_{t-1}$	8	67	1.184	1.474	0.804	0.422
(b)							
5km	$CONTROL_t$	16	137	-0.419	2.811	-0.149	0.881
10km	$CONTROL_t$	13	112	-1.423	3.119	-0.456	0.648
20km	$CONTROL_t$	8	67	-1.927	2.990	-0.644	0.519

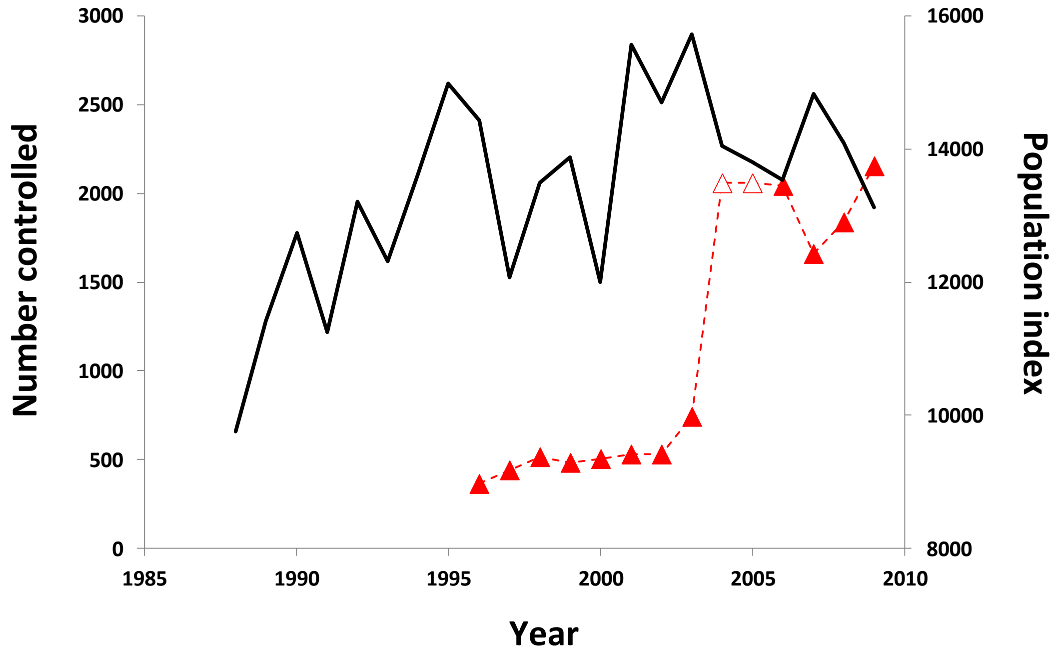

Fig. 1. Total annual inland winter Cormorant population index (solid line) and the annual number of Cormorants controlled under licence (dashed line). Note that due to difficulties in assigning numbers controlled to a given year in 2004 and 2005, the mean value over the two years is presented for each (open triangles). The Cormorant population index is taken from Chamberlain et al. (2012).

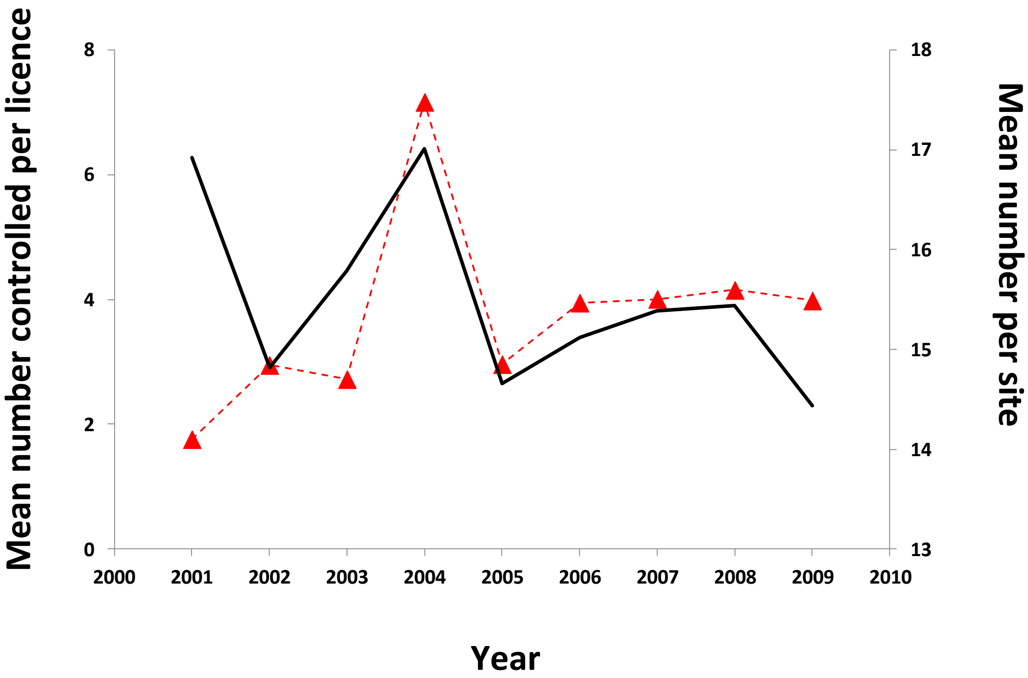

Fig. 2. Mean number of Cormorants per site per year (solid line) and the annual number of Cormorants controlled per licence (dashed line) for the period for which there were spatially referenced control data, and for sites used in the site-level control intensity analysis where control measures could be assigned to specific years; *n* = 695 sites overall (525-592 per year).

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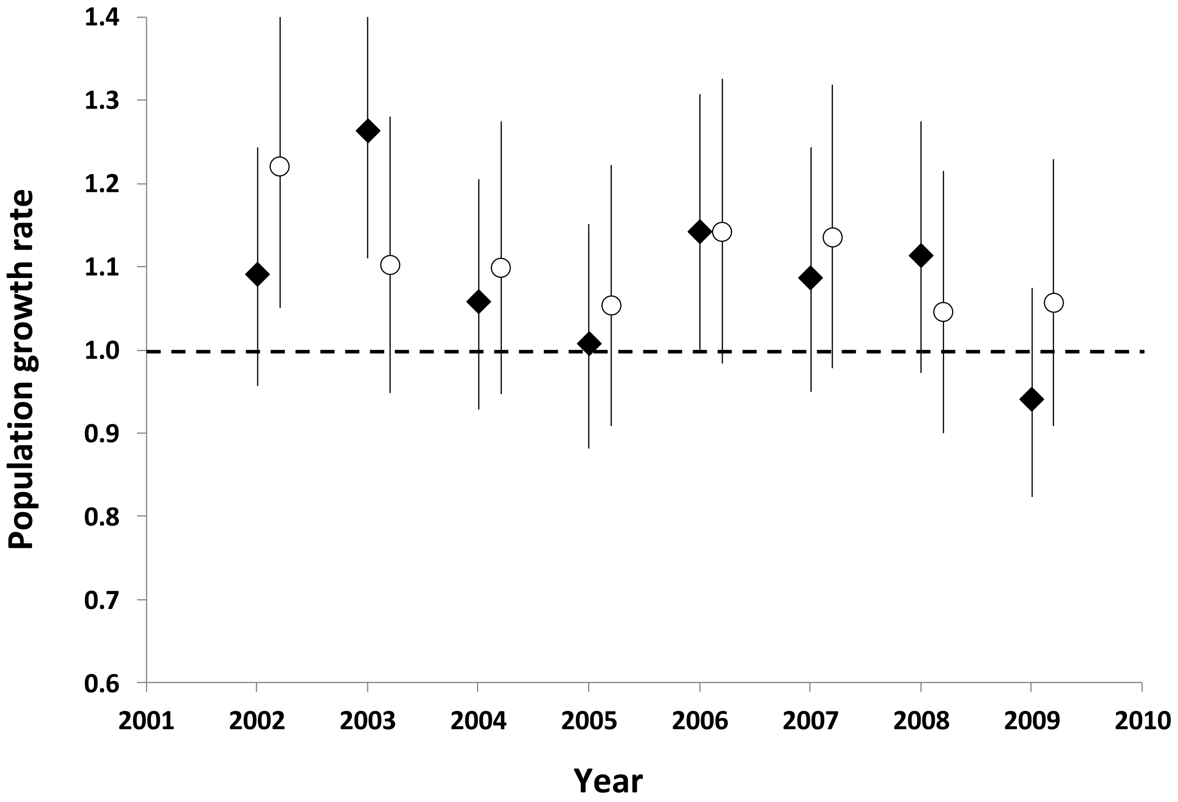

Fig. 3. Estimated population growth rates of winter Cormorant populations in sites where no control ever took place (black diamonds) and those where control took place in at least one year (open circles) within a 5-km radius of the site. Estimates were back-transformed from a negative binomial model of year-to-year change. The dashed line at 1.0 indicates zero population growth. The models included site as a fixed effect and water area within a 5km radius (set to zero in this model) as a covariate. Error bars represent 95% confidence intervals. *n* = 5753 observations from 917 sites.

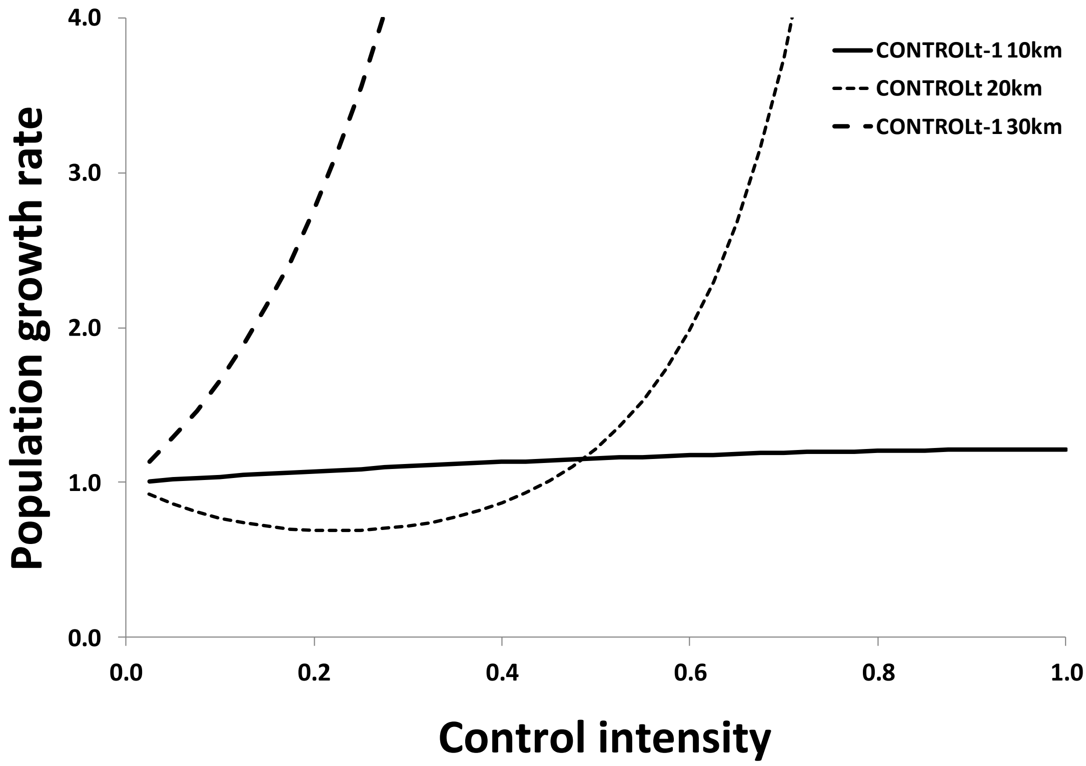

Fig. 4. Predicted relationships between Cormorant control intensity in the previous year (delayed control - $CONTROL_{t-1}$) and in the current year concurrent control - ($CONTROL_t$) within different radii of a given WeBS site, and Cormorant relative population rate of change derived from the models presented in Table 1. All other variables in the model (site effects, water, urban and landscape class, and region) have been set at zero. Relationships were significant ($P < 0.05$) in each case.

Supporting information

Table S1. Modelled rate of cormorant winter population growth in relation to control intensity, and urban habitat category, water cover category ('high' is the reference category for both, with Parameter = 0), landscape class (LS; 'Upland' reference category), and region (East Anglia reference category) of the central 1-km square of each WeBS site. Models assumed negative binomial errors and included fixed site effects. r1 to r8 represents the estimated rate of change from year to year, where r1 is from 2001 to 2002. (a) Effects of numbers controlled in the previous winter (delayed control - $CONTROL_{t-1}$). (b) Effects of control in the winter concurrent with the Cormorant counts (concurrent control - $CONTROL_t$). N = 695 sites, 4354 observations.

	Parameter estimate	SE	z	P
(a)				
$CONTROL_{t-1}$	-0.030	0.068	-0.445	0.657
Urban(medium)	-0.019	0.014	-1.355	0.175
Urban(low)	-0.004	0.015	-0.250	0.802
Water(medium)	0.004	0.013	0.299	0.765
Water(low)	0.007	0.012	0.530	0.596
LS(lowland)	-0.097	0.029	-3.371	0.001
r1	0.125	0.062	2.000	0.045
r2	0.255	0.062	4.128	0.000
r3	0.113	0.062	1.834	0.067
r4	0.034	0.063	0.536	0.592
r5	0.178	0.063	2.808	0.005
r6	0.130	0.063	2.068	0.039
r7	0.146	0.063	2.328	0.020
r8	0.008	0.062	0.128	0.899
London	-0.068	0.026	-2.625	0.009
Southeast	-0.033	0.016	-2.077	0.038
Southwest	-0.030	0.021	-1.393	0.164
Midlands	-0.009	0.017	-0.543	0.587
Northeast	-0.051	0.018	-2.811	0.005
Northwest	0.006	0.022	0.259	0.795
Intercept	-1.569	0.736	-2.133	0.033

	Parameter estimate	SE	z	P
(b)				
CONTROL$_t$	-0.071	0.091	-0.777	0.437
Urban(medium)	-0.019	0.014	-1.321	0.186
Urban(low)	-0.004	0.015	-0.241	0.809
Water(medium)	0.004	0.013	0.313	0.755
Water(low)	0.007	0.012	0.556	0.578
LS(lowland)	-0.097	0.029	-3.368	0.001
r1	0.124	0.062	1.980	0.048
r2	0.255	0.062	4.123	0.000
r3	0.113	0.062	1.832	0.067
r4	0.033	0.063	0.528	0.598
r5	0.177	0.063	2.801	0.005
r6	0.130	0.063	2.060	0.039
r7	0.146	0.063	2.318	0.020
r8	0.008	0.062	0.134	0.894
London	-0.067	0.026	-2.604	0.009
Southeast	-0.033	0.016	-2.072	0.038
Southwest	-0.029	0.021	-1.377	0.169
Midlands	-0.009	0.017	-0.526	0.599
Northeast	-0.051	0.018	-2.786	0.005
Northwest	0.005	0.022	0.255	0.799
Intercept	-0.067	0.026	-2.604	0.009

Table S2. Relationships between the number of local cormorants controlled at different radii around the count sites and cormorant count per winter. Models assumed negative binomial errors and included fixed site effects. (a) Relationships with control intensity in the previous winter (delayed control - $CONTROL_{t-1}$). (b) Relationships with control intensity in the winter concurrent with the cormorant counts (concurrent control - $CONTROL_t$). N_{sites} is the number of sites in the model, N_{obs} is the number of observations (i.e. site/years).

Scale	Variable	N_{sites}	N_{obs}	Parameter estimate	SE	χ^2	P
(a)							
5km	$CONTROL_{t-1}$	695	4354	0.011	0.007	2.52	0.113
10km	$CONTROL_{t-1}^2$	506	3225	0.009	0.006	2.43	0.119
20km	$CONTROL_{t-1}$	211	1406	0.007	0.004	2.70	0.101
30km	$CONTROL_{t-1}$	57	417	0.005	0.005	1.13	0.289
(b)							
5km	$CONTROL_t$	695	4354	0.010	0.009	1.49	0.223
10km	$CONTROL_t$	506	3225	0.001	0.006	0.05	0.820
20km	$CONTROL_t$	211	1406	0.006	0.006	1.03	0.311
30km	$CONTROL_t$	57	417	0.003	0.008	0.10	0.756

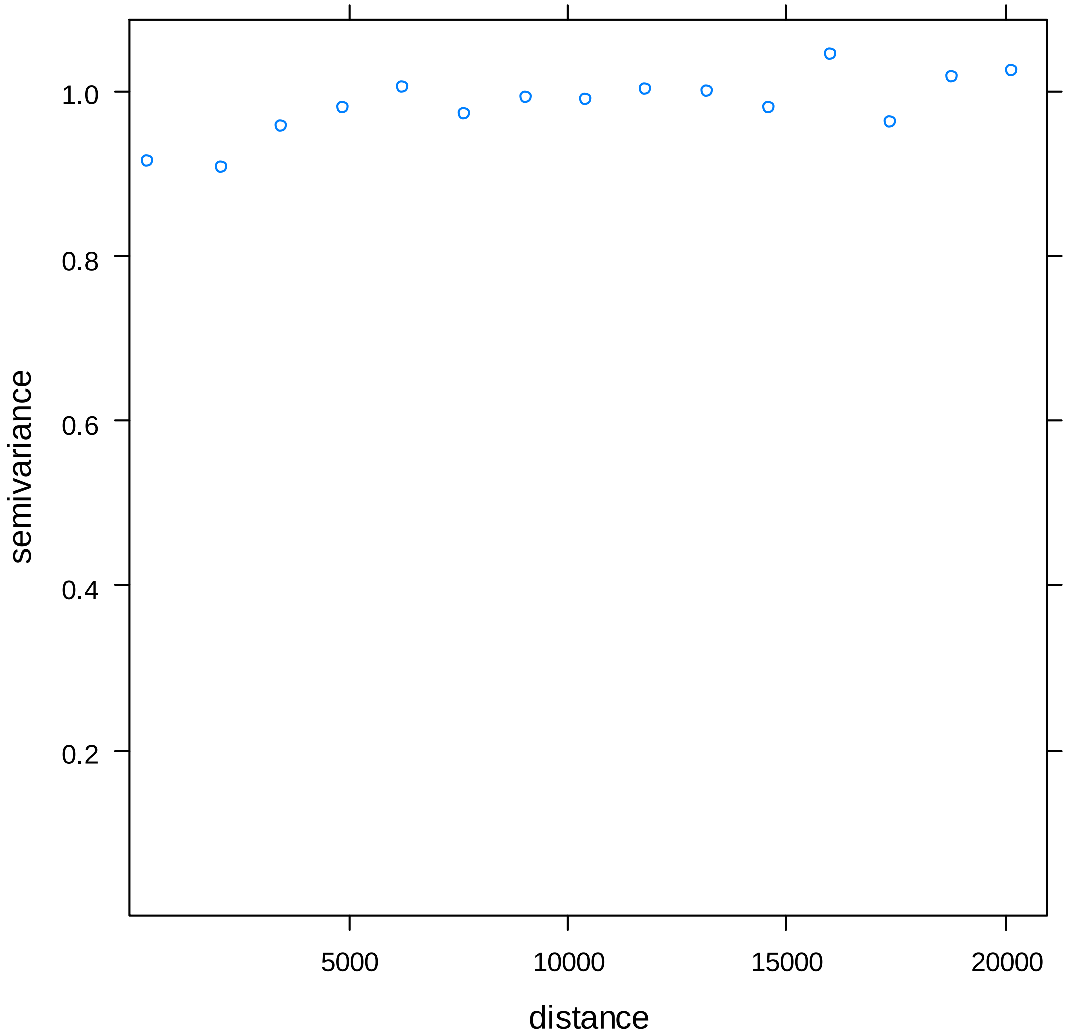

Figure S1. Variogram of residuals plotted against distance derived from the model of $CONTROL_{t-1}$ at the 5km scale. There was some slight positive correlation at small scales and at larger scales, but overall the evidence for spatial autocorrelation was weak.

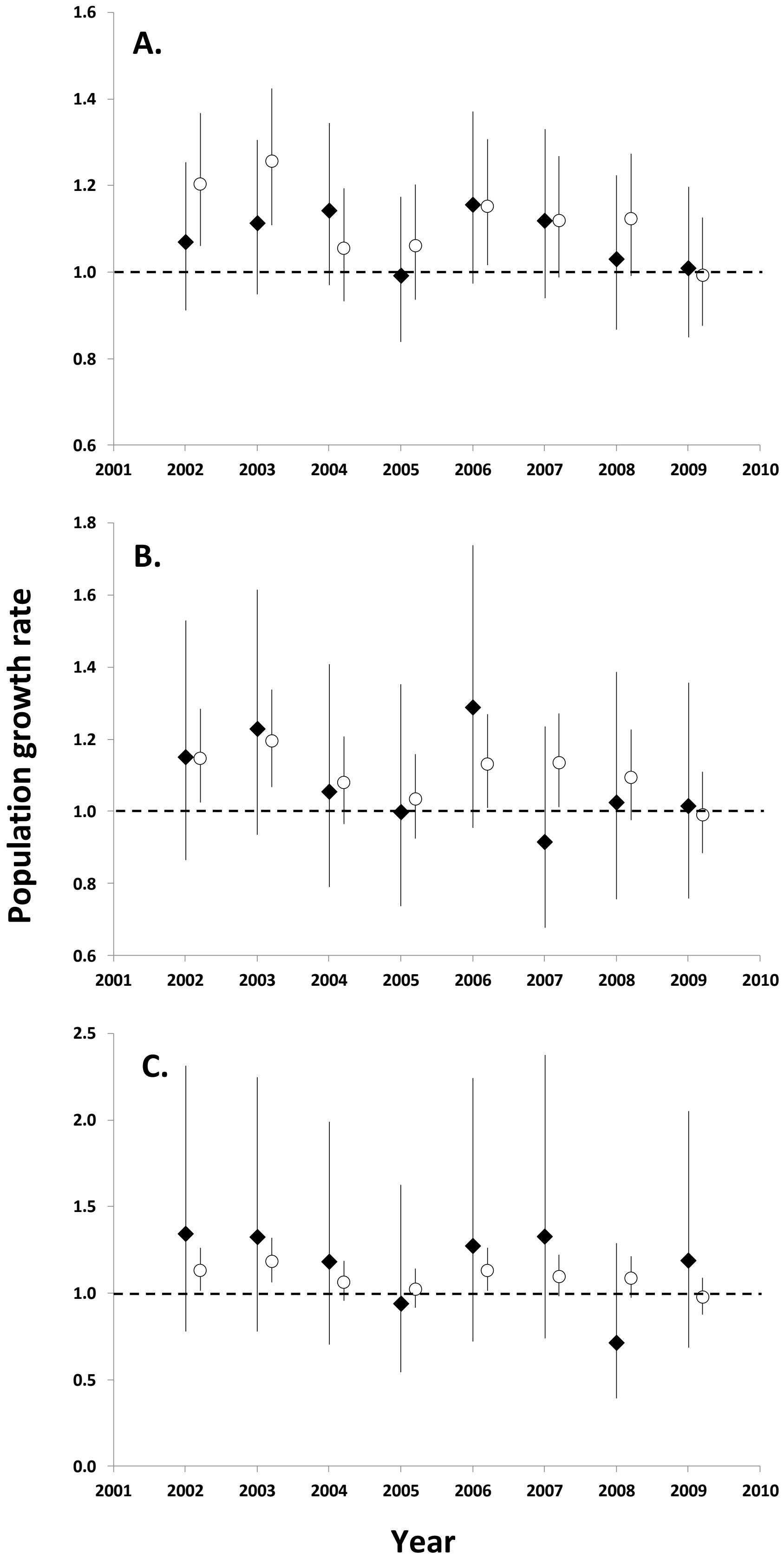

Figure S2. Estimated population growth rates of winter cormorant populations in sites where no control took place (black diamonds) and those without control (open circles at different radii around each site. A 10km, B 20km, C 30km. N = 5753 observations from 917 sites. Note that the number of no control sites decreases (and hence errors increase) as the radius increases. Other details as per Fig. 2.