Management options for a resident Barnacle Goose population in Flanders

A comparison of different scenarios using population modelling

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MANAGEMENT OPTIONS FOR A RESIDENT BARNACLE GOOSE (BRANTA LEUCOPSIS) POPULATION IN FLANDERS

A comparison of different scenarios using population modelling

Frank Huysentruyt, Pieter Verschelde, Toon Van Daele, Jim Casaer, Axel Neukermans and Tim Adriaens

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Flanders
State of the Art

Acknowledgments/Introduction

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Abstract

In Flanders, Barnacle Goose wintering numbers have, in line with a pan-European trend for the species, drastically increased over the last decades. Simultaneously, a local breeding population has emerged, probably of mixed feral and wild origin. The species is protected by the European Union Birds Directive with options for crop damage compensation payments in Flanders. The presence of large numbers of other breeding goose species in Flanders already led to various problems, resulting in an ongoing broad-scale management of these populations. Both the rise in Barnacle Goose wintering numbers and the emergence of a breeding population have therefore raised concerns, mainly in the light of diminishing Canada Goose numbers as a result of management, where an increase in breeding Barnacle Goose numbers could potentially counter the benefits gained from this management.

The complex situation of Barnacle Goose in Flanders, combined with anticipated conflicts in management objectives and means, fuels the need for a better informed management. This study examines different Barnacle Goose life-history parameters in the Flanders' breeding population under the hypothesis that the absence of migratory efforts would increase reproductive output and population growth rate. Demographic data were derived using detailed monitoring and standard capture-recapture analyses and further used to construct a stage-based population model. This model allows examination of the expected development of the breeding population and the evaluation of different management scenarios.

Contrary to the initial hypothesis, neither reproductive output nor population growth rate exceeded those of other Barnacle Goose populations, both migratory and non-migratory. The yearly average growth rate of the population was 1.12, which corresponds to a population doubling every 6.12 years. At the current breeding population level and expected thresholds for acceptance of goose damage, we estimate a 5 to 10 year time frame for implementation of a management scheme.

Finally, elasticity analysis shows adult survival has the greatest impact on population growth rates. Therefore, a management scheme aimed at the reduction or entire removal of the Barnacle Goose breeding population, should consider methods that impact adults rather than chick, juvenile or subadult survival. A simulation of three different management scenarios (nest destruction, culling and a combined scenario) showed that culling birds through moult captures is currently the most effective method to affect population growth rate. Also, this method is practical to implement and has immediate effect on population numbers.

Nederlandstalige samenvatting

De aantallen overwinterende brandganzen in Vlaanderen namen de laatste decennia sterk toe, in overeenstemming met Europese trends voor de soort. Tegelijk ontwikkelde zich een lokale broedpopulatie, vermoedelijk van zowel wilde als verwilderde herkomst. De soort geniet bescherming binnen de Europese Vogelrichtlijn gecombineerd met de mogelijkheid tot het vergoeden van landbouwschade in Vlaanderen. De aanwezigheid van grote aantallen broedende ganzen van andere soorten leidde in het verleden in Vlaanderen al tot verschillende problemen, die resulteerden in een breedschalig beheer van deze soorten, dat tot op vandaag wordt toegepast. Hierdoor hebben zowel de toename van het aantal overwinterende brandganzen als het ontstaan van een broedpopulatie tot bezorgdheid geleid. Vrees is daarbij vooral dat positieve effecten van afnemende aantallen Canadese ganzen als gevolg van populatiebeheer, teniet zouden kunnen worden gedaan door een toename van het aantal broedende brandganzen.

De complexe toestand van brandganzen in Vlaanderen, in combinatie met de verwachte conflicten tussen beheerdoelen en -middelen, voedt de nood aan een beter geïnformeerd beheer. Deze studie onderzoekt verschillende parameters van de populatieontwikkeling bij in Vlaanderen broedende brandganzen. Hierbij wordt uitgegaan van de hypothese van een verhoogde reproductie en groeisnelheid van de populatie door het feit dat de vogels geen energie in migratie moeten investeren. Populatie-demografische parameters werden verzameld via gedetailleerde monitoring en standaard vangst-hervangst analyses en werden verder gebruikt om een populatiemodel te ontwikkelen. Dit model laat toe de verwachte groei van de broedpopulatie te onderzoeken en verschillende beheerscenario's te simuleren.

In tegenstelling tot de initiële hypothese bleek noch de productiviteit noch de groeisnelheid van de populatie hoger te liggen dan in andere populaties van brandganzen, zowel migratorisch als niet-migratorisch. De vastgestelde jaarlijkse populatiegroei was 1.12, wat overeenkomt met een populatie die elke 6.12 jaar in aantal verdubbelt. Met het huidige niveau van de broedpopulatie en met de verwachte drempelwaarden voor het aanvaarden van ganzenschade, schatten we een tijdsraam in van 5 tot 10 jaar waarbinnen een beheerplan kan worden geïmplementeerd.

Mocht een dergelijk beheerplan, gericht op de reductie of totale verwijdering van de broedpopulatie van brandgans in Vlaanderen, worden beoogd, dan toont een analyse van drie verschillende beheerscenario's (nestvernieling, doden en een gecombineerd scenario) dat het doden van vogels via ruivangsten momenteel het meest wenselijke scenario is om tot snelle en effectieve resultaten te komen.

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1 INTRODUCTION

The increase in population sizes of many western Palearctic goose species presents several challenges to decision makers and wildlife managers (Madsen et al. 1999, Fox et al. 2010). Expanding populations of farmland foraging geese are causing human-wildlife conflicts, not only due to crop damage, but also biodiversity interests can be harmed (Adriaens et al. 2012, Van Daele et al. 2012, Johnson et al. 2014, Buij et al. 2017). At the same time, migratory geese populations are regarded as a high conservation priority due to their strong reliance on restricted sites along their migration routes, their vulnerability to environmental change and to over-harvesting (Kirby et al. 2008, Madsen et al. 2015). Hence, the presence and increase in geese populations often results in conflicting interests and objectives for geese management (Tombre et al. 2013, Fox et al. 2017). The need for improved management of goose populations, balancing viable population sizes with acceptable levels of damages, has triggered international cooperation. In Europe, a coordinated mechanism is ensured through the Goose Management Platform within the African-Eurasian Migratory Waterbird Agreement (AEWA; http://www.unep-aewa.org/). This platform aims to address sustainable use of goose populations and solve human-goose conflicts. As a matter of priority, management plans are currently under development for Barnacle Goose (*Branta leucopsis*) and Greylag Goose (*Anser anser*) populations and for Taiga Bean Goose (*Anser fabalis*) and the Svalbard population of the Pink-Footed Goose (*Anser brachyrhynchus*), plans are already in place (Madsen et al. 2015).

In Flanders, most wintering goose species are legally considered game species but only Canada Goose (*Branta canadensis*) and Greylag Goose have open hunting seasons. All other goose species, game species or not, with the exception of non-native species, are protected. For Barnacle Goose, the emergence of a local breeding population has challenged this status, especially during summer months. To counter this, the option of a derogation for management by trapping has been provided within Flemish species protection laws (BVR 2009). As Barnacle Goose is regionally protected (BVR 2009), listed on Annex I of the European Union Birds Directive (EU Directive 2009/147/EC), such diversion from the protected status should be based on a thorough evidence base (Adriaens et al. 2012). However, scientific data on local population dynamics are currently lacking for the Flanders' breeding population. This research project aimed to fill this gap by providing the necessary life history characteristics, modelling the population development based on these parameters and using this output for the comparison of different potential management strategies.

1.1 CURRENT STATUS, MANAGEMENT AND OBJECTIVES

Until the early 1990s Barnacle Goose was an uncommon wintering migrant in the coastal plains of Flanders, Belgium (Devos et al. 1998). Coinciding with a rapid increase in European population numbers and the expansion of the breeding ranges into lower latitudes, wintering numbers in Flanders started to rise from the late 1990s on (Larsson et al. 1988, van der Jeugd & Litvin 2006, Fox et al. 2010, Van Daele et al. 2012, Jonker et al. 2013, Devos & T'Jollyn 2017) (Figure 1). The first record of Barnacle Geese breeding in Flanders dates back to 1992, with numbers slowly increasing up to an estimated 200 breeding couples in recent years (Anselin & Vermeersch 2005, Van Daele et al. 2012). Although the population currently breeding in Western Europe is commonly presumed feral, some evidence exists that part of this population emerged from migratory populations (Ouweneel 2001, Anselin & Vermeersch 2005, Feige et al. 2008, Jonker et al. 2012). This fits the general trend of spring migration delay and shortening of migration distances observed in all Barnacle Goose populations (Jonker et al.

2013). Since the 1980s breeding colonies have been established at various locations along the traditional migration route, including the Zeeland Delta region (North Sea population) (Jonker et al. 2011). It is assumed that the current Flemish breeding population originated from escaped or released birds, which was documented at various locations throughout the region (Beck et al. 2002, Devos & Kuijken 2008). There is, however, some evidence that wild geese also breed in Flanders and there are strong suspicions of exchanges between the Flemish and Zeeland population (Devos 2011). As such, Barnacle Geese breeding in Flanders are a mixture of feral birds originating from escaped Barnacle Geese which are usually perceived as problem birds, supplemented with wild birds originating from the Russian-Baltic population which are under a strict protection regime. On the European scale, the species is protected by the European Union Birds Directive and cannot be harvested through regular hunting (EU Directive 2009/147/EC on the conservation of wild birds, Annex I). In Flanders, the species is protected by a Decree on Species Conservation (BVR 2009) including the possibility for compensation payments to farmers in case of damage. The presence of large numbers of breeding goose species like non-native Canada Goose, Egyptian Goose (*Alopochen aegyptiaca*) and native Greylag Goose in Flanders already led to various problems, resulting in an ongoing broad-scale management of breeding goose populations (Van Daele et al. 2012, Reyns et al. 2018). The rise in Barnacle Goose numbers has therefore also raised concerns, mainly in the light of diminishing Canada Goose numbers as a result of ongoing management actions, where a broad scale presence of the former species during the breeding season could potentially counter the benefits gained from the management of the latter species (see Reyns et al. 2018). The lack of knowledge on the origin of resident populations and the exchange of summering birds with wild breeding or wintering populations currently impedes a clear policy choice. As a result, there is currently no systematic coordinated management of the species in Flanders and Barnacle Goose problems are mainly tackled ad hoc in light of damage reduction.

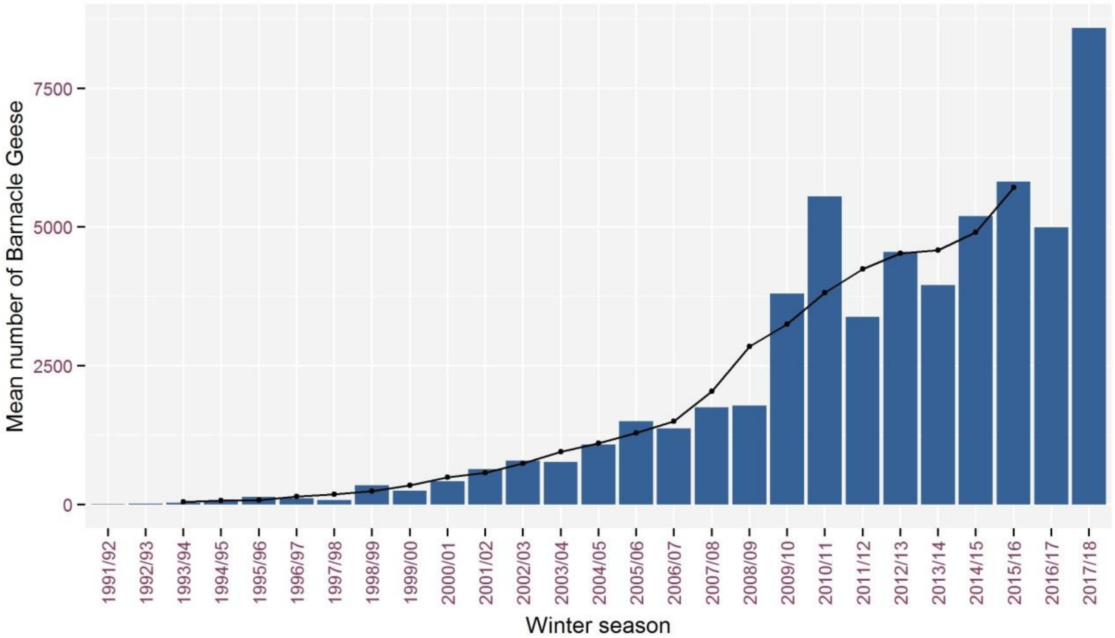


Figure 1: Mean numbers of Barnacle Geese (bars) and five year smoothed average (black line) present during the winter season in Flanders (source: watervogeltellingen INBO).

1.2 POPULATION MODELS AS A TOOL TO EXAMINE MANAGEMENT STRATEGIES

The complex situation of dealing with a Barnacle Goose population of mixed origin in Flanders, combined with anticipated conflicts in management objectives and means, fuel the need for a better informed management. Several modelling techniques have been applied to better inform decision making on goose management. These include species distribution models and spatially explicit resource depletion models which have been used in combination to identify suitable foraging and refuge areas (Baveco et al. 2017). However, for decisions on active population management through fertility reduction or culling, as applied for Canada Goose and other species in Flanders (Reyns et al. 2018), demographic population models are applied more widely to gain insight in population development and the effectiveness of management options (e.g. Schekkerman et al. 2000, Gauthier & Lebreton 2004, Klok et al. 2010, Beston et al. 2016). In stage structured populations, stage specific transition matrices are widely used to model population growth (Caswell 2001). The use of such population models to inform value-based decision making is increasingly becoming a part of species management in general (Madsen et al. 2015, Cohen et al. 2016) and goose management in particular (Menu et al. 2002, Coluccy et al. 2003, Beston et al. 2016, Baveco et al. 2017).

To accurately project population growth, matrix population models consider reproduction and survival at different life stages (De Kroon et al. 1986, Caswell 2001). Therefore, age or stage-specific survival and fecundity parameters are essential to build the transition matrices used in these models (Gauthier & Lebreton 2004). Such knowledge was not available for the Flemish breeding Barnacle Goose population.

This study aims to provide the required information to construct a model which allows examination of the expected development of the Flemish breeding population and the evaluation of different management scenarios. This in order to make a first assessment of the risk of rapid growth of the summering population and the potential for increased management of the species. We report life-history data on breeding success and survival of Barnacle Geese in the study area. This information was derived using detailed monitoring and standard capture-recapture analyses. The results are compared with documented populations elsewhere along the flyway. These data are subsequently used to parameterize stage-based population models that can be used to inform decisions on Barnacle Goose management in the region.

1.2.1 The Greylag Goose model as a template

The model developed in this study is based on the matrix model developed by Klok et al. (2010) for Greylag Goose in the Netherlands. Initially, similar assumptions as for the Greylag model were made (Figure 2). This means a three stage structured, female only, year-based model with post-breeding census was constructed assuming a closed population with no immigration to compensate for mortality (Klok et al. 2010). Klok et al. (2010) argument that food for fledged geese, nesting habitat and habitat for rearing goslings are the only possible limiting factors for geese. These authors also refer to the high amount of land under agricultural practice (60%) to account for sufficient availability of food for fledged geese in the Netherlands (Klok et al. 2010). This is comparable to the situation in Flanders, where 45% of the surface area is under agricultural practice, of which 36% is grassland (Danckaert et al. 2018). The availability of grasslands in the proximity of water, the required habitat for nesting and subsequent rearing of goslings is not expected to be a limiting factor in Flanders since this habitat type has been shown to be widely available for Canada Goose (Van Daele et al. 2012).

Given this, in correspondence to the Greylag Goose model, we also expected little to no density dependent effects and only included a maximum of 2,500 nests as a (high) density dependent threshold.

However, given the observed variability of the different parameters studied in this project, the model template by Klok et al. (2010) was largely abandoned here, with the exception of some basic assumptions. This way, contrary to the model by Klok et al. (2010), we did incorporate demographic stochasticity as Klok et al. (2010) already suggested this would improve the model. However, as mentioned by Caswell (2001) and further documented by Cooch et al. (2003), exploration of stochastic models is indeed recommended but deserves caution with respect to potential covariance among individual matrix elements. To counter this, we further opted to divert from the model by Klok et al. (2010) by altering the census timing in our model to pre-breeding, separating adult survival and reproductive output (Cooch et al. 2003), which leads to the pre-breeding life cycle graph as presented in Figure 3. Finally, we also opted to abandon the female only approach and only used adult sex ratio, combined with the fraction of adults reproducing, in the estimation of the number of nests in a given year.

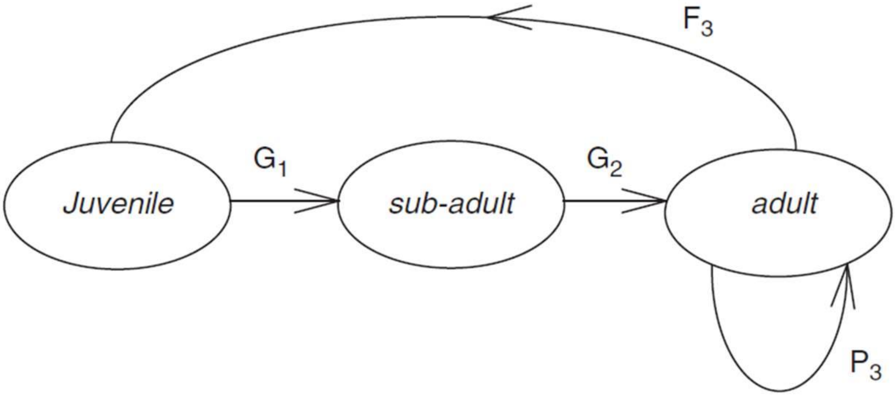


Figure 2: Life cycle graph of the Greylag Goose female population after Klok et al. (2010) (F_3: the number of fledged female young produced per adult female which survived the year, G_1: probability of a fledged young surviving the first year, G_2: the probability that a sub-adult matures to become adult, P_3: the probability that an adult survives the year and remains in the adult stage. F_3 and P_3 are functions of underlying parameters; F_3 is the product of the probability that an adult female survives up to the breeding season, the clutch size times 0.5 (sex ratio), hatching success and chick survival; P_3 depends on the number of years spent in the adult stage and the yearly adult survival probability).

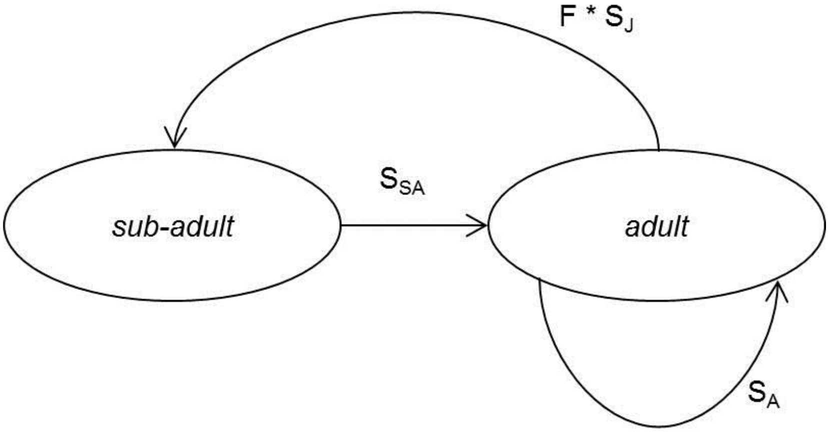


Figure 3: Life cycle graph of the Barnacle Goose model for both sexes used for Flanders (F: the number of fledged young produced per adult female, S_J: probability of a fledged young surviving the first year and reaching the subadult stage at the next census, S_{SA}: the probability that a sub-adult survives to become adult, S_A: the probability that an adult survives the year and remains in the adult stage (modified from Klok et al. 2010)).

2 MATERIAL AND METHODS

2.1 INTRODUCTION

To construct the population model shown in Figure 3, we need to asses four different main parameters: reproduction (*F*), juvenile survival (S_J), subadult survival (S_{SA}) and adult survival (S_A).

In the Greylag Goose life cycle presented by Klok et al. (2010) (Figure 2), reproduction is presented as a single parameter (F_3) and defined as the product of the probability that an adult female survives (δ_a), the clutch size (φ) * the female sex ratio (set at 0.5), hatching success (δ_e) and chick survival (δ_c). The model further defines three survival parameters (G_1; G_2; P_3), where P_3 depends on the number of years spent in the adult stage (d_3) and the yearly adult survival probability (δ_a) (Klok et al. 2010).

As mentioned, for the Barnacle Goose model here (Figure 3), we opted to model both sexes using a pre-breeding census where the number of nests is equal to the number of reproducing adult females, and thus defined F as the product of adult female sex ratio, the fraction reproducing, clutch size, hatching success and chick survival. Under this definition, all chicks, both male and female are modelled, but the number of nests is limited by the number of reproducing female birds. As a result of the pre-breeding modelling, juvenile survival S_J (G_1 in the model by Klok et al. (2010)) is also added to the reproduction step, since all surviving juveniles become subadult birds in the next pre-breeding census.

As a basis for assessing the different model parameters, and for comparison with other Barnacle Goose populations outside Flanders, the initial reproductive parameters were calculated based on the definitions provided by Rockwell et al. (1987):

Total clutch laid (TCL): The total number of eggs laid in a nest by the breeding female.

Nest success (NS): proportion of nests in which a minimum of one egg hatches.

Clutch size at hatching (CSH): The number of eggs laid in a nest in which at least one egg hatches (= *TCL* * *NS*).

Goslings leaving the nest (GLN): The total number of goslings leaving a nest. This is only calculated for nests where CSH > 0. As mentioned above, this number was assessed on site and based on the number of eggs hatched (identified by marked shell fragments), additional non-marked fragments and the number of egg membranes found in the nest. Any chicks found dead in the nest, often the result of late hatching, were treated as unhatched.

Brood size at fledging (BSF): The total number of goslings that survive to just before fledging.

Out of these parameters, the necessary reproductive parameters for the population model were calculated as:

- Clutch size (φ): TCL
- Hatching success (δ_e): GLN/TCL
- Chick survival (δ_c): BSF/GLN

The use of this and similar prior definitions of hatching success as a measure of brood success has been debated by Mayfield (1961) and Johnson (1979), who have proposed alternative methods for determining egg and nest success to account for nests that were not found and for infrequent visits by the researcher. Mayfield (1961), however, has argued that, if all nests were discovered with the first wisp of building material and followed through to termination, analysis would present few difficulties and this discussion would be unnecessary. In the case of the Barnacle Geese studied here, these conditions are clearly met, since all nests were located on small islands, were frequently visited and were all found from the earliest stages of nest building on, so hatching success could be calculated as described by Rockwell et al. (1987) (also see Gosser & Conover 1999). In addition to the parameters described by Rockwell et al. (1987) we also calculated nesting success as the proportion of the breeding couples that hatch at least one egg, following Cooper (1978).

In our approach we also chose to combine nesting success en hatching success as defined by Rockwell et al. (1987) into a single hatching success parameter in which unsuccessful nests were included. Since a low number of unsuccessful nests were found, this not only reduces the number of parameters in F, it also allows to analyse covariate effects in a single model, where this would not be possible for the low number of unsuccessful nests.

Additional survival parameters (S_J; S_{SA}; S_A) were all estimated using observations and recaptures of marked individuals with Capture-Mark-Recapture models (Laake 2013). For S_J and S_{SA} this was limited to known-age individuals. For S_A, δ_a was estimated based on observations and recaptures of marked unknown-age adult individuals. Klok et al. (2010) suggest to further assess S_A, using yearly adult survival (δ_a) and adult stage duration (d_3) with the following equation by Caswell (2001):

$$P_3 = \frac{\left(1 - \delta_a^{d_3 - 1}\right)}{1 - \delta_a^{d_3}} \, . \, \delta_a$$

Which implies that for large values of the stage duration (d_3), P_3 approaches the yearly survival probability (δ_a). Given that geese are long lived, often reaching a very high age, we simplified S_A in our model to:

$$S_A = \delta_a$$

2.2 STUDY AREA

This study was carried out from 2014 until 2018 at three different locations in the province of East Flanders, Belgium: Haasdonk (51°10’22”N, 4°13’44”E), Uitbergen (51°01’01”N, 3°56’43”E) and Lochristi (51°08’33”N, 3°52’51”E) (Table 1). All locations were selected based on the documented presence of nesting Barnacle Geese in previous years. To minimize the influence of predation and estimate maximal reproductive potential, only insular nesting locations totally surrounded by water were included in the study (see Gosser & Conover 1999). In Flanders, Barnacle Geese mainly breed on small islands in lakes and ponds, like in our study setting, and since we monitored a yearly average of 88 nests out of an estimated number of 200 breeding pairs for Flanders, we consider the monitored breeding colonies as a representative sample for of the entire Flemish population. During the first two years of the study, all three locations were visited. In the second year severe nest predation was observed at the Haasdonk location. This location differed from the other two sites in the fact the island still had a bridged connection to the mainland, which, although closed off by a gate, possibly facilitated the entrance of larger predators such as Red Fox (*Vulpes vulpes*) and Stone Marten

(*Martes foina*). In addition, the island was much larger, held an abandoned military fortress and meadows with rabbits and grazing sheep, which also provided ample habitat for smaller nest predator species like Brown Rat (*Rattus norvegicus*) and Carrion Crow (*Corvus corone*). Therefore, this location was dropped from the study from 2016 on. In 2017, data on clutch size, nesting success en hatching success had shown to be fairly constant over time and location. From that year on, only the Lochristi location, which by far held the largest breeding population and where gosling survival could also easily be monitored, was retained for field observations.

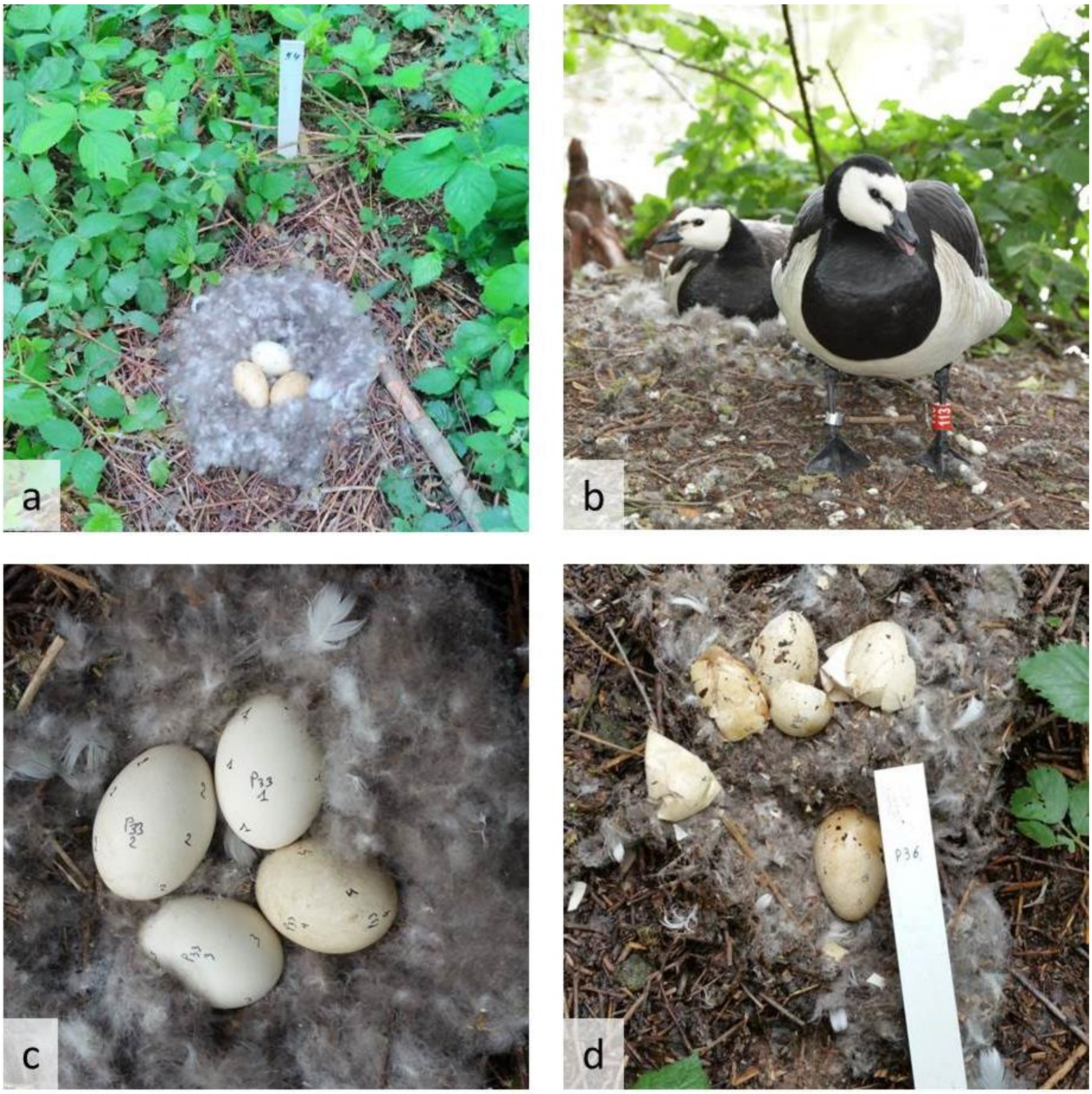
Table 1: Number of Barnacle Goose nests studied per location per year, with reference to the location code and area size.

	Area (ha)	2014	2015	2016	2017	2018	Total
Haasdonk (F)	4,07	15	14				29
Uitbergen (H)	0,16	15	18	14			47
Lochristi (P)	0,15	70	93	66	69	68	366
Total		100	125	80	69	68	442

2.3 MONITORING POPULATION PARAMETERS

All locations were visited from late March on. Once nest building was observed, nest controls were performed weekly, starting from the earliest stages of nest building until the last nest present had fully hatched. All nests were given a unique code (location code + nest number_year, e.g. F01_2014) and all nests were marked by placing white plastic plant markers with the location code and nest number next to the nests (Picture 1a). This allowed us to identify individual nests from a distance using binoculars and provided opportunity to identify any leg banded parents that might be present (Picture 1b).

Each visit, all eggs present in a nest were counted and new eggs were numbered on several sides of the egg shell and marked with the respective location code and nest number (Picture 1c). This was done to check for eggs disappearing during the breeding stage and to maximize the chance that hatching success could be determined correctly. After hatching, all fragments of egg shell were gathered and checked for markings (Picture 1d). Based on the number of eggs hatched (= egg shell fragments with numbers), additional non-marked fragments and the number of egg membranes found in the nest, the total number of eggs hatched was assessed. When the total number of both hatched and unhatched eggs differed from the original nest size or where obvious predation events had occurred, hatching success was labelled as unclear and these results were omitted from any analysis beyond clutch size.



Picture 1: Barnacle Goose nest monitoring: (a) marked nest site, (b) identification of parent birds, (c) marked eggs in a nest, (d) final control of hatching success (this example: eggs #1,2,3,4 clearly hatched, #5 unhatched, nest code P36 is indicated on plastic plant marker and on the top of egg #3, central in the picture).

When possible, parent birds were caught on the nest, leg banded and colour marked with red-white coloured leg rings (Picture 1b, Picture 2a,b,d, Table 2). This allowed identification of the parent birds, both on the nest and during the rearing period from hatching to fledging. During the rearing stage, goslings stay in family groups with the parents, which allowed us to control gosling survival on a weekly basis (Picture 2d). Actual control of gosling survival was only possible for those nests with at least one identifiable (colour marked) parent, where the number of goslings was counted weekly from hatching until early July, just a few weeks prior to fledging. Early July, both goslings and adult birds moult and are flightless. At this moment, captures were performed yearly at the Lochristi site (Picture 2c, Table 2). In 2015, unmarked adult birds were colour marked to broaden the number of possible marked breeding birds in the next year. All goslings were colour marked, which provided a pool of marked known-age individuals, suited for the estimation of first and second year survival. From 2016 on, only goslings were colour marked during the captures. Non-breeding adult birds that were caught during moult trapping sessions in those years were removed from the site and euthanized as part of a broader management approach for the species in Flanders and to reduce goose damages at the request of the site manager.

At the end of the project period, as part of the Flanders management approach, a final moult trapping session was performed at the Lochristi site, at which point all Barnacle Geese caught, both marked and unmarked, were euthanized. This presented us with the opportunity to perform additional biometric analyses of all marked birds in the lab, where animals were sexed to check for errors made while sexing the live animals at the time of banding. All animals collected were aged into age classes, weighed and tarsus, head and culmen length were measured.

Picture 2: Banding and colour marking of Barnacle Geese: (a) net capture on the nest, (b) ringing and colour marking, (c) moult trapping, (d) identification of couples with their offspring using spotting scopes.

Table 2: Number of Barnacle Geese trapped for leg banding per year, age class and trapping type combined with the number of unmarked Barnacle Geese removed from the population at Lochristi. In 2018 all birds that could be trapped were removed (both marked and unmarked) and collected for biometric analysis (2^{nd} CJ = second calendar year, recapture of known age individuals banded as juveniles the year before).

Year	Adult (on nest)	(Sub)adult (moult)	Gosling (moult)	2^{nd} CJ (recap)	(Sub)adult (recap)	Removed (moult)
2015	14	105	72	-	1	184
2016	1	17	33	34	80	184
2017	2	7	52	18	106	72
2018	-	-	-	7	110	310

2.4 ANALYSIS

All analyses were performed using R version 3.5.1 (R Core Team 2019). A detailed description of specific analysis methods is provided at each respective results section further in this report.

3 RESULTS: POPULATION PARAMETERS FOR FLANDERS

3.1 REPRODUCTION AND CHICK SURVIVAL (*F*)

In feral Canada Goose, the potentially high clutch size, hatching success and gosling survival and the low first-year mortality all contribute to a rapid population increase (Allen et al. 1995). This can also be expected for non-migratory Barnacle Geese since the absence or decrease of migratory efforts, and associated high energy costs, can result in higher reproductive output as well as in higher adult, juvenile and offspring survival (Bromley & Jarvis 1993, van der Jeugd et al. 2009). This is due to the fact that Arctic-nesting geese are generally classified as capital breeders, relying on stored energy levels and thus independent of nutritional intake during the pre-laying period (Ryder 1970, MacInnes & Dunn 1988, Bromley & Jarvis 1993, Layton-Matthews et al. 2019). Therefore, reproductive success, reflected in laying date and clutch size, depends mainly on female pre-migration body condition, migratory behaviour and resulting energy reserves on arrival in the breeding area (Ankney & MacInnes 1978, Cooper 1978, Prop et al. 1984, Lessels 1986, Alisauskas & Ankney 1990, Kostin & Mooij 1995, Prop & Black 1998, Bêty et al. 2003, Prop et al. 2003, Hübner 2006). Given the absence of migration in the Flemish breeding population, we expected higher reproductive outputs in comparison to migratory conspecifics.

First, we expected a higher nesting success (proportion of the breeding couples that hatch at least one egg) than in migrating populations. Moreover, Barnacle Geese exhibit an evolutionary trade-off between a late start (favourable foraging conditions resulting in improved maternal body condition with enhanced probability of successful hatching) and an early start (resulting in a high survival rate of the offspring) of incubation in their breeding grounds (Finney & Cooke 1978, Prop & de Vries 1993). This trade-off occurs in response to unfavourable feeding conditions due to snow cover during the early stage and the need for sufficient growth time for the offspring prior to fall migration. This leads to geese adjusting lay date and clutch size according to body condition (condition dependent optimization model) (Bêty et al. 2003, 2004). The penalty of laying late is particularly severe in Arctic-nesting geese and Barnacle Geese experience a strong selection for early laying (Prop et al. 1984).

Since both constraints on snow cover and fall migration timing acting on Arctic-nesting geese are lacking in the Flanders' breeding population, we expected a significant earlier onset of egg-laying and incubation. Given that improved body condition of the female results in larger clutch sizes (Ryder 1970, Lessels 1986, Bêty et al. 2003), we also expected that non-migratory geese, breeding in good feeding habitats, would have larger clutch sizes than their migratory conspecifics. Although there is little evidence that suggests a direct impact of clutch size on hatch success, some authors mention possible indirect effects (Prop et al. 1984, Rockwell et al. 1987). Nest attentiveness has been shown to positively impact hatching success and is, like clutch size, also related to female body condition (Aldrich & Raveling 1983). Therefore, we also expected differences in hatching success between migratory populations and the non-migratory Barnacle Geese examined in our study. Finally, as reproductive output is often evaluated at the time of fledging and reported as the number of fledglings per successful pair (Larsson & Forslund 1994), we also calculated fledging success. We did not expect differences in fledging success between migratory and non-migratory populations since this is irrespective of maternal condition and thus of migratory efforts.

3.1.1 Sex ratio

We performed a post mortem section on 325 Barnacle Geese caught during moult trapping session. Given the difficulties with identifying second calendar year birds late in the season during moulting, birds were either categorized as *adult* or *pullus*. The female/male ratio in pulli (0.59) was slightly higher than in adult birds (0.53) but this difference was not significant (χ^2=0.78, *df*=1, *P*=0.38). Sex ratio over all age categories was 0.55.

3.1.2 Fraction reproducing

We had few data to assess the fraction of female adult birds that participate in reproduction. In 2015 we caught 303 non-juvenile birds at the Lochristi site and recorded 93 nests in that year. We estimated about 20% subadult birds and given the sex ratio of 0.55 this would make 133 female adult birds present, of which 93 entered reproduction, resulting in a fraction reproducing of 0.70. We are, however, very aware of the roughness of this estimate, which calls for extra attention in the further evaluation of the elasticity of this parameter in the model.

3.1.3 Clutch size

During the five year period, clutch size could be determined in 394 Barnacle Goose nests on 3 different locations. In all years and locations examined, egg laying started around week 16 and in 88% (*n* = 350) the first egg was laid between week 16 and 19. On some occasions, eggs were added to a clutch late in the breeding process, leaving insufficient time for hatching. Incubation time has been shown to be dependent of timing in the breeding season but has a minimum of 22 days (Dalhaug et al. 1996). In 14 different nests, a total of 20 eggs were known to be laid in the last two weeks before hatching and these eggs were omitted from the analysis. The nests themselves, with their adjusted nest sizes, were retained for further analysis.

Overall average clutch size was 4.06 ± 1.28 (SD). Generalized linear model analysis *(family = poisson (link = log))* of clutch size with backward model selection was performed to test the importance of hatching year, hatching week and location (McCullagh & Nelder 1989). The null model outcompeted all other models indicating no proof for variation in clutch size between years, weeks or locations. Mean modelled clutch size was 4.11 ± 0.10 (SE).

In the population model, since no significant variables were identified, clutch size was added as a discretized sample from a normal distribution with a mean of 4.06. SD was lowered to 1 to narrow the distribution around a nest size of 4. A comparison of the distribution of 10,000 clutch size samples generated this way to the observed distribution in our data is presented in Figure 4.

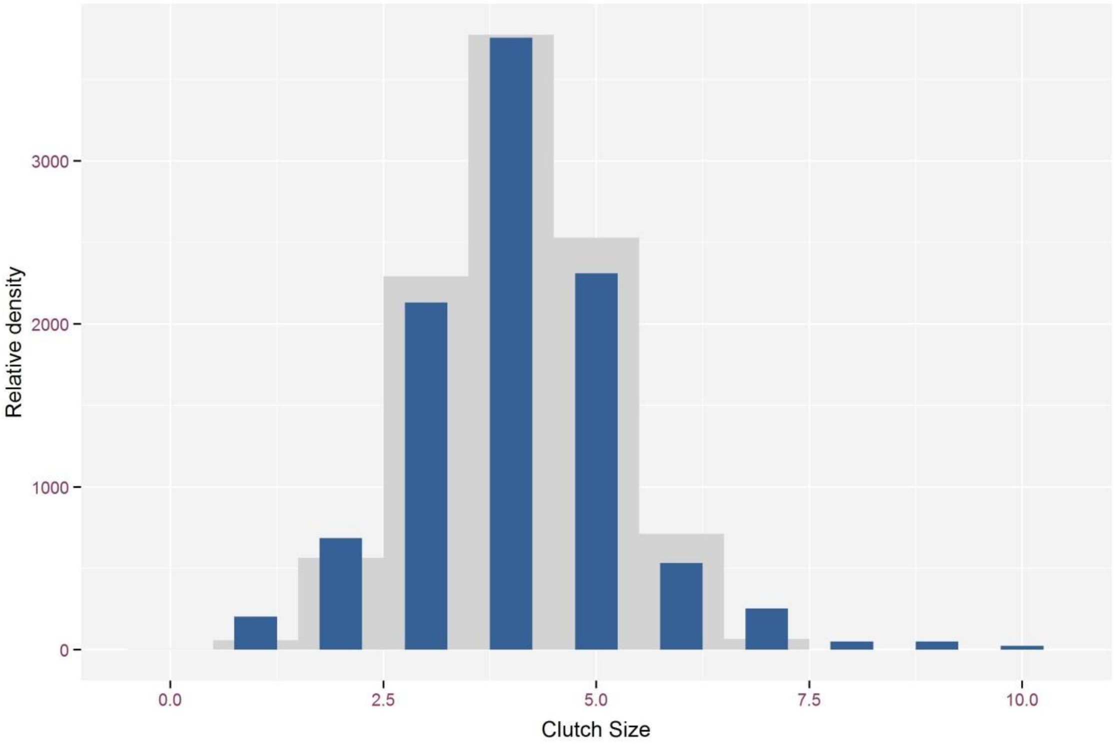


Figure 4: Comparison of the distribution of the observed (blue) to 10,000 modelled (grey) clutch size samples under the model assumptions described in the text.

3.1.4 Nest success, clutch size at hatching and hatching success

As mentioned under 2.1, results from the Haasdonk location were omitted for analysis beyond clutch size since severe nest predation was observed at that site in 2015. This left a total of 371 nests suitable for the analysis of hatching success. In 24 of these nests, not a single egg had hatched, resulting in an overall nest success of 93%. The number of hatched eggs, or goslings leaving the nest (GLN) was estimated on site and hatching success was calculated as GLN/TCL (see 2.1). For comparison with other studies, clutch size at hatching (CSH) was also calculated under the definition provided in 2.1.

We modelled nest success as a binomial response using generalized linear modelling to test for the effect of clutch size and year. The model with both variables *(Nest Success ~ Clutch Size + Year, family = binomial (link = logit))* was selected as the best model. The inclusion of Year in this model was almost entirely attributed to the fact that in 2018, nest success was 100%. Since, for all other years, the effect of clutch size was much higher than the year effect we opted to simplify the population model and only include the effect of clutch size on nest success (see 3.1.5).

In the 347 successful nests, mean clutch size at hatching was 4.08 ± 1.27. On average hatching success in these nests was 0.85 ± 0.21. A binomial GAM-analysis with stepwise backward model selection was performed to test the effects of week, year, location and clutch size on hatching success in these successful nests. In 110 nests, the nesting couple was known and added as a random effect in the model since breeding behaviour could very well be couple specific (in the remaining nests unique couple identifiers were added). The use of a GAM enabled us to allow non-linear effects of week and clutch size, since in both cases we expected an optimum value. The analysis showed that the model with year outcompeted other models *(cbind(Hatched,Unhatched) ~ Year, family = binomial (link = logit))*. This showed that average hatching success ranged between 0.82 and 0.89 in the different years studied with an overall average of 0.84 ± 0.21 (Figure 5).

Figure 5: Observed distribution (grey) and modelled average (yellow) of hatching success per year under the model assumptions described in the text.

3.1.5 Goslings leaving the nest

Overall, we observed an average number of 3.21 ± 1.50 successfully hatched eggs per nest (=goslings leaving the nest). For comparison we combined the modelled distribution of clutch sizes (see 3.1.2) with a modelled hatch success. For this, nest success was first modelled in relation to clutch size as modelled under 3.1.4. This step was afterwards combined with a normally distributed hatching success of 0.84 ± 0.21. The distribution of the number of goslings leaving the nest under these assumptions (*n* = 10,000) was compared to the relative observed distribution of GLN in Figure 6. This comparison shows good overlap between both distributions with an average GLN of 3.17 ± 1.48, very similar to the observed average of 3.21 ± 1.50 (χ^2=12.49, *df*=7, *P*=0.13).

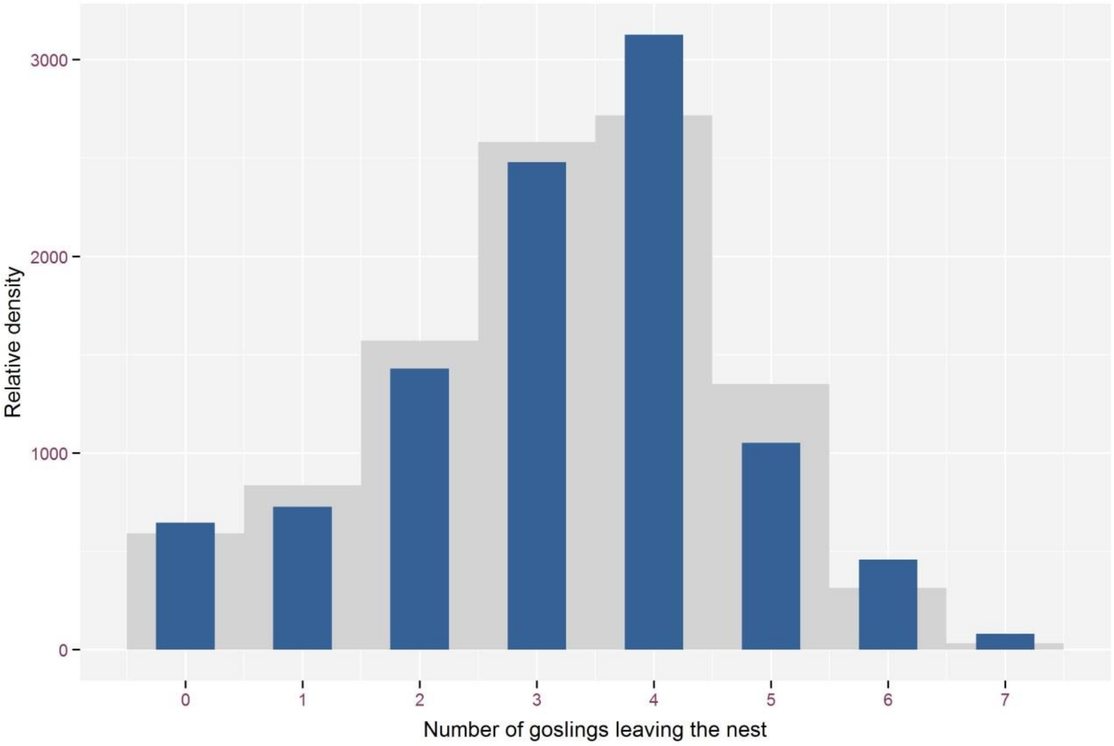


Figure 6: : Comparison of the distribution of the observed (blue) to 10,000 modelled (grey) goslings leaving the nests under the model assumptions described in the text.

3.1.6 Chick survival

In 110 of 371 nests that were used to determine hatching success, at least one of the parent birds was leg banded and could be individually identified. In 107 of these nests, young fletched and were observed with their parents after hatching. In two of these nests, adoption of chicks from another nest was observed. From the remaining nests, six nests were only observed in the first three weeks after hatching, which we considered as an insufficient period to determine chick survival. However, in five out of these six nests, all chicks had already died, which makes these results valid for chick survival analysis. This left 104 successfully hatched nests with identifiable parents in which chick survival could be determined up to at least the third week after hatching. 90 of these nests (86%) even provided data up to the sixth week after hatching.

These 104 nests had an average GLN of 3.58 ± 1.21 of which on average 1.47 ± 1.45 chicks survived. This results in an overall chick survival until fledging (fledging success) of 0.37 ± 0.35 and number of fledglings/successful couple (at least one chick survived until fledging) of 2.35 ± 1.12. We used a zero-inflated Poisson model for analysing the possible effect of year, hatching week and GLN on the number of chicks surviving in a nest. Models which included couple as a random variable in the model failed to converge so this variable was omitted from further analyses. To account for exposure we used GLN as an offset in the Poisson part of our model. The optimal model incorporated an effect of GLN on the zero-inflated part of the model and an effect of year in the Poisson part *(Chick Survival ~ Year + offset(log(GLN)), ziformula = ~ GLN, family = poisson (link = log))*. This implies that smaller nests with a GLN of one or two have a high probability that not a single chick will survive up to fledging, with low variation between years. Additionally, the year effect implies that overall success varied significantly in nests with a GLN of three or more between the different years studied (Figure 7).

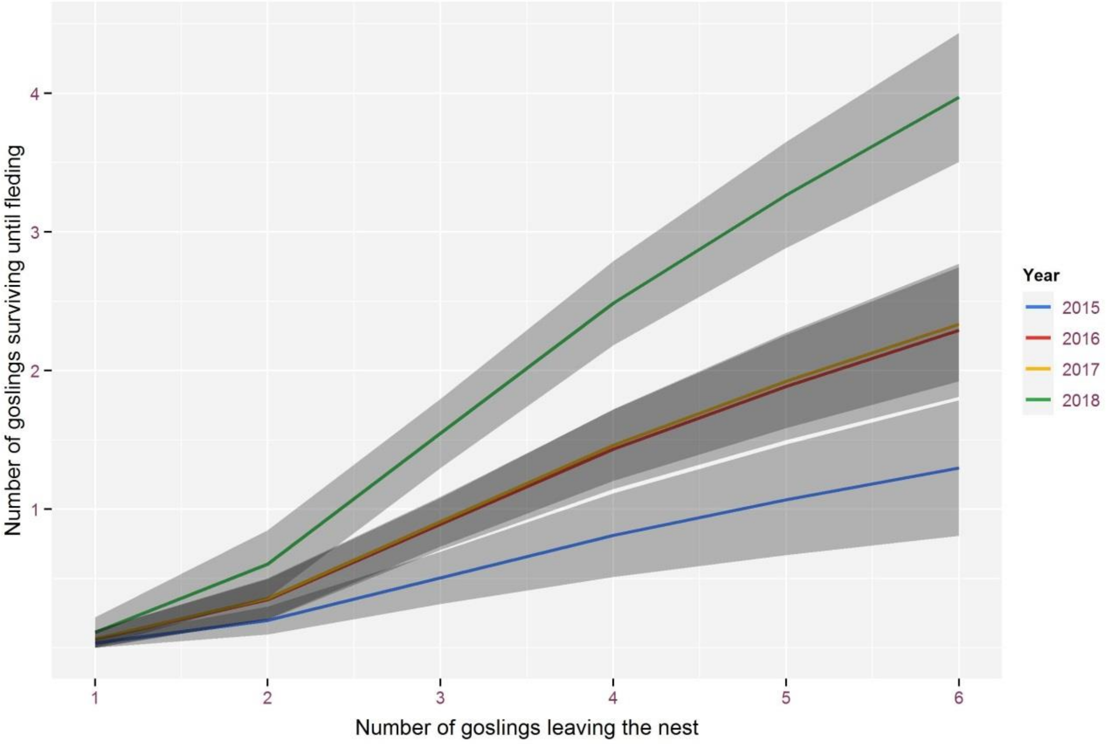


Figure 7: Modelled number of goslings surviving in relation to GLN and year under the model assumptions described in the text.

For the population model, we modelled the chance of no chicks surviving as a separate binomial response to GLN and further modelled survival in remaining nest as a binomial response to year. This showed that the effect in 2018 was much larger than in other years with an overall survival probability in successful nests of 0.70. In contrast to the Poisson part in the zero-inflation model, the difference in survival probability between 2015 (0.50) and 2016 (0.54) or 2017 (0.54) was far less pronounced. For model parsimony, we opted to assign a chick survival probability of 0.50 ± 0.10 in nests in which at least one chick survives for all years, but incorporated a 25% chance of a very successful year with a survival probability of 0.70 ± 0.10 (Figure 8). This approach yielded a slight underestimation of the total number of chicks surviving, but still gave an overall sufficient fit to the observed data (Figure 8).

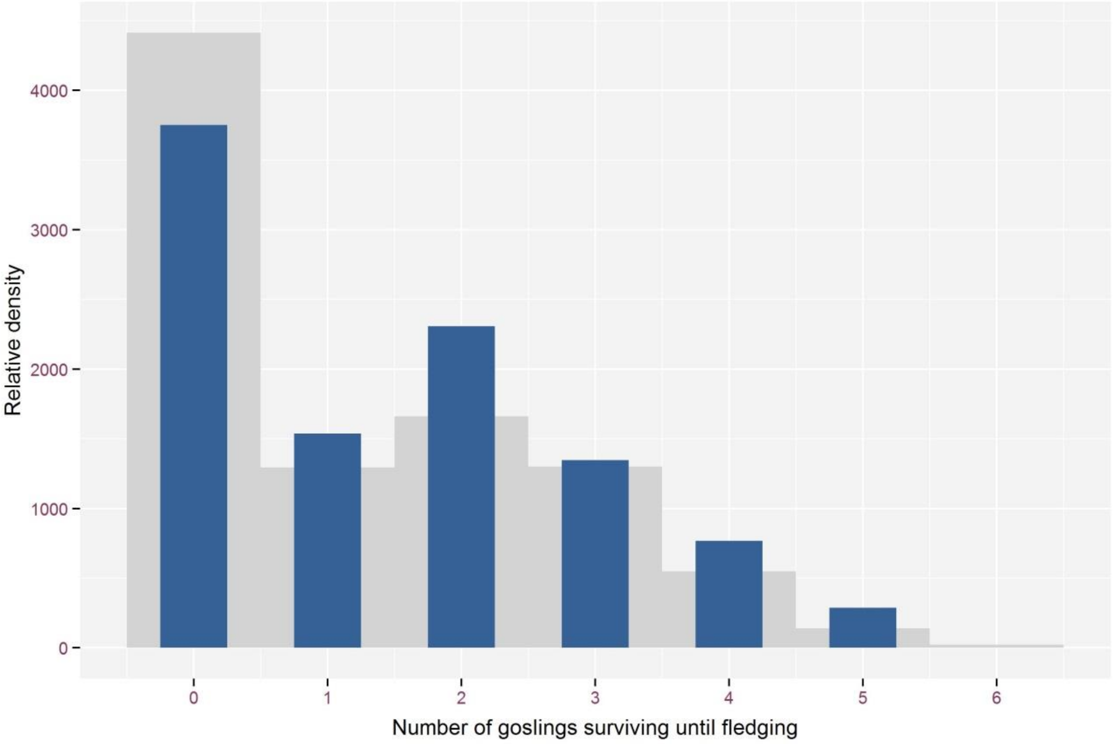


Figure 8: Comparison of the distribution of the observed (blue) to 10,000 modelled (grey) goslings surviving until fledging, in nests in which at least one chick had hatched, under the model assumptions described in the text.

3.2 SURVIVAL (S_{SA}, S_A)

As for reproduction, we expected survival of the Flanders' resident breeding population to be higher than for their migratory conspecifics, given the absence of energy investment in migration (van der Jeugd et al. 2009). In addition, resident breeding Barnacle Geese in Flanders are not hunted which would again increase the expected annual survival probability.

During the study, a total of 308 Barnacle Geese were captured for leg banding and/or colour marking from 2015 on. 303 of these were captured at the Lochristi site and 5 at the Uitbergen site. The latter were all adult birds captured on the nest. 157 of the geese captured at the Lochristi site were juveniles, 146 were older. In 2015, all geese caught, with the exception of one adult bird, were colour marked. From 2016 on, only a few additional adult birds were colour marked, while all juvenile birds were colour marked (for an overview of trapping and banding results at the Lochristi site see Table 3 and Table 2).

Table 3: Number of Barnacle Geese leg banded at the Lochristi site by age class, trapping type and year of capture with indication of the number of geese that were also colour marked (LB = leg banded, CM = colour marked).

Year	(Sub)adult				Juvenile	
	Nest trapped		Moult trapped		Moult trapped	
	LB	CM	LB	CM	LB	CM
2015	14	14	105	104	72	72
2016	1	1	17	2	33	33
2017	2	2	7	0	52	11

3.2.1 Juvenile and subadult survival

Juvenile and subadult survival were estimated using recapture and resighting data from known age individuals in our study population. To harmonize the detection probability, the analysis was further limited to data from the colour marked individuals. This resulted in a dataset with mark-recapture data from 116 Barnacle Geese that were colour marked as juveniles between 2015 and 2017 at the Lochristi site (see also Table 3). 80 of these Barnacle Geese were sexed on site, but, since sexing of juvenile waterfowl is difficult and can result in misidentification (Brown & Brown 2002, Van Dijk 2016) only the animals that were later captured and euthanized could be assessed with certainty. This was the case for 40 of 116 animals, which did not allow to test sex-related survival. The sex determined at trapping for 80 geese was, as suggested earlier, also not useful, since 27 of these geese allowed comparison to lab results, showing a sex misidentification rate of 41% (11 geese).

To account for potential errors in visual colour marking identifications, all records of a single resighting of an individual throughout the study period were removed from the dataset. This way, 14 resightings of known age individuals were removed.

Both juvenile and subadult survival were analysed with a Cormack-Jolly-Seber model (CJS) using the RMark software package (Laake 2013) in which both φ (apparent survival) and p (detection probability) are estimated.

Since the main interest of this analysis was to distinguish differences in survival between juvenile and subadult birds, age was added as a variable to the survival estimate. As sampling was not similar in all three years of colour marking juvenile birds, we did not test for time or cohort effects in survival.

Analysis showed an overall high detection probability of 0.90 ± 0.03 (SE). First year (juvenile) survival was estimated at 0.78 ± 0.04 and second year (subadult) survival at 0.88 ± 0.05. For animals captured in 2015, resighting data from 2018 also provided a third year survival estimate, which was 0.88 ± 0.07. This value is highly comparable to the subadult survival estimate and should correspond to the adult survival estimate calculated for non-known-age individuals (see 3.2.2). This would then indicate that survival is overall very high and reaches a constant level from the second year on.

3.2.2 Adult survival

This analysis was performed on 123 non-juvenile Barnacle geese colour marked at the Lochristi site during the study period (Table 3). Since it is very difficult to discern subadult from adult birds at that stage, some of these birds caught in 2015 were potentially subadult birds. This number was probably limited and since these birds were also followed for several years and we did not expect large differences between subadult and adult survival, we assumed this effect to be minimal. Similar to the juvenile and subadult survival analysis, all records of a single resighting were removed from the dataset. This way, 6 resightings of adult individuals were removed. For all other individuals combined, a total of 1,858 resightings were available, of which 1,655 were done in years following colour marking, allowing, combined with recapture data, assessment of yearly survival. Adult survival was also analysed with a CJS-model using the RMark software package (Laake 2013). For 120 of the individuals analysed, sex was known, which allowed us to analyse differences in survival between sexes. No effect of sex nor year on adult survival was found.

As in juvenile and adult birds, analysis showed an overall high detection probability of 0.93 ± 0.02 (SE). Adult survival was estimated at 0.93 ± 0.02. This value is slightly higher than the adult survival estimate for known age birds (see 3.2.1), but still well within the error margins of this latter value. Since the difference with estimated survival in known age subadult birds was more explicit, juvenile (0.78 ± 0.04), subadult (0.88 ± 0.05) and adult survival (0.93 ± 0.02) were included as separate survival parameters in the population model.

4 THE POPULATION MODEL

4.1 THE BASELINE MODEL

Based on the obtained parameter estimation obtained (see 3 and 3.2) we developed a population model to estimate population size of Barnacle Geese in both pre-breeding age classes over the period 2021-2050 (Figure 9). For this model, we assumed a 2020 starting population of 800 Barnacle Geese for Flanders, consisting of 600 adult birds and 200 subadult birds. Given a female sex ratio of 0.55 and a fraction of 0.70 reproducing, 600 adults roughly correspond to the current breeding population estimate of 200 breeding pairs (see 1.1). Given an overall reproductive output of 1.47 per breeding pair (see 3.1.5) and a juvenile survival of 0.78 (see 3.2.1), 200 breeding pairs would generate approximately 200 subadults in the following year. As mentioned, the model assumes a closed population with a density dependence threshold for the number of nests in Flanders. We assumed that a maximum of around 25 nesting sites with on average 100 nests would be a reasonable maximal nest occupancy for Barnacle Goose in Flanders, so we included a maximum of 2,500 nests in the model. The model further does not account for senescence effects and assumes a constant adult survival rate of 0.93. This value was calculated based on a population which probably holds few very old birds so that adult survival in het higher ages classes is probably lower. Also, reproductive output in these age classes could be lower. However, the model presented was constructed to test management approaches on a shorter term than 20-30 years so that these effects would have little impact on management scenarios. In Figure 9, it is clear that the threshold level for the number of nests in Flanders is reached much quicker, around the year 2035. Since this does have implications on management scenarios in which reproductive output is targeted, this density dependent effect was kept in the model for further analysis. Overall, the baseline model presented here shows a population with an estimated average lambda over 30 years (= finite rate of increase per time step, i.c. year) of 1.12 ± 0.07 SE. This implies that the population would double every 6.12 years.

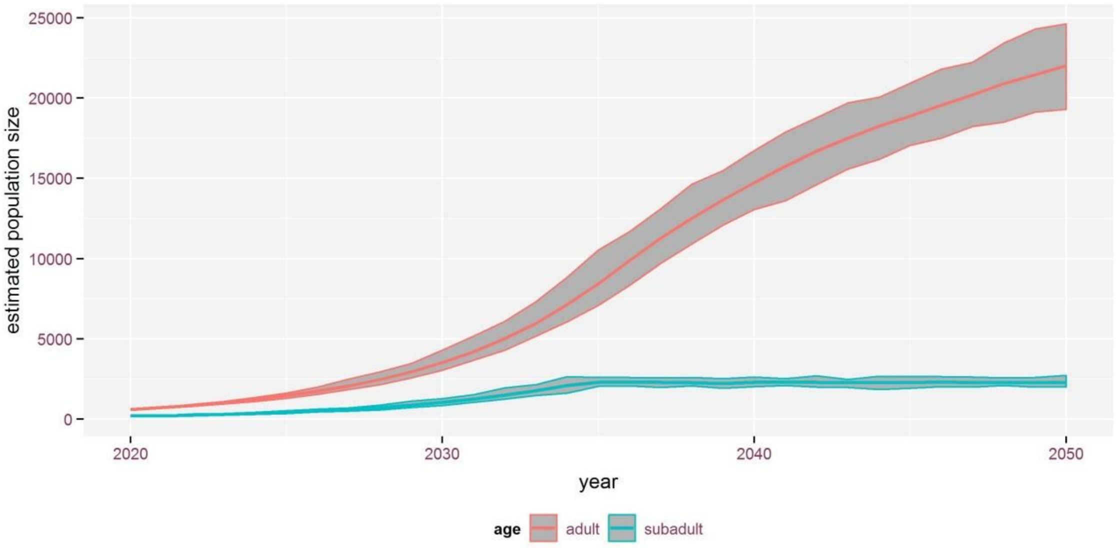


Figure 9: Evolution of estimated population size based on the population model output for the period 2020-2050.

4.2 ELASTICITY OF MODEL PARAMETERS

An analysis of the elasticity (= proportional change in lambda for a proportional change in a single parameter) of the different parameters in the model shows that changes in adult survival have the highest impact on lambda values (Figure 10). From a 20% decrease in the original estimated survival, lambda values start to differ significantly from the average lambda under the baseline model. With a 30% decrease of adult survival, lambda drops below 1, implying population decline. Clutch size and hatching success also show a relevant impact, which is quite similar for both of these parameters. Diminishing one of both values with 30% usually results in a lower lambda than the baseline model and under a 40-50% decrease, lambda values approach 1. In all these cases, the impact of clutch size variation is higher than the impact of variation in hatching success. To test for the impact of a potential increase in nest size (e.g. as a density dependent effect) we also increased nest size up to 30% with 10% intervals (not shown in Figure 10). These increases showed no differences in lambda values in comparison to the baseline model. For chick, juvenile or subadult survival, even a reduction of 50% did not result in lambda values approaching 1. Finally, the fraction of birds reproducing, which had high uncertainty (see 3.1.2), also shows little impact, with only a value diminished with 30-40% resulting in a lower than baseline lambda value and not approaching 1 under a 50% reduction.

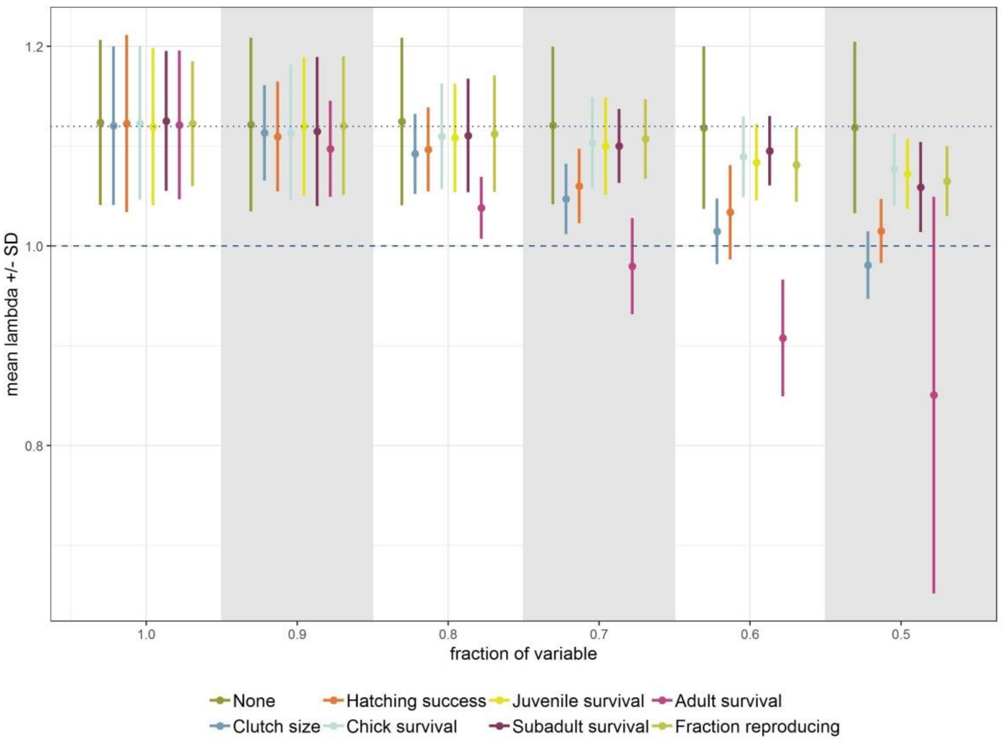


Figure 10: Elasticity analysis of seven model parameters. Original parameter values were diminished to 50% by 10% intervals. Each value represents the average lambda of 10 model runs in which a single parameter was altered while all other parameters were kept at the original levels (dotted line represents the estimated baseline lambda of 1.12).

5 MANAGEMENT SCENARIOS

The findings under 4.2 imply that any management scheme should target adult survival first and nest size and hatching success second. Any management plan that would only target chick, juvenile or subadult survival would be very inefficient at reducing population size. To assess the combined effect of realistic management impact on these variables combined, we first discuss each management separately and then discuss a combined approach.

Although in 4.2, elasticity values for each of the parameters are described already, it is recommended that, using stochastic models as this one, the effect of changes on lambda is evaluated through simulations (Caswell 2001, Gauthier & Lebreton 2004). Each of the scenarios were modelled under the same assumptions and starting population as the baseline model (see 4.1). All scenario analyses presented below are the result of 50 simulations of population model projections 2021-2050.

5.1 NEST DESTRUCTION

In the first management scenario we modelled the removal of nests, which impacts both nest size and hatching success. This management approach can be done by oiling, shaking or pricking of eggs or by total destruction of the nests (Klok et al. 2010, Van Daele et al. 2012). The effectiveness of these different options can differ, but for this scenario analysis we assumed that all nests that received treatment could be fully destroyed, would not be replaced and would thus not yield any chicks. This simulates a maximal impact of this management scenario on reproductive performance. To model this management approach, the number of nests produced was lowered by 10% intervals starting from 0 to 90%. Nests success and hatching success were not altered for the unmanaged nests.

Figure 11 shows that average lambda only starts to decrease drastically from a reduction of 50% of the total number of nests onward. To establish a mean lambda under 1, resulting in a population reduction, more than 75% of all nests would need to be destroyed.

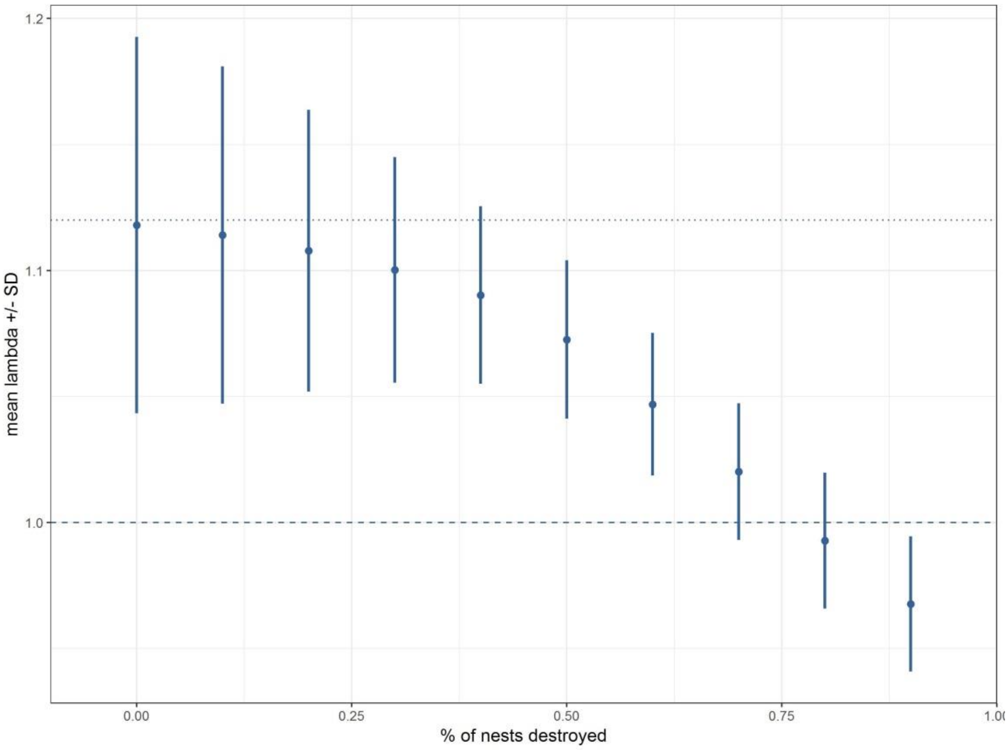


Figure 11: Mean lambda for 50 simulations of a 2020-2050 population projection for Barnacle Goose in Flanders under a nest destruction management ranging from 0 to 90% reduction of the number of nest by 10% intervals (dotted line = baseline model lambda of 1.12).

5.2 REMOVAL OF FLEDGED GEESE

A second scenario involves the removal of birds from the population from around the moment of fledging up to the census point just prior to the breeding season. This can be achieved through either trapping of moulting flocks or through shooting.

Since the Barnacle Goose is protected by the European Union Birds Directive and the Flemish species protection legislation, which does not allow derogation shooting, no shooting of Barnacle Geese is currently done in Flanders. However, from a modelling perspective, the capture of moulting flocks is probably highly comparable to the removal of birds by shooting them after the moulting season. This would, however, given that the Flemish breeding population is enriched with wintering birds from October to May, only be the case for a hunting season outside of this period and under the assumption they both evenly target survival in all age classes. Given the timing of spring migration of wintering birds (February – March) and the onset of the breeding season in the resident population (April – May), shooting of Barnacle Goose in Flanders would thus have to be limited to the period July – September to target the local population of fledged geese. Since no hunting of this species is currently performed in Flanders and no monthly bag numbers are available for other huntable goose species in Flanders (Canada Goose and Greylag Goose), it is very difficult to assess the numbers that could be targeted through shooting during this period. Therefore, we opted to only model the removal of fledged birds through the scenario of moult trapping.

Capturing moulting flocks is a management technique which has been applied for the management of greater Canada Goose in addition to hunting in Flanders since 2010 (Van Daele et al. 2012, Reyns et al. 2018). This technique has, since 2015, also been applied on an *ad hoc* basis to lower the population of resident Barnacle Geese. Part of the moult capture efforts in the 2015-2017 period have gone into the capture/recapture part of this study, but also helped to show the potential capacity of this management method for Flanders. Table 2 shows that at a single moulting site in Flanders, captures of over 400 birds per year are possible. In Flanders, including the removal of birds used in this study in 2018, an average of 467 Barnacle Geese were removed from the population yearly using this method. To evaluate this scenario, we modelled the effect of capturing a fixed percentage of the total moulting population on a yearly basis ranging from 0 to 90% by 10% intervals. Since birds of different age classes group together in moulting flocks, capture probability in the model was evenly distributed over all age classes. The results presented in Figure 12 show that from a moult capture regime of 20% on, average lambda drops below 1 and the population decreases. Furthermore, lambda drops drastically from yearly reduction of 50% on, resulting in average lambdas of below 0.5, although margins of error also increase from this point on. Given the stochasticity we included in this yearly percentage and that this value is relative to the total population size, lambda stabilizes at very low values from an 80% yearly capture effort on.

To counter this, we also modelled the effect of a fixed number of Barnacle Geese captured yearly over a shorter time frame (2021-2030) (Figure 13). This approach shows a more rapid decrease in lambda, reaching values of below 0.5 from a fixed capture effort of 200 Barnacle Geese per year. Lambda further stabilizes at very low levels from an average capture level of 400 geese on, comparable to the yearly average number of Barnacle Goose currently trapped in Flanders.

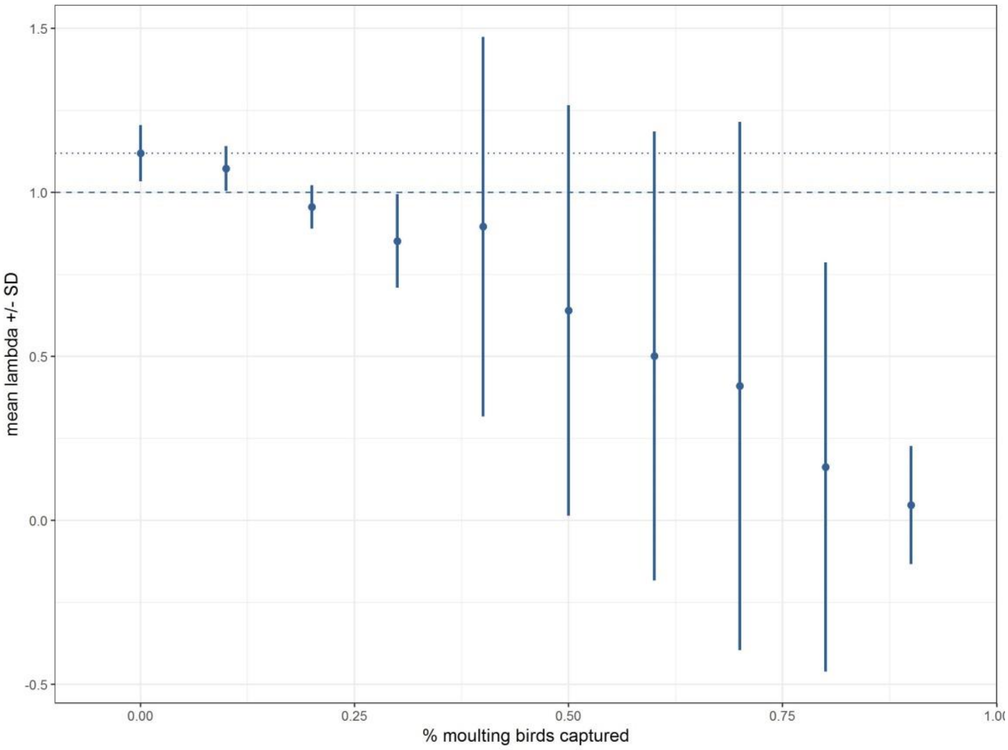


Figure 12: Mean lambda for 50 simulations of a 2020-2050 population projection for Barnacle Goose in Flanders under a moult capture regime ranging from 0 to 90% reduction of the moulting population by 10% intervals (dotted line = baseline model lambda of 1.12).

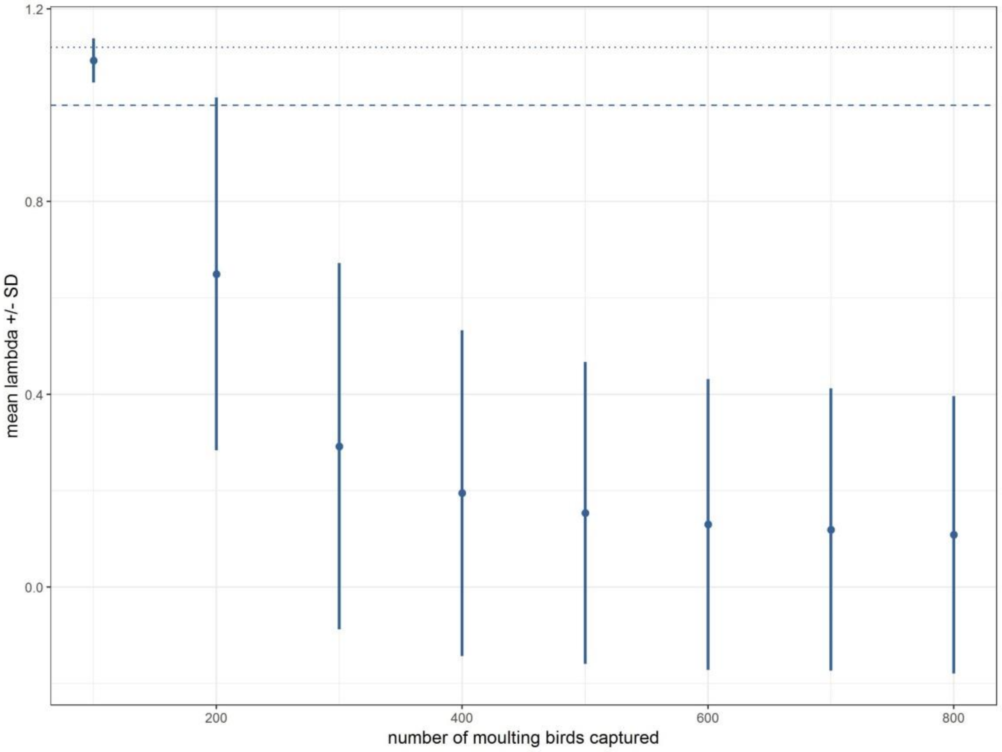


Figure 13: Mean lambda for 50 simulations of a 2020-2030 population projection for Barnacle Goose in Flanders under a yearly moult capture regime ranging from 100 to 800 Barnacle Geese by intervals of 100 (dotted line = baseline model lambda of 1.12).

5.3 COMBINED

Finally, we modelled a combined effect of moult trapping and nest destruction. Therefore, we modelled the fixed effect of destroying between 0 and 400 nests yearly over the period 2021-2030 under a yearly moult capturing scenario of 0, 100, 200 and 300 birds (Figure 14). We included the zero capture option to demonstrate the effect of a yearly fixed number of nests destroyed, which was not yet visualized under 0. This combined approach shows that any nest destruction regime has limited impact on altering lambda under any of the different moult trapping regimes. Even under the zero trapping scenario, 200 nests need to be destroyed yearly, which, given the starting values used, is a very high percentage of the nests present (see also Figure 11). From a yearly capture of 200 birds on, half of what has been caught in recent years, the population drastically decreases, regardless of nest destruction regimes.

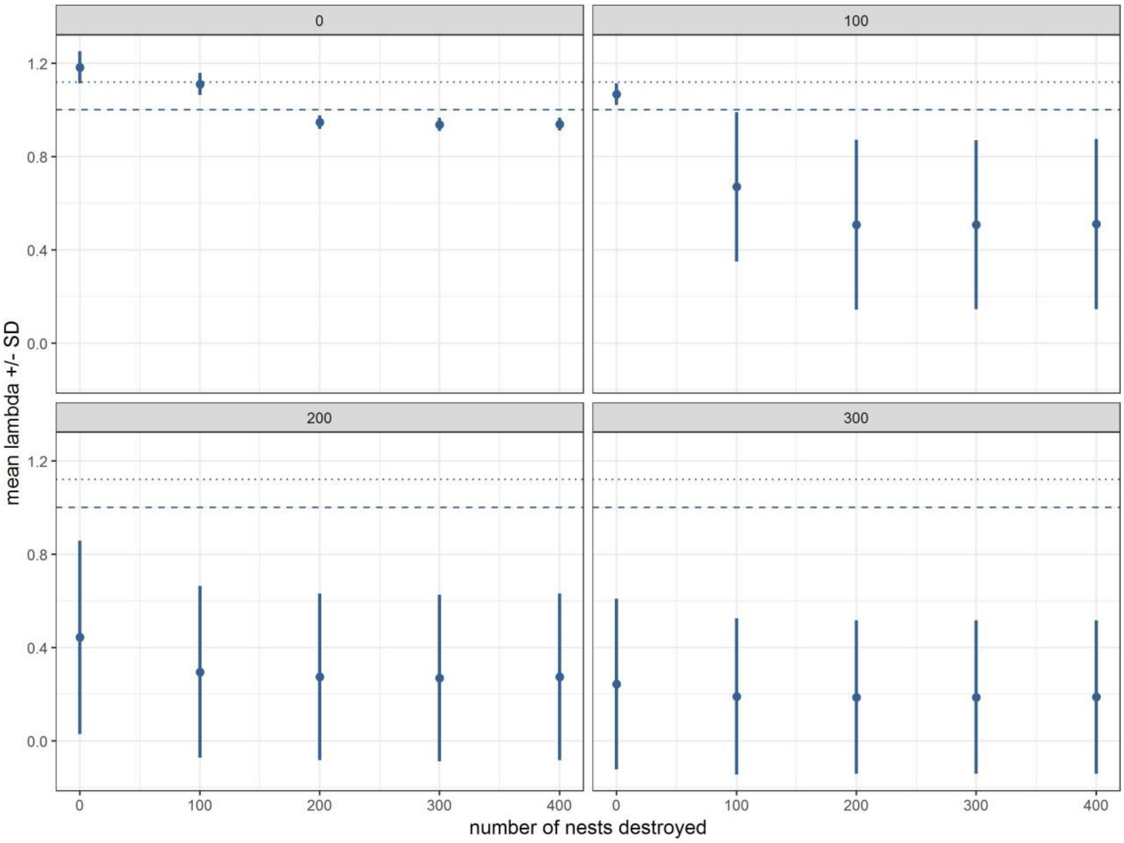


Figure 14: Mean lambda for 50 simulations of a 2020-2030 population projection for Barnacle Goose in Flanders under a yearly moult capture regime ranging from 0 to 300 Barnacle Geese (facet) by intervals of 100 combined with a nest destruction regime ranging from 0 to 400 nests (x-axis) (dotted line = baseline model lambda of 1.12).

6 SUMMARY AND DISCUSSION

6.1 POPULATION PARAMETERS

Egg laying in Flanders started on average in week 17 (end of April), and incubation at around week 19 (second week of May). This is at the same time Barnacle Geese start nesting in the Netherlands and the Baltic area, but much sooner than in the Arctic breeding populations, where breeding only starts at the end of May or early June (Prop et al. 1984, Larsson & Forslund 1994, Ganter et al. 1999, Owen & Black 1999, Van Der Jeugd et al. 2006, 2009, Hahn et al. 2011, Layton-Matthews et al. 2019).

The average clutch size of 4.06 observed in Flanders was lower than the yearly averages reported in several studies from Gotland (4.4-4.8) and from the Netherlands (5.2-4.6) (Larsson et al. 1988, Forslund & Larsson 1991, Larsson & Forslund 1994, Pouw et al. 2005, van der Jeugd et al. 2006,). In comparison to the Arctic breeding colonies, the overall mean total clutch laid in Flanders was higher than found in several studies on Svalbard (3.20, 3.56, 3.77) (Dalhaug et al. 1996, Loonen et al. 1997, Hübner et al. 2002), Greenland (3.74) (Ogilvie et al. 1999) and in the Russian Kolokolkova bay (2.77,) (van der Jeugd et al. 2003). In other studies, similar values to the value reported here were found at Svalbard (4.0-4.5) (Owen & Norderhaug 1977, Hahn et al. 2011) and values of 3.8-4.7 were found in Russian breeding colonies (Filchagov & Leonovich 1992, Gurtovaya 1997 in Van De Jeugd et al. 2003, Ganter et al. 1999). All these values, with the exception of 2.77 at Kolokolkova bay show a rather rigid clutch size of around 4 eggs. This seems to indicate that the effect of high energy reserves at the onset of breeding and earlier laying date due to the absence of migratory efforts is not reflected in clutch size. Although geese indeed rely on pre-laying stored energy levels for reproduction, which could benefit from the absence of migration, it has also been shown that female Barnacle Geese can vary the use of these resources for egg laying, potentially allowing for adaptive adjustments to local environmental conditions just before breeding (Choinière & Gauthier 1995, Hahn et al. 2011). It thus seems likely that Barnacle Geese allocate energy towards a more optimal clutch size of around 4 eggs, rather than investing in brood enlargement.

Out of the 371 nests studied here, 24 nests did not hatch a single egg, which results in an overall nesting success of 93%, higher than the 71% observed in Gotland (Larsson & Forslund 1994). At Kolokolkova Bay, a weighted mean of 70% was reported, but with high variation among colonies due to very high local levels of nest predation and levels going up to 88% nest success in colonies with the lowest levels of predation (van der Jeugd et al. 2003). Similarly, in various studies on different colonies breeding on Svalbard, very high, breeding site specific, variation in breeding success is reported, with values ranging from 18 up to 98% (Prop et al. 1984, Loonen et al. 1997, Owen & Black 1999). This indicates that potential nest success is probably equally high across the breeding range with values up to around 90%. In this study, since we targeted Barnacle Goose colonies in which expected predation risk was low, to assess maximal reproduction potential, it is difficult to compare these values but maximal values indicate that the absence of migratory behaviour has little impact on overall nesting success.

In Flanders, the number of goslings leaving the nest was on average 3.21. This is comparable to the mean yearly number of hatched young described for the Gotland population (3.30-3.14) and Barnacle Geese breeding at Hellegatsplaten in the Netherlands in 2005 (3.01) but higher than values observed at the same location in 2004 (2.69) (Forslund 1993, Larsson & Forslund 1994, Pouw et al. 2005, van der Jeugd et al. 2006). For Greenland, 3.13 and 3.42 goslings leaving the nest were observed in two separate studies and in Svalbard and Kolokolkova bay, mean brood sizes at hatching of 3.32 and 2.87 were reported respectively (Dalhaug et al. 1996, Ogilvie et al. 1999, van der Jeugd et al. 2003). Since the number of goslings leaving the nest is the result of clutch size, nest success and hatching success, the value provides a good overview of initial reproductive success in all the different populations studied. Again, contradictory to our initial hypothesis, comparison of the different values show little evidence of influences of difference in migratory behaviour on reproductive success.

We did not expect differences in fledging success between migratory and non-migratory populations since we expected this is irrespective of maternal condition and thus of migratory efforts. We observed a brood size at fledging of 1.47 and a number of fledged young/successful pair of 2.35. Again, this value corresponds to brood sizes at fledging found on Gotland (2.4-3.0), Svalbard (2-2.66) and is similar to average brood sizes observed on the wintering grounds in Islay, Scotland (2.11) and Ireland (1.91) (Boyd 1968, Cabot & West 1973, Owen & Norderhaug 1977, Black & Owen 1995, Loonen et al. 1997). Only Choudhury et al. (1993) found a higher value of 3.05 at Svalbard, but given the time of the study and the limited sample size, this value represents a subsample of the study by Black & Owen (1995). Overall, large differences in survival of chicks from hatching to fledging were reported, mostly attributed to high predation levels, next to chilling, starvation and diseases (Forslund 1993, Larsson & Forslund 1994). However, Larsson & Forslund (1994) state that the underlying cause of chick mortality may be food competition. One of the main characteristics of the pattern of gosling losses between hatching and fledging was that losses were very unevenly distributed, with many pairs losing all their young. This suggests that parents are unequally competent to protect and lead their goslings to good grazing sites (Prop et al. 1984, Owen & Black 1989, Prop & de Vries 1993). This is also clear from our results since we observed that the chance of not a single chick surviving was higher in nests with clutch sizes of 1 or 2 eggs, which are known to often be produced by the younger goose couples who also have less experience in raising goslings (Forslund & Larsson 1992).

In the resident breeding population in Flanders, juvenile survival was estimated at around 78%, subadult survival at 88% and yearly adult survival at 93%. These values closely match the values observed in the non-migratory population in the Netherlands where juvenile survival was estimated at 67-76% and subadult and adult survival at 85-91% (van der Jeugd & Kwak 2017). In Barnacle Geese breeding on Gotland, first year survival was estimated at 83% and annual survival in older birds at around 95% by Larsson et al. (1988, 1998) and van der Jeugd & Larsson (1998) reported a combined juvenile/subadult survival (pre-breeding survival) of 73-95%. In Arctic breeding, migratory Barnacle Geese, overall annual mortality over longer periods has been estimated at around 5-10% for Svalbard breeding birds (Ogilvie & Owen 1984, Owen & Black 1989) and annual survival was estimated at 86-92% in Greenland breeding birds and at 90% for Russian breeding birds (Cabot & West 1973, Ebbinge et al. 1991). Similar to the observations from the reproductive output, there is no indication that the absence of migratory efforts in the Flanders' breeding population would result in a higher survival probability at any of the age classes considered.

Overall, the combination of these parameters and their stochasticity result in a modelled average yearly growth rate (lambda) of 1.12 ± 0.07 for the Barnacle Goose breeding population in Flanders. As could be expected based on the high degree of similarity of the different life history parameters considered this value does not differ much from the growth rates observed in other Barnacle Goose populations, both migratory and non-migratory. Various studies reported Barnacle Goose steady state population growth rates of 1.12 and 1.17 at Svalbard, 1.04 in Greenland and 1.07 in Russia (Cabot & West 1973, Owen & Norderhaug 1977, Feige et al. 2008). The most recent and comprehensive study on this topic for the Svalbard breeding population reported an average growth rate of 1.05, within a 90% confidence interval of 0.92 to 1.17, which is somewhat lower than the value obtained for Flanders here (Layton-Matthews et al. 2019). For the Dutch breeding population, an initial exponential growth phase with rates up to 1.48 were reported in the first decade of the 21st century, which was attributed to a higher absolute fitness in the North Sea population in comparison to Arctic breeding populations (van de Jeugd et al. 2009). However, after this initial phase, the Dutch Barnacle Goose breeding population continued to double between 2007 and 2014 (van der Jeugd & Kwak 2017), which corresponds to an overall growth rate of around 1.10.

In conclusion, although we studied animals in situations with low predatory pressure to assess maximal reproductive potential, neither reproductive output nor population growth rate exceeded those of other Barnacle Goose populations, both migratory and non-migratory. It is well known that in long-lived species such as geese, population growth is often more sensitive to variation in survival than reproduction (Layton-Matthews 2019), but estimated survival rates in the Flanders' population also closely matched other values reported for the species. This shows that the absence of migratory efforts has little impact on the potential output and growth of the Flanders' breeding population. However, survival of chicks from hatching to fledging showed a much higher degree of variability over all populations than other parameters considered. Due to the low average chick survival in Flanders, elasticity analysis showed little impact of a further chick survival reduction, but the low values do allow occasional increases in this survival. In the years studied here, chick survival was 0.5 on average but increased up to 0.7 in 2018. Although the reason for this is unclear, probably environmental (weather) conditions play an important role. Given that the months May and June 2018 were extraordinarily (once in every 30 year) dry in Flanders (KMI 2019) may well have had an important impact on chick survival in that year. Given the low elasticity of his single parameter and the fact that the model currently already incorporates a 25% chance of such high chick survival, which is much higher than the expected 3% of a similarly dry spring (once every 30 years), we do not expect that, even under changing climate conditions for Flanders, population growth would increase much as a result of increased chick survival in the near future.

6.2 MANAGEMENT SCENARIOS

Elasticity of adult survival was much higher than any of the other parameters evaluated for Barnacle Geese in Flanders (Figure 10). This further translates into the three management scenarios we evaluated, where it was shown that the use of moult captures clearly outcompetes management options that did not include culling.

Evaluation of management through nest destruction showed that a very high effort would be required, targeting over 75% of all nests, to achieve a population decrease. Nest management can be done by shaking, collecting, destroying, puncturing, or treating the eggs with, for example, corn oil (Hindman et al. 2014). The effect of shaking or puncturing of the eggs is that they will not hatch. Shaking causes the death of the embryo, puncturing causes the egg to rot. Treating the eggs with (corn) oil, a pore closing agent, causes the death of the embryo by a lack of oxygen (Voslamber 2010). According to Van Daele et al. (2012) there are some points to keep in mind while treating the eggs, these comprise: not damaging the nest, leaving the eggs seemingly intact and covering the eggs after puncturing. However, these methods are not flawless. Primarily, manually reducing eggs is very labour-intensive (Voslamber 2010). Even after shaking or puncturing, a small proportion of the eggs still hatches. This is because the measures are not always perfectly executed and or not all nests are found (Schekkerman et al. 2000, Voslamber 2010). Furthermore, reducing eggs will only influence the population a few years after the treatment, due to the time delay of the treatment (Klok et al. 2010). Another disadvantage is that, while reducing the eggs, other breeding bird species are disturbed (Van Daele et al. 2012). This can negatively influence other conservation plans specifically designed for these species, which is undesirable.

Since Barnacle Geese in Flanders are considered a protected species, hunting was not evaluated in this report. However, since hunting also targets (sub)adult survival, it can be expected that hunting could also succeed in impacting population growth, as is the case in other goose species and is probably a general feature of long-lived species (Gauthier et al. 2001, Menu et al. 2002, Madsen et al. 2012). Hunting also has the benefit of low cost, since it can be done by voluntary hunters. It is nonetheless questionable whether hunting could attain similar effectiveness as moult captures in the case of the Flanders' breeding population. From October on, the resident breeding population becomes enriched with migratory birds with numbers regularly exceeding 5,000 Barnacle Geese (Devos & T'Jollyn 2017). During this period, breeding birds form mixed flocks with migratory birds, which would make it impossible to target high numbers of local birds through hunting during this period. In addition, shooting birds just after the breeding season would also target higher proportions of non-adult Barnacle Geese, which could lower the impact of hunting. This is evidently also the case for moult captures, but is countered there by the high total number of birds targeted under this management approach. Similar findings were reported by van der Jeugd & Kwak (2017) for Barnacle Geese in the Netherlands, where the authors evaluated derogation shooting and claim that current harvest levels seem insufficient to reverse population growth and that the effect of shooting is low, caused by a disproportionate number of immature individuals just after the breeding season, and by shooting individuals that are not belonging to the target population during late winter. Van der Jeugd & Kwak (2017) therefore advise that, in order to resort maximum effect, derogation shooting should be directed towards adult, reproducing breeding birds at the start of the breeding season. Given the protected status of the species in Flanders, management by shooting should also need to be organized through derogation shooting and would probably, given the strict conditions related to this type of management, not result in numbers comparable to the number of adults trapped during the moulting season. Additionally, although hunting is viewed as a direct killing agent, the pellets also unintentionally wound other animals or only cripple the targeted animals (van Eerbeek 2013, Clausen et al. 2017). Additionally, the shooting not only influences the population size, it also potentially influences the spatial distribution (van Eerbeek 2013). Casas et al. (2009) showed that, by shooting, animals can perceive humans as potential predators and can alter their behaviour in their presence, which can potentially negatively interfere with other management measures.

The alternative, as is currently applied in Flanders and modelled under 0, is to trap and kill geese during the moulting period. Killing is done by gassing with CO_2. According to the Dutch RDA (2012) using CO_2 is the least animal-unfriendly method to kill a large number of geese. The use of CO_2 is approved by the European Commission and allows the meat, if desired, to be further processed for human consumption (van Eerbeek 2013). Furthermore, Reyns et al. (2018) constructed a cost-benefit analysis for the management of greater Canada Goose in which they compared different scenarios in which they showed that the moult capture scenario resulted in the highest avoided cost due to damage.

This leads to the overall conclusion that, when the aim is to reduce or remove entirely, the resident population of Barnacle Geese in Flanders, culling birds through moult captures is currently the most feasible method to achieve rapid and effective effects. Unlike egg reduction, culling has an immediate effect in the year after application (Klok et al., 2010). Moult trapping also has the advantage of being less labour-intensive, and causes less disturbance of other breeding bird species (Van Daele et al. 2012). For Flanders, a level of

2,000 breeding pairs of Canada Goose was considered well above acceptance levels in terms of both agricultural as ecological damage (Van Daele et al. 2012). Therefore, the current population levels and reproductive output for Barnacle Goose in Flanders, when unmanaged, would leave a window of 5 to 10 years before the species levels would exceed similar acceptance levels. Currently, the first efforts to manage this small population in Flanders and the Netherlands are undertaken, but ongoing management will remain necessary in the years to follow.

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