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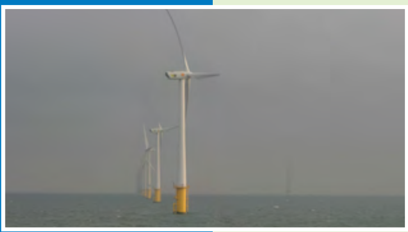
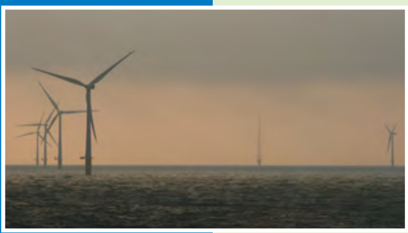
A modelling approach to estimate effects on population levels in seabirds





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commissioned by: Noordzeewind

date: 18 November 2011
report nr: 11-026
OWEZ_R_212_T1_20111118_Cumulative effects

Status:	Final report
Report nr.:	11-026 OWEZ_R_212_T1_20111118_Cumulative effects
Date of publication:	18 November 2011
Title:	Effect studies Offshore Wind Egmond aan Zee: cumulative effects on seabirds
Subtitle:	A modelling approach to estimate effects on population levels in seabirds
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Number of pages incl. appendices:	247
Project nr:	06-466
Project manager:	drs. M.J.M. Poot
Name & address client:	Noordzeewind, ing. H.J. Kouwenhoven 2e Havenstraat 5B 1976 CE IJmuiden
Reference client:	Framework agreement for the provision of “MEP services” 30 May 2005
Signed for publication:	Team manager bird ecology department Bureau Waardenburg bv drs. T.J. Boudewijn
Initials:	

The Quality Management System of Bureau Waardenburg bv has been certified by CERTIKED according to ISO 9001:2000.

Photos frontcover; M. Poot,Bureau Waardenburg

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Preface

The Dutch government has granted 'Noordzeewind' (Nuon and Shell Wind Energy) the possibility to build a wind farm consisting of 36 wind turbines off the coast of the Netherlands, near Egmond. This project serves to evaluate the economical, technical, ecological and social effects of offshore wind farms in general. To gather knowledge in these areas, a Monitoring and Evaluation Program (NSW-MEP) has been developed. The knowledge gained by this project will be made available to all parties involved in the realisation of large-scale offshore wind farms. Bureau Waardenburg and IMARES Wageningen UR, in cooperation, have been commissioned to execute the study of the effects on flight paths, flight altitudes and flux of migratory and non-migratory birds.

Here we present results in which the cumulative effects of several wind farms on the Dutch continental shelf on birds are modelled. This study has been carried out by means of creating population models for different bird species. The effects of wind farms on these species, as the (estimated) number of victims, was fed into the population models. Most of these estimates were based on the radar measurements and field observations gathered from OWEZ, the operational offshore wind farm Egmond aan Zee (Krijgsveld *et al.* 2011, and Leopold *et al.* 2010) and less from the baselline study near Meetpost Noordwijk (Krijgsveld *et al.* 2005).

Many national and international colleagues have made contributions to this report and are thanked in the acknowledgements. Most of them were met at a workshop on cumulative effects in May 2007 in Peterborough, which was organised by COWRIE around the time that the present study was started. Since that time information on developments and techiques for assessing cumulative effects have been shared, however, it is strongly recommended that in a wider international context further investigations on the cumulative effects of multiple wind farms are initiated. The urgency for such collaborative studies became clear at the Conference for Wind energy and Wildlife impacts, organised by NINA, in Norway in May 2011. The preliminary results of the present study represented one of the few studies addressing the issue of cumulative effects that were presented. Furthermore, in many countries around the North Sea, plans and developments in offshore wind energy are at a much higher level than in 2007.

The Offshore Wind Farm Egmond aan Zee has a subsidy of the Ministry of Economic Affairs under the CO_2 Reduction Scheme of the Netherlands.

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Summary

Framework
In order to increase the supply of renewable energy in the Netherlands, the Dutch government has supported the construction of a near-shore wind farm of 36 Vestas V90/3MW wind turbines 10-15 km off the coast of Egmond aan Zee, in the Netherlands (OWEZ, Offshore Wind farm Egmond aan Zee). This project served as a demonstration project to build up knowledge and experience with the construction and exploitation of large-scale offshore wind farms. In order to collect this knowledge, an extensive Monitoring and Evaluation Program (MEP-NSW) has been designed in which the economical, technical, ecological and social effects of the wind farm are gathered. Within this framework a baseline (Krijgsveld *et al.* 2005, Leopold *et al.* 2005) as well as an effect study (Leopold *et al.* 2010, Krijgsveld *et al.* 2011) have been carried out to measure the impact of the wind farm on birds. Those studies describe the impact of a single wind farm. In the present study we attempt, for the first time, to estimate the cumulative effects of multiple offshore wind farms in part of the North Sea on the population levels for a range of bird species.

The offshore wind farm at Egmond aan Zee was the first offshore wind farm built in the Netherlands, with a second one completed one year later (but not studied as part of the OWEZ report). The Dutch government supports plans to build more turbines at sea in the coming years. As described in the following paragraphs a single wind farm will have certain effects on birds by means of collision, disturbance and/or barrier effects. Single wind farms might have a minor impact on the reproduction and survival (and thus population sizes) of birds as shown in several studies on single wind farms. Numerical impacts are mainly on a local scale by changes in distribution. The greater the effect, in terms of a decrease in reproduction and/or survival, the greater the impact will be on the population size. To this end, the construction of multiple wind farms at sea has the potential to reach the level above which survival and reproduction are significantly affected, which could potentially lead to a decrease in population levels at the wider (international) scale. With plans and proposals for expanding the number of wind farms in the Dutch part of the North Sea, the question is:

> *What are the cumulative effects (as quantitative as possible) of multiple wind farms in the Dutch North Sea on the population levels of bird species?*

General approach
This is the first attempt to model the cumulative effects of multiple offshore wind farms as impacts on the population level for a range of species in part of the North Sea. Based on the effect study in and around OWEZ the potential effects due to increased mortality resulting from collisions have been calculated (Krijgsveld *et al.* 2011).

The cumulative effects on birds, as estimated in this report, are derived on the basis of impacts measured at the wind farm OWEZ during the effect study in 2007-2010. In this report these effects are extrapolated in order to represent multiple wind farms on

the Dutch continental shelf. We consider two scenarios, the first with multiple wind farms near-shore (all comparable in their effects with OWEZ) and the second with multiple wind farms scattered across the Dutch North Sea area, thus a proportion in deeper offshore areas (partly comparable with OWEZ and partly corrected for differences in species composition and abundance). The current study focuses on seabirds and to a lesser extent also deals with migrant species (passerines, waders, etc.). The seabirds considered are those that breed in coastal areas in the Netherlands (e.g. cormorants, gulls and terns) and those that regularly migrate or winter in the Dutch North Sea (e.g. divers, fulmars, gannets, ducks, gulls and alcids).

The cumulative effects were assessed for a selection of the most relevant and vulnerable bird species and were assessed at the population level with the aid of population models. The approach consisted of constructing population models, which were tested alongside known population trends. The data used in constructing the models were obtained from both published and unpublished field studies and from the relevant populations, and included parameters such as reproduction rate, mortality by age class, age at first breeding, proportion of non-breeding birds, etc. The potential effects of a number of wind farms, such as an increase in mortality, could then be applied to these populations. The potential effects of a number of wind farms were calculated based on the results obtained from the study at OWEZ. To this end the scenarios of multiple wind farms were based on the having a number of wind farms with the same configuration of OWEZ.

In this study a multi-step modelling approach was adopted in order to estimate the cumulative effects on the population levels of seabirds.

1. Step one consisted of the construction of population models for the species concerned, which described the known population trends in recent decades; the 0-model.
2. Step two involved calculating the levels of species-specific mortality resulting from two scenarios of multiple wind farms in the Dutch part of the North Sea (near-shore and more offshore).
3. In step three the levels of increased mortality were applied to the 0-models. Results provided an indication of the size of the effect of multiple wind farms at the population level. This provided an effect-model.
4. Step four involved calculating the amount of additional mortality needed in order to reach zero growth in each of the 0-models. This provided an indication of the level of additional mortality that could be sustained by the population without it showing a decline. This resulted in a 0-growth-model.
5. In step five we calculate, by means of the Potential Biological Removal (PBR) approach, the maximum sustainable harvest; e.g. the number of victims that can be sustained by the population without serious effect on the population size.
6. Finally the outputs from the steps three, four and five were compared in order to provide a number of different perspectives into the cumulative effects of multiple wind farms at the population level.

The methods described above are based on known techniques that have been proven in earlier studies with similar questions. Population models were based on Leslie matrix models, a proven simple and robust way of modelling animal populations. These models allow for density dependence and immigration and emigration where necessary. A major aspect in the structure of many of the bird populations modelled in this study is the existence of floaters. Floaters are essentially non-breeding adults that have the capability of compensating for a loss of breeding individuals from the population by joining the breeding population and, therefore, acting as a buffer for the population under periods with increased mortality.

Floaters and the breeding population
The loss of adult birds as a result of fatal collisions with wind turbines has direct consequences for the population, as the number of reproducing pairs may decrease and the growth of the population affected. It is also anticipated that the loss of breeding individuals from the population can be buffered by the non-breeding proportion of the adult population, termed floaters. This report presents an overview of this phenomenon based on a comprehensive literature review. Depending on the population structure the proportion of non-breeding birds can vary, although particularly in long-lived species may be several tens of percents. In many populations this remains complex and poorly studied. This means that despite an increase in the mortality of breeding adult individuals a population can remain stable.

Effects of offshore the OWEZ wind farm
Species differ in their response to wind farms; some fly straight through it, whereas at the other extreme, others avoid it entirely. These responses may also affect the foraging behaviour of birds in the area of the wind farm. Consequently, three types of potential negative effects on birds have been identified, which we define as follows:

- *collision of flying bird*s - being the numbers of individuals of each species that physically collide with the turbines or that are mortally injured by encounters with the air vortices associated with the revolving rotor blades;
- *disturbance* - being the displacement from the spatial arrangement of resting and/or feeding birds caused by the construction of the turbines, represented by differences in these distributions between the baseline pre-construction condition and those post-construction (typically a reduction in numbers of birds);
- *occurrence of barrier effects* - being the changes in flight trajectories within the construction area post-erection of turbines (in terms of flux, flight paths and altitudes) relative to pre-construction conditions.

These effects can have a negative impact on the survival and/or reproductive output of individuals, which in turn can be reflected in their populations. This may especially be true if numerous wind farms are present within the distribution range or flyway of a species.

To assess the sensitivity of the modelled populations to the potential effects of multiple wind farms a number of scenarios were calculated with respect to collision victims.

During surveys of local birds around OWEZ, no significant avoidance of the wind farm by foraging birds was identified. Nevertheless, there are indications that the distributions of some species have been altered due to the loss of habitat associated with the wind farm. Little is known over barrier effects, although the increased energetic costs of flying around a wind farm or the possibility that birds decide not to utilise the area beyond a wind farm may reduce their reproductive output or in extreme cases reduce survival. The observations gathered around OWEZ are not suitable for assessing the consequences of barrier effects at the population level, however, compared to the direct mortality associated with collisions with turbines the consequences of barrier effects are considered to be negligible.

Estimating effects by means of population models

The construction of population models will enable the assessment of the impact of additional mortality on the population. First, the additional mortality was estimated as described above, and subsequently this estimate fed into the population models. In this report the effect of additional, wind farm-related, mortality on a population is simulated for two multiple offshore wind farm scenarios. In order to investigate the effects on species specific populations the following approaches have been followed:

- Estimating the response of a population to a certain amount of victims based on calculations using parameters on avoidance behaviour and fluxes of birds as determined in the field at OWEZ. This is called the effect-model.
- Estimating the amount of additional mortality to reach zero growth. This estimate is an indication for the level of the maximum sustainable effect; maximum in the sense that a greater level of additional mortality will lead to a decreasing population. This is called the 0-growth-model.
- For a few species, offshore wind farms may be a threat because of (significant) disturbance from feeding areas (habitat loss; guillemots, razorbill, gannet, greater skua). Data from OWEZ were not able to support this hypothesis. Therefore, the 0-growth-model has been used to provide an indication as to the maximum (acceptable) levels for these species. This has also been done for species with low, but variable fluxes during migration (Bewick's swan and brent goose) because of the presumed barrier effects.

In this report an overview of the different models used are given in table 5.1.0, the overview of input parameters of the different models are presented in table 5.1.2. In table 5.2.1, the calculted levels of Potential Biological Removal (PBR) are presented for selected species for populations occurring in the Dutch part of the North Sea together with the calculated number of collision victims (expressed as breeding pairs) for OWEZ alone and the two scenarios of multiple offshore wind farms in the Dutch part of the North Sea.

In this study we have followed a worst case scenario approach in that;

- As a precautionary approach we have attributed all victims to females with breeding status. Also for Dutch breeding populations the modelling did not take into account the potential that collisions involved birds of a foreign origin (thus outside of the modelled populations) or with birds of a non-breeding, juvenile or subadult status.

- For seabird species breeding outside the Netherlands, the impacts of new Dutch offshore wind farms were restricted to one geographical population (mostly Scotland), while in reality a much wider breeding range with different geographical populations might be involved.
- In most models a floater population of 10 or 30% has been chosen. Published research has shown that higher percentages can occur, especially in many long-lived species, meaning that a larger buffer function could be present in the floater population.
- In the models used, density dependence is modelled in relation to reproduction only. Density dependence can also act on mortality via the process of intra-specific competition between individual birds outside the breeding season. In a case in which the carrying capacity decreases the intra-specific competition on resources will increase, with the consequence of a potentially lowered survival of birds. This would imply that in this situation the victims occurring due to human-related impacts such as from collisions with wind turbines could have a so-called compensatory effect, as victims taken out from the population will reduce the intra-specific competition.
- The levels of additional collision-related mortality applied to the population models have been kept stable over time. This assumes that a decrease in the population due to collisions does not affect the intensity of flight movements in and around the wind farms. This situation is possible in cases where wind farms are developed in high quality foraging areas with birds from low quality areas replacing those victims in these high quality areas.

Regarding the floaters (non-breeding adults) in the population, collision victims have been calculated as breeding adults only. This implies the situation that floaters are not directly affected, but immediately take the empty places in the breeding population. In reality floaters can also collide with wind turbines, especially in those situations when new offshore wind farms are located in areas where disproportional more floaters are present. In such a situation the impacts on the level of the population also occur; with floaters disappearing from the population as a result of collisions the recruitment of new breeding birds is ultimately hampered. Our models also describe this strong connection between floaters and breeding birds yet in order to illustrate a worst case scenario, we have chosen to concentrate all victims in the group of breeding birds. This also has an immediate consequence on reproduction by assuming the failure of the brood.

Calculating effects of multiple offshore wind farms from findings from OWEZ

Estimates for the numbers of collision victims for each species for each scenario were calculated using the SNH-Band model. The SNH-band model calculates the probability of collision of a certain species on the basis of the physical characteristics of the wind turbines and the species in question. The single most important aspect in calculating this collision probability is the level of avoidance that the bird shows. In general, estimates for the levels of avoidance of birds are largely based on estimated figures and seldom on field studies, particulaly in the case of offshore situations. In the OWEZ field studies both radar and visual data were used to determine the level of avoidance at an offshore wind farm. This was combined with species- and species-group- specific fluxes determined with the same combination of visual and radar observations, meaning that realistic data from an actual and relevant situation were used. Also, species group and species-specific fluxes were determined by the same combination of visual and radar observations. Again these situation-specific field data proved valuable in calculating the number of collision victims at OWEZ.

Effects at the population level
When the extrapolated numbers of victims calculated for multiple offshore wind farm scenarios are applied to the population models, those species that currently have a stable or increasing popualtion trend do not show any decline in numbers. For these populations the influence of the additional mortality resulting from victims of the wind farms is very limited. Instead, the population trends appear to be dominated by ecological changes in the environment, such as is known from the decline in the numbers of kittiwakes in Scotland in response to changes in food availability.

In this study two of the populations studied are currently undergoing a decline, namely the international Bewick's swan population and the Dutch breeding population of the herring gull. The population model outcomes in these two species show that the influence of the increased mortality due to new offshore wind farm developments is relatively small in relation to their current trends. We conclude that stochastic incidents are not likely to be more influential in situations with co-current impacts, including offshore wind farms. In the case of long-lived species, as studied in this report, such scenarios with consecutive years of strongly decreased recruitment is rare, and most of the time not caused by a natural phenomenon.

The baseline population models highlighted that two of the species, herring gull and Bewick's swan, both had very negative trends even before the effects of the wind farms were applied. The calculated additional mortality furthered this trend. In the case of the herring gull the calculated number of collision victims was within the limit of the Potential Biological Removal level for a species with a 'near threatened' status, even though it is classed above this criteria according to the IUCN and is still very common in northwest Europe. The Potential Biological Removal is a calculation based on species-specific maximum population growth rate and a minimum population estimate and calculates the total number of victims feasible without the population becoming into danger.

At the end of this summary we present a copy of table 5.2.2 from chapter 5. Here, the estimated number of collision victims, as calculated using the Band model, are presented for the studied species along with an indication as to the effects of the two modelled wind farm scenarios on the studied populations.

Limitations of the findings
The fluxes and densities of local seabirds as measured by the effect studies have proven to be extremely location-specific; especially based on the ship-based surveys that were conducted across a much larger area than OWEZ itself. The habitat features that determine the distribution patterns of foraging seabirds, for both breeding and non-breeding birds, are: distance to the coast; water depth; salinity; turbidity; and the presence and availability of food, the latter being of paramount importance. This limits the certainty with which the findings from OWEZ can be applied to locations further offshore. The effects of multiple wind farms might not be simply additive but could also be multiplicative or non-linear depending on the presence or absence of birds. The effect studies on OWEZ do not yield data with which these effects can be assessed, therefore, the assumption that the effects of multiple wind farms were additive was used for the purpose of this report. In terms of species and numbers of birds present, however, the knowledge gained from OWEZ may not be applicable to areas further offshore.

Band model and avoidance figures
The outcomes of the model presented in this study are based on calculations that include stochastic variability in both mortality and reproduction, thus the extremes are incorporated. Furthermore, the macro and micro avoidance figures based on the OWEZ field study must be regarded as conservative. We have found higher avoidance figures than those assumed for some species by SHN for use with the Band model, however, we feel that for most species avoidance rates are in reality even higher. Limitations in spatial resolution of the radar data and the difficulty of species identification of individual radar targets limited the calculation of species-specific avoidance rates and specifically avoidance at close vicinity to the rotors. With a higher resolution of data in the analysis of micro avoidance more birds can be identified as flying outside the rotor area. It is therefore reasonable to expect that in future, better (and probably higher) avoidance rates will be determined through the use of technical innovations in radar ornithology or alternative studies of individual flight paths (e.g. GPSlogger studies). Based on the sensitivity analysis of Chamberlain *et al.* (2006) reproduced in table 2.3.1 this will result in a lower number of estimated collisions.

For the Dutch breeding birds, but also for foreign breeding populations, it is only possible to make predictions of population growth at the larger scale. On a smaller scale, such as at a colony or regional level (e.g. North Holland), the calculation of cumulative effects using population models is not possible as relevant data for individual colonies are largely unknown. A second important limitation concerns the spatial restriction of the calculation by cumulative scenarios of wind power development for only the Dutch North Sea. This means that developments in the neighbouring parts of the North Sea are not included. This limitation is largely motivated by the lack of knowledge on how offshore wind farms are being developed outside of Dutch waters. In a wider international context it is desirable to initiate further investigation on the cumulative effects of multiple wind farms in within the total distribution range of species, or in case of waterbirds and other migrant species in the entire flyway of a species.

Conclusions

This study represents the first attempt to estimate the cumulative effects of multiple offshore wind farms in part of the North Sea on the population levels of a range of species. The analyses in this report have shown that the effects of the multiple offshore wind farm scenarios are far away from the levels above which decreasing trends occur and as such, this might be representative for multiple wind farms in the Dutch North Sea. This conclusion was confirmed by using the Potential Biological Removal approach; a method for estimating the level of sustainable mortailty without causing a negative population trend. Emphasis should be placed on the fact that calculations were carried out conservatively and followed precautionary assumptions. Future research related to monitoring the effects around new offshore wind farms in deeper waters would likely yield results to confirm that in this report a worst-case approach has been followed.

Table 5.2.2 Summary of cumulative effects due to multiple wind farms for the two scenarios. The impact of cumulative effects on populations were determined through populations models (for species for which sufficient data were available; see species accounts in chapter 5) and/or based on calculations of the level of Potential Biological Removal as presented in table 5.2.1. See paragraph 6.1 for an overview of the worst case scenario followed in the population modelling and paragraph 6.3 for an explanation of the limitations of the conclusions presented here based on the scenarios studied. The figures of number of victims in this table are taken from table 5.2.1.

species	Dutch name	region	current population trend	Scenario 1 (11 OWEZ-like farms 10-20 km offshore) n victims	Cumulative effects due to multiple wind farms of scenario 1 (taken into account worste case scenario of the models)	Scenario 2 (11 offshore farms across Dutch North Sea, thus largely > 20 km	Cumulative effects due to multiple wind farms of scenario 2 (taken into account worste case scenario of the models)
red-throated diver	roodkeelduiker	North sea basin	unknown	1.8	highly unlikely	9.2	highly unlikely
cormorant	aalscholver	Netherlands	stable	332.2	positive	unknown	positive?
shag	kuifaalscholver	Scotland	stable	0.0	highly unlikely	0.0	highly unlikely
gannet	jan van gent	Scotland	stable	17.2	highly unlikely	199.2	highly unlikely
fulmar	noordse stormvogel	Scotland	stable	0.0	highly unlikely	0.0	highly unlikely
Bewick's swan	kleine zwaan	NW-Europe	decrease	5.0	highly unlikely	5.0	highly unlikely
brent goose	rotgans	NW-Europe	stable	5.0	highly unlikely	5.0	highly unlikely
shelduck	bergeend	NW-Europe	stable	0.0	none	0.0	none
eider	eider-eend	NW-Europe	stable	0.0	highly unlikely	0.0	none
common scoter	zwarte zee-eend	NW-Europe	unknown	1.0	highly unlikely	1.0	highly unlikely
great skua	grote jager	Scotland	stable	0.8	highly unlikely	39.6	highly unlikely
great black-backed gull	grote mantelmeeuw	NW-Europe	stable	209.4	highly unlikely	134.9	highly unlikely
herring gull	zilvermeeuw	Netherlands	decrease	585.6	highly unlikely	698.1	highly unlikely
lesser black-backed gull	kleine mantelmeeuw	Netherlands	stable	776.8	highly unlikely	875.8	highly unlikely
little gull	dwergmeeuw	NW-Europe	unknown	172.3	highly unlikely	75.1	highly unlikely
common gull	stormmeeuw	NW-Europe	stable	355.7	highly unlikely	152.7	highly unlikely
kittiwake	drieteenmeeuw	Scotland	decrease	345.6	highly unlikely	217.1	highly unlikely
Sandwich tern	grote stern	Netherlands	increase	28.8	highly unlikely	154.5	highly unlikely
common tern	visdief	Netherlands	increase	2.8	highly unlikely	60.5	highly unlikely
little tern	dwergstern	Netherlands	stable	0.0	none	0.0	none
guillemot	zeekoet	Scotland	increase	0.1	highly unlikely	0.1	highly unlikely
razorbill	alk	Scotland	increase	0.1	highly unlikely	0.1	highly unlikely
puffin	papagaaiduiker	Scotland	stable	0.0	none	0.0	none
knot	kanoet	Can./Greenl./Russia	decrease	0.0	none	0.0	none
redwing	koperwiek	Scandinavia	unknown	max. 3,400	highly unlikely	max. 3,400	highly unlikely
starling	spreeuw	Central Europe/Russia	unknown	max. 3,400	highly unlikely	max. 3,400	highly unlikely
skylark	veldleeuwerik	Scan./Russia	unknown	max. 3,400	highly unlikely	max. 3,400	highly unlikely
meadow pipit	graspieper	Scandinavia	unknown	max. 3,400	highly unlikely	max. 3,400	highly unlikely

Nederlandse samenvatting

Doel en kader onderzoek cumulatieve effecten OWEZ

In 2007-2010 heeft uitgebreid veldonderzoek plaatsgevonden om de effecten van het Offshore Windpark bij Egmond aan Zee (OWEZ) op vogels te meten. Het gaat hierbij enerzijds om zeer grote aantallen vogels van uiteenlopende soortsgroepen die in voor- en najaar langstrekken, en anderzijds om lokaal actief rondvliegende en foeragerende zeevogels. Binnen deze laatste categorie kan weer onderscheid gemaakt worden in enerzijds zeevogels die langs de Nederlandse kust broeden en op zee voedsel vinden voor zichzelf en hun jongen en anderzijds in zeevogels die buiten Nederland broeden maar in de Nederlandse wateren komen ruien en/of overwinteren. Van deze verschillende categorieën vogels worden in dit rapport op basis van de gemeten effecten aan het eerste gerealiseerde offshore windpark in het Nederlands deel van de Noordzee de cumulatieve effecten op populatieschaal doorgerekend indien meer offshore windparken ontwikkeld worden.

Aanpak doorrekenen cumulatieve effecten van meerdere offshore windparken

De cumulatieve effectinschatting is uitgevoerd voor een selectie van de meest relevante en kwetsbare vogelsoorten. De doorrekening van effecten is uitgevoerd met behulp van wiskundige populatiemodellen. De aanpak bestond eruit om wiskundig het populatieverloop van tientallen jaren terug tot heden modelmatig te simuleren. Hierbij werden als basis populatiedynamische gegevens gebruikt uit gepubliceerde en ongepubliceerde veldonderzoeken. Het gaat hierbij om verschillende parameters zoals reproductie, mortaliteit per leeftijdsklasse, jaar van eerst broeden, de hoeveelheid niet-broeders in de populatie, etc. Vervolgens zijn op basis van de uitkomsten van het veldonderzoek rond en in OWEZ het aantal potentiële slachtoffers van meerdere nieuwe windparken zoals OWEZ in de populatiemodellen gebracht om het effect in de toekomst te simuleren. Er is hierbij uitgegaan van het scenario waarbij windparken ontwikkeld worden van een vergelijkbare configuratie als het OWEZ windpark.

Alleen aanvaringsslachtoffers kunnen doorrekenen

Om de gevoeligheid van de modellen te testen hebben we verschillende populatie- en effectscenario's doorgerekend met betrekking tot aanvaringsslachtoffers. Uit de module tellingen van lokale zeevogels vanaf schepen in en rond OWEZ zijn geen statistisch significante effecten vastgesteld die kunnen duiden op vermijding van het windpark door foeragerende zeevogels. Niettemin zijn er wel voor sommige individuele soorten aanwijzingen dat er verstoringseffecten optreden die duiden op habitatverlies door de aanwezigheid van een windpark. Over de effecten van barrière-werking is eveneens nog weinig bekend. Energetische consequenties van het omvliegen zouden kunnen betekenen dat vogels een hoger energieverbruik ondervinden, dan wel besluiten niet meer gebruik te maken van bepaald foerageergebied omdat een windpark dit belemmert, al dan niet door verhoogde vliegkosten. In populatiedynamische termen zou dit kunnen betekenen dat vogels eerder komen te overlijden of dat broedvogels minder efficiënt voedsel kunnen aandragen voor hun jongen en dat via een verminderde reproductie effecten kunnen optreden. De waarnemingen die ten aanzien van verstoring en barrière-werking rond OWEZ verzameld zijn, zijn niet geschikt om het effect op populatie door te rekenen doordat deze niet één op één te vertalen zijn in verhoogde sterfte. Tevens wordt onderbouwd dat de meeste soorten waarvoor extra vliegkosten een rol zouden kunnen spelen, broedvogels zijn die op dagbasis verschillende foerageervluchten maken. Juist voor die soorten is gebleken dat deze niet of nauwelijks het OWEZ windpark mijden (het gaat hierbij om zilver- en kleine mantelmeeuw en aalscholver). OWEZ en toekomstige parken zijn gesitueerd op te grote afstand van kolonies van sterns om binnen het foerageerbereik te liggen. Effecten ten aanzien van barrière-werking zullen daarom marginaal zijn in vergelijking tot de directe effecten van mogelijk verlies van individuen door aanvaringen.

Het gebruik van populatiemodellen

Er is in deze studie gebruik gemaakt van Leslie matrix modellen. Dit is een wiskundige berekeningsmethode waarbij op basis van de basisparameters reproductie en sterfte het populatieverloop van een soort kan worden berekend. Bij het modelleren hebben wij rekening gehouden met het optreden van stochastische variabiliteit in de natuur en deze ook in de doorrekeningen toegepast. Dit houdt in dat random 'trekkingen' zijn gedaan uit de verdeling van mogelijke reproductie- en mortaliteitgetallen (op basis van het gemiddelde getal en standaard deviatie gevonden in de literatuur). Uit de 'stochastisch' gestuurde modeluitkomsten is vervolgens het gemiddelde populatieverloop bepaald. De validatie van deze gemiddelde trend bleek bij de verschillende gemodelleerde soorten goed overeen te komen met de historische trends.

Aangenomen is dat in deze parameters geen langjarige trends of langzaam golvende fluctuaties voorkomen, wat mogelijk wel het geval kan zijn, maar waarover bij veel soorten geen informatie beschikbaar is. Voor sommige soorten bleek een gemiddeld getal voor nationale of internationale populaties al niet goed bekend, de reden waarom robuuste methoden als de Potential Biological Removal methoden zijn ontwikkeld. In internationaal verband wordt dan ook met het beschikbaar komen van deze robuuste analyse methoden in de context van beoordeling van ecologische effecten van windparken hiervan steeds meer gebruik gemaakt.

Er bestaan modellen waarmee het verloop van de gehele populatie op basis van individuele vogels wordt gemodelleerd en waarbij meer basisparameters ten aanzien van gedrag moeten worden ingevuld. Kennis over de waarde en variabiliteit van gedragsparameters per individu ontbreekt doorgaans, wat de reden is geweest dat deze modellen niet gebruikt zijn. Kennis over de relatie van deze parameters met reproductie en mortaliteit is vaak onbekend en ook hoe eventueel deze relaties beïnvloed worden door effecten van offshore windparken.

Omdat onbekend is in welke leeftijdsverdeling of geslachtsverdeling aanvaringsslachtoffers vallen is uitgegaan van een negatief scenario waarbij alle slachtoffers adult en een reproducerend vrouwtje zijn. Hierdoor moeten de in deze studie voorspelde uitkomsten worden beschouwd worden als zeer negatieve scenario's. De toegepaste "worst-case" benadering in de voorspelde uitkomsten van deze studie reduceert daarmee de kans op onderschatting van de werkelijke risico's op effecten bij vogels.

De rol van 'floaters' in de populatie
Het directe effect van het wegvallen van adulte vogels door fatale aanvaringen met windturbines heeft direct consequenties voor de populatie aangezien het aantal reproducerende broedparen kan afnemen alsmede de aanwas van de populatie wordt beïnvloed. Beide aspecten zijn als effect gemodelleerd. Hierbij is er tevens rekening mee gehouden dat wegvallen van reproducerende adulte vogels uit de populatie direct kan worden gebufferd doordat de opengevallen plek ingenomen wordt door een geslachtsrijpe vogel uit het aandeel niet-broedvogels. In dit rapport wordt een overzicht van dit fenomeen gegeven aan de hand van een uitgebreid literatuuronderzoek. Afhankelijk van de populatieopbouw kan het aandeel niet-broedvogels variëren. Met name bij langlevende soorten blijkt uit literatuuronderzoek dat hoewel dit bij veel populaties complex ligt, en daardoor slecht onderzocht is, dit aandeel uit enkele tientallen procenten kan bestaan. Dit betekent dat indien er genoeg aanwas is en er dichtheidsafhankelijke regulatie plaatsvindt een populatie in evenwicht blijft. Om de rek in de verschillende populaties van de geselecteerde soorten te onderzoeken, is onderzocht bij welk aantal slachtoffers de populatie gelijk blijft (dus waarbij sterfte en aanwas met elkaar in evenwicht zijn). Dit scenario is voor een aantal soorten vervolgens doorgerekend voor een populatie met respectievelijk 0, 10 en 30 % niet-broedvogels in de populatie. De 10 en 30 % niet-broedvogels zijn alleen te modelleren als de populatietrend dat toelaat. Met een neergaande populatie door bijvoorbeeld een slechte reproductie of hoge sterfte slinkt het niet-broedvogel percentage tot een laag niveau, omdat in het ene geval er niet genoeg aanwas is of omdat er onder adulte vogels een hoge sterfte heerst waardoor deze vogels direct kunnen deelnemen aan het broedproces door opengevallen plekken in te nemen.

Cumulatieve effecten geëxtrapoleerd op basis van metingen in OWEZ
Op basis van het veldonderzoek verricht in en rond OWEZ zijn voor verschillende soorten het aantal aanvaringsslachtoffers berekend met behulp van het Band-model. Het Band-model is een rekenmethode die de kans op een aanvaring berekent op basis van fysische kenmerken van de windturbines en van de vogelsoort in kwestie. Een belangrijk aspect waar dit model rekening mee houdt is het vermijdingsgedrag van de vogel van het rotoroppervlak. In het veldonderzoek van Krijgsveld *et al.* (2011) is specifiek naar dit aspect gekeken, waarbij op basis van een combinatie van visuele en radarwaarnemingen per soortgroep en vaak ook op soortniveau vermijdingsgedrag is bepaald. Tevens zijn soortsgroep- en soortsspecifieke fluxen bepaald door middel van dezelfde combinatie van visuele en radarwaarnemingen, zodat het aantal aanvaringsslachtoffers voor OWEZ kon worden berekend.

Voor twee scenario's zijn vervolgens de aantallen aanvaringsslachtoffers geëxtrapoleerd voor 10 extra offshore windparken van het OWEZ type (en configuratie) in het Nederlandse deel van de Noordzee, een getal dat overeenkomt met de orde van grootte van windparkplannen voor zo ver bekend. Het eerste scenario betreft het uitbreiden van het gebied met nieuwe windparken direct in de omgeving van OWEZ, omdat de bestudeerde effecten van OWEZ daarmee direct toepasbaar zijn. Het tweede scenario betreft de mogelijke ontwikkeling van windparken verder uit de kust in een gebied met een andere zeevogelgemeenschap dan bestudeerd in en rond OWEZ. Langjarige tellingen vanuit het vliegtuig zijn gebruikt om de compositie en voorkomen van zeevogels voor dat uitgestrektere gebied te beschrijven en de resultaten van OWEZ te corrigeren voor een extrapolatie verder op zee. Hier zitten nogal wat haken en ogen aan (zo is niet bekend wat de flux is van vliegbewegingen van zeevogels), maar dit is bij gebrek aan een vergelijkbaar effectonderzoek ver op zee, zoals uitgevoerd in OWEZ, het best mogelijke wat gedaan kan worden.

Het doorrekenen van effecten door middel van populatie modellen
Door het maken van een wiskundig populatiemodel kan doorgerekend worden wat het effect is op de populatie van het aantal vogels dat jaarlijks slachtoffer wordt door de windturbines van meerdere toekomstige windparken. Het aantal vogels dat additioneel sterft kan ingevuld worden in de modellen. Dit is gebeurd voor twee scenario's van ieder een totaal van 11 windparken. Om het effect op vogelpopulaties te beschrijven zijn de volgende benaderingen gekozen:

- Allereerst is het populatieverloop doorgerekend op basis van het geëxtrapoleerde berekende aantal slachtoffers van OWEZ. Dit model is het effectmodel genoemd.
- Als tweede is het voor iedere vogelsoort opgestelde populatiemodel gebruikt om te bepalen bij welk aantal slachtoffers de populatie stabiel blijft, het zogenaamde nul-groei-model.
- Voor een aantal soorten waarbij potentieel verstoring door windparken kan optreden (zeekoet, alk, jan van gent, grote jager) zijn de uitkomsten van het nul groei-model gebruikt als indicatie voor de orde van grootte van het aantal vogels dat in een negatief scenario verstoord zou mogen worden om een stabiele populatie te behouden. Hetzelfde werd gedaan voor kleine zwaan en rotgans, omdat bij deze soorten in potentie barrière-werking zou kunnen optreden. In OWEZ en onmiddelijke omgeving werden deze soorten in dermate lage aantallen vastgesteld dat geen aanwijzingen werd verkregen dat verstoring dan wel barrière-werking optreedt.

In dit rapport wordt in tabel 5.1.0 een overzicht gegeven van de bovenstaande modellen, waarbij in tabel 5.1.2 voor ieder model de gebruikte parameterwaarden worden gepresenteerd. In tabel 5.2.1 wordt vervolgens een overzicht gegeven van het aantal slachtoffers dat in de Leslie modellen is ingevuld, en deze worden vergeleken met de uitkomsten van het zogenaamde PBR – Potential Biological Removal niveau, waarbij onderscheid is gemaakt in de twee scenario's van ieder een totaal van 11 windparken.

In deze studie hebben we een worst case scenario gevolgd waarbij;

- Voor de Nederlandse broedvogelpopulaties (kleine mantelmeeuw, zilvermeeuw, aalscholver, sterns) alle slachtoffers zijn toegekend aan de broedende vrouwtjes tijdens het broedseizoen, terwijl in werkelijkheid vogels

jaarrond, van een buitenlandse oorsprong, en/of uit de andere leeftijds- (juveniele en onvolwassen vogels tot ongeveer vier jaar oud) en/of geslachtscategorie als slachtoffer kunnen vallen.
- Voor zeevogelsoorten die buiten Nederland broeden zijn de impacts van nieuwe Nederlandse offshore windparken doorgerekend op een geografisch afgebakende regio, meestal Schotland. Dit is waarschijnlijk onrealistisch omdat waarschijnlijk sprake is van effecten op vogels uit een veel groter verspreidingsgebied waarbij effecten op populaties meer gespreid worden.
- In de meeste modellen is gekozen voor een floater populatie van 10 of 30 %. Literatuur onderzoek zoals uitgevoerd binnen deze studie heeft laten zien dat in veel langlevende soorten een hoger percentage floaters kan voorkomen, hetgeen betekent dat een sterkere bufferende werking kan uitgaan van de floater populatie.
- In de hier gebruikt modellen is dichtheidsafhankelijkheid alleen gemodelleerd voor reproductie. Dichtheidsafhankelijkheid kan ook optreden bij sterfte door middel van verhoogde of in het geval van additionele sterfte door uitval van aanvaringsslachtoffers door verlaagde concurrentie.
- In onze berekeningen hebben we het aantal berekende slachtoffers dat ingevuld wordt in het model gelijk gehouden, terwijl bij een eventueel afnemende populatie het aantal slachtoffers ook zal afnemen.

Zoals hierboven aangegeven laten we alle slachtoffers vallen onder de broedende vrouwtjes van een vogelsoort. Dit betekent dat de floaters niet direct worden beïnvloed, maar alleen een rol spelen bij het innemen van de vrijgevallen plekken van weggevallen broedvogels. In werkelijkheid kunnen floaters wel degelijk direct in aanvaring komen met een windturbine, speciaal wanneer offshore windparken worden gerealiseerd in gebieden waar zich proportioneel meer floaters bevinden. Het effect dat dan zou kunnen optreden is dat de recruitment in de populatie wordt beïnvloed. Door alle slachtoffers onder de broedende vrouwtjes te laten vallen, nemen we ook mee dat er een effect is van het mislukken van broedsels indien één van de partners van een broedpaar wegvalt door een aanvaring tijdens het broedseizoen.

Resultaten cumulatieve effecten
Wanneer de geëxtrapoleerde slachtofferaantallen berekend met het Band-model voor de twee scenario's ingevuld worden in de populatiemodellen, blijkt dat bij die soorten die een stabiele of stijgende populatietrend vertonen, niet te leiden tot terugval van populaties. De invloed van de berekende slachtofferaantallen is heel beperkt. De populatietrends lijken te worden overheerst door ecologische veranderingen in de omgeving. Veelal zal dit voedsel gestuurd zijn, zoals bekend is bij de grote neergang van de aantallen drieteenmeeuwen in Schotland. Bij het opstellen van de populatiemodellen bleek dat, dus nog voor effecten van het windpark in rekening werden gebracht, met name ook bij de kleine zwaan en de zilvermeeuw een zeer negatieve populatietrend optreedt. De geëxtrapoleerde slachtofferaantallen berekend met het Band-model voor de twee scenario's versterken die trend enigszins. Voor de zilvermeeuw bleek als enige soort dat het aantal berekende aanvaringsslachtoffers

beneden de grens van de Potential Biological Removal (PBR) niveau te liggen als ervan uitgegaan wordt dat de soort een 'bedreigde' status heeft. Volgens IUCN criteria heeft de zilvermeeuw deze status nog niet omdat de soort in noordwest Europa nog steeds zeer algemeen is. De Potential Biological Removal is een berekeningswijze die op basis van de soortspecifieke maximum populatiegroeisnelheid en een minimale populatieschatting het totaal aantal slachtoffers berekent zodanig dat de populatie niet in gevaar is. Bij de berekening wordt rekening gehouden met de beschermingsstatus zoals opgesteld door de IUCN. Voor alle soorten die in dit rapport zijn meegenomen geldt voor de noordwest Europese populaties dat deze niet bedreigd zijn (*least concern*). Voor alle overige soorten liggen de berekende aantallen aanvaringsslachtoffers ruim binnen de PBR voor deze beschermingsstatus, overeenkomstig de bevindingen voor die soorten waarvoor een populatiemodel kon worden opgesteld. Voor de zeer talrijke zangvogelsoorten veldleeuwerik, koperwiek, graspieper en spreeuw konden geen populatiemodellen worden opgesteld, maar de PBR waarden bleken ver boven de berekende aantallen slachtoffers te liggen.

Discussie

Er zijn verschillende beperkingen aan te geven voor de doorgerekende scenario's. Voor de Nederlandse broedvogels, maar ook voor de buitenlandse populaties is het alleen mogelijk op een grote ruimtelijke schaal voorspellingen te doen van het populatieverloop. Op een kleinere schaal, zoals bijvoorbeeld op kolonieniveau of op regionaal niveau (Noord-Holland), zijn doorrekeningen van cumulatieve effecten met behulp van populatiemodellen niet mogelijk omdat dan ook op die schaal empirische gegevens over reproductie, mortaliteit per leeftijdsklasse, jaar van eerst broeden, de hoeveelheid niet-broeders in de populatie, etc. noodzakelijk zijn. Sommige van deze gegevens (zoals mortaliteit per leeftijdsklasse, of het aandeel niet-broedvogels in de populatie) zijn zelfs voor sommige Nederlandse broedvogelpopulaties niet bekend, waardoor gegevens van naburige landen moesten worden gebruikt. Eén van de belangrijkste beperkingen van de hier gepresenteerde doorrekeningen is dat deze gebaseerd zijn op onderzoek aan het windpark OWEZ, een windpark met een specifieke configuratie en op een unieke locatie. Dit betekent dat de geldigheid van de bevindingen beperkt is tot de scenario's die hier behandeld zijn. Grotere windparken en windparken met afwijkende configuraties ten opzichte van OWEZ zullen met dezelfde intensiteit onderzocht moeten worden om opnieuw cumulatieve effecten door te rekenen. Tenslotte, de laatste beperking betreft de ruimtelijke begrenzing van het doorrekenen van cumulatieve scenario's van te ontwikkelen windenergie voor alleen het Nederlandse deel van de Noordzee. Dit houdt in dat de ontwikkelingen in de naburige delen van de Noordzee niet meegenomen zijn. Deze beperking is voor het grootste deel ingegeven door het ontbreken van kennis over hoe effecten van offshore windparken zullen doorwerken op populaties vogels in buitenlandse wateren die op enig moment binnen de territoriale wateren van Nederland kunnen voorkomen. Een dergelijke exercitie is wel noodzakelijk, waarbij een brede internationale aanpak noodzakelijk is. In ieder geval is voor het Nederlandse deel door middel van voorliggend rapport een eerste inschatting gemaakt van de effecten van meerdere windparken bovenop het nu operationele OWEZ windpark.

Band-model en vermijdingsgedrag
De model uitkomsten die in deze studie gepresenteerd worden, zijn gebaseerd op berekeningen waarbij rekening is gehouden met stochastische variabiliteit in zowel sterfte als reproductie, zodat extremen zijn meegenomen. De macro- en microvermijdings getallen zoals vastgesteld in de OWEZ veldstudie moeten beschouwd worden als conservatief. Vergeleken met het SHN 2010 rappport betreffende het Band-model is er voor een aantal soorten sterker vermijdingsgedrag vastgesteld. We denken dat voor de meeste soorten het vermijdingsgedrag nog sterker is door de beperkingen in ruimtelijke resolutie om radargegevens te analyseren en de moeilijkheid om onderscheid te maken tussen soorten. Wanneer de resolutie verbetert verwachten we dat van nog meer vogels bepaald kan worden dat zij buiten het rotoroppervlak passeren. Daarom is te verwachten dat wanneer door technische innovatie in de radar ornithologie of in dit kader alternatieve studies naar individuele vliegpaden (bijvoorbeeld door middel van GPSloggers) worden uitgevoerd, in de toekomst betere (lees hogere) schattingen van vermijdingsgedrag zullen worden vastgesteld. Dit zal leiden tot lagere berekende aantallen slachtoffers met mogelijk een relatief grote impact gezien de gevoeligheidsanalyse van Chamberlain *et al.* (2006) in tabel 2.3.1 voor de factor vermijdingsgedrag.

Eindconclusie en aanbevelingen
Dit is de eerste keer dat cumulatieve effecten van meerdere offshore windparken zijn onderzocht op verschillende vogelsoorten op populatie niveau met name ten aanzien van zeevogels. Het gebruik hierbij van soortspecifieke populatiemodellen is uniek. Hoewel een aantal beperkingen gelden voor de hier gevolgde aanpak, is de eindconclusie dat het aantal aanvaringsslachtoffers dat berekend wordt voor de verschillende onderzochte vogelsoorten voor alle soorten niet leidt tot negatieve populatie trends. Deze bevinding kon worden bevestigd met de *Potential Biological Removal* –berekening, een alternatieve benadering om het effect te toetsen op het ontstaan van negatieve populatietrends. Aan de andere kant moet benadrukt worden dat het doorrekenen van slachtofferaantallen op een zeer conservatieve wijze is uitgevoerd. De belangrijkste aanbeveling betreft het uitvoeren van een vergelijkbaar effectonderzoek als uitgevoerd in OWEZ bij een toekomstig windpark ver op zee. Naar verwachting kunnen dan de fluxen gemeten worden die bevestigen dat in dit rapport een *worst-case* benadering is gevolgd. Tevens zouden dan voor een aantal offshore zeevogelsoorten mogelijk statistisch significante verstoringseffecten kunnen worden vastgesteld, iets wat in en rond OWEZ niet lukte doordat te lage aantallen van deze soorten aanwezig waren.

1 Introduction

1.1 Background

Framework

In order to increase the supply of renewable energy in the Netherlands, the Dutch government has supported the construction of a near-shore wind farm of 36 turbines 10-15 km off the coast of Egmond in the Netherlands (OWEZ, Offshore Wind farm Egmond aan Zee). This project serves as a demonstration project to build up knowledge and experience with the construction and exploitation of large-scale offshore wind farms. The knowledge gained through this project will be made available to those parties that are involved in the realisation of large-scale offshore wind farms. In order to collect this knowledge, an extensive Monitoring and Evaluation Program (MEP-NSW) under supervision of the Ministry of Water & Transport has been designed in which the economical, technical, ecological and social effects of the wind farm are gathered. Carrying out this MEP serves 'learning goals' for future wind farms further offshore as well as 'effect assessment goals' for the near-shore wind farm itself. Within this framework a baseline and effect study has been carried out to measure the impacts of the wind farm on birds.

Baseline and effect study

The baseline and effect study design carried out in and around OWEZ follows recommended set ups as proposed by Exo *et al.* (2003) and Drewitt *et al.* (2006), consisting of:

(1) transect studies to analyse the distribution and density of seabirds before and after construction (Leopold *et al.* 2010);
(2) radar studies to analyse fluxes of birds, flight altitudes and flight paths during day and night (Krijgsveld *et al.* 2011);
(3) visual observations and flight call recordings to detect movements of passage migrants and foraging birds including avoidance behaviour (Krijgsveld *et al.* 2011).

Prior to construction, in 2003-2004, the 'reference situation' was established. For flying birds, the results of the baseline study are reported in Krijgsveld *et al.* (2005) and Dirksen *et al.* (2005). Krijgsveld *et al.* (2005) describe fluxes, flight altitudes and flight paths as they were measured at Meetpost Noordwijk, approximately 40 km south of the OWEZ wind farm area, using both radar and a range of visual observation techniques. Leopold *et al.* (2004) describes 'the reference situation' of occurrence of 'local' seabirds.

Over the summer of 2006, the turbines of OWEZ were constructed and by September 2006, the first energy was produced at the wind farm. With the wind farm constructed and operational, observations to establish the effects of the wind farm on flying birds began in April 2007. Since then, various types of surveys were underway to assess the numbers, distribution, behaviour and flight movements within the framework of the effect study. The primary goal of the effect study was to measure direct effects of the wind farm on birds.

Derived from research results on land, the MEP-NSW required research to enable an analysis of three types of possible negative effects on birds, which we define as follows:

- *collision of flying birds* – being the numbers of individuals of each species that physically collide with the turbines or that are mortally injured by encounters with the air vortices associated with the revolving rotor blades;
- *disturbance* - being the displacement from the spatial arrangement of resting and/or feeding birds caused by the construction of the turbines, represented by differences in these distributions between the baseline pre-construction condition and those post-construction (typically a reduction in numbers of birds);
- *occurrence of barrier effects* – being the changes in flight trajectories within the construction area post erection of turbines (in terms of flux, flight paths and altitudes) relative to pre-construction conditions.

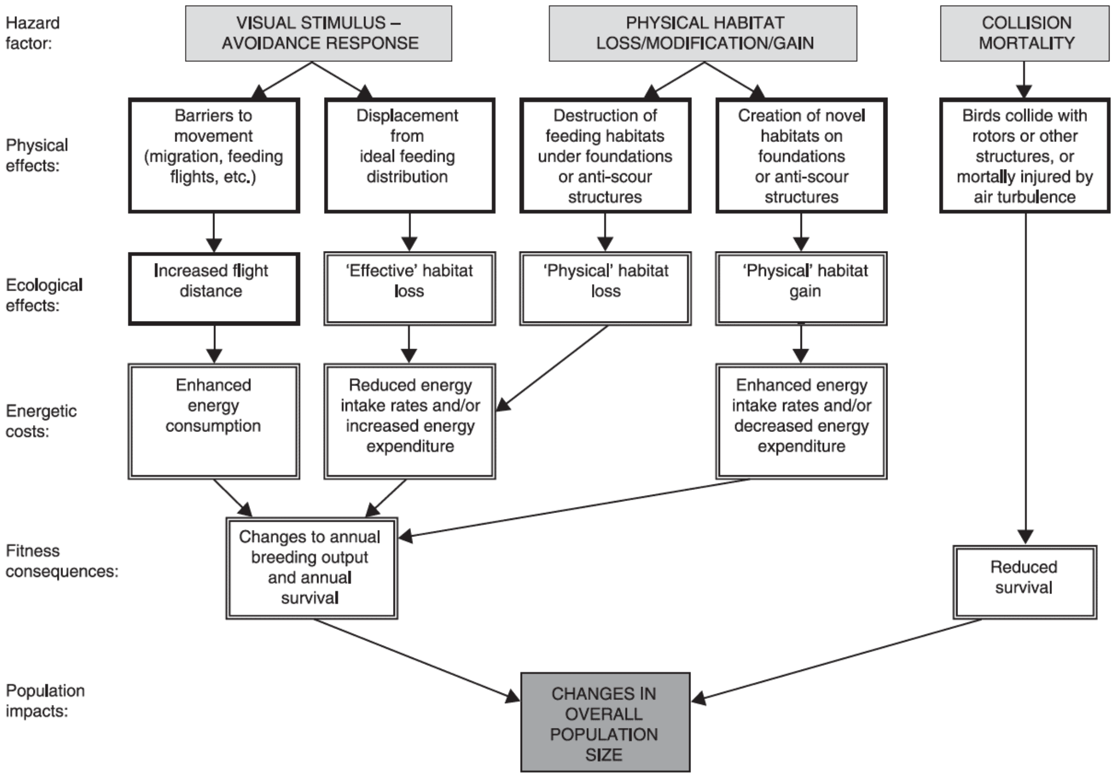


Figure 1.1.1. Flow chart describing the three major hazard factors (light shaded boxes) presented to birds by the construction of offshore wind farms, showing their physical and ecological effects on birds, the energetic costs and fitness consequences of these effects, and their ultimate impacts on the population level (dark shaded box). The boxes with a heavy solid frame indicate potentially measurable effects; the double-framed boxes indicate processes that need to be modelled (from Fox et al. 2006).

The ultimate effects of these three themes at the population level, collisions, disturbance and barrier effects, can have their impacts through different ecological pathways. An overview of these different pathways has been made by Fox *et al.* (2006) and is reproduced in figure 1.1.1.

The overview in figure 1.1.1 is more extensive than the three themes mentioned above, furthermore also the potential positive effects are described. However, ultimately these three effects can be transformed into a decrease in survival and/or reproduction of the species involved. On the contrary, positive effects can become visible in an increase in survival and/or reproduction. So, finally both positive and negative effects can have their impact on the population size. Figure 1.1.1 indicates also which effects can be measured in the field and which have to be modelled.

Collision has a direct impact on the survival of birds and thus on population size. Disturbance (or attraction) and barrier effects are translated into the fitness of a species by a cascade of steps (figure 1.1.1.); e.g. ecological and energetic effects follow physical effects before being transformed into fitness consequences. Furthermore, these hazard factors can interact. Birds avoiding a wind farm have a lower risk on collision than birds being ignorant or attracted by the turbines. Avoidance leading to a prolonged flight (barrier effect) lowers the risk on collision but increases the energy expenditure (see Masden *et al.* 2009, 2010). Species differ in their response to the wind farm; some will fly straight through it, whereas at the other extreme, others will avoid entirely (Krijgsveld *et al.* 2011, Leopold *et al.* 2010). So the relevance of different hazardous effects (cf. Figure 1.1.1) might differ from species to species and, therefore, effect studies have to be species specific, if possible.

The effects of the OWEZ wind farm have been measured, and evaluated, by means of a before and after construction setup. Based on:

- ship based local surveys (Leopold *et al.* 2010);
- radar studies (Krijgsveld *et al.* 2011);
- visual observations (Krijgsveld *et al.* 2011);

we are well informed about:

- species composition (quantitative) on sea and in the lower air layers;
- fluxes and flight altitudes;
- flight paths;
- avoidance on macro (wind farm) and micro scale (turbines within the farm) in flight and at sea(feeding, resting).

Based on these outcomes the collision risk can be, and will be, estimated quantitative.

1.2 Cumulative effects of wind farms on birds

The OWEZ Egmond aan Zee was the first offshore wind farm built in the Netherlands; with a second one completed one year later (figure 1.4.1). The government is intended to build more turbines on sea in the forthcoming years. As pointed out in previous paragraph one wind farm will have certain effects on birds by means of collision, disturbance and/or barrier effects. The question is whether these changes of single offshore wind farms will affect total populations by means of a lowered survival or reproduction. It is assumed that single wind farms likely have a minor impact on population sizes based on the measured effects in several studies on single offshore wind farms. Numerical impacts of single offshore wind farms are on a local scale by changes in distribution and flight paths (e.g. Masden 2009, 2010, Percival 2009, 2010, Pettersson 2005, Petersen *et al.* 2006a,b). However, the bigger the effect, in terms of decrease in reproduction and/or survival, the bigger the impact will be on the population size. Potentially multiple wind farms erected at sea might reach the limits of the effects above which survival and reproduction are affected significantly, leading to a decrease on population levels on a national and international scale. With plans and proposals for expanding the number of wind farms in the Dutch part of the North Sea, the question arises:

> *What are the cumulative effects (as quantitative as possible) of multiple wind farms in the Dutch North Sea on the population level of bird species?*

The principle of cumulative effects of wind farms on birds can be explained as follows: a single wind farm 'A' leads to a small increase in mortality due to collision, This small increase lies well within the capacity of that population for compensating for additional losses (regenerating) and hence has little or no effect on the overall population level. The same would apply to a second wind farm 'B', taken in its own. However, the increase in mortality resulting from wind farms 'A' and 'B' together may exceed the capacity of the population for regeneration. In this case the population will start to decline. Whereas the impact of 'A' and 'B', each on their own, did not lead to changes in population size, the cumulative impact of 'A' + 'B' will cause a decrease in bird populations.

Within the MEP-NSW (Krijgsveld *et al.* 2005) it was originally intended to focus the study of cumulative effects just on additional mortality due to collision. The relative importance of the two other major factors (disturbance, barrier effect, Figure 1.1.1) was assumed to be lower. The current studies on seabird distributions and flight movements in the OWEZ wind farm (and some other studies) have yielded data that make it possible:

- to make estimates of the number of victims due to collision;
- the relevance of disturbance and barrier effects, especially for offshore seabirds, could not yet assessed properly (see Krijgsveld *et al.* 2011 and Leopold *et al.* 2010, and further explanation see below and in section 2.3.2 and 2.3.3).

So finally, this report addresses the question on cumulative effects due to additional mortality (collision) on population levels. Disturbance and barrier effects as cumulative effect will be addressed and discussed for relevant species.

1.3 General approach

The OWEZ wind farm is the first offshore wind farm in the Dutch coastal waters. More offshore wind farms are expected in the Dutch North Sea in the near future. The North Sea is within the migration routes of many seabird species and in addition many migrating land bird species cross this area body in large numbers (Platteeuw *et al.* 1998, Wernham *et al.* 2002, Lensink *et al.* 2002). Consequently, multiple wind farms in the North Sea may affect many species and populations. Cumulative effects should be estimated as loss of birds, e.g. increased mortality by collisions and/or loss of habitat.

The field studies of MEP-NSW have yielded information on different aspects of local and migrant birds in a near-shore area of the Dutch coast. This information is used for as much as possible in the 'cumulative effects report'. Information from other sources (e.g. bimonthly aerial surveys by Ministry of Water & Transport, etc), is used when necessary; for instance, to get an idea about the seabird community in *offshore* parts of the North Sea. Data from the MEP-NSW are for many species just representative for near-shore.

Assessing the cumulative effects on population level by a modelling approach
Currently, there is an absence of data on impacts from actual situations of multiple offshore wind farms at the population level. To determine the cumulative effects of collision risk, disturbance and barrier effects caused by multiple offshore wind farms, a modelling approach must be applied. Modelling of cumulative risks is founded on the modelling of the three effects on birds that are posed by individual wind farms. It requires initial modelling of effects for each wind farm within the range of the species of interest. For that purpose effects have to be measured in the field, which can be used in a mathematical population model to determine the impacts on species-specific bird populations.

Effects of multiple wind farms finally lead to an effect (decrease) on reproduction and/or survival. Those are the two parameters that can be used in population models of species. Therefore, the number of victims is translated into a decrease in survival and/or reproduction. Collision gives a strait forward estimate for reduction in survival (and if relevant reproduction); e.g. a dead adult in de breeding season leads also to zero reproductive success. Translating disturbance or barrier effects into a decrease in reproduction and survival is more difficult. The energetic costs of a prolonged flight (barrier effect) have no straightforward relation with survival (or reproductive success), although the relation is evident (Piersma & Van Gils 2010, see also Masden 2008, 2009). To assess the effect of disturbance one could assume that the birds leaving the area (being disturbed), become a dead bird. Since most seabirds, especially outside the breeding season, are opportunistic, they will leave the disturbed area for another area. So, calculating all disturbed birds as dead birds is certainly not realistic. The effects due to energy budget interference are difficult to model in terms of population dynamics. In our study, based on the OWEZ findings these will probably be very small in the long-lived species present in the OWEZ area. The impacts of disturbance and barrier effects on the populations studied in this report (by increased survival and/or decreased reproduction by means of increased energy expenditure) are expected to be marginal in relation to direct effects by fatal collisions.

In conclusion, an increase in mortality by collision is relative easy to estimate and is for most species the major factor when assessing cumulative effects of multiple wind farms in the current Dutch situation.

Estimates for collision

Collision risk is the chance that a bird passing the wind farm will collide with a turbine or with the turbulence behind the turbine. Collision risk is determined by measuring both collision rate and the flux of birds passing the wind farm. For this purpose, fluxes of birds, flight altitudes, and avoidance rate were measured (Krijgsveld *et al.* 2011).

Collision rate was intended to be measured by quantifying numbers of birds colliding with wind turbines in the OWEZ wind farm. WT-Bird, a method to measure collision rate, has been developed by the Energy Research Centre of the Netherlands (ECN) (Wiggelinkhuizen *et al.* 2006 a, b), but was at the time of installation of OWEZ, and still is, not available for the Vestas V90/3MW wind turbine as applied in the OWEZ wind farm. To date no other proven bird collision measurement technologies are available in the OWEZ wind farm. Therefore, other routes are taken to get estimates of the number of victims due to collision (see section 2.3).

In this report collision victims are calculated by means of the Band model (SHN 2010), based on the field measurements of fluxes and avoidance rates of Krijgsveld *et al.* (2010).

Setup of modelling cumulative effects

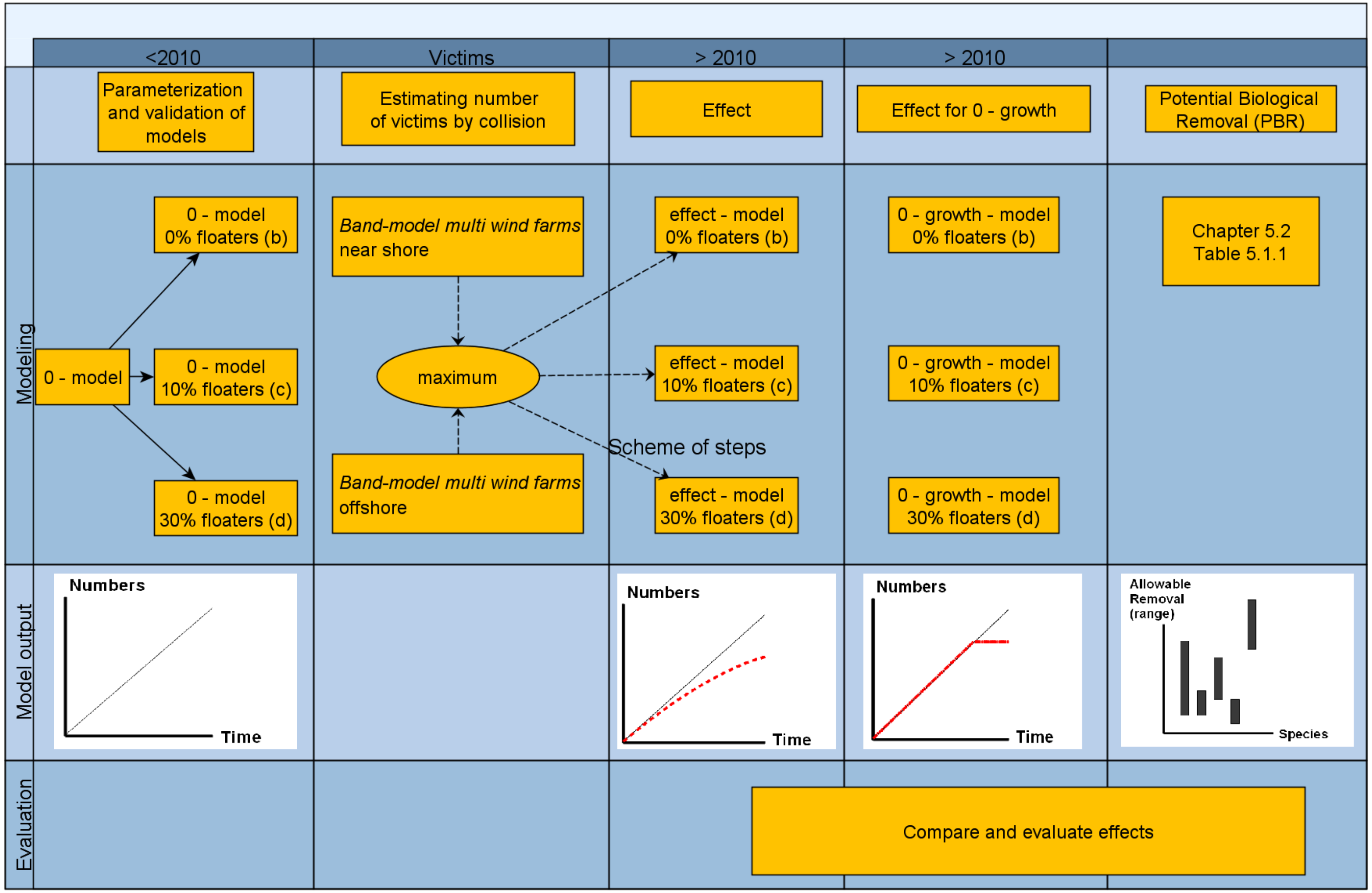
This is the first attempt to model cumulative effects of many offshore wind farms as impacts on the population level for a range of species in a part of the North Sea. Based on the effect study in and around OWEZ the potential effects due to increased mortality resulting form collisions have been calculated (Krijgsveld *et al.* 2011).

As a next step we have used the population models to calculate at which level of numbers of victims for the selected species a stable population arise, under the condition that many species (e.g. gannet, great skua, guillemot, sandwich tern etc.) have shown or show increasing population trends. This gives the possibility to see whether the estimated number of collisions victims (effect-model) lies well within the range of acceptable numbers of victims (0-growth-model, PBR-approach). The latter two models pinpoint also to the limits of expected (and acceptable) effects of disturbance and barrier effects (added up to effects of collision).

The methodology to estimate cumulative effects will consist of a multi-step modelling approach for each of the three types of effects of wind farms on birds (Figure 1.3.1).

1. Step one consists of the construction of a population model, which describes the population trends in recent decades. For this purpose, literature data on life history and population variables, such as mortality and reproduction, were gathered, and models were built and validated with past population trends. This is called the 0-model.
2. Step two involves calculating realistic levels of mortality resulting from collision for two scenarios for multiple wind farms in the Dutch part of the North Sea (see further section 1.4 for a description of the two).
3. In Step three the levels of increased mortality are put into a 0-model; thus named effect-model. The results give an idea on the size of the effect of multiple wind farms.
4. In step four we calculate the amount of additional mortality to reach zero-growth in the realistic 0-model used for the species. This provided an indication as to the size of potential effects without serious impacts on current population levels. This is called the 0-growth-model
5. In step five we calculate, by means of the Potential Biological Removal (PBR) approach, the maximum sustainable harvest; e.g. in this the number of victims that may occur, without serious effects on the population size.
6. Finally the output from the steps three, four and five were compared in order to provide a number of different perspectives into the cumulative effects of multiple wind farms at the population level.

Figure 1.3.1 *Overview of the approach in this report; explanation in the text (§ 1.3). The 0-model is the basic model (with parameters) is validated on population development in recent decades (<2010) and used for further exploration of effects. In graphics the black dashed line. The effect-model is used to estimate the effect of additional mortality (>2010) as estimated (in graphic the red line). The 0-growth-model is used to determine the size of additional mortality (>2010) to reach 0-growth (in graphic the red line). The Potential Biological Removal approach is another way to look for the maximum amount of additional mortality (in graphics comparison with other methods, blocks defining ranges of mortality). The letters b, c, and d refer to 0%, 10% and 30% floaters.*



For those species for which disturbance effects (effectively summounting to habitat loss) are expected, the number of birds that might be displaced was calculated. This number of birds must be regarded as an *worst case* scenario, namely total avoidance and subsequently dying of the birds (under the assumption of a full satisfied system on carrying capacity and severe intraspecific competition leading to starvation of the losers), based on the proportional surface area of 10 new offshore wind farms of the size of OWEZ.

1.4 Scenarios multiple new offshore wind farms in Dutch waters

At the moment of publication of this report, it is largely unknown which scenarios for the development offshore wind farms will be most realistic for the Dutch part of the North Sea. At the moment two offshore wind farms are operational, OWEZ and Prinses Amalia Wind farm. Another eleven wind farms have a construction permit (indicated in green in fig 1.4.1).

In this report we chose to calculate the effects of 10 additional OWEZ-like wind farms in the Dutch part of the North Sea. They will be developed in the same depth range and distance class from the coast (near-shore, scenario 1), or will be developed scattered over the North Sea, thus mainly in deeper water and at greater distances from the coast (offshore, scenario 2). For the first it is clear that data from OWEZ are representative. Since the some bird community might be expected in an expanded OWEZ. In the latter case most turbines will be constructed in water deeper than 20 m. Here the bird community might differ from OWEZ; so results from OWEZ are not to be translated directly (see further in § 2.5 and chapter 4). Despite these difficulties we have made an attempt under strict assumptions.

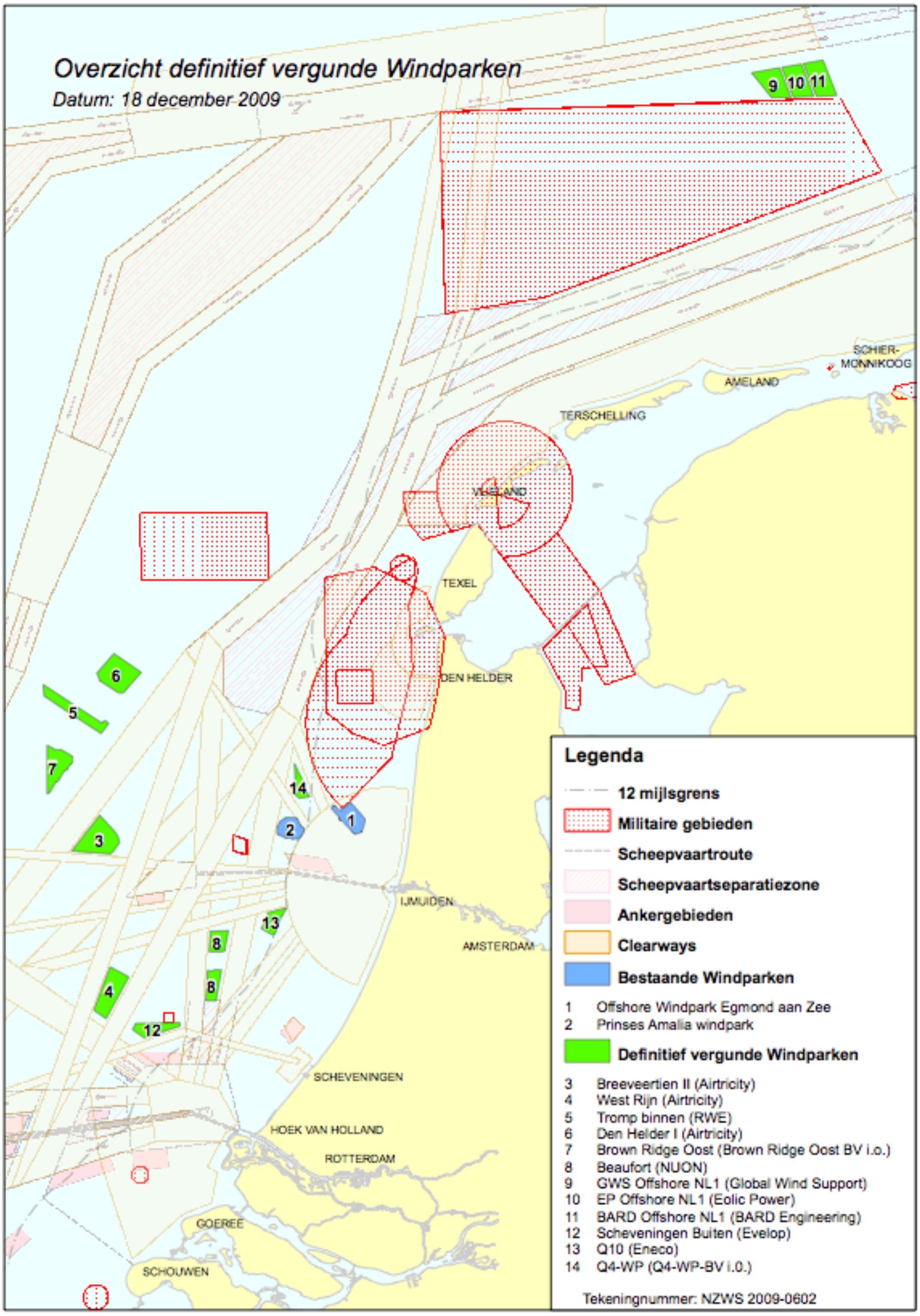


Figure 1.4.1 *Overview of wind farms in operation (blue) and wind farms with a construction permit (green) in the Dutch part of the North Sea (source www.noordzeeloket.nl). Also indicated are main shipping lanes (the different bands) and military zones (red dotted areas).*

2 Materials and methods

In this chapter first the species of interest are addressed (§ 2.1). Thereafter we describe the steps taken to construct population models for selected bird species. With these models historic and actual population trends are described (0-model in § 1.3). For this purpose data on life history traits and population variables from literature were gathered. In the next paragraph (§ 2.3) we describe how we calculated the estimate for the number of victims due to collision and how we dealt with birds involved in disturbance and barrier effects. In § 2.4 we present the way we assessed the cumulative effects of multiple wind farms; applying various levels of mortality resulting from collisions, disturbance and barrier effects.

2.1 Species of interest

To be able to evaluate whether significant cumulative effects on birds will result from the presence of wind farms, we need to gather information on those species of birds that are relevant to the ecosystem of the North Sea. These species include marine birds as well as non-marine migrating birds. Marine birds are those bird species that are entirely or partially reliant upon the sea. They include local breeding birds foraging at sea, and migrating seabirds. Non-marine migrating birds include all other species flying over the study area mainly during the migration periods in spring and autumn, towards and from their breeding and wintering grounds. For the purpose of this study, all birds passing the study area in the North Sea and its immediate surroundings are included.

However, because some species groups or species have a higher ecological relevance than others, based on for instance abundance in the area and in respect to population size, the effect study, similar to the baseline study, will focus on the species listed in table 2.1. The main argument for the selection of species and species groups is that, based on the monitoring both during the baseline as well as during the effect study, those species are more or less abundant in the area of the wind farm during at least part of the year. Some species have been added to the list as they are considered as vulnerable (e.g. because of a strong negative trend) and relevant in relation to international conservation policy (e.g. Nature 2000), and, despite being rare or rarely occurring in the studies areas, potentially can be affected by cumulative impacts offshore wind farms. Flight patterns of the various species or species groups in relation to the wind farm will be determined visually.

2.2 Data for parameterization of population models

Data on life history traits and population variables were collected for use in the population models. The primary means of acquiring data was through a literature search. This search included both an online literature search as well as a trawl through specific publications in the fields of seabirds and general avian ecology and population studies, such as 'Atlantic Seabirds' and 'Bird Study'. In addition to books and journals, other sources such as websites, online databases, reports and theses were used. Furthermore experts[1] on specific species were contacted. These specialists included authors of referenced publications or persons who were known to have knowledge or experience of the specific species or populations in question.

Table 2.2.1 Selection of the species groups of birds for which cumulative effects will be studied thoroughly based on a population models of selected flag species. Assessment of cumulative effects in other species (Table 2.2.2) are based on the outcomes of these selected species (groups).

species or group	flag species	*(Dutch name)*
local and migrating marine birds:		
cormorant	cormorant	*(aalscholver)*
divers	red-throated diver	*(roodkeelduiker)*
alcids –guillemot, razorbill, puffin	guillemot	*(zeekoet)*
gannet	gannet	*(jan van gent)*
sea ducks – scoter & eider	common scoter	*(zwarte zee-eend)*
ducks	shellduck	(bergeend)
terns	sandwich tern	*(grote stern)*
large gulls	lesser black-backed gull	*(kleine mantelmeeuw)*
small gulls	little gull	*(dwergmeeuw)*
skuas	greater skua	*(grote jager)*
migrating birds:		
swans	Bewick's swans	*(kleine zwaan)*
geese	brent goose	*(rotgans)*
waders	knot	(kanoet)
thrushes	redwing	*(koperwiek)*
starling	starling	*(spreeuw)*

The data search focused on seabird species found in and around the North Sea and on studies of the populations from North Sea countries. The key species for which data were sought are listed in Table 2.2.1. Assessment of cumulative effects in other species studied in OWEZ (Table 2.2.2) are based on the outcomes of these selected species (groups). Although the data search was focused on the North Sea populations, data from related seabird species and from other geographical areas were also collected to permit the validation of data for the focus populations through comparisons between similar species and across different geographical areas (Table 2.2.3).

[1],[3] R. Barrett, R.-J. Buijs, J. Calladine, C.J. Camphuysen, S. Garthe, O. Hüppop, M.F. Leopold, T. Reiertsen, A. Spaans, E. Stienen and A. Webb

Table 2.2.2 Overview of the species and groups of birds that were studied in the field studies in the OWEZ wind farm.

species or group	*Dutch name*
local and migrating marine birds:	
cormorant	*aalscholver*
grebes	*futen*
divers	*duikers*
alcids –guillemot, razorbill, puffin	*alkachtigen*
gannet	*jan van gent*
sea ducks – scoter & eider	*zee-eenden & eider*
swimming ducks	*zwemeenden*
shelduck	*bergeend*
terns	*sterns*
large gulls	*grote meeuwen*
skuas	*jagers*
small gulls	*kleine meeuwen*
fulmar & shearwaters	*noordse stormvogel & pijlstormvogels*
storm petrels	*stormvogeltjes*
migrating birds:	
swans	*zwanen*
geese	*ganzen*
diving ducks	*duikeenden*
dabbling ducks	*grondeleenden*
waders	*steltlopers*
swift	*gierzwaluw*
larks	*leeuwerikken*
thrushes	*lijsters*
crows	*kraaiachtigen*
starling	*spreeuw*
small songbirds	*zangvogels*

Life history traits and population parameters presented in the literature varied between studies. To ensure that data were comparable, the exact parameters that the data referred to were noted. The variables recorded and the life stage group (age groups and breeding/non-breeding status) to which these refer, are given in table 2.2.4. In addition to the variation in parameters stated, many studies referred to experimental or atypical situations, such as comparison studies between two colonies or years of extremely poor breeding success. Data from such studies were identified as reliable to use in the population models. Furthermore, the data from some studies were occasionally presented in more than one report or were adopted by other studies and were thus duplicated; again these data were identified as such to allow future removal of any duplication.

Table 2.2.3 Species of birds that formed the focus of the literature search for life history and population parameters for use in population models.

group	English name	scientific name	Dutch name
divers	red-throated diver	*Gavia stellata*	roodkeelduiker
tubenoses	northern fulmar	*Fulmarus glacialis*	noordse stormvogel
gannets	northern gannet	*Morus bassanus*	jan van gent
cormorants	great cormorant	*Phalacrocorax carbo*	aalscholver
	European shag	*P. aristotelis*	kuifaalscholver
geese & swans	dark-bellied brent goose	*Branta bernicla*	rotgans
	Bewick's swan	Cygnus bewickii	kleine zwaan
swimming ducks	Eurasian Shellduck	*Tadorna tadorna*	bergeend
sea ducks	common scoter	*Melanitta nigra*	zwarte zee-eend
	common eider	*Somateria mollissima*	eider
skuas	great skua	*Stercorarius skua*	grote jager
gulls	lesser black-backed gull	*Larus fuscus*	mantelmeeuw
	great black-backed gull	*L. marinus*	mantelmeeuw
	herring gull	*L. argentatus*	zilvermeeuw
	little gull	*L. minutus*	dwergmeeuw
	common gull	*L. canus*	stormmeeuw
	kittiwake	*Rissa tridactyla*	drieteenmeeuw
terns	common tern	*Sterna hirundo*	visdief
	sandwich tern	*S. sandvicensis*	grote stern
	little tern	*S. albifrons*	dwergstern
alcids	guillemot	*Uria aalge*	zeekoet
	razorbill	*Alca torda*	alk
	puffin	*Fratercula arctica*	papegaaiduiker

In the population models used survival and reproduction are the most essential life history traits (chapter 3). In the search for data not all could be used in a strait forward way. If necessary, data were transformed into the right unity (e.g. survival = 1 – mortality). Data on reproduction were brought back to the number of fledglings/pair/year including failed nests (Figure 2.1.2).

Table 2.2.4 Life history traits and population parameters that were collected for use in the population models.

life stage group	parameter variant
survival	age
	life span
	maximum age
	first year survival
	juvenile survival
	adult survival
	adult mortality
	annual survival
	annual mortality
reproduction	breeding success
	annual reproductive success
	chicks per nest
	daily nest survival
	hatching probability
	overall success
	age at first breeding
	generation length
	non-breeding frequency
	productivity
	percentage young
	chick survival
	young per pair
	young per nest
	young per pair paying
	number of broods
	clutch size
	brood size
population	dispersal
	distribution
	population size non-breeding
	population size breeding
	population trend non-breeding
	population trend breeding

2.3 Estimating the effect of wind turbines on birds

2.3.1 Estimating collision-related mortality

Definition and expected effects

The rate at which birds collide with wind turbines at sea is largely unknown. Although figures of collision rates exist for turbines on land (Winkelman 1992a,b, Krijgsveld *et al.* 2009), no figures exist for offshore turbines as technical devices lack to detect victims before they fall in the water (Dirksen 2009). Consequently, assessments of the level of collision mortality in offshore situations have been predicted using various models. In the absence of empirical data for collision rates in offshore situations, the 'Route 3' model outlined by Troost (2008) provides a suitable method to estimate the potential mortality rate through collisions with turbines. Route 3 stands for the SNH Band Model (SHN 2010). In this model it is assumed that the ratio of birds colliding with the turbines, specifically with the rotating blades and associated turbulence, and the rate of mortality is 1:1. The 'Route 3' model or SNH 'Band' model (Troost 2008) can be shown as;

$$c = b * h * a\_macro * r * e * a\_micro * p$$

where:

c =	collision (and thus mortality) rate
b =	number of bird crossings per time (usually one year)
h =	fraction of birds at rotor height
a_macro =	rate of avoidance of the entire wind farm
r =	ratio of rotor area to side area of entire farm
e =	number of turbines per crossing
a_micro =	rate of avoidance of individual turbines
p =	probability of collision when travelling through rotor-sweep area

The probability of collision whilst travelling through the area swept by the rotor area can be calculated using the SNH/Band model.

The SNH Band model

The SNH Band model (also known as the Band model or the SNH Collision Risk Model) provides a means of estimating the probability of collision for birds that fly through the rotor area of an operating wind turbine (SHN 2010). The model requires input variables related to the turbine, such as the rotor diameter, rotor width and rotation speed as well as information about the bird species of concern, such as length, wingspan and flight speed. The basis of the model is concerned only with calculating the number of birds passing through that collide with the rotors. The model is therefore frequently used in conjunction with other models or estimates, such as the number of birds passing the area, the proportion of those flying at rotor height and the level of avoidance. In the case of avoidance, this factor has been commonly applied in estimates employing the SNH Band model, in which instances it has a large bearing on the estimated number of collisions (Chamberlain *et al.* 2006). Chamberlain *et al.* (2006) highlight the sensitivity of the SNH Band model to changes in the various input parameters. Table 2.3.1 taken from Chamberlain *et al.* (2006), illustrates the sensitivity of estimates of collision risk as derived by the SNH Band models and incorporating avoidance. With the exception of avoidance rate, a 10% change to any of the input variables in turn altered the estimated risk of collision by between 1.48 and 10.20% (table 2.3.1). For avoidance rate the effect was much greater, with an increased collision risk of over 2500% for a 10% reduction in avoidance.

Collision risk model parameter selection

In the case of the selection of parameters for use in the collision risk model, figures on measurements of different bird species were taken from literature. Figures for the level of avoidance were taken from measured values at OWEZ (table 14.1 in Krijgsveld *et al.* (2011)). When figures were not available for the species in question, those for the species group were used (SHN 2010).

Table 2.3.1 Sensitivity of SNH Band model incorporating avoidance for estimates of collision risk for Bewick's swan at a site in England. Effects of 10% variation in input parameters on predicted mortality rates of Bewick's Swans at Little Cheyne Court (Percival 2004) Taken from Chamberlain et al. (2006). Additional remark: The Pitch angle of 30.0° used in the table below is unrealistic in the OWEZ case. In this report for OWEZ wind turbines a pitch angle of 0.5° has been used in the calculations.

Input variable	Baseline′	Baseline ± 10%	Collision risk	Revised collisions	% increase
Max. chord (m)	5.00	5.50	0.153	0.063	5.62
Pitch angle (°)	30.00	33.00	0.150	0.062	3.55
Bird length (m)	1.21	1.33	0.151	0.063	4.24
Wingspan (m)	1.96	2.16	0.147	0.061	1.48
Bird speed (m/s)	20.00	18.00	0.158	0.065	9.07
Rotor diameter (m)	92.00	82.80	0.150	0.062	3.55
Rotation speed (/s)	3.00	2.70	0.158	0.065	9.07
Bird count	109.00	120.00	0.145	0.066	10.20
Avoidance rate	0.9962	0.897	0.145	1.628	2613.19

Variables were changed by 10% (increased or decreased) so that mortality rates increased. The original collision risk was 0.145 and the original number of predicted collisions was 0.06 (Table 1).

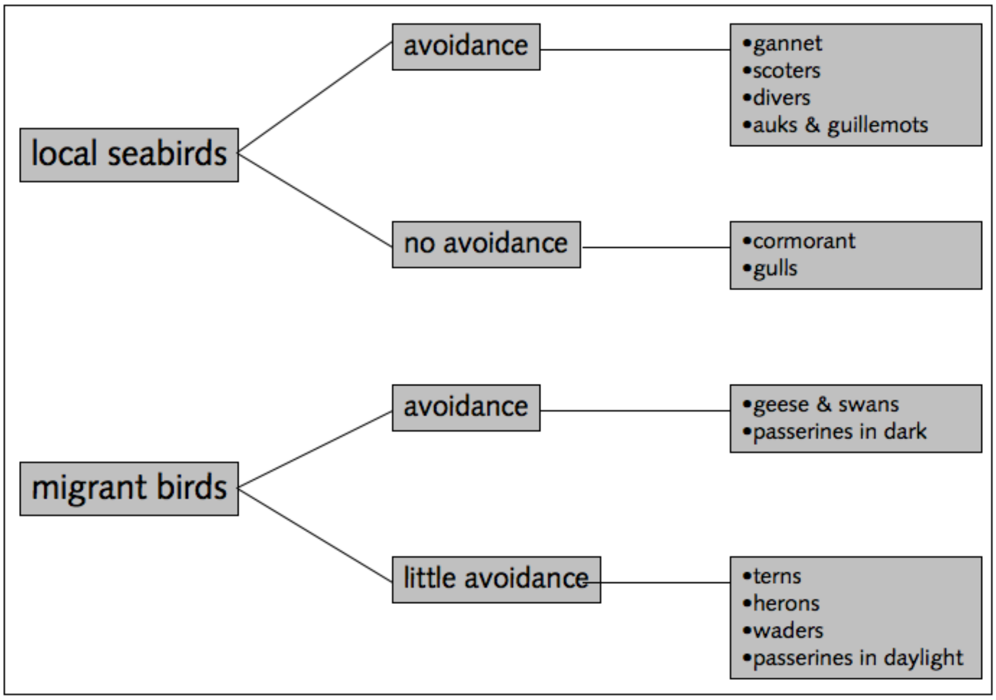


Figure 2.3.1 Schematic overview of species that did or did not avoid the wind farm, separated into (mostly local) seabird species and migrating land birds (reproduced from Krijgsveld et al. 2011, figure 15.1).

Collision related mortality estimated through models
The total number of collisions for 14 species and species groups were calculated for OWEZ using figures from studies at the wind farm itself. Data for each of the 14 species and species groups were taken from available literature or from field studies at the wind farm at Egmond aan Zee (table 2.3.2). Figures for bird length and wingspan were taken from Snow & Perrins (1997a, 1997b) and those for flight speed from Alerstam *et al.* (2007).

Figures for flux, that is the number of birds flying through the wind farm, have been calculated from fluxes measured at OWEZ by radar. These figures are for birds flying between 25 m and 150 m altitude. For birds flying within this height band it was assumed that birds were evenly distributed. The rotors at OWEZ are between 25 m and 115 m, which is 72% of the height in which flux is measured. Therefore, a correction factor of 0.72 was used to calculate the number of birds at rotor height. Figures for both macro-avoidance, birds that avoid the entire wind farm, and micro-avoidance, birds within the wind farm that avoid individual rotors, were taken from figures calculated at OWEZ (table 14.1 in Krijgsveld *et al.* (2011)). Basic data for Band model calculations of victims as used in this report are presented in appendices 6, 7 and 8.

Effects of collision-related mortality of bird populations
The effects of the calculated mortality rates of modelled scenarios at the population level can be predicted by applying this additional level of mortality to the species population models. By applying the additional mortality from two or more wind farms, the cumulative effect of collisions can be estimated. Because of the direct consequence of a bird being killed through collision with a turbine, it would be expected that the effect of collisions at the population level will be greater than those from the increased mortality due to causes such as habitat loss or increased energetic costs from disturbance and/or barrier effects. Indeed, current studies suggest that the consequences from disturbance and barrier effects are less obvious than for collisions (Drewitt & Langston 2006; Fox *et al.* 2006).

2.3.2 Disturbance effects

Definition and expected effects
Although the presence of turbines will reduce the amount of habitat that is physically available, the amount lost in this way is likely to be negligible, particularly in relation to the area covered by the wind farm (Fox *et al.* 2006). More importantly will be the loss of the habitat in the vicinity of the turbines through processes such as disturbance, most likely due to the presence of the moving rotors. In addition to any disturbance caused by the actual presence of turbines, boat or other traffic associated with the wind farm may also cause disturbance to birds within the area (Fox *et al.* 2006). The surface of OWEZ amounts 21.5 km^2 within the borders of the farm, which is 0.00375% of the total surface of the Dutch part of the North Sea (being 572,000 km^2 based on

www.compendiumvoordeleefomgeving.nl) and 0.04491% of the total surface of the Dutch coastal zone less than 20 m depth NAP.

The extent to which wind farms disturb different species at sea was assessed by comparing the changes in distributions of birds around an existing wind farm (Leopold *et al.* 2010). The change in numbers and behaviour of specific species around a constructed wind farm served as a reference for which the level of disturbance can be assessed. The effect of disturbance may become visible in a lowered density (with ultimately the complete absence of birds).

The consequent effect on mortality cannot easily be measured at the population level. In the case of seabirds with in many species increasing populations, indicating that carrying capacity has not been reached yet, only displacement could be at hand without any effect on mortality.

Information from OWEZ
An important finding of the field studies (Leopold *et al.* 2010) was that in general a low abundance of local sea birds occurred in the wider area where OWEZ is located. This low abundance was related to the location of the wind farm rather than the presence of the wind farm itself: near-shore species remained closer to the coast, while the more offshore species were low in abundance and showed no difference in abundance in the wind farm area versus further away from it. Leopold *et al.* (2010) found indications of disturbance in *alcids* in some surveys, but the numbers were too low to reach statistical significance and to quantify the effect of disturbance.

The research on flight paths and bird fluxes in and around OWEZ showed that offshore seabird species like gannets and auks had the highest avoidance levels (Krijgsveld *et al.* 2011, see figure 2.3.1). This indicates that these species avoid the OWEZ wind farm, which in case of gannets translates directly into disturbance to foraging birds as this species forage in flight. Auks however can also enter the OWEZ wind farm by swimming, and guillemots and razorbills regularly forage inside OWEZ boundaries (Leopold *et al.* 2010). However, as numbers of birds like gannets and auks in the area were low due to reasons other than the presence of the wind farm, the numbers of birds that were disturbed were probably relatively low. This however prevents us to be able to extrapolate effects and quantify the impact on populations in a realistic way. For those species potential disturbance effects are to be expected, the possible maximum number of victims are calculated with a so-called zero growth model, which indicates at what level of impacts the population would start to decline, see further the species accounts of gannet, great skua, guillemot and razorbill and results in the appendix 3. On the other side, species groups gulls and cormorants entered the wind farm without any major macro avoidance, with for some species like little gull and kittiwake even potential higher densities inside the wind farm, but this latter can be coincidental (Krijgsveld *et al.* 2011). For these species we assume no disturbance/displacement effects and are therefore also not further investigated in relation to cumulative effects of new multiple offshore wind farms.

2.3.3 Barrier effects; disturbance of flight paths

Definition and expected effects

Barrier effects can also be better termed as 'disturbance to flight paths' ultimately resulting in a barrier (Fox *et al.* 2006). In this respect the wind farm acts as a barrier to flying birds, which consequently results in altered, typically longer, flight paths during both local movements and during migration. The consequence of increases in flight durations and distances are likely to be an effective reduction in habitat (birds searching for food in flight) and increased energy expenditure (foraging birds and migrating birds) due to larger distances of flight (Masden *et al.* 2009, 2010).

Besides influencing the flight paths of local birds, a wind farm may also act as a barrier to birds during migration. Birds may, therefore, have to change their flight paths during migration, thus increasing their total journey length. For wind farms situated in the Dutch sector of the North Sea the most likely populations affected will be those that regularly migrate between the Netherlands and Britain and Ireland, as well as those that migrate offshore along the Dutch coast. These species are likely to include swans, geese, ducks, waders, gulls, terns and some passerines.

Because of the higher flight altitude of migrant birds compared to foraging birds (Alerstam 1990), wind farms are less likely to act as a barrier to migrants than to local birds. In addition, migrant birds are known to deviate from a straight-line route, usually as a consequence of wind-related drift and navigational corrections (Alerstam 1990), and as such distances during migration may typically be 0,75-1.6% longer than the straight-line route (Desholm 2003). The additional distance taken in flying around a wind farm is likely to be small compared to the total distance during migration.

The presence of wind turbines at sea may have the effect of reducing the availability of this habitat to birds both directly and indirectly. In addition to habitat loss through disturbance effects birds can be excluded from areas of habitat through barrier effects or more specifically by separating two or more ecologically linked areas, such as nesting and foraging areas. The amount of habitat affected by such a barrier will be dependent on the location and size of the wind farm. In turn, the effect on a population of birds will depend on the number of individuals affected as well as the value of the habitat lost.

Information from OWEZ

Species specific flight paths and fluxes measured in the field indicate that for species specific avoidance behaviour occurs. Other species however do enter the OWEZ wind farm with none or hardly any avoidance behaviour (Krijgsveld *et al.* 2011). Along the coast of the Netherlands at the same altitude as OWEZ along the coast cormorants, herring gulls, lesser black-backed gulls, common gulls, and common tern occur as breeding seabirds. The foraging ranges of breeding common terns and common gulls are too short to reach OWEZ or other new offshore wind farms, so barrier effects are not to be expected in this species in the Dutch situation which might lead to ultimate impacts by increased mortality. Cormorant, herring gull and lesser black-backed gull did not show main avoidance behaviour in relation to OWEZ. These species however suffer the highest collision risks, with probably much larger impacts on the population level compared to the potential effects of increased energy expenditures due to barrier effects (Masden *et al.* 2009, 2010). This also holds for many migrant species, and for further discussions and findings on barrier effects in migrant birds, including swans and geese we refer to Krijgsveld *et al.* (2011). Based on the specific situation of species composition and expected flight behaviour of local seabirds (both breeding as non-breeding seabirds) as observed around OWEZ, and expected around new offshore wind farms in the Dutch part of the North Sea, the impacts of barrier effects are expected to be marginal in comparison to the calculated number of victims due to collisions, for the two scenarios evaluated in this report. This is however under the assumption that the effects of 10 individual offshore wind farm in either scenario are additive. See section 2.5 for a further discussion on the limitations and problems of extrapolating the findings of OWEZ to multiple offshore wind farm scenarios. But with the lack of specific information about future offshore wind farm developments, and the expected impacts of barrier effects could not be investigated (with the remark that these are expected to be small compared to the much larger impacts of collision).

2.4 Assessing the cumulative effects at the population level

Cumulative Impact Assessment

Quantifying the level of the impact of multiple wind farms in the Dutch Part of the North Sea on bird populations requires several steps:

- how many birds from a specific population use the lower air layers (<200 m) in and/or around these offshore wind farms?
- what are the estimated effects (number of victims, increased mortality, reduced reproduction) on these species?
- What are the estimated effects of increased mortality and/or reduced reproduction at the population level?

The mortality caused by collisions, disturbance and barrier effects will not be mutually exclusive. Individual birds that avoid the wind farm during migration will be subject to higher energetic costs but also to a lower collision risk. Similarly, birds may be attracted into the wind farm area due to increased food supply or opportunities to rest but this could potentially lead to increased collision risk as the birds are spending more time close to the turbines. In addition, birds leaving an area because of one wind farm may encounter a second one as a consequence.

For each type of effect, collisions with turbines, disturbance and barrier effects, both a most realistic effect scenario based on the outcomes of the field research at OWEZ will be modelled as well as a maximum effect scenario. In the latter case all birds affected will assume to be lost from the population. In the case of collisions this is a realistic assumption. In the case of disturbance and barrier effects, mortality of all birds affected is only expected in situations with a strictly limited carrying capacity in which the loss of an area of habitat would result in the death of all birds that use that area. In the situation of many increasing populations this is unlikely to be the case for most species, so maximum effect scenarios presented in this report are unrealistic. They give us a first indication about the limits of maximum impact.

2.4.1 Estimating effects by means of population models

The construction of population models will enable assessment of the impact of additional mortality on species. First the additional mortality is estimated as described above. Second this estimate is fed into the population models. In this report the effect of additional, wind farm-related, mortality on a population is simulated for two multiple offshore wind farm scenarios (§ 1.4). In order to investigate the effects on species specific populations the following approaches have been followed:

- Estimating the response of a population to a certain amount of victims based on calculations using parameters on avoidance behaviour and fluxes of birds as determined in the field at OWEZ (Krijgsveld *et al.* (2011). This is called the effect-model (Figure 1.3.1).
- Estimating the amount of additional mortality to reach zero growth. This estimate is an indication for the level of the maximum effect; maximum in the sense that a larger additional mortality will lead to a decreasing population. This is called the 0-growth-model (§ 1.3).
- For a few species offshore wind farms might be a threat because of (significant) disturbance from feeding areas (habitat loss; guillemots, razorbill, gannet, greater skua). Data from OWEZ were not able to support this hypothesis. Therefore, the 0-growth-model is used to get an idea about maximum (acceptable) levels for these species. This is also done for species with low, but variable fluxes during migration (Bewick's swan and brent goose) because of presumed barrier effects.

The effects at the population level will depend on a number of factors, such as the age and sex of the birds affected, their breeding status and the time of year. In this approach we have assumed that all estimated victims are adult female birds. This implies a worst case scenario as in reality adult males, and sub-adult and juvenile birds of both sexes will also be victim. In long living species (such as seabirds) additional mortality of adults has a larger impact on population levels compared to additional mortality of juvenile and/or sub adult birds (Newton 1998), also illustrated in the sensitivity analysis of the Lesser Black-backed Gull model in appendix 2.

For species breeding at the Dutch coast, it is realistic that reproduction will be lowered due to the loss of a brood in case a victim falls during the breeding season. During breeding in those species concerned both partners are necessary to raise young successfully. Therefore, as a worst case scenario, by assuming that all victims are females a failure of broods is included (and more).

2.4.2 The Potential Biological Removal approach

If we approach this problem from the viewpoint of a bird population using the North Sea we could turn around the question and try to answer the question: At what impact (number of victims, increased mortality) is the effect on a bird population unacceptable large? This approach can be compared with the outcomes of the effects calculated via the species-specific population models (effect-model and zero-growth model) (Figure 1.3.1).

To answer this we use the approach followed by Lebreton (2005), Niel & Lebreton (2005) and Dillingham & Fletcher (2008). In Dillingham & Fletcher (2008) the number of additional casualties (increased mortality) that can be sustained each year by a population is expressed as the Potential Biological Removal (PBR):

$$PBR = 0.5 * R_{max} * N_{min} * rf$$

Where R_{max} is the maximum annual recruitment rate, N_{min} is a conservative estimate of population size and *rf* is a recovery factor between 0.1 and 1 (Dillingham & Fletcher 2008, Wade 1998, Niel & Lebreton 2005). Half of the R_{max} is the net recruitment rate at maximum net productivity level, with the 0.5 in the formula determined by Wade (1998). This method provides a conservative estimate of the PBR assuming a convex or logistic density dependent growth curve (Dillingham & Fletcher 2008).

R_{max} and maximum annual population growth rate (λ_{max}) are related through $R_{max} = \lambda_{max} - 1$. If sufficient demographic information is available matrix population models can be constructed to estimate λ_{max}. If sufficient data on population size are available R_{max} can be estimated from these data. Niel & Lebreton (2005) propose estimates of a theoretical maximum growth rate (λ_{max}) based on age at first reproduction (α) and adult survival (s) for bird species (see Niel & Lebreton 2005 for details).

N_{min} is a conservative estimate of the population size, suggested by Wade (1998) to be the lower bound of the 60% confidence interval, to be regarded as an important precautionary step to compensate for the uncertainty of the few data used in the PBR approach.

The factor *rf* is a management factor, *rf*= 0.1 provides a minimal increase in recovery time for a depleted population or near threatened population (IUCN criterion), to maintain a population size close to carrying capacity or to minimize the extinction risk for a population with a limited range. A value of *rf*=1.0 could be used to maintain a growing population at or above its maximum net production level, recommended to use for a population with a least concern status with a stable or increasing population trend (Dillingham & Fletcher 2008).

2.5 Limitations and problems of extrapolating findings of OWEZ

The baseline and effect studies aimed at monitoring the effects of the OWEZ wind farm are presented in Krijgsveld *et al.* (2011) and Leopold *et al.* (2010). The findings in relation to flight behaviour, avoidance behaviour and presence of birds in and around this wind farm are used in this current report to estimate the potential impacts on bird population levels in cases of multiple offshore wind farms as are planned for the Dutch part of the North Sea. With the extrapolation of the results from the effect studies of OWEZ we are confronted with a serious limitation that the findings are in principle only applicable to the specific situation of OWEZ. This is determined by the specific configuration of the OWEZ wind farm (number of turbines, their size, rotor diameters, spacing, etc.) as well as the specific location of OWEZ in relation to the sea habitat-related occurrence of the bird species and related behaviour of the birds.

Findings of OWEZ are location and configuration specific

The fluxes and densities of local seabirds as measured by the effect studies have proven to be extremely location-specific; especially based on the ship-based surveys that were conducted across a much larger area than OWEZ itself (Leopold *et al.* 2010). The measured densities (and related fluxes as studies in Krijgsveld *et al.* (2011) can be explained by habitat-related species specific distribution patterns at sea (offshore species versus more coastal species) and for species occurring in OWEZ that breed on the coast, the distances to the nearest colonies. OWEZ lies within the flight range of only a very limited number of species of breeding birds (lesser black-backed gull, herring gull and cormorant). The habitat features that determine the distribution patterns of foraging seabirds, both breeding on the coast as well as non-breeding birds, are: distance to the coast; water depth; salinity; turbidity; and presence and availability of food, the latter being of paramount importance.

Lack of field data on the effects of a situation of multiple offshore wind farms

In a situation under which individual wind farms are erected in close proximity of each other the avoidance of wind turbines by flying birds, for example, might increase. Such cases suggest that the effects of multiple wind farms are not simply additive but could also be multiplicative or non-linear. The effect studies on OWEZ do not yield data with which these effects can be assessed, although the Princess Amalia Wind Farm does lie within the vicinity of OWEZ. Despite this relative close proximity (12 km), Princess Amalia Wind Farm was far outside the range of the radar in the study of Krijgsveld *et al.* (2011) and in and around Princess Amalia Wind Farm no comparable study (including radar and/or visual observations on flight movements) has been carried out as in OWEZ. Therefore, no study is available that combines the observations and draws conclusions on the potential cumulative effects of these two offshore wind farms. With a lack of data on the effects of multiple offshore wind farms the approach has been adopted to assume additive effects and for the purpose of this report.

Extrapolating findings OWEZ to a wider near-shore situation

The first scenario that wind farms are developed in the near-shore zone along the Dutch coast can be done under the assumption that multiple offshore wind farms will be developed in more or less the same area that OWEZ is situated and with:

- a species composition and behaviour being very similar to OWEZ
- a comparable size and configuration of the multiple wind farms as OWEZ

- the effects of multiple wind farms are additive, due to sufficient distances between the wind farms

A scenario of ten wind farms will yield a tenfold increase in the number of related collision victims for the relevant species. For the near-shore situation no significant displacement effects have been found (Leopold *et al.* 2010). Both radar and visual observations have indicated that some seabird species clearly avoid the wind farm (e.g. gannet, alcids), while others, including the most numerous species present (e.g. gulls and cormorants), show almost no avoidance of the OWEZ wind farm. Although the extrapolation of the findings of OWEZ for a near-shore scenario are limited, we know that it is highly unrealistic that scenario 1 would be realised as many other activities prevent the construction of more offshore wind farms in the near-shore areas of the Dutch coast (see figure 1.4.1 for these activities, such as military areas, shipping lanes and mining).

Extrapolating findings OWEZ to an offshore situation
A major problem for the extrapolation of the findings of OWEZ for a scenario of multiple offshore wind farms further out at sea is that another seabird community, than that found at OWEZ, will be present in and around such wind farms (Leopold *et al.* 2010). As no comparable data to those gathered in the effect studies in and around OWEZ are available for these offshore areas we do not know how the fluxes of flying birds differ to the near-shore situation. In order to be able to calculate the potential impacts of an offshore scenario we have used a long-term database of aerial surveys available for the total Dutch North Sea in order to translate the findings of OWEZ to a situation involving another seabird community. One limitation that remains is that we do not know whether the proportions in numbers present of different species, as determined by aerial surveys, reflects the proportions of the fluxes of different species, and in turn the potential number of victims. However, the majority of species involved are those that forage in flight, such as gannet, skuas, gulls and terns. The number of collision victims can only be calculated under the assumption that the numerical proportion has a strong correlation with the amount of flight activity of the species. In chapter 4 calculations are carried out using the Band model assuming a similar total number of seabird victims per wind turbine as calculated for OWEZ and taking into account the species specific avoidance behaviour as determined by Krijgsveld *et al.* (2011). We are aware of the speculative nature of this exercise. Nevertheless, in light that the aerial database of the total Dutch North Sea represents the most most appropriate data currently available, we have chosen to make this first attempt and have applied the observations from OWEZ to other areas further offshore. In addition, the offshore scenario 2 is more likely to be developed than the near-shore scenario 1.

3 Population models

In this chapter we first give a detailed outline of the way models are build and the models are run (§ 3.1). Thereafter we present a review on floaters, a major element in the way we deal with populations (§ 3.2).

3.1 General approach

Basic population model
Methods to predict the impact of human-caused mortalities on bird populations are described extensively in literature (e.g. Newton 1998). To quantify the impact at population level knowledge of population dynamics is necessary.

In this study, the assessment of the effect of increased mortality on seabird populations has been carried out with Leslie matrix modelling. These are relative simple, robust models describing the change of a population through time based on rates of reproduction, survival, immigration and emigration (Caswell 2001). Alternative modelling approaches such as population dynamic modelling (bifurcation or stochastic modelling) have been considered, but would require more information on individual bird behaviour. In the Dutch situation relatively detailed information is available via two studies using GPS loggers but still only for one species (lesser black-backed gull) and even here the amount of information is insufficient. This is due to the need to model the birds distribution and behaviour at sea and relate this to the influence of offshore wind farms (for which many assumptions would be needed) as well as how the individual's behaviour relates to mortality and reproductive success in order to ultimately estimate effects on a population level.

Population modelling is generally done by the projection of vital population parameters over time (Perrins *et al.* 1991; Akçakaya *et al.* 1999). If an accurate historical record of population size is known, a model describing the historical population size can be constructed and validated. Under the assumption that the same population parameters (and their relative importance) describe the population size in the future these models can be used to quantify the effect of changing vital rates on a population size. The size of a population increases by births and through immigration while deaths and emigration decrease the size of a population according to:

$$N_{t+1} = N_t + \text{births} - \text{deaths} + \text{immigration} - \text{emigration} \quad (1)$$

Where t = time step and N is population size.

For animal populations with an annual cycle of reproduction, such as birds, time step t represents one year. Population sizes are usually measured in the breeding season (number of breeding pairs) or at wintering or staging locations (total number of wintering/staging individuals in an area).

Equation 1 can be formulated as:

$$N_{t+1} = R * N_t + N_t$$
(2)

Where R is the net per capita rate of recruitment.

If the vital parameters of a bird population (births, deaths, immigration and emigration) and the size of a population at the start (N_0) are known, equation 2 can be projected over several time steps (years) to produce a population size over time. If R is >0 the population size will increase (exponential), if R is <0 the population size will decrease and if R is 0 the population size will be stable.

The growth of animal populations is limited (e.g. Newton 1998; Perrins *et al.* 1991). Feedback mechanism usually occur where the rate of reproduction or deaths are related to the size of a population. A very common feedback mechanism is the decrease of reproduction rate with increasing population size. If a population grows the pressure on resources (e.g. food to feed chicks or space to breed) becomes larger thereby increasing the competition between individuals of that (and sometimes other) population(s). At a certain point the consequences of this increased competition affect the individual such as either through reduced productivity or increased mortality. If at a large enough scale, these consequences on individuals can be seen at the population level and is termed as density dependence. How this mechanism affects different species determines the different type of feedback mechanisms seen in different populations.

Model assumptions, limitations and evaluation
The modelling of populations depends on the available information on vital rates of each population (Perrins *et al.* 1991). For most species considered in this study there is a fairly good knowledge of population size since 1960 (either breeding pairs or wintering population), average annual adult survival, average annual sub-adult survival (survival until breeding age), age of first breeding and average yearly reproductive values. For swans and geese detailed information on year-to-year fluctuations in reproduction is available.

For each species a model has been built with three stages (first year, sub adult and adult) where average yearly vital rates are used to predict the historical population size through time (figure 3.1.1). Available stochastic variation on demographic rates was available and was used to examine the effect of fluctuation on the results.

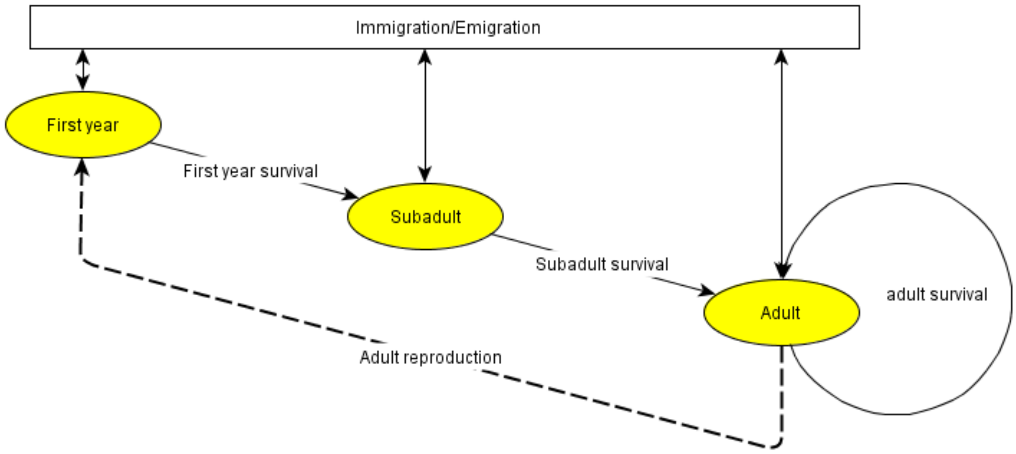


Figure 3.1.1 Basic model layout.

Models describing the population size through time are run 100 times allowing stochastic fluctuation. For the species considered here (long living species with relatively low annual reproduction rates), fluctuations in adult and sub adult survival are likely to be small, while fluctuation in reproductive rates are likely to be relatively large (Lebreton & Clobert 1991).

Carrying capacity (K) is used to calculate the density dependence in the model, where adult reproduction is related tot population density and where density dependence affects the reproductive rates (figure 3.1.2). The value of K is set at the maximum of the measured population size.

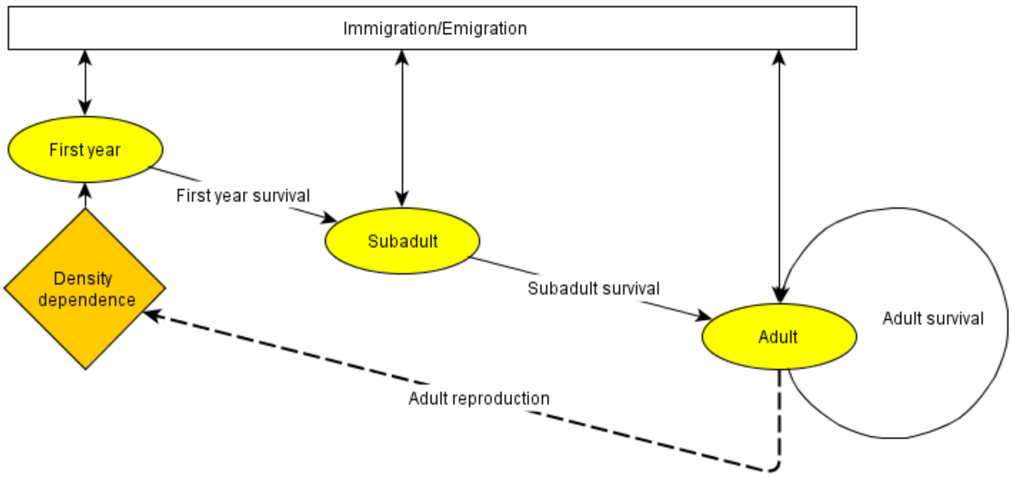


Figure 3.1.2 Model layout including density dependence.

Some species have a population structure where a considerable group of the adults within the population does not participate in breeding during each annual cycle. These individuals are known as floaters. This group of individuals does function as a buffer in a population and can compensate for increased mortality among breeding adults (figure 3.1.2; for details see § 3.2 on floaters). This leads to a stable breeding population whereas the fluctuations are transposed tot the group of non-breeding adults (floaters).

Model results are validated on measured population size. For each model, graphs with median, 25 and 75 percentile per year are created (see chapter 5 and appendix 2 and 3).

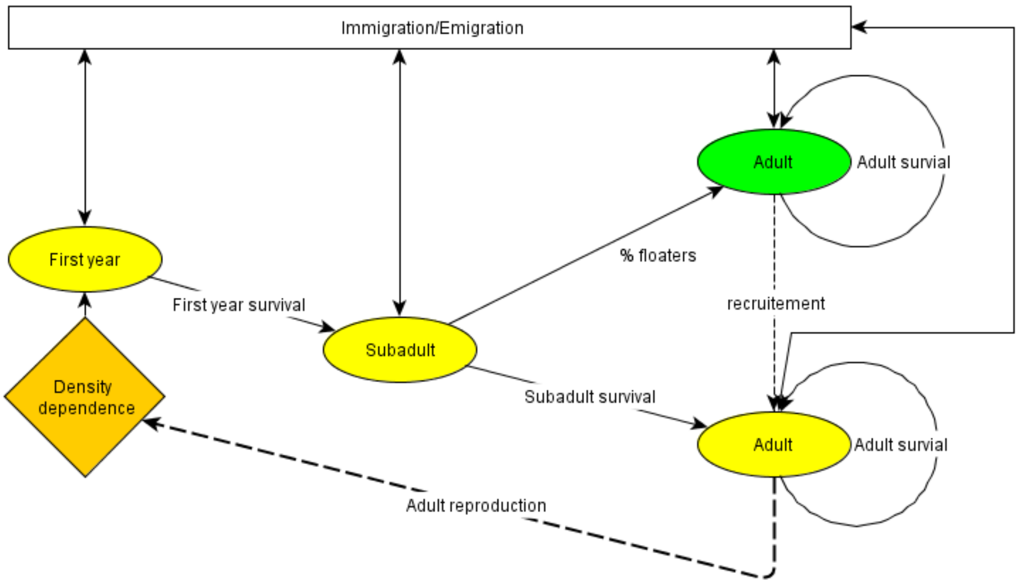


Figure 3.1.3 Model layout with density dependence and floaters.

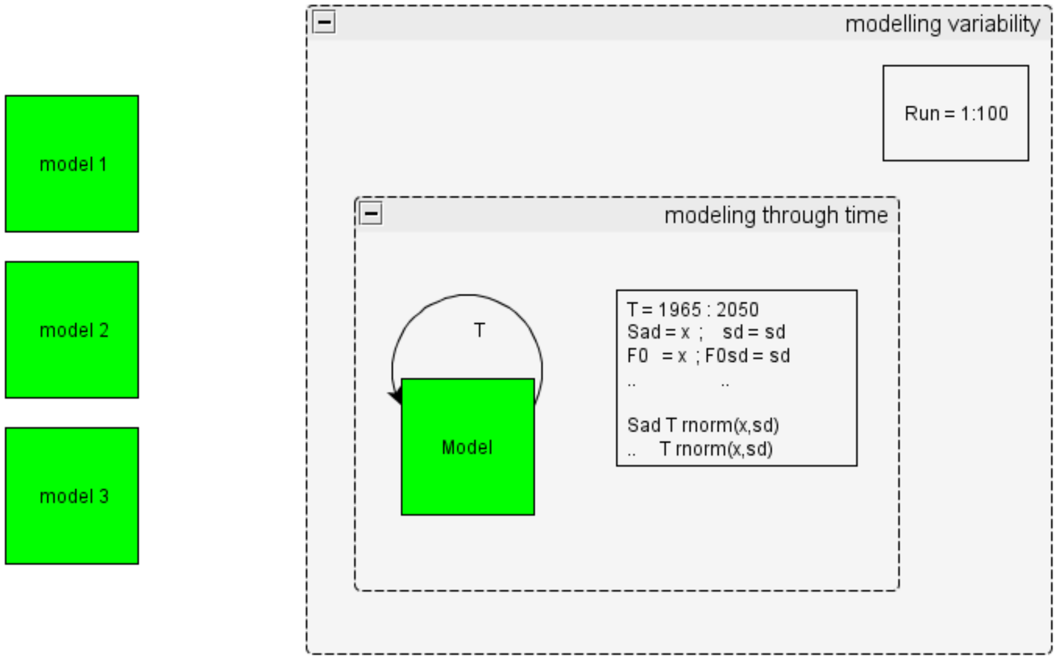


Figure 3.1.4 Framework modelling variability.

The effect of parameter variability is taken into account in the modelling (figure 3.1.4). Essentially the projection of the population in time is run in the 'inner' box where for each parameter stochasticity is introduced. For each parameter (in our case mortality and reproduction) where a set of different values was available, each single model run drew randomnly a parameter value from a normal distribution with mean and standard deviation based on literature sources (appendix 5, with main sources mentioned in the species accounts in chapter 5; means and standard deviations for every species presentend in table 5.1.2). Parameters are not correlated between years (so for every time step a new random value was taken). The 'inner' box is run 100 times (in the 'outer' box) to produce 100 realisations with stochasticity of the models. Output graphs are produced where 25/50/75 percentiles per year are drawn as model results (see chapter 5, and appendix 2 and 3), thus allowing insight in the effect of demographic stochasticity on the output of the models (flowchart of modelling process in figure 3.1.5). No environmental stochasticity is taken into account.

Scenarios

Scenarios with increased mortality can be evaluated with the models. Increased mortality is incorporated in the models by subtracting matrix M that contains the number of birds for each stage. Increased mortality is only applied in the scenarios after the year 2010. The number of victims in the model is expressed as females (or breeding pairs). The modelling predicts the number of breeding pairs (and or floaters, also pairs) after the breeding season and consequently the increased mortality is evaluated after the breeding season. Presently no provision in the modelling is made to predict the effect of decreased reproduction due to increased mortality in the breeding season. Presently increased mortality is only evaluated for the adult breeding stage.

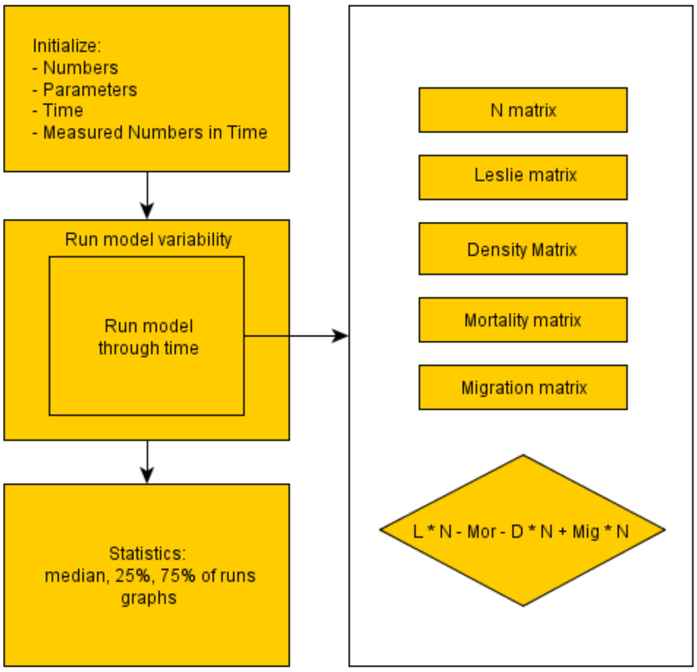


Figure 3.1.5 Flowchart of modelling process.

After prediction of population size, the floater surplus is calculated (with floaters – victims). If the floater surplus is > 0, the number of victims is taken from the floaters stage and added to the breeders stage, effectively the victims are breeding birds only and they are replaced with floaters when these are available. If the floater surplus is <=0, the number of floaters is set to 0 and the chance to become a floater is set to zero, this is done to prevent adult breeders to emerge in the floater stage after this stage is consumed by loss of breeders. If birds in the floater stage are all transformed into breeders, the model effectively becomes a model without a floater stage.

Population models per species

For all species, the male to female ratio is assumed 1:1, as in most bird species (Newton 1998). All modelling is done for the female part of the population only. If the sex ratio is 1:1 the models about breeding females alternatively can be read as pairs (female with a male). Reproduction is expressed as the number of chicks per pair, for both successful and failed breeding attempts, that survive to the reproductive age (fecundity rates). With a sex ratio of 1:1, the value fed into models is half of the value measured in the field (halve of the young fledged are female). Survival is expressed as the chance of survival to the next time step (t; here 1 year). Immigration is expressed as number of females at breeding age, entering the population. As the population models are structured in age classes (1^{st} yr, sub-adult, adult), reproduction and survival rates are age class specific. In many seabird species the sub-adult phase last two or more years. The difference in the survival rates between the sub-adult and the adult age classes is of particular importance, as is the age at first breeding.

The prediction of the size and changes in a population with population models is not without uncertainty. Depending on the quality of the input parameters (e.g. adult survival rate) differences between the predicted and the known populations can be expected. Since several parameters, each with some level of variation from the actual values are often used, this variation can accumulate, leading to even greater differences between the predicted and known population, see appendix 2 for a sensitivity analysis on the parameters reproduction and survival (with a distinction in immature and adult survival). Population models, however, provide a recognised method to investigate the influence of changes in mortality rates on the population size (Caswell 2001).

Parameter selection per species
Where possible parameter values from Dutch breeding populations were used. Where this was not possible, either as the species does not breed in the Netherlands or as appropriate figures were not available, data from populations in other countries were used; most notably from Belgium, Germany, Ireland, Norway and United Kingdom. For these species, it was assumed that the populations of these species breeding within the North Sea basin would have similar parameter values and be influenced by similar factors to those birds occurring in the Dutch North Sea region. Selected mean and standard deviation of parameters for every species are presentend in table 5.1.2. These are based on literature sources of which an overview of all consulted sources is given in appendix 5, with the main sources mentioned in the species accounts in chapter 5.

Validation of the species-specific models
To ensure that the population models produced were comparable to the known populations, they were validated against the observed population development in the past decades. Information on the status and historical changes of the relevant populations were gathered from the published literature. For species that breed in the Netherlands, data were collected on Dutch breeding populations. For species that only occur within the Dutch North Sea region as non-breeding birds, data were based, where known, on the relevant breeding populations. When uncertainty existed as to the origins of these individuals, data were based on breeding populations from around the North Sea. In addition, data were gathered on the status of each species within the Dutch North Sea region outside of the breeding season.

For the population modelling several basic assumptions are made (with exceptions).

- The average model parameters of mortality and reproduction are assumed to be constant through time, implying that no trends occur in the course of years. For most species limited information is available based on e.g. a fraction of colonies studied in some years, let alone that trend information is available.
- The effect of parameter variability is taken into account in the modelling by introducing stochasticity for each parameter. This was done by drawing randomnly a parameter value (in our case mortality and reproduction) from a normal distribution with mean and standard deviation (based on literature sources) for for each time step. This implies that the stochasticity of parameters was not correlated between years.

- Models are calibrated on breeding populations/winter populations on national scale.
- Survival parameters are relatively well known and stable (especially for long living animals). Reproduction is relatively unknown en can vary between years.
- Sensitivity analysis is a technique for systematically changing variables in a model to determine the effects of such changes. Here we use a simple change one factor at a time (OAT) approach, presented in appendix 2 on the model of the Lesser black-backed gull. Essentially the method applied in the species modelling were input parameters are drawn from a defined distribution (mean and sd) gives insight in the variability of the model outcomes based on the stochasticity of the combination of the parameters mortalitity and reproduction (graph 1 in appendix 2). In the following graphs 2 -4 in appendix 2 one can see that as expected in long-lived species the model outcome is most sensitive to changes in adult mortality (Sad, widest range in outcomes), then to immature mortality (S0) and least to reproduction (Fad).
- Survival can depend on density but here we choose to model the effect through reproduction only, i.e. the closer the population size is to the carrying capacity the lower the reproduction, until it reaches carrying capacity (as seen in many species, e.g. Newton 1998, Schreiber & Burger 2002).
- For breeding populations in the Netherlands immigration and emigration is assumed to be zero (0). Immigration and emigration parameters are only properly known for a few species. At first models are build under the assumption that emigration + immigration = 0. For two species it was not possible to reach a good fit on the observed trend in de past decades. While adding immigration (within the limits of observed values), a good fit was reached.
- In two cases sudden increases in population growth rate (sandwich tern in the Netherlands and gannets from Bass Rock) have been simulated with an adaptation of reproduction and immigration rate based on literature sources.
- Most populations contain floaters, being an extra stage in the model containing adult birds that do not reproduce. These birds function as a buffer in a population and can compensate for additional mortality among breeding adults. Birds in the floaters stage are augmented from the population through a slightly increased reproduction. Null-models are constructed with 0-10-30% floaters (Figure 1.3.1).
- In population modelling proper parameters that describe these populations need to be selected. Quality of these parameters varies between species. In general survival rates (both adult and sub adult) are relatively stable through time while reproduction can vary greatly over time (Newton 1998, Schreiber & Burger 2002).
- The number of victims is held constant through time, despite possible changes in relation to changing population size through time.

Technical description of the modelling
Numerical modelling with (2) is usually done with matrix algebra. Most commonly used is the Leslie matrix L, as this matrix offers suitable accuracy against reasonable computing efficiency. Equation 2 is rewritten as:

$$N_{t+1} = \lambda * N_t$$
(3)

Where λ = (R + 1). If we consider 3 stages (1=first year, 2=sub adult, and 3= adult) in matrix notation (3) is written as:

$$N_{t+1} = L * N_t$$
(4)

For three stages this leads to:

$$\begin{bmatrix} N_{k=1,t+1} \\ N_{k=2,t+1} \\ N_{k=3,t+1} \end{bmatrix} = L * \begin{bmatrix} N_{k=1,t} \\ N_{k=2,t} \\ N_{k=3,t} \end{bmatrix}$$
(5)

where L is the Leslie-matrix:

$$L = \begin{bmatrix} F_{k=1} & F_{k=2} & F_{k=3} \\ S_{k=1} & 0 & 0 \\ 0 & S_{k=2} & S_{k=3} \end{bmatrix}$$
(6)

F_k = fecundity (for stage *k*) and is the number of offspring per year and S_k is the yearly survival rate for stage *k*. If only adults reproduce $F_{k=1} = F_{k=2} = 0$.

Stages are defined here as age classes and therefore birds can only move in one direction through these stages (that is aging) or stay in the final class with only adult animals. With this definition birds also have to move between stages (with exception of the adult stage). This is in accordance with general assumptions concerning stage based population modelling (Akçakaya *et al.* 1999).

In this system density dependence is incorporated through fecundity (Jensen 1997, Brandon & Wade 2006):

$$F_t = f_0 + (f_{max} - f_0) * [1 - (N_{t-1}/K)^z]$$
(7)

Fecundity (F_t) is now affected by the density at t-1 and scaled by the minimum (f_0) and maximum fecundity possible (f_{max}) given the population parameters for survival and age at first breeding. Z is a shape parameter for the steepness of increase in the past decades. In this report Z is derived from fitting the observed population growth in the past decades.

Fecundity (F_t) is now affected by the density at t-1 and scaled by the minimum (f_0) and maximum (f_{max}) fecundity possible given the population parameters for survival and age at first breeding.

$$N_{t+1} = L_{Ft} * N_t$$
(8)

Floaters in a population model
In the Leslie matrix stages for non-breeding adults can be constructed (Akçakaya *et al.* 1999, Runge *et al.* 2006, Cooch *et al.* 2010)

Thus floaters are modelled with:

$$L = \begin{bmatrix} F_{k=1} & F_{k=2} & F_{k=b} & F_{k=f} \\ S_{k=1} & 0 & 0 & 0 \\ 0 & S_{k=2} & S_{k=b}bp & S_{k-f}fp \\ 0 & 0 & S_{k=b}(1-bp) & S_{k=f}(1-fp) \end{bmatrix}$$

(9)

Where $F_{k=1}$, $F_{k=2}$ and $F_{k=f}$ are 0 and breeders proportion (bp) and floater proportion (fp) are complementary. *k=b* is breeder stage and *k=f* is floater stage. This allows exchange of adults between de breeder and floater stage and *vice versa*.

Immigration and emigration
With IMM and EM a matrix with yearly fraction of immigration or emigration for each specific stage is constructed. In the Leslie-matrix, stages for non-breeding adults are constructed (Cooch *et al.* 2010). For all species in this report IMM + EM is held at 0, with the exception of the sandwich tern and gannet. For this species IMM + EM > 0 (see § 5).

Scenarios
Assessment of effects of increasing mortality is done by incorporation of a 'victims' matrix in the modelling process:

$$N_{t+1} = L * N_t - M$$

M has the same size as the N matrix and is filled with the number of assumed victims per stage class. These victims are subtracted in each time step. In the scenarios the victim matrix is incorporated after year 2010. In this study a worst case scenario has followed that all estimated victims are only taken from the adult female breeding population.

Results

Results (in figures in chapter 5 and appendix 3) are expressed as the number of breeding pairs. The main source of information about past en recent population size was the number of breeding pairs in relevant areas of origin. These 'breeding' data were used to validate the zero-model. Only for brent geese and Bewick's swan results are expressed as number of birds, since reliable source on population size are about number of birds (in winter). These non-breeding data were used to validate the zero-model.

3.2 Non-breeding adults or 'floaters' in bird populations

Introduction

Bird populations are structured in age classes, with the main categories juveniles, sub adults, and adults (Figure 3.2.1, 3.2.2). Depending on species, birds are mature (adult) and able to breed after one or more years up to eight years (Newton 1998). Not all individuals enter the breeding population in the first year they are sexually mature (adult). Among most species there is a difference between the average age of sexual maturity and the average age of first breeding (Becker & Bradley 2007). The birds that enter the breeding population not in their first year of sexual maturity are called non-breeding adults. Also adult birds which later in their life loose their partner or being not in good shape become a non-breeding adult for one or more years. In later years these birds can return to breeding again. Entering the breeding population is a delicate process, known as recruitment (Becker & Bradley 2007). This process depends on several factors, especially the quality of bird itself, environmental factors ahead of the breeding season and the density of the breeding population (Penteriani *et al.* 2007, Becker & Bradley 2007). These non-breeding adults are a surplus in the population that can buffer incidences and catastrophes in the breeding population (Newton 1998, Grimm *et al.* 2005). So the breeding population might stay stable whereas the non-breeding component meets (heavy) fluctuations. Even there are individuals within a population that will never breed, and so never contribute to the next generation. In some long-lived species adults do breed only every two years; i.c. some large raptors and albatrosses. The birds that skip a breeding season are non-breeding adults. All those non-breeding adults are also known as floaters (Newton 1998).

How many floaters are there?

The proportion of non-breeding adults or floaters varies between species, ranging from some percentages in smaller species to more than 50% in larger species (Table 3.2.1). There is a positive correlation between body mass and the proportion of floaters (Figure 3.2.3). In general larger species (heavy weighted) live longer compared to smaller species (lean weighted) (Newton 1998). In order to get a feeling for the variance in life span at one hand and the adult survival, compare the average life span of Blue Tit (2,1 years, adult survival 30-50%, 11 gr) with that of the Imperial Eagle (>12,7 years, adult survival >95%, 3.000 gr) or Wandering Albatross (>14 years, adult survival 94%, 9.000 gr) (Dhondt 1989, Watson 1997, Weimerskirch 1992, Schreiber & Burger 2002). Longer living species invest more energy in their own life compared to their offspring whereas short living species invest more energy in their offspring (Newton 1998). In general, seabirds are long-lived species, with small clutches and a long fledging period (Weimerskirch 2002).

There is a negative relation between the average number of eggs per year and adult survival among species (Perrins 1991). This is also the reason why in bad years larger species have a larger tendency to abandon their nest with young than in small species (Newton 1998). Therefore long living species can choose each breeding season between the status of breeding or non-breeding. This choice is influenced by genetic quality, body condition en feeding conditions just before breeding (Newton 1998). In fact, short living species do not have any choice. If they want to contribute to the next generation, they will have to breed in the one or two years before leaving life as a breeding adult. In long-lived species survival rates and reproductive success increase with a delayed first time breeding, as is shown for the Wandering Albatross (Crespin *et al.* 2006), and some *Alcidae* (Netlleship & Birkhead 1988). Investments in ones own life pay back!

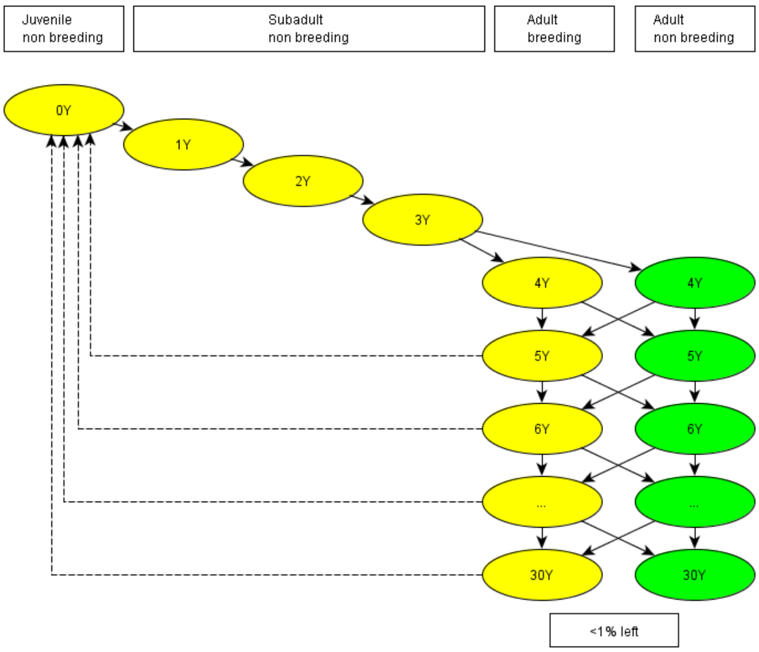


Figure 3.2.1 *Basic population structure of a long living bird species, sexual mature after four winters, with a certain proportion of non-breeding adults. Arrows denote possible transitions between stages in the course of a bird life. Dashed lines = reproduction. After 30 years <1% of the initial number of fledged juveniles is left.*

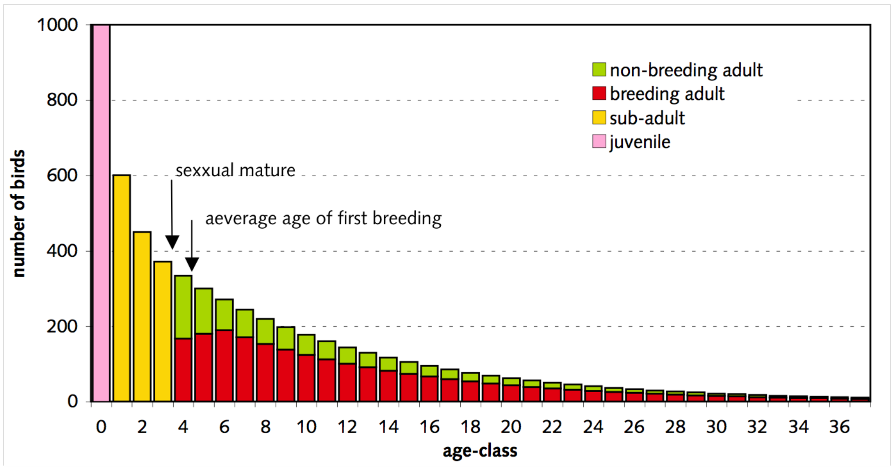


Figure 3.2.2 Age-class distribution of a long-lived bird species, with 32% non-breeding adults, sexual mature after 4 winters, average age of first breeding after 4 winters. After 37 years < 1% of the initial number of birds fledged is left.

Most seabirds are long living species (Nettleship & Birkhead 1988). All data found so far (Table 3.2.1) fall within the variability in the proportion of non-breeding adults (Figure 3.2.3). Among non-seabirds the full range of weights is included in the sample. For this reason the correlation between body mass and % floaters is much stronger for this group compared to seabird species.

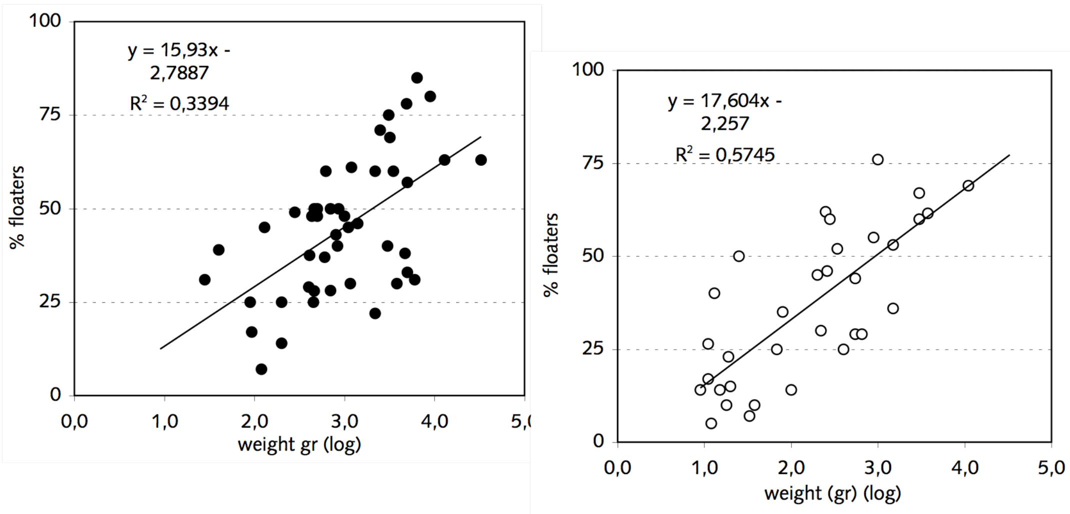


Figure 3.2.3 Relation between body mass (gr, log) and proportion of floaters (%) among bird species (data in Table 3.2.1). Left seabirds; based on log-log regression R^2=0,301, r=0,548, df=46, p<0,001, right non-sea-birds R^2=0,518, r=0,720, df=33, p<0,001. Data in Table 3.2.1.

Variation in the proportion of floaters

Despite the positive correlation there is much variation between species in the proportion of floaters. Factors, determining the proportion, might vary between species, between years and between locations. For Common Guillemots accidents like wreck or oil spills causing relative high mortality among breeding adults, cause an influx of floaters into the breeding population, reducing the size of the non-breeding group (Votier *et al.* 2008). The number of floaters in the population of Great-horned Owls fluctuates between 0 and 60% on the 3-year cycle of the main food resource, the Snowshoe Hare (Rohner 1996).

Most studies on the non-breeding component in a population are conducted in a limited breeding range or in one or more colonies. In colonial species most non-breeders attend the colony (Nettleship & Birkhead 1988), but not all. Not all studies have made corrections for the amount of non-breeders not seen in the breeding areas or colony. Especially in seabirds, birds spend a major part of their life far out at sea, without being noticed by any ornithologist. In Short-tailed Shearwaters it is shown that every breeding season about 7% of the expected adults does not attend the colony (Wooller *et al.* 1989, Bradley *et al.* 1999). Nettleship & Birkhead (1988) mention figures above 90% of colony attendance among adults of *Alcidae.* Furthermore, if a floater population is present, it might be reported, as is the case in increasing or stable populations. In decreasing populations floaters might be absent when all sexually mature birds take advantage of the lack of experienced breeders (i.c. Rutz & Bijlsma 2006). In such cases the proportion of birds in nearly adult plumages in the breeding population increases (Newton 1979, Ferrer *et al.* 2003). But there are examples of the contrary; e.g. the population of the globally threatened Balearic shearwater is declining with a very low number of breeders (Oro *et al.* 2004) while the proportion of non-breeders is substantial based on total numbers estimated (pers. com. J.M. Arcos).

In cave breeding species the number of suitable nest sites might be limited, leading to heavy competition for breeding sites. In such species first time breeding can be delayed due to lack of nest sites. In a smaller species like the Tree Swallow first year birds (after one winter) are sexual mature (Stutchbury & Robertson 1987, Kempenaers *et al.* 2001). They form the majority in the group of non-breeding adults; estimated in total at 26%. Among Red-winged Blackbirds two type of floaters are distinguished, the shallow floaters and the deep floaters (Shutler & Weatherhead 1991). Both types differ in some morphological measures (wing length, size of epaulets in red and yellow) from territorial birds, but not in body condition. As in tree swallows, most of the floaters among Red-winged Blackbirds are second year birds (in total 25%). In seabirds the average weight (as an indicator for condition) of non-breeding adults during the breeding season is most times higher than of breeding adults, up to 20% (Nettleship & Birkhead 1988, Bell *et al.* 2005). This shows the costs of investment of being breeding adult!

In a study on Red Grouse non-breeding adults had a far lower winter survival compared to breeding adults (Watson 1985). Therefore the number of non-breeding adults sharply declined due to mortality and emigration (96,6%), whereas the number of breeding adults hardly differed between autumn and spring (1,7%). In this study the number of territorial adults in autumn was a good predictor for the number of breeding adults in the next spring. These results were consistent with the theory of the doomed surplus (living with little or no perspective). In another study on Red Grouse the outcome was completely different due to ecological conditions (Park *et al.* 2002). Among non-breeding and breeding adults the winter survival rates were about the same. The main explanation for this outcome was the differences in predation pressure between both study areas and study moments (1957-67 versus 1986-93). In populations without predation the status in autumn was a good predictor for the number of breeding adults next spring. Under a higher predation pressure (fox, raptors) both adult groups are victim. From both studies it became clear that in the event of losses under territorial adults, the open places are filled up quickly by surviving non-breeding adults (male or female).

In Britain farmland species have declined strongly since the seventies. Although the main factors were equal for the species, the timing of the decline differed between species (Le V. dit Durell & Clarck 2004). It was show that the structure of the population was a major factor in the timing of the decline. Species with a relative large proportion of non-breeding adults, and thus delayed first time breeding, showed the decline later in time compared with species with a small fraction of non-breeding adults. In the nineties by Ens *et al.* (1995) the queuing-hypothesis was formulated, based on an extensive colour-ringing program among Oystercatchers. It was show that this species can choose between high quality breeding sites and low quality sites. The competition for the high quality sites is heavy; high quality would say survival of young almost for granted and on the low quality sites totally not. So, birds should wait for years to get a high quality nest site or take a low quality site immediately. There was a queue for the high quality sites; waiting was profitable; thus forming a large group of non-breeding adults (46% Table 3.2.2).

Table 3.2.1 Proportion of floaters (%) and body mass (g) in different seabird species. Exact figures given, some species just qualitative.

seabird species	% floaters	weight	source
European storm petrel	31	28	Hamery *et al.* 1986
Wilson's storm petrel	39	40	Ainley *et al.* 1984
least auklet	25	90	Jones 1992
Bulwer's petrel	17	93	Mougin *et al.* 1997
common diving petrel	7	120	Chastel *et al.* 1995b
common tern	45	130	Dittman & Becker 2003, Becker & Bradley 2007
blue petrel	14	200	Chastel *et al.* 1995b
Cassin's auklet	25	200	Manuwal 1974a, 1974b
red-billed gull	49	280	Mills 1989
puffin	29	400	Nettleship & Birkhead 1989, Ashcroft 1979
kittiwake	38	410	Cam *et al.* 1998, Coulson 1998
Hawaiian dark-rumped petrel	48	434	Simons 1985
short-tailed shearwater	25	450	Wooler *et al.* 1989, Bradley *et al.* 1999
cape petrel	50	460	Weidinger 1996
Manx shearwater	28	460	Brooke 1990
Magenta petrel	50	500	Imber *et al.* 2005
snow petrel	48	500	Chastel *et al.* 1993
northern fulmar	37	600	Coulson 1972
great-winged petrel	60	620	Chastel 1995
black petrel	28	700	Bell *et al.* 2009, Bell *et al.* 2007
white-headed petrel	50	700	Zotier 1990
southern fulmar	43	800	Jenouvriers *et al.* 2003
Cory's shearwater	40	840	Mougin *et al.* 1997, Halley *et al.* 1995
common guillemot	50	860	Votier *et al.* 2008
grey petrel	48	1.000	Chastel 1995, Bell 2002
herring gull	45	1.100	Shugart *et al.* 1987
southpolar skua	30	1.156	Ainley *et al.* 1984
white-chinned petrel	61	1.200	Chastel 1995a
great skua	46	1.400	Klomp & Furness 1992, Catrey *et al.* 1998
northern gannet	40	3.000	Nelson 1966
common eider	22	2.200	Coulson 1984
yellow-nosed albatross	60	2.200	Jouventin & Weimerskirch 1988
sooty albatross	71	2.500	Cuthbert & Summers 2004
light-mantled sooty albatross	75	3.100	Jouventin & Weimerskirch 1988
laysan albatross	69	3.200	Fisher 1969
waved albatross	60	3.500	Anderson *et al.* 2002
black-browed albatross	30	3.800	Croxall *et al.* 1998
Adelie penguin	38	4.700	Ainley *et al.* 1984
grey-headed albatross	78	4.900	Prince *et al.* 1994, Converse *et al.* 2009
northern giant petrel	57	5.000	Hunter 1984, 1985, Voisin 1988
southern giant petrel	33	5.000	Hunter 1984, 1985, Voisin 1988
yellow-eyed penguin	31	6.000	Efford *et al.* 1994
Amsterdam albatross	85	6.400	Weimerskirch *et al.* 1997
wandering albatross	80	9.000	Weimerskirsch 1992, Gauthier *et al.* 2010a,b
king penguin	63	13.000	Le Bohec 2007
emperor penguin	63	33.000	Ainley *et al.* 1984
arctic tern	+	110	Bertram *et al.* 1934
long-tailed skua	+	295	Bertram *et al.* 1934
parasitic skua	+	450	Catry *et al.* 1998
long-tailed duck	+	730	Bertram *et al.* 1934
pomarine skua	+	750	Bertram *et al.* 1934
common scoter	+	1.000	Bertram *et al.* 1934
red-throated diver	+	1.600	Bertram *et al.* 1934
king eider	+	1.600	Bertram *et al.* 1934
glacous gull	+	1.600	Bertram *et al.* 1934
shag	+	1.900	Potts 1980
great northern diver	+	5.100	Piper *et al.* 2006

Table 3.2.2 *Proportion of floaters (%) and body mass (g) in different seabird species. Exact figures given, some species just qualitative.*

non-seabird species	% floaters	weight	source
house wren	14	9	Kendeigh 1942
blue tit	17	11	Dhondt 1989
nuthatch	26	18	Mathyssen 1989
indigo bunting	5	12	Payne 1989
pied flycatcher	40	13	Sternberg 1989
song sparrow	14	15	Tompa 1964, Smit & Arcese 1989.
house martin	10	18	Bryant 1989
tree swallow	23	19	Stutchbury & Robertson 1985, Kempenaers *et al.* 2001
silvereye	15	20	Catterall *et al.* 1982
ruffous-collared sparrow	50	25	Smith 1978
Arabian babbler	7	33	Zahavi 1989
skylark	10	38	Delius 1965
red-winged blackbird	25	68	Shutler & Weatherhead 1990
Florida shrub jay	35	80	Woolfenden & Fitzpatrick 1991
blackbird	14	100	Ribaut 1964
sparrowhawk	45	200	Newton 1985, Newton & Rothery 2003
magpie	30	220	Birkhead *et al.* 1986, Bayens 1981
Australian magpie	62	250	Carrick 1963
oystercatcher	46	260	Ens *et al.* 1995, Heg 1999
African black oystercatcher	60	280	Summers & Cooper 1977
goshawk	52	340	Kenward *et al.*1991
hen harrier	25	400	Temeles 1989, 1990
red grouse	29	550	Park *et al.* 2002
mountain duck	44	550	Riggert 1977
American coot	29	650	Alisauskas 1987
common buzzard	55	890	Kenward *et al.* 2000.
black kite	76	1000	Sergio *et al.* 2009
great-horned owl	53	1500	Rohner 1996, 1997
osprey	36	1500	Bretagnolle *et al.* 2008
Spanish imperial eagle	67	3000	Penteriani *et al.* 2009
eastern imperial eagle	60	3000	Rudnick *et al.* 2007
eagle owl	62	3750	Campioni *et al.* 2010
mute swan	69	11000	Jenkins *et al.* 1976, Brown & Brown 1993, Meek 1993
willow tit	+	12	Eckman *et al.* 1981
black-capped chickadee	+	13	Smith 1987, 1989
crested tit	+	13	Eckman *et al.* 1981
great tit	+	18	Drent 1983
barn swallow	+	20	Crook & Shields 1987
chaffinch	+	24	Seather & Fonstad 1981
songsparrow	+	25	Hochachka *et al.* 1989
white-crowned sparrow	+	26	Petrinovich & Patterson 1982
yellow-breasted chats	+	27	Thompson 1977
mountain bluebird	+	30	Power 1975
red-cockaded woodpecker	+	47	Walters *et al.* 2002
purple martin	+	52	Stutchbury 1991
grey starling	+	75	Saitou 2001
starling	+	78	Tobler & Smith 2003
pied babbler	+	85	Ridley *et al.* 2008
nutcracker	+	128	Rolando & Carisio 2003
boat-tailed crackle	+	170	Poston 1997
Eurasian kestrel	+	205	Village 1983
willow ptarmigan	+	450	Hannon 1983
ruffed grouse	+	600	Gullion 1981
red grouse	+	600	Watson 1985
peregrine falcon	+	850	Monneret 1988
blue grouse	+	1025	Jamieson & Zwickel 1983
raven	+	1200	Ratcliff 1997
ring-necked pheasant	+	1200	Burger 1966
Bonelli's eagle	+	3000	Caretta *et al.* 2006
golden eagle	+	4500	Watson 1997
black eagle	+	4500	Gargett 1975
bearded vulture	+	6.000	Carrete *et al.* 2006

Recruitment of non-breeding adults into the breeding population

In recent years many seabird studies have been published about the structure of the population. Since the introduction of CMR-models (based on Capture-Mark-Recapture; Lebreton *et al.* 1992, Converse *et al.* 2009), the stage of non-breeding adults is inevitable to make proper estimates for adult survival (Crespin *et al.* 2003, Jenouvriers *et al.* 2003, 2005, Converse *et al.* 2009). In case of the dead or emigration of a breeding adult, a non-breeding adult often fills an open space. The number of non-breeding adults and their role in population dynamics is difficult to estimate and understand (Weimerskirsch 1992). Only a part of the non-breeding adults (but the major part) is attending breeding sites (cf. Shugart *et al.* 1987, Bell *et al.* 2009, Birkhead & Nettleship 1988); the others stay elsewhere. Nonetheless, the existence of floaters has never been in discussion. In recent years more theoretical analysis have been made about the role of floaters in a population (Pen & Weising 2000, Lopez-Sepulcre & Kokko 2005, Pol *et al.* 2007, Blas & Hiraldo 2010), showing that they are a essential component in the dynamics of a population (Penteriani *et al.* 2011).

Non-breeding adults are capable of breeding; even they compete for nest sites, and if possible take over the partner or nest site, even within the breeding season, as shown by removal experiments (Cassin's Auklet, Manuwal 1974a; Grey Starling, Saitou 2001, Kestrel, Village 1983, American Kestrel, Bowman & Bird 1986, Sparrowhawk, Newton 1991), and breed successfully thereafter. Although, not all floaters will have a chance to enter the breeding population, they still are present around or near potential breeding sites or colonies. Among smaller species many floaters stay around breeding sites. One of the explanations for this behaviour is to gain information and experience, about future breeding opportunities, scanning for open space in order to be well prepared in the next breeding season (e.g. Piper *et al.* 2009). A second explanation is extra-pair copulation, and thus investment in the next generation without having a partner or a nest site (Saitou 2001, Stutchbury & Robertson 1987, Kempenaers *et al.* 2001).

On Skomer Island (Wales) a large colony of Common Guillemots is located (Votier *et al.* 2008). Outside the breeding season the birds spread over the adjacent seas, immature, and non-breeding adults further away than breeding adults. During the breeding season most of the non-breeding adults attend the colony and after one or two years they are seen as breeding adults, showing the natural succession in breeding status. Between 1984 and 2004 four large oil spills occurred in the winter areas of the birds. Due to these spills the survival rate of breeding adults decreased. Non-breeding adults quickly filled the open space. Resightings of ringed birds revealed a large influx of non-breeding adults into the breeding population after each oil spill. Due to this the colony size was hardly affected by this spills as adults took the empty places from the buffer of non-breeding-adults. Therefore, the number of sub adults and non-breeding adults decreased sharply. Similar findings of Potts *et al.* (1980) showed that the death of relative large numbers of breeders (shags) by an oil spill was followed by a relative strong increase in recruitment of new breeders.

Further north, in Scotland, a colony of common guillemots has been subject to large scale ringing (Crespin *et al.* 2006). Data from 1983-2001 revealed that the probability of entering the pool of non-breeding adults at the colony was negatively correlated with colony size. This could be explained by competition for resources like food between immatures and pre-breeders. Furthermore this study showed a positive relation between the probability of returning to the colony and the NAO (North Atlantic Oscilation). Return rates are high when the index is high (i.c. warm and windy weather). Although not clear, this could point towards higher food availability during a high return rate in the NAO.

In recent years effort has been put into understanding the process of recruitment into the breeding population (Becker & Bradley 2007). They found among Common Tern that the older birds arrive in the breeding area relatively early and in a better condition than the younger ones. The sub adult birds were the last group to arrive in spring, and at the same time with less weight compared to adults.

Conclusion

Non-breeding adults play a major role in bird populations and are characterized in their first years by less experience compared to breeding adults. In later years most of them will enter the breeding stage. Depending on several factors the size of the non-breeding stage may differ between sites and years. Nonetheless, weight (and probably life expectancy) is a predictor for the proportion of floaters in a species. In many studies it appeared that floaters are a natural buffer (or surplus) which can met the yearly starvation among breeding adults as well as extra mortality due to incidences.

Consequences for this report and models used

As has been shown, a proportion of floaters have to be taken into account when impacts have to be estimated on total number of breeding birds. The impact of increased mortality due to e.g. collisions will however both affect adult breeding as non-breeding birds, as well as sub adult and juvenile birds. Furthermore it should be taken into account when modelling populations that non-breeding adults can take the place of breeding birds disappearing from the population due to impacts of future wind farms. In the current situation in the Netherlands and Europe many seabird populations are still increasing after a long recovery of bad times due to high pressure of human activities or are at a peak level. Many seabird species in the past were suffering of high intensity of harvest of young birds or hunting of adult birds in the breeding colonies (e.g. Nelson 1979). Also disastrous oil spills have been diminished in the last decades, strengthening the positive trends, not meaning to say that seabirds do not suffer any more of human induced impacts (Harris & Birkhead 1988). Nevertheless, most of the time it is not known how the floater percentage relates to the status of the population, whether it is increasing, stable or decreasing. In case of increasing or stable populations one can argue that the floater population is highest, while with a decreasing population the buffer role is apparent and many non-breeding adults immediately can fill in breeding territories or sites.

As in many cases for the seabird species occurring in the Dutch part of the North Sea no information is available on the percentage of floaters we have chosen to use a conservative floater percentage of 10 and 30 % when modelling future trends in breeding birds incorporating increased mortality due to wind farm impacts. In our literature review we found a total of 36 out of 46 seabird studies a floater percentage higher than 30%. In this way we will be able to show what the potential buffering effect will be of floaters in the population for different species. These models can be compared with models not taking into account of the presence of floaters.

4 Effects of multiple offshore wind farms

In this chapter we first present results from the study in and around OWEZ (Krijgsveld *et al.* 2011, § 4.1). In the next paragraph we translate these results into effects of multiple wind farms (§ 4.2).

4.1 Occurrence and effects on birds in and around OWEZ

By using a systematic observation methodology during daylight we have determined species composition of flying birds in and around OWEZ (table 4.1.1). In table 4.1.1 an overview of observed total numbers are presented of the relevant species for a cumulative approach (the same species groups as in table 2.2.1.) In table 4.1.2 an overview is given of all observed species and the total numbers recorded with the visual method. In table 4.1.1 it is clear that for five selected species no observations were gathered at all. From table 4.1.2 it can be concluded that the most numerous species during the day consist of different gull species, cormorants and gannets. For a thorough description of the methodology used, and analysis and discussion of the data, we refer to Krijgsveld *et al.* (2011).

Table 4.1.1 Total number of birds per species observed in and near OWEZ in panorama scans in the period February 2007 – October 2009 during daylight (n panorama scans total = 405). The species selection is similar as presented in table 2.2.1. See for methodologies and further analysis Krijgsveld et al. (2011).

group / subgroup	English name	*scientific name*	Dutch name	total n
gulls	lesser black-backed gull	*Larus fuscus*	kleine mantelmeeuw	1,516
gulls	herring gull	*L. argentatus*	zilvermeeuw	1,143
cormorants	great cormorant	*Phalacrocorax carbo*	aalscholver	1,020
gulls	common gull	*L. canus*	stormmeeuw	993
gulls	kittiwake	*Rissa tridactyla*	drieteenmeeuw	965
gulls	great black-backed gull	*L. marinus*	grote mantelmeeuw	532
gulls	little gull	*L. minutus*	dwergmeeuw	481
gannets	northern gannet	*Morus bassanus*	jan van gent	329
terns	sandwich tern	*S. sandvicensis*	grote stern	209
sea ducks	common scoter	*Melanitta nigra*	zwarte zee-eend	140
geese & swans	dark-bellied brent goose	*Branta bernicla*	rotgans	138
terns	common tern	*Sterna hirundo*	visdief	20
alcids	razorbill/guillemot	*aalge/torda sp.*	alk/zeekoet	17
divers	red-throated diver	*Gavia stellata*	roodkeelduiker	13
tubenoses	northern fulmar	*Fulmarus glacialis*	noordse stormvogel	11
sea ducks	common eider	*Somateria mollissima*	eider	11
alcids	guillemot	*Uria aalge*	zeekoet	11
alcids	razorbill	*Alca torda*	alk	10
terns	"comic tern"	*S. Hirundo/arctica*	visdief/noordse stern	9
divers	diver sp.	*G. stellata/arctica*	duiker sp.	6
divers	black-throated diver	*Gavia arctica*	parelduiker	1
cormorants	European shag	*P. aristotelis*	kuifaalscholver	1
geese & swans	Bewick's swan	*Cygnus bewickii*	kleine zwaan	0
swimming ducks	Eurasian Shellduck	*Tadorna tadorna*	bergeend	0
skuas	great skua	*Stercorarius skua*	grote jager	0
terns	little tern	*S. albifrons*	dwergstern	0
alcids	puffin	*Fratercula arctica*	papegaaiduiker	0

Table 4.1.2 *Total number of birds per species observed in and near OWEZ in panorama scans in the period February 2007 – October 2009 during daylight (with n=140 panorama scans in spring, n=71 in summer, n=121 in autumn, and n=73 in winter). Only flying birds within 3 km distance from the metmast. In light blue the relevant species are highlighted, with in green groups of unidentified birds which might consist of relevant species. See for methodologies and further analysis Krijgsveld et al. (2011).*

group	species	spring	summer	autumn	winter	total
landbirds	starling	675	2	2151	23	2851
gulls	large gull	847	266	293	209	1615
gulls	lesser black-backed gull	892	406	208	10	1516
gulls	herring gull	749	119	75	200	1143
cormorants	great cormorant	238	362	268	152	1020
gulls	common gull	237	6	100	650	993
gulls	kittiwake	18		468	479	965
gulls	great black-backed gull	113	10	181	228	532
gulls	little gull	465			16	481
gulls	black-headed gull	181	93	21	72	367
gannets	northern gannet	135	9	154	31	329
gulls	small gull	81		19	125	225
terns	sandwich tern	42	124	43		209
sea ducks	common scoter	130	4	2	4	140
geese & swans	dark-bellied brent goose	24		26	88	138
gulls	black-backed gull spec.	66	12	35	12	125
gulls	gull spec.	33		72	17	122
landbirds	thrush spec.			73		73
waders	dunlin	31				31
other ducks	northern pintail			30		30
landbirds	swallow	18				18
alcids	razorbill/guillemot			3	14	17
landbirds	blackbird	6		9		15
divers	red-throated diver	2			11	13
landbirds	skylark			13		13
other ducks	red-breasted merganser	8		4		12
sea ducks	velvet scoter	12				12
alcids	guillemot			1	10	11
sea ducks	eider			10	1	11
terns	common tern	10	1			11
tubenoses	northern fulmar	1		1	9	11
alcids	razorbill		1	2	7	10
landbirds	jackdaw			10		10
other ducks	scaup	8			1	9
terns	common/arctic tern	6	2	1		9
landbirds	meadow pipit			8		8
landbirds	yellow wagtail			8		8
other ducks	teal	8				8
waders	calidris spec.	8				8
landbirds	pipit spec.			7		7
divers	diver spec.	5			1	6
landbirds	song thrush			6		6
landbirds	homing pigeon	1		4		5
geese & swans	goose spec.				4	4
other ducks	goosander				4	4
other ducks	Eurasian wigeon			3	1	4
raptors & owls	merlin	4				4
waders	Eurasian curlew		4			4
landbirds	grey heron		3			3
landbirds	chaffinch	1		2		3
landbirds	redwing			3		3
terns	black tern	3				3
geese & swans	greylag goose				2	2
landbirds	wood pigeon			2		2
landbirds	house martin	2				2
landbirds	redpoll			2		2
landbirds	songbird spec.	1		1		2
landbirds	swift		2			2
other ducks	duck spec.		1	1		2
terns	arctic tern	2				2
terns	tern spec	1		1		2
waders	Eurasian golden plover	2				2
waders	lapwing	2			0	2
waders	wader spec.	2				2
cormorants	European shag				1	1
divers	black-throated diver				1	1
grebes	great crested grebe			1		1
gulls	common/herring gull			1		1
gulls	Sabine's gull			1		1
landbirds	carrion crow	1				1
landbirds	pigeon spec.				1	1
landbirds	pied wagtail			1		1
raptors & owls	goshawk			1		1
raptors & owls	kestrel	1				1
raptors & owls	marsh harrier		1			1
raptors & owls	peregrine			1		1
skuas	arctic skua	1				1
waders	grey plover	1				1
waders	oystercatcher		1			1

From collision risk to estimated number of victims at OWEZ

Collision rates of bird species with the OWEZ wind farm have been calculated based on the abundance of bird flights of the different species (general fluxes determined by radar, see further Krijgsveld *et al.* (2011) to arrive at species specific fluxes), and the level of both macro-and micro-avoidance of these species (also determined by radar, see further Krijgsveld *et al.* (2011) to arrive at species specific avoidance figures). As in OWEZ it was not technically possible to measure collisions the Band model method has been applied to estimate number of victims per species and/or species group.

Migrant passerines passing the area reached high numbers in spring and autumn and dominate the number of estimated collision victims (table 4.1.3). A considerable number, approximately one million bird flocks, passed the OWEZ area at rotor height. Flock size varied between 1 and >5,000 individuals (starling). Nocturnally migrating passerines mostly migrate as single birds (Berthold 1999). Because of this, and because of the high level of variation in flight altitude, the highest number of collisions is expected to fall among the migrating passerines. Among passerines, rough estimates suggest an order of magnitude of some hundreds of collision victims on an annual basis, among all species of birds passing the area. In § 4.2 the Band model is used to calculate the numbers of victims of a cumulative scenario of 11 wind farms like OWEZ in the Dutch part of the North Sea. For passerines no distinction can be made between the near-shore scenario 1 and the offshore scenario 2 as no data are (yet) available on fluxes of migrant birds offshore.

Table 4.1.3 Species-specific flux and estimated number of collision victims in the OWEZ wind farm (based on Krijgsveld et al. 2011, table 15.2). Given are: proportional presence of species in the wind farm area as observed in panorama scans; species-specific flux in the wind farm area at rotor height, based on an overall flux of 1,866,040 bird groups; macro-avoidance and adjustments based on flight altitudes in the wind farm area; flux through the wind farm after correction for macro-avoidance and flight altitudes; crude estimate of the number of collision victims using the Band model. Fluxes are rounded off to the nearest decimal.

species group	%. of birds	flux in area	macro avoidance	prop.not @rotor	flux corrected	estimated # of Band model
divers	0.06	1,130	0.68		360	0.2
grebes	0.00	50	0.28	0.98	1	0.0
tubenoses	0.03	540	0.28	0.5	200	0.0
gannets	0.92	17,160	0.64		6,090	1.6
cormorants	4.20	78,430	0.18	0.5	32,160	30.2
geese & swans	0.35	6,500	0.68	0.5	1,040	0.9
sea ducks	0.41	7,590	0.71		2,170	0.1
other ducks	0.19	3,520	0.28	0.5	1,320	0.6
raptors & owls	0.02	360	0.28		270	0.1
waders	0.12	2,300	0.28		1,730	0.4
skuas	0.00	90	0.28		70	0.1
gulls	32.75	611,120	0.18		501,120	234.3
terns	0.57	10,660	0.28		7,990	2.9
alcids	0.38	7,000	0.68	0.98	50	0.0
passerines	60.00	1,119,600	0.28	0.5	419,850	309.9
total	100.00	1,866,040			974,420	581.2
# victims/wind turbine/year						16.1

Effect of habitat loss

Disturbance effects on local seabirds are reported in Leopold *et al.* (2010), who found a low abundance of local sea birds in the OWEZ wind farm area. This low abundance was related to the location of the wind farm rather than the wind farm itself: near-shore species remained closer to the coast, while the more pelagic species are abundant further away. Therefore, OWEZ is located in a dip in the density gradient perpendicular to the coast. Nevertheless, they found strong indications of disturbance in *alcids*, but numbers were too low to determine disturbance effects statistically significant.

The results of Krijgsveld *et al.* (2011) show that pelagic seabird species had the highest avoidance levels. This indicates that these species avoided the wind farm area, which may result in disturbance to foraging birds. However, as numbers of foraging birds in the area were low, the numbers of birds that were disturbed were probably limited. Gannets, alcids and marine ducks were all seen foraging within or near the wind farm on rare occasions.

4.2 Assessing the effects based on multiple wind farm scenarios

For multiple wind farms two scenarios exist:

1. all new farms about 10-20 km out of the coast in near-shore (shallow) waters
2. all new farms scattered over the Dutch part of the North Sea in offshore (deeper) waters

Collision risk model parameter selection in seabird species

For scenario 1 the number of estimated collision victims is a matter of applying a tenfold increase in the figures of the Band model calculations as presented in table 4.1.2 . Based on the species composition of the different species based on field observations (table 4.1.1 and 4.1.2) the total number of victims per species is estimated for OWEZ. This mainly applies to the species group gulls and terns, and yields species-specific numbers of collision victims as presented in table 4.2.1.

For scenario 2 for the seabird species in principle the same calculation approach has been followed, under the assumption that the total flux flying all over the North Sea is comparable to that of OWEZ. As it is clear from table 4.1.3 that more than 50% of the victims consist of mostly nocturnal migrant birds mainly consisting of passerines and other non-seabird species (e.g. 8.6 victims per turbine per year). The remaining part consists of seabirds (e.g. 7.5 victims per turbine per year). It is possible that outside the OWEZ area further at sea the total flux of seabirds is different from the OWEZ area, at least higher for offshore species and lower for coastal species. With the lack of data on total fluxes in the offshore situation, we have chosen to calculate species specific numbers of collision victims for wind farms in scenario 2 based on the same total number of collision victims in seabirds per wind turbine per year as estimated for OWEZ. This number is about 7.5 victims per wind turbine per year (less than half of the total of 16 victims as presented in table 4.1.3).

We have used the long-term aerial monitoring dataset available for the Dutch part of the North Sea (Arts 2010) to describe the proportions of occurrence of the different seabird species in the Dutch North Sea (figure 4.2.1) (Arts 2010, with an additional analysis by Poot *et al.* 2010). The aerial survey design of this monitoring program of the Dutch government is equally covering the complete area, in this way yielding a representative picture of the species composition of the seabird community (figure 4.2.2). We subsequently recalculated with the Band model species-specific collision victims, taking into account different avoidance behaviours, in this way proportionally divided the 7.5 victims per wind turbine per year over the species. An important assumption here is that the proportion of the different species over the flux is strongly related to the numerical proportion as determined during the aerial surveys. Based on the sensitivity indexes for collisions of Garthe & Hüppop (2004) fulmar, guillemot and razorbill have been ruled out as victims, and only for those species that potentially fly regularly at rotor height, e.g. during foraging, have estimates of casualties have been generated. In this way the proportion of calculated proportion of the different species over the flux is potentially related with the numerical proportion as determined based on the aerial surveys, as all the species involved are mainly foraging in flight. Nevertheless, the outcomes of this approach should be treated as a preliminary analysis of collision for a multiple offshore wind farm scenario.

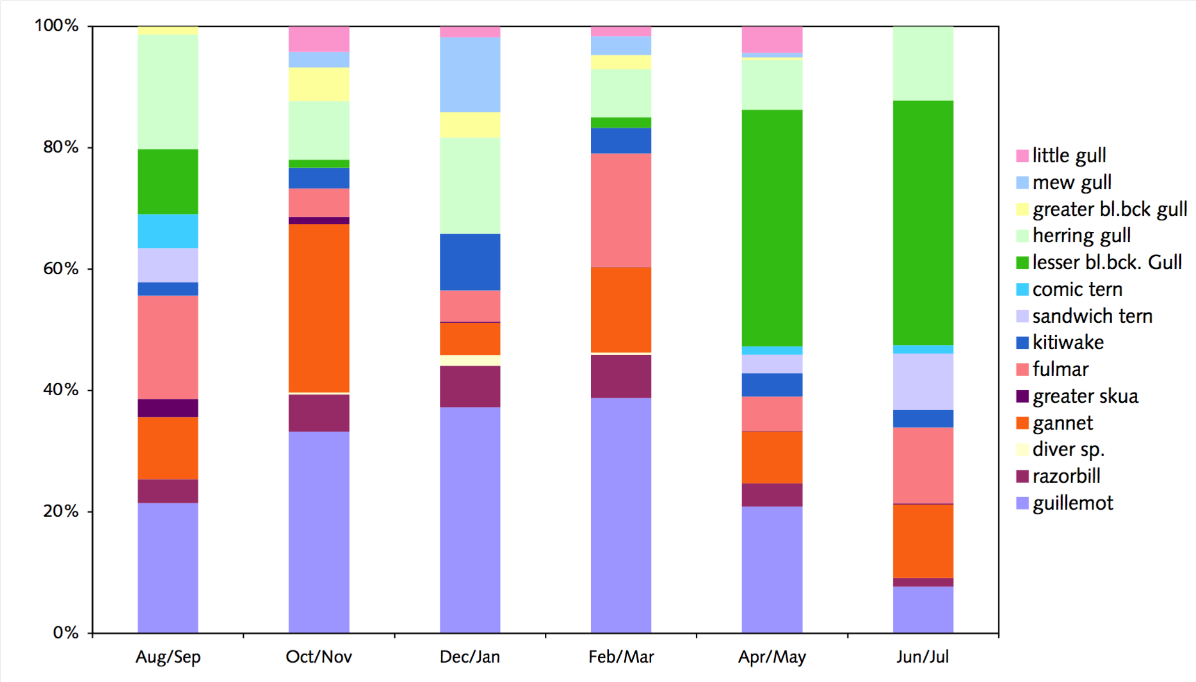


Figure 4.2.1 *Composition of the seabirds community present offshore in the course of the year in bimonthly periods (excluding sea ducks) based on an analysis of the long-year monitoring of seabirds in the Dutch part of the North Sea (Arts 2010, additionally analysed in Poot et al. 2010).*

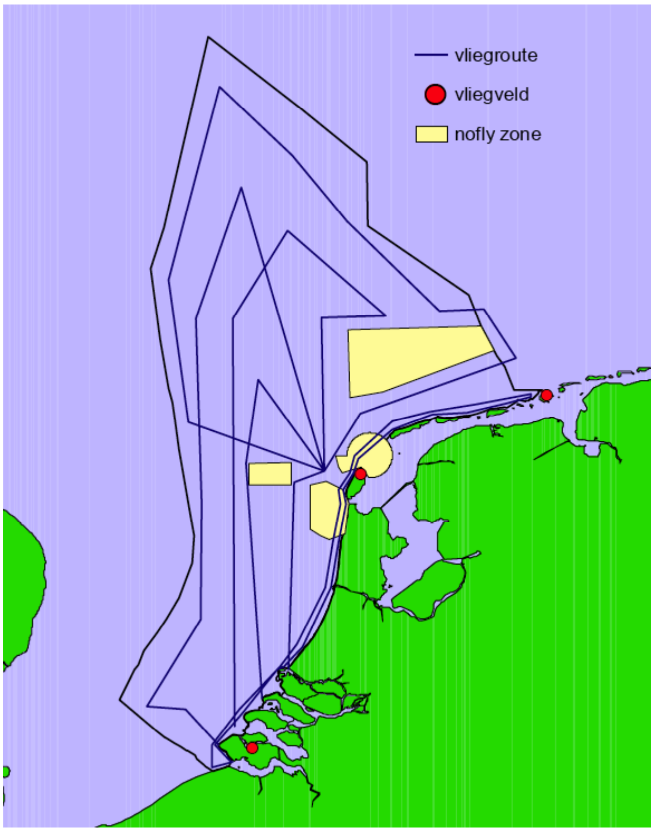


Figure 4.2.2 Survey design of the long-year monitoring of seabirds by the Ministry of Water and Transport in the Dutch part of the North Sea (taken from Arts 2010).

Table 4.2.1 Total number of estimated collision victims on the Dutch North Sea with 10 more offshore OWEZ wind farms developed; a near-shore scenario and an offshore scenario. Numbers of casualties are calculated according the Band model approach. For low flying guillemot, razorbill, and fulmar no casualties are expected offshore. Fraction at sea is based on the extensive data set of the long year aerial monitoring of the Dutch part of the North Sea (Arts 2010, Poot et al. 2010). See text for explanation of the calculation methodology.

			total # collision victims in 1 year	
	fraction at sea	fraction of risky species	Scenario 1 in OWEZ area	Scenario 2 outside OWEZ
black-headed gull	0.001		131.5	0.1
comic tern	0.014	0.024	2.8	60.5
diver sp.	0.004	0.007	1.8	9.2
fulmar	0.104		0.1	0.1
cormorant	0.010	?	332.2	?
gannet	0.134	0.232	17.2	199.2
greater black-backed gull	0.024	0.041	209.4	134.9
greater skua	0.008	0.014	0.8	39.6
guillemot	0.269		0.1	0.1
herring gull	0.120	0.207	585.6	698.1
kitiwake	0.043	0.074	345.6	217.1
lesser black-backed gull	0.153	0.264	776.8	875.8
little gull	0.021	0.037	172.3	75.1
mew gull	0.030	0.052	355.7	152.7
razorbill	0.049		0.1	0.1
sandwich tern	0.028	0.049	28.8	154.5

For those species with enough life history these figures has been used in the population models to estimate the implication at population level (section 5.1), otherwise only the Potential Biological Removal approach could be used to get an indication of the order of magnitude of cumulative effects on population levels. The results of this exercise are presented in section 5.2.

For cormorant for scenario 2 no number of victims have been estimated as the distribution of the species is nowadays still very coastal and therefore not well covered by the aerial surveys. The species shows an expansion further from the coast related to the presence of platforms and the offshore wind farms, but the question is how far from the coast the expansion will go.

In table 4.1.3 the total number of estimated collisions of migrant passerines in OWEZ is presented based on a calculation with the Band model. For passerines no distinction can be made between the near-shore scenario 1 and the offshore scenario 2 as no data are (yet) available on fluxes of migrant birds further offshore. Therefore, the numbers of victims of a cumulative scenario of 11 wind farms like OWEZ in the Dutch part of the North Sea are for both scenarios the same, resulting in an estimated total number of 3400 victims on a yearly basis.

Worst case approach habitat loss in some seabird species
For some species disturbance effects are to be expected. There is a lack on quantitative estimates of disturbance effects (leading to habitat loss) (Leopold *et al.* 2010) and lack of data how to translate the number of displaced birds into lowered survival or reproduction. For these species the results from the zero-growth model have been used in order to calculate the level of impact before growing or stable populations start to decline. See further the species account of gannet, great skua, guillemot and razorbill for this topic.

5 Effects on bird populations

5.1 Species specific population information and effects

In this chapter for selected species information is given on population size and trends. The selection contains those species that are numerous near-shore and offshore and have a high chance of interacting with offshore wind farms based on distribution, flight range and flight behaviour. For those species breeding in substantial numbers in The Netherlands population models were created. The same has been done for those species occurring as non-breeding species. Those models (Figure 1.3.1) are used to estimate the effect by different routes. Details of the parameters used in the models are given in appendix 1. For the scarcer species the species account is limited to a description of the population size and trends. For all species the cumulative effects on the (international) population has been estimated by applying the Potential Biological Removal approach (§ 5.2); basic parameters are found in appendix 4.

In every species account the following topics are presented:

- distribution, abundance and population trend in the Netherlands;
- description and limitations of the population models constructed;
- effects of OWEZ extrapolated, i.c. cumulative effects on selected populations.

For a selection of species the following maximum set of population models have been constructed (figure 1.3.1):

- 0-model (no victims, 0% floaters) by 0-models with 0%, 10% and 30% floaters;
- effect-model, by using the figures about collision from the Band model;
- 0-growth model (in order to determine the amount of victims above which population decline starts).

In the species accounts in principle only the most reliable models are presented on the general population trend, for instance for lesser black-backed gull only the models with 30% floaters are presented in the species account below, where the other models with 0% and 10% floaters are available for consult in appendix 3 (Table 5.1.0). In this appendix also detailed graphics are presented showing the variability of the different runs and the detailed outcomes per year classes (structure of the population).

In models with density dependence incorporated, formula 7 is used (§ 3.1). In this formula the factor Z plays an important role. The values for Z were obtained by validating the zero-model and are given in Table 5.1.1. Z is a measure for the steepness of the increase in the past decades (e.g. the period as used in the models).

The models with 0 % floaters, 10 % floaters and 30 % floaters are only available for those species with an increasing or a stable population trend. In appendix 3 also basic information on model input and the layout of graphics are presented. An overview of the parameters used of the different models that have been constructed is presented in Table 5.1.2. An overview of the literature sources of these parameters is given in appendix 5, with specific references in the species accounts in chapter 5.

Table 5.1.0 *Overview of different models that have been constructed and run (green). In dark green the most realistic model, which is explained and used in the text of this chapter. Other models are visualized in appendix 3 with additional information on parameter input (light green).*

Model (Fig 1.3.1)	0-model			Effect-model			O-growth model		
% floaters	0%	10%	30%	0%	10%	30%	0%	10%	30%
figure name in appendices	0b	0c	0d	6b	6c	6d	3b	3c	3d
Bewick's swan	+								
brent geese	+								
common tern			+			+			
cormorant	+								
diver sp.									
fulmar									
gannet			+			+			
greater bl.bck gull									
greater skua		+			+				
guillemot		+							
herring gull	+			+					
kittiwake		+			+				
lesser bl.bck. Gull			+			+			+
mew gull									
razorbill		+							
sandwich tern	+			+					

Table 5.1.1 *Values of Z obtained after validating a model with density dependence. The formula (from Jensen 1997, Brandon & Wade 2006):*

$$F_t = f_0 + (f_{max} - f_0) * [1 - (N_{t-1} / K)^z]$$

Further explanation can be found in § 2.4.

% floaters	0%	10%	30%
Bewick's swan	0,8		
brent geese	1		
comon tern	4	1,35	3
cormorant	1,5		
diver sp.			
fulmar			
gannet	5	4	1
greater bl.bck gull			
greater skua	7	3,5	2,5
guillemot	1,5	1,9	5
herring gull	0,25		
kittiwake	1	1	
lesser bl.bck. Gull	0,9	0,75	1,1
mew gull			
razorbill	5	7	10
sandwich tern	0,5	0,5	

Table 5.1.2 Overview of the parameters used of the different models that have been constructed. In dark green the most realistic model, which is explained and used in the text of this chapter. Other models are visualised in appendix 3 with additional information on parameter input. An overview of the literature sources is given in appendix 5, with specific references in the species accounts in chapter 5.

Model	parameter	0-model			Effect model			zero-growth		
% floaters		0	10	30	0	10	30	0	10	30
name in appendices		0b	0c	0d	6b	6c	6d	3b	3c	3d
Bewicks	S0 (sd)	0,66 (0,01)			0,66 (0,01)			0,66 (0,01)		
	S1 (sd)	0,822 (0,01)			0,822 (0,01)			0,822 (0,01)		
	Sad (sd)	0,822 (0,0,079)			0,822 (0,0,079)			0,822 (0,0,079)		
	Fad	0,5 (0,185)			0,5 (0,185)			0,5 (0,185)		
	B1	4			4			4		
Brent	S0	0,6 (0,01)			0,6 (0,01)			0,6 (0,01)		
	Sad	0,9 (0,036)			0,9 (0,036)			0,9 (0,036)		
	Fad	0,875 (0,39)			0,875 (0,39)			0,875 (0,39)		
	B1	2			2			2		
common tern	S0 (sd)		0,67 (0,05)	0,67 (0,05)	0,67 (0,05)	0,67 (0,05)	0,67 (0,05)	0,67 (0,05)	0,67 (0,05)	0,67 (0,05)
	Sad (sd)		0,898 (0,05)	0,898 (0,05)	0,898 (0,05)	0,898 (0,05)	0,898 (0,05)	0,898 (0,05)	0,898 (0,05)	0,898 (0,05)
	Fad		0,405 (0,25)	0,425 (0,25)	0,35 (0,25)	0,405 (0,25)	0,425 (0,25)	0,35 (0,25)	0,405 (0,25)	0,425 (0,25)
	B1		2	2	2	2	2	2	2	2
cormorant	S0 (sd)	0,58 (0,1)								
	S1 (sd)	0,88 (0,1)								
	Sad (sd)	0,88 (0,1)								
	Fad	0,625 (0,1)								
	B1	3								
diver sp										
fulmar										
gannet	S0 (sd)	0,542 (0,01)	0,542 (0,01)	0,542 (0,01)	0,542 (0,01)	0,542 (0,01)	0,542 (0,01)	0,542 (0,01)	0,542 (0,01)	0,542 (0,01)
	S1 (sd)	0,779 (0,01)	0,779 (0,01)	0,779 (0,01)	0,779 (0,01)	0,779 (0,01)	0,779 (0,01)	0,779 (0,01)	0,779 (0,01)	0,779 (0,01)
	S2 (sd)	0,859 (0,01)	0,859 (0,01)	0,859 (0,01)	0,859 (0,01)	0,859 (0,01)	0,859 (0,01)	0,859 (0,01)	0,859 (0,01)	0,859 (0,01)
	S3 (sd)	0,863 (0,01)	0,863 (0,01)	0,863 (0,01)	0,863 (0,01)	0,863 (0,01)	0,863 (0,01)	0,863 (0,01)	0,863 (0,01)	0,863 (0,01)
	Sad < 1965	0,907 (0,002)	0,907 (0,002)	0,907 (0,002)	0,907 (0,002)	0,907 (0,002)	0,907 (0,002)	0,907 (0,002)	0,907 (0,002)	0,907 (0,002)
	Sad=> 1965	0,919 (0,002)	0,919 (0,002)	0,919 (0,002)	0,919 (0,002)	0,919 (0,002)	0,919 (0,002)	0,919 (0,002)	0,919 (0,002)	0,919 (0,002)
	Fad < 1965	0,36 (0,1)	0,38 (0,1)	0,455 (0,1)	0,36 (0,1)	0,38 (0,1)	0,455 (0,1)	0,36 (0,1)	0,38 (0,1)	0,455 (0,1)
	Fad => 1965	0,475 (0,1)	0,58 (0,1)	0,8 (0,1)	0,475 (0,1)	0,58 (0,1)	0,8 (0,1)	0,475 (0,1)	0,58 (0,1)	0,8 (0,1)
	B1	5	5	5	5	5	5	5	5	5
greater bl.bck gull										
greater skua	S0 (sd)	0,8 (0,006)	0,8 (0,006)	0,8 (0,006)	0,8 (0,006)	0,8 (0,006)	0,8 (0,006)	0,8 (0,006)	0,8 (0,006)	0,8 (0,006)
	Sad (sd)	0,888 (0,006)	0,888 (0,006)	0,888 (0,006)	0,888 (0,006)	0,888 (0,006)	0,888 (0,006)	0,888 (0,006)	0,888 (0,006)	0,888 (0,006)
	Fad	0,8 (0,185)	0,805 (0,185)	0,985 (0,185)	0,8 (0,185)	0,805 (0,185)	0,985 (0,185)	0,8 (0,185)	0,805 (0,185)	0,985 (0,185)
	B1	7.5	7.5	7.5	7.5	7.5	7.5	7.5	7.5	7.5
guillemot	S0 (sd)	0,89 (0,05)	0,89 (0,05)	0,89 (0,05)				0,89 (0,05)	0,89 (0,05)	0,89 (0,05)
	Sad (sd)	0,946 (0,05)	0,946 (0,05)	0,946 (0,05)				0,946 (0,05)	0,946 (0,05)	0,946 (0,05)
	Fad	0,235 (0,06)	0,250 (0,06)	0,275 (0,06)				0,235 (0,06)	0,250 (0,06)	0,275 (0,06)
	B1	4,5	4,5	4,5				4,5	4,5	4,5
herring gull	S0 (sd)	0,78 (0,01)			0,78 (0,01)			0,78 (0,01)		
	Sad (sd)	0,88 (0,013)			0,88 (0,013)			0,88 (0,013)		
	Fad	0,675 (0,25)			0,675 (0,25)			0,675 (0,25)		
	B1	4.5			4.5			4.5		
kittiwake	S0 (sd)	0,79 (0,01)	0,79 (0,01)		0,79 (0,01)	0,79 (0,01)		0,79 (0,01)	0,79 (0,01)	
	Sad (sd)	0,941 (0,01)	0,941 (0,01)		0,941 (0,01)	0,941 (0,01)		0,941 (0,01)	0,941 (0,01)	
	Fad	0,06 (0,2)	0,072 (0,2)		0,06 (0,2)	0,072 (0,2)		0,06 (0,2)	0,072 (0,2)	
	B1	4	4		4	4		4	4	
lesser blk bck gull	S0 (sd)	0,78 (0,01)	0,78 (0,01)	0,78 (0,01)	0,78 (0,01)	0,78 (0,01)	0,78 (0,01)	0,78 (0,01)	0,78 (0,01)	0,78 (0,01)
	Sad (sd)	0,913 (0,012)	0,913 (0,012)	0,913 (0,012)	0,913 (0,012)	0,913 (0,012)	0,913 (0,012)	0,913 (0,012)	0,913 (0,012)	0,913 (0,012)
	Fad	0,69 (0,01)	0,75 (0,1)	0,825 (0,1)	0,69 (0,01)	0,75 (0,1)	0,825 (0,1)	0,69 (0,01)	0,75 (0,1)	0,825 (0,1)
	B1	4	4	4	4	4	4	4	4	4
mew gull										
razorbill	S0 (sd)	0,785 (0,05)	0,785 (0,05)	0,785 (0,05)				0,785 (0,05)	0,785 (0,05)	0,785 (0,05)
	Sad (sd)	0,900 (0,05)	0,900 (0,05)	0,900 (0,05)				0,900 (0,05)	0,900 (0,05)	0,900 (0,05)
	Fad	0,32 (0,13)	0,335 (0,13)	0,425 (0,13)				0,32 (0,13)	0,335 (0,13)	0,425 (0,13)
	B1	4	4	4				4	4	4
sandwich tern	S0 (sd)	0,358 (0,219)	0,358 (0,219)		0,358 (0,219)	0,358 (0,219)		0,358 (0,219)	0,358 (0,219)	
	S1 (sd)	0,741 (0,206)	0,741 (0,206)		0,741 (0,206)	0,741 (0,206)		0,741 (0,206)	0,741 (0,206)	
	Sad (sd)	0,898 (0,029)	0,898 (0,029)		0,898 (0,029)	0,898 (0,029)		0,898 (0,029)	0,898 (0,029)	
	Fad	0,35 (0,25)	0,75 (0,25)		0,35 (0,25)	0,75 (0,25)		0,35 (0,25)	0,75 (0,25)	
	B1	3	3		3	3		3	3	
	imm<1982	0,18	0,16		0,18	0,16		0,18	0,16	
	imm=>1982	0,02	0,02		0,02	0,02		0,02	0,02	

5.1.1 Seabird species (mainly) breeding in the Netherlands

Cormorant *Phalacrocorax carbo*

Occurrence and population trend in the Netherlands

Numbers of breeding cormorants in the Netherlands have increased since the 1970s, at which time the breeding population was around 2,500 pairs (Bijlsma *et al.* 2001, Bregnballe 1996). Numbers rose in the early 1980s to over 7,000 pairs and ten years later reached over 19,000 pairs (Bregnballe 1996). In the 1990s between 14,000-21,000 pairs bred at around 25 colonies within the Netherlands, mainly located in fresh water marshes inland. In the nineties the initiation of coastal colonies occurred, with after the year 2000 a substantial increase of coastal breeding pairs and subsequently an increase of cormorants foraging at sea (Leopold & Van Damme 1999). With exploring the marine habitat cormorants also learned to fish behind trawlers (Camphuysen 1999).

The initial increase is due to better protection measures (Newson *et al.* 2007), better fishing opportunities (Bijlsma *et al.* 2001) and a decline in the concentration of pollutants such as PCB's (Boudewijn & Dirksen 2001). Since the mid-1990s, the number of breeding birds has stabilised with an estimated 21,000 pairs breeding in 2007 (Van Dijk *et al.* 2009). The known and modelled trends for the breeding population of cormorant are shown in figure 3.1.1. The model reflects the period of growth up to the beginning of this centuary and the subsequent stabilisation. However, individual colonies showed different developments, with inland colonies having low reproduction and decreasing total numbers and coastal colonies showing a substantial increase, especially the colonies on the Wadden Isles (Van Dijk *et al.* 2009).

With increasing coastal colonies since 2000 at sea cormorants can be found foraging up to 25 km from the coast, with the larger numbers found in the coastal zone and in particular around the Delta and the Wadden Islands (Bijlsma *et al.* 2001, Leopold & Van Damme 1999).

Description and limitations of the constructed population model

A population model was built for the total Dutch breeding population as reproduction and other parameters were hardly available for the individual coastal colonies. Also for the total Dutch population dynamic parameters are scarce. The following parameters have been used: Adult survival before 1980 is set at 0.82 (+/- 0.1) and after 1980 set at 0.88 (+/- 0.1), 1st year survival set at 0.58 (+/-0.1), sub adult yearly survival set at adult survival (Frederiksen & Bregnballe 2000). Year of 1st breeding set at 3 year (BTO bird facts) and maximum age set at 23 year (Euring). Reproduction was set at 1.25 (Van Eerden & Van Rijn 2004).

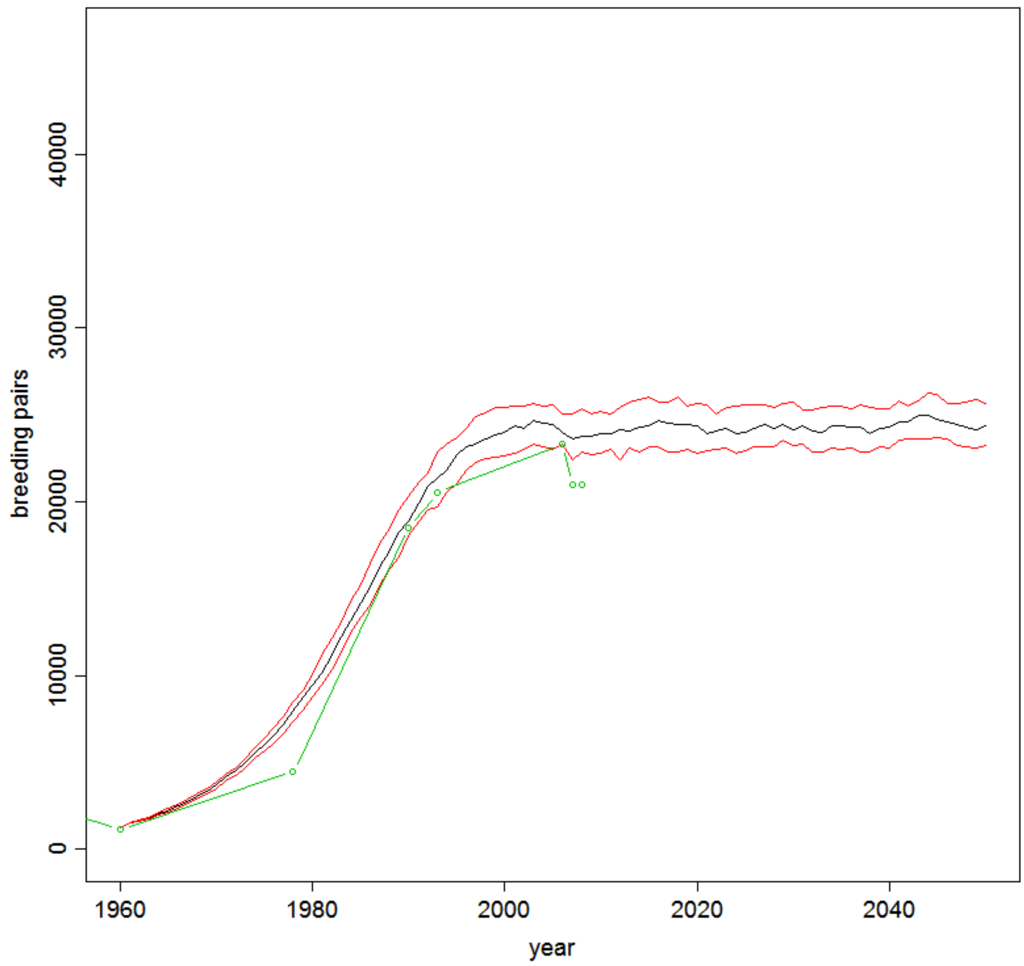


Figure 5.1.1 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for cormorant.*

Effects of OWEZ extrapolated - cumulative effects on the selected population
Cormorants are one of the most common species in the OWEZ wind farm area. Birds are foraging within the wind farm and resting on the meteorological mast and wind turbine access platforms on a daily basis, with the largest numbers during summer. This is a recent development in the Dutch part of the North Sea, as cormorants were not used to occur so numerously so far out at sea. The development is also in line with a levelling off of the Dutch breeding population of which the majority still breeds inland in fresh water marshes and forages in fresh water. The wind farm with its availability of resting posts and possibly growing availability of fish is another chapter in the expansion story of the cormorant in Dutch coastal waters.

Regularly cormorants fly from the metmast, OWEZ wind turbines and adjacent gas platforms to the feeding areas in the wind farm, but also in the surrounding waters, either to forage alone or in small to medium-sized flocks, with sometimes tens to over hundred of birds behind fishing vessels. Based on the recorded flight movement with radar and the visual observations it cannot be ruled out that the flight movements will yield collision victims. Based on the calculations done with the Band model the numbers of estimated collision victims are substantial. However, it is clear that the main effect of OWEZ is a positive one and must be regarded as habitat expansion. Without the presence of the metmast and the wind turbines, the distribution of cormorants would not be so far out at sea (see further Leopold *et al.* 2010). Potentially the collision victims in this respect must be regarded as a side effect of a larger positive effect. The main effect of OWEZ therefore must firstly be expressed as an increase in carrying capacity for foraging breeding birds, e.g. of the colony in the dunes of Castricum, based on flight paths to and from OWEZ and this colony (own observations). The increase in carrying capacity can occur by a direct increase of breeding pairs, but also via an increase of the reproductive output of breeding birds, and/or by an increased survival of both breeding as well as non-breeding birds using OWEZ (non-breeding adults, as well as juvenile and sub adult birds). However, modelling the effects of multiple new offshore wind farms is not possible in this species as it is unknown in which quantitative way and even in which direction (positive or negative) this occurs.

Shelduck *Tadorna tadorna*

Occurrence and population trend in the Netherlands

In the middle of the 20th century around 2,500-3,300 pairs of Shelduck bred in the Netherlands. By the 1970s numbers had increased to around 3,500-4,500 pairs and by the early 1980s colonisation of inland waters had helped the population reach 6,000-9,000 pairs (Bijlsma *et al.* 2001). The current population is estimated at around 11,000 pairs, the vast majority of which are found around inter-tidal coastal areas, although smaller numbers are found inland (Alterra 2009).

During late June and early July almost the entire North Sea population of Shelducks gather in the German Wadden Sea to moult (Wernham *et al.* 2002; Blew & Südbeck 2005). From Britain, non-breeding and immature birds migrate from mid-June with the peak in early July involving birds that have finished breeding. Many adults and all juvenile birds do not make this migration, instead remaining within Britain and Ireland to moult. Birds are thought to cross the North Sea from Britain in a single nighttime flight (Wernham *et al.* 2002). Birds return to Britain more gradually but most have done so by December or January (Wernham *et al.* 2002).

During winters of extreme cold weather birds may move out of the Wadden Sea to the Dutch Delta or even further south along the coast or across the North Sea to England (Blew & Südbeck 2005). During winter around 22,000 Shelducks can be found in the Wadden Sea and a further 7,000 in the Delta (Van Dijk *et al.* 2009).

Effects of OWEZ extrapolated - cumulative effects on the selected population

With no observations of migrating shelducks in or in the vicinity of OWEZ, the next round of new offshore wind farms not being developed within the flight range of shelducks (in general being very coastal or over land) in the Dutch part of the North Sea, no cumulative negative effects are being expected for the Dutch breeding population. Also migratory movements of international, continental populations will mainly be located along the coast or over land (Lensink *et. al.* 2002, Camphuysen & Van Dijk 1983) with potentially most birds flying at night like for British birds coming to the international Wadden Sea to moult (Wernham *et al.* 2002).

Common eider *Somateria mollissima*

Occurrence and population trend in the Netherlands
Eiders were first recorded breeding in the Netherlands shortly after the turn of the 19th century (Swennen 1991). The breeding population remained small until an initial increase to almost 6,000 pairs in 1960. Shortly afterwards, the population declined dramatically by almost 80 percent in the following eight years. This decline has been attributed to pollution by pesticides and mostly affected breeding females (Swennen 1991; Bijlsma *et al.* 2001). In the following decades the population recovered and reached 10,000 pairs in the mid-1990s. Nowadays the number lies between 6,000 – 7,000 pairs (www.sovon.nl). Nearly all birds of the Dutch breeding population can be found around the Wadden Sea region, the main colonies being on the Wadden Islands.

During winter, Eider are found along the entire Dutch coast, although highest numbers are found in the Wadden Sea, offshore of the Wadden Islands and offshore of the delta (Camphuysen & Leopold 1994; Swennen 1991).

Effects of OWEZ extrapolated - cumulative effects on the selected population
This species strongly avoid offshore wind turbines during both diurnal as well as nocturnal migration over sea (Desholm & Kahlert 2005, as found in other seabird species by the study at OWEZ reported by Krijgsveld *et al.* (2011)). The migratory movements of international, northern populations will mainly be located along the coast, with relatively low numbers passing further south than the Wadden Sea (Lensink *et. al.* 2002, Camphuysen & Van Dijk 1983). In line with this hardly no observations of eiders were recorded in or in the vicinity of OWEZ, so that no cumulative negative effects are being expected for this species because the next round of new offshore wind farms not being developed within the regular distribution and flight range (being very coastal) of the species. This conclusion holds for both the Dutch and foreign eiders passing by in the Dutch coastal zone of the North Sea.

Herring gull *Larus argentatus*

Occurrence and population trend in the Netherlands
During the first part of the 20th century the Dutch breeding population of herring gulls was around 1,600 pairs. Numbers later increased to around 13,000 in the 1930s and

to 24,000 in the 1970s. Breeding numbers then increased further following various conservation measures, the reduction in persecution and the ban on pesticides (Camphuysen & Leopold 1994; Spaans 1998a; Bijlsma *et al.* 2001). In the early 1980s the Netherlands held around 90,000 breeding pairs. During the mid-1980s, however, numbers decreased, which has been attributed to predation pressure from foxes in the mainland colonies (Bijlsma *et al.* 2001). Over the last 20 years the population has shown a slight decline with the 2007 breeding population estimated at between 40,000 and 49,000 pairs (Van Dijk *et al.* 2009). The known and modelled trends for the breeding population of herring gull are shown in figure 5.1.2. The model reflects the period of growth up to the 1980s and the decline thereafter.

The breeding distribution is largely coastal with the most important breeding areas are Schiermonnikoog, Terschelling, Texel in the Wadden Sea region and Saeftinghe, Maasvlakte, and Kop van Schouwen in the Delta (Strucker *et al.* 2005; Van Dijk *et al.* 2009). The majority of birds forage up to 50 km from the colony with 100 km thought to be the maximum range for breeding birds (Ens *et al.* 2009). Camphuysen and Leopold (1994) suggest that most herring gulls scavenging at trawls were found within 10 km of the coast and most foraging birds within 5 km.

Outside the breeding season, numbers offshore peak during December and January, when an estimated 117,500 individuals could be found in Dutch North Sea waters in the eighties/early nineties (Camphuysen & Leopold 1994), but since then also the numbers offshore have declined seriously (Arts 2010). Most Dutch herring gulls winter in the Netherlands. During winter a peak occurs offshore explained by the arrival of birds of a northern origin, joining the many ten thousands of Dutch origin (Bijlsma *et al.* 2001).

Description and limitations of the constructed population model

A population model was built for the total Dutch breeding populations as during the course of the year the majority of birds occurring in the Dutch part of the North Sea are of Dutch origin, and therefore having the highest chance of interactions of potential impacts of new offshore wind farms. For the total Dutch population dynamic parameters are scarce. Therefore the following parameters have been used in order to reproduce the population trend for the total population in the Netherlands: Adult survival is set at 0.880 (+/- 0.13) based on Wanless *et al.* (1996). Sub adult survival is set at 0.78 (+/- 0.01) (Migot 1992). Age at first breeding is set at 4.5 based on Chabrzyk & Coulson (1976). Maximum age is 34 years (Euring). Reproduction is poorly described in the Netherlands. On the Wadden islands reproduction in 1967-1969 was 1.25 – 1.5, in 1983-1984 0.34 – 0.43 and 0.1 in the early 90ties (Spaans 1998c). Since the early 80ties the population is in decline. In the model reproduction was set to 1.35 before 1980 and 0.6 after 1980 en the carrying capacity value after 1980 was set to 20,000 breeding pairs. Since it is not expected that a declining population hold many floaters, as these are probably already for a great amount 'consumed' since the decline, only models with 0% floaters are made.

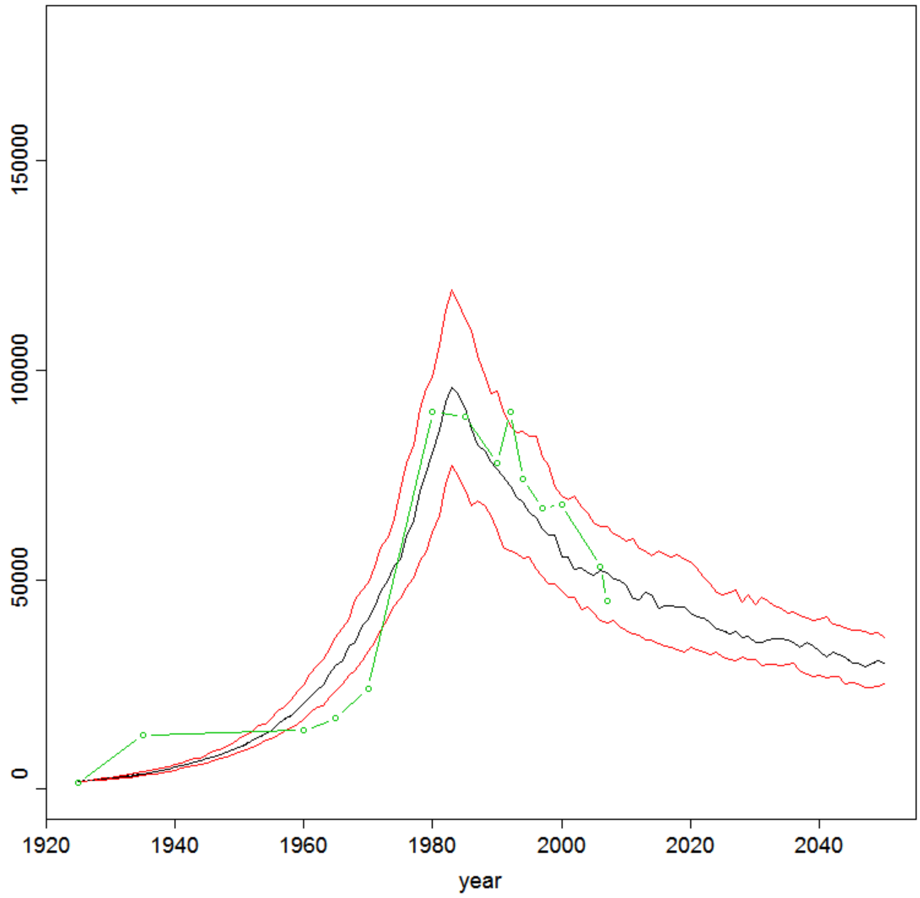


Figure 5.1.2 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for herring gull. Since it is not expected that a declining population holds any floaters, these are probably already 'consumed' since the decline, only models with 0% floaters are made.*

Effects of OWEZ extrapolated - cumulative effects on the selected population

In figure 5.1.3 the impact of the increased mortality of multiple wind farms in offshore waters is depicted. As can been seen the current decline of the Dutch breeding population speeds up by the additional impact of the estimated extra 700 female victims. One should bear in mind, that this impact must be regarded as a worst case scenario, as in the model the situation has been simulated that all collisions occur on breeding females during the breeding season with an extra effect on failure of 700 broods. The modelling did not take into account potential collisions with herring gulls of a foreign origin outside the breeding season or with birds of a non-breeding or juvenile status.

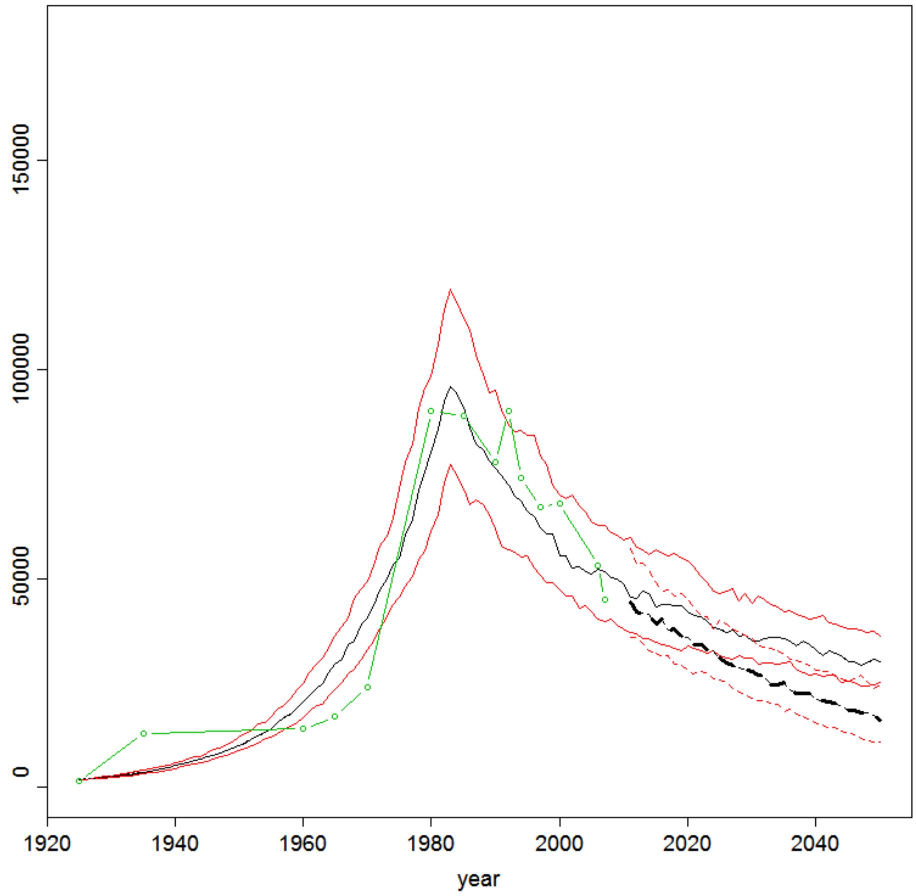


Figure 5.1.3 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for herring gull. With the effect of worst-case increased mortality, number of yearly collisions based on band model calculation (n female victims = 700), of a scenario with 11 offshore wind farms.*

Lesser black-backed gull *Larus fuscus*

Occurrence and population trend in the Netherlands

Lesser black-backed gulls first bred in the Netherlands in the mid-1920s and by the 1960s numbers had increased to around 80 pairs. During this period, the breeding population was limited to the coasts of the Wadden Sea. By the late 1960s the breeding population had increased to over 600 pairs and in the 1970s a period of exponential growth began, resulting a population reaching 11,000 pairs by the end of the decade. This increase continued into the following decades with the population reaching 23,000 pairs by the end of the 1980s and 57,000 pairs in the 1990s (Spaans 1998a; Bijlsma *et al.* 2001). In the following years up to 2007, numbers have continued to increase but then seemed to level off with an estimated breeding population of between 82,000 and 92,000 pairs (Van Dijk *et al.* 2009). The known and modelled trends for the breeding population of lesser black-backed gulls are shown in figure 5.1.4. The model reflects the period of exponential growth of the

Dutch breeding population that began in the late 1960s and levelling off around 2000. With reaching the peak numbers of breeding pairs also discussions about the reliability of the census numbers began, especially because in most colonies mixed breeding with herring gulls occurs. Different counting methods, especially in relation to differentiating between the two species have proven to yield substantial differences in census results (pers. com. C.J. Camphuysen). This fact also hampers the ability to conclude on whether already a decrease in breeding pairs has started or not, and made us decide not to try to produce a perfect fit of the modelling trend with the census data. Detailed breeding biology studies at Texel during most recent years in this species have shown low breeding success and that a decrease of this important colony is to be expected in coming years (Camphuysen *et al.* 2008, Camphuysen 2010). At present it is unclear whether this is a density dependent phenomenon or that other factors play a role (e.g. changing fishery activities and the related potential decrease of availability of discards).

The majority of breeding colonies in the Netherlands lie along the Wadden Sea coast, with the largest colonies on Texel, Terschelling and Schiermonnikoog. Around half of the Dutch breeding population occurs in South Holland and the Delta with the largest colony in the Maasvlakte (Strucker *et al.* 2007). Numbers here have stabilised in the past few years at around 25,000 pairs (Strucker *et al.* 2008). During the breeding season lesser black-backed gulls can forage up to 100-200 km from the colony with the maximum foraging distances being estimated at 400 km (Ens *et al.* 2009). Most birds, however, forage within 60-90 km (Camphuysen *et al.* 2008).

Most lesser black-backed gulls leave the Netherlands on southward migration during late summer to early winter. During this time many birds take a route westwards across the North Sea to England before heading in a more southerly winter quarters such as Spain and Portugal (Ens *et al.* 2009). Other individuals take a route along the west coast of continental Europe or partly overland. Few birds are present between December and February with the first birds returning from the end of February and March (Hustings *et al.* 2008; Bijlsma *et al.* 2001).

Description and limitations of the constructed population model

The model is based on the breeding population in the Netherlands. All parameters follow Lensink *et al. (*2010). Adult survival set at 0.913 (+-0.012 based on Wanless *et al.* (1996) and sub adult survival is set at 0.78 (+/- 0.01) based on Wanless *et al.* (1996) and Migot (1992). Age at first breeding is set at 4 years (BTO bird facts). Maximum age is 34 years (Euring). Reproduction in the Netherlands is described for a couple of colonies on the Wadden Isles (mainly in Spaans *et al.* 1994, Spaans & Spaans 1975, Spaans 1998b). Maximum reproduction determined in the Netherlands is 1.55 - 1.77. Reproduction for 0-model with 0% floaters set at 1.38 (+/-0.1).

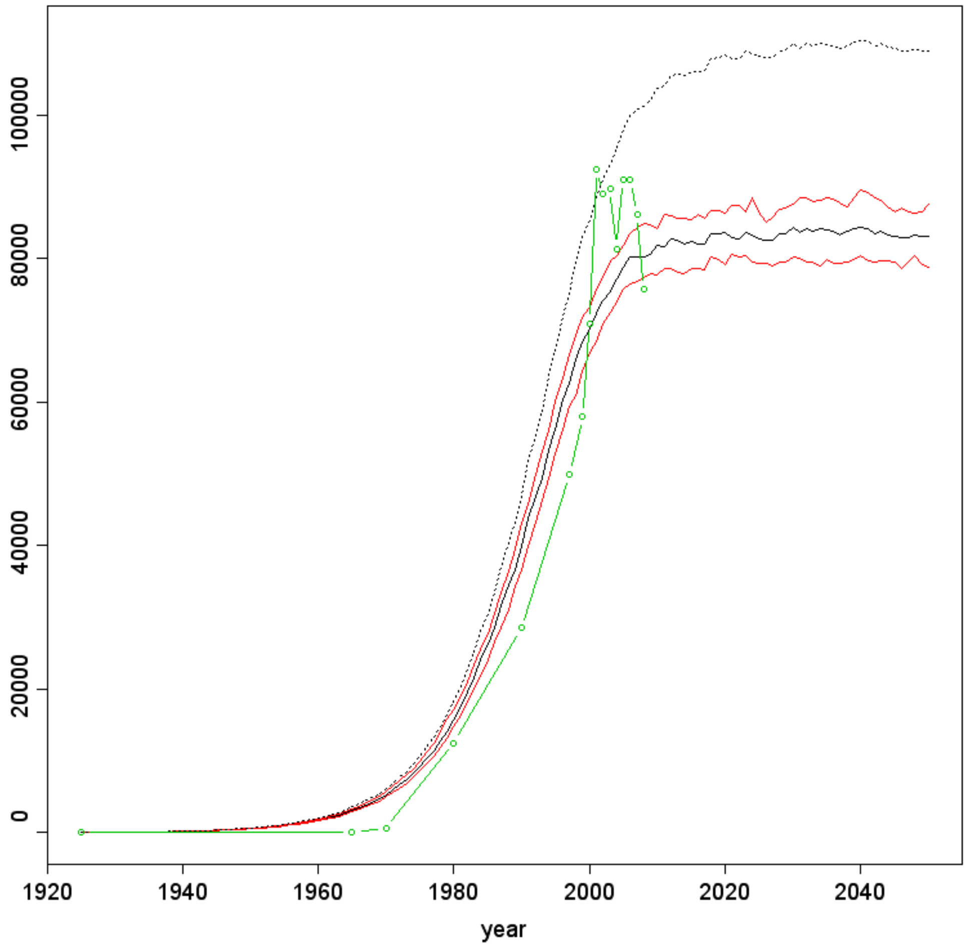


Figure 5.1.4 *Counted (green circles and line) and modelled population trend for lesser black-backed gull in number of pairs (black line=median; red lines=25 and 75 percentile; black broken line= floater pairs 30%) (source www.sovon.nl).*

Effects of OWEZ extrapolated - cumulative effects on the selected population

In figure 5.1.5 the impact of the increased mortality of multiple wind farms in offshore waters is depicted. As can been seen the total Dutch breeding population will remain on a stable level despite the additional impact of the estimated extra 875 female victims, under the condition that carrying capacity for the species will remain the same. It is discussed earlier that low breeding success at least in the Texel colonies indicates otherwise. In figure 5.1.6. the results are presented of a modelling exercise in which the level of the number of victims has been determined before the population goes into decline. This level with a population model with 30% floaters yields a level of 1,790 female birds. Later these figures of numbers of victims are compared to the number of victims determined with the Potential Biological Removal approach.

One should bear in mind that the imposed impacts in the models must be regarded as worst-case scenarios, as in the models the situation has been simulated that all collisions occur on breeding females during the breeding season with an extra effect on failure of

875 broods. Thus we did not take into account potential collisions of lesser black-backed gulls of a foreign origin outside the breeding season or with birds of a non-breeding or juvenile status, resulting in less impacts as now calculated for the total Dutch breeding population.

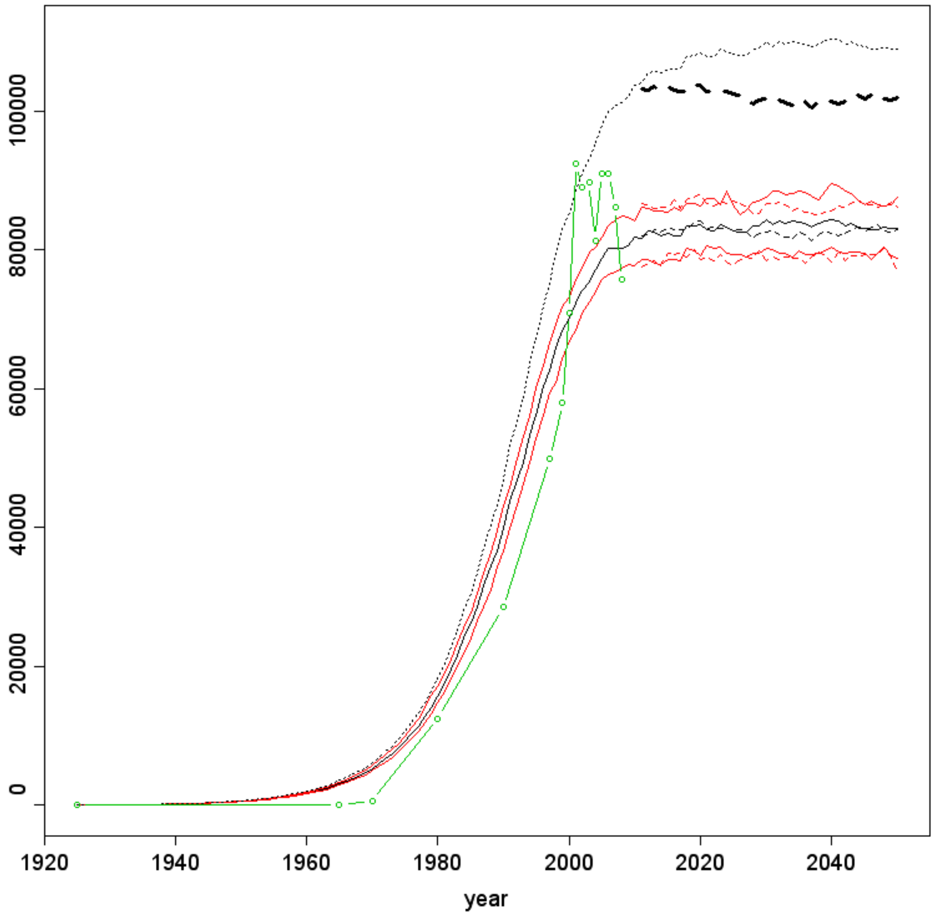


Figure 5.1.5 *Counted (green circles and line) and modelled population trend for lesser black-backed gull in number of pairs (black line=median; red lines=25 and 75 percentile; black broken line= floater pairs 30%) (source www.sovon.nl). With the effect of worst-case increased mortality, number of yearly collisions based on band model calculation (n victims = 875) for a scenario with 11 wind farms in offshore waters.*

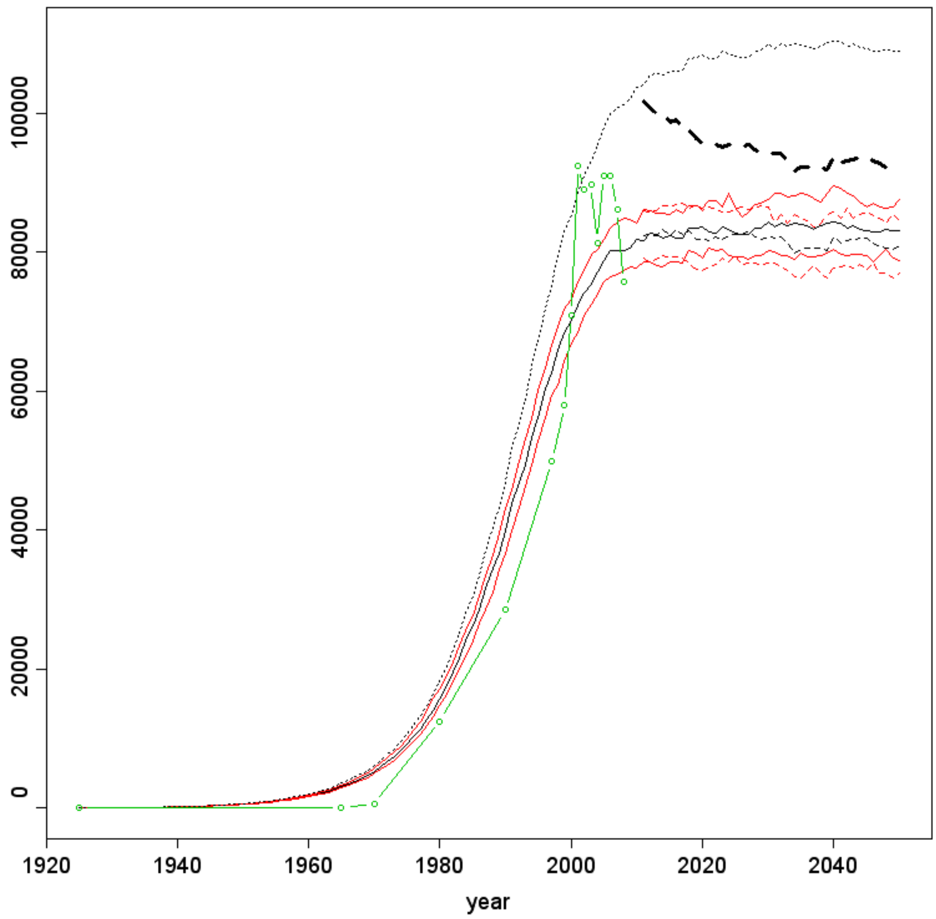


Figure 5.1.6 *Counted (green circles and line) and modelled population trend for lesser black-backed gull in number of pairs (black line=median; red lines=25 and 75 percentile; black broken line= floater pairs 30%) (source www.sovon.nl). With the effect of worst-case increased mortality, number of yearly collisions based on a scenario that the population reach a 0-growth status (resulting in a number of potential victims of 1,750 female birds, implying a scenario of more than 20 new wind farms in offshore waters).*

Sandwich tern *Sterna sandvicensis*

Occurrence and population trend in the Netherlands

Sandwich terns breeding in the Netherlands winter mainly along the coasts of West Africa. Although Sandwich terns are a fairly common breeding bird in the Netherlands, their breeding colonies are restricted to the coast. In the 1930s and 1950s, the Dutch breeding population was between 30,000 and 46,000 pairs (Bijlsma *et al.* 2001). However, the breeding population crashed to only 900 pairs in the 1960s due to poisoning from pesticides (Bijlsma *et al.* 2001). By the 1990s, the population had shown signs of recovery and was estimated at between 9,400 to 14,600 pairs. In 2007 an estimated 18,900 pairs bred in the Netherlands. The known and modelled trends for the breeding population of Sandwich terns are shown in figure 3.1.4. The model reflects the period of growth since the 1960s and shows two levels of growth represented by a higher immigration rate prior to the early 1980s.

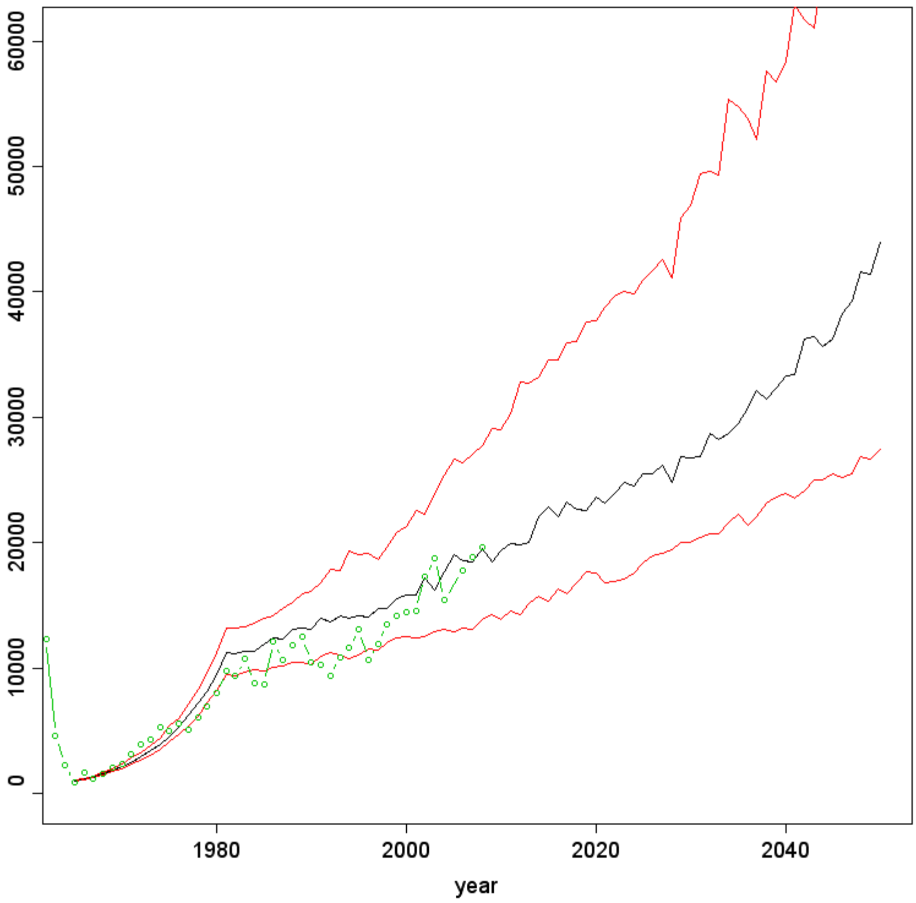


Figure 5.1.7 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for Sandwich tern with no floaters.*

Almost three-quarters of the Dutch population breed in the Wadden Sea, the largest colonies being on Griend, Terschelling, Ameland and Texel. Nearly all the remaining birds breed in the Delta, the largest colonies here in 2007 being at Stellendam, Serooskerke and Westerschelde (Strucker *et al.* 2008). During the breeding season, Sandwich terns can forage up to 30 to 45 km from the colony. Most birds, however, forage much closer to the colony with numbers at 5-10 km from the colony being half of those within 5 km of the colony (Garthe & Flore 2007). Relatively few Sandwich terns occur offshore. Most birds stay close to the coast and in particular near the breeding colonies (Camphuysen & Leopold 1994).

Description and limitations of the constructed population model

The model was built on the Dutch breeding population. Adult survival is set at 0.898 (+/- 0.029) (1st year survival is set at 0.358 (+/- 0.219) and 2nd year survival at 0.741

(+/- 0.206) (Robinson, 2010). Age at first breeding is set at 3 year (BTO bird facts), maximum age at 30 (Euring). Reproduction is set at 0.7 (+/- 0.25) (Stienen 2006). The measured population cannot be modelled without net immigration in the Dutch breeding population as assumed by Stienen (2006). Up until 1980 immigration is set to be 18%, after 1980 immigration is set at 2% in order to fit the population trend in the Netherlands. Recruitment is split in immigration and reproduction in this model. Assumed reproduction is just around the level in order to sustain a stable population (Stienen 2006), thus the 'local' Dutch population growth is assumed to been driven by a continuous low rate of net immigration. A model variant with 10% floaters can only be constructed if an unrealistic high level of reproduction is assumed (1.5 p/pair).

Effects of OWEZ extrapolated - cumulative effects on the selected population

The migratory movements of Dutch and international, more northerly populations are mainly nocturnal and occur most often at high altitudes. However, migratory movements can also occur during the day, but those are likely more concentrated along the coast, the area where also most foraging occurs for fuelling the long-distance flights (Lensink *et. al.* 2002, Camphuysen & Van Dijk 1983). Nocturnal migration does not have to be confined to the coast or above sea (Camphuysen 1992). With the next round of new offshore wind farms not being developed within the regular distribution and flight range (being very coastal) of locally foraging or diurnal migrating sandwich terns of non-breeding status or in the non-breeding season in the Dutch coastal zone of the North Sea (Bijlsma *et al.* 2001, Camphuysen & Leopold 1994), no cumulative negative effects are being expected for foreign populations. Therefore we have assumed that all potential negative impacts of new offshore wind farms will affect only Dutch birds. As new offshore wind farms will be not within the foraging range of sandwich terns, this could potentially only hold for two wind farms (Vlakte van de Raan and north of Schiermonnikoog), the only risky period for sandwich terns is when the birds arrive in spring and after they wander around after breeding. But as is clear from figure 5.1.8 the increase in the population, although at the moment partly dependent on the immigration of birds outside the Netherlands, will not be stopped due to the numbers of victims calculated.

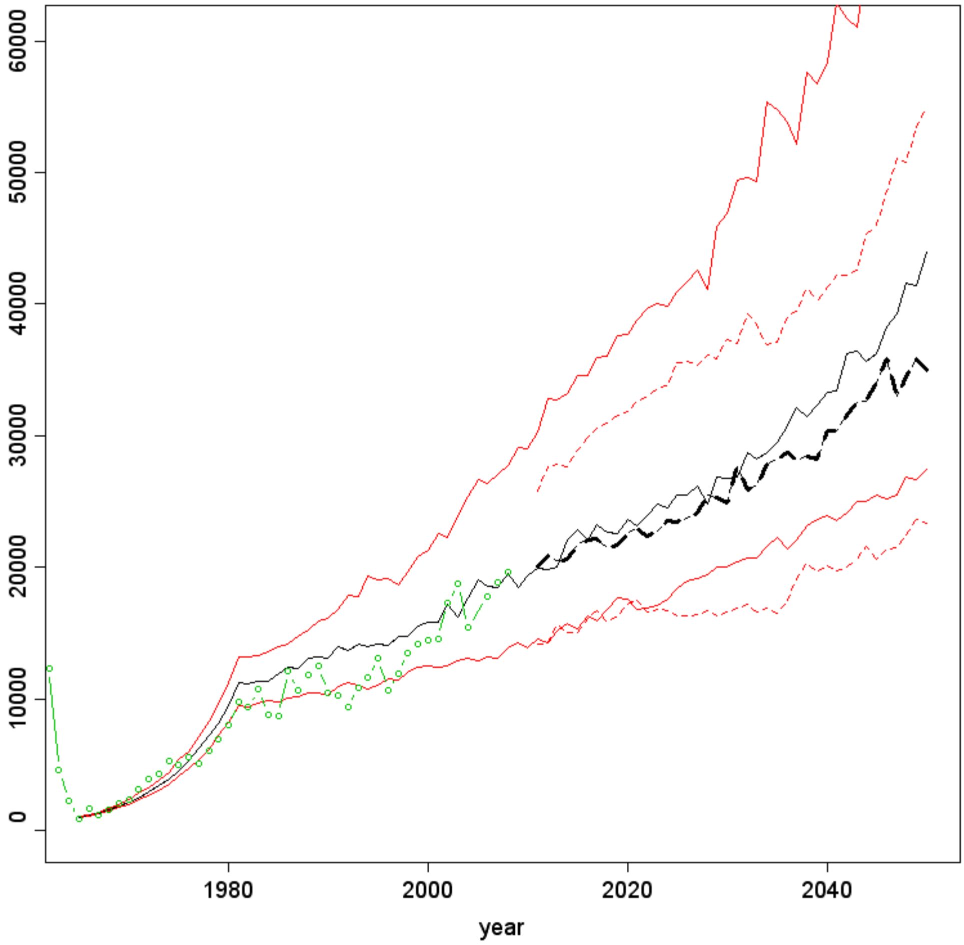


Figure 5.1.8 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for Sandwich tern with no floaters. With the effect of worst-case increased mortality, number of yearly collisions based on band model calculation (n victims = 150) for a scenario with 11 wind farms (like OWEZ) in offshore waters.*

Common tern *Sterna hirundo*

Occurrence and population trend in the Netherlands

Comparable to the Sandwich tern population, the number of common terns breeding in the Netherlands crashed during the 1960s due to pollution from pesticides (Bijlsma *et al.* 2001). Since then, the number of breeding birds has increased, reaching 16,000 to 18,000 pairs in 1992-1997 and 21,000 in 2007 (Bijlsma *et al.* 2001; Van Dijk *et al.* 2009). The known and modelled trends for the breeding population of common terns are shown in figure 5.1.9. The model reflects the period of growth since the 1960s.

The Kreupel in the IJsselmeer currently holds the largest breeding colony. Other key locations include Friesland, Griend, Rottumerplaats as well as the Maasvlakte and the Delta; which holds approximately one-third of the Dutch population (Strucker *et al.* 2005). The birds of colonies along the Dutch coast forage within 10 km of the shoreline; therefore the risk of effects by offshore wind farms mainly is confined to the period of arrival when birds can migrate over sea, and during the post-breeding period when birds can forage further from the coast.

Description and limitations of the constructed population model

Model based on breeding population. Adult survival is set at 0.898 (+/- 0.05) (Wendeln & Becker, 1998), sub adult survival is set at 0.67 (+/- 0.05) Wendeln & Becker 1998. Age at first breeding is set to 2 years (BTO facts); maximum age is set to 23 years (Euring database). The Dutch reproduction figures are deduced to lie around 0.7 and are used in the modelling (0.7 +/- 0.25) (Stienen *et al.* 2009).

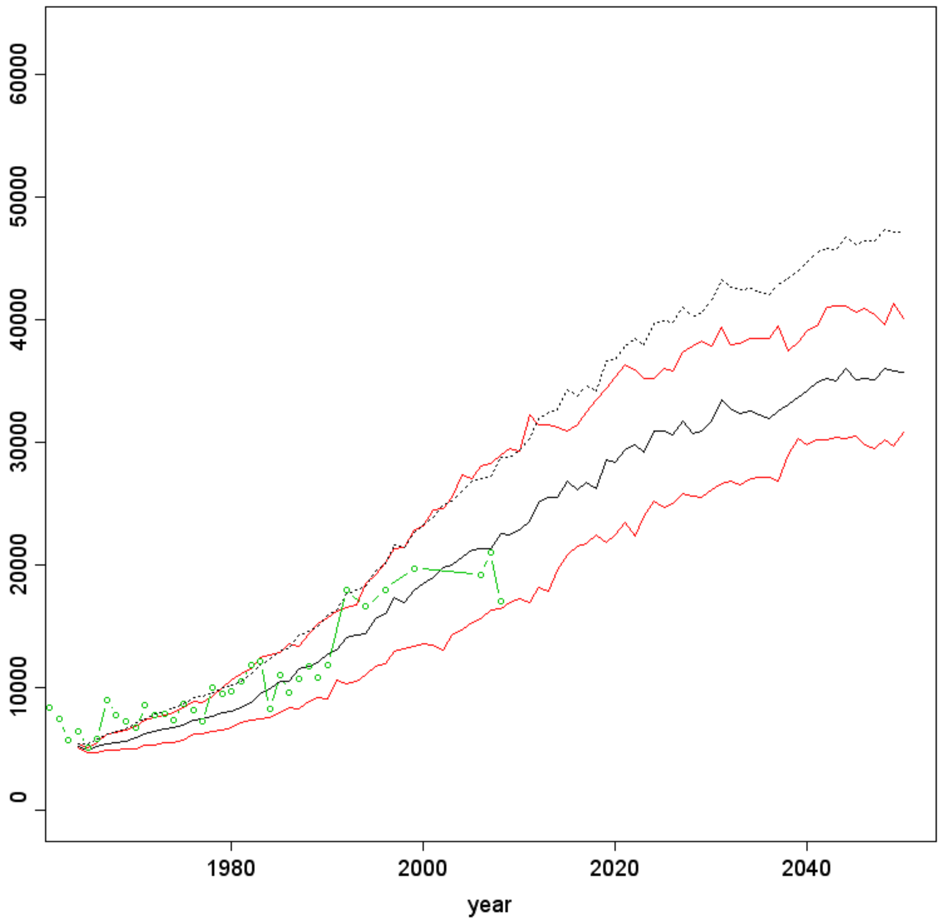


Figure 5.1.9 Counted (green circles and line) and modelled population trend for common tern (black line=median; red lines=25 and 75 percentile; black broken line= floater pairs 30%).

Effects of OWEZ extrapolated - cumulative effects on the selected population

The migratory movements of Dutch and international, more northerly populations are mainly nocturnal and occur most often at high altitudes. However, migratory movements can also occur during the day, but those are likely more concentrated along the coast, the area where also most foraging occurs for fuelling the long-distance flights

(Lensink *et. al.* 2002, Camphuysen & Van Dijk 1983). Therefore no cumulative negative effects are being expected for foreign populations and consequently we have assumed that all potential negative impacts of new offshore wind farms will affect only Dutch birds. As is clear from figure 5.1.10 the increase in the population, will not be stopped due to the numbers of victims calculated.

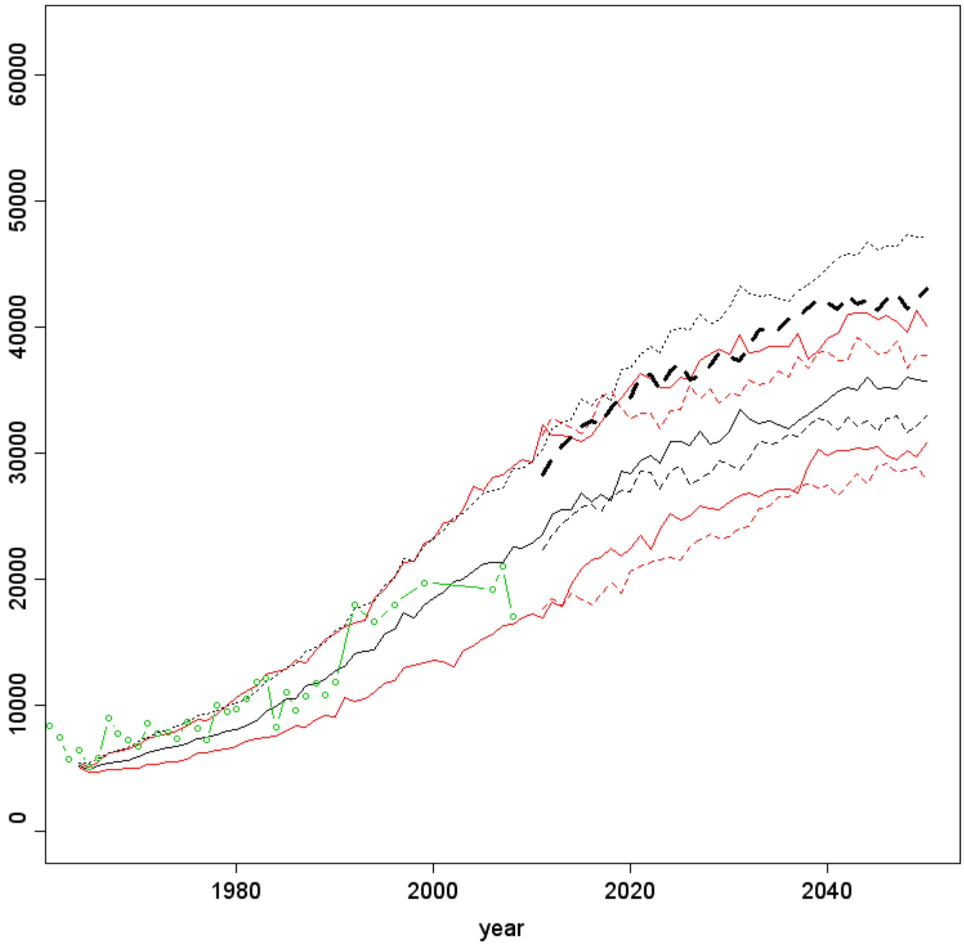


Figure 5.1.10 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for Sandwich tern with 30% floaters. With the effect of worst case increased mortality, number of yearly collisions based on band model calculation (n victims = 73) for a scenario with 11 wind farms like OWEZ in offshore waters.*

Little tern ***Sterna albifrons***

Occurrence and population trend in the Netherlands

Currently between 465 and 850 pairs of Little Tern breed in the Netherlands (SOVON). Most colonies are present in the Wadden Sea and the Delta areas, the latter holding around 300 pairs (Strucker *et al.* 2005). During the first half of the 20th century up to

1,000 pairs bred in the Netherlands, however, numbers dropped in the 1960s due to pollution. Since then, numbers have gradually recovered to their current levels and nowadays disturbance and vegetation succession pose the most threat to breeding colonies. Few birds are seen offshore, most being recorded within a few kilometres of the coast (Camphuysen & Leopold 1994). During the observations in and around OWEZ no observations of this species were made (Krijgsveld *et al.* 2011, Leopold *et al.* 2010). Birds from the Wadden Sea and international populations most likely migrate nocturnally at high altitudes or fly very coastal during the day (Lensink *et. al.* 2002, Camphuysen & Van Dijk 1983).

Effects of OWEZ extrapolated - cumulative effects on the selected population
With no observations of little terns in or near OWEZ, and the next round of new offshore wind farms not being developed within the distribution and flight range of little terns in the Dutch part of the North Sea (the species being very coastal), no cumulative negative effects are expected for this species.

5.1.2 Seabird species (mainly) breeding outside The Netherlands

Red-throated diver *Gavia stellata*

Occurrence and population trend in the Netherlands
Red-throated Divers are found in the coastal waters around the Netherlands, mostly within 20 km of the shore (Bijlsma *et al.* 2001). Fewer birds are found further offshore (Camphuysen & Leopold 1994). The species is most abundant between October and May, when passage birds pass the Dutch coast. Up to 10,000 individuals are estimated to winter in Dutch waters (Camphuysen & Leopold 1994). Favoured foraging areas include the channels between the Wadden Islands and off the Delta (Camphuysen & Leopold 1994; Poot *et al.* 2006). Up to 1,500 birds can occur offshore of the Delta during winter (Poot *et al.* 2006).

Effects of OWEZ extrapolated - cumulative effects on the selected population
With the lack of reliable (international and national) population census data, the uncertainty of the origin of birds passing through and wintering in the Dutch part of the North Sea, and the lack of reliable population parameters, prevented us to construct a reliable population model. Different levels of potential cumulative effects of multiple new offshore wind farms will therefore be calculated with the Potential Biological Removal approach, see section 5.2.

Shag *Phalacrocorax aristotelis*

Occurrence and population trend of non-breeding birds in the Netherlands
In the Netherlands, Shags are a non-breeding species that are almost exclusively confined to coastal and offshore areas. Although the species is recorded year round,

most records are from the end of August until the middle of February (Bijlsma *et al.* 2001). In general, less than a hundred individuals are recorded each year. Over half of all Shags nest within the UK, where the species is largely sedentary and most movements of any distance refer to post-breeding dispersal (Wernham *et al.* 2002).

Effects of OWEZ extrapolated - cumulative effects on the selected population
Only a few shags were once in a while present in or near OWEZ. Similar to cormorant potentially only positive cumulative effects are expected, as the development of more offshore wind farms will imply habitat expansion for this species due to the increased availability of resting and foraging opportunities.

Gannet *Morus bassanus*

Occurrence and population trend in the Netherlands
Largest numbers of gannets occur in Dutch waters during late summer and in autumn (Camphuysen & Leopold 1994). Smaller numbers occur during the rest of the year (Bijlsma *et al.* 2001). In general, gannets are widespread in low densities, although concentrations can occur in good feeding areas and sometimes at fishing trawls (Camphuysen & Leopold 1994; Bijlsma *et al.* 2001).

Key breeding colonies around the North Sea include Bass Rock, Bempton Cliffs and the Shetlands in Britain and Helgoland in Germany. The numbers of breeding pairs at these sites have increased over recent years. On Bass Rock, the largest colony, numbers had risen from around 8,000 in 1968-70 to over 44,000 in 1998-2000, whilst during the same period numbers had reached over 2,500 pairs from an initial 18 (Mitchel *et al.* 2004). Gannets first bred on Helgoland in 1991. Numbers have increased to over 125 in 2002 (Schneider 2002). Gannets rarely forage more than 150 km from the colony (Tasker *et al.* 1985). The known and modelled trend for the breeding population of gannets on Bass Rock is shown in figure 5.1.11. These colonies were selected as being examples of a large and a small colony, respectively, within the North Sea and potentially being affected in case multiple offshore wind farms will be realised in the Dutch part of the North Sea. Both models reflect the periods of growth witnessed at each colony. The colony at Bass Rock has shown a higher rate of growth since the 1970s, which is reflected in the model through a higher adult survival as an assumed consequence of reduced human related mortality (Nelson 1978). Much of the growth at Helgoland is driven by immigration, as reproduction figures are too low to explain the population trend (data O. Hüppop, Garthe 2010).

Outside the breeding season young gannets are known to winter further south than adults and are thought to remain south of the North Sea for their first one to two years and generally subsequently winter further north with increasing age (Wernham *et al.* 2002). Both birds that breed in Scotland and Norway pass through the North Sea during migration. Data logger studies of birds breeding on Bass Rock have revealed that around 1 in 5 Gannets wintered within the area of the North Sea and English

Channel. Furthermore, although many passed through the North Sea during post-breeding migration most favoured the route via the west of the UK during the return migration (Kubetzki *et al.* 2009). Gannets from colonies along the coast of Norway are also known to pass through the North Sea during migration (Barrett 1988).

In order to investigate the cumulative effects of multiple offshore wind farms in the Dutch part of the North Sea we have chosen to model a restricted population, in this case the one of Bass Rock in Scotland, being a large colony possibly supplying the largest part of the birds occurring along the Dutch coast or birds which stay longest in the area. Possibly the Bass Rock colony has the relatively largest proportion of wintering birds in the North Sea compared to e.g. Norwegian colonies, as northern breeding birds winter more south, like the largest part of the Bass Rock population does ((Kubetzki *et al.* 2009)).

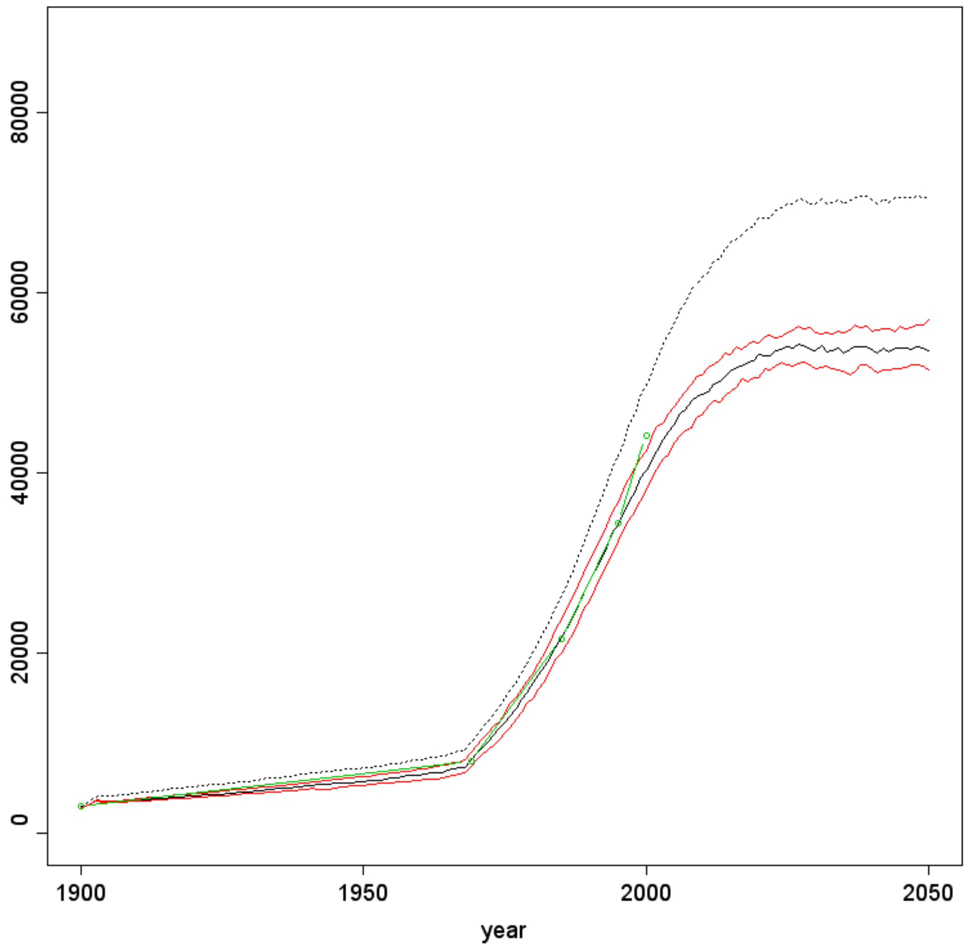


Figure 5.1.11 Counted (green circles and line) and modelled population trend for gannet on Bass Rock (black line=median; red lines=25 and 75 percentile; black broken line= floater pairs 30%).

Description and limitations of the constructed population mode

A model is made for the Bass Rock population, one of the most important and well-studied colonies in Scotland. Before 1965 adult survival is set to 0.907 (+/-0.002), after 1965 adult survival is set to 0.919 (+/- 0.002) (Wanless 2006). 1st year survival is set to 0.542 (+/-0.002), 2nd year survival is set to 0.779 (+/-0.002), 3rd year survival is 0.859 ((+/-0.002) and 4th year survival is set to 0.863 (+/-0.002) (Wanless 2006). Year of first breeding is set to 5; maximum age is 37 (Euring). Reproduction before 1965 is set to 0.72 (+/- 0.1) and reproduction after 1965 is set to 0.95 (+/- 0.1).

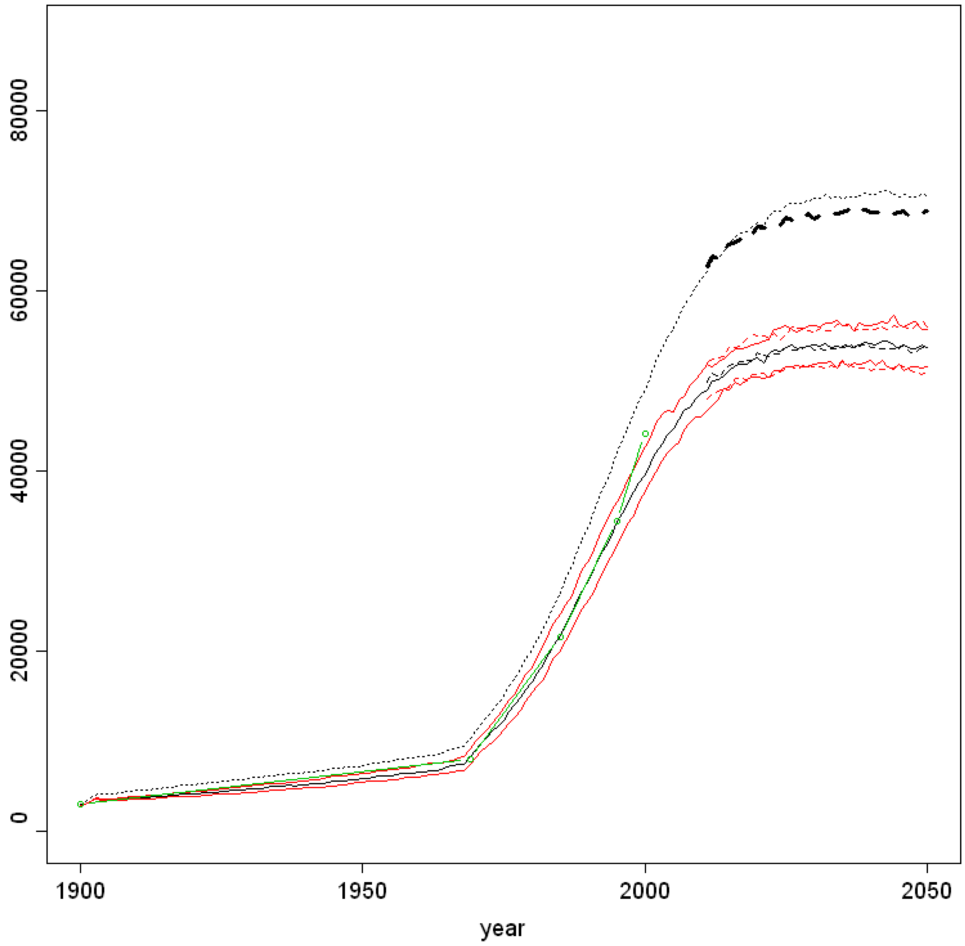


Figure 5.1.12 *Counted (green circles and line) and modelled population trend for the gannet in number of pairs (black line=median; red lines=25 and 75 percentile; black broken line= floater pairs 30%). With the effect of worst case increased mortality, number of yearly collisions based on band model calculation (n victims = 200) for a scenario with 11 wind farms like OWEZ in offshore waters.*

Effects of OWEZ extrapolated - cumulative effects on the selected population

The impact of the increased mortality due to the number of about 200 yearly collisions, based on band model calculation for 11 wind farms like OWEZ in offshore waters, is very limited. A calculation of the number of victims in order to let the population of

Bass Rock to decline amounts over 1,800 birds on a yearly basis. Clearly these calculations are to be regarded as a worst-case scenario, as now all potential victims are attributed to the Bass Rock population, although also gannets from other colonies occur in the Dutch part of the North Sea.

With the finding that densities in the OWEZ area are relatively very low compared to deeper water we are not able now to estimate any potential disturbance/displacement effects of scenarios of new offshore wind farms. Therefore we refer to the results of the zero growth models in appendix 3, to be regarded as a worst case scenario with total avoidance of new offshore wind farms with, through intense intraspecific competition, all displaced birds dying due to starvation (assuming a satisfied carrying capacity). Although this is a very unrealistic scenario this approach shows that a dramatic collapse of the population will occur at a level between 1,800 (0% floaters) and 2,800 (30% floaters) number of victims, implying a displacement of maximum over 250 birds per new offshore wind farm (see appendix 3). Future research around an offshore wind farm in higher density areas than OWEZ is necessary to obtain results in order to estimate population impacts of displacement on a larger scale for this species than now is possible.

Fulmar *Fulmaris glacialis*

Occurrence and population trend in the Netherlands

In the Dutch North Sea area, Fulmars occur in their highest numbers between August and October and February and March (Camphuysen & Leopold 1994). Numbers are thought to increase in the southern and eastern North Sea from late spring onwards in response to increased food and in summer due to movements of moulting birds (Wernham *et al.* 2002). During late summer and early August numbers reached around 114,000, while in late winter 111,000 are thought to occur in the Dutch North Sea (Camphuysen & Leopold 1994). Fulmars occur throughout the Dutch North Sea, but more frequently relatively far offshore (Arts 2010). Densities, however, are thought to be lowest all along the coast, from the north of the Wadden islands and off the southern Delta. The pattern of distribution is general patchy and variable. The species can often be found scavenging behind trawlers offshore but still is rarely seen in these situations close to the coast.

The nearest breeding colonies are in eastern England and Helgoland in Germany, although birds occurring in Dutch waters are likely also to include birds breeding in Norway, Scotland and beyond. Fulmars are thought to spend their entire first four to five years at sea during which time they can roam widely (Wernham *et al.* 2002).

Effects of OWEZ extrapolated - cumulative effects on the selected population

The numbers of fulmars observed near OWEZ both from the metmast (Krijgsveld *et al.* 2011) as well as from the ships (Leopold *et al.* 2010) were so low that no conclusion can be drawn on the potential effect of disturbance of wind turbines in this species.

OWEZ is definitely situated outside the core area for fulmars in the Dutch part of the North Sea. The only potential impact might be expected from habitat loss; by the effect that birds find wind farms that disturbing that a wind farm area is avoided. Also with the uncertainty of the origin of birds passing through and wintering in the Dutch part of the North Sea, and the likely vast area of breeding origin of birds occurring in Dutch waters, no population model was developed for this species. Different levels of potential cumulative effects of multiple new offshore wind farms will therefore only be evaluated with the Potential Biological Removal approach, see section 5.2.

Common Scoter *Melanitta nigra*

Occurrence and population trend of non-breeding birds in the Netherlands
Common scoters are widespread throughout the Dutch North Sea, although the species is most abundant close to shore. Birds occur year round, although numbers peak during the winter, typically February and March (Camphuysen & Leopold 1994). Common Scoters often occur in very large groups of up to 15,000 to 75,000 birds and exceptionally 125,000 (Camphuysen & Leopold 1994). Occasionally large groups can be present during the summer; these are typically moulting birds. Most common scoters in the Netherlands winter off the coasts of the Wadden Islands, but up to 25,000 were estimated off the coast of the delta in the early 1990s (Camphuysen & Leopold 1994). Most movements of common scoter are relatively close to shore of birds moving between these concentration areas.

Effects of OWEZ extrapolated - cumulative effects on the selected population
With the lack of reliable population census data, the uncertainty of the origin of birds passing through and wintering in the Dutch part of the North Sea, and the lack of reliable population parameters, prevented us to construct a reliable population model. However, in line with relatively low numbers of scoters in the coastal area near OWEZ, and hardly no observations of scoters in or near OWEZ, and the next round of new offshore wind farms not being developed within the distribution range of wintering scoters in the Dutch part of the North Sea, no cumulative negative effects of habitat loss are expected for this species. Furthermore, the species showed a strong avoidance behaviour during the day, which is likely to occur at night as well which results in very low risk of collisions in this species (see table 4.1.3) and therefore also low risks of cumulative negative effects.

Great Skua *Catharacta skua*

Occurrence and population trend in Scotland
Great skuas are widespread, yet in relatively low numbers, around the Dutch North Sea, particularly in autumn (Camphuysen & Leopold 1994). The species is found both offshore and along the coast and at time at trawlers (Bijlsma *et al.* 2004). During migration birds are less frequently seen within 2-5 km of the coast and birds in the

southern North Sea are most likely from colonies in the east of Scotland (Wernham *et al.* 2002).

The world population of great skuas was estimated at 16,000 in 1999-2000, of which the majority bred in Scotland and Iceland (Mitchell *et al.* 2004). The numbers breeding in Scotland have increased over the past forty years and reached almost 10,000 in 1999-2000. Relatively small numbers breed in Norway and it is likely that most of the birds present in the North Sea originate from Scottish breeding populations. The known and modelled trends for the breeding population of great skua in Scotland are shown in figure 5.1.13. The model reflects the period of growth witnessed since the 1970s.

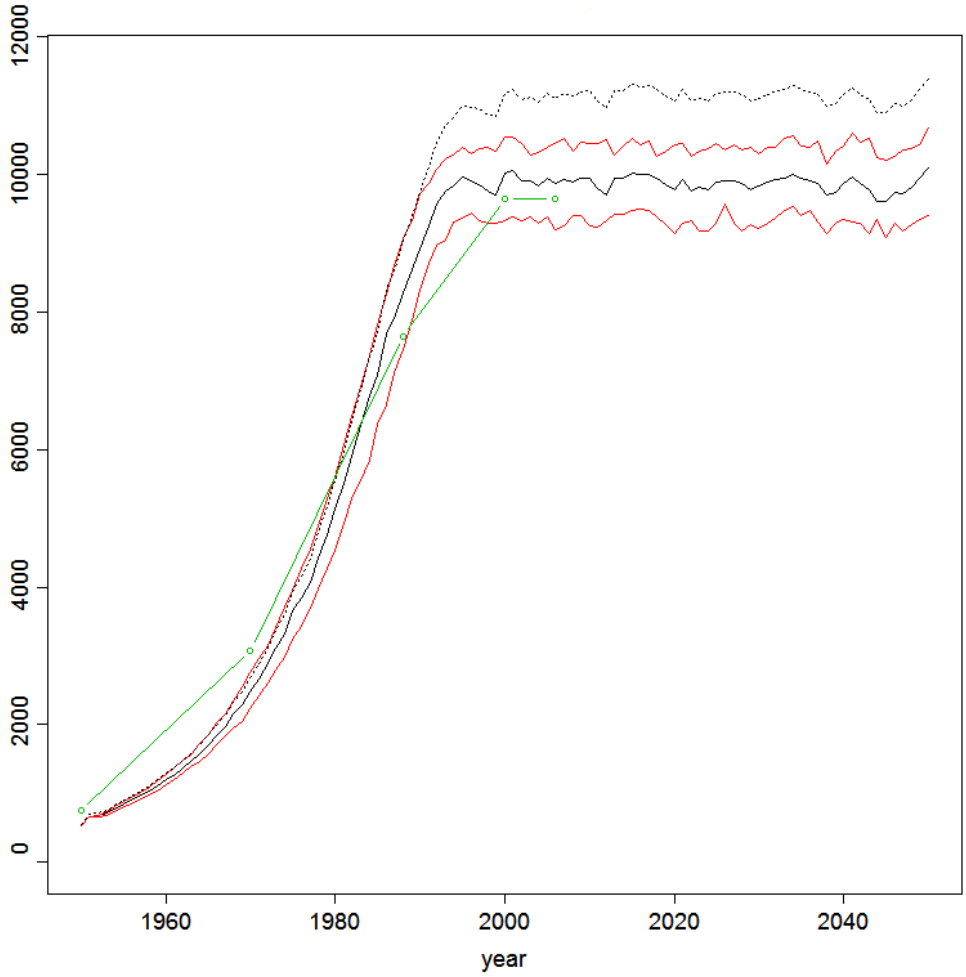


Figure 5.1.13 Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for great skua, with an assumed 10% floater population (dotted line).

Description and limitations of the constructed population model
Models are built for the Scottisch breeding population. Adult survival is set at 0.888 (+/- 0.006), 1st year survival at 0.800 (+/- 0.006) Ratcliff *et al.* (2002). Sub adult yearly survival (up to year 7-8 is set to 0.85 (+/- 0.006). Age at first breeding is set to 7.5 (BTO bird facts) and maximum age is set at 32 year (Euring). Reproduction on Shetlands and Orkneys varies between 0 – 2.0 (2005) and 0 – 1.33 (2006).

Effects of OWEZ extrapolated - cumulative effects on the selected population
The 0-growth-model yield a total of 150 collision victims as being the level after which a real population decline will commence due to the development of multiple offshore wind farms (graph not depicted as the figure hardly differs from figure above). The 40 collisions victims based on a calculation for scenario 2 is, therefore, well below the limit at which a serious impact on the population occurs.

With the finding that densities in the OWEZ area are relatively very low compared to deeper water (Leopold *et al.* 2010, Poot *et al.* 2010) we are not able to estimate any potential disturbance/displacement effects of scenarios of new offshore wind farms. Therefore, we refer to the results of the zero growth models in appendix 3, to be regarded as a worst case scenario with total avoidance of new offshore wind farms with, through intense intra-specific competition, all displaced birds dying due to starvation (assuming a satisfied carrying capacity). Although this is a very unrealistic scenario this approach shows that a collapse of the population will occur at a level of 250 (30% floaters) of displaced and starved victims, implying a displacement of maximum over 20 birds per new offshore wind farm (see appendix 3).

The total of 150 annual collision victims is well above the level at which a decline in the population will be seen due to the development of multiple offshore wind farms (graph not depicted as the figure hardly differs from figure above). The 40 collisions victims based on a calculation for multiple wind farms in offshore waters is, therefore, well below the limit at which a serious impact on the population occurs (figure 5.1.14).

Future research around an offshore wind farm in higher density areas than OWEZ is necessary to yield results in order to estimate population impacts of displacement on a larger scale for this species than possible now.

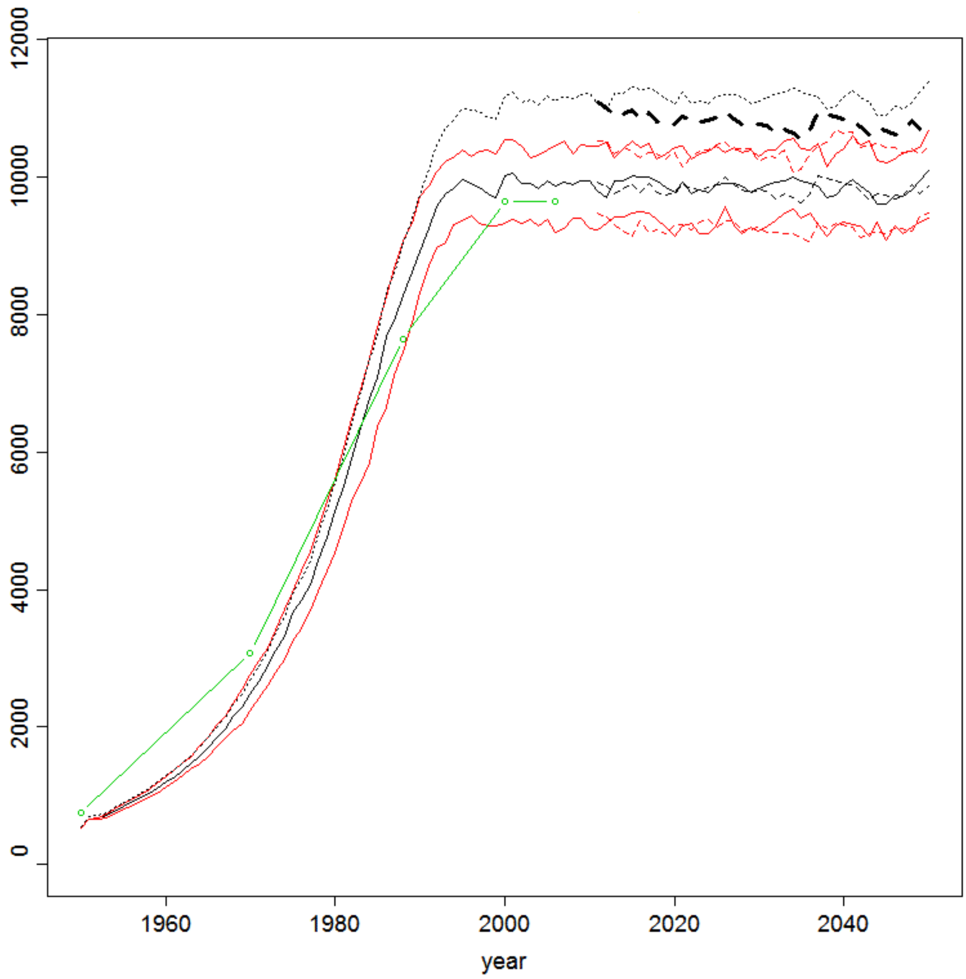


Figure 5.1.14 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for great skua, with an assumed 10% floater population (dotted line). Also depicted is are the lines with the effect of worst case increased mortality due to yearly collisions based on band model calculation (n victims = 40) for a scenario with 11 wind farms like OWEZ in offshore waters.*

Great black-backed gull *Larus marinus*

Occurrence and population trends

The first breeding record of Great Black-backed Gull for the Netherlands was in 1993 when one pair bred in the Delta (Bijlsma *et al.* 2001). Since then, the species has bred annually in this area but in low numbers (maximum 12 pairs) (Strucker 2005) and also in low numbers in the Wadden Sea and IJsselmeer (Bijlsma *et al.* 2001). The estimated breeding population in 2007 was between 25-35 pairs (Van Dijk *et al.* 2009).

In the Dutch North Sea zone, Great Black-backed Gulls occur predominantly as a non-breeding species and mainly are present between August and May (Camphuysen &

Leopold 1994). Peak numbers occur during autumn when up to 60,000-90,000 was estimated to be in the region (Bijlsma *et al.* 2001).

Effects of OWEZ extrapolated - cumulative effects on the selected population
The origin of the Dutch wintering population at sea consists of birds coming from a very wide range of northern breeding areas, covering mainly the coasts of Scandinavia and northern Russia. This breeding population consist of a total of more than 300,000 breeding pairs. As population trends and population dynamic parameters are uncertain for this vast area, different levels of potential cumulative effects of multiple new offshore wind farms will be calculated with the Potential Biological Removal approach, see section 5.2.

Kittiwake *Rissa tridactyla*

Occurrence and population trend
Kittiwakes are widespread in the Dutch North Sea throughout the year, although are present in higher numbers during autumn and winter. Numbers can peak at 53,000 during autumn (Camphuysen & Leopold 1994). Kittiwakes can be found throughout Dutch North Sea waters, although favour offshore areas and can be seen scavenging at trawls (Camphuysen & Leopold 1994). Since a couple years the species breeds in small numbers in Dutch territorial waters at platforms (Camphuysen & de Vreeze 2005). The then nearest breeding colonies are in eastern and North-eastern England and Helgoland in Germany. The largest colonies around the North Sea are in Scotland and Norway (Mitchell *et al.* 2004). Based on the lack of a substantial increase in wintering numbers of kittiwakes in the Dutch part of the North Sea, while breeding numbers in the UK showed a threefold increase, it is concluded that the breeding origin of birds wintering in Dutch waters is much wider than the British Isles alone (Bijlsma *et al.* 2001). However, in the period 1992-2004 an increase was occurring in the Dutch part of the North Sea, indicating a relation with UK breeding bird numbers. After 2004 the onset of a decrease occurred (Arts 2010), completely in line with the dramatic decline of East Scottish colonies (JNCC 2010, Fredriksen *et al.* 2004) (figure 5.1.15).

Description and limitations of the constructed population model
The model was built on the Scottish/ Eastern UK breeding population. Adult survival is set at 0.941 (+/- 0.01); yearly sub adult survival is set at 0.79 (+/- 0.01) (Frederiksen *et al.* 2004). Year at 1st breeding is set 4 and maximum age is 28 years (BTO bird facts). Reproduction of breeders in the North Sea basin is fluctuating between 1.24 and 0.02, and shows a negative trend (Isle of May, Frederiksen *et al.* 2004). The decline in population is modelled by adjusting the reproduction in time. Before 1988 0.26 fledged per pair, after 1988 0.12 fledged per pair.

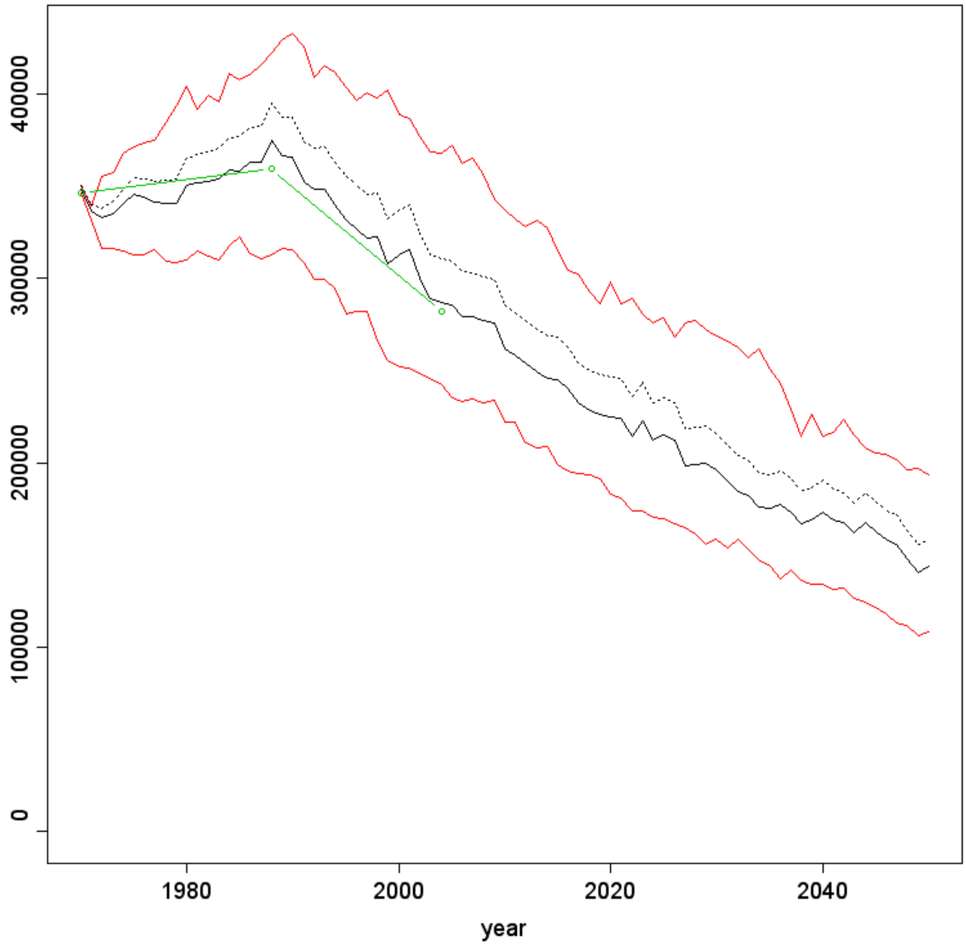


Figure 5.1.15 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for kittiwake in Scotland, with an assumed 10% floater population (fine dotted line).*

Effects of OWEZ extrapolated - cumulative effects on the selected population

The numbers of calculated victims for multiple wind farms in near-shore waters are higher than for multiple wind farms in offshore waters. This is surprising, as we would expect that the proportion of kittiwakes in waters deeper than 20 m would be larger. Krijgsveld *et al.* (2011) however found indications that Kittiwakes possibly were attracted to OWEZ (as they are to platforms), so this might be an artefact due to the numbers observed around OWEZ in relation to other seabird species. Nevertheless, the maximum number of collision victims does contribute to the decline as is currently going on (see figure 5.1.16), with the ecological changes happening in the food chain in the North Sea being the main driving force for the decline (Fredriksen *et al.* 2004).

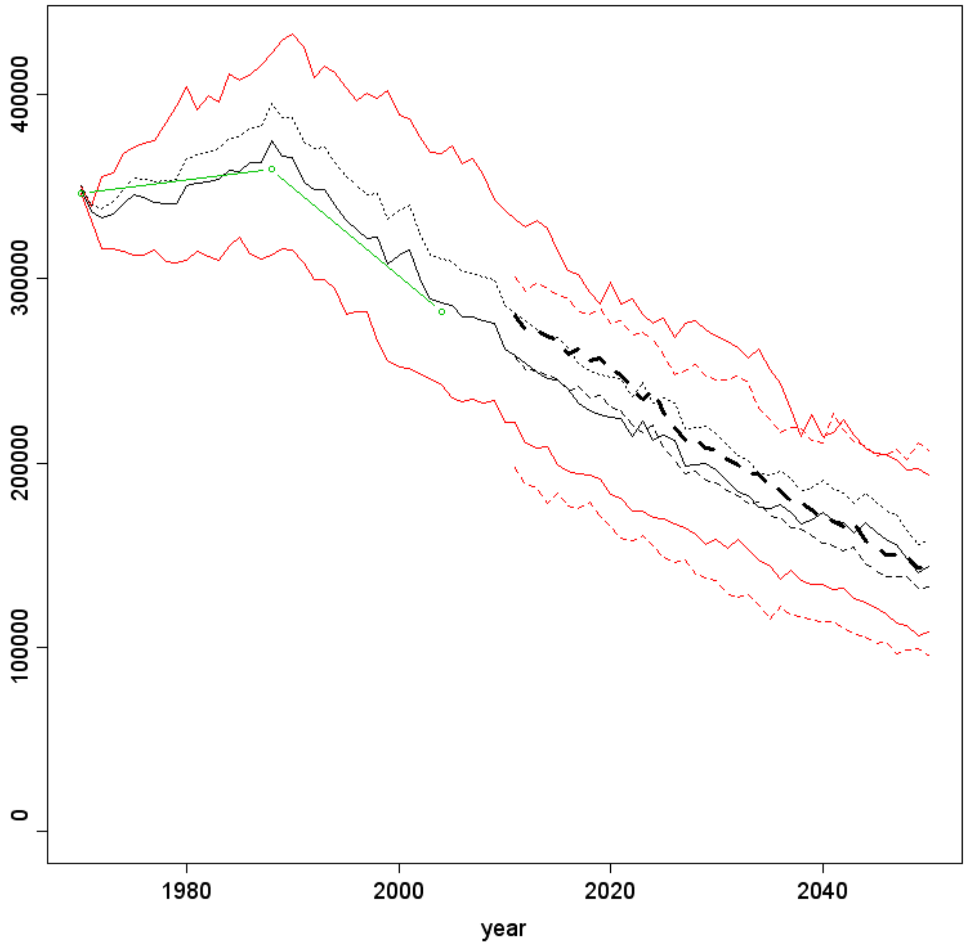


Figure 5.1.16 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for kittiwake on the east coast of Scotland, with an assumed 10% floater population (fine dotted line).*

Common gull *Larus canus*

Occurrence and population trend in the Netherlands

Common Gulls first bred in the Netherlands in the early 1900s. The population increased steadily and reached around 1,000 pairs in the 1960s. Shortly afterwards numbers increased exponentially and peaked at around 11,500 pairs in the mid-1980s. Since then, the population declined to around 6,000-7,900 in the mid- to late-1990s. This decline was partly attributed to predation by foxes (Bijlsma *et al.* 2001). In 2007, breeding numbers were estimated at 4,300 pairs (Van Dijk *et al.* 2009). Most of the population is restricted to the coast, the majority breeding colonies being nowadays around the Wadden Sea, North-Holland and the Delta; the latter holding between 600-700 pairs in recent years (Strucker 2005).

The species is most abundant offshore during winter (Bijlsma *et al.* 2001). Around 60,800 Common Gulls occur in the Dutch North Sea, most of these birds along the coastal zone (Camphuysen & Leopold 1994). In summer breeding birds mainly feed in coastal waters, inland waters, intertidal areas and terrestrial. The total wintering population in the Netherlands is much larger with up to 190.000 birds staying inland (Bijlsma *et al.* 2001).

Effects of OWEZ extrapolated - cumulative effects on the selected population
Because of the very coastal behaviour of the species during breeding and the relatively small breeding population of the Netherlands with nowadays a very restricted distribution the risk of population effects of multiple new offshore wind farms is negligible. On the other hand, the origin of the Dutch wintering population at sea consists of birds coming from a very wide range of northern breeding areas, covering Scandinavia and large parts of Russia. This breeding population consists of a total of more than one million breeding pairs. As population trends and population dynamic parameters are uncertain for this vast area, different levels of potential cumulative effects of multiple new offshore wind farms will be calculated with the Potential Biological Removal approach, see section 5.2.

Little gull *Larus minutus*

Occurrence and population trend in the Netherlands
Little gulls first bred in the Netherlands during the early 1940s. The species has been an irregular breeder since then with very few breeding attempts recorded until the mid-1970s, when a maximum of 61 pairs bred in the Lauwersmeer (Bijlsma *et al.* 2001). Breeding remains sporadic with only two pairs recorded in 2007 (Van Dijk *et al.* 2009). Most breeding attempts are in the Wadden Sea or Delta areas.

Little Gulls are most common in the Dutch North Sea during the autumn migration period in October and November. Up to 4,500 are present, during winter mostly in the coastal zone (Camphuysen & Leopold 1994). Numbers increase again during spring migration although this is mainly limited from the end of April to early May.

Effects of OWEZ extrapolated - cumulative effects on the selected population
With the lack of reliable population census data, the uncertainty of the origin of birds passing through the Dutch part of the North Sea, and the lack of reliable population parameters, prevented us to construct a reliable population model. With a couple of positive observations of flocks of little gulls foraging within the OWEZ, potentially positive cumulative effects could be expected for this species. However, migrating little gulls could potentially collide with the turbines during migration. As population trends and population dynamic parameters are uncertain for this vast area, different levels of potential cumulative effects by collisions of multiple new offshore wind farms will be calculated with the Potential Biological Removal approach, see section 5.2.

Guillemot *Uria aalge*

Occurrence and population trend on the east coast of Scotland

The species does not breed in the Netherlands. Numbers generally build up during late summer as breeding birds arrive with their offspring. An estimated number of 15,000-45,000 chicks can be present in Dutch waters during this time (Bijlsma *et al.* 2001). Peak numbers occur during October and November when an estimated 240,000 individuals may be present in the southern North Sea (Camphuysen & Leopold 1994). Numbers gradually decrease throughout the winter, although winter storms may drive birds into the Dutch North Sea region again. Most birds occur in the offshore zone with fewer birds nearer to the coast.

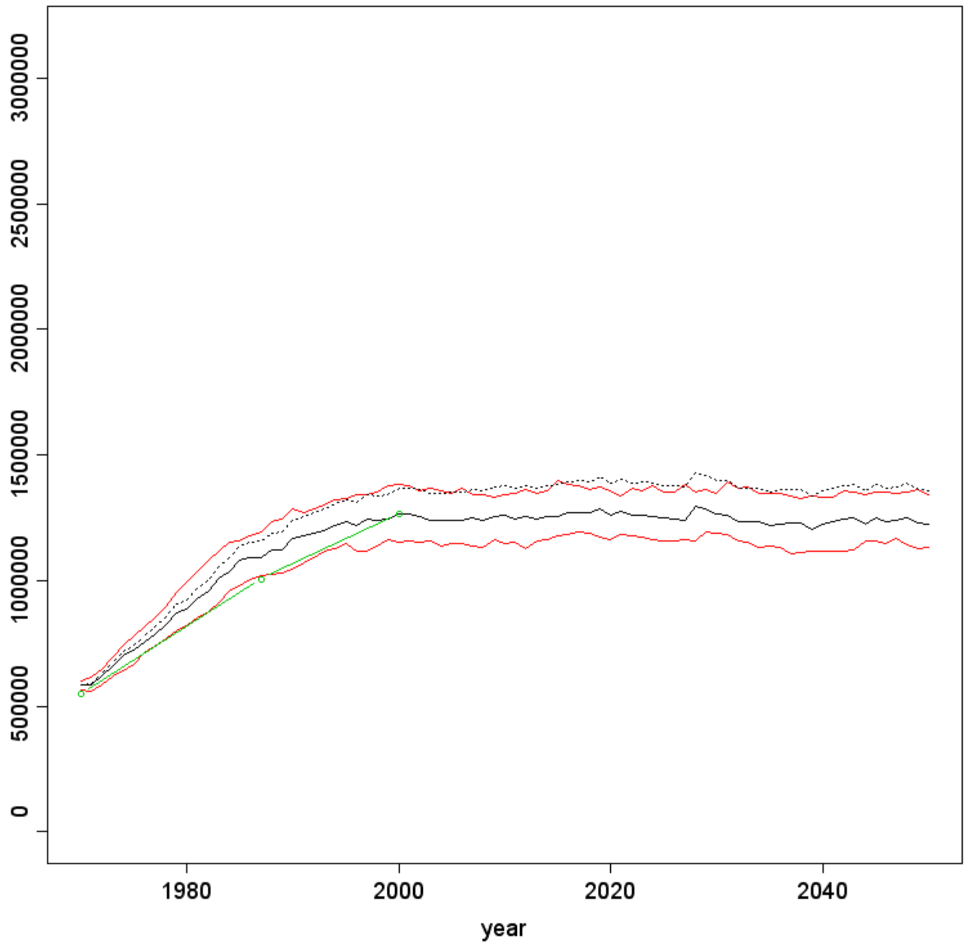


Figure 5.1.17 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for guillemot in Scotland, with an assumed 10% floater population (fine dotted line).*

The Dutch North Sea is of importance to birds breeding in eastern England and southeastern Scotland, particularly in late summer and early autumn. In addition, birds from Western Scotland are also known to occur in Dutch waters during winter

(Camphuysen & Leopold 1994). In order to investigate the cumulative effects of multiple offshore wind farms in the Dutch part of the North Sea we have chosen to model a restricted population, in this case the population of Scotland, being a wide area covering the largest part of the region from which ringing recoveries have been made of dead birds found along the Dutch coast.

Description and limitations of the constructed population model

A population model was built for the Scottish population. Although the information on the population trend of this region is limited to three data points, we have constructed a population model as was clear that a steady population increase is still going on (JNCC 2010). We have assumed a carrying capacity of the population at the level of the population estimate of 2002 (Mitchel *et al.* 2004). Adult survival set at 0.946 (+/- 0.05), yearly sub adult survival set at 0.89 (+/- 0.005) (Harris *et al.* 2000). Age at first breeding is set at 4.5 years (BTO birdfacts, Russell 1999), maximum age 42 years (Euring). Reproduction is set at 0.47 (+/-0.06) (Mavor 2008).

Effects of OWEZ extrapolated - cumulative effects on the selected population

With the finding that densities in the OWEZ area are relatively very low compared to deeper water (Leopold *et al.* 2010) we are not able to estimate any potential disturbance/displacement effects of scenarios of new offshore wind farms. Therefore we refer to the results of the zero growth models in appendix 3, to be regarded as a worst case scenario with total avoidance of new offshore wind farms with, through intense intra-specific competition, all displaced birds dying due to starvation (assuming a satisfied carrying capacity). Although this is a very unrealistic scenario this approach shows that a collapse of the population will occur at a level between 4,000 (0% floaters) and 38,000 (30% floaters) number of victims, implying a displacement of maximum over 3,400 birds per new offshore wind farm (see appendix 3). Future research around an offshore wind farm in higher density areas than OWEZ is necessary to yield results in order to estimate population impacts of displacement on a larger scale for this species than now possible.

Razorbill *Alco torda*

Occurrence and population trend on the east coast of Scotland

The nearest breeding colonies are in northeast England and Highland in Germany. However, many birds from breeding colonies in western Britain are also known to winter in the Dutch North Sea (Campuses & Leopold 1994), although in smaller numbers (Wenham *et al.* 2002). It is estimated that around 44,000 razorbills are present within Dutch North Sea waters during the winter, most between February and March (Campuses & Leopold 1994). Most birds can be found offshore but higher densities occur to the northwest of the Warden Islands. Unlike guillemots, few young are seen in Dutch waters (Campuses & Leopold 1994). In order to investigate the cumulative effects of multiple offshore wind farms in the Dutch part of the North Sea we have chosen to model a restricted population, in this case the population of

Scotland, being a wide area covering the largest part of the region from which ringing recoveries have been made of dead birds found along the Dutch coast.

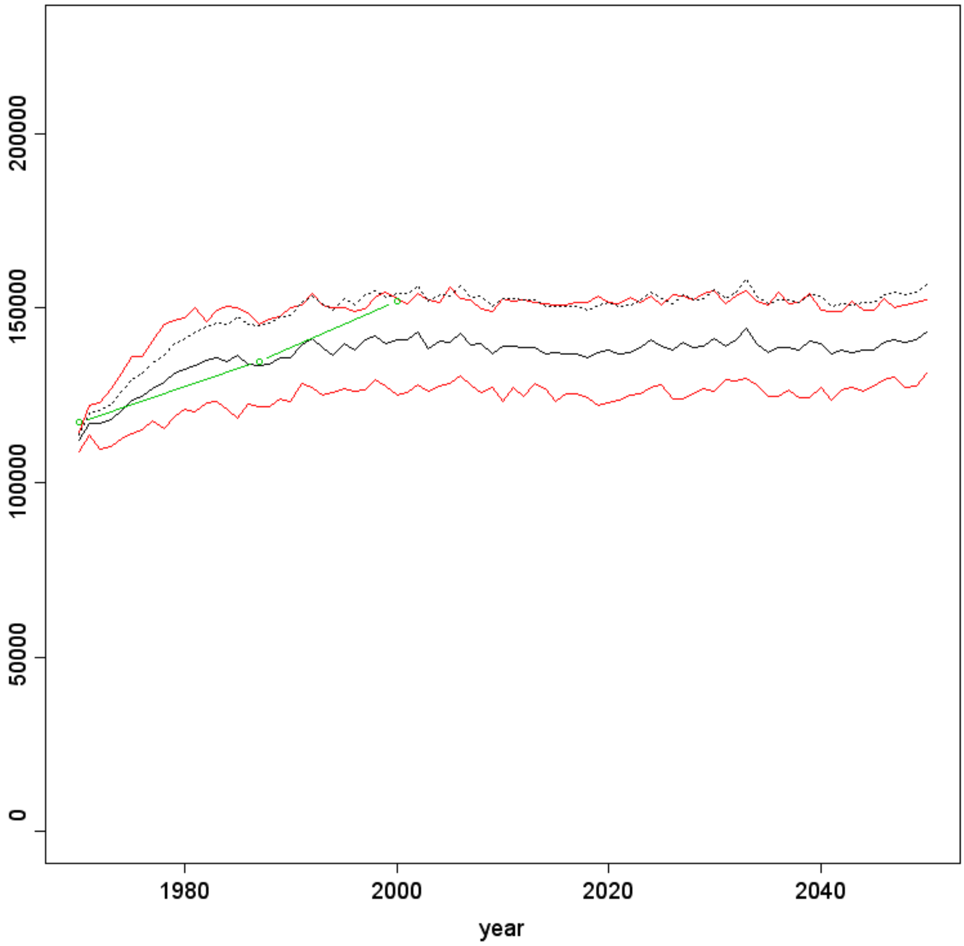


Figure 5.1.18 *Counted (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) population trend for razorbill on the east coast of Scotland, with an assumed 10% floater population (fine dotted line).*

Effects of OWEZ extrapolated - cumulative effects on the selected population

With the finding that densities in the OWEZ area are relatively very low compared to deeper water we are not able to estimate any potential disturbance/displacement effects of scenarios of new offshore wind farms. Therefore we refer to the results of the zero growth models in appendix 3, to be regarded as a worst case scenario with total avoidance of new offshore wind farms with, through intense intra-specific competition, all displaced birds dying due to starvation (assuming a satisfied carrying capacity). Although this is a very unrealistic scenario this approach shows that a collapse of the population will occur at a level between 300 (0% floaters) and 5,900 (30% floaters) number of victims, implying a displacement of maximum over 500 birds per new offshore wind farm (see appendix 3). Future research around an offshore wind farm in higher density areas than OWEZ is necessary to yield results in order to estimate population impacts of displacement on a larger scale for this species than now possible.

Puffin *Fratercula arctica*

Occurrence and population trend as winter visitor in the Netherlands
Within the Netherlands, Puffins are mostly found in the northwest region of the Dutch North Sea, with fewer birds present closer to the coast (Bijlsma *et al.* 2001). Up to 4,000 individuals are thought to be present during the period from October to May, with numbers peaking at 7,000 in February and March (Bijlsma *et al.* 2001).

Effects of OWEZ extrapolated - cumulative effects on the selected population
With no observations of puffins in or near OWEZ, and the next round of new offshore wind farms not being developed within the distribution range of wintering puffins in the Dutch part of the North Sea, no cumulative negative effects under the condition of current plans for offshore wind farm developments are being expected for this species.

5.1.3 Migrant species (mainly) breeding outside The Netherlands

Bewick's Swan *Cygnus bewickii*

Occurrence and population trend of wintering birds in the Netherlands
Bewick's swans winter in the Netherlands with the first birds arriving in the Lauwersmeer and Randmeer from Siberia at the end of September or beginning of October and most birds arrive in the second half of October (Voslamber *et al.* 2004). Birds start to leave at the end of February, or in cold winters as late as March and in mild winters as early as December (Bijlsma *et al.* 2001; Hustings *et al.* 2008).

In the mid-1970s the number of Bewick's swans wintering in the Netherlands increased from around 2,000 to over 6,000. This increase continued into the mid-1990s, when numbers peaked at 17,000 to 19,000 although during this time there were several periods with lower totals, most noticeably during the late 1980s (Bijlsma *et al.* 2001). Numbers have gradually declined since the turn of the century and more recently a maximum of 11,000 were counted in November 2006 (Hustings *et al.* 2008). The majority of birds can be found inland throughout the country although the areas south of the IJsselmeer have become an important staging area for this species (Bijlsma *et al.* 2001). Up to 7,500 Bewick's swans cross the North Sea to winter in eastern England. However, the numbers of birds making this journey have fallen to around 3,500 in recent years possibly as a response to milder winters (Austin *et al.* 2008).

The world population of Bewick's swan is currently estimated at around 20,000 birds (Delany & Scott 2006). The known and modelled trends for the population of Bewick's swan and for those that breed are shown in figure 5.1.19. The model reflects the gradual increase in the population over the past decades and the recent decline.

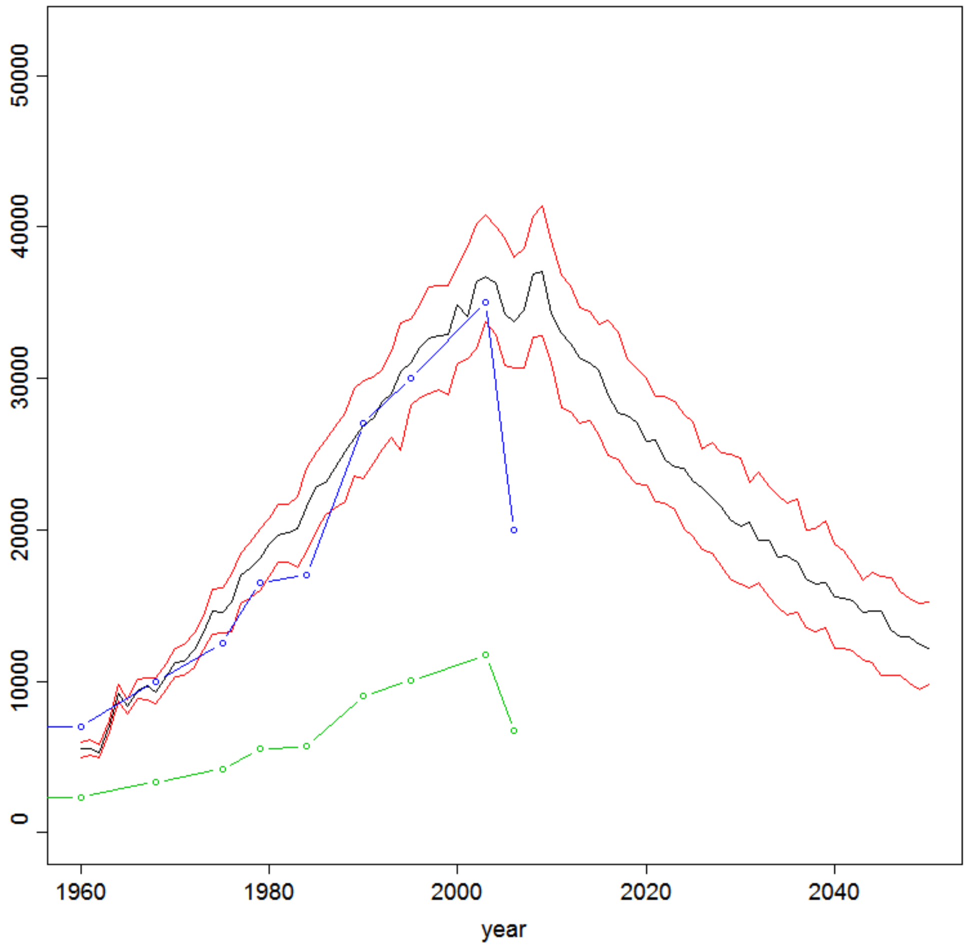


Figure 5.1.19 Counted breeding (green circles and line) and total (blue circles and line), and modelled (black line=median; red lines=25 and 75 percentile) population trend for Bewick's swan.

Description and limitations of the constructed population model

The trend of the population is based on the winter counts of the world population (mainly Netherlands and United Kingdom. For the modelling a percentage of 67% breeders in the population is used (Rees 2006). Adult yearly survival is set at 0.849 (+/- 0.079) (Scott 1990) and sub adult survival for the first 2 years at 0.66 (+/- 0.01), yearly values 0.812 (Rees, 2006) and for years 3 and 4 at a yearly survival of 0.822 (+/- 0.1) Rees (2006). Age at first breeding is 4 years (BTO birdfacts). Maximum lifespan is 24 years (Euring). Reproduction figures between 1964 and 2009 taken from UK and Dutch counts in wintering areas (Collier *et al.* 2008 in series, SOVON in series, Dirksen 1991, Evans 1979, Thijssen 2010). It is clear that recently there is a severe decline in numbers, however last available reproduction figures in 2009 were still relatively high. For the time period after 2009 and in future reproduction was set to 1 juv/pair. It is clear that with this reproduction figures the model does not properly describes the current situation; here our knowledge is too short to understand this phenomenon.

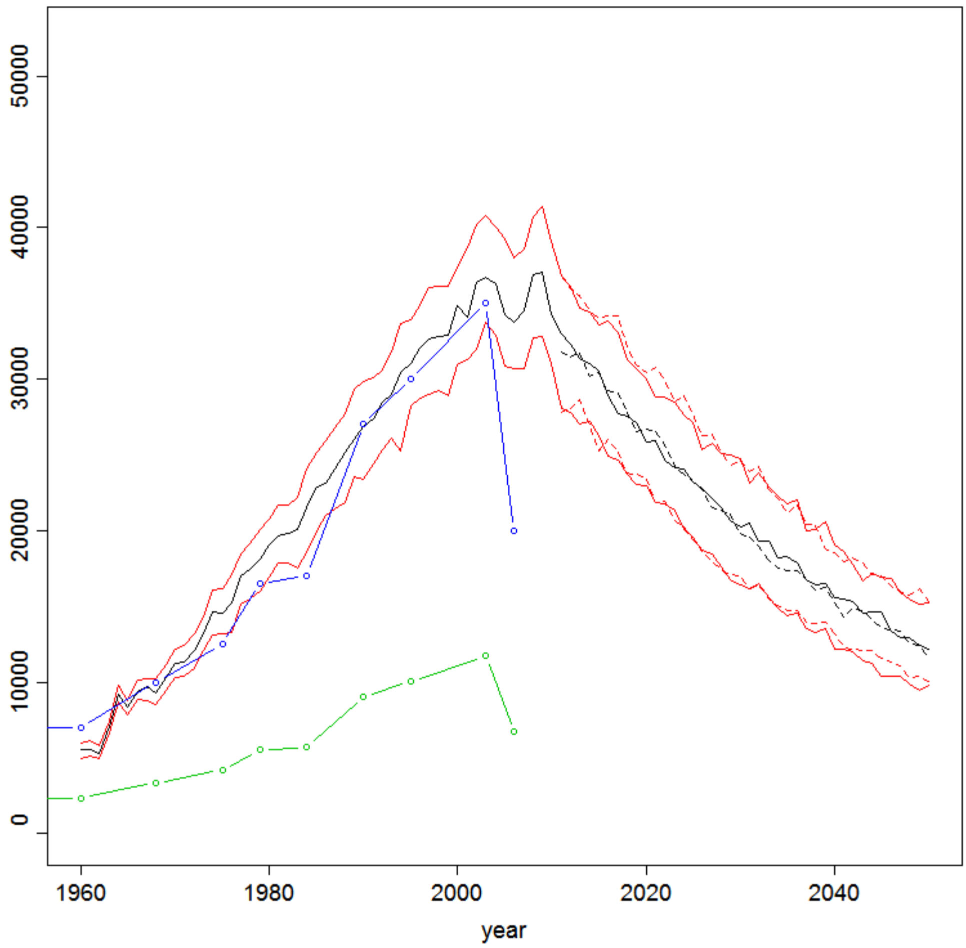


Figure 5.1.20 *Counted breeding (green circles and line) and total (blue circles and line), and modelled (black line=median; red lines=25 and 75 percentile) population trend for Bewick's swan. With the effect of worst case increased mortality, number of yearly collisions based on band model calculation (n victims = 10) for a scenario with 11 wind farms like OWEZ in offshore waters.*

Effects of OWEZ extrapolated - cumulative effects on the selected population

During the observations from OWEZ at the far distance only one unidentified swan has been observed in December, which flew above OWEZ. The chance that this solitary swan was a Bewick's is fairly high, as sea crossings from the Dutch coast to the UK by whooper and mute swans are relatively rare. As indicated above, sea crossings of birds wintering in the UK occur on a yearly basis with up to a few thousand birds wintering in the UK. Potentially most birds fly at altitudes above rotor height or when at lower level shows avoidance behaviours similar as observed in geese in and around OWEZ.

Estimated potential collision victims in Bewick's swans are calculated based on the fluxes and avoidance behaviour determined for geese, see further Krijgsveld *et al.* (2011). When using geese figures, the number of victims estimated for OWEZ and multiple new offshore wind farms yield hardly any population effect based on the population model at hand. The Potential Biological Removal approach yield that the number of 20 victims on a yearly basis is still within the level that an immediate recovery of the population is possible.

Brent Goose *Branta bernicla*

Occurrence and population trend of wintering birds in the Netherlands
Brent geese are winter and passage visitors in the Netherlands. Up to 63,000 individuals occur in the country during the winter and up to 120,000 on passage to the UK and France (Bijlsma *et al.* 2001; Hustings *et al.* 2008). In the Netherlands, birds are predominantly from the western Siberian breeding population, which begin to arrive from September onwards. Peak numbers are generally recorded during the end of April and the beginning of May as birds from wintering areas further south and west pass through the Netherlands (Wernham *et al.* 2002; Voslamber *et al.* 2004).

The most important areas are the Delta and Wadden Sea and their surrounding polders. During winter up to 90% of the Dutch wintering population may be found around the Wadden Sea and 50% of the total in Friesland (Voslamber *et al.* 2004). In general, the number of birds wintering in the Netherlands has increased, however, numbers can vary annually depending on weather conditions. During the 1960s and early 1970s maximum numbers were below 10,000. Since the mid-1970s maximum numbers increased steadily to around 120,000 in the mid-1990s but have fallen in recent years (Bijlsma *et al.* 2001), perhaps reflecting the change in the entire population (Banks *et al.* 2006).

Description and limitations of the constructed population model
The model constructed is based on the late winter population counted in the Netherlands. Presumed percentage of breeders in population amounts 70 %. Adult survival and standard deviation 0.9 +/- 0.036 from Sedinger *et al.* (2002). Sub adult survival based on Ebbinge (1992) and adjusted to 0.6 +/- 0.01. Age at first breeding is set at 2 years (BTO bird facts). Maximum age is set at 28 years (Euring). Reproduction is 1.75 (+/- 0.39) juv/pair from Collier *et al.* (2008) and reports on Waterbirds in the UK (03/04, 04/05, 05/06, 07/07).

Effects of OWEZ extrapolated - cumulative effects on the selected population
Based on the research in and around OWEZ the strong avoidance behaviour and low fluxes at rotor height of brent geese means that the numbers colliding with offshore wind farms is very low. The impact of these low numbers therefore on the population is negligible.

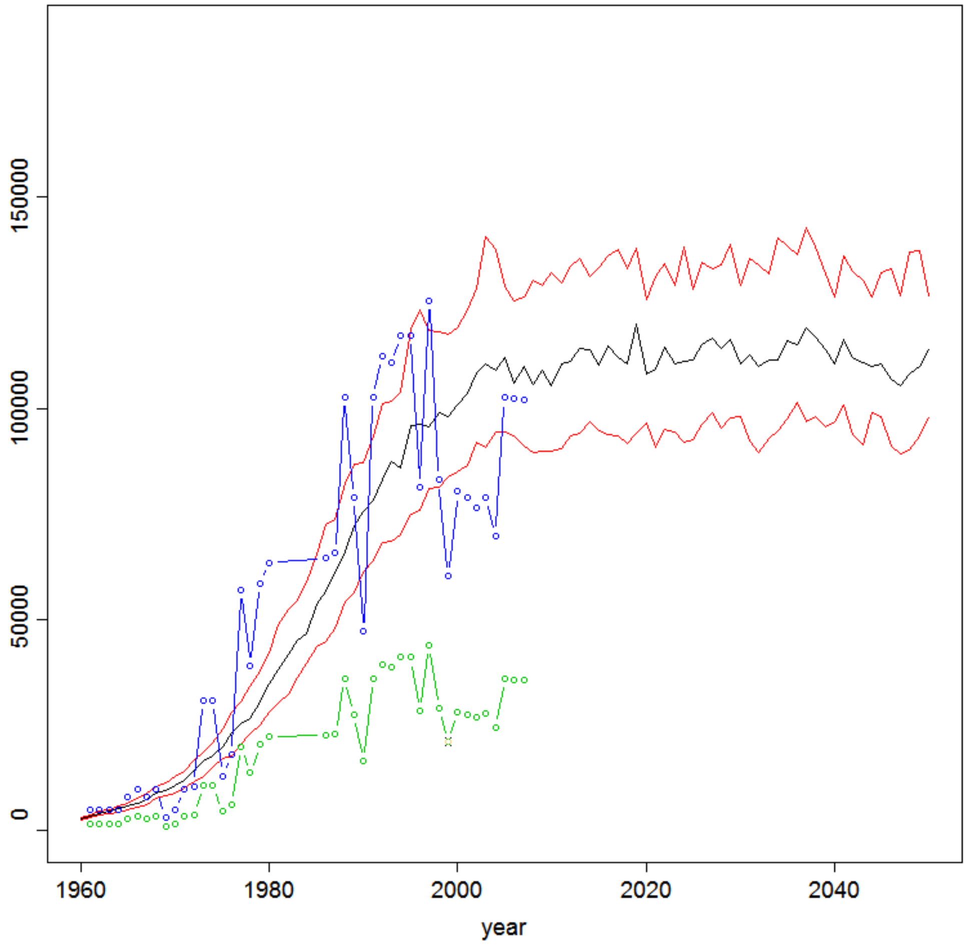


Figure 5.1.21 Counted total (blue circles and line) and modelled (black line=median; red lines=25 and 75 percentile) total population trend for brent geese. Fraction of breeding birds depicted with green circles and line.

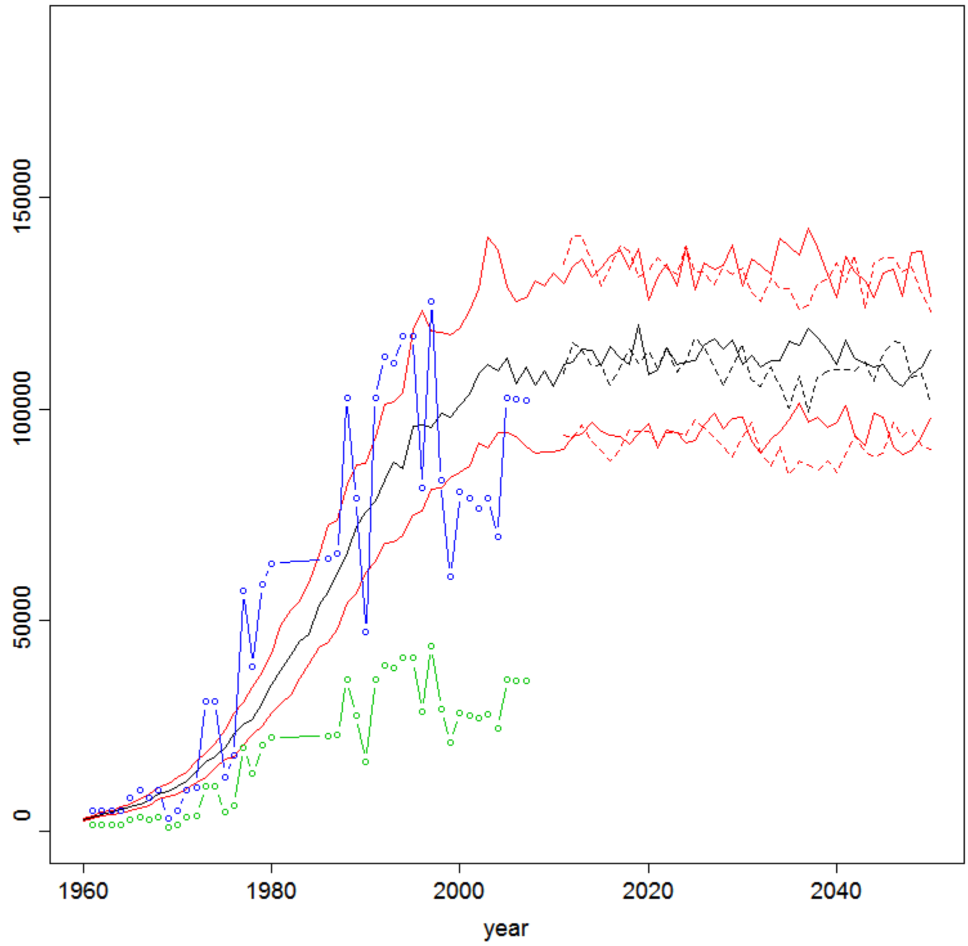


Figure 5.1.22 *Counted total (green circles and line) and modelled (black line=median; red lines=25 and 75 percentile) total population trend for brent goose. Fraction of breeding birds depicted with green circles and line. With the effect of worst case increased mortality, number of yearly collisions based on band model calculation (n victims = 10) for a scenario with 11 wind farms like OWEZ in offshore waters.*

Knot *Calidris canutus*

Occurrence and population trend in the Netherlands

The knots present in the Netherlands are from breeding populations in High Arctic Canada and Greenland (*Calidris canutus islandica*) and Central Siberia (*C. c. canutus*) (Delany & Scott 2006). The latter winter in West Africa and occur in the Netherlands laregly as a passage migrant. Birds from the *islandica* population winter in Europe with numbers up to 90,000 present in the Wadden Sea (Bijlsma *et al.* 2001). During prolonged cold periods the number of wintering birds in the Wadden Sea may fall as birds relocate to other sites, such as in the UK. Numbers in the Dutch Delta typically remain fairly constant, between 15,000 and 28,000 (Bijlsma *et al.* 2001).

Effects of OWEZ extrapolated - cumulative effects on the selected population
The migratory movements of this species in autumn are mainly nocturnal (with departures from the Wadden Sea) and occur then most often at high altitudes. However, migratory movements can also occur during the day, mostly during spring in head wind situations when birds arrive from southern wintering areas, but those are likely more concentrated along the coast (Camphuysen & Van Dijk 1983). Since during the field studies no observations on this species have been made in OWEZ no cumulative negative effects of multiple offshore wind farm scenarios are being expected for this species.

Skylark *Alauda arvensis*

Occurrence and population trend in the Netherlands
Skylarks are found throughout the Netherlands and are a fairly abundant breeding and wintering species. Between the early 1970s and the mid-1080s numbers fell from over 500,000 pairs to less than 300,000 pairs (Bijlsma *et al.* 2001). Breeding numbers are still considered to be in decline (Boele *et al.* 2011). In some areas numbers remain stable or are possible increasing. During winter, it is thought that a small proportion of Dutch population may cross the North Sea to Britain and Ireland, although many migrants are likely to be of Scandinavian or more eastern origin (Bijlsma *et al.* 2001; Wernham *et al.* 2002).

Effects of OWEZ extrapolated - cumulative effects on the selected population
The population migrating in extremely large numbers over the Dutch part of the North sea consists of birds coming from a very wide range of northern breeding areas in Scandinavia and possibly also Russia. This population consists of a total of over 7 million breeding pairs. As population trends and population dynamic parameters are uncertain for this huge population of this vast area, different levels of potential cumulative effects of multiple new offshore wind farms are calculated with the Potential Biological Removal approach, see section 5.2.

Meadow Pipit *Anthus pratensis*

Occurrence and population trend in the Netherlands
Meadow Pipits are widespread throughout the year in the Netherlands. During the mid-1980s the breeding population was estimated at between 70,000 and 100,000 pairs (Bijlsma *et al.* 2001). Breeding numbers in the Netherlands have declined slightly, although may be increasing locally (Bijlsma *et al.* 2001; Boele *et al.* 2011). During winter, it is thought that a proportion of Dutch population may cross the North Sea to Britain, although the huge numbers of migrants passing by are more to be of Scandinavian origin (Bijlsma *et al.* 2001; Wernham *et al.* 2002).

Effects of OWEZ extrapolated - cumulative effects on the selected population
The population migrating in extremely large numbers over the Dutch part of the North sea consists of birds coming from a very wide range of northern breeding areas in Scandinavia and possibly also Russia. This population consists of a total of over 3 million breeding pairs. As population trends and population dynamic parameters are uncertain for this huge population of this vast area, different levels of potential cumulative effects of multiple new offshore wind farms are calculated with the Potential Biological Removal approach, see section 5.2.

Redwing *Turdus iliacus*

Occurrence and population trend in the Netherlands
Redwings are a common passage migrant and winter visitor in the Netherlands with birds typically present between September to March. Redwings in the Netherlands are most of Scandinavian origin or from further east. During migration it is estimated that a million birds can be present in the Netherlands (Bijlsma *et al.* 2001). Numbers during winter are variable due to pending weather and food conditions at the larger scale.

Effects of OWEZ extrapolated - cumulative effects on the selected population
The population migrating in extremely large numbers over the Dutch part of the North sea consists of birds coming from a very wide range of northern breeding areas in Scandinavia and possibly also Russia. This population consists of a total of several million breeding pairs. As population trends and population dynamic parameters are uncertain for this huge population of this vast area, different levels of potential cumulative effects of multiple new offshore wind farms are calculated with the Potential Biological Removal approach, see section 5.2.

Starling *Sturnus vulgaris*

Occurrence and population trend in the Netherlands
Starlings are an abundant breeding and wintering species in the Netherlands. Breeding were estimated at between 750,000 and 1,300,000 pairs in the mid-1980s (Bijlsma *et al.* 2001). Numbers are thought to have declines slightly in recent years (Boele *et al.* 2011). Wintering and passage birds are mostly considered to be of eastern origin, with many continuing their migration to Britain and Ireland (Bijlsma et al 2001; Wernham *et al.* 2002). During winter large numbers are found in open agricultural areas.

Effects of OWEZ extrapolated - cumulative effects on the selected population
The population migrating in extremely large numbers over the Dutch part of the North sea consists of birds coming from a very wide range of breeding areas, covering large parts of central Europe and Russia. This population consists of a total of more than 20 million breeding pairs. As population trends and population dynamic parameters are uncertain for this huge population of this vast area, different levels of potential

cumulative effects of multiple new offshore wind farms are calculated with the Potential Biological Removal approach, see section 5.2.

5.2 Results from the Potential Biological Removal approach

The calculation of Potential Biological Removal level (PBR) yields the number of additional casualties (increased mortality) that can be sustained each year by a population based on

- R_{max} - the maximum annual recruitment rate;
- N_{min} - a conservative estimate of the population size and;
- *rf* - a recovery factor between 0.1 and 1 depending on the status;

(Dillingham & Fletcher 2008; Wade 1998; Niel & Lebreton 2005) (see § 2.4.2 for the formula).

The factor *rf* is a recovery factor depending on the population status, *rf*= 0.1 provides a minimal increase in recovery time for a depleted population chosen for those species with a near threatened status (according to the IUCN), in order to maintain a population size close to carrying capacity or to minimize the extinction risk for a population with a limited range. A value of *rf*=1 could be used to maintain a healthy growing population at or above its maximum net production level; *rf*=0.5 is an arbitrary intermediate stage for species with a least concern status but with unstable or a decreasing population trend.

The PBR approach has the advantage that only few demographic parameters are necessary to calculate the order of magnitude of sustainable mortality limits. Because the method relies on few demographic parameters, the precautionary approach is guaranteed by using minimum population estimates and a recovery factor depending on the population status. At one hand the PBR approach gives the opportunity to calculate sustainable mortality limits for those species where detailed information on population parameters were lacking and in section 5.1 no detailed population model could be constructed, on the other hand for those species these models could be constructed, the modelling outcomes in section 5.1 can be compared with the outcomes of this approach as well.

In table 5.1 the number of estimated collision victims for a scenario of a total of 11 offshore OWEZ wind farms based on the OWEZ field studies (Krijgsveld *et al.* 2011, and calculated with the Band model) are compared with the number of victims with three different recovery stages of species-specific populations.

The imposed impacts in the models in section 5.1 must be regarded as worst-case scenarios because:

- in the models the situation has been simulated that all collisions occur on breeding females or pairs during the breeding season and
- only reproducing units become a victim, as every victim is probably a partner of a unique breeding pair.

The chance that during the breeding season both partners of the same breeding pair are lost due to colliding with a wind farm is very low, therefore the approach chosen here is truly precautionary as one can argue that the double the number of victims could occur (namely including the partner of the pairs affected by a first victim). For a comparison with the impacts of effects by the constructed population models, also the PBR levels are expressed as breeding pairs. Furthermore, the modelling did not take into account potential collisions with birds of a foreign origin outside the breeding season or with birds of a non-breeding or juvenile status.

All northwestern European populations of the selected species studied in this report have the IUCN least concern status. This implies that all calculated numbers of victims for OWEZ alone, and the two scenarios should be compared with the calculated sustainable PBR level for *rf* = 0.5 (and for most species even *rf* = 1.0) (in table 5.1 indicated with a different yellow colours). However, for three species as a precautionary approach we have made an exception because of recent dramatic declines in the population by treating bewick's swan, herring gull and knot as potentially near threatened species. This implies that the calculated number of victims in relation the sustainable level of increased mortality is judged for the recovery factor of *rf* = 0.1. In that case for herring gull the numbers of victims of the two scenarios for multiple offshore wind farms are higher than the calculated sustainable PBR level.

Based on the findings of Krijgsveld *et al.* (2011) more than 50% of the victims consist of mostly nocturnal migrant birds mainly consisting of passerines and other non-seabird species. The numbers of passerine victims of a cumulative scenario of 11 wind farms like OWEZ in the Dutch part of the North Sea is an estimated total number of 3400 passerine victims on a yearly basis. Skylark, meadow pipit, redwing and starling are likely the most numerous passerine species flying over sea with for each species several millions of birds involved. The exact proportion for every species however is unknown, so for a comparison of the Potential Biological Removal the maximum number of passerine victims is compared with the PBR values per species. As can be seen in table 5.1 for all four species the PBR values are far above the total number of victims calculated for the multiple offshore wind farm scenario.

Table 5.2.1 *PBR- Potential Biological Removal level for selected species for populations occurring in the Dutch part of the North Sea compared to respectively the calculated number of collision victims (expressed as breeding pairs), for OWEZ alone, and two scenarios of multiple new offshore wind farms in the Dutch part of the North Sea. All selected species have an IUCN least concern status in NW-Europe, most with a stable or increasing population trend (recovery PBR factor rf = 1.0) or a least concern with unstable or decreasing population trend (recovery PBR factor rf = 0.5) (IUCN 2011). In green indicated the number of victims lying well within the sustainable mortality limits. Bewick's swan, herring gull and knot, because of a strong negative population trend, are treated as a precautionary approach as near threatened species, resulting that for the Dutch population of the herring gull the calculated number of collision victims for both scenarios potentially are beyond the sustainable mortality limits of the PBR approach (indicated with a purple colour) (based on rf = 0.1). R_{max} calculated based on parameters in appendix 4.*

							PBR			Collision victims		
species	Dutch name	region	Nmin	Rmax	year	source	*rf=0.1*	*rf=0.5*	*rf=1.0*	OWEZ	Scenario 1	Scenario 2
red-throated diver	roodkeelduiker	North sea basin	55900	0.25	2006	1a	700	3400	6900	0.2	1.8	9.2
cormorant	aalscholver	Netherlands	21000	0.16	2006	8	200	900	1700	30.2	332.2	unknown
shag	kuifaalscholver	Scotland	21400	0.17	2006	1a	200	900	1800	0.0	0.0	0.0
gannet	jan van gent	Scotland	167300	0.12	2006	1b	1000	5200	10400	1.6	17.2	199.2
fulmar	noordse stormvogel	Scotland	281700	0.07	2006	6	1000	4900	9800	0.0	0.0	0.0
Bewick's swan	kleine zwaan	NW-Europe	7200	0.06	2006	3	20	110	230	0.5	5.0	5.0
brent goose	rotgans	NW-Europe	76000	0.20	2004	1a	800	3800	7600	0.5	5.0	5.0
shelduck	bergeend	NW-Europe	118000	0.21	2006	1a	1300	6300	12500	0.0	0.0	0.0
eider	eider-eend	NW-Europe	304000	0.13	2002	1a	1900	9600	19200	0.0	0.0	0.0
common scoter	zwarte zee-eend	NW-Europe	640000	0.21	2006	1a	6600	33000	66000	0.1	1.0	1.0
great skua	grote jager	Scotland	7600	0.07	2004	2	30	130	260	0.1	0.8	39.6
great black-backed gull	grote mantelrmeeuw	NW-Europe	130900	0.15	2006	1a	1000	4900	9800	19.0	209.4	134.9
herring gull	zilvermeeuw	Netherlands	50000	0.10	2008	5	200	1200	2400	53.2	585.6	698.1
lesser black-backed gull	kleine mantelmeeuw	Netherlands	89900	0.13	2006	4	600	2800	5600	70.6	776.8	875.8
little gull	dwergmeeuw	NW-Europe	28200	0.23	2006	1a	300	1600	3200	15.7	172.3	75.1
common gull	stormmeeuw	NW-Europe	472800	0.17	2006	1a	4100	20600	41200	32.33	355.7	152.7
kittiwake	drieteenmeeuw	Scotland	281600	0.10	2002	7	1300	6700	13500	31.4	345.6	217.1
Sandwich tern	grote stern	Netherlands	16700	0.15	2010	8	100	600	1300	2.6	28.8	154.5
common tern	visdief	Netherlands	17000	0.16	2010	8	100	700	1400	0.3	2.8	60.5
little tern	dwergstern	Netherlands	500	0.15	2006	8	4	18	35	0.0	0.0	0.0
guillemot	zeekoet	Scotland	504200	0.11	2002	7	2700	13700	27400	0.0	0.1	0.1
razorbill	alk	Scotland	110800	0.10	2002	7	600	2900	5800	0.0	0.1	0.1
puffin	papagaaiduiker	Scotland	409400	0.06	2004	1b	1300	6600	13200	0.0	0.0	0.0
knot	kanoet	Can./Greenl./Russia	238000	0,233	2006	1a	2780	13800	27798	0.0	0.0	0.0
redwing	koperwiek	Scandinavia	3980000	0,755	1995	9	150000	750000	1500000	max. 309.9	max. 3,400	max. 3,400
starling	spreeuw	Central Europe/Russia	29900000	0,559	1995	9	840000	4180000	8370000	max. 309.9	max. 3,400	max. 3,400
skylark	veldleeuwerik	Scan./Russia	7970000	0,69	1995	10	278000	1390000	2780000	max. 309.9	max. 3,400	max. 3,400
meadow pipit	graspieper	Scandinavia	3200000	0,68	1995	10	108000	540000	1080000	max. 309.9	max. 3,400	max. 3,400

Sources population size: (1a) Wetlands International 2006. Waterbird population estimates - Fourth Edition. Wetlands International, Wageningen, Netherlands. (www.wetlands.org). (1b) BirdLife International (2004) Birds in Europe. Population estimates, trends and conservation status. Cambridge, UK: BirdLife International. (BirdLife Conservation Series No. 12), (2) Camphuysen 2002, (3) Petkov, Rees & Solokha Overview of the status of the NW European population of Bewick's Swan, (4) SOVON broedvogels in nederland 2006 - 2008/01 (5) SOVON broedvogels in nederland 2010/01, (6) Baker et al 2006; British Birds. (7) Mitchel et al. 2004, (8) SOVON Vogelonderzoek Nederland, (9) Hagemeijer & Blair 1997, (10) Cramp 2000. R_{max}: based on Neil & Lebreton 2005, Rmax Bewick's swan calculated from reproduction figures between 1964 and 2009 taken from UK and Dutch counts in wintering areas (Collier et al. 2008 in series, SOVON in series, Dirksen 1991, Evans, 1979 and Thijssen 2010)

Table 5.2.2 Summary of cumulative effects due to multiple wind farms for the two scenarios. The impact of cumulative effects on populations were determined through populations models (for species for which sufficient data were available; see species accounts in chapter 5) and/or based on calculations of the level of Potential Biological Removal as presented in table 5.2.1. See paragraph 6.1 for an overview of the worst case scenario followed in the population modelling and paragraph 6.3 for an explanation of the limitations of the conclusions presented here based on the scenarios studied. The figures of number of victims in this table are taken from table 5.2.1.

species	Dutch name	region	current population trend	Scenario 1 (11 OWEZ-like farms 10-20 km offshore) n victims	Cumulative effects due to multiple wind farms of scenario 1 (taken into account worste case scenario of the models)	Scenario 2 (11 offshore farms across Dutch North Sea, thus largely > 20 km	Cumulative effects due to multiple wind farms of scenario 2 (taken into account worste case scenario of the models)
red-throated diver	roodkeelduiker	North sea basin	unknown	1.8	highly unlikely	9.2	highly unlikely
cormorant	aalscholver	Netherlands	stable	332.2	positive	unknown	positive?
shag	kuifaalscholver	Scotland	stable	0.0	highly unlikely	0.0	highly unlikely
gannet	jan van gent	Scotland	stable	17.2	highly unlikely	199.2	highly unlikely
fulmar	noordse stormvogel	Scotland	stable	0.0	highly unlikely	0.0	highly unlikely
Bewick's swan	kleine zwaan	NW-Europe	decrease	5.0	highly unlikely	5.0	highly unlikely
brent goose	rotgans	NW-Europe	stable	5.0	highly unlikely	5.0	highly unlikely
shelduck	bergeend	NW-Europe	stable	0.0	none	0.0	none
eider	eider-eend	NW-Europe	stable	0.0	highly unlikely	0.0	none
common scoter	zwarte zee-eend	NW-Europe	unknown	1.0	highly unlikely	1.0	highly unlikely
great skua	grote jager	Scotland	stable	0.8	highly unlikely	39.6	highly unlikely
great black-backed gull	grote mantelrmeeuw	NW-Europe	stable	209.4	highly unlikely	134.9	highly unlikely
herring gull	zilvermeeuw	Netherlands	decrease	585.6	highly unlikely	698.1	highly unlikely
lesser black-backed gull	kleine mantelmeeuw	Netherlands	stable	776.8	highly unlikely	875.8	highly unlikely
little gull	dwergmeeuw	NW-Europe	unknown	172.3	highly unlikely	75.1	highly unlikely
common gull	stormmeeuw	NW-Europe	stable	355.7	highly unlikely	152.7	highly unlikely
kittiwake	drieteenmeeuw	Scotland	decrease	345.6	highly unlikely	217.1	highly unlikely
Sandwich tern	grote stern	Netherlands	increase	28.8	highly unlikely	154.5	highly unlikely
common tern	visdief	Netherlands	increase	2.8	highly unlikely	60.5	highly unlikely
little tern	dwergstern	Netherlands	stable	0.0	none	0.0	none
guillemot	zeekoet	Scotland	increase	0.1	highly unlikely	0.1	highly unlikely
razorbill	alk	Scotland	increase	0.1	highly unlikely	0.1	highly unlikely
puffin	papagaaiduiker	Scotland	stable	0.0	none	0.0	none
knot	kanoet	Can./Greenl./Russia	decrease	0.0	none	0.0	none
redwing	koperwiek	Scandinavia	unknown	max. 3,400	highly unlikely	max. 3,400	highly unlikely
starling	spreeuw	Central Europe/Russia	unknown	max. 3,400	highly unlikely	max. 3,400	highly unlikely
skylark	veldleeuwerik	Scan./Russia	unknown	max. 3,400	highly unlikely	max. 3,400	highly unlikely
meadow pipit	graspieper	Scandinavia	unknown	max. 3,400	highly unlikely	max. 3,400	highly unlikely

Sources population trends, see table 5.2.1.

6 Discussion

6.1 Limitations of models used and the precautionary approach

The species-specific models used in this report to calculate cumulative effects on the population were shown to mirror the actual historic population trends for each population. Therefore, for those species populations for which good information exist, they can be considered to be robust instruments for investigating whether essential population impacts are to be expected in case multiple wind farms are being developed. As outlined in the first chapters, however, many population parameters for individual colonies of seabirds are often lacking, so that for selected species breeding in the Netherlands only the total population could be modelled.

For several species we could not use the modelling approach because too much uncertainty exists about the origin of birds occurring in the Dutch part of the North Sea. This was particularly true for those species with a wide breeding distribution and with indications that birds from the entire breeding range use the Dutch part of the North Sea, e.g. red-throated diver, common scoter, great black-backed gull, little gull, and common gull. For other species, such as gannet, kittiwake, razorbill and guillemot, we chose to model the cumulative effects on specific regional populations, for example Scotland. This was because ringing recoveries or observations via data loggers have shown that a substantial proportion of birds turning up in the Dutch part of the North Sea originate from that area.

This approach must be regarded as worst case scenario approach in that;

- For Dutch breeding populations we have modelled breeding females and as a precautionary approach we have attributed all victims to breeding females. The modelling also did not take into account potential collisions of birds of a foreign origin (outside of the modelled populations) outside the breeding season or with birds of a non-breeding or juvenile status.
- For seabird species breeding outside the Netherlands, the impacts of new Dutch offshore wind farms were restricted to one geographical population (mostly Scotland), while in reality a much wider breeding range with different geographical populations might be involved.
- In most models a floater population of 10 or 30 % has been chosen. Published research has shown that higher percentages can occur, especially in many long-lived species, meaning that a larger buffer function could be present in the floater population.
- In the models used, only density dependence is modelled in relation to reproduction. Density dependence can also act on mortality via the process of intra-specific competition between individual birds outside the breeding season. In a case in which the carrying capacity decreases the intra-specific competition on resources will increase, with the consequence of a potentially lowered survival of birds. This would imply that in this situation the victims

occurring due to human caused impacts like through collisions with wind turbines could have a so-called compensatory effect, as victims taken out from the population will reduce the intra-specific competition.

- The number of calculated collisions put into the populations models have been kept stable over the years, assuming that a decrease in the population due to collisions does not affect the intensity of flight movements in and around wind farms. This situation is possible in case wind farms are developed in high quality foraging areas with birds shifting from low quality areas to these high quality areas.

Regarding the floaters (non-breeding adults) in the population, collision victims have been calculated as breeding adults only. This implies the situation that floaters are not directly affected, but immediately take the empty places in the breeding population. In reality floaters can also collide with wind turbines, especially in those situations when new offshore wind farms are located in areas where disproportional more floaters are present. In such a situation the impacts on the level of the population also occur; with floaters disappearing from the population as a result of collisions the recruitment of new breeding birds is ultimately hampered (Penteriani *et al.* 2011). Our models also describe this strong connection between floaters and breeding birds yet in order to illustrate a worst case scenario, we have chosen to concentrate all victims in the group of breeding birds. This also has an immediate consequence on reproduction by assuming the failure of the brood.

Band model and avoidance figures

The model outcomes presented in this study are based on calculations including stochastic variability in both mortality and reproduction, so the extremes are incorporated. Furthermore, the macro and micro avoidance figures based on the OWEZ field study must be regarded as conservative. Compared to the SHN 2010 report on the Band model for some species we have found higher avoidance figures as already assumed by the SHN group, but we feel that for most species they are in reality even higher due to the limitations in spatial resolution of the radar data and the difficulty of species identification of individual radar targets. The idea is that with a better resolution in the analysis of micro avoidance more birds can be identified as flying outside the rotor area. We therefore think that due to technical innovations in radar ornithology or alternative studies in individual flight paths (e.g. GPS logger studies) in future better (read higher) estimates of avoidance rates will be determined. Based on the sensitivity analysis of Chamberlain *et al.* (2006) reproduced in table 2.3.1 this will have an impact by a lower number of estimated collisions.

6.2 Uncertainty about disturbance and barrier effects

Disturbance

In the area where OWEZ is located, offshore species, such as gannets, great skuas, guillemots, razorbills and other species, occur in low densities for reasons other than the presence of the wind farm (Leopold *et al.* 2010), therefore, the numbers of birds that were disturbed were probably relatively low. This has prevented us in extrapolating the potential effects to impacts on the populations of these species in a realistic way. However, the calculations with the developed populations models and the PBR approach have shown that for the relevant species, in addition to the impacts of collisions, a large buffer exists without the risks of serious impacts on total populations. We will have to wait for the opportunity to measure and quantify disturbance effects in species groups like auks in future new offshore wind farm in deeper waters in order to be able to calculate impacts due to cumulative effects of disturbance/displacement.

Barrier effects

Species-specific flight paths and fluxes measured in the field indicate that at a large-scale avoidance behaviour occurs in some species (Krijgsveld *et al.* 2011). Energetic considerations, like those calculated by Masden *et al.* (2009, 2010) to cover increased flight distances, probably only play a role in cases where very large wind farms are developed and breeding birds with daily foraging fights are involved. However, Krijgsveld *et al.* (2011) have shown that for the species involved for the Dutch situation no strong avoidance behaviour was found, so that population impacts of barrier effects are limited, certainly compared to the population effects of increased collision risks. The calculations with the developed populations models and the PBR approach have shown that in addition to the impacts by collisions in the relevant species a large buffer exists for impacts by barrier effects. Species such as cormorant, herring gull and lesser black-backed gull, the few species with daily foraging flights from breeding colonies, did not show avoidance behaviour in relation to OWEZ, therefore, the occurrence of barrier effects in these species of multiple offshore wind farms are expected to negligible compared to collision impacts in these species.

6.3 Application of findings in relation to new offshore wind farms

Configuration of OWEZ

The configuration of OWEZ should be regarded from a bird perspective as very open. We expect that the measurements of avoidance levels of flying birds in different species groups of Krijgsveld *et al.* (2011) and the first indications of disturbance/displacement effects of Leopold *et al.* (2010) are very specific to the configuration of OWEZ. In particular, the wide spacing between the four rows of wind turbines in OWEZ makes it potentially a very open wind farm; e.g. easy to enter by flying birds. As no data are available on configuration related effects, for this report we can only extrapolate the effects as measured in and around OWEZ and, therefore, we can only evaluate scenarios of new wind farm developments of wind farms under the assumption of the same configuration and size as OWEZ. Thus the findings of this report can certainly not be extrapolated if future new offshore wind farm developments deviate strongly from the two scenarios now evaluated (e.g. wind farms with a higher density of turbines than OWEZ). The number of wind farms, the location of the wind farms, the size and configuration of the individual wind farms will ultimately determine the scale of impacts relative to what is presented in this report. The quantitative effects of other configurations of offshore wind farms should be determined first before such an exercise is possible.

Other impacts on seabirds

In this report only the impact of two scenarios of future new offshore wind farms have been studied. Besides new wind farm development, also other anthropogenic pressures must be taken into account when evaluating the effects of wind farms on a population level for different species. Especially the developments of shipping, fishery and mining and related potential pollution and disturbance need to be taken into account. The incorporation of these impacts however was outside the scope of this report.

Cumulative effects of offshore wind farms outside the Dutch EEZ

In this report only cumulative effects have been investigated for scenarios of offshore wind farm developments within the boundaries of the Dutch EEZ. Such an analysis was beyond the scope of this report, although we strongly recommend that in near future more international effort should be undertaken in this field.

7 Conclusions and recommendations

This is the first attempt to estimate cumulative effects of many offshore wind farms in a part of the North Sea on the population level for a range of species. We used a simple and robust population model as a basis and fed the effects, in terms of additional mortality resulting from numerous wind farms, into the models. Krijgsveld *et al.* (2011) and Leopold *et al.* (2010) have shown the effects of OWEZ, a single wind farm, on seabirds and other bird species. The analyses in this report have shown that for a single wind farm the effects on the population are far from those levels at which serious negative impacts with decreasing trends occur. Furthermore, for most species even a tenfold extrapolation of the effects are still within those limits, except for the herring gull. The Dutch breeding population of this species shows a strong decrease, which is likely to be related to major changes in the ecological conditions for the species and not related to the presence of the two active offshore wind farms. As such, this might be representative for multiple wind farms in the Dutch North Sea. This conclusion was confirmed by using the Potential Biological Removal approach; another way for estimating the size of effects on populations. It should be emphasized that all calculations have always been carried out conservatively. Following precautionary assumptions in different aspects, future research related to monitoring of effects around new offshore wind farms in deeper waters would probably yield results that confirm that in this report a worst-case approach has been followed.

In this study we describe two cases of decreasing populations, namely the international Bewick's swan population and the Dutch breeding population of the herring gull. The population model outcomes in these two species show that the influence of the increased mortality due to new offshore wind farm developments are relatively small in relation to these natural trends in decreasing populations. We conclude that stochastic incidents are likely not more influential in case of parallel impacts including offshore wind farms. In case of long-lived species as studied in this report such scenarios with consecutive years of strongly decreased recruitment is rare, and most of the time not caused by a natural phenomenon.

For the scenario of multiple wind farms in near-shore waters all information from OWEZ (Krijgsveld *et al.* 2011, Leopold *et al.* 2010) and the baseline study at Meetpost Noordwijk (Krijgsveld *et al.* 2005) could be extrapolated straight forward. For the scenario of multiple offshore wind farms scattered over a wide area of the Dutch North Sea assumptions have been made about the fluxes of bird movements of about the seabird species composition in offshore waters. These assumptions were derived from all available information about fluxes and species in near-shore and offshore waters. At the moment of writing a radar study is currently under way at an offshore platform about 150 km northwest of OWEZ, in a set up that is comparable with the one in and around OWEZ as well as Meetpost Noordwijk. With the results from this offshore study the impact assessment of several offshore wind farms with an OWEZ configuration could be improved.

Furthermore, when developing offshore wind farms in offshore deeper waters, a different seabird community is likely to be present. This implies the presence of higher densities of guillemots, razorbill, great skua, gannet, and possible others species compared to near-shore waters (Leopold *et al.* (2010). Furthermore, for real coastal species, the densities in offshore waters will be lower, or may not even be present at all. The ship-based surveys to quantify the disturbance/displacement effect in and around OWEZ encountered low densities of offshore species in this area. Therefore, it was not possible to show disturbance/displacement effects among offshore species statistical significant. So, this type of survey is recommended to carry out when a new offshore wind farm is to be developed in deeper waters in the Dutch North Sea. At this moment it is not possible to fully quantify impacts of disturbance/displacement on population levels for new offshore wind farms in deeper waters. Firstly, future research around an offshore wind farm in higher density areas (for offshore species) than OWEZ is necessary to obtain results in order to determine the disturbance/habitat loss effects in terms of density decrease. Subsequently the question has to be answered, and that is much more difficult, what will be the ultimate effect of the displacement of birds involved? Will the displacement truly lead to miss foraging opportunities with consequently a decrease in food intake on one hand and an increased energy expenditure on the other? Or will the effect be negligible because the numbers of guillemots, razorbills and gannets are still far from from the carrying capacity in the non-breeding season?

Based on the species composition and flight behaviour of local seabirds around OWEZ, it is expected that around an offshore wind farm in the Dutch part of the North Sea the impact of barrier effects will be negligible in comparison to the expected number of victims due to collisions. Therefore, with the lack of precise information about future offshore wind farms, the expected negligible impacts of barrier effects could not been investigated any further.

Recommendations

As indicated in the discussion already, in a wider international context it is desirable to initiate further investigation on the cumulative effects of multiple wind farms in within the total distribution range of species, or in case of waterbirds and other migrant species in the entire flyway.

The acquisition of local (national) knowledge of the different countries involved is a first step. Basic information on numbers, trends, life history traits and movements should be sampled. Secondly, measured effects of existing offshore wind farms and realistic scenarios of wind farm developments are the necessary ingredients for a successful enterprise in future. In such an approach the cumulative impacts of multiple wind farms within the flyway or range of a species come into focus.

For species like red-throated diver, it is more worthwhile to gather knowledge on the movement ecology (with detailed information on flight activity patterns, flight routes, flight altitudes, stop over ecology, etc.) than on trying to measure impacts around new offshore wind farms. This is an example of a species for which basic knowledge about their ecology outside of the breeding areas is missing. The same is true for a number of other migrant species (e.g. swans, geese, scoters). For some of these species, detailed studies with gps-loggers and satellite transmitters are under way. They will hopefully yield the data that will make it possible to estimate potential effects of all three themes of effects of offshore wind farms. If more basic information on the ecology of these species becomes available, this will facilitate effect studies in and around wind farmes.

8 Acknowledgements

The authors thank everyone who has contributed to this report, especially the species specialist working on seabirds in- and outside the Netherlands. They put us on the track of useful references or delivered unpublished data. Special thanks to Rob Barrett, Roland-Jan Buijs, John Calladine, Kees Camphuysen, Stefan Garthe, Omo Hüppop, Mardik Leopold, Tone Reiertsen, Arie Spaans, Eric Stienen and Andy Webb.

Within Bureau Waardenburg Karen Krijgsveld, Ruben Fijn and Jan van der Winden contributed with knowledge and providing suggestions. Quality control was made by Theo Boudewijn and Tony Fox (NERI, Denmark).

During the different stages of this project feedback has been received from Ministry of Water & Transport/Rijkswaterstaat Waterdienst (Suzan van Lieshout, Marcel van der Tol, Joop Bakker).

This project was commissioned by Noordzeewind and was under supervision by Henk Kouwenhoven. His critical and positive feedback was always a stimulus during this challenging project. Jesper Kyed Larsen (Vattenfall) also thoroughly reviewed the report and made valuable suggestions to improve it.

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Appendices

Appendix 1 – explanation population models and parameters

Short rationale and explanation of models:
All numbers refer to the female part of the population or can be read as breeding pairs. This means that the number of floaters are also expressed as females or pairs.

For a selection of species the following maximum set of population models have been constructed:

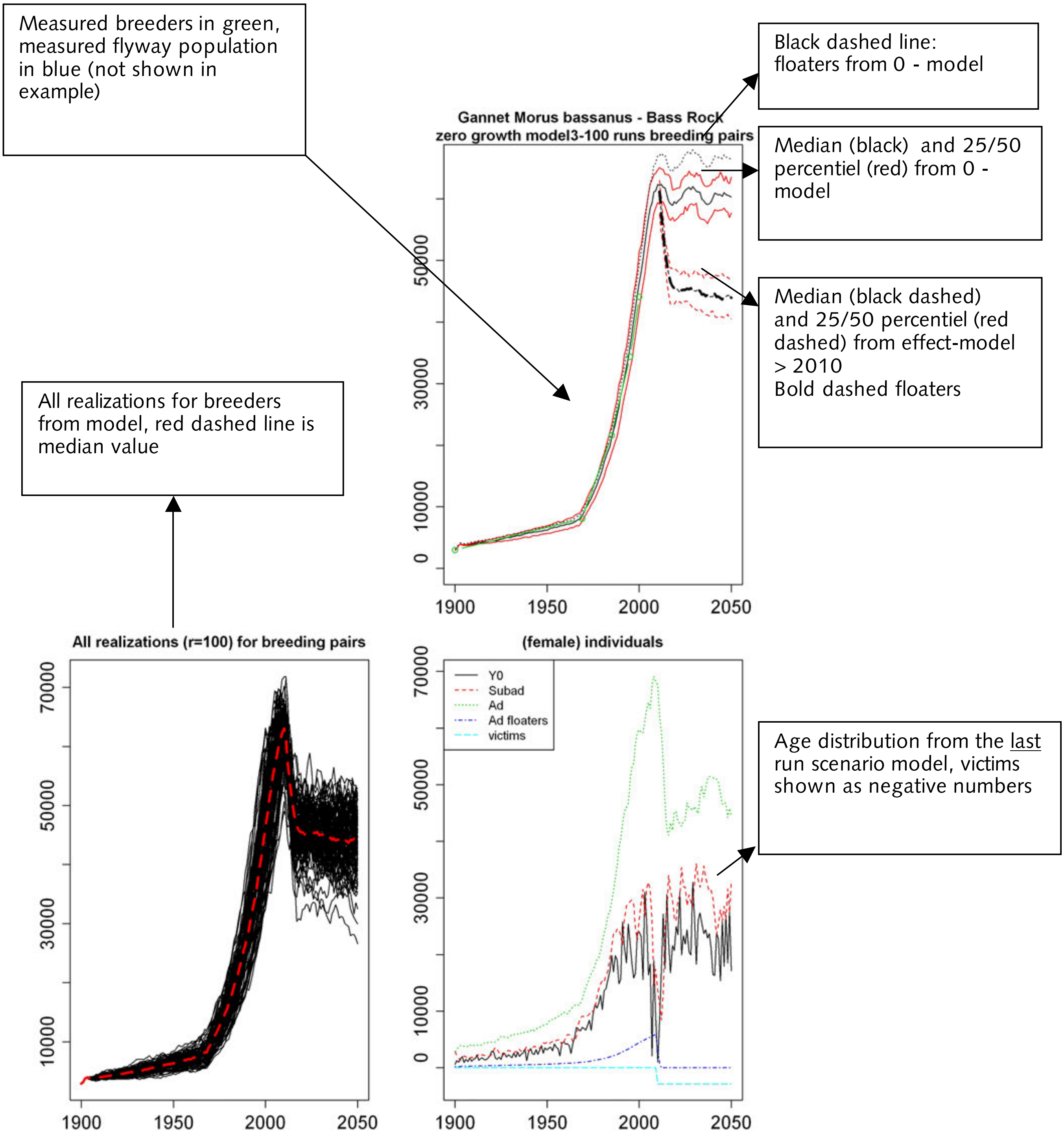
- 0-model (no victims, 0% floaters)
- 0-growth (in order to determine # victims above which population decline starts)
- # victims - band model (n victims on yearly basis based on outcomes field studies OWEZ, Krijgsveld *et al.* (2011))
- scenario maximum effect for bewick's swan and brent goose

For those species with increasing or stable population trends submodels were generated with 0 % floaters, 10 % floaters and 30 % floaters.

Explanation of the log file per model:

S0/S0sd	Year0 survival with standard deviation
S1/S1sd	Year1 survival with standard deviation
etc ...	
Sad/Sadsd	Adult survival with standard deviation
F0/F0sd	fecundity ~recruitement with standard deviation
B1	year of first breeding
Floaters –	proportions of floaters in the population
K	Carrying capacity
Init pop -	start number of population per age group
Victims -	number of victims per age group
population –	numbers in the population per agegroup in 2050

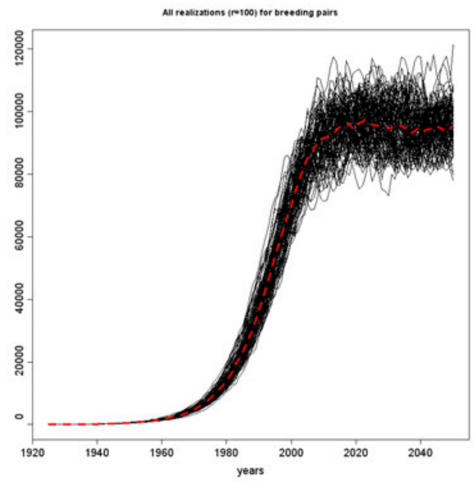
Appendix 2 - explanation graphs population models and sensitivity analysis

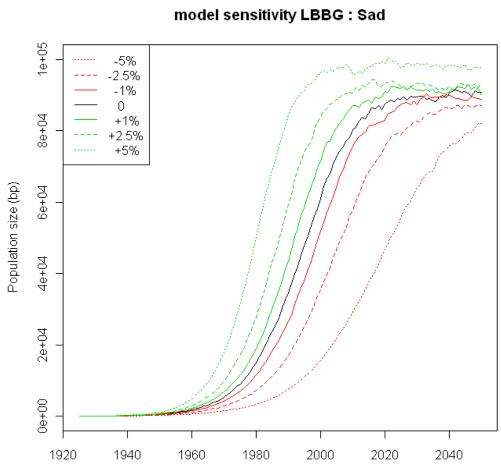

Measured breeders in green, measured flyway population in blue (not shown in example)
Black dashed line: floaters from 0 - model
Gannet Morus bassanus - Bass Rock
zero growth model3-100 runs breeding pairs
Median (black) and 25/50 percentiel (red) from 0 - model
Median (black dashed) and 25/50 percentiel (red dashed) from effect-model > 2010
Bold dashed floaters
All realizations for breeders from model, red dashed line is median value
All realizations (r=100) for breeding pairs
(female) individuals
Y0
Subad
Ad
Ad floaters
victims
Age distribution from the last run scenario model, victims shown as negative numbers
0
10000
30000
50000
70000
1900
1950
2000
2050

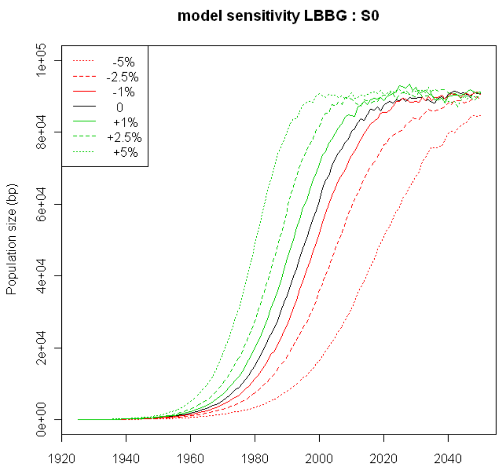
Sensitivity analysis in lesser black-backed gull

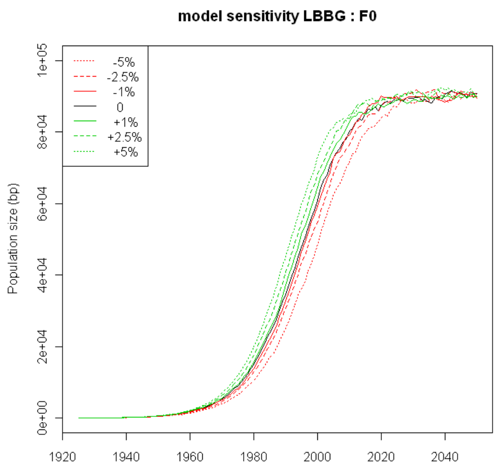
Sensitivity analysis is a technique for systematically changing variables in a model to determine the effects of such changes. Here we use a simple change one factor at a time (OAT) approach, presented below on the model of the lesser black-backed gull. Essentially the method applied in the species modelling were input parameters are drawn from a defined distribution (mean and sd) gives insight in the variability of the model outcomes based on the stochasticity of the combination of the parameters mortalitity and reproduction (graph 1). In the following graphs 2-4 one can see that as expected in long-lived species the model outcome is most sensitive to changes in adult mortality (Sad, widest range in outcomes), then to immature mortality (S0) and least to reproduction (Fad).

Graph 1 shows 100 runs with input parameters drawn from defined normal distributions for Sad, S0 and F0. The next 3 graphs show yearly (median) size of population for each adjustment of Sad, S0 and F0. Due to the density dependence mechanism numbers converge around 92,000 breeding pairs. The main effect of the variability in parameters is that timing of reaching the carying capacity changes.









Appendix 3 – species specific population models

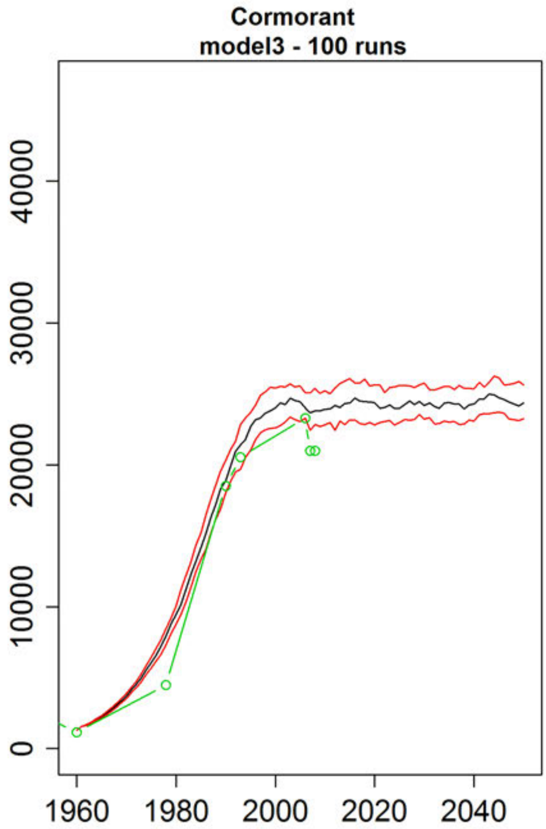
Cormorant – the Netherlands

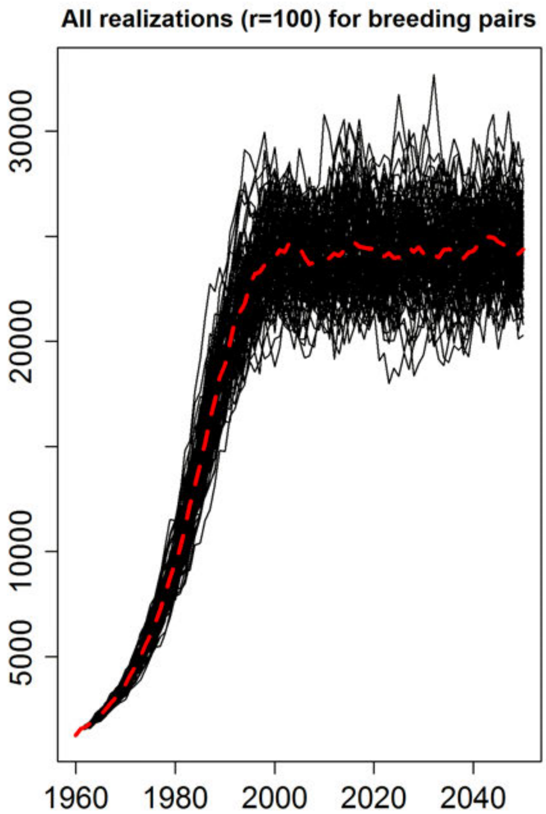
0 model

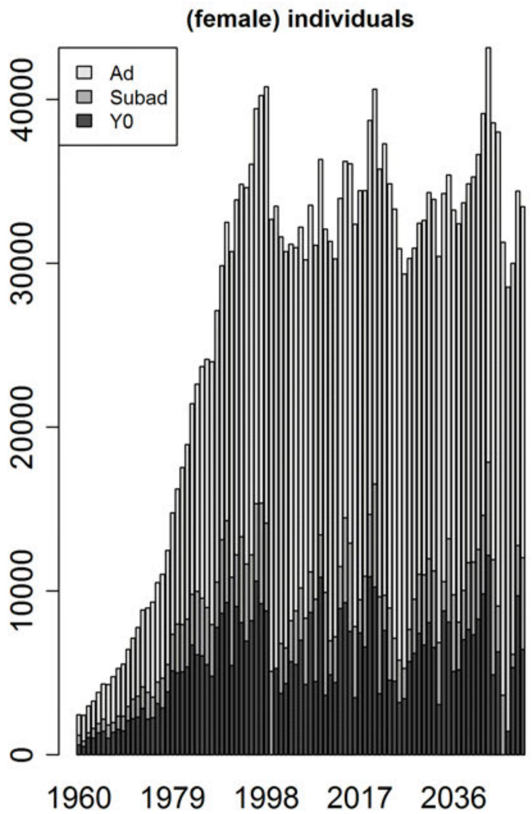
S0 : 0.58 S0sd : 0.1
S1 : 0.88 S0sd : 0.1
Sad: 0.88 Sadsd: 0.1
F0 : 0.625 F0sd : 0.1

K: 23300

Init population: 1000 500 1000







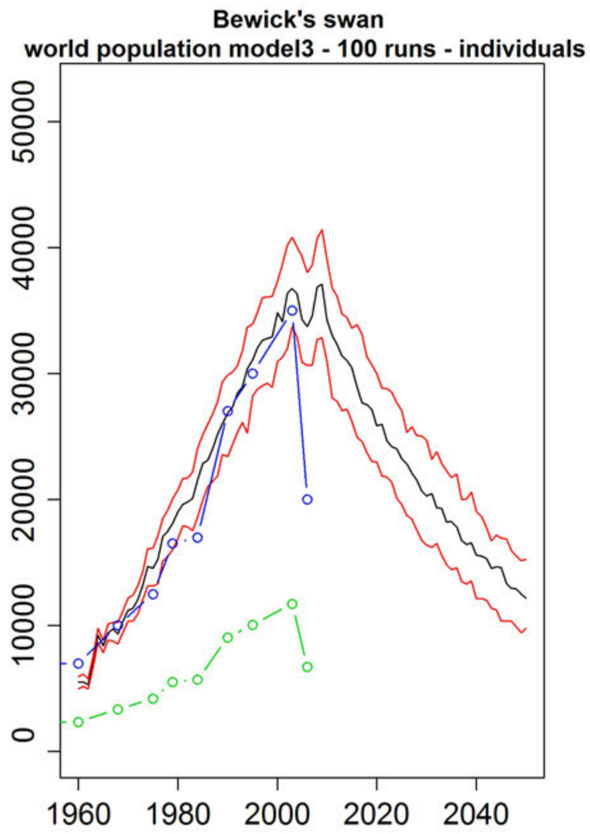
Bewicks swan – NW-Europe

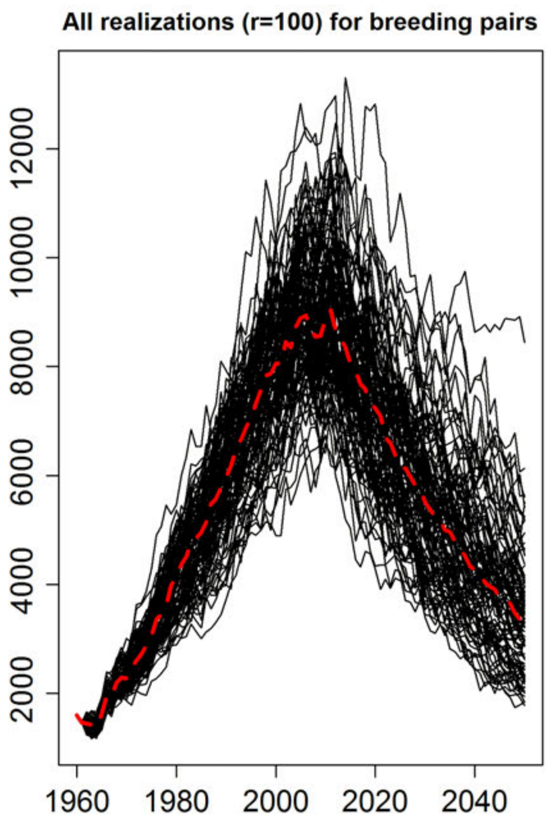
0-model

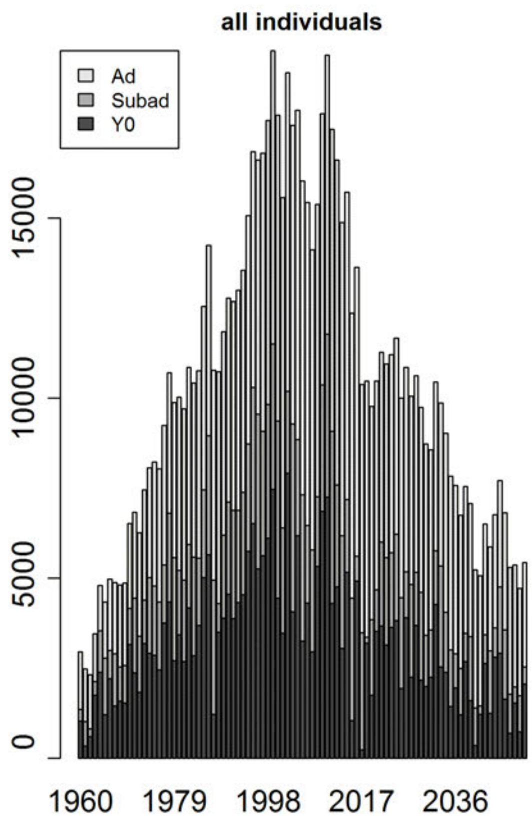
S0 : 0.66 S0sd : 0.01
S1 : 0.822 S0sd : 0.01
Sad: 0.822 Sadsd: 0.079
F0 : 0.5 F0sd : 0.185

K: 15000

Init population: 502.5 502.5 1675





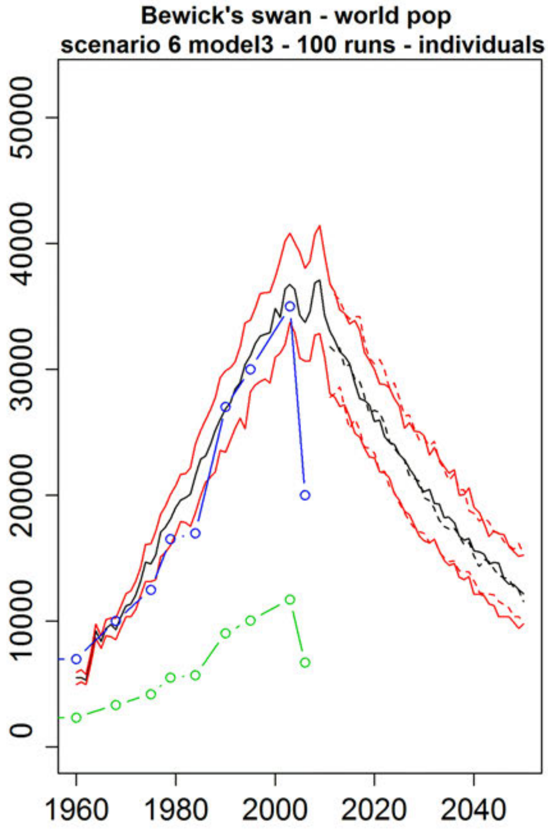


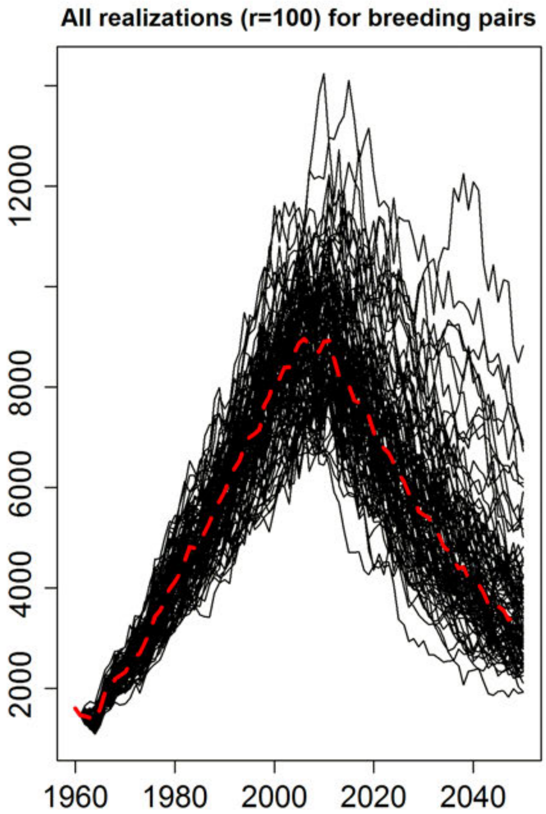
Scenario collisions Band-model

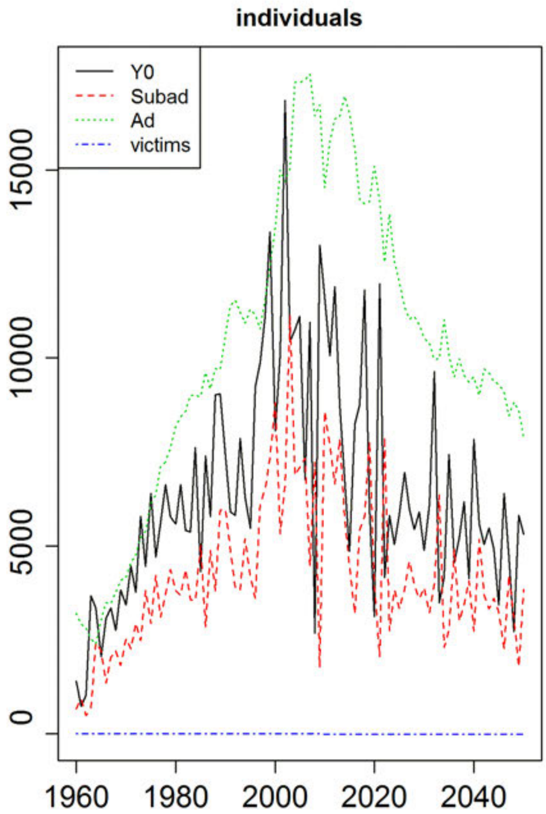
S0 : 0.66 S0sd : 0.01
S1 : 0.822 S0sd : 0.01
Sad: 0.822 Sadsd: 0.079
F0 : 0.5 F0sd : 0.185

K: 15000

Init population: 502.5 502.5 1675
Victims: 0 0 5





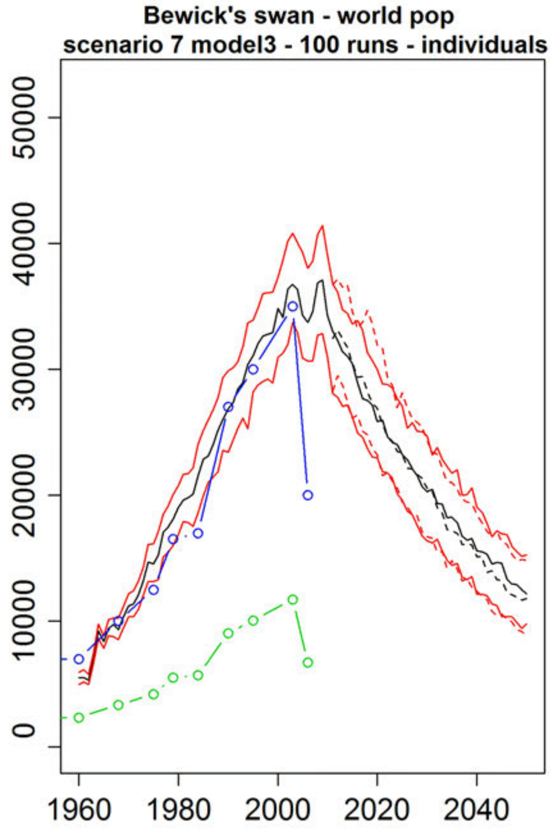


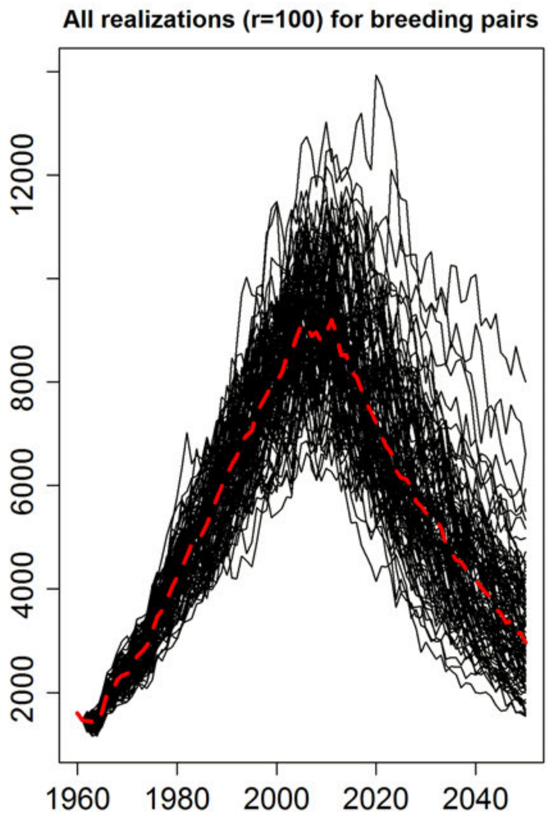
Scenario maximum effect scenario

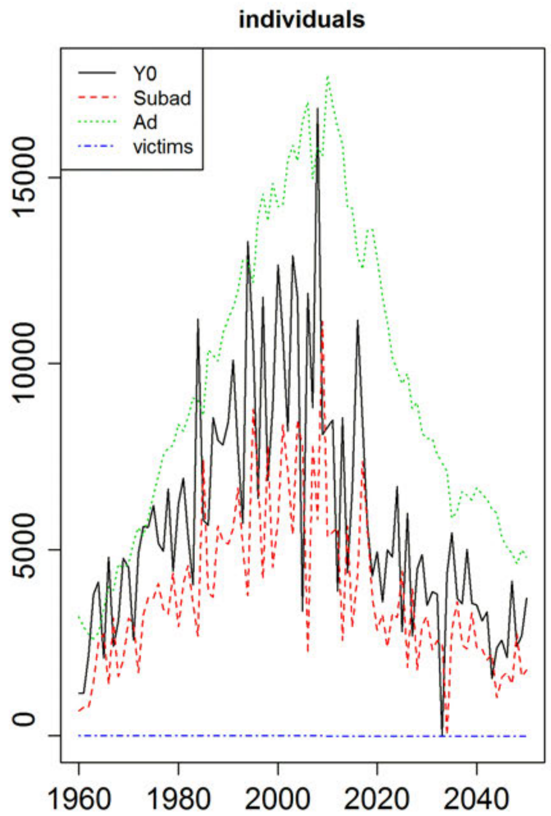
S0 : 0.66 S0sd : 0.01
S1 : 0.822 S0sd : 0.01
Sad: 0.822 Sadsd: 0.079
F0 : 0.5 F0sd : 0.185

K: 15000

Init population: 502.5 502.5 1675
Victims: 0 0 11







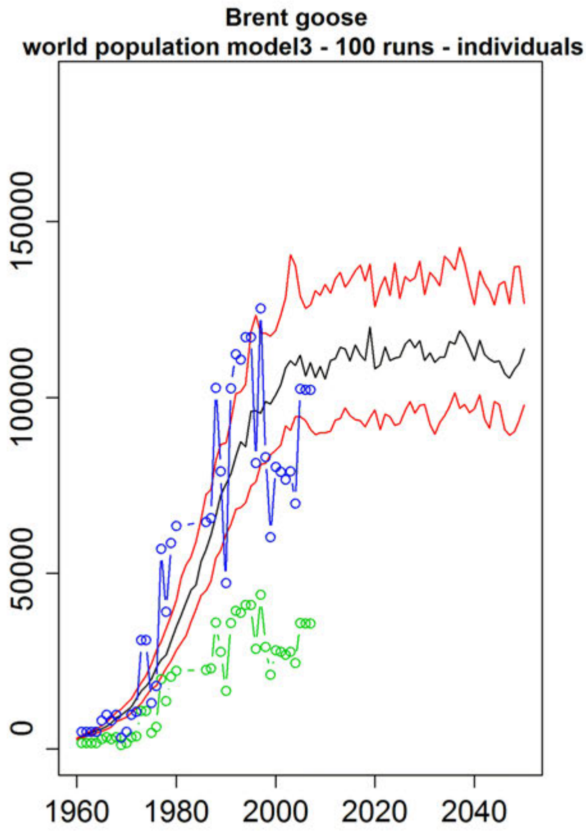
Brent goose – NW-Europe

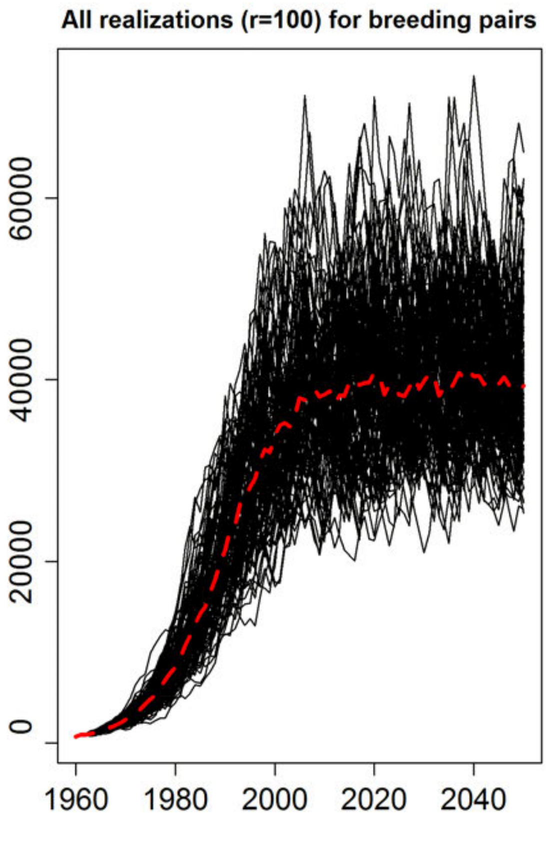
0-model

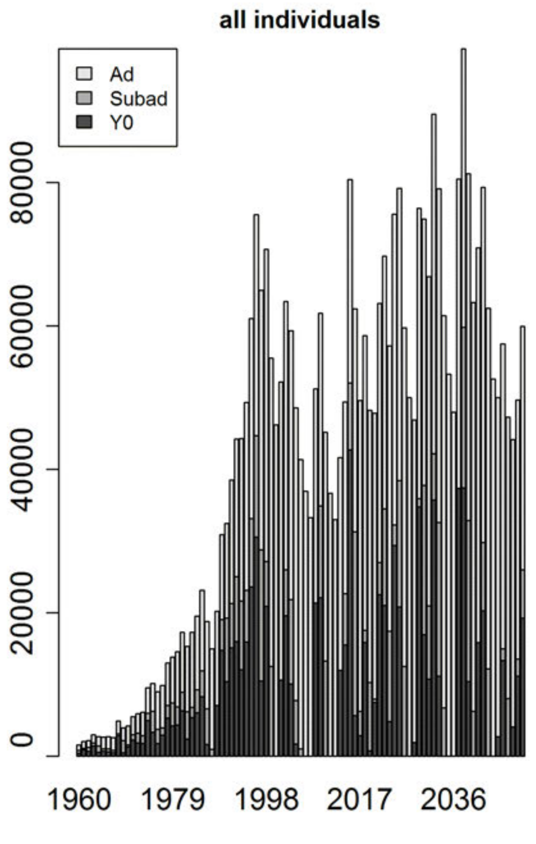
S0 : 0.6 S0sd : 0.01
Sad: 0.9 Sadsd: 0.036
F0 : 0.875 F0sd : 0.39
B1 : 2

K: 35000

Init population: 700 700 350





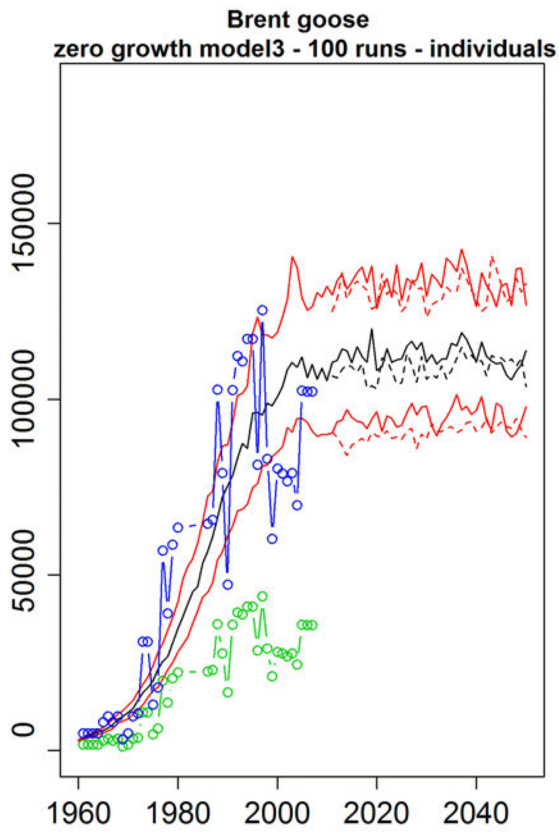


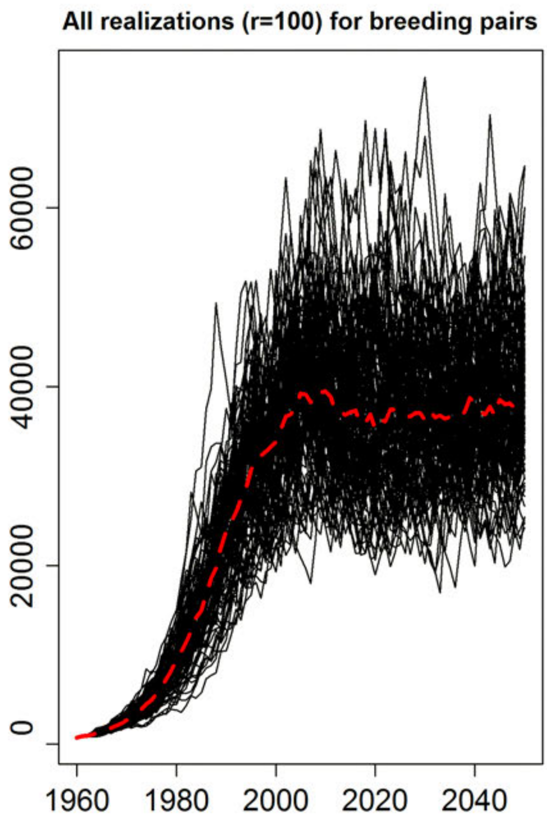
Zero growth

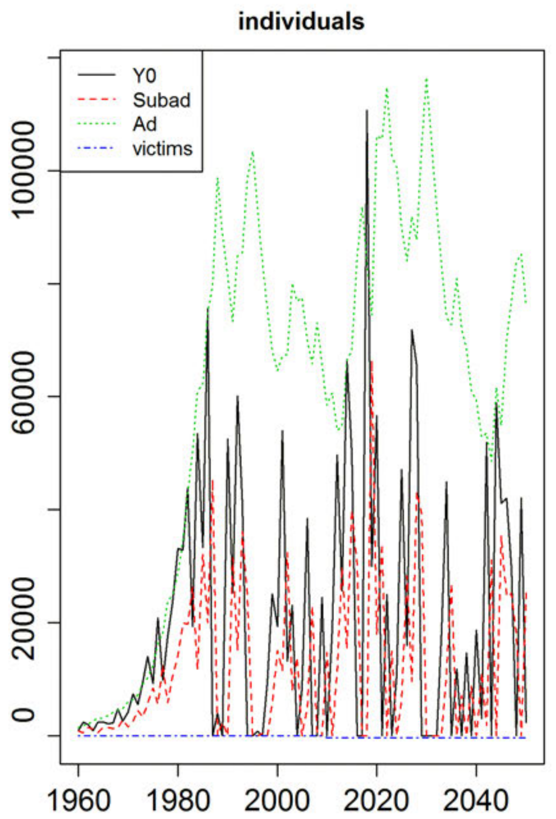
S0 : 0.6 S0sd : 0.01
Sad: 0.9 Sadsd: 0.036
F0 : 0.875 F0sd : 0.39

K: 35000

Init population: 700 700 350
Victims: 0 0 350





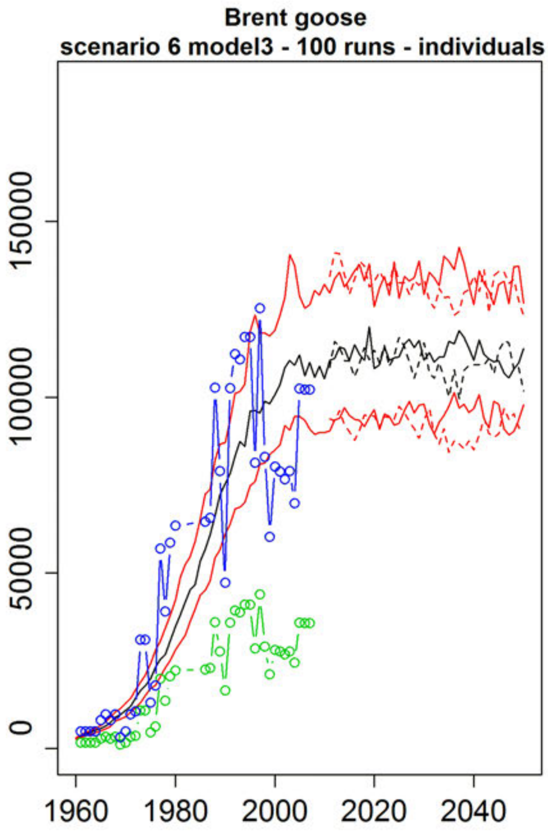


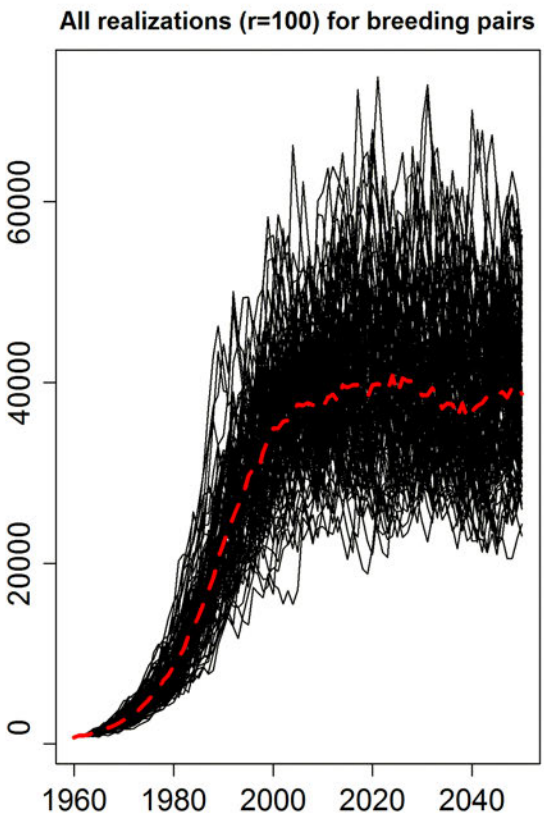
Scenario collissions Band-model

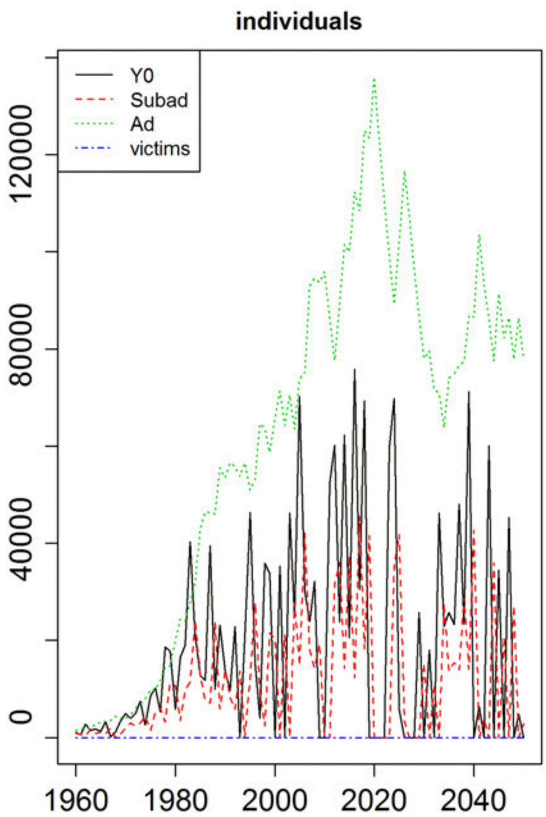
S0 : 0.6 S0sd : 0.01
Sad: 0.9 Sadsd: 0.036
F0 : 0.875 F0sd : 0.39

K: 35000

Init population: 700 700 350
Victims: 0 0 5





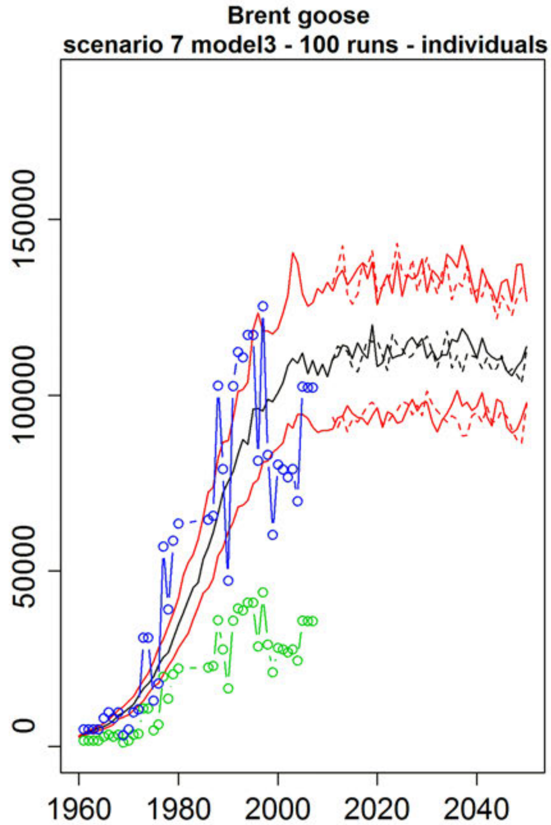


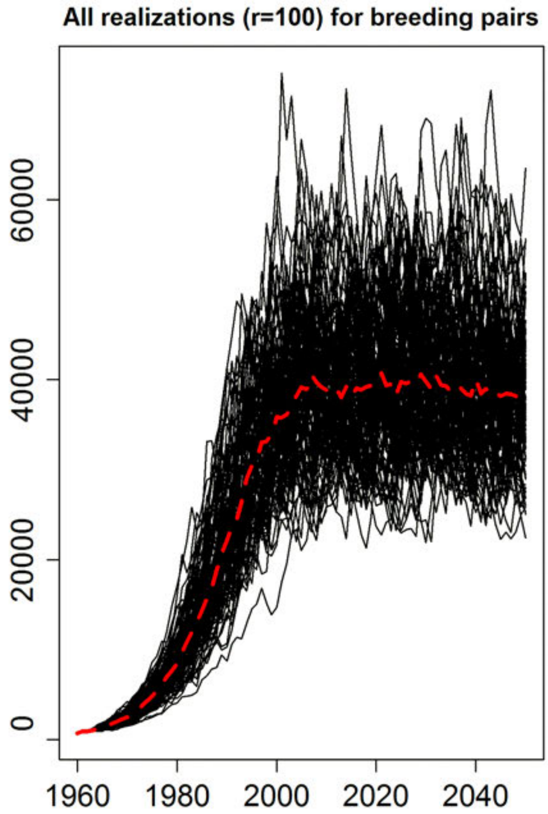
Scenario maximum effect

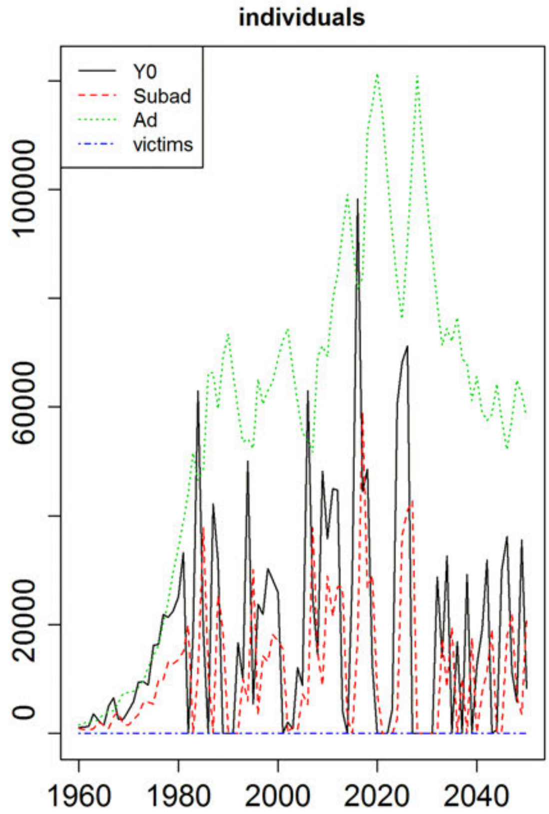
S0 : 0.6 S0sd : 0.01
Sad: 0.9 Sadsd: 0.036
F0 : 0.875 F0sd : 0.39

K: 35000

Init population: 700 700 350
Victims: 0 0 11







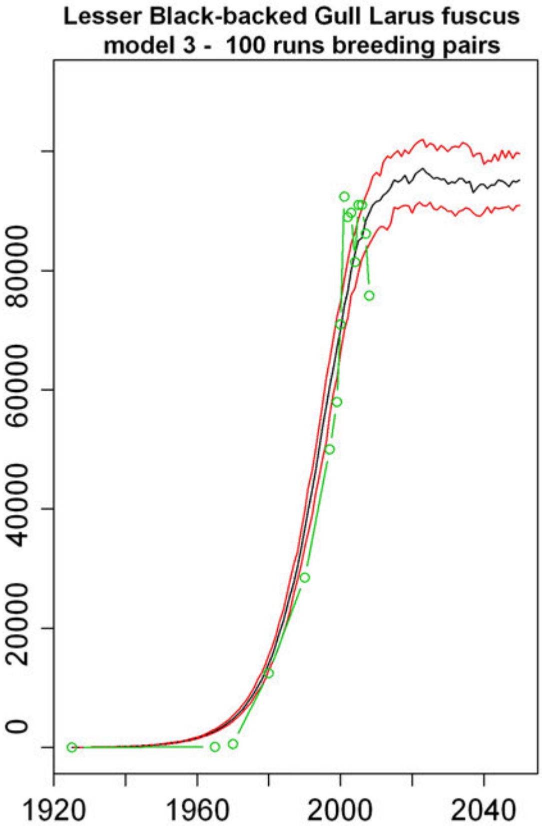
Lesser black-backed gull – the Netherlands

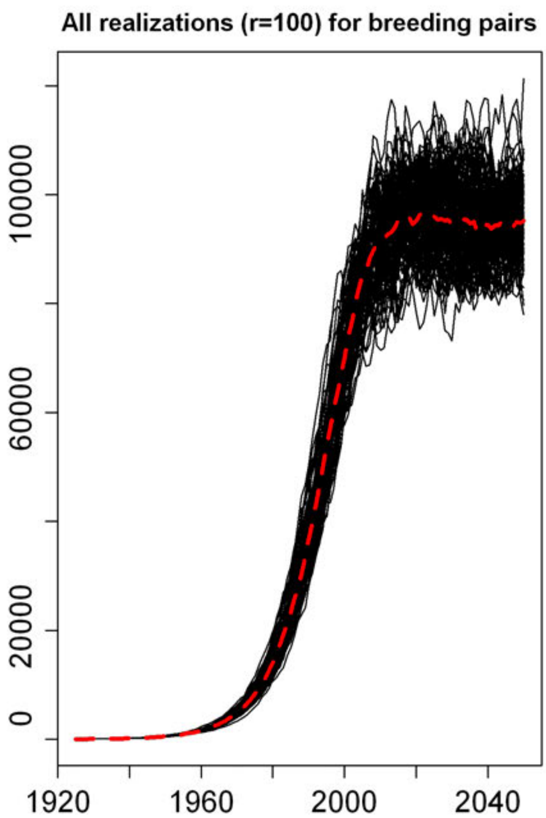
0-model – 0% floaters

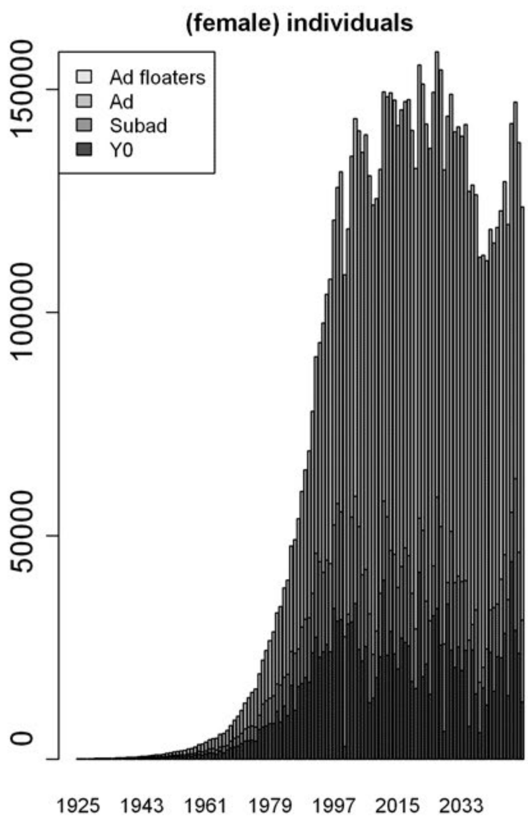
S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.69 F0sd : 0.1
B1 : 4
Floaters % >=2010: 0

K: 92400

Init population: 50 25 15 2





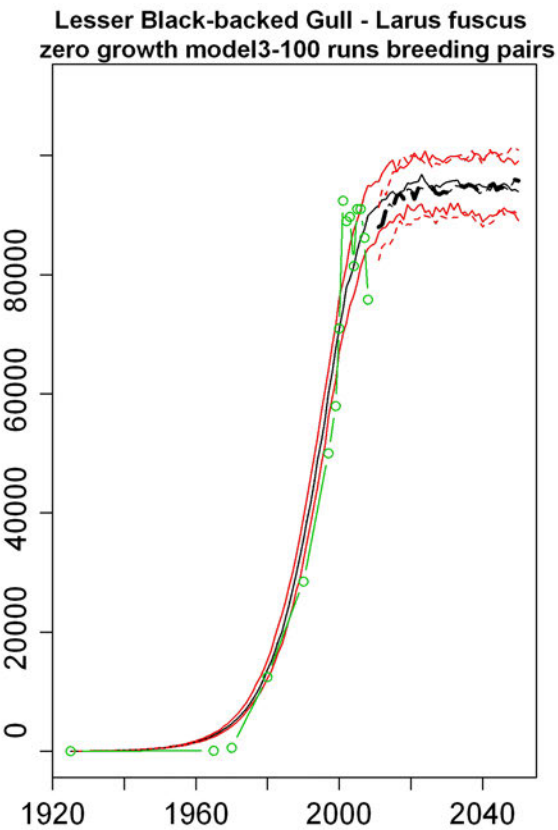


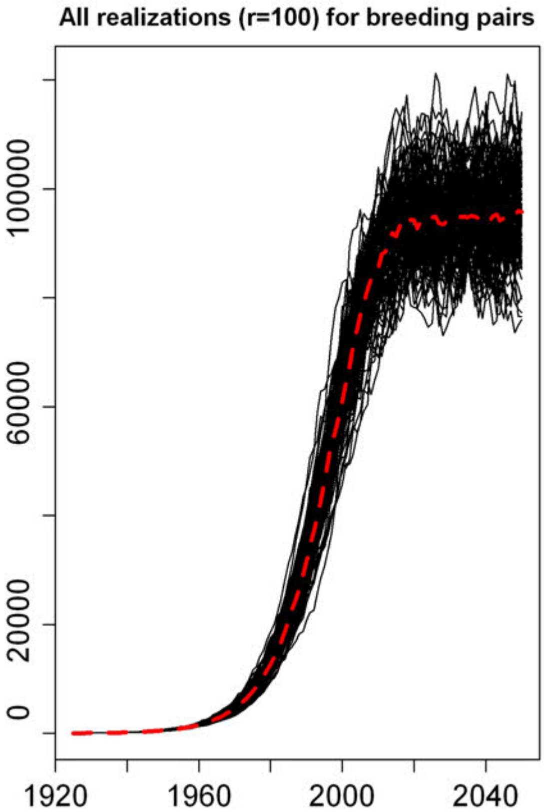
Zero growth – 0% floaters

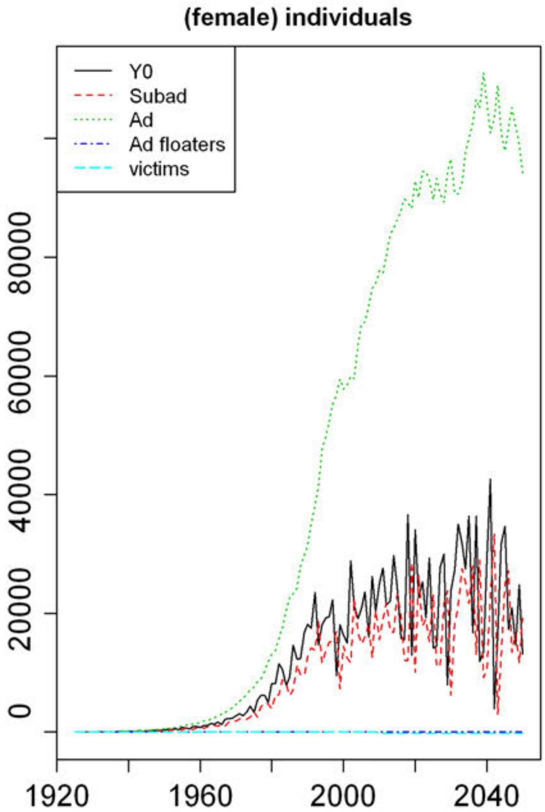
S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.69 F0sd : 0.1
B1 : 4
Floaters % >=2010: 0

K: 92400

Init population: 50 25 15 2



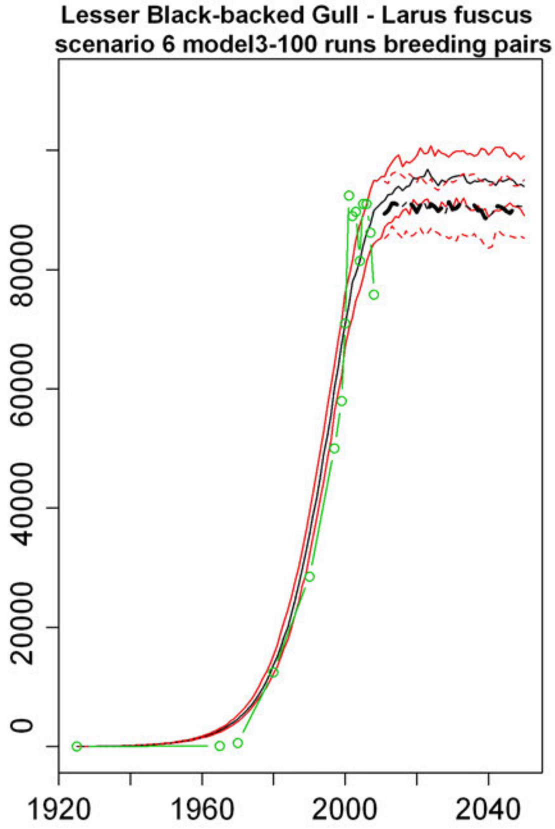


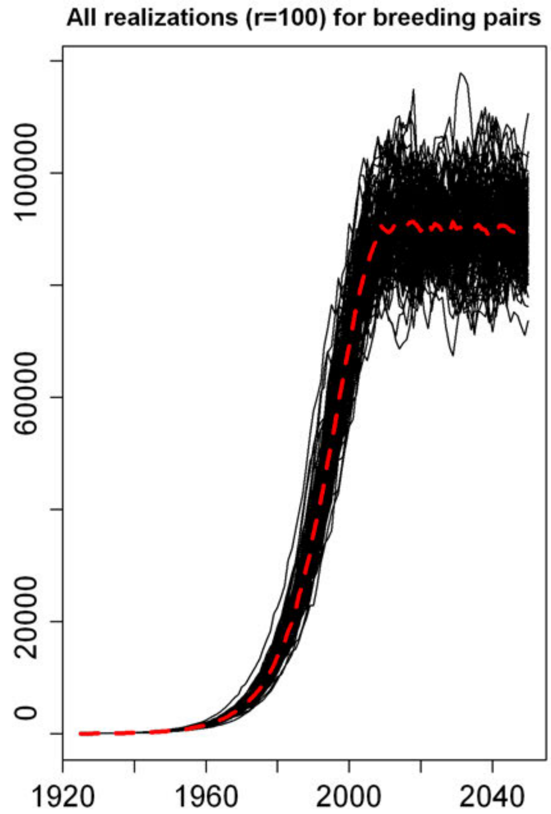


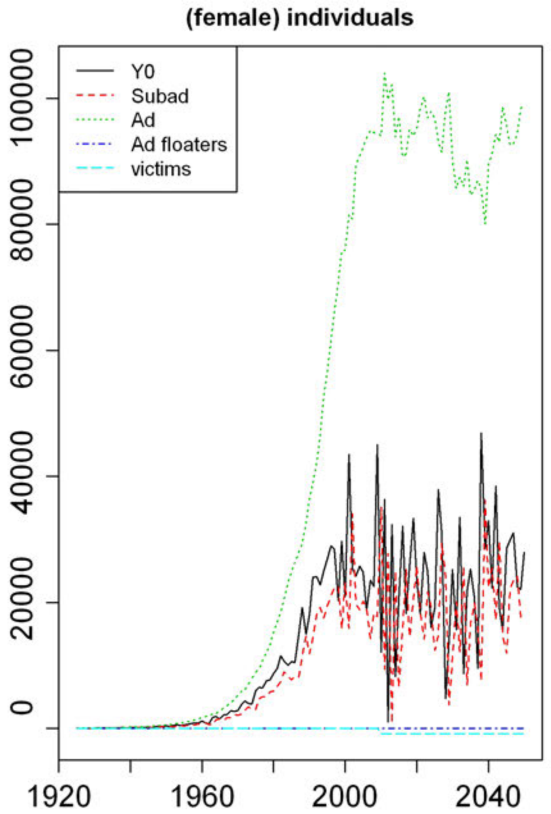
Scenario collisions Band-model – 0 % floaters

S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.69 F0sd : 0.1
B1 : 4
Floaters % >=2010: 0

K: 92400
----
victims : 0 0 875 0
Init population: 50 25 15 2





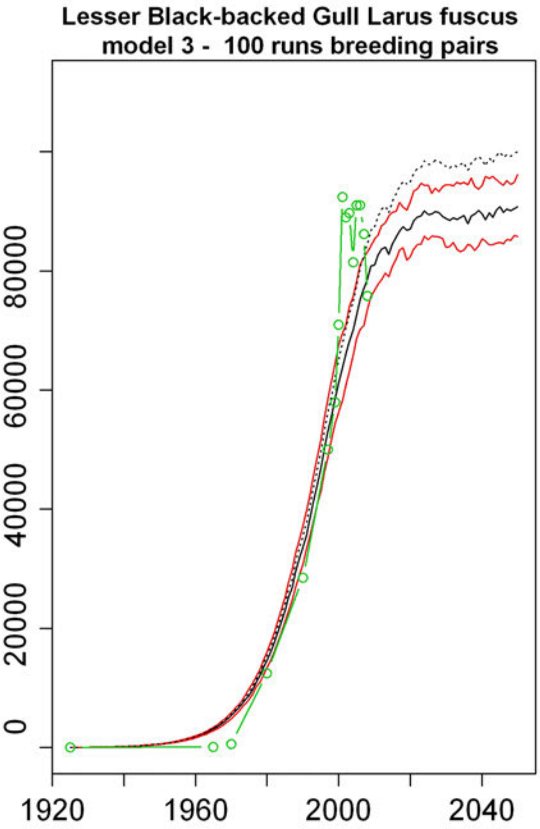


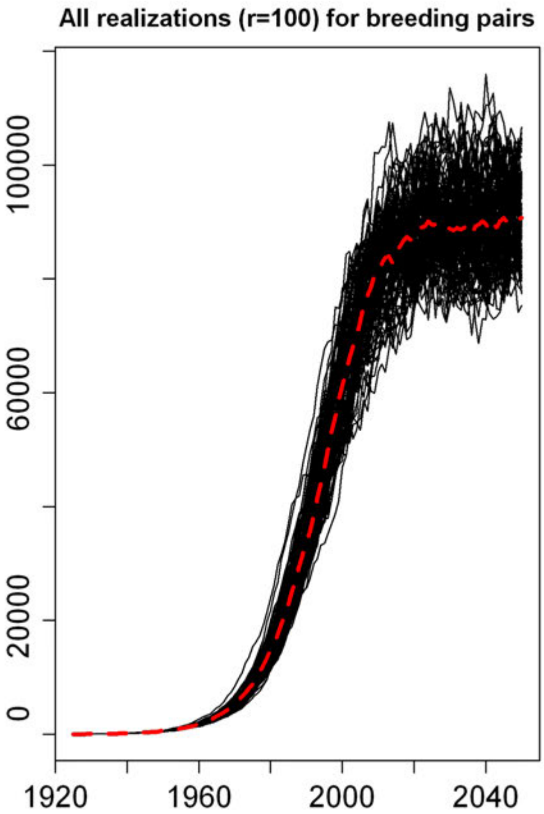
0-model – 10% floaters

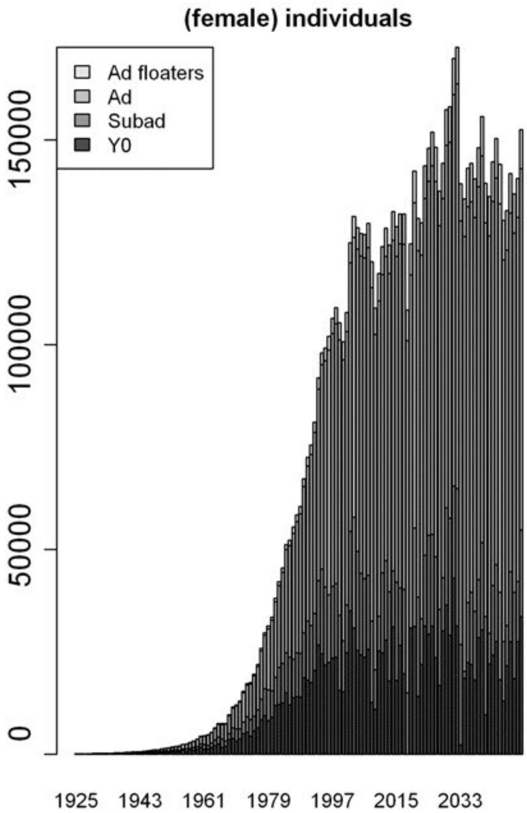
S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.75 F0sd : 0.1
B1 : 4
Floaters % >=2010: 10

K: 92400

Init population: 50 25 15 2





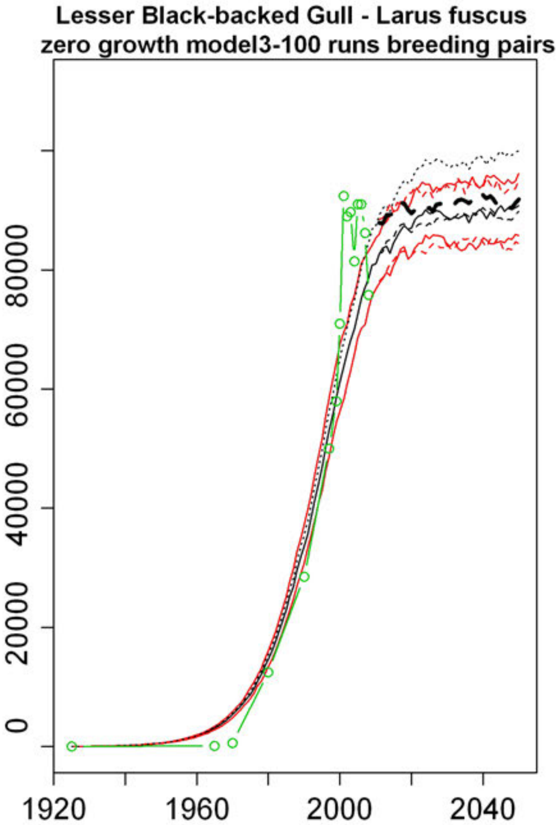


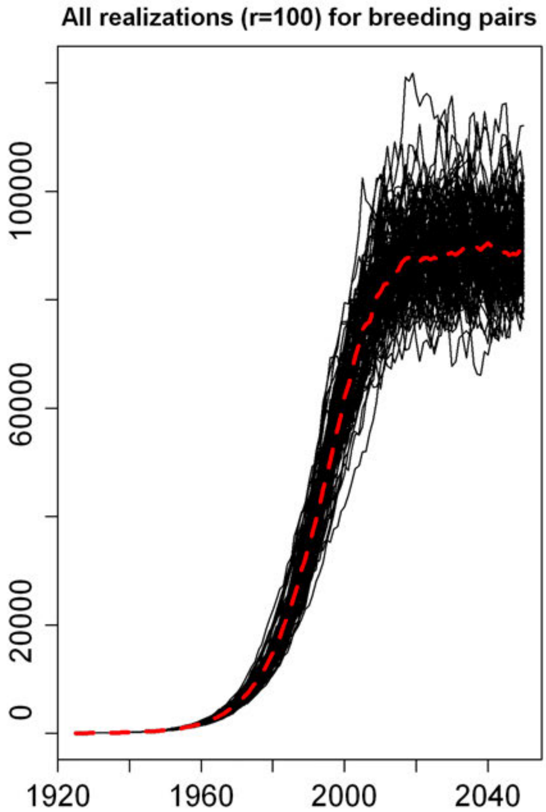
Zero growth – 10% floaters

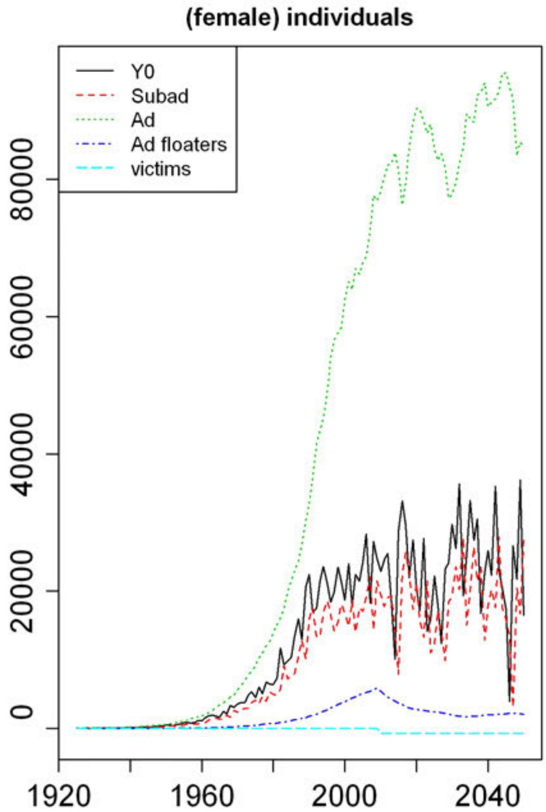
S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.75 F0sd : 0.1
B1 : 4
Floaters % >=2010: 3

K: 92400

Init population: 50 25 15 2





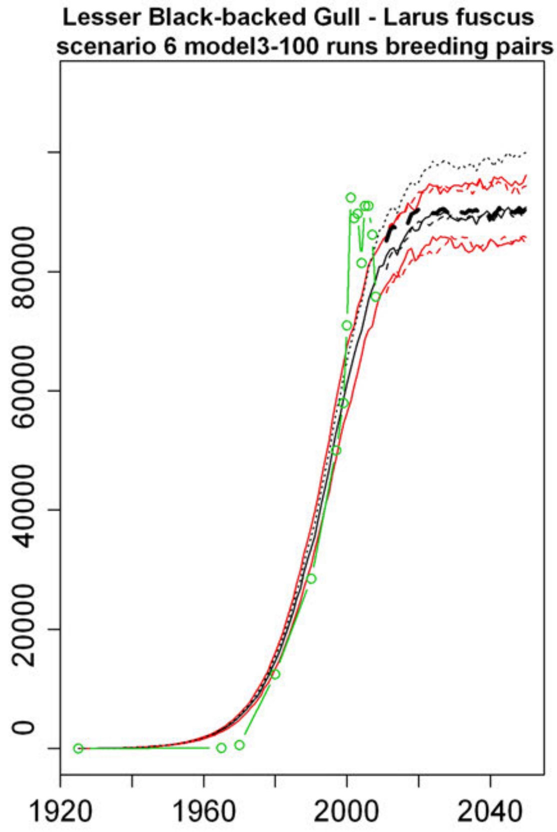


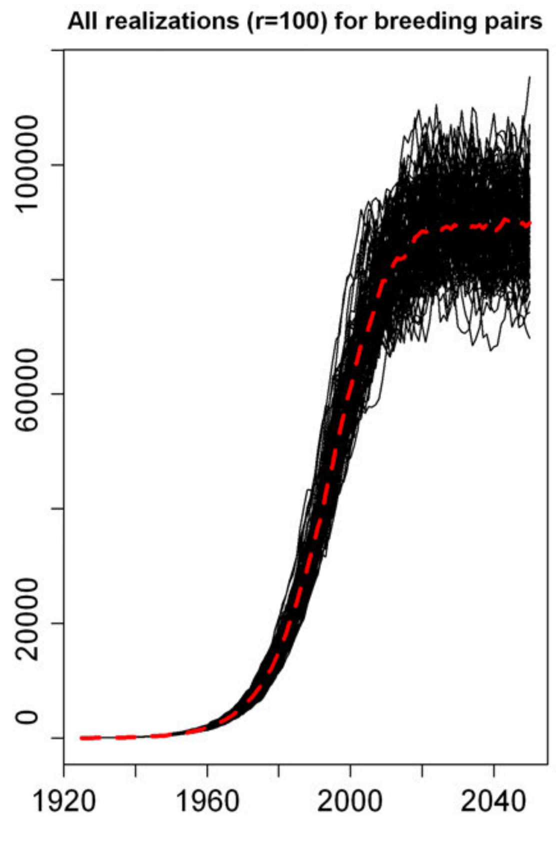
scenario collisions Band-model – 10% floaters

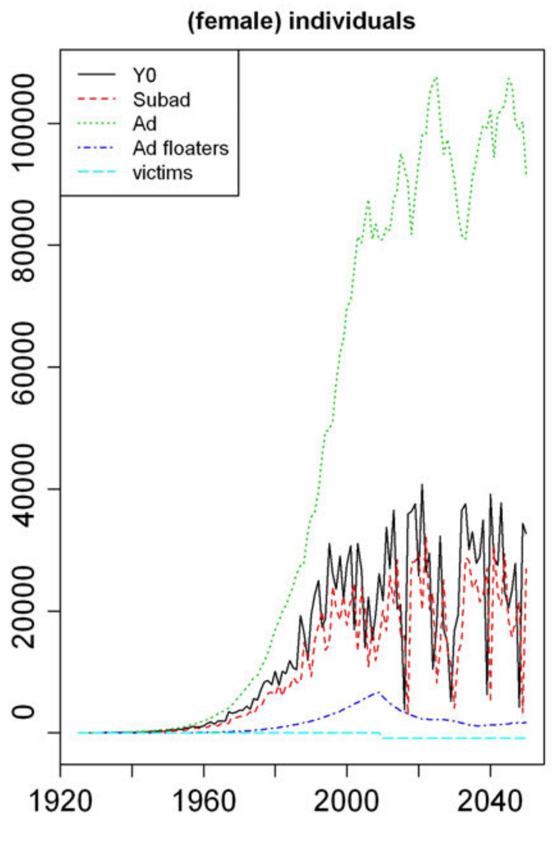
S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.75 F0sd : 0.1
B1 : 4
Floaters % >=2010: 2

K: 92400

Init population: 50 25 15 2
victims : 0 0 875 0





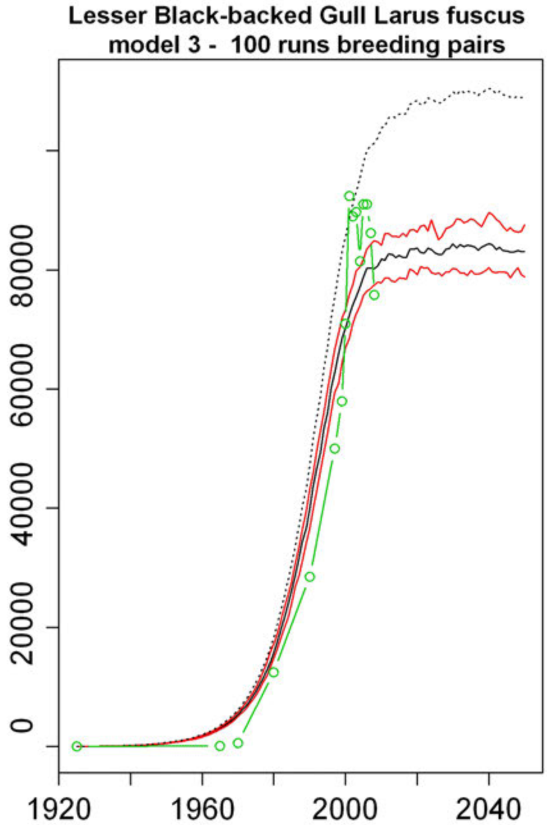


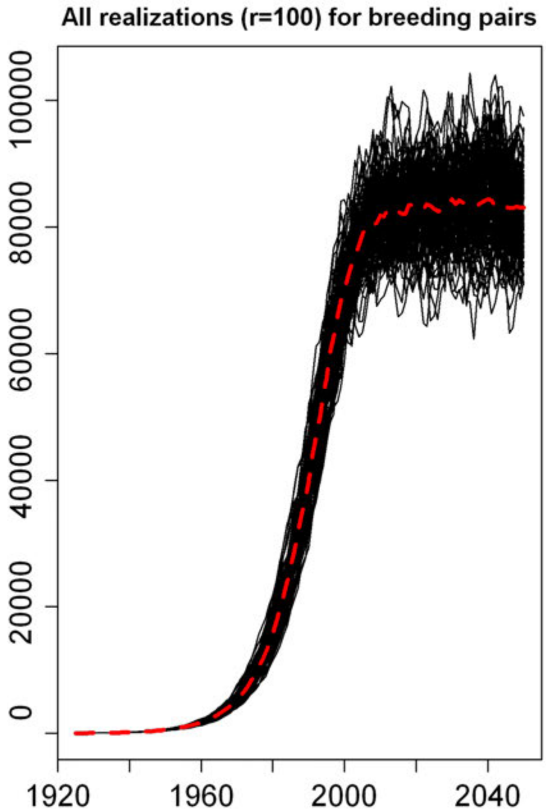
0 scenario - 30% floaters

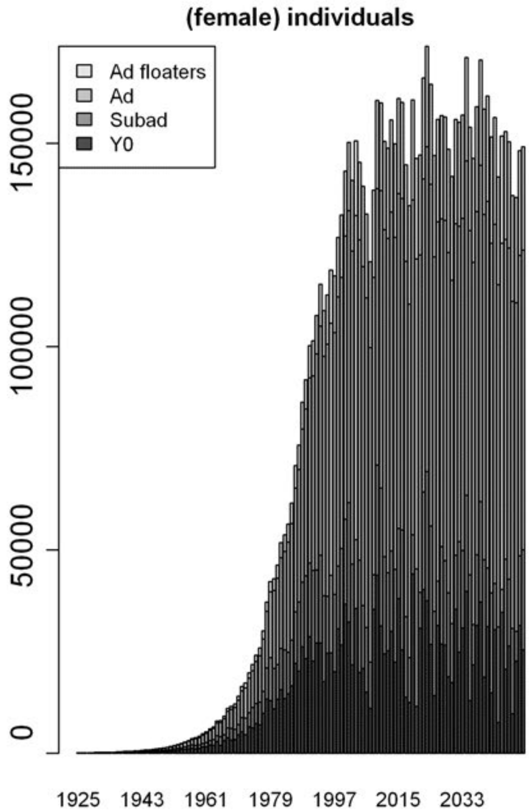
S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.825 F0sd : 0.1
B1 : 4
Floaters % >=2010: 30

K: 92400

Init population: 50 25 15 2





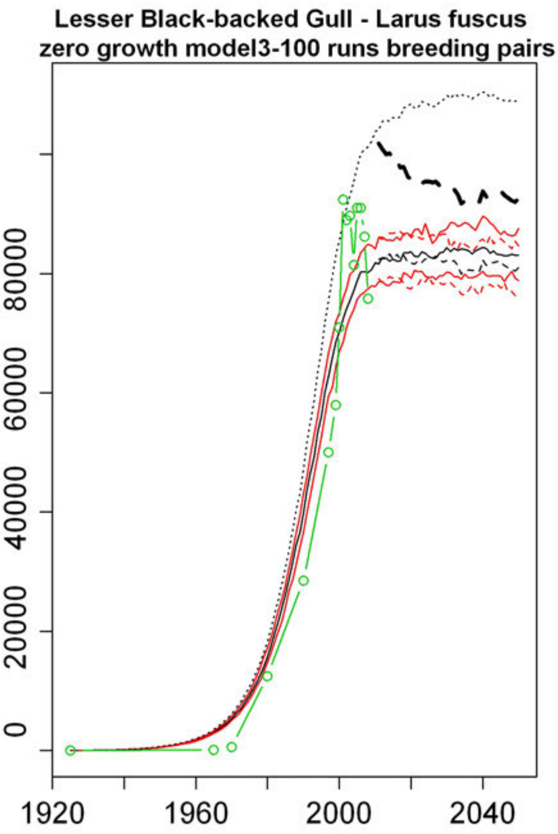


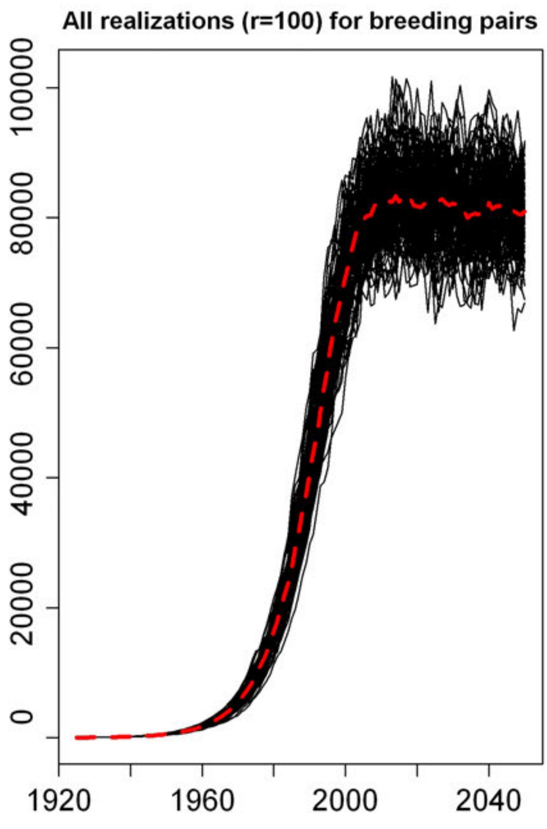
zero growth – 30% floaters

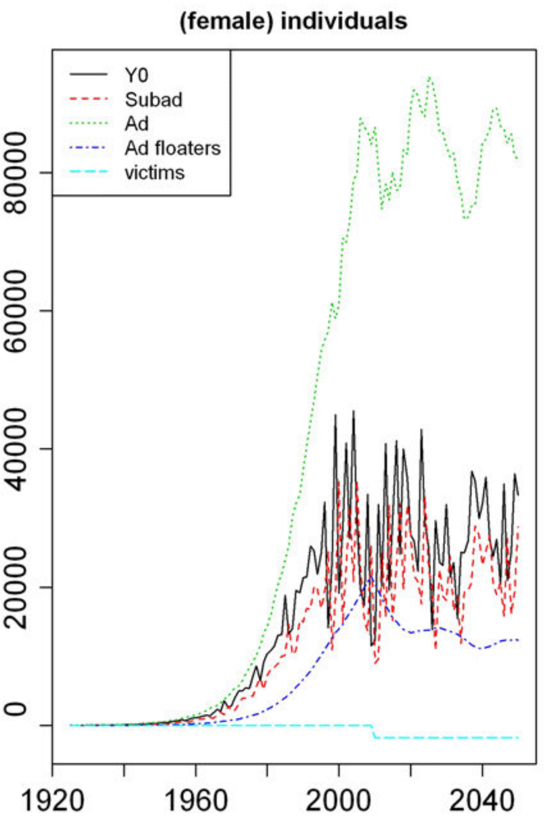
S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.825 F0sd : 0.1
B1 : 4
Floaters % >=2010: 16

K: 92400

Init population: 50 25 15 2







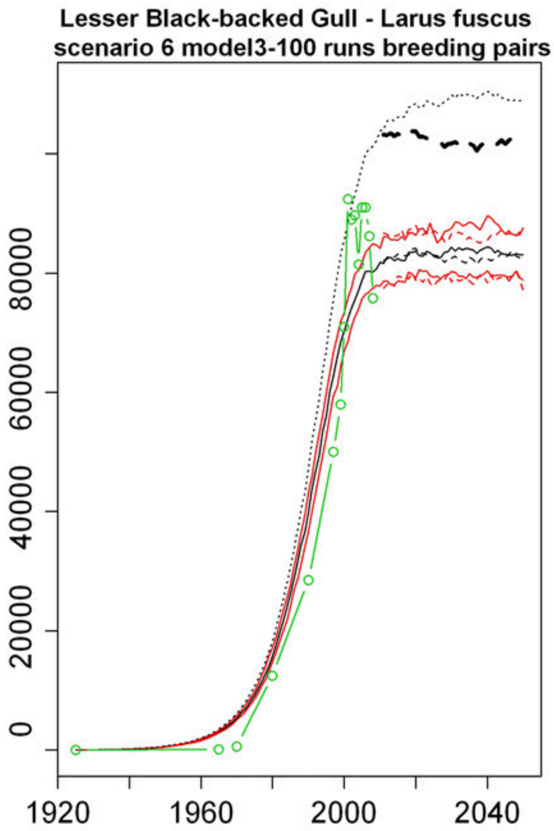
Lesser black-backed gull

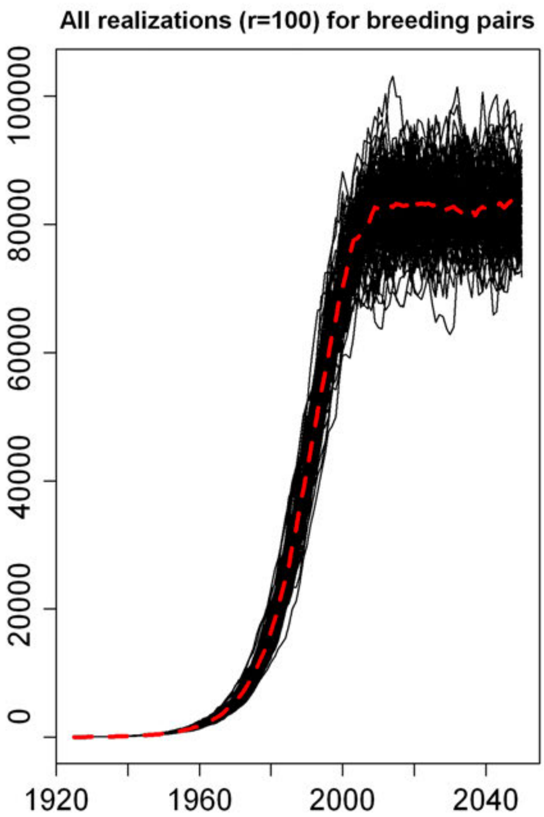
scenario collisions Band-model – 30% floaters

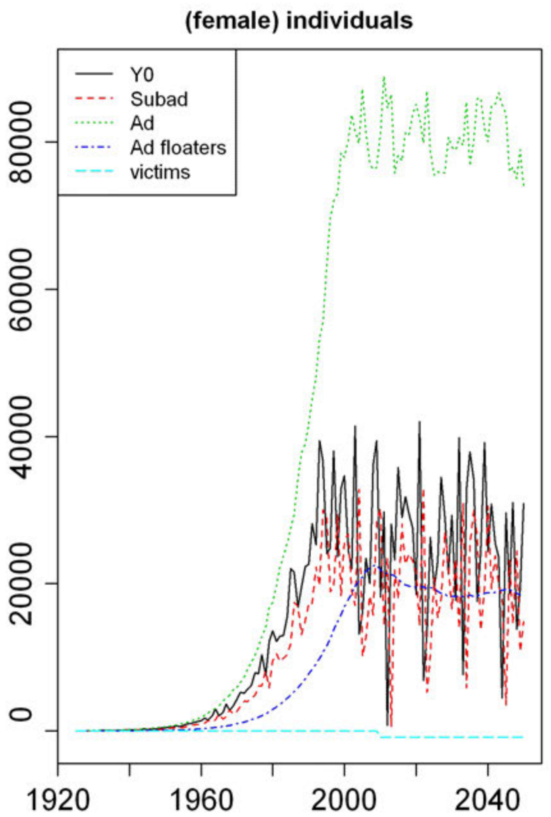
S0 : 0.78 S0sd : 0.01
Sad: 0.913 Sadsd: 0.012
F0 : 0.825 F0sd : 0.1
B1 : 4
Floaters % >=2010: 24

K: 92400

Init population: 50 25 15 2
victims : 0 0 875 0







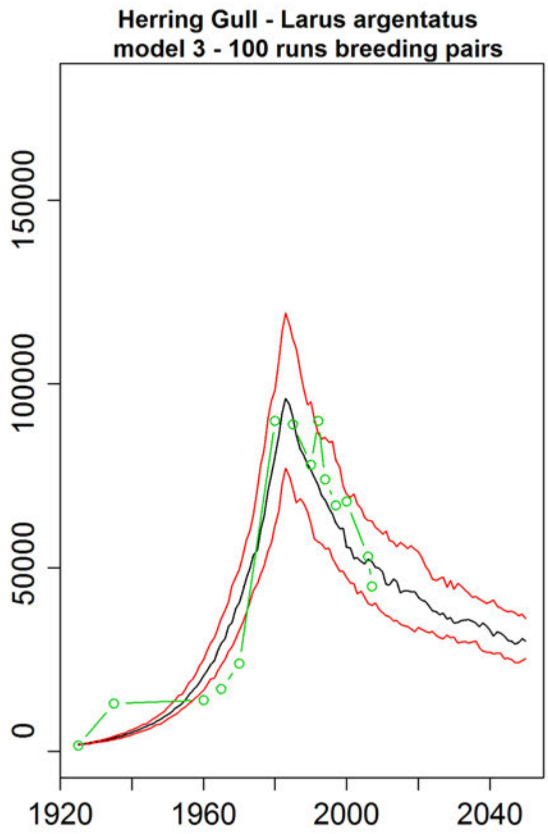
Herring gull – the Netherlands

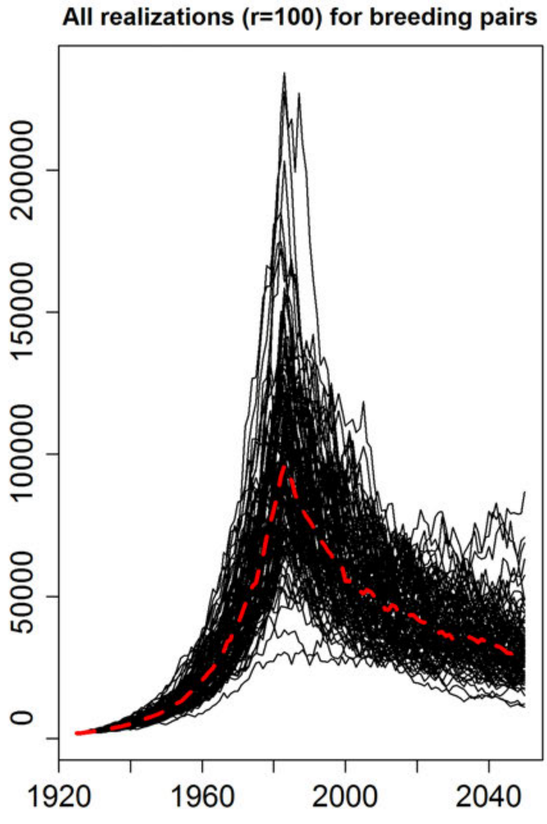
0-Scenario – 0% floaters

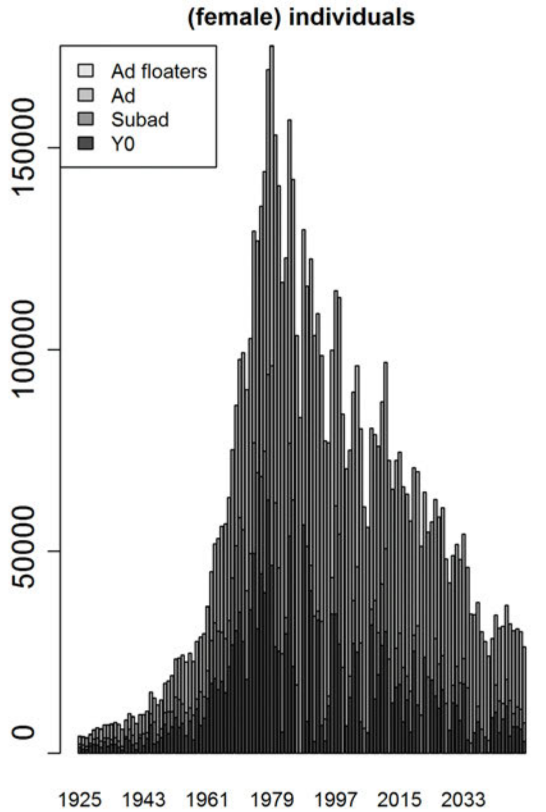
S0 : 0.78 S0sd : 0.01
Sad: 0.88 Sadsd: 0.013
F0 : 0.675 F0sd : 0.25
B1: 4.5
Floaters % : 0

K (> 1981): 20000

Init population: 1000 1000 1600 160





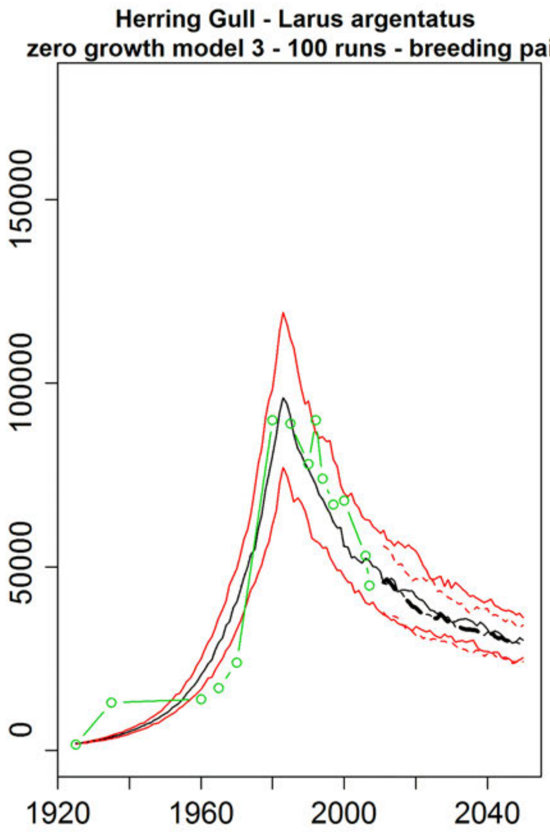


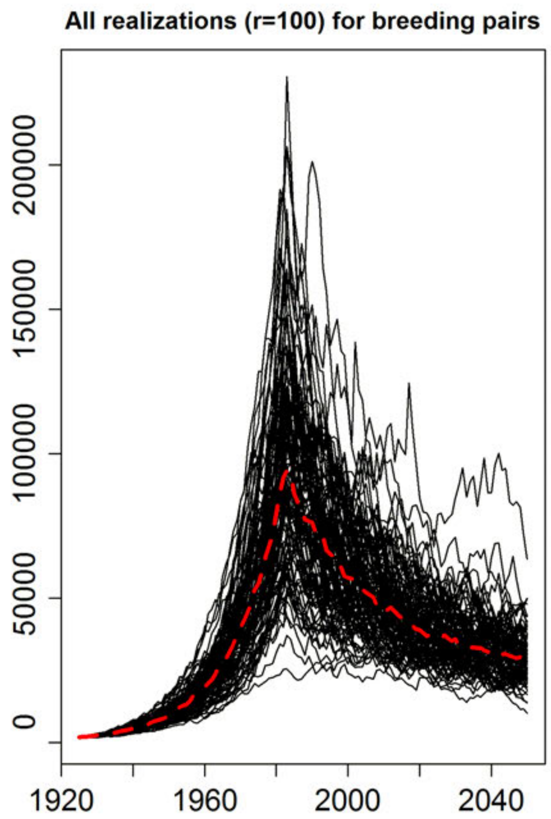
Zero growth – 0% floaters

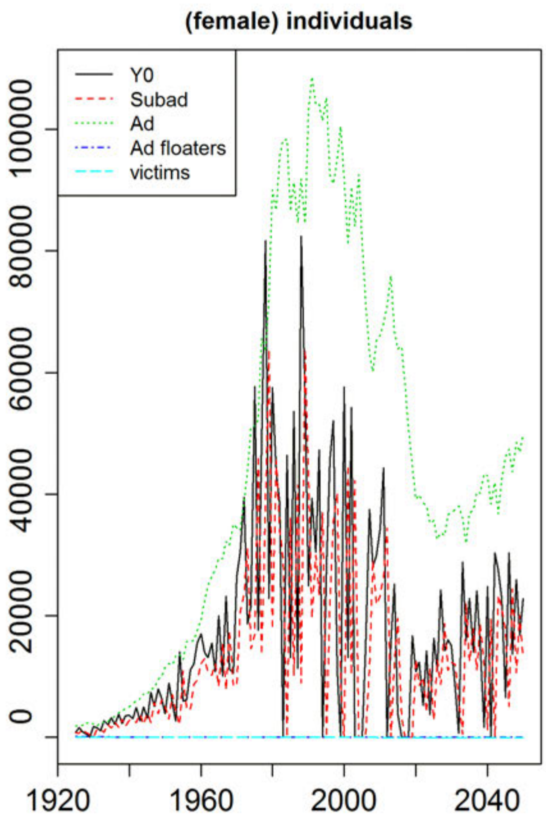
S0 : 0.78 S0sd : 0.01
Sad: 0.88 Sadsd: 0.013
F0 : 0.675 F0sd : 0.25
B1: 4.5
Floaters % >=2010: 0

K (> 1981): 20000

Init population: 1000 1000 1600 0
Victims: 0 0 62.5 0





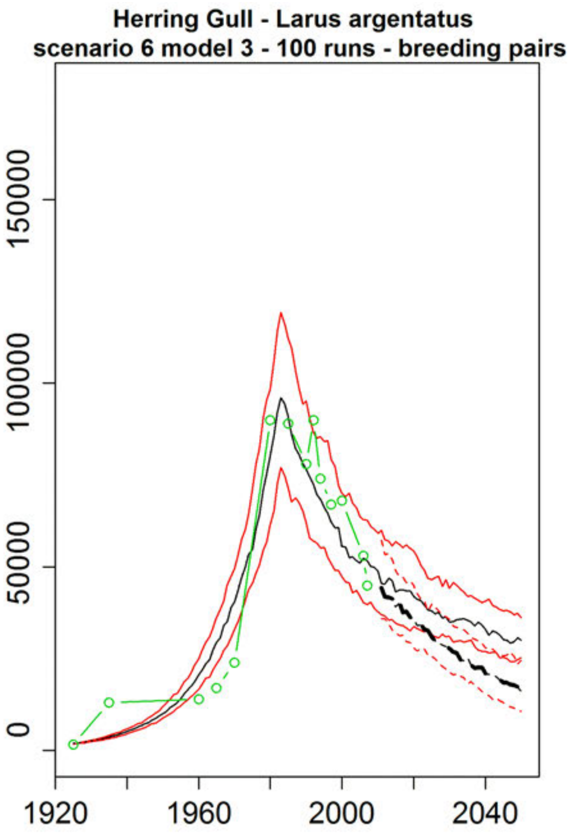


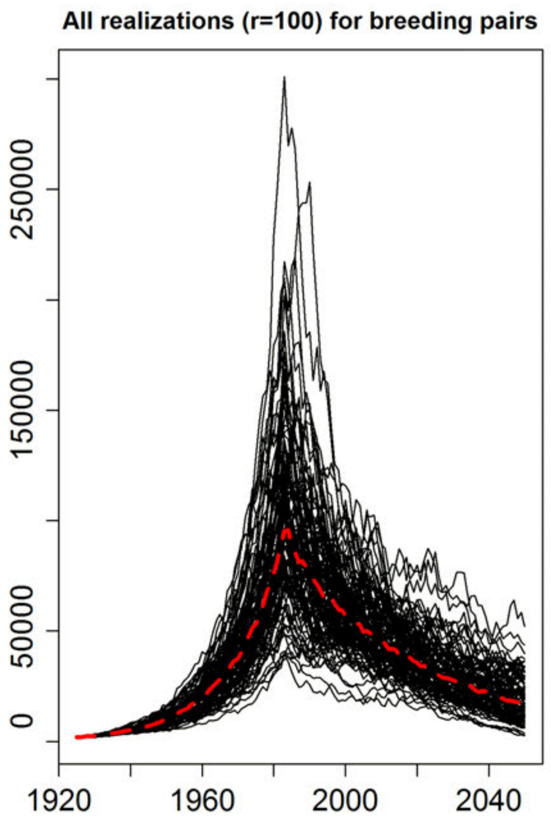
Scenario collisions Band-model – 0% floaters

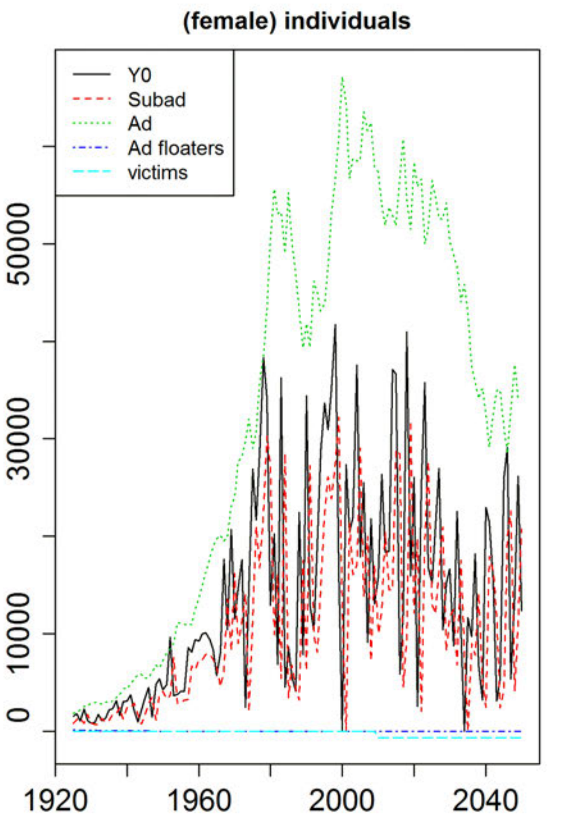
S0 : 0.78 S0sd : 0.01
Sad: 0.88 Sadsd: 0.013
F0 : 0.675 F0sd : 0.25
B1: 4.5
Floaters % >=2010: 0

K (> 1981): 20000

Init population: 1000 1000 1600 160
Victims: 0 0 698 0







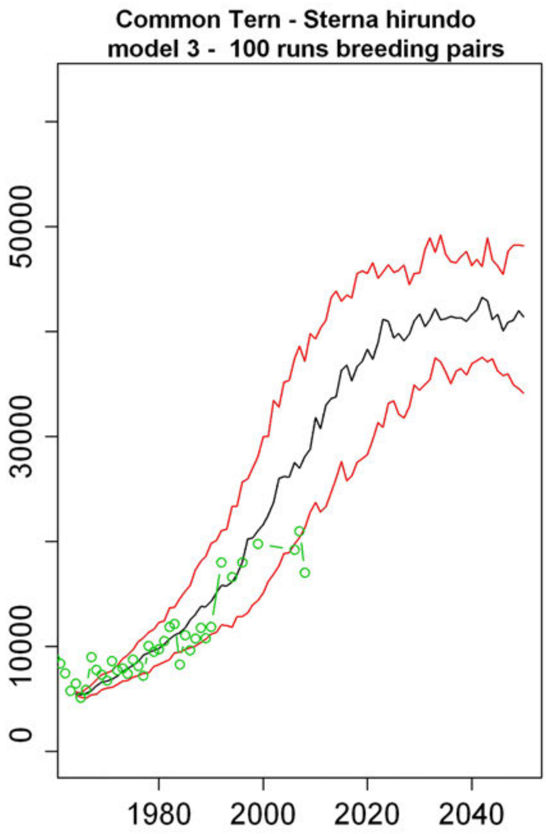
Common tern – the Netherlands

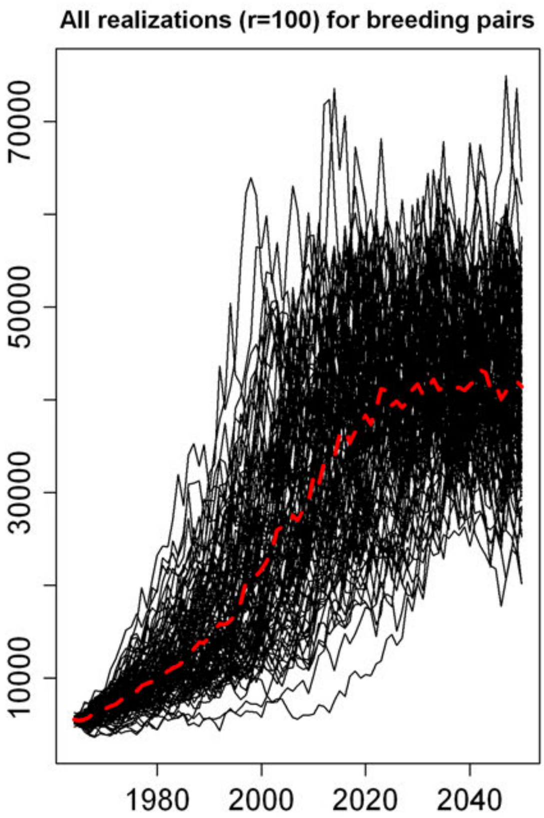
0 model - 0% floaters

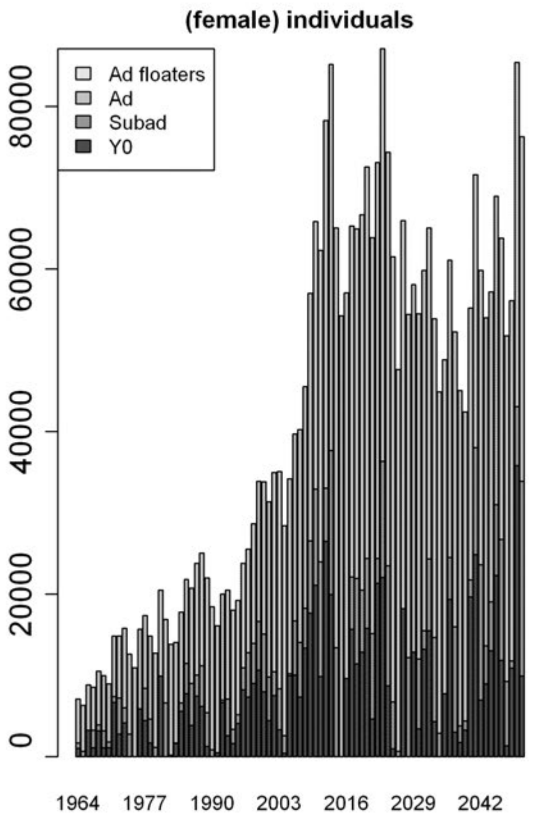
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.35 F0sd : 0.25
B1: 2
Floaters % >=2010: 0

K: 40000

Init population: 1000 1500 5000 0
victims   : 0 0 0 0





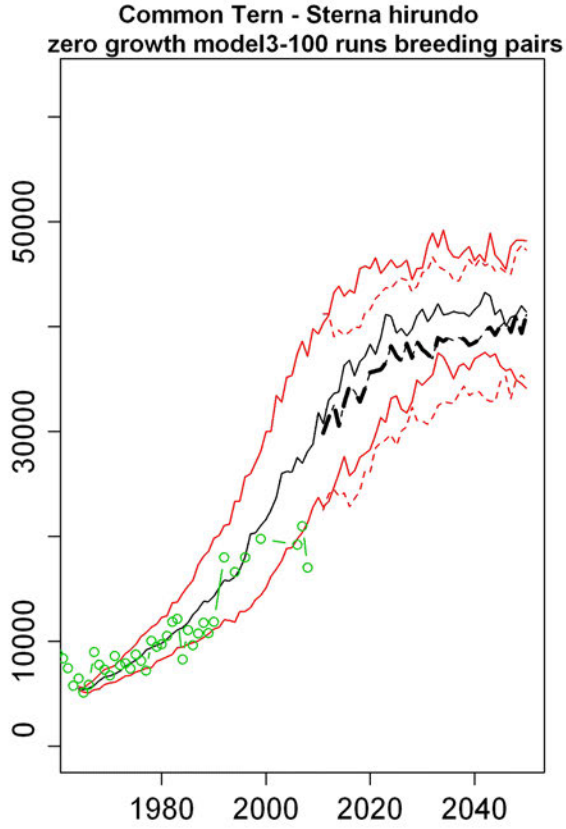


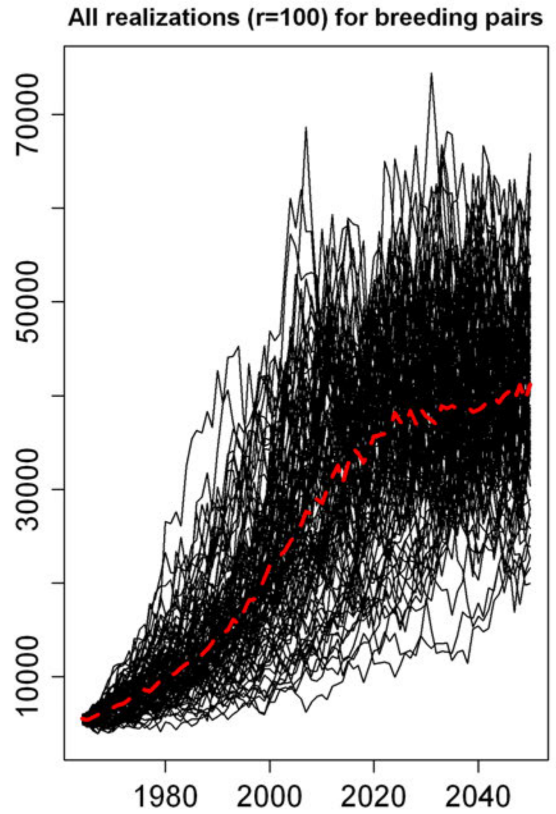
zero growth – 0% floaters

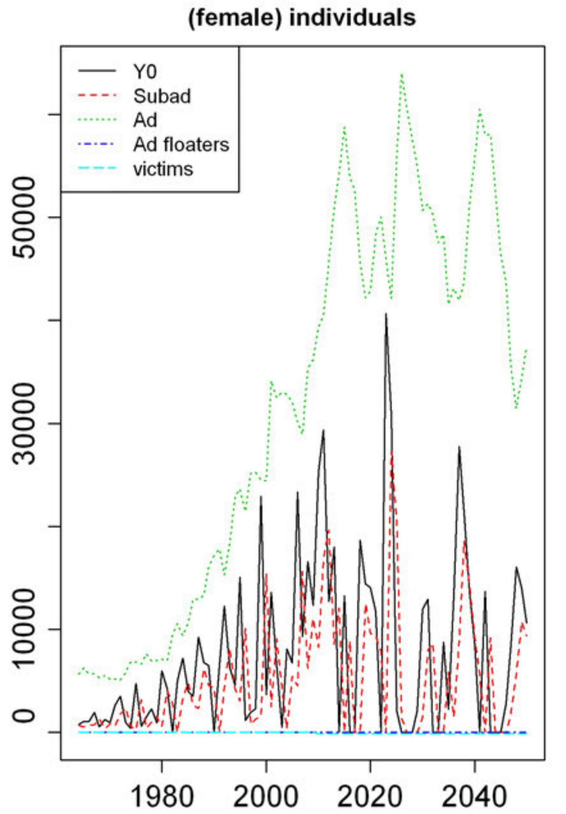
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.35 F0sd : 0.25
B1: 2
Floaters % >=2010: 0

K: 40000

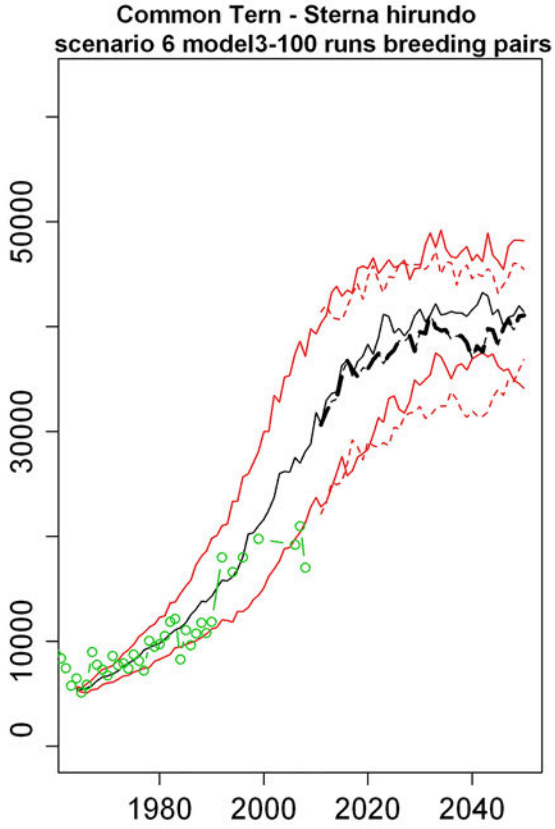
Init population: 1000 1500 5000 0
victims : 0 0 125 0

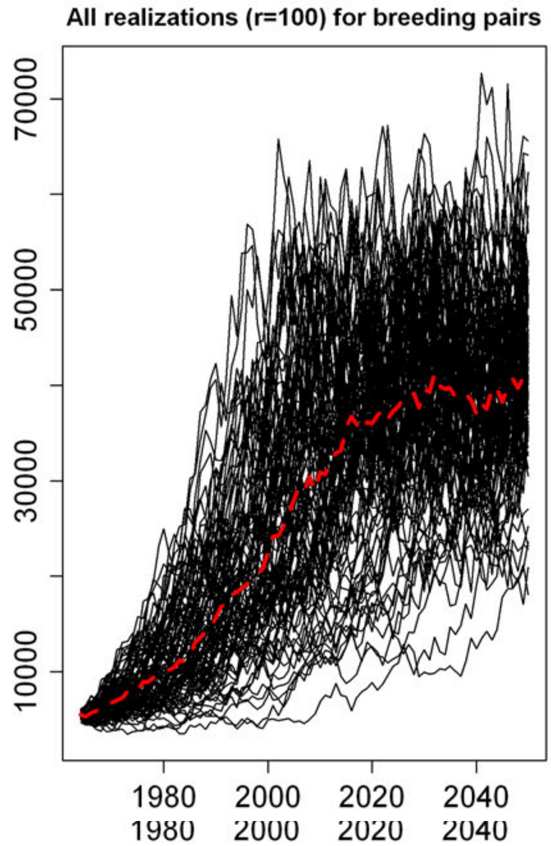


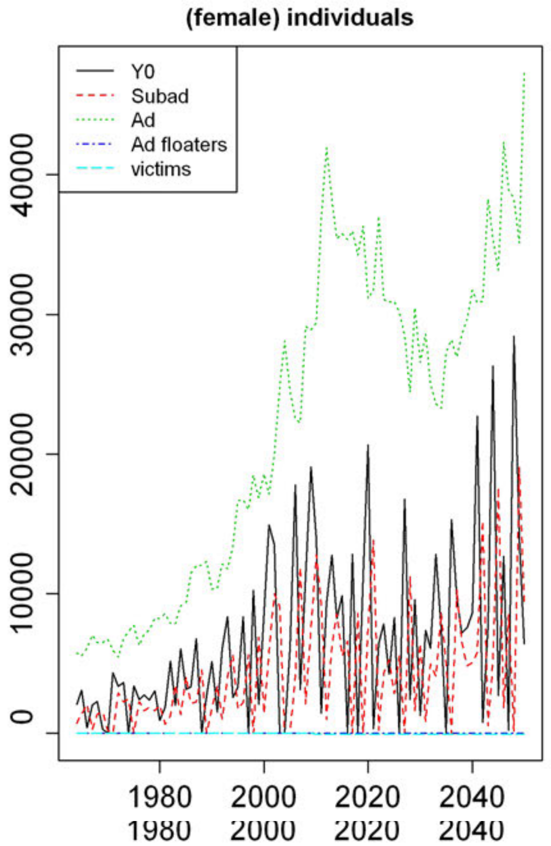




```
scenario collisions Band-model – 0% floaters
-------------------------
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.35 F0sd : 0.25
B1: 2
Floaters % >=2010: 0
-------------------------
K: 40000
-------------------------
Init population: 1000 1500 5000 0
victims        : 0 0 60 0
population     : 6382.81 9379.57 47299.34 0
-------------------------
```







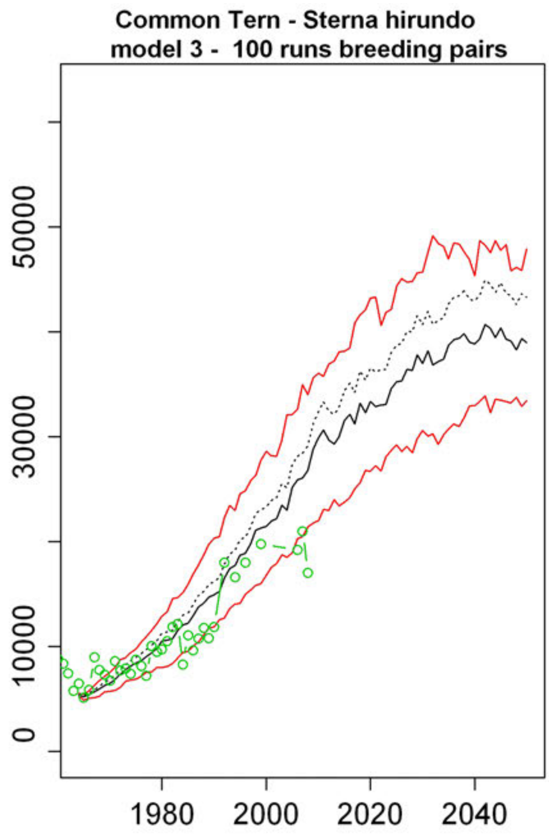
0 model – 10% floaters

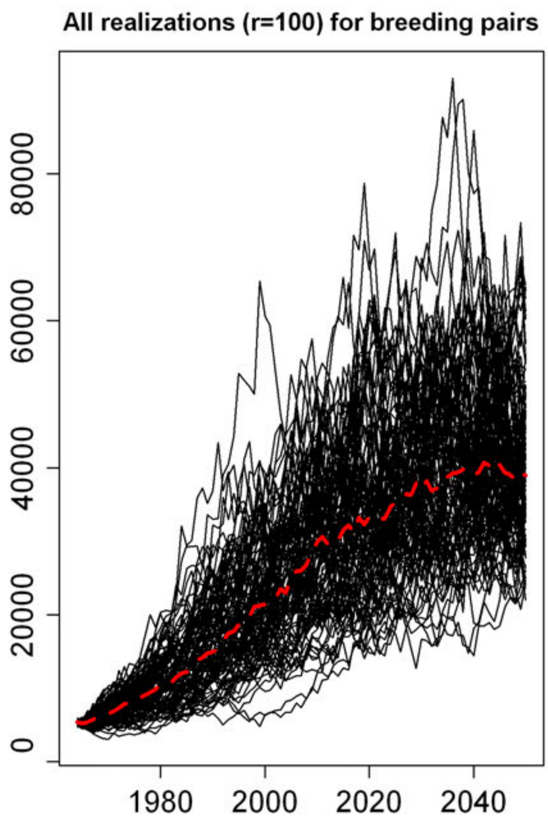
--- ----------------------

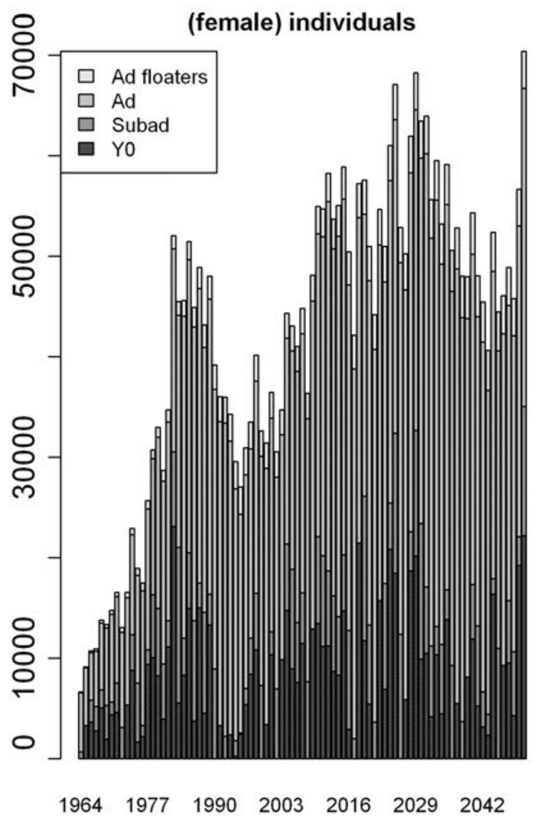
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.405 F0sd : 0.25
B1: 2
Floaters % >=2010: 10

K: 40000

Init population: 1000 1500 5000 0
victims   : 0 0 0 0





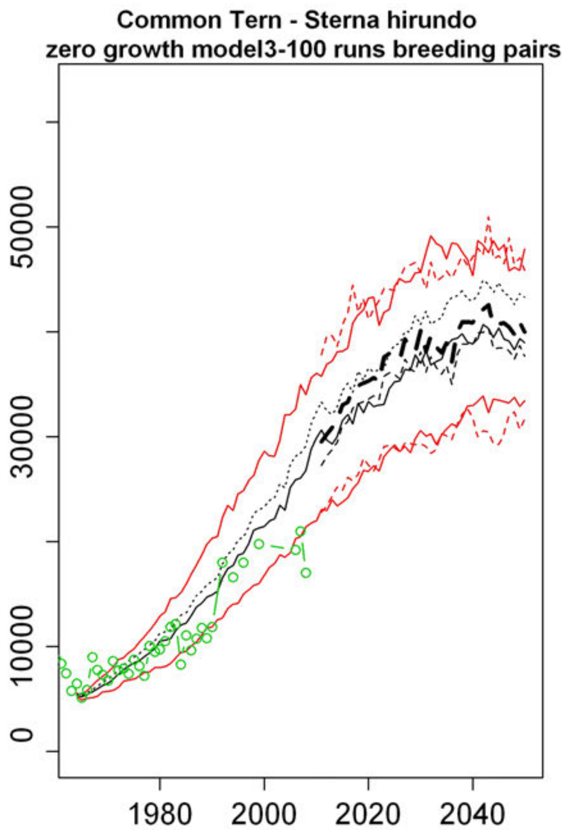


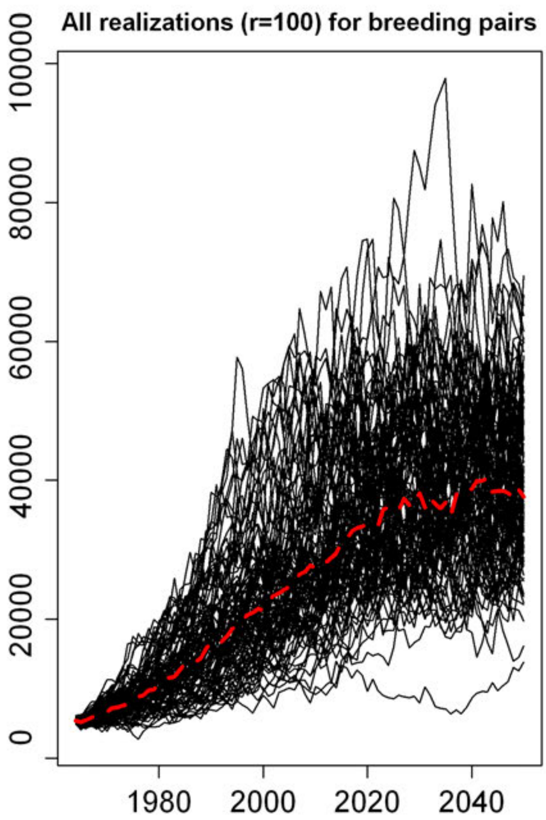
zero growth – 10% floaters

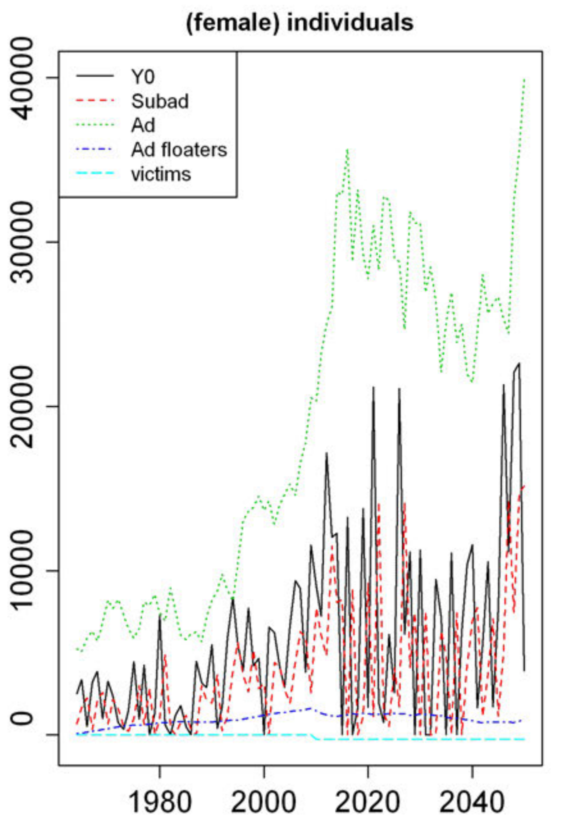
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.405 F0sd : 0.25
B1: 2
Floaters % >=2010: 6

K: 40000

Init population: 1000 1500 5000 0
victims : 0 0 250 0





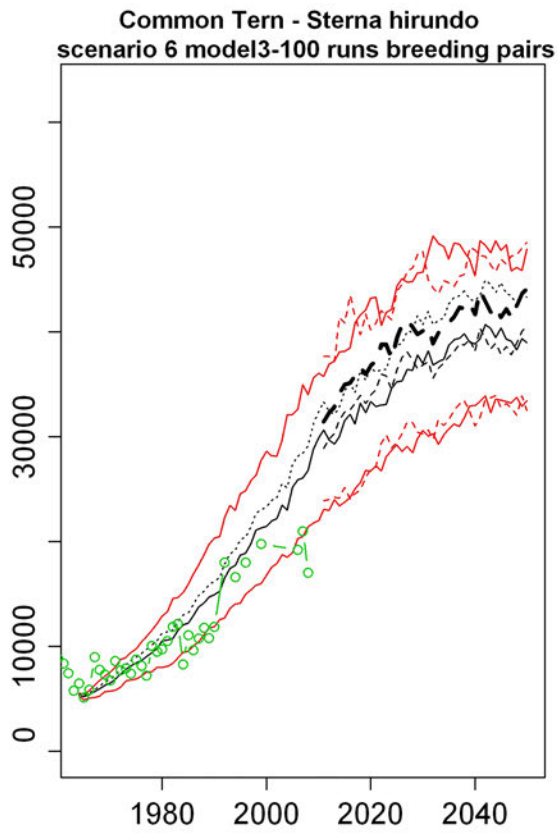


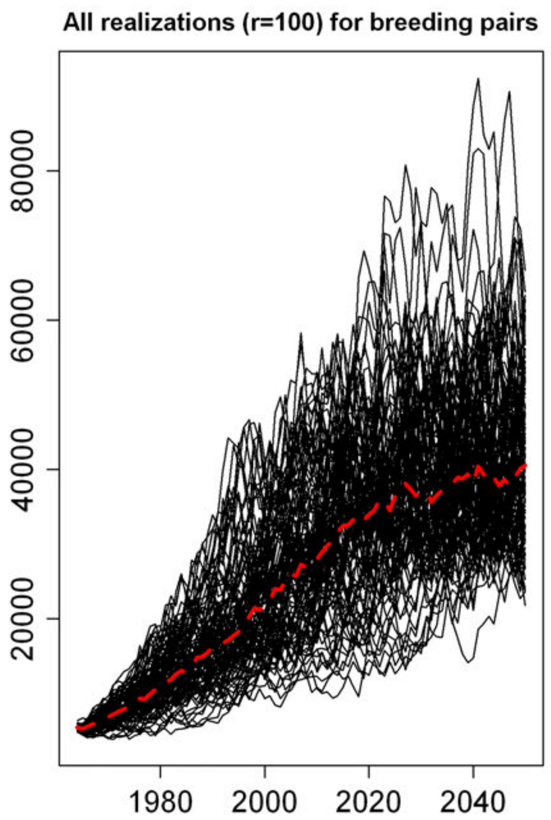
scenario collisions Band-model – 10% floaters

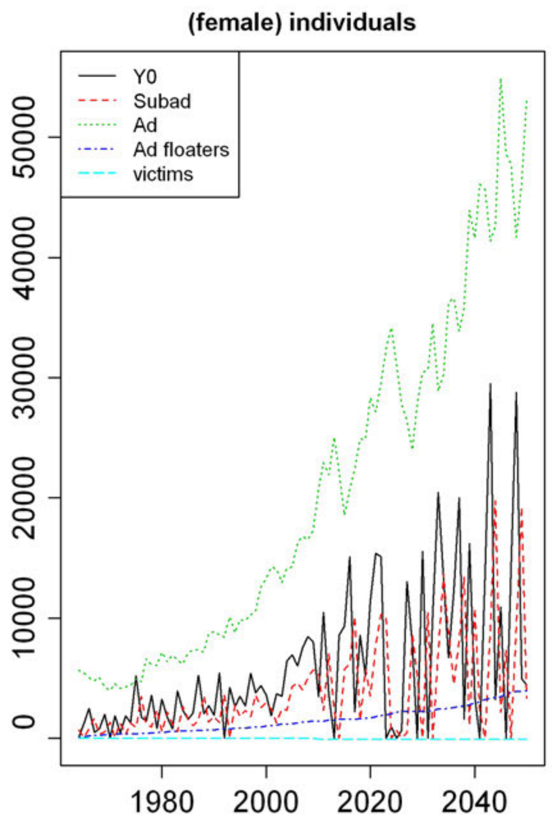
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.405 F0sd : 0.25
B1: 2
Floaters % >=2010: 9

K: 40000

Init population: 1000 1500 5000 0
victims : 0 0 60 0





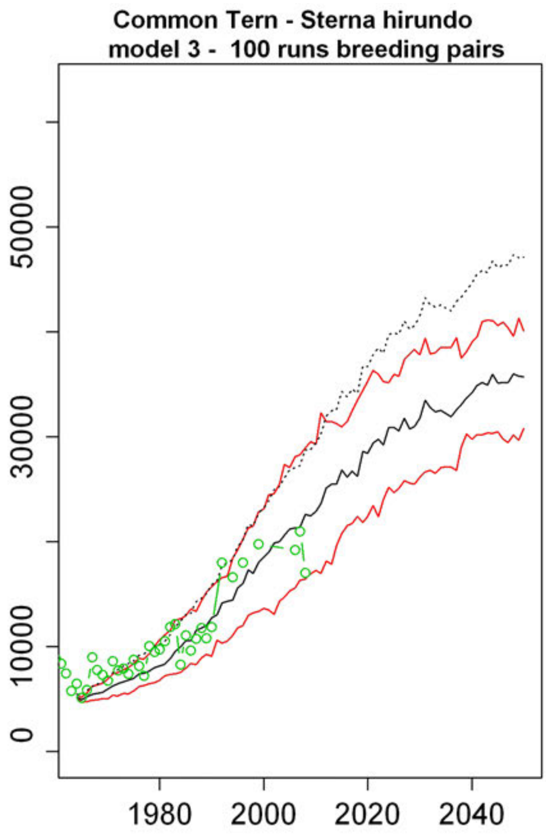


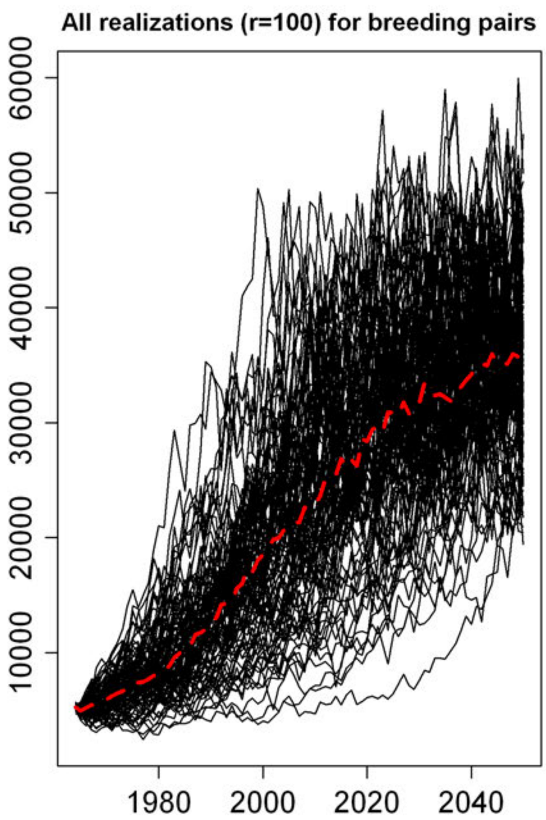
0 model – 30% floaters

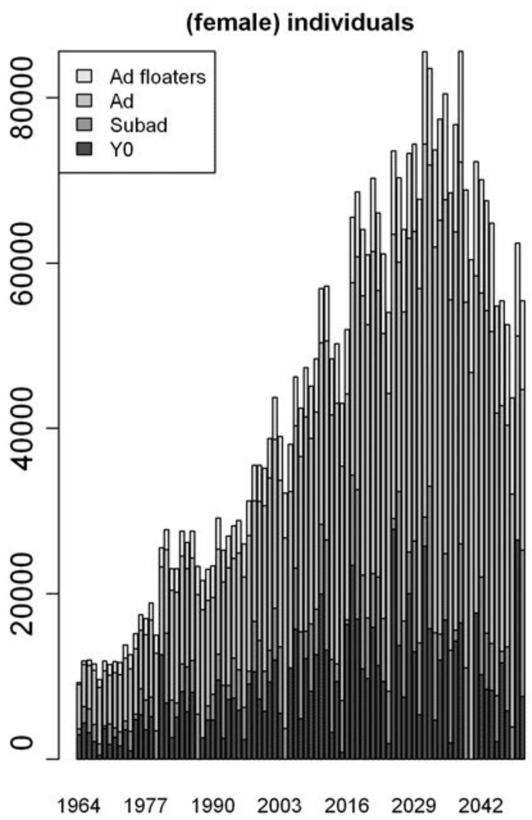
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.425 F0sd : 0.25
B1: 2
Floaters % >=2010: 30

K: 40000

Init population: 1000 1500 5000 0
victims : 0 0 0 0





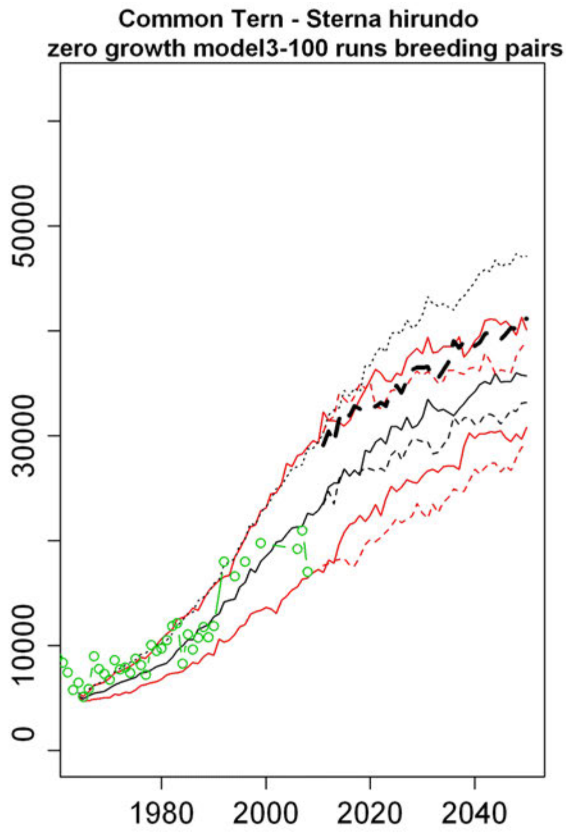


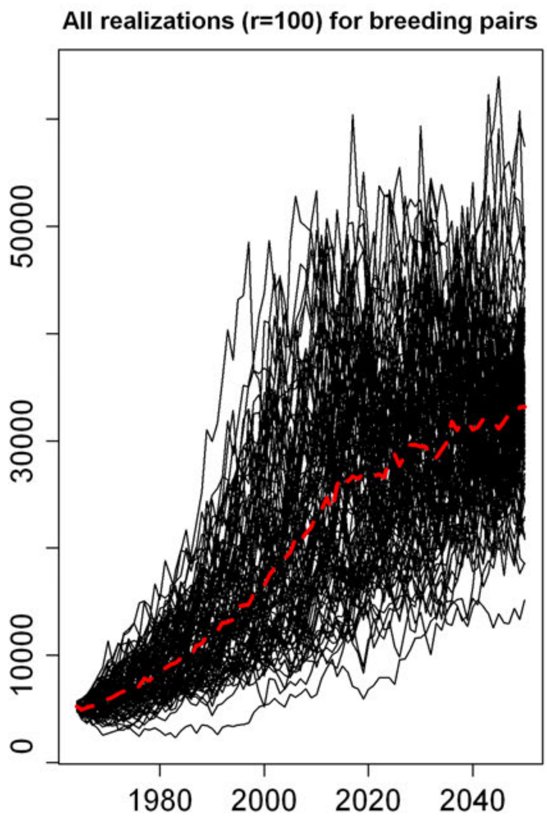
zero growth – 30% floaters

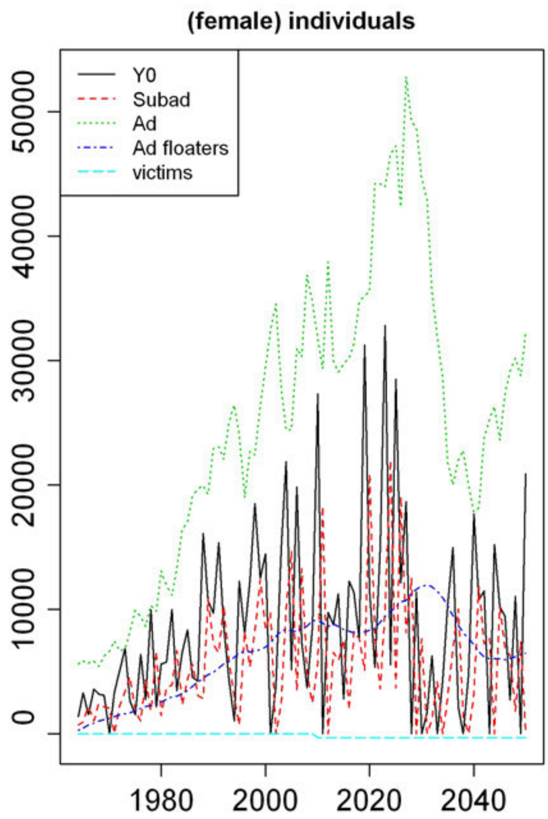
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.425 F0sd : 0.25
B1: 2
Floaters % >=2010: 24

K: 40000

Init population: 1000 1500 5000 0
victims : 0 0 325 0





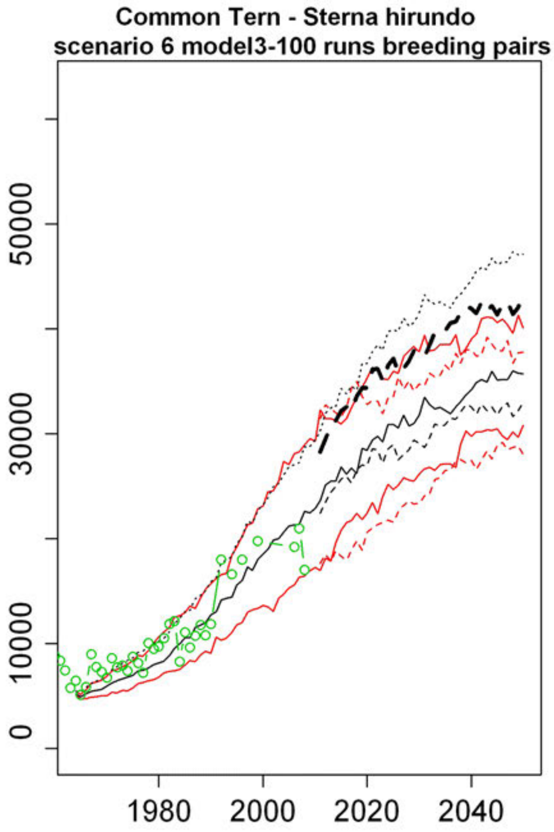


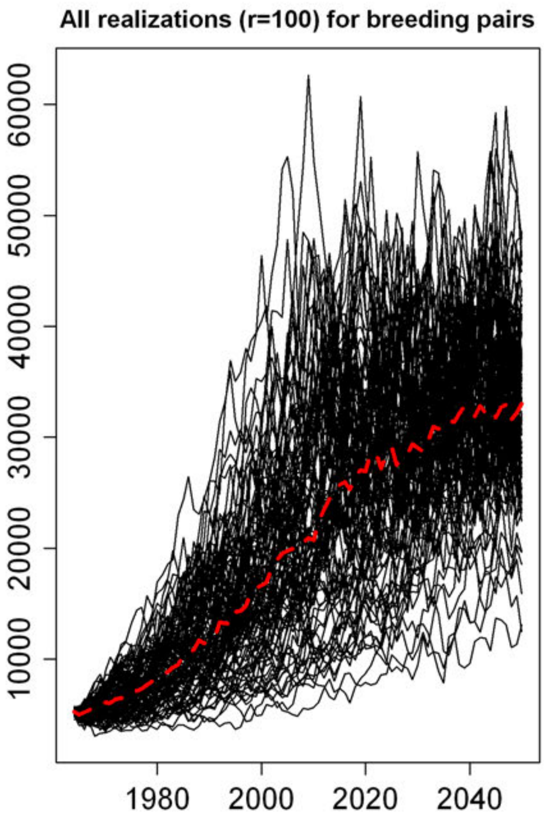
Scenario collisions Band-model – 30% floaters

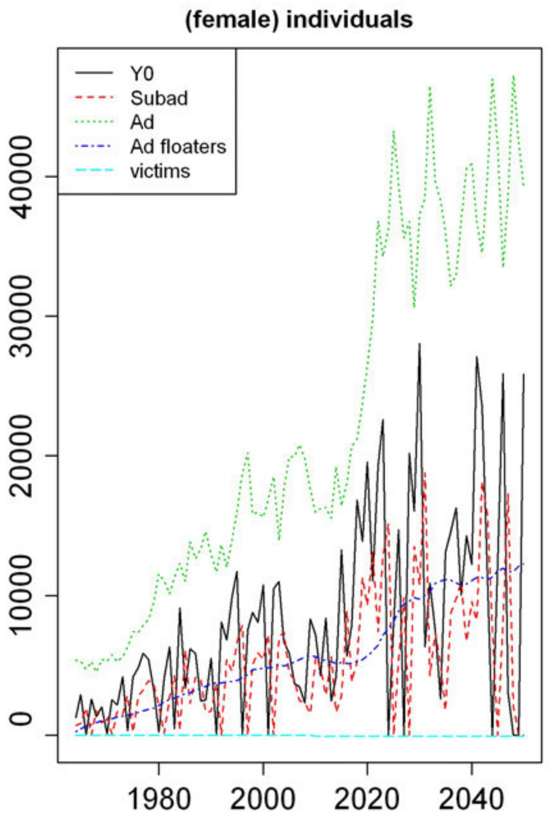
S0 : 0.67 S0sd : 0.05
Sad: 0.898 Sadsd: 0.05
F0 : 0.425 F0sd : 0.25
B1: 2
Floaters % >=2010: 29

K: 40000

Init population: 1000 1500 5000 0
victims   : 0 0 60 0







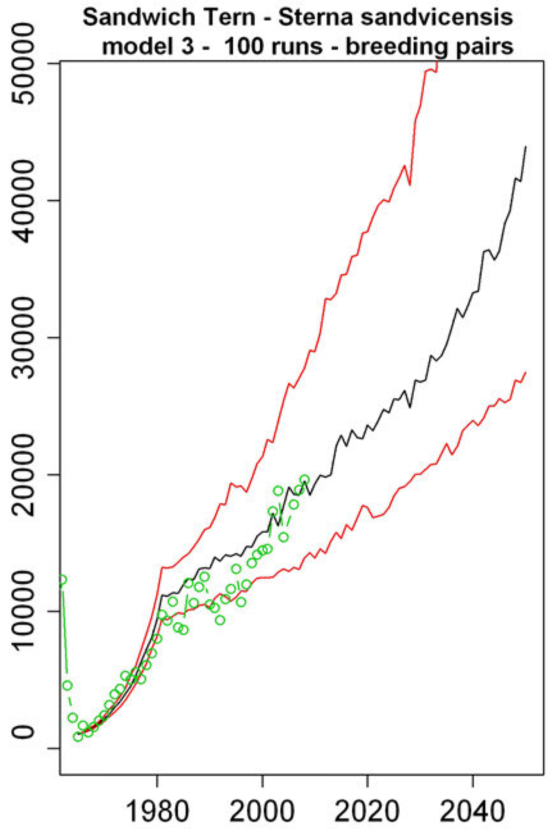
Sandwich tern – the Netherlands

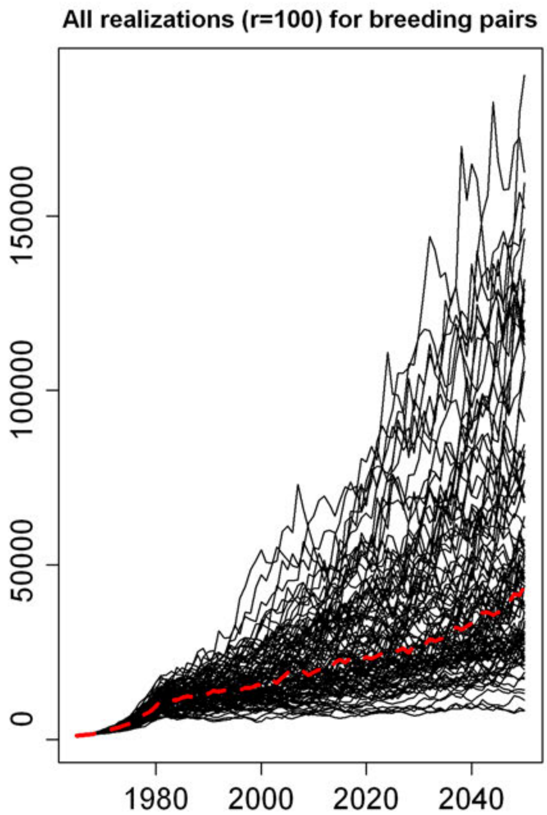
0-model 0% floaters

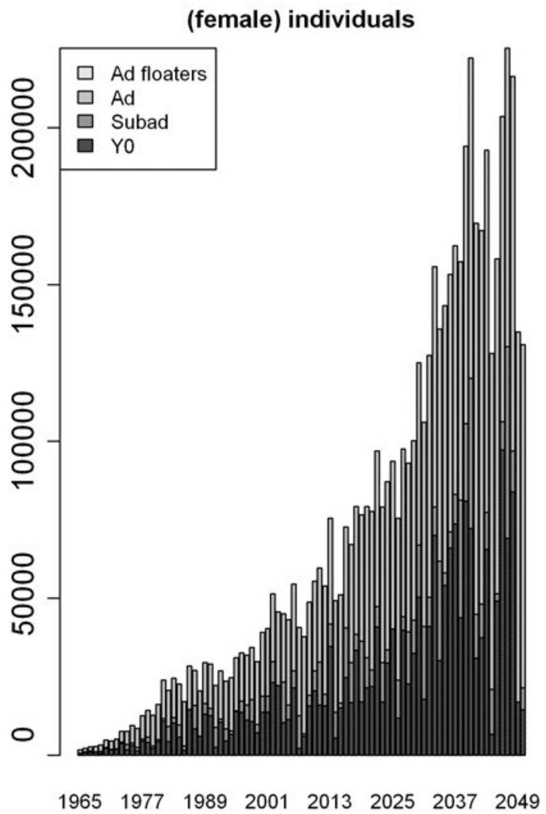
S0 : 0.358 S0sd : 0.219
S1_2: 0.741 S1_2sd: 0.206
Sad: 0.898 Sadsd: 0.029
F0 : 0.35 F0sd : 0.25
Imm < 1982: 0.18
Imm => 1982: 0.02
B1: 3
Floaters % >=2010: 0

K: 40000

Init population: 200 200 865 0
victims : 0 0 0 0





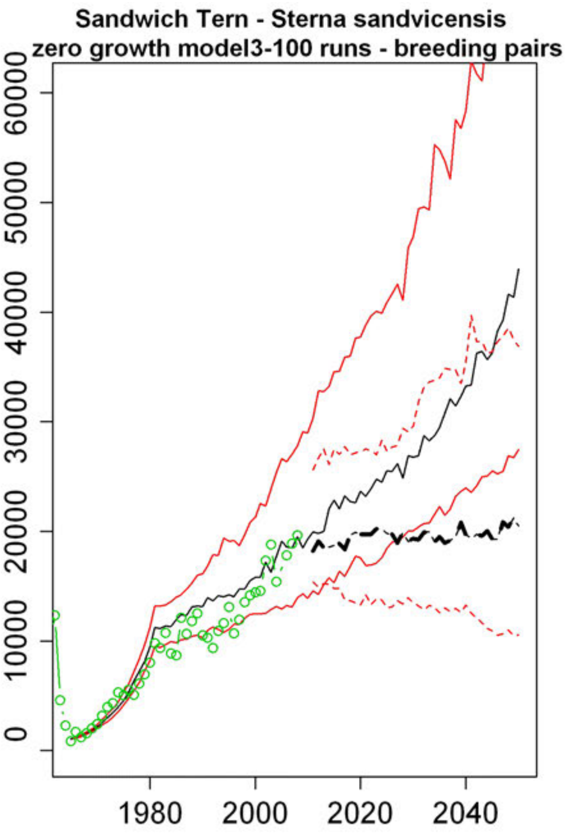


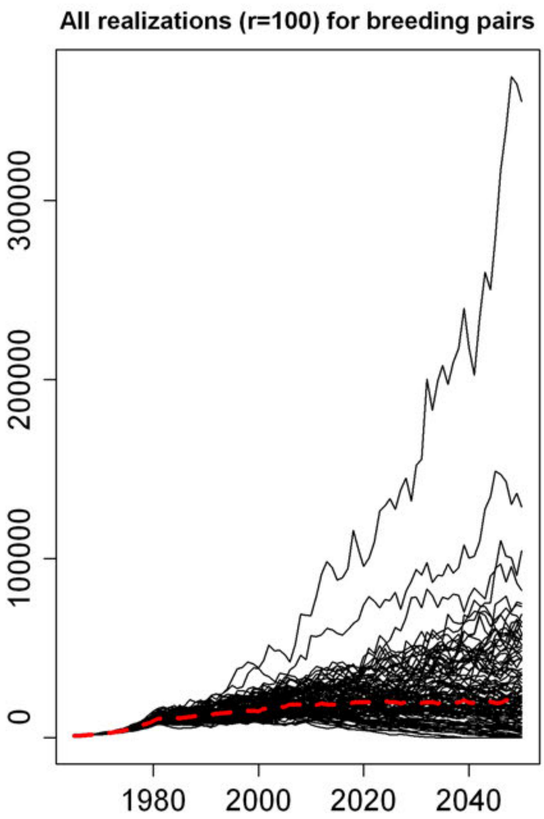
Zero growth - 0% floaters

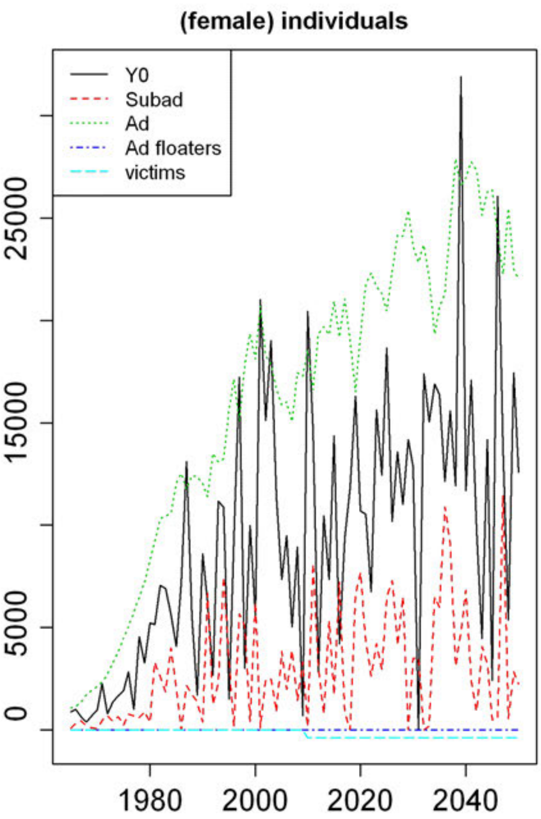
S0 : 0.358 S0sd : 0.219
S1_2: 0.741 S1_2sd: 0.206
Sad: 0.898 Sadsd: 0.029
F0 : 0.35 F0sd : 0.25
Imm < 1982: 0.18
Imm => 1982: 0.02
B1: 3
Floaters % >=2010: 0

K: 40000

Init population: 200 200 865 0
victims : 0 0 375 0





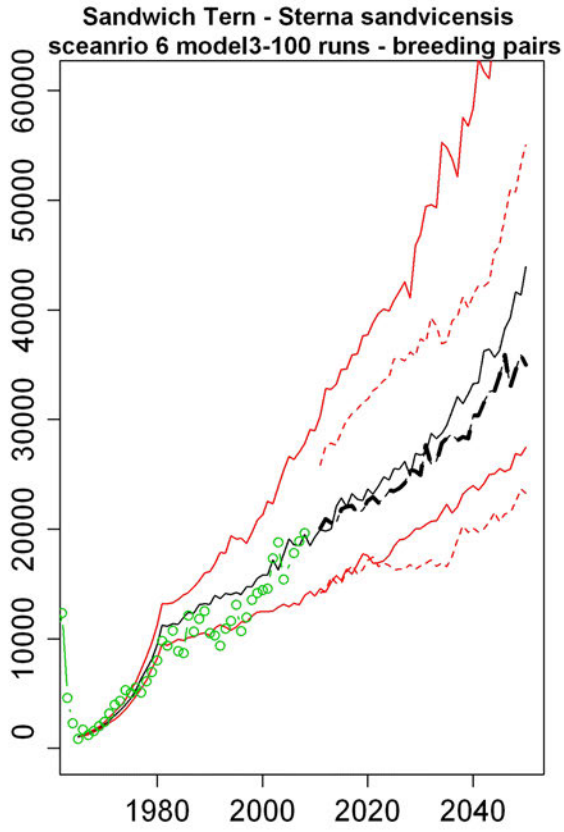


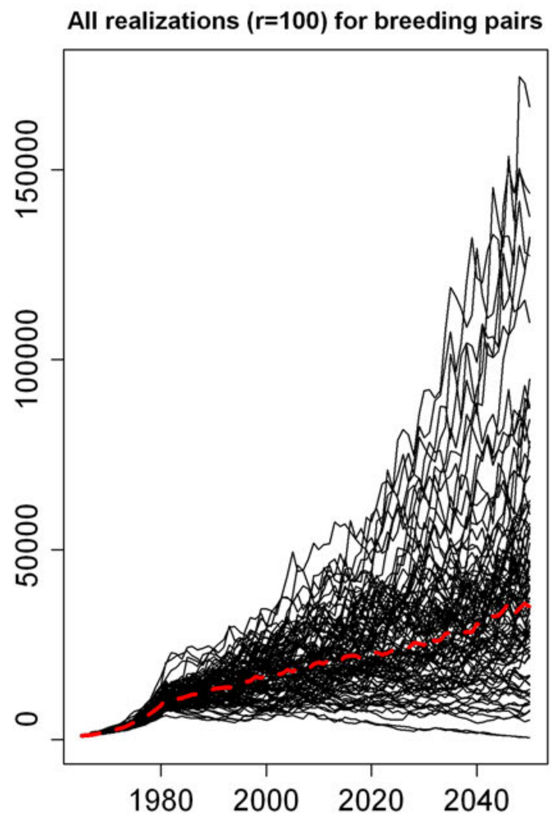
scenario collisions Band-model - 0% floaters

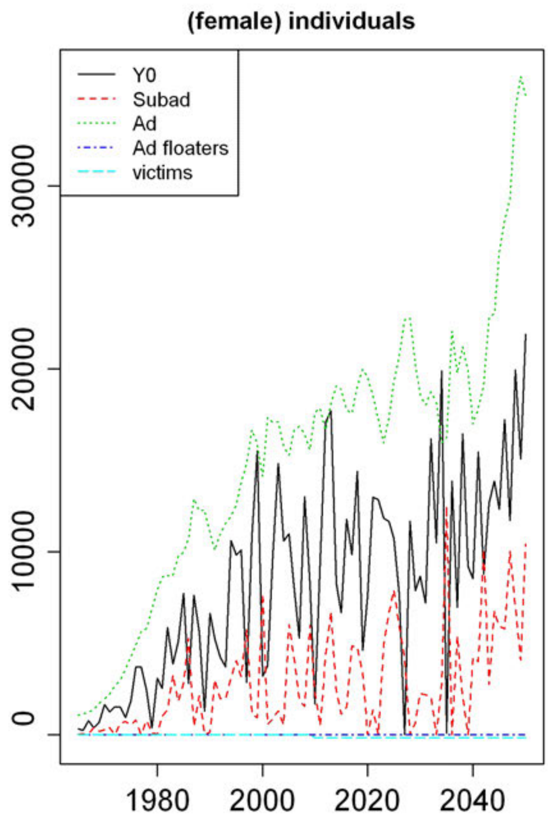
S0 : 0.358 S0sd : 0.219
S1_2: 0.741 S1_2sd: 0.206
Sad: 0.898 Sadsd: 0.029
F0 : 0.35 F0sd : 0.25
Imm < 1982: 0.18
Imm => 1982: 0.02
B1: 3
Floaters % >=2010: 0

K: 40215

Init population: 200 200 865 0
victims : 0 0 155 0





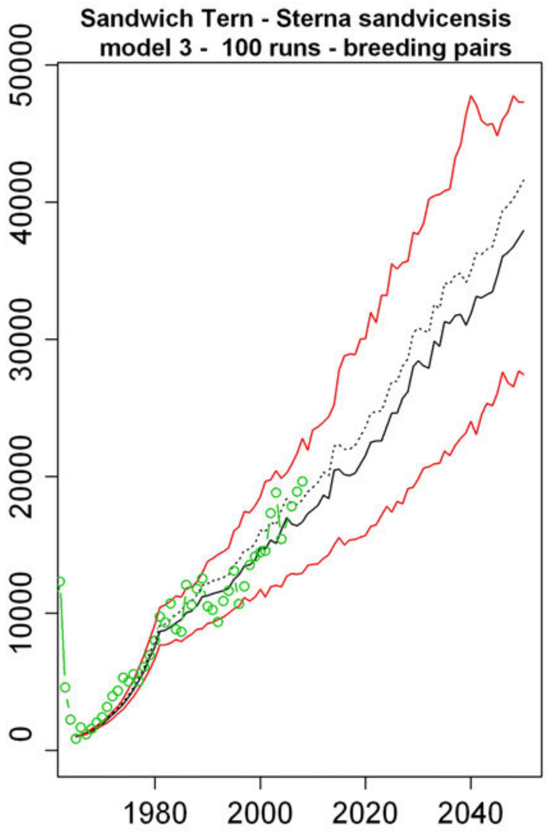


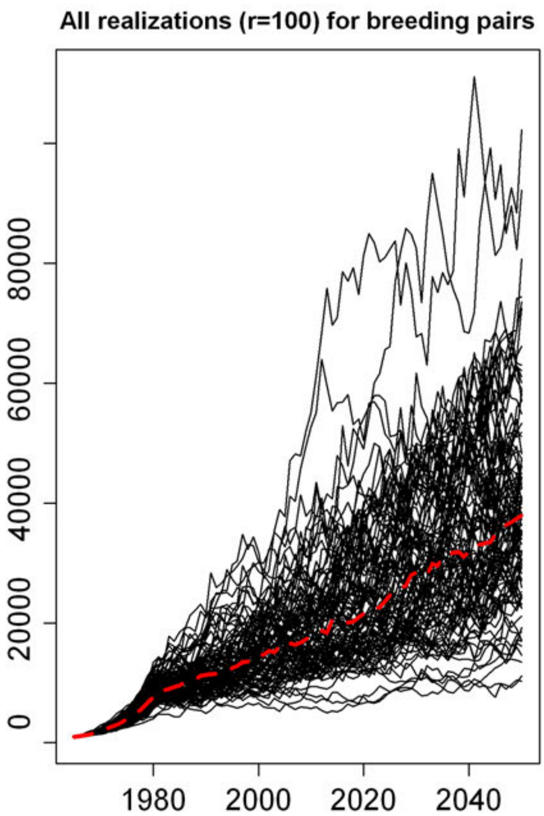
0 model – 10% floaters

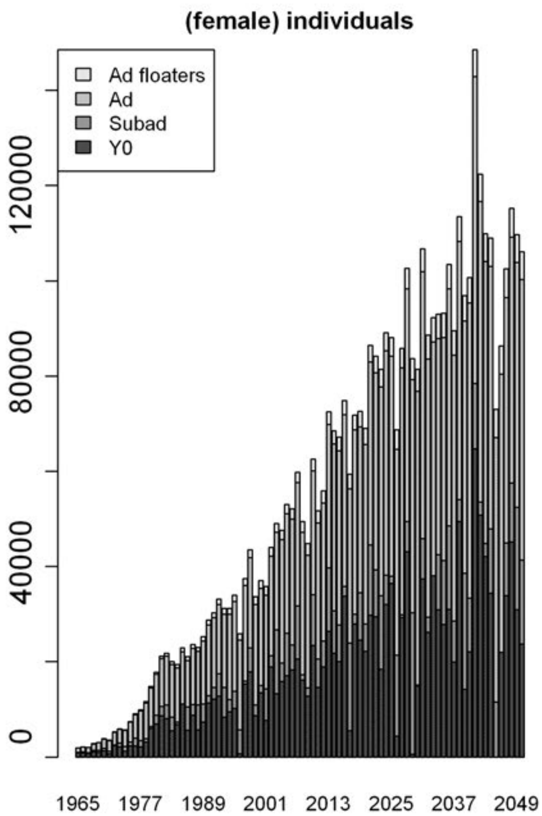
S0 : 0.358 S0sd : 0.219
S1_2: 0.741 S1_2sd: 0.206
Sad: 0.898 Sadsd: 0.029
F0 : 0.75 F0sd : 0.25
Imm < 1982: 0.16
Imm => 1982: 0.02
B1: 3
Floaters % >=2010: 9

K: 40000

Init population: 200 200 865 0
victims : 0 0 0 0





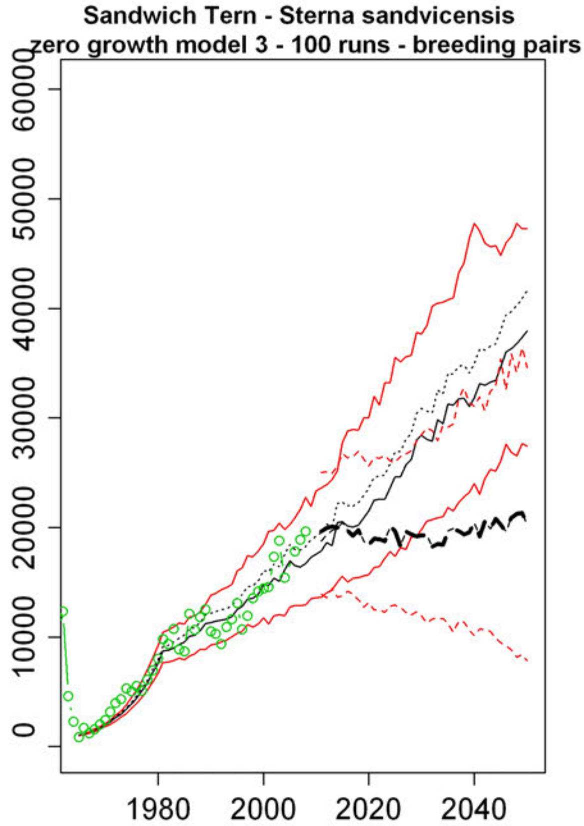


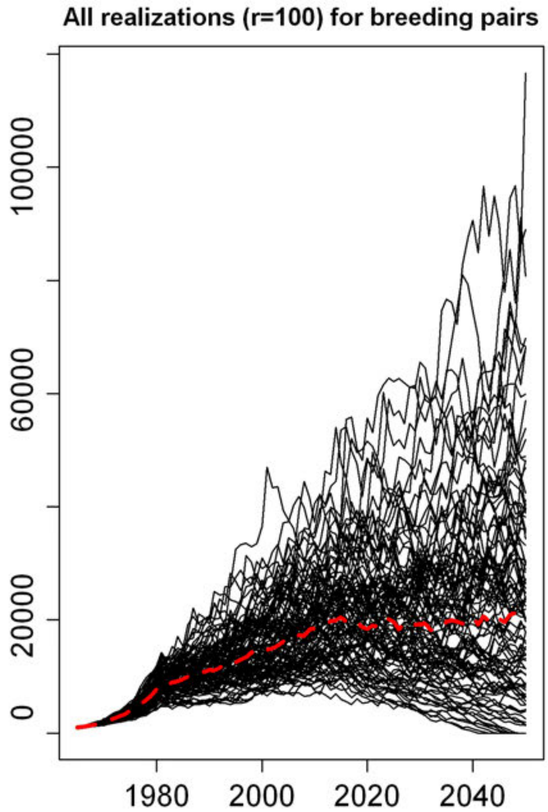
Zero growth – 10% floaters

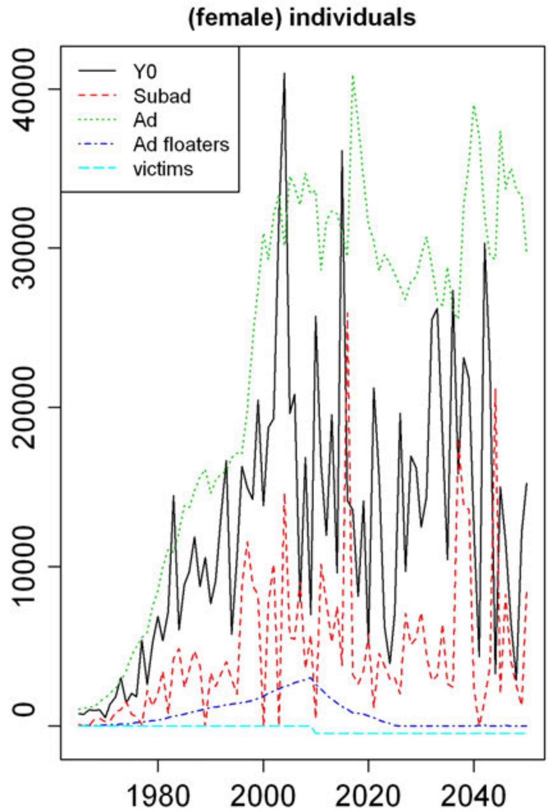
S0 : 0.358 S0sd : 0.219
S1_2: 0.741 S1_2sd: 0.206
Sad: 0.898 Sadsd: 0.029
F0 : 0.75 F0sd : 0.25
Imm < 1982: 0.16
Imm => 1982: 0.02
B1: 3
Floaters % >=2010: 0

K: 40000

Init population: 200 200 865 0
victims : 0 0 450 0





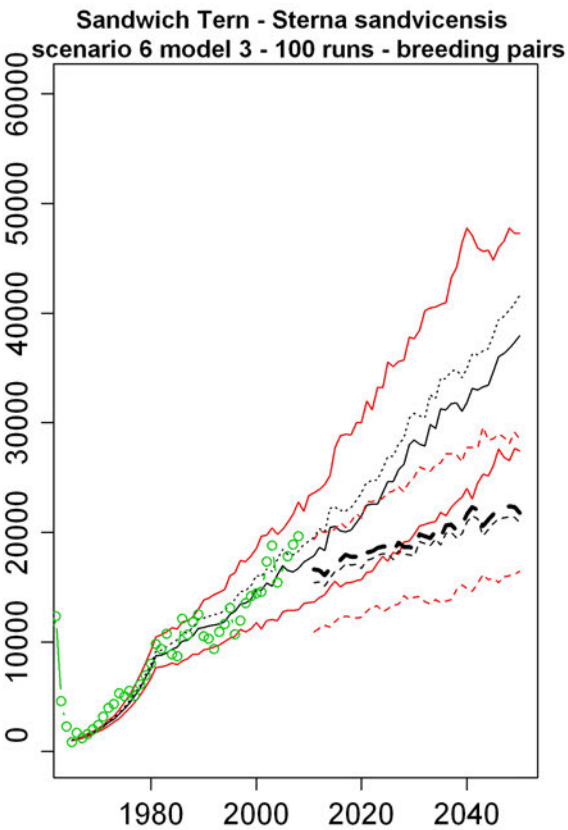


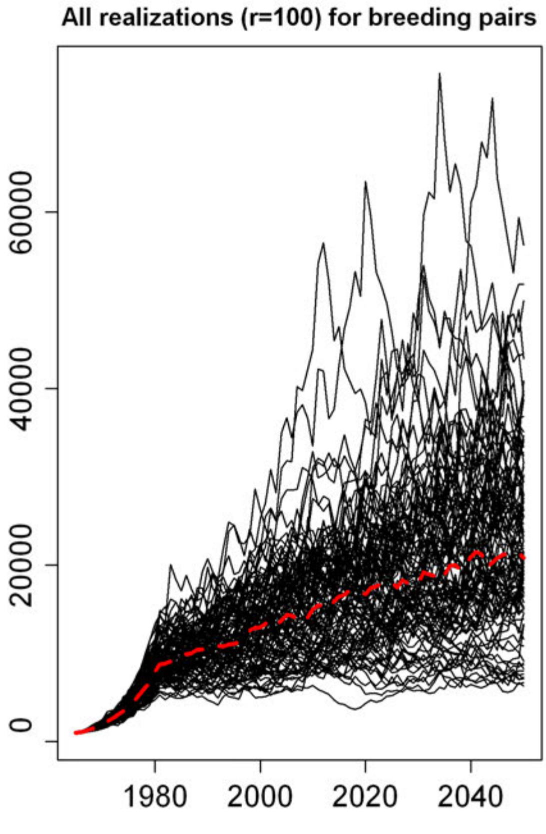
Scenario collisions Band-model – 10% floaters

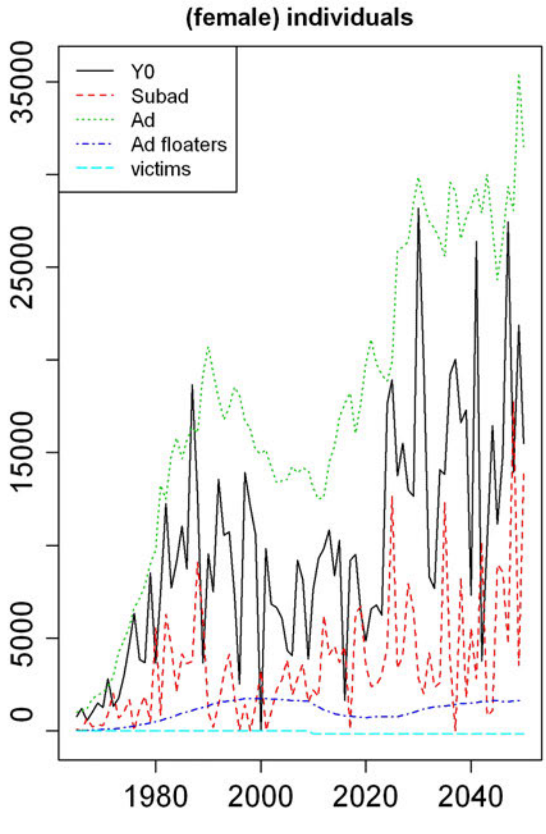
S0 : 0.358 S0sd : 0.219
S1_2: 0.741 S1_2sd: 0.206
Sad: 0.898 Sadsd: 0.029
F0 : 0.75 F0sd : 0.276
Imm < 1982: 0.16
Imm => 1982: 0.02
B1: 3
Floaters % >=2010: 4

K: 19635

Init population: 200 200 865 0
victims : 0 0 155 0







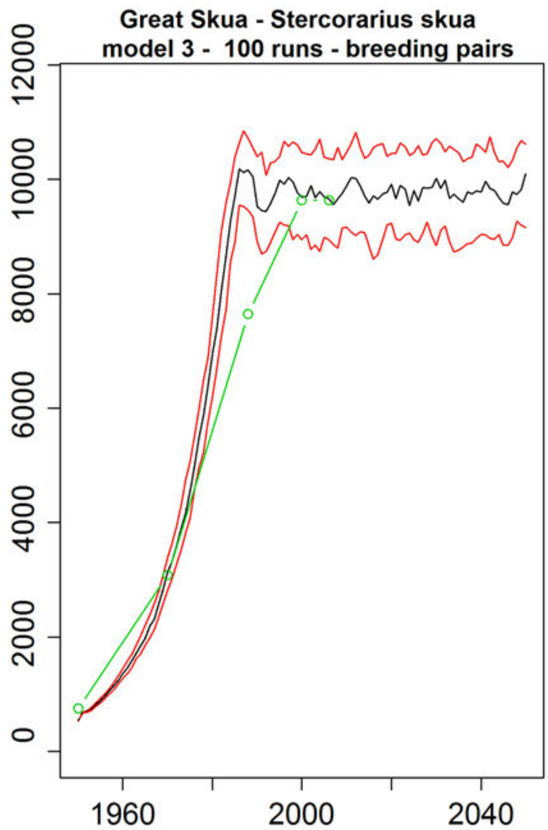
Great skua - Scotland

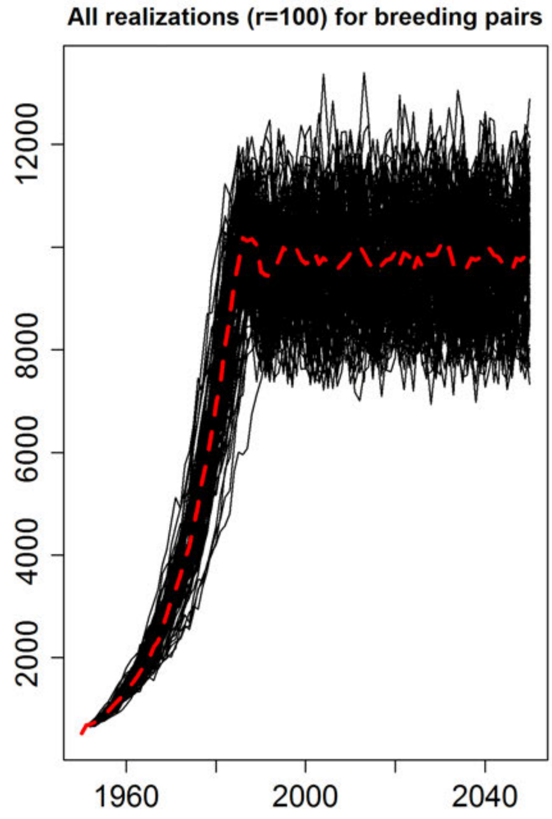
0 model – 0% floaters

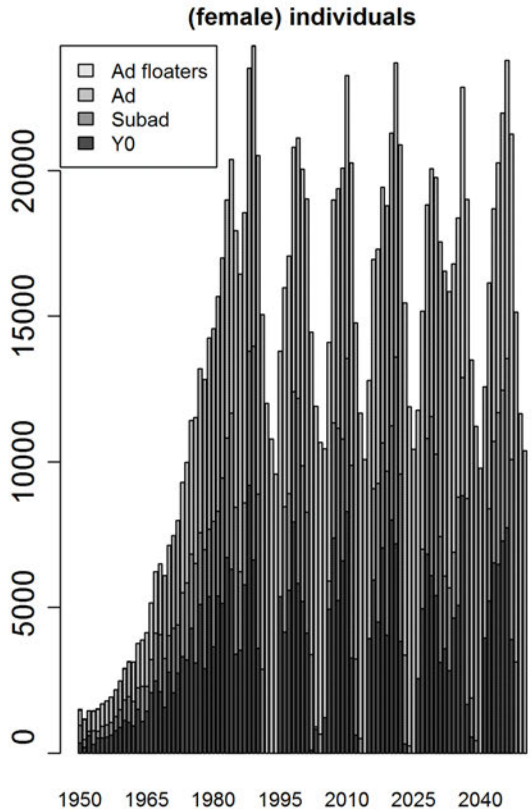
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.8 F0sd : 0.185
B1: 7.5
Floaters % >=2010: 0

K: 9635

Init population: 750 500 400 20





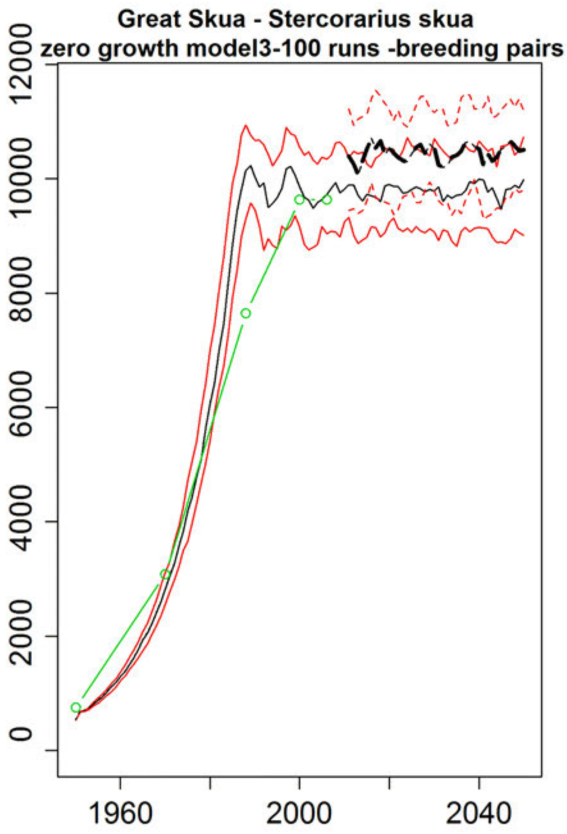


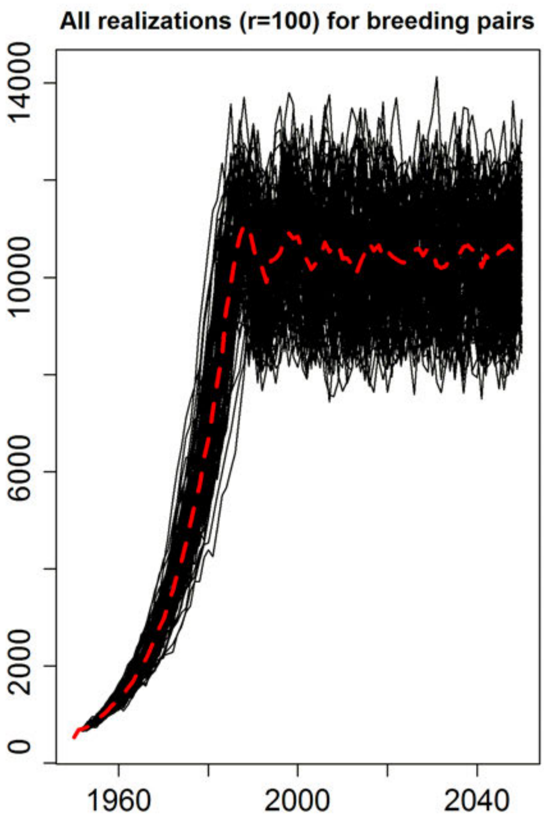
zero growth - 0% floaters

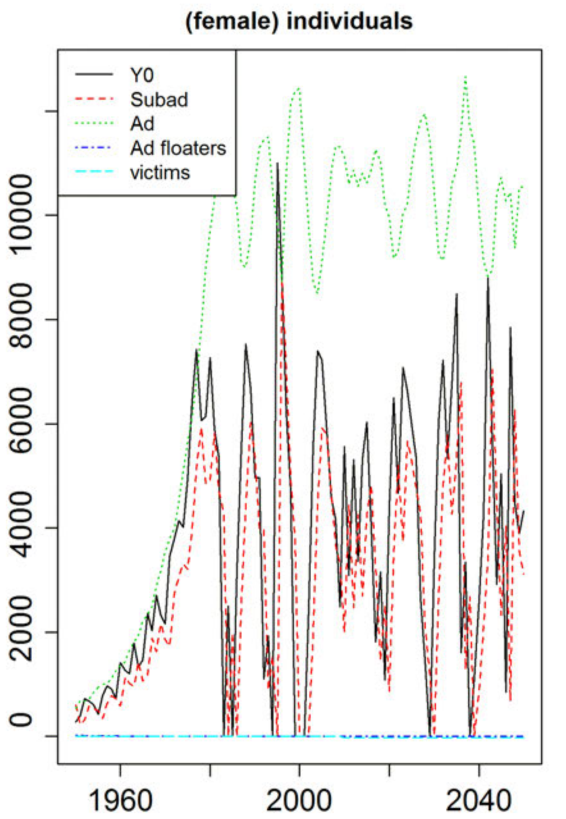
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.8 F0sd : 0.185
B1: 7.5
Floaters % >=2010: 0

K: 9635

Init population: 750 500 400 20
victims : 0 0 25 0





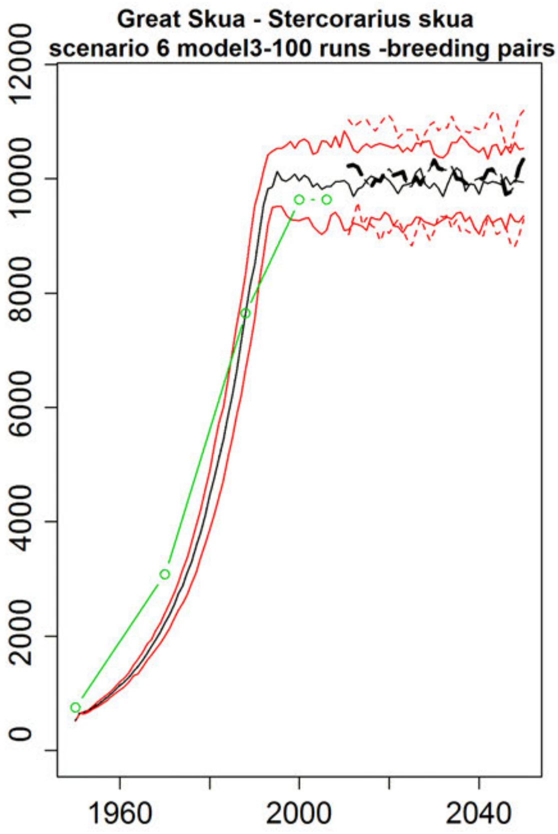


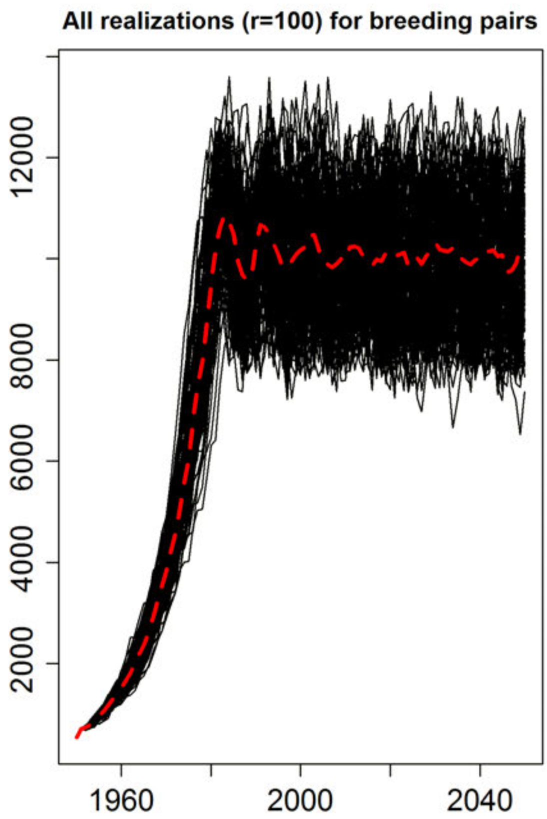
scenario collisions Band-model – 0% floaters

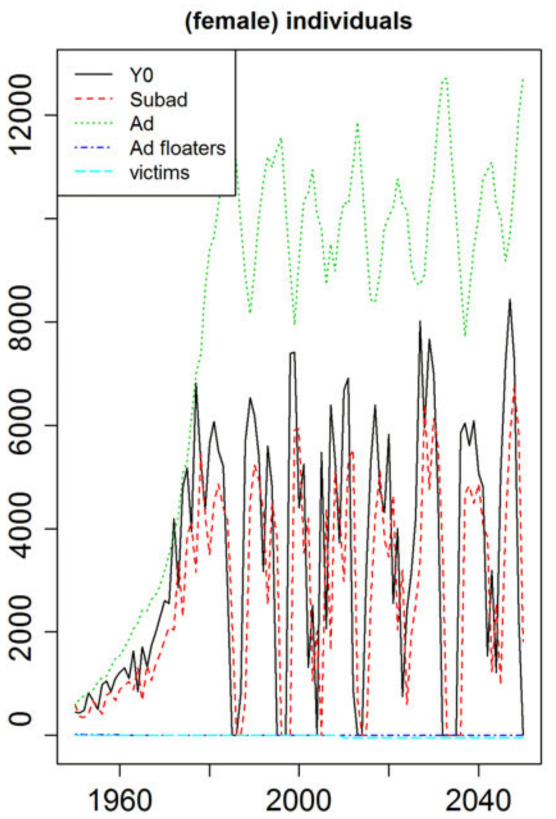
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.8 F0sd : 0.185
B1: 7
Floaters % >=2010: 0

K: 9635

Init population: 750 500 400 20
victims : 0 0 40 0





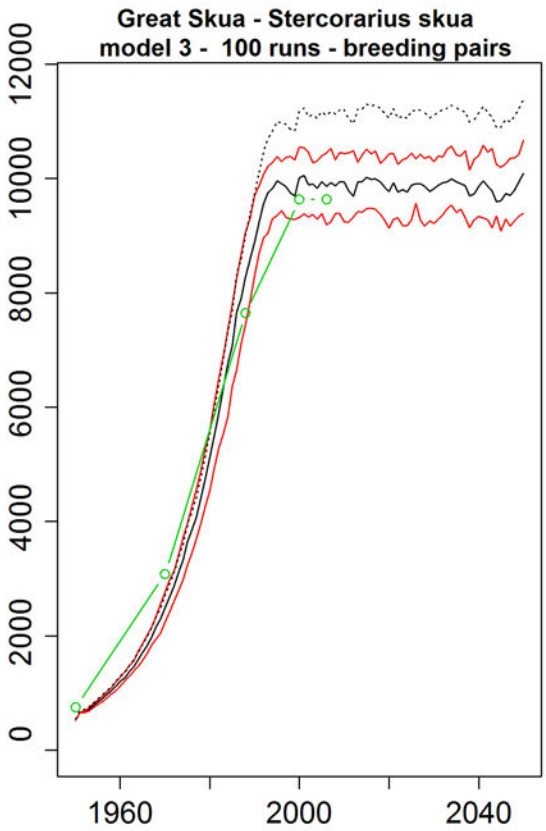


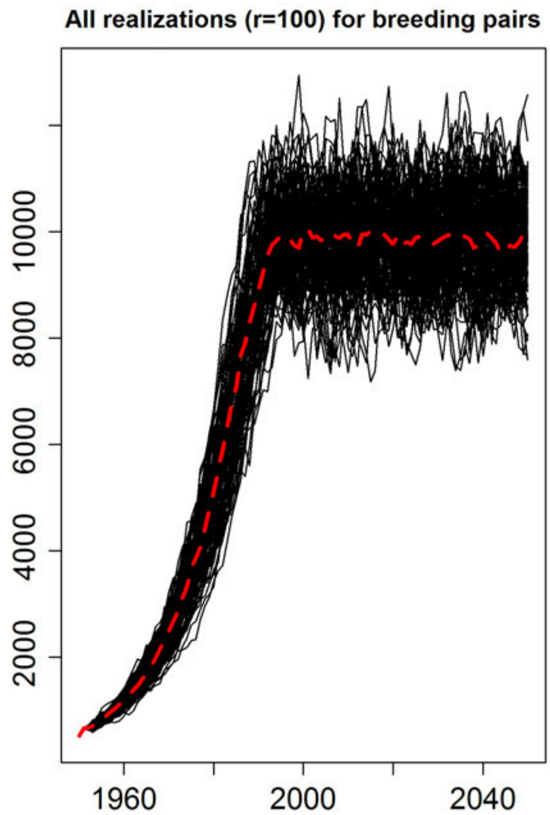
0 model – 10% floaters

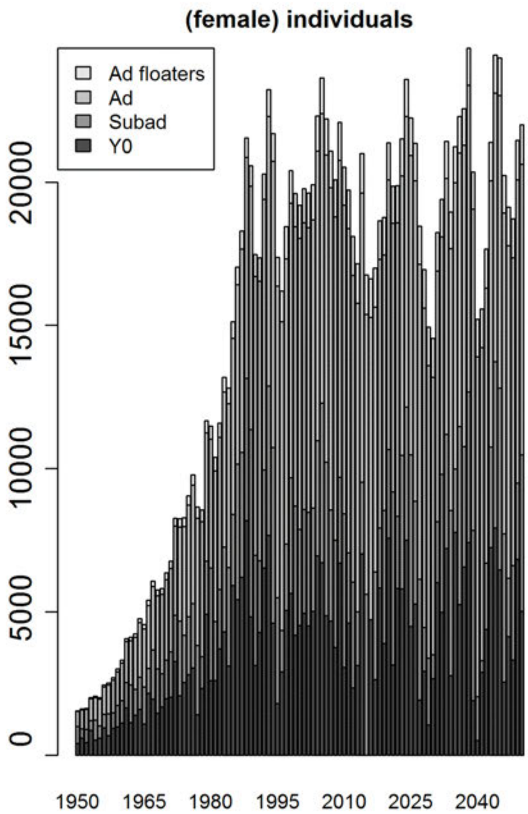
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.805 F0sd : 0.185
B1: 7.5
Floaters % >=2010: 10

K: 9635

Init population: 750 500 400 20





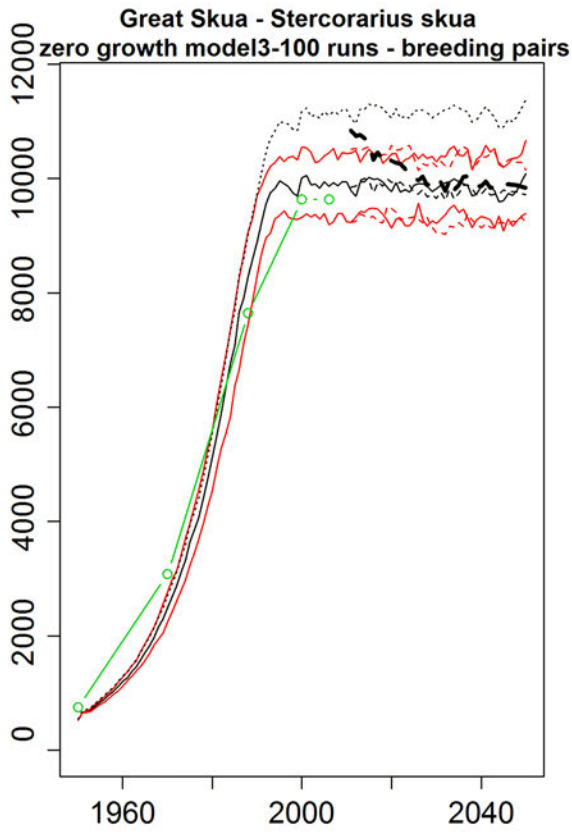


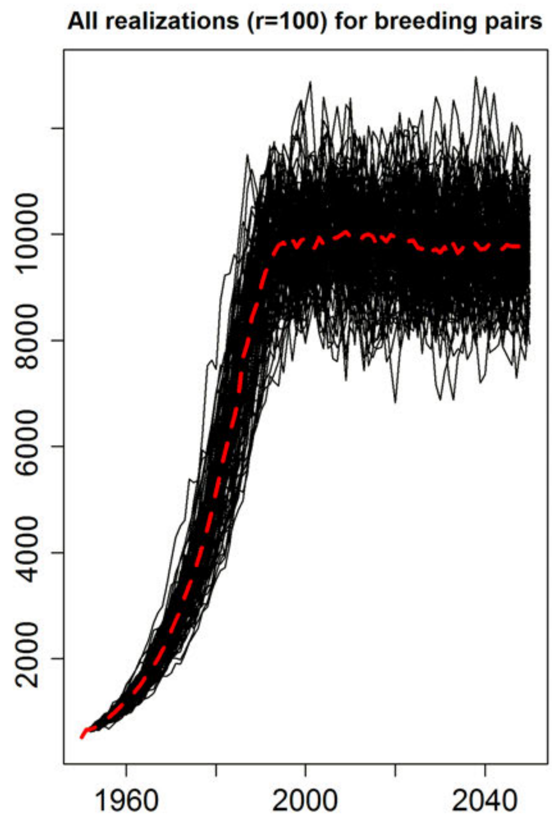
zero growth – 10% floaters

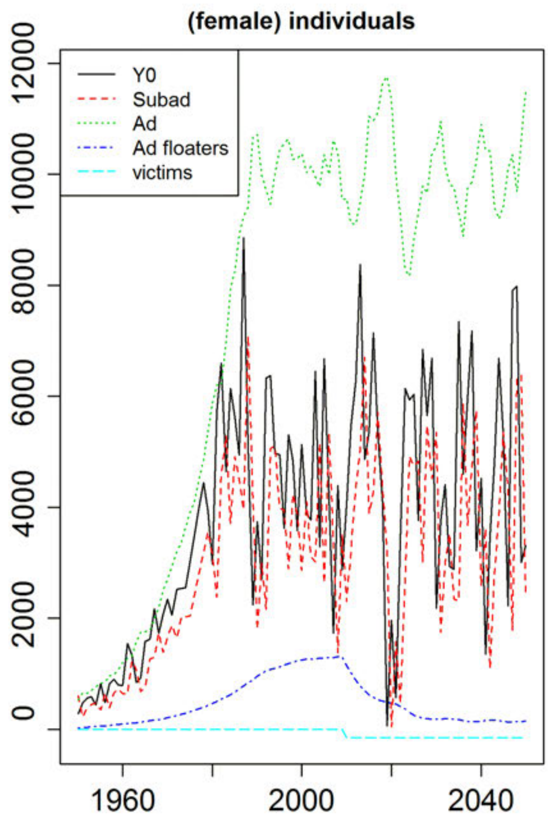
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.805 F0sd : 0.185
B1: 7.5
Floaters % >=2010: 3

K: 9635

Init population: 750 500 400 20
victims : 0 0 150 0





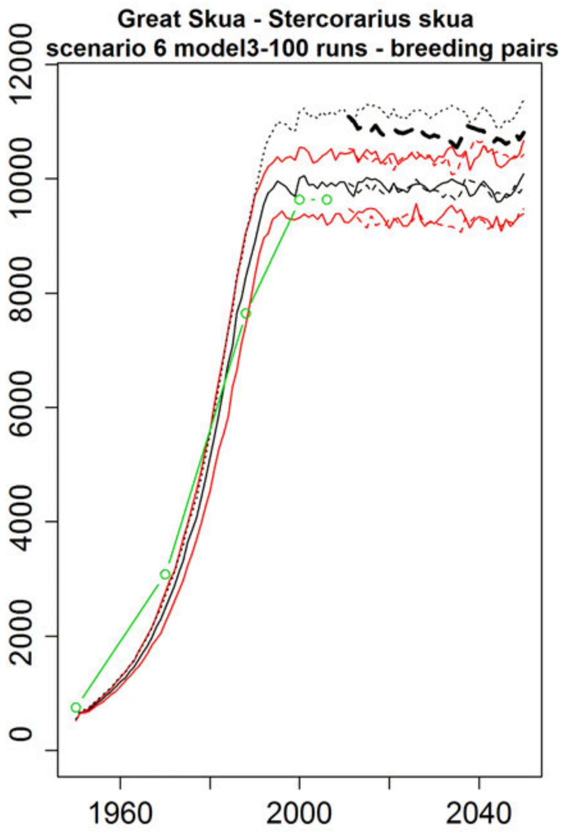


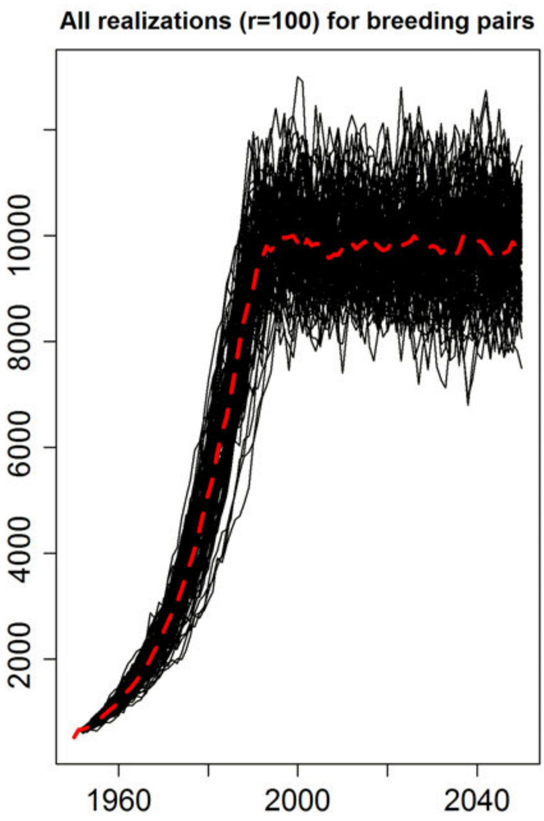
scenario collisions Band-model – 10% floaters

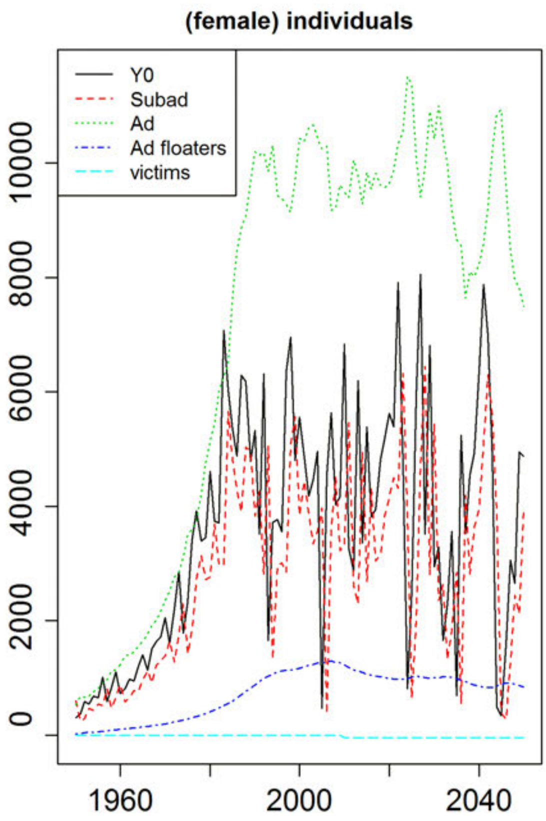
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.805 F0sd : 0.185
B1: 7.5
Floaters % >=2010: 10

K: 9635

Init population: 750 500 400 20
victims   : 0 0 40 0





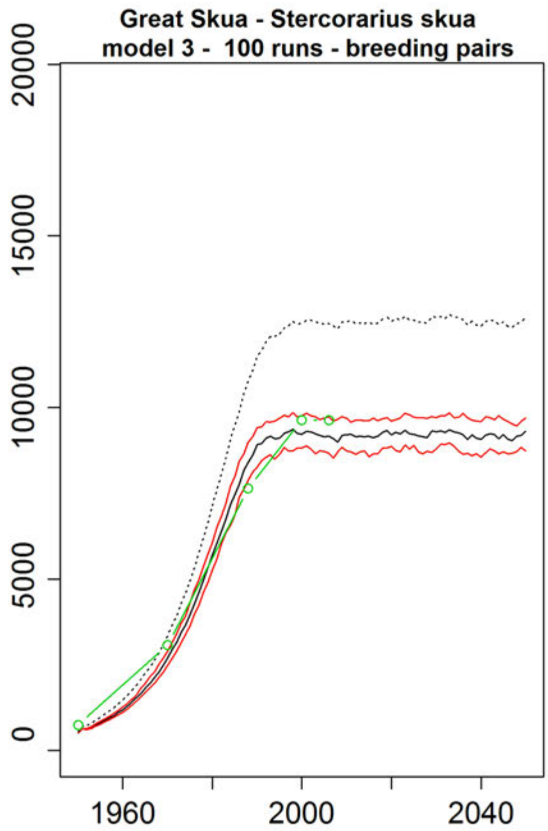


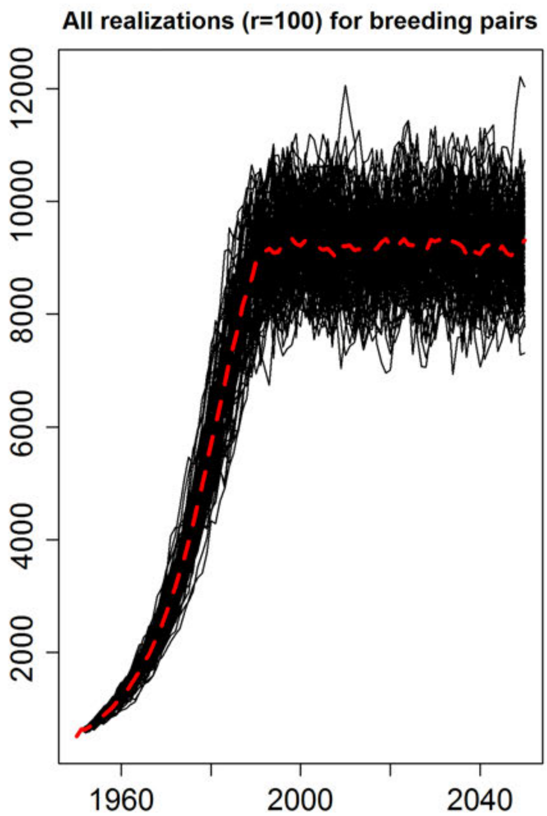
0 model – 30% floaters

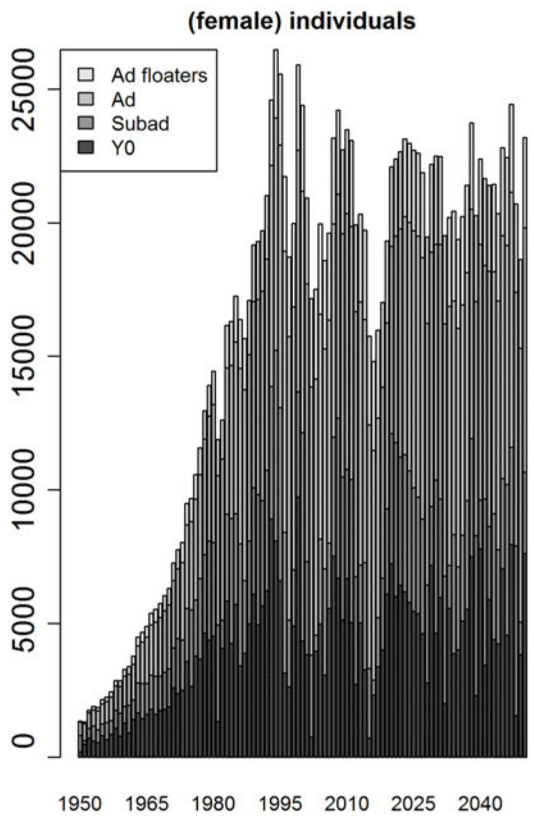
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.985 F0sd : 0.185
B1: 7.5
Floaters % >=2010: 33

K: 9635

Init population: 750 500 400 20





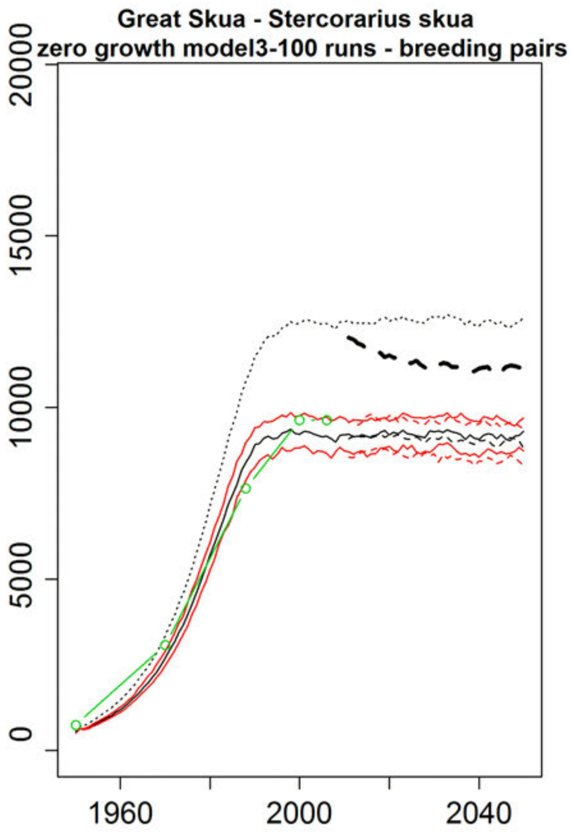


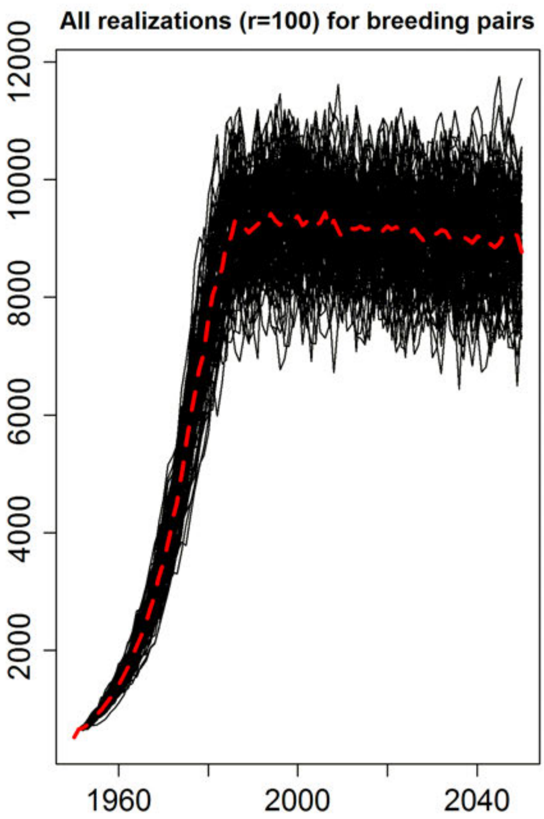
zero growth – 30% floaters

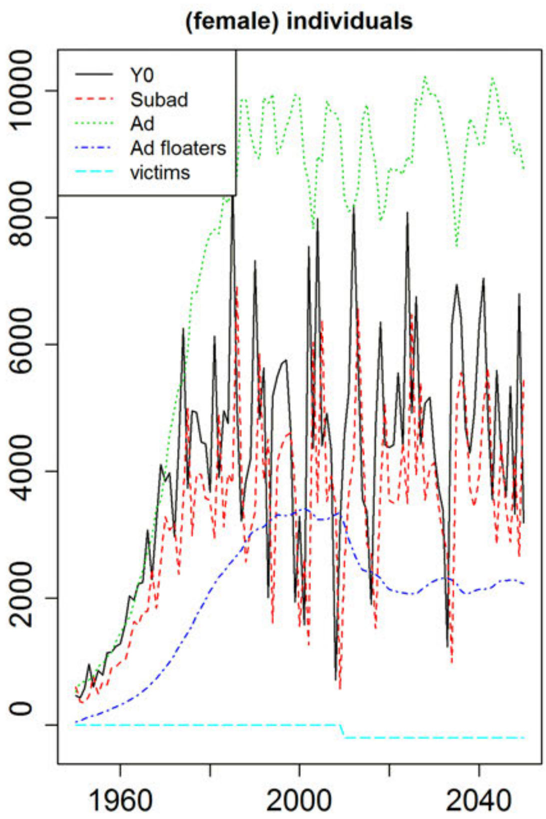
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.985 F0sd : 0.185
B1: 7
Floaters % >=2010: 25

K: 9635

Init population: 750 500 400 20
victims : 0 0 200 0





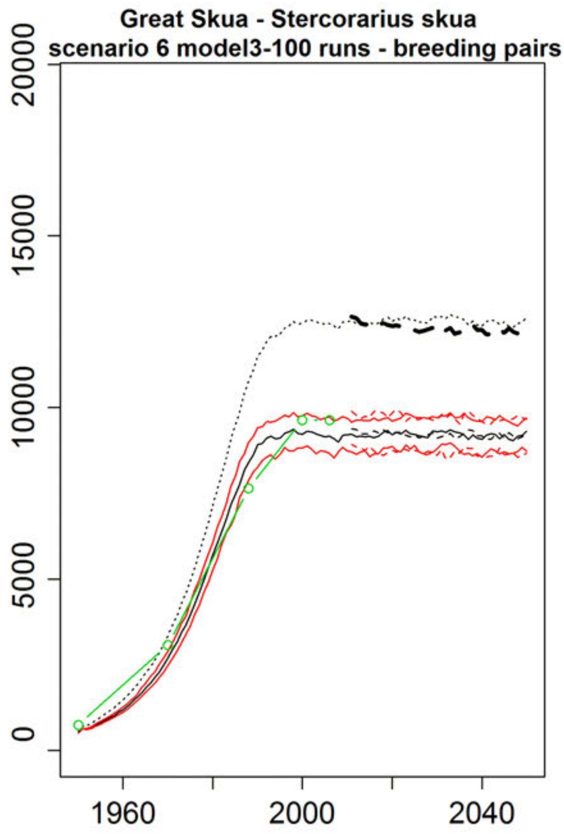


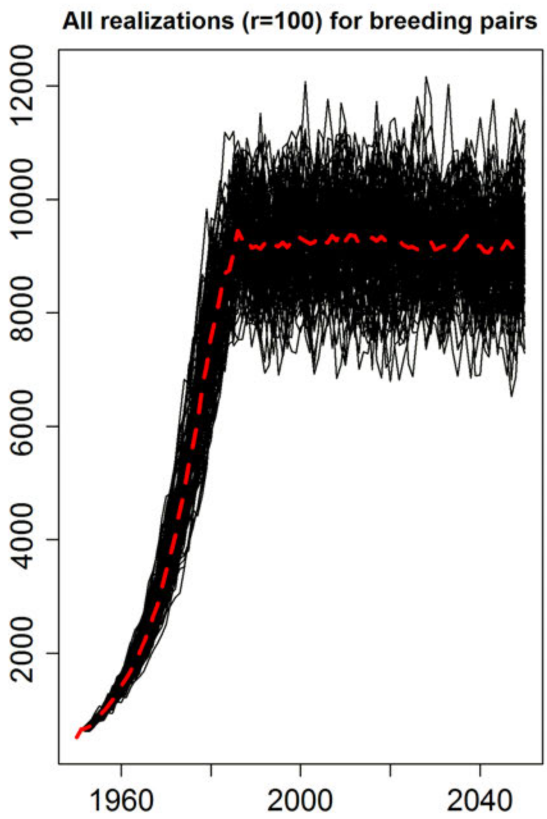
scenario collisions Band-model – 30% floaters

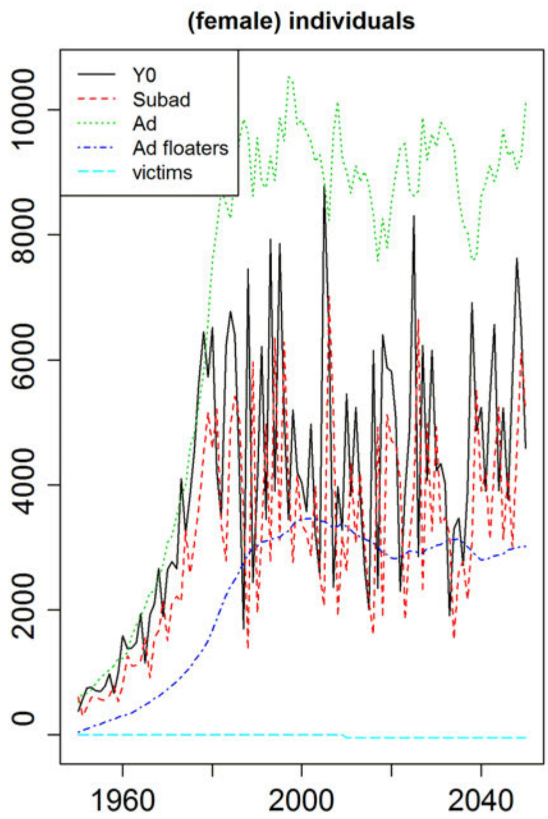
S0 : 0.8 S0sd : 0.006
Sad: 0.888 Sadsd: 0.006
F0 : 0.985 F0sd : 0.185
B1: 7
Floaters % >=2010: 30

K: 9635

Init population: 750 500 400 20
victims : 0 0 40 0





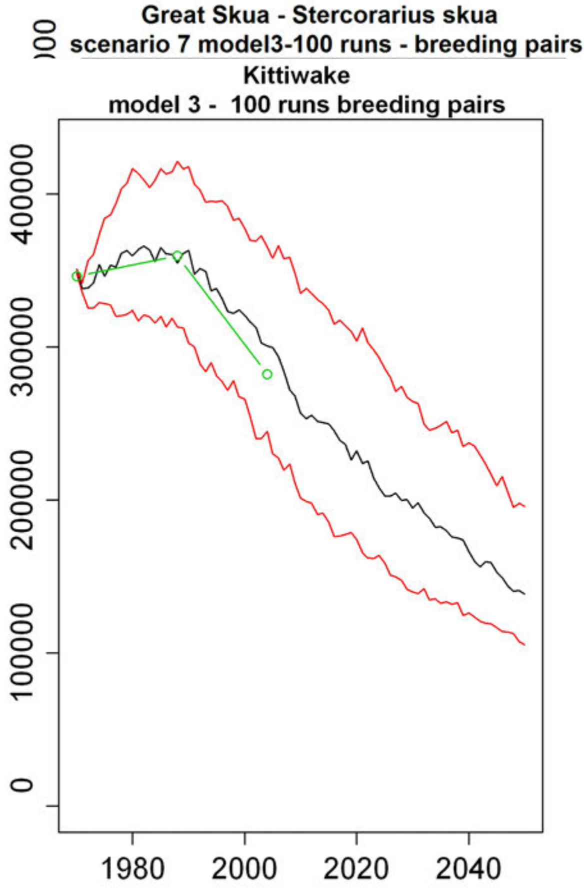


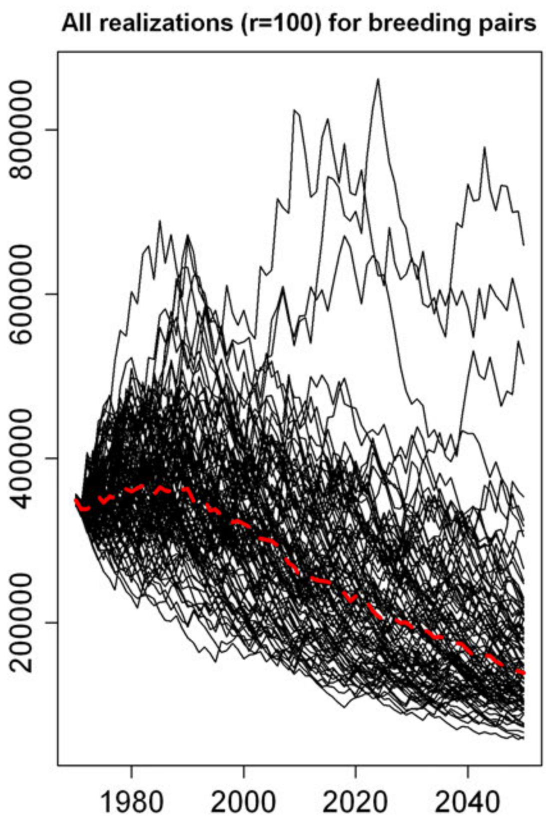
Kittiwake - Scotland

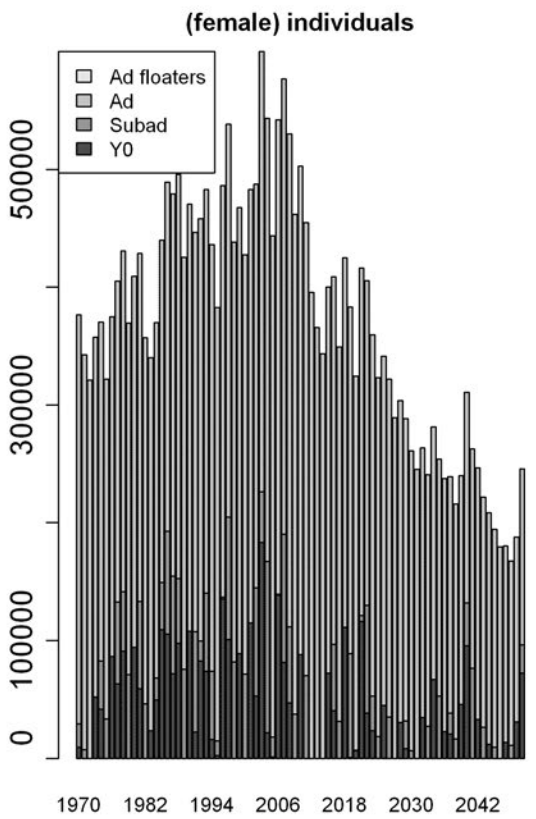
0 model

S0 : 0.79 S0sd : 0.01
Sad: 0.941 Sadsd: 0.01
F0 : 0.06 F0sd : 0.2
B1 : 4
Floaters % >=2010: 0

Init population: 25000 50000 345000 0





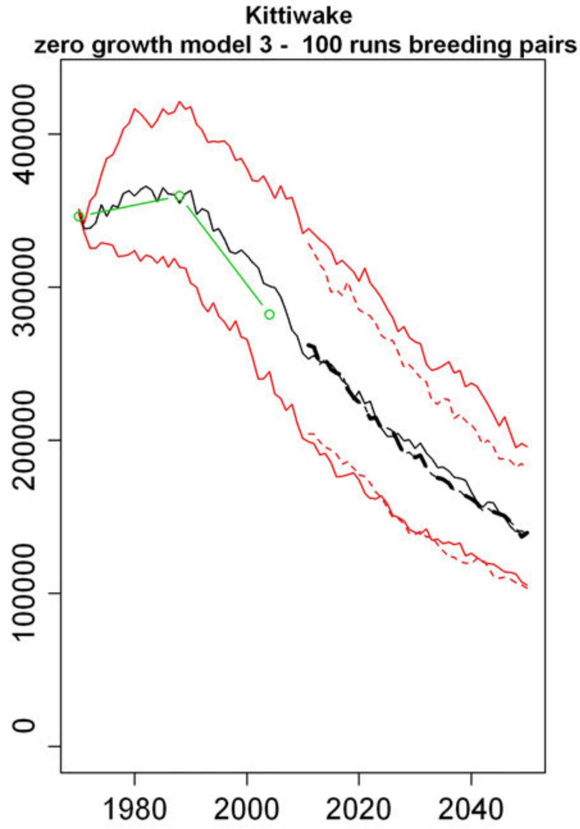


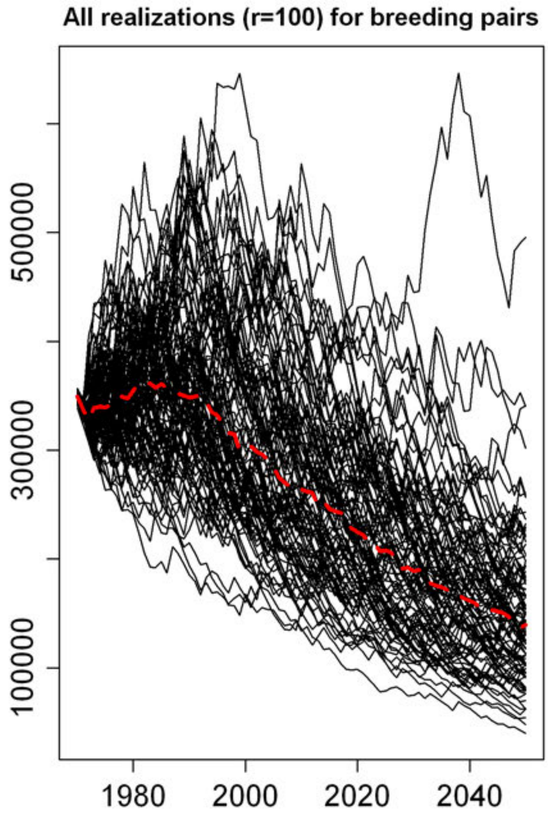
Zero growth model

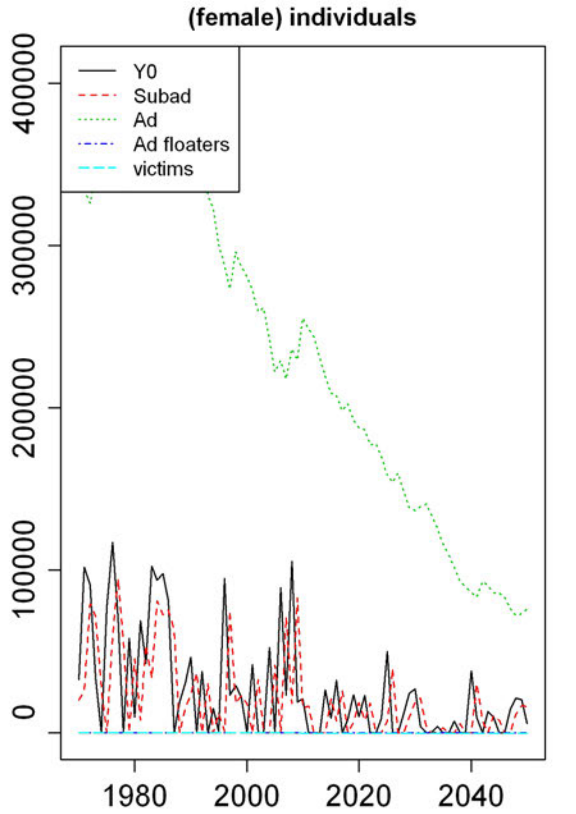
S0 : 0.79 S0sd : 0.01
Sad: 0.941 Sadsd: 0.01
F0 : 0.06 F0sd : 0.2
B1 : 4
Floaters % >=2010: 0

K: 359425

Victims : 0 0 750 0
Init population: 25000 50000 345000 0





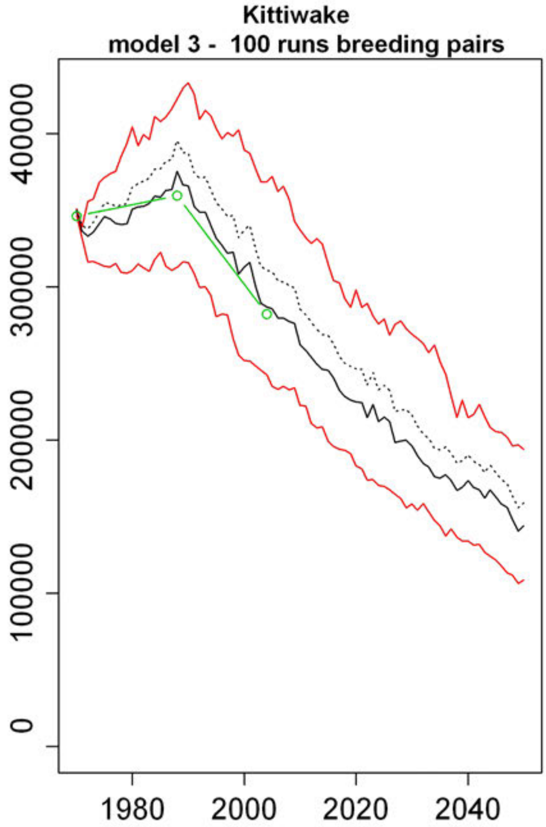


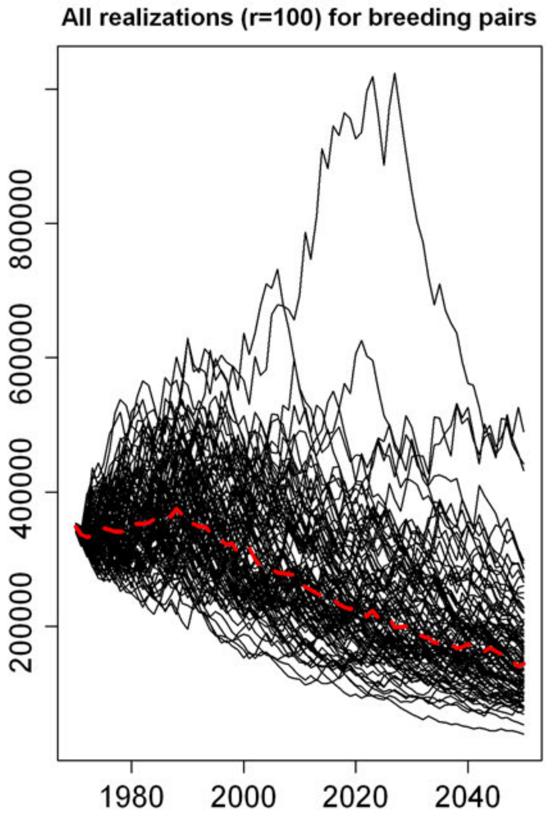
model : 10% floaters

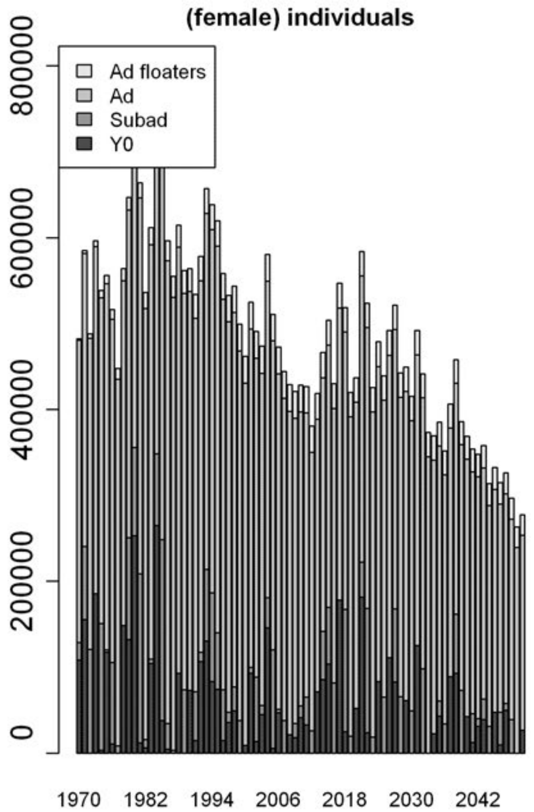
S0 : 0.79 S0sd : 0.01
Sad: 0.941 Sadsd: 0.01
F0 : 0.072 F0sd : 0.2
B1 : 4
Floaters % >=2010: 10

K: 359425

Init population: 25000 50000 345000 0





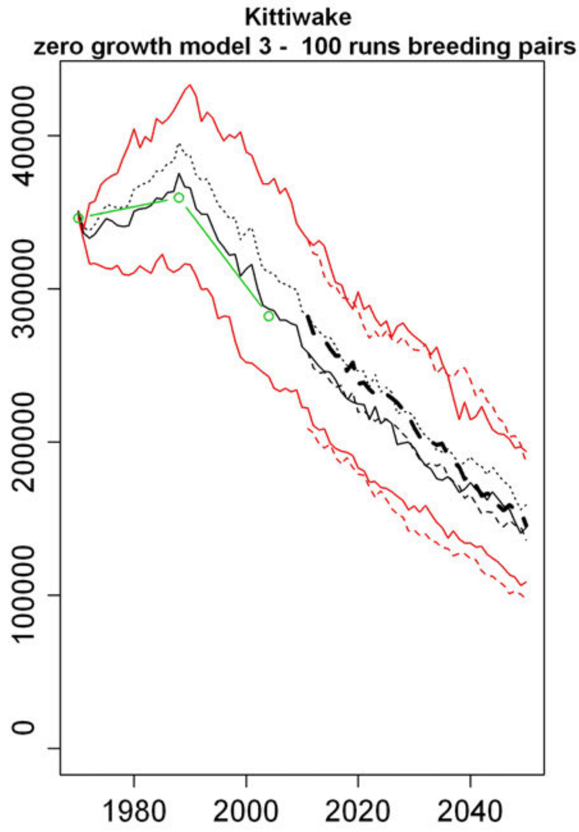


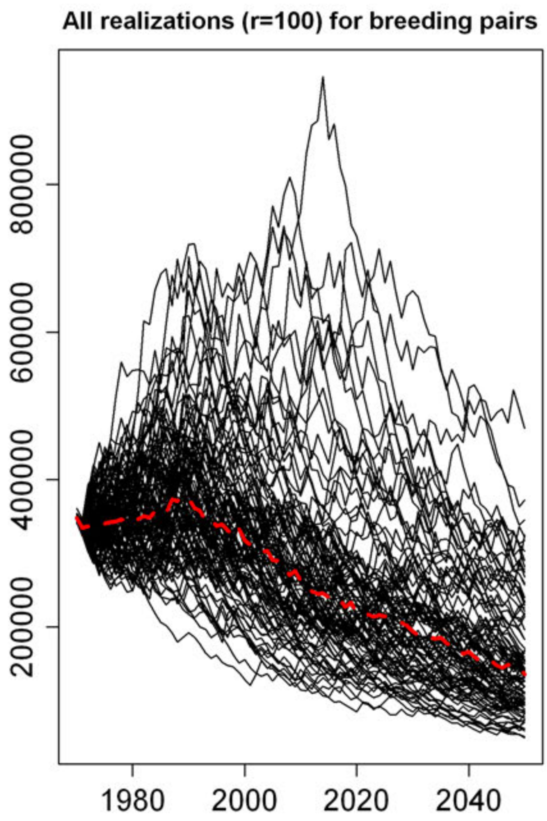
Zero Growth : 10% floaters

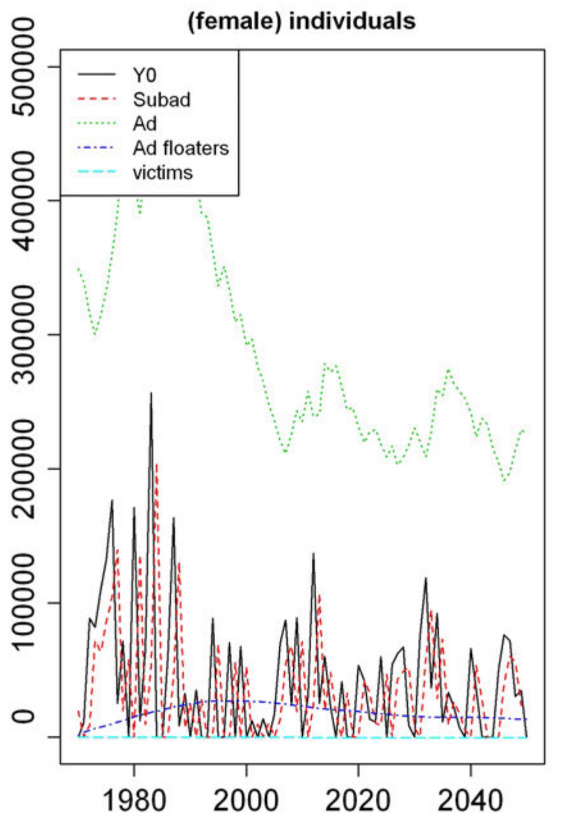
S0 : 0.79 S0sd : 0.01
Sad: 0.941 Sadsd: 0.01
F0 : 0.072 F0sd : 0.2
B1 : 4
Floaters % >=2010: 8

K: 359425

Victims : 0 0 750 0
Init population: 25000 50000 345000 0





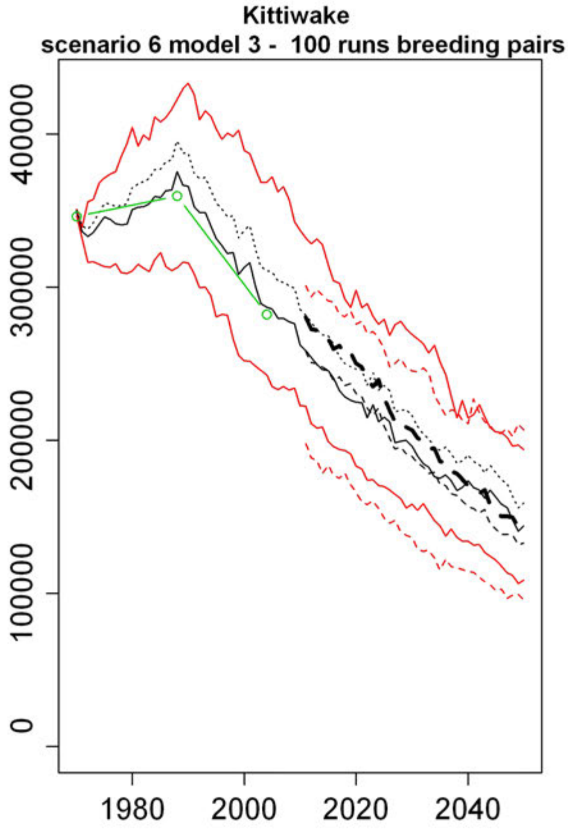


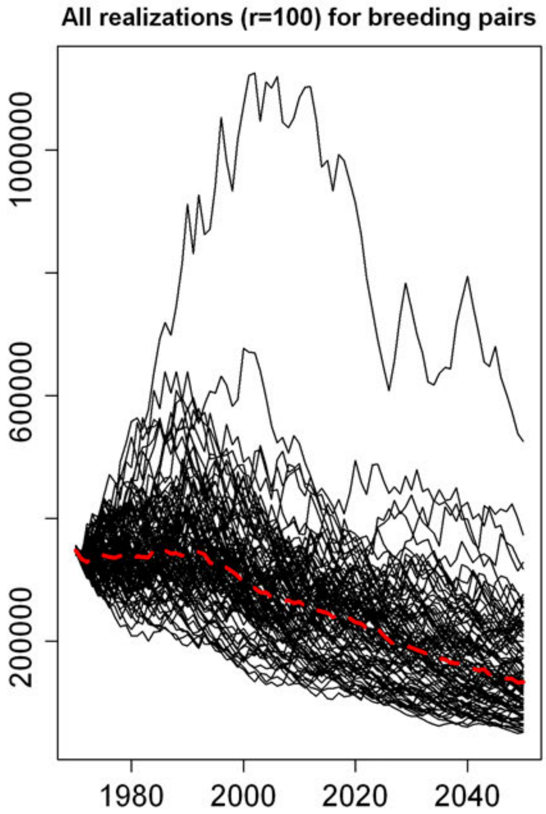
Scenario collisions Band-model : 10% floaters

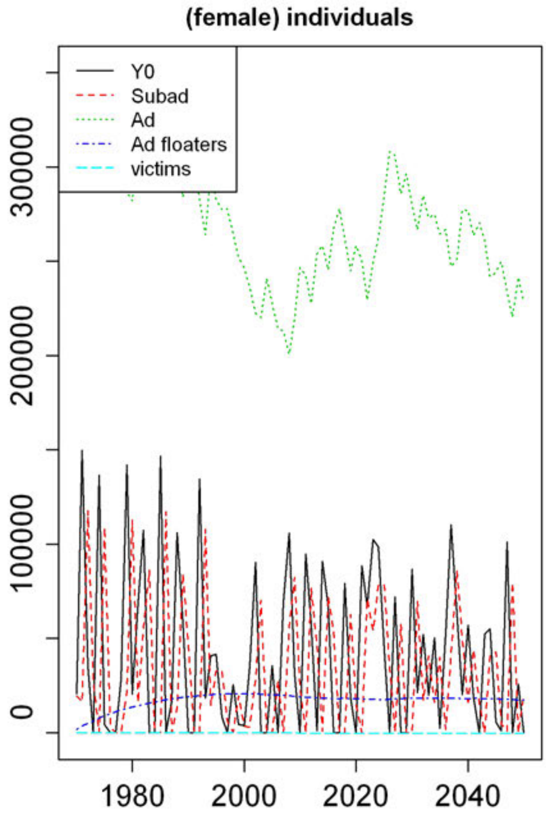
S0 : 0.79 S0sd : 0.01
Sad: 0.941 Sadsd: 0.01
F0 : 0.072 F0sd : 0.2
B1 : 4
Floaters % >=2010: 9

K: 359425

Victims : 0 0 345 0
Init population: 25000 50000 345000 0





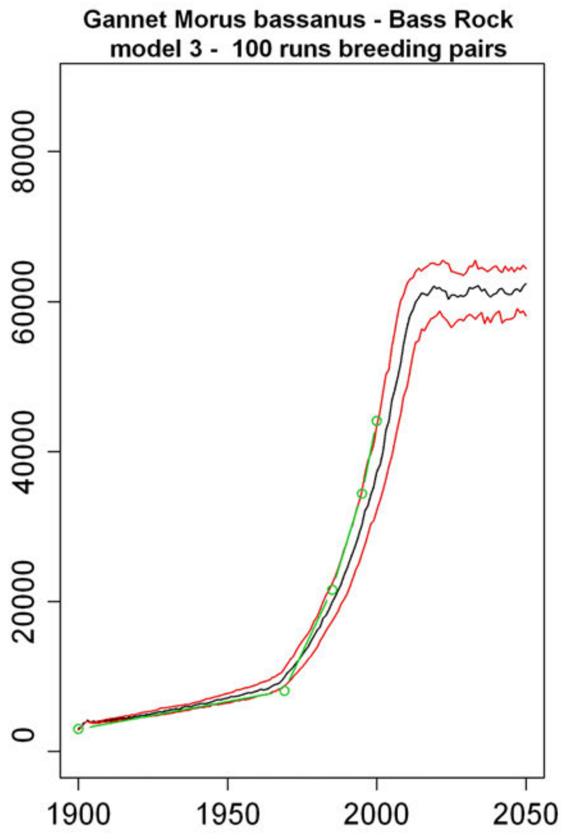


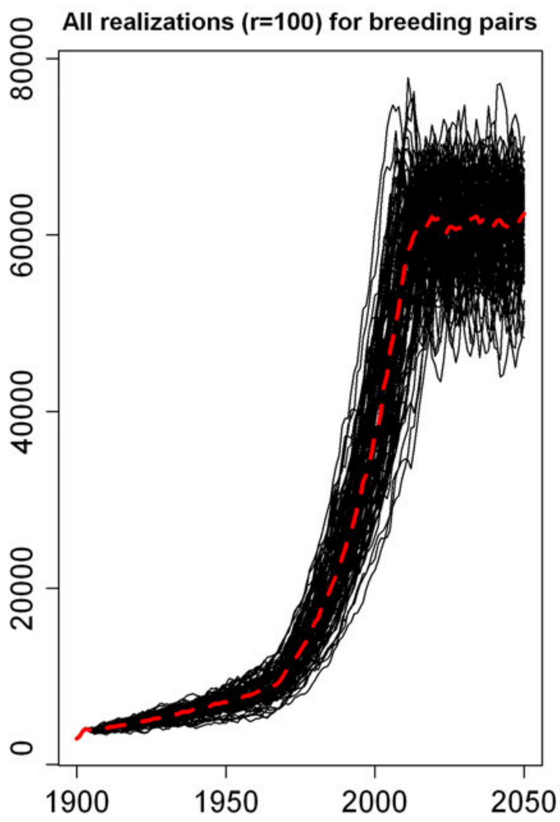
Gannet - Bass Rock

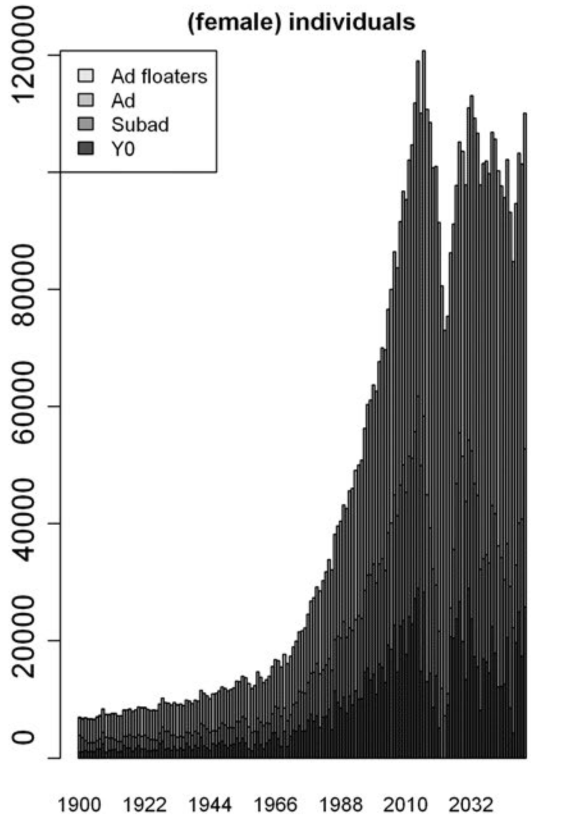
0 model – 0% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0 < 1965 : 0.36 F0sd : 0.1
F0 >= 1965 : 0.475 F0sd : 0.1
B1: 5
Floaters % >=2010: 0

Init population: 2000 1500 750 750 2500 200



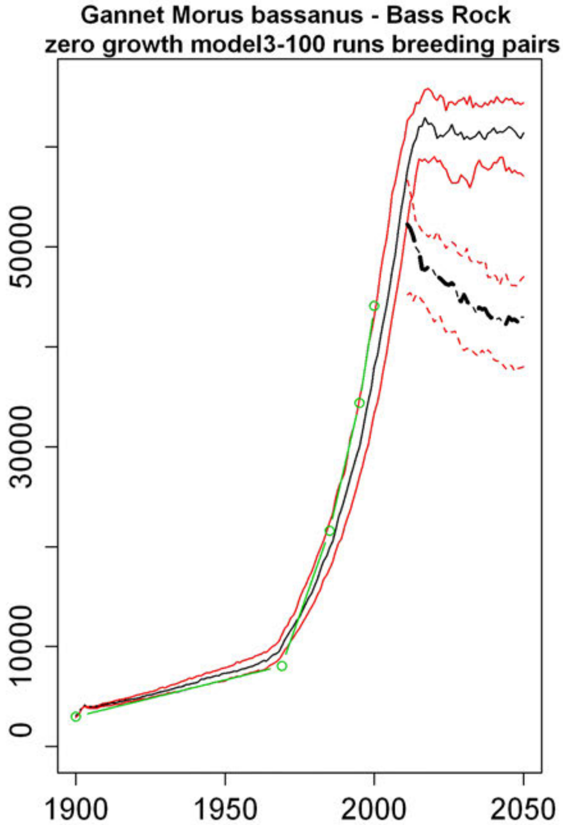


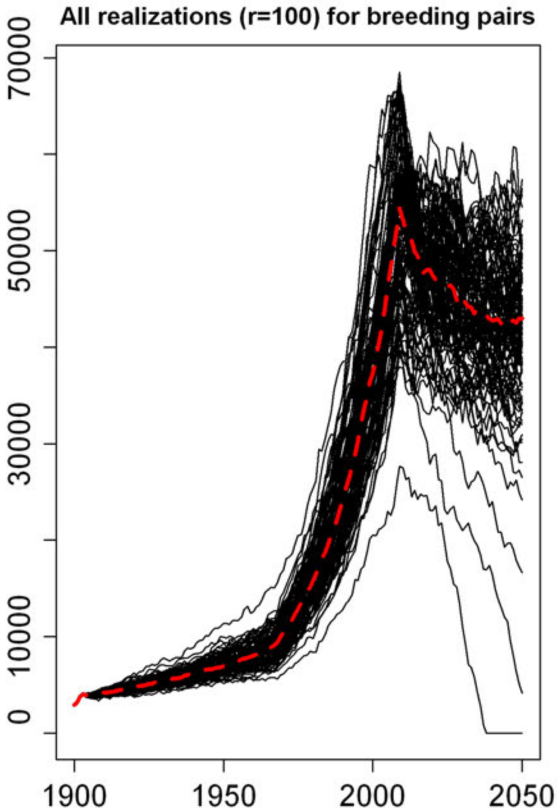


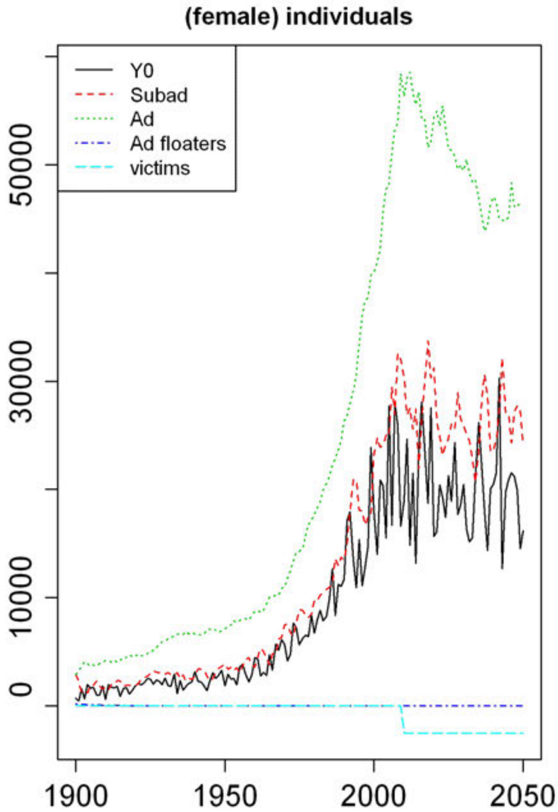
zero growth – 0% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0 < 1965 : 0.36 F0sd : 0.1
F0 >= 1965 : 0.475 F0sd : 0.1
B1: 5
Floaters % >=2010: 0

K: 61350
victims : 0 0 0 0 2500 0
Init population: 2000 1500 750 750 2500 200



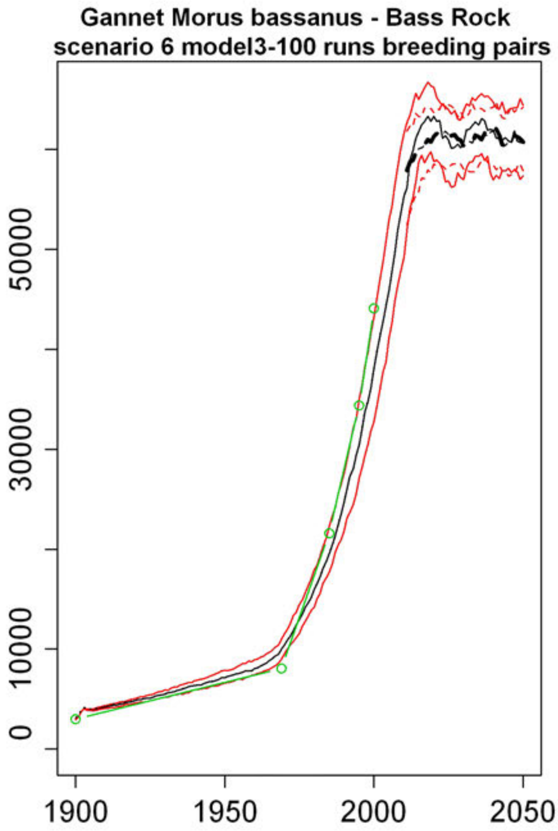


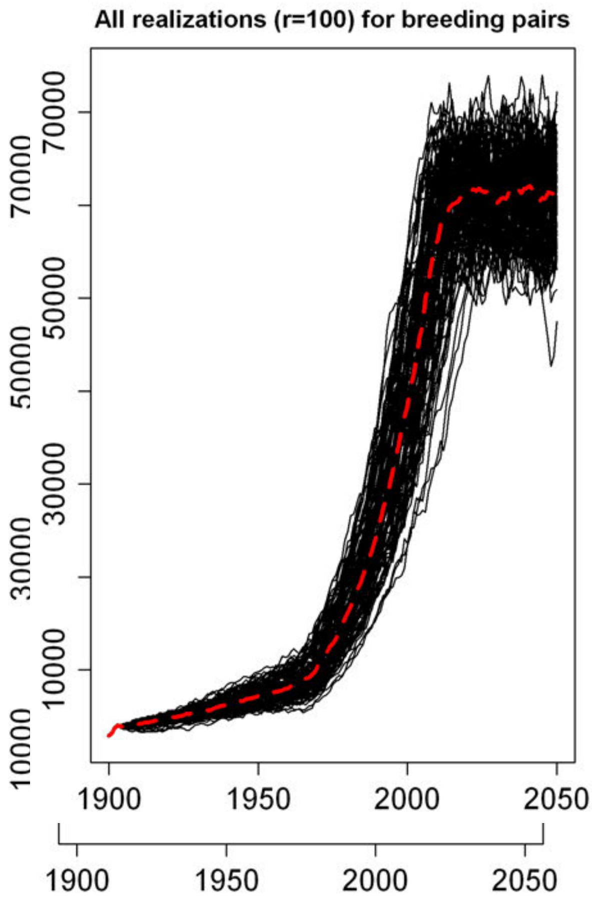


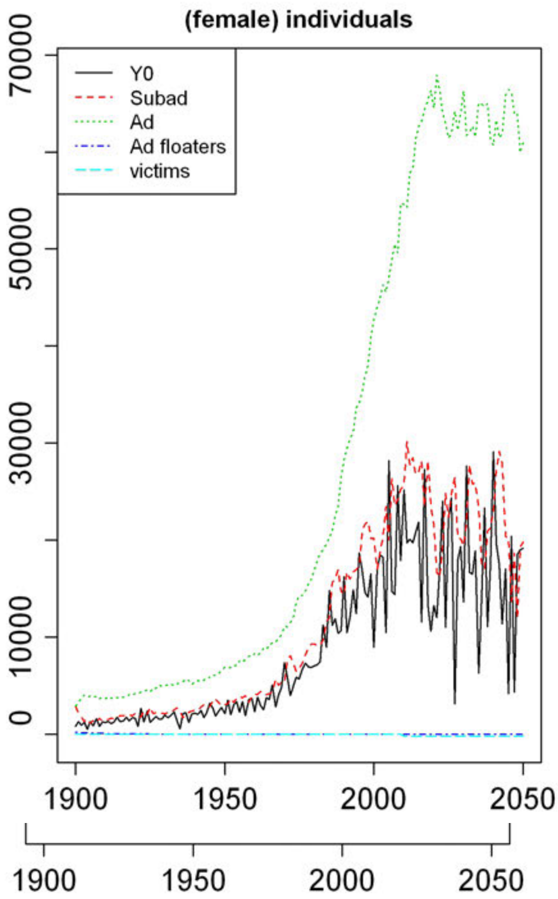
scenario collisions Band-model – 0% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0   < 1965 : 0.36 F0sd : 0.1
F0 >= 1965 : 0.475 F0sd : 0.1
B1: 5
Floaters % >=2010: 0

K: 61350
victims   : 0 0 0 0 200 0
Init population: 2000 1500 750 750 2500 200



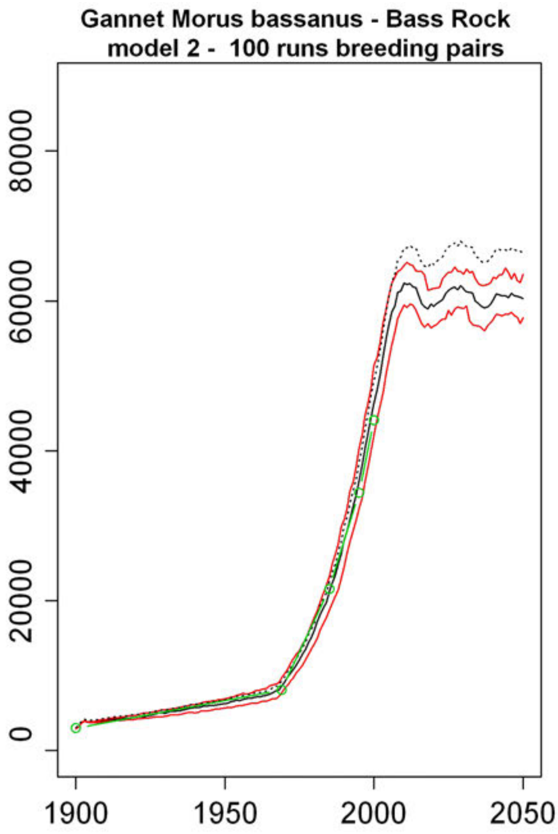


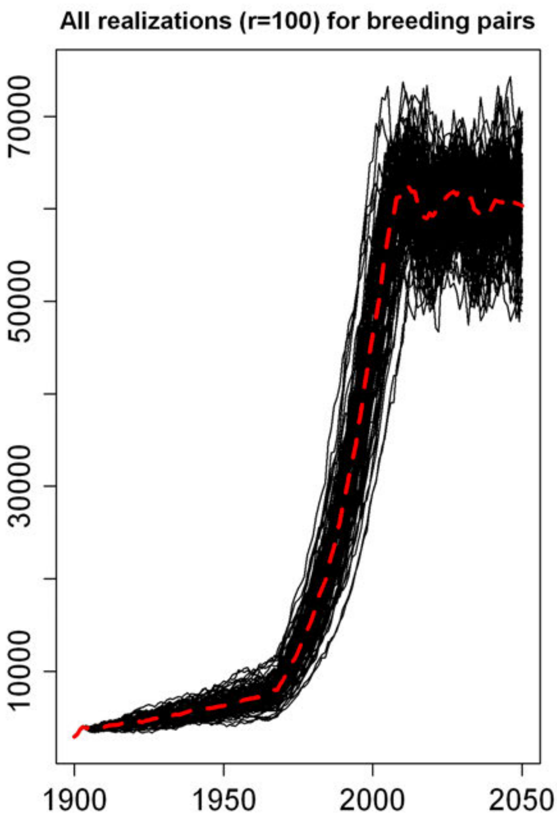


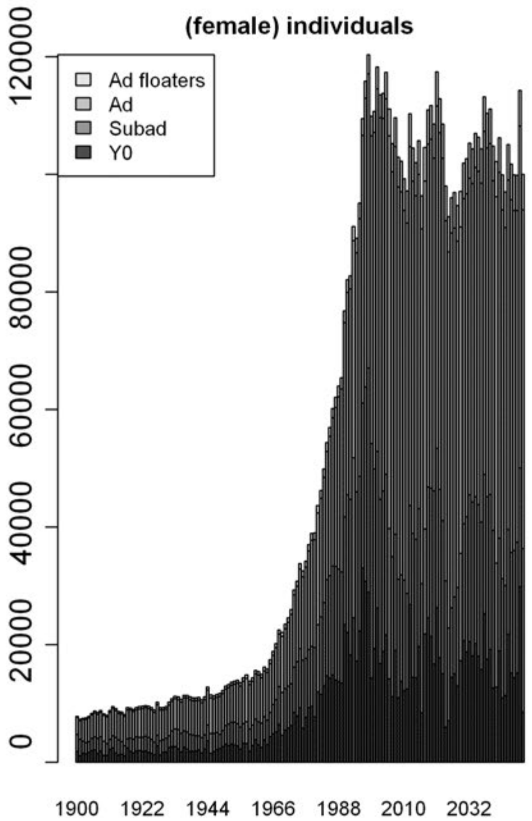
0 model – 10% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0 < 1965 : 0.38 F0sd : 0.1
F0 >= 1965 : 0.58 F0sd : 0.1
B1: 5
Floaters % >=2010: 10

Init population: 2000 1500 750 750 2500 200



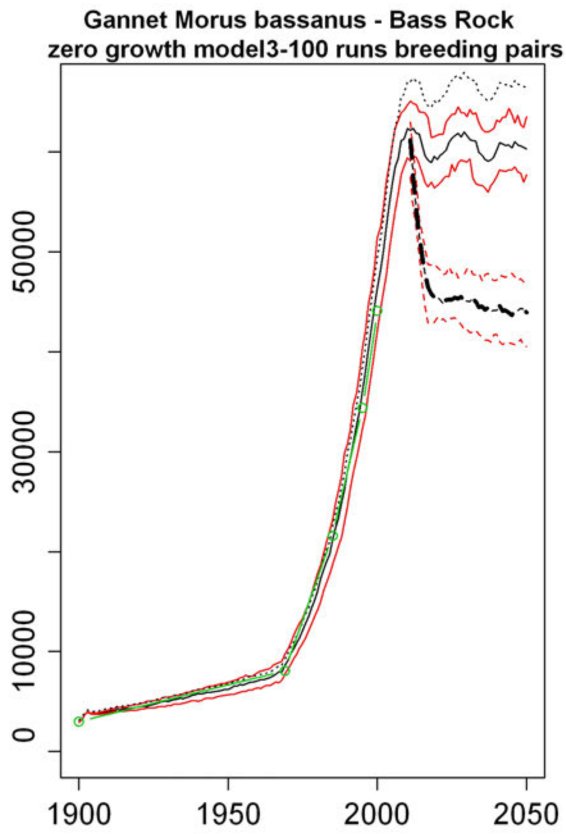


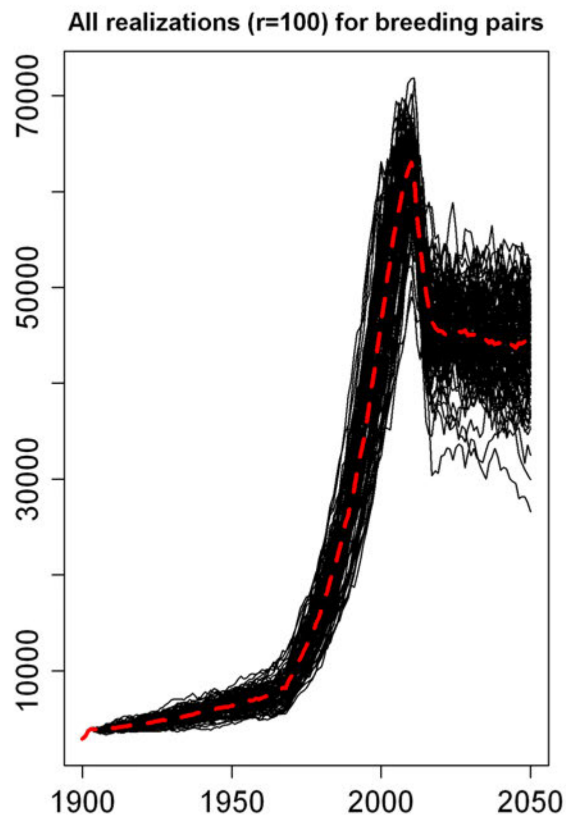


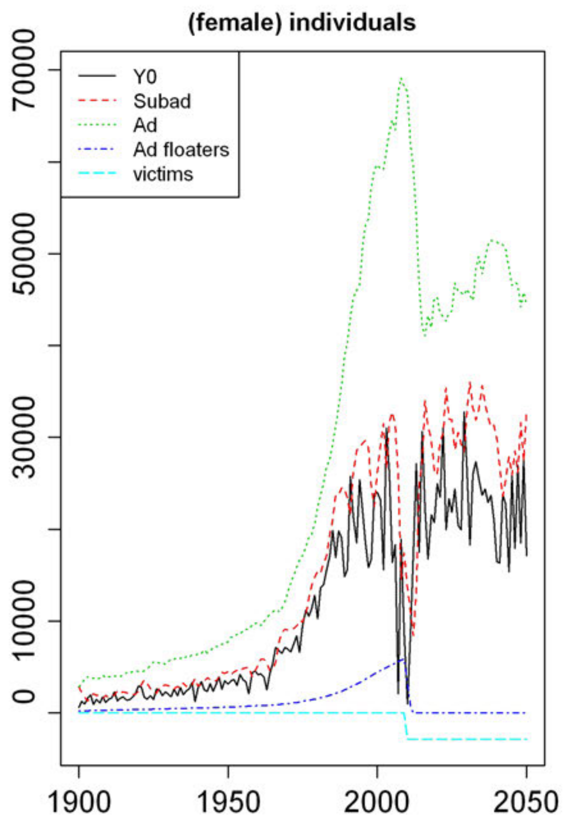
zero growth - 10% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0   < 1965 : 0.38 F0sd : 0.1
F0 >= 1965 : 0.58 F0sd : 0.1
B1: 5
Floaters % >=2010: 0

K: 61350
victims   : 0 0 0 0 2850 0
Init population: 2000 1500 750 750 2500 200



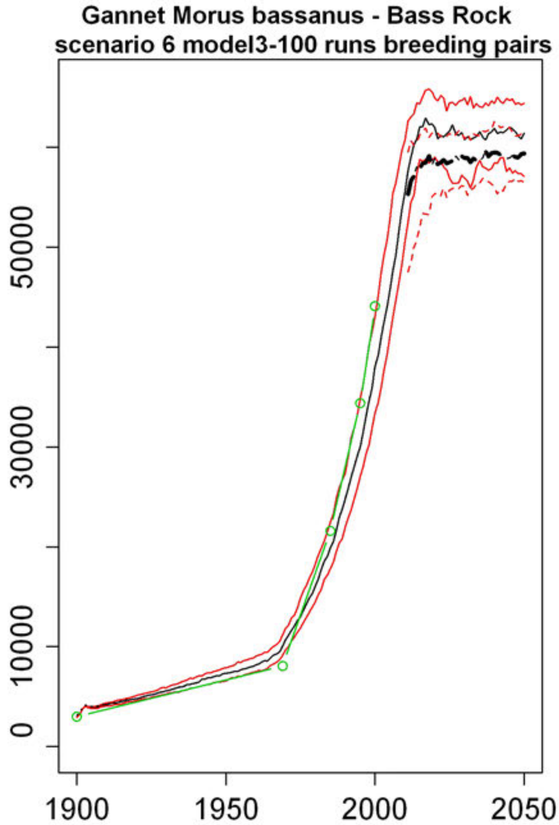


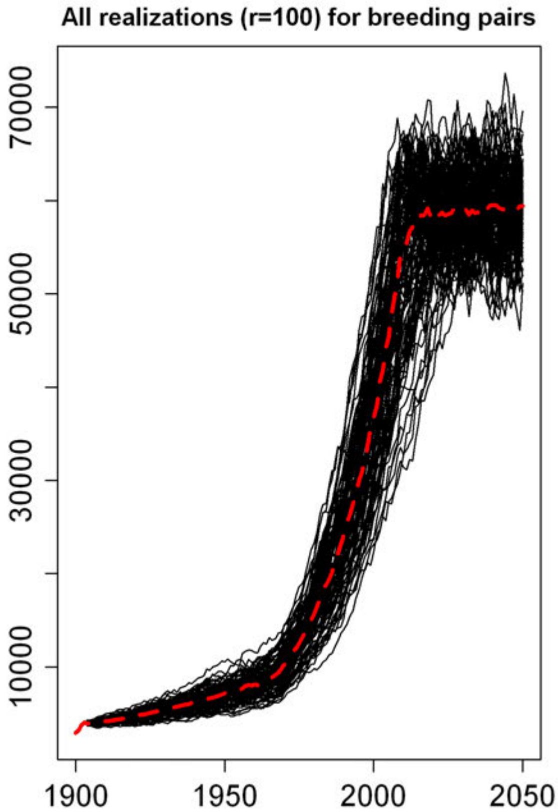


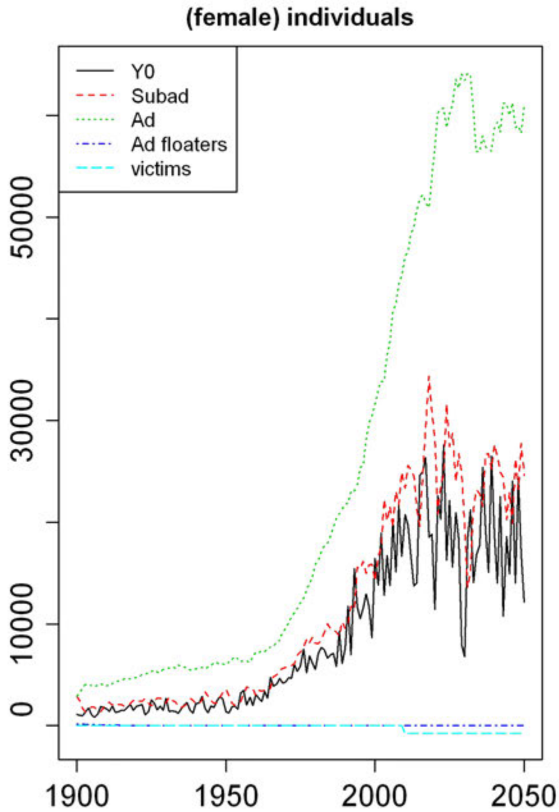
scenario collisions Band-model – 10% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0   < 1965 : 0.36 F0sd : 0.1
F0 >= 1965 : 0.475 F0sd : 0.1
B1: 5
Floaters % >=2010: 0

K: 61350
victims   : 0 0 0 0 200 0
Init population: 2000 1500 750 750 2500 200



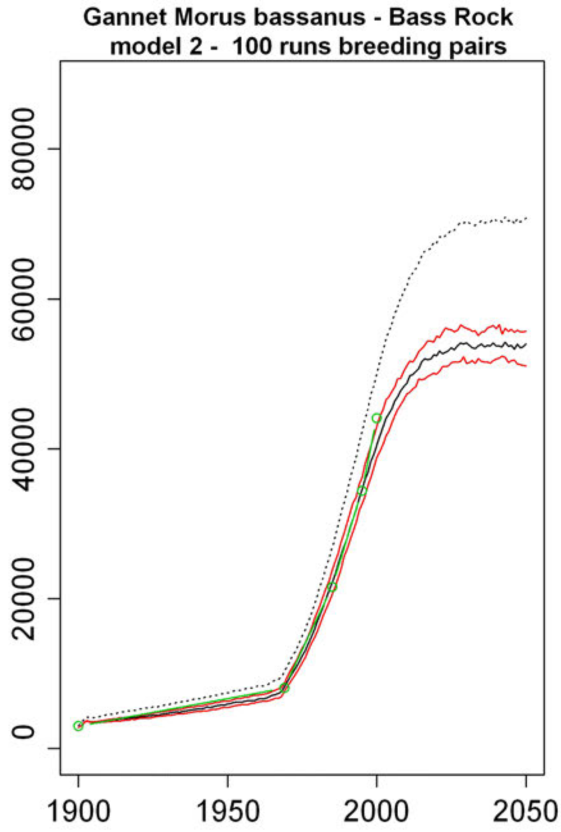


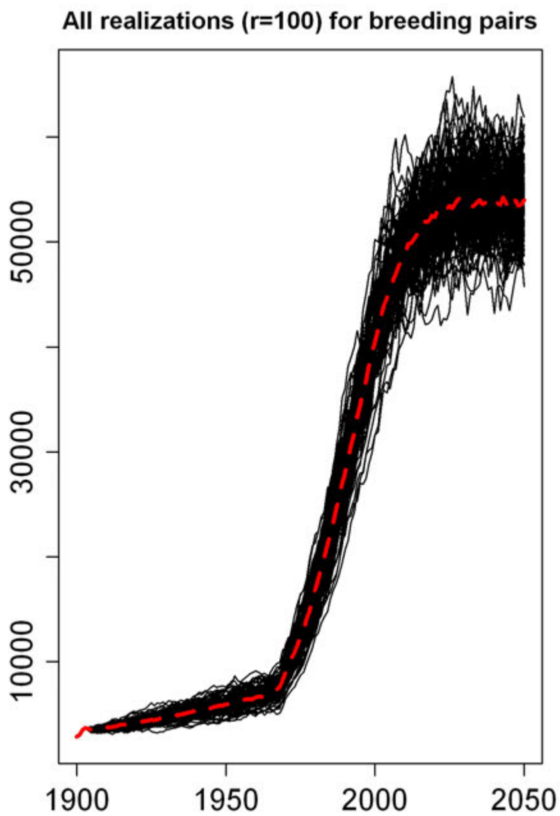


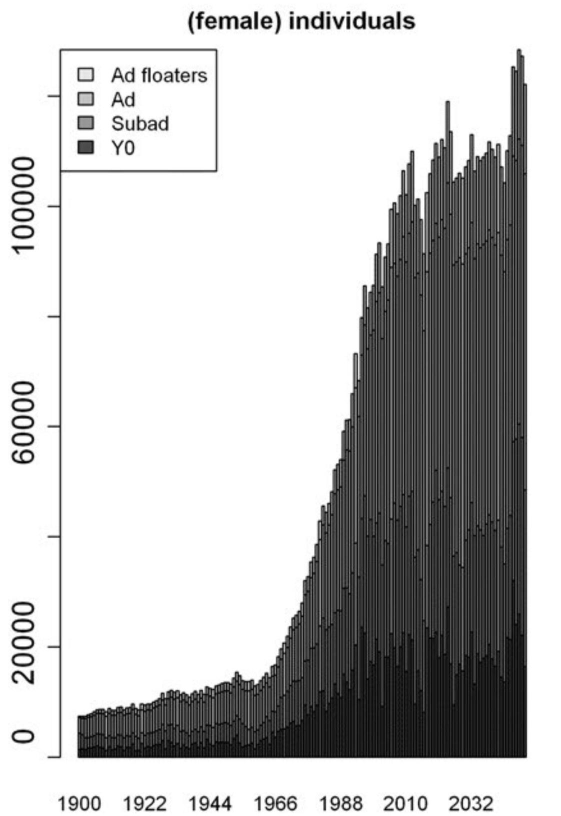
0-model – 30% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0 < 1965 : 0.455 F0sd : 0.1
F0 >= 1965 : 0.8 F0sd : 0.1
B1: 5
Floaters % >=2010: 30

Init population: 2000 1500 750 750 2500 200



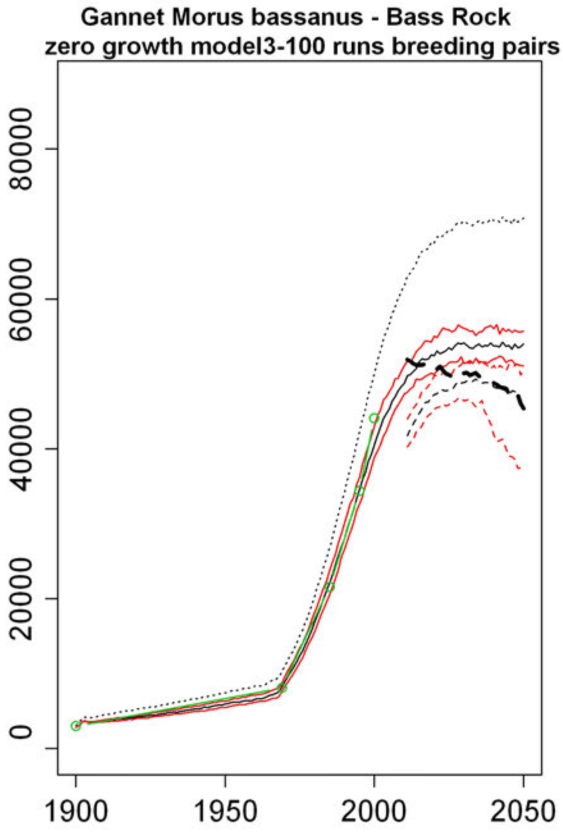


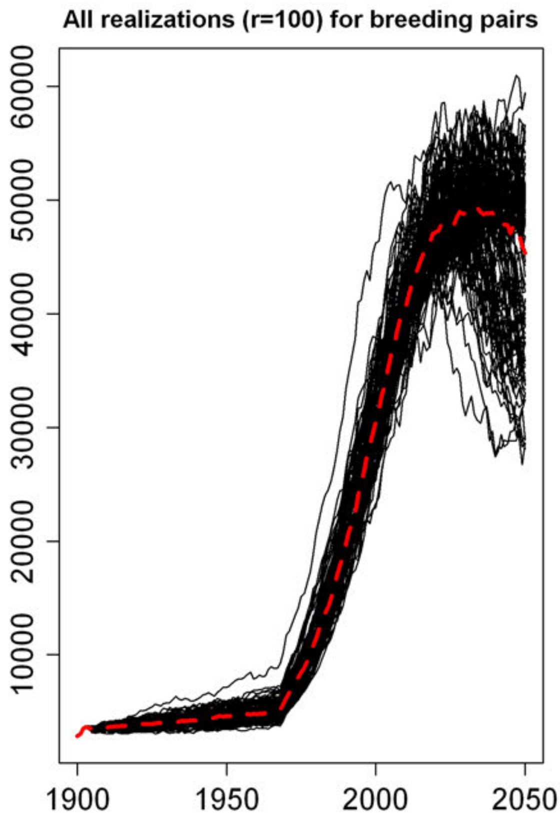


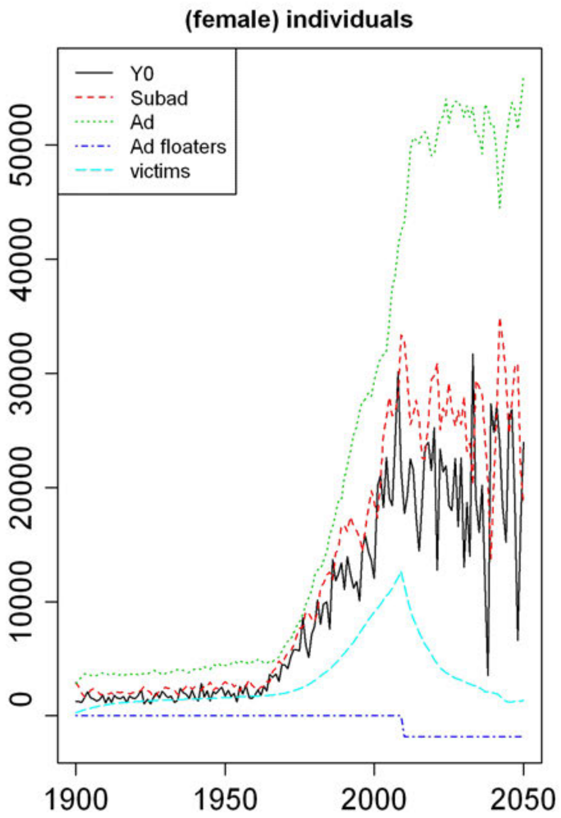
zero growth – 30% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0   < 1965 : 0.455 F0sd : 0.1
F0 >= 1965 : 0.8 F0sd : 0.1
B1: 5
Floaters % >=2010: 5

K: 61350
victims   : 0 0 0 0 1812.5 0
Init population: 2000 1500 750 750 2500 200



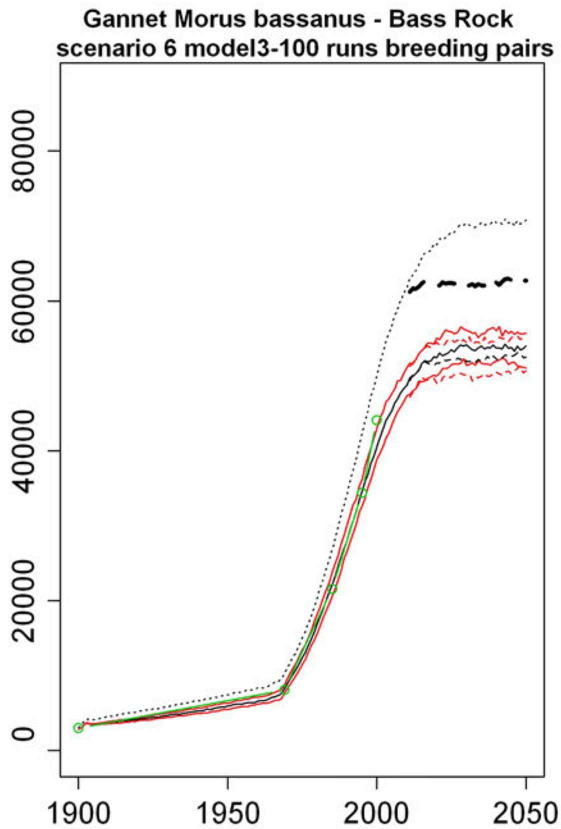


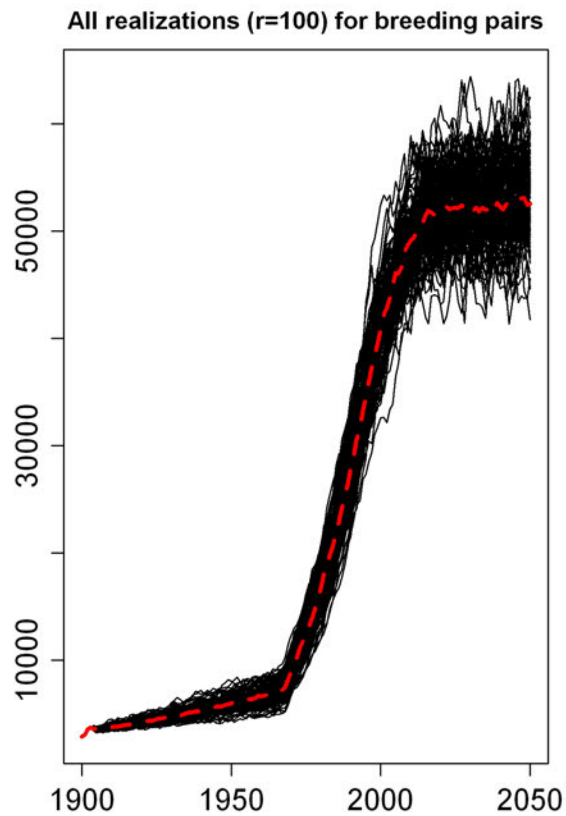


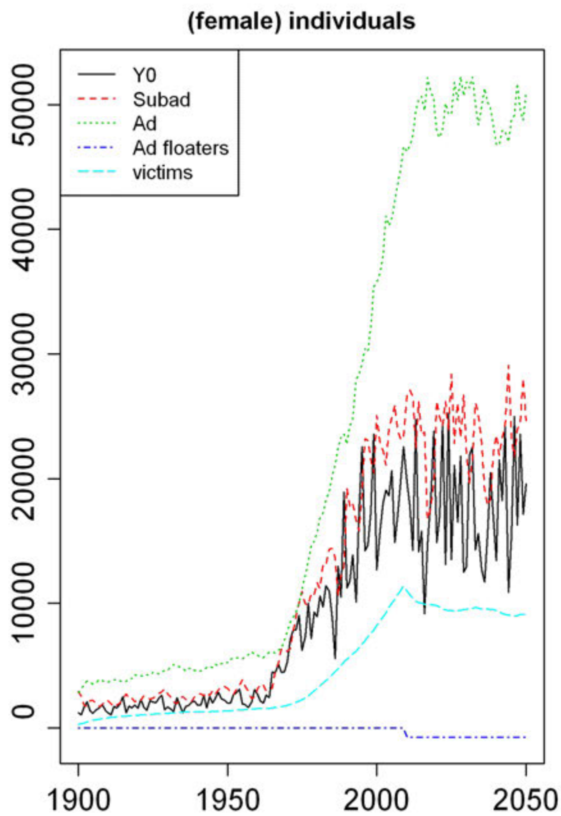
scenario collisions Band-model – 30% floaters

S0 : 0.542 S0sd : 0.01
S1 : 0.779 S1sd : 0.01
S2 : 0.859 S2sd : 0.01
S3 : 0.863 S3sd : 0.01
Sad < 1965 : 0.907 Sadsd: 0.002
Sad >= 1965 : 0.919 Sadsd: 0.002
F0   < 1965 : 0.455 F0sd : 0.1
F0 >= 1965 : 0.8 F0sd : 0.1
B1: 5
Floaters % >=2010: 20

K: 61350
victims   : 0 0 0 0 200 0
Init population: 2000 1500 750 750 2500 200





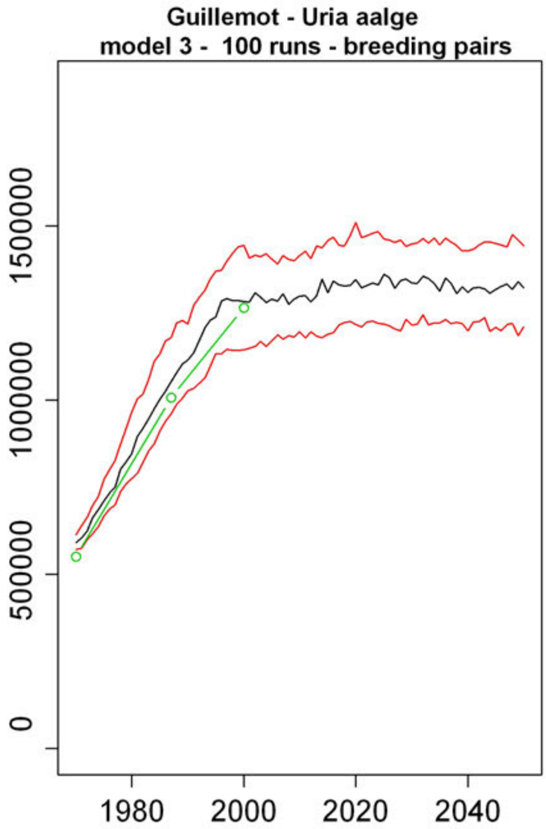


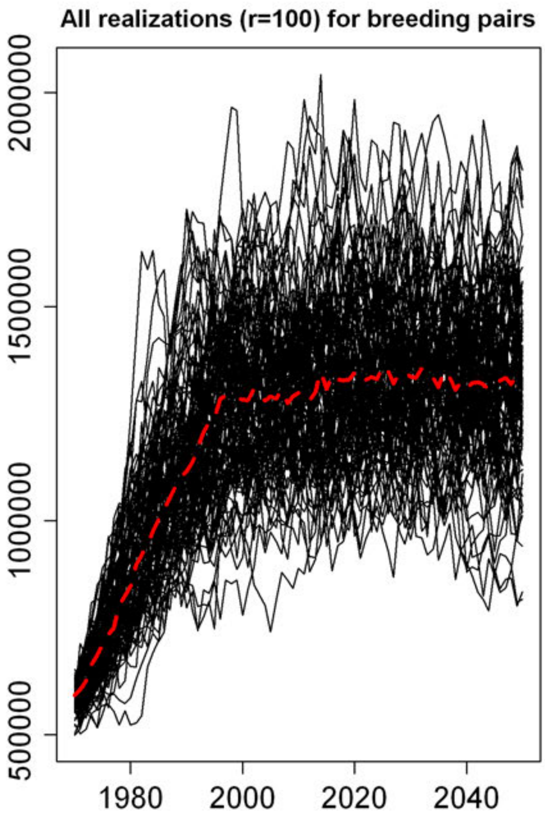
Guillemot - Scotland

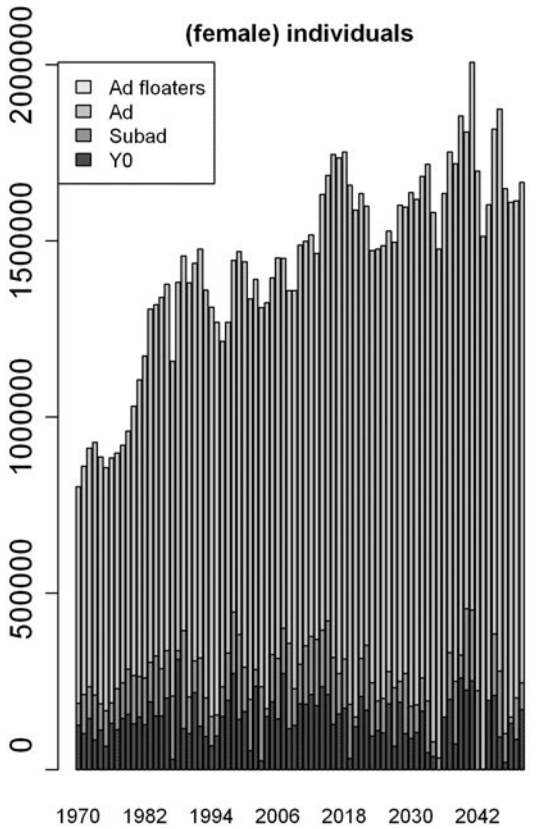
0 model – 0% floaters

S0 : 0.89 S0sd : 0.05
Sad: 0.946 Sadsd: 0.05
F0 : 0.235 F0sd : 0.06
B1: 4.5
Floaters % >=2010: 0

K: 1264869
Init population: 71000 100000 550000 0
victims : 0 0 0 0



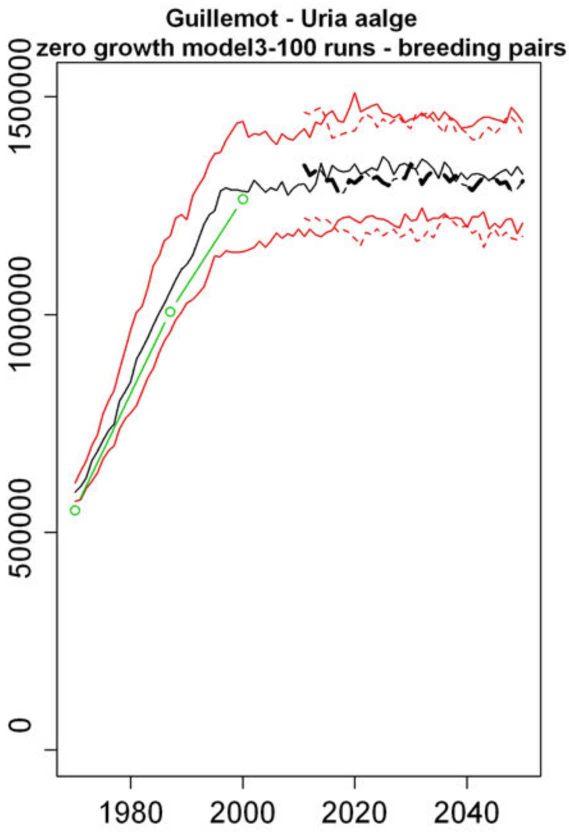


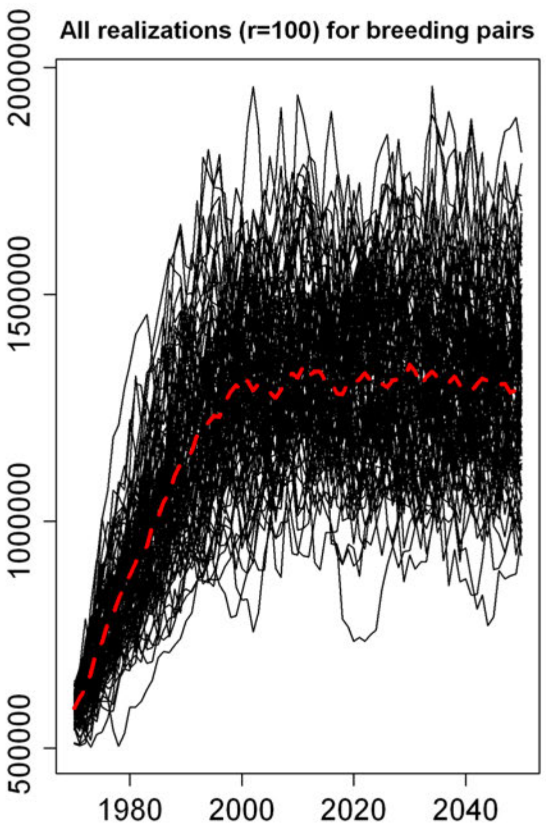


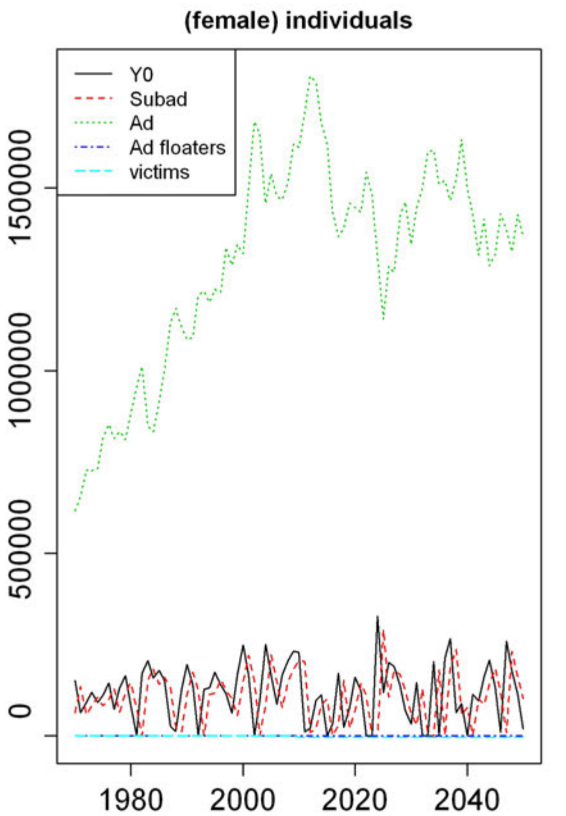
Zero growth – 0% floaters

S0 : 0.89 S0sd : 0.05
Sad: 0.946 Sadsd: 0.05
F0 : 0.235 F0sd : 0.06
B1: 4.5
Floaters % >=2010: 0

K: 1264869
Init population: 71000 100000 550000 0
victims : 0 0 4000 0



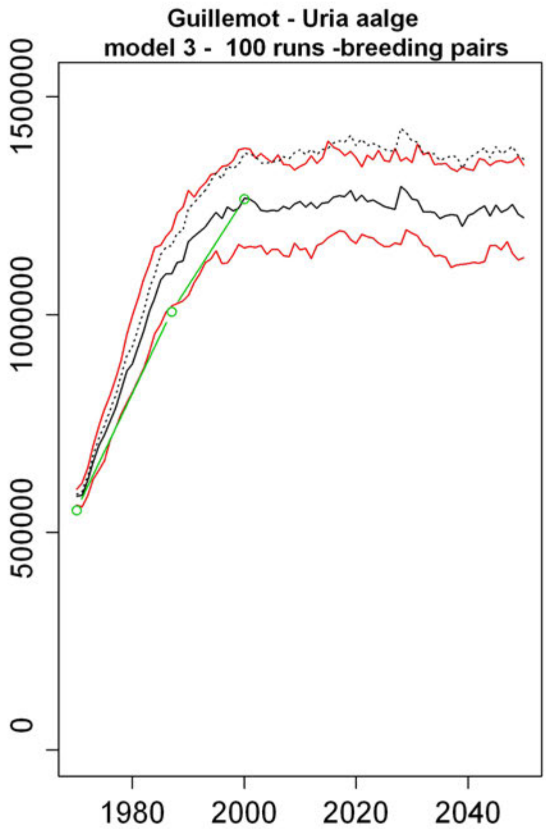


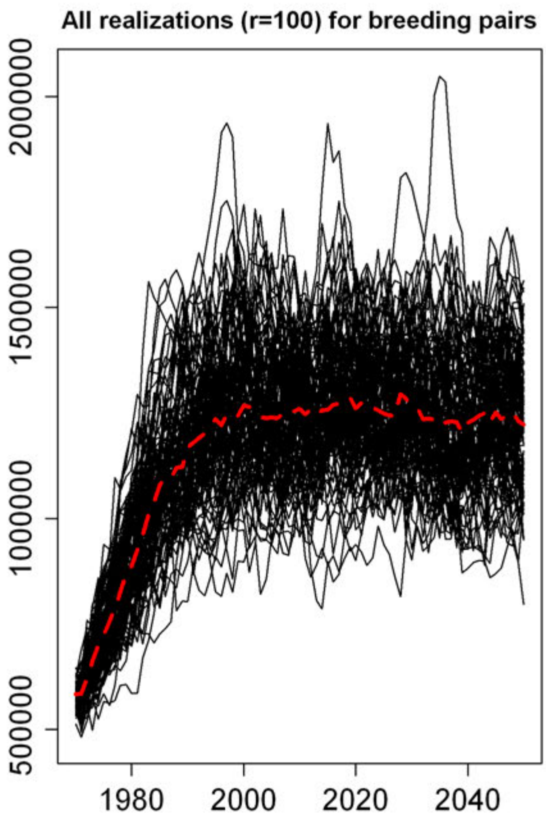


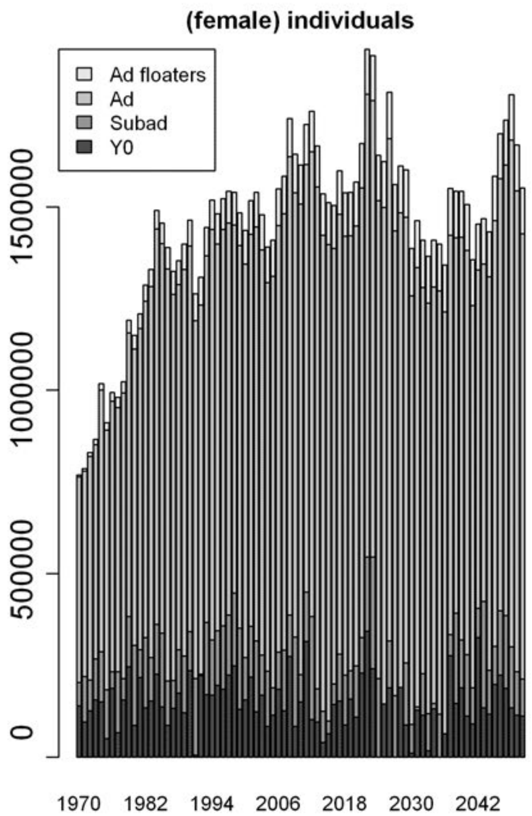
0 model – 10% floaters

S0 : 0.89 S0sd : 0.05
Sad: 0.946 Sadsd: 0.05
F0 : 0.25 F0sd : 0.06
B1: 4.5
Floaters % >=2010: 10

K: 1264869
Init population: 71000 100000 550000 550
victims : 0 0 0 0



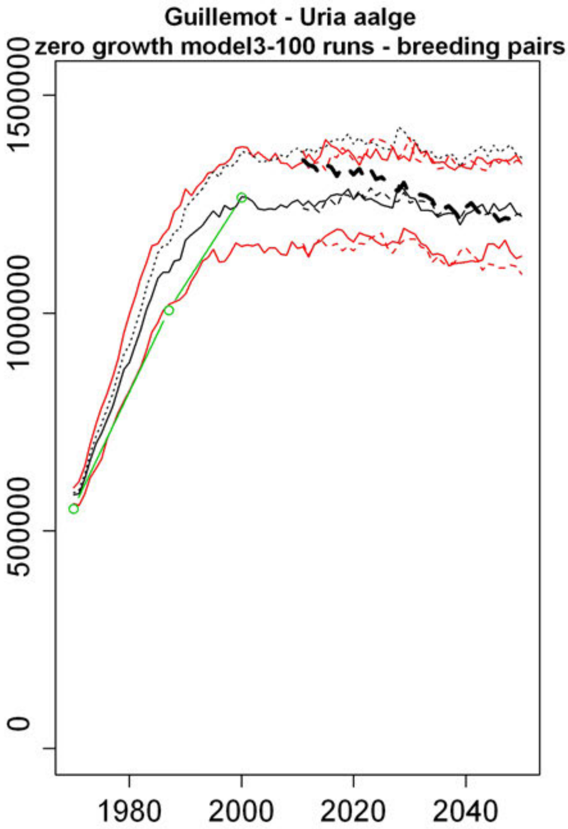


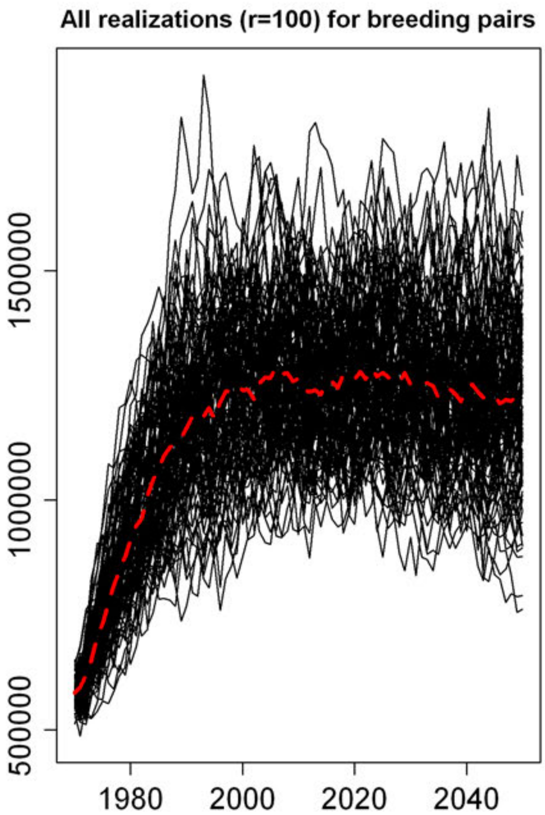


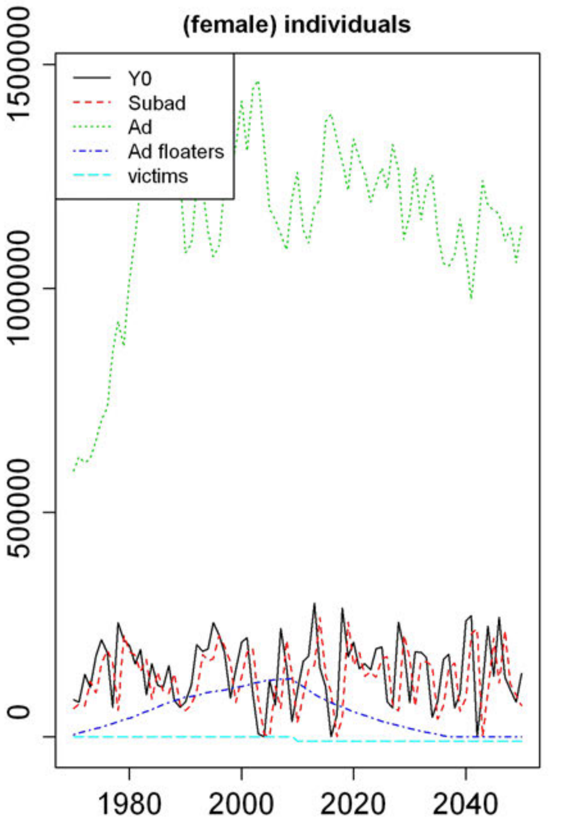
Zero growth – 10% floaters

S0 : 0.89 S0sd : 0.05
Sad: 0.946 Sadsd: 0.05
F0 : 0.25 F0sd : 0.06
B1: 4.5
Floaters % >=2010: 3

K: 1264869
Init population: 71000 100000 550000 550
victims : 0 0 10000 0



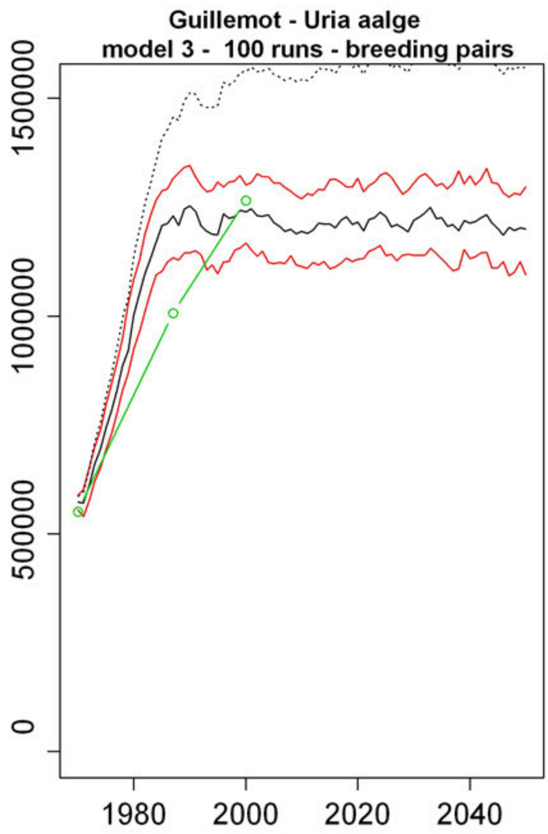


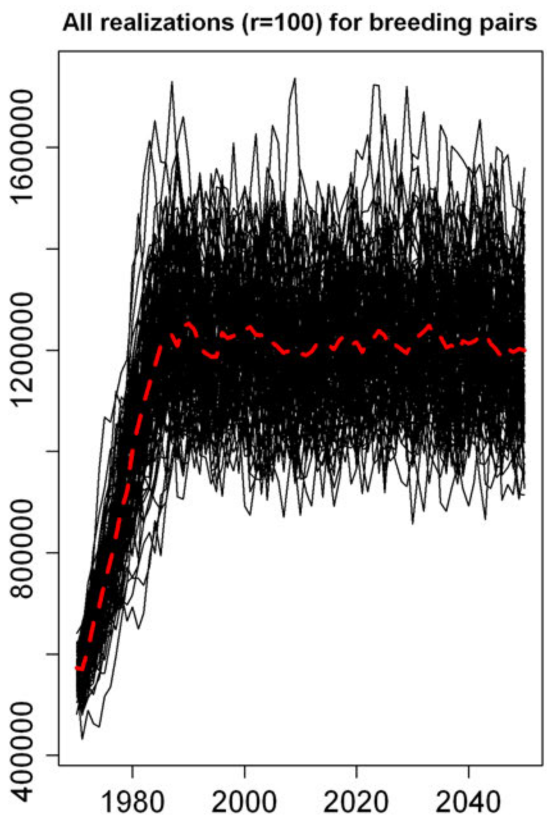


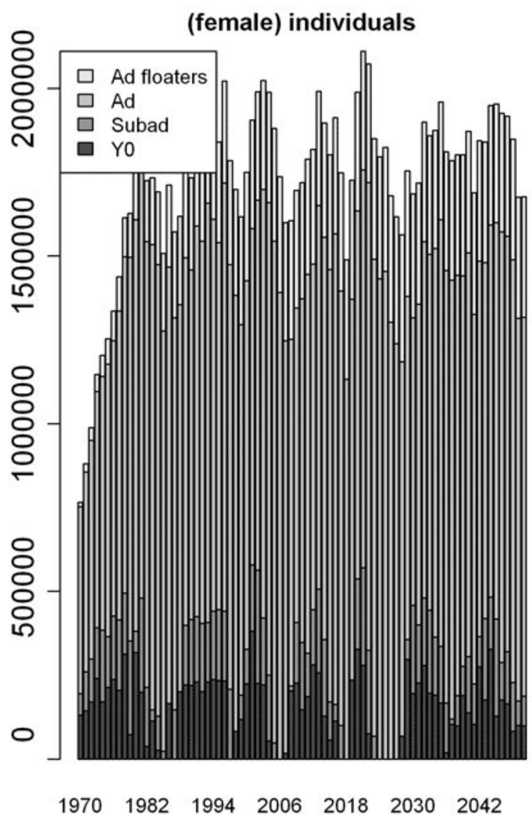
0 model – 30% floaters

S0 : 0.89 S0sd : 0.05
Sad: 0.946 Sadsd: 0.05
F0 : 0.275 F0sd : 0.06
B1: 4.5
Floaters % >=2010: 30

K: 1264869
Init population: 71000 100000 550000 550
victims : 0 0 0 0



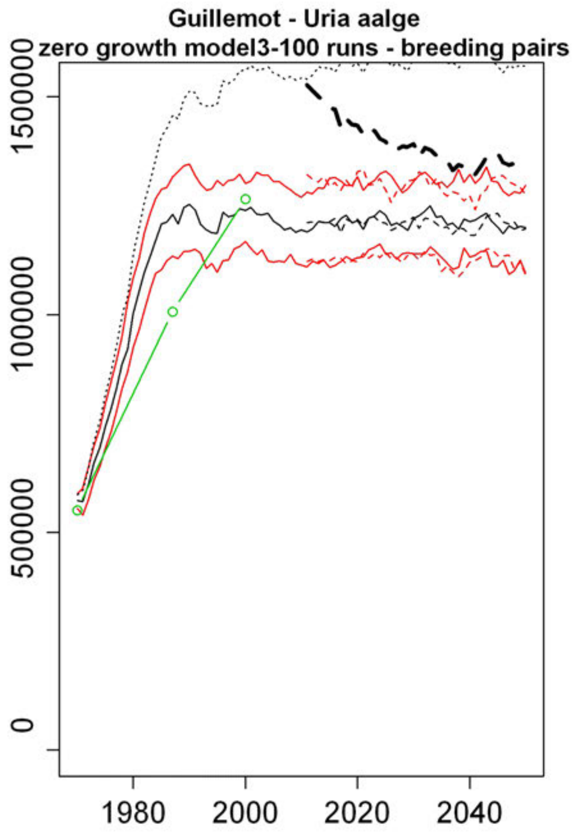


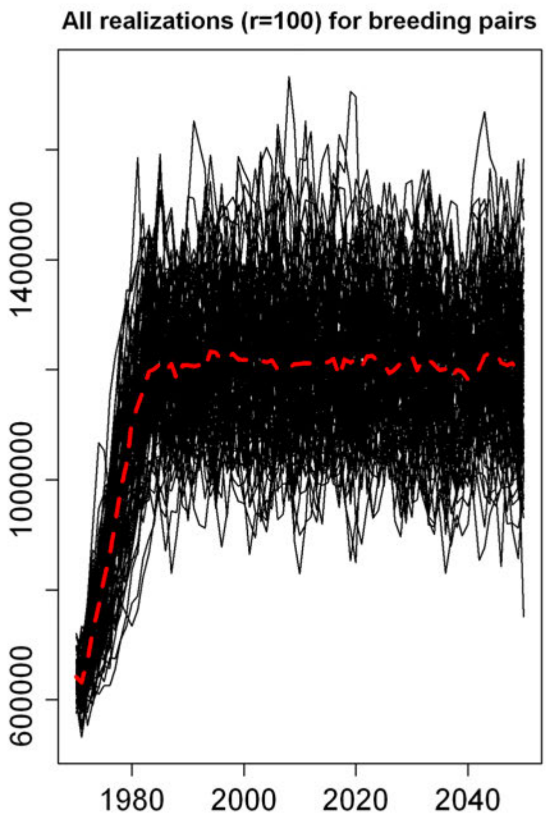


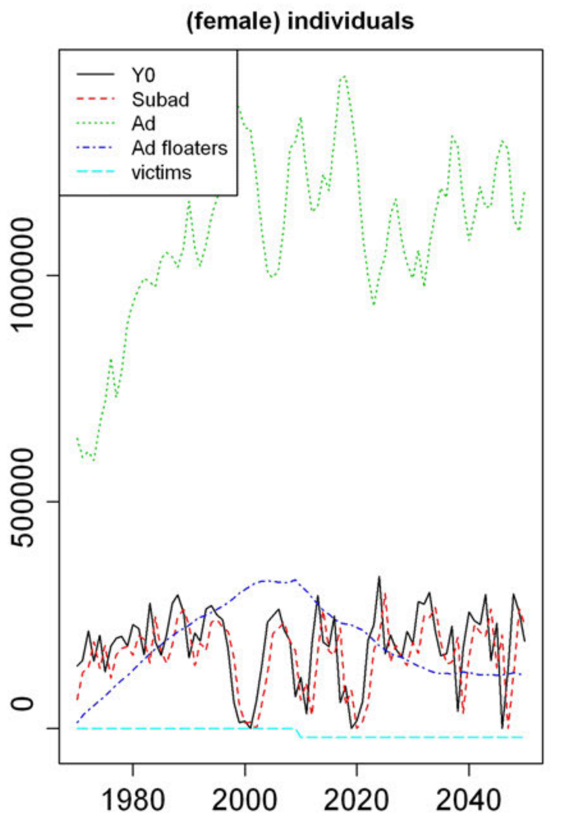
Zero growth – 30% floaters

S0 : 0.89 S0sd : 0.05
Sad: 0.946 Sadsd: 0.05
F0 : 0.275 F0sd : 0.06
B1: 4.5
Floaters % >=2010: 16

K: 1264869
Init population: 71000 100000 550000 550
victims : 0 0 19000 0





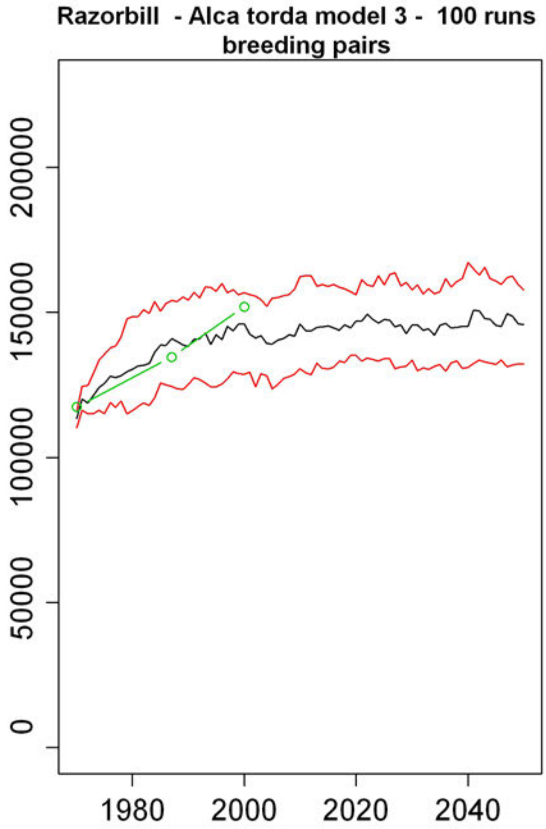


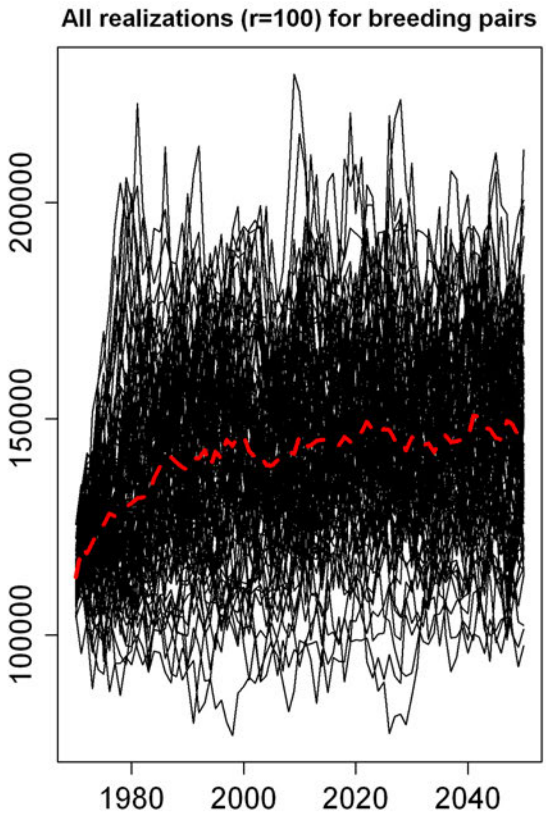
Razorbill - Scotland

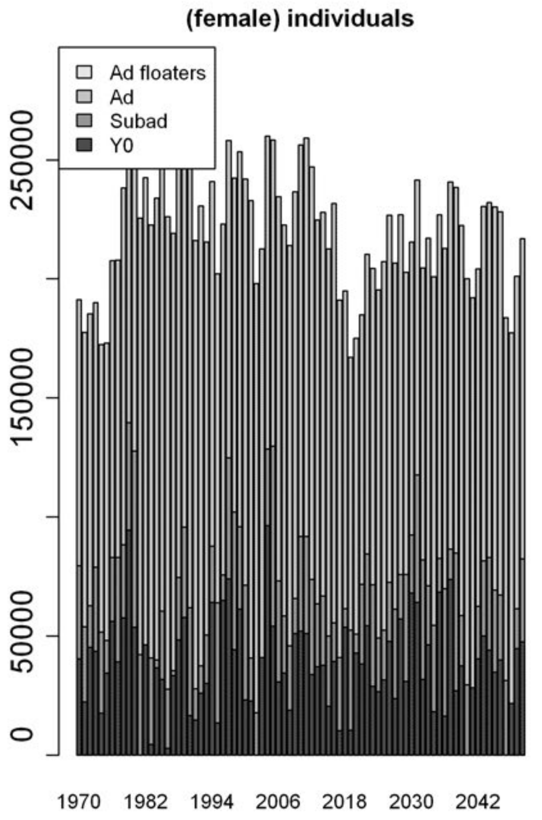
0 model – 0% floaters

S0 : 0.785 S0sd : 0.05
Sad: 0.9 Sadsd: 0.05
F0 : 0.32 F0sd : 0.13
B1: 4
Floaters % >=2010: 0

K: 151919
Init population: 50000 75000 85000 0
victims : 0 0 0 0



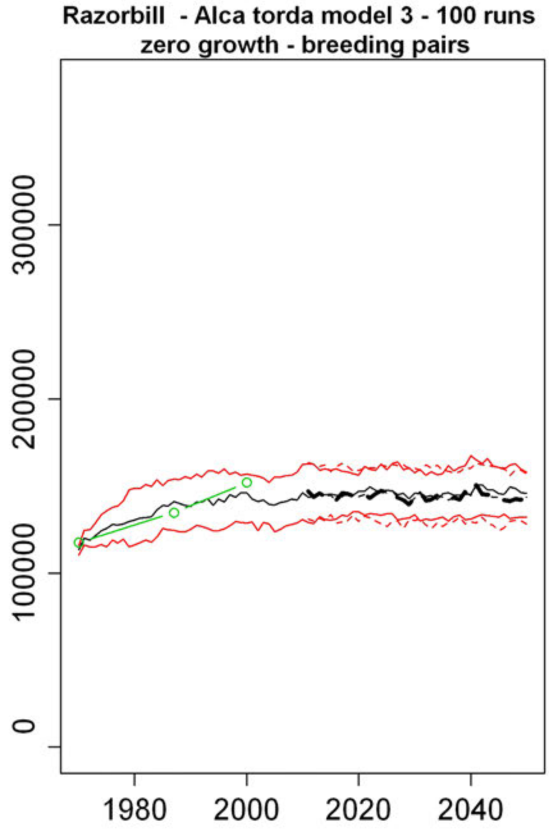


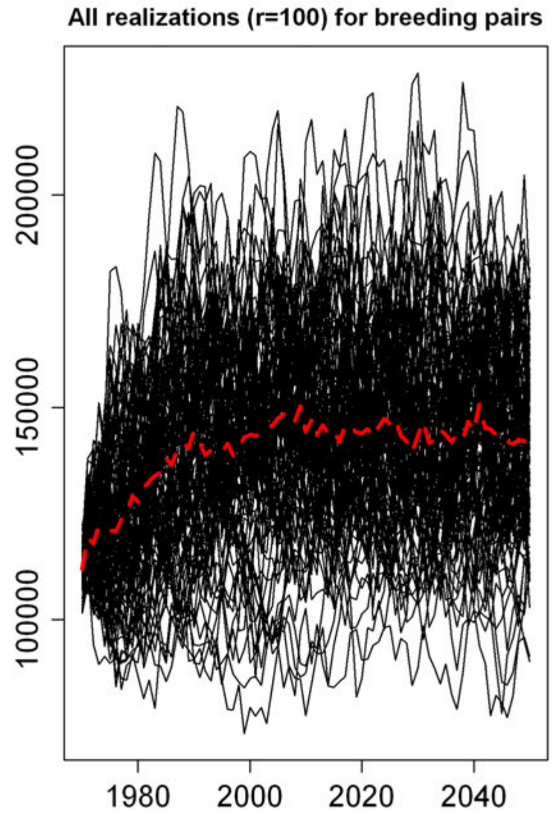


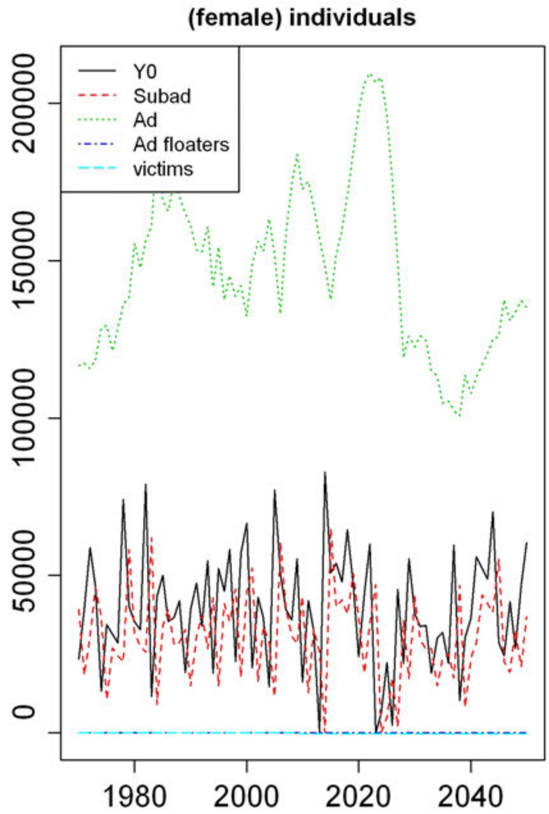
Zero growth – 0% floaters

S0 : 0.785 S0sd : 0.05
Sad: 0.9 Sadsd: 0.05
F0 : 0.32 F0sd : 0.13
B1: 4
Floaters % >=2010: 0

K: 151919
Init population: 50000 75000 85000 0
victims : 0 0 300 0



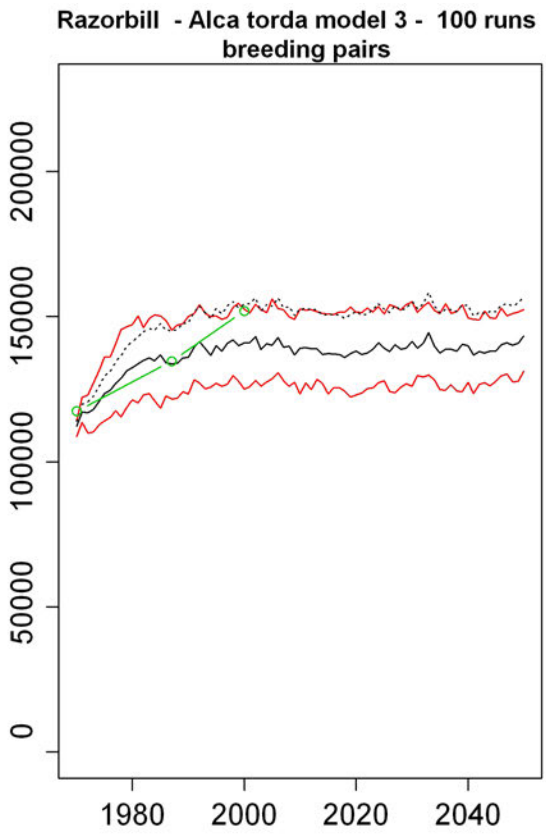


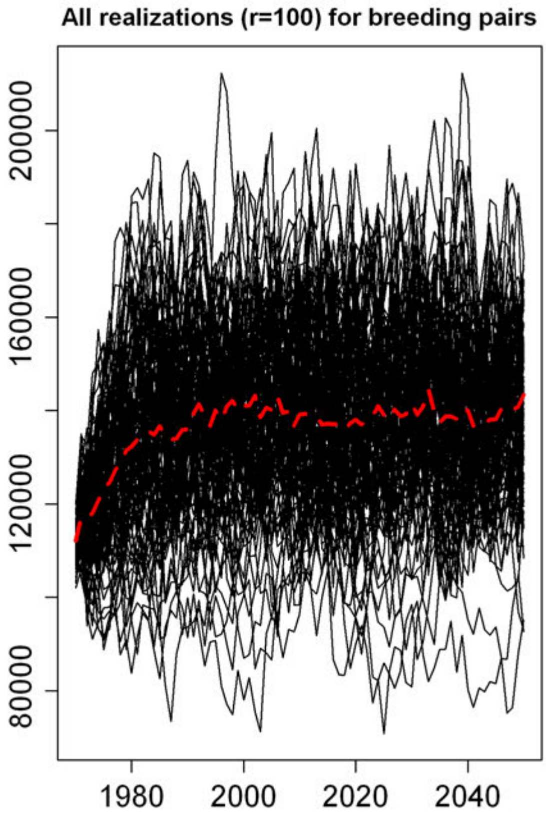


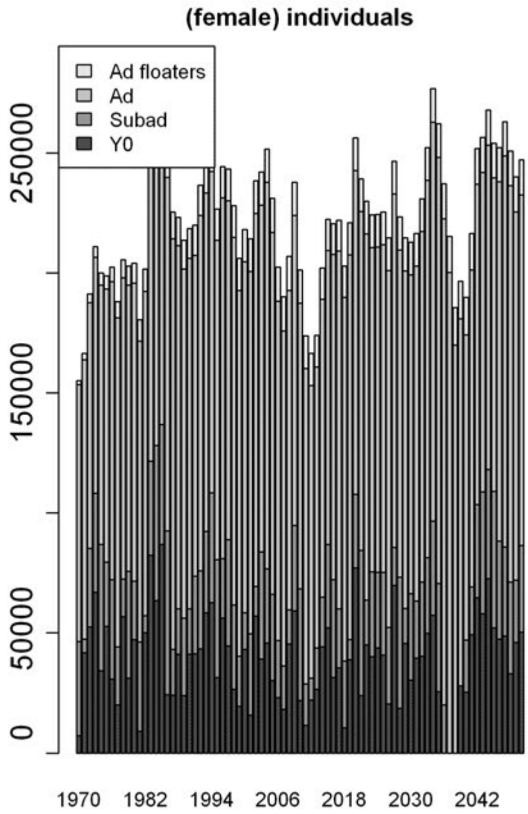
0 model – 10% floaters

S0 : 0.785 S0sd : 0.05
Sad: 0.9 Sadsd: 0.05
F0 : 0.355 F0sd : 0.13
B1: 4
Floaters % >=2010: 10

K: 151919
Init population: 50000 75000 85000 850
victims   : 0 0 0 0



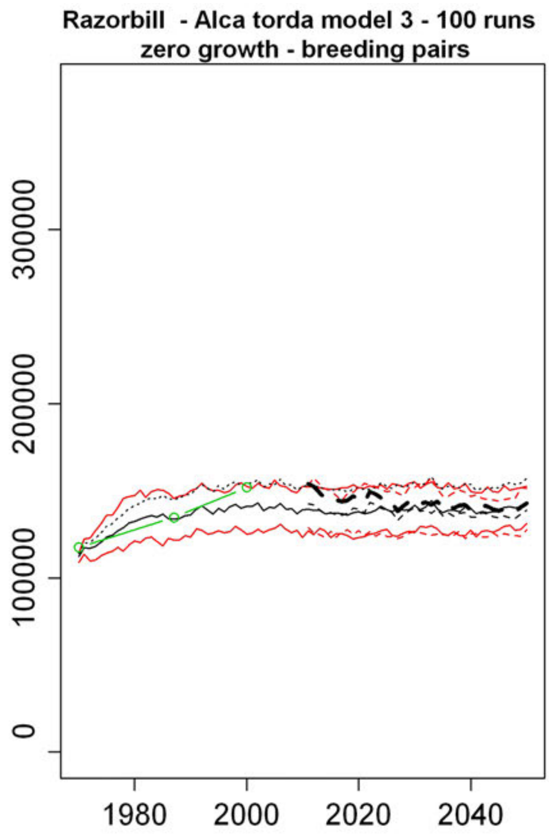


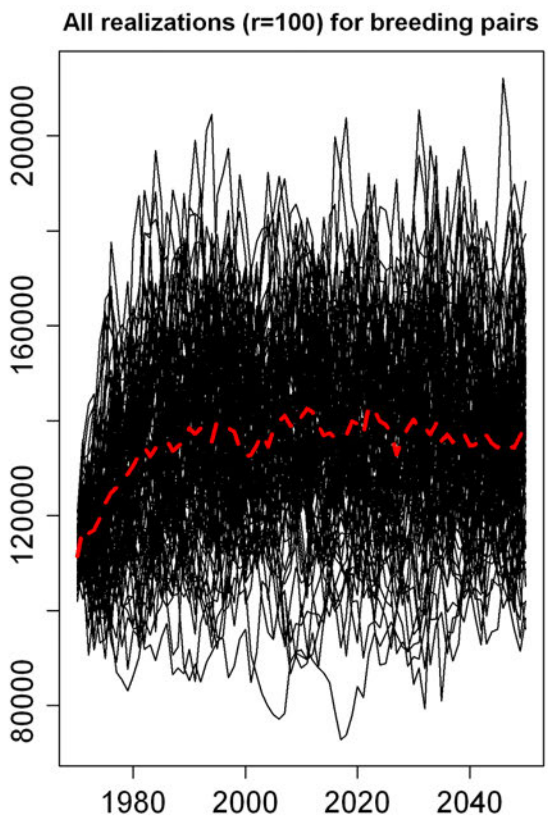


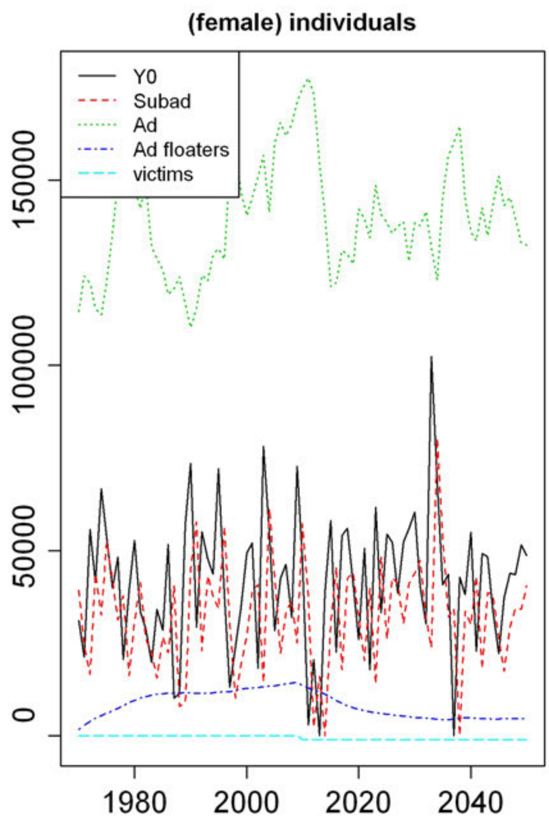
Zero growth – 10% floaters

S0 : 0.785 S0sd : 0.05
Sad: 0.9 Sadsd: 0.05
F0 : 0.355 F0sd : 0.13
B1: 4
Floaters % >=2010: 4

K: 151919
Init population: 50000 75000 85000 850
victims : 0 0 1050 0



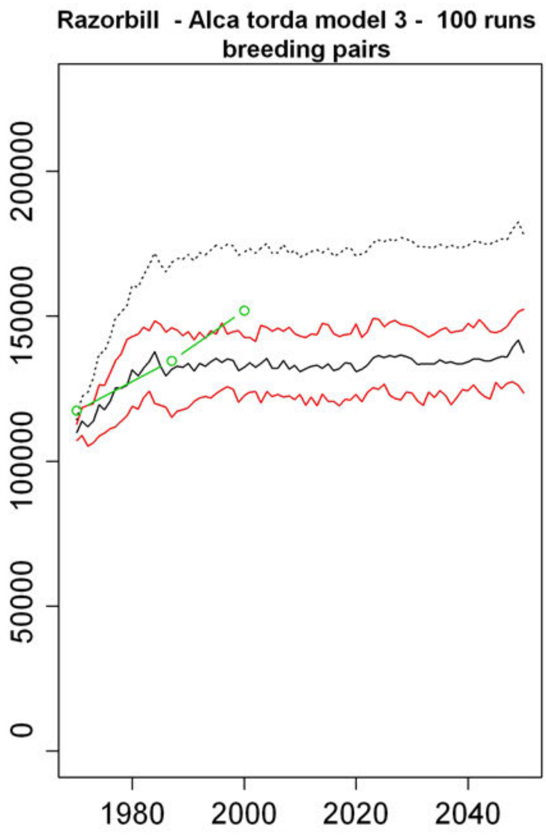


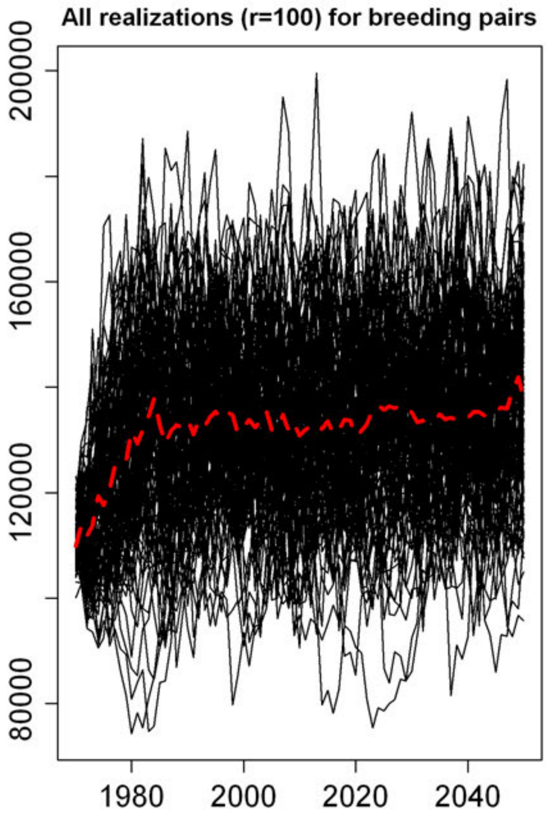


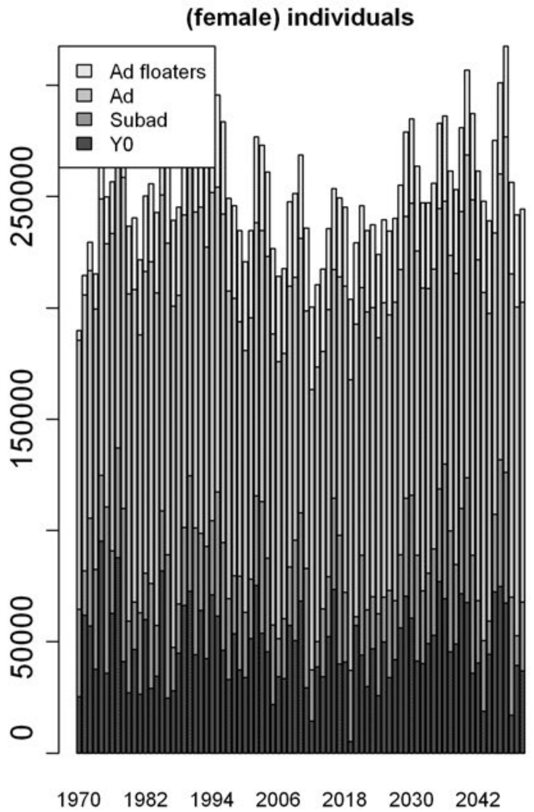
0-model – 30% floaters

S0 : 0.785 S0sd : 0.05
Sad: 0.9 Sadsd: 0.05
F0 : 0.425 F0sd : 0.13
B1: 4
Floaters % >=2010: 30

K: 151919
Init population: 50000 75000 85000 850
victims : 0 0 0 0



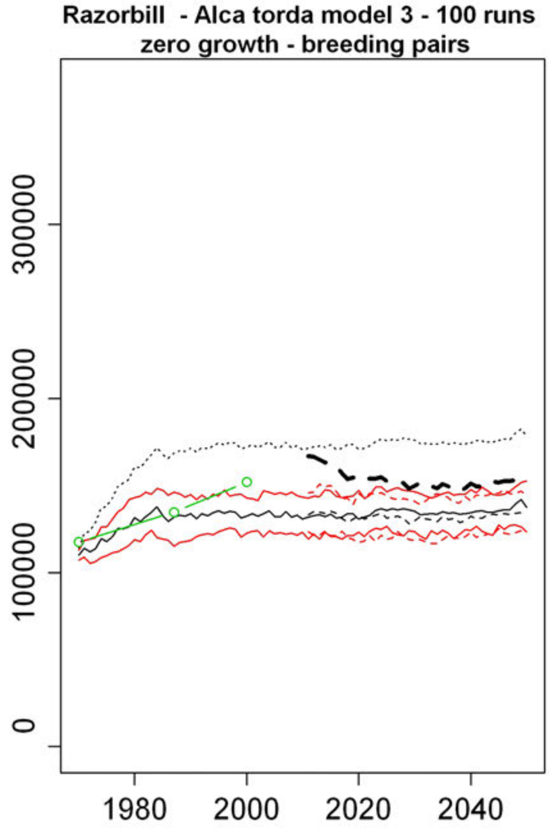


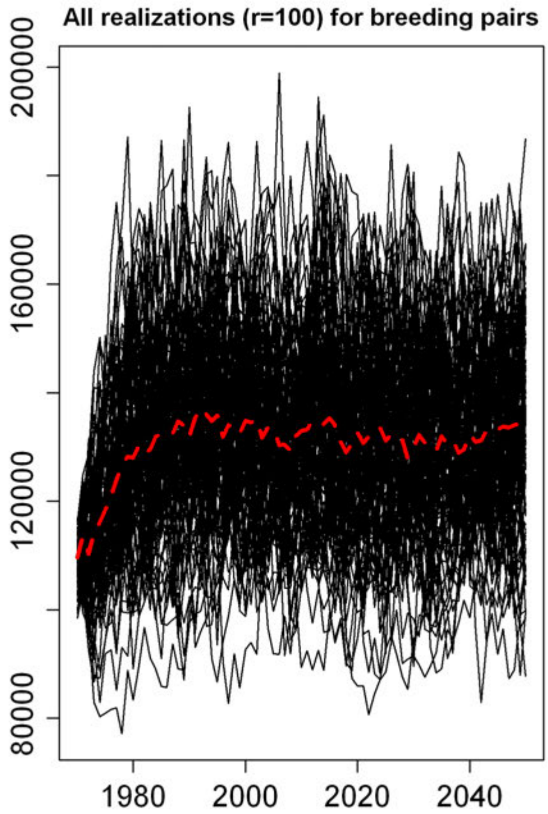


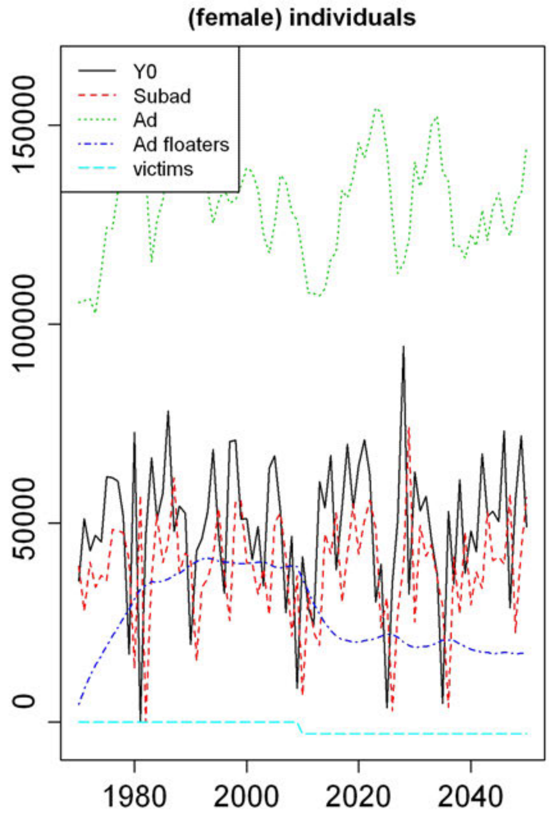
Zero growth – 30% floaters

S0 : 0.785 S0sd : 0.05
Sad: 0.9 Sadsd: 0.05
F0 : 0.425 F0sd : 0.13
B1: 4
Floaters % >=2010: 17

K: 151919
Init population: 50000 75000 85000 850
victims : 0 0 2950 0







Appendix 4 – species specific recruitement parameters, used in PBR

PBR = 0.5 * R_{max} * N_{min} * *rf*   where R = (λ - 1)

$$\lambda_{max} \approx \frac{(s\alpha - s + \alpha + 1) + \sqrt{((s - s\alpha - \alpha -1)^2 - 4s\alpha^2)}}{2\alpha}$$

where s = adult survival and α = age at first breeding

$$T_{op} \approx \frac{1}{(\lambda_{max} -1)}$$

where T_{op} is the mean optimal generation length knowing only age at first reproduction and adult survival

see § 2.4 for further explanation

name	naam	R_{max}	λ_{max}	T_{op}	age at 1st breeding (α)	adult survival (s)	source
bewick's swan	kleine zwaan	0,15	1,15	6,88	4,0	0,850	2
brent goose	rotgans	0,20	1,20	5,00	2,0	0,900	2
common gull	stormmeeuw	0,17	1,17	5,74	3,0	0,860	2
common scoter	zwarte zee-eend	0,21	1,21	4,85	3,0	0,783	2
common tern	visdief	0,16	1,16	6,10	3,0	0,880	5
cormorant	aalscholver	0,16	1,16	6,10	3,0	0,880	2
eider	eider	0,13	1,13	7,92	3,0	0,936	2
fulmar	noordse stormvogel	0,07	1,07	14,34	8,0	0,924	1
gannet	jan van gent	0,12	1,12	8,03	4,0	0,901	1
great black-backed gull	grote mantelrmeeuw	0,15	1,15	6,65	4,0	0,835	4
great skua	grote jager	0,07	1,07	14,75	8,0	0,930	1
guillemot	zeekoet	0,11	1,11	9,21	4,0	0,930	1
herring gull	zilvermeeuw	0,10	1,10	10,25	4,5	0,935	2
kittiwake	drieteenmeeuw	0,10	1,10	10,46	5,0	0,926	1
lesser black-backed gull	kleine mantelmeeuw	0,11	1,11	9,15	4,5	0,913	2
little gull	dwergmeeuw	0,20	1,20	4,90	2,5	0,850	*
little tern	dwergstern	0,15	1,15	6,54	3,0	0,899	2
puffin	papagaaiduiker	0,06	1,06	15,48	6,0	0,963	1
razorbill	alk	0,10	1,10	9,53	5,0	0,905	2
red-throated diver	roodkeelduiker	0,25	1,25	4,07	2,0	0,840	3
sandwich tern	grote stern	0,15	1,15	6,51	3,0	0,898	2
shag	kuifaalscholver	0,17	1,17	6,06	3,0	0,878	1
shelduck	bergeend	0,21	1,21	4,72	2,0	0,886	2

sources

1 Russel 1999
2 BTO bird facts (www.bto.org)
3 Garthe S. & O. Hüppop 2004
4 Wernham *et al.* 2002
5 Becker & Wendeln 1997
* estimate

Appendix 5 – Data sources for population models

The following table lists the sources of data from which figures used in the population models were taken or derived.

Source	Population	Reproduction	Survival
Andrews & Day 1999		x	
Anker-Nilssen & Aarvak 2009		x	
Austin *et al*. 2008	x		
Baker *et al*. 2006	x		
Banks *et al*. 2006	x	x	
Barrett 1988	x	x	
Barret & Bakken 1997	x	x	
Barrett *et al*. 2006	x		
Becker & Wendeln 1997	x		
Beekman & Laubek 1997	x		
Bijlsma *et al*. 2001	x		
BirdLife 2004	x		
Blanco *et al*. 1999		x	
Boere *et al*. 2007		x	
Boudewijn & Dirksen 1991		x	
Boudewijn *et al*. 1991		x	
Boudewijn & Dirksen 1993		x	
Boudewijn & Dirksen 1994		x	
Boudewijn & Dirksen 1995		x	
Boudewijn *et al*. 1996		x	
Boudewijn & Dirksen 1997		x	
Boudewijn *et al*. 1997		x	
Boudewijn & Dirksen 1998		x	
Boudewijn & Dirksen 1999		x	
Boudewijn & Dirksen 2001		x	
Braasch *et al*. 2009		x	
Bukacinski *et al*. 1998		x	
Burger & Schreiber 2002		x	x
Buxton *et al*. 2004			x
Cadiou *et al*. 2009		x	
Cadiou & Monnat 1996		x	
Calbrade *et al*. 2010	x	x	
Calladine 1997a		x	
Calladine 1997b			x
Camphuysen *et al*. 2002			x
Camphuysen & Leopold 2004	x		x
Camphuysen *et al*. 2008		x	
Catry *et al*. 1998		x	
Chabrzyk & Coulson 1976		x	x
Clutton-Brock 1988		x	
Collier *et al*. 2005	x	x	
Coulson 1984		x	
Coulson 1991	x		x
Craik 1999		x	
Craik 2000		x	

Cranswick 2002	x		
Cranswick *et al*. 2006	x		
Croxall 1987		x	x
Croxall 1991		x	x
Delany & Scott 2006	x		
van Dijk 1998		x	
van Dijk *et al*. 2007a		x	
van Dijk *et al*. 2007b		x	
van Dijk *et al*. 2008		x	
van Dijk *et al*. 2009		x	
van DIjk *et al*. 2010		x	
Dirksen *et al*. 1991	x		
Dirksen *et al*. 2005	x		
Drewitt & Langston 2006	x		
Durinck *et al*. 1993	x		
Ebbinge 1992	x		
Ebbinge & Spaans 2002	x		x
van Eerden *et al*. 1995	x		x
Euring 2010			x
Evans 1979		x	x
Ewins *et al*. 1999		x	x
Flint *et al*. 2000			x
Fox *et al*. 2006	x		x
Frederiksen & Bregnballe 2000			x
Frederiksen & Petersen 1999			x
Garthe & Flore 2007	x	x	x
Garthe & Hüppop 2004	x		
Golet *et al*. 2000		x	x
Grantham 2004			x
Green *et al*. 2009		x	
Hall & Kress 2004		x	
Hancock 2000		x	
Harris 1983		x	
Harris *et al*. 1994			x
Harris *et al*. 1997			x
Harris *et al*. 2000a			x
Harris *et al*. 2000b			x
Harris *et al*. 2003	x		
Harris *et al*. 2007			x
Harris & Wanless 1995			x
Harris & Wanless 1996			x
Harris & Wanless 2004			x
Harrison *et al*. 1998		x	
Hatch 1987		x	x
Hatchwell & Birkhead 1991	x	x	x
Hipfner & Bryant 1999		x	
Hipfner 2001		x	
Holt *et al*. 2009	x	x	
Hustings *et al*. 2008	x	x	
Hustings *et al*. 2009	x	x	
Jackson 2005		x	
Johnsgard 1987		x	x
Jones *et al*. 2008		x	
Kirby *et al*. 1995	x		

Kjeil & Arts 1998		x	
van Klinken 1992		x	
Koffijberg *et al*. 1997	x	x	
van der Kolff et al. 2010	x		
Koks 1998		x	
Kondratiev 1991		x	
Krijgsveld *et al*. 2005	x		
Kubetzki *et al*. 2009	x		
Lavers & Jones 2007		x	
Leopold *et al*. 1992		x	
Litvin *et al*. 1999	x	x	
Lloyd 1979		x	x
Lloyd & Perrins 1977		x	x
Lyngs 2006			x
Madsen *et al*. 1999	x		
Markones *et al*. 2009	x		
Mavor *et al*. 2008	x	x	
Meininger *et al*. 2002		x	
Meininger *et al*. 2004		x	
Mendenhall & Milne 1985			x
Migot 1987	x	x	x
Migot 1992	x	x	x
Mineyev 1991	x		
Mitchell *et al*. 2004	x	x	x
Moe *et al*. 2009	x		
Musgrove *et al*. 2007	x	x	
Nelson 1967	x	x	x
Nelson 1978		x	
Newson *et al*. 2005	x	x	
Nisbet & Drury 1972		x	x
O'Brien *et al*. 2008	x		
Offringa & Miere 1999	x		
Ogilvie 1997a	x	x	x
Ogilvie 1997b	x	x	x
Ogilvie & St Joseph 1976	x	x	
Oosterhuis & van Dijk 2002		x	
Oro 1996		x	
Österblom *et al*. 2004			x
Petkov *et al*. 2009	x	x	
Phillips *et al*. 1999	x		
Ratcliffe *et al*. 1998		x	
Ratcliffe *et al*. 2000	x		
Ratcliffe *et al*. 2002	x	x	
Raven & Coulson 1997		x	
Rees 2006	x	x	x
Rees & Beekman 2010	x	x	x
Richards & Morris 1984		x	
Ricklefs 2000			x
van Rijn *et al*. 2004	x	x	
Robert & Ralph 1975		x	
Robinson 2005	x	x	x
van Roomen *et al*. 2004a	x	x	
van Roomen *et al*. 2004b	x	x	
van Roomen *et al*. 2005	x	x	

van Roomen *et al*. 2006	x	x	
van Roomen *et al*. 2007	x	x	
Rothery *et al*. 2002	x	x	x
Russell 1999	x	x	x
Sandvik *et al*. 2005	x		x
Sandvik & Erikstad 2008	x	x	x
Schekkerman & Slaterus 2007	x		
Schreiber & Kissling 2005		x	
Seys *et al*. 1998		x	
Shchadilov *et al*. 2002		x	
Snow & Perrins 1997	x	x	x
Spaans 1998a	x	x	
Spaans 1998b	x	x	
Strucker *et al*. 2005		x	
Strucker *et al*. 2006		x	
Swennen 1983		x	
Swennen 1991	x	x	x
Syroechkovsky *et al*. 2002		x	
Thyen & Becker 2006	x	x	
Vickery & Sutherland 1996	x	x	
Vigfúsdóttir *et al*. 2009		x	
Voskamp & Driessen 2003	x	x	
Votier *et al*. 2009	x	x	
Wahl & Degen 2009	x	x	
Wanless *et al*. 1996	x	x	
Wanless *et al*. 2006	x		x
Ward 2004	x	x	x
Wendeln & Becker 1998		x	
Wernham & Bryant 1988		x	
Wernham *et al*. 2002	x	x	x
Wiggelinkhuizen *et al*. 2006a			x
Wiggelinkhuizen *et al*. 2006b			x
Wooler & Coulson 1977			x
WWT 2010	x	x	
Žydelis *et al*. 2009			x

Appendix 6 Variables for the wind farm OWEZ used in the calculation of the number of collision victims using the Band-model

Number of blades	Maximum chord (m)	Pitch (º)	Rotor diameter (m)	Axis height (m)	Rotation period (sec)	Mean minimum distance between turbines (m)	Number of turbines
3	3.5	0.5	90	70	5	540	36

Appendix 7 Variables for species of seabirds used in the calculation of the number of collision victims using the Band-model. Fluxes of species-groups in italics are based on those for 'all other non-passerines' and relate to the relative abundance of each species-group.

Species-group	Species	Bird length (m) [1]	Wing span (m) [2]	Flight speed (m/s) [3]	Flux Scenario 1 [4]	Flux Scenario 2 [5]	Proportion at rotor height [6]	Macro avoidance [7]	Micro avoidance [8]
geese & swans	Bewick's swan	1.27	1.95	18.5	17809	-	0.72	0.68	0.976
geese & swans	brent goose	0.62	1.17	17.7	17809	-	0.72	0.68	0.976
sea ducks	common scoter	0.43	0.69	16.2	3960	-	0.72	0.71	0.976
divers	red-throated diver	0.94	1.49	17.9	6105	32000	0.72	0.68	0.976
gannets	gannet	0.97	1.92	14.2	46816	542200	0.72	0.64	0.976
cormorants	cormorant	0.94	1.49	15.2	427427	-	0.72	0.18	0.976
skuas	great skua	0.58	1.40	15.6	1276	66600	0.72	0.28	0.976
gulls	kittiwake	0.42	1.05	13.1	554478	348200	0.72	0.18	0.976
gulls	herring gull	0.60	1.48	12.8	816754	973600	0.72	0.18	0.976
gulls	lesser black-backed gull	0.56	1.34	11.9	1101320	1241700	0.72	0.18	0.976
gulls	common gull	0.46	1.08	13.4	560703	240650	0.72	0.18	0.976
gulls	great black-backed gull	0.74	1.66	13.7	275675	177600	0.72	0.18	0.976
gulls	little gull	0.28	0.69	11.5	306436	133650	0.72	0.18	0.976
gulls	black-headed gull	0.39	0.99	11.9	211900	240	0.72	0.18	0.976
terns	Sandwich tern	0.43	0.97	10.9	50820	272500	0.72	0.28	0.976
terns	common tern	0.37	0.80	10.9	5390	112850	0.72	0.28	0.976
passerines	passerines	0.23	0.34	13.8	7692223	-	0.72	0.28	0.976
all other non-passerines		0.46	1.08	13.4	23738	23520	0.72	0.28	0.976
grebes	*great crested grebe*	*0,51*	*0,73*	*13,9*	*1980*	-	*0.72*	*0.68*	*0.976*
tubenoses	*fulmar*	*0,52*	*1,17*	*9,7*	*528*	-	*0.72*	*0.68*	*0.976*
other ducks	*wigeon*	*0,50*	*0,85*	*20,6*	*33550*	-	*0.72*	*0.71*	*0.976*
raptors & owls	*peregrine*	*0,51*	*1,13*	*12,1*	*2145*	-	*0.72*	*0.68*	*0.976*
waders	*redshank*	*0,27*	*0,53*	*9,6*	*22000*	-	*0.72*	*0.68*	*0.976*
alcids	*guillemot/razorbill*	*0,50*	*0,73*	*17,9*	*1100*	*1100*	*0.72*	*0.68*	*0.976*

1 Based on values given in Cramp *et al.* 2000.
2 Based on values given in Cramp *et al.* 2000.
3 Based on values given in Allerstam *et al.* 2007.
4 Calculated as 11 times the flux as recorded at OWEZ. Species and species-group specific fluxes are based on the relative abundance of species and species-groups at OWEZ. Krijgsveld *et al.* 2011.
5 Based on the total flux at OWEZ and the relative abundance of species and species groups in the Dutch North Sea (based on long year aerial surveys, see Arts 2010).
6 Flux measured at OWEZ was based on radar data for an altitude of 25-150m. In the absence of detailed information on species-specific flight altitudes the assumption that species were evenly distributed within the altitude range 25-150m was made. The correction for proportion at rotor height was based on the span of the rotor-swept area (90m) being 72% of this altitude.
7 Based on figures calculated for OWEZ wind farm. Krijgsveld *et al.* 2011.
8 Based on the figure calculated for OWEZ wind farm. Krijgsveld *et al.* 2011.
\- Not calculated.

Appendix 8 Avoidance rates used in the calculation of collision victims, based on macro- and micro-avoidance figures calculated at OWEZ (Krijgsveld et al. 2011). The total avoidance is given to compare to the recommended avoidance figure given by SNH for use with the Band model1.

Species-group	Species	Macro avoidance	Micro avoidance	Total avoidance	Recommended SNH avoidance	SNH species or species groups
geese & swans	Bewick's swan	0.68	0.976	0,992	0,98	Whooper swan
geese & swans	brent goose	0.68	0.976	0,992	0,99	Barnacle goose
sea ducks	common scoter	0.71	0.976	0,993	(0,98)	SHN *default* figure
divers	red-throated diver	0.68	0.976	0,992	0,98	Divers
gannets	gannet	0.64	0.976	0,991	(0,98)	SHN *default* figure
cormorants	cormorant	0.18	0.976	0,980	(0,98)	SHN *default* figure
skuas	great skua	0.28	0.976	0,983	0,98	Skua (all species)
gulls	kittiwake	0.18	0.976	0,980	0,98	Gulls (all species)
gulls	herring gull	0.18	0.976	0,980	0,98	Gulls (all species)
gulls	lesser black-backed gull	0.18	0.976	0,980	0,98	Gulls (all species)
gulls	common gull	0.18	0.976	0,980	0.98	Gulls (all species)
gulls	great black-backed gull	0.18	0.976	0,980	0,98	Gulls (all species)
gulls	little gull	0.18	0.976	0,980	0,98	Gulls (all species)
gulls	black-headed gull	0.18	0.976	0,980	0,98	Gulls (all species)
terns	Sandwich tern	0.28	0.976	0,983	0,98	Tern (all species)
terns	common tern	0.28	0.976	0,983	0,98	Tern (all species)
passerines	passerines	0.28	0.976	0,983	(0,98)	SHN *default* figure
all other non-passerines		0.28	0.976	0,983	(0,98)	SHN *default* figure
grebes	*great crested grebe*	*0.68*	*0.976*	0,992	(0,98)	*SHN default figure*
tubenoses	*fulmar*	*0.68*	*0.976*	0,992	(0,98)	SHN *default* figure
other ducks	*wigeon*	*0.71*	*0.976*	0,993	(0,98)	SHN *default* figure
raptors & owls	*peregrine*	*0.68*	*0.976*	0,992	*0,98*	*Peregrine*
waders	*redshank*	*0.68*	*0.976*	0,992	*0,98*	*Greenshank*
alcids	*guillemot/razorbill*	*0.68*	*0.976*	0,992	(0,98)	SHN *default* figure

1 As given in http://www.snh.gov.uk/docs/B721137.pdf (accessed 17-10-2011).

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