A revised methodology for quantifying ‘Acceptable Level of Impact’ from offshore wind farms on seabird populations

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Contents

Summary

The Acceptable Level of Impact (ALI) methodology defines acceptable limits for the predicted population effect of mortality imposed by offshore wind farms (OWFs) on marine bird populations and was first developed in 2021 as a replacement for the Potential Biological Removal (PBR) method. The ALI is a probabilistic framework and defined as: '*The probability of a population decline of X% or more, 30 years after the impact, cannot exceed Y'.* The methodology is based on a comparison of future population abundance between two scenarios: one unimpacted scenario without impact from OWFs, and one impacted scenario which includes additional mortality resulting from bird collisions with offshore wind turbines and/or habitat loss from avoiding OWFs. In the ALI definition, X represents the threshold value above which the population effects of OWFs are considered undesirable and Y is the threshold value for the probability that a population decline larger than X would still occur.

Following its development and first use, several reviews and an in-depth analysis revealed a number of methodological issues. These were related to the use of a causality measure. This quantified the probability that an unacceptable decline in population abundance was caused by OWFs, instead of by uncertainty or biological variability (*e.g.* environmental stochasticity). Discounting the effect of uncertainty and variation was considered contrary to the precautionary principle. In addition, this may lead to the counterintuitive result that a more strict threshold for the acceptable level of population decline (X) results in a lower probability that such a decline is judged as unacceptable. Thus, only choosing a more strict X threshold would permit larger impacts of OWFs. Although these dependencies can be accounted for through the choice of threshold values, this requires expert judgement and complicates the methodology. It was therefore recently recommended to revise the ALI methodology and abandon the use of a causality measure.

The current report proposes a revision for the ALI methodology based on a comparison between many (100,000) impacted and many unimpacted simulations using stochastic population models. Here, impacted simulations include additional OWF-induced mortality. The adopted approach is similar to the original methodology, but the crucial difference is that this comparison is made per replicate simulation. Essentially, a comparison between scenarios is made while keeping constant all other processes that affect the predicted development of the population. Therefore, variation between the simulations do not contribute directly to the differences between scenarios. This approach makes the ALI methodology more user-friendly and it no longer uses a measure of causality. Furthermore, the X and Y threshold values of the probabilistic ALI framework are no longer interrelated, which obscured the use and applicability of the original ALI. We present an updated set of recommendations for choosing these threshold values.

Samenvatting

De plannen voor grootschalige uitrol van windparken op zee hebben mogelijk negatieve effecten voor vogels die gedurende (een deel van) hun leven de zee gebruiken als leef- en/of doortrekgebied. Sommige (zee)vogelsoorten vermijden windparken en verliezen daardoor een deel van hun leefgebied, ook wel habitatverlies genoemd. Andere soorten vliegen juist wel windparken in, met het risico om geraakt te worden door draaiende windturbines, veelal met dodelijk afloop. Zowel dit soort aanvaringen als habitatverlies leiden tot verhoogde sterfte.

Binnen het project Kader Ecologie en Cumulatie (KEC) wordt een inschatting gemaakt van de mogelijke effecten van windparken op zee door habitatverlies en aanvaringen, voor veelal op zee levende vogelsoorten. Hiervoor is in eerste plaats een inschatting nodig van het aantal slachtoffers per soort als gevolg van aanvaringen en habitatverlies. Vervolgens wordt onderzocht in welke mate het voorspelde aantal slachtoffers een negatieve invloed heeft op de toekomstige ontwikkeling van de vogelpopulatie. Dit wordt gedaan met zogenaamde stochastische populatiemodellen, die rekening houden met onzekerheden over de toekomstige aantalsontwikkeling van populaties, o.a. door omgevingsvariatie (bijvoorbeeld schommelingen in weersomstandigheden of voedselaanbod). De toekomstige populatiegrootte wordt hiermee voorspeld als een range van uitkomsten (een kansverdeling), in plaats van als een enkele waarde. Tot slot wordt beoordeeld of het berekende effect op de populatie acceptabel is of niet. Hiervoor wordt de ALI-methodiek gebruikt, wat staat voor '*Acceptable Level of Impact'*. Deze methodiek helpt beleidsmakers te beoordelen of een vastgestelde norm voor de maximaal toelaatbare impact op vogelpopulaties, die mogelijk hinder ondervinden van (toekomstige) windparken op zee, wordt overschreden. Het KEC wordt uitgevoerd in opdracht van Rijkswaterstaat.

De ALI methodiek is in 2021 ontwikkeld als vervanging van de *Potential Biological Removal* (PBR) methodiek. Sindsdien hebben er verschillende reviews en een gevoeligheidsanalyse plaatsgevonden. Op basis hiervan is een aanbeveling gedaan om de ALI methodiek verder te ontwikkelen. Het huidige rapport beschrijft de aanpassingen die hiervoor zijn gedaan.

De definitie van de ALI-norm is in 2021 vastgesteld door Rijkswaterstaat als "*De kans op een populatieafname van X% of meer, 30 jaar na het begin van de impact, mag niet groter zijn dan Y."* Hierbij verwijst de impact naar het negatieve effect van windparken op zee. De waardes voor X en Y dienen per soort te worden vastgesteld door beleidsmakers. Dit gebeurt in de praktijk door het ministerie van Landbouw, Natuur en Voedselkwaliteit (LNV). De huidige soort-specifieke keuzen voor X en Y zijn onder andere gebaseerd op ecologische toestand van een soort (Staat van Instandhouding, of IUCN rode lijst status) en de mate van onzekerheid over de kwetsbaarheid van een soort voor effecten van windparken op zee. De ALI is gedefinieerd voor een tijdsperiode van 30 jaar, maar dit kan eenvoudig worden aangepast. Voor de aankomende versie van het KEC (KEC5) zal een tijdsperiode van 40 jaar worden gebruikt, omdat windparken nu voor 40 jaar vergund worden.

Oorspronkelijke ALI methodiek

De ALI methodiek zoals deze in 2021 is ontwikkeld vergelijkt de uitkomst van een scenario mét impact met een nulscenario zónder deze impact. Hierbij beschrijft de X drempel van de ALI norm de maximaal toelaatbare afname (in procenten) van een populatie in het geval van een impact gedurende 30 jaar, ten opzichte van een scenario zonder impact. De Y-drempel heeft betrekking op de kans dat een achteruitgang van meer dan X% toch optreedt in het scenario met impact. Echter, in het scenario met impact is een deel van overschrijdingen van de X drempel niet het gevolg van de impact, maar van onzekerheid en omgevingsvariatie. Dit deel is gelijk aan de fractie overschrijdingen die plaatsvindt in het scenario zonder impact. De zogenaamde 'causaliteitskans' geeft aan welk deel van de overschrijdingen het gevolg is van de impact van windparken op zee, en niet van onzekerheid of omgevingsvariatie. Naast een keuze voor X kiezen beleidsmakers per soort een drempelwaarde voor deze causaliteitskans, waaruit de drempelwaarde Y wordt berekend. Door de drempelwaarde Y af te

leiden uit deze 'causaliteitskans' corrigeert de oorspronkelijk ALI methodiek voor overschrijdingen van de X drempel door onzekerheid of omgevingsvariatie.

Echter, deze manier van berekenen zorgt ervoor dat de X en Y drempels niet onafhankelijk van elkaar zijn. Een strengere (lagere) X-waarde zorgt namelijk voor een hoger aantal 'toevallige' overschrijdingen in het nulscenario. Bij dezelfde impact van wind op zee, is hiermee het aandeel overschrijdingen van de X-drempel als gevolg van onzekerheid en variatie relatief groter dan bij een minder strenge drempelwaarde voor X, waardoor de causaliteitskans (en de Y drempelwaarde) minder snel wordt overschreden. Dit gebeurt omdat de strengere X norm met minder zekerheid gedetecteerd kan worden. Met andere woorden, voor dezelfde drempelwaarde voor de causaliteitskans leidt een strengere X tot een hogere Y drempelwaarde. Op eenzelfde manier leidt meer onzekerheid in de populatievoorspelling zonder impact voor een hogere Y waarde. In beide gevallen (strengere X of meer onzekerheid) wordt bij dezelfde causaliteitsdrempel de ALI pas overschreden bij een hogere windpark-gerelateerde sterfte. Dit betekent een minder strenge bescherming tegen negatieve effecten van windparken op zee.

Dit effect is meegenomen in de vaststelling van de drempelwaarden, door bij een strengere (lagere) X-waarde, of meer onzekerheid, ook te kiezen voor een strengere (lagere) causaliteitsdrempel. Echter, het is niet eenduidig hoe de keuze van de causaliteitsdrempelwaarde moet worden aangepast, om voor het effect van onzekerheid en X op de Y drempelwaarde te corrigeren, en in de praktijk gebeurt dit door middel van een expertoordeel. Daarnaast zijn er enkele andere overwegingen die meespelen bij het kiezen van de causaliteitsdrempelwaarden, zoals het risico op aanvaringen met windturbines en de onzekerheid in (natuurlijke variatie) van parameters die de natuurlijke populatieontwikkeling beïnvloeden. De ALI methodiek zou meer transparant worden als het effect van onzekerheid in de methodiek zelf wordt meegenomen, in plaats van in de vaststelling van de drempelwaarden. Ook wordt de ALI door de gebruikers al als ingewikkeld ervaren. Er is daarom besloten om de methodiek te versimpelen, op een manier waarbij er geen sprake meer is van een causaliteitskans (waarbij effecten van windparken op zee vermengd zijn met onzekerheden/variatie), en er een directe keuze voor Y kan worden gemaakt.

Herziening van de ALI methodiek
De herziene ALI zoals beschreven in dit rapport gebruikt geen causaliteitskans. In de herziene methodiek wordt het effect van windparken op zee bekeken per individuele simulatie met een stochastisch populatiemodel. Hierbij worden alle andere processen die van invloed zijn op het voorspelde verloop van de populatie constant gehouden tussen de twee scenario's (mét en zónder impact). Dit heeft als voordeel dat deze processen geen invloed hebben op de vergelijking van de scenario's. Dit betekent dat alle overschrijdingen van de X drempel worden veroorzaakt door de effecten van windparken (in tegenstelling tot variatie/onzekerheid), waardoor de oorzakelijkheid van een overschrijding irrelevant wordt.

Het effect van windparken op zee wordt dus beoordeeld voor een zekere (gekozen) realisatie van omgevingsvariatie en onzekerheid. Dit leidt tot een relatief verschil in populatiegrootte tussen de twee scenario's. Dit wordt herhaald voor veel verschillende realisaties van omgevingsvariatie en onzekerheid. Hierdoor ontstaat een kansverdeling voor het relatieve verschil in populatiegrootte. De kans op een overschrijding van de X drempelwaarde is vervolgens gelijk aan de fractie van alle simulaties waarin dit verschil boven de X drempelwaarde uitkomt. De ALI wordt overschreden wanneer deze kans op een overschrijding van X groter is dan Y. Deze nieuwe manier van berekenen verkleint de spreiding in uitkomsten, waardoor het cruciaal wordt om onzekerheid in de impact van windparken op zee expliciet mee te nemen. Ook wordt het voorzorgsprincipe gehanteerd doordat meer onzekerheid / variatie leidt tot overschrijding van de ALI bij lagere sterfte door windparken op zee.

In de herziene methodiek zijn de waarden voor X en Y onafhankelijk te kiezen. Hiermee wordt het effect van X zoals bedoeld; een strengere X zorgt voor een strengere ALI toets, waarbij de overschrijding van de ALI plaatsvindt bij lagere sterfte door windparken op zee. Daarnaast hoeft er alleen een soort-specifieke keuze gemaakt te worden voor X. Voor Y volstaat één enkele waarde die gelijk is voor alle soorten. Deze waarde moet laag zijn zodat aan het voorzorgsprincipe (eerdere ALI overschrijding bij meer onzekerheid / variatie) wordt voldaan. Bij de keuze voor X dient rekening gehouden te worden met de ecologische toestand van een soort, bij voorkeur op basis van de Nederlandse Staat van Instandhouding (SvI), of anders via internationale criteria zoals de IUCN-status of de Kaderrichtlijn Mariene Strategie of Ospar. Let wel, ook met de herziene methodiek blijft de ALI een methode die het *relatieve* effect van windparken op zee kwantificeert, ten opzichte van een scenario zonder windparken op zee. Idealiter zou naast de huidige ALI methodiek ook getoetst worden ten opzichte van een vaste referentiewaarde, zoals de Gunstige Referentiewaarde uit de SvI, welke voor veel soorten een minimaal vereiste populatieomvang beschrijft. Dit vergt een verdere doorontwikkeling van de methodiek. We beschrijven een aantal situaties waarbij een dergelijke aanvullende toetsing relevant zou zijn, en noemen een aantal punten waarmee rekening gehouden dient te worden in een dergelijke aanpak. Zo vereist een toetsing ten opzichte van absolute referentiekaders een beter inzicht in de verschillende processen die de ontwikkeling van populaties beïnvloeden.

1 Scope and outline

A revision for the Acceptable Level of Impact (ALI) methodology is proposed in the current report. This methodology can be used to define acceptable limits for the predicted population effects of mortality imposed by offshore wind farms (OWFs) on marine bird populations. It was first developed in 2021 as a replacement for the Potential Biological Removal (PBR) method (Potiek, et al., 2022). Following its development and first use, several reviews and an in-depth analysis (Hin, et al., 2023) revealed a number of methodological issues, which are addressed in the current report.

In section 2, we describe the ALI methodology as originally proposed and its current use so far. We also summarize the key findings from the reviews and the in-depth sensitivity analysis by Hin et al. (2023), together with the associated recommendations to revise the methodology (section 2.3). Subsequently, we present in section 3 a revised methodology and show how this methodology overcomes the methodological issues identified in the original framework. Lastly, we provide an updated set of recommendations for choosing threshold values within the revised methodology (sections 4 and 5).

The ALI methodology is designed to help policy makers in defining limits on the effects of OWFs on seabird populations. The current report discusses the details of the methodology, but does not aim to quantify population effects of OWFs. Results of simulations presented in the current report therefore do not reflect true estimates of the effects of OWFs on seabird populations, but are hypothetical impacts used to test the methodology as a proof of concept. A detailed assessment of the effects of OWFs on seabird populations is done within the '*Kader Ecologie en Cumulatie*'[1] (KEC) assessment, which makes use of the ALI methodology. The most recent version of this assessment was KEC 4.0, see Rijksoverheid (2021), Potiek, et al. (2021) and Soudijn, et al. (2022).

[1] 'Framework Ecology and Cumulation'

2 Original ALI methodology

2.1 Definition of the ALI

The original ALI was defined as: '*The probability of a population decline of X% or more, 30 years after the impact, cannot exceed Y'.* This definition was formulated by the Dutch government (RWS/LNV), while the underlying ALI methodology was developed by Potiek et al. (2022). This methodology is based on a comparison of future population abundance between two scenarios: one unimpacted scenario without impact from OWFs, and one impacted scenario which includes additional mortality resulting from collisions with offshore wind turbines and/or habitat loss from avoiding OWFs. Furthermore, the ALI is a probabilistic framework, formulated in terms of a probability of a certain population decline. In this way, the ALI can account for (biological) variation and uncertainty associated with predictions of future population abundance. Accordingly, the ALI is used in combination with stochastic matrix population models (Box 1) that generate distributions of future population abundance for each scenario (with and without impact from OWFs). Derivation and details of the specific population models used within the ALI framework are given by Van Kooten, et al. (2019) and Hin, et al. (2023).

In the original ALI definition, X represents the threshold value above which the population effects of OWFs are considered undesirable. The value of X is measured as the relative difference (expressed as percentage) between the impacted population abundance after 30 years and the *median* population abundance in the unimpacted scenario after 30 years. For example, an X value of 25 means that the impacted population abundance can at maximum be 25% lower than the median population abundance in the unimpacted scenario. The X threshold value (acceptable decline over the time frame of 30 years) is derived from a policy decision for X', which is the acceptable decline over three times the generation time of the relevant species or ten years, whichever is longer (Potiek et al., 2022).

In both the unimpacted and the impacted scenario, some replicate simulations will exceed the X threshold (Box 2). Under the unimpacted scenario this is solely the effect of uncertainty or variability and under the impacted scenario this is the joined effect of uncertainty, variability and impact from OWFs. In the original ALI methodology, the relative contribution of the OWF-impact on the probability of violating the X threshold is referred to as the causality level P_C. Hence, P_C quantifies the probability that a difference in population abundance larger than X can causally be attributed to the impact from OWFs. The threshold value Y is derived from a threshold value for P_C, termed P_T, which is decided on by policymakers (see Box 2). Note that Y is a threshold value for the total probability of violation in the impacted scenario, both due to impact as well as due to uncertainty/variation. The ALI is said to be violated if the obtained value for P_C exceeds the set threshold value P_T, but see Box 2 for an alternative method to assess whether the ALI is violated.

Box 1 (next page)*: Stochastic matrix population models and methods of parameter sampling*

Matrix population models describe transitions of individuals between different life stages (*e.g.* yearlings/juveniles, immatures, adults) based on parameters that represent life history processes (*e.g.* the probability of breeding, breeding success, survival) (Caswell, 2001). The core of a matrix population model is the projection matrix, which describes the transitions of individuals between life stages, the creation of new individuals (reproduction), and the probability of remaining in the same life stage. An example is the projection matrix for the northern gannet, as derived by Van Kooten et al. (2019), which contains six life stages:

$$\mathbf{A} = \begin{pmatrix} 0 & 0 & 0 & 0 & 0 & 0 \\ S_0 & 0 & 0 & 0 & 0 & S_0 \frac{F_A}{2}(1-P_F) \\ 0 & S_1 & 0 & 0 & 0 & 0 \\ 0 & 0 & S_2 & 0 & 0 & 0 \\ 0 & 0 & 0 & S_3 & 0 & 0 \\ 0 & 0 & 0 & 0 & S_A & S_A \end{pmatrix} \qquad \text{eq. 1}$$

In this matrix, the entry of row *i* and column *j* describes the fraction of individuals in life stage *i* that were generated by individuals in life stage *j* the previous time step (usually one year). Parameters S represent survival, F_A is breeding success and P_F is the probability of breeding. To project forward the population a single time step, the matrix $\mathbf{A}$ is left-multiplied by the population state vector n_t that holds the number of individuals per life stage:

$$n_{t+1} = \mathbf{A}_t n_t, \qquad \text{eq. 2}$$

To account for uncertainty about the future development of populations, parameters of the population models can be generated from statistical distributions (see Hin, et al. 2023 for examples). These distributions are derived from literature data on (variation in) reproductive success and survival of seabirds (Van Kooten, et al., 2019). This essentially creates a stochastic population model. A Monte Carlo approach can then be used to generate very many population projections, from which a distribution of future population abundance can be derived (Figure 1).

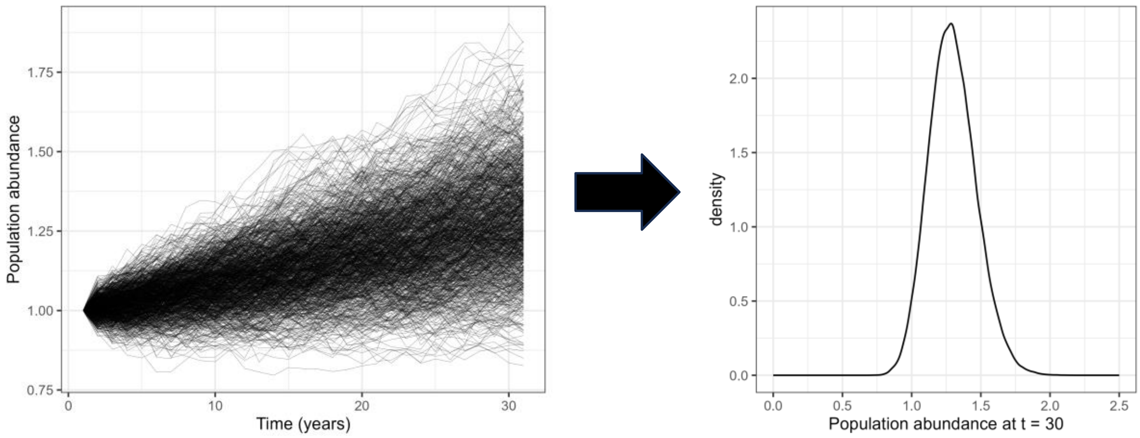
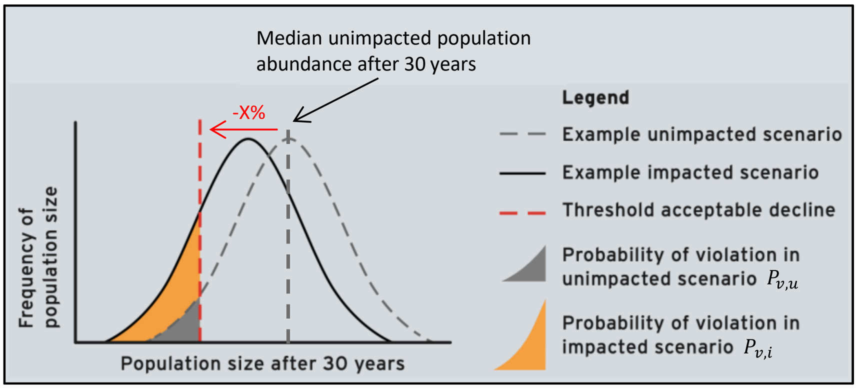


Figure 1*: left: spaghetti plot with one thousand replicate simulations using the stochastic matrix population model for the northern gannet (Van Kooten, et al. 2019) with annual parameter sampling starting at an initial abundance at t = 0 of 1. Right: the resulting distribution of population abundance at t = 30.*

In the original ALI methodology, many replicate population trajectories were simulated, each with different parameters. However, parameters were only resampled between different simulations and were held constant across different years within a single simulation. This way of sampling was termed the "initial" parameter sampling method by Hin et al. (2023) and leads to a rapid increase in the variation of population abundance through time. An alternative method is to sample parameters each year, as was done in the figure above and termed the "annual" method of parameter sampling. This creates considerably less variation in projected population abundance if the same probability distributions are used for the parameters. Although normally different distributions should be used as initial and annual sampling address different types of uncertainty/variation in the parameters, in practice the variation in projected abundance will still often be smaller with annual sampling.

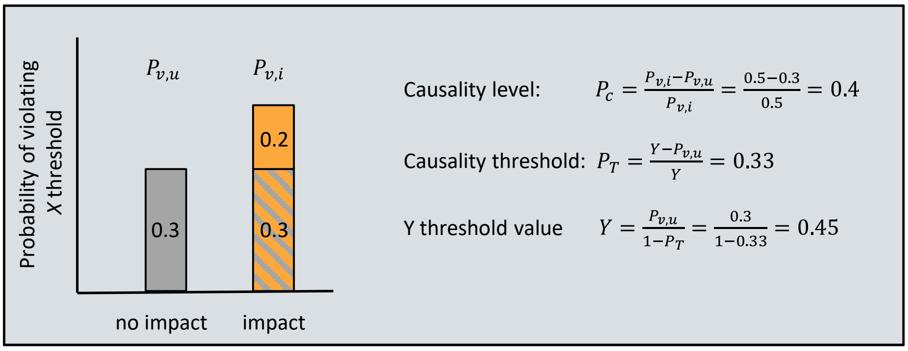
***Box 2**: The original ALI methodology*

In the original ALI methodology as defined by Potiek et al. (2022), stochastic population models (Box 1) are used to generate distributions of future population abundance for two scenarios: one without impact from OWFs (unimpacted; gray dashed distribution) and one scenario with impact from OWFs (impacted scenario; solid black distribution). The median population abundance in the unimpacted scenario (vertical gray dashed line) is used to calculate the population threshold abundance N_T (red dashed line) corresponding to an X decline in population abundance. The probability of violating this threshold value is evaluated for both scenarios as the fraction of many (100,000) simulations with stochastic population models that end up below N_T. These probabilities are termed $P_{v,u}$ and $P_{v,i}$, for, respectively, the unimpacted and impacted scenario.



Subsequently, the probability that a violation results from OWF impact (probability of causality: P_C) is calculated as the difference between $P_{v,i}$ and $P_{v,u}$, relative to the total probability of violation $P_{v,i}$ (see numerical example below). The resulting value of P_C can then be compared directly with the threshold causality level P_T that was chosen by policy (0.33 in the example below). The causality threshold P_T is a threshold for the probability that a decline larger than X is due to OWF impact. The ALI is violated in case $P_C > P_T$.

Potiek et al. (2022) originally presented a slightly different method to evaluate whether the ALI was violated, but with identical outcome. This method used the threshold P_T to calculate the Y threshold value, the latter being a threshold value for the probability $P_{v,i}$ that a negative impact occurs, irrespective of whether it is due to impact or not. The equation that relates P_T to Y can be derived by substituting $P_{v,i}$ = Y and P_C = P_T in the equation for P_C and rearranging. This reveals that Y is directly proportional to $P_{v,u}$: the probability of an X threshold violation in the unimpacted scenario. This probability in turn depends on the value for X and the amount of variation and uncertainty associated with the population projections.



2.2 Current use of the ALI

The original methodology was reviewed in November 2021 – January 2022 (see section 2.3). The general conclusion from the reviews was that the approach and assumptions of the proposed ALI methodology were a considerable improvement in relation to existing impact-evaluation frameworks (ORNIS, PBR). However, the reviews also raised concerns about, among others, the definition of the causality level P_C and the interpretation and interrelationship between threshold values X and Y. Several comments raised in the reviews and in feedback from government officials were addressed, resulting in an updated report for the ALI methodology (Potiek, et al., 2022).

Although the development of ALI methodology was still in progress, it was considered a better alternative than the PBR method and was therefore applied for the calculations of the "*Aanvullend Ontwerp bij het Programma Noordzee 2022-2027*" (KEC 4.0) (Potiek, et al., 2021; Rijksoverheid, 2021; Soudijn, et al., 2022) and in several Environmental Impact Assessments (EIA) for individual wind farms. In order to apply the methodology, the Dutch Ministry of Agriculture, Nature and Food Quality (LNV) defined preliminary species-specific thresholds values based on the IUCN red list status. In the meantime, Sovon was asked by LNV to give an ecological advice on these thresholds (Sovon, 2022b), which led to an update of the thresholds in 2023 (LNV, 2023).

2.3 Review and sensitivity analysis

The main points of the review related to the original ALI methodology included:

1) *Derivation of P_C*: Concerns were raised about the use and definition of the causality level P_C. It was questioned whether P_C was defined correctly and how it relates to other statistical measures, such as the risk ratio, and that the use of P_c introduced a dependence of Y on the choice of X.
2) *Method of parameter sampling:* The review questioned the original method of sampling parameters only at the start of each simulation, in contrast to sampling parameters each year. Specifically, it was questioned how this method influences the amount of uncertainty concerning future population trajectories and the consequences for the probability of ALI violation.
3) *Definition of generation time:* The review disagreed with the adopted definition of generation time, which differed from the definition used by the IUCN, although the related IUCN criteria was proposed as a basis to define X. Accordingly, the ALI methodology should use the same definition of generation time as was used in the IUCN criterium, which is the *mean age at which a cohort of individuals produce offspring* (Appendix A).
4) *Threshold values X:* The review argued that the allowable decline of X should be calculated relative to a fixed reference value representing some desired minimum population size, instead of relating it to an unimpacted scenario. As argued in the review, the choice to compare two scenarios does not guarantee that the conservation status or even survival of a species will be maintained. This would mean that, for species with an unfavourable conservation status, X should be equal to 0%, and not 15% as proposed by Potiek et al. (2022). The review also proposed that if a '*Favourable Reference Value'* (FRV) has been defined for a species (by the Dutch government), this value should take priority over the IUCN 30% threshold as a basis for choosing X.
5) *Threshold value Y:* In the original methodology, the threshold value for Y is derived from the causality threshold value P_T (Box 2), and P_T could be chosen based on the amount of available knowledge, such as uncertainty in model parameters, as well as on the importance of a specific species in context of the Dutch nature policy, conservation status, population trend and potential for compensation (Potiek, et al. 2022). The reviewers argued that 1) it is unclear to what extent a stricter choice of P_T will compensate the effect of parameter uncertainty on Y, 2) the choice of P_T remains open to interpretation and discussion, 3) the approach is complicated, 4) P_T and Y will vary between species while a uniform Y criterion should be preferred and 5) the definition of Y based on P_T is contrary to the precautionary

principle, because more uncertainty will result in larger OWF impacts to be judged as acceptable.

Several of these points were addressed in a follow-up project that started in 2023. In this project a framework was developed that addressed the possibility of using the FRV as a basis for choosing X (Potiek, et al., 2023). This project also included a discussion of the legal tenability and an in-depth sensitivity analysis of the ALI methodology (Hin, et al., 2023). The most important conclusions were:

6) *Legal tenability.* A discussion with stakeholders and legal advisors took place on June 29th 2023 to discuss the compatibility of the method with current legal and policy frameworks. The conclusion of the meeting was that the ALI methodology was an improvement from the earlier methods (PBR and ORNIS-criterion). There were some concerns about the reference value being relative to the unimpacted scenario, instead of an absolute value. Nonetheless, continued use of the current methodology was approved within the Wozep-KEC projects, as well as within the cumulative environmental impact assessments required for the commissioning of future wind farms.
7) *Definition of P_C.* The causality threshold P_C was derived from theory of conditional probabilities (section 3.1 in Hin, et al. 2023). This derivation showed that P_C in itself was correctly defined, and related to other commonly used risk measures, such as the 'attributable fraction among the exposed' and the 'relative risk ratio.'
8) *Sensitivity analysis: effect of uncertainty.* An analysis of the effect of uncertainty on the outcome of the ALI was performed by comparing 1) 'initial' versus 'annual' parameter sampling (see Box 1) and 2) changing the standard deviation of model parameters. Initial sampling led to considerably more variation in predicted population abundance than annual sampling. This increased the probability of an X threshold violation in the unimpacted scenario ($P_{v,u}$), which decreased the causality probability P_C (Box 2). Increasing the standard deviations of model parameters resulted in a similar effect. Taken together, more uncertainty reflected by increased variation in predicted population abundance might decrease the probability of an ALI violation (P_C). This was considered an undesirable property of the methodology and contrary to the precautionary principle (Hin, et al., 2023). With a precautionary approach, increased uncertainty about the future development of populations should increase, instead of decrease, the probability of an ALI violation.
9) *Sensitivity analysis: X threshold.* A more strict (lower) X threshold resulted in an increased probability of an X threshold violation in the unimpacted scenario ($P_{v,u}$). With the Y threshold being directly proportional to $P_{v,u}$ (Box 2), a more strict X threshold leads to a less strict (higher) Y threshold and, as a result, a lower probability of an ALI violation. This was considered an undesirable property of the methodology, as a more strict X threshold should increase the likelihood of an ALI violation (Hin, et al., 2023).

Both 8) and 9) stem from the use of the causality measure P_C, which attempts to correct for false positive outcomes (threshold violation without impact) that are caused by variation (*e.g.* environmental stochasticity) or uncertainty (Hin, et al., 2023). Although the relationship between X and Y could potentially be accounted for by adjusting the causality threshold value P_T (as initially proposed by Potiek et al. 2022), this heuristic approach would further complicate an already complex methodology. Furthermore, it would remain arbitrary how such an adjustment should be made in practice (see point 5 above). Therefore, Hin, et al. (2023) recommended to revise the ALI methodology according to the following points:

10) Avoid the use of a causality measure (such as P_C) that attempts to correct for uncertainty. Instead, in the revised methodology, increased uncertainty should lead to a more cautious approach, in the sense that the maximum allowable impact from OWF decreases with increasing uncertainty.
11) Simplify the methodology
12) Use annual parameter sampling instead of initial parameter sampling, as this better reflects the year-to-year variation in seabird vital rates (breeding success and survival).

In the next section we present an alternative methodology for the ALI that adheres to these three recommendations. Subsequently, we discuss the implications of this new method for the interpretation of X and Y and present guidelines on how to choose these threshold values (section 4).

3 The revised ALI methodology

The revised ALI methodology follows the recommendation set out by Hin, et al. (2023), while also taking into account the comments and suggestions from several reviewers. The most important recommendation was to abandon the use of a causality measure (P_C), as this led to several undesirable properties in the original framework (Hin, et al. 2023 and section 2.3). In addition, Hin et al. (2023) advised to sample population parameters each time step (year), instead of only at the beginning of each simulation ('initial' method).

The proposed revision of the ALI does not affect the definition of the ALI as given by Potiek et al. (2022). The revised ALI is also based on a comparison between impacted and unimpacted scenarios and formulated in a probabilistic manner. The essential difference, however, is that this comparison is made for each replicate simulation individually. This is achieved by generating a single sequence of projection matrices (Box 1) with yearly varying population parameters, through 'annual' parameter sampling. This form of parameter variation represents the effect of environmental stochasticity on individual vital rates. From this sequence, an unimpacted simulation is generated, leading to a prediction of unimpacted population abundance thirty years later, termed N_{30}^{U}. Subsequently, the sequence of projection matrices are modified to include effects of OWFs on survival. These modified projection matrices are then used to calculate the impacted simulation, leading to a prediction of impacted population abundance thirty years later, termed N_{30}^{I}. From these two related, but different simulations, the population impact is calculated as the relative difference (in percentage) between N_{30}^{U} and N_{30}^{I}, termed Δ_{30} (Box 3). In this manner, apart from the OWF-effect on survival, the impacted and unimpacted scenarios use otherwise identical sequence of population parameters. Essentially, this approach evaluates the OWF effect on the population *for the same realization of environmental stochasticity*. This approach lowers the overall variability produced by the methodology, while maintaining natural variability and inherent uncertainty.

This procedure is repeated for many (100,000) possible realizations of environmental stochasticity, resulting in a distribution of the relative difference in population abundance at t = 30 (Δ_{30} in percentage). This distribution can then be directly compared against the X threshold value (Box 3). The probability of exceeding the X threshold is derived as the fraction of Δ_{30}–values larger than X, i.e. $P_{v,i} = P(\Delta_{30} > X)$. In case $P_{v,i} > Y$, the ALI is violated. Hence, the threshold values X and Y are uncorrelated and should be chosen independently (see section 4). In addition, a more strict (smaller) X threshold will lead to a larger value for $P_{v,i}$ and increase the probability of an ALI violation (*i.e.* decrease the level of OWF-induced mortality that leads to violation of the ALI).

Note that in this procedure there is no possibility of exceeding the X threshold without impact from OWFs: there is no $P_{v,u}$. This is even true when the population is already declining in the unimpacted scenario, as X refers to the relative difference between the unimpacted and impacted scenarios. Likewise, there is no causality measure, such as P_C. Instead, all violations of the X threshold are caused by the impact, as violations arising from uncertainty or variability are impossible. The new ALI also does not use a statistical measure as a benchmark to calculate the probability of violating the X threshold. In the original framework, the median unimpacted population abundance was used for this (Box 2), but this choice is rather arbitrary. For example, Green, et al. (2016) suggest to compare impacts of OWFs against the mean population abundance of the unimpacted scenario. The revised approach does not require a choice about which statistical measure to use as benchmark to test against.

Box 3: *The revised ALI methodology*

The revised ALI methodology is also based on a comparison between an impacted and unimpacted scenario. However, in the revised method the comparison is made *for each* replicate simulation instead of by comparing the final outcome of the two scenarios. The idea behind this approach is that the OWF effect on the population is evaluated *for the same realization of environmental stochasticity*. In population models, environmental stochasticity is simulated as the annual variation in population parameters. The figure below shows the development in population abundance of a single replicate simulation, with and without OWF impact. The pattern in the population trajectory is the same in both scenarios and the OWF impact decreases the overall trend.

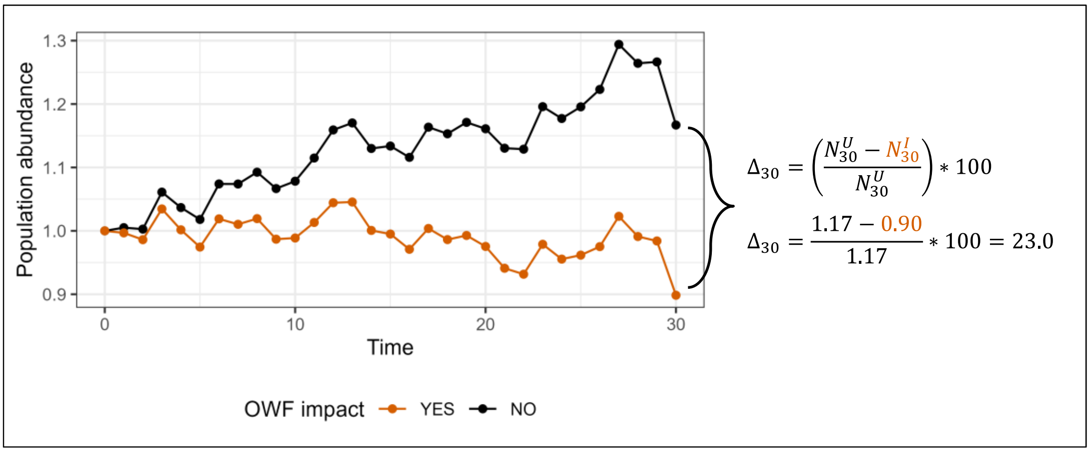


Figure 2: *Simulations of population abundance with population parameters sampled each year. The trajectory with OWF impact is derived by adding OWF-induced mortality to the sequence of annual projection matrices used to generate the unimpacted scenario. Simulations were performed using the northern gannet population model (Van Kooten, et al. 2019; Hin, et al. 2023).*

The population impact of OWFs is now evaluated as the relative difference in population abundance between the two scenarios, thirty years after the onset of the impact, termed Δ_{30}. This procedure is repeated many (100,000) times, which results in a distribution of Δ_{30} (Figure 3). This distribution can be compared directly against a chosen X threshold value. The probability of a relative decrease exceeding X, defined as $P_{v,i} = P(\Delta_{30} > X)$, can then be evaluated directly against an Y threshold value. The ALI is violated if $P_{v,i} > Y$.

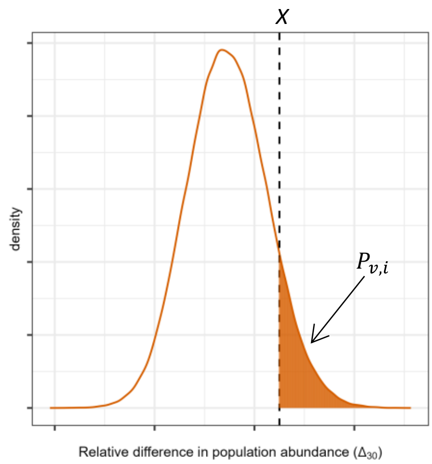


Figure 3: *Hypothetical distribution of the relative difference due to OWF impact thirty years after the start of the impact, with a hypothetical X value. The orange area is the probability $P_{v,i}$ of an X threshold violation. The ALI is violated if $P_{v,i} > Y$.*

Another important difference with the original ALI is that a comparison between scenarios is made, while keeping constant all other processes that affect the predicted development of the population. Apart from environmental stochasticity, such processes could also include uncertainty in parameter estimates, or other processes that generate variability or uncertainty in the projection of the unimpacted population. Any assessment that utilizes the ALI should account for the various processes that generate biological variation and uncertainty in a consistent manner. To achieve this, we recommend an approach in which processes that affect the prediction of unimpacted population abundance are kept equal between scenarios, while processes that are affected by OWFs should be applied to the impacted scenario only. It cannot be specified beforehand which forms and types of processes that generate uncertainty should be included, as this depends on the details of a particular assessment, and the availability of (species-specific) data to parameterize these processes.

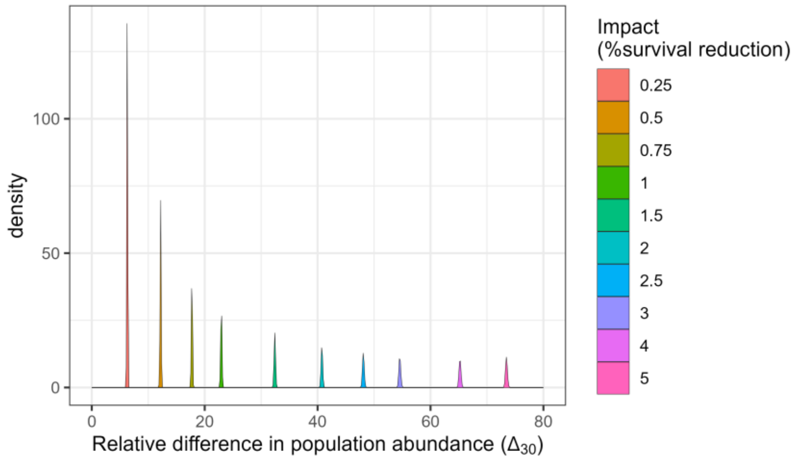


Figure 4: *The distribution of Δ_{30} (relative difference in percentage in population abundance between impacted and unimpacted scenarios, thirty years after onset of impact) for various levels of a fixed OWF-impact (% of survival reduction) applied to birds of 2 years of age and older. Results were obtained using the northern gannet population model with 'annual' parameter sampling and 100,000 replicate simulation per impact level (Hin, et al., 2023; Van Kooten, et al., 2019).*

The framework uses 'annual' sampling of parameters by default. This, however, is more a property of the population models than it is of the ALI framework. The new framework can still be applied using initial parameter sampling, and which sampling method is preferred also depends on the available information about yearly variation in vital rates (e.g. breeding success and survival), which may differ per species and colony. For instance, if for a certain parameter only an estimate for the mean (with standard error) is available instead of annual estimates, one may use initial sampling and use the standard error as the standard deviation of the parameter's probability distribution.

If uncertainty or variation in the OWF-impact is ignored, the revised methodology results in a very narrow distribution of Δ_{30} (Figure 4 and Figure 5). This happens because most of the variation created by environmental stochasticity cancels out, as this variation is applied in the same manner in both scenarios. This means that while environmental stochasticity affects the trajectory of the population, it has only little impact on the relative population effect of OWFs. Indeed, applying a randomly varying OWF-effect, instead of a fixed one, results in a much wider distribution of Δ_{30} (Figure 5). Thus, taking account of the variation and uncertainty associated with the size of the OWF impact is crucial in any assessment that utilizes this ALI methodology and not doing so will create the false impression that OWF effects can be predicted with seemingly high accuracy.

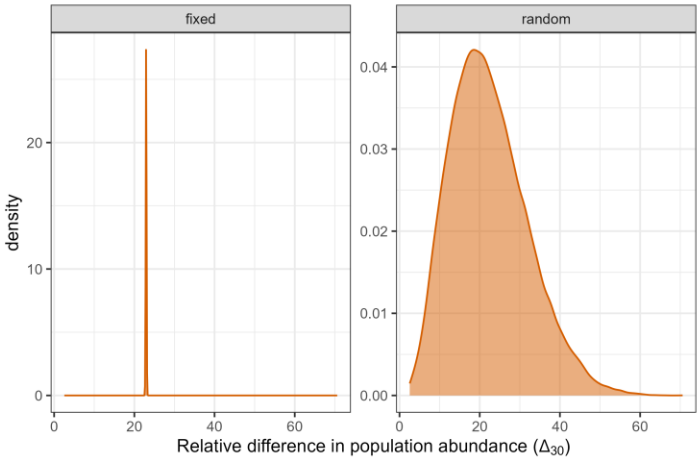


Figure 5: *Distribution of Δ_{30} (relative difference in percentage in population abundance between impacted and unimpacted scenarios, thirty years after onset of impact) for either a fixed OWF-impact (left: 1% decrease in survival) or a random OWF-impact (right), modelled as a decrease in survival generated from a beta distribution with mean 1% and standard deviation 0.005.*

Precautionary principle and implications for Y
For the revised ALI to adhere to the precautionary principle, Y should be small enough. With a precautionary approach, increased uncertainty about either the OWF-impact or the future development of populations should increase the probability of an ALI violation. The amount of uncertainty associated with the development of populations is represented by the standard deviation (SD) of population parameters, which scale the amount of year-to-year variability in individual vital rates (Hin, et al., 2023). A higher SD reflects more variability and leads to a wider distribution of Δ_{30} (Figure 6). For high values of Y, it might occur that increased variability or uncertainty (larger SD) does not lead to an ALI violation, while the ALI would be violated at low variability / uncertainty (low SD; Table 1). To adhere to the precautionary principle therefore requires that Y should be well below 0.5, because of the symmetric nature of the distribution of Δ_{30}. We expect this to rarely be controversial in practice, as it is unlikely that a probability of exceeding the X threshold larger than 50% would be considered acceptable (by society).

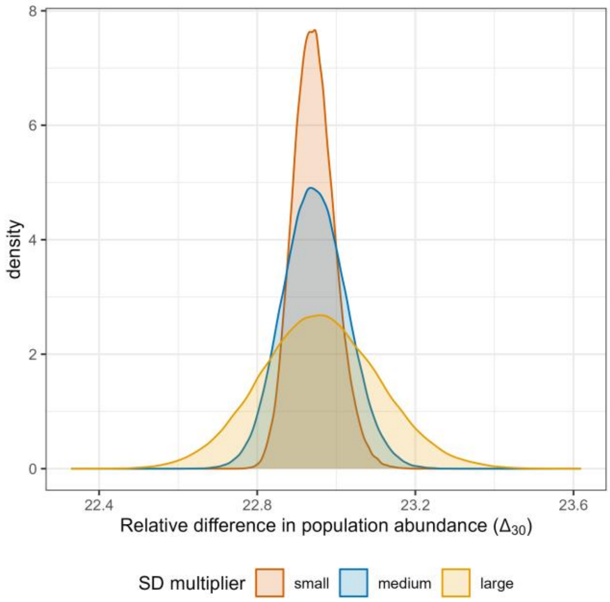


Figure 6: *Distribution of Δ_{30} (in percentage) for different levels of year-to-year variation in vital rates, modelled by applying a multiplier to the standard deviations of all population parameters (SD multiplier). Levels of the SD multiplier are 0.5 (small), 1 (medium, default) and 2 (large). Distributions are derived from the northern gannet population model using a fixed OWF-impact of 1% survival reduction applied to birds of age ≥ 2 (see Hin, et al. 2023 for details).*

Table 1: *The outcome of the ALI (violation occurs if $P_{v,i} > Y$) for the distributions in Figure 6 for two hypothetical values of Y (0.7 and 0.05) and a hypothetical value of X = 22.9%. $P_{v,i}$ is calculated as the area under the curve in Figure 6 above 22.9. For Y = 0.7 increased variability / uncertainty (SD = 2.0) changes the outcome of the ALI from 'violation' (TRUE) to 'no-violation' (FALSE). Adopting Y << 0.5 prevents this.*

X	*SD multiplier*	$P_{v,i}$	*Y = 0.7*	*Y = 0.05*
22.9	Small (0.5)	0.797	TRUE	TRUE
22.9	Medium (1.0)	0.714	TRUE	TRUE
22.9	Large (2.0)	0.648	FALSE	TRUE

Effect of unimpacted population growth rate

Part of the variation in the relative population impact (Δ_{30}) is explained by the mean population growth rate of the unimpacted scenario ($\bar{\lambda}^{U}$). This population growth rate is calculated as the geometric mean annual increase in population abundance over thirty years: $\bar{\lambda}^{U} = \left(N_{30}^{U}\right)^{1/30}$, given that initial population abundance was set to 1 ($N_0^U = 1$). As shown in Figure 7, the relative population impact after thirty years (Δ_{30}) decreases with increasing values of $\bar{\lambda}^{U}$. This implies that populations with a higher growth rate can sustain a higher mortality impact from OWFs before the ALI is violated. Overestimating the true growth rate of the population would therefore impose a risk of overestimating the number of casualties that would lead to violation of the ALI threshold. Consequently, it is key that the population models associated with the ALI do not overestimate the true population growth rate and any assessment that utilizes the ALI should check whether this is indeed the case.

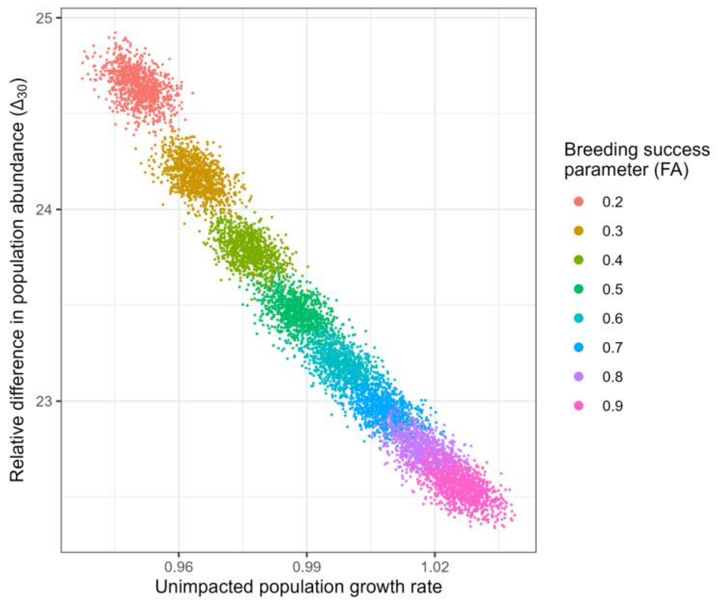


***Figure 7**: The relative population impact of OWFs decreases with the population growth rate in the unimpacted scenario. The latter was quantified as $(N_{30}^{U})^{1/30}$, given that $N_0^U = 1$. The relative population impact was derived from the northern gannet population model using a fixed OWF-impact of 1% decrease in baseline survival applied to birds with age ≥ 2. The mean breeding success (F_A) was changed to simulate a larger range of unimpacted population growth rates. The standard deviation of breeding success was kept constant.*

Time frame of the ALI

The revised ALI is presented here for a time frame of thirty years, but this can easily be changed to other time frames. Here, thirty years is used to connect to the original ALI methodology, but the definite time period might change depending on the assessment that utilizes the ALI methodology. However, it should be realized that the ALI is based on density-independent population models that exhibit exponential growth or decline. The assumption of exponential growth might only be a reasonable approximation under short time frames, and will be of increasing influence as the time frame of the ALI gets extended. This should be accounted for when choosing threshold values for X, as discussed in the next section.

4 Thresholds for X and Y: Guidelines

The revision of the ALI methodology also changes the considerations on how to choose the threshold values X and Y. Note that setting these thresholds is outside the scope of our assignment, and threshold values should be set by policymakers. In this section we present several considerations that should be taken into account when choosing values for X and Y for the revised ALI methodology.

4.1 Choosing X

The interpretation of X does not change with the proposed revision of the ALI. Similar to the original ALI methodology, the X threshold value in the revised methodology relates to the relative difference between unimpacted and impacted population abundance 30 years later. As discussed in section 3, this time frame can be changed depending on the requirements of the specific assessment that utilizes the ALI.

4.1.1 Time frame of X

In the original ALI methodology (section 2.1), the threshold value X is associated with a time period of thirty years, and X was derived from a policy decision for X’, which is the acceptable decline over a reference time (T_{ref}) of three generations or ten years, whichever is longest (Potiek, et al. 2022). In this way, the decision about what is an acceptable decline relates to a time scale relevant to the subject species, instead of the time frame used within assessments that utilize the ALI. The equation to calculate X (30 years) from X’ (T_{ref} years) is:

$$X = 1 - (1 - X')^{\frac{30}{T_{ref}}}$$

The decision to relate X to the generation time of a species is maintained for the revised version of the ALI, as generation time is indicative of the recovery potential and vulnerability of a species to anthropogenic disturbances, with shorter generation times being associated with higher recovery potential and increased resilience to disturbance. This means that the <u>reference time associated with X’ should be three generations or ten years, whichever is longer</u>. This choice of reference time is based on the widely accepted reference times for species trends in classifying the species’ vulnerability as used by the International Union for Conservation of Nature and Natural Resource (IUCN, 2012).

Most seabird species included in the KEC assessment have a generation time exceeding 10 years. The X’ threshold values based on generation time should be recalculated to the relevant time frame of the assessment, which was 30 years in the latest KEC4 assessment. To be consistent with the IUCN definition this should be done by applying the definition of generation time as used by the IUCN, which is *the mean age at which a cohort of individuals produce offspring* (see point 3 in section 2.3). For most species included in the KEC4 assessment, using this definition would result in a slightly lower estimate of generation time compared to the definition used by Potiek, et al. (2022), which was *the time required by the population to increase by a factor of* R_0 (mean lifetime reproductive output; see Appendix A).

4.1.2 Considerations for X

For the choice of the threshold of an unacceptable decline, the conservation status should be taken into account. For species with an unfavourable conservation status, the level of X should be stricter. An indicator of conservation status on the international scale is the IUCN Red List status (IUCN, 2012). On the national scale the Dutch Favourable Conservation Status (*Staat van Instandhouding, SvI,* (Sovon, 2022)) can be used, which is used as input for species conservation laws and policies regarding birds in the Netherlands. On an international scale indicators for conservation status are defined by the Marine Strategy Framework Directive (MSFD) criterion D1C2 (Population abundance of seabirds) for 'Good Environmental Status' (OSPAR, 2023), or the IUCN threshold for category 'Least Concern'.

We suggest making the choice for X solely dependent on the conservation status. This conservation status already takes into account the following factors: population trend, distribution, habitat and future perspective. This combination of factors presents a thorough basis for a well-considered decision. In case the different indicators (SvI, MSFD or IUCN) disagree in the assessment of conservation status, expert judgement should be used to decide on a precautionary value of X.

Note that with the currently (and previously) presented approach, the reference point against which the unacceptable decline is measured is the final population size after 30 years. Using this reference point after 30 years instead of the current population size, the effect of the impact is tested, and any current population decline is not considered. Hence, it is important that an unfavourable population status as a result of a population decline is reflected in a stricter level of X. If a population status is currently unfavourable or is likely to become unfavourable, it is ecologically more relevant to relate the unacceptable decline to a favourable reference value, as described in more detail in section 5.

4.1.3 The implications of exponential growth

The ALI methodology uses population models without density dependence, and population abundance therefore grows or declines exponentially. This means that the modelled populations may grow infinitely, without being limited at high population abundances. Especially on longer time-scales this is an unrealistic assumption, as in reality population growth will decline at high population numbers. However, there is generally little information on the relationship between population growth and population abundance (the form of density dependence) and even less so about the exact mechanisms that are responsible for this relationship (the type of density dependence) and this is also the case for the bird species most likely to be relevant in OWF impact assessments. Casualties from OWFs have a more severe effects on population growth in populations that grow exponentially. In models that include density-dependent processes, (part of) the additional mortality from OWFs is compensated for, because (other) vital rates (reproduction, baseline survival) increase at lower population abundance. Wrongly specifying the shape and type of density dependence therefore introduces a risk of underestimating the true population effect of additional mortality. The assumption of exponential growth is therefore often retained for precautionary reasons as we want to be as certain as possible that we do not underestimate the impact.

The consequence of the assumption of exponential growth is that the relative population effect increases over time (Figure 8), as there is no population compensation. Over longer time periods relatively small levels of additional mortality can lead to large differences between the unimpacted and impacted scenario. In the example in Figure 8, 1% additional mortality leads to a 23% lower population abundance after 30 years compared to the unimpacted scenario.

It should be noted that despite any large difference in final population abundance between the two scenarios, the absolute trend in population abundance with impact from OWFs might still be increasing. For example, an OWF-induced mortality of 0.01 that reduces the population growth rate from 1.03 to 1.02 would result in a population abundance after 30 years that is $1 - (1.02/1.03)^{30} =$ 25% lower with impact than without impact, but still $1.02^{30} = 1.8$ times higher than at the start of the impact. This is a direct consequence of the fact that the ALI is based on the relative difference between two scenarios, instead of the absolute development of the population over time.

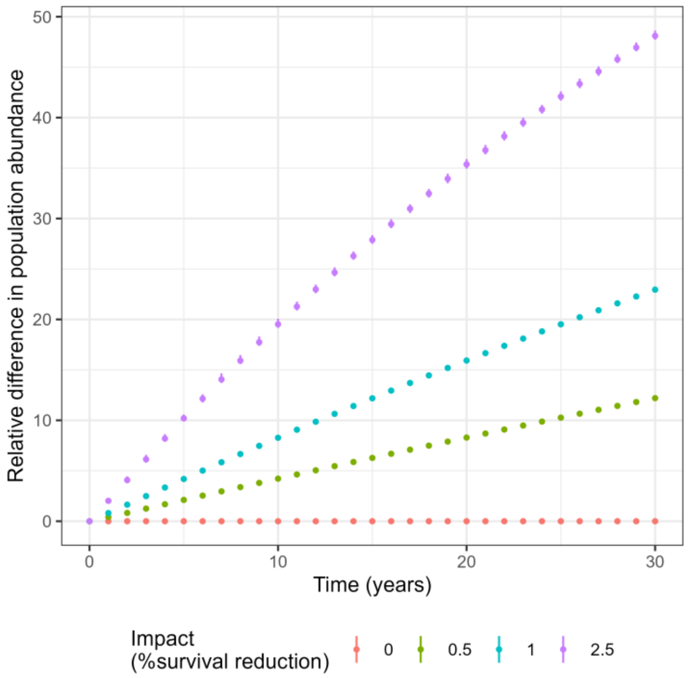


***Figure 8**: Development of relative population impact over time for different levels of impact (colours). The X-axis represents time in years after the start of the impact (OWF construction). The y-axis represents the relative reduction in population abundance compared to development of the population without impact (the red line with impact = 0).*

4.2 Choosing Y

In the original ALI, the Y threshold was derived from a causality threshold (Box 2) and therefore reflected the probability that an unacceptable outcome (violation of the X threshold) was caused by an OWF-related impact. This focused the burden of proof on being sure the effect is due to the windfarm and not potentially some knowledge uncertainty. The key issue with this line of reasoning is that under higher uncertainty or a stricter X it becomes harder to prove the found effect was truly caused by the tested impact of the windfarm.

In the revised methodology, Y is not derived from a threshold for causality, but rather represents the probability that a population effect of OWFs that is deemed unacceptable (larger than X), still occurs. This would correspond to statistical errors in hypothesis testing and we therefore suggest to define Y using a commonly accepted level of statistical error, such as 0.05. Such a low value for Y also guarantees that the revised ALI methodology adheres to the precautionary principle regarding the effect of uncertainty (see section 3). It should be noted that choosing very low values for Y will make the outcome of the ALI more dependent on the number of replicate simulations performed. It is therefore advised to check whether the number of replicate simulations has a large influence on the outcome and how this depends on the adopted Y value.

Because we suggest Y to represent a simple statistical error, we also suggest that Y should not be made dependent on any species-specific considerations such as population trends. Hence, following a recommendation from the reviewers, we recommend policymakers to keep Y constant for all considered species.

5 Further consideration: Status-approach

In the presented methodology, as well as in the original methodology, the unacceptable decline is measured in relation to the unimpacted scenario after 30 years. Using this reference point after 30 years instead of the current population size, the effect of the impact is tested, and any current or prior population decline is not taken into account (however, note that an unfavourable population status as a result of a population decline should result in a stricter level of X). Although the thresholds are species-specific and take into account the species' status, the methodology focuses on the relative impact, and can be referred to as the '**impact-approach**'. Within this approach, the modelled effect of the impact is purely based on the final population sizes for impacted and unimpacted simulations.

Note that when a population is declining, the unimpacted final population size is already lower than the current population size. This means that even if the chosen thresholds are more strict, the reference point (i.e. unimpacted final population size) may be an undesirable population level. For that reason, we here advise an additional approach.

If a population status is currently unfavourable or threatens to become so due to a modelled population decline, it is ecologically more relevant to relate the unacceptable decline to a favourable reference value, which are absolute reference points for population abundance. For that reason, we suggest to develop an additional approach that focuses on the conservation status of a species, which is a leading criterium for Dutch and European nature legislation. This so-called '**status-approach**' was also suggested by one of the reviewers and would require a species-specific reference value that defines the minimum population size deemed acceptable. The maximum acceptable impact of OWFs could then be equal to the relative difference between the current population size and this reference value, or to 0 if the current population is below the reference value.

To illustrate the status-approach and its merits in addition to the impact-approach, we describe three simple examples in Figure 9, each with a relative impact of 20%. Note that this means that the impact-approach would in these different situations render the same outcome, assuming the thresholds are constant. However, the suggested status-approach would take into account the status in relation to the favourable reference value (FRV). In the left figure the population starts in a favourable status and is still in a favourable status after 30 years with OWF-impact. In the right figure, the OWF-impact causes a population decline to a point below the FRV while the unimpacted population is still above the FRV. In the scenario in the middle the population is currently in an unfavourable status, but the population projection without OWFs predicts a favourable status after 30 years; however, the additional OWF-impact results in the population no longer reaching a favourable status within 30 years.

In such a status-approach, the allowable X will depend on the relative difference between the unimpacted scenario and the FRV. In case of the impact-approach, this is not explicitly the case. However, note that the allowable X can be chosen based on a combination of the current status and the projected population trend.

As described in Potiek et al. (2023), in order to be applicable for the assessment using the current population models, such a reference value must meet the following requirements:

a) reference values need to be available for all species
b) the spatial scale of the population to which the favourable reference value applies should be equivalent to the scale used within the ALI methodology. Within the KEC studies, the population is defined as 'all individuals making use of 1. the Dutch continental plate (national scenario) or 2. the southern and central North Sea (international scenario)'.

c) the methodology behind defining the reference value should be relatable to the population sizes used within the ALI methodology. Within the population models behind the ALI methodology, the population size is based on MWTL/ESAS density maps, and calculated as the maximum bimonthly population estimate.

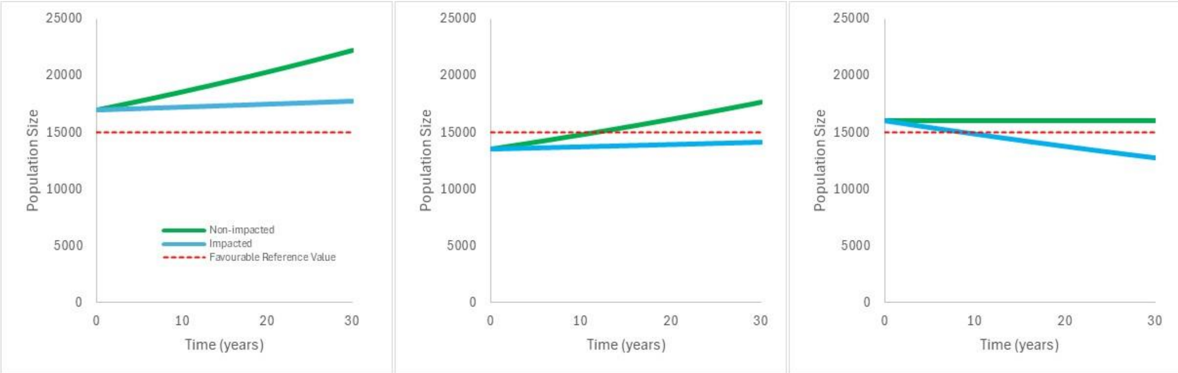


Figure 9*: Illustration of three scenarios. Implications for a status-approach are described in the text. Green line represents scenario without impact, blue line with impact and red dotted line presenting the favourable status (FRV). The acceptable decline X is constant in all figures (20%).*

Regarding the impact assessment of offshore wind farms, the 'status-approach' and 'impact approach' address different research questions, respectively:

1. What is the impact of offshore wind farms on the population of a species? For this question, the outcome of the scenario with wind farms is compared to the outcome of the scenario without wind farms. In other words, in order to answer this question, the impacted scenario is tested in relation to the scenario without impact. For this research question, the focus is on the effect of the impact. In this approach, which is the point of departure for the ALI methodology, the magnitude of the impact that is considered 'acceptable' should be used to define the threshold for X.
2. Will development of offshore wind energy, potentially in cumulation with other effects, affect the conservation status of a population? Note that the population status may already be unfavourable without additional mortality due to OWFs. In addition, the impact of OWFs may be small compared to some of the other cumulative effects. This research question focuses on the conservation status per se (for the scenario with impact), which is of key relevance for the legal tenability of the activity, instead of on the effect of the impact.

The currently presented approach, as well as the original ALI methodology, relates to the first research question, with a focus on the impact. The ALI methodology originates from the KEC studies, and aimed at focusing on the impact of solely offshore wind energy. Hence, within the original and current ALI methodology the effect of a single impact, in this case offshore wind energy, is assessed. The current aim is not to predict future population development considering a multitude of (cumulative) impacts. Note that in the current impact-approach the relative difference between the impacted and null scenario is assessed, which means that the outcome is independent of the initial population size, and that the baseline growth rate and size of a population have only a modest effect on the outcome (Figure 7). This also means that any under- or overestimation of initial population size does not affect the outcome of the population models, apart from potential under- or overestimation of the modelled impact (*e.g.* casualties from collision or habitat loss).

As described in this chapter, we recommend extending the approach with an additional status-approach, in which the conservation status of a species is the central point of interest. Note that such a status-approach requires predictions of future population development in an absolute sense. This means that this approach is more sensitive to under- or overestimation of the population size as well as the baseline growth rate. For this status-approach, the population model needs to be validated with the observed population growth rate. Moreover, this approach requires a more thorough understanding of the different processes that might affect population dynamics.

6 Conclusions and recommendations

We present a revised methodology for defining acceptable levels of impact from OWFs on seabird populations. The following points summarize the most important properties of the revised methodology, and their associated recommendations.

1. Similar to the original ALI methodology, the revised ALI is a probabilistic threshold that assesses the relative population effect of an impact by comparing an impacted scenario to an unimpacted scenario.
2. The impact from OWFs on the population is evaluated while keeping other processes that affect the population constant. These processes include environmental stochasticity, which should be simulated by varying population parameters on a yearly basis, but could also include uncertainty in model parameters. We recommend an approach in which processes that affect the prediction of unimpacted population abundance are kept equal between scenarios.
3. The result of 2) is that most of the variation created by environmental stochasticity cancels out, because this variation is applied in the same manner in both scenarios. If variation or uncertainty in the OWF impact is not explicitly accounted for, there is very little variation in the distribution of the relative impact. Therefore, taking account of the variation and uncertainty associated with the size of the OWF impact is crucial in any assessment that utilizes this approach
4. Unlike the original ALI methodology (Potiek et al. 2022), the revised ALI does not use a causality measure. Instead, all violations of the X threshold are due to the OWF-impact. The revised framework also does not include a probability of violating the X threshold based on variation or uncertainty ($P_{v,u}$).
5. In the revised methodology, the threshold values X and Y are independent and the revised ALI adheres to the precautionary principle if $Y < 0.5$. In addition, choosing a more strict X threshold will lead to a more strict ALI (*i.e.* the ALI is violated at a lower OWF impact).
6. The relative impact on the population declines as a function of population growth rate in the unimpacted scenario. It is therefore key that the population models associated with the ALI do not overestimate the true population growth rate. This can be achieved by comparing the population growth rate predicted by the unimpacted scenario model with the observed population trend.

In addition, we present new considerations for choosing threshold values X and Y, which are:

7. The value of X should be defined on a time frame that is relevant for the species. The IUCN criterium of the maximum of three generations or ten years as used in the original methodology is therefore still appropriate. The same definition of generation time as used by the IUCN should be adopted.
8. The choice of X should take account of the conservation status of a species. For species with an unfavourable conservation status, X should be stricter (lower). X should be defined per species/population, also because of 7). If the status is currently favourable, but the population is declining to such an extent that an unfavourable status is likely to occur, X should be stricter as well.
9. The choice of Y should reflect a commonly accepted level of statistical error, such as 0.05. The value of Y can be equal for all species.
10. We suggest to develop an additional status-approach, in which the focus is on whether the conservation status of a species is favourable at the end of the time period, as described in section 5. Considering species' conservation status is required to ensure the legal tenability of offshore wind energy development.

Appendix A: Generation time

Caswell (2001, p. 128) presents three commonly used definitions of generation time for age-structured population models:

1. The time T required for the population to increase by a factor of R_0 (the reproductive rate), which satisfies $\lambda^T = R_0$ and hence $T = \frac{\log R_0}{\log \lambda}$
2. The mean age μ of the parents of the offspring produced by a cohort over its lifetime
3. The mean age $\bar{A}$ of the parents of the offspring produced by a population at the stable age distribution.

In stationary populations ($\lambda = 1$) definitions 2 and 3 are equal. The difference between these measures of generation time are likely greater for species with higher mortality rates and population growth rates farther from 1 (Caswell, 2001).

The IUCN uses definition 2) to calculate generation time, while the original ALI methodology used definition 1). As outlined in the ALI review, definition 1) is invalid for stationary ($\lambda = 1$) or declining populations. For most species included in the KEC4 assessment, definition 1) yields a higher estimate of generation time than definition 2) (Figure A1). Exceptions are the species with an estimated population growth rate > 1 (Figure A2). It should be noted that for species with declining growth rates ($\lambda < 1$), definition 1) still results in a estimate of generation time that is approximately equal to the midpoint of the two other definitions, because $\log R_0$ and $\log \lambda$ always have equal sign (Figure A1).

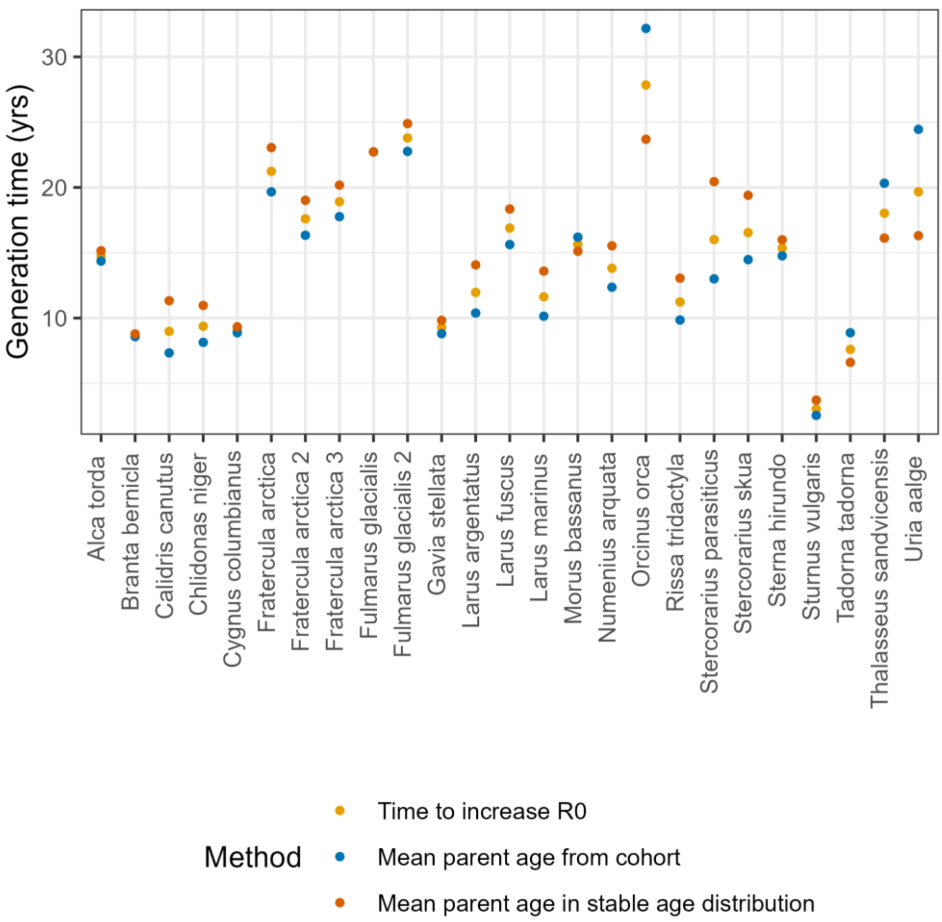


Figure A1*: The three measures of generation time for the species included in the KEC4 assessment, plus the killer whale (Orcinus orca). Definition 1) = "Time to increase R_0", definition 2) = "mean parent age from cohort" and definition 3 = "mean parent age in stable age distribution".*

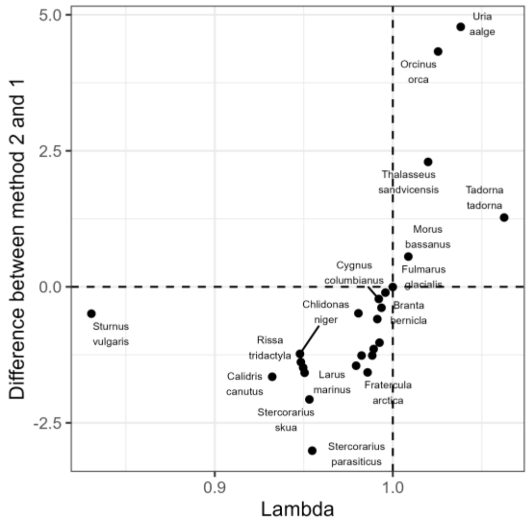


***Figure A2**: Difference between generation time measures 1) and 2) as a function of the predicted population growth rate ('Lambda'). If 'Lambda' > 1, definition 2 (based on a cohort of individuals) yields a higher generation time than definition 1 (based on net reproductive rate).*

7 Quality Assurance

Wageningen Marine Research utilises an ISO 9001:2015 certified quality management system. The organisation has been certified since 27 February 2001. The certification was issued by DNV.

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Justification

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The scientific quality of this report has been peer reviewed by a colleague scientist and a member of the Management Team of Wageningen Marine Research

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