# Assessing the demographic impact of bird collisions with wind turbines.

## *State of the art and methodological recommendations.*

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# 1. Introduction

In France and elsewhere in the world, the development of wind energy stands as one of the pillars of the energy transition (Teske et al. 2019). However, this mode of energy production is also a source of negative impacts on biodiversity, particularly for birds (Drewitt and Langston 2006, Schuster et al. 2015, Serrano et al. 2020). These impacts include (i) direct impacts, i.e. mortality caused by bird collisions with turbines; and (ii) indirect impacts, such as habitat loss, disturbance and barrier effects (Drewitt and Langston 2006, Fox et al. 2006, Fox and Petersen 2019). The MAPE[1] research project, for which this literature review was written, focuses only on direct impacts, i.e. mortality due to collisions with wind turbines.

Wind farms, such as defined by the French law of 10 February 2000[2] (hereinafter referred to as "wind turbines" or "wind farms"), are Installations Classified for Environmental Protection (ICPE) since the decree of 23/08/2011. They are subject to the principle of Avoidance, Mitigation and Compensation[3] of their impact on the environment (a principle called the "ERC[4]" sequence, in French; L.110-1 and L. 122-3 CE). For each wind farm project (with masts higher than 50 m) subject to authorisation and according to certain criteria in the context of repowering, an impact study must therefore be carried out (L. 122-1 EC) in order to assess its effects on the environment, including the impacts on biodiversity and in particular on birds.

If protected species[5] occur[6] on the site and a risk of collision has been established, the project owner is also obliged to request an exemption authorising the destruction[7] of protected species (DEP; L.411-1 and L.411-2 EC). The application for a DEP exemption is necessary when there is a risk of destruction of one or more of these protected species despite avoidance and mitigation measures, even if this risk only concerns one pair or one individual. The project owner must then demonstrate that there are no impacts on protected species if he/she decides not to apply for a DEP exemption. It is the impact assessment study that will assess the risk of mortality for the birds concerned. Recent case law[8] has re-emphasized the *obligatory and systematic* nature of this procedure, and has highlighted the fact that no suspensive condition can be retained. Nonetheless, the proper implementation of this procedure remains highly variable among different regions of France. In theory, the DEP exemption can only be granted by the French State services if the following three conditions (L.411-2 CE) are met:

1. the wind farm project must be carried out for imperative reasons of **major public interest**
2. there is **no alternative** satisfactory solution
3. the DEP exemption is not detrimental to **maintaining the populations** of the species concerned in a favourable conservation status, within their natural range.

In addition to assessing fatality risk (i.e. number of individuals at risk of collision), it is therefore necessary, in the context of this last point, that the impact of a wind farm project be assessed at the

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[1] MAPE : « réduction de la **M**ortalité **A**viaire dans les **P**arcs **É**oliens en exploitation » ("Reduction of Bird Mortality in Operating Wind Farms"). More information here: https://mape.cnrs.fr/

[2] In French legal texts, wind farms are precisely defined as "terrestrial electricity production installations that use the mechanical energy of the wind and that are made of one or more turbines". https://www.legifrance.gouv.fr/loda/id/LEGITEXT000005629085

[3] French law states that 100% of the impacts caused by the building of an infrastructure should be avoided or at least mitigated; and any residual impact, after mitigation, should be compensated for (L.110-1 and L. 122-3 CE).

[4] ERC stands for "Avoiding, Mitigating and Compensating" ("Eviter, Réduire, Compenser" in French).

[5] Species included in the list established by the Decree of 29 October 2009: https://www.legifrance.gouv.fr/loda/id/JORFTEXT000021384277

[6] Nesting, feeding or migration.

[7] The term "destruction" in its broadest sense, as defined in Article L. 411-1 of the French Environmental Law.

[8] See the following documents: CAA NANTES, 06/10/2020, 19NT02389 et CAA BORDEAUX, 17/11/2020, 19BX02284.

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scale of the bird population. More generally, this shift from the individual to the population scale is crucial if we wish to better understand the consequences of wind power development on biodiversity (May et al. 2019). Indeed, the major challenge for biodiversity conservation is to maintain populations in "good conservation status", i.e. making sure that they remain *viable* in the short and the long run (Sanderson 2006, Traill et al. 2010).

To understand the consequences of collisions on the conservation status of populations, estimating the number of mortalities (individual scale) is not sufficient (Diffendorfer et al. 2015, May et al. 2019). Indeed, the impact severity of additional mortalities on a bird population depends on three things: (i) the size of the impacted population, (ii) its current demographic trend and (iii) the species vital rates (survivals and fecundities).

First, an abundant population will be able to withstand mortalities better than a small population. The effect of mortalities at the population level should be considered in terms of mortality rates (number of fatalities/population size), rather than in terms of raw numbers of mortalities. Here, the choice of a relevant spatial scale to delimit the population that needs to be considered in the impact assessment is a crucial but difficult element. The issues related to this choice of scale, and some avenues for reflection, are discussed below (in section 3 "Delineation of the population: spatial scale and cumulative impacts").

Second, the demographic trend of the population also plays a role. A growing population will suffer less from additional mortality than one that is already declining. This is because a growing population produces a 'surplus' of individuals, which partly offsets the additional mortalities. Nonetheless, the *relative impact*[9] of fatalities on future population size can be quite similar regardless of the trend.

Finally, additional mortalities do not have the same demographic impact depending on the species of interest, as this impact depends on its survival and fecundity rates. Long-lived[10] species, such as large raptors, are characterised by low population growth potential due to their low fecundity and relatively long immaturity period (Stearns 1992, Caswell 2001). They produce few young each year and are therefore much more vulnerable to additional mortality than short-lived species (which have higher productivity), such as passerines.

Assessing impacts at the population scale requires an analytical step to translate a number of fatalities into a *change* of population trajectory. Currently, however, impact studies are generally limited to an interpretation of individual mortalities (May et al. 2019). The questions addressed are generally the following: how many individuals of a given species frequent the target implantation site? how many carcasses were observed during post-implementation monitoring, and therefore how many mortalities are estimated (sometimes after correcting for various sources of bias such as the persistence of carcasses or their non-exhaustive detection; Bernardino et al. 2013)? Or what is the collision risk of any single individual (Band et al. 2007)? Based on this information, the "significance" of the impact for the population or species is then assessed from an "expert's opinion", without use of any structured methodology[11]. This procedure, which lacks rigour and standardisation, is therefore subject to numerous biases (Flyvbjerg 2007, Gigerenzer and Gaissmaier 2011, Sörqvist 2016, Williams and Dupuy 2017, Enríquez-de-Salamanca 2018).

In this context, one of the objectives of the MAPE project is to provide a standardised and objective methodological framework that allows a reliable and rigorous assessment of the impact of collisions on bird populations. To this end, we have carried out a review of the literature dealing with the

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[9] i.e., the *relative drop* in population size caused by collision fatalities. The relative impact on the population can be measured by comparing two scenarios: with and without additional mortalities. This is the approach we recommend to take (see below).

[10] "Long-lived": species with a life expectancy generally over 15 years.

[11] There are, however, so-called 'structured' approaches to expert elicitation that are recognised as effective (Martin et al. 2012, Frick et al. 2017), but they are rarely used in impact assessments.

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problem of dealing with additional mortalities caused by human activities. Indeed, a rich literature exists on this subject, particularly in the context of animal populations exploited for hunting and fishing. This document summarises the topic's state of the art and presents the methodological framework that we propose to use to assess the impact of collisions at the population level in the context of wind energy projects.

# 2. Literature review

The question of the impact and sustainability[12] of human-caused mortality on animal populations is not new (Errington 1945, Schaefer 1954). Historically, it has been addressed in two ways.

First, in the context of exploited species (hunting, fishing), methods for calculating quotas were developed to determine harvest thresholds that were sustainable for populations (Ricker 1954, Schaefer 1954, Milner-Gulland and Akçakaya 2001, Sutherland 2001, Beverton and Holt 2012). These analytical methods have subsequently been adapted in the context of the conservation of non-exploited marine species (cetaceans, sharks, seabirds, sea turtles) that are victims of bycatch mortalities (NMFS 1994, Wade 1998, Dillingham and Fletcher 2008, 2011, Zhou and Griffiths 2008, Curtis and Moore 2013, Moore et al. 2013). The most widely used method for calculating so-called 'sustainable' removal quotas is the 'Potential Biological Removal' - PBR (Wade 1998, Dillingham and Fletcher 2008). This is the first method discussed below.

The second approach found in the literature consists of explicitly modelling the trajectory of a population over time in order to assess the consequences of different disturbance or management scenarios (Lande et al. 2003). This method, known as "population projections", is widely used in conservation biology to carry out population viability analyses[13] (Boyce 1992, Beissinger and McCullough 2002). It is the second approach that we will describe.

## 2.1. The Potential Biological Removal (PBR)

The PBR method was developed in North America in the 1990s in the context of incidental catches of cetaceans in the nets of commercial fishing vessels (NMFS 1994, Wade 1998). This approach is mainly used in the case of non-exploited species that are victims of accidental mortalities induced by human activities (Dillingham and Fletcher 2011). There are a few examples in the international literature of the use of PBR in the context of avian mortalities caused by wind turbines: (i) mainly in published study reports (Poot et al. 2011, Leopold et al. 2014, NIRAS 2016), but also (ii) in some scientific publications (Bellebaum et al. 2013, Busch and Garthe 2016).

### 2.1.1. Principle

The PBR is an analytical method for calculating a harvest quota based on the following formula (Wade 1998):

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[12] A population subject to harvesting (voluntary: hunting, fishing; or involuntary: accidental mortality) is "sustainable" if its long-term persistence is not compromised (Sutherland 2001, Fryxell et al. 2010). In reality, the boundary between 'sustainable' and 'unsustainable' is very difficult to determine due to the probabilistic and uncertain nature of population persistence (Sanderson 2006, Traill et al. 2007).
[13] "Population viability analysis" (PVA) is a method to assess the risk of extinction of a population over a given time horizon.

$$PBR \;=\; F\frac{R_{max}}{2}N_{min} \qquad (1)$$

where:

- $R_{max}$ corresponds to the theoretical maximum growth rate of the population, i.e. when it is at low density and in the absence of anthropogenic mortalities;

- $N_{min}$ is a "conservative" estimate of the population size; and

- $F$ is an arbitrary parameter, often referred to as the "recovery factor", to adjust the quota for various sources of uncertainty[14].

This formula relies on the assumption that the population of interest follows a "logistic" growth curve (Wade 1998, Eberhardt et al. 2008), as shown in Figure 1, and that it is able to achieve its theoretical maximum growth rate ($R_{max}$). If these conditions are met (but see limitations below), then by applying a removal rate equal to the PBR, the population should move towards a new equilibrium as shown in Figure 2. In theory, if we set F = 1 in the formula, the new equilibrium point of the population will be equal to **half the carrying capacity** (K/2) of the environment, where carrying capacity (K) is defined as the maximum size that the population can reach in its local environment (Figure 1). If a value F < 1 is used, then the new population equilibrium point will be between K/2 and K (Wade 1998), but in all cases the underlying logic remains the same. The calculated quota relies on the assumption that the population will tend towards a **new stable equilibrium point**[15], regardless of its initial state (Figure 2). In other words, a population below the equilibrium point will be expected to continue to grow, despite additional mortalities, until it reaches this new equilibrium point, whereas a population above the equilibrium point will decline towards this equilibrium point. This notion of equilibrium point is therefore central to the logic of the PBR, but it is often misunderstood by users, who tend to interpret the PBR as a quota to "*protect against the risk of decline*" (e.g. Leopold et al. 2014). This interpretation of the PBR is wrong. The PBR is a value which, if the assumptions underlying the mathematical calculation are correct, just prevents the population from falling below the pre-determined threshold of K/2. One must understand that, before reaching its new equilibrium point, a population might decline sharply for some time, especially if it was close to the carrying capacity.

### 2.1.2. Advantages

The main advantages of this approach are (i) its simplicity of implementation and interpretation, and (ii) the fact that it requires little data. Indeed, applying this simple formula will directly provide a removal quota, i.e. the number of mortalities below which it is considered that the persistence of the population will be assured. The result is therefore very simple to interpret: below the PBR, the level of harvest is considered "sustainable"; above this quota, the harvest rate is "unsustainable". Moreover, the only data required to calculate this quota are the $R_{max}$ and an estimate of the population size. The $R_{max}$ is rarely known, but it can be estimated, via empirical relationships, from the adult survival and the age of first reproduction of the species (Niel and Lebreton 2005); two parameters that are well known for most bird species.

These are the two main reasons why this approach is so attractive to wildlife managers and decision makers (Dillingham and Fletcher 2008, Lonergan 2011, Cooke et al. 2012, Moore et al. 2013).

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[14] All sources of uncertainty inherent to the input data, without any distinction. See text below.
[15] A '*stable*' equilibrium point is an equilibrium point towards which the population will always tend to return to after a disturbance.
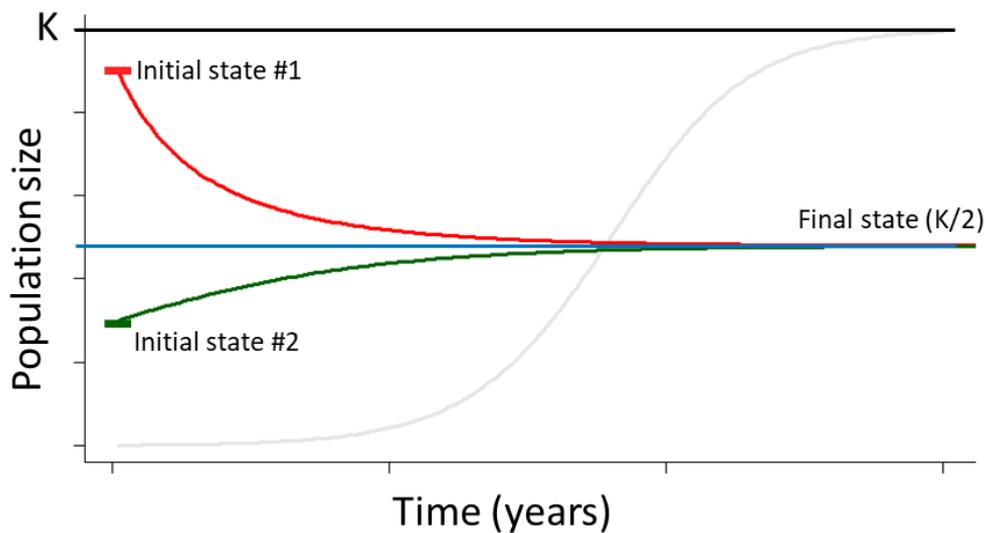
## Figure 1 : Logistic (theoretical) growth curve of a population.
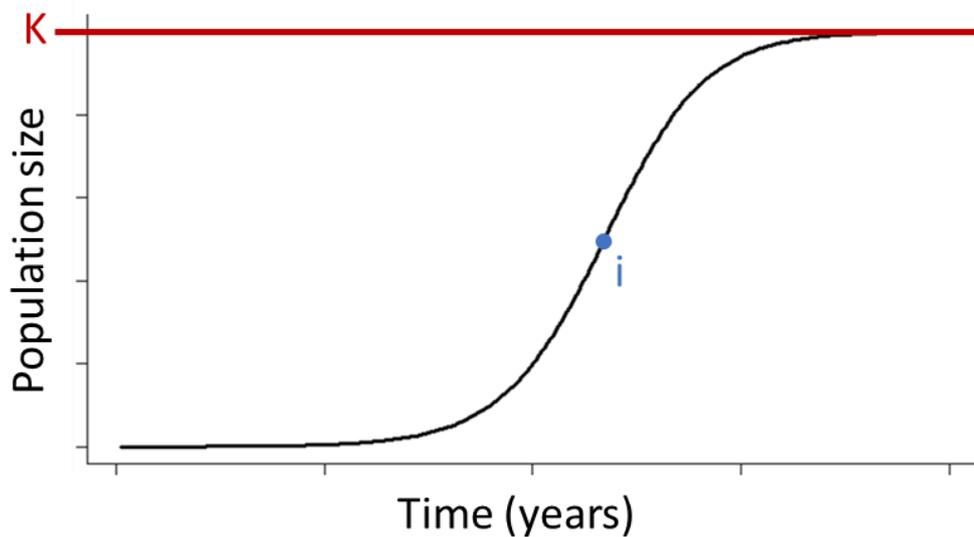


A logistic growth is characterised by this "S" shape curve, which consists of three distinct phases: (i) rapid exponential growth at low density, as long as the population is below the inflection point "i"; (ii) slower growth when the population has passed the inflection point "i"; (iii) stabilisation of the population size when it has reached the carrying capacity of the environment (K), which corresponds to the maximum size that the population can reach in its local environment.

This "S" shaped trajectory corresponds to the classical picture of "natural" population growth in ecology, due to the existence of competition for resources. When the population is small, the resources available to each individual are large, so the survival and fecundity of these individuals are good and the population grows rapidly. As the population size increases, the competition for resources increases, resulting in lower survival and fecundity of individuals and thus slower population growth.

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**Figure 2 : Theoretical trajectory of a population with a levy equal to the PBR.**



Theoretical trajectory of a population, according to two scenarios (two initial states, in red and in green), to which we would apply a removal rate equal to the PBR quota (with F = 1). Independently from its initial state, the population should tend towards the same equilibrium, which is equal to half the carrying capacity (K/2, blue line). The "S" curve, in light grey in the background, represents the theoretical logistic growth curve.

### 2.1.1. Limits

Several criticisms have been made against this approach, and more specifically against its use in impact assessment studies of wind energy projects (Green et al. 2016, O'Brien et al. 2017, Schippers et al. 2020).

On the one hand, the validity of the quota provided by the PBR calculation relies on several assumptions that are in fact rarely verified in animal populations (O'Brien et al. 2017, Miller et al. 2019, Punt et al. 2020). The first assumption is that the population must be free of any threats or disturbances other than the source of mortality of interest, so that it can achieve its optimal growth rate ($R_{max}$, at low density) in the absence of that source of mortality. In other words, it is assumed that in the absence of fatalities *due to this single source*, the population would grow optimally. In the context of wind energy, this means that bird populations must not be subject to any other anthropogenic threats that would cause mortality or reduce fecundity. This is far from being the case. Bird populations face a multitude of human-induced disturbances other than wind energy (Van

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Bommel 2004, BirdLife International 2018): habitat loss and degradation (urbanisation, deforestation), invasive species, climate change, collisions with other structures (power lines, buildings, vehicles, etc.), poisoning (pesticides and other sources), hunting, poaching and other forms of intentional or accidental destruction. The PBR approach is therefore not suitable if we are looking at a single additional source of mortality among others (Green et al. 2016).

Secondly, the PBR formula assumes that the population follows a logistic type of growth (Figure 1; (Wade 1998, Eberhardt et al. 2008), which assumes the existence of a compensatory density-dependency relationship (Herrando-Pérez et al. 2012). This means that the population growth rate is expected to increase when the population size decreases (e.g. in response to harvesting), thus partially or fully compensating the additional mortality (Rose et al. 2001, Beverton and Holt 2012). However, several scientific studies have shown that in bird populations this type of compensatory relationship does not always exist (Horswill and Robinson 2015, Horswill et al. 2017). In this case, the very basis of the PBR, i.e. the predicted equilibrium point, is no longer valid. Without a compensatory relationship, the population will not be able to stabilise at a new equilibrium point and any decline will eventually lead to extinction. Applying the PBR approach in such situations can therefore have catastrophic consequences (O'Brien et al. 2017). If a population is already declining, even without any wind energy related mortalities, the implementation of a quota based on the PBR approach will only amplify and accelerate the decline.

Another criticism concerns the fact that the parameter F ("recovery factor") of equation (1) has no real biological meaning (Punt et al. 2018, 2020). In the original formulation of the PBR (Wade 1998), this parameter was presented as a simple way to revise the quota value downwards to (1) apply a precautionary principle given sources of uncertainty (e.g. on population size or on the $R_{max}$) that cannot be incorporated into the calculation; and (2) allow for faster recovery of depreciated populations (Dillingham and Fletcher 2008). To their credit, the authors of this formula carried a large simulation study to come up with relevant recommendations of F values to use for the conservation of cetaceans threatened by bycatch from North American fisheries (Wade 1998). However, this effort has not been repeated to assess the validity of these recommendations in other contexts where the PBR is used (Punt et al. 2018; O'Brien et al. 2017). Therefore, the value of the parameter F is often set arbitrarily, without any robust justification.

Furthermore, the PBR does not offer any flexibility in the way the trajectory of a population is modelled. As this method is based on a simple formula and constrained theoretical relationships (notably logistic growth), it is impossible to include local population specificities, or even well-known demographic mechanisms, such as the age structure of populations (Caswell 2001) or environmental or demographic stochasticity (Beissinger and McCullough 2002, Saeher and Engen 2002). The PBR is therefore also heavily criticised for its inability to predict what happens to populations when mortality levels are below the PBR threshold (Green et al. 2016, O'Brien et al. 2017).

In addition to these general criticisms, there are some limitations that are more specific to the wind energy context and its French and European regulatory framework. First of all, by providing a quota that can only be interpreted in a binary way (acceptable or unacceptable level of mortality), the PBR is poorly adapted to the ERC sequence, which is based on the notion of *impact relative to an initial state*. In theory, this impact should be measured quantitatively. However, it could be argued that in the context of a DEP exemption request, it is indeed a binary response that is required in order to make a decision: the granting or refusal of the exemption.

By applying the PBR formula, one implicitly accepts to use K/2 (half the carrying capacity) as the *critical limit* for a sustainable population. This value of K/2 is indeed the equilibrium point at which the population is expected to stabilise if the actual rate of removal exactly equates the PBR quota

and if all the assumptions underlying the calculation are verified[16]. The validity of this prediction is strongly constrained by the model assumptions discussed above: (i) the absence of other disturbances; if this assumption is false, it is not relevant to rely on the $R_{max}$ to calculate a quota; and (ii) the presence of compensatory « density-dependence, without which compensation for mortality is impossible. But, beyond these technical limitations, this implicit conservation objective (i.e. K/2 as a *limit*) raises a more fundamental question: what is the justification and relevance of using this K/2 value (or any other fraction of K) as the minimum critical size of a protected population? Indeed, there is no justification, either scientific or regulatory, for considering this K/2 value as a universal target for bird or wildlife conservation in general (Schippers et al. 2020). The use of this K/2 value stems from the fact that historically the PBR formula has been based on the maximum yield[17] calculation methods used to determine fisheries quotas (Beverton and Holt 2012), i.e. for exploited species and not protected species. In the case of a population that is close to its carrying capacity, the use of a quota equal to the PBR implies an acceptance of a 50% reduction in its size. However, there is no guarantee that this will maintain the population at "good conservation status", as dictated by French and European regulations. For protected species, it would seem more appropriate to use the scientific notion of "population viability[18]" (Thomas 1990, Sanderson 2006, Traill et al. 2007, 2010).

## 2.2. Population projections

The population projection method is widely used in ecological research and conservation biology (Morris and Doak 2002, Lande et al. 2003) as it is central to population viability analyses (Boyce 1992, Beissinger and McCullough 2002). This approach makes it possible to predict the trajectory of a population over time, according to any type of scenario defined by the investigator. It is therefore a very flexible and explicit approach that allows assessing the demographic impact of avian collisions by comparing two scenarios (Frick et al. 2017): (i) a scenario *with* additional collision fatalities and (ii) a scenario *without* additional fatalities (Figure 3). In the international literature, there are several examples of the use of population projections in the context of avian wind turbine fatalities: (i) in study reports (Poot et al. 2011, Rydell et al. 2012, Grünkorn et al. 2016, Korner-Nievergelt et al. 2016), and (ii) in various international scientific publications (Carrete et al. 2009, Masden 2010, García-Ripollés and López-López 2011, Schaub 2012, Sanz-Aguilar et al. 2015). However, there is a significant disparity in the level of complexity and realism of the demographic models used, with some studies remaining very simplistic (no age structure, no stochasticity; e.g. Grünkorn et al. 2016), while others employ complex individual-based simulations (e.g. Masden 2010, Schaub 2012).

### 2.2.1. Principle

Projecting a demographic trajectory involves simulating changes in the size of a population over time (Boyce 1992). This is done by setting an initial population size (usually the current population size) and then applying a 'mathematical' model to calculate changes in population size from year to year. These population changes are driven by two population processes: (i) the addition of new individuals,

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[16] i.e., if the population follows a logistic growth, determined by a linear density-dependence relationship; and in the case where F = 1 and the actual population size (N) is exactly equal to the value $N_{min}$ used in the formula. Since the logic of the PBR is to use a precautionary principle, it is generally considered that in the majority of cases, one will have F < 1, N > $N_{min}$ and/or a form of density-dependence such that the equilibrium point will tend to be greater than K/2 (Wade 1998, Dillingham and Fletcher 2008), but the logic remains the same.

[17] Notion of "maximum sustainable yield", which has itself been criticized (Larkin 1977).

[18] The "viability" of a population corresponds to its probability of persistence over a given time horizon. A population is viable if it has a high probability of persistence on the long term. There are three components to the viability of a wild population: demographic, genetic and ecological (Sanderson 2006).

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linked to births and modelled through fecundity parameters; and (ii) the loss of individuals, due to fatalities. In general, mortality of "natural[19]" origin is modelled via a survival probability[20], then mortality due to the cause of interest can be added by including another specific parameter (e.g. additional mortality rate).

## Figure 3 : Population trajectories with and without additional mortality.
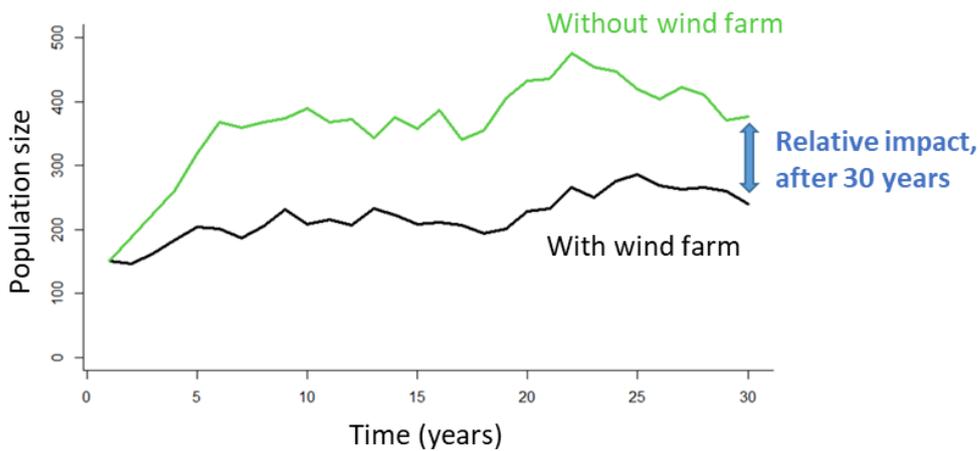


Illustration of population trajectories, according to two scenarios: (i) without a wind farm, i.e. without additional mortality (in green) and (ii) with additional mortality due to the presence of a wind farm (in black). The relative impact corresponds to the percent difference in population size between these two scenarios after 30 years. The time horizon can be different depending on the needs of the study. In addition, it is possible to use this approach to assess cumulative impacts, by summing up the fatalities of several wind farms.

### 2.2.2. Advantages

This method has many advantages over the PBR approach (Beissinger and McCullough 2002, Green et al. 2016). First, all assumptions of the model are under control. Unlike with the PBR, the user is not forced to assume "optimal" population growth, or even a logistic growth. Populations with sub-optimal growth, that are stable or even already in decline can be modelled. Density-dependence can be excluded or included, and can be modelled in any way we choose (Eberhardt et al. 2008). This approach is therefore adaptable to any situation. In the context of wind energy, it can be used to quantify collision impacts, even if the target populations are subject to other disturbances (Green et

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[19] "Natural" here means that we consider all causes of mortality (which may include anthropogenic causes) other than the source of additive mortality we are interested in (here, collisions).
[20] Probability of survival = 1 − probability of mortality.

al. 2016, Frick et al. 2017); provided, of course, that the appropriate data (see below) are available. For data that are not available, it is always possible to use a structured approach of expert elicitation (Martin et al. 2012) to fill these gaps[21].

In addition, the population projection method offers a great deal of flexibility in the demographic processes that can be included, which allows for more realism (Boyce 1992, Lande et al. 2003). For example, age structure on survival and fecundity parameters is usually included to properly account for delayed maturity[22] and the fact that the different age groups have different contributions to the dynamic of the population (Caswell 2001). Stochasticity, i.e. variability due to demographic and environmental random events, can also be included (Morris and Doak 2002, Lande et al. 2003, Saether et al. 2016). These random events are especially important to consider in the case of a small population, as they greatly increase its risk of extinction (Lande 1993). This flexibility in the model construction also makes it possible to include local specificities (e.g. local values of survival and fecundity) when they are available.

Finally, when based on a comparative approach between two scenarios (e.g. with vs. without additional fatalities), this method provides a measure of *relative impact*, which is less sensitive to uncertainties on demographic parameters and mechanisms (Green et al. 2016). Indeed, if certain assumptions are wrong or imprecise, they will apply equally to both scenarios, which will limit their influence on the final result in terms of relative impact. This simulation approach also allows for sensitivity analysis, and thus allows exploring the consequences of the different assumptions that were made when building the model (Mills and Lindberg 2002).

Finally, no implicit threshold (critical population size limit) is required to quantify the impact (Grant et al. 2016). This method therefore provides a result that is more "objective" than the one obtained from the PBR approach (Lonergan 2011).

### 2.2.3. Limits

The main limitation of this approach, compared to the PBR, is that it requires more data. In addition to the current population size, it requires an estimate of the population growth rate and associated survival and fecundity values. Ideally, vital rates should be estimated on the population of interest, thereby providing a precise estimate of the local population growth rate. However, the data needed to estimate vital rates (e.g. individual capture-mark-recapture monitoring data) are expensive and complex to acquire in the field, making it difficult to obtain these values at the local scale. Without this information, the model may lead to erroneous results (Reed et al. 2002). In the absence of local information though, estimates of vital rates can often been obtained from the literature on the same or a closely related species (Kindsvater et al. 2018). This is a good alternative because the values of these parameters are relatively stable within a species (Stearns 1992). However, in this case, it is preferable to know at least the general trend of the target population to adjust vital rate values to the local population growth rate. Indeed, it is usually much easier to get estimates of local population trends than local vital rates. As an alternative, expert elicitation can be used to fill in the gaps (Martin et al. 2012, Frick et al. 2017).

Another disadvantage of this method is the computational time required, which can sometimes be high. Indeed, in the presence of stochasticity in the model, it is necessary to repeat the simulations many times (e.g. >100 simulations) for each scenario if reliable results are to be obtained (Boyce

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[21] The so-called "structured" expert elicitation approaches are a set of standardized methods for translating the information provided by experts into quantitative expressions, which can then be used in a model. These methods aim to (i) reduce bias when extracting information from experts and (ii) combine the opinions of several experts into a probabilistic distribution so as to realistically represent uncertainties.
[22] Number of years between birth and age at maturity (first possible reproduction).

1992, Lande et al. 2003). This is due to the fact that each simulation will provide a different result because of random events (hazards). However, this limitation should be put into perspective for two reasons: first, because the simulation programming can often be optimised to reduce this duration; second, because nowadays it is quite cheap to get access to high computing power.

The final disadvantage is that the quantitative measure of impact does not provide a clear-cut, binary decision rule, unlike the PBR. Indeed, there is a continuous gradient of possible impacts, which implies a *post hoc* interpretation of the result in order to decide on its "acceptability" or not. This method therefore provides a result that is more difficult to interpret for the decision-maker. However, it does have the merit of separating the steps leading to a final decision: (i) on the one hand, the quantification of the impact, which is an objective exercise of purely technical and scientific nature; (ii) on the other hand, the interpretation of its acceptability, a more subjective step which is supposed to reflect a societal or political stance, rather than being an arbitrary choice (Lonergan 2011).

## 2.3. Comparison

A comparative summary of the advantages and disadvantages of the two methods is provided in Table 1.

The PBR requires less input information than the population projection method, and the binary output it provides (a quota) is easier to interpret. But this apparent simplicity comes at the expense of its reliability. As discussed above, the PBR threshold is based on an implicit objective that is not necessarily adapted to the problem of bird conservation in the face of the risk of collisions with wind turbines (Lonergan 2011, Green et al. 2016). Furthermore, this tool has been heavily criticised for its inability to predict what happens to populations when fatalities are below the threshold provided (Green et al. 2016, O'Brien et al. 2017).

**Table 1. Summary of the advantages and disadvantages of each method.**

| PBR (Potential Biological Removal) | Population projections |
|---|---|
| Requires less data | Requires more data |
| Fast (simple analytical formula) | Longer computation time (simulations) |
| Easy to interpret (binary result) | No clear-cut result (impact: non-binary) |
| Not suited to the ERC[23] sequence | Suitable for the ERC sequence |
| Implicit objective (K/2) | No implicit objective |
| Fixed assumptions: optimal growth and compensatory dependence density | Assumptions fully controlled |
| Unsuitable for a single cause of mortality | Suitable for any situation |

In green: advantages. In red: limitations and major disadvantages. In yellow-orange: minor disadvantages.

On the contrary, the method of population projections does not set any implicit objective a priori. With this approach, one simply quantifies, objectively, the impact of additional mortalities. The decision as to whether or not this impact is acceptable is then made independently of its calculation.

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[23] ERC stands for "Avoidance, Mitigation and Compensation" ("Eviter, Réduire, Compenser" in French). French law states that 100% of the impacts caused by the building of an infrastructure should be avoided or at least mitigated; and any residual impact, after mitigation, should be compensated for (L.110-1 and L. 122-3 CE).

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It seems preferable to separate the calculation stage (objective approach) and the decision stage (partly subjective) as they are not the same thing (science vs. societal/political choice).

The method of population projections is also better suited to the ERC ("Avoid, Mitigate, Compensate") sequence as it provides a quantitative measure of the expected impact.

Furthermore, the reliability of the PBR approach has also been questioned on a technical level, as it is based on theoretical relationships and simplistic assumptions (Green et al. 2016, Horswill et al. 2017, O'Brien et al. 2017, Punt et al. 2020, Schippers et al. 2020). To summarise what has been detailed above, the result provided by the PBR will only be valid if:

- the population is in an optimal growth regime, which excludes, de facto, stable and declining populations;

- individuals are not subject to any other anthropogenic threats, thus excluding virtually any bird population (BirdLife International 2018).

These restrictions do not apply to population projections, as with this method the assumptions are entirely under the control of the user. The flexibility of this approach also allows for more realism (age structure, stochasticity, various forms of density-dependence) to be included in the modelling of demographic trajectories, which is absolutely not possible with PBR (Horswill et al. 2017, O'Brien et al. 2017).

# 3. Delineation of the population: spatial scale and cumulative impacts

As mentioned above, in order to assess a demographic impact, it is essential to provide information on the size of the target population, i.e. the population to which individuals at risk of collision belong. This requires a clear definition of the target population and its boundaries. This exercise is often difficult because individuals of a species are usually not spatially distributed in isolated clusters, and the degree of interaction (genetic exchanges, etc.) between individuals is not known. Moreover, there is no consensus in the scientific literature on the definition, even theoretical, of a population in ecology (Debouzie 1999; Berryman 2002). A very generic definition considers the population as a "group of individuals of the same species" (Berryman 2002; Jax 2006), but a more precise definition is needed to reflect the emergent properties specific to the population scale, which cannot be apprehended at the individual or species scale (Debouzie 1999). Some define it as "a group of individuals of a given species living in a particular locality whose spatial limits are determined by natural barriers [...]" (Andrewartha & Birch 1954; Den Boer 1968); or as "a group of conspecific individuals occupying a natural habitat [...] and whose dynamics are largely dominated by survival and fecundity processes" (Huffaker 1999; Berryman 2002).

The idea of *population* is thus a loosely defined concept, and the relationships between individuals of the same species leading to *population functioning* are in fact expressed at different spatial scales. Four scales can be distinguished: the individual, the local population, the regional population and the global population (Goodwin & Fahrig 1998). The two extreme scales, the individual and the global population (worldwide distribution of the species), are not relevant for an analysis of the demographic impact of collisions with wind turbines; only the local and regional scales are relevant. The local population includes only those individuals with a high probability of encountering each other, much higher than the probability of interacting with individuals from another population

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cluster. At this scale, we therefore choose to ignore "long-distance[24]" inter-individual relationships (i.e. dispersal) because they are too infrequent. The regional population scale includes all local population clusters within a particular region, thus including a significant fraction of dispersal movements, even those rare ones occurring at fairly long distances. In this regard, it is very equivalent to the notion of metapopulation. The distinction between local and regional scales is thus determined by the amount of interactions between individuals or groups of individuals, and is therefore dependent on the dispersal capabilities (dispersal distance distribution) of the species considered. The notion of dispersal distance is therefore central to the definition of a population.

Since the levels of inter-group exchange (immigration, emigration) are almost never known, it is not realistic to try to explicitly model immigration and emigration processes at the regional metapopulation level in a *demographic impact analysis*. We must thus make a choice of scale (local or regional) and apply a demographic model in which the population is assumed to be 'closed', in terms of emigration/immigration, at that scale. This choice of scale is partly subjective, but it should be chosen to reflect the issues at stake in the wind energy project being assessed and the conservation objectives for the species under study. For example, if the impact assessment focuses on a single wind farm and/or if all population clusters have a strong value (ecological, functional, cultural, aesthetic; Sanderson 2006), then a 'local' scale analysis is appropriate. If, on the other hand, one is interested in the cumulative impacts of several wind farms, and the objective is to conserve the species on the scale of an administrative or ecological region, then the (meta-)population contours should be delineated on a regional scale. It should be noted that the population projection method allows, without any particular difficulty, to evaluate the cumulative impact of several wind farms at any scale. The difficulty lies not so much in the method of analysis as in the access to the necessary data (fatalities on several farms and species headcounts over a fairly large area).

This choice of scale must nevertheless be made with caution. Indeed, if the objective is to evaluate the impact of local collisions on a specific population cluster, certain biases may be induced by the choice of a spatial scale that is too large or too small. If a small population cluster is taken as being a single closed population (choice of a very local scale), this amounts to ignoring the exchanges that may exist with individuals located outside of that particular population cluster. In this case, the projections will tend to overestimate the local impact, as it ignores the fact that external exchanges will partly compensate for local losses via immigration (compensation effect). Moreover, this will lead to completely ignoring the fact that local collisions also have an indirect impact on these other population clusters that export individuals to the focal cluster ("draft" effect). By defining a larger population, on a regional scale, we can avoid these biases, but we will lose resolution because the estimated impact will not necessarily accurately reflect the losses suffered locally by a specific population cluster. For example, one could very well have situations in which the impact on the regional population is low, but the impact on a specific cluster would be much greater and could even lead to a local extinction. As mentioned above, in the case of a regional scale analysis, it will be important to consider the cumulative impacts of all wind farms in the defined region. In any case, the choice of the area to be considered for the population delineation should always lead to the inclusion of all wind farms located inside the area into the impact analysis.

Once the relevant spatial scale has been chosen, there are two possible approaches to setting the spatial boundaries and defining the population to be considered in the analysis: (i) if sufficient knowledge is available, the contours of a coherent ecological unit for the species under study in the region or locality concerned can be used; otherwise, (ii) spatial boundaries can be set based on the dispersal distances (statistical distribution) of the species in question. Work on this issue has been

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[24] The concept of "long distance" will not be the same from one species to another. This distance will depend on the movement capabilities and behavior, including dispersal, of the species.

carried out within the MAPE project and has resulted in a set of specific recommendations that will be made available to the public[25].

# 4. Conclusion

In view of this critical analysis, we conclude that the population projection method is much better suited to the objective of assessing the consequences of wind turbine collisions on bird populations than the PBR. The analytical method developed within the framework of the MAPE project, which has been embedded in a computer program (EolPop) freely available online[26], is based on this approach.

This tool allows to carry out demographic simulations to compare the trajectory of a population exposed to collision fatalities with the trajectory of a reference scenario, without additional mortality (see Figure 3). The impact is quantified using the relative difference between these two scenarios in terms of predicted population size after several years (e.g. 30 years), depending on the expected operating time of the wind farms concerned. The demographic model used includes (i) age structure, (ii) environmental stochasticity and (iii) demographic stochasticity in the case of small populations. Regarding mortality compensation, a compensatory density-dependence relationship has been included but only for growing populations as this is the only case where the relationship can actually be assessed (Horswill et al. 2017, O'Brien et al. 2017).

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[25] These recommendations can be found here: https://mape.cnrs.fr/autres-telechargements/applications/
[26] The EolPop program is available here: https://shiny.cefe.cnrs.fr/en_eolpop/

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