



Short communication

Wind turbines impact bat activity, leading to high losses of habitat use in a biodiversity hotspot

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ABSTRACT

Previous studies have mainly focused on bat mortality through collision by wind turbines, and very few studies have assessed the indirect impacts on bat activity and on foraging habitat availability. Also, there is a global lack of knowledge on the vulnerability of tropical bat fauna due to wind energy production, even though it is well known that windpower can affect bat communities and biodiversity hotspots are widespread in the tropics. We present one of the first studies to quantify the indirect impact of wind farms on insectivorous bats in tropical hotspots of biodiversity. Bat activity was compared between wind farm sites and control sites, via ultrasound recordings at stationary points. The activity of bent winged bats (*Miniopterus* sp.) and wattled bats (*Chalinolobus* sp.) were both significantly lower at wind turbine sites. The result of the study demonstrates a large effect on bat habitat use at wind turbine sites compared to control sites. Bat activity was 20 times higher at control sites compared to wind turbine sites, which suggests that habitat loss is an important impact to consider in wind farm planning. We strongly recommend that the loss of the foraging habitat loss is considered in mitigation hierarchy (avoiding, reducing, offsetting) when compensating for negative impacts of wind farms.

1. Introduction

Global wind capacity grew by 16.1% between mid-2015 and mid-2016 (<http://www.wwindea.org/2016/>). The wind energy industry is well known to impact biodiversity, mainly for birds and bats. The impacts are both direct, such as mortality due to collision or barotrauma (Rydell et al., 2010; Grodsky et al., 2011; Huso et al., 2016), and indirect: habitat loss, behavioral changes and reduced population viability (Zimmerling et al., 2013; Arnett and May, 2016; Frick et al., 2017). Yet various factors make it difficult to ascertain bat fatality rates (Zimmerling et al., 2013), while the indirect impacts are often overlooked (Minderman et al., 2012; Arnett and May, 2016). Today, while wind turbines are recognized to be one of the primary causes of bat mortality (O'Shea et al., 2016), it is still direct mortality that is principally assessed (Arnett et al., 2016; Schuster et al., 2015). Indirect impacts on bats include the destruction of habitat and roosts during the construction of the wind farm and behavioral changes caused during the operational phase (Arnett and May, 2016; Frick et al., 2017). Currently, only 4 studies (Barré et al., 2017; Millon et al., 2015; Minderman et al., 2012, 2017) have dealt with the assessment of habitat loss due to wind farms or individual wind turbines. All these studies, carried out in

Europe, reported a reduced bat activity within a wind farm compared to outside the wind farm. While most studies on the impact of wind turbines on bats come from Europe or North America (Arnett et al., 2016; O'Shea et al., 2016; Frick et al., 2017) a recent review (Arnett et al., 2016) highlighted the need to assess the vulnerability of tropical bat fauna. These assessments should be performed preferably before extensive wind facilities will be planned particularly because the ecological requirements and sensitivity to new human pressure are relatively poorly known, for the species involved. However, wind farm are already operational in tropical islands and many more installations are planned, in order to make such isolated territories energetically autonomous (Weisser, 2004). Our study aims to evaluate the indirect impact of wind turbines on bat activity in New-Caledonia. Using a paired-site design sampling, we compared bat activity measured in sites close to wind turbine to control sites (i.e. site outside the wind farm and including the same habitat cover). Bat activity was assessed using ultrasound recordings at stationary points. New Caledonia is one of the world's smallest biodiversity hotspots and hosts nine species of bat, six of them endemic (Myers et al., 2000). Of the Pacific islands, New-Caledonia has the highest electricity needs, mainly due to nickel mining (Hourçourigaray et al., 2014). In 2015, only 2% of its total energy

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demand was covered by wind energy, but new projects are underway in order to increase the part of renewable energy in the New-Caledonia electricity production (<http://www.isee.nc/>) thus assessment of wind turbine impact on wildlife is urgently needed.

2. Materials and methods

2.1. Sampling design

The two wind farms studied were localized on the summit of two Neo Caledonian medium mountain (up to 375 m high for Prony and up to 216 m for Mont Mau). The wind farms were made up by 66 wind turbines (Supporting Information 1), 31 of which were 50 m high, in operation since 2004–2005, and 35 of which were 55 meters high, in operation since 2006–2007 (Vergnet, 2016). Wind turbines were placed along the ridges of this mountain ((Supporting Information 1), on ultramafic soil with naturally low or no vegetation (<http://explorateur-carto.georep.nc/explorateur-carto/>). Remnants of forests and maquis vegetation are found in the valley (<http://explorateur-carto.georep.nc/explorateur-carto/>). To assess the potential disturbance from the wind turbines, we used a paired survey design to quantify bat activity at sites close to wind turbine (WT) and sites outside the wind farm (control). Each control sites was positioned in the vicinity of a site close to wind turbine (700 ± 140 m from the wind farm, minimum distance between a control site and a wind turbine was 170 m). Recorders (full-spectrum bat detector SM2BAT Wildlife Acoustics Inc. USA) at wind turbine sites were placed 15 m from the wind turbine. Microphones were placed 2.5 m from the ground. Two sites within a pair were 996 ± 210 m apart (238 m minimum). Upstream, we have paid a particular attention to ensure that control sites had similar altitude and habitat cover than WT sites (no significant difference was detected between WT sites and control sites: Wilcoxon test: $W = 24$, p -value = .43 for the altitude, Kruskal test: $ks = 2.649$, $P = .10$; $ks = 0.820$, $P = .37$; respectively for forest cover in a radius of 200 and 1000m; Kruskal test: $ks = 1.118$, $P = .29$; $ks = 0.365$, $P = .55$; respectively for open land cover in a radius of 200 and 1000m). One or two pairs were sampled during the same night. To ensure the independence of the data, the minimum distance between two sampling points was 250 m. Eight pairs in total were sampled. Fieldwork was carried out during two periods: May–June 2015 (cold season) and November 2015 (dry season). Each pair was sampled once per period thus the dataset included 32 efficient recording samples.

2.2. Bat activity

We used echolocation recordings at stationary points to sample bat activity, the most widely used method for a standardized insectivorous bat activity survey (Stahlschmidt and Brühl, 2012). We recorded ultrasounds higher than 8 kHz during the three first hours after sunset, with Song Meter 2 Bat detectors (<http://www.wildlifeacoustics.com/>). The beginning of the night was chosen because it is known to be when bat activity first peaks (Froidevaux et al., 2014; Griffiths, 2007; see Supporting Information 2). Focusing on the first three hours of the night also allows optimization of the time resources allocated to audio file analysis. We used Kaleidoscope (version 4.0.0, <http://www.wildlifeacoustics.com/>) and Scan'R (version 1.7.6, <http://binaryacoustictech.com/>) to isolate bat calls. All files labeled as containing bat calls were manually verified with Batsound (version 4.03, <http://www.batsound.com/>). Calls higher than 50 kHz were assigned to the genus *Miniopterus*, and calls between 30 kHz and 40 kHz were assigned to the genus *Chalinolobus* (Kirsch et al., 2002). In southern New-Caledonia, *Miniopterus* genus is composed of the little bentwinged bat (*M. australis*) and the small melanesian bentwinged bat (*M. macrocneme*), considered as least concern and as data deficient respectively (IUCN, 2017). Although the small melanesian bentwinged bat is believed to occur across a very wide range, the difficulties inherent in its

identification prevent an accurate evaluation of trends and threats for this species. Locally, bentwinged bats are considered as vulnerable (Kirsch et al., 2002). *Chalinolobus* genus is only composed of the New-Caledonia wattled wat (*C. neocaledonicus*, Kirsch et al., 2002), a New-Caledonian endemic species listed as endangered (IUCN, 2017), due to a constant decline in the extent and the quality of its habitat. One other species, the New-Caledonia Long-eared bat (*Nyctophilus nebulosus*), an endemic critically endangered species (IUCN, 2017), may have been present in the study area (Parnaby, 2002) but was not detected in our recordings. As it is impossible to determine the number of individual bats from their echolocation calls, we calculated a bat activity metric (bat passes), calculated from the total number of contacts per genus during the three first hours after sunset. A bat pass is defined as a single or several echolocation calls during a five second interval (Millon et al., 2015). If it was obvious that several bats emitted calls at the same time, an additional contact was counted.

2.3. Statistical analyses

Generalized Linear Mixed Models (GLMM) were used to assess variation in bat activity (i.e., the response variable) as a function of site type (WT versus control, the explanatory variable, R package *glmmADMB*, Fournier et al., 2012). Due to the nature of the data (i.e.: count data) and its overdispersion, we performed GLMMs (package *glmmADMB*) with a negative binomial error distribution (Crawley, 2009). Due to the difference in detection distance for *Miniopterus* and *Chalinolobus*, per genus modelling was performed. In view of the hierarchical sampling design (up to two pairs – i.e. a pair means one WT site and one control site – were sampled at the same date) and the repetition of the sampling during two seasons, we used a nested random effect: pairs within recording sessions and season. Since bat activity can depend on landscape variables, we add as fixed effect landscape co-variable: the percentage of open area (with little or no vegetation) and the percentage of forest area each calculated at 2 different buffers of 200 and 1000 m (<http://www.oeil.nc/geoportail>, see Supporting Information 3). To avoid over-parametrization, we only include one co-variable in models and choose the best model (among the 4 model: open land cover and forest cover at 2 radius distance), according to AIC criterion. Thus, our statistical models were structured in the following way:

[Bat activity] ~ site type + co- variable + 1 | Season/Date/Pair

where bat activity could be either *Miniopterus* or *Chalinolobus* activity.

We performed variance-inflation factors (VIF) on each model (Fox and Monette, 1992). All variables showed a VIF value of < 3 and the mean of VIF values was < 2, meaning there was no striking evidence of multicollinearity (Chatterjee et al., 2000). All analyses were performed with R statistical software (R Core Team, 2016).

3. Results

In total, more than 3 000 contacts were recorded at wind turbine sites (WT) and sites outside the wind farm, the majority of them emitted by *Chalinolobus* (80%). Both genera were present during the two seasons, with more contacts recorded during the dry season (72%) than the cold season. For both *Chalinolobus* and *Miniopterus* the landscape co-variable selected was the percentage of open area, with a significant negative effect.

For both *Chalinolobus* and *Miniopterus*, wind turbines induced a significant difference in bat activity (Table 1, Fig. 1). The mean activity of *Chalinolobus* was tenfold lower at WT sites than at control sites (mean observed values: Control: 147 [min: 0–max: 790]; WT: 14 [min: 0–max: 120]; mean predicted values: Control: 265 ± 72; WT: 21 ± 4), while the mean activity of *Miniopterus* was, 20-fold lower at WT than at control sites (mean naïve values: Control: 39 [min: 0–max: 240]; WT: 2

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