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ORIGINAL ARTICLE

Wildlife and infrastructure: impact of wind turbines on bats in the Black Sea coast region

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Abstract

In Eastern Europe, wind energy production is currently promoted as an important source of renewable energy, yet in most cases without appropriate consideration of the negative impacts wind turbines (WT) may have on protected species such as bats. Here, we present first data on fatality rates, fatality factors and the likely origin of bats killed by WT in the Dobrogea region (Romania), located in a major migratory corridor for wildlife in Eastern Europe. Over a 4-year period, we found a total of 166 bat carcasses from 10 species, mostly representing migratory species such as *Pipistrellus nathusii* and *Nyctalus noctula*. Most fatalities at WT occurred in July and August. We documented 15 cases of barotrauma and 34 cases of blunt-force trauma in carcasses found below WT. After adjusting for carcass removals and variations in searcher efficiency, we estimated for the 4-year study period a total of 2394 bat casualties at the studied WT facility consisting of 20 units, resulting in a mean fatality rate of 30 bats/WT/year, or 14.2 bats/MW/year. By implementing a curtailment measure at wind speeds below 6.5 m/s, we reduced fatality rates by 78%. Isoscape origin models based on hydrogen stable isotope ratios in fur keratin revealed that the majority of *N. noctula* that were killed by WT or captured nearby in mist nets originated from distant areas in the North (Ukraine, Belarus, Russia). The estimated high fatalitjegangy rates of bats at WT in this area have far-reaching consequences, particularly for populations of migratory bats, if no appropriate mitigation schemes are practised.

Keywords Bat migration · Wind energy · Infrastructure · Post-construction monitoring · Stable isotopes · *Nyctalus noctula*

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Introduction

Intensive development of infrastructures represents one of the major global drivers of biodiversity loss (Benítez-López and Verweij 2010). Many studies have demonstrated biodiversity loss in proximity to infrastructure facilities (Spellerberg 1998; Nellemann et al. 2001; Fahrig and Rytwinski 2009; Benítez-López and Verweij 2010). Nevertheless, production and distribution of new energy infrastructures worldwide are in high demand, as the human population continues to grow (Sánchez-Zapata et al. 2016). At the same time, national energy production schemes shifted from fossil to renewable sources, such as wind energy, which has emerged over the past decades as potentially very important worldwide (Dincer 2000).

The European Union (EU) has initiated a directive (2009/28/EC) to substantially reduce carbon emission rates in order to counter the threats of global climate change. As part of this action, wind energy production is promoted in EU countries, especially in those where energy production from renewable sources was only of minor or no importance in the past, such as in Eastern Europe. However, operating wind turbines (WT) may kill wildlife species, such as bats (Cryan and Barclay 2009; Hayes 2013; Voigt et al. 2015; Arnett et al. 2016; O'Shea et al. 2016). A recent meta-analysis about mass mortalities in bats confirmed that WT are one of the leading causes for multiple mortality events in bats (O'Shea et al. 2016). Cumulative numbers of casualties at WT may impact bat populations on a large geographical scale (Frick et al. 2017), since individuals may originate from near and distant source populations (Voigt et al. 2012; Baerwald et al. 2014; Lehnert et al. 2014). Therefore, WT may add substantially to local and regional biodiversity losses.

Within the EU, all bat species are legally protected based on the Habitat Directive 92/43/CEE (Annexes II and IV). Therefore, post-construction monitoring and mitigation measures, such as increased cut-in speeds of wind turbines, that proved to be effective in reducing fatality rates in central Europe and elsewhere (Arnett et al. 2011; Brinkmann et al. 2011), should be mandatory in all EU countries. Bats are also protected by the 'UN Convention on the Conservation of Migratory Species of Wild Animals' (CMS, Bonn 1979; London, 1991). Recently, EUROBATS, the European agreement of the CMS convention which is targeting specifically migratory bats, has received feedback to a survey which was sent out to all 63 range states of the EUROBATS agreement, including almost all EU countries. According to this survey, post-construction monitoring of bat activities was practiced only in 11 out of a total of 63 EUROBATS agreement range states (17%), probably owing to a lack of administrative power to support the EUROBATS guidelines and the implementation of Habitat Directive 92/43/CEE at times when wind energy production expands at an unprecedented rate (Rodrigues et al. 2017). Furthermore, in 13 range states, mitigation measures are regularly commissioned, but only in 7 states mitigation measures are being monitored for their efficiency (Rodrigues et al. 2017). Considering the deficiency in the implementation of Habitat Directive 92/43/CEE in many EU member states and of the EUROBATS guidelines in many of the agreement range states, there is a large uncertainty regarding the impact of operating WT on bats and whether or not mitigation measures, if practised at all, are efficient to protect bat species on a local and continental scale.

Over the past decade, the western part of the Black Sea region witnessed a rapid development of the wind energy sector. This infrastructure expansion was facilitated by the favourable wind conditions in combination with areas that have a low human population density, which mitigates the potential conflict between wind energy companies and local communities (Croitoru and Zamfir 2014; Nghiem et al. 2017). Information on post-construction fatality rates of bats at WT in Eastern Europe is very limited (Carmen and Chachula 2013; Vlaschenko et al. 2013; Măntoiu et al. 2015; Rodrigues et al. 2017), even though the assessment of such negative effects is crucial for ensuring the favourable conservation status of bats as requested by the EU Habitat Directive (92/43/CEE).

The western coast of the Black Sea has been called Via Pontica for its preeminent value as a migratory corridor for wildlife (Moreau 1972). Several migratory bat species have been observed in this area, including *Nyctalus noctula*, *Pipistrellus pipistrellus* s.l. and *P. nathusii* (Strelkov 1969; Panutin 1980; Hutterer et al. 2005; Gashchak et al. 2015). Yet, our knowledge about the connectivity between breeding and hibernation areas of these long-distance migratory species remains very scant for Eastern Europe, since recaptures of banded bats have rarely been reported for this part of Europe (Strelkov 1969; Panutin 1980; Strelkov 2002; Hutterer et al. 2005).

Our study aimed at collating first information about migratory bats in general and the wind energy-bat conflict in particular for the understudied Eastern Europe region (Dobrogea, Romania). We aimed at measuring fatality rates at WT situated in the Black Sea region to test if peak fatality rates would coincide with migration activity and if fatality rates in this area are higher than those reported for central Europe. Moreover, we aimed to assess the likely cause of death of bats at WT and estimated the origin of casualties of common noctules (*N. noctula*) as one of the most abundant species affected by WT infrastructure in Europe (Voigt et al. 2014). Based on previous banding data, we hypothesised that the western part of the Black Sea region is a migratory route for bats from northern areas of the Ukraine, Belarus and the European part of Russia (Strelkov 2002; Vlaschenko et al. 2016) and that WT operating without a mitigation scheme might kill large numbers of bats in this region, yet we assumed that elevated cut-in speeds will reduce the fatality rates significantly (Arnett et al. 2011; Brinkmann et al. 2011). Therefore, we quantified the efficacy of curtailment measures to assess if increased cut-in speeds might reduce the fatality rates of local WT.

Methods

Study site

We carried out our study in the western part of the Black Sea region, Northern Dobrogea, Romania, Eastern Europe Region, at the Babadag Wind Park (44° 53′ 0.88″ N, 28° 44′ 51.37″ E, Fig. 1). All work was conducted under the permit #3660/22.11.2012 issued by the Natural Monuments Commission of the Romanian Academy. The studied WT facility consisted of 20 WT (2.1 MW/WT, type Suzlon S88, commissioned in 2011), with a nacelle height of about 79 m, a rotor diameter of 88 m and a total rotor swept area of about 122,000 m^2. The turbines were distributed over two separate areas located near the town of Babadag (Babadag 1—western block—16 turbines, Babadag 2—eastern block—4 turbines), within a silvo-steppe hilly landscape, with small limestone outcrops. The turbines were located on pastures, surrounded by farmland and vineyards. In the north and south, the facility is flanked by two Natura 2000 sites of community importance (ROSCI0065 and ROSCI0201), one consisting of deciduous temperate forests and the other of a lake with loess cliffs. The latter is also included within the UNESCO Danube Delta Biosphere Reserve.

Carcass searches

We conducted our study over a period of 4 consecutive years (2013–2016), taking into account periods when WT operated without and with curtailment. Over the study period, we performed weekly systematic bat carcass searches, between April and November. Searches were done below all WT. Two days were assigned for each of the 30 annual visits, searching 10 WT per day, with a total effort summed up to 240 days in the field and 2400 bat carcass surveys. At each WT, we searched for carcasses within a rectangular area (141 m diagonal or 100 m length per side). A team of 4 searchers walked along transects with a distance of 5 m between adjacent transects (2.5 m/side). In some cases, transects varied slightly in length or distance to adjacent transects due to vegetation cover or because of the topography, e.g. steep slopes. Small steel poles, which were capped by red plastic tubes for better visibility, were installed in each corner of the rectangular search area, in order to ensure consistency across all surveys. The spatial position of poles and carcasses was measured using a Trimble GeoXH (DGPS, Differential Global Positioning System) with sub-meter precision. Each time a carcass was found, its visibility in the field was classified using the percentage of vegetation cover. This was obtained using a 1-m^2 reference grid with a 10-cm^2 mesh. The classes were split into easy (< 10%, RP), moderate (< 50%, M) and difficult (> 50%, D). The distance from the carcass to the WT was measured using a telemeter (Nikon Forestry 550T) with slope correction function.

Estimation of fatality rates

As advised by the EUROBATS WT guidelines (Rodrigues et al. 2014), we applied different fatality estimators in order to predict the number of bat fatalities at the studied WT: GenEst v.1.4 - USGS (Dalthorp et al. 2018) and Fatality Estimator USGS (Huso et al. 2012). We have quantified carcass persistence (CP), searcher efficiency (SE) and detection probability (DP) by laying out 30 *Mus musculus* carcasses three times per season (spring, summer and autumn). This sums up to a total of 36 search trials over a period of 4 years. Akaike information criterion (AIC) was used to determine the best model for carcass persistence and searcher efficiency trials (Burnham and Anderson 2002). Density-weighted proportion (DWP) was used in order to identify the proportion of total carcasses which are expected at each individual WT, taking into account the distance from the WT and the aforementioned visibility classes in the search area. This enabled us to reduce overestimations, when assuming the same dispersion of carcasses within the search plot (Huso and Dalthorp 2014). The estimated number of carcasses/MW/year was obtained by dividing the annual site-specific fatality estimate by the maximum-rated capacity of the wind park, expressed in MW. We chose mice carcasses as test subjects for accessibility and similarity in size and colour with bat carcasses. The test carcasses were classified as pathogen-free and were obtained from the "Cantacuzino" National Institute for Medical and Military Research and Development, Romania, which produces them for research purposes. No animals were killed specifically for our study. The trials were performed under random WT with various degrees of vegetation cover (open spaces, pastures and crops). The test carcasses were classified according to their visibility as mentioned for the bat carcasses. In order to minimise bias, when manually laying out the test carcasses, we used a spatial random point function (ArcGIS 10.3), generating plots of 30 points within a survey unit. Accurate field positioning of each test carcass was ensured using the DGPS. Multiple origin reference points were selected for each search area, and survey shots were recorded for each planted carcass, using a compass with a clinometer (Suunto Tandem 360PC/360R G) and an optical telemeter. After mice carcasses were brought out randomly within the search area, each searcher who had not yet

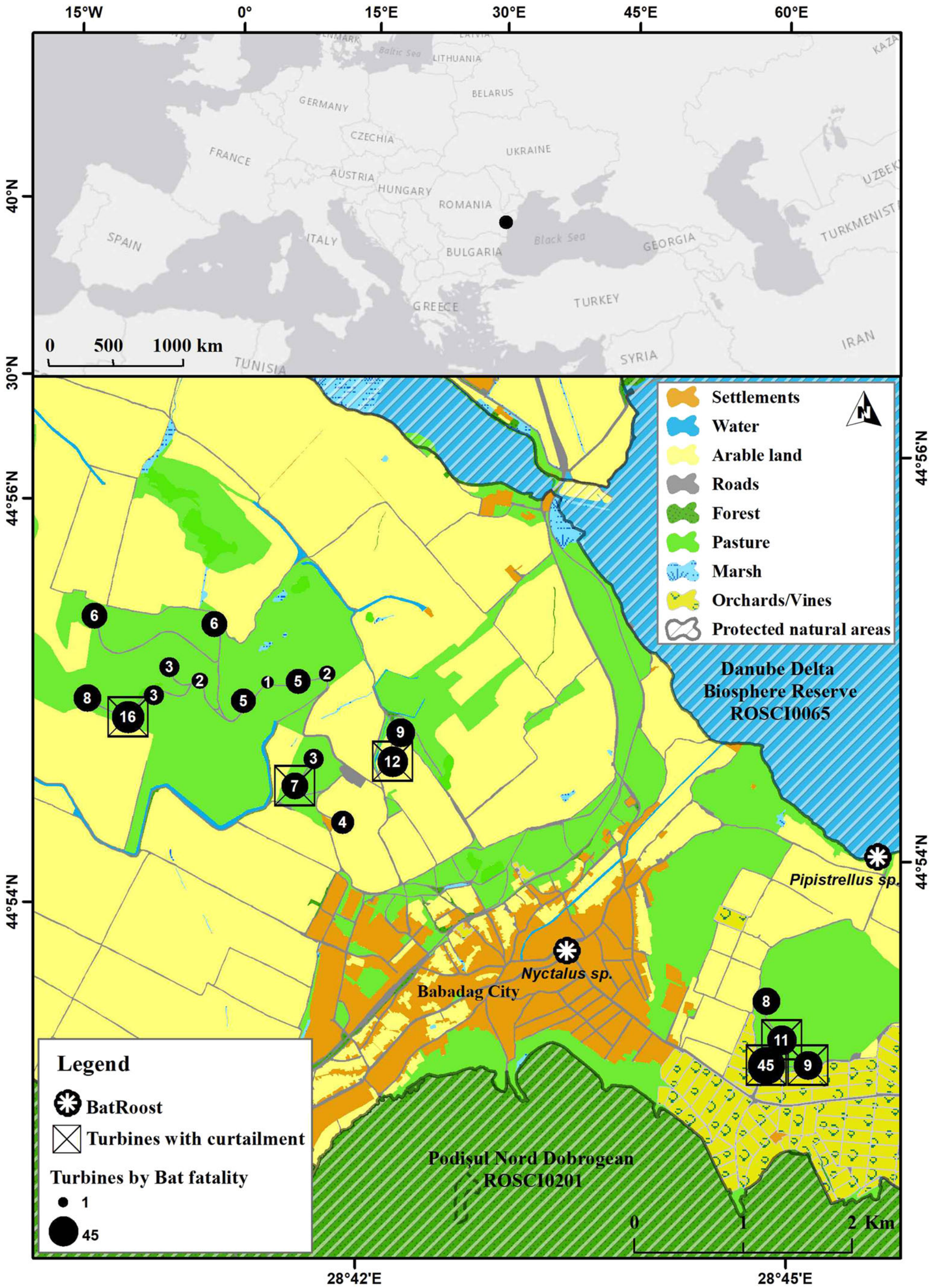

Fig. 1 Upper section—the study site location (black marker) in southeastern Europe; Lower section—map of the Babadag Wind Park; number of bat carcasses recorded per WT and the WT where mitigation measures were applied

participated in such a survey event was tested. For this, we left test carcasses in the field and checked their presence periodically at intervals of 1, 2, 7, 8, 14, 15, 21 and 22 days, in order to identify the time at which they were carried away, consumed by scavengers or disappeared from the site due to other reasons such as rain or wind. Scavengers were identified to the species level in spring, summer and autumn using two camera traps (LTL Acorn 6210 MC Trail Cameras mounted on 1.5-m wooden poles). The cameras were left in the field for 22 days per season in a test plot, for each monitoring year.

Pathological evaluation of carcasses

All bat carcasses were brought to the Faculty of Veterinary and Agricultural Sciences, University of Bucharest (Romania), where necropsies were performed in well-preserved specimens to determine the cause of death. The individuals that were found alive under WT were brought to the "Visul Luanei" Animal Rehabilitation Centre in Bucharest (L'ARC). We collected fur samples from the dorsal pelvic region of all carcasses. The time of death was difficult to establish with high accuracy due to the small body size of animals (Kunz and Parsons 2009). Carcasses were classified as fresh if no maggots or signs of decomposition (smell, deformation) were observed, according to the method described by Rollins et al. (2012). Most of the carcasses were identified at species level, using standard identification keys (Dodelin 2002; Dietz and Helversen 2004). In some cases, we were not able to report the species, sex and age (juvenile, adult) due to the different stages of decomposition or due to partial consumption by scavengers (Appendix 1). The sex distribution was estimated using a binomial test (function “binom.test”) from “stats” package in R 3.6.0 (R Core Team 2020). When possible, we assessed the age of carcasses based on species-specific information, biometrics, the degree of fusion of the phalangeal epiphyses and dental rigidity of the canine teeth (Haarsma 2008).

Bat mist-netting

Nyctalus noctula and *Pipistrellus nathusii*, two species with high fatality risk at WT in Europe (Rydell et al. 2010), were mist-netted to improve our knowledge on the phenology of migration and the likely origin of these bats for the study area. Live animals were treated following published ethical guidelines (Sikes and Gannon 2011). We set up 12-m × 3-m monofilament mistnets between poles during whole nights. The mist-netting sites were located 129 km south from Babadag Wind Park, near Băneasa village, during 12 and 13 October 2014, and included areas with known roosts of *N. noctula*. We collected fur samples from the dorsal pelvic region of 18 captured bats from the study site.

Stable isotope analysis

Fur samples from carcasses were sorted according to the estimated time of death of the individual. Only fur samples from carcasses which had remained < 7 days in a field were used for isotope analysis. The fur samples were analysed at the stable isotope laboratory of the Leibniz Institute for Zoo and Wildlife Research (Germany). Samples were washed with 2 : 1 chloroform-methanol solution and then placed in an oven for 10 days in order to dry, at a temperature of 50° C. Afterwards, fur samples were weighed using a microbalance (Sartorius ME5, Göttingen, Germany) to a subsample of 0.274 mg ± 0.01 mg and transferred singly into silver-foil capsules (IVA Analysetechnik e.K. Meerbusch, Germany). The samples were flushed for at least 1 h with chemically pure helium (Linde, Leuna, Germany) in the autosampler (Zero Blank autosampler; Costech Analytical Technologies Inc., Italy) of the elemental analyser (HT Elementaranalysator HEKAtech GmbH, Wegberg, Germany). For further analyses, we used a Delta V Advantage isotope ratio mass spectrometer (ThermoFischer Scientific, Bremen, Germany) that was connected via an interface (Finnigan Conflo III, ThermoFisher Scientific, Bremen, Germany) with the elemental analyser. All stages of the laboratory procedures and laboratory standards are described in detail in Popa-Lisseanu et al. (2012), Voigt et al. (2012) and Lehnert et al. (2014). Values were calculated in the delta notation with per mill deviation from the international hydrogen standard V-SMOW as the unit of measurement. Based on the analysis of keratin laboratory standards, we estimated the analytical precision to be ± 3 per mil (one standard deviation).

Modelling the origin of bats based on δ^2H of fur keratin and data analysis

Our isoscape origin model was based on stable isotope ratios in the non-exchangeable portion of hydrogen (δ^2H_f) in fur keratin of common noctules (Sullivan et al. 2012; Voigt et al. 2012; Voigt et al. 2014). We selected this species because it migrates over long distances (Lehnert et al. 2018) and because it is more active during spring and the end of summer in the Dobrogea area (Măntoiu et al. 2015). For comparison, we used fur samples from *N. noctula* bats captured with mist nets in southern Dobrogea, as a reference dataset to compare predicted breeding origins of deceased versus alive conspecifics. In Eastern Europe, stable hydrogen isotope values in precipitation water (δ^2H_p) are known to follow a latitudinal gradient with δ^2H_p values increasing from north to south (Bowen et al. 2005). Stable isotope ratios of non-exchangeable hydrogen in inert keratin tissues, such as fur, reflect the variation of stable isotope ratios in precipitation assimilated along the food chain (Hobson 1999; Popa-Lisseanu et al. 2012) and can be used to trace movement of migratory wildlife. *Nyctalus noctula* usually moults in its summering area before the onset of migration (Ilyin 1990; Cryan et al. 2004; Blohm and Heise 2008; Fraser et al. 2013). We used the R-package IsoriX (Courtiol et al. 2019) and the transfer function which was recently established for common noctule bats (Lehnert et al. 2018), which incorporates data points from central and Eastern Europe.

All statistical analyses were performed with R 3.3.2 environment. We excluded geographic areas with elevations higher than 500 m a.s.l. from our predictive maps, using an overlaying polygon on the map, because *N. noctula* forms maternity colonies predominantly in lowland areas (Dietz et al. 2009). We have chosen a map extent which engulfs banding data for common noctules in Eastern Europe (Hutterer et al. 2005).

Testing of curtailment measures

During the last 2 years of the monitoring scheme (2013–2014 = period 1, no curtailment, 2015–2016 = period 2, with curtailment), we tested the efficacy of a mitigation design to reduce fatality rates. The mitigation scheme consisted of raising cut-in speeds of WT by 30%, from 4 to 6.5 m/s. This scheme was applied in high-risk periods covering the summer and autumn migration (July 15 to September 15), when most carcasses were found during period 1. Also, selected WT generated the highest number of bat fatalities during period 1, summing up to half of the total fatality values of the wind park. The daily operational time was set to start half an hour before sunset and to end half an hour after sunrise. The WT were programmed to stop if the wind speed was below the threshold value (< 6.5 m/s), as measured by real-time climatic sensors installed on the nacelle of each turbine. To estimate the effect of the mitigation measures, we applied a general linear model (GLM) with before-after-control-impact (BACI) design. For that, we selected data for the period of increased fatality from the 15th of July to the 15th of September, for all the 4 years of the study. The first 2 years (2013–2014) were considered "before" and the last two (2015–2016), the "after" input data. Next, we called the WT to which mitigation was applied as an "impact" and all the WT without mitigations measures as a "control". We used a GLM with Poisson distribution, where the number of bat carcasses under each turbine was a response variable and the period ("before" or "after"), the WT group ("impact" or "control") and also their interaction were the explanatory variables.

To estimate the differences in number of carcasses amongst different monitoring years and months, we used Pearson's chi square test for independence (χ^2). The level of significance was defined as 0.05 for all statistical tests.

Results

Carcass searches

During the 4-year study period, we found a total of 166 bat carcasses at Babadag Wind Park. We identified 10 bat species belonging to 6 genera (Table 1). Most carcasses belonged to the following species in decreasing abundance: *P. nathusii* ($n = 58$; 35%), *N. noctula* ($n = 45$; 27%) and *Pipistrellus sp.* ($n = 27$; 16%). We observed differences between the raw annual number of bat carcasses amongst the monitoring years (Pearson's chi square test for independence, $\chi^2 = 19.783$, $p < 0.001$): 48, 72, 21 and 25 in 2013, 2014, 2015 and 2016. We also found differences in the number of carcasses amongst months ($\chi^2 = 44.83$, $p < 0.0001$). More than 33% of carcases were recorded in August (Table 1).

We were able to identify the sex of bats in 96 out of 166 cases. Thirty-seven carcasses (38%) were females and 59 (62%) were males, which was significantly different from a balanced ratio (binomial test, $p = 0.03$). The total number of bat fatalities recorded per WT for the whole study period varied from 1 to 45 ($\bar{x} = 8.3$, $n = 20$ turbines). The majority of carcasses (44%, $n = 73$) were found in a small area of the wind park (Babadag 2, eastern block) that consisted only of 4 WT (Fig. 1).

We estimated the age of bats (juvenile and adult) in 105 out of 166 carcasses. Our survey revealed 4 juvenile individuals (*Pipistrellus* sp.) and 101 adult bats (Appendix 1).

Fifteen individuals showed signs of barotrauma, such as hemothorax, and 34 cases exhibited symptoms of blunt-force trauma such as fractures of wing bones and opened wounds, caused by direct collisions. Two bats were found alive under WT, but later died due to complications from pulmonary haemorrhage. We never observed combination of symptoms, i.e. combined barotrauma and blunt-force trauma. One carcass could not be spatially positioned due to scavenger interference while performing the carcass searches, and therefore, this carcass was excluded from the fatality estimator databases.

Carcasses were found at distances varying between 2 and 162 m from the WT ($\bar{x} = 26.7 \pm 25.7$, $n = 166$). The detection probability of bat carcasses for the entire WT facility was biased towards open areas lacking any vegetation, particularly on some parts of the platforms and maintenance roads.

The carcass persistence (CP) was modelled using a logistic distribution and a visibility variable. The time of removal averaged 10.5 ± 7.3 days, with a maximum of 22 days and a minimum of 1 day. Searcher efficiency models (SE) took into account visibility and were higher for the easy (RP) visibility class. The detection probability trials ($n = 36$, 30 carcasses per trial) resulted in 24.7% DP in open spaces (RP), 13.9% in moderately cluttered areas (M) and 8.3% in difficult areas, covered by vegetation. Trail cameras recorded the following wildlife scavenging on test carcasses ($n = 1200$, in order of decreasing frequency): stray dogs (78%), red foxes (*Vulpes vulpes*, 14%), common buzzards *(Buteo buteo*, 4%), snakes (*Natrix natrix*, 3%) and wild cats (*Felis sylvestris*; 1%).

Estimation of fatality rates

Based on GenEst and Fatality Estimator, we present two fatality estimates for each site. In total, we estimated 1581 bat carcases with GenEst and 2394 bat carcasses with Fatality Estimator. These values resulted for Babadag Wind Park in the 4-year study period, with a 95% CI and 5000 bootstraps (1198–3594 GenEst, 1765–3400 Fatality Estimator), which is the equivalent of 9.4 bat carcasses/MW/year for GenEst and 14.2 bat carcasses/MW/year for Fatality Estimator or 19.8 carcasses/WT/year for GenEst and 29.9 carcasses/WT/year for Fatality Estimator (Table 2). A detailed per species estimation can be seen in Table 3. Most of the carcasses were found in July and August.

Table 1 Bat fatality at Babadag Wind Park, 2013–2016, by species, month and season (Fatality/WT 8.3 ± 9.5, $n = 20$ turbines)

Species	Spring		Summer		Autumn			Winter	Total	
	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	No. carcasses	No./%
*Nyctalus noctula**	2	1	2	6	22	8	4		45	27.1
*Nyctalus leisleri**			2	4					6	3.6
*Pipistrellus nathusii**	16	8	6	4	14	6	3	1	58	34.9
*Pipistrellus pygmaeus**		1			*2*	*2*			5	3
*Vespertilio murinus**	1	1		4	4				10	6
*Eptesicus serotinus***				1					1	0.6
*Hypsugo savii***					1	1			2	1.2
*Pipistrellus kuhlii***		2		1	3				6	3.6
*Pipistrellus pipistrellus***	2		1		1				4	2.4
Myotis sp.		2							2	1.2
Pipistrellus sp.	1	5	1	7	8	5			27	16.3
Total	22	20	12	27	55	22	7	1	166	100
No./%	13.3	12	7.2	16.3	33.1	13.3	4.2	0.6		

Mitigation measures

The implementation of curtailment measures caused a reduction in the number of found carcasses (Table 2 and supplementary materials 1 and 2) and also a drop in the estimated fatality rates per WT from the first period with no curtailment to the second period with curtailment. One WT (T19) had relatively high numbers of carcasses ($n = 45$). The estimated fatality per WT for the entire study period varied significantly (supplementary materials 3), with most of the carcasses belonging to Babadag 2 facility (eastern block—T17, 18, 19, 20). The area also had a higher species diversity regarding fatality, including *Myotis* sp. and *Hypsugo savii*. One WT from the Babadag 1 facility (western block, T9) showed also high fatality numbers. A comparison of fatality rates across seasons (Spring: April–May, Summer: June–August, Autumn: September–November) revealed that T19 produced high numbers throughout all seasons, while the other WT showed some seasonal variation, with a higher number of carcasses observed or estimated during the migration period in summer and autumn (supplementary materials 4).

Geographic origin of bats

Based on stable isotope ratios of the non-exchangeable hydrogen in fur keratin, we defined individuals as of regional origin if the predicted geographic place of origin included the place of sampling; otherwise, individuals were categorised as migrants. For *N. noctula*, we estimated that 18 (84%—3♀, 15♂) individuals captured by mist nets and 17 (94%—8♀, 9♂) of carcasses found below WT were of distant origin and thus potentially migratory. For each of the two groups, we built a single probability map for the likely summer origin of the candidate bat (Fig. 2). According to the results of isoscape origin modelling, migratory individuals most probably originated from Ukraine, Belarus and the European part of Russia (Fig. 2).

Table 2 Fatality estimates per year using GenEst 1.4 and Fatality Estimator USGS - 5000 bootstraps, CI 95%—one carcass left out due to scavenger interference

		GENEST 1.4			Fatality Estimator			
Number.	Year	Estimation (no. of carcasses)	No. of carcass/ MW/year	No. of carcasses /WT	Estimation (no. carcasses)	No. of carcass/ MW/year	No. of carcasses /WT	Curtailment measure
1	2013	457.2	10.8	22.8	782	18.6	39.1	No
2	2014	640.5	15.2	32	904	21.5	45.2	No
3	2015	201.5	4.8	10	274	6.5	13.7	Yes
4	2016	281.4	6.7	14	434	10.3	21.7	Yes
	Total/Avg.	1580.7	Avg. 9.4	Avg. 19.7	2394.00	Avg. 14.2	Avg. 29.9	

Table 3 Estimated number of bat carcasses produced at the Babadag Wind Park, during the 4-year study period, listed according to species/groups (decreasing number); *GE* GenEst 1.4, *FE* Fatality Estimator

	Year	2013		2014		2015		2016		Total	
No.	Species	GE	FE	GE	FE	GE	FE	GE	FE	GE	FE
1	*Pipistrellus nathusii*	137	241	365	497	55	75	67	94	624	907
2	*Nyctalus noctula*	75	97	135	178	60	93	119	188	389	556
3	*Pipistrellus* sp.	88	161	24	27	78	95	46	62	236	345
4	*Vespertilio murinus*	77	146			8	12	10	17	95	175
5	*Nyctalus leisleri*			60	97			5	7	65	104
6	*Pipistrellus pygmaeus*	23	44	4	6			27	50	54	100
7	*Pipistrellus kuhlii*	35	69	17	20					52	89
8	*Pipistrellus pipistrellus*	22	26					7	20	29	46
9	*Hypsugo savii*			14	45					14	45
10	*Myotis* sp.			18	20					18	20
11	*Eptesicus serotinus*			4	7					4	7
	Total	457	784	641	897	201	275	281	438	1580	2394

Efficiency of the mitigation scheme

The recommended mitigation scheme consisted of increasing the cut-in wind speed of 6 WT (Fig. 1) to 6.5 m/s from mid-July to the end of September when peak fatality rates were observed. According to the results of the GLM model (Tables 4 and 5), the interaction between the period and the WT group was significant. The predicted number of carcasses after the mitigation measure was applied for the “impact” turbines dropped to 0.5, which is 9 times lower than the estimated value (4.66) before the implementation of the mitigation scheme. After applying this scheme during period 2, the number of carcasses recorded for the whole WT park dropped by 62% compared with that of period 1.

Discussion

Our study is the first comprehensive work conducted on bat fatality at a wind park in Eastern Europe, the Black Sea region. We demonstrated that a relatively small wind park, consisting of 20 turbines, may cause high fatalities of bats if no curtailment measures are applied. The estimated fatality rates ranged between 9.4 carcasses/MW/year (GenEst) and 14.2 carcasses/MW/year (Fatality Estimator). Given that the Fatality Estimator may overestimate the results when carcass persistence is low and search intervals are short (Barnes et al. 2018), GenEst was considered a more stable estimator for our study; thus, the following discussions take GenEst values into account.

Estimated fatality rates for the unmitigated period (2013–2014) were high: 13 carcasses/MW/year (22.9 carcasses/WT/year). These values recorded in the period without curtailment are higher than those reported for Canada (Zimmerling and Francis 2016) and most values reported for central Europe, where the highest documented values reached 7.6–10.5 bats per MW/year, equalling 18–19 carcasses/WT/year (Rydell et al. 2010). Peak values in the USA reached 10 bat carcasses/MW/year (Hayes 2013). These differences may also be caused by the use of specific estimators (Bernardino et al. 2013), by local habitat diversity, the proximity to favourable landscape features with high bat activity (cliffs, lakeshores, forest edges) and by the type of collision. Thus, in our study, one-third of all carcasses observed inside the search areas exhibited symptoms of barotrauma. Even though the proportion of proximate causes of death at WT is unknown (Rollins

Table 4 Results of GLM with BACI analysis

Number of bat mortalities			
Predictors	Estimated coefficients	Conf. int (95%)	*P* value
Intercept	0.75	0.49–1.15	0.187
Turbine: impact	6.22	3.77–10.27	*< 0.001*
Period: after	1.05	0.58–1.90	0.879
Interaction: after × impact	0.10	0.04–0.29	*< 0.001*
Observations	80		
R^2 adjusted	0.726		

Table 5 Predicted by the GLM number of bat carcasses per control and impact turbines based on the number of carcasses found

Period	Group	Predicted	Std. error	Conf. low	Conf. high
Before	Control	0.75	0.21	0.48	1.15
Before	Impact	4.66	0.13	3.59	6.06
After	Control	0.78	0.21	0.51	1.19
After	Impact	0.50	0.40	0.22	1.11

et al. 2012), we expect a bias in the detection probability of carcasses within 50-m radius of search area around the WT. Bats with injuries from direct collisions, such as broken wings, are likely to be found on the ground (Voigt et al. 2015), while the proportion of WT-caused fatalities could be significantly underestimated if some bats were exposed to mild barotrauma. The latest may have been able to continue their flight for short periods before dying (US Fish and Wildlife Service 2011; Voigt et al. 2015). The current fatality estimators do not account for this additional disappearance of affected individuals (Huso et al. 2016). Therefore, reliable numbers of affected bats per WT could be underestimated and further studies of relative proportions of trauma types obtained from WT are required. There is also a chance that the local bat populations has been reduced by the functionality of the WT in the first study period (2013–2014), or that their roosts may have been destroyed or abandoned, leading to lower results in the second study period (2015–2016).

We consider that the high values recorded for the Babadag wind park are caused by the positioning of the WT within the migratory routes of bats from the genera *Pipistrellus*, *Nyctalus* and *Vespertilio*. Stable isotope analysis of the non-exchangeable hydrogen in fur keratin for *N. noctula* showed that most studied animals were migrants, with most likely places of origin ranging from Ukraine, Belarus to the European parts of Russia. The most affected bat species were *P. nathusii*, *N. noctula*, *Vespertilio*

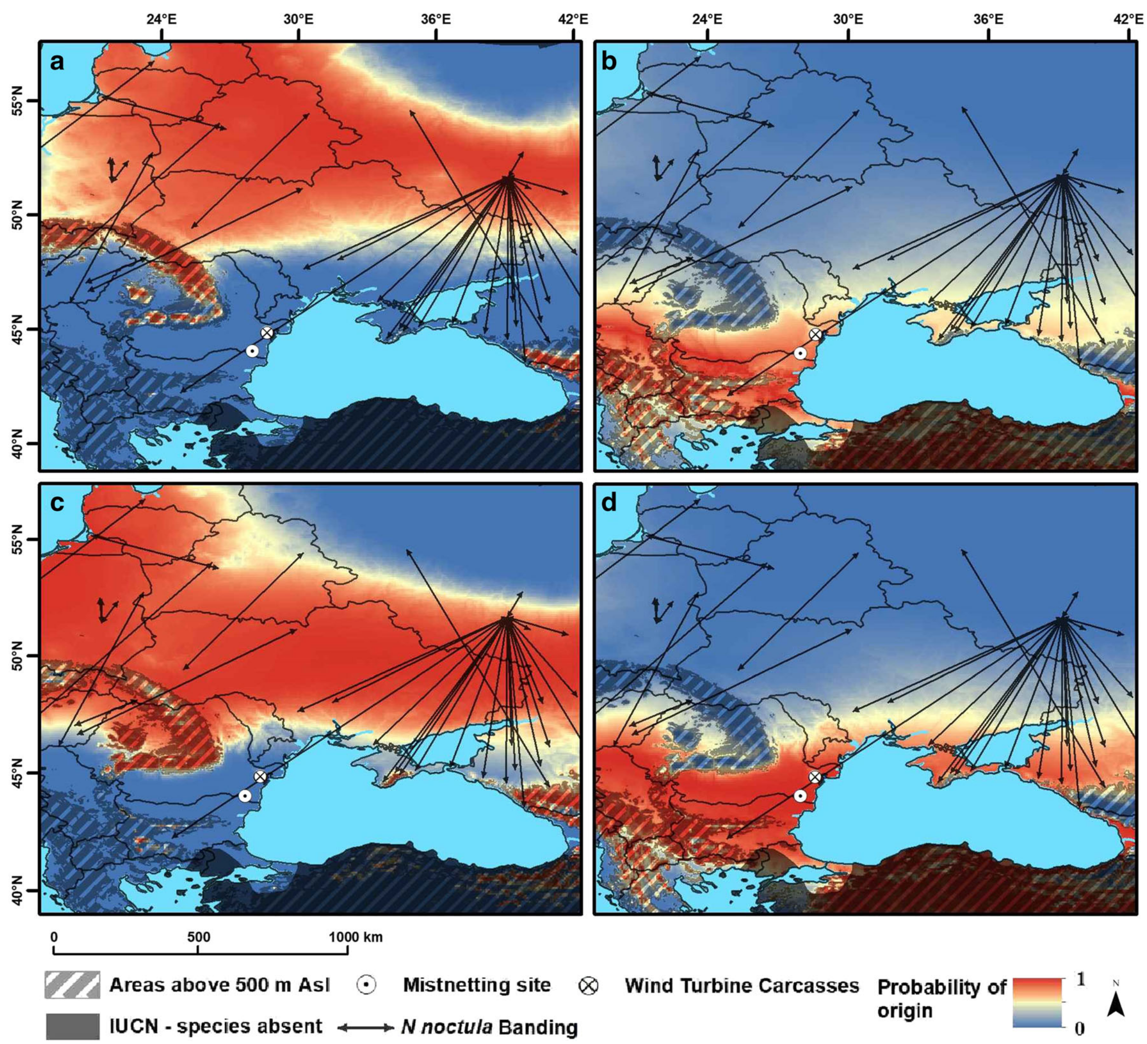

Fig. 2 Predicted geographic provenance of *N. noctula* from Dobrogea based on isoscape origin models ((**a**) carcasses of migratory bats ($n = 16$), (**b**) carcass of local bats ($n = 1$), (**c**) live migratory bats ($n = 15$), (**d**) alive regional ($n = 3$)). Values closer to 1 indicate likely areas of summer origin. Arrows indicate previous records of recaptured banded data according to Panutin (1980), Hutterer et al. (2005) and Gashchak et al. (2015)

murinus and *N. leisleri*. These species usually hunt in open space, at relatively high altitudes, and undertake long-distance migrations. Also, these species have been reported as common casualties at WT throughout Europe (Rodrigues et al. 2017). Accordingly, they were classified as high-risk species at WT by Rydell et al. (2010). The seasonal pattern of bat fatalities in Babadag was similar to the one observed across Europe (Rydell et al. 2010); fatality rates started to increase in mid-July and peaked in August, during the migration period. This period also coincides with a higher bat abundance due to the onset of independent hunting of juveniles. This factor might add to the observed seasonal peak of bat fatalities at WT (Barclay and Baerwald 2017). Lower fatality peaks were also observed during spring migration, mostly consisting of *P. nathusii*.

Curtailment of turbines with particularly high fatality rates proved to be successful since the fatality rates could be lowered significantly for the whole wind park. The decrease in fatality rates for WT operating under the mitigation scheme decreased by a factor of 9 and had similar values compared with the control WT. After implementing mitigation measures at the Babadag Wind Park, the fatality rates decreased substantially for *P. nathusii, N. noctula*, *V. murinus* and *N. leisleri*. Nevertheless, the tested mitigation measures revealed a positive effect by significantly reducing fatality rates of bats. The energy production of the wind facility was marginally affected by the curtailment measures, with an average of 0.35% loss/year (wind energy production data from the Babadag wind facility).

We consider that further modification of the operational algorithm might decrease fatality rates even beyond the reduced number of fatalities. Thus, the reduction of WT functionality during times of high bat activity, such as the migratory periods, can further be optimised by switching off the operation of WT during favourable climatic conditions for bat foraging and migration (at low wind speeds, high air temperatures, stable air pressure, lack of precipitation or moonlight intensity; Erickson and West 2002; Cryan and Brown 2007; Amorim et al. 2012; Behr et al. 2017; Martin et al. 2017). Furthermore, sensitivity maps based on mortality observations, species distribution and habitat suitability models, but also home range mapping of local bat populations, can help identify areas where current or future WT infrastructure operations need to be mitigated (Santos et al. 2013; Măntoiu et al. 2015; Bosso et al. 2018).

Stable isotope analysis of the non-exchangeable hydrogen in fur keratin for *N. noctula* has shown that about 90% of the tested individuals were migratory, with breeding origins ranging most likely from the Ukraine, Belarus and the European parts of Russia. It is noteworthy that the proportion of migratory individuals amongst bats captured alive and those recorded dead below WT was similar. Based on the reference dataset from individuals captured in mist nets, we suggest that the proportion of migratory individuals found at the WT reflects the proportion of migrant individuals present in the area. The proportion of migratory individuals observed in Dobrogea is considerably higher compared with that of North-Eastern Germany (Lehnert et al. 2014; Voigt et al. 2014), where only a quarter of *N. noctula* carcasses originated from a distant population. These results may reflect differences in the spatial and temporal patterns of bat migration. For example, Northern Germany has been identified to be both a breeding area for non-migratory populations and a transit area for migratory conspecifics (Lehnert et al. 2014). At the same time the conclusion formulated by Strelkov (1997, 1999, 2000), Strelkov and Abramov (2001), summarised by Hanák et al. (2001) and Benda et al. (2003) suggested that the Balkan Peninsula, Dobrogea included, is a hibernation area for long-distance migrant species, which mostly breed in the boreal zone of Eastern Europe.

Our study further corroborates earlier studies from other EU countries that fatalities at WT involve transboundary effects on wildlife far beyond the political borders of the EU (Voigt et al. 2012; Lehnert et al. 2014). These transboundary effects of wind facilities on bat populations may become especially severe when conservation measures as practiced for threatened species in non-EU countries of Eastern Europe are weak, less regulated by legislation norms and likely ignored by companies and authorities (Rodrigues et al. 2017). The catchment area of WT in Dobrogea is not part of the legislation range of Habitat Directive 92/43/CEE (Annexes II and IV) and Environmental Impact Assessment (EIA) Directive 85/337/EEC (amended to Council Directive 97/11/EC in 2011), being composed of areas within Ukraine, Belarus and Russia, which are not a part of EU. Several international agreements such as EUROBATS and Bern Convention provide some basis for conservation of European species outside the EU, but these are rather interpreted as recommendations or guidelines.

The fatality rates observed in our study were high, even though we presented data from a relatively small wind park, consisting only of 20 WT with a total capacity of 42 MW net energy production. The current capacity of operating wind facilities of the whole Dobrogea alone amounts to 3028 MW of installed net energy production by the end of 2016 (WWEA, http://www.wwindea.org), with 2944 MW installed net energy production (www.thewindpower.net). Based on our estimation of fatality rate per MW and taking into account common capacity of WT facilities in the region, we strongly support the implementation of mitigation measures. Potentially, the high numbers of bat fatalities can have cumulative effects on source populations and may lead to a decline in bat populations over large geographic ranges. We plead for a large-scale implementation of curtailment speeds, possibly adjusted to threshold values of ambient temperature, not only in the Dobrogea but also in the entire Black Sea region in order to make the Via Pontica flyway safe again for migratory bats.

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Appendix

The list of bat mortalities at the Babadag Wind Park, 2013–2016

Abbreviations: Bat species acronyms: ESER – *Eptesicus serotinus*; HSAV – *Hypsugo savii*; MYOSP – *Myotis sp.*; NLEI – *Nyctalus leisleri*; NNOC – *Nyctalus noctula*; PKUL – *Pipistrellus kuhlii*; PNAT – *Pipistrellus nathusii*, PPIP – *Pipistrellus pipistrellus*; PPYG – *Pipistrellus pygmaeus*, PIPSP – *Pipistrellus sp.*; VMUR – *Vespertilio murinus*. Structure of citation: date (mm/yy), species, no. individuals (in the case of isolated individuals, no numbers have been aded), sex (♀- female, ♂ - male), age (ad – adult, sad-sub adult, juv – juvenile). In case of missing data, the n/a (not available) symbol was used, for example: PNAT n/a – no information about sex and age, or PNAT ♂n/a – no information about age. Each sex-age-n/a group was divided by a coma. Species belonging to the same date were delimited by semicolons and different months were delimited by a point

4.13 – PNAT ♂n/a, ♀n/a, n/a; PPIP ♂n/a, n/a ad; PIPSP n/a; VMUR n/a ad; NNOC ♂n/a. 05.13 – PNAT 2n/a. 06.13 – PNAT 2n/a; PPIP n/a; PIPSP n/a. 07.13 – PKUL ♂ad; PIPSP n/a ad, 2n/a; VMUR 3n/a. 08.13 – PNAT 3♂ad, 2♀ad, n/a juv; PKUL 2♂ad, n/a ad; PPYG ♂ad, n/a ad; PIPSP ♂juv, n/a juv, n/a ad, 3 n/a; NNOC ♂ad, 3♀ad, 2n/a; VMUR ♂ad, 3n/a. 04.14 – PNAT 7♂ad, ♂n/a, 4♀ad. 05.14 – PNAT 3♂ad, ♀ad; PKUL 2♂ad; PIPSP ♂ad; MYOSP n/a ad, ♂ad. 06.14 – PNAT 2♂ad, 2♀ad; NNOC ♀ad; NLEI 2♀n/a. 07.14 – PNAT ♂ad, n/a; PIPSP ♀ad; ESER ♂ad; NNOC ♂ad, ♂n/a, n/a ad, 2n/a; NLEI ♂ad, ♀n/a, n/a ad. 08.14 – PNAT ♀ad, ♂juv, 2n/a ad; HSAV ♂n/a; NNOC 2♂ad, ♀ad, n/a ad, 2n/a. 09.14 – PNAT 2♂ad, 3♀ad; PPYG n/a ad; PIPSP n/a ad; NNOC ♂ad, ♀ad, 2n/a ad, n/a; HSAV n/a. 10.14 – PNAT ♂ad, ♂n/a, n/a ad; NNOC ♀ad, 2n/a ad, n/a. 11.14 – PNAT ♀ad. 04.15 – PNAT ♂ad; NNOC ♀ad. 05.15 – PNAT ♂ad, PIPSP 2n/a, VMUR n/a ad. 06.15 – NNOC n/a. 07.15 – PNAT ♂ad; PIPSP 3n/a. 08.15 – PNAT ♂ad, 2♀ad; NNOC 2♂ad, 2n/a ad. 09.15 – PIPSP 3n/a. 5.16 – PNAT ♂ad; PPYG ♂n/a, PIPSP 2n/a; NNOC ♀ad. 7.16 – PNAT ♂ad; NNOC ♀ad; NLEI ♂ad.; VMUR n/a. 8.16 – PNAT ♂n/a; PPIP ♂ad; PIPSP 3n/a; NNOC 3♂ad, ♀ad, 2n/a. 9.16 – PNAT ♂ad; PPYG ♀n/a; PIPSP n/a; NNOC 2♀ad, n/a.

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