Effects of wind turbine noise on songbird behavior during nonbreeding season

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Article impact statement

Wind turbine noise negatively and dramatically affects songbirds. Noise must be considered when planning wind power infrastructure.

Abstract

Anthropogenic noise is one of the fastest growing, globally widespread pollutants, affecting countless species worldwide. A rapidly growing source of noise pollution is wind turbine infrastructure. Despite accumulating evidence on the negative impacts of wind turbines on wildlife, little is known about how the noise they generate impacts ecological systems. Songbirds may be susceptible to noise pollution due to their reliance on vocal communication and thus, in this field study, we examined how songbirds are impacted by wind turbine noise. We broadcasted noise produced by one wind-turbine in a migratory stopover site during the nonbreeding season. Throughout the study, we repeatedly monitored the acoustic environment and songbird community before, during and after the noise treatments. We found a significant decrease of approximately 30% in the daily number of birds in the presence of wind turbine noise, compared to the before or after phases. This reduction had a significant spatial pattern with the strongest decrease closer to the speaker and on its downwind side, fitting measured sound propagation. Although we found no impact on species diversity, two out of three most common species at our site showed clear avoidance behavior: 45% and 36% decrease in abundance for the Lesser whitethroat (*Sylvia curruca*) and Sardinian warbler (*Sylvia melanocephala momus*), respectively, with lingering effects on the Lesser whitethroat in the 'after' phase. Additionally, the age structure of the lesser whitethroat population was impacted, as only juvenile birds showed avoidance behavior. No difference in avoidance extent was found between migratory and nonmigratory species, but the impacts of displacement on migrants during stopover are especially troubling from a conservation perspective. Our results stress the need to address the impacts of noise pollution on wildlife when planning noise-

generating infrastructures, such as wind turbines, to allow for sustainable development without threatening already declining songbird populations.

Introduction

In recent decades, evidence for various negative impacts of wind turbines on birds has accumulated (e.g., increased mortality, reduced breeding success, altered feeding rate, lower population densities; Dai *et al.* 2015, Shaffer & Buhl 2016, May *et al.* 2017, Allison *et al.* 2019), yet the mechanisms leading to these impacts remain unclear. Better understanding of the factors contributing to these negative impacts is key for the development of applicable mitigation tools (May *et al.* 2017) and for better planning and placement of infrastructure. In this respect, without empirical evidence, several studies have suggested that wind turbine noise may play a major role in displacing birds from their habitats(Dooling 2002, Stevens *et al.* 2013, Wang *et al.* 2015, Zwart *et al.* 2016).

Animals use acoustic information from their surroundings to assess habitat quality and resource distribution (Farina *et al.* 2011, Pijanowski *et al.* 2011). Recent decades have brought a rapid increase in anthropogenic sounds (Barber *et al.* 2010), commonly termed "noise pollution". This pervasive and important human-caused element degrades habitat quality and affects many species worldwide, even in supposed refuges such as protected habitats (Swaddle *et al.* 2015, Buxton *et al.* 2017, Senzaki *et al.* 2020a). Noise can be perceived as a threat by animals, it may distract them and reduce the efficiency of their actions, and it may mask their communication, obscuring acoustic signals upon which they may rely (Dominoni *et al.* 2020). A common behavioral reaction to noise pollution is avoidance of noise-polluted areas (McClure *et al.* 2013, Shannon *et al.* 2016, Cinto Mejia *et* *al.* 2019). Displacement caused by noise reduces the availability of otherwise suitable habitat, potentially forcing individuals into habitats of lower quality. However, even if an individual stays in a disturbed habitat, it does not mean that it does not suffer from the negative effects of this disturbance (Swaddle *et al.* 2015, Dominoni *et al.* 2020).

To successfully use their environment, animals must optimize their rate of information gain to maximize their exploitation of resources in ever-changing habitats (Berger-Tal *et al.* 2014). This is especially true for migrating and dispersing individuals for which the environment may be completely novel. Specifically, it has been shown that many birds rely on vocal cues for stopover site selection (Mukhin *et al.* 2008, Németh & Moore 2014) and for decision-making within sites (Németh & Moore 2007). However, as previously mentioned, exposure to noise pollution might disrupt these crucial communication behaviors (Barber *et al.* 2010, Shannon *et al.* 2016), leading to a reduction in perceived habitat quality and increasing the chances that individuals will avoid the area (McClure *et al.* 2013, Shannon *et al.* 2016, Cinto Mejia *et al.* 2019). Such behavioral changes can scale up from the individual level by modifying population structure, shaping communities and, in the long run, altering entire ecosystems (Francis & Barber 2013).

A rapidly growing source of noise pollution worldwide is wind turbines (Dai *et al.* 2015, Zwart *et al.* 2016). Despite their recognized environmental benefits, wind turbines can negatively impact wildlife populations; either through direct mortality and disturbance but also by reducing breeding success, altering movement patterns, and causing large-scale habitat loss via avoidance (Stevens *et al.* 2013, Dai *et al.* 2015, May 2015, Wang *et al.* 2015, Zwart *et al.* 2015, Fernández-Bellon *et al.* 2019).. Specifically, wind turbine noise substantially increases ambient sound levels (Dai *et al.* 2015) within the hearing range of many animals, including most bird species (Dooling 2002). The magnitude of noise created by a wind turbine can remain above 30dB(A) even at a distance of 800m (Rogers *et al.* 2006, Katinas *et al.* 2016). This distance (800m) fits the range of avoidance behavior found in several wind turbine studies (Percival 2005, Fernández-Bellon et al. 2018, Santos et al. 2021). Moreover, wind turbine noise differs from other sources in three important aspects when considering noise impacts on the acoustic environment or on wildlife (Gomes *et al.* 2021): 1) it is noticeable even when the measured level is below the natural background sound level because of its amplitude modulation (a repeating “swooshing” sound, with regularly repeated peaks higher by 15-20 dB compared to minimum SPL at different frequency bands; Katinas *et al.* 2016), 2) it has a broadband frequency range (mainly 1-5kHz but often reaching 8 kHz and more; Dai *et al.* 2015, Katinas *et al.* 2016), and 3) it is influenced by wind conditions (Rogers *et al.* 2006), leading to an unpredictable variation between hours and days. These acoustic features distinguish wind turbine noise from traffic noise, for example (Pedersen & Van Den Berg 2010), the most extensively studied source of anthropogenic noise (Jerem & Mathews 2021). Altogether, the high intensity and the unique acoustic signature make it an obvious knowledge gap in the noise-pollution literature and limit our understanding of the possible impacts of such noise on the movement and the habitat selection of wildlife (Zwart *et al.* 2015, May *et al.* 2017).

While studies examining bird mortality as a result of collisions with turbines are relatively abundant, only a few have investigated the displacement of birds due to nonlethal impacts of wind farms (Coppes *et al.* 2020, Taubmann *et al.* 2021). Studies investigating the effects of turbine-generated noise on birds are even fewer and mostly focus on breeding behavior (Szymański *et al.* 2017, Whalen *et al.* 2019, Gómez-Catasús *et al.* 2022). The limited available literature suggests that wind turbine noise affects bird vocal communication and habitat use (Zwart *et al.* 2016, Szymański *et al.* 2017, Whalen *et al.* 2019, Gómez-Catasús *et al.* 2022). Previous studies trying to estimate the spatial extent of avoidance of wind turbines by birds, possibly but not necessarily due to noise, found varying results with some species showing larger avoidance distance than others (ranging from several dozens to 850 m). Variation in the range of impact among species, including closely related ones, may be attributed to life history of the species, internal state factors (e.g., age and sex biases), wind farm location or micro-site conditions within an individual wind farm (Percival 2005, Fernández-Bellon *et al.* 2019, Santos *et al.* 2021). In the absence of empirical data on the impact of wind turbine noise pollution on migratory songbirds, the conservation of vital stopover sites may be hampered. To fill this knowledge gap, we broadcasted the sound of a wind turbine in a migratory stopover site during the nonbreeding (migration) season to quantify the effects of wind turbine noise on the natural acoustic environment and consequently, on songbird species diversity, population structure, and avoidance behavior.

Methods

We tested the effects of wind turbine noise on songbirds by creating a 'phantom wind turbine' (*sensu* McClure *et al.* 2013; “phantom-road” experiment): an acoustic experimental manipulation based on broadcasting wind turbine noise, isolating the effects of noise from other effects (such as visual or tactile cues). We measured both the surrounding acoustic environment and the status of the songbird community before, during, and after the noise treatment.

We repeated the experiment six times; each repeat included three 48 hours phases ("before", "noise-treatment", "after"). We did not take measurements during Saturdays, which are the national weekend in Israel and may thus be characterized by different human-induced background noise levels. We repeated the experiment every second week to allow migratory birds to continue their migration and for new birds to arrive. Similarly, nonmigratory species were able to redistribute naturally in the habitat between repeats. The one-week gap between each experimental repeat ensured that we were testing a different migratory bird community every time (and this was supported by the extremely low recapture incidents of marked individuals between repeats; Tables S1 and S2), thus allowing the use of a single site and eliminating among-sites variation (in species composition, and in noise propagation due to differences in topography, but see 'limitations of the study'). Overall, we conducted six repeats of the experimental protocol during the fall of 2018, between August 20th and November 2nd.

Study site

We conducted our study at Horesh Adulam Nature Reserve in central Israel (Fig. 1A). This migratory stopover site is characterized by a generally homogenous Mediterranean Maquis habitat enclosed by three minor roads. The closest human settlement is a small village situated at a distance of more than 1.5 km. These conditions make this a quiet site (Table S3) and thus ideal for sound recordings and manipulations. Weather measurements (including wind direction and speed) were obtained from the nearest weather station, situated <4 Km away at Beit Govrin, with clear line of sight to the study site (Vantage Pro2 Plus, Davis Instruments, California, USA; Weather station data available on

Wunderground.com, station ID: IDAROMBE2). We calculated the average hourly wind speed for all monitoring hours.

Phantom wind turbine

To create the phantom turbine during “noise-treatment” days, we broadcasted wind turbine noise (Fig S1) at a typical level of 102 dB(A) L_{eq} (Katinas *et al.* 2016) measured for 30 sec on each placement at a distance of 1 m from the speaker (Soundboks2, 40-20,000Hz, Soundboks, China). The WAV recordings of wind turbine noise were provided by Timothy Van Renterghem (for recording details see Van Renterghem *et al.* 2013) and were played using a SanDisk Clip player. The recording accurately recreates real wind turbine noise in respect to the range of frequencies and amplitudes that are relevant to both the avian and the human auditory system, only differing from the original sounds by not including infra-sound (which was not the focus of this study and is below the hearing range of songbirds; Dooling 2002, Rogers *et al.* 2006, Dooling & Popper 2007). We broadcasted the noise for the typical operation duration of wind turbines in the region based on wind speed- from one and a half hours before sunrise (landing time of migratory birds; Chernetsov & Mukhin 2006) until one hour after sunset. This range covers the resting and refueling hours of migrants at this stopover site as well as the foraging time of local birds. We used a recording from a single turbine to prevent a nonrealistically varying acoustic stimulus caused by merging recordings from different sources and recording sites. During the “before” and “after” phases, we placed a “dummy” speaker at the same location.

We placed the phantom turbine on a hill-top with the speaker positioned facing NNE (30 degrees; calculated wind direction) because sound propagates from wind turbines such that most of the sound energy is directed downwind (Rogers *et al.* 2006) . Wind direction was calculated as median wind direction during daylight hours in the previous season. The speaker was placed 2 m off the ground, above the main vegetation level, allowing unobstructed sound propagation.

Acoustic monitoring

We recorded the acoustic environment continuously during each experimental repeat in MP3 format, at a sampling rate of 48 kHz, and at a bit rate of 128 kbps at 12 locations using calibrated Acoustic Recording Units (Roland R05 audio recorder) (Mennitt & Fristrup 2012). This setting allowed us to measure absolute background sound levels instead of relative measurements. We deployed the recorders inside a fabric windscreen within Palestine Oaks, *Quercus calliprinos,* 1.5 meters above the ground, at varying angles and distances from the phantom turbine (0, 90, 180 and 270 degrees; 5, 70, 140 and 280 m; Fig 1B).

Sound pressure level (SPL) was measured in A-weighted decibels (hereafter dB(A)) suited for human-related sound measurements and relevant for songbirds that have a similar hearing curve (Dooling & Popper 2007). To describe the wind turbine noise, we used the hourly L_{50} (median noise exceedance level, in dB(A) SPL), which is suited to describe continuous sources of noise, using custom programs (Damon Joyce, NPS, AUDIO2NVSPL and Acoustic Monitoring Toolbox).

To account for the noise measurements' artifacts created by strong winds, we calculated Cook's distance (Cook 1977) to identify outliers of wind speed's effect on sound level (L_{50}) during daytime when wind turbine noise was not played. We then found that wind speeds of 8 m/sec or higher produced >10% outliers and consequently removed noise measurements obtained at these wind speeds from all experimental phases. We do acknowledge that this removed data that corresponds to conditions when turbines will be operating in the landscape, but since we expect greater impacts with increasing sound levels, it means our approach is conservative.

Measuring the effects of noise on songbirds

To measure the impact of the "noise-treatment" on the songbird community, we conducted bird-ringing sessions on the second day of each experimental phase. Each session included 12 mist-nets of 12m length and 2 mist-nets of 18m length at fixed locations set 45 minutes before sunrise and opened for 4.5 hours (hereafter 'bird-ringing hours'; Table S4). The nets were inspected every 60 minutes, except for rare cases of strong wind or high temperatures in which we inspected the nets every 30 minutes and summed every two extractions. The nets were placed at a gradient of distance (20m-120m), and at varying angles from the phantom wind turbine. We opted to divide the capturing area into 4 "slices" and placed 4 nets in each slice – two "close" and two "far". By doing so, we aimed to minimize bias in the setting while dealing with a heterogenous natural study site (with all the benefits of conducting a field experiment). Additionally, the nets were placed in three directions (upwind, downwind, and perpendicular to wind direction; covering <180 degrees), and not in a complete circle around the phantom wind turbine with the assumption that noise propagations to both sides will be similar (as supported by the continuous sound measurements; Fig. 1B) and to assure birds' safety (by keeping the distance between the nets relatively short and thus reducing the time the birds spent in the nets). None of the songbird species on site is found in flocks (Shirihai *et al.* 1996), and thus we assume their movements are made at the individual level. For each captured individual we collected date and time of capture, mist-net location, species identification, age and sex (when possible based on plumage differences), maximum wing chord and tail length (0.5 mm accuracy), and body mass (0.1 g accuracy). Species identification also provides the classification to categories of migrant or nonmigrant, as there are no species with both migratory and nonmigratory populations at this site at this time of the year. All birds were released at the study site shortly after their capture. To avoid pseudo-replications and to minimize the inclusion of data from individuals who experienced repeated exposure to the “noise-treatment” (Harding *et al.* 2019), all re-captured birds were excluded from the analysis (using only first capture data). All birds were handled following the national ethical guidelines (Israeli Nature and Parks Authority bird ringing permit A-302 and research permit 2018/41922).

Data analysis

We analyzed the data using R version 3.4.4 (R Core Team (2020), Vienna, Austria) in RStudio (RStudio Team (2016), Boston, MA) with packages lme4 (Bates *et al.* 2015) and lmerTest (Kuznetsova *et al.* 2017). We used generalized linear mixed effects models (GLMM) to test the effect of experimental phase (categorical variable; “before”, “noise-treatment”, and “after”, with the “before” phase being the reference condition) on different response variables: using Poisson distribution for abundance (total number), age, and sex, Gamma distribution for species diversity (calculated as Fisher’s alpha), and Binomial distribution for the ratio of migratory to nonmigratory individuals and species. All models included the experimental repeat number (1-6; the repeated measures) as a random factor to account for possible temporal variation in the bird community and habitat along the season and for biases that may be caused by overdispersion in the Poisson models (Harrison 2014). First, each model was compared to a null model which only included the random factor (i.e.

experimental repeat). Using a likelihood ratio test (Zuur *et al.* 2009, Alday 2016) we checked whether the insertion of the experimental phase factor significantly improved the model compared to the null model. Finally, for models which were significantly better than the null model, we report the p-values for each experimental phase.

For the three most abundant species in the area (see below) we separately tested the effects of the experimental phase on each response variable. If the experimental phase significantly affected categorical internal-state factors (i.e., age or sex group), we built separate models for each group (i.e., juveniles\adults, females\males) using Poisson distribution and followed the abovementioned process to explore specifically if and how each group was affected by the "noise-treatment". We had six repeats for each experimental phase. For the lesser whitethroats (*Sylvia curruca)* we only used the first four experimental repeats because none were caught during the last two experimental repeats.

Lastly, we constructed a model to examine the spatial distribution of the noise impact. To do so, we used the daily number of birds captured at each net, correcting the number of captured birds at the two 18 m nets by dividing it by 1.5 and rounding to the nearest whole number, allowing the use of Poisson distribution. This model included the following explanatory variables: experimental phase, distance from the phantom wind turbine (in meters), direction from phantom wind turbine downwind axis (i.e., sound propagation direction; in degrees), the interaction between these three factors, and the experimental repeat number as a random factor. Distance and direction were scaled and centered. We chose not to use sound levels directly as a predictor for bird' abundance at each net due to the relatively sparse distribution of recorders compared to mist-nets (Fig. 1). Table S5 specify the full list of models we've used to analyze our data.

Results

Acoustic monitoring

After removing 27.5% of all recorded data due to wind speed exceeding 8m/sec (see Methods), we were left with a total of 1337 hourly values collected during the bird-ringing hours (Table S6). Specifically, we collected 85-127 hourly L_{50} (median exceedance level) sound level measurements over 21-28 days at each of the 12 recorders.

The sound level of the acoustic environment was similar at all locations during "before" and "after" phases (except for the relatively high values from location 'Bird-ringing side 140m' that was more exposed to wind and to the nearby road; Table S3). During the "noise-treatment", sound levels were considerably higher at 11 of the 12 recording locations, compared to the sound levels before or after the treatment (Table S3). Altogether, the distribution of change in sound levels results from a combination of distance and direction from the phantom wind turbine (Fig. 1B): The mean L_{50} sound level was higher during "noise-treatment" phase than "before" phase in all directions but not at all distances.

Birds' response to noise

We caught and ringed a total of 670 birds (652 new and 18 recaptured from previous ringing days that were removed from the data analysis; see Methods) from 26 different species (Table S1). We examined the effects of the noise treatment at the community level for all birds and at the population level for the three most common species, all belonging to the *Sylvia*-warblers family. The noise significantly reduced the number of birds we sampled daily, including lowering the abundance of two of the three *Sylvia*-warbler species. This reduction in bird abundance had a distinct spatial pattern. We did not find any significant changes to species diversity or species composition.

Abundance and distribution

Bird abundance, measured as the mean daily number of all songbirds caught in the nets, showed a significant decline of approximately 30% during the “noise-treatment” compared to the “before” and “after” phases (“before”: 41.33 ± 7.58; “noise-treatment”: 28.33 ± 5.35; “after”: 39.0 ± 4.95 (Mean ± SE, here and throughout the text); likelihood ratio test, $\chi^2_{(df=1)}$= 16.54, p<0.0003;Table 1; Fig. 2).

The spatial model showed an impact of “noise-treatment” on the daily number of captured birds, with a significant interaction among treatment phase, distance from the phantom turbine, and direction, revealing a stronger displacement of birds from areas closer and downwind of the phantom wind turbine. This pattern was found significant during “noise-treatment” phase, and also during the “after” phase (With smaller impact during “after” phase; likelihood ratio test, $\chi^2_{(df=7)}$= 23.35, p<0.001; Table 2; Fig. 1C).

Species composition

Although we found a small reduction in the number of sampled species using Fisher’s α measure of diversity during the “noise-treatment” phase, this reduction was not significant (“before”: 3.14 ± 0.52; “noise-treatment”: 2.77 ± 0.68; “after”: 3.5 ± 0.67; likelihood ratio test, $\chi^2_{(df=1)}$= 2.91, p=0.23). Altogether, 21 migratory species were ringed out of 26 species in total (Table S1). The ratio of migratory and nonmigratory species did not change during the “noise-treatment” phase compared to “before” or “after” phases (likelihood ratio test, $\chi^2_{(df=2)}$= 0.2, p=0.91). Moreover, we found no difference in the reaction of individuals of migratory species and individuals of nonmigratory species to the noise treatment as implied by the constant ratio between the numbers of caught individuals from the two species' groups: approximately 70% of captured birds were individuals of migratory species and 30% were individuals of nonmigratory species regardless of the experimental phase ("before": 0.68 ± 0.05; "noise-treatment": 0.70 ± 0.03; "after": 0.70 ± 0.03; likelihood ratio test, $\chi^2_{(df=1)}$= 0.17, p=0.92). Similarly, the "noise-treatment" did not have a significant impact on the ratio of individuals from migratory and nonmigratory species when excluding the three most common *Sylvia* warblers (likelihood ratio test, $\chi^2_{(df=2)}$= 1.72, p=0.42).

Population level responses – Sylvia warblers

Three species of *Sylvia*-warblers: Eurasian blackcap (*Sylvia atricapilla)*, lesser whitethroat (*S. curruca)*, and Sardinian warbler (*S. melanocephala momus),* were the most common species in this study, with more than 100 sampled individuals for each species (Table S2). Two of these species are migratory: Eurasian blackcap and lesser whitethroat. Individuals of the third species, the Sardinian warbler belong to a nonmigratory sub-species (Shirihai *et al.* 2001). We were able to identify the age of most blackcap and lesser whitethroat individuals (97.4% and 99.2%, respectively) but only of two-thirds of Sardinian warblers (66.9%; Table S7). Moreover, we visually identified the sex of 258 warblers from two species: Blackcap and Sardinian warbler, representing 99.5% and 99.4% of the individuals from each species, respectively (Table S8). The extensive data for these species allowed us to further examine the effects of the "noise-treatment" on them.

The noise treatment had a different effect on each of the species (Fig. 3). For the lesser whitethroat the decline in abundance (i.e., daily mean number of individuals) was consistent during the "noise-treatment" and the "after" phases when compared to the "before" phase ("before": 13.25 ± 6.99; "noise-treatment": 8.5 ± 2.99; "after": 8.0 ± 1.96;

likelihood ratio test, $\chi^2_{(df=2)} = 6.49$, $p<0.05$; Table S9; Fig. 3A). Similarly, for the Sardinian warbler we found a significant decline during the “noise-treatment” when compared to the “before” phase, but the “after” phase was not significantly different from both other phases (“before”: 11.0 ± 2.41; “noise-treatment”: 6.0 ± 1.65; “after”: 8.17 ± 1.8; likelihood ratio test, $\chi^2_{(df=1)}= 9.01$, $p<0.02$; Table S10; Fig. 3B). The effect of the “noise-treatment” was not found to be significant for the blackcap compared to the “before” or the “after” phases (“before”: 10.83 ± 2.95; “noise-treatment”: 9.0 ± 2.11; “after”: 11.67 ± 3.13; likelihood ratio test, $\chi^2_{(df=1)}= 2.17$, $p=0.34$; Fig. 3C).

For the lesser whitethroat only, we found a significant difference in the reaction of different age groups to the “noise-treatment” with a reduction in the ratio of juveniles to adults during “noise” and “after” phases compared to “before” (“before”: 0.81 ± 0.11; “noise-treatment”: 0.53 ± 0.17; “after”: 0.62 ± 0.11; likelihood ratio test, $\chi^2_{(df=2)}= 6.61$, $p=0.04$). This was caused by the fact that fewer juvenile individuals were found at the site during the “noise-treatment” and the “after” phases compared to the “before” phase (“before”: 9.75 ± 4.33; “noise-treatment”: 3.75 ± 1.38; “after”: 5.25 ± 1.44; likelihood ratio test, $\chi^2_{(df=2)}= 12.04$, $p<0.005$; Tables S7 and S11; Fig. 4A). This trend was not observed in adult lesser whitethroats, which showed no significant change in their daily number in relation to the experimental phase (“before”: 3.5 ± 2.87; “noise-treatment”: 4.5 ± 2.33; “after”: 2.75 ± 0.85; likelihood ratio test, $\chi^2_{(df=1)}= 1.72$, $p=0.42$; Table S7; Fig. 4B). We did not find age effects for Blackcaps or Sardinian warblers. Also, in both species where sex can be visually identified (blackcap and Sardinian warbler), we found no significant role of sex on the reaction to noise.

Discussion

Our results show a direct and strong negative impact of wind turbine noise on songbirds. By broadcasting wind turbine noise, we were able to isolate the impact of noise from other possible effects of wind turbines, and found an approximate 30% decline in mean daily bird abundance while the wind turbine noise was broadcasted. This decline had a distinct spatial pattern: we found stronger displacement of birds from areas closer and downwind of the phantom wind turbine.Individuals of migratory and nonmigratory species reacted to the change in the acoustic environment in a similar fashion. Correspondingly, the noise treatment did not cause a significant change in species composition. Nevertheless, we found a substantial reduction in the daily abundance of two of the three most common species we examined.

Acoustic monitoring

We quantified sound levels of the experimentally-manipulated acoustic environment while measuring changes in the bird community and population structure to better understand the relations between anthropogenic noise pollution and its ecological consequences (McKenna *et al.* 2016, Shannon *et al.* 2016, Lehnardt *et al.* 2019). Specifically, noise generated by wind turbines has a typical spatial propagation pattern according to the wind direction and speed (Dai *et al.* 2015). Here, the measured sound levels from the phantom wind turbine fit the general expectations from an operating wind turbine: during the “noise-treatment” phase, sound levels increased throughout the study site in all directions around the phantom wind turbine. The highest sound levels were measured downwind and the lowest were measured upwind from the phantom wind turbine and with a gradual reduction in the level of noise from the center outwards (Fig. 1B). This result strongly supports the validity of this method, allowing future studies to continue distinguishing noise effects from other confounding effects generated by the establishment of wind turbines, as was previously done for roads (McClure *et al.* 2013) or natural gas fields (Cinto Mejia *et al.* 2019). It is important to note though, that specific wind turbine models placed at particular sites may produce different noise levels at the given distances used in this study (Dai *et al.* 2015). In addition, noise propagation of real wind turbines is expected to reach much further distances than our speaker due to their substantially greater size and height above the ground. At a larger scale, sound levels from entire wind farms can be modelled for specific sites to allow for a realistic impact evaluation (Sun *et al.* 2018). Accordingly, when using our data for wind infrastructure planning, it is vital to extrapolate the expected sound levels for the planned wind farm’s specific circumstances and spatial distribution. This procedure will ensure better precision of models and ecological impact assessments when developing windfarms in remote or nonurban open landscapes.

Overall, the experimentally broadcasted wind turbine noise strongly impacted the natural acoustic environment, increasing the sound levels by up to 60 dB(A) (Fig. 1B). In this respect, it is important to realize that while Environmental Impact Assessments often consider a change of a few decibels as having a “negligible impact”, a change of 3 dB(A) already reduces the hearing range of birds by half, a change of 6 dB(A) reduces the hearing range by three-quarters, and a change greater than 12dB(A) reduces the hearing range of a bird by more than 90% (Barber *et al.* 2010). This means that at the core of the wind turbine noise-polluted area, which covers several hundred square meters, birds are expected to barely perceive any acoustic cues in their environment (within the frequency band of the noise). This limited ability to sense acoustic signals and cues may elicit various behavioral responses such as avoidance behavior, reduced feeding rate, elevated anti-predator behavior, and impaired communication (Barber *et al.* 2010, Francis & Barber 2013, Shannon *et al.* 2016). In addition to the masking of acoustic cues, noise pollution may also distract exposed birds, reducing their ability to efficiently use the habitat (Dominoni et al. 2020). Lastly, exposure to noise pollution might negatively impact (directly or through complementary mechanisms) many physiological processes and lead to high stress hormone levels, accelerated DNA damage, and reduced sleep quality, immune response, cognitive ability, and fertility (Kight & Swaddle 2011).

Bird abundance and distribution

We found a decrease of ca. 30% in the mean daily abundance of songbirds when broadcasting wind turbine noise. Interestingly, similar rates of displacement in birds were found in previous studies on other sources of anthropogenic noise pollution such as roadway traffic (McClure *et al.* 2013, Ware *et al.* 2015, Senzaki *et al.* 2020b) and natural gas extraction fields (Cinto Mejia *et al.* 2019) despite major differences in the studied species, habitat, and source of noise. Two out of the three most common species in our study showed an even greater rate of decline with 45% and 36% reduction in abundance of lesser whitethroats and Sardinian warblers, respectively. Despite the existence of extensive literature on wind turbine noise in terrestrial environments, we found no study which examined the impact of such noise on bird habitat selection. Notwithstanding, our findings contradict the claim raised by (May *et al.* 2017), in their comprehensive review focusing on offshore wind turbine–wildlife interactions, that wind turbine noise will likely cause no displacement in birds (although we note that the impacts offshore and onshore turbines are not necessarily comparable). While the habitat surrounding wind turbines may seem to remain intact, it is no longer suitable for some birds because of an invisible pollutant: noise.

As mentioned above, each of the three focal species in our study reacted differently to the added wind turbine noise, despite having a similar life-history (e.g., habitat preferences, diet, vocalizations; Shirihai 2001).While we need a better understanding how these species interact with the acoustic niche during the post-breeding season to explain the differences in their responses, it is worth noting that this result aligns with previous studies on the variation among displaced birds from noisy areas in general (McClure *et al.* 2013, Ware *et al.* 2015, Cinto Mejia *et al.* 2019) and wind farms in particular (without specific emphasis on noise; Percival 2005, Stevens *et al.* 2013, May 2015, Fernández-Bellon *et al.* 2019). This result also stresses the need to identify noise-sensitive species of conservation concern, and use this knowledge to define habitats which are more vulnerable to noise pollution.

Inherently, the meaning of the decrease in bird abundance is that wind turbine noise reduces the number of available and suitable habitats for many birds. Our results therefore support recent studies that have highlighted the role of wind turbines in causing habitat loss and fragmentation in birds (Fernández-Bellon *et al.* 2019, Marques *et al.* 2019) and suggest that at least to some extent, the observed displacement found in wind farms is caused by noise. This rate of displacement occurs in addition to the hazardous effects of other factors such as wake effects, pulsing lights or others (Lovich & Ennen 2013, Sun *et al.* 2018). Specifically, our results are in agreement with the work of (Leddy *et al.* 1999) and of (Stevens *et al.* 2013) on the impacts of wind turbines on grassland songbirds and the work of (Fernández-Bellon *et al.* 2019) on forest songbirds where a clear displacement gradient was observed at similar distance ranges as in this experiment (several dozens to a couple hundred meters). Up-scaling the impacts of a single wind turbine to entire wind farms, especially given their rapidly increasing global distribution (He & Li 2020), points to a large-scale source of functional habitat loss and fragmentation worldwide. Since a primary driver of this fragmentation, noise pollution, is invisible, the risk of it being overlooked by landscape-centered fragmentation metrics is high (Berger-Tal & Saltz 2019). Importantly, this impact is especially troubling given the fact that wind turbines are usually placed away from human settlements or other infrastructures, pushing wind farms to natural sites with minimal anthropogenic noise pollution. We strongly recommend that the impacts of noise pollution on wildlife should be included in ecological impact assessments for wind power infrastructure. Moreover, the predictable spatial pattern of wind turbine noise (Rogers *et al.* 2006, Katinas *et al.* 2016) may allow for the mitigation of its impacts through proper planning and placement of turbines.

A question that remains open is what causes the reduction in the number of captured birds during our noise broadcasts – Do birds leave the noisy area altogether, or are they just less active and therefore less likely to be captured by the mist-nets? We suggest that for the most part, our results are due to habitat avoidance and displacement. Thanks to the setting of the experiment, it is evident that the number of the migratory Lesser whitethroats remains low also in the last phase, after the noise treatment ended. Additionally, there are significantly fewer juveniles both during and after experimental noise exposure compared to their rate before it. These findings support the notion that the population of Lesser whitethroats was altered by the broadcast of wind turbine noise and did not shift back to baseline because the birds had left the site. Our finding that the abundance of nonmigratory Sardinian warblers only partially recovered after the noise exposure also supports this argument. We suggest that the displaced Lesser whitethroats, a migratory species, moved further away (to other stopover sites or continued migrating) whereas the displaced Sardinian warblers, stayed close-by and returned faster to the site after the noise receded. Future studies should further test the underlying mechanism of the reduced numbers of observed birds, for example, by using innovative tracking devices (Toledo *et al.* 2014, Taylor *et al.* 2017).

Species composition and population level responses

In contrast to previous studies on the effects of noise pollution on songbirds communities (Francis *et al.* 2009, McClure *et al.* 2013, Senzaki *et al.* 2020b) we found no significant change to the songbird species diversity. However, when examining the response rate in the three most common species, all belonging to the *Sylvia* warbler family, we found a clear difference among them with only two species out of three showing a reduction in abundance due to noise. This implies that the lack of significant change in community structure in our study might result from a small sample size of species (the abovementioned studies had more than 40 species whereas we had 26) or individuals. Surprisingly, previous studies on the effects of wind turbines on songbirds did not test for changes in community structure (but see (Leddy *et al.* 1999, Villegas-Patraca *et al.* 2012). Therefore, we recommend that further work be done to determine whether wind turbine noise causes changes in songbird species composition, to be able to provide valid conservation recommendations.

We found that in the lesser whitethroat the response to noise is dependent on the age of the bird, with a distinct reduction in the number of juveniles during and after noise exposure, but with no significant change in the abundance of adults. This finding stands in contrast to many studies on bird populations showing that less experienced individuals are the ones to normally stay in disrupted or lesser-quality habitats during migration or right upon arrival to breeding ground (Yong *et al.* 1998, Habib *et al.* 2007, McClure *et al.* 2017). This result perhaps reflects a higher tendency of juveniles to move when redistributing at stopover sites (Chernetsov 2006), this time in response to noise pollution. Nonetheless, it is yet to be tested whether inexperienced juveniles are less capable of withstanding the cost of noise compared to adult individuals, and therefore make the effort of avoiding the noisy habitat.

A recent study, testing the effects of anthropogenic noise pollution on 322 bird-species, showed that noise is shaping the *breeding* distribution of migratory species to a similar extent as nonmigratory species (Klingbeil *et al.* 2020). However, the effects of noise on migrants during their migratory journey was not tested. Here, we found no difference among migrants and nonmigrants in their response to wind turbine noise. Furthermore, of the two focal species whose numbers significantly decreased during the noise broadcast, one was migratory and the other nonmigratory. This result contradicts our expectation that migrants will show higher rate of avoidance during migration, based on greater reliance on acoustic communication or on acoustic environmental cues in a novel environment (Berger-Tal *et al.* 2014), or on their state of mobility (McClure *et al.* 2013). This result may be at least partially explained by the fact that nonmigratory species during post-breeding season do not defend territories and roam locally, which means that both migrants and nonmigrants during this time are not fully familiar with their surroundings or have a similar state of mobility. Importantly, despite similar degree of reaction to noise between migratory and nonmigratory species, the decline in the numbers of migrants is especially alarming. Migratory bird populations are globally declining faster than nonmigratory birds, with notable declines in populations migrating to East Africa (Sanderson *et al.* 2006, Laaksonen & Lehikoinen 2013, Vickery *et al.* 2023), as do the populations examined in this study. This decline is partly because of habitat quality degradation at stopover sites (Sillett & Holmes 2002). Our results demonstrate that by degrading habitat quality, anthropogenic noise can be a key factor threatening migrating birds.

Limitations of the study

While our study addresses an important knowledge gap on the impacts of wind turbine noise in a natural environment during the nonbreeding season, it is essential to acknowledge its limitations. Firstly, the study was conducted at a single site with a specific habitat type and a limited number of songbird species, thereby restricting the generalizability of the findings. Additionally, we broadcasted a recording from a single turbine, as outlined in the methods. Although it is recommended to use multiple recordings and replicate them several times (Kroodsma *et al.* 2001), we faced limitations due to the unavailability of high-quality recordings from various turbine models. Moreover, the short duration of the post-breeding season limited us to only six repetitions, a low number that is suitable for statistical analyses for a single site and recording (based on expected residual variance values). Hence, we chose to repeatedly examine the response to a single recording at a single site. Furthermore, the study was conducted within a single year, overlooking potential inter-annual variation that could impact the observed outcomes. Importantly, these limitations do not invalidate our results, but they constrain their generalizability and emphasize the need for further research to better evaluate the ecological impacts of wind turbine noise in other conditions.

Summary

The impact of noise pollution on wildlife is rarely considered in environmental impact assessments for most types of infrastructures, and especially for wind turbines (Teff-Seker et al. 2022). Despite the remaining knowledge gaps, decision makers can already rely on existing mapping tools, sound propagation models, and the vast noise pollution scientific literature to minimize hazardous impacts of wind turbine noise on biodiversity. Our results stress the need to consider these noise impacts on the environment when planning wind farms or other noisy infrastructures. Doing so will help preserve stopover sites for migratory birds and assure the availability of habitats used by birds and wildlife in general.

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Table 1. GLMM results for birds' abundance: in this model we explored how bird abundance (daily number of birds caught) was affected by wind turbine noise. Three experimental phases were compared: "before", "noise-treatment", and "after". The model also included the experimental repeat number (1-6) as a random factor to account for possible temporal variation in the birds' community and habitat along the season and for biases that may be caused by overdispersion in the Poisson model. The model was structured as following: Daily bird number ~ Treatment phase + (1|WeekNum). *Error Distribution:* Poisson. *Link function:* log. Table sections A and B present the results of the same model with the intercept including the "before" phase in A and the "after" phase in B. Variables in bold were found to significantly affect bird abundance and the stars at the rightmost column represent the level of significance: * - p < 0.05, ** - p < 0.01, *** - p < 0.001.

A	**Exp(estimate)**	**Estimate**	**Std. Error**	**z value**	**Pr(>|z|)**	
Intercept	38.99	3.66	0.16	23.49	0	***
Treatment phase Noise-treatment	0.69	-0.38	0.1	-3.8	0.0001	***
Treatment phase After	0.94	-0.06	0.09	-0.64	0.52	
B						
Intercept	36.79	3.61	0.16	23	0	***
Treatment phase Noise-treatment	0.73	-0.32	0.1	-3.18	0.001	**
Treatment phase Before	1.06	0.06	0.09	0.64	0.52	

Table 2. GLMM results for birds' abundance and spatial distribution: in this model we explored how bird abundance (daily number of birds caught) was affected by wind turbine noise. Three experimental phases were compared: "before", "noise-treatment", and "after". We also included in the model distance (m) and direction (deg) from the phantom wind turbine, their interactions with treatment phase ("before", "noise-treatment", or "after") and the three-way interaction between distance, direction, and treatment phase. In the two-way interactions, the "noise-treatment" and "before" phases were each compared to the intercept ("after" phase and its interactions with distance and direction). The model also included the experimental repeat number (1-6) as a random factor to account for possible temporal variation in the birds' community and habitat along the season and for biases that may be caused by overdispersion in the Poisson models. The model was structured as following: birds' abundance ~ treatment phase:Distance:Direction - Distance:Direction + experimental repeat number (random factor). Colons represent interactions, variables in bold were found to significantly affect bird abundance and the stars at the rightmost column represent the level of significance: * - p < 0.05, ** - p < 0.01, *** - p < 0.001.

	Exp(estimate)	**Estimate**	**Std. Error**	**z value**	**Pr(>|z|)**	
Intercept	2.37	0.86	0.15	5.6	<0.001	***
Before	1.08	0.08	0.1	0.83	0.41	
Noise-treatment	0.70	-0.36	0.11	-3.22	<0.01	**
Direction	1.00	0	0.07	-0.06	0.95	
Distance	1.16	0.15	0.07	2.16	0.03	*
Before:Distance	0.94	-0.06	0.1	-0.62	0.53	
Noise-treatment:Distance	1.16	0.15	0.11	1.32	0.19	
Before:Direction	0.96	-0.04	0.1	-0.46	0.65	
Noise-treatment:Direction	1.10	0.1	0.11	0.87	0.38	
Before:Distance:Direction	0.90	-0.1	0.07	-1.6	0.11	
Noise-treatment:Distance:Direction	0.77	-0.27	0.08	-3.43	<0.001	***
After:Distance:Direction	0.85	-0.16	0.07	-2.35	0.02	*

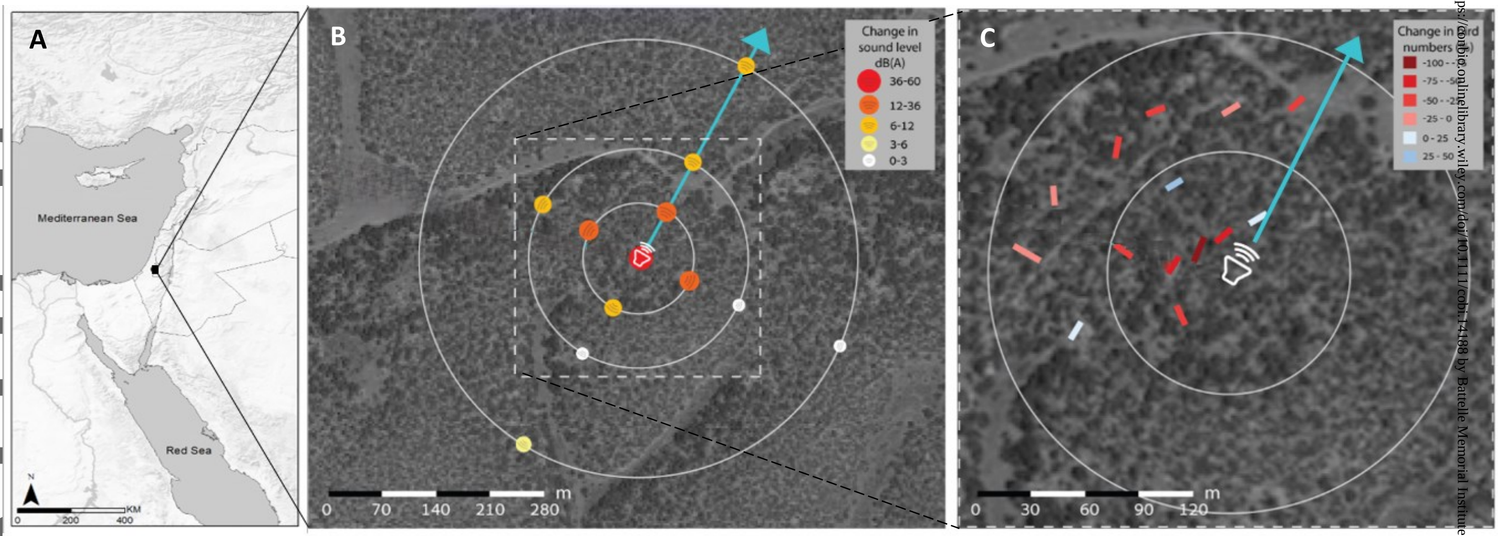

Figure 1. **A.** Map of the region. Study site location marked with a black rectangle. **B**. The change in ambient sound level between the “noise-treatment” and “before” phases as measured by the difference between the mean L_{50} (median) noise level in dB(A) during bird-ringing hours (4.5 hours starting 45 min before sunrise; Table S4). The "after" phase is absent due to its similarity to the "before" phase. Redder colors represent larger changes in sound levels. The change in ambient sound level was measured in three distances (grey circles; 70, 140 and 280 meters from speaker) and four directions based on expected wind direction (light blue arrow; pointing NNE). Dashed square is enlarged in C. **C.** The change in number of birds between “noise-treatment” and “before” phases, as measured by the reduction of the mean number of birds between these two phases, is presented according to the sampling location (mist nets). The "after" phase is absent due to its overall similarity to the "before" phase. These changes were pooled into colored groups with darker red colors representing a stronger decline in the number of birds whereas blue representing an increase in the number of birds. A steeper decline was found based on the combination of a shorter distance to the phantom wind turbine and sound direction (blue arrow; based on the expected downwind side) during the “noise-treatment” compared to when no noise was broadcasted (see statistical model in Table 2).

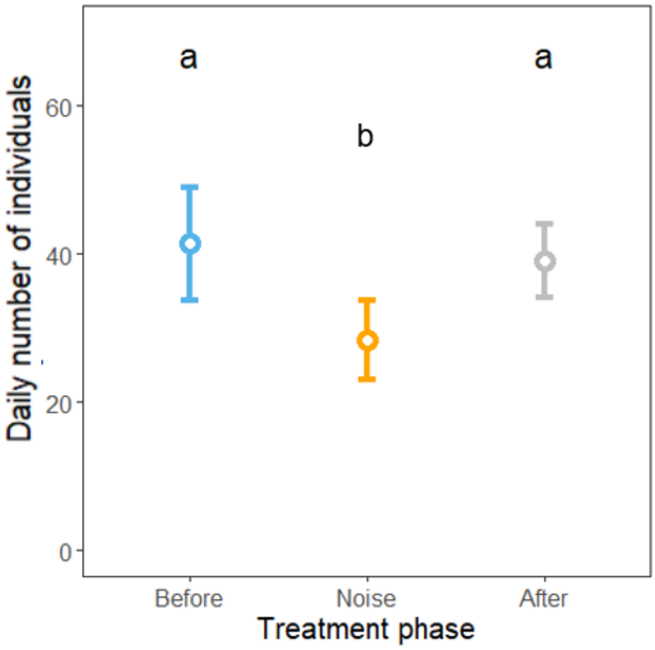

Figure 2. Songbird abundance (y-axis) measured as daily number of individuals of all species, across experimental treatment phases (x-axis): "before" (blue), "noise-treatment" (orange), and "after" (grey). Each phase lasted 48 hours, with the "noise-treatment" involving broadcasting of wind turbine noise. The experiment was repeated six times during the fall (post-breeding and migration) season of 2018. The circle represents the mean, the vertical lines represent the standard error. Different superscript letters indicate significant difference (p<0.005; see statistical model in Table 1).

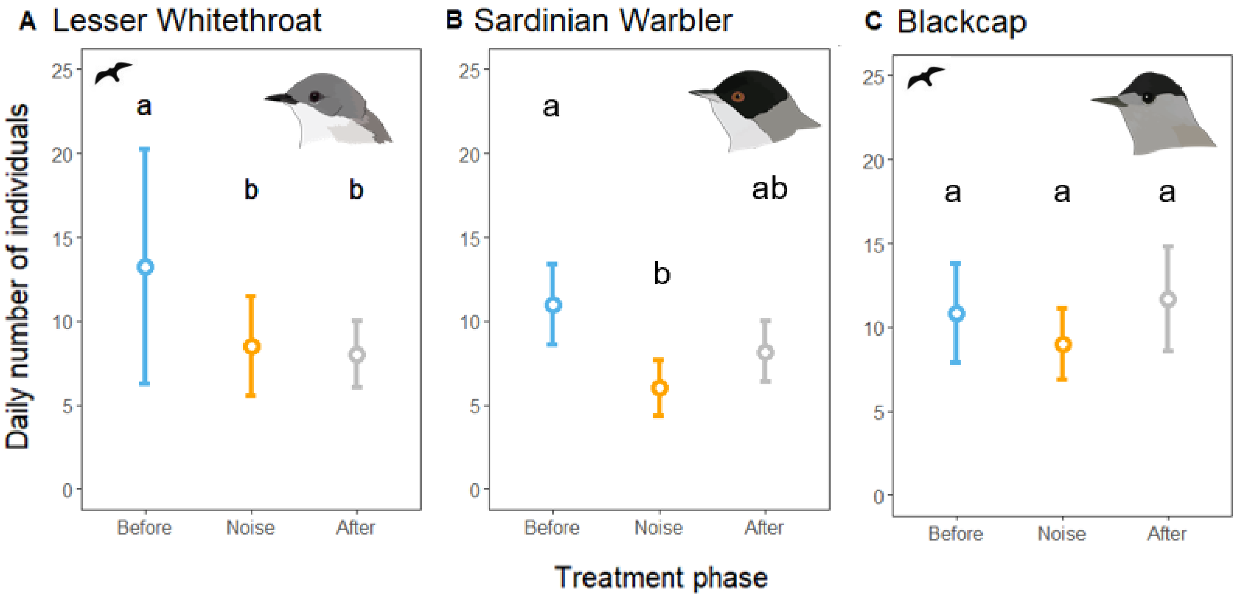

Figure 3. The abundance (y-axis) of *Sylvia*-warbler species: **A.** lesser whitethroat (*Sylvia curruca; migratory)* **B.** Sardinian warbler (*S. melanocephala momus; nonmigratory)*, and **C.** Eurasian blackcap (S. *atricapilla; migratory)* measured as the daily number of individuals of each species, across experimental treatment phases (x-axis): "before" (blue), "noise-treatment" (orange), and "after" (grey). The circle represents the mean, the vertical lines represent the standard error. Different superscript letters indicate significant differences for each species (p<0.05; see statistical models in Tables S9 and S10). The flying bird icon at the upper left corner of panels A and C represents a migratory species.

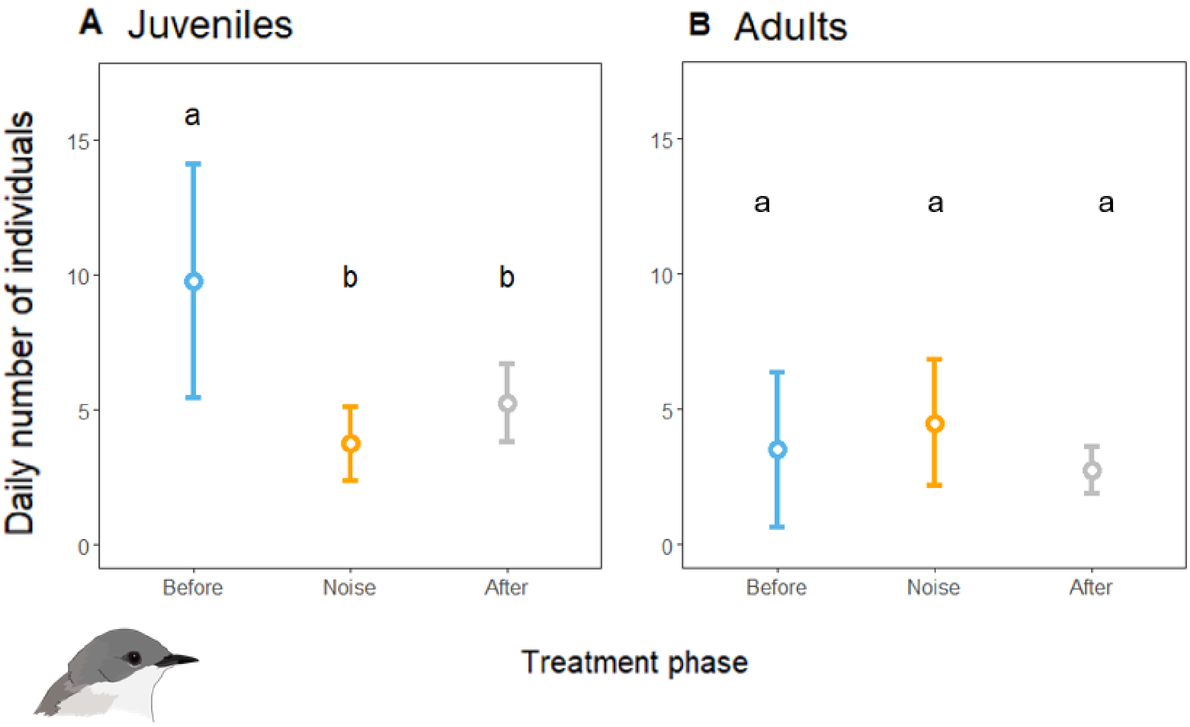

Figure 4. The abundance (y-axis) of **A.** juvenile and **B.** adult lesser whitethroats (*Sylvia curruca)* measured as their daily number across experimental treatment phases (x-axis): "before" (blue), "noise-treatment" (orange), and "after" (grey). The circle represents the mean, the vertical lines represent the standard error. Different superscript letters indicate significant difference for each age group (p<0.05; see statistical model in S12).